Paludiculture as paludifuture on Dutch peatlands: An environmental and economic analysis of *Typha* cultivation and insulation production

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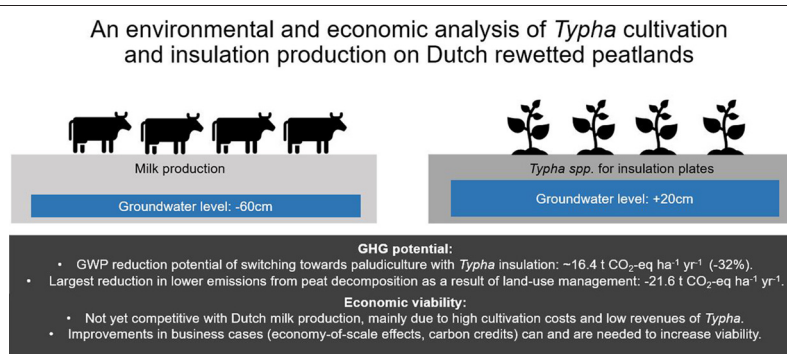
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HIGHLIGHTS

- Drained agricultural peat soils cause high GHG emissions (GHGe).
- Rewetting peatlands for *Typha* insulation production can reduce GHGe significantly.
- *Typha* insulation only achieves high GHGe reductions when including biogenic carbon.
- Economically, *Typha* cultivation is not competitive with Dutch dairy production.
- The viability is, however, accompanied by and sensitive to many uncertainties.

GRAPHICAL ABSTRACT

An environmental and economic analysis of *Typha* cultivation and insulation production on Dutch rewetted peatlands

GHG potential:
- GWP reduction potential of switching towards paludiculture with *Typha* insulation: ~16.4 t CO_2-eq ha^{-1} yr^{-1} (-32%).
- Largest reduction in lower emissions from peat decomposition as a result of land-use management: -21.6 t CO_2-eq ha^{-1} yr^{-1}.

Economic viability:
- Not yet competitive with Dutch milk production, mainly due to high cultivation costs and low revenues of *Typha*.
- Improvements in business cases (economy-of-scale effects, carbon credits) can and are needed to increase viability.

ARTICLE INFO

ABSTRACT

Paludiculture, the cultivation of crops on rewetted peatlands, is often proposed as a viable climate change mitigation option that reduces greenhouse gas emissions (GHGe), while simultaneously providing novel agricultural business options. In West Europe, experiments are ongoing in using the *paludicrop* cattail (*Typha* spp.) as feedstock for insulation panel material. Here, we use a Dutch case study to investigate the environmental potential and economic viability of shifting the use of peat soils from grassland (for dairy production) to *Typha* paludiculture (for cultivation and insulation panel production). Using a life cycle assessment and cost-benefit analysis, we compared the global warming potential (GWP), yearly revenues and calculated Net Present Value (NPV) of 1 ha Dutch peat soil used either for dairy production or for *Typha* paludiculture. We estimated that changing to *Typha* paludiculture leads to a GWP reduction of ~32% (16.4 t CO_2-eq ha^{-1}), mainly because of lower emissions from peat decomposition as a result of land-use management (-21.6 t CO_2-eq ha^{-1}). If biogenic carbon storage is excluded, the avoided impact of conventional insulation material is insufficient to compensate the impact of cultivating and processing *Typha* (9.7 t CO_2-eq ha^{-1}); however, this changes if biogenic carbon storage is included (following PAS2050 guidelines). *Typha* paludiculture is currently not competitive with dairy production, mainly due to high cultivation costs and low revenues, which are both uncertain, and will likely improve as the system develops. Its NPV is negative, mainly due to high investment costs. This can be improved by introducing carbon credits, with carbon prices for *Typha* paludiculture (30 years) comparable to EU-ETS prices.

In conclusion, Dutch *Typha* paludiculture has a significant climate change mitigation potential by reducing emissions from deep drained peatlands. Nevertheless, attention is needed to increase its economic viability as this is a key aspect of the system change.

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1. Introduction

Peat soils cover only 3% of the world's land surface but store approximately one third of the world's soil carbon, which amounts to at least 450 Gigaton of carbon (Yu et al., 2011). The use of peat soils for agriculture is often facilitated by drainage, in which the water table is lowered to make these nutrient-rich soils more suitable for cultivation (Rienks and Gerritsen, 2005; Wilson et al., 2016). In the Netherlands, the majority of peat soils are drained up to 1 m below the soil surface, using ditches, to enable dairy farming (Rienks and Gerritsen, 2005; Van den Born et al., 2016). However, drainage exposes peat to oxygen, which causes it to rapidly decompose and which reduces peat volumes, leading to land subsidence that damages both the infrastructural and the natural environment (Rienks and Gerritsen, 2005; Fritz et al., 2014). Moreover, decomposition of peat combined with agricultural use causes the release of large quantities of stored carbon in the form of CO_2 combined with the more potent greenhouse gas (GHG) nitrous oxide (N_2O). While covering less than 10% of the Dutch agricultural area, peat soils are responsible for more than half of the greenhouse gas emissions (GHGe) from agricultural soils (Fritz et al., 2014) and roughly 2.5% of the total Dutch anthropogenic GHGe (Van den Akker et al., 2010). This is why preserving and protecting peatlands is a key priority in mitigating climate change (Leifeld and Menichetti, 2018), and the Dutch government has set itself the goal of reducing emissions from peatlands by 1 Mt CO_2 by 2030 (Klimaatakkoord, 2019).

Both land subsidence and release of GHGs result from oxidation caused by drainage, and thus they can be mitigated by rewetting peat soils (Joosten et al., 2012; Geurts et al., 2019). However, rewetted peat soils are less suitable for agriculture and dairy farming, as common agricultural crops are not adapted to such wet conditions and the limited load-bearing capacity of wet peat soils restricts their accessibility for agricultural machinery (Schröder et al., 2015). To avoid peatland losing its value for agriculture completely, agricultural practices adapted to wet peatlands, i.e. paludiculture, are currently being studied and refined (Wichmann, 2017). In addition to the climate mitigation achieved by rewetting peat soils, paludicrops absorb substantial amounts of nutrients, thus providing an effective way to control nutrient effluxes from rewetted sites as well as to sequester carbon in biomass products (Fig. 1; Geurts et al., 2020). However, a potential trade-off of rewetting peatlands is the formation of the GHG methane (CH_4), which is formed
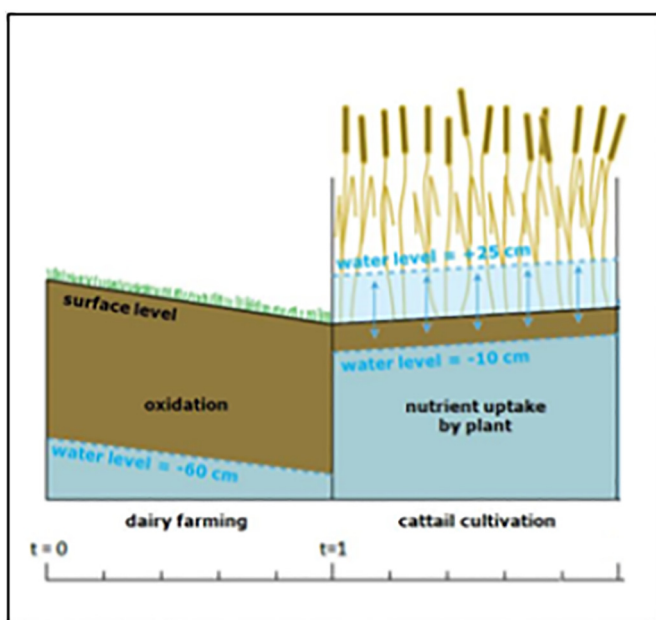
in anaerobic conditions when water levels are around or above the soil surface (Günther et al., 2015; Bansal et al., 2019). If land users, who are mostly farmers, are to adopt alternative practices on agricultural land, these practices must be feasible and economically competitive with current practices (Wichmann, 2017; Van der Hilst et al., 2010).

Cattail (*Typha* spp.) is a promising paludicrop that is easily recognized by its unique cigar-shaped head (Heinz, 2011; Geurts et al., 2019). It grows naturally in nutrient-rich ponds, paddies and lakes, which suggests its suitability to grow on rewetted peat soils. The peat-forming paludicrops that are the optimal compromise between biomass production, climate mitigation and peat preservation mostly grow at groundwater levels at or slightly below the soil surface; however, cattail performs better at higher water levels of 0 to 20 cm (Geurts et al., 2019) or of even up to 25 cm above soil surface (Van de Riet et al., 2014) (Fig. 1), which may lead to CH_4 emissions (Geurts et al., 2019). Its characteristics make cattail a suitable material for producing insulation plates with competitive insulation properties (Luakanchanaphana et al., 2012) such as a lower thermal conductivity than polystyrene (Georgiev et al., 2014). Compared to common insulation materials, bio-based materials such as *Typha* have the advantage of being renewable, being highly resistant to mould, and having a good hygrothermal performance that is beneficial to humidity control and indoor air quality, while at the same time storing sequestered biogenic carbon for decades (Krus et al., 2014; Pittau et al., 2018). Currently, most buildings are insulated using materials such as polyurethane or mineral wool (La Rosa et al., 2014), which are commonly produced from petrochemicals (mainly polystyrene) or which require high energy input during production (glass and rock wools) (Asdrubali et al., 2015). The rising demand for insulation material and the targets set for reducing the environmental impact of its production (Pavel and Blagoeva, 2018) has resulted in a growing demand for eco-friendly thermal insulators, and insulation panel material based on *Typha* may well meet this demand.

Cultivating and producing *Typha* on peat soils which are currently drained for dairy production requires a system change. Previous research has illustrated the impact of relevant parts of this system change, such as rewetting peatlands (e.g. Erkens et al., 2016; Wilson et al., 2016) and the environmental performance of bio-based insulation (e.g. Asdrubali et al., 2015; Casas-Ledón et al., 2020). While *Typha* cultivation is proposed as a viable mitigation option for rewetted peat soils (Vroom et al., 2018), little information is available on the economic viability of a system change to paludiculture, which is likely to involve many uncertainties (Wichmann, 2017). However, such information is essential to assess the potential of the system, as economic viability is considered to be one of the key factors within a system change (Van der Hilst et al., 2010).

Such a change has far-reaching consequences, for example for land use as well as for the markets for insulation materials and the markets for milk and meat; this is why we consider a systems approach to be essential to assess the suitability and potential of both rewetting peat soils and paludiculture. Therefore, the aim of this paper is to holistically assess the environmental and economic potential of changing the use of Dutch peat soils by performing a life cycle assessment (LCA) and cost-benefit analysis of peatland use for dairy and *Typha* insulation panel production.

2. Methods

2.1. Environmental impact

The difference in environmental impact was explored by comparing the impact of the reference system (RS) of dairy production to that of the alternative system (AS) with *Typha* production. For this, we used an LCA, which is a holistic approach to assess environmental impacts throughout a product's life-cycle (Finnveden et al., 2009). Following LCA standards (Finnveden et al., 2009; Singh et al., 2010), the goal and scope of this LCA are defined below, followed by the inventory analysis

Fig. 1. Transition from dairy farming to cattail cultivation in rewetted peatlands (Van de Riet et al., 2014).

of both RS (Section 2.1.1) and AS (Section 2.1.2). The impact assessment is described in the results and the interpretation of the results in the discussion.

Both systems require land-use management, either related to grassland management of dairy production (RS) or *Typha* cultivation (AS) (Fig. 2). To reflect a change in land use, the functional unit was set to 1 ha of Dutch peatland. As the AS was proposed to offset the contribution of peatland management to GHGe, this LCA was performed for the indicator 'global warming potential' (GWP) for a 100-year time horizon (Wilson et al., 2016). This indicator, expressed in CO_2-eq, adds up the three main GHGs, namely CO_2, CH_4 and N_2O, that are related to peatland use and conditions, and its potencies are based on a relative scale which compares the specific GHG with an equivalent mass of CO_2, whose GWP is by definition equal to 1 (Myhre et al., 2013).

The common system boundaries were assumed for each system, i.e. cradle-to-farm-gate for dairy production (Doornewaard et al., 2019) and cradle-to-gate for insulation (Asdrubali et al., 2015; Casas-Ledón et al., 2020). This implies that the impact of cultivation and processing were considered as well as the input needed for these processes. Within the cradle-to-gate approach for *Typha* panel insulation, installation and maintenance of the panel as well as the disposal stage at the end of its life (grave) were not considered because of the novelty of the material and the consequent lack of data about these stages; this is in line with LCAs on bio-based insulation material (Zampori et al., 2013; Casas-Ledón et al., 2020). If there were multifunctional processes at dairy production, environmental impacts were allocated over the multiple outputs with economic allocation for feed and biophysical allocation for milk and meat in compliance with the International Dairy Federation (IDF, 2015). For multifunctional processes within the *Typha* lifecycle, we assumed that by-products from cultivation, processing and manufacturing did not have a significant economic value; thus, all impacts were assigned to the *Typha* insulation panels. In line with the IPCC recommendations, the consequences of the system change beyond the system boundary were accounted for by using system expansion (Ekvall and Finnveden, 2001). System expansion assumes that the input and output changes of a system are compensated by adding or subtracting the impact of the most likely alternatives (Ekvall and Finnveden, 2001).

2.1.1. Reference system

2.1.1.1. Land-use management. Throughout the Netherlands there is a variation in the drainage depth and soil properties of dairy grasslands, and this influences the extent of peat oxidation and GHGe (Kuikman et al., 2005). In this study, we assumed an annual GHGe of 32.7 t CO_2-eq ha^{-1}, which was based on a specific case study in Zegveld, the Netherlands. The field under consideration had a peat soil without clay layers with a mean subsidence rate of 0.0118 m yr^{-1}, an average drainage depth of 60 cm below soil surface, 80% assumed soil organic matter, a bulk density 140 kg m^{-3} and a carbon content of 55% of the peat material with a C/N ratio of 20. This resulted in a C-mineralisation of 7.3 t ha^{-1} yr^{-1} and a N-mineralisation of 363 kg ha^{-1} yr^{-1} (Kuikman et al., 2005). The total GHGe consisted of 26.6 t CO_2 ha^{-1} and 6.1 t CO_2-eq ha^{-1} N_2O. In the current study we assumed a replacement of peat soil without ditches by growing cattails. Therefore, in the RS, CO_2 emissions from dissolved organic carbon (DOC) were not taken into account, and CH_4 emissions were assumed to be insignificant (Jurasinski et al., 2016; Tiemeyer et al., 2020; Van den Pol-Van Dasselaar et al., 1999). This field and the related GHGe were considered to be representative of other Dutch peatlands with a similar drainage depth.

2.1.1.2. Dairy production. The assumed GWP of dairy production was 19.4 t CO_2-eq ha^{-1} yr^{-1}, based on the average GWP of milk produced in the Netherlands of 1.195 kg CO_2·eq kg^{-1} Fat and Protein Corrected Milk (FPCM) with an average allocation of 86% of emissions to milk and 14% to livestock and meat production in 2018 (Doornewaard et al., 2019). Stocking density was assumed at 1.81 dairy cows ha^{-1} with a milk yield of 8960 kg cow^{-1} yr^{-1}, based on the reference year 2018 (Dutch Agro and Food portal from Wageningen University & Research, 2020).

2.1.2. Alternative system

2.1.2.1. Land-use management. In this study, we focused on the species *Typha latifolia*. *T. latifolia* is native to the Netherlands and adapted to wide habitat variations; most *Typha* research in North-West Europe has been focused on this species (Geurts and Fritz, 2018; Geurts et al., 2020). For Dutch paludiculture, an average water level of about 20 cm above soil surface was assumed as the condition for optimal growth (Wichtmann and Joosten, 2007; Bansal et al., 2019). We assumed that this water level resulted in 11.1 t CO_2-eq ha^{-1} from CH_4 emissions only (Jurasinski et al., 2016), as rewetted peatlands generally emit negligible N_2O emissions (Tiemeyer et al., 2020; Vroom et al., 2018) and dissolved organic carbon (DOC) was excluded (Kuikman et al., 2005). Moreover, we considered carbon sequestration in the root system in the peatland, with a one-time average of 20 t CO_2-eq ha^{-1} additional underground carbon storage compared to grassland (Geurts and Fritz, 2018). Adjusted to the assumed lifetime of the rewetting project
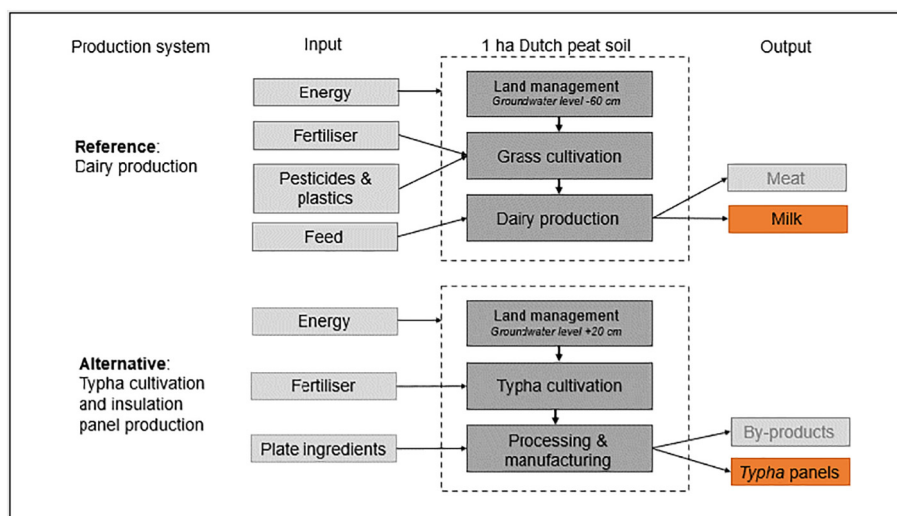
Fig. 2. Schematic overview of considered categories and inflow and outflow in the RS and the AS.

(30years) (Van de Riet et al., 2014), this results in a one-time carbon sequestration of 0.67 t CO_2-eq ha^{-1} yr^{-1}.

2.1.2.2. Cultivation of Typha. Planting was assumed to be done once in the lifetime of the *T. latifolia* crop, with a planting density of 2 plants per m^2, using a two-row planter (Ecoinvent, 2013). The energy needed for producing the *Typha* plants (rhizomes) and related water use was excluded. As harvesting results in removed minerals and depleted reserves in the *Typha* root system (Bansal et al., 2019; Geurts et al., 2020), we assumed that fertilizers need to be applied annually between plant re-growth and biomass harvesting, with a N:P:K ratio of 8:1:8 (Pijlman et al., 2019) (see Table A1 in the Appendix). The crop rotation period was assumed to be 30 years, which is equal to the project lifetime of rewetting projects (Van de Riet et al., 2014), although long-term data of *T. latifolia* yields are scarce (e.g. Geurts et al., 2020). Mowing was assumed to take place annually in February when the crop is as dry as possible (Bestman et al., 2019; Geurts et al., 2020), and it was performed with a light-weight rotary mower. Fuel consumption under wet conditions was assumed to be 14 l h^{-1} for mowing and 18 l h^{-1} for loading (Wichmann, 2017). We assumed that mowing required 1.3 h ha^{-1} yr^{-1} and loading 3.1 h ha^{-1} yr^{-1}, based on the harvesting and transport of reeds on rewetted peatlands (Wichmann, 2017). The impact of diesel was assumed to be 3.23 kg CO_2-eq well-to-wheel. Detailed information about the different categories and related impacts can be found in Table A1 (Appendix).

Annual *T. latifolia* winter dry matter (DM) yields were assumed to be 8.5 t ha^{-1} (Heinz, 2011; Geurts et al., 2020). This yield was assumed to include 10% other plants (e.g. grasses or wetland plants such as *Sparganium* or *Phragmites australis*). Additionally, 5% DM was assumed to be lost during harvest (Wageningen Livestock Research, 2019). In total, the impact of cultivation of 7.3 t DW ha^{-1} *Typha* was calculated as being 2.6 t CO_2-eq ha^{-1} (Table 1).

2.1.2.3. Processing and panel manufacturing. *Typha* was assumed to be transported over 50 km with a diesel tractor and trailer (loading capacity of 26.2 t) to the processing plant, with an impact of 0.102 kg CO_2-eq t^{-1} km^{-1}. To process the harvested *Typha* into an insulation panel component, it was screened, cleaned and dried; we assumed that 8% of the yield was rejected before this process. The average moisture content at harvest was assumed to be 49% (Geurts et al., 2020); as a result, we assumed that it was necessary to dry the biomass to 6% moisture, which is appropriate for insulation panels (Luakanchanaphana et al., 2012). The impact of drying was assumed to be 2.6 t CO_2-eq, which was based on the moisture content and energy consumption for grass drying at 110–120 °C (Table 1) (Ecoinvent, 2013). During this process, 6% of the dry mass was lost as dust (Bajwa et al., 2015; Grosshans et al., 2013), resulting in 6.3 t ha^{-1} biomass assumed to be used for *Typha* insulation panels.

Typha was then screened and dried biomass was pressed into insulation panels (Krus et al., 2014). On a weight basis, these panels contained 82% *Typha* biomass and the remainder consisted of 14% binder (polyester polymer fibre), 0.5% fungicide (thiocarbamate) and 3.5% flame retardant (borax) based on hemp and kenaf insulation panels (Ardente et al., 2008; Zampori et al., 2013). The shredding process required 0.6 t CO_2-eq ha^{-1}, based on 187 l diesel needed for hemp shredding, adjusted for

yield variations (Zampori et al., 2013). Panel manufacturing, including fibre preparation, carding, thermal binding, cutting of the panel and packaging, resulted in 0.81 kg CO_2-eq per *Typha* panel and 3.9 t CO_2-eq ha^{-1} in total (Table A2), 40% of which attributed to the production of the *Typha* panel itself and 60% allocated to the production of the other components (see Fig. A1). Production impacts were based on hemp panel production (Zampori et al., 2013) as no data on the impact of *Typha* panel production are available, and data were combined using the Dutch energy mix (0.556 kg CO_2 kWh^{-1}; Table A2). Table A2 (production process) and Table A3 (assumed properties) provide information and inventory data for panel manufacturing and related input, and the assumed properties for and output of *Typha* panels per ha^{-1}.

Different processes and related CO_2-eq emissions within the *Typha* lifecycle are given in Table 1. In total, the *Typha* lifecycle involved emissions of 9.73 t CO_2-eq.

2.1.2.4. Reference insulation material to be substituted. The *Typha* insulation produced was assumed to substitute the most-used insulation material in Europe, which is a mixture of low-density 50% glass and 50% stone wool (Papadopoulos et al., 2002). For equal substitution, the inorganic insulation should have the same thermal value as the *Typha* insulation, which was compared using the panel weight needed to achieve a thermal value (R) of 1 m^2 K W^{-1}. Based on their difference in thermal value (Table A3), the manufactured *Typha* panels replace 2447 panels of insulation mixture with an impact of 3.41 kg CO_2-eq per panel (Ecoinvent, 2013), thus avoiding a total of 8.36 t CO_2-eq.

2.1.2.5. Replacing dairy production. Assuming that the demand for milk will remain after the system change, we replaced the former dairy production to elsewhere in West Europe with an average impact of 1.47 kg CO_2-eq kg^{-1} FPCM (FAO, 2010, after an allocation correction to milk and meat). The total impact of this dairy production was 20.9 t CO_2-eq ha^{-1} yr^{-1}.

2.2. Economic analysis: comparing yearly income and Net Present Value

We performed a cost-benefit analysis by comparing revenues from 1 ha Dutch dairy production and *Typha* paludiculture and calculating the Net Present Value (NPV) (Blok and Nieuwlaar, 2020).

2.2.1. Comparison of net yearly income from dairy production

As reference we used the business case of Vogelzang and Blokland (2011), which was based on a Dutch dairy farm in a peatland area (*Groene Hart*) in the period 2001–2009. The dairy farm had an average of 69 cows and 45 ha of cultivated land, of which 35 ha were in use as permanent grassland. Milk intensity was more than 11,000 kg milk ha^{-1} yr^{-1}, which implies that this reference farm was slightly less intensive than the dairy production assumed in the LCA; however, these data were used as representative of a Dutch dairy farm on peatland. This business case assumed a net revenue of €8.35 per 100 kg milk, based on €42.35 revenues, €12.09 costs and €21.91 non-allocated costs per 100 kg milk (Vogelzang and Blokland, 2011), resulting in a net revenue of approximately €1350 ha^{-1} for 16,218 kg FPCM milk ha^{-1} on peatlands.

2.2.2. Net Present Value of Typha paludiculture

We assumed that projects are only undertaken if the total benefits exceed the costs resulting in a NPV value that is positive if a project is economically viable. The NPV can be calculated with Eq. (1), with initial investment (*I*), annual benefits and costs (*B* and *C*), discount rate (*r*) and lifetime of the project (*n*). The discount rate, used to determine the current value of future cashflows, was assumed to be 6% for paludiculture and this is based on realistic farmer loans (5.5% in Van der Hilst et al., 2010) and a 30-year project lifetime, which is comparable to other rewetting projects (30–50 years) (Van de Riet et al., 2014).

Table 1
Overview of *Typha* lifecycle and associated GHGe in ha^{-1}.

Process	t CO_2-eq ha^{-1}
Cultivation of *Typha*	2.64
Processing and panel manufacturing	
Shredding fibres	0.60
Drying	2.56
Panel manufacturing	3.89
Transportation	0.04
Total	9.73

Calculation of Net Present Value (Blok&Nieuwlaar, 2020)

$$NPV = -I + \sum_{i=1}^{n} \frac{B-C}{(1+r)^i} \qquad (1)$$

The costs and revenues of crop production are regionally specific and depend on aspects such as farm management, soil, climate and economic environment (Van der Hilst et al., 2010). Costs include variable costs (i.e. cultivation costs), fixed costs, investment costs and overhead costs. Cultivation costs are costs incurred for planting (year 1), products, and the labour and energy needed for fertilizing, harvesting and transportation (years 2–30). The price of diesel was assumed to be €1.24 l^{-1}, based on the average Dutch diesel price in December 2020 (Global Petrol Prices, 2020). Together, the costs were assumed to be ~€1100 ha^{-1} yr^{-1} on average (Table 2). Investment costs included €7300 ha^{-1} for dams and pumps (Van de Riet et al., 2014). The benefits from *Typha* cultivation relate to the sale of biomass after cultivation (7.3 t DM ha^{-1} yr^{-1}), which takes place from year 2 onwards; these benefits were assumed to be €150 t^{-1} DM yr^{-1} (Van Duursen and Nieuwenhuijs, 2016). The area-based subsidies which were provided for former dairy production on Dutch peat soils were assumed to remain equal.

2.3. Scenario and sensitivity analysis

Certain assumptions and/or limitations in data and methodology may influence the results; therefore, these were assessed in a scenario and sensitivity analysis. This analysis relates to methodology, i.e. assessing the potential with other water levels, including biogenic storage, and to the sensitivity of the key parameters in both the environmental and the economic system. These have been selected based on expected fluctuations or uncertainty in specific parameters and/or the expected effect of the key parameter on the result (biomass yield, benefits, discount rate and reference insulation material).

Table 2
Individual contributions of cost categories for *Typha* cultivation and benefits from *Typha* biomass, during years 2–30.

	Costs		Benefits	
			Typha biomass	€150 t^{-1} DM yr^{-1}
	Fertilizer and application[a]	93		
	Harvesting[b]	92		
	Transportation[c]	102		
	Labour costs[d]	220		
	Total cultivation costs	507		
	Depreciation of plant material[e]	100		
	Fixed costs[f]	340		
	Overhead costs[g]	150		
Total (€ ha^{-1} yr^{-1})		±1100		1095

[a] Fertilizer was assumed to cost €20 per 100 kg for N, P, K (Agrimatie Wageningen University Prijsontwikkeling van kunstmest, n.d.).

[b] Cultivation processes combined require 4.4 h ha^{-1} yr^{-1}, with fuel consumption under wet conditions assumed to be 14 l h^{-1} for mowing and 18 l h^{-1} for loading (Wichmann, 2017).

[c] Yield was transported by truck for 50 km to the nearest processing plant, with no load for return trips. Diesel use depended on the mass of the transported biomass (empty: 0.2 l km^{-1} and full: 0.4 l km^{-1}) (Smeets et al., 2009).

[d] Total labour hours were calculated by multiplying the time that a machine is used, ~7 h h^{-1} yr^{-1} for fertilization, harvesting and transportation. Labour costs were taken to be €33 h^{-1} and unproductive time required for travelling, servicing, lubricating and training was taken into account (Smeets et al., 2009).

[e] No replanting was needed within 30 years, although long-term data of *Typha* is scarce (Geurts et al., 2020). The assumed costs were €3000 for planting material (Van Duursen and Nieuwenhuijs, 2016), equal to an annual depreciation of €100 ha^{-1} yr^{-1}. Planting was assumed to require ~3 h ha^{-1} and a fuel consumption of 28 l h^{-1} (Ecoinvent, 2013).

[f] Fixed costs relate to machinery costs and depreciation of buildings and machines (Van Duursen and Nieuwenhuijs, 2016), but exclude land rent.

[g] Overhead costs include costs for administration and office (Van Duursen and Nieuwenhuijs, 2016).

2.3.1. Rewetting with other reference drainage depths

As peat oxidation levels depend on drainage depth, the mitigation potential of the proposed system change depends on the depth to which a field is currently drained. While the drainage depth assumed in the RS is within the most common depth range in the Netherlands of 30–60 cm below soil surface (approximately 34% of Dutch peatlands), followed by >90 cm (21%) and 60–90 cm (19%), with shallow drainage mostly occurring in the west and deep drainage in the north of the country (Van den Born et al., 2016). To illustrate the potential in different regions and the relative importance of land management in the proposed systems change, we assessed two alternative reference scenarios with a drainage depth of 30 cm (RS −30 cm) and 90 cm below soil surface (RS −90 cm). We assumed that a linear relationship between groundwater level and soil organic matter decomposition (Kuikman et al., 2005; Drösler et al., 2014; Martens et al., 2021) results in an average GHGe increase of 4.5 t CO_2-eq ha^{-1} yr^{-1} per 10 cm drainage (Jurasinski et al., 2016; Kuikman et al., 2005); consequently, drainage to −30 cm resulted in 19.0 t CO_2-eq ha^{-1} yr^{-1} and drainage to −90 cm in 46.2 t CO_2-eq ha^{-1} yr^{-1}.

2.3.2. Including biogenic carbon storage

Biogenic carbon uptake is one of the main advantages of bio-insulation (Zampori et al., 2013). This uptake was considered in an additional scenario following the Publicly Available Specification 2050-methodology (PAS205). In PAS2050, the part of the carbon that is not emitted to the atmosphere during the 100-year assessment period is treated as stored carbon (for more information on PAS2050 and calculations, see García and Freire, 2014). The lifespan of *Typha* panels was assumed to be 50 years, equal to the lifetime of buildings, with an expected durability of the panel similar to other panels manufactured with natural fibres (e.g. kenaf and hemp) (Ardente et al., 2008; Casas-Ledón et al., 2020). It was assumed that 1 g of dry *Typha* biomass contains 0.412 g of carbon, based on an average carbon content in *Typha* of 41.2% (Grosshans et al., 2013), and this results in 2.6 t carbon stored in harvested yield of 6.3 t DM *Typha* fibres (Grosshans et al., 2013).

2.3.3. Effects of key parameter sensitivities in LCA

- *Typha processing and panel manufacturing*: Renewable energy (solar or wind energy) or residual heat from industrial processes, assumed to have zero carbon impact resulting from residual energy or minimum impact for renewable energy (CE Delft, 2020), could be used for processing (replacing burning light fuel for drying) and/or panel manufacturing (replacing natural gas). Impacts for the production of the polyethylene packaging film and additional ingredients were assumed to remain the same.
- *Reference insulation material*: We assumed two different compositions of the reference material which *Typha* substitutes, namely one with 100% glass wool and one with 100% stone wool. The production of glass wool involved an associated GWP to produce the assumed amount of insulation of 12.7 t CO_2-eq for glass wool, based on an average impact of 5.1 t CO_2-eq per panel (Ecoinvent, 2013) and 2400 panels to be replaced. For stone wool, this was 3.8 t CO_2·eq, based on an average impact of 1.5 t CO_2-eq per panel (Ecoinvent, 2013) and 2595 panels to be replaced.
- *Yield*: Long-term data of *T. latifolia* biomass yields are scarce and highly dependent on the crop stand age (Geurts et al., 2020). The sensitivity of the yield parameter within the LCA of *Typha* was assessed with varying yields of −50% and +50% of the original value (8.5 t DM ha^{-1} yr^{-1}), resulting in varying yields of 4.3 t DM ha^{-1} yr^{-1} and 12.3 t DM ha^{-1} yr^{-1}. Changes in yield affected the fuel and energy needed for cultivation as well as for transport, panel processing and manufacturing, and the amount of potentially avoided conventional insulation; however, the effect of biogenic storage was excluded. A linear relationship was assumed for the *Typha* cultivation impacts (harvesting, rhizomes planting), with the exception of fertilizer input, as *Typha* can absorb nutrients from the groundwater after soil

infiltration (Geurts et al., 2020), thus reducing the need of an increased amount of artificial fertilizer. The impact for transport, drying and panel manufacturing were adjusted to the related yield.

2.3.4. Key parameters economic analysis

Biomass yield and price per tonne, as well as discount rate, strongly influence the profitability of paludiculture (Wichmann, 2017), and cultivation costs are likely to be reduced if the practice is commercialised. Therefore, fluctuations in these key parameters are expected to influence the NPV. The sensitivity of these parameters was assessed, with a variation of −50% and +50% of the original values, which were considered realistic cost reductions and were based on the values mentioned in previous research on average yields (Geurts et al., 2020) and prices per tonne (Van de Riet et al., 2014).

3. Results

3.1. Comparative analysis of the system change to Typha insulation panel production from paludiculture

In the RS, the total GWP amounts to 52.1 t CO_2-eq ha^{-1} (Fig. 3), of which 62.8% originates from draining peat soils (32.7 t CO_2-eq) and 37.2% from dairy production (19.4 t CO_2-eq). In the AS, the total GWP amounts to 35.6 t CO_2-eq (Fig. 3), of which 31.1% originates from draining peat soils (11.1 t CO_2-eq), −1.9% from carbon sequestration in peatlands (−0.7 t CO_2-eq), 27.3% from Typha cultivation and panel production (9.7 t CO_2-eq) and 66.9% from the replacing milk produced in West Europe (23.8 t CO_2-eq); 8.4 t CO_2-eq (−23.4%) was avoided by substituting mineral wool insulation.

A system change from using drained peatland (−60 cm drainage depth) for dairy farming towards Typha insulation panel production can reduce the GWP by 31.6% (16.4 t CO_2-eq ha^{-1}). This reduction is mainly the result of the decrease in emissions from land-use management, which were 21.6 t CO_2-eq (−66.0%) lower in the paludiculture system. However, this reduction is partly offset by the higher impact of the replacing milk from West Europe and the additional impact of Typha cultivation and panel production.

Within the Typha panel lifecycle (with a total GWP of 9.7 t CO_2-eq ha^{-1}), processing and panel manufacturing accounts for 72% of the GWP, followed by cultivation of Typha with 27% of the GWP (see Fig. A2). In the cultivation of Typha, fertilizer application and production represent the highest GWP, which is approximately 24% of the total GWP from the Typha panel lifecycle. Within the processing and panel manufacturing, the highest GWP is associated with fossil energy use for panel manufacturing (40% of GWP of the total GWP from the Typha panel lifecycle) and drying (26% of the total GWP from the Typha panel lifecycle). The fossil-based production of the additional

ingredients in Typha insulation panels is a large contributor in panel manufacturing, with 40% attributed to energy needed for the production of the plates itself and 60% to the production of the other components (see Fig. A1).

3.2. Economic viability of Typha paludiculture

The current revenue from dairy production on Dutch peatlands (~€1350 ha^{-1} yr^{-1}) is higher than the predicted revenue from Typha paludiculture, whose net yearly revenue per ha is slightly negative (Table 2). This is due to relatively low benefits from Typha biomass and insulation compared to high cultivation costs due to labour-intensive harvesting. Moreover, the current NPV for Typha paludiculture is negative (Fig. 4), which indicates that investing in Typha paludiculture is not viable. The negative NPV is mainly caused by high investment costs related to landscape design with paludiculture, and again high cultivation costs compared to low benefits.

3.3. Scenario and sensitivity analysis

3.3.1. Rewetting with other reference drainage depths

The total GWP is 65.6 t CO_2-eq ha^{-1} and 35.0 t CO_2-eq ha^{-1} for −90 cm drainage depth (RS −90 cm) and −30 cm (RS −30 cm), respectively (Fig. 5).

3.3.2. Biogenic storage

Inclusion of biogenic storage, which is equal to 4.7 t CO_2-eq following PAS2050, results in a decrease of the GWP in the AS to 30.9 t CO_2-eq ha^{-1} (Fig. 6).

3.3.3. Effects of key parameter sensitivities in LCA
- Typha processing and panel manufacturing: Assuming the use of renewable energy or residual heat for processing (i.e. drying biomass), for panel manufacturing, or for both combined reduces the GWP of the Typha lifecycle by 2.6, 1.5 and 4.1 t CO_2-eq, respectively. On a systems scale (GWP ha^{-1} in AS), the GWP is reduced by 12% if both sustainable drying and panel manufacturing is assumed; if only sustainable drying or panel manufacturing is assumed, the GWP is reduced by 7% and 4%, respectively (Fig. 6).
- Reference insulation material: Substituting Typha with 100% stone wool panels results in a GWP of 40.2 t CO_2-eq ha^{-1} in the AS (Fig. 6), thus decreasing the environmental potential of the Typha paludiculture system change. Assuming the Typha panel substitutes a glass wool panel results in a larger reduction, of a total of 31.3 t CO_2-eq ha^{-1} (Fig. 6). The two different assumptions show an increase (stone wool) or decrease (glass wool) of ±12% in the AS.
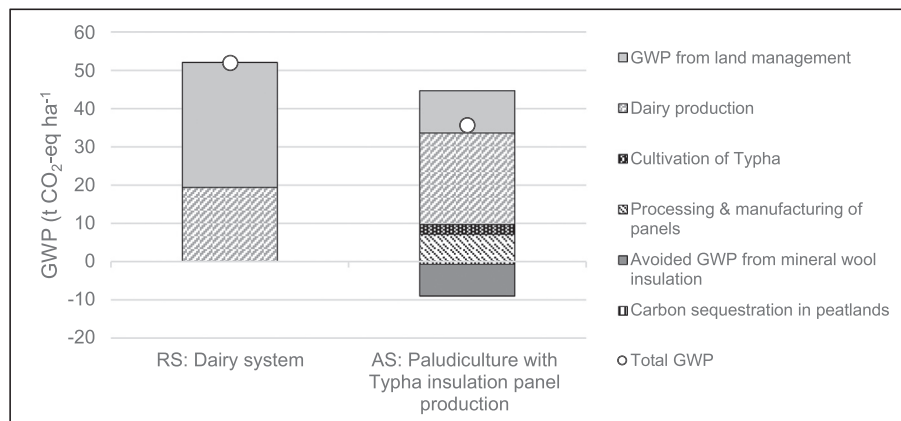

Fig. 3. Total GWP ha^{-1} from different categories in RS (dairy production) and AS (paludiculture with Typha insulation panels).
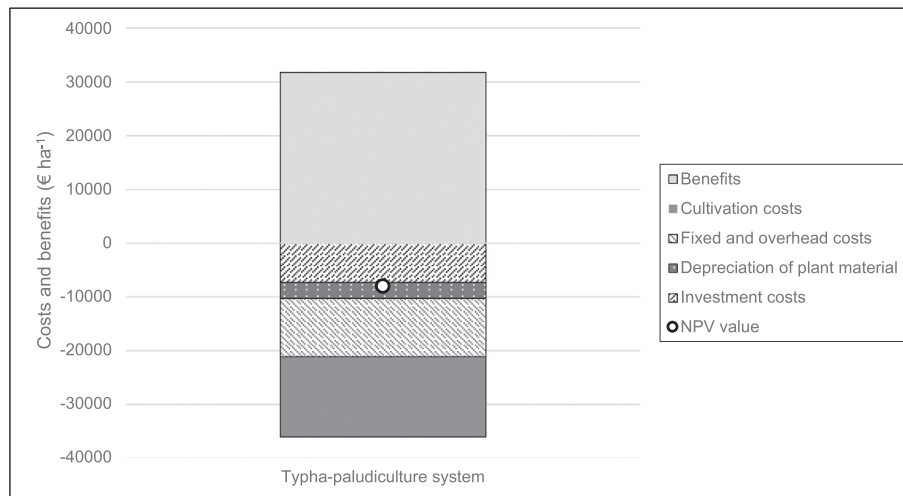
Fig. 4. Individual contributions of cost items and benefits to Net Present Value (NPV) (project lifetime of 30 years, discount rate of 6%).

- *Yield*: The analysis shows that the environmental potential from the system change is slightly sensitive to changes in yield (Fig. 7, blue dotted line), with a decreasing impact at increasing yields. Total GWP per ha^{-1} varies from 34.6 to 35.8 t CO_2-eq in the AS.

3.3.4. Key parameters economic analysis

The NPV was found to be very sensitive to changes in the assumed benefits and cultivation costs from *Typha*. This is due to the relatively intensive management and high related costs, rather than to changes in the discount rate (Fig. 7). The sensitivity analysis shows that an increase in the price t^{-1} DM significantly influences the NPV of *Typha*.

4. Discussion

4.1. GWP reduction potential of Typha paludiculture

This research confirms that rewetting Dutch peatlands currently used for dairy farming (with a reference water level of −60 cm drainage depth) with a view to *Typha* insulation panel production from paludiculture is an effective GHG mitigation option, as indicated by Martens et al. (2021) & Vroom et al. (2018). However, the mitigation potential of Dutch paludiculture is most sensitive to the initial drainage depth and less so to assumptions related to *Typha* cultivation and the replacement of dairy production.

Although different methodologies and uncertainties were used in calculating carbon fluxes from rewetted peatlands (Motelica-Wagenaar and Beemster, 2020) and the associated GHGe were highly dependent on local soil properties (Berglund and Berglund, 2011), previous literature confirms the relationship between an increasing water level and decreasing GHGe from peatlands (Wilson et al., 2016; Martens et al., 2021). Thus, the GWP reduction potential of rewetting peatlands is the highest in areas which have a deep drainage. This implies that the potential of *Typha* insulation panel production from paludiculture also depends on current drainage depths (Fig. 5), which are highly regional. Based on the GHGe assumptions used in the current study, Dutch *Typha* insulation panel production on peatlands should therefore be mainly applied in regions with deep drainage, while switching in areas with a current shallow drainage – a mean groundwater table of about −30 cm or higher – results in almost no net GWP reduction (Fig. 5). Tiemeyer et al. (2020) reported that mean CO_2 emissions versus mean water table level of drained peatlands can be described by a Gompertz function, with a (near) maximum CO_2 emission of about 37 t ha^{-1} reached at a water table of around −40 cm, although they did not observe a relationship between the mean water table and N_2O emissions. Following these assumptions, the effects of water levels will be less important for GHGe if the reference system is less deeply drained than the assumption currently used.

In this study, we investigated a paludiculture system with an assumed water level of +20 cm above soil surface to facilitate *Typha* cultivation. From a climate change mitigation perspective, however, the assumed water level is not optimal. Water levels of around 10–20 cm
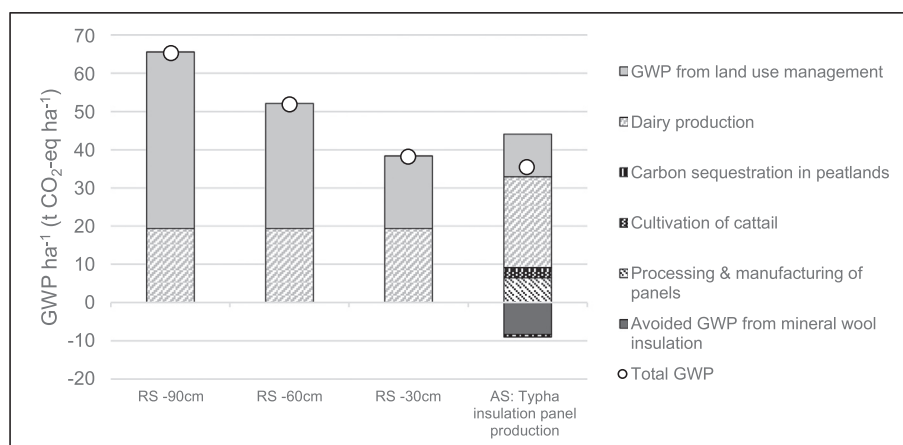

Fig. 5. Total GWP ha^{-1} with two different reference water levels (−90 cm and −30 cm drainage depth) and the used reference water level in this study (−60 cm drainage depth).
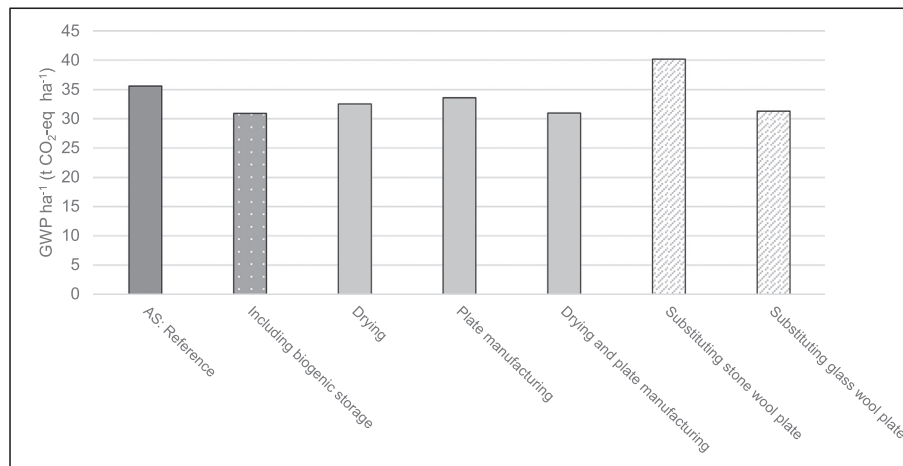
Fig. 6. Climate impact of Typha panels (GWP ha^{-1}) in AS for the scenarios of including biogenic storage, of assuming renewable energy for drying or panel manufacturing or combined, and of substituting either a 100% stone wool or glass wool panel.

below soil surface generally show lower GHGe (8.9 t CO_2-eq ha^{-1} yr^{-1}; Motelica-Wagenaar and Beemster, 2020; Tiemeyer et al., 2020). They are assumed to be optimal in terms of GHGe from peatlands, compared to water levels above the soil surface as these result in CH_4 emissions and thus they decrease the GWP reduction potential from the system change. This is confirmed by Bansal et al. (2019), who concluded in a review study that in many studies higher CH_4 emissions were observed for increased *Typha* biomass production. Even if the assumed carbon sequestration in the *Typha* rhizomes and roots (-0.67 t CO_2-eq) is included, this by far does not compensate these higher emissions. Moreover, the quantity of this carbon sequestration is an uncertain assumption, as no field studies on carbon sequestration in soils and roots under *Typha* cultivation conditions have been conducted, despite several studies in natural conditions (e.g. Bonneville et al., 2008).

While putting an end to draining peat soils leads to clear GHG benefits, the benefits of *Typha* insulation panel production are less pronounced. The assumed impact of cultivating and processing *Typha* into insulation material results in an insulation material with a higher environmental impact (9.73 t CO_2-eq ha^{-1}) than the conventional insulation material it is assumed to replace (8.36 t CO_2-eq ha^{-1}). The main uncertainties of these assumptions – made to enable the comparison of the GWP of these insulation materials – are related to the production system, as it is still in the early stages of development. The limited availability of robust empirical data of the manufacturing processes of *Typha* insulation results in uncertainties in the LCA methodology, which have been assessed in the sensitivity analysis.

Related to uncertainties in the yields, the GWP of the AS as a whole decreases with increasing yields (Fig. 7, blue dotted line). This decrease can be mainly attributed to the lower potentially avoided GHGe from mineral insulation at lower yields. However, we excluded the impact of biogenic storage from this analysis, and this is likely to lower the GWP per ha^{-1} with increasing yields as more carbon is stored. With increased yields, the high impact of cultivating *Typha* offsets the increased potentially avoided environmental burdens from the substituted insulation. The assumed GWP of cultivating and processing 1 kg of *Typha* are relatively high, as cultivation impacts were based on experimental data and the assumed energy use was from fossil energy. Assuming renewable energy use for both drying and manufacturing reduces the impact of *Typha* insulation with 4.1 t CO_2-eq, resulting in a GWP of ~5.6 t CO_2-eq ha^{-1}, and this results in *Typha* insulation being more environmentally friendly than the insulation it is assumed to replace. However, at manufacturing there are the additives (binder and other ingredients) that contribute to the majority of the GWP (~60%); consequently, exploring alternative additives and binders could reduce environmental burdens. Starch or gypsum are promising alternative binders from an environmental point of view, but research is needed into their influence on various properties of bio-composites (Bumanis et al., 2020). Magnesite has also been mentioned as a mineral-based binding alternative, and is already being used in *Typhaboard*, a building material from 50% chopped *Typha* leaves and 50% caustic magnesite, which is commercialised on a small scale (Krus et al., 2015). Moreover, it has been claimed that the material is produced in a simple process and
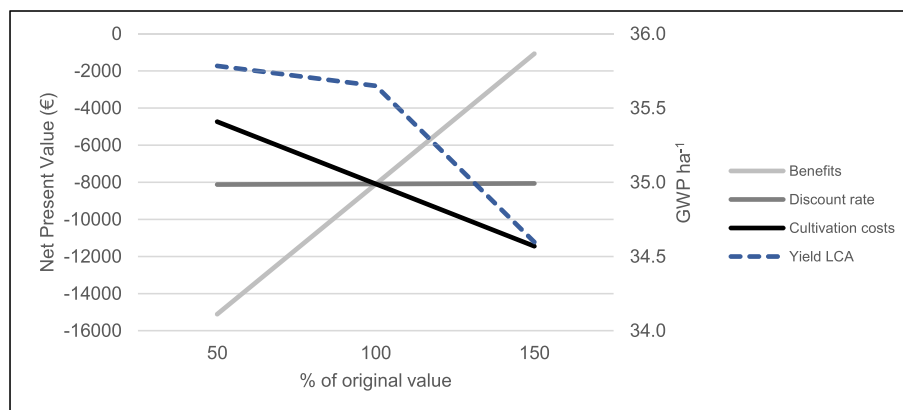

Fig. 7. Sensitivity analysis for the key parameters of the economic analysis on the primary axis (benefits, discount rate, cultivation costs) and LCA on secondary axis (biomass yield) with values varying between -50% and $+50\%$ of the original value.

under low pressure (Georgiev et al., 2019), resulting in low energy consumption during production (Krus et al., 2015). However, availability of data on these processes as well as on the environmental burdens of the production of the caustic magnesite used was too limited to integrate this into the current study.

The long-term carbon storage in *Typha* is one of the major environmental benefits of biobased insulation, and it is not taken into account in a standard LCA. When considering this stored carbon (using the PAS2050-methodology), the impact of the *Typha* insulation is reduced to a GWP of ~5 t CO_2-eq ha^{-1}, making it more environmentally friendly than the reference material. Including biogenic storage reduces the environmental impact of the AS by 4.7 t $CO_2 \cdot$eq to 30.9 t CO_2-eq ha^{-1}, which is 41% lower than the RS. However, it should be taken into account that this is an approximation of the benefits of carbon storage, as the assumed fate of the *Typha* insulation is highly uncertain (García and Freire, 2014). Furthermore, there is no clear consensus yet on how carbon storage should be accounted for in LCA (Brandão et al., 2013). In the chosen 100-year perspective, carbon storage in the insulation panels may not be highly relevant, as the lifespan of the panels was assumed to be shorter than 100 years. However, this may change if a shorter time perspective is chosen. In follow-up research, it would be of interest to include an analysis on a shorter time horizon, potentially leading to a higher total GWP due to higher GHGe as a result of higher relative potencies of GHGe on a shorter time horizon. Not only the assumed impact of the *Typha* insulation but also that of the assumed replaced insulation material affects the environmental potential of the system change. Substituting 100% stone wool leads to less GHGe being avoided (increase of ~12% in the AS, Fig. 6), while substituting glass wool leads to more GHGe being avoided (reduction of ~12% in GWP in AS, Fig. 6); this results in an increasing environmental benefit of the system as impacts of *Typha* cultivation and manufacturing are fully compensated in the latter case.

Lastly, we assumed that Dutch dairy production must be replaced, which is shown to be an environmental burden in this analysis because the GHG emissions per kg FPCM in the Netherlands are relatively low compared to other countries. However, it should be taken into account that the carbon footprint of milk production is highly country- and/or site-specific (Yan et al., 2011; Gerber et al., 2011), and assuming averages may well be a simplification of reality. In this study, we assumed an average Dutch dairy farm and its related carbon footprint. Ideally, data on the footprint of dairy farms on peatlands should be used; unfortunately, such specific data were not available.

We assumed that sufficient land is available in West Europe for the replacement of the dairy production, but recent developments show a rise in competition for farmland within Europe, mainly due to urbanisation (Van der Zanden et al., 2017). At the same time, there is an increasing trend of agricultural abandonment where control over land is given up, and agricultural abandonment is one of the dominant land-use change processes in Europe (Van der Zanden et al., 2017). This land is expected to be less productive, and as a result, the related animal productivity is expected to be lower, which results in a higher GWP per kg FPCM (Gerber et al., 2011). Consequently, the assumed higher average impact of the replaced milk production might be an underestimation. Nowadays, Dutch dairy production on peat soils still has a favourable economic outlook; however, this might change as a result of the pressure to decrease GHGe from food production and land-use emissions. With the current pressure on agriculture to efficiently produce food, reinforced by the potential of rewetting peatlands and the overconsumption of proteins in the Netherlands (Tijhuis et al., 2011), it is debatable whether and for how long this favourable outlook will hold.

Overall, implementing the cultivation of *Typha* on Dutch peatlands for insulation may substantially contribute to achieving the Dutch goal of reducing emissions from peatlands by 1 Mt CO_2 by 2030 (Klimaatakkoord, 2019). With roughly 235,000 ha of Dutch peatland (Lof et al., 2017) and approximately 19% of this peatland within the

drainage depth category of −60–90 cm (Van den Born et al., 2016), rewetting and cultivating *Typha* on 10% of these peatlands could result in a GHG reduction of 0.07 Mt yr^{-1}. In fact, the reduction could be increased to 0.15 Mt yr^{-1} if *Typha* is cultivated on 10% of the peatlands with a deep drainage depth (−90 cm or more; 21% of all Dutch rural peatlands); this clearly shows the high mitigation potential of rewetting peatlands.

Focusing on *Typha* on Dutch peatlands could provide synergies with other Dutch goals, such as the goal of a climate neutral agriculture and land use by 2050 (Klimaatakkoord, 2019); in addition, it has the advantage of contributing to the Dutch circular (building) economy. Furthermore, paludiculture benefits biodiversity as it allows for a more diverse and abundant mire biodiversity than found in the drained state (Tanneberger et al., 2020). However, from a climate change perspective, it may be more interesting to focus on peatland restoration and *Sphagnum* (Van de Riet et al., 2014) as climate-neutral options, as may focusing on other paludicrops which grow at water levels which are optimal from a GHGe perspective, such as reed (−20 to +20 cm) (Geurts et al., 2019). Growing these other paludicrops may also be interesting as other potential business cases, for example with reed (Wichmann, 2017) used for biogas production, direct combustion or thatching, or *Sphagnum* replacing peat in horticulture (Wichmann et al., 2020).

4.2. Factors influencing the economic viability of paludiculture

The outcomes of this economic analysis should be interpreted with care, as *Typha* is not commercially grown and estimates are provisional, which results in uncertainty regarding management data and benefits. Different assumptions in costs, benefits and yield may thus explain why our outcomes could well be different from other literature. The reference business case (dairy production) is not without uncertainties, as differences between individual companies are large in terms of operational management, firm size and financial situation; as a result, outcomes are not directly applicable to each existing dairy farm (Vogelzang and Blokland, 2011). The assumed revenue from Dutch dairy production may have been slightly underestimated, with a small increase in the revenue from milk production in recent years (Agro and Food portal, 2020). We excluded the drainage costs needed to enable the dairy production, but it is expected that if drainage continues along the same lines, the levies of the Dutch water boards will rise further because of peat subsidence (Van den Born et al., 2016). For the Dutch province of Friesland, it was estimated that water management costs will increase by 30% compared to 2010; for an area of 50,000 ha of peatland, annual additional costs were estimated at €3.5 million in 2050 (Van de Riet et al., 2014). Potentially, these costs could be avoided if peatlands are rewetted, although previous research has concluded that these costs are expected not to be decisive for future decisions about the use of peatlands (Van den Born et al., 2016). Lastly, although we assumed that the main driver for the adoption of paludiculture is economic performance, other factors may influence land use, such as farmers' personal preferences, long-term agreements with producers of processing chains, previous investments, specific land use due to policy measures, subsidies, potential market value changes of agricultural peatlands (Van der Hilst et al., 2010), but also the cultural value of the present landscape and the specific biodiversity on the grasslands (Tanneberger et al., 2020). Related to the economic aspect, the current market values of Dutch agricultural peatland may be a limiting factor, as these are the highest in Europe, with prices of nearly €60,000 per ha^{-1} of agricultural land (Silvis and Voskuilen, 2018). Nevertheless, these values could decrease with large-scale GHG mitigation measures involving increased groundwater levels. The high social costs of for example water and infrastructure management, which are expected to rise to billions of euros in 2050 (Van den Born et al., 2016), once more trigger the question whether and for how long the favourable outlook of Dutch dairy production will hold from an economic perspective.

This situation may well lead to a more favourable outcome for alternatives such as paludiculture.

Our research indicates that at present *Typha* paludiculture is not economically competitive with current land use. Our assumed yearly costs of €1100 are within the range mentioned in Westerhof (2018), who estimates €250–1500 ha^{-1} yr^{-1} for *Typha* cultivation, but they are less than values mentioned by Van Duursen and Nieuwenhuijs (2016), who report an average of €2250 ha^{-1} yr^{-1} for the exploitation of *Typha*, including a land rent price of €360 ha^{-1}. Moreover, Van Duursen and Nieuwenhuijs (2016) state that revenues of *Typha* vary from €-750 up to €750 ha^{-1}, depending on *Typha* biomass prices. However, they assumed a higher average biomass yield than in the present study (15 t DM ha^{-1}), which results in higher potential revenues of €1500–3000 ha^{-1}, depending on prices per t DM (€100 or €150 t^{-1} DW). Van de Riet et al. (2014) even assume prices of €300–500 t^{-1} DM, yielding a revenue of €4800 ha^{-1} yr^{-1}. However, revenues are highly dependent on assumptions of yields and prices; with a lack of empirical data, it is difficult to make clear-cut statements. Moreover, it may be concluded that investing in *Typha* is not economically viable, mainly due to high investment and cultivation costs, combined with low benefits, although the cultivation costs and the benefits are highly uncertain, and assumptions in these parameters greatly affect the NPV of *Typha* (Fig. 7). It is well-established that the early stages of science-based innovation are characterised by uncertainty in both technology and market, often leading to high costs with no revenues in the early stages of an innovation; this phase is known as the *Valley of Death* for innovation (Ellwood et al., 2020) and can also be observed in the business case of *Typha* insulation. Upscaling of the cultivation and maturing of the production process are expected to lead to decreased costs as a result of economy-of-scale effects, with for example improvements in wetland-adapted machines, which are currently often immature prototypes (Wichmann, 2017).

Concerning the benefits, with the assumed yields a commodity price of ~€240 t^{-1} DM *Typha* is needed for the NPV to reach break-even point. Given the current data, this is not a profitable option as this price is relatively high. Increasing revenues, in combination with decreasing cultivation costs, can significantly improve economic viability (Fig. 7). A combined production of *Typha* products, for example as building material, feed or shredded *Typha* for bioenergy (Bestman et al., 2019; Pijlman et al., 2019), may be beneficial for the business case. This study assumed that 1 ha is completely assigned to growing *Typha* for bio-insulation purposes, which may well be optimal from a climate perspective but can result in a limited added value and economic business case. In addition, carbon credits may be an option to compensate for GHGe elsewhere to increase revenues. Carbon credits are a well-known tool for accounting for environmental costs, based on the principle of compensating or 'offsetting' GHGe by paying someone else to absorb or avoid the release of GHGe elsewhere (Kollmuss et al., 2008). With large amounts of carbon stored in peatlands, peatland rewetting may constitute a considerable source of carbon credits in the future (Günther et al., 2018). Thus, introducing carbon credits is beneficial for the economic viability of rewetting projects; however, carbon prices are very uncertain (Van den Born et al., 2016). If carbon credits are assigned to the GWP reduction resulting from *Typha* insulation panel production on peatlands (16.4 t $CO_2 \cdot$eq ha^{-1} yr^{-1}), a yearly carbon credit price of ~€85 t^{-1} avoided GHGe will be needed to compete with dairy production on a yearly basis. A carbon commodity price of ~€38 t^{-1} avoided GHGe is needed for *Typha*-focused paludiculture to become economically viable for a project period of 30 years. These prices are in line with, for example, prices within the current EU Emission Trading Scheme, whose prices are currently just above €30 per t^{-1} CO_2-eq (Ember, 2020). Moreover, the price of MoorFutures, which is the first worldwide carbon certificate sold to offset unavoidable emissions by corporations or individuals especially for peatland rewetting, is €35–67 per t^{-1} avoided GHGe (Günther et al., 2018), which is of the same order of magnitude as the necessary carbon commodity price that we calculated. We

based the carbon credit price on the potentially avoided emissions from rewetting peatlands and growing *Typha*; the inclusion of carbon credits for negative emissions from for example carbon storage in *Typha* insulation could further enhance the business case. The necessary yearly carbon commodity prices are relatively high; however, if a net zero emission society is aimed for, and thus compensation for human-induced GHGe is sought for elsewhere, it is expected that this will have to be done through negative emissions, which are likely to be more expensive. Land-use emissions resulting from the loss of soil organic carbon are difficult to reduce or compensate, and options with carbon storage, perhaps combined with bio-energy (BECCS) are needed to achieve a zero emissions society.

Lastly, if the Dutch authorities intend to reduce GHGe by rewetting peatlands, it will probably be necessary to compensate Dutch peatland farmers for the loss of the value of their land. Policy instruments may vary, from phasing out fossil-based insulation materials to providing a subsidy for bio-insulation material, in combination with the recognition of *Typha* for agricultural payments in the EU's Common Agricultural Policy, which will lead to extra revenue (Wichmann, 2017). Compensating Dutch farmers may contribute to the acceptance of paludiculture, although more attention for stakeholder acceptance is needed. As the Dutch farmers are raised with the philosophy of using agricultural land to its maximum value, switching towards paludiculture requires a paradigm shift, with a way of farming that is more suited to extensive farming.

5. Conclusion

We estimated that *Typha* cultivation and insulation panel production instead of dairy production on deep drained peat soil results in a significant GWP reduction potential, mainly as a result of a large decrease in emissions from peat decomposition at a higher groundwater level. Putting an end to deep draining peat soils shows clear GHG benefits, but the benefits of *Typha* insulation panel production are less pronounced. The impact of cultivating and processing *Typha* is not compensated by the avoided impact of conventional insulation material. However, the production process of *Typha* is in the early stages of development and different assumptions may well lead to different conclusions. Moreover, the reduction potential is highly sensitive to the choice of the water level in the reference situation, with a substantial increase in the reduction potential if paludiculture is adopted in reference areas with currently deep drainage depths (RS −90 cm); however, there is almost no environmental benefit when areas with shallow drainage and dairy production (RS −30 cm) are chosen as reference. Choices in the substituted insulation material highly affect potentially avoided GHGe; they may not only lead to other conclusions related to the environmental profile of *Typha* insulation but could also shed a different light on the total potential reduction of the system change.

In the competitiveness analysis of *Typha* production with current land use, revenue calculations show that a system change towards *Typha* paludiculture cannot compete with Dutch dairy farming under the present conditions and with the present commodity prices. In addition, the lower income estimation is combined with a negative NPV, which implies that *Typha* paludiculture is currently not economically attractive. Current expected market commodity prices for harvested *Typha* are too low for *Typha* insulation panel production to become profitable. Nevertheless, the economic performance is subject to many uncertainties and is highly sensitive to fluctuations in the cultivation costs and benefits, which are likely to improve as the system develops. The business case could be improved by increasing revenues, for example as a result of focusing on a varied product range of *Typha* biomass and/or introducing carbon credits for rewetting peatlands. The necessary carbon commodity prices related to rewetting peatlands for the business case to become viable are relatively high compared to cap-and-trade carbon credit prices (EU ETS), but are in line with expected carbon prices in rewetting projects. In addition, these land-use emissions cannot be directly compensated by renewable energy, in contrast

to fossil emissions in the energy sector. If a net zero emission society is aimed for, and compensation for human-induced GHGe is sought for elsewhere, it is expected that this will have to involve negative emissions which are likely to be more expensive. Additional research is required, for example so that more empirical data can be gathered to improve the predictive economic analyses, as well as more in-depth data about yield levels and related prices and stakeholder acceptance.

Overall, this research has shown promising results related to GHGe reduction and could support a step towards more sustainable agricultural peat land use, while at the same time working towards a biobased economy.

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CRediT authorship contribution statement

Marle de Jong: Conceptualization, Methodology, Writing – original draft. **Ollie van Hal:** Writing – review & editing, Methodology, Validation. **Jeroen Pijlman:** Writing – review & editing, Validation. **Nick van Eekeren:** Writing – review & editing, Supervision. **Martin Junginger:** Writing – review & editing, Supervision.

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.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2021.148161.

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