Author's response

Reviewer (R#1) comments and author responses and changes to ms bg-2020-230

We highly appreciate the very helpful and constructive comments of the anonymous referee, which helped us to further improve the manuscript. We tried to consider all of them.

Reviewer comments are given in italic and with author responses in normal style

Sub-soil irrigation does not lower greenhouse gas emission from drained peat meadows

by Stefan Weideveld et al.

General comments:

The authors investigate the GHG reduction potential of drained peatlands by using sub-soil irrigation. The topic of the paper is of relevance to Biogeosciences and will be of interest to an audience interested in mitigating greenhouse gas emissions from agriculturally used peatland. It is a novel approach, which needs further research. For the evaluation of the effect of sub soil irrigation on GHG emissions, a paired design of a control site and a sub-soil-irrigated site is used. Four different sites were investigated.CO2, CH4 and N2O fluxes were measured with chambers over a two 2 years period. Carbon and greenhouse gas budgets are determined and compared for the paired sites.

> **Response(1):** We thank the reviewer for the positive comments and constructive inputs. This helped us improve the manuscript.

I do not understand the experimental setup: The basic hypotheses of the manuscript that main GHG emissions comes from soil layers deeper than 70 cm is not well explained.

> **Response(2):** In the Netherlands, the aim of the government is to reduce CO_2 emission from peat meadow areas by 1 Mt by 2030, from which halve is expected to be achieved with the SSI technique (PBL, 2018). To come to this reduction, an area of 50.000 ha with SSI drainage pipes are planned and a CO_2 reduction of 50% is expected from this area. This technique has, however, never been validated by measured CO_2 emission data. Expectations are based on pilots with only soil subsidence measurements. In these pilots a relation between lowest GWT and soil subsidence is found, therefore the elevation of summer GWT is expected to contribute most to the reduction of CO_2 emission. So, our hypothesis is based on the state of the SSI technique according to policy in practice. The current state of application and the basic hypothesis are explained in introduction (L71 – 89).

Moreover, no information about soil properties and soil moisture of this relevant depth are given in the manuscript. As these soil data are missing, it is not clear, which amount of soil organic carbon is exposed to oxygen due to the alterated ground-water level. Often the bulk density is low in deeper peat layers.

> **Response(3):** We agree that the table providing soil data was inadequate. We replaced the averaged soil properties with data of a higher resolution per soil layer. More details are provided on the mineral

cover layer, the schalter layer, the degraded peat layer and the less degraded peat layer in Appendix B. (Page 31)

It would be interesting to calculate the additional % of aerated carbon due to the alteration of the groundwater level and to compare it to GHG emissions.

Response(4): It would have been interesting to measure the change in soil moisture though out the and time with fluctuating water tables, in combination with soil oxygen. To see what the true effects are on the aeration of the soil as a result of the SSI. However, we did not measure it during this field experiment.

Moreover, the authors should estimate if the small differences in groundwater level (<20 cm) can lead to a theoretical GHG reduction, which can be measured with this method (and associated uncertainties) and experimental set up.

Response(5): GWL are in summer even elevated for up to 60 cm difference. This is again not our own expectation, but the expectations are based on previous pilot studies which are now commonly accepted in policy. The basis of this expectation is now explained in detail (L71 – 89)

Unfortunately, soil moisture was only measured to soil depth of20cm. (At site A, C, D, soil moisture and temperature is measured only in the mineral soil cover). Moreover, these data are not presented in the paper, although the importance of soil moisture is discussed in the discussion section.

Response(6): Soil moisture data is now included in the table in appendix B (Page 31) to expand the soil properties. This is data from a sampling done during the peak of summer period indicating the effect of SSI throughout the soil profile.

The main conclusion that SSI does not lower GHG emissions cannot be drawn from the presented data as most of time the difference in ground water level between the treatments was relatively small. However, when the differences in ground water level were > 20 cm a reduction of GHG emissions was observed. In my view, the conclusion from this paper would be that a substantial increase in groundwater level is needed to allow large enough effects in the emissions to be measured.

Response(7): The current design of SSI, at a depth of -70 cm and spaced 6 or 5 meters apart, was not capable of raising the water table to a level to have a sufficient effect on the GHG emission. Even with a flexible ditch water level, inflow of water as not able to raise the water table to higher level (L509). The conclusion was intended to state that the current way that the SSI was implemented does not allow for large enough effects on the groundwater table to have a measurable effect on the emission. Optimization of the SSI technique was not part of the main conclusion, but indeed a substantial higher water table is needed (L513).

For two sites, the comparability of control and SSI treatments is not given. This may influence the mineralization processes and thus the results. In particular, Site A: SSI has considerably higher organic matter content (39 vs 27%) as the control, and C/N ratios (29 vs 20) indicate different organic matter quality. Moreover, Site D: Control *site has nearly the double amount of organic matter than SSI (38 vs 61%). This aspect is not discussed in the manuscript and might bias the results.*

Response(8): The differences in organic matter content are largely due to the thickness of the mineral top layer. However, the soil organic carbon stock is of a similar size for both sites. To avoid confusion, the indication of soil organic matter is changed into g/l soil (Table 1, Page 5). And it is now indicated for all the different soil types to give a better sign of the comparability between the treatment and control and the different sites. Appendix B (Page 31)

In particular, the methods used to measure the carbon and greenhouse gas fluxes and management are not described in sufficient detail.

Response(9): The method is expanded upon, and described in further detail (L166).

The annual N2O budget was calculated based on only few measurement campaigns. In my opinion, it is not possible to calculate annual N2O budgets from 6-9 daily values, which were measured within 6 month in 2017. This is most evident at Site B Control: Linear interpolation of the high N2O emission in March probably overestimate the N2O emissions for the whole winter time. Moreover the material and methods section is misleadingly stating that N2O was measured for each measuring camping, but at Site B 38 campaigns were made, whereas I counted only 17 data points in Figure C1.Researchers reading the manuscript without looking at the supplementary data could extract the N2O data for annual budgets. Due to the low temporal resolution of the N2O data, it is not possible to distinguish between background N2O emissions and fertilizer-induced N2O emissions.

Response(10): In 2017 we experienced infrastructural constraints to measure N_2O fluxes more frequently. The extended winter gap is a consequence of mal-functioning of the Picarro 2508 under field conditions with low temperature. We agree that 7 flux days and 90 measurements are too few for year budget estimation. We present an average measured N2O flux in table 4 (Page 21). The methods and results will be adjusted so that it becomes clear that measurements of 2017 are a rough estimation based on average fluxes from 7 flux days (L78). However, we believe that the measured data is still valuable for evaluating the N_2O emissions under influence of SSI. The results show no structural higher or lower N_2O emissions between the control and SSI sites. The measured data fits our expectations and references of these types of systems. Clarification be added to methodology and discussion (L494) to stress the low temporal resolution of our measurements, and daily measured data will be presented. The moments between frost and thaw was measured for Farm B and C in the beginning March 2018. However due to technical difficulties with low temperatures and the gas measure equipment these moments were still sparse.

As no daily data are presented for the CH4 budget, the data coverage and thus quality of annual budget cannot be evaluated.

Response(11): Daily CH4 data is added to the appendix D (Page 35) of the manuscript to improve the data evaluation.

The description of management is very short and important information about cutting days, fertilization events, and amount of applied fertilizer are missing, which makes it difficult to understand N2O and NEE data.

Response(12): Cutting days and fertilization events are added to Figure 7, Appendix B1, C1. Furthermore, fertilizer information is included in the methods. (L229)

E.g. Why are the cutting days not visible in the GPP data? In other studies, the decrease in GPP after management events can be nicely seen (Poyda et al. 2016 or Beetz et al 2013,). In comparison to their data, the GPP stayed rather constant and relatively high (-10 g CO2 m-2 d-1) throughout the year. Accordingly, it is difficult to evaluate the quality of GPP modeling.

Response (13): We re-calculated the GPP for the 2017 and 2018 campaign-wise 1) with data from adjacent campaigns clustered; 2) with inclusion of the cutting events where the model parameters ($R_{eco,Tref}$, GPP_{max}, and α) are reduced based on linear relationships between grass height and model parameters. In this way, better model performance is achieved and the influence from plant biomass is accounted for (L229). The harvest dates are included in figure 7 and in appendix C to visualize these moments. And to give a better estimate for the total emission.

The uncertainty assessment is nicely done for the gap filling method of NEE, but the uncertainty estimates are not integrated in the results and transferred to GPP and Reco. For NEE, Reco, and GPP an uncertainty is indicated, but it is not stated what it is(error of SD or 95% confidence interval...). The uncertainty range is given for NEE as3-16 t CO2 ha-1 yr-1, (L 370), but NEE uncertainty from NEE gap filling is given as 14-25 t CO2 ha-1 yr-1). For N2O and CH4 the uncertainty assessment is missing. Other sources of uncertainty (systematic errors of the use of chamber methods or random errors) are not discussed. Please provide a more thorough uncertainty estimation of all component of the net ecosystem carbon balance and included this values in Table2 and

Response(14): Uncertainty is discussed and quantified in more detail. Specifically, the uncertainty is considered in two aspects, 1) model error interpolated for the year and 2) extrapolation uncertainty which was already calculated as the uncertainty from gap-filling model selection. The two sources are then combined following the law of error propagation (L244).

3.Specific comments:

Experimental set up and management Table 1: please provide more information about soil properties of the mineral soil cover, and underlying peat layers (carbon content, bulk density, C/N and carbon stock) in a higher resolution for the entire aerated soil depth. Please state how many soil samples were taken per depth and where. Please also add the information of the depth location of the schalter.

Response(15): A table with a higher resolution of soil characteristics is added Appendix B. The methodology is updated (L138)

Figure 2: please provide information about the location of the chambers of the control site relative to the main ditch.

Response(16): The distance to the main ditch is added (L128). The distance variated between 25 and 40 meters. The location was chosen to exclude a direct effect of the ditch on the water table in the control sites.

Please provide data of cuttings days, fertilization events and measurement campaigns for CO2, N2O and CH4 as the growth of the grass and thus GPP strongly depend on time of measurement (days after cutting). Information can be added in Figure 7, Appendix B1, C1.

Response(17): We included the harvesting and fertilization events in the figure 7, appendix C1. Fertilization events where added to the figures in appendix D1 and E1. To account for the influence from plant biomass on the CO2 fluxes, linear relationships between grass height and model parameters ($R_{eco,Tref}$, GPP_{max}, and α) were developed (L229)

Please add information about amount and determination of N und C input through slurry application. Please add information about the determination of the yield (dry mass of the grass).

Response(18): This information is added to the methods and results (L164). From every manure application manure samples were taken. Bulk-density was determined, Total nitrogen (TN) and total carbon (TC) was determined in dry slurry material (3 mg) using an elemental CNS analyzer (NA 1500, Carlo Erba; Thermo Fisher Scientific, Franklin, USA) (L160)

According to Table 1 Site A und B were grazed. How was carbon import through cattle manure determined? How was the carbon export through grazing determined? Was the yield only determined within the chambers or for the entire grassland? How was the grass height determined?

Response(19): The management of the whole field was grazing, however our field site was fenced off to prevent the mentioned problems. The yield was determined inside the chamber frames, to close the carbon budget. Grass height was estimated using a straight scale with a plastic disk with a diameter of 30cm to determine the top of the grass(L187). The management description is updated in the manuscript (L157)

Gas fluxes Chambers: Was the location of the frames fixed over the two years? Did the vegetation change within the chambers during the experiment?

Response(20): The frames where fixed trough out the two measurement years (L129). The vegetation though out the years remained dominated by *Lolium perenne.* However in spring there were always other species coming up in the frame. However after the first harvest these species disappeared.

Please add the transparency of the chambers? Was a correction term introduced due to a reduced transparency?

Response(21): We corrected the PAR values outside the chamber since the acrylic glass of the transparent chambers reflected or absorbed at least 8% of the incoming radiation (L184)

Please add information about the used sign convention, positive fluxes= loss of carbon? Please add information about the used equation for the calculation of GHG balances and assumptions (harvest is assumed to be released as CO2?, loss of dissolved organic carbon?

Response(22): The methodology is expanded upon. The atmospheric sign convention was used. All C fluxes into the ecosystem where defined as negative (uptake from the atmosphere into the ecosystem), and all C fluxes from the ecosystem to the atmosphere are defined as positive. This also holds for non-atmospheric inputs like manure (negative) and outputs like harvests (Positive). Both harvest and manure input are expected to be released as CO_2 again (L233). Dissolved organic carbon was not sampled during the experiment.

L242ff: what is the accuracy of precipitation data derived from satellite images?

Response(23): The accuracy is nine square kilometer. Giving a precipitation value every 3 hours (L154).

L243: June 2017 seemed to have received more than the average precipitation June is included in the drought period?

Response(24): The average precipitation in June was higher than average, however this is due two days with heavy rain at the end of the month, ending the drought (L279)

Figure 4: As there were 3 groundwater measurements per site, it is not clear which groundwater table is presented, average of all 3?, what is the SD of the three wells? How is the variability of groundwater level of the control site? Please explain DRN

Response(25): The presented data is data from the logger in the field site (L289), the other groundwater measurements are manual dip wells, recorded each measurement campaign. The data shown in figure 4 is a good depiction of the situation in the control site. Only close to the ditch (Less than 10 meters) there is a higher groundwater table in the summer and lower in the winter.

L276: I do not understand the sentence "There is variation.." please clarify.

Response(26): There is difference (variation) between the SSI and control site on the different days in regards to temperature and grass height.

L327-330: What is meant by uncertainty of 3-16 t CO2 ha-1 yr-1. What is represented by 1.6 t CO2 ha-1 yr-1 I for NEE in 2017?L326-332: What is the difference between annual NEE of 47 t CO2 eq. ha-1 yr-1(L327) and emissions of 62 t CO2 eq. ha-1 yr-1 (L313)

Response(27) This part and other parts are rewritten with updated values to specify the values and uncertainties of R_{eco}, GPP and NEE.

L 334-338: Please provide daily CH4 data.

Response(28): Daily data are added to the manuscript in Appendix D

Table 2 and Table 3: Please add uncertainty estimates for all components of GHG balance.

Response(29): Modeling and gap-filling uncertainties are updated and added to table 3 for R_{eco}, GPP and NEE.

L. 380: Reco was lower when the differences of groundwater level was >20 cm

Response(30): Correct, this is adjusted in the manuscript (L406).

L. 420: N2O emissions are not only driven by fertilization events, but also by soil moisture, which should be differ by the treatment. Thus, the comparison can be biased by missing peak events.

Response(31): See response(10) Soil moisture is an important driver for the N2O fluxes from these drained peatland systems. We assume that with the method used we missed peaks induced by fertilization and rewetting (L452). However the comparison between the treatments effects on the basis of the different measurement campaigns still provides insight into the effect of SSI on N_2O emissions.

L428: please use the same sign convention for all cited references.

Response(32): The references are updated.

Technical comments: L310: Please state was the 4 t are, SD?,...,

Response(33): this has been clarified in the manuscript. (L347)

Please indicate A und B in Figure 6

Response(33): A and B are included in figure 6

Figure 7: please use colors, which can be clearly distinguished

Response(34): Figure 7 is improved to increase the understandability of the figure.

Reviewer (R#2) comments and author responses and changes to ms bg-2020-230

We highly appreciate the very helpful and constructive comments of the anonymous referee, which helped us to further improve the manuscript. We tried to consider all of them.

Reviewer comments are given in italic and with author responses in normal style

Sub-soil irrigation does not lower greenhouse gas emission from drained peat meadows

by Stefan Weideveld et al.

Generally, the manuscript will be of interest for readers of Biogeosciences, and the topic of adequate mitigation strategies for drained organic soils is one of high relevance. While the overall result that there is no difference in GHG emissions of this sub-surface irrigation (SSI)system and the control seems to be robust, there are, in my opinion, still four major issues which need to be solved before the manuscript could be considered for publication in BG:

> **Response (1)** We thank the reviewer for the positive comments and constructive inputs. This will help us improve the manuscript.

• *The authors appear to be surprised that SSI does not result in lowered GHG emissions, but this "surprise" is rather unfounded as the water table is raised only slightly towards a target level of 60 cm below ground, which I would –in line with the IPCC Wetlands Supplement (IPCC, 2014) –still regard as "deeply drained".*

> **Response (2)** We recognize that part of the questions raised are a result of an inadequate framing of the experiment. The introduction is rewritten to improve the framing of current state of the SSI technique. (L71 -L89) In the Netherlands, the aim of the government is to reduce CO_2 emission from peat meadow areas by 1 M t by 2030, from which halve is expected to be achieved with the SSI technique (PBL, 2018). To come to this reduction, an area of 50.000 ha with SSI drainage pipes are planned and a CO_2 reduction of 50% is expected from this area. However, the current design of the SSI technique aims to increase the lowest water table while maintaining the agricultural function as "business as usual". Also, this technique has never been validated by measured CO_2 emission data. Expectations are based on pilots with only soil subsidence measurements. In these pilots a relation between lowest GWT and soil subsidence is found, therefore the elevation of summer GWT is expected to contribute most to the reduction of CO_2 emission. The current set-up tested in our experiment aims to explore the effectiveness of SSI on GHG emission for the first time by measurements, also on a large scale on sites representative for the Frisian peat meadows. Therefore, our hypothesis is based on the state of the SSI technique according to policy in practice (L91). And our set-up was made based on the current policy status rather than the scientific exploration of the optimal use of rewetting to mitigate the emissions.

• *According to Figure C1, there were partially only 7 measurement dates for N2O in 2017 and afterwards a gap of five months. Given the highly episodic nature of N2O fluxes, this is absolutely inadequate for the calculation of annual balances in a strongly fertilized grassland.*

Response (3): In 2017 we experienced infrastructural constraints to measure N_2O fluxes more frequently. The extended winter gap is a consequence of mal-functioning of the Picarro 2508 under field conditions with low temperature. We agree that 7 flux days and 90 measurements are too few for year budget estimation. We present an average measured N2O flux in table 4 (Page 21). The methods and results will be adjusted so that it becomes clear that measurements of 2017 are a rough estimation based on average fluxes from 7 flux days (L78). However, we believe that the measured data is still valuable for evaluating the N_2O emissions under influence of SSI. The results show no structural higher or lower N_2O emissions between the control and SSI sites. The measured data fits our expectations and references of these types of systems. Clarification be added to methodology and discussion (L494) to stress the low temporal resolution of our measurements, and daily measured data will be presented. The moments between frost and thaw was measured for Farm B and C in the beginning March 2018. However due to technical difficulties with low temperatures and the gas measure equipment these moments were still sparse.

• *For the interpolation of GPP, all measurement campaigns have been pooled for 2017 and harvests have not been accounted for when interpolating GPP despite the large influence of above ground biomass on maximum photosynthetic rates.*

Response (4): We appreciate the reviewer's suggestions for improved gap-filling strategies. The data for GPP gap-filling was available for a recalculation of the GPP balance for 2017 for a better estimate of the GPP. Pooling of measurement campaigns is improved based on conditions during the measurements. The amount of biomass and the harvest are key in understanding the GPP flux. We have included the harvest moments in the interpolations of GPP and corrected for the Interpolated Reco.(L229)

Title and assumption that this specific SSI system would lower GHG emissions

The SSI system studied here has a target water level of -60 cm. Given the limited hydraulic conductivity of the peat and the "exit resistance" of the pipes, a water level of -60 cm in the ditches results in even deeper field water levels in summer. This target seems to be based on the assumption that CO2emissions originate from deeper peat (see below). Thus, the authors state that a WT rise of 6-18 cm in summer compared to an even lower level "unexpectedly" (line 22) or "contrary to our expectations" (line 29) does not lower GHG emissions. In my opinion, this is absolutely no surprise, but should be expected as laboratory studies often show highest respiration rates at medium water content and as field studies, on average, showed an asymptotic rather than a linear response of CO2 emissions to water table depth (too dry, no more peat exposed, Tiemeyer et al., 2020).

Response (5): See response(2) for a full response. This is not our own expectation, but the expectations are based on previous pilot studies and now common accepted in policy.

Thus, the title needs to be changed to "Sub-soil irrigation with target water levels of 60 cm does not lower carbon dioxide emissions from drained peat meadows" or something similar, as the experiments do not allow for conclusions on SSI in general. Further, if the authors are really surprised by their results, they will need to convince the reader why. In this context, it also needs to be discussed why such low target water levels have been chosen at all. At least for meadow use as in 2018, such low water levels are technically not needed when adequate machinery (low weight, double tyres, etc.) is used.

Response (6): A change will are made in the title (L1) and conclusion (L525), to clarify that the current design of SSI is the commonly applied compromise between additional drainage and increased infiltration during summer and that this technique may fall short to have a significant effect on the GHG balance.

Peat layers below -70 cm contribute most to GHG emissions

In the introduction, there is no reasoning why this should be the case at all. Many studies have shown that top soils show higher respiration rates than subsoils e.g. due to higher nutrient contents or generally more favourable conditions for microbial activity(e.g. Bader et al., 2018). This is indeed briefly discussed on page 22, but the whole "story"of the manuscript (and probably also the design of the sub-surface irrigation system) builds on this assumption. Thus, either it needs to be substantiated by peer-reviewed (!) literature, or the manuscript needs to be restructured based on more adequate hypotheses.

Response (7): We agree with the reviewer that the manuscript needed a clear distinction between current knowledge in peatland sciences and (current) assumptions of land authorities and Dutch governmental institutions responsible for emissions reporting from peatlands. In our own scientific reporting (van den Berg et al. 2018) we show that the top 20 cm of peat revealed the highest CO_2 production potential. In contrast, the current estimation methods in the Netherlands make no use of CO_2 flux data but rely on soil volume – soil carbon models. It is assumed that soil subsidence is quasi 1:1 related to carbon losses in form of CO_2 without taking volume changes of the peat and changes in the carbon density into account. Based on that 1:1 soil subsidence-soil carbon relationship it has been inferred that soil subsidence is stronger when groundwater resides during summer (L71 -L89)

Frequency of N2O flux measurements

According to FigureC1, there seem to be only 7 measurement dates for N2O in some cases in 2017, then a gap of more than 5 months in winter and finally a further gap of two months at the end of the study period. This contradicts the text that N2O was measured at each campaign, i.e. supposedly bi-weekly in summer and monthly in winter(page 8). If Figure C1 is actually correct, this data may not be used for the calculation of annual balances as effects of fertilisation cannot be captured adequately with such a low temporal resolution. Further, I would suspect that the first fertilisation event took place before April and was thus missed by the campaigns. In any case, fertilisation dates should be indicated in Figure C1.

Even more important, it is well-known that high N2O emissions may occur when temperatures change between frost and thaw(e.g. Koponen and Martikainen, 2004), especially under wetter conditions, and that maximum N2O fluxes of drained peatlands may occur in winter also under temperate climatic conditions (e.g. Flessa et al., 1988). Therefore, the authors should refrain from calculating annual balances from a dataset without winter data. The N2O data could, however, be used to compare treatment effects on the basis of campaigns. In consequence, this means that GHG balances cannot be calculated from the presented data, but only C balances.

Response (8): See response (3) for the elaboration about the N_2O choices that were made in the manuscript and the changes that we made.

GPP modelling

In my opinion, pooling all summer data as done for 2017 is not an adequate gap-filling strategy as GPP max and α strongly depend on vegetation development. This strategy of pooling might be valid for (semi-)natural vegetation, but no for intensively used grasslands with frequent harvests.

Further, it seems that parameters are generally interpolated across harvests which does not capture the effects on GPP, which should be very low after harvests. Harvests are unfortunately not indicated in Figure 7 and Appendix B. I would strongly suggest using an interpolation approach suited for highly managed systems (e.g. Eickenscheidt et al., 2015). If this should not be possible due to inadequate PAR ranges during measuring campaigns, only campaign data (instead of annual balances) may be evaluated.

> **Response (9):** We re-calculated the GPP for the 2017 and 2018 campaign-wise with improved pooling of campaign-wise data and the inclusion of the cutting events where the GPP will be reduced. The GPP estimation has been improved with better parameter fitting. The influence from plant biomass on the CO2 fluxes is now accounted based on linear relationships between grass height and model parameters ($R_{eco,Tref}$, GPP_{max}, and α) (L229). The harvest dates are included in figure 7 and in appendix C to visualize these moments. And to give a better estimate for the total emission.

Further comments

• Line 59: Better cite the most recent Dutch inventory data instead of an "old" (2009) paper.

> **Response(10):** The most recent Dutch inventory is used to have an indication of the national emissions from drained peatlands. (L58)

Table 1:

• Details (e.g. SOC, clay content) on the "mineral top layer" would be helpful.• Soil properties averaged for 0 to 70 cm are not really informative, better provide data on the top soil and on depths where the water level/moisture changes actually occurred.

> **Response(11):** We agree that the current table is inadequate. A table is added to the appendix B to provide additional information on the soil properties for the mineral cover layer, the schalter layer, the degraded peat layer and the less degraded peat layer. The mineral content was determined, however the fractions of the mineral top layer where not determined.

• How comparable are SSI and control when they partially strongly differ in SOM content (location D) or C:N ratio (location A)?

> **Response(12):** The differences in organic matter is largely due to the thickness of the mineral top layer. However, for the soil organic carbon stock is of a similar size for both sites. The soil organic matter in table 1 is indicated as g/l soil.

• Data on hydraulic conductivity or at least on the degree on decomposition are needed to discuss the contrasting hydrologic effects of SSI at the four locations.

> **Response**(13): The hydraulic conductivity was not measured during the experiment. However, the dip wells that we used to measure the water table for the different frames could give an indication for the processes in the field. Only one location had a good horizontal water flow in the peat layer, that was location B. Location A saw a strong effect of the SSI in the water table. However, in some places there was a large difference in the water table at different distances from the pipes.

• Does the "schalter" layer have any effect on the sites' hydrology?

> **Response(14):** Schalter is known to limit vertical water flow, due to its laminated structure. However, there is little documented about the properties and processes (L108). In our case, the locations with "schalter" seem to have lower effects from the SSI

• Line 140: How was the C-export actually determined? For the frames (line 166 ff) or for the whole field as it is implied here? Were the frames fenced off from grazing?

> **Response(15):** The experiment sites were fenced off from grazing. The C-Export was determined inside the frames. However, on harvest days the whole experimental site was cut. (L157)

• If the C-export had to be excluded from statistical analysis, how could the GHG balance containing the C-export be analysed?

> **Response(16):** With regard to the uncertainties on C-export and CH4/N2O budgets, the statistical analysis is now focused only on annual NEE and measured CH4 and N2O fluxes.(L244 – L260)

Line 145: Flux measurements and modelling

While I understand that not all details can be provided to limit the length of the manuscript, lots of information is missing which would allow assessing the quality of the data.

• Were the chambers cooled and vented for pressure equilibration?

> **Response(17):** The chambers where not cooled. The pressure inside was equilibrated when placing the chamber on the frames. (L178)

• Was PAR outside the chamber corrected for the light transmittance of the chamber before interpolating GPP? Which light transmittance was assumed/measured?

> **Response(18):** We corrected the PAR values outside the chamber since the acrylic glass of the transparent chambers reflected or absorbed at least 8% of the incoming radiation. (L184)

• Was there any quality control procedure for flux calculation (linearity, outliers, leakage...)?

> **Response(19)**: Each flux was checked, for the dark measurements a only fluxes with a R-squared of 0.99 or higher where used. For the light measurements the majority of the fluxes that were used had a R-

squared of 0.95. The exception where the fluxes with slopes close to zero or zero (equilibrium between gross primary production – GPP – and R_{ECO}) were not discarded.

• *Was there any minimum temperature difference within one Reco campaign to avoid artefacts due to extrapolation of Eq. 2?*

Response(20): There was no minimum temperature difference set within the Reco campaign. The measurement campaigns where planned to have a range in light variation for the GPP calculations, this resulted in a good temperature range during the day. (L208)

• *Which Reco(nearest?) was subtracted from NEE to yield GPP (line 192)?*

Response(21): The Reco closest in time was used for subtraction of NEE to yield GPP. During the campaigns light and dark measurements were always conducted in the same time frame. (L212)

• *The unit of α is wrong (line 200), it should be mg CO2-C m-2h-1/ µmol m-2s-1*

Response(22): Is adjusted in the material and methods. (L220)

• *Why did you choose to interpolate parameters and not weighted fluxes(line 204)?*

Response(23): We have updated the interpolation method and adjusted in the manuscript, for the recalculations we used weighted fluxes instead of interpolating the parameters since it has been suggested to be better (Hoffmann et al. 2015). Extrapolated values at times between two adjacent models are weighted averages of the estimates from these two models, where the weights are temporal distances of the extrapolated time spots to both of the measurements (L227)

• *Why did you use GPPmax and not GPPopt(Falge et al., 2001)which is less susceptible to extrapolation errors?*

Response (24): According to Falge et al. (2001), GPPmax (or saturation value of GPP) has less explanatory worth for real systems since PAR will not reach infinite, therefore the author switched to GPPopt which provides a reference value at a certain PAR level. GPPmax was used in numerous fluxes modeling works, and we did not find argument from literature stating significantly larger uncertainty from the use of GPPmax. Thereby, if a GPPopt is used, there should have enough data in a specific PAR value (e.g. 2000 µmol m^{-2} s^{-1}). With eddy covariance (where this gap filling paper of Falge et al. is written for) this is not a problem, but with chamber measurement data is limited. The accuracy will therefore be better to fit the light response curve with the GPPmax.

Line 254 ff: "Drainage" and "irrigation" periods

• *From my understanding, "drainage" and "irrigation" periods are not defined correctly. While it remains unclear which WT (0.5, 1.5 or 3.0 m from the pipe) was used for this calculation, it is of course useful to differentiate whether SSI was dryer or wetter than the control when comparing Reco. However, "drainage" and "irrigation" periods can only be identified by using absolute heads, i.e. by comparing the field WT to the ditch water level!*

Response(25): The definition of these periods is clarified (L292). ’Drainage’ periods refer to moments when there is drainage to the ditch and ‘irrigation’ periods refer to moments with water infiltration from the ditch to the field. Here the aim is to differentiate between SSI and control.

• *In this context, it also remains unclear why the SSI system works better at some of the fields in terms of hydrology –is there always enough water in the ditches, how is the hydraulic conductivity or at least the degree of decomposition of the peat, or are there strong differences in WT at 0.5, 1.5 and 3.0 m difference to the pipes which could be used to deduce information on the hydraulic conductivity?*

Response(26): The water table in the ditch was maintained at a level between -60 to -20 cm from the soil surface. It was never a limiting factor for the functioning of SSI. The hydraulic conductivity of the peat soil was not measured during the experiment. However the functioning of the SSI gives an indication of the conductivity. This is closely related to the type of peat present. Farm A, C and D all have Sphagnum peat, with the layer where the pipe is present being moderately decomposed (H5-H7). We suspect there are some macro cracks in the peat soil of farm A, that help infiltration. For location B the peat soil consists of Alder peat. The layer where the pipe is present is moderately decomposed but with a large presence of wood/branches. For this location the SSI seems to work best. With a strong drainage and infiltration effect.

• *Furthermore, it is rather difficult to compare results to other studies. Therefore, please give numbers for the mean and the summer mean water level.*

Response(27): The mean annual average GWT table is added in Table 2 to increase the comparability between the different sites and to other studies.

Table 2 and Table 3

•Should be merged and N-fertilisation should be added. •Uncertainties should be added.

Response(28): The tables where not merged, the readability was in proved, to make a division between the Net ecosystem carbon balance (NECB) and the CH4 and N_2O emissions Modeling and gap-filling uncertainties have been added to R_{eco}, GPP and NEE. (Table 3). N-Fertilization is added to the methodology (L164)

• ***Line 425 ff:*** *There are some comparisons to other studies, but the authors do not try to explain the differences in emissions between their sites.*

Response(29): The differences between the sites are largely because of the soil conditions (Appendix B). The locations with a mineral topsoil seem to respond stronger to drought. Furthermore, there was a difference between the starting conditions of the sites. The sites A and B where grazed before the experiment and site C and D where only mown Table 1. This resulted in a difference in the grass structure, where the grazed grass forms a more dense vegetation structure than the mown grass.

• ***Line 439 :*** *How do you know that moisture conditions were optimal in 2017?*

Response(30): The indication of 'optimal' come from observation of the conditions in the field, for example the grass growth that we observed during the field experiment. This was also determined in contact with the farmers who judged a better year for grass growth. We will reconsider the wording 'optimal', but the point was that we expect that the moisture levels were not a limiting factor during this summer period. (L489)

• ***Line 451:*** *What do you mean with "abnormal data points"?*

Response(31): The "abnormal data points" refer to measurement that did not fit into the temperature dependent function of R_{eco} or light response curve of GPP, due to the extreme drought that limited soil respiration's response to higher temperature, or reduced the photosynthetic rate. Wording has been adjusted to avoid confusion (L501).

• ***Line 486*** *: Effects of land-use intensity and land-use history should be discussed in the context of general emission level (section 4.3) as these aspects do not fit to the section "costs and benefits SSI".*

Response (32): We chose to discuss about land use in the 'cost and benefit' section because of the possibility of SSI to be beneficial for the intensive land use. Due to the increased load boarding capacity of the fields and the drainage in Spring and Autumn, it is possible to extend the periods that the field can be managed. We consider this as a possible benefit from the SSI, however we didn't observe this during the experiment.

• *Besides methodological issues, the manuscript seems to be hastily prepared which results in many inaccuracies especially regarding the references (list might not be exhaustive): • Several references mentioned in the text are not in the list of references (Hoffmann et al., 2015, Tiemeyer et al., 2020)• One reference appears twice (Berglund and Berglund, 2011)•References are incomplete (Couwenberg, 2009, Tanneberger et al., 2017)*

Response(33): The references have been updated.

• *Generally, there is some tendency to cite non-peer reviewed literature (Joosten and Clarke, 2002, Joosten, 2009, Jurasinski et al., 2016, Hendriks et al., 2007b, Hoving et al., 2015, van den Akker et al., 2008, van den Born et al., 2016). In many cases, peer-reviewed papers could easily be found and should be cited instead.*

Response(34) : The current references will be updated. The choice for the non-peer reviewed literature is largely due to the current condition that many of the decisions made for the SSI-experiment by the local government were based on these references. And some indicate the aim of the national and provincial government to implement SSI on a large scale as a way of mitigating problems that occur with management of these Peat meadows.

• *Furthermore, table and figure headings are often very brief or contain abbreviations, sometimes also such which are not used in the manuscript(e.g. location "Ger" in Appendix B).*

Response(35): The table and figure headings are expanded upon the improve the understandability of the figures and the abbreviations will be written full out in the headings.

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Conventional s~~S~~ub-soil irrigation techniques do~~es~~ not lower greenhouse gas emission from drained peat meadows

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Abstract

Current water management in drained peatlands to facilitate agricultural use, leads to soil subsidence and strongly increases greenhouse gas (GHG) emission. High-density, sub-soil irrigation~~/drainage~~ (SSI) systems have been proposed as a potential climate mitigation measure, while maintaining high biomass production. In summer, ~~sub-soil irrigation~~SSI ~~can potentially~~was expected to reduce peat decomposition by preventing groundwater tables to drop below -60 cm.

In 2017-2018, we evaluated the effects of ~~sub-soil irrigation~~SSI on GHG emissions (CO_2, CH_4, N_2O) for four dairy farms on drained peat meadows in the Netherlands. Each farm had a treatment site with perforated pipes at 70 cm below soil level spacing 5-6 m to improve both drainage (winter- spring) and irrigation (summer) of the subsoil, and a control site drained only by ditches (ditch water level -60/-90 cm, 100 m distance between ditches). GHG emissions were measured using closed chambers (0.8 x 0.8 m) every 2-4 weeks for CO_2 and CH_4. C inputs by manure and C export by grass yields were accounted for. Unexpectedly, ~~sub-soil irrigation~~SSI hardly affected ecosystem respiration (R_{eco}) despite raising summer groundwater tables (GWT) by 6-18 cm, and even up to 50 cm during drought. Only when the groundwater table of ~~sub-soil irrigation~~SSI sites was substantially higher than the control value (> 20 cm), R_{eco} was significantly lower ($p<0.01$), indicating a small effect

of irrigation on C turnover. During wet conditions sub-soil pipes lowered water levels by 1-20 cm, without a significant effect on R_{eco}. As a result, R_{eco} differed little (>3%) between ~~sub-soil irrigation~~SSI and control sites on an annual base.

CO_2 fluxes were high at all locations, ~~exceeding~~ ranging from ~~45~~35 – 66 and 20 – 50 t CO_2 ha^{-1} ~~a~~yr^{-1}~~,~~ in 2017 and 2018, respectively, even where peat was covered by clay (25-40 cm). Despite extended drought episodes and lower water levels in 2018, we found lower annual CO_2 fluxes than in 2017 indicating drought stress for microbial respiration. Contrary to our expectation, there was no difference between the ~~yearly greenhouse balance~~annual CO_2 fluxes of the sub-soil irrigated (~~64~~ 40 and 30 t CO_2–eq ha^{-1} yr^{-1} in 2017~~, 53 in~~ and 2018) and control sites (~~61~~ 38 and 34 t CO_2–eq ha^{-1} yr^{-1} in 2017~~, 51 in~~ and 2018). Emissions of N_2O were lower with average emissions were measured (2.9±1.8~~3~~ mg N_2O. m^{-2} d^{-1} for 2017~~±1 t CO_2–eq ha^{-1} yr^{-1}~~) in 2017 than in 2018 (3.6±3.3~~5±2~~ mg N_2O. m^{-2} d^{-1}~~t CO_2–eq ha^{-1} yr^{-1}~~), without treatment effects. The contribution of CH_4 to the total GHG budget was negligible (<0.1%), with lower GWT favoring CH_4 oxidation over its production. ~~Even during the 2018 drought, sub-soil irrigation had only little~~No effect was found on yields~~ (9.7 vs. 9.1 t DM ha^{-1} yr^{-1})~~, suggesting that the increased GWT in summer ~~failed to~~did not increase plant water supply. This is probably because GWT increase only takes place in deeper soil layers (60-120 cm depth), which also indicates that peat oxidation is hardly affected.~~, probably because GWT increase only takes place in deeper soil layers (60-120 cm depth).~~

We conclude that, although our field-scale experimental research revealed substantial differences in summer GWT and timing/intensity of irrigation and drainage, ~~sub-soil irrigation~~SSI fails to lower annual GHG emission and is unsuitable as a climate mitigation strategy. Future research should focus on potential effects of GWT manipulation in the uppermost organic layers (-30 cm and higher) on GHG emissions from drained peatlands.

1 Introduction

Peatlands cover only 3% of the land and freshwater surface of the planet, yet they contain one third of the total carbon (C) stored in soils (Joosten and Clarke, 2002). Natural peatlands capture C by producing more organic material than is decomposed due to waterlogged conditions (Gorham et al., 2012;Lamers et al., 2015). Drainage of peatlands for agricultural purposes leads to aerobic oxidation of organic material resulting in soil subsidence and the concomitant release of CO_2 and N_2O (Regina et al., 2004;Joosten, 2009;Hoogland et al., 2012;Lamers et al., 2015;Leifeld and Menichetti, 2018). Soil subsidence occurs when the groundwater table (GWT) drops through drainage, leading to physical and chemical changes of the peat. This results in consolidation, shrinkage, compaction and increased decomposition (Stephens et al., 1984;Hooijer et al., 2010). Soil subsidence increases the risk of flooding (frequency and duration) in areas where soil surface subsides below river and sea levels (Syvitski et al., 2009). In the Netherlands, 26% of the surface area is currently below sea level, an area currently inhabited by 4 million people (Kabat et al., 2009). This area is expected to increase due to further land subsidence, while sea level is rising at the same time, which is a general issue of coastal peatlands (Erkens et al., 2016). Additionally, peatland subsidence alters hydrology, leading to drainage problems, salt water intrusion and loss of productive land (Dawson et al., 2010;Herbert et al., 2015). This will result in strongly increased societal costs and difficulties in maintaining productive land use (Van den Born et al., 2016;Tiggeloven et al., 2020).

The peatland area used for agriculture is estimated at 10% for the USA and 15% Canada, and varies from less than 5 to more than 80% or Europe (Lamers et al., 2015). In the Netherlands, 85% of the peatland areas are in agricultural use (Tanneberger et al., 2017), leading to CO_2 emissions of 7 Mt CO_2-eq per year, amounting to 4% of total national greenhouse gas (GHG) emissions (Arets et al., 2020). Fundamental changes in the management of peatlands are required if land use, biodiversity and socio-economic values including GHG emission reduction are to be maintained.

Carbon dioxide emissions from peatlands are related to the water table position, which affects oxygen intrusion, moisture content and temperature. There is ample evidence that elevating water levels to 0-20 cm below the land surface results in substantial reduction of CO_2 emissions from (formerly) managed peatlands (Hendriks et al., 2007b;Hiraishi et al., 2014;Jurasinski et al., 2016;Tiemeyer et al., 2020) Increasing water levels close to the surface not only worsens conditions for aerobic CO_2 production and rapid gas exchange but also reduces land-use intensity (fertilization, tillage, planting, grazing). Additionally, high water levels favor vegetation assemblages with a higher carbon sequestration potential (e.g. peat forming plants) compared to common fodder grasses and crops Experimental research using water table manupulations stresses the importance of rewetting the upper 20-30 cm to achieve noteworthy CO_2 emissions reduction (Regina, 2014;Karki et al., 2016) which seems in line with the meta-analysis of field CO_2 emission data by Tiemeyer et al. (2020)

Dutch water- and land-authorities have relied on height measurements of the peat surface rather than CO_2 flux measurements to estimate CO_2 emissions from peatlands (Arets et al., 2020) and the effects of elevated water levels on CO_2 emissions. The soil-carbon-water model used is based on two assumptions. Firstly, multi-year changes ~~is 1:1~~are directly related to carbon losses from peatlands, eventhough elevation changes are small in magenitude in the range of ~~mm~~milimeters per year. Each ~~mm~~milimeter of height loss is translated into carbon emissions equalling 2.23 t CO_2 ha^{-1} ~~a~~yr^{-1} (Kuikman et al., 2005;Van den Akker et al., 2010)). Secondly, the average lowest summer ~~water levels~~GWT is assumed to be a major control of subsidence rates of peat surface elevation and henceforth CO_2 emissions based on the first assumption ~~1~~ above (Arets et al. 2020). As a consequence of both assumptions, Dutch climate mitigation frameworks focus on elevating summer ~~water levels~~GWT in peatlands rather than mean annual ~~water levels~~GWT (Querner et al., 2012;Brouns et al., 2015)~~,~~. Dutch water- and land-authorities expect that increasing the average lowest summer ~~water level~~ GWT by 20 cm would result in an emission reduction equalling 10.5 t CO_2 ha^{-1} ~~a~~yr^{-1} (Van den Akker et al., 2007;Brouns et al., 2015;Van den Born et al., 2016). ~~The second assumption is currently under investigation (Stowa 2020).~~

~~To elevate summer water levels by 10 to 40 cm above the lowest summer level t~~The use of ~~sub-soil irrigation~~ SSI systems ~~(SSI)~~ has been proposed since the early 2000's (van den Akker et al. 2008;~~,~~ (Querner et al., 2012)~~)~~. ~~An overal 50% reduction of carbon emissions from peatlands was assumed after implementing SSI ((Querner et al., 2012;Van den Born et al., 2016).~~ SSI works by installing drainage/irrigation pipes ~~some~~at around 70 cm below the surface and at least 10 cm below the ditch water level~~, which requires ditch water levels high enough~~. ~~During summer w~~Water from the ditch can infiltrate ~~from the ditch through the pipes~~ into the adjacent peat and thereby ~~limiting groundwater table drawdown by 10-20 cm and more during drought~~ limite GWT drawdowns during summer (c.f. (Hoving et al., 2013)~~)~~., ~~However, annual groundwater table in the peatlands remains little affected by sub-soil irrigation as~~while the pipes also full-fill a drainage function when the ~~groundwater table~~GWT is above the ditch water level ~~(which is often 50 cm below the surface)~~. Therefore, the SSI was assumed to have an effect of 50% overall reduction of carbon emissions from peatlands (Querner et al., 2012;Van den Born et al., 2016), based on

~~HypothesesThe hypothesis for the effectiveness of SSI is based on~~ the soil-carbon-water model assumptions that peat layers below -70 cm contribute largely to GHG emissions and that surface elevation differences can be translated ~~1:1~~directly into CO_2 emissions.

The aim of our study was~~,~~ to quantify the effects of SSI on the GWT and the GHG ~~balance~~emissions, in particular the net ecosystem carbon balance (NECB). We questioned 1) to what extent can SSI elevate water levels in two summers that differed in drought duration, 2) whether the SSI can substantially reduce (30-50%) CO_2 emission compared to traditional ditch drainage, and 3) whether nitrous oxide peaks are lowered by SSI. To adress these questions we directly compared GHG emissions from a control grassland (traditional ditch drainage) with a treatment grassland (SSI) on four farms over a periode of 2 years (16 site-years).

~~A higher groundwater table (GWT) creates anaerobic conditions (Berglund and Berglund, 2011), which could lower peat oxidation rates and therefore CO_2 emissions and soil subsidence (Van den Bos and van de Plassche, 2003;Lloyd, 2006b;Wilson et al., 2016b;Van Huissteden et al., 2006).~~

~~To reduce peat oxidation, drastic rewetting (raising the water table to -20 cm below soil surface or higher) would be the ideal option (Hendriks et al., 2007a;Jurasinski et al., 2016). However, current agricultural use would then no longer be feasible. Therefore, there is a incentive to explore options where the effects of peat oxidation are mitigated but land use is not changed. A solution suggested to reduce C loss and land subsidence, which is already in use in the Netherlands, is sub-soil irrigation (SSI). The aim of this management option is to raise the GWT during summer when CO_2 emissions are highest due to high temperatures in concert with low GWT. Raising the GWT in the summer could prove effective to limit aerobic peat oxidation (Hoving et al., 2015;Kechavarzi et al., 2007). Irrigation pipes are placed in the soil at a depth of 70 cm below the soil surface, and 10 cm below ditchwater level. This will have two effects: drainage when there is excess water (mostly in autumn, winter and spring), and irrigation in dry periods (summer). This will force the GWT towards the ditch water level at around -60 cm below the soil surface. The drainage effect results in more of the peat being exposed to oxygen, but since this happens in a colder period, it is expected that the effect of irrigation on CO_2 emissions during summer will be much larger. There are,~~

~~however, few comprehensive studies that report on the effect of sub-soil irrigation on total GHG emissions and C balances for peat soils (Van den Akker et al., 2010;Hendriks et al., 2007b). The hypothesis for the effectiveness of SSI is based on the assumption that peat layers below -70 cm contribute most to GHG emissions. However, this is only based on soil subsidence data, and until now there have not been any studies that directly measured GHG fluxes to test the expected GHG reduction.~~

~~The aim of our study was therefore to quantify the effect of sub-soil irrigation as an alternative drainage technique on the GWT and the GHG balance. The main research questions were whether, compared to traditional drainage, sub-soil irrigation of peat meadows can 1) achieve the intended regulation of GWT within each year and between years (i.e. irrigation during summer and drainage during winter), and 2) lead to a significant reduction of peat oxidation and GHG emission?~~

2 Material and methods

2.1 Study area

The study areas are located in a peat meadow area in the province of Friesland, the Netherlands. The climate is humid Atlantic with an average annual precipitation of 840 mm and an average annual temperature of 10.1°C (KNMI, reference period 1999-2018).

About 62% of the Frisian peatland region is now used as grassland for dairy farming (Hartman et al., 2012). Agricultural land in Friesland is farmed intensively, with high yields, and intensive fertilization (>230 kg N ha^{-1} yr^{-1}). It is characterized by large fields with deep drainage, as one third of the fields are drained to -90 – -120 cm below soil surface. Large parts of these grasslands are covered with a carbon rich clay layer, ranging from 20–40 cm thick. The peat layer below has a thickness of 80–200 cm, which consists of sphagnum peat on top of sedge, reed and alder peat. The top 30 cm of the peat layer is strongly humified (van Post H8-H10) and the peat below 60 – 70 cm deep is only moderately decomposed (van Post H5-H7). On two locations (C and D, see below), there is a 'schalter' peat layer present, highly laminated peat (compacted/ hydrophobic layers of *Sphagnum cuspidatum* remnants) with poor degradability and poor water permeability. The grasslands are dominated by *Lolium Perenne*; other species such as *Holcus lanatus, Elytrigia repens, Ranoculus acris* and *Trivolium repens* are present in a low abundance.

Table 1 Soil and land use characteristics of the research sites in the peat meadows of Friesland, the Netherlands.* Displayed concentrations of the top 70 cm.

Location	Farm type	management	Treatment	Field size ha	mineral top layer thickness m	schalter present	thickness peat layer m	Organic matter ~~%~~ *g/l	Carbon content kg C-m^2-70cm	C:N*
A	Organic	Grazing	SSI	2	0.35	-	1.6	132.9~~38.6~~	53.4	29.2
			Control	0.6	0.40	-	2.0	141.2~~26.8~~	47	19.8
B	Conventional	Grazing	SSI	2.3	-	-	1.4	190.7~~76.8~~	68.1	34.6
			Control	2.3	-	-	1.4	175.9~~80.6~~	74.9	32.8
C	Conventional	Mowing	SSI	1.2	0.30	yes	1.3	141.7~~47.9~~	56.3	23
			Control	1.8	0.30	yes	1.0	133.4~~50.4~~	60.5	23.5
D	Conventional	Mowing	SSI	2.4	0.30	yes	0.9	161.9~~37.5~~	59.6	23.3
			Control	3.5	0.25	yes	0.9	151.5~~60.8~~	63.4	26.9

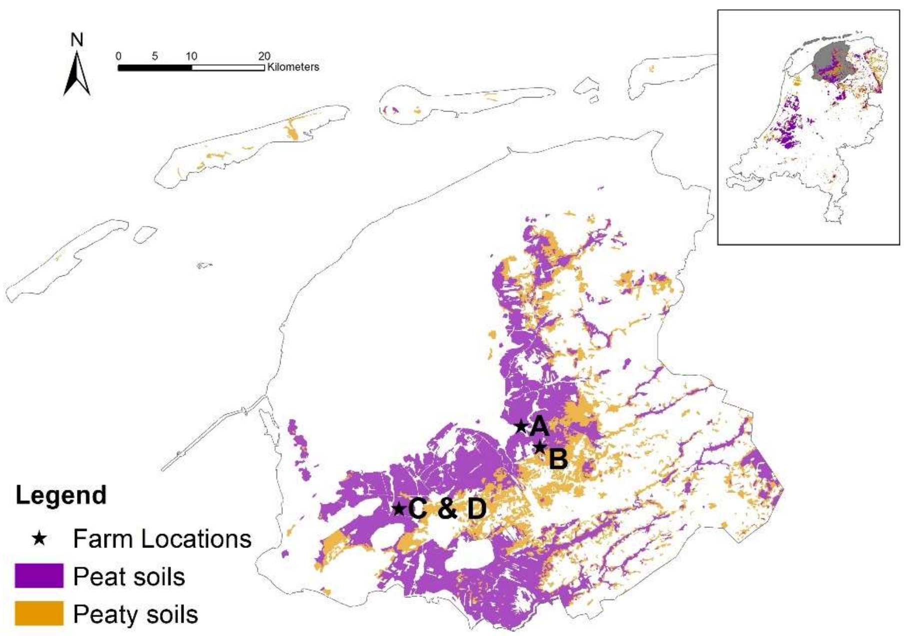

Figure 1 Field locations situated in the province of Friesland, with soil types. Peat soils refer to soils with an organic layer of at least 40 cm within the first 120 cm, while peaty soils are soils with an organic layer of 5-40 cm within the first 80 cm. Insert shows these soil types in the Netherlands, with the location of the field locations in grey.

2.2 Experiment setup

Four sites were set up at dairy farms with land management and soil types representative for Friesland (see Table 1 and Fig. 1). Each location consisted of a treatment site with ~~sub-soil irrigation~~SSI pipes and a control site. The irrigation pipes were installed at a depth of 70 cm below the surface and 6 m (2,000 m drains ha^{-1}) apart from each other, except for the D location where pipes were 5 m apart. The pipes were either directly connected to the ditch (A and C) or connected to a collection tube before connected into the ditch (B and D). The connections with ditches were placed 10 cm below the maintained ditchwater level. The control sites are fields that have traditional drainage, through a system with deep drainage ditches (32 – 42 meter from the main ditch) with convex fields and small shallow ditches.

On the treatment sites, three gas measurement frames in 80x80 cm squares were placed for the duration of the experiment on 0.5 m, 1.5 m and 3 m distance from the chosen irrigation pipe (Fig. 2), representing best the variation in the environmental conditions and vegetation. Dip well tubes were installed to monitor water levels 0.5, 1.5 and 3 m from the pipe, pairing with the locations of gas measurement frames (Fig. 2). The nylon coated tubes were 5 cm wide and perforated filters placed in the peat layer. The tube 1.5 m from the irrigation pipe was equipped with a pressure sensor and a data logger (ElliTrack-D, Leiderdorp instruments, Leiderdorp, Netherlands) that measures and records the GWT every hour. Ten more dip well tubes were further placed at intervals 0.5 and 3 m from the pipes in the field, which were manually sampled every 2 weeks during gas sampling campaigns, to obtain the variation on field scale.

Soil samples were taken using a gouge auger three replicas where taken, from 1.5 meter from the irrigation pipes. To determine moisture content, sediment samples were weighed and subsequently oven-dried at105°C for 24 h. Organic matter content was determined via loss on ignition. Dried sediment samples were incinerated for 4 h at 550°C (Heiri et al., 2001). Total nitrogen (TN) and total carbon (TC) was determined in soil material (9-23 mg) using an elemental CNS analyzer (NA 1500, Carlo Erba; Thermo Fisher Scientific, Franklin, USA)

Soil temperature at -5, -10 and -20 cm depth and soil moisture were continuously measured (12-Bit Temperature sensor -S-TMB-M002 and 10HS Soil Moisture Smart Sensor, Onset Computer Corporation, Bourne, USA) and recorded every 5 min on a data logger (HOBO H21-USB Micro Station Onset Computer Corporation, Bourne, USA). Because of the frequent failure of sensors, extra temperature sensors (HOBO™ pendant loggers, model UA-002-64, Onset Computer Corporation, Bourne, USA) were placed in the soil at a depth of -10 cm.

At farms A and D, sensors were set up at 1.5 m above ground to measure photosynthetically active radiation (PAR, Smart Sensor S-LIA-M003, ONSET Computer Corporation, Bourne, USA), air temperature and air humidity (Temperature/Relative Humidity Smart Sensor, S-THB-M002, Onset Computer Corporation, Bourne, USA). Data were logged every 5 minutes (HOBO H21-USB Micro Station, Onset Computer Corporation, Bourne, USA). Average air temperature and precipitation from the weather station Leeuwarden (18 to 30 km distance from research sites) were used. (KNMI, data). The location specific precipitation was estimated using radar images with a resolution of 3x3 km~~.~~

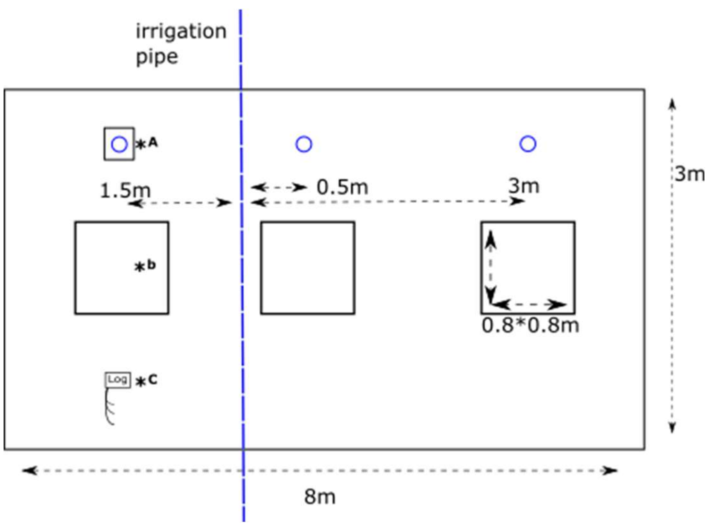

Figure 2 Overview field site SSI. Blue dashed line = irrigation pipe, blue circle = dipwell, A – dipwell with data logger, B – gas measurement frame, C – data logger, -5 -10 -20 soil temperature and soil moisture

C-export was determined by harvesting the frames eight times in 2017 and five times in 2018, the whole field site were managed with 4-5 cuts per year to have a similar grass height with the surrounding field. The biomass was harvested ~~five times per year. These samples~~ where weighed and dried at 70 °C until constant weight. Total nitrogen (TN) and total carbon (TC) was determined in dry plant material (3 mg) using an elemental CNS analyzer (NA 1500, Carlo Erba; Thermo Fisher Scientific, Franklin, USA)

~~The sites were managed with 4-5 cuts per year.~~ Due to grazing disturbance in 2018, an estimation instead of measurements was made for the C-export of location A in consultation with the farmer, but excluded from statistical analysis. Four times per year slurry manure from location C was applied to all plots. The slurry was diluted with ditchwater (2:1 ratio) and applied above ground in the gas measurement frames and the surrounding area (119 – 181 kg N ha^{-1} yr^{-1} for 2017 and 129 – 162 kg N ha^{-1} yr^{-1} for 2018 with a C/N ratio of 16.3±1.3)~~. (0.61 kg m^{-2} yr^{-1} dw for 2017 and 0.62 kg m^{-2} yr^{-1} dw for 2018 with a C/N ratio of 16.3±1.3~~

2.3 Flux measurements

CO_2 exchange was measured from January 2017 to December 2018, at a frequency of two measurement campaigns a month during growing season (April – October) and once a month during winter. This resulted in 34 (A), 35 (C and D) and 38 (B) campaigns over the two years for CO_2 and CH_4. N_2O was measured with a lower frequency with 22 (A), 20 (B and C) and 17 (D) campaigns over the two years.. A measurement campaign consisted of flux measurements with opaque (dark) and transparent (light) closed chambers (0.8x0.8x0.5 m) to be able to distinguish ecosystem respiration (R_{eco}) and gross primary production (GPP) from net ecosystem exchange (NEE). During winter an average of 9 light and 10 dark measurements, and during summer 18 light and 20 dark measurements were carried out over the course of the day, to achieve data over a gradient in soil temperature and PAR.

The chamber was placed on a frame installed into the soil and connected to a fast greenhouse gas analyzer (GGA) with cavity ring-down spectroscopy (GGA-30~~24~~EP, Los Gatos Research, Santa Clara, CA, USA) to measure CO_2 and CH_4 or to a G2508 gas concentration analyzer with cavity ring-down spectroscopy (G2508 CRDS Analyzer, Picarro, Santa Clara, CA, USA) to measure N_2O. To prevent heating and to ensure thorough mixing of the air inside the chamber, the chambers where equipped with two fans running continuously during the measurements. For CO_2 and CH_4, each flux measurement lasted on average

180s. N_2O fluxes were measured on all frames at least once during a measurement campaign, with an opaque chamber for 480s per flux.

PAR was manually measured (Skye SKP 215 PAR Quantum Sensor, Skye instruments Ltd, Llandrindod Wells, United Kingdom) during the transparent measurements, on top of the chamber. The PAR value was corrected for transparency of the chamber. Within each measurement, a variation in PAR higher than 75 µmol m^{-2} s^{-1} would lead to a restart of the measurement. Soil temperature was measured manually in the frame after the dark measurements at -5 and -10 cm depth (Greisinger GTH 175/PT Thermometer, GMH Messtechnik GmbH, Regenstauf, Germany). Crop height was measured using a straight scale with a plastic disk with a diameter of 30 cm before starting the measurement campaign. ~~The biomass was harvested five times per year. These samples where weighed and dried at 70 °C until constant weight. Total nitrogen (TN) and total carbon (TC) was determined in dry plant material (3 mg) using an elemental CNS analyzer (NA 1500, Carlo Erba; Thermo Fisher Scientific, Franklin, USA)~~

2.4 Data analyses

2.4.1 Flux calculations

Gas fluxes were calculated using the slope of gas concentration over time (Almeida et al., 2016) (eq.1).

$$F = \frac{V}{A} * slope * \frac{P * F1 * F2}{R * T} \tag{1}$$

Where F is gas flux (mg m^2 d^{-1}), V is chamber volume (0.32 m^3), A is the chamber surface area (0.64 m^2), slope is the gas concentration change over time(ppm second^{-1}); P is atmospheric pressure (kPa); F1 is the molecular weight, 44 g mol^{-1} for CO_2 and N_2O and 16 g mol^{-1} for CH_4; F2 is the conversion factor of seconds to days; R is gas constant (8.3144 J K^{-1} mol^{-1}); and T is temperature in Kelvin (K) in the chamber.

2.4.2 R_{eco} modeling

To gap-fill for the days that were not measured for an annual balance for CO_2 exchange, R_{eco} and GPP models needed to be fitted with the measured data for each measurement campaign. R_{eco} was fitted with the Lloyd-Taylor function (Lloyd and Taylor, 1994) based on soil temperature (Eq. 2):

$$R_{eco} = R_{eco,Tref} * e^{E_0 * \left(\frac{1}{T_{ref} - T_0} - \frac{1}{T - T_0}\right)} \tag{2}$$

where R_{eco} is ecosystems respiration, $R_{eco,Tref}$ is ecosystem respiration at the reference temperature (T_{ref}) of 281.15 K and was fitted for each measurement campaign, E_0 is long term ecosystem sensitivity coefficient (308.56, (Lloyd and Taylor, 1994)), T_0 Temperature between 0 and T (227.13, Lloyd and Taylor, 1994), T is the observed soil temperature (K) at 5 cm depth and T_{ref} is the reference temperature (283.15 K). If it was not possible to get a significant relationship between the T and the R_{eco} with data from a single campaign, data were pooled for two measuring days to achieve significant fitting (Beetz et al., 2013;Poyda et al., 2016;Karki et al., 2019)

2.4.3 GPP modeling

GPP was obtained by subtracting the measured R_{eco} (CO_2 flux measured with the dark chambers) from the measured NEE (CO_2 flux measured with the light chambers) according to measurement time. For the days in between the measurement campaigns, data were modeled with the relationship between the GPP and PAR using a Michaelis–Menten light optimizing response curve (Kandel et al., 2016;Beetz et al., 2013). For each measurement location per measurement campaign, the GPP was modeled by the parameters α and GPP_{max} (maximum photosynthetic rate with infinite PAR) of (eq.3):

$$\cancel{NEE}GPP = \frac{\alpha * PAR * GPP_{max}}{GPP_{max} + \alpha * PAR} \cancel{- R_{eco}} \tag{3}$$

where NEE is the measured CO_2 flux with light chamber, α is ecosystem quantum yield (mg CO_2 - C m^{-2} h~~s~~$^{-1}$)/(µmol m^{-2} s^{-1}) which is the linear change of GPP per change in PAR at low light intensities (<400 µmol m^{-2} s^{-1} as in (Falge et al., 2001), PAR is measured photosynthetic active radiation (µmol quantum m^{-2} s^{-1}), GPP_{max} is gross primary productivity at its optimum. ~~, R_{eco}~~

~~is ecosystem respiration measured for light response curve and for the year budget calculated with the Lloyd–Tayler function where used. The fitted parameters were linearly interpolated between the measurement campaigns.~~ Due to low coverage of the PAR range in a single measurement campaign, data from multiple campaigns were pooled according to dates, vegetation, and air temperature. ~~in data from year 2017, the complete data set of 2017 were divided into summer and winter periods, and the two datasets (instead of every field campaign) were fitted for the corresponding period per location.~~

2.4.4 ~~Yearly budget calculations~~NECB calculations

~~The calculated parameters were used to interpolate the data for a yearly budget. For the GPP, an important factor of grass growth was added by assuming a linear development of the model parameters α and GPP max, since the plant biomass continued growing between the measurement dates. The NEE year budgets were calculated using the interpolated hourly R_{eco} and GPP values.~~ The NEE is the sum of R_{eco} and GPP values, calculated by applying the hourly monitored soil temperature and PAR data to the models developed per campaign. Extrapolated values at times between ~~two modeled measurements~~two adjacent models are weighted averages of the estimates from these two models, where the weights are temporal distances of the extrapolated time spots to both of the measurements. To account for the influence from plant biomass on the CO2 fluxes, linear relationships between grass height and model parameters ($R_{eco,Tref}$, GPP_{max}, and α) were developed. Models developed for the campaign before harvesting were then corrected using the slopes of the linear regressions as the models after the harvest to be applied in the extrapolation. Unrealistic parameters after correction were discarded, and instead adopted from parameters from campaigns with low grass height at the same plot. The annual CO_2 fluxes were thus summing of the hourly R_{eco}, GPP and NEE values.

The atmospheric sign convention was used for the calculation of NECB. All C fluxes into the ecosystem where defined as negative (uptake from the atmosphere into the ecosystem), and all C fluxes from the ecosystem to the atmosphere are defined as positive. This also holds for non-atmospheric inputs like manure (negative) and outputs like harvests (positive). Both harvest and manure input are expected to be released as CO_2.

~~Besides the campaign-wise gap-filling strategy introduced above, other approaches exist to calculate NEE year budget that may result in different values (Karki et al. 2019), which is considered an important source of uncertainty in our study. To quantify this uncertainty, six R_{eco} models and four GPP models were select from Karki et al. (2019) and fitted with annual data~~

~~(Supplement Table 1). The models with Nash–Sutcliffe modeling efficiencies (NSE) larger than 0.5 (Hoffmann et al. 2015) was accepted and calculated into gap-filled NEE. Not all sites and years have acceptable models due to large variations of measured fluxes within a year. The remaining NEE values were averaged per site per year and compared with the campaign-wise NEE year budgets as a range of uncertainty.~~

2.4.5 CH_4 and N_2O fluxes

CH_4 and N_2O fluxes per site and measurement campaign were averaged per day. The annual emissions sums for CH_4 where estimated by linear interpolation between the single measurement dates. Global Warming Potential (GWP) of 34 t CO_2-eq and 298 t CO_2-eq per ton for CH_4 and N_2O was used according to IPCC standards (Myhre et al., 2013) to calculate the yearly GHG balance.

2.4.6 Uncertainties

The estimation of total uncertainties of the yearly budget should include multiple sources of error, where both model error and uncertainty from extrapolations in time are the most important (Beetz et al., 2013). Therefore, we included these two sources of error and combined them into a total uncertainty in three steps.

First, we calculated the model error, which would cover the uncertainties from replications (between the three frames) and the random errors from the measurements, the environmental conditions at the time, and the parameter estimation of R_{eco} and GPP. Standard errors (SE) of the prediction were calculated for each measurement campaign / pooled dataset as the SEs of the midday of the campaign dates. The hourly SEs were then extrapolated linearly between modeled campaigns. Total model error of the annual NEE was therefore calculated following the law of error propagation as the square root of the sum of squared SEs.

Second, we attribute the uncertainty from extrapolation to the variations from selecting different gap-filling strategies, since other approaches of annual NEE estimation including different R_{eco} and GPP models would result in different values (Karki et al., 2019).~~Besides the campaign-wise gap-filling strategy introduced above, other approaches exist to calculate NEE year budget that may result in different values (Karki et al. 2019), which is considered an important source of uncertainty in our study~~. To quantify this uncertainty, six R_{eco} models and four GPP models were select from Karki et al. (2019)) and fitted with annual data (Supplement Table 1). The models ~~with Nash–Sutcliffe modeling efficiencies (NSE) larger than 0.5~~were evaluated following the thresholds of performance indicators in Hoffmann et al. (2015). R_{eco} and GPP models that were above the ‘satisfactory’ rating was~~was~~ accepted and calculated into gap-filled NEEs. Based on all the annual NEEs per site and year, standard deviations from the means were considered as the extrapolation uncertainty. In the year 2018, the control site of farm D did not yield any satisfactory R_{eco} model. The uncertainty was thus calculated as the average of all sites.~~Not all sites and years have acceptable models due to large variations of measured fluxes within a year. The remaining NEE values were averaged per site per year and compared with the campaign-wise NEE year budgets as a range of uncertainty.~~

Finally, we calculated the total uncertainties per site and year following the law of error propagation with the uncertainties from the previous steps.

2.5 Statistics

The effect of the treatment on gap-filled annual R_{eco} and GPP, the resulting NEE, the C-export data, the NECB, and the measured CH_4, N_2O exchanges~~ and the combined GHG balance~~ were tested by fitting linear mixed-effects models, with farm location as a random effect. Effectiveness of the random term was tested using the likelihood ratio test method. Significance of the fixed terms was tested via Satterthwaite's degrees of freedom method. General linear regression was used instead when the mixed effect model gives singular fit. The treatment effect was further tested using campaign-wise R_{eco} data. Measured R_{eco} fluxes from SSI and Control were calculated into daily averages and paired per date. The data pairs were grouped based on the GWT differences between SSI and control of the dates. Differences between treatments were then analyzed by linear regression of the R_{eco} flux pairs without interception and testing the null hypothesis ‘slope of the regression equals to 1’. All statistical analyses were computed using R version 3.5.3 (Team, 2019) using packages lme4 (Bates et al., 2014), lmerTest (Kuznetsova et al., 2017), sjstats (Lüdecke, 2019), and car (Fox and Weisberg, 2018).

3 Results

3.1 Weather conditions

Mean annual air temperature was 10.3 °C for 2017 and 10.7 °C for 2018, which were higher than the 30-year average of 10.1 °C. The growing season (April–September) in 2017 was slightly cooler with 14.3 °C than the average of 2018 at 14.6 °C, while the temperature during the growing season in 2018 was 1.1 °C warmer than average. Precipitation was slightly higher for 2017 840-951 mm compared to the 30-year average of 840 mm (KMNI ~~data~~). There was a small period of drought in May and June, ending in the last week of June (see Fig.3). In contrast, 2018 was a dry year with average of 546-611 mm. The year is characterized by a period of extreme drought in the summer, from June to the beginning of August, and precipitation lower than average in the fall and winter.

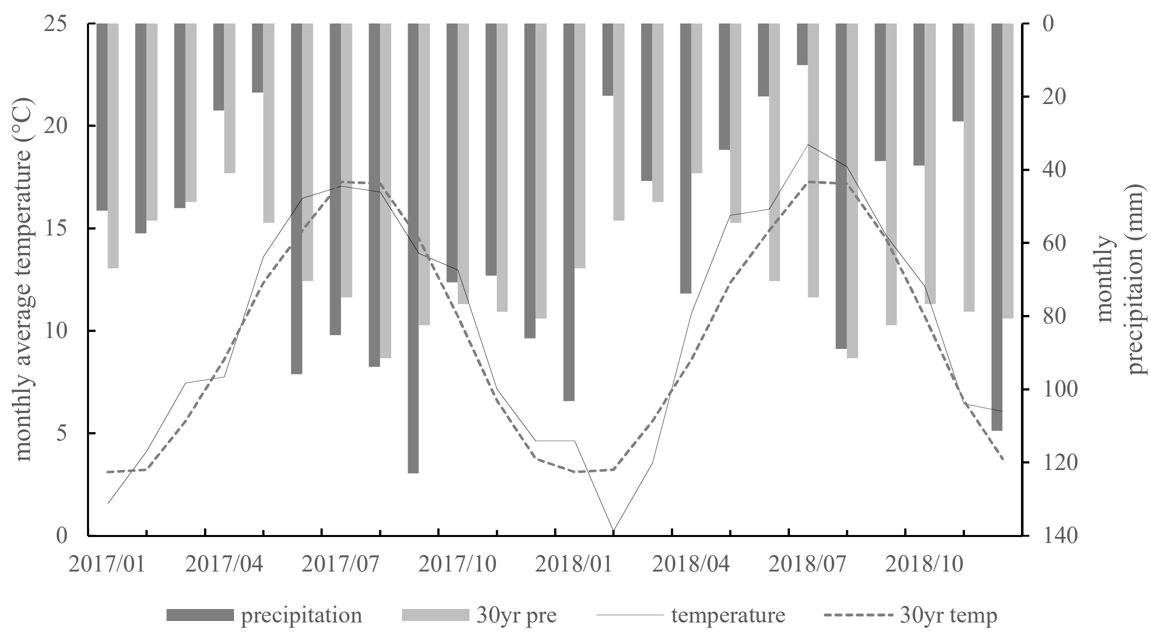

Figure 3 Monthly average air temperature at weather station Leeuwarden (18 to 30 km distance from research sites), and the 30-year average. Sum precipitation at weather station Leeuwarden, and the 30-year average.

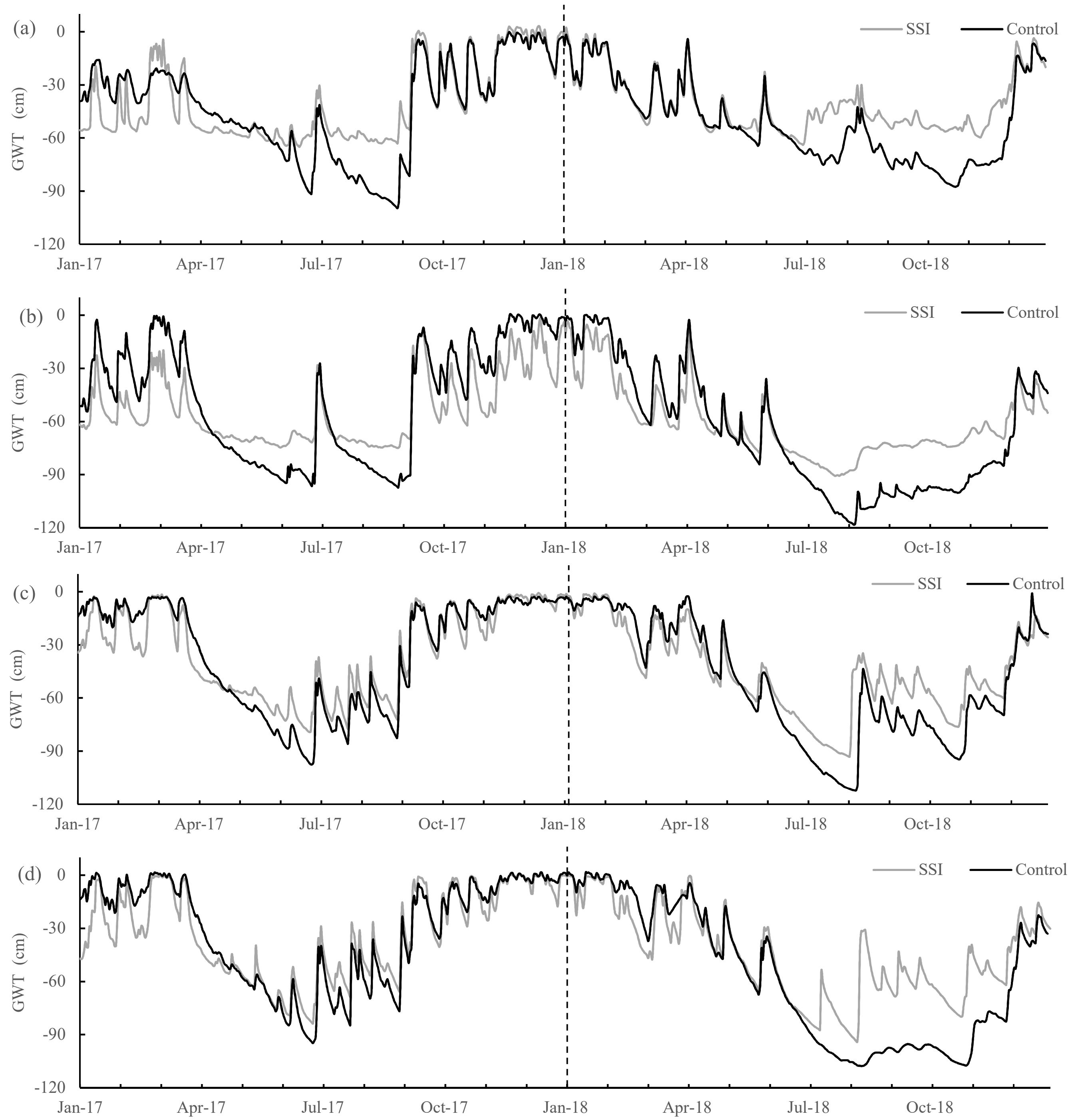

Figure 4 Groundwater table (GWT, ~~below~~ from soil surface) during the measuring period per farm (letter), per graph SSI (m~~M~~easured 1.5 m from the irrigation pipe) and control.

3.2 Groundwater table (GWT)

Deploying ~~subsurface irrigation (SSI)~~SSI systems affected the GWT during the two years for all farms (Fig. 4). However, there was a large variation in effect-size between years and locations. The effect of SSI can be divided into two types of periods. Periods with drainage (decreased GWT), in the wet periods, coincided with the autumn (in 2017) and winter period (2017 and 2018). Irrigation (increased GWT) periods, where the SSI leads to a higher water table than control, occurred during spring and summer when the GWT dipped below the ditch water level. In 2017, the effectiveness differed per farm. For locations A and B, GWT was more stable in summer around the -60 and -70 for SSI compared to the control, while locations C and D the GWT fluctuated more like in the control fields. During the dry summer of 2018, in contrast, all locations showed a strong effect of irrigation, especially after the dry period in the beginning of august. In this period the water table recovered quickly while the control lagged behind.

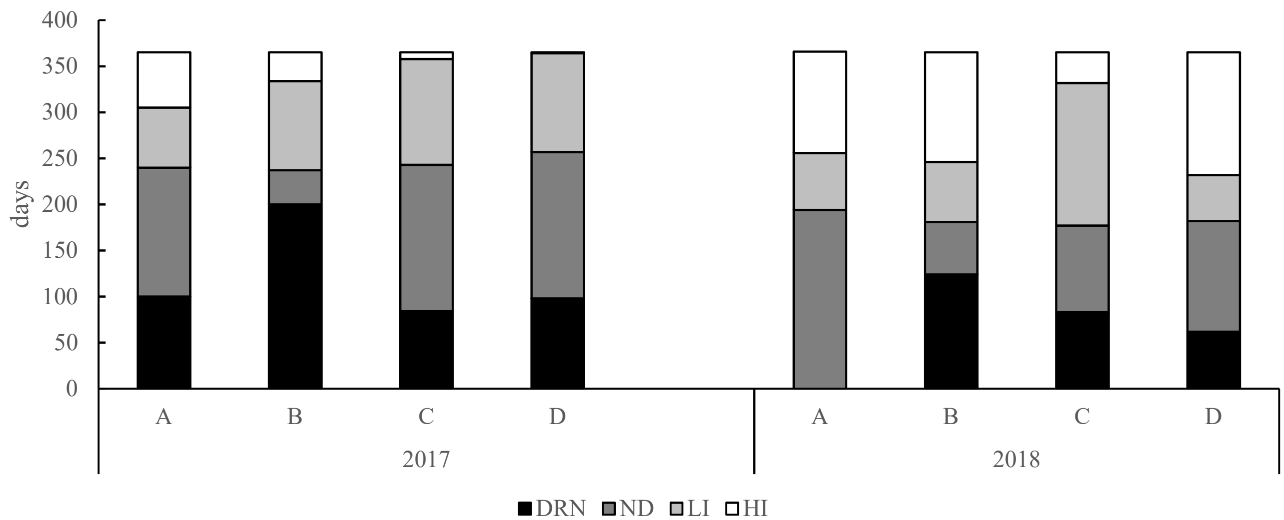

Figure 5 Days with effective drainage/ irrigation for the four locations. drainage (DRN, <-5 cm), no difference (ND, -5 ~ 5 cm), low to intermediate irrigation (LI, 5 ~ 20 cm) and high irrigation (HI, > 20 cm) 1.5 m from the irrigation pipe.

Although there was hardly any difference in annual average GWT between control and SSI (Table 2), drainage and irrigation effects could be observed when dividing the calendar year into seasons. The effective days of the SSI are summarized in Fig. 5 according to four categories, based on practical definitions of drainage and irrigation: drainage (DRN, <-5 cm), no difference (ND, -5 ~ 5 cm), low to intermediate irrigation (LI, 5 ~ 20 cm) and high irrigation (HI, > 20 cm). These categories are also used in the statistical analysis of R_{eco} measurements (see 3.7 Seasonal R_{eco}). In 2017 there were 17 days more without any GWT difference than in 2018. There was a much stronger irrigation effect in the dry year of 2018, with 61 more irrigated days comparing to 2017, and the number of irrigation days was constantly similar to, or higher than the number of drainage days, except for site B in 2017 which had a long period showing a drainage effect.

Table 2:Average ground-water table during the measuring period per farm. Summer ground-water table ranges from April till October~~-~~. Measured 1.5 meter from the irrigation pipe.

Location	Treatment	Average 2017	Summer 2017	Average 2018	Summer 2018
A	SSI	-43	-52	-51	-48
	Control	-40	-63	-41	-59
B	SSI	-47	-64	-67	-71
	Control	-53	-73	-61	-83
C	SSI	-35	-54	-51	-56
	Control	-34	-61	-45	-67
D	SSI	-31	-51	-59	-56
	Control	-32	-56	-45	-77

3.3 Measured R_{eco}

~~Despite these observed differences in GWT, there was no overall effect of SSI on the total C budget.~~ Fig. 6 ~~C~~comparesing ~~days and locations where~~ the measured R_{eco} fluxes ~~was measured~~ with the corresponding GWT measurements, ~~d the GWT, provide insight of the effect of SSI. There is variation differences between emission rates depending on temperature and grass height, but these differences were small during the measurement day due to the regular harvests. However, the R_{eco} values for the measurement days can~~which could give an indication for the ~~effectivity~~ effectiveness of the GWT differences ~~in GWT (Fig. 6)~~. The division between the groups was based on ~~the function of the irrigation pipes,~~ the GWT differences ~~of the GWT~~ between the SSI and control sites on the measurement days (~~similar to~~ the same~~the~~ groups used in Fig. 5~~2~~). There was a slightly higher R_{eco} for SSI during drainage periods when GWT was lower, which compensates for the lower R_{eco} during summer. For moments where there was no GWT difference and those showing moderate irrigation, there was no effect of SSI on R_{eco}. However, when the GWT of the SSI was more than 20cm higher than the control, the emissions of the control where significantly higher than SSI ($p < 0.01$), indicating an effect of the irrigation. However, this effect of the raised GWT was small, even though in some cases the GWT was raised more than 60 cm. According to Fig. 5, ~~shows how often the different groups of GWT effects occurred. For 2017~~ in 2017, the majority of the days were dominated by drainage (increasing R_{eco}), or by no difference or small irrigation resulting in no effect on the R_{eco}. However, the moments with increased irrigation, when there was a reduced R_{eco} effect of SSI, were sparse compared to the other dominating periods.

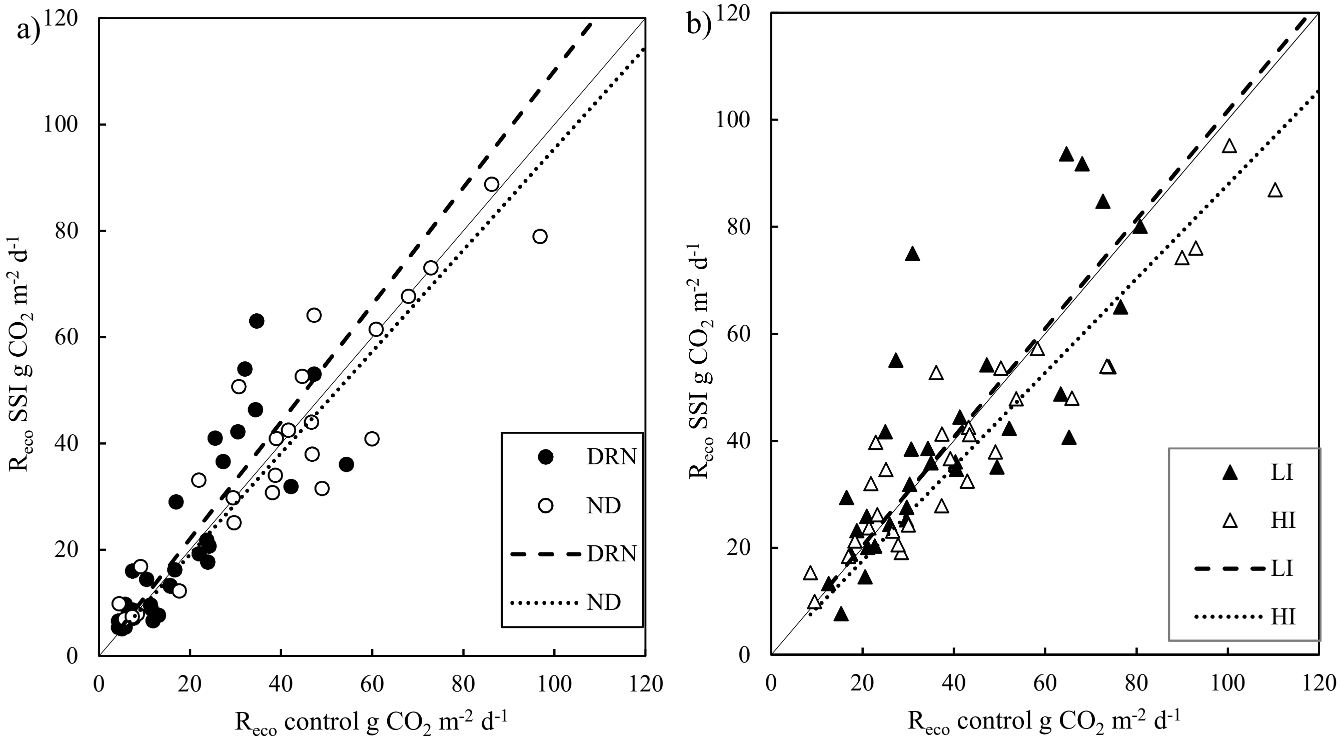

Figure 6 Measured fluxes for ecosystem respiration (R_{eco}), one-to-one comparison in which daily averages where used. a) Values divided into two groups: where the ground water table was lower due to the effect of drainage, and where there was a limited difference. B) Values divided into two groups with irrigation effects, moderate infiltration with more than 5–20 cm difference and high infiltration with more than 20cm difference between SSI and Control. Black filled line is the 1:1 line.

3.4 Annual ~~carbon exchange rates~~carbon fluxes

3.4.1 Gross primary production (GPP)

GPP was high for all locations in both years, showing a clear seasonal pattern with the highest uptake at the start of the summer (Fig.7). GPP was 30% lower in the dry year 2018 ($p < 0.001$) compared to 2017 (see Table 2) and differed between locations (random effect p = ~~0.009~~0.006). ~~The average GPP over all location for the SSI treatment was -80±4 t CO_2 ha^{-1} yr^{-1} for 2017, and -58±4 t CO_2-eq. ha^{-1} yr^{-1} for 2018.~~ There was, however, no treatment effect on GPP (p = 0.~~733~~3101). Average GPP values for all ~~control and~~ SSI and control plots were -88.3±7.5~~-81±4 t CO_2 eq. ha^{-1} yr^{-1}~~ and -89.2±13~~-80±4~~ t CO_2 ~~eq.~~ ha^{-1} yr^{-1} for 2017, -71.7±6.6~~-55±3 t CO_2 eq. ha^{-1} yr^{-1}~~ and -65.7±4.9~~-58±4~~ t CO_2 ~~eq.~~ ha^{-1} yr^{-1} for 2018, respectively.

3.4.2 Ecosystem respiration (R_{eco})

R_{eco} was generally high for all the farms measured during the two years, with the average R_{eco} of ~~131~~128.4±~~1~~4.6 t CO_2 ha^{-1} yr^{-1} for 2017 being significantly higher than 100.8~~101~~±114 t CO_2 ha^{-1} yr^{-1} for 2018 ($p < 0.001$) (Table 2). Different seasonal patterns were also observed between the two years, where in 2017 R_{eco} peaked in June and July, while in 2018 the highest R_{eco} was found in May (Fig. 7, Appendix B). However, no effect of SSI on R_{eco} was found (p = 0.~~350~~6191), with average R_{eco} values for all SSI and control plots as 128.7±9.2 and 126.7±9.5 t CO_2 ha^{-1} yr^{-1} in 2017, 102.1±14.1 and 99.6±13.5 t CO_2 ha^{-1} yr^{-1} in 2018.~~no difference among farm locations (random effect p = 0.627). R_{eco} showed a strong seasonal pattern; in 2017 R_{eco} peaked in June and July, while in 2018 the highest R_{eco} was found in May (Fig. 7 Appendix B).~~

~~3.4.3 C-export (yield)~~

~~C-exports (i.e. yields) differed between years without treatment effect of SSI (p = 0.691). Following the drought in 2018, C export (13.8±0.6 t CO_2 ha^{-1} yr^{-1}) was significantly lower (p < 0.001) than in 2017 (18.0±1.4 t CO_2 ha^{-1} yr^{-1}). These values corresponded to dry matter yields of 9.4±0.6 t DM ha^{-1} yr-1 in 2018 and 12.6±1.1 t DM ha^{-1} yr^{-1} in 2017. The year-effect differed per location (random effect p < 0.001). We found a solid relationship between C-export and GPP (p < 0.001, r^2 = 0.942; linear-mixed modeling).~~

3.4.~~4~~ 3 Net ecosystem exchange (NEE)

All locations functioned as large C sources during the measurement period. The average annual NEE of all sites ~~and years~~ amounted~~, on average,~~ to ~~47.1~~39.7±11 and 31.8±8.4 t CO_2 ha^{-1} yr^{-1}~~,~~ in 2017 and 2018, respectively ~~with an uncertainty of 3-16 t CO_2 ha^{-1} yr^{-1}~~. The overall explanatory power of year, treatment and location was low~~, (conditional r^2 = 0.531 for fixed and random effects combined) after combining R_{eco} and GPP into NEE. There was, again,~~with no yearly difference between 2017 and 2018 (p = 0.1813), or any treatment effect of SSI (p = 0.~~329~~9805). The average NEE values for all SSI and control plots are 40.4±11.9 and 37.5±16.1 t CO_2 ha^{-1} yr^{-1} in 2017, 30.4±15.6 and 34±14.5 t CO_2 ha^{-1} yr^{-1} in 2018, respectively.~~, but there were small differences between both years~~ or~~(p = 0.040)~~. ~~NEE values were 67.9±1.6 t CO_2 ha^{-1} yr^{-1} in 2017 and 56.4±5.1 t~~

~~CO_2 ha^{-1} yr^{-1} in 2018 for the treatment plots. No differences between locations were observed (random effect $p = 0.076$). On average, for all sites and both years, the emission was 62 t CO_2 eq. ha^{-1} yr^{-1} with an uncertainty of 3–16 t CO_2 ha^{-1} yr^{-1}~~

3.4.~~3~~4 C-export (yield)

C-exports (i.e. yields) differed between years without treatment effect of SSI ($p = 0.691$). Following the drought in 2018, C export (13.8±0.6 t CO_2 ha^{-1} yr^{-1}) was significantly lower ($p < 0.001$) than in 2017 (18.0±1.4 t CO_2 ha^{-1} yr^{-1}). These values corresponded to dry matter yields of 9.4±0.6 t DM ha^{-1} yr-1 in 2018 and 12.6±1.1 t DM ha^{-1} yr-^{1} in 2017. The year-effect differed per location (random effect $p < 0.001$). We found a solid relationship between C-export and GPP ($p < 0.001$, $r^2 = 0.942$; linear-mixed modeling).

3.4.5 Net ecosystem carbon balance (NECB)

All sites are large carbon sources, without an effect of SSI ($p = 0.9446$) which was consistent for all farms (Table 3). However, there was a significant difference between the two years, with higher carbon emission rates in 2017 amounting to 49.6±11 t CO_2 eq. ha^{-1} yr^{-1} on average, compared with 36.9±7.6 t CO_2 eq. ha^{-1} yr^{-1} for 2018 ($p=0.0277$).

3.5 Methane exchange

The total exchange of CH_4 was very low during both years with no effect from the SSI ($p=0.1147$) or difference between years ($p=0.1253$). During most periods, the locations functioned as a sink of CH_4. The annual fluxes were -0.01±0.01 t CO_2 eq. ha^{-1} yr^{-1} (-0.25 kg CH_4 ha^{-1} yr^{-1}) for 2017 and -0.06±0.05 t CO_2 eq. ha^{-1} yr^{-1} (-1.8 kg CH_4 ha^{-1} yr^{-1}) for 2018 (~~table~~ Table ~~3~~4). Such exchange did not play a significant part in the total GHG emissions~~balance~~ (comparable to less than 0.4% of the annual ~~GHG balance~~NECB)~~, and was not influenced by SSI (p = 0.232) or farm location (random effect p = 0.726). Fluxes only differed between years (p = 0.027)~~.

3.6 Nitrous oxide exchange

~~The fluxes for N_2O showed a high spatial variability between (random effect p = 0.010) and within all locations, and showed an erratic pattern with mostly low emissions with some high peaks.~~

There was no treatment effect ($p=0.5640$) or inter-annual difference ($p=0.4414$) detected. The highest average emissions were measured on the SSI plot of location D, with 5.78±5.9 mg N_2O. m^{-2} d^{-1} for 2017 and 10.7±17.4 mg N_2O. m^{-2} d^{-1} for 2018.~~The highest emissions were measured on the frame closest to the irrigation pipe in the treatment plot of location D, with 4.4 t CO_2 eq. ha^{-1} yr^{-1} for 2017 and 4.9 t CO_2 eq. ha^{-1} yr^{-1} for 2018.~~ The highest peak was measured on the frame closest to the irrigation pipe in August for SSI of location D, showing 55±15 mg N_2O m^{-2} d^{-1}. The peaks observed were erratic~~, and cannot be explained by year or treatment effect ($p = 0.060$ and $p = 1.000$ respectively, marginal $r^2 = 0.107$ for the fixed effects). Emissions~~ and did not correspond to fertilization management with slurry before measurement campaigns.

~~**3.7 Total GHG balance**~~

~~All sites showed high emissions, without an effect of SSI ($p = 0.332$) which was consistent for all farms, without location effect (random effect $p = 0.099$) (table 3). However, there was a large difference between both years, with higher emission rates in 2017 amounting to 63±2 t CO_2 eq. ha^{-1} yr^{-1}, compared 52±3 t CO_2 eq. ha^{-1} yr^{-1} for 2018 ($p<0.001$).~~

Table 3 Overview of all processes contributing to the carbon balance calculated for both years. Ecosystems respiration (R_{eco}), gross primary production (GPP), net ecosystems exchange (NEE, sum of GPP and R_{eco}), C-exports ~~(from~~ harvest)~~, and~~ C-~~addition~~ manure ~~from~~ (carbon addition from manure application), and net ecosystem carbon balance (NECB, sum of all fluxes) for subsoil irrigation (SSI) and control plots at farm locations A-D. The **range of R_{eco}, GPP and NEE represent the combination of model error and extrapolation uncertainties following the law of error propagation.**

-	-	-	Carbon exchange				
Year	Location	treatment	R_{eco}	GPP	NEE	C-export	C-manure
-	-	-	t CO_2 ha^{-1} yr^{-1}-	t CO_2 ha^{-1} yr^{-1}-	t CO_2 ha^{-1} yr^{-1}-	t CO_2 ha^{-1} yr^{-1}-	t CO_2 ha^{-1} yr^{-1}-
2017	A	SSI	123.1±3.4129.4	-88.9±2.7-75.2	34.2±4.454.2	16.6	-6.9
-		Control	133.7±6.5134.1	-81.3±7.9-79.8	52.4±10.254.3	19.3	-6.9
-	B	SSI	125±5.8133.7	-98.5±3-80.6	26.5±6.553.1	15.3	-5.3
-		Control	123.2±5.8125.9	-92.6±2.9-74	30.6±6.551.9	15.5	-5.3
-	C	SSI	132.1±4.6136.3	-87.7±5.7-91.4	44.5±7.444.9	22.1	-10.9
-		Control	122.3±3.2129.3	-100±8.3-92.6	22.3±8.936.7	23.3	-10.9
-	D	SSI	134.5±4.2134.1	-78.6±2.8-74.6	56±559.5	15.7	-9.3
-		Control	127.9±2129.1	-82.9±5.3-77.4	44.9±5.651.6	16.3	-9.3
2018	A	SSI	98.3±6.598.3	-74.7±2.5-59.3	23.6±739	14	-7.4
-		Control	101.3±5.5102.8	-68.9±3.1-63.3	32.4±6.439.5	14	-7.4
-	B	SSI	117.5±10.1117.5	-73.4±3.4-60.1	44.2±10.757.4	13.8	-9.3
-		Control	111.4±10.5112.5	-64.5±2.8-53.5	46.9±10.959	12.2	-9.3
-	C	SSI	109.6±5.8109.7	-82.4±4.6-65.6	27.3±7.444.1	15.7	-9.3
-		Control	99.2±1.390	-73.7±0.6-58.5	25.5±1.531.6	15.8	-9.3
-	D	SSI	82.9±4.584.2	-56.1±2.2-45.2	26.8±539	13.4	-9.3
-	-	Control	86.6±6.389.6	-55.5±2.4-46.8	31.1±742.8	12	-9.3

			Carbon exchange					NECB
Year	Location	treatment	R_{eco} t CO_2 ha^{-1} yr^{-1}	GPP t CO_2 ha^{-1} yr^{-1}	NEE t CO_2 ha^{-1} yr^{-1}	C-export t CO_2 ha^{-1} yr^{-1}	C-manure t CO_2 ha^{-1} yr^{-1}	CO_2 t CO_2 ha^{-1} yr^{-1}
2017	A	SSI	123.1±3.4	-88.9±2.7	34.2±4.4	16.6	-6.9	46.8
		Control	133.7±6.5	-81.3±7.9	52.4±10.2	19.3	-6.9	65.7
	B	SSI	125±5.8	-98.5±3	26.5±6.5	15.3	-5.3	37.4
		Control	123.2±5.8	-92.6±2.9	30.6±6.5	15.5	-5.3	41.4
	C	SSI	132.1±4.6	-87.7±5.7	44.5±7.4	22.1	-10.9	55.8
		Control	122.3±3.2	-100±8.3	22.3±8.9	23.3	-10.9	35
	D	SSI	134.5±4.2	-78.6±2.8	56±5	15.7	-9.3	62.4
		Control	127.9±2	-82.9±5.3	44.9±5.6	16.3	-9.3	52.2
2018	A	SSI	98.3±6.5	-74.7±2.5	23.6±7	14	-7.4	29.7
		Control	101.3±5.5	-68.9±3.1	32.4±6.4	14	-7.4	38.5
	B	SSI	117.5±10.1	-73.4±3.4	44.2±10.7	13.8	-9.3	48.8
		Control	111.4±10.5	-64.5±2.8	46.9±10.9	12.2	-9.3	49.8
	C	SSI	109.6±5.8	-82.4±4.6	27.3±7.4	15.7	-9.3	32.6
		Control	99.2±1.3	-73.7±0.6	25.5±1.5	15.8	-9.3	31.5
	D	SSI	82.9±4.5	-56.1±2.2	26.8±5	13.4	-9.3	31
		Control	86.6±6.3	-55.5±2.4	31.1±7	12	-9.3	33.3

Table 4 The average measured CH_4 and N_2O emissions subsoil irrigation (SSI) and controls for the four locations (A-D) for both years in mg m^{-2} d^{-1}. The total CH_4 balance in CO_2 equivalents, using radiative forcing factors of 34 for CH_4 according to IPCC standards (Myhre et al., 2013). The ranges of CH_4 and N_2O represent the standard deviation (SD) of the measured fluxes.~~All GHG emissions contributing to the total GHG balance for subsoil irrigation (SSI) and controls for the four locations (A-D) for both years. The sum of NEE, C-export and C-manure form the total CO_2 flux. The total GHG balance per year, location and treatment is the sum of CO_2, CH_4 and N_2O fluxes in CO_2 equivalents, using radiative forcing factors of 34 for CH_4 and N_2O 298 according to IPCC standards (Myhre et al., 2013).~~

			GHG fluxes		Balance
Year	Location	treatment	CH_4 mg CH_4 m^{-2} d^{-1}	N_2O mg N_2O m^{-2} d^{-1}	CH_4 t CO_2 eq. ha^{-1} yr^{-1}
2017	A	SSI	-0.44±0.5	0.02±0.7	-0.01
		Control	-0.54±0.9	1.46±1.8	-0.05
	B	SSI	-0.43±0.4	3.81±3.3	-0.04
		Control	-0.27±0.9	2.30±4.9	-0.02
	C	SSI	-0.43±1.0	2.48±1.5	-0.03
		Control	-0.40±0.5	2.56±2.0	0.01
	D	SSI	-0.50±0.8	5.78±5.9	0.01
		Control	0.72±2.7	4.81±2.3	0.06
2018	A	SSI	-0.39±0.7	0.15±0.8	-0.05
		Control	-0.67±1.2	0.80±0.9	-0.12
	B	SSI	-0.40±0.3	2.08±3.7	-0.04
		Control	-0.30±0.9	4.88±3.9	0
	C	SSI	-0.73±0.9	3.27±3.0	-0.11
		Control	-0.66±0.9	4.46±3.7	-0.07
	D	SSI	-0.91±0.6	10.7±17.4	-0.09
		Control	-0.14±0.8	2.69±2.2	0.02

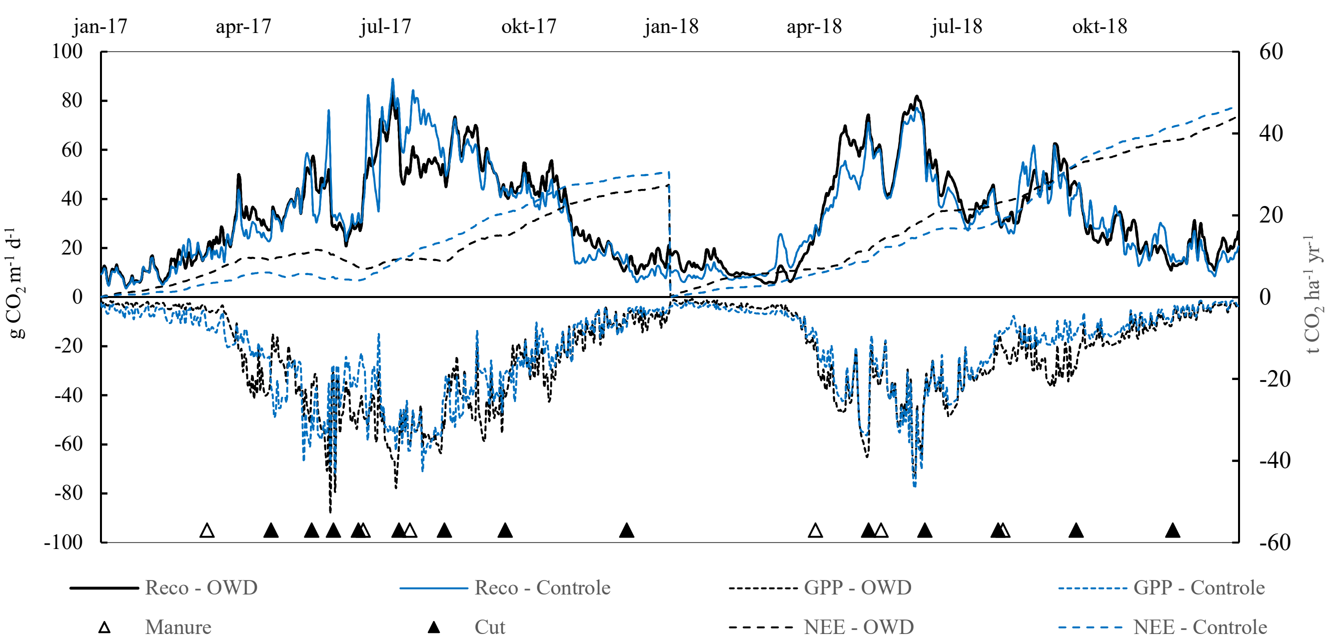

Figure 7 R_{eco} and GPP for location B in g CO_2 m^{-2} d^{-1} on the primary y-axis, for control and SSI. Accumulative NEE in t CO_2 ha^{-1} yr^{-1}, for control and subsoil irrigation (SSI), every year starting at 0.

4 Discussion

For both years, SSI had a clear irrigation effect during summer ~~at the four farms~~, increasing the averages of GWT during summer period ~~on average~~ by 6–18 cm at the four farms. During winter, there was a moderate but consistent drainage effect, reducing the average GWT in the wet/winter period by 1–20 cm. Despite the irrigation effects and higher water levels in summer, there was no effect of SSI on R_{eco} and total GHG balances remained high (62 t CO_2 eq. ha^{-1} yr^{-1} on average of all sites and years with an uncertainty of 3–16 t CO_2 ha^{-1} yr^{-1}). We found no evidence for a reduction of CO_2 emissions, nor for higher yields, on an annual base by implementing SSI.

4.1 SSI does not reduce annual R_{eco}

~~Despite the higher summer GWT, there was no effect of SSI on the annual R_{eco} at all sites.~~ Based on the direct comparison using measured R_{eco} fluxes (Fig. 6), ~~W~~we found a modest 5–10% reduction in R_{eco} only when GWT differences were larger than 20 cm~~, based on the direct comparison using measuredraw R_{eco} fluxes (Fig. 6)~~. When the irrigation effect was smaller, no effect on the R_{eco} was found. An earlier study in the Netherlands on the role of GWT~~-~~ also showed small effects of higher summer GWT on R_{eco} and NEE ~~(Net Ecosystem Exchange)~~ despite substantial differences in soil volume changes/soil subsidence (Dirks et al., 2000). Similarly, a~~the~~ 4-year study (Schrier-Uijl et al., 2014) found little differences in NEE estimates despite substantial large variations in summer GWT and soil moisture contents.

Our findings contradict the general assumption that a higher GWT leads to lower CO_2 emissions, which is often found in near-natural peatlands with the presence of peat-forming vegetation (Wilson et al., 2016a;Lloyd, 2006a;Moore and Dalva, 1993). However, most studies discuss the effect of ~~lower~~ annual average GWT, instead of seasonal changes in GWT. In addition, there are also studies that did not find an effect of GWT on CO_2 emissions during the season (Parmentier et al., 2009;Lafleur et al., 2005;Nieveen et al., 2005)~~)~~. This lack of effect is explained by the fact that there is only a small variation in soil moisture values above the GWT. A large number of studies report lower CO_2 emissions when water levels were structurally elevated, concomitant with substantial differences in vegetation/land use following higher water levels (Beetz et al., 2013;Schrier-Uijl et al., 2014;Wilson et al., 2016a). In our study, SSI seems to have an effect of a similar magnitude trending towards higher emissions during periods with lower GWT at the SSI sites.

The small effect size in our study can most probably be explained by differences in peat oxidation rates along the soil profile. Some other studies suggest that the top 30–40 cm layer of the peat profile plays an important role in C turnover rates in drained peatlands, due to more readily decomposable C sources and higher temperatures (Saeurich et al., 2019;Karki et al., 2016;Lafleur et al., 2005;Moore and Dalva, 1993). This soil layer was, however, not affected by higher summer GWTs in our study. Moreover, the ~~top soil~~topsoil layer was even exposed to oxygen for longer periods due to extra drainage during wet seasons. As the infiltrating water will affect the soil moisture content of these layers, it is even expected that this content will approach the optimum for C mineralization more often at the locations where SSI is applied. Saeurich et al. (2019) speculated that the highest CO_2 production in the top 10 cm is reached when GWTs are approximately 40 cm below the surface (Silvola et al., 1996).

In contrast to surface irrigation, where the topsoil is replenished with moisture, the SSI effect is limited to deeper parts of the peat soils, at -60–-100 cm depth. However, the role of this deeper layer as a C source is only limited. Its potency to act as a C source is reduced by lower temperatures, limited O_2 intrusion, and the fact that water content of this layer is already close to saturation (Berglund and Berglund, 2011;Taggart et al., 2012;Saeurich et al., 2019). This layer shows low levels of stronger electron acceptors such as O_2 and nitrate used for the microbial oxidation of organic compounds, and of labile organic matter (Fontaine et al., 2007;Leifeld et al., 2012). Visually, the layers deeper than 60 cm are less decomposed (plant macrofossils still visible) compared to the highly degraded uppermost 40 cm.

In addition, lower CO_2 production in ~~the~~ deeper peat layers that are saturated due to the higher water level may be compensated ~~for~~ by the increased CO_2 production in the top 20–40 cm due to the higher moisture levels resulting from elevated water levels. The dry year of 2018 with very low GWT in the control sites (and thus an expected maximized effect of SSI) provides additional evidence that SSI contributes little if any to the mitigation of CO_2 emission from drained peatlands. Such understanding of the processes of CO_2 emissions in relation to soil profiles, along with the~~The second~~ assumption from the Dutch soil-carbon-water model that the average lowest summer GWT is the major control of CO_2 emissions, is currently under investigation (STOWA, 2020)

4.2 SSI effects on CH_4 and N_2O emissions

The magnitudes of measured CH_4 and N_2O fluxes are substantially lower than CO_2, which would thus lead to negligible contributions to the total GHG emissions in our case. Looking directly at the measured fluxes, no SSI effect was detected for neither CH_4 or N_2O. Findings of this experiment agree with the generally accepted idea that intensively drained peatlands have low levels of CH_4 emissions, and often these systems even function as a small CH_4 sink (Couwenberg et al., 2011;Couwenberg and Fritz, 2012;Tiemeyer et al., 2016;Maljanen et al., 2010). The SSI site in farm D showed the highest N_2O emissions with 10.7±17.4 mg N_2O m^{-2} d^{-1} for 2017~~The SSI site in farm C showed the highest N_2O emissions with 23 kg N_2O ha^{-1}yr^{-1} for 2017~~. In the current study the average measured N_2O emission ~~measured from the drained peatland grasslands~~ was 12 kg N_2O ha^{-1} yr^{-1} falling with the range of annual N_2O emissions from drained peatlands in Northern Europe (4-18 kg N_2O ha^{-1}) ~~In the current study the average N_2O emission from the drained peatland grasslands was 9 kg N_2O ha^{-1}yr^{-1} falling with the range of annual N_2O emissions from drained peatlands in Northern Europe (4-18 kg N_2O ha^{-1})~~ (Kandel et al., 2018;Leahy et al., 2004;Maljanen et al., 2010). Fertilization, temperature and water table fluctuations play major roles in the total N_2O emission (Regina et al., 1999;Van Beek et al., 2011). No distinct peaks were measured after application of fertilizer, and fertilizer was applied on all locations on the same day, so missing peak fluxes would not influence the comparison. The mechanisms of N_2O production and consumption in organic soils are, however, complex and there is high temporal and spatial variability as influenced by site conditions and management (Leppelt et al., 2014;Taghizadeh-Toosi et al., 2019). Because of the low measurement interval for both years in the winter period, there is ~~large~~high chance of an underestimation of the N_2O~~N_2O~~ emission, although this would not result in noticeable changes on the total GHG emissions. It is well studied that periods ~~where temperature changes for~~with frost and~~to~~ thawing result in high N_2O emissions (Koponen and Martikainen, 2004).

~~**4.3 High CO**2 emissions, but lack of effect of SSI on GHG emission~~

4.3 Reasonably high NEE

In contrast to the expected function of the SSI technique based on land subsidence data, no effect has been found on either promoting the yield/GPP nor reduction on NEE and other GHG emissions. Our NEE estimates from all sites and years at 35.8 (22.6 – 56.0) t CO_2 ha^{-1} yr^{-1} has exceeded the ranges reported for drained temperate peatlands, where Tiemeyer et al. (2020) reported 30.4 (5.1 – 40.3) t CO_2 ha^{-1} yr^{-1} for the German drained organic soils, and ~~Jacobs et~~ Veenendaal et al. (2007) reported ~~8.1 (±3.3)~~ 4.9 t CO_2 ha^{-1} yr^{-1} in an earlier analysis at an intensively managed Dutch peat meadow measured with eddy covariance~~eight Dutch grasslands~~.

Looking into GPP and R_{eco} individually, on the one hand, ~~T~~the GPP of the sites ~~(-45.2 – -92.6-80.7 and -56.5 t CO_2 ha^{-1} yr^{-1} in 2017 and 2018, respectively)~~ was ~~in line with~~higher than values found by Tiemeyer et al. (2016) for productive and drained peatlands (-70 ± 18 t CO_2 ha^{-1} yr^{-1}) ~~and within the range of grasslands from Europe (45-78 t CO_2 ha^{-1} yr^{-1}) (Eze et al., 2018;Ma et al., 2015;Byrne et al., 2005)~~especially in the year 2017 (-88.7±7.2 t CO_2 ha^{-1} yr^{-1}), and falls back to the range in 2018 (-69.0±8.9 t CO_2 ha^{-1} yr^{-1}) due to the drought induced decline of CO_2 uptake (Fu et al., 2020). This could be simply explained by the high productivity of the sites, where the C-export in 2017 (on average 18.0 t CO_2 ha^{-1}) was substantially larger than the 8.5 t CO_2 ha^{-1} reported by Tiemeyer et al. (2016) for grassland on organic soils.

On the other hand, ~~T~~the R_{eco} values of the sites (~~131.5~~128.4±4.6 and 100.~~6~~8±11 t CO_2 ha^{-1} yr^{-1} in 2017 and 2018, respectively) are~~, however,~~ also at the higher end of the range (97 ± 33 t CO_2 ha^{-1} yr^{-1} in Tiemeyer et al. (2016)). ~~This leads to a relatively high NEE contributing to the generally large annual GHG budgets found in our study. There was, however, a large difference between 2017 and 2018 (-80.7 and -56.5 t CO_2 ha^{-1} yr^{-1}, respectively), which was due to the strong drought effect in 2018. In contrast to our expectations, no effect of SSI was found on GPP. The net GHG budgets from the current study (42.4 – 70.4 t CO_2-eq. ha^{-1} yr^{-1}) fall in the upper range of reported emissions from *drained?* temperate peatlands (Hiraishi et al. 2014, Wilson et al. 2016a). Intensively drained peatlands with productive grassland vegetation tend to emit more CO2 (40–70 t CO_2 ha^{-1} yr^{-1}) (Hoffmann et al., 2015;Tiemeyer et al., 2016;Wilson et al., 2016a;Tiemeyer et al., 2020) than IPCC Tier default values (Hiraishi et al. 2014). Emissions found in the current study were substantially higher than those reported earlier for drained peatlands in the Netherlands (20–25 t CO_2 ha^{-1} yr^{-1} in (Jacobs et al., 2007;Schrier-Uijl et al., 2014).~~ Extrapolation bias was excluded as a possible reason for this high CO_2 emission, since testing of different R_{eco} modeling approaches (including different model selection, data clustering procedure and removal of raw data outliers) did not yield substantially difference R_{eco} values (Järveoja et al., 2020) discovered in a boreal natural peatland strong diel patterns of R_{eco} with peaks at both midnight and midday, which could lead to overestimation of daily fluxes when models are developed with data collected around the peaks. Although this process is not clear for temperate productive peatland systems, representativeness of the campaign could be a reason for the high R_{eco} estimates. Besides the methodological speculations, ~~T~~there are also a number of biochemical reasons for the high emissions found here. Abiotic conditions that favor high CO_2 emissions were present, with high temperatures for both years and ~~optimal~~ non-limiting moisture conditions for 2017. Research from (Pohl et al., 2015) found a high impact of dynamic soil organic carbon (SOC) and N stocks in the aerobic zone on CO_2 fluxes. In our case, the peat soils contained a high amount of C, especially in the upper 20 cm layer. This layer was also aerobic for long periods during the experiment, thus promoting C formation and transformation processes in the plant–soil system.

4.4 Uncertainties

GHG emissions on peat grasslands are highly variable (Tiemeyer et al., 2016) given the uncertainties from the wide ranges of land use and management activities (Renou-Wilson et al., 2016) and gap filling techniques (Huth et al., 2017). In this study, ~~only~~ besides the model errors inherent in the model development process, uncertainties from gap-filling techniques in terms of data-pooling strategies and model selections were also considered.

Campaign-wise fitting of R_{eco} and GPP models can best represent the original data sets, while pooling data for a longer period can provide better model fitness and less bias toward single measurements (Huth et al., 2017;Poyda et al., 2017). However, in this study, different responses of vegetation and soil processes to drought, especially to the extreme drought in 2018, caused ~~abnormal data points that do not fit~~data points that could not be represented by the classic models, resulting in the generally poor performances of annual models. For this reason, we reported the annual budgets with campaign-wise gap-filled NEE values. The uncertainties of NEE estimates from model differences were on average 14 tons and up to 25 tons of CO_2. Nevertheless, no SSI effect was found considering NEE estimates from annual models. The model differences quantified here were in good agreements with other model tests (Karki et al., 2019;Görres et al., 2014) and match the magnitude of NEE uncertainties calculated with other methods (e.g. the 23–30 tons CO2 variances reported by (Schrier-Uijl et al., 2014) using eddy co-variance techniques).

4.5 Costs and benefits of the SSI

The intensity of land use (intensity and timing of drainage and fertilization, plant species composition, mowing and grazing regimes) influence the grassland's ability to accumulate or lose C (Renou-Wilson et al., 2016;Smith, 2014;Ward et al., 2016). SSI can increase the load-bearing capacity of the field surface for fertilizing equipment, facilitating earlier fertilization compared to management under current drainage systems. This can also cause increased leaching of water due to earlier drainage in a wet spring. However, the general land-use intensity will not change with the use of SSI. It was expected that C-export via crop yields due to extra drainage could increase in a wet autumn. However, we did not find any indication for an increase in land-use intensity or yield as a result of SSI.

The use of SSI is considered impractical for use in most regions outside of the Netherlands due to the high investment costs for irrigation pipes and the intensive water infrastructure needed for controlling the water level. In addition, irrigation pipes will increase the water demand in summer for these agricultural fields. Both land-use intensity and an increase in yield are related to an increase in CO_2 emissions on drained peat (Beetz et al., 2013;Couwenberg, 2011). The land-use history of our sites favors high CO_2 emission: tillage (cultivators, sod-renewal, and some plowing), cumulative fertilization and well-maintained drainage (Provincie Fryslân ,2015).

5 Main ~~conclusion~~conclusions

~~Unfortunately, t~~The implementation of SSI technique with the current design does not lead to a reduction of GHG emissions from drained peat meadows, even though there was a clear increase in GWT during summer (especially in the dry year of 2018). We therefore conclude that the current use of SSI with the aim to raise the water table to -60 cm is ineffective as a mitigation measure to sufficiently lower peat oxidation rates and, therefore, also soil subsidence. Most likely, the largest part of the peat oxidation takes place in the top 70 cm of the soil, which stays above the GWT with the use of SSI. This layer is still exposed to higher temperatures, sufficient moisture, oxygen and alternative electron acceptors such as nitrate, and nutrient input. We expect that SSI may only be effective when the GWT can be raised permanently to levels close to the soil surface (-20–35 cm below the surface).

Data availability. The data are available on request from the corresponding author, (S.T.J. Weideveld).

CRediT authorship contribution statement:

SW: Investigation, Data curation, Writing – original draft, Visualization, Methodology. WL: Investigation, Data curation, Writing – original draft, Visualization. MB: Data curation, Writing – original draft, Visualization. LL: Writing - review & editing, Supervision.CZ: Conceptualization, Methodology, Writing - original draft, Supervision

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Appendix A ~~annual~~Annual -models

Table A1. Model selected for annual-model gap-filling approach of year budgets (adopted from Karki et al. 2019), as a measure of extrapolation uncertianties.

Model		Structure	Description
R_{eco}	1	$Reco_{T_{ref}} * e^{E_0*\left(\frac{1}{T_{ref}-T_0}-\frac{1}{T-T_0}\right)}$	Arrhenius function as used for the campaign-wise model fit. Parameters follow descriptions in Material and Methods.
	2	$(Reco_{T_{ref}} + (\alpha * GH)) * e^{E_0*\left(\frac{1}{T_{ref}-T_0}-\frac{1}{T-T_0}\right)}$	Model 1 adding GH (grass height) as a vegetation factor. α is a scaling parameter of GH.
	3	$Reco_{T_{ref}} * e^{E_0*\left(\frac{1}{T_{ref}-T_0}-\frac{1}{T-T_0}\right)} + (\alpha * GH)$	Different form of vegetation included Model 1.
	4	$R_0 * e^{bT}$	Exponential function. R_0 is respiration at 0 °C, b is a temperature sensitivity parameter.
	5	$(R_0 + (\alpha * GH)) * e^{bT}$	Model 4 with vegetation included.
	6	$R_0 + (b * T) + (\alpha * GH)$	Linear function.
GPP	1	$\frac{\alpha * PAR * GPP_{max}}{GPP_{max} + \alpha * PAR}$	Michaelis-Menten light response curve as used for the campaign-wise model fitting.
	2	$\frac{\alpha * PAR * GPP_{max} * GH}{GPP_{max} * GH + \alpha * PAR} * FT$	Model 1 with vegetation and air temperature included. FT is a temperature dependent function of photosynthesis set

			to 0 below - 2 °C and 1 above 10 °C and with an exponential increase between - 2 and 10 °C.
	3	$\frac{GPP_{max} * PAR}{\kappa + PAR} * (\frac{GH}{GH + a})$	Another form of the Michaelis-Menten light response curve with a vegetation term included. a is a model-specific parameter.
	4	$\frac{GPP_{max} * PAR}{\kappa + PAR} * \left(\frac{GH}{GH + a}\right) * FT$	Model 3 with air temperature included.

Appendix B – Soil type decryption

Table B1 Soil characteristics of the research sites in Frisian peat meadows. Average per soil type, gravimetric soil moisture content taken August 2017, Dry bulk density, Organic matter content, and elemental Carbon content.

Farm	Treatment	Soil type	Depth	Soil moisture %	Bulk density g DW/cm3	Organic matter g Org/L	Carbon content g C/L
A	SSI	Mineral	0-35	38.1	0.99	122.6	52.3
		Peat	35-60	77.1	0.23	144.0	77.1
		Peat	60-80	82.1	0.14	130.4	67.9
	Control	Mineral	0-40	37.6	0.93	130.1	53.6
		Peat	40-60	59.2	0.24	156.0	82.9
		Peat	60-80	85.3	0.16	153.8	98.1
B	SSI	Peat	0-20	51.0	0.44	270.3	107.6
		Peat	20-60	79.3	0.19	168.9	76.6
		Peat	60-80	88.4	0.12	118.3	59.9
	Control	Peat	0-20	50.1	0.49	273.4	138.3
		Peat	20-60	77.7	0.17	140.6	72.0
		Peat	60-80	86.5	0.13	122.0	66.9
C	SSI	Mineral	0-30	36.0	0.71	127.9	58.2
		Schalter	30-40	79.2	0.19	176.9	87.5
		Peat	40-60	82.2	0.18	128.5	64.2
		Peat	60-80	87.5	0.11	132.9	81.4
	Control	Mineral	0-30	38.0	0.75	141.7	59.2
		Schalter	30-40	78.7	0.19	176.9	92.4
		Peat	40-60	84.3	0.12	116.3	59.9
		Peat	60-80	89.2	0.10	133.6	71.5
D	SSI	Mineral	0-30	37.7	0.85	154.5	73.7
		Schalter	30-40	63.9	0.30	266.5	85.2
		Peat	40-60	84.3	0.19	137.0	73.1
		Peat	60-80	80.2	0.14	129.6	54.6
	Control	Mineral	0-25	32.9	0.82	140.7	73.3
		Schalter	25-35	70.0	0.27	172.6	85.9
		Peat	35-60	84.1	0.15	141.9	82.7
		Peat	60-80	81.9	0.11	108.5	69.5

Appendix ~~B~~ C Reco ,GPP and NEE

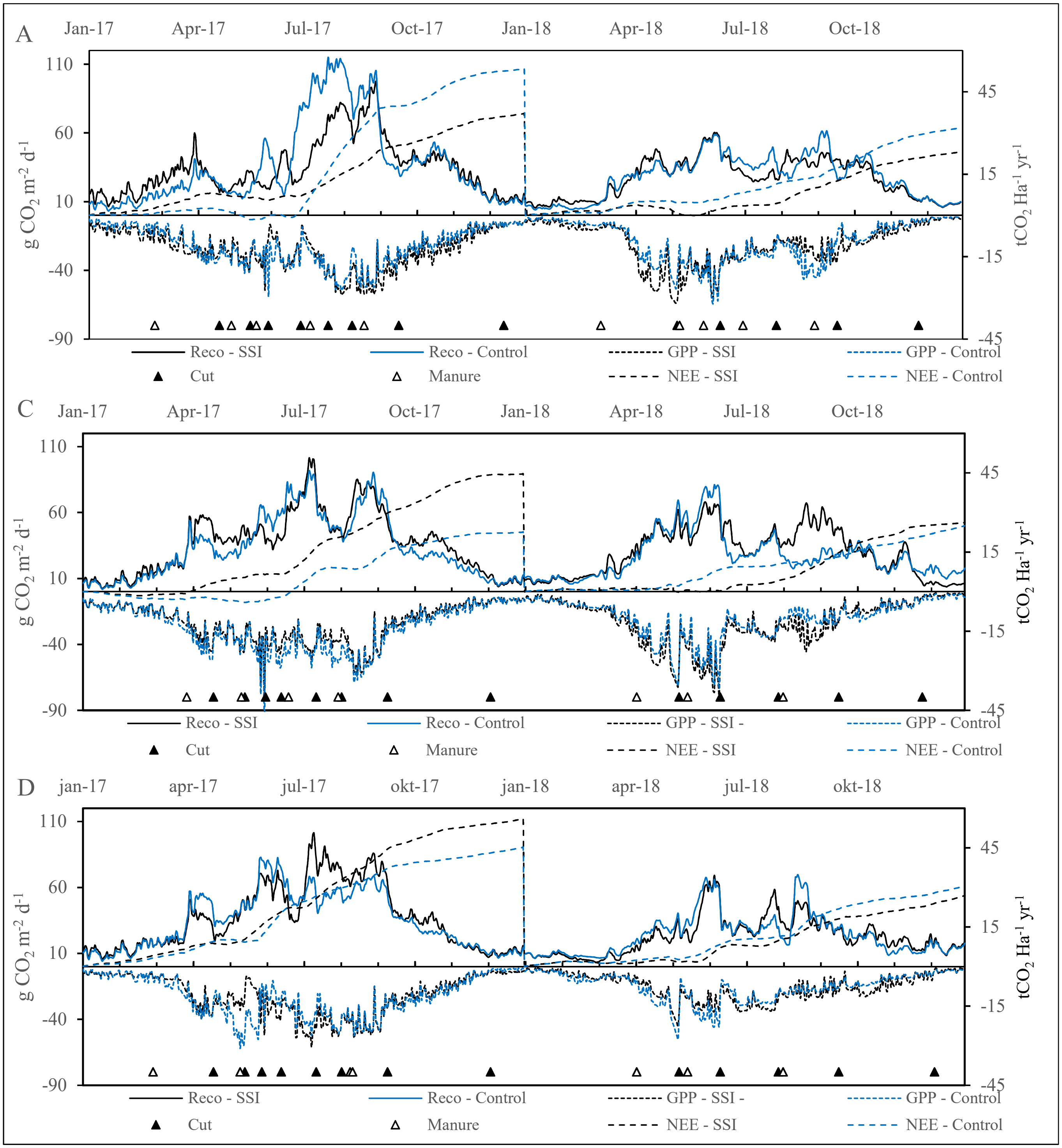

Figure C~~B~~1 Daily R_{eco} and GPP for location in g CO2 *m-1 d-1* on the primary y-axis, for control and SSI for locations A,C and D ~~Ger~~. Accumulative NEE in tCO2 Ha-1 yr-1, for control and SSI, every year starting at 0.

Appendix D CH_4 exchange

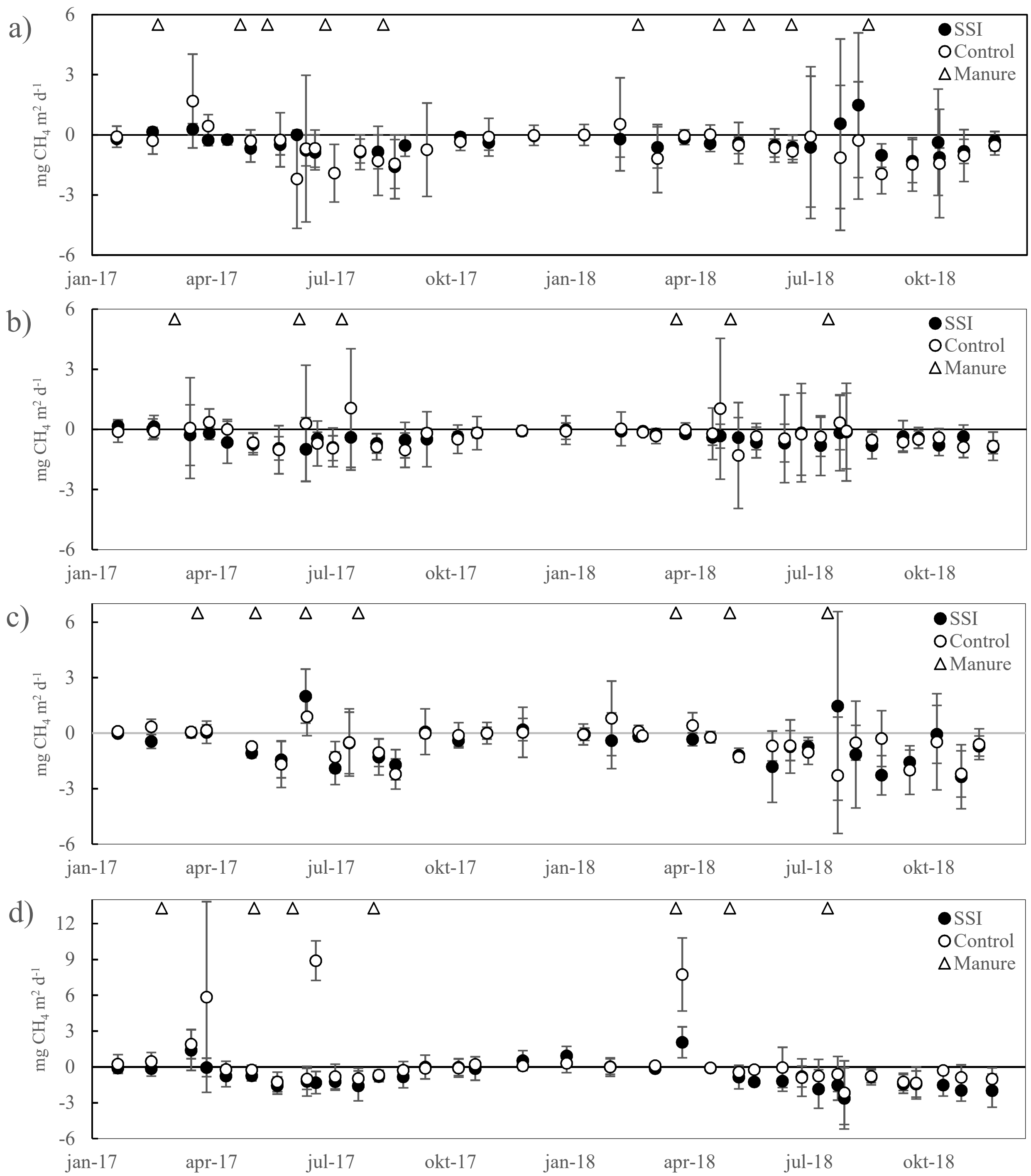

Figure D1 CH_4 exchange throughout 2017 and 2018 in mg CH_4 m^{-2} d^{-1}

Appendix ~~C~~ E N_2O exchange

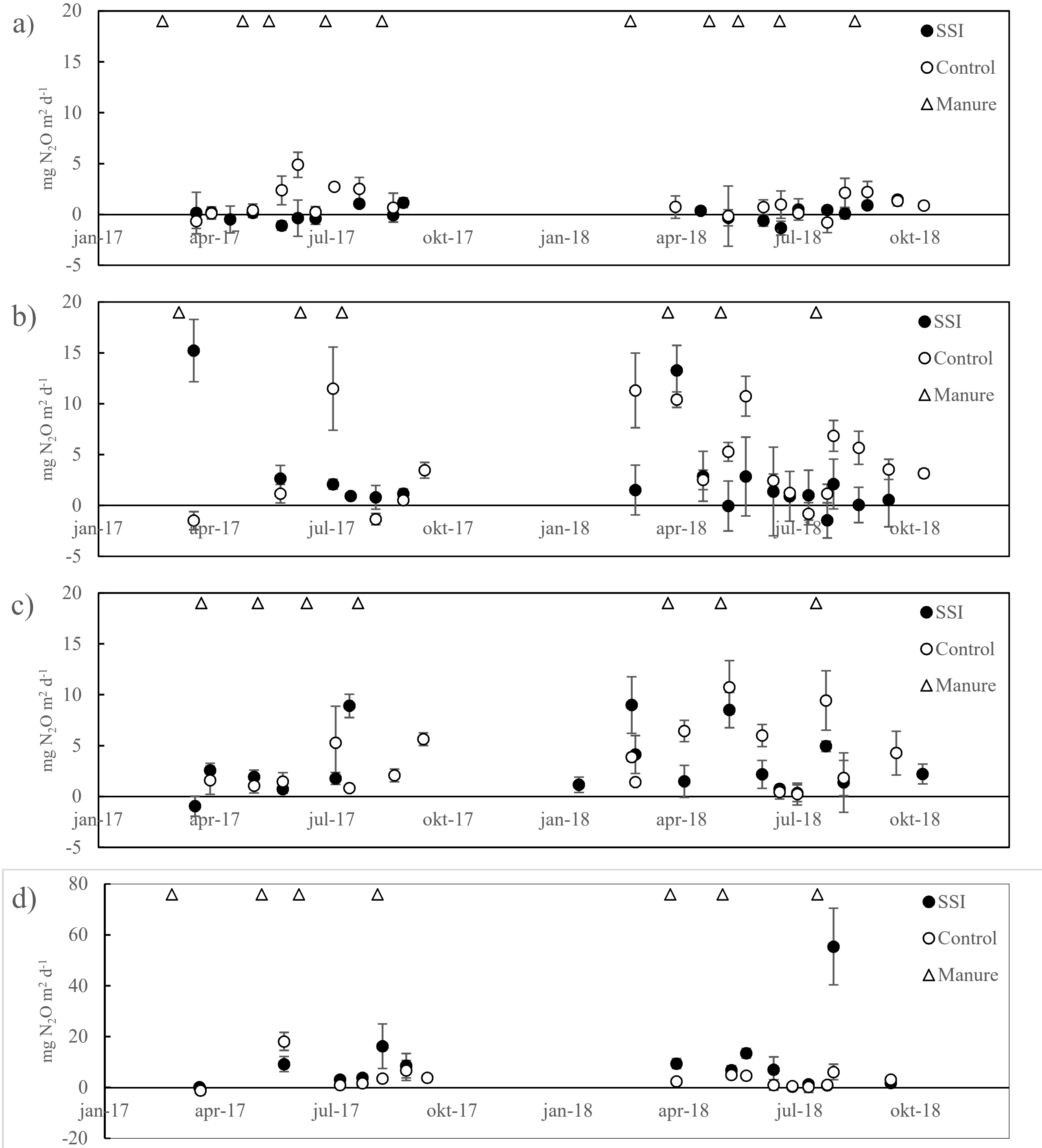

Figure E~~C~~1 N_2O exchange throughout 2017 and 2018 in mg N_2O m^{-2} d^{-1}.

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