Conventional sub-soil irrigation techniques do not lower ~~greenhouse gas~~carbon emission from drained peat meadows

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Abstract

The focus of ~~C~~current water management in drained peatlands is to facilitate ~~agricultural use~~optimal drainage, which has~~,~~ le~~a~~d~~s~~ to soil subsidence and a strong~~ly~~ ~~increases~~ increase~~d~~ of greenhouse gas (GHG) emission. ~~High-density,~~The Dutch land and water authorities proposed the application of sub-soil irrigation (SSI) system~~s~~ on a large scale ~~in high density~~ ~~have been proposed~~ to potentially reduce GHG emission~~as a potential climate mitigation measure~~, while maintaining high biomass production. ~~In summer, SSI was~~Based on model results, ~~They~~the expect~~ed~~ation was that SSI would~~to~~ reduce peat decomposition in summer by preventing groundwater tables (GWT) to drop below -60 cm. In 2017–~~-~~2018, we evaluated the effects of 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 SSI installation and a control site drained only by ditches (ditch water level -60/-90 cm, 100 m distance between ditches). The SSI system consisteds of perforated pipes~~ at~~ 70 cm below ~~ground~~surface~~soil~~ level with spacing of 5–~~-~~6 m to improve ~~both~~ drainage during ~~(~~winter–spring~~)~~ and irrigation in ~~(~~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 and N_2O. ~~C inputs by manure and C export by grass yields were accounted for. Unexpectedly, 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~~ Measured ecosystem respiration (R_{eco}) ~~did not show significant difference between SSI and control sites only except~~only showed a small difference between SSI and control sites when the ~~groundwater table~~ (GWT) of SSI sites ~~was~~ were substantially higher than the control site~~value~~ (> 20 cm difference).~~,~~ Over all years and locations, however, there was no significant difference found, despite the 6–18 cm higher GWT in summer and 1–20 cm lower GWT in wet conditions at SSI sites ~~on average in the four farm locations~~. Differences in mean annual GWT remained low (< 5 cm). Direct comparison of measured N_2O and CH_4 fluxes between SSI and control sites did not show any significant differences. CO_2 fluxes varied according to temperature and management events while differences between control and SSI sites remained small. ~~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 SSI and control sites on an annual base. CO_2 fluxes were high at all locations, ranging from 35 – 66 and 20 – 50 t CO_2 ha^{-1} 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, Overall~~Therefore, there was no difference between the annual ~~CO_2 fluxes~~gap-filled net ecosystem exchange (NEE) ~~GHG fluxes~~ of the ~~sub-soil irrigated~~SSI and control site. The net ecosystem carbon balance was on average ~~(~~40 and 30 t CO_2~~-eq~~ ha^{-1} yr^{-1} in 2017 and 2018 on the SSI sites ~~)~~ and ~~control sites (~~38 and 34 t CO_2~~-eq~~ ha^{-1} yr^{-1} in 2017 and 2018 on the control sites~~)~~. ~~SSI did not affect average N_2O emissions, although the number of flux measurements were limited to draw strong conclusions about the treatment effects.~~

~~Emissions of N_2O were lower with average emissions were measured (2.9±1.8 mg N_2O. m^{-2} d^{-1} for 2017) in 2017 than in 2018 (3.6±3.3 mg N_2O. m^{-2} d^{-1}), without treatment effects. No SSI effect was detected for CH_4, Tthe contribution of CH_4 to the total GHG budget was negligible (<0.1%), with lower GWT favoring CH_4 oxidation over its production. Moreover, NEE was summed up with C inputs by manure and C export by grass yield into net ecosystem carbon balance (NECB), whereas Nno treatment effect was found on yields, suggesting that the increased GWT in summer did not increase plant water supply.~~ This lack of SSI effect is probably because the GWT increase remains limited to ~~s only takes place in~~ deeper soil layers (60–~~-~~120 cm depth), which ~~also indicates that~~~~could hardly affect~~ ~~contribute~~contribute little to peat oxidation ~~is hardly affected~~~~and plant water supply~~.

We conclude that~~, although our field-scale experimental research revealed substantial differences in summer GWT and timing/intensity of irrigation and drainage,~~ SSI modulates water table dynamics but fails to lower annual ~~GHG~~ carbon emission. SSI ~~and is~~seems 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 and increased gas exchange ~~resulting in soil subsidence and the concomitant release~~releasing ~~of~~ CO_2 and N_2O at high rates (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 including microbial breakdown of organic matter. 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;Herrera-García et al., 2021). Additionally, peatland subsidence alters hydrology on various scales, which lead~~ing~~ to frequent drainage failure~~ problems~~~~difficulties~~, ~~salt water~~saltwater intrusion and loss of productive lands (Dawson et al., 2010;Herbert et al., 2015). ~~This will result in strongly~~ Ongoing peatland subsidence ~~increased~~ inflict high 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% for Canada, and varies from less than 5 to more than 80% ~~or~~ in European countries (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~~ accounting for >25% of total ~~to 4% of total national~~ greenhouse gas (GHG) emissions from Dutch agriculture (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~~CO_2 emissions from peatlands are related to the ~~water table position~~GWT position below surface, which affects oxygen intrusion, moisture content and temperature. There is ample evidence that elevating ~~water levels~~GWT to 0-20 cm below ~~the land~~ surface results in substantial reduction of CO_2 emissions from (formerly) managed peatlands (Hendriks et al., 2007;Hiraishi et al., 2014;Jurasinski et al., 2016;Tiemeyer et al., 2020) Increasing ~~water levels~~GWT close to the surface does not only ~~worsens conditions for~~constrains aerobic CO_2 production and rapid gas exchange but also reduces land-use intensity (fertilization, tillage, planting, grazing). Additionally, high ~~water levels~~GWT could favor vegetation assemblages with a higher carbon sequestration potential (e.g. peat forming plants) compared to common fodder grasses and crops. Experimental ~~research using~~ studies on water table man~~u~~ipulation~~s~~ stressed~~s~~ 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 correlation of CO_2 emissions with GWT based on a meta-analysis of field CO_2 emission data by Tiemeyer et al. (2020).

Dutch water- and land-authorities have relied on ~~height~~ ground surface elevation measurements to estimate peat loss~~of the peat surface~~ rather than CO_2 flux measurements to ~~estimate~~ ~~come to~~calculate CO_2 emissions from peatlands (Arets et al., 2020) and the effects of elevated ~~water levels~~GWT on CO_2 emissions. Two assumptions are generally made when inferring ~~translating~~ surface elevation data into CO_2 emission from surface elevation changes: 1) Elevation changes are directly related to C losses from peatlands within a time frame of years ignoring physical changes of peat following drainage. As a conversion factor 2.23

t CO_2 ha^{-1} per mm subsidence is assumed (Kuikman et al., 2005;Van den Akker et al., 2010). 2) The average lowest summer GWT (GLG) is assumed to be a major control factor of subsidence rates of peat surface elevation and henceforth CO_2 emissions based on the first assumption above (Arets et al., 2020). As a consequence of both assumptions, Dutch climate mitigation frameworks focus on elevating summer GWT in peatlands rather than mean annual GWT (Querner et al., 2012;Brouns et al., 2015). Dutch water- and land-authorities expect that increasing the average lowest summer GWT by 20 cm would result in an emission reduction equalling 10.5 t CO_2 ha^{-1} yr^{-1} (Van den Akker et al., 2007;Brouns et al., 2015;Van den Born et al., 2016).

The use of subsoil irrigation and drainage systems (SSI) ~~SSI systems has~~ have been proposed to elevate summertime GWT and thereby presumably reducing CO_2 emissions ~~since the early 2000's~~ (Van den Akker et al., 2010;Querner et al., 2012). SSI works by installing ~~perforated~~ drainage/irrigation pipes at around 70 cm below the surface ~~and~~ or at least 10 cm below the ditch water level. Water from the ditch can infiltrate into the ~~adjacent~~ peat adjacent to ~~drainage~~SSI pipes and thereby limite GWT drawdowns during summer (c.f. (Hoving et al., 2013). Next to irrigation, ~~drainage~~ SSI pipes primarly fulfill a drainage function when the GWT is above the ditch water level. Based on the elevating effects on summer groundwater table SSI was assumed to reduce of C emissions from peatlands by 50% according to the soil-carbon-water model (Querner et al., 2012;Van den Born et al., 2016). However, th effect of SSI on C emissions has nog yet been tested by field measurements of C-fluxes. ~~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 directly into CO2 emission~~

The aim of our study was to quantify the effects of SSI on the GWT and GHG emissions, ~~in particular~~ with consideration of the farm field net ecosystem carbon balance (NECB). We questioned 1) to what extent can SSI regulate GWT, especially during dry conditions in summer, 2) whether the SSI can substantially reduce (up to 50% as assumed by authorities) 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.

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~~combined with ~~large~~ wide fields with deep drainage.~~,~~ O~~as o~~ne 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, which is highly laminated peat (compacted/ hydrophobic layers of *Sphagnum cuspidatum* remnants) with poor degradability and poor water permeability. The grasslands ~~are~~ were dominated by *Lolium ~~Perenne~~perenne*; other species such as *Holcus lanatus*, *Elytrigia repens*, *Ran~~o~~unculus acris* and *Trif~~v~~olium repens* ~~are~~ were present in a low abundance in 2017-2019.

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.~~ Averages per soil type, gravimetric soil moisture content taken August 2017, Dry bulk density, Organic matter content, and elemental Carbon content.

~~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~~	~~53.4~~	~~29.2~~
~~-~~	~~-~~	~~-~~	~~Control~~	~~0.6~~	~~0.40~~	~~-~~	~~2.0~~	~~141.2~~	~~47~~	~~19.8~~
~~B~~	~~Conventional~~	~~Grazing~~	~~SSI~~	~~2.3~~	~~-~~	~~-~~	~~1.4~~	~~190.7~~	~~68.1~~	~~34.6~~
~~-~~	~~-~~	~~-~~	~~Control~~	~~2.3~~	~~-~~	~~-~~	~~1.4~~	~~175.9~~	~~74.9~~	~~32.8~~
~~C~~	~~Conventional~~	~~Mowing~~	~~SSI~~	~~1.2~~	~~0.30~~	~~yes~~	~~1.3~~	~~141.7~~	~~56.3~~	~~23~~

~~-~~	~~-~~	~~-~~	~~Control~~	~~1.8~~	~~0.30~~	~~yes~~	~~1.0~~	~~133.4~~	~~60.5~~	~~23.5~~
~~D~~	~~Conventional~~	~~Mowing~~	~~SSI~~	~~2.4~~	~~0.30~~	~~yes~~	~~0.9~~	~~161.9~~	~~59.6~~	~~23.3~~
~~-~~	~~-~~	~~-~~	~~Control~~	~~3.5~~	~~0.25~~	~~yes~~	~~0.9~~	~~151.5~~	~~63.4~~	~~26.9~~

Farm	Treatment	Field size	peat thickness	Soil type	Soil depth	Soil moisture	Bulk density	Organic matter	Carbon content	Carbon content
		ha	m		cm	%	g DW/cm3	g Org/L	g C/l	g C/kg
A)	SSI	2 ha	1.6 m	Mineral	0 – 35	38.1	0.99	123	52	53
Organic				Peat	35 – 60	77.1	0.23	144	77	335
Grazing				Peat	60 – 80	82.1	0.14	130	68	485
	Control	0.6 ha	2 m	Mineral	0 – 40	37.6	0.93	130	54	58
				Peat	40 – 60	59.2	0.24	156	83	345
				Peat	60 – 80	85.3	0.16	154	98	613
B)	SSI	2.3 ha	1.4 m	Peat	0 – 20	51	0.44	270	108	245
Conventional				Peat	20 – 60	79.3	0.19	169	77	403
Grazing				Peat	60 – 80	88.4	0.12	118	60	499
	Control	2.3 ha	1.4 m	Peat	0 – 20	50.1	0.49	273	138	282
				Peat	20 – 60	77.7	0.17	141	72	424
				Peat	60 – 80	86.5	0.13	122	67	515
C	SSI	1.2 ha	1.3 m	Mineral	0 – 30	36	0.71	128	58	82
Conventional				Schalter	30 – 40	79.2	0.19	177	88	461
Mowing				Peat	40 – 60	82.2	0.18	129	64	357
				Peat	60 – 80	87.5	0.11	133	81	740
	Control	1.8 ha	1 m	Mineral	0 – 30	38	0.75	142	59	79
				Schalter	30 – 40	78.7	0.19	177	92	486
				Peat	40 – 60	84.3	0.12	116	60	499
				Peat	60 – 80	89.2	0.1	134	72	715
D	SSI	2.4 ha	0.9 m	Mineral	0 – 30	37.7	0.85	155	74	87
Conventional				Schalter	30 – 40	63.9	0.3	267	85	284
Mowing				Peat	40 – 60	84.3	0.19	137	73	385
				Peat	60 – 80	80.2	0.14	130	55	390
	Control	3.5 ha	0.9 m	Mineral	0 – 25	32.9	0.82	141	73	89

				Schalter	25 – 35	70	0.27	173	86	318
				Peat	35 – 60	84.1	0.15	142	83	551
				Peat	60 – 80	81.9	0.11	109	70	632

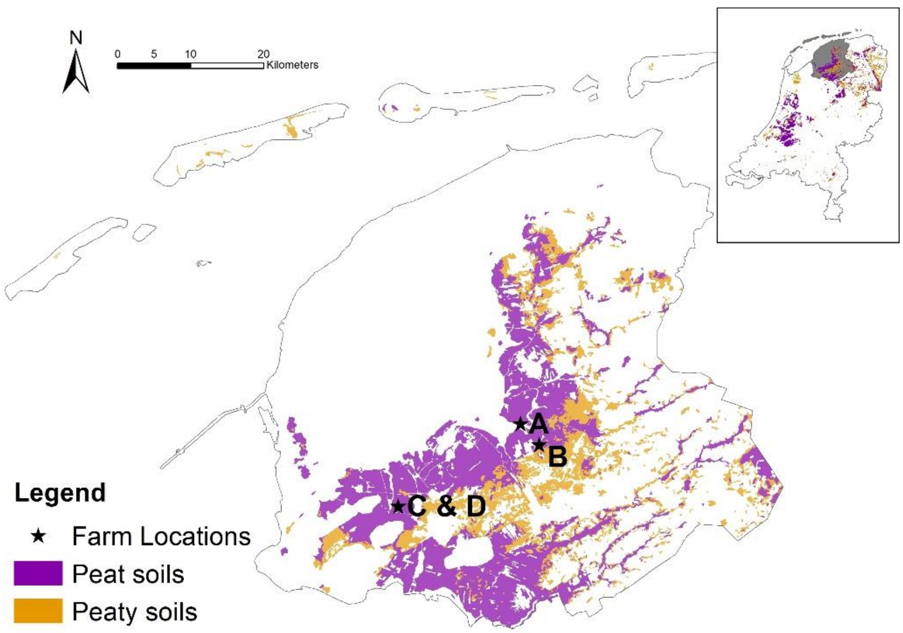

Figure 1 Soil map with ~~F~~field locations situated in the province of Friesland, ~~of~~ the Netherlands~~, with soil types~~. Peat soils refer to soils with an organic layer of at least 40 cm within the first 120 cm, while peaty soils are soils with an organic layer of 5-40 cm within the first 80 cm. ~~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 SSI pipes and a control site. The ~~irrigation~~ SSI pipes were installed at a depth of 70 cm below the surface and 5–6 m ~~(2,000 m drains ha-1)~~ apart from each other, except for the D location where pipes were 5 m apart. The overall drainage intensity was around 2000 m ha-1. The pipes were either directly connected to the ditch (A and C) or connected to a ~~collection~~ collector tube ~~before connected into the~~that was connected to a ditch (B and D). The connections with ditches were placed 10 cm below the ~~maintained~~ targeted ditchwater level that was regulated by a complex network of water inlet and pumps at the lowest parts of the polder. The control sites are fields that have traditional drainage~~, through a system~~ with deep drainage ditches with convex fields and small shallow ditches (furrows).

On the treatment sites, three gas measurement frames ~~in 80x80~~ 80 × 80 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~~ SSI pipe (Fig. 2), representing best the variation in the environmental conditions and vegetation. The control sites where located 32 – 42 m from the ditch. Dip well tubes were installed to monitor water ~~level~~tables 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 (130-150 cm length) were placed in the peat layer. The tube 1.5 m from the ~~irrigation~~SSI 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.

To determine soil properties, ~~S~~soil samples were taken using a gouge auger in three replicates ~~till …~~0.8 m depth ~~where taken~~, ~~from~~ 1.5 meter from the ~~irrigation~~ SSI pipes ~~every… moment in year~~taken in august 2017~~?~~. ~~To determine moisture content, s~~For soil moisture, sediment samples were weighed and subsequently oven-dried at 105°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) during the run time of the experiment and recorded every 5 minutes 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 -5 and -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 relative 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 3 x 3 km

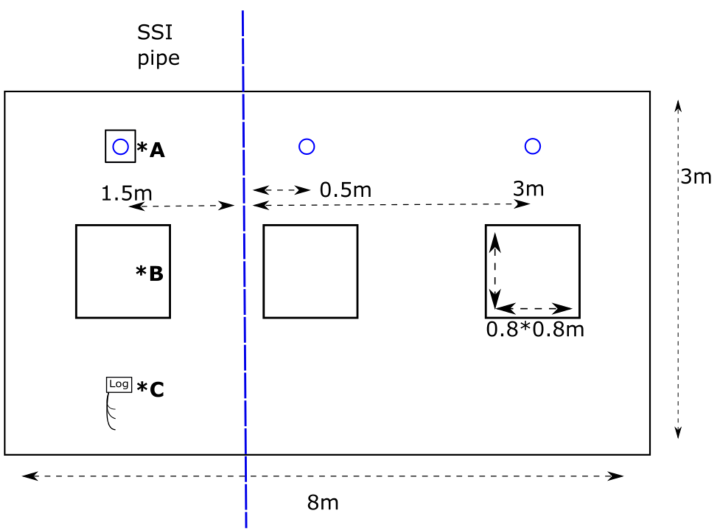

Figure 2 Overview field site SSI. Blue dashed line = ~~irrigation~~ SSI pipe, blue circle = dipwell, *A – dipwell with data logger, *B – ~~gas~~ greenhouse gas flux measurement frame, *C – data logger, -5 -10 -20 soil temperature ~~and soil moisture~~

C-export from frames used GHG measurements was determined by harvesting ~~the frames~~ the standing biomass eight times in 2017 and five times in 2018.~~,~~ ~~t~~Two ~~extra~~of the harvest moments ~~where implemented~~ in 2017 were extra planned, once in May ~~to~~ because of the fast grass growth and ~~extreme~~ grass height exceeding 30 cm, and the other in December ~~to come back to the~~in order to reset the grass height ~~at~~to the start of the experiment for next year. ~~The surrounding~~Surrounding the frames ~~A~~an area of 8 x~~×~~ 3 m was fenced off ~~field site surrounding the measurement frames~~ to avoid disturbance from grazing and other field ~~managements~~activities (Fig. 2). ~~This~~The ~~field~~fenced-off area outside the frames was~~the whole field site were~~ managed with ~~4~~ 5 cuts per year to have a similar grass height with the farmland ~~the surrounding field~~. The biomass was harvested, ~~where~~ weighed for fresh weight 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). 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

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. The N_2O emissions w~~h~~ere 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-30EP, 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~~ were 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~~ Grass height was measured using a straight scale with a plastic disk with a diameter of 30 cm before starting the measurement campaign.

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):

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

W~~w~~here 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 (Beetz et al., 2013;Kandel et al., 2016). 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):

$$GPP = \frac{\alpha * PAR * GPP_{max}}{GPP_{max} + \alpha * PAR} \tag{3}$$

where ~~NEE~~ GPP is the ~~measured~~ ~~corrected~~ CO_2 flux~~ with light chamber~~ measured with ~~light~~transperant chambers and corrected with R_{eco}, α is ecosystem quantum yield (mg CO_2 - C m^{-2} h^{-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. 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.

2.4.4 ~~NECB~~ Net ecosystem carbon balance calculations

The NEE is the sum of R_{eco} and GPP values, calculated by applying the hourly monitored soil temperature (-5 cm) and PAR data to the models developed per campaign. 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. 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. The loss of biomass was therefore accounted according to lowered grass height, different from the studies where model parameters are to zero after harvest (e.g. Beetz et al. 2013). 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 w~~h~~ere 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.

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~~ were 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).. To quantify this uncertainty, six

R_{eco} models and four GPP models were select from Karki et al. (2019)) and fitted with annual data (Appendix Table A1). The models were evaluated following the thresholds of performance indicators in Hoffmann et al. (2015). Fitted parameters of R_{eco} and GPP models that ~~were~~ performed above the 'satisfactory' rating ~~was~~ were accepted and ~~calculated into~~used to gap-fill~~ed~~ 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.

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 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 (K~~M~~NM~~N~~I). There was a small period of drought in May and June, ending in the last week of June (See ~~see~~ Fig.3). In contrast, 2018 was a dry year with average precipitation of 546-611 mm (range of 2 sites in Friesland). 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.

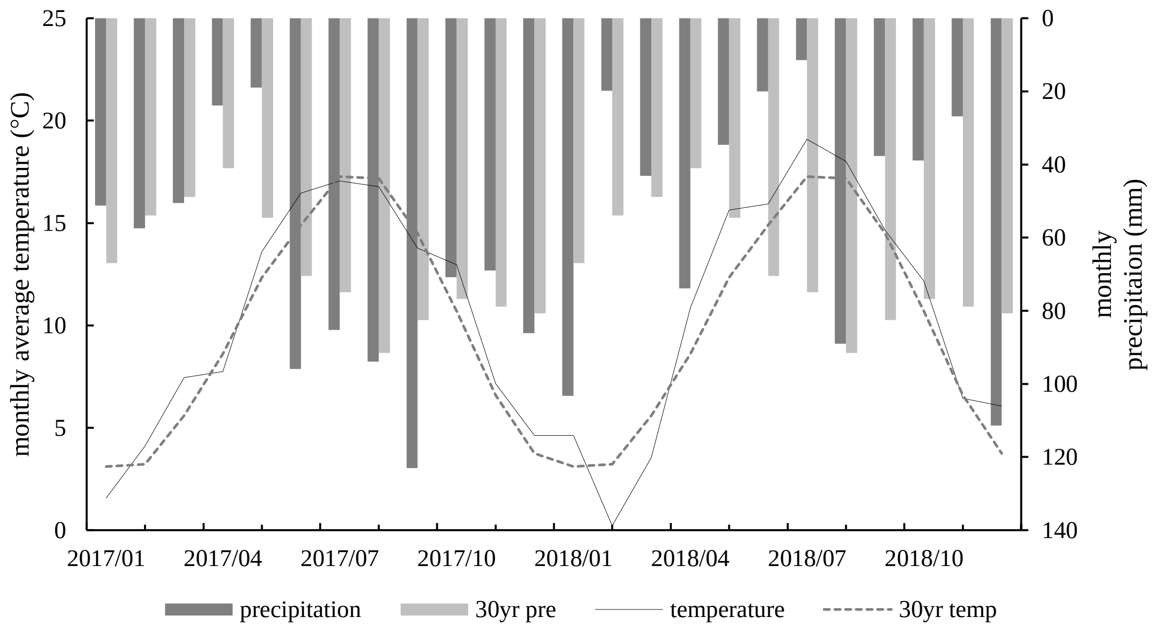

Figure 3 Monthly average air temperature at weather station Leeuwarden (18 to 30 km distance from research sites), and the 30-year average. Sum precipitation at weather station Leeuwarden, and the 30-year average.

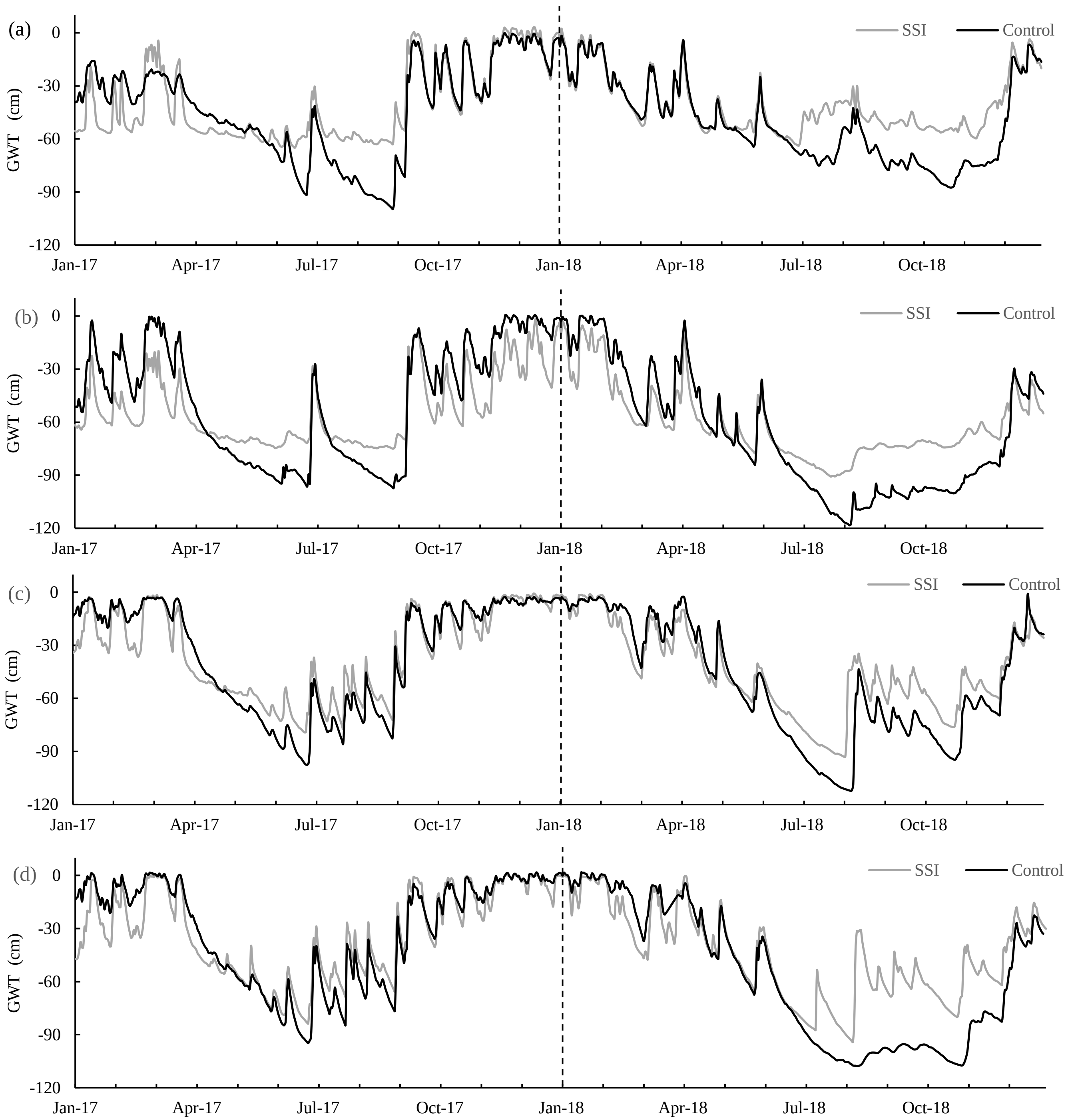

Figure 4 Groundwater table (GWT, from soil surface) during the measuring period per farm (letter), per graph SSI (measured 1.5 m from the irrigation pipe) and control.

3.2 Groundwater table (GWT)

Deploying SSI systems affected ~~the~~ GWT dynamics 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.

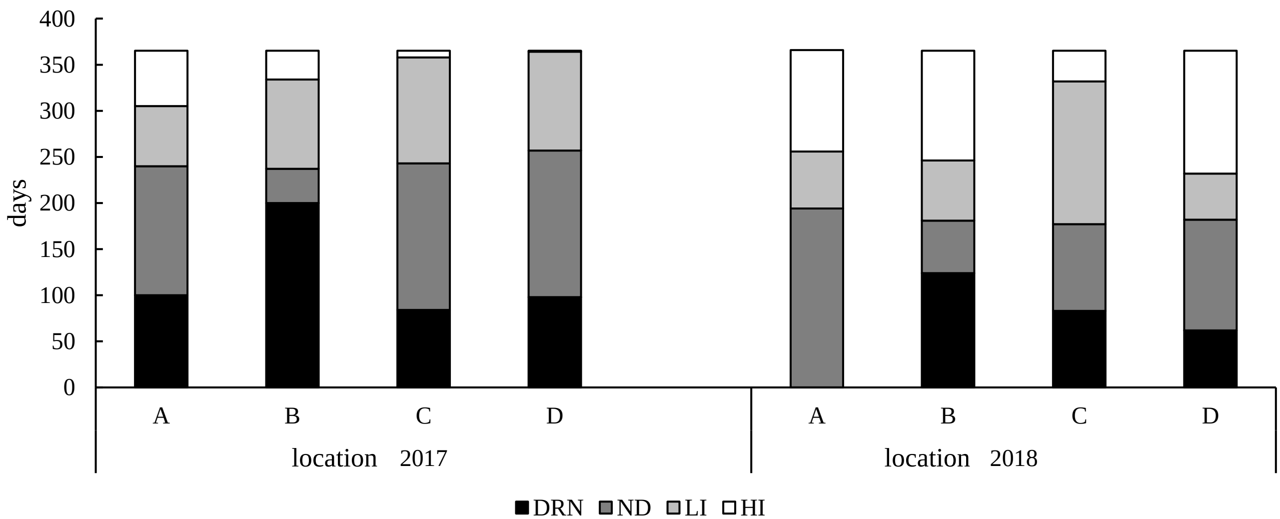

Figure 5 Days with effective drainage/ irrigation for the four locations. drainage (DRN, <-5 cm), no difference (ND, -5 ~ 5 cm), low to intermediate irrigation (LI, 5 ~ 20 cm) and high irrigation (HI, > 20 cm) 1.5 m from the ~~irrigation~~ SSI pipe.

Although there was hardly any difference in annual average GWT between control and SSI (< 5 cm; 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~~classes, 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~~

classes 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 Groundwater table (~~below~~cm from ~~soilground~~the surface level~~-~~) ~~groundwater table~~ during the measuring period per farm. Summer groundwater table ranges from April till October. Measured 1.5 meter from the ~~irrigation~~ SSI 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}

Figure. 6 compares the measured R_{eco} fluxes with the corresponding GWT measurements, which could give an indication for the effectiveness of the GWT differences. ~~The division between t~~The ~~groups~~classes where~~as~~ based on the GWT differences between the SSI and control sites on the measurement days (the same ~~groups~~ classes used in Fig. 5). There was a slightly higher R_{eco} for SSI during drainage periods when GWT was lower (DRN), which compensates for the lower R_{eco} during summer. For moments where there was no GWT difference (ND) and those showing moderate irrigation (LI), there was no effect of SSI on R_{eco}. However, when the GWT of the SSI was more than 20 cm higher than the control (HI), the emissions of the control w~~h~~ere 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, 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 periodes~~periods with increased irrigation (Fig. 5), when there was a reduced R_{eco} effect of SSI, were sparse compared to the other dominating periods.

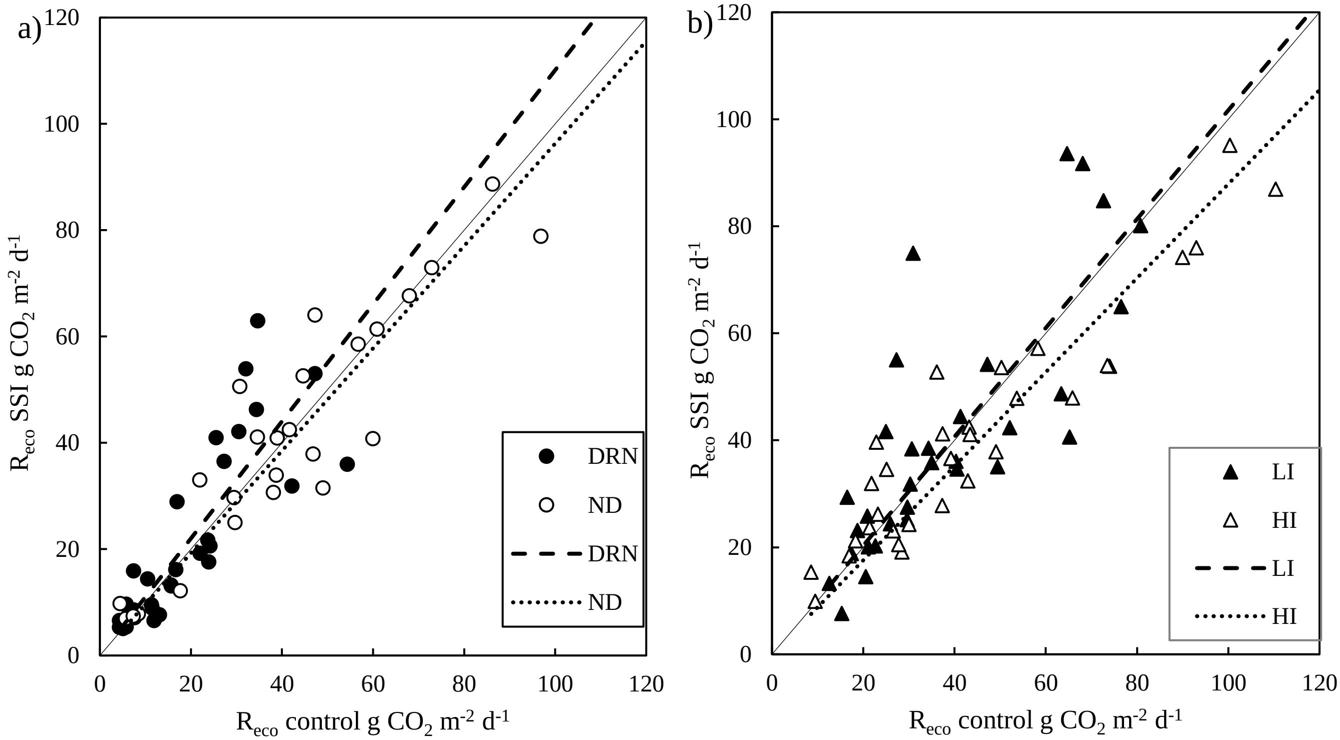

Figure 6 Measured fluxes for ecosystem respiration (R_{eco}), one-to-one comparison in which daily averages where used. a) Values divided into two groups: where the ground water table was lower due to the effect of drainage (DRN), and where there was a limited difference (ND). B) Values divided into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

3.4 Annual 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 $p = 0.006$). There was, however, no treatment effect on GPP ($p = 0.3101$). Average GPP values for all SSI and control plots were -88.3±7.5 and -89.2±13 t CO_2 ha^{-1} yr^{-1} for 2017, -71.7±6.6 and -65.7±4.9 t CO_2 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 128.4±4.6 t CO_2 ha^{-1} yr^{-1} for 2017 being significantly higher than 100.8±11 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 ($p = 0.6191$), 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.

3.4.3 Net ecosystem exchange (NEE)

All locations functioned as large C sources during the measurement period. The average annual NEE of all sites amounted to 39.7±11 and 31.8±8.4 t CO_2 ha^{-1} yr^{-1} in 2017 and 2018, respectively. The overall explanatory power of year, treatment and location was low, with no yearly difference between 2017 and 2018 ($p = 0.1813$), or any treatment effect of SSI ($p = 0.9805$). 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. ~~or.~~

3.4.4 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 4). Such exchange did not play a significant part in the total GHG emissions (comparable to less than 0.4% of the annual NECB).

3.6 Nitrous oxide exchange

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 peak was measured on the frame closest to the ~~irrigation~~ SSI pipe in August for SSI of location D, showing 55±15 mg N_2O m^{-2} d^{-1}. The peaks observed were erratic and did not correspond to fertilization management with slurry before measurement campaigns.

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 (harvest), C-manure (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					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	125.9~~123.1~~±3.4	-88.8~~-88.9~~±2.7	37.1~~34.2~~±4.4	16.6±0.4	-6.9±0.1	46.8±4.4
		Control	134.8~~133.7~~±6.5	-81.5~~-81.3~~±7.9	53.3~~52.4~~±10.2	19.3±0.7	-6.9±0.1	65.7±10.2
	B	SSI	125.2~~125~~±5.8	-97.8~~-98.5~~±3	27.4~~26.5~~±6.5	15.3±1.1	-5.3±0.1	37.4±6.6
		Control	123.4~~123.2~~±5.8	-92.2~~-92.6~~±2.9	31.2~~30.6~~±6.5	15.5±0.0	-5.3±0.1	41.4±6.5
	C	SSI	132.5~~132.1~~±4.6	-87.9~~-87.7~~±5.7	44.6~~44.5~~±7.4	22.1±0.2	-10.9±0.2	55.8±7.4
		Control	122.7~~122.3~~±3.2	100.1~~-100~~±8.3	22.6~~22.3~~±8.9	23.3±0.9	-10.9±0.2	35±8.9
	D	SSI	134.6~~134.5~~±4.2	-78.6~~-78.6~~±2.8	56~~56~~±5	15.7±1.4	-9.3±0.2	62.4±5.2
		Control	127.9~~127.9~~±2	-82.7~~-82.9~~±5.3	45.2~~44.9~~±5.6	16.3±0.6	-9.3±0.2	52.2±5.6
2018	A	SSI	98~~98.3~~±6.5	-74.9~~-74.7~~±2.5	23.1~~23.6~~±7	14±0.0	-7.4±0.1	29.7±7
		Control	101.1~~101.3~~±5.5	-69.3~~-68.9~~±3.1	31.9~~32.4~~±6.4	14±0.0	-7.4±0.1	38.5±6.4
	B	SSI	118.1~~117.5~~±10.1	-73.8~~-73.4~~±3.4	44.3~~44.2~~±10.7	13.8±0.6	-9.3±0.2	48.8±10.7
		Control	111.5~~111.4~~±10.5	-64.6~~-64.5~~±2.8	46.9~~46.9~~±10.9	12.2±1.2	-9.3±0.2	49.8±11
	C	SSI	109.2~~109.6~~±5.8	-83~~-82.4~~±4.6	26.2~~27.3~~±7.4	15.7±1.0	-9.3±0.2	32.6±7.5

	Control	99.2~~99.2~~±1.3	-74.2~~-73.7~~±0.6	25~~25.5~~±1.5	15.8±0.4	-9.3±0.2	31.5±1.6
D	SSI	82.9~~82.9~~±4.5	-56~~-56.1~~±2.2	26.9~~26.8~~±5	13.4±0.23	-9.3±0.2	31±5
	Control	86.5~~86.6~~±6.3	-55.9~~-55.5~~±2.4	30.6~~31.1~~±7	12±0.32	-9.3±0.2	33.3±7

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.

			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	0.00
	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

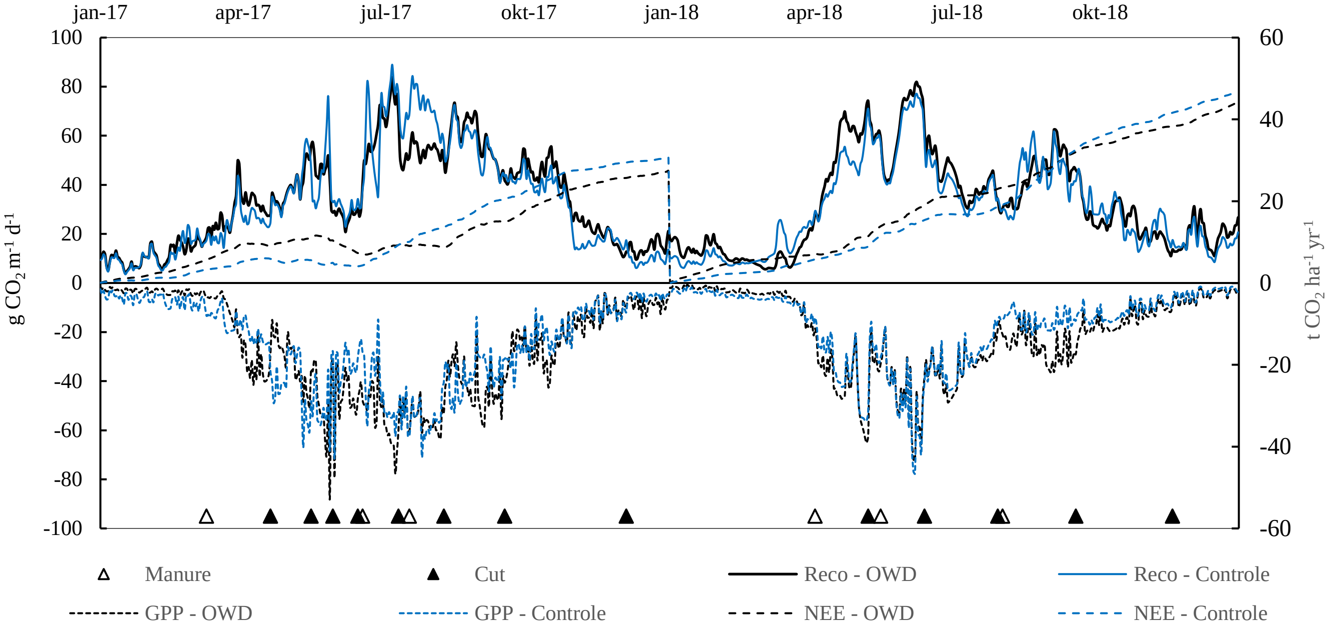

Figure 7 R_{eco} and GPP for location B in g CO_2 m^{-2} d^{-1} on the primary y-axis, for control and SSI. Accumulative NEE in t CO_2 ha^{-1} yr^{-1}, for control and subsoil irrigation (SSI), every year starting at 0.

4 Discussion

In this experimental research we found effects of subsoil irrigation (SSI) on water table dynamics without changing carbon dynamics profoundly. For both years, SSI had a clear irrigation effect during summer, increasing the averages of GWT during summer period 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. Mean annual GWT was little affected by SSI. Despite the irrigation effects and higher water ~~levels~~ tables in summer, there was no effect of SSI on R_{eco}~~-~~, GPP and NEE ~~on~~in neither of the two years ~~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~~yield improvements, on an annual base by implementing SSI.

4.1 SSI does not reduce annual R_{eco}

We identified ~~3~~three conditions that can explain the limited effect of SSI on carbon fluxes most prominent peat decomposition. Firstly, the uppermost 30-40 cm remain drained in both treatments throughout large parts of the year (220-255 days) facilitating increased CO_2 fluxes. Secondly, gas exchange from lower layers (60 cm and below) was presumably low due to moisture levels close to saturation that limit diffusion of CO_2 and O_2 effectively. Thirdly, the deliberate increase in drainage in the SSI treatment frustrate the irrigation effect on GWT. As a consequence, mean annual GWT was similar for both treatments.

Based on the direct comparison using measured R_{eco} fluxes (Fig. 6), we found a modest 5–10% reduction in R_{eco} only when GWT differences were larger than 20 cm. When the irrigation effect was smaller, no effect on the R_{eco} was found. An earlier study on intensively managed peat pastures in the Netherlands on the role of GWT also showed small effects of higher summer GWT on annual R_{eco} and NEE despite substantial differences in soil volume changes/soil subsidence (Dirks et al., 2000). Similarly, a 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 . However, most studies discuss the effect of annual average GWT, instead of seasonal changes in GWT.~~ It is generally assumed that higher GWT (mean annual or actual) -leads to lower CO_2 emissions according to laboratory data ~~(~~(Moore and Dalva, 1993)~~Moore and Dalva, 1993)~~ and correlations between annual CO_2 fluxes and mean annual GWT ~~(~~(Wilson et al., 2016;Tiemeyer et al., 2020)~~). In addition~~However, there are also studies that did not find an effect of GWT on CO_2 emissions during the growing season (Lafleur et al., 2005;Nieveen et al., 2005;Parmentier et al., 2009). This lack of effect is explained by the fact that there is only a small ~~variation~~ difference in soil moisture values above the GWT~~ between SSI and Control sites~~. ~~A large number of studies report~~The lower CO_2 emissions reported with~~when water levels were~~ structurally elevated GWT are often~~,~~ concomitant with substantial differences in vegetation/land use that are adapted to the higher GWT~~following higher water levels~~ (Beetz et al., 2013;Schrier-Uijl et al., 2014;Wilson et al., 2016), which could confound the effects of GWT change. 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~~small treatment effect on measured R_{eco} (Fig. 6) 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 play~~s~~ an important role in C turnover rates in drained peatlands, due to more readily decomposable C sources and higher temperatures (Moore and Dalva, 1993;Lafleur et al., 2005;Karki et al., 2016;Säurich et al., 2019). This soil layer was, however, not affected by higher summer GWTs in our study. SSI even reduced the number of days (24-27 days) that the top 30-40 cm remained saturated, mostly in the wet season. ~~Moreover, the 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.~~ Moreover, Säurich 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)~~. As the infiltrating water will affect the soil moisture content of these layers, it is possible that SSI could even facilitate rather than mitigate summer emissions by approaching the optimum for C mineralization more often.

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 prominent C source for emissions to the atmosphere is ~~only~~ supposed to remain limited. ~~Its potency to act as a C source is reduced~~CO_2 production and export from deeper layers is prevented by lower temperatures, limited O_2 intrusion, and the fact that water content of this layer is already close to saturation frustrating gas diffusion (Berglund and Berglund, 2011;Taggart et al., 2012;Säurich 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 at our sites deeper than 60 cm ~~are~~ were less decomposed (yellow-brown with plant macrofossils still visible) compared to the highly degraded peat in the uppermost 40 cm.

~~In addition, lower~~In our case, although CO_2 production in deeper peat layers could be lower due to~~that are~~ saturat~~ed~~ion after SSI induced~~due to the higher water level~~ GWT elevation, this reduction may be compensated 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 as low as -120 cm 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 assumption from the Dutch soil-carbon-water model that the average lowest summer GWT (i.e. GLG 'gemiddeld laagste grondwaterstanden') is the major control of CO_2 emissions, is currently under investigation (STOWA, 2020)

4.2 SSI effects on CH_4 and N_2O emission~~s~~

The magnitudes of measured CH_4 and N_2O fluxes are substantially lower than CO_2 fluxes, 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). Drainage ditches, in contrast, emitted ~~methane~~CH_4 at high rates ~~(~~(Kosten et al., 2018;Lovelock et al., 2019) ~~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.~~ In the current study the average of all measured N_2O ~~emission~~ fluxes was 3.3 mg N_2O m^{-2} d^{-1} (12 kg N_2O ha^{-1} yr^{-1}), which fall~~ing~~ with~~in~~ the range of annual N_2O emissions from drained peatlands in Northern Europe (4-18 kg N_2O ha^{-1}) (Leahy et al., 2004;Maljanen et al., 2010;Schrier-Uijl et al., 2014;Kandel et al., 2018). Fertilization, temperature and water table fluctuations play major roles in the total N_2O emission (Regina et al., 1999;Van Beek et al., 2011;Poyda et al., 2016). ~~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). It is well studied that periods with frost and thawing result in high N_2O emissions (Koponen and Martikainen, 2004). In this study, the low measurement frequency in both years does not allow annual estimations of N_2O with enough representation of peak N_2O emission. However, SSI effect still cannot be expected according to the direct comparison of measured fluxes. ~~annual N_2O emissions may have been underestimated by linear interpolation missing potential N_2O emissions peaks in late winter and spring.~~ ~~Because of the low measurement interval for both years~~The low measurement interval in both years ~~in the winter period, there is~~ a ~~high chance of an underestimation of the N_2O emission~~. which makes it hardlimits this study testing the effects of SSI on N_2O emissions extensively.~~However, this would not influence our conclusion on the absent of SSI treatment effect.~~ ~~, although this would not result in noticeable changes on the total GHG emissions.~~ ~~It is well studied that periods with frost and thawing result in high N_2O emissions (Koponen and Martikainen, 2004).~~

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 estimate~~s~~ ~~from~~ averaging all sites and years at 35.8 (22.6 – 56.0) t CO_2 ha^{-1} yr^{-1} ~~has exceeded~~is at the higher end of the ranges reported for drained temperate peatlands ~~(Wilson et al. 2016~~(Wilson et al., 2016)~~)~~. ~~, 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 in Germany. In a Dutch case study authors found a NECB of 20.1 t CO_2 ha^{-1} yr^{-1} average over the years 2005-2008 ~~(Schrier-Uijl et al. 2014~~(Schrier-Uijl et al., 2014)~~)~~. ~~, and Veenendaal et al. (2007) reported 4.9 t CO_2 ha^{-1} yr^{-1} in an earlier analysis at an intensively managed Dutch peat meadow measured with eddy covariance.~~ ~~Looking into~~ Comparing GPP and R_{eco} estimates with earlier reports we find that ~~individually, on the one hand,~~ ~~the~~ GPP of the sites was higher than values found by Tiemeyer et al. (2016) for productive and drained peatlands (-70 ± 18 t CO_2 ha^{-1} yr^{-1}) 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~~Higher GPP estimates seems reasonable give the high ~~the~~ C-export in 2017 (on average 18.0 t CO_2 ha^{-1}) that 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, the R_{eco} values of the sites (128.4±4.6 and 100.8±11 t CO_2 ha^{-1} yr^{-1} in 2017 and 2018, respectively) are also at the higher end of the range (97 ± 33 t CO_2 ha^{-1} yr^{-1} in Tiemeyer et al. (2016)). 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 different~~ce~~ R_{eco} values. Järveoja et al. (2020) ~~discovered~~ reported in a boreal natural peatland strong diel patterns of R_{eco} with emission peaks at both midnight and midday. The authors show that daily carbon fluxes were overestimated- ~~, which could lead to overestimation of daily fluxes~~ when models ~~are~~ were developed ~~with data collected around the peaks~~including peak emission. -In case a ~~Although this process the samea~~ similar pattern of R_{eco} applies to~~is not clear for~~ temperate ~~productive~~ highly productive and drained peatlands~~-systems~~ the flux measurements with opaque chambers to estimate R_{eco} would need to be spread more evenly during day (and ideally throughout the night) In our case, the flux measurements were unevenly distributed and concentrated around midday, which may have led to overestimation of R_{eco} and, therefore, NEE overestimation~~to avoid overestimation.~~ ~~, representativeness of the campaign could be a reason for the high R_{eco} estimates.~~ Besides ~~the~~ general methodological ~~speculations~~limitations of the close-chamber method, there are also a number of biochemical ~~reasons for~~mechanisms that may explain the high emissions found here. Abiotic conditions that favor high CO_2 emissions were present, with high temperatures for both years and non-limiting moisture conditions for 2017. Research from Pohl et al. (2015) found in a drained peatland 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 high rates of C ~~formation~~ sequestration and decomposition. ~~transformation processes in the plant–soil system.~~In conclusion, NEE estimates in the current study are high owing to systemic overestimation of -R_{eco} and conditions promoting high soil CO_2 production and release.

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, 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 data points that could not be ~~represented~~ explained 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 (Görres et al., 2014;Karki et al., 2019) and match the magnitude of NEE uncertainties calculated with other methods (e.g. the 23–30 tons CO_2 variances reported by (Schrier-Uijl et al., 2014) using eddy co-variance techniques). Additionally, CO_2 fluxes and annual budgets derived from eddy co-variance approach in 2019 at location ~~'~~A~~'~~ support findings of the present study ~~(~~(Van den Berg and Kruijt, 2020)~~)~~. The eddy co-variance revealed virtual identical flux patterns for both control and SSI field despite drastic differences in summer GWT surpassing 80 cm at the height of the vegetation period.

4.5 ~~Costs and benefits of the SSI~~The effects of SSI on land use

The intensity of land use (intensity and timing of drainage and fertilization, plant species composition, mowing and grazing regimes) ~~influence~~ is a major driver of ~~the~~ carbon turnover in grasslands~~'s ability to accumulate or lose C~~ (Renou-Wilson et al., 2016;Smith, 2014;Ward et al., 2016). SSI facilitates earlier fertilization compared to management under current drainage systems ~~SSI can~~ by ~~increase~~ increasing the load-bearing capacity of the field surface for fertilizing equipment. ~~,~~ We expect nutrient accumulation to continue that can lead to high CO_2 losses accelerated by nitrogen or phosphorus ~~(~~(Tiemeyer et al., 2016;Säurich et al., 2019)~~Tiemeyer et al., 2016, Säuerich et al., 2019) 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. ~~However, the general~~In summary, land-use intensity will ~~not change with the use of SSI~~remain high in SSI treatments without substantial changes to carbon sequestrating vegetation (e.g.(Couwenberg et al., 2011;Schrier-Uijl et al., 2014;Tiemeyer et al., 2020)~~ Couwenberg et al., 2011; Schrier-Uijl et al., 2014; Tiemeyer et al.,~~

~~2020~~), tillage (Smith, 2014) or potential nutrient accumulation ~~(~~(Pohl et al., 2015;Vroom et al., 2020)~~Pohl et al., 2015;~~. ~~Vroom et al., 2020)~~.

The implementation of SSI may further inflict high costs on land users. Next to investing in 1800 to 2500 m of extra drainage pipes per hectare maintenance costs rise. Drain pipe inspection, cleaning and maintenance cost range between 0.30 to 0.90 € per m with an incurrence interval of 3-6 years depending on abiotic conditions (Klaas Kooistra pers. communication). SSI inflict practical challenges in all catchments where ditch~~er~~ water levels are difficult to control and where water needs to pumped in during summer. Groundwater extraction has been suggested as an alternative which will further increase direct costs (pumping infrastructure, fuel) and indirect costs including land-subsidence following groundwater extraction ~~(~~(Herrera-García et al., 2021)~~)~~. A large rolle out of SSI seems costly, impractical and holds only few benefits for land use on peatlands.

~~The use of SSI is considered impractical for use in most regions outside of the Netherlands due to the high investment costs for irrigation~~ SSI ~~pipes and the intensive water infrastructure needed for controlling the water level. In addition, irrigation~~ SSI ~~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 (Couwenberg, 2011;Beetz et al., 2013). 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 conclusions

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 40 cm of the soil, which ~~stays above the GWT with the use of SSI~~remained drained. 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 water tables~~levels~~ close to the soil~~-~~ 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~~CF: Conceptualization, Methodology, Data curation, Writing - original draft, Supervision

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Appendix A 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~~uncertainties.

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 ~~C~~ B Reco ,GPP and NEE

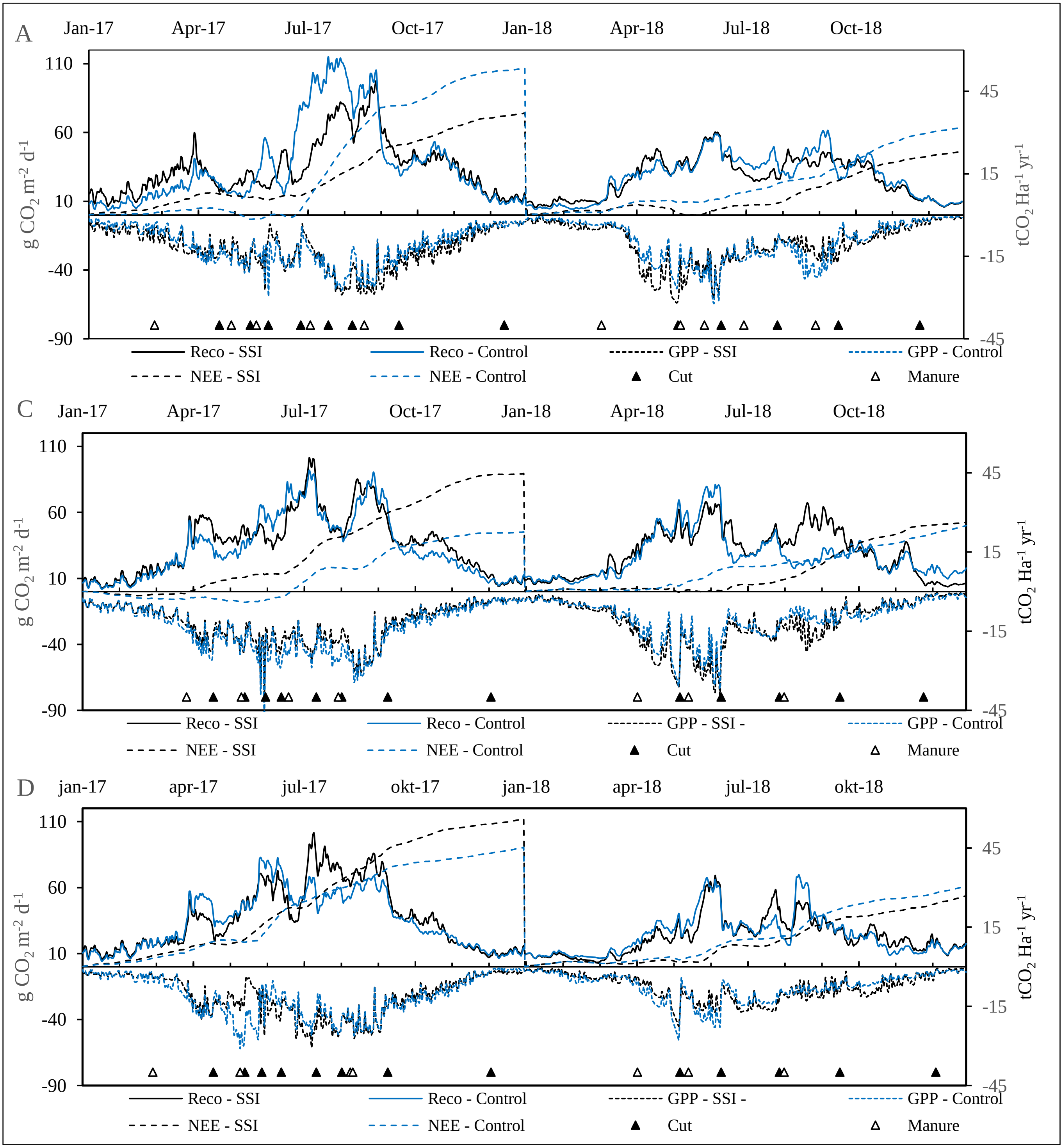

Figure B~~C~~1 Daily R_{eco} and GPP for location in g CO^2 m^{-1} d^{-1} on the primary y-axis, for control and SSI for locations A,C and D. Accumulative NEE in tCO^2 Ha^{-1} yr^{-1}, for control and SSI, every year starting at 0.

Appendix ~~D~~ C CH_4 exchange

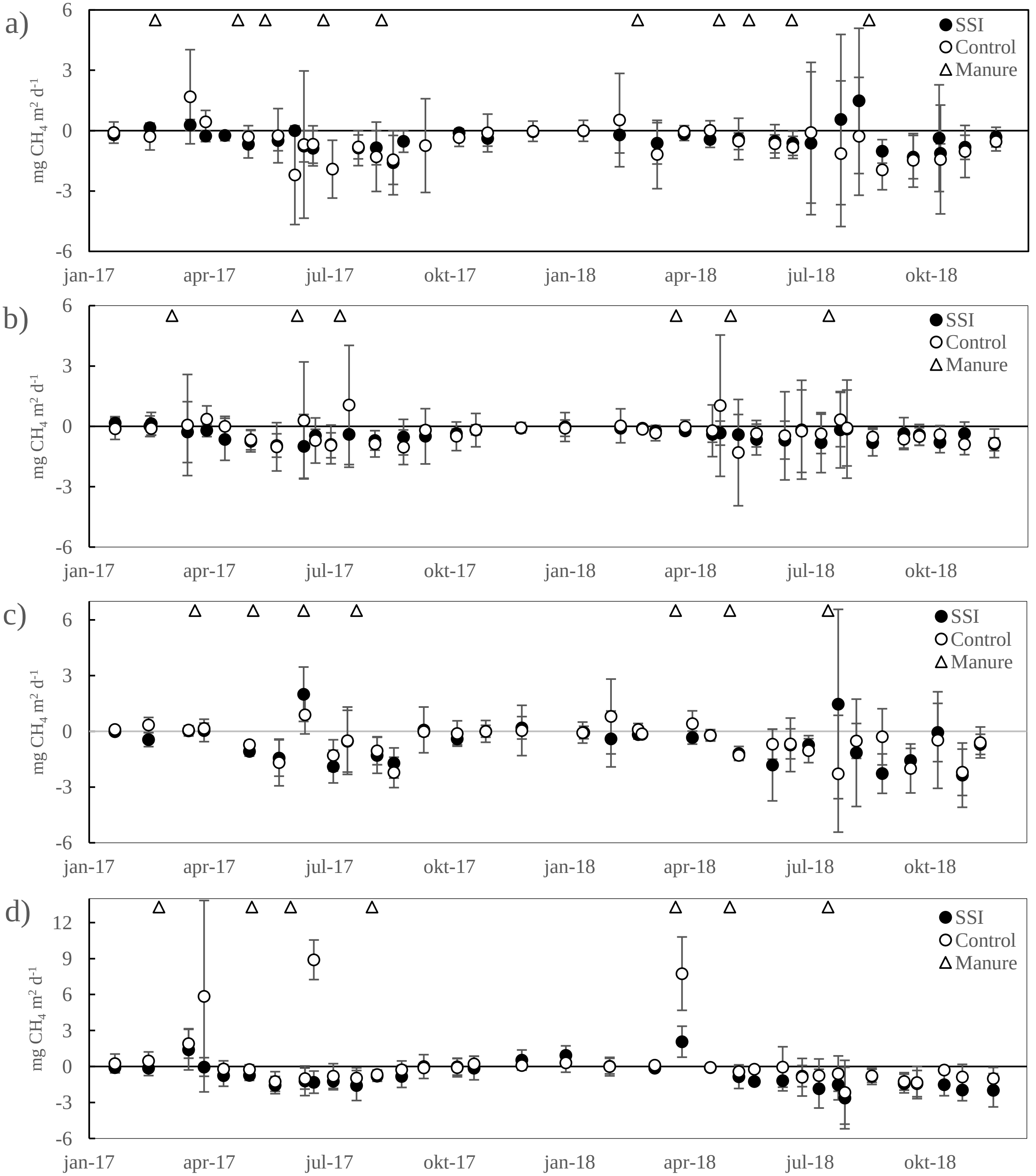

Figure C~~D~~1 CH_4 exchange throughout 2017 and 2018 in mg CH_4 m^{-2} d^{-1}

Appendix ~~E~~ D N_2O exchange

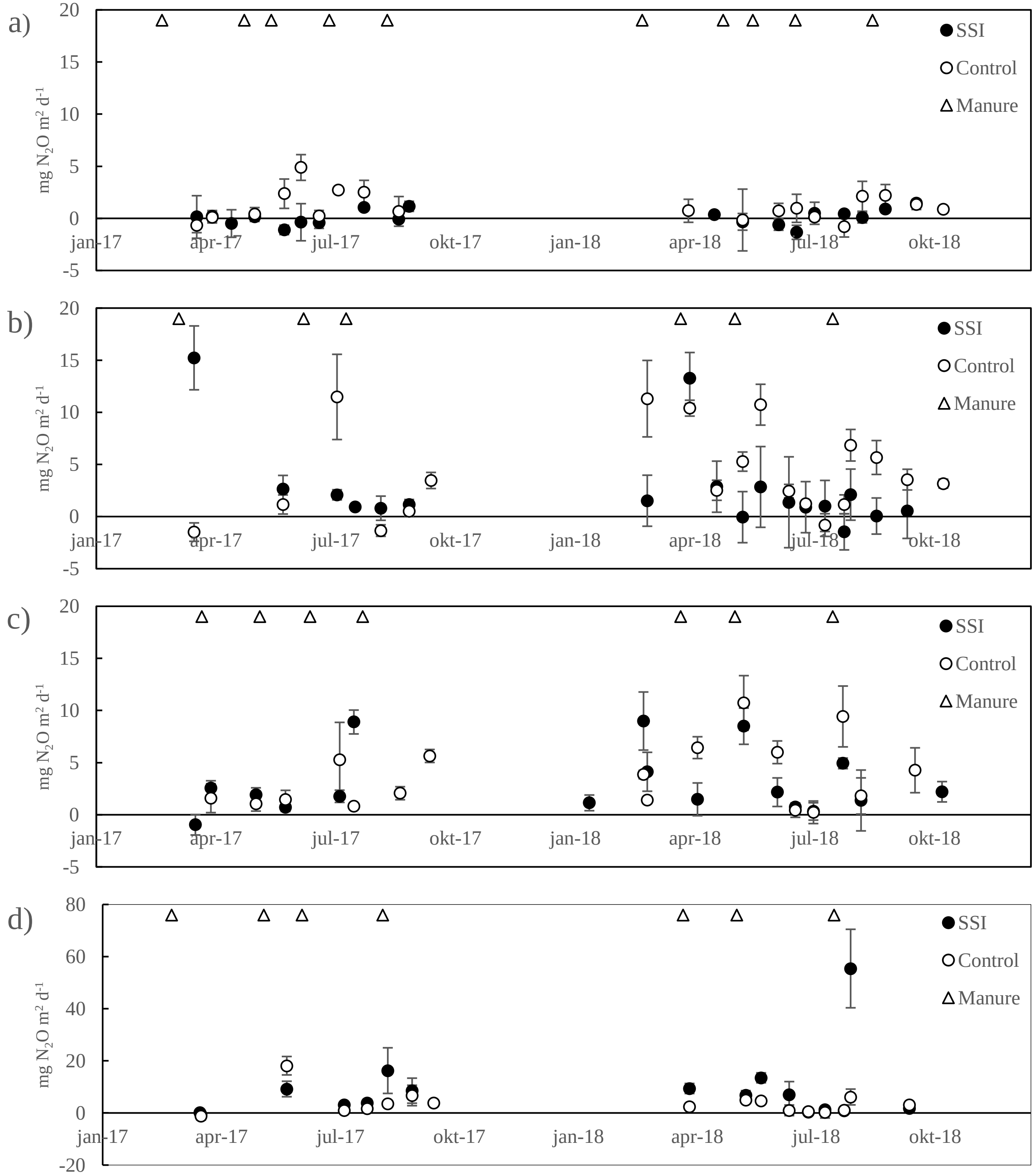

Figure ~~E1~~ D1 N_2O exchange throughout 2017 and 2018 in mg N_2O m^{-2} d^{-1}.

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