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GERT VEBER

Greenhouse gas fluxes in natural and drained peatlands: spatial and temporal dynamics

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Press

Department of Geography, Institute of Ecology and Earth Sciences, Faculty of Science and Technology, University of Tartu, Estonia.

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ORIGINAL PUBLICATIONS

This thesis is based on the following publications, which are referred to in the text by Roman numerals:

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I **Veber, G**., Paal, J., Läänelaid, A., Sohar, K., Tampuu, T., Salm, J.-O., Maddison, M., Teemusk, A., Mander, Ü., Kull, A. (202X) Simple vegetation and soil parameters determine greenhouse gas fluxes along drainage gradient in peatlands. Manuscript.

II **Veber, G**., Kull, A., Paal, J. (2021) Spatio-temporal variability of greenhouse gases along drainage gradient in 17 peatlands across Estonia. *Proceedings of the 16th International Peatland Congress 2021, Tallinn, Estonia.* 120–126.

III Viru, B., **Veber, G**., Jaagus, J., Kull, A., Maddison, M., Muhel, M., Espenberg, M., Teemusk, A., Mander, Ü. (2020) Wintertime greenhouse gas fluxes in hemiboreal drained peatlands. *Atmosphere,* 11, 731. https://doi.org/doi:10.3390/atmos11070731

IV Burdun, I., Kull, A., Maddison, M., **Veber, G.,** Karasov, O., Sagris, V., Mander, Ü. (2021) Remotely sensed land surface temperature can be used to estimate ecosystem respiration in intact and disturbed northern peatlands. *Journal of Geophysical Research – Biogeosciences.* Accepted for publication 14.10.2021.

V **Veber, G**., Kull, A., Villa, J., Maddison, M., Paal, J., Oja, T., Iturraspe, R., Pärn, J., Teemusk, A., Mander, Ü. (2018) Greenhouse gas emissions in natural and managed peatlands of America: Case studies along a latitudinal gradient. *Ecological Engineering* 114, 34–45. https://doi.org/10.1016/j.ecoleng.2017.06.068

VI Pärn, J., Verhoeven, JT.A., Butterbach-Bahl, K., Dise, N.B., Ullah, S., Aasa, A., Egorov, S., Espenberg, M., Järveoja, J., Jauhiainen, J., Kasak, K., Klemedtsson, L., Kull, K., Laggoun-Défarge, F., Lapshina, E.D., Lohila, A., Lõhmus, K., Maddison, M., Mitsch, W., Müller, C., Niinemets, Ü., Osborne, B., Pae, T., Salm, J.-O., Sgouridis, F., Sohar, K., Soosaar, K., Storey, K., Teemusk, A., Tenywa, M., Tournebize, J., Truu, J., **Veber, G**., Villa, J., Zaw, S.S., Mander, Ü. (2018) Nitrogen-rich organic soils under warm welldrained conditions are global nitrous oxide emission hotspots. *Nature Communications* 9, 1135. https://doi.org/10.1038/s41467-018-03540-1

Author’s contribution to the articles denotes: ‘*’ a minor contribution, ‘**’ a moderate contribution, ‘***’ a major contribution.

	Articles					
	I	II	III	IV	V	VI
Original idea	**	**	*	*	***	*
Study design	**	**	**	**	***	*
Data processing and analysis	***	***	**	**	***	**
Interpretation of the results	***	***	*	**	***	**
Writing the manuscript	***	***	**	*	***	*

ABBREVIATIONS AND ACRONYMS

ANOVA	analysis of variance
APEA	abandoned peat extraction area
CH_4	methane
CO_2	carbon dioxide
DB	Downy birch (*Betula pubescens*)
DC	dissolved carbon
DIC	dissolved inorganic carbon
DM	soil dry matter content
DN	dissolved nitrogen content
DOC	dissolved organic carbon
DPF	drained peatland forest
EVI	enhanced vegetation index
FAO	Food and Agriculture Organization
GAM	generalized additive model
GEST	greenhouse gas emission site type
GHG	greenhouse gas
GWP	global warming potential
LIDAR	Light Detection and Ranging
LST	land surface temperature
MODIS	Moderate Resolution Imaging Spectroradiometer
NH_4-N	ammonium nitrogen
NO_3-N	nitrate nitrogen
N_2O	nitrous oxide
NS	Norway spruce (*Picea abies*)
ORP	oxidation-reduction potential
PCA	principal component analysis
PERMANOVA	permutational multivariate analysis of variance
Ra	autotrophic respiration
R_{eco}	ecosystem respiration
Rh	heterotrophic respiration
rmR	repeated measures correlation
RMSE	root-mean-square error
R^2_{adj}	adjusted determination coefficient
r_s	Spearman's rank correlation coefficient
SOM	soil organic matter content
TN	peat water total nitrogen content
VWC	volumetric water content
WT	water table
WTD	water table depth
WTL	water table level

1. INTRODUCTION

Peatlands are the most widespread of all wetland types in the world, representing over 50% of global wetlands (Joosten & Clarke, 2002; Parish *et al.*, 2008). Despite comprising merely 3% of land area (Leifeld & Menichetti, 2018; Xu *et al.*, 2018), they store nearly a third of global soil carbon (Post *et al.*, 1982; Gorham, 1991) and 10% of global freshwater resources (Joosten & Clarke, 2002). Peatlands are important natural ecosystems with high value for climate regulation, biodiversity conservation, flood control, maintenance of natural processes and human welfare (Costanza *et al.*, 1997; De Groot *et al.*, 2006; Erwin, 2008).

Mires are peatlands (Fig. 1) where accumulation of organic matter (peat) from dead and decaying plant material occurs under conditions of permanent or almost permanent water saturation (Holden *et al.*, 2004). Dead organic matter which forms peat usually originates from *Sphagnum* and many other non-moss species, depending both on climate and on eco-hydrological conditions. Peatlands are adapted to the extreme conditions of high water and low oxygen content; their water chemistry varies from alkaline to acidic and often has low availability of plant nutrients.

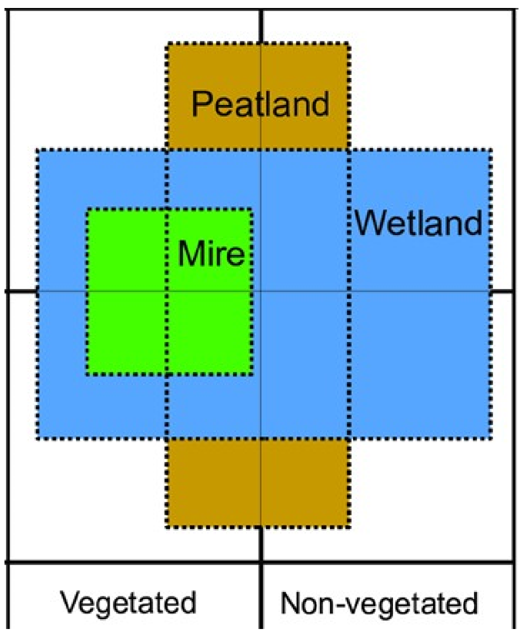

Figure 1. The relationships between mire, peatland and wetland, distinguished on the basis of presence/absence of vegetation. Peatlands outside wetland borders have a heavily altered water regime (adapted from Bragg & Lindsay, 2003).

Due to dependence on water saturation, mire ecosystems are very sensitive to changes in hydrology that may be brought about by climate or anthropogenic land use change, and thus in delicate balance between peat accumulation and decomposition. While climate change has an effect on peatlands at global scale (Swindles *et al.*, 2019; Humphrey *et al.*, 2021), the single most important factor affecting peatlands at any scale is drainage (Holden *et al.*, 2004; Limpens *et al.*, 2008; Leifeld & Menichetti, 2018). Considering that, globally, soil organic carbon in active exchange with the atmosphere constitutes approximately two-thirds of the carbon (C) in terrestrial ecosystems (Post *et al.*, 1982), and that a large amount of soil carbon is stored in histosols, the wise management of this large peatland carbon pool with long residence time (of the order of 1,200 yr) is required to make it a potentially important sink for carbon released to the atmosphere. Similarly, the amount of nitrogen (N) stored in soil is also related to climate through biotic processes associated with productivity of vegetation and decomposition of organic matter. Post *et al.* (1985) quantified global N stock in soils and found that: (1) relatively large amounts of soil nitrogen in wet tropical regions are associated with recalcitrant humic materials in an advanced state of decay, with low C/N ratios; (2) the seasonal climate contrast in temperate regions, combined with variable litter quality due to the mix of coniferous and deciduous species, results in moderate carbon and nitrogen storage in soil and variable C/N ratios; and (3) slow decomposition in wet tundra regions results in high carbon and nitrogen storage, with high C/N ratios. Organic soils with moderate or high N content could contribute to greenhouse gas (GHG) balance as potential sink or source of N_2O depending on land use management or climate change, but this requires better understanding of spatial and temporal distribution of N_2O and processes affecting its fluxes. N_2O has up to 265 (GWP_{100}) or 153 (GWP_{500}) times higher radiative efficiency than CO_2 and much shorter lifetime in the atmosphere (Myhre *et al.*, 2013), which makes it an increasingly dangerous GHG (Ravishankara *et al.*, 2009) if its concentration continues to increase due to emissions from drained peatlands.

Intact peatlands bind atmospheric carbon dioxide (CO_2) as carbon and accumulate it as peat (Maljanen *et al.*, 2010; Günther *et al.*, 2018). Meanwhile peatlands lose C with methane (CH_4) emissions due to shallow water table depths (WTD) and anoxic conditions in peat layers (Waddington & Roulet, 2000). CH_4 has more significant radiative efficiency – 32 (GWP_{100}) (Etminan *et al.*, 2016) or 7.6 (GWP_{500}) (Myhre *et al.*, 2013) – than CO_2, but a much shorter lifetime (appr. 12 years) in the atmosphere, which should be addressed when assessing the long-term effect of methane emissions from bogs as resilient ecosystems (Lynch *et al.*, 2020). Therefore, over the millennial time scale, intact peatlands have a cooling effect on the Earth climate even though they are a source of CH_4 (Günther *et al.*, 2020).

It is assumed that undisturbed peatlands are currently a weak carbon sink (~0.1 Pg C y^{-1}), a moderate source of methane (~0.03 Pg CH_4 y^{-1}), and a very weak source of nitrous oxide (~0.00002 Pg N_2O–N y^{-1}), but anthropogenic disturbance, primarily agriculture and forestry drainage (10–20% of global peatlands), results in net CO_2 emissions, reduced CH_4 emissions, and significantly increased N_2O emissions (Frolking *et al.*, 2011). Land use change and associated drainage of mires have turned carbon accumulating peatlands to significant sources of CO_2 and N_2O (Joosten, 2009; Maljanen *et al.*, 2010; Günther *et al.*, 2018; Leifeld *et al.*, 2019). Changes in wetland water levels alter the ratio of CH_4 and CO_2 fluxes. Dryer conditions in peatlands reduce CH_4 emissions and increase CO_2 loss from the soil (Turetsky *et al.*, 2014). However, Petrescu *et al.* (2015) showed that even drained peatlands can emit methane.

Pristine oligotrophic peatlands in boreal zone have rather low atmospheric N deposition (Nordin *et al.*, 1998); these peatlands are generally N limited and are usually not sources of N_2O (Martikainen *et al.*, 1993). Drained peatlands, on the other hand, have high N stocks and are therefore significant sources of N_2O (Frolking *et al.*, 2011; Carter *et al.*, 2012; Liimatainen *et al.*, 2018).

Locating and quantifying the sources and sinks of GHG, as well as understanding their spatial and temporal distributions both at local and global levels, is critical to predicting C and N responses to peatland status and management or climate change. Carbon stock and dynamics of temperate and boreal peatlands is relatively well studied (Turunen *et al.*, 2002; Abdalla *et al.*, 2016; Magnan *et al.*, 2020; Beaulne *et al.*, 2021). Less is known of tropical peatlands distribution (Leifeld & Menichetti, 2018; Xu *et al.*, 2018) and carbon stock and GHG fluxes in tropical peatlands (Jauhiainen *et al.*, 2012; Hodgkins *et al.*, 2018; Ribeiro *et al.*, 2021), and very limited information is available on GHG emissions along altitudinal gradient (Hribljan *et al.*, 2016; Veber *et al.,* 2018; Lamprea Pineda *et al.*, 2020). Regardless of geographical origin, especially scarce are studies that address simultaneously drainage or other peatland management effects on both biotic and abiotic factors and their interactions (Braekke, 1983; Coulson *et al.*, 1990; Laine *et al.*, 1995; Paal *et al.*, 2016). However, better understanding of processes and effects of the climatic change or drainage on the chemical and physical properties of peatlands and their microbial activity, gas exchange, tree stand and ground vegetation, biodiversity, and faunal change would improve peatlands management and increase efficiency of rewetting or restoration of peatlands.

Most commonly in previous studies, the water table is considered the major control on CH_4 emissions from northern peatlands during the summer months (Waddington *et al.*, 1996). Water table position has large spatial and temporal fluctuations over the frost free season (Bubier, 1995). Therefore, these fluctuations are challenging to capture using manual or automatic site specific water level measurements with little spatial coverage.

Previous research has demonstrated a positive exponential relationship between ecosystem respiration (R_{eco}) and both soil and air temperature (Bubier *et al.*, 2003; Alm *et al.*, 2007; Maljanen *et al.*, 2010; Davidson *et al.*, 2019; Järveoja *et al.*, 2020). R_{eco} (Fig. 2) is a good summary indicator to characterize drainage effect on peatland ecosystems as it incorporates both soil respiration and plant respiration. Thus it is more sensitive to water level and drainage-induced vegetation changes than the net ecosystem exchange (NEE) of CO_2 or heterotrophic respiration.

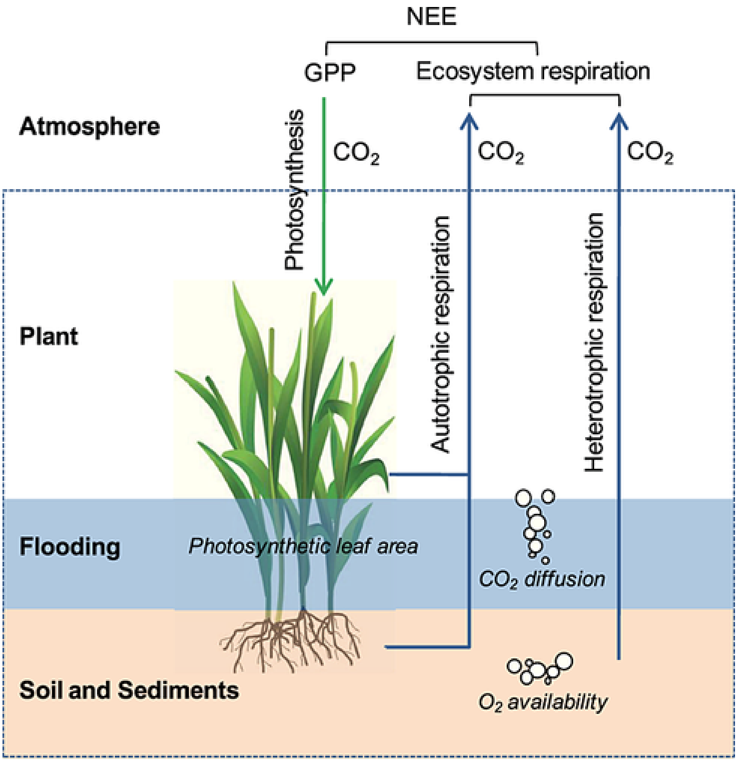

Figure 2. Conceptual scheme of autotrophic, heterotrophic and ecosystem respiration in peatlands (adapted from Han *et al.*, 2015).

Ecosystem respiration (R_{eco}) is the product of respiration by plants (autotrophic respiration, Ra) and soil microorganisms (heterotrophic respiration, Rh) (Bond-Lamberty *et al.*, 2004). The magnitudes of each of these are the result of multiple soil biogeochemical and plant physiological processes (Mäkiranta *et al.*, 2009; Wang *et al.*, 2013). This implies that the individual contributions of Rh and Ra to R_{eco} vary over space and time depending on biotic (e.g. plant species composition, biomass pools, phenology) and abiotic conditions (e.g. air and soil temperature, solar radiation, water availability; Järveoja *et al.*, 2018). While R_{eco} is a summary indicator, and it is relatively easy to measure compared to NEE, direct measurements of CO_2 fluxes are still challenging, expensive and with limited spatial coverage. By understanding the temporal and spatial processes controlling CO_2 fluxes (e.g. soil temperature, water table depth and vegetation composition), establishing a relationship between R_{eco} and other components of NEE in test areas for main vegetation communities may be possible using remote sensing methods to upscale measurements to whole ecosystem extents (Kelly *et al.*, 2021).

The main aim of the study was to quantify GHG fluxes from natural and drained peatlands and analyse the role of environmental factors on the fluxes. The specific objectives were to: (1) quantify the impact of drainage on GHG fluxes, (2) select environmental factors of high predictive capacity for potential use in GHG estimations using remote sensing, (3) assess the range of GHG fluxes in different climate zones.

Based on previous studies the hypotheses were:

(1) drainage impact varies among different GHG fluxes both spatially and temporally
(2) predictive value of environmental indicators affected by drainage differs among GHGs
(3) easily detectable environmental variables applicable for remote sensing describe a major part of GHG fluxes
(4) principal driving factors for GHG fluxes are similar at both local and global scale.

2. MATERIALS AND METHODS

2.1 Description of study sites

2.1.1 Study sites in Estonian peatlands (Publications I–IV)

There are 1 009 100 ha of peatlands in Estonia, covering 22.3% of the country's territory (Orru & Orru, 2008). Approximately 70% of them have been affected by drainage (Salm *et al.*, 2012).

A typologically and geographically representative selection of 17 peatlands from all over Estonia was chosen for the first and second studies (Fig. 3; Publications I and II). A stratified sampling method, based on data of the Estonian mires inventory (Paal & Leibak, 2011), amelioration information system and landscape regions were used to select study sites. Specific criteria included: mire trophic types (transitional bogs and raised bogs), bedrock type of the region (limestone or sandstone), landscape region, minimum diameter of the undrained part of the bog (>1.5 km), drainage type, and prevailing land use type in neighbourhood of studied drainage system of the bogs. The selection of peatlands was compiled on the basis of criterion that groups with different trophic and drainage types should include at least two repetitions in order to assess the variability inside the groups while still representing the groups' overall share among drained peatlands in Estonia.

Transects with sampling plots were established in each of these peatlands. This selection contained 10 raised bogs, two raised bogs with an old overgrown ditch and 5 transitional bogs (Tuhu, Tuudi, Hindaste, Kassisaare, Kaseraba). Seven transects were located entirely on a deep peat soil, three of them were drained for forestry (Tellissaare, Vedelsoo, Selisoo), two were located next to the peat extraction areas (Umbusi and Laukasoo), and two study sites were affected by an old overgrown ditch in the middle of the peatland (Maarjapeakse and Ullika). Five transects were in peatlands representing classical transition from transitional bog to raised bog, where the ditching was on the edge of the raised bog (Esäkeste, Allipa, Koordi, Musa, Mõksi).

Considering that all 17 studied peatlands have been affected by drainage for at least 35 years, we have in all cases almost stabilized ecosystems where changes caused by drainage have already become evident. According to Laine *et al.* (1995), typical replacement of species assemblages caused by drainage can take about 30 years in Finnish ombrotrophic bogs, and this is expected to be valid also in Estonia. Moreover, none of these studied peatlands have been grazed or subjected to tree removal and thus only the effect of drainage is present.

The third study (Publication III) was conducted at two abandoned peat extraction areas (APEAs) – Ess-soo and Laiuse – and two typical drained peatland forests (DPFs) – Norway spruce (*Picea abies*; NS) and Downy birch (*Betula pubescens*; DB) sites in Järvselja – all located in eastern and south eastern Estonia (Fig. 3). At each site, two replicate sampling plots were established.

The fourth Estonian study (Publication IV) on ecosystem respiration and methane fluxes was conducted at seven peatlands with different types of management including abandoned peat extraction areas Ess-soo, Laiuse, Kildemaa, Kõima and Maima; and pristine parts of Linnussaare and Männikjärve bogs (Fig. 3).

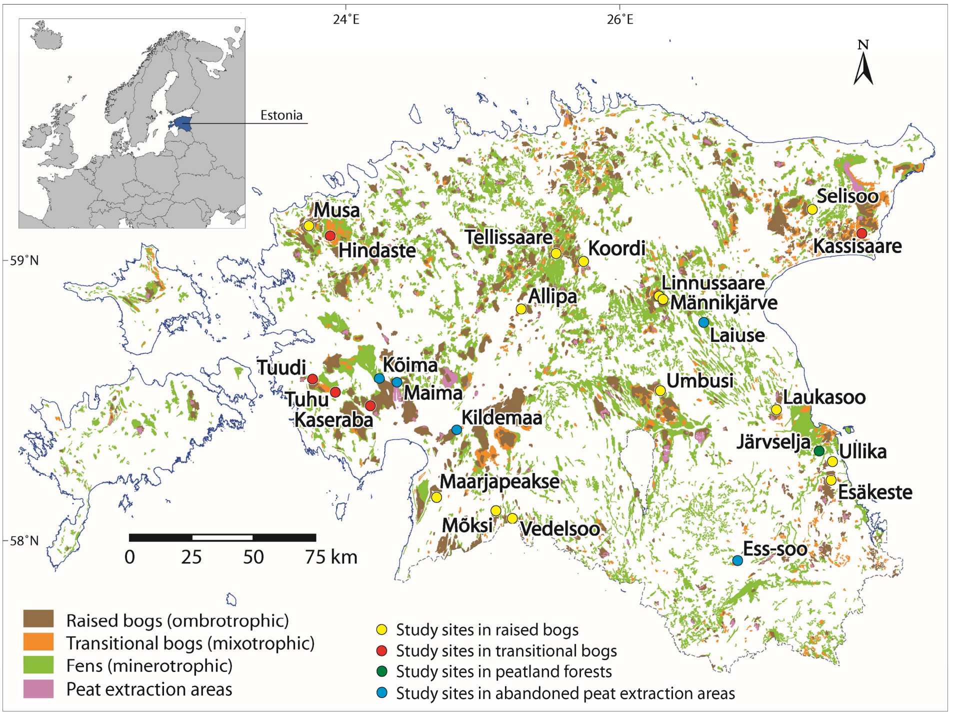

Figure 3. The location of study sites in Estonian peatlands (Publications I–IV).

For Publications I and II sampling was performed along a straight line, transects aligned in peatlands perpendicular to the drainage ditch toward the bog centre following the water regime changes gradient. Depending on the site conditions, 6–7 sampling plots were established on each transect until natural mire conditions were reached. According to the expected decrease in drainage effect, the distance between sampling plots was set to increase from ditch to bog centre. The first sampling plot/node was set at 5 m from the ditch, the others were spaced at successive intervals in the order of 10 m, 25 m, 50 m, 100 m and 250 m (Fig. 4). Water depth and samples were also collected from the drainage ditch.

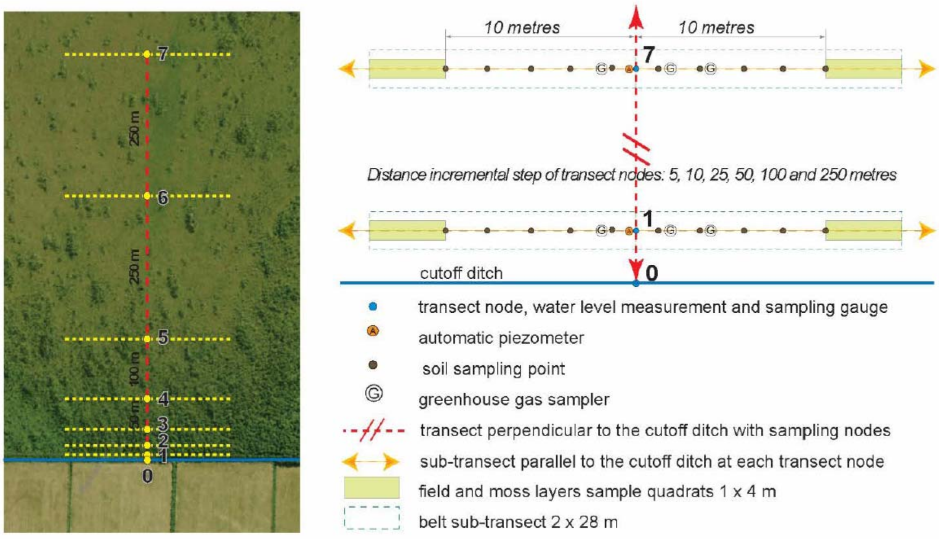

Figure 4. Study transect and layout of sampling sites with nodes and sampling plots along sub-transects (Publications I and II; adapted from Paal *et al.*, 2016).

2.1.2 Study sites in global peatlands project (Publications V–VI)

Our global soil- and gas-sampling campaign was conducted during the vegetation periods from August 2011 to March 2017, following a standard protocol. 58 organic-soil sites were sampled using criteria for organic soils adapted from the FAO World Reference Base for Soils (>12% soil carbon content in the upper 0.1 m) in 23 regions throughout the A (rainy tropical), C (temperate), and D (boreal) climates of the Köppen classification (Fig. 5). Both natural and artificially drained sites were identified, based on the proximity of drainage ditches, water table depth, and characteristic vegetation. The hydrology and trophic status of the natural sites ranged from groundwater-fed swamps and fens to ombrotrophic bogs. The most important environmental parameters were measured.

At each site, 1 to 4 sampling plots were established 15–500 m apart from each other to maximise environmental variance. Each plot was instrumented with 3–5 white opaque PVC 65 L truncated conical chambers 1.5–5 m apart and a 1 m deep observation well (a 50 mm diameter perforated PP-HT pipe wrapped in geotextile). GHG fluxes were measured using the static chamber method using PVC collars of 0.5 m diameter and 0.1 m depth installed in the soil. A stabilisation period of 3–12 h was allowed before gas sampling in order to reduce the disturbance effect on fluxes from inserting the collars. At least three sampling sessions per location were conducted over 3 days. The gas samples were brought to the University of Tartu for the laboratory analyses.

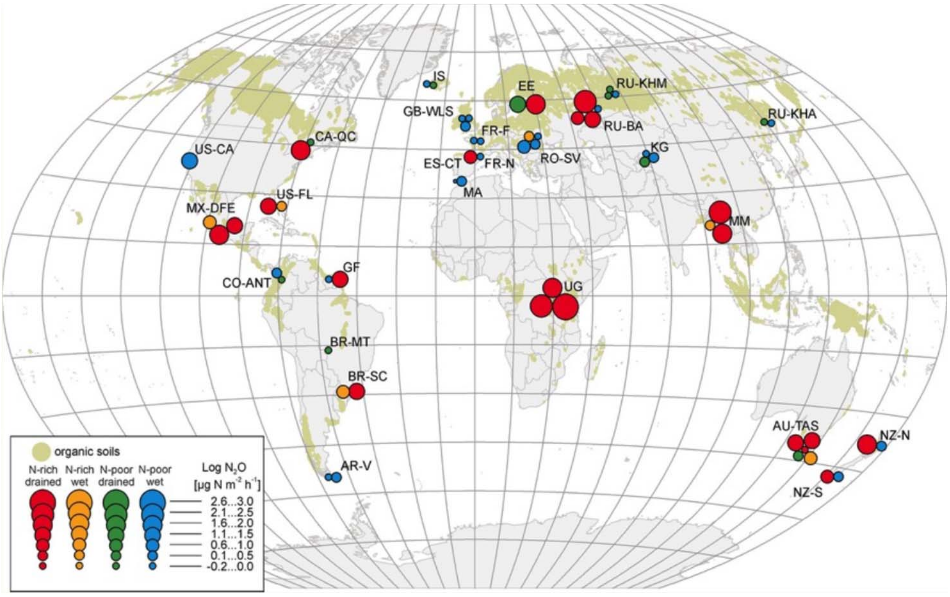

Figure 5. Site-mean N_2O fluxes by study region superimposed on a global organic-soil map. Country and region codes are defined after ISO 3166-2. The distribution of organic soil was defined as >150 t C_{org} ha^{-1} from the Global Soil Organic Carbon Estimates (courtesy of the European Soil Data Centre) + 0.5 geographical-degrees buffer for visual generalization. Based on Publication VI.

2.2 Sampling and field analyses

2.2.1 Gas sampling and analyses

Gas sampling was carried out on a monthly basis from July 2012 to August 2016 in 17 peatlands for the first and second studies (Publications I and II), twice a month in the winter period from October 2014 to April 2019 in 4 peatlands for the third study (Publication III), and during the vegetation period (March – November) in 2017–2020 in 7 peatlands for the fourth study (Publication IV). The closed static chamber method (Hutchinson & Livingston, 1993) was used for the measurement of ecosystem respiration (CO_2), CH_4 and N_2O fluxes. The gas samplers were closed chambers made of white PVC, with height 40 cm, diameter 50 cm and volume 65 L, sealed with water-filled PVC collars, which were installed at 10 cm depth in each sampling plot in three replicates (Fig. 6). In the winter season, with thick snow coverage (up to 40 cm), chambers were placed on the snow. Gas was sampled using pre-evacuated (0.3 mbar) 100 mL or 50 mL glass vials every 20 minutes during a one-hour session (at 0, 20, 40 and 60 minutes from the enclosure of samplers).

The gas concentration in the collected air was determined using the gas chromatography system Shimadzu GC-2014 equipped with an electron capture detector (ECD), flame ionization detector (FID), and Loftfield autosampler, in the laboratory of the Department of Geography, Institute of Ecology and Earth Sciences, University of Tartu, Estonia. The emission rate of the gas in the chamber was calculated on the basis of a linear change in the gas concentration, taking into account the volume of the chamber and the area covered by the chamber. The emission rate for one sampling plot was calculated as the average of the three chambers at the sampling plot.

Figure 6. Sampling plot on a study transect in Estonian transitional mire (left), closed static chamber on a collar for GHG sampling (right). Photos by Gert Veber.

2.2.2 Soil and water sampling and analyses

During each gas sampling session at every sampling plot, the depth of the groundwater table (cm) in the observation wells (∅75 mm, up to 1.3 m deep polypropylene pipes (PP-HT) perforated and covered with Typar SF 20 filter cloth) and soil temperature were measured at four depths (10, 20, 30, 40 cm) by a handheld Comet S0141 temperature logger with Pt1000TG8 sensors. In observation wells, water temperature, dissolved oxygen content O_2 (mg/l), dissolved O_2 saturation level (%), pH, conductivity (µS/cm) and redox potential (ORP, mV) were measured using a handheld YSI Professional Plus Multiparameter Water Quality Instrument with a Quatro field cable (Fig. 7). Concurrently, water samples were collected for analysis of total nitrogen (TN), dissolved nitrogen (DN), dissolved organic carbon (DOC), dissolved inorganic carbon (DIC) and dissolved carbon (DC) after the filtration of the water samples with 0.45 µm MN CROMAFIL Xtra PET-45/25 filter with the Vario TOC cube analyser (Elementar GmbH, Germany) in the laboratory of the Department of Geography, Institute of Ecology and Earth Sciences, University of Tartu, Estonia.

Figure 7. Water level observation and sampling well with YSI Water Quality Instrument and soil temperature logger. Photo by Gert Veber.

Soil and water samples for chemical analysis were collected during the period with the lowest water level, at the end of August.

In the Estonian studies (Publications I and II), at every sampling node ten sub-samples of soil were taken at 2 m intervals along a sub-transect (Fig. 4) running at 90 degrees to the main transect and parallel to the ditch. The volume of each sub-sample was 125 cm^3, and they were taken from depth 0–40 cm; as zero level, the base of the living moss layer was considered. These ten sub-samples were mixed and homogenised before analysis in the laboratory. The analyses were carried out at the Plant Biochemistry Laboratory of the Estonian University of Life Sciences.

The contents of dry matter, ash (3 hours at 550 ºC) and total-C of the homogenised soil samples were determined using a dry combustion method on a varioMAX CNS elemental analyser. The soil samples were analysed for total nitrogen (N) according to Kjeldahl (Tecator ASN3313), NH_4-N (Tecator ASN 65-32/84) and NO_3-N (Tecator ASN 65-31/84). Available (ammonium lactate extractable) phosphorus (P) was measured by flow injection analysis (Tecator ASTN 9/84). The same solution was analysed for available K, Ca and Mg content using the flame photometric method.

Water samples were analysed for total-N (ISO 11905-1), NH_4-N (ISO 11732), NO_3-N (ISO10304-1), NO_2-N (ISO 13395), total-P (ISO 15681-2), PO_4-P (ISO 15681-2) and SO_4-S (ISO 10304-1) at the Tartu Laboratory of the Estonian Environmental Research Centre.

In the studies conducted abroad (Publications V and VI), soil samples of 150–200 g were collected from each chamber location at two depths: from the permanently aerated 0–10 cm layer and from the layer directly above the groundwater level (in most cases 20–30 cm at each sampling plot) after the final gas sampling. The hermetically packed soil samples were kept in a thermal box at a low temperature and transported to the Laboratory of Plant Biochemistry of the

Estonian University of Life Sciences in Tartu, Estonia for chemical and physical analyses. All of the samples were analysed for pH, dry matter content (DM), soil organic matter content (SOM), P, K, Ca, Mg, NH_4-N, NO_3-N, total N and total C. Soil pH was determined from a potassium-chloride solution. SOM was determined as loss on ignition at 360 °C for two hours of incubation. Plant-available phosphorus (P) was determined on a FiaStar5000 flow-injection analyser (ammonium-lactate extractable). Plant-available potassium (K) was determined from the same solution using the flame-photometric method. Plant-available magnesium (Mg) was determined from a 100 mL ammonium-acetate solution with a titanium-yellow reagent on a flow-injection analyser. Calcium (Ca) was determined from a 100 mL ammonium-acetate solution using the flame-photometric method. Soil ammonium (NH_4-N) and nitrate (NO_3-N) were determined from a potassium-chloride solution via flow-injection analysis. The total nitrogen (N) and total carbon (C) content of oven-dry samples were determined using the dry-combustion method on a varioMAX CNS elemental analyser.

2.2.3 Data analysis

In statistical analyses of the Publication I, the mean values of the repeatedly measured environmental parameters on the specific sampling plots were used. The statistical analysis was carried out using STATISTICA 7.1 (StatSoft Inc.). The normality of the variable distributions was verified using the Kolmogorov–Smirnov, Lilliefors and Shapiro–Wilk's tests. In terms of gas emission data, the distribution differed from the normal, and hence nonparametric tests were performed. The Kruskal–Wallis ANOVA test was used to check the significance of differences between the measured gas emissions at different distances on transect. Spearman's rank order correlation, stepwise multiple regression models and generalized linear and nonlinear models were used to analyse the relationship between measured GHG emission and environmental parameters. In all cases, $p < 0.05$ was the standard by which statistical significance was accepted. Except the GHG fluxes, where samples with the determination coefficient $R^2 \geq 0.90$ for goodness of fit to linear regression and samples with near-zero fluxes below the gas-chromatograph accuracy (change of concentration < 20 ppb) with $R^2 < 0.90$ were used. The values of distance and the emission values of CH_4 and N_2O were log transformed prior to analysis to reduce heterogeneity of variances.

Multiple Regression models with forward stepwise inclusion of parameters for predicting CO_2-C, CH_4-C and N_2O-N fluxes were generated. Akaike Information Criterion was used to prevent over-parameterization of the models.

In Publication III the principal component analysis (PCA) was performed on environmental parameters using the R package ade4 v. 1.7–15 (Dray & Dufour, 2007). Differences in PCA between the sites were evaluated using PERMANOVA with 9999 permutations. The pairwise comparisons were corrected with the Bonferroni method using the vegan v. 2.5–6 R package (Oksanen *et al.*, 2019). The normality of distributions was checked using the Shapiro–Wilk, Anderson–

Darling, Kolmogorov–Smirnov, Lilliefors and Jarque–Bera tests. The distribution of gas data deviated from normal, and hence non-parametric tests were performed. The Kruskal–Wallis analysis of variance (ANOVA) test and Dunn's multiple comparison test were used to check the significance of differences between gas fluxes for different land-use categories and between different years at each study site, and the Spearman's rank correlation to analyse the relationship between GHG fluxes and environmental parameters. P-values were considered statistically significant after Benjamin–Hochberg correction. Statistical analysis was carried out using Statistica and XLSTAT. The level of significance of $p < 0.05$ was accepted in all cases, except in evaluating CH_4 and N_2O flux regressions, when $p < 0.1$ was accepted.

In Publication IV the collar flux data for replicates in each site was averaged for further statistical analysis to avoid pseudo-replication. Further, principal component analysis was applied to derive information about the relationships among all in-situ measured variables and cluster data depending on the relevance of different variables for four different studied groups: hummocks, hollows, drained and extracted sites. Before PCA analysis, the variables were standardized to zero mean. To estimate the common linear associations in paired repeated measures data, the repeated measures correlation, rmR (Bakdash & Marusich, 2017), between CO_2 and CH_4 fluxes and in situ measured parameters, and between in situ temperatures, MODIS LST, and Landsat LST was calculated. Pearson correlation (R) for the data originating from one sampling site was calculated. The goodness of R_{eco} model performance was evaluated with R-squared (R^2) and root-mean-square error (RMSE) statistics. All statistics were computed using R software.

In Publication VI, the principal component analysis, Spearman's rank correlation and stepwise multiple regression of site-mean efflux vs. the environmental parameters were used. The tests were run using both untransformed and log-transformed N_2O fluxes. Normality of the variables and the residuals were checked by the Shapiro–Wilk test. Neither the N_2O fluxes nor their logarithms were normally distributed ($p < 0.05$). Therefore, only a nonparametric test such as Spearman's rank correlation and generalised additive models (GAM) could be applied. The mgcv package of the R Project was used to calculate the GAM regressions using minimal smoothness ($k = 3$). P-values (significance level $p < 0.05$) from the summaries of the GAM regressions produced by the summary.gam package of the R Project were reported. Only GAM regressions when the residuals were normally distributed were reported. As a presumption for the stepwise multiple regression, the independent variables were checked for GAM concurvity – only multiple relationships with a variance inflation factor <10 between the independent variables were reported. The presence of a boundary in our data was tested. The test compared the density of points in the region of the data set's upper envelope to the expected density of the upper envelope of a bivariate normally distributed data set of the same size (Milne *et al.*, 2006).

3. RESULTS AND DISCUSSION

3.1 Simple vegetation and soil parameters determine greenhouse gas fluxes along drainage gradient in peatlands (Publication I)

Locating and quantifying spatial and temporal distributions of the sources and sinks of GHG at local level is critical to understanding processes affecting GHG fluxes, and helps to predict carbon and nitrogen cycle response to peatland status, management or climate change.

There are several studies relating GHG fluxes, especially those of CH_4 to plant species (Yavitt & Knapp, 1998; Fritz *et al.*, 2011) or vegetation communities (Bubier, 1995), microtopography (Frenzel & Karofeld, 2000; Whalen & Reeburgh, 2000; Bubier *et al.*, 2011), or a number of physico-chemical properties (Liu, 1996; Korrensalo *et al.*, 2017; Garnett *et al.*, 2020), but usually the established statistical relationship or model is valid only in a particular site or bog type and could not be expanded to other regions. Despite well-known relationship between soil nutrients level (particularly nitrogen content) and vegetation communities or indicator species it is rarely used in peatland studies. There are almost no studies relating vegetation indicators to N_2O emissions, or studies relating them at the landscape level in peatlands (Voigt *et al.*, 2017; Luan *et al.*, 2019).

While water level and pH control the vegetation composition in peatlands, the vegetation composition affects the C balance by affecting net primary productivity (NPP), the amount and ratio of CO_2:CH_4 released into the atmosphere and peat's physical properties. As plants and vegetation types generally form easily recognizable units for both land based and airborne assessments, and are closely coupled to soil processes, vegetation changes are a convenient monitoring tool for environmental changes (Limpens *et al.*, 2008).

Direct measurements of GHG fluxes are still challenging, expensive (due to large seasonal and inter-annual variation in temperate climates) and time consuming, and have limited spatial coverage. An understanding of the temporal and spatial processes controlling GHG fluxes (e.g. soil temperature, water table depth and vegetation composition) and well established relationships in test areas enable application of models in wider areas within similar climate zones and habitats.

One such more advanced indicator matrix method called GEST (*Greenhouse gas Emission Site Type*) – has been developed by Couwenberg *et al.* (2011). It is mainly based on water level class, the presence of aerenchymous shunts, nutrient status, pH and land use. However, it requires significant input of labour- or time-costly parameters (e.g. time series for defining water level classes, soil chemical parameters). Likewise, it still lacks spatial resolution to describe drainage effects in most affected zones which may extend at least 250 m from the ditch (Paal *et al.*, 2016).

3.1.1 Drainage effect in raised bogs

Based on the Spearman's rank order correlations in drained raised bogs, the soil CO_2 efflux was positively related to tree canopy cover ($r_s = 0.54$) and sum of tree girth ($r_s = 0.66$), as well as soil nitrogen ($r_s = 0.47$) and dry matter content ($r_s = 0.63$). Strong negative correlations occurred in distance from the drainage ditch ($r_s = -0.64$), minimum water level (average of annual minimum water levels; $r_s = -0.57$) and the cover of *Sphagnum* spp. ($r_s = -0.55$). Methane emissions correlated positively to the distance from the ditch, minimum water level and the cover of *Sphagnum* spp. ($r_s = 0.66$, $r_s = 0.76$, $r_s = 0.76$ respectively), but negatively to the tree canopy cover ($r_s = -0.46$), sum of tree girth ($r_s = -0.47$) and soil dry matter content ($r_s = -0.73$). Similarly to CO_2-C, the emission of N_2O-N correlated positively to soil dry matter content ($r_s = 0.46$) and negatively to minimum water level ($r_s = -0.51$) and the cover of *Sphagnum* spp. ($r_s = -0.55$) (see Table 2 in Publication I).

Median ecosystem respiration in raised bogs varied between 78.4 close to the drainage ditch and 48.7 mg C m^{-2} h^{-1} in the natural part of the transect. According to the Kruskal-Wallis test, the fluxes in the 5 m sampling plot significantly differed from those in the sampling plots at 90 m and 440 m correspondingly. Differences were significant between the fluxes in 40 m and 440 m sampling plots as well (Fig. 8a).

Median CH_4-C emission in raised bogs ranged from 45.5 to 2714.6 µg C m^{-2} h^{-1} as the distance from the drainage ditch increased. Maximum average emission 15355.6 µg C m^{-2} h^{-1} was measured in the Selisoo 440 m sampling plot, situated next to a pool. Negative fluxes occurred at the distances of 5 and 15 metres from the drainage ditch, being −38.2 µg C m^{-2} h^{-1} in the Selisoo 5 m sampling plot. Spatial variability of CH_4-C fluxes was high; significant difference occurred between the fluxes at the furthest distance of 440 m and at the distance groups of 5 and 15 metres (Fig. 8d).

Median fluxes of N_2O-N in raised bogs remained close to zero, varying from 0.2 to 1.5 µg N m^{-2} h^{-1}. Increased maximum average fluxes occurred in the 5 and 15 m sampling plots. The flux in the 5 m sampling plot differed significantly from the fluxes in the 40 and 90 m; the flux in the 15 m differed statistically significantly from the flux in the 40 m plot (Fig. 8g).

Based on the results of the statistical models generated by multiple regression analysis with forward stepwise inclusion of the variables, the combination of easily detectable and most predictable models for GHG emissions in drained raised bogs and drained transitional bogs was selected.

In raised bogs, the flux of CO_2-C was best predicted ($R^2_{adj} = 0.47$; $p < 0.000$) by the combination of log-distance from the drainage ditch, cover of shrub layer and tree height (see Fig. 5a in Publication I). Alternatively, soil CO_2-C efflux is predictable with the minimum water level and the cover of the tree layer ($R^2_{adj} = 0.43$). The emission of CH_4-C in raised bogs was best modelled with the cover of *Sphagnum* spp., soil temperature at 40 cm depth and minimum water level ($R^2_{adj} = 0.66$; $p < 0.000$; see Fig. 5c in Publication I). While using the cover of *Sphagnum* spp. and soil temperature at 40 cm depth alone, the model explained 60% of the variation. The best model for predicting N_2O emission in raised bogs was mostly based on soil characteristics, using the cover of *Sphagnum* spp., soil NO_3-N content, SOM and soil pH, resulting in adjusted $R^2 = 0.37$ ($p < 0.000$; see Fig. 5e in Publication I).

3.1.2 Drainage effect in transitional bogs

The Spearman's rank correlation analysis revealed strong correlations between all three GHG emissions and the cover of *Sphagnum* spp. ($r_s = -0.67$ CO_2-C, $r_s = 0.76$ CH_4-C, $r_s = -0.77$ N_2O-N), mean annual increment ($r_s = 0.68$, $r_s = -0.44$, $r_s = 0.54$), soil dry matter ($r_s = 0.70$, $r_s = -0.79$, $r_s = 0.60$) and the depth of bog peat ($r_s = -0.60$, $r_s = 0.55$, $r_s = -0.52$ respectively). Distance from the drainage ditch correlated negatively ($r_s = -0.51$) to CO_2-C and positively ($r_s = 0.69$) to CH_4-C fluxes. Soil water O_2 content correlated positively ($r_s = 0.55$) to CO_2-C and negatively ($r_s = -0.77$) to CH_4-C fluxes. In drained transitional bogs, the soil NO_3-N content correlated significantly to the GHG emissions ($r_s = 0.64$ CO_2-C, $r_s = -0.45$ CH_4-C, $r_s = 0.79$ N_2O-N) (see Table 2 in Publication I).

Median R_{eco} in transitional bogs varied between 105.0 in the 5 m and 52.0 mg C m^{-2} h^{-1} in the 440 m sampling plot, the two places on the transect where the fluxes had statistically significant differences (Fig. 8b).

Median emissions of CH_4-C in transitional bogs ranged from 53.8 to 5168.3 µg C m^{-2} h^{-1}. Maximum average emission occurred in the Hindaste 440 m sampling plot. Similarly to the raised bogs, the 5 and 15 m sampling plots had negative median fluxes, −28.3 and −5.3 µg C m^{-2} h^{-1} respectively, though lacking statistically significant differences (Fig. 8e).

In transitional bogs the median emissions of N_2O-N ranged from 0.5 to 10.3 µg N m^{-2} h^{-1}. The emissions were remarkably higher, though not statistically significantly differing, in the 5, 15 and 40 m sampling plots with fen peat and greater fluctuations in ground water level. Maximum average emissions reached up to 46.2 and 32.3 µg N m^{-2} h^{-1} in the Hindaste 15 and 40 m sampling plots respectively (Fig. 8h).

In transitional bogs the flux of CO_2-C is predictable by minimum water level alone ($R^2_{adj} = 0.54$). The explanatory power of the model improved when the log-distance from the drainage ditch and tree height was also accounted for ($R^2_{adj} = 0.58$; $p < 0.000$; see Fig. 5b in Publication I). The emission of CH_4-C was best predicted ($R^2_{adj} = 0.64$; $p < 0.000$) with the combination of minimum water level and tree canopy cover (see Fig. 5d in Publication I). Meanwhile the minimum water level alone explained 56% of the variation of methane emission. The cover of *Sphagnum* spp. and the cover of tree canopy predicted 42% of the variation respectively. The emission of N_2O-N in transitional bogs was explained by the cover of *Sphagnum* spp., minimum water level and the content of soil organic matter ($R^2_{adj} = 0.52$; $p < 0.001$; see Fig. 5f in Publication I). The cover of *Sphagnum* spp. alone predicted 43% of the variation in the emission of N_2O-N.

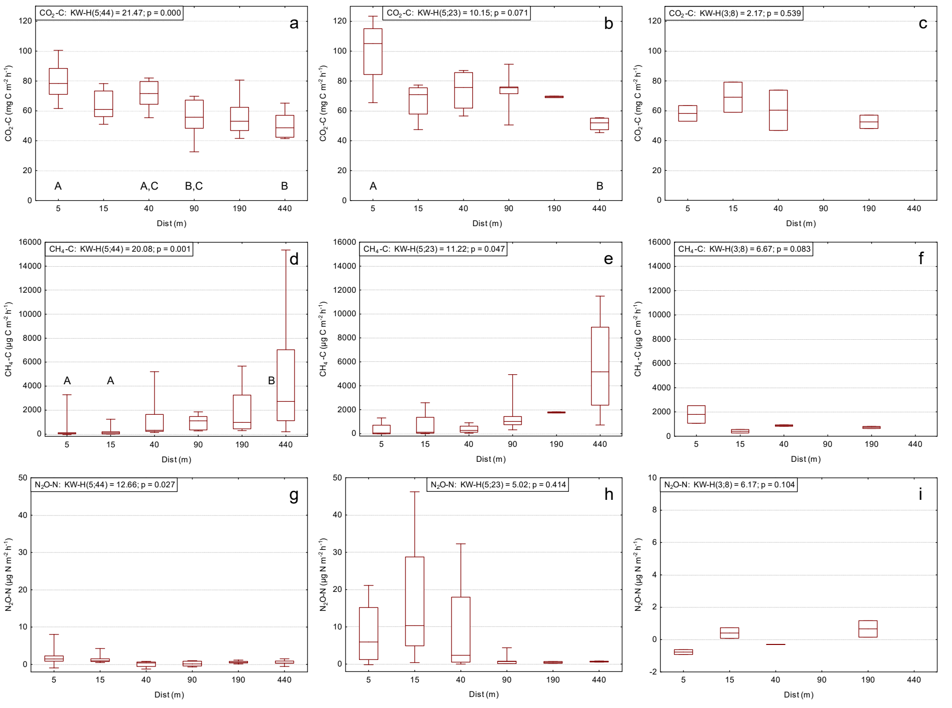

Figure 8. CO_2-C (mg C m^{-2} h^{-1}), CH_4-C (µg C m^{-2} h^{-1}), N_2O-N (µg N m^{-2} h^{-1}) fluxes from raised bogs (a, d, g), transitional bogs (b, e, h) and raised bogs with old overgrown ditches (c, f, i). Notice difference in scale of y-axis for (i). Median, minimum and maximum values and interquartile range (from 25 to 75%) are shown. A and B – significantly differing values ($p < 0.05$) by Kruskal-Wallis ANOVA test. Figure from Publication I.

3.1.3 Drainage effect in raised bogs with overgrown ditches

The statistically significant correlations were scarce in raised bogs with old overgrown ditches, where environmental parameters had minor variability compared to raised bogs and transitional bogs. The flux of CO_2-C had strong positive correlations with tree layer parameters and strong negative correlations with soil-water nitrogen content and soil temperature (see Table 2 in Publication I). Methane emission only correlated negatively with minimum water level and positively with total soil water P content. The emission of N_2O-N had strong positive correlations with soil temperature at all four measured depths and strong negative correlations with the sum of tree girth, total soil water P content and the soil NO_3-N content (see Table 2 in Publication I).

The median R_{eco} in bogs with an old overgrown ditch varied between 58.3 and 52.6 mg C m^{-2} h^{-1}, resulting in statistically insignificant differences between sampling plots (Fig. 8c). Opposite to raised bogs and transitional bogs, the emission of CH_4-C in bogs with an old overgrown ditch was highest in the 5 m sampling plot. The emissions of methane between different sampling plots had no statistically significant difference in these bogs (Fig. 8f). The median fluxes of N_2O-N in bogs with an old overgrown ditch varied between −0.8 and 0.7 µg N m^{-2} h^{-1}, having statistically non-existent differences between sampling plots at any distance from the ditch (Fig. 8i).

3.1.4 Main factors affecting GHG-s in drained peatlands at local level

The decrease of CH_4 emission, increase of R_{eco} and the sum of tree girth, as well as the height and cover of tree layer in drained bogs and drained transitional bogs, are related to water level draw-down creating better conditions for tree growth. Vegetation is found to be a strong indicator for mean water levels (Laine *et al.*, 1995) and can provide a robust proxy for spatially explicit estimates of methane emissions (Couwenberg, 2009; Couwenberg *et al.*, 2011) and CO_2 fluxes (Karu *et al.*, 2014) over larger areas. Furthermore, plant species composition has been regarded as the best single predictor of mean CH_4 flux (Levy *et al.*, 2012). In our study, the cover of *Sphagnum* spp. or the cover of tree layer could explain 42% of the variation in methane emission in transitional bogs; the explanatory power of the statistical model improved significantly when the abiotic minimum water level was included ($R^2_{adj} = 0.64$; $p < 0.000$). Higher water table generally corresponds to increased CH_4 emissions and reduced CO_2 emissions, as evident in other studies as well (Wilson *et al.*, 2016; Swenson *et al.*, 2019). Pelletier *et al.* (2007) and Salm *et al.* (2012) reported that CH_4 fluxes could be considered as zero when the water level is below −30 cm. However, our study shows average emissions of 784 and 1097 µg CH_4-C m^{-2} h^{-1} remaining in drained bogs and drained transitional bogs respectively, despite the water level below −30 cm (Publications I, II, IV).

Based on our study we can state that tree canopy cover is an important indicator of methane flux, particularly in drained transitional bogs ($r_s = -0.68$; $p < 0.000$). We also determined temperature at 40 cm depth and minimum water level to be important abiotic predictors of methane emission. Our results are in conjunction with the study of Ojanen et al. (2010), where summer mean water table depth and mire vegetation cover explained the annual CH_4 flux.

In well-drained peatlands, temperature is found to be the only significant variable explaining temporal variation in momentary soil respiration (Mäkiranta *et al.*, 2008). Lafleur et al. (2005) discovered in their study, conducted in a cool, relatively dry (water table fluctuating between 30 and 75 cm below ground) temperate bog, that ecosystem respiration depended on peat temperature, not water table depth. If the water table level is constantly low or high (Nieveen *et al.*, 1998), it does not vary enough to produce an effect on ecosystem respiration. With a high water level, the peat decomposition is inhibited by flooding of peat by water; with a low water level, the peat decomposition is limited by insufficient peat moisture. With a moderate water level, an optimum is reached between these two factors (Kurbatova *et al.*, 2013). The soil CO_2 emissions have been found to be impeded when the water table drops 60–70 cm below ground surface (Mäkiranta *et al.*, 2009; Ojanen & Minkkinen, 2019), which is confirmed also by our results (Publication I).

The emission of N_2O being remarkably higher in drained transitional bogs than in raised bogs in our study was supported by the main findings of Minkkinen et al. (2020), where drainage increased N_2O emission from nutrient-rich boreal peatlands without any effect in nutrient-poor sites. In our study, the drainage induced rise in N_2O emission was up to tenfold in nutrient-rich transitional bogs compared to raised bogs. The outcome of this study – that any single variable fails to explain the emissions of N_2O very well – was supported by Minkkinen *et al*. (2020).

Tree canopy height and cover, moss cover (especially *Sphagnum* moss cover), and water table are the most indicative parameters describing the GHG fluxes in both drained bogs and transitional bogs. These parameters are relatively easily detectable manually or by remote sensing methods, thus their indicative value to describe GHG fluxes is potentially high. Remote detection of *Sphagnum* mosses is more complicated but methods are under development (Arroyo-Mora *et al.,* 2018; Räsänen *et al.,* 2020; Yuwen *et al.,* 2020). However, environmental factors determining the GHG fluxes in drained raised bogs and drained transitional bogs are different. Environmental factors easily determinable by remote sensing are applicable to estimating R_{eco} and CH_4 emissions, but not the N_2O fluxes. Simple parameters alone tend to have insufficient explanatory power; estimations improve when water level and soil temperature are also accounted for, and are further enhanced when pH and soil organic matter content are defined.

3.2 Spatio-temporal dynamics of GHG fluxes in natural and drained peatlands in Estonia (Publication II)

Natural wetlands are sources of CH_4 and sinks of CO_2; changes in the water level alter the ratio of CH_4 and CO_2 fluxes. Dryer conditions in peatlands reduce CH_4 emissions and increase CO_2 loss from the soil (Petrescu *et al.*, 2015), converting efficient carbon sink ecosystems to large CO_2 sources (Maljanen *et al.*, 2010). Most commonly in previous studies, the water table position (Dise *et al.*, 1993; Moore & Roulet, 1993; Waddington *et al.*, 1996; Ojanen & Minkkinen, 2019) and soil temperature (Nykanen *et al.*, 1995; Le Mer & Roger, 2001; Pelletier *et al.*, 2007; Salm *et al.*, 2012; Ballantyne *et al.*, 2014; Levy & Gray, 2015) are considered the major control on GHG emissions from northern peatlands. While main factors controlling GHG fluxes in peatlands are known, there is still high uncertainty about how far the drainage effect reaches, and, especially, how it affects spatial and temporal dynamics of GHG fluxes.

Based on the results of our study, the temporal variability of R_{eco} based CO_2 flux and CH_4 emission in different sampling plots on transects in drained bogs, drained transitional bogs and bogs with an old overgrown ditch had strong correlations to water table position, along with strong correlations to both water and soil temperature. The N_2O flux in studied nitrogen-poor peatlands was low or insignificant, and statistically significant correlations between the environmental variables and the emission of N_2O were scarce. According to PCA, the N_2O emission was most often related to higher dissolved nitrogen and dissolved organic carbon concentration, particularly in drained transitional bogs in zones of strong drainage effect and large seasonal water level fluctuations.

3.2.1 Ecosystem respiration

The R_{eco} had a strong seasonal pattern, almost 90% was emitted during the growing season (Fig. 9a, b, c). In all studied peatlands, August was the month with the highest CO_2 flux (Fig. 9a, b, c), having also the highest peat temperature at 20 cm depth and the lowest water table level (Fig. 9d, e, f). Fluxes were higher closer to the drainage ditches, and seasonal amplitude in R_{eco} decreased towards mire in natural status (440 m from ditch) in all studied peatlands. In drained transitional bogs, R_{eco} was on average 139.1 mg CO_2-C $m^{-2}h^{-1}$ during the growing season while being only 20.9 mg CO_2-C $m^{-2}h^{-1}$ during the winter. The highest mean monthly flux was reached in natural background areas in July, but in drainage affected zones in August. Drained transitional bogs had the highest R_{eco} (202.8 mg CO_2-C $m^{-2}h^{-1}$) at a distance up to 15m from the ditch (Fig. 9c), while in raised bogs it remained at 169 mg CO_2-C $m^{-2}h^{-1}$. Water table level had large spatial and temporal fluctuations over the frost free season, in both drained raised bogs and transitional bogs. Only in raised bogs with overgrown ditches was the water table depth on average higher than −30 cm at 15m distance from the ditch all year round (Fig. 9d). High R_{eco} in drainage-affected zones of transitional and raised bogs is related to faster C turnover, and is indicated also by higher pore water dissolved O_2 content, DOC and DN concentration.

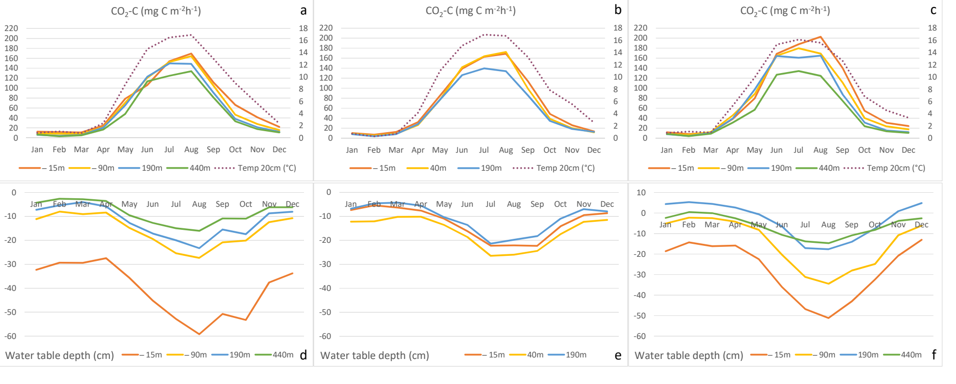

Figure 9. Seasonal dynamics of CO_2-C fluxes by distance from the drainage ditch, and mean soil temperature at 20 cm depth (right axis) in drained raised bogs (a), raised bogs with an old overgrown ditch (b) and drained transitional bogs (c). Seasonal dynamics of mean water table depth by distance from the ditch in drained raised bogs (d), raised bogs with an old overgrown ditch (e) and drained transitional bogs (f). Figure from Publication II.

3.2.2 Methane fluxes

Methane emissions had remarkable seasonal and spatial variability, being lowest in the winter and near the drainage ditches, and increasing up to 20310.3 µg CH_4-C $m^{-2}h^{-1}$ by August in the pristine parts of the raised bogs with ridge-hollow-pool subtype (Fig. 10a, c). On the contrary to the raised bogs and transitional bogs, the emission of methane in bogs with old overgrown ditches was higher close to the drainage (Fig. 10b), though staying considerably lower than the emission in pristine parts of raised bogs and transitional bogs. The *Sphagnum*-filled ditches kept an optimal water table level for methane production during the summer months (Fig. 9e) and avoided the major water table lowering present in other peatlands of the study (Fig. 9d, f). However, *Sphagnum cuspidatum*-filled old shallow ditches and partly blocked water movement had throughout the year sufficient dissolved oxygen level for oxidation of CH_4 produced in inundated zone (Raghoebarsing *et al.*, 2005).

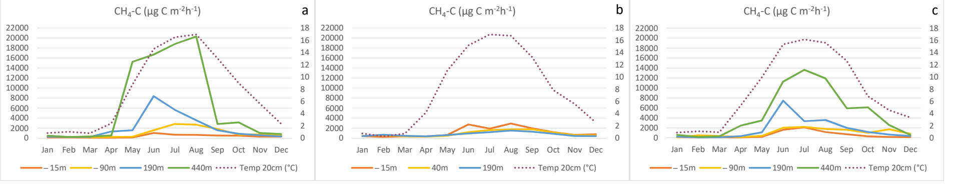

Figure 10. Seasonal dynamics of CH_4-C emission by distance from the drainage ditch (left axis) and mean soil temperature at 20 cm depth (right axis) in drained raised bogs (a), raised bogs with an old overgrown ditch (b) and drained transitional bogs (c). Figure from Publication II.

3.2.3 Nitrous oxide fluxes

Emission of N_2O was insignificant and without any clear seasonal pattern, staying below 3 µg N_2O-N $m^{-2}h^{-1}$ the whole year in nitrogen-poor raised bogs (Fig. 11a, b). The emission had a slight tendency to increase in the warm period since May, next to the ditch in the most drainage-affected zone where peat decomposition is faster, dissolved nitrogen is more readily available and pore water content is highly fluctuating according to added rainwater. In areas less affected by drainage or in natural background sites, N_2O emission was slightly higher only in months of the lowest water table level, and the sole determining factor was the fluctuation of pore-water content according to added rainwater. In drained transitional bogs, soil nitrogen content was higher and emission reached 20.7 µg N_2O-N $m^{-2}h^{-1}$ at the distance up to 15 m from the ditch (Fig. 11c). The N_2O emission in strongly drainage-affected zones up to 90 m in transitional bogs was higher all year round; it had a distinctive short-lasting peak in winter with freeze-thaw cycles when peat pores started to fill up with snow melt water, and a longer- lasting peak in summer when water table level was low but fluctuant and DN was increased in pore water after rainfalls. Further from the ditch in more intact parts of the transitional bogs, the emission of N_2O was similar to that in raised bogs (Fig. 11c).

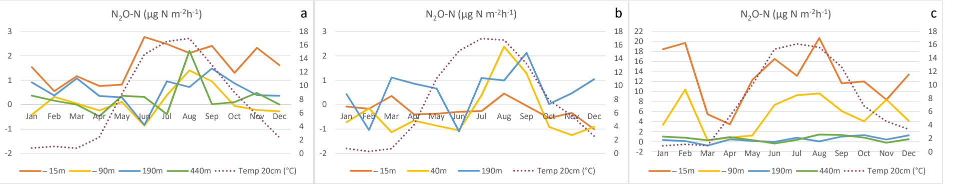

Figure 11. Seasonal dynamics of N_2O-N emission by distance from the drainage ditch (left axis) and mean soil temperature at 20 cm depth (right axis) in drained raised bogs (a), raised bogs with an old overgrown ditch (b) and drained transitional bogs (c). Notice difference in scale of y-axis for (c). Figure from Publication II.

There is a clear seasonal and spatial pattern in the fluxes of R_{eco} and CH_4. During the winter months, the fluxes are similarly low regardless of drainage effect along transects. In spring, when soil starts to warm up, the fluxes of GHG-s start to increase (Fig. 9a, b, c). During the summer, the differences in fluxes of GHG-s along the drainage gradient emerged, and this is in good agreement with other studies (Metzger *et al.*, 2015). Areas with lower water table levels have higher fluxes of CO_2, rather than more natural parts of transects with higher water table levels (Fig. 9). The relationship between the water table level and methane emission was the opposite: the lower the water level dropped, the smaller the emission. Areas unaffected by drainage with natural water tables showed an increase in methane emission following the seasonal course of the surface temperature (Fig. 10a, b, c and 9d, e, f). Although the emissions of CH_4 in the natural parts of the peatlands are noticeable, the amount of carbon lost as CH_4-C is not comparable to the CO_2-C fluxes from the areas with lowered water table levels.

The N_2O emission from peatlands is closely related to peat nitrogen content, and thus raised bogs, transitional bogs and fens emissions vary highly after drainage. This is supported by previous studies by other authors (Nykanen *et al.*, 1995), and furthermore, peat degradation in drainage affected zones is the main driver of N_2O and DOC release (Liu *et al.*, 2019). In general, raised bogs are a very weak source of nitrous oxide, but their share increases with anthropogenic disturbance (Frolking *et al.*, 2011; Salm *et al.*, 2012; Publication V). Drained peatlands with high N stocks have increased N_2O emissions (Liimatainen *et al.*, 2018); this approach is applicable also in association with nutrient-rich organic soils characteristic to drained transitional bogs (Wilson *et al.*, 2016).

A strong seasonal and spatial pattern of GHG emission characteristic for northern peatlands is amplified in drainage affected zones. R_{eco} and N_2O emissions increased up to 190 m and 90 m, respectively, from functioning drainage ditches in studied peatlands, while raised bogs with shallow overgrown ditches had still increased R_{eco} but reduced methane emission and N_2O emission at the level typical of natural mires. The main drivers of CH_4 and CO_2 emissions were water table level and soil temperature. Dissolved nitrogen content explained the differences in N_2O emissions between raised and transitional bogs, and between drainage affected zones and natural background areas. Distance from the ditch and dissolved O_2 content were also relevant in explaining the emissions.

3.3 Comparative analysis of GHG fluxes from drained peatland forests and abandoned peat extraction areas (Publication III)

Peatlands with significant anthropogenic impact contribute about 10% of greenhouse gas emissions from the land use sector (Buckmaster *et al.*, 2014). Drainage causes intensive mineralization (decomposition) of organic matter accumulated in peat, which results in significant loss of carbon and plant nutrients, especially nitrogen, from the drained area, while CH_4 emissions usually decrease. These processes were also evident in Publications I and II. According to Raudsaar *et al.* (2014), the drained swamp (decayed-mire) forests cover 14.8% (328,300 ha) of the total forest area. The area of abandoned peat extraction sites amounts to ca 9400 ha of 1 009 100 ha of peatlands in Estonia (Orru & Orru, 2008).

Wintertime GHG fluxes are an important part of the global carbon and nitrogen budget. Snow cover significantly influences wintertime emissions of CO_2, CH_4 and N_2O (Winston *et al.*, 1997; Alm *et al.*, 1999a; Groffman *et al.*, 2001; Maljanen *et al.*, 2007), especially due to global warming affecting snow cover and soil freezing-thawing cycles (Groffman *et al.*, 2001; Kull *et al.*, 2008; Congreves *et al.,* 2018). The aim of the study in Publication III was to estimate emissions of greenhouse gases CO_2, N_2O and CH_4 in winter in Estonian abandoned peat extraction areas and drained peatland forests, and to study how snow cover influences these emissions.

3.3.1 Ecosystem respiration

The daily average soil flux of carbon dioxide varied between −1.1 and 106.0 mg CO_2-C m^{-2} h^{-1} on individual sampling plots. The highest values were measured in October and April, and the lowest values were measured in February (Fig. 12a). The median values of CO_2-C fluxes were similar between the abandoned peat extraction areas (APEA) and drained peatland forests (DPF). The emissions were slightly higher at the DPFs (Fig. 12a). CO_2 emissions had stronger correlation with air and soil temperatures in drained forests than in APEAs. In all areas, the correlation was strongest with the topsoil (5–10 cm) temperatures and lower with deeper (40 cm) soil temperatures.

Easily biodegradable upper layer was removed in APEAs, the remaining deeper layers likely consisted of recalcitrant peat, which is not favourable for CO_2 production as demonstrated by Hilasvuori *et al.* (2013) and Mastný *et al.* (2016). Dense woody root system in DPFs likely contributed to the higher CO_2 flux. The flux was lower in the APEAs without well-developed plant roots. The disturbance of fine roots can be a source for fresh available carbon for microorganisms (Comerford *et al.*, 2013; Wang *et al.*, 2013).

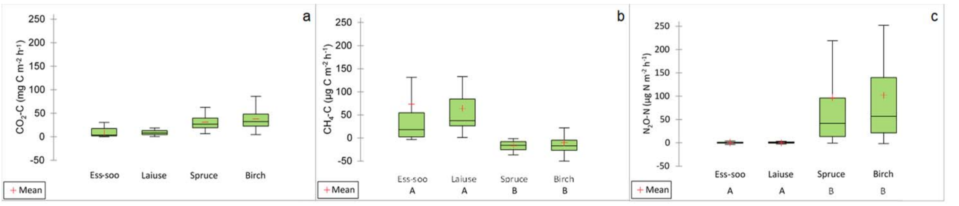

Figure 12. Soil CO_2-C flux (a), CH_4-C (b) and N_2O-N (c) emissions in abandoned peat extraction areas – Ess-soo (n = 25) and Laiuse (n = 25) and drained peatland forests – Spruce (n = 41) and Birch (n = 41). Median, minimum and maximum values and interquartile range (from 25 to 75%) are shown. A and B – statistically significantly differing values (Kruskal–Wallis ANOVA test and multiple comparison of mean ranks test.) Adapted from Publication III.

3.3.2 Methane fluxes

The average CH_4-C fluxes varied between −97.7 and 1201.9 µg CH_4-C m^{-2} h^{-1} on individual sampling plots. There was a significant difference between the APEAs and DPFs. Methane emissions from the APEAs were the highest (Fig. 12b). Methane consumption was observed mainly in DPFs. There was no clear temporal pattern, however, the fluxes were slightly higher from October to January and the lowest in February and March (Fig. 12b). Wintertime CH_4 release from the APEAs (ES and LA site) on average accounted for 31 and 52% of the total annual emission, respectively, and wintertime atmospheric CH_4 consumption in the drained forests (Spruce and Birch) on average accounted for 46 and 33% of the total annual consumption, respectively.

Negative correlation between methane consumption and soil temperature (20–40 cm) in drained forests and positive correlation with water table depth was observed. In APEAs the CH_4 fluxes correlated positively with air, water and soil temperature and water table depth. Lower soil temperature inhibited CH_4 production in the APEAs, demonstrated also in earlier studies (Melloh & Crill, 1996; Alm *et al.*, 1999a).

3.3.3 Nitrous oxide fluxes

The average fluxes of N_2O-N varied between −32.2 and 1440.8 µg N_2O-N m^{-2} h^{-1} on individual sampling plots. There was a significant difference between the APEAs and DPFs (Fig. 12c). Emissions of N_2O correlated negatively with water table depth in all sites. The N_2O emissions were the highest at soil temperatures around 0 °C and 8 °C (see Fig. 10b in Publication III). Water table depth significantly affected N_2O fluxes. The highest fluxes were observed at the water table depths from −30 to −40 cm. This is associated with the optimal soil moisture level for microbial activity in the soil matrix. Wintertime N_2O release from the Norway spruce and Downy birch DPF sites accounted for 87% of the total annual emission.

The N_2O fluxes were influenced by soil properties, the ecosystem and the weather. Relationships between these factors are not straightforward, correlations are usually moderate as demonstrated by Flessa *et al.* (1995). The highest peaks of N_2O emission occurred during freeze–thaw events in November and March, the months with the highest emissions. A similar wintertime peak in N_2O emission was also observed in Publication II. The high N_2O peaks during the freeze–thaw events can be explained by a release of fresh organic carbon as energy source for denitrification, by decomposing soil aggregates and destructing cells of soil organisms and plants (Oechel *et al.*, 1997; Hao *et al.*, 2006).

3.4 Ecosystem CO_2 emission in intact and disturbed peatlands determined using the remotely sensed land surface temperature (Publication IV)

The thermal regime is among the main factors controlling CO_2 and CH_4 fluxes in peatlands. This is evident from Publications I, II, III and V as from previous studies. Carbon fluxes were shown to have positive exponential relationships with peat temperatures at different depths – −20 cm (Helbig *et al.*, 2019), −10 cm (Davidson *et al.*, 2019) and −5 cm (Acosta *et al.*, 2017) – as well as with surface temperature (Huang *et al.*, 2020b). However, the limited spatial coverage of in-situ temperature measurements enables the modelling of C fluxes only at the plot scale. Instead, the application of remotely sensed parameters, including land surface temperature (LST), can be used to model R_{eco} in peatlands globally (Lees *et al.*, 2018). We studied potential of satellite based LST to assess the ecosystem respiration in disturbed (drained and extracted) peatlands. Previous studies have shown the importance of LST for R_{eco} estimations in different ecosystems (Nykanen *et al.*, 1995; Le Mer & Roger, 2001; Levy & Gray, 2015). So far, none of the studies has addressed the potential use of satellite based LST as a proxy of in-situ measured temperatures for modelling R_{eco} in disturbed peatlands.

MODIS and Landsat have long-term LST time series with high temporal frequency of images. The profiles of simultaneously in-situ measured temperatures at different depths together with remotely-sensed Landsat and MODIS LST values are shown in Figure 13. Median peat temperatures decreased with depth, while the highest temperature difference occurred between T_0 and T_{10}. In drained and extracted sites peat surface temperature variability has bimodal distribution (Fig. 13d-e). In contrast, hummocks, hollows and flooded sites have lower temperature variability and close to normal temperature distribution at almost all depths (Fig. 13a–c).

Repeated measures correlation (rmR) as well as Pearson's R between LST and in-situ measured temperatures were for all sites (except flooded), and both for MODIS (Fig. 13b–e) and Landsat LST (Fig. 13b–c) the highest with T_0. Correlations between LST and in-situ temperatures were higher for disturbed sites than for intact ones.

We used daytime MODIS LST data with synchronized ground based surface temperature measurements. Both, LST and measured surface temperatures reflected similar temporal dynamics at plot scale (Fig. 13). This is particularly important for disturbed sites, where R_{eco} was mainly driven by thermal conditions (see Fig. 5 in Publication IV).

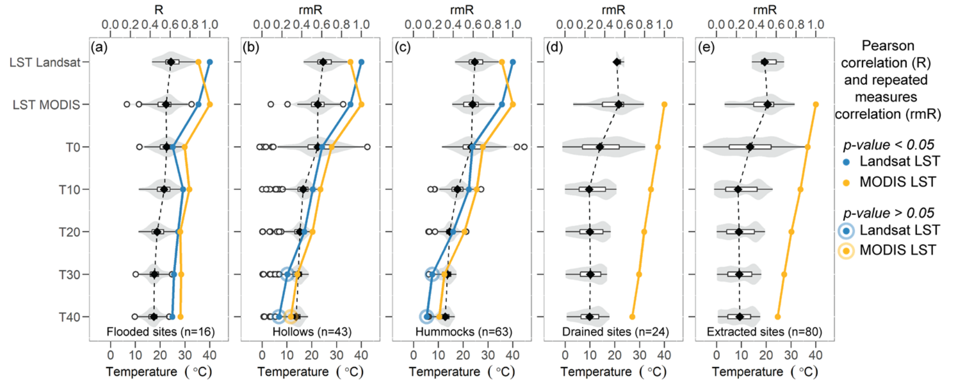

Figure 13. Profiles of temperature variation (boxplot) and distribution (shaded area) sensed by Landsat and MODIS, measured at the surface level (T_0) and 10–40 cm depths in the peat (T_{10}–T_{40}) for five studied groups. The median values (black diamond) for the temperatures are connected with a dashed line. Blue and orange dots represent Pearson correlation (R) and repeated measures correlation (rmR) between Landsat LST and MODIS LST correspondingly and in situ measured temperatures. Figure from Publication IV.

To estimate the potential of LST to be used instead of in situ measured temperatures in R_{eco} modelling, CO_2 fluxes with T_0 as well as with MODIS LST data were modelled. R_{eco} values were generally modelled with higher accuracy for disturbed peatlands (Fig. 14) where T_0 had stronger relationship with CO_2 fluxes.

Comparison between measured and modelled CO_2 fluxes reveals that generally, we fail to catch the variability of CO_2 in intact sites (Fig. 14a). In intact sites modelling approach gave only moderate results as model failed to predict R_{eco} CO_2 fluxes higher than 100 mg C m^{-2} h^{-1}. This was evident both when using T_0 and MODIS LST as model input in the intact sites. Modelled CO_2 fluxes were in better agreement with measured data in the disturbed sites (Fig. 14b, c).

After model modification we used MODIS LST instead of T_0 and obtained R^2 equal to 0.26 for modelled R_{eco} in intact sites. In drained and extracted sites R^2 was 0.65 and 0.69, correspondingly. Junttila *et al.* (2021) demonstrated that using remotely sensed LST and EVI data jointly, the average R^2 was 0.56 among five peatlands. In the same study the lowest R^2 was obtained for a bog site (0.23), in fen sites R^2 was varying from 0.51 to 0.85.

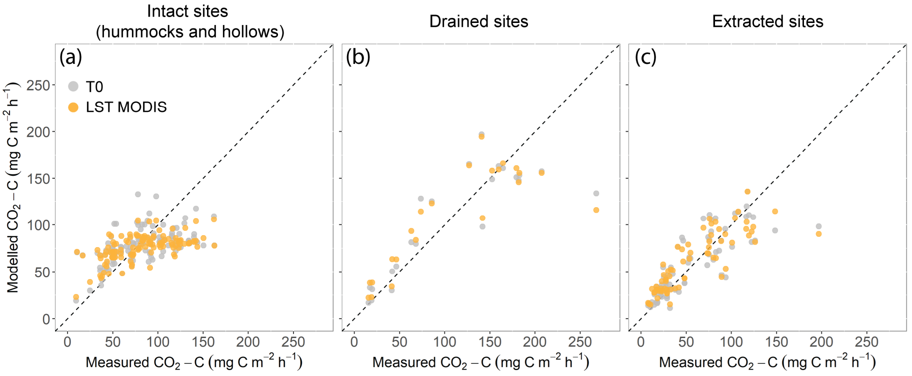

Figure 14. CO_2 fluxes measured in situ and modelled with surface temperature – T_0 (grey circle) – and remotely sensed MODIS LST (orange circle) for intact (joint hummocks and hollows), drained and extracted sites. The dashed line shows a 1:1 line. Figure from Publication IV.

A weak rmR was observed both between LST and in situ temperatures, between in situ temperatures and CO_2, and CH_4 fluxes in intact sites. LST has a weak to moderate association with soil temperatures in bogs (Burdun *et al.*, 2019). The strength of this association decreases with soil depth. Huang *et al.* (2020a) showed that LST dynamics is highly dictated by incident solar radiation, while deeper soil temperatures react slowly with fewer fluctuations.

LST reveals weaker rmR with T_0 in intact sites; presumably, this was primarily caused by more diverse vegetation cover and more pronounced microtopography. Through evapotranspiration, mosses cool the surface and perform as a thermal insulation layer (Kim & Verma, 1996; Blok *et al.*, 2011). In opposite, the disturbed sites with deeper WTD, covered sporadically with sedges and open peat surface, had higher rmR between LST and T_0–T_{40}.

Our study demonstrated that remotely sensed LST is a powerful tool for modelling R_{eco}, particularly in disturbed peatlands and could be potentially applied in large peatlands without ground based observation data.

3.5 Latitudinal gradient of GHG fluxes in peatlands (Publication V)

Processes affecting CO_2 and CH_4 emissions and their budgets have been relatively well studied in northern temperate peatlands (Frolking *et al.*, 2011; Haddaway *et al.*, 2014; Oertel *et al.*, 2016), whereas similar studies are almost absent in southern Patagonia and the high-altitude Andean peatlands, both of which are currently under heavy anthropogenic pressure. Global studies on GHG fluxes and their drivers have showed significant latitudinal and altitudinal dependence (Turetsky *et al.*, 2014; Humphrey *et al.*, 2021; Ribeiro *et al.*, 2021; Publication VI).

Thus, the objectives of Publication V were to compare greenhouse gas emissions in natural and managed peatlands to examine the effect of management on GHG emissions and identify the environmental parameters affecting them. We analysed CO_2, CH_4 and N_2O emissions related to the physical and chemical conditions of the peat: in a natural and managed transitional bog in Quebec, Canada, a natural páramo and grazed peatland in the Colombian Andes, and a bog and a fen in Tierra del Fuego, Argentina.

3.5.1 Ecosystem respiration

Local hydroclimate and vegetation are important factors determining R_{eco} along latitudinal and altitudinal gradients. Temperature and solar radiation were the main parameters differentiating studied areas in Argentina, Colombia and Canada. Median R_{eco} (CO_2-C) fluxes varied between 26.8 in the Argentinean natural and 572.4 mg C $m^{-2}h^{-1}$in the Canadian managed sites (Fig. 15A1). While the median fluxes in the natural sites remained below 50 mg C $m^{-2}h^{-1}$, the fluxes in the managed sites increased more than tenfold from Argentina to Canada along the temperature and management gradient. The variability of the fluxes within the studied sites increased with more continental climate and management intensity (Fig. 15A1). In the natural sites, the variation of fluxes was low (6.4 to 87.3 mg C $m^{-2}h^{-1}$). R_{eco} fluxes had the strongest positive correlations with water temperature and soil temperature (at 10, 20, 30 and 40 cm depth), in both the natural and managed sites (see Table 3 in Publication V). The correlations between R_{eco} and environmental parameters were stronger in the managed sites. Non-linear regression formula between CO_2-C and soil temperature at 10 cm depth ($r = 0.92$; $p < 0.05$) is presented in Fig. 15A2. Soil dry matter content played an important role in explaining CO_2-C emissions (see Table 3 in Publication V; Fig. 19A).

3.5.2 Methane fluxes

Methane fluxes are mainly dependent on water table depth and temperature. In Colombia, both in páramo and managed peatland pasture, constantly humid climate with fluctuating temperature prevails. Thus the methane fluxes are also with high variability, in managed site from 1600.6 to 3128.3 and from −1.5 to 2252.2 µg CH_4-C $m^{-2}h^{-1}$ in natural site (Fig. 15B1). High varability was observed also in Canadian managed peatland site (−9.8 to 2414.3 µg CH_4-C $m^{-2}h^{-1}$), while median flux of CH_4-C in Canadian natural site was −0.8 µg CH_4-C $m^{-2}h^{-1}$.

In natural sites the CH_4-C emissions were positively correlated with water table depth. Soil temperatures at different depths did not affect emissions in natural sites, whereas in the managed sites soil temperature positively correlated with methane emission (see Table 3 in Publication V). Contrarily, in managed sites water level had no significant correlation with methane flux. Soil water oxygen content correlated negatively ($r = -0.55$; $p < 0.05$ in managed sites)

(Fig. 15B2) and soil log (C/NO_3-N) ratio positively with CH_4-C emission both in natural and managed sites.

The Colombian natural site, with moderate temperature and high water table, had relatively high CH_4-C emissions. Meanwhile, the Argentinean natural site, having a high water table, also had low CH_4-C emissions, probably limited by the low temperature. Any management system that involves large N inputs into the soil, like the intensively grazed pasture in Colombia, may have a significant effect on the production of CH_4 in the soil (Gregorich *et al.*, 2005).

3.5.3 Nitrous oxide fluxes

Nitrous oxide fluxes are usually less affected by temperature than fluctuating water regime and soil properties, particularly nitrogen availability. Therefore managed nutrient rich peatlands are normally performing as emitters while natural sites act as low emitters or sinks for N_2O-N. Median N_2O-N emission was high in Canadian managed site (58.4 µg N_2O-N $m^{-2}h^{-1}$; Fig. 15C1). In all other sites median N_2O-N fluxes remained close to zero (from −8.4 in the Colombian managed to 9.8 µg N_2O-N $m^{-2}h^{-1}$ in the Argentinean natural site). N_2O flux had significant correlation with water temperature, soil temperatures at all measured depths, soil Ca and Mg content and C/N ratio in managed peatland sites (see Table 3 in Publication V; Fig. 15C2). Ca and Mg content, and C/N ratio are all parameters describing soil fertility, thus supporting microbial processes which produces nitrous oxide under fluctuating water regime.

The Canadian continental summer with high temperatures and precipitation variability creates suitable conditions for fast mineralization and N_2O-N emission. Other studies on N_2O emissions from farmed peat soils give similarly high values as in our Canadian drained peatland site (Regina *et al.*, 2004; Publication VI). Disturbances such as the ploughing on organic soils can significantly increase N_2O and CO_2 effluxes, whereas CH_4 emission may also appear (Merbold *et al.*, 2014). N_2O emission is often localized in hotspots, the occurrence of which may be related to the distribution of anaerobic microsites and carbon availability (Gregorich *et al.*, 2005). Higher rates of denitrification have been linked to higher levels of mineralizable organic matter (Tenuta *et al.*, 2000). In Colombia and Argentina both the temperature and precipitation regime is more stable, resulting in minor N_2O fluxes.

The water table was relatively uniform and high both in the natural and managed sites during study period in the humid climates of Patagonia and páramo, but significantly lower in the Canadian natural bog (see Table 2 in Publication V). Canada experienced a prolonged hot and dry period preceding our sampling in July 2012. This had a stronger effect on the mainly precipitation-fed natural site than on the heavily degraded and compacted managed peatland, partly fed by groundwater. That was also confirmed by lower dissolved oxygen content (2.06 vs 4.02 mg l^{-1}) and a nearly uniform soil temperature profile in the upper 40 cm layer (from 21.8 to 19.6 °C) in the managed site compared to a steep soil temperature gradient (from 19.2 to 13.1 °C) in the natural site.

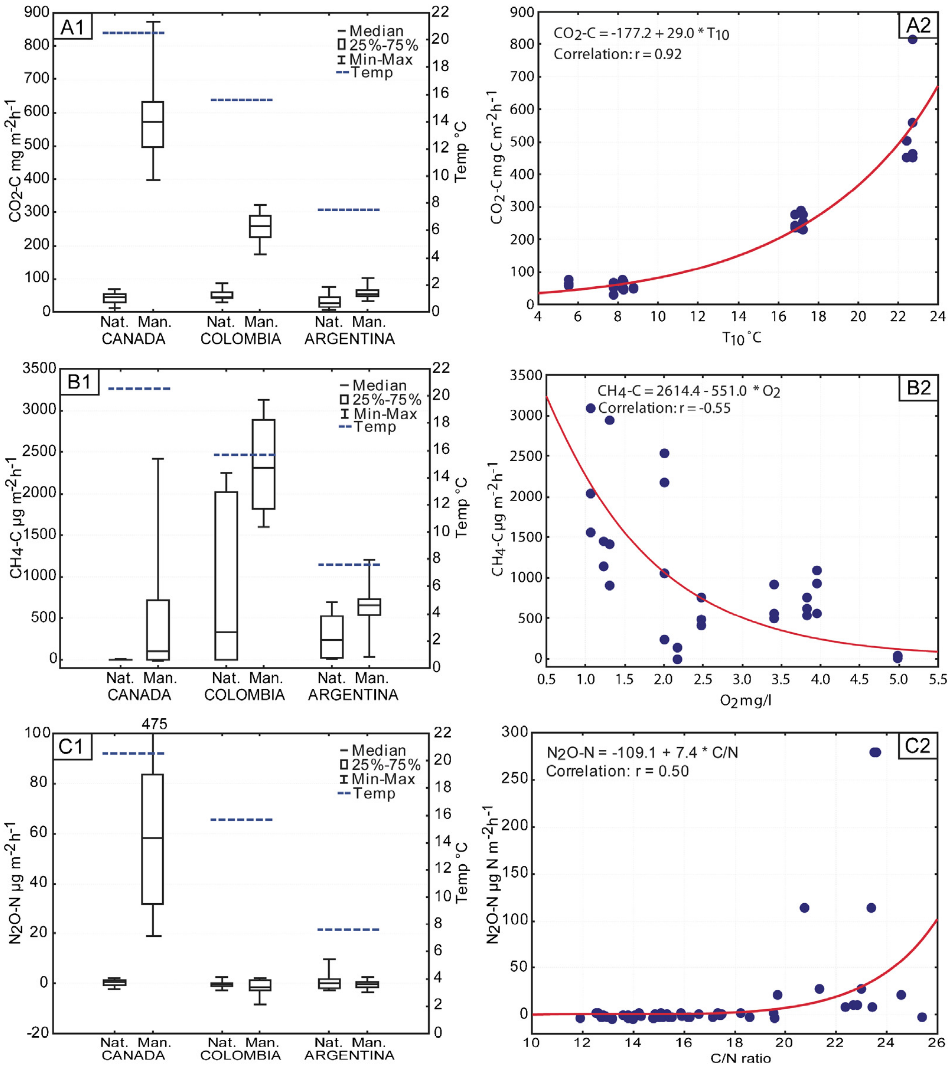

Figure 15. CO_2-C (mg C m^{-2} h^{-1}) (A1), CH_4-C (µg C m^{-2} h^{-1}) (B1) and N_2O-N (µg N m^{-2} h^{-1}) (C1) emissions from natural (Nat.) and managed (Man.) sites and the mean soil temperature at 10 cm depth averaged over the time of measurement sessions in Canada, Colombia and Argentina. The strongest relationships in managed sites were between CO_2-C efflux and temperature (°C) at 10 cm depth (A2); CH_4-C emission and dissolved oxygen content (mg/l) in water (B2); N_2O-N emission and C/N ratio (C2). Figure from Publication V.

3.6 The impact of soil moisture and temperature on N_2O fluxes from global organic soils (Publication VI)

Organic soils make up more than one-tenth of the world's soil nitrogen pool (Batjes, 2014) and are a significant global source of the greenhouse gas nitrous oxide. Quantifying the influence of both increasing rates of land drainage and climate change on organic soil N_2O fluxes is thus critically important (Butterbach-Bahl *et al.*, 2013; Publication III). So far N_2O fluxes from different land uses and ecosystems are usually assessed based on the application rate of fertiliser (or atmospheric reactive N deposition for non-cultivated soils) and broad land-use categories; these models also do not take into account climate-related changes (Alm *et al.*, 2007). To map N_2O fluxes from organic soils we undertook a standardised global survey of in situ measurements together with ancillary key environmental drivers. Based on that we developed a statistical model of N_2O emissions that would be applicable to a wide range of biomes and environmental conditions.

Of the 18 parameters assessed, soil NO_3^- was the strongest predictor of site-mean N_2O, explaining 60% of the variation in log N_2O flux (see Fig. 3a in Publication VI). Inclusion of site-mean volumetric water content (VWC) raised the explanatory power of the multiple regression generalised additive model (GAM) to 72% ($n = 58$; $R^2 = 0.72$; $p < 0.001$; Fig. 16a). The relationship between the mean N_2O fluxes and VWC was best described by a bell-shaped GAM regression curve ($R^2 = 0.78$; $p < 0.001$).

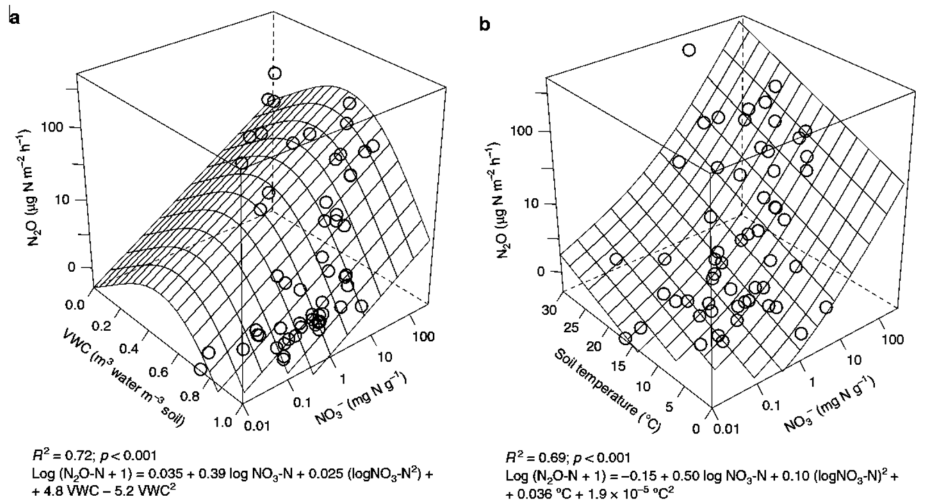

Figure 16. Site-mean N_2O flux multiple regression models. Soil nitrate (NO_3^-) and volumetric water content (VWC) (a); soil nitrate and temperature at 40 cm depth (b). N = 58. Adapted from Publication VI.

Fluctuation around the intermediate VWC (~0.5 m^3 m^{-3}) creates variability in the oxygen content within the soil profile, stimulating mineralisation and nitrification. This contributes to higher NO_3^- content (Linn & Doran, 1984; Kasimir-Klemedtsson *et al.*, 1997; Golovchenko *et al.*, 2007; Rubol *et al.*, 2012) and intermediate VWC also promotes incomplete denitrification (Davidson *et al.*, 2000; Ball *et al.*, 2007; Farquharson & Baldock, 2008; Balaine *et al.*, 2013; Benanti *et al.*, 2014; Leppelt *et al.*, 2014). The maximum N_2O emission at the intermediate VWC means that both wetting from lower moisture values and drying from higher moisture will increase N_2O emissions. At a VWC of ~0.8 m^3 m^{-3}, oxygen concentration in the pore water is 5–9% of saturation, which is low enough to trigger N_2O production but insufficient for complete denitrification (Linn & Doran, 1984; Rubol *et al.*, 2012; Liang *et al.*, 2016).

We found only a weak relationship between N_2O fluxes and soil temperature (40cm-depth temperature log GAM $R^2 = 0.21$, $p < 0.001$; see Fig. 3c in Publication VI). The soil temperatures normalised to local annual air temperature maxima gave even lower correlation values (e.g. with temperature at 40 cm-depth log GAM $R^2 = 0.09$, $p = 0.018$). This may have been partially due to the short time span of our measurements per site and also suggests that the high N_2O fluxes were measured in soils where temperature was not the limiting factor. A multiple regression GAM model containing soil temperature at 40 cm depth and log NO_3^- explained 69% of log N_2O fluxes ($n = 58$; $R^2 = 0.69$; $p < 0.001$; see Eq. (2) in Publication VI; Fig. 16b). Within our drained sites ($n = 27$) the temperature relationship was somewhat stronger ($R^2 = 0.27$; $p < 0.0078$). This shows that organic soils exposed to warmer conditions, such as in the tropics, can act as N_2O emission hotspots where soil moisture is optimal and NO_3^- is above a threshold of 5 mg N kg^{-1} (Fig. 16a). This is in good agreement with the study by Jauhiainen *et al.* (2012), conducted in tropical peatland. There the average N_2O fluxes and also the total of monitored GHG emissions were highest in a drainage-affected forest, which was characterized by continuously labile nitrogen availability from vegetation and water table typically below the surface.

Our findings highlight the importance of soil nitrate, moisture, and temperature in organic soils as potential significant contributors to N_2O emissions. Our global-scale models show that constantly high soil moisture results in low N_2O emissions, whereas drainage creates fluctuation around the intermediate soil moisture and thus increases N_2O emissions from organic soils.

3.7 Environmental indicators of GHG fluxes across the scales: Synthesis of derived results for further use in remote sensing and modelling

Peatland-atmosphere carbon exchange depends largely on the thermal and hydrological regimes of the ecosystem throughout the scales as evident from Publications I–VI. Increasing air and soil temperatures can stimulate microbial activity, resulting in higher CO_2 emissions (Juszczak *et al.*, 2013).

Methane fluxes from peatlands depend primarily on water table, which define in large part the net balance of CH_4 production by methanogenic archaea and CH_4 oxidation by methane oxidizing bacteria. Methane production is the final step in anaerobic organic matter breakdown (Fig. 17), whilst CH_4 oxidation occurs preferentially under aerobic conditions, mainly in acrotelm (Le Mer & Roger, 2001). Methane production and oxidation is a complex interaction influenced by biotic diversity and activity, soil water, oxygen concentrations within the soil and redox potential and transport processes within the soil-water-plant-atmosphere (diffusion, ebullition and transport via aerenchyma and xylem) (Lai, 2009; Levy *et al.*, 2012; Turetsky *et al.*, 2014).

The N_2O emission from peatlands is closely related to peat nitrogen content and thus with peatland types (bogs and fens), and further-more, peat degradation is the main driver of N_2O and DOC release (Liu *et al.*, 2019).

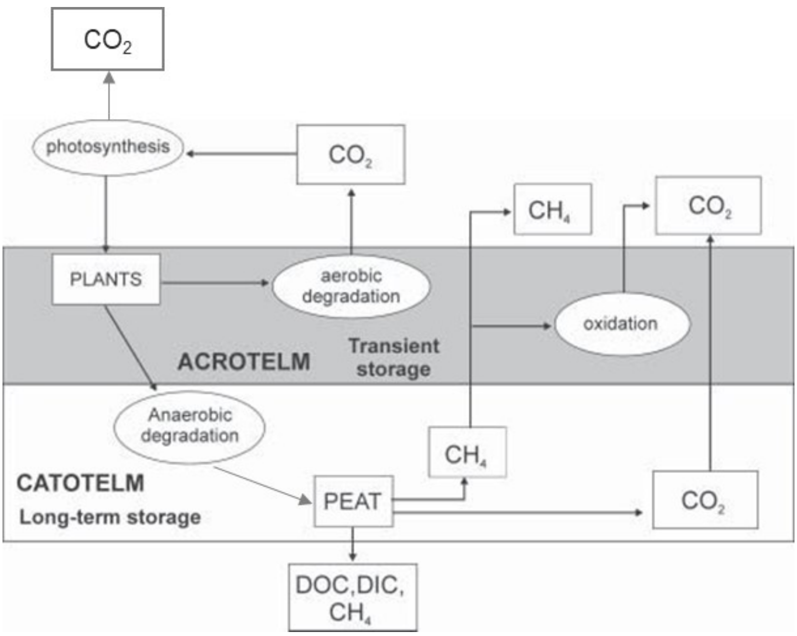

Figure 17. Peatland carbon cycle (acrotelm represents oxidized region of peat and catotelm represents anoxic region of peat). Adapted from Brown (1998).

In agreement with previous studies on the specific aerobic-anaerobic conditions in peat deposits and the microbiological processes affecting GHG fluxes in peatlands, the most universal environmental indicators across the scales from local to global were temperature and water table depth (Fig. 18; 19; 20; 21).

CO_2 fluxes are positively associated with temperature increase from high latitudes towards low latitudes (Publication VI), with the highest fluxes of CO_2 in the summer months in temperate zone. In contrast, the lowest values of CO_2 fluxes are present during the winter season with frozen ground (Publications II, III, IV). In addition to the positive association present in both CO_2 and CH_4 fluxes and temperature, the WTD affects the fluxes in the opposite way. Lower WT increases CO_2 emission and inhibits CH_4 flux; higher WT increases CH_4 emission and inhibits CO_2 emission. Similarly to CO_2, the highest CH_4 fluxes occur in the summer period (Publications II, III, IV). The relations of T_0 and WTD involved in CO_2 and CH_4 fluxes for five studied groups of peatlands in Estonia are presented in Figure 18 (Publication IV).

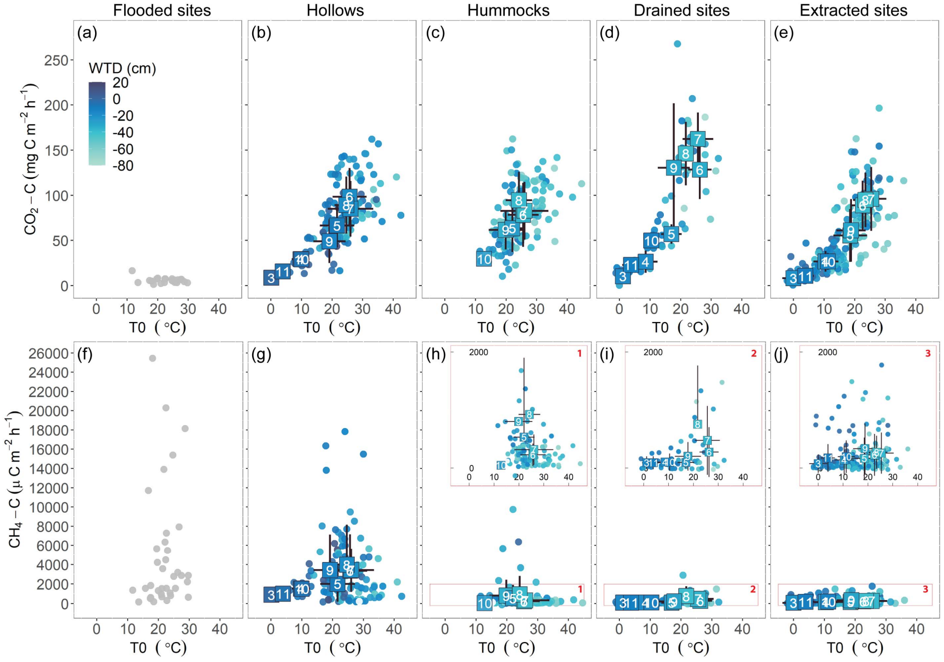

Figure 18. Scatterplots of surface temperature (T_0), CO_2 and CH_4 fluxes (circles). Monthly fluxes and T_0 averages (square shapes with month numbers) are also given with monthly standard deviations (error bars). Colours indicate the water table depth (WTD) except for flooded sites, where no WTD data are available. Inset graphs in panels (h), (i), and (j) present zoomed-in areas in red rectangles. Figure from Publication IV.

The results from Publication I emphasized the importance of vegetation composition and parameters as indicators of GHG fluxes at the micro-scale, as they affect but also reflect peat chemical and physical properties, water movement and nutrient status. The cover of *Sphagnum* spp. and the height and cover of tree canopy are significant proxies for estimating GHG emission in northern peatlands. The Sphagnum cover is the most universal indicator for all GHG-s, while tree canopy height and cover on the other hand are easily captured with remote sensing techniques with only marginally inferior performance as indicators. Still, as demonstrated in Publication IV, remotely sensed surface temperature may significantly contribute to simple GHG flux models, especially in heavily drainage-affected peatlands.

Intensive peatland management alters the soil C/N balance, increases emissions of CO_2-C, N_2O-N, and leads to higher variability of GHG emissions. More specifically, intensive management of peatlands has a negative effect on carbon balance and increases N_2O-N emissions. In natural areas, the effect of any single environmental variable on GHG fluxes is smaller and emissions are lower than in managed areas where the ecosystem balance is affected and several key factors (e.g. water table level, increased soil temperature, oxygen content, nitrogen availability) contribute to higher emissions. In the study conducted on both the natural and managed sites of the Americas (Publication V), the fluxes of CO_2-C and CH_4-C were explained most importantly by land use and temperature (Fig. 19). Soil dry matter content played an important role in explaining the CO_2-C fluxes, redox potential of soil water was related to CH_4-C fluxes, and the fluxes of N_2O-N were most affected by environmental conditions related to soil nitrogen content and pH (Fig. 19A).

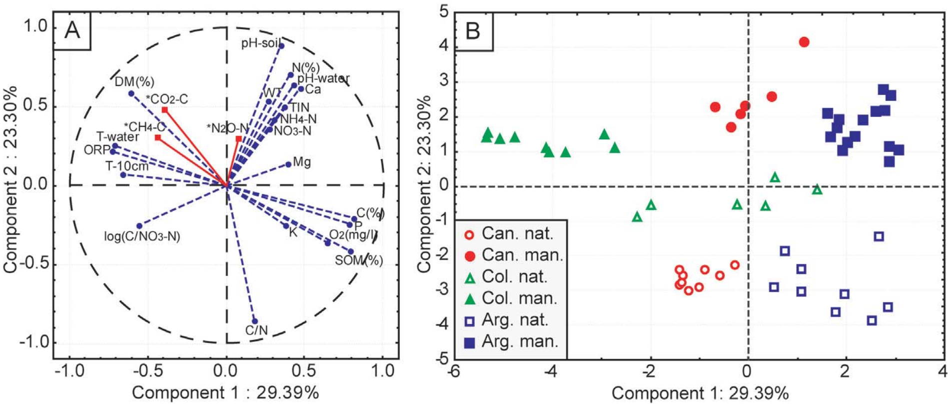

Figure 19. Ordination diagrams of PCA of environmental variables with GHG flux data in natural and managed areas, grouping by variables (A) and by sites (B). See Publication V Fig. 2 for abbreviations of environmental variables. Figure from Publication V.

Both the peatland management intensity and climatic factors on GHG emissions were addressed in Publication VI. The PCA analysis differentiated tropical sites from temperate and boreal ones, and low agricultural-intensity sites (index 0 and 1) from arable sites (index 3) (Fig. 20a, b). Soil temperature and NO_3^- were positively correlated to N_2O emissions, whereas water table, VWC, and soil C were negatively related to N_2O emission (Fig. 20c). The difference between N_2O emissions from drained and natural sites was clear in all three major climate types. The effect of agriculture on N_2O emissions was mainly related to cultivation (Fig. 20b). The more the water regime is affected through intensive management, the more substantial the change of the thermal regime and physico-chemical properties of the soil, resulting in greater GHG emissions. Particularly sensitive to intense management practices is the N_2O flux (Publications III, V, VI).

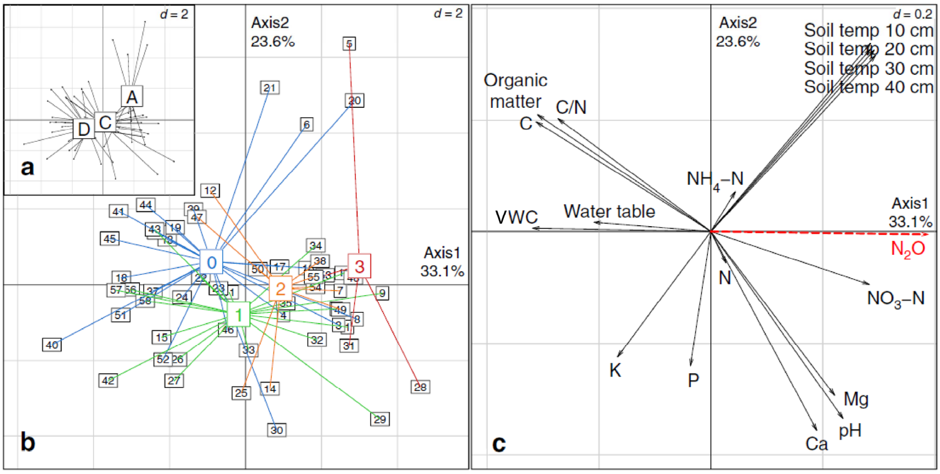

Figure 20. Ordination plots based on principal component analysis grouping sites and variables. Köppen climates (A) tropical, (C) temperate and (D) boreal (a); intensity of agricultural use (0) no agriculture, (1) moderate grazing or mowing, (2) intensive grazing or mowing and (3) arable (b); soil physical and chemical parameters (c). Figure from Publication VI.

According to Publication VI, the fluxes of GHG-s are climate dependent (Fig. 20a), with annually high fluxes due to intensive cycling of substances in the tropics and seasonally fluctuating emissions in the temperate and boreal zone. Furthermore, Alm *et al.* (1999b) have demonstrated in boreal bogs that severe C losses can occur during extended summer droughts as well, without the influence of artificial drainage. Relatively less studied winter season GHG emissions, and their relations to environmental characteristics on different land use types, were addressed in Publication III. Varying environmental factors between the sites such as groundwater depth, temperature, pH and water electrical conductivity (Fig. 21), result in higher CO_2-C and N_2O-N, and lower CH_4-C emissions in peatland forests compared to abandoned peat extraction areas, for instance.

The contribution of different GHG-s to the global warming potential (GWP) differs among peatlands. The components of the determination of wintertime GWP were different for the abandoned peat extraction areas and drained peatland forests. In the DPFs, the flux of N_2O (GWP_{100} 265 times CO_2 equivalent; Myhre *et al.,* 2013) was the main component, showing 3–6 times higher values than that of the R_{eco} CO_2 flux (without accounting for CO_2 uptake by plants). The role of CH_4 was of little importance in DPFs. In the APEAs, CO_2 as R_{eco} and CH_4 made up almost equal parts, whereas the impact of N_2O on global warming was minor. In none of the study sites could the slight consumption of CH_4 compensate for the

GWP from R_{eco} (CO_2) and N_2O emissions. Addressing boreal peatland ecosystems in the context of climate warming, anticipating milder winters in northern regions (Jaagus *et al.*, 2017; Rimkus *et al.*, 2018), it is extremely important to be careful in managing the drained peatland forests where unfrozen ground with lowered water level enables extreme emissions of N_2O-N, accounting 87% of total annual emission in both studied sites (Publication III).

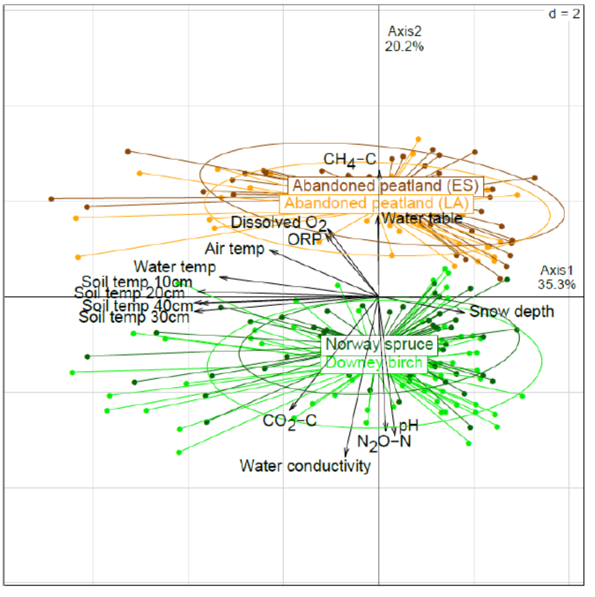

Figure 21. Principal component analysis ordination plot with 95% confidence ellipses showing the grouping of studied sites according to environmental characteristics (n = 215). Figure from Publication III.

Furthermore, the management practices of drained peatlands as well as low yield peatland forests should move towards the restoration of these ecosystems to as close to a natural state as possible. The benefits from these ecosystems as long term carbon sinks etc. are overwhelming their limited wood production, not to mention the direct effect of drained peatlands on climate warming. The prevalent fear of increased methane emissions following peatland restoration is reduced by Järveoja *et al.* (2016), suggesting that in non-flooded conditions WTL changes following peatland restoration on APEA have a limited effect on the CH_4 emissions. However, rewetting decreased the N_2O emissions by the order of 1–2 magnitudes which indicates a high potential of peatland restoration in reducing the N_2O emissions commonly occurring in drained peatlands. The same is demonstrated in our study areas, both in natural peatland regeneration around old overgrown ditches (Publications I, II) and in flooded shallow peat extraction sites (Publication IV).

Interactions between the GHG emissions and various environmental parameters in disturbed homogenous peatlands are amplified and thus more distinctive compared to more diverse peatlands in natural condition (Publications I, IV, V). Therefore, the evaluation of GHG emissions via environmental parameters, including by the use of remote sensing techniques, is more feasible in peatlands with anthropogenic disturbances.

4. CONCLUSIONS

The results of this dissertation demonstrate the extensive impact of artificial drainage on the fluxes of CO_2, CH_4 and N_2O, despite the type of peatland, climate zone or continent studied. Furthermore, the principal driving factors of GHG fluxes are similar across the globe. The studies conducted on Estonian peatlands (Publications I, II and IV) showed the considerable potential of estimating GHG fluxes by indicative environmental parameters, a great number of them detectable by remote sensing techniques.

Based on the results of this thesis and hypotheses tested, it can be concluded:

(1) In general, the 1st hypothesis – drainage impact varies among different GHG fluxes both spatially and temporally – was supported by the results.

Lowered water table increases CO_2 and N_2O emission, and decreases CH_4 emissions. Furthermore, in drained peatlands, minimum water level is a better predictor of GHG fluxes than the average water level.

In temperate peatlands, the seasonal dynamics of CO_2 and CH_4 are strongly temperature-related, while N_2O is more dependent on water-level fluctuations, particularly those caused by artificial drainage.

The drainage-impacted zone is widest for CO_2 and narrowest in the case of N_2O. However, the change in emission intensity within this strongly drainage-impacted zone is the greatest for N_2O and CH_4.

(2) The 2nd hypothesis – predictive value of environmental indicators affected by drainage differs among GHGs – was also supported.

Tree canopy height and cover, moss cover, and water table depth are the most indicative parameters describing the fluxes of CO_2 and CH_4 in both drained bogs and transitional bogs.

In terms of N_2O, soil parameters like organic matter and NO_3 content are also required for flux estimations.

(3) The results support the 3rd hypothesis – easily detectable environmental variables applicable for remote sensing indicate a major part of GHG fluxes.

Tree layer parameters based on airborne LIDAR-data and the remotely sensed land surface temperature can both explain more than half of R_{eco} and CH_4 emissions.

Other simply detectable environmental factors such as tree canopy height and cover, moss cover, and minimum water level have strong indicative capacity for CO_2 and CH_4 fluxes.

For N_2O flux detection, the easily measurable environmental indicators are not sufficient; soil and water chemistry parameters as well as water level fluctuation data are needed.

(4) The 4^{th} hypothesis was supported. The results confirm that leading driving factors such as water table depth or soil water content, soil temperature, soil carbon and nitrogen content are universal indicators for predicting GHG fluxes at both local and global scale.

The results of this thesis provide new findings into enhanced models describing temporal and spatial GHG dynamics, and contribute to the more efficient use of remote sensing methods for large-scale and accurate assessment of peatlands' GHG fluxes, as well as to the more efficient monitoring of status of restored drained bogs. Further research should focus on more precise detection and incorporation of remotely-sensed environmental factors into the GHG flux models.

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SUMMARY IN ESTONIAN

Kasvuhoonegaaside voogude ajaline ja ruumiline käik looduslikes ja kuivendatud soodes

Looduslikud sood on ökosüsteemid, millel on oluline roll kliima reguleerimisel, sidudes atmosfäärist süsinikku ja talletades seda turbana anaeroobsesse veega küllastunud keskkonda, samuti leevendavad sood tulvasid ning säilitavad bioloogilist- ja maastikulist mitmekesisust (Costanza *et al.*, 1997; De Groot *et al.*, 2006; Erwin, 2008). Soode tähtsust on laialdaselt tunnustatud ning tegeletakse nende kaitse ja taastamisega (Joosten & Clarke, 2002; Paal & Leibak, 2011). Turbaalad katavad vaid 3% maailma maismaast (Leifeld & Menichetti, 2018), kuid nendes on talletunud ligikaudu kolmandik globaalsest süsinikuvarust (Post *et al.*, 1982; Gorham, 1991) ja 10% mageveest (Joosten & Clarke, 2002).

Kuna sooökosüsteemid sõltuvad maapinna veega küllastatusest, on nad väga tundlikud hüdroloogiliste muutuste osas, mis võivad olla põhjustatud nii klimaatilistest kui ka inimtekkelistest teguritest, balansseerides seega pikemaajaliselt õrnal turba akumuleerumise ja seda ületava lagunemise vahelisel piiril. Kui globaalsel tasandil mõjutavad soid ja seal turba ladestumist kliimamuutused (Swindles *et al.*, 2019; Humphrey *et al.*, 2021), siis kõigil tasanditel on soode olulisimaks mõjutajaks kuivendus (Holden *et al.*, 2004; Limpens *et al.*, 2008; Leifeld & Menichetti, 2018). Maakasutusmuutus koos kaasneva kuivendusega, eeskätt põllumajanduseks ja metsanduseks, on muutnud süsinikku (C) talletavad sood olulisteks CO_2 ja N_2O allikateks (Joosten, 2009; Maljanen *et al.*, 2010; Günther *et al.*, 2018; Leifeld *et al.*, 2019).

Eestis katavad turbaalad 1 009 100 ha (22,3% riigi pindalast; Orru & Orru, 2008). Peamiselt põllumajandusliku ja metsandusliku kuivenduse tõttu on jätkuvalt looduslikus seisundis soid säilinud vaid 5,5 % alal (Paal & Leibak, 2011). Kuivenduse tulemusel tekkinud kõdusoometsad hõlmavad 328 300 ha (14,8% metsadest; Raudsaar *et al.*, 2014) ning turba tootmisalad ligemale 30 000 ha, millest ligi 10 000 ha on korrastamata hüljatud endised tootmisalad (Paal & Leibak, 2011).

Kasvuhoonegaaside (KHG) – CO_2, CH_4, N_2O – allikate ja heitkoguste määramine ning nende ajalis-ruumiline jaotumine nii lokaalsel kui globaalsel tasandil on äärmiselt oluline, et hinnata süsiniku ja lämmastiku vooge vastavalt turbaala seisundile, majandamisele või seoses kliimamuutustega.

Doktoritöö eesmärgiks oli täpsustada kasvuhoonegaaside vooge looduslikes ja kuivendatud soodes, analüüsida neid vooge mõjutavaid keskkonnaparameetreid ning luua statistilised mudelid, mis võimaldaks kaudsete indikaatorite abil gaasivooge hinnata. Uurimisülesanded olid: (1) kvantifitseerida kuivenduse mõju KHG voogudele; (2) selekteerida kõrge ennustusvõimega keskkonnaparameetrid, mida saaks kaugseire vahenditega mõõtes kasutada KHG voogude hindamiseks; (3) hinnata KHG vooge erinevates kliimavöötmetes ja nende sõltuvust majandamisvõtetest.

KHG voogude, mulla- vee- taimkatte ning kaasnevate keskkonnaparameetrite andmeid koguti 26 turbaalalt Eestist (Publikatsioonid I–IV: 14 raba, 5 siirdesood, 5 jääksood, 2 kõdusoometsa) ning 58 proovialalt 23 erinevast regioonist üle kogu maailma (Publikatsioonid V ja VI).

Nelja aasta jooksul staatilise pimekambrimeetodi igakuiste KHG mõõtmiste põhjal oli aasta keskmine ökosüsteemihingamise (CO_2) mediaanväärtus kuivendusest mõjutatud Eesti rabades 48,7–78,4 mg C m^{-2} h^{-1} suurenedes looduslikus seisundis säilinud soo keskosast kraavituse poole liikudes. Ökosüsteemihingamine oli kraavitusest mõjutatud kuni 190 m kauguseni. Siirdesoodes oli kuivendusest mõjutatud CO_2 voo mediaanväärtus 105,0 mg C m^{-2} h^{-1}, jäädes looduslikus osas (sarnaselt rabadele) 52,0 mg C m^{-2} h^{-1} juurde. Ökosüsteemihingamine on aastaajaline, ligemale 90% emiteerus kasvuperioodil, mil mulla temperatuur oli kõrgeim ja veetase madalaim (Publikatsioonid I, II).

Metaani (CH_4) voog sõltus samuti aastaajast ja muutus kraavide suhtes kaugusvööndiliselt, olles madalaim talvel kuivenduskraavide lähedal ning kõrgeim suve teises pooles looduslikus kõrge veetasemega vööndis. Aasta keskmine metaani emissiooni mediaanväärtus varieerus rabades vahemikus 45,5–2714,6 µg C m^{-2} h^{-1}. Kõige suuremad keskmised vood 15355,6 µg C m^{-2} h^{-1} mõõdeti laugaste servades ja älvestes paiknenud proovivõtupunktides, madalaimad vood aga vahetult kuivenduskraavide lähedal. Siirdesoo metaanivood ületasid rabade omi nii vahetult kraavidest mõjutatud kui ka looduslikus seisundis aladel, jäädes mediaanväärtustelt vastavalt 53,8 ja 5168,3 µg C m^{-2} h^{-1} (Publikatsioonid I, II). Soode metaanivood sõltuvad ruumiliselt peamiselt veetasemest, mis määrab anaeroobses keskkonnas elutegevuse käigus metaani emiteerivate arhebakterite ja aeroobses keskkonnas metaani oksüdeerivate bakterite vahelise tasakaalu ning sõltub temperatuuri aastaajalisest muutusest. Metaani produktsioon ja oksüdatsioon on kompleksne ja seda mõjutab mikrobioloogiline mitmekesisus ja aktiivsus, pinnase vee- ja hapnikusisaldus, redokspotentsiaal ning mulla-vee-taimede-atmosfääri vaheline gaasivahetus (Lai, 2009; Levy *et al.*, 2012; Turetsky *et al.*, 2014).

Kui aasta keskmised dilämmastikoksiidi (N_2O) e. naerugaasivood olid rabades marginaalsed ja aastaajalise varieeruvuseta, jäädes vahemikku 0,2–1,5 µg N m^{-2} h^{-1}, siis siirdesoodes, kus mulla lämmastikusisaldus on suurem, olid need vastavalt 0,5–10,3 µg N m^{-2} h^{-1}, ulatudes kuivendusest enim mõjutatud piirkondades keskmisena kuni 46,2 µg N m^{-2} h^{-1}. Kuivendusest tugevalt mõjutatud tsoonis (kuni 90m), kus turba lagunemine on kiirem, lahustunud lämmastik poorivees kättesaadavam ning veetase lisanduva sademevee näol tugevalt kõikuv, oli N_2O voog aastaringselt kõrgem. Voog intensiivistus lühiajaliselt talvisel ajal seoses maapinna külmumise-sulamise tsüklitega ning saavutas pikemaajaliselt keskmisest kõrgema fooni suveperioodil, mil veetase oli madal, kuid sademetest tulenevalt muutlik (Publikatsioonid I, II). Talvised N_2O vood võivad kuivendatud kõdusoometsade näitel moodustada lausa 87% aastasest emissioonist (Publikatsioon III). Kuivendusest tingitud lämmastikurikka turba lagunemine on peamine turbaalade N_2O emissiooni põhjustaja (Liu *et al.*, 2019) ning suurema lämmastikusisaldusega kuivendatud turbaalad, sealhulgas siirdesood, emiteerivad rohkem naerugaasi (Wilson *et al.*, 2016; Liimatainen *et al.*, 2018) kui rabad või teised toitainevaesemad sookooslused.

Intensiivne turbaalade majandamine muudab mulla C/N tasakaalu, suurendab CO_2 ja N_2O emissioone ja nende varieeruvust. Looduslikel aladel on üksikute keskkonnaparameetrite mõju KHG emissioonidele väiksem kui majandatavatel aladel, kus ökosüsteemi tasakaal on rikutud ning võtmetegurid, nagu sügavam veetase, suurenenud pinnase temperatuur ja hapnikusisaldus ning lämmastiku kättesaadavus, tingivad kõrgema emissiooni nii lokaalsel (Publikatsioonid I–IV) kui globaalsel (Publikatsioonid V, VI) tasandil. Kuivenduse mõju on väga tugev kuni 50 m laiuses tsoonis kraavi ümber ning mõju on tugevam toitainerikkamates soodes.

Kambripõhised KHG mõõtmised on väga kohaspetsiifilised, väikese ruumilise katvusega, aeganõudvad, tööjõumahukad ning seeläbi kulukad. Selleks, et saada adekvaatseid KHG voogude hinnanguid suuremate ja iseäranis raskesti ligipääsetavate piirkondade kohta, oleks tarvis neid hinnata kasutades selleks iseloomulikke gaasivooga tihedalt seotud keskkonnaparameetreid, eelistatult kaugseire vahendite abil seiratavaid.

Nii varasematest uurimustest kui ka käesolevast tööst tuleb esile, et veetase (Dise *et al.*, 1993; Ojanen & Minkkinen, 2019) ja temperatuur (Levy & Gray, 2015; Davidson *et al.*, 2019; Helbig *et al.*, 2019) on olulisimad keskkonnaparameetrid, mis määravad soode KHG vooge. Mõlema eelmainitud parameetri ajaline ja ruumiline varieeruvus on suur ning seetõttu korrektselt kohapealsete vahenditega raskesti mõõdetavad. Kuna soode taimkate on otseselt mõjutatud veetasemest ja mullaprotsessidest ning kaugseire vahenditega lihtsasti tuvastatav, on taimkattemuutuste kasutamine keskkonnamuutuste määramiseks igati asjakohane lahendus (Limpens *et al.*, 2008).

Kuivendatud Eesti soode andmetele tuginedes leiti mitmest regressioonmudelit koostades (Publikatsioon I) rabade ökosüsteemihingamise kirjeldamiseks sobivaim kuivenduskraavi kauguse, puhmarinde katvuse ja puurinde kõrguse kombinatsioon ($R^2_{adj} = 0,47$; $p < 0,000$), 60% metaani voost oli kirjeldatav turbasammalde katvuse ja 40 cm sügavusel mõõdetud mullatemperatuuriga. Siirdesoode CO_2 voogu kirjeldab minimaalse veetaseme, kuivenduskraavi kauguse ja puurinde kõrguse kombinatsioon ($R^2_{adj} = 0,58$; $p < 0,000$), metaani voogu aga minimaalne veetase koos puurinde katvusega ($R^2_{adj} = 0,64$; $p < 0,000$). Nii turbasammalde kui ka puurinde katvus kumbki eraldi võetuna kirjeldasid 42% siirdesoode metaanivoost. N_2O voo hindamiseks on lisaks vaja konkreetsemate mullaparameetrite laboris määramist. Rabades on vaja teada turbasammalde katvust, mulla nitraatlämmastiku ja orgaanilise aine sisaldust ja pH-d, ent mudeli kirjeldusvõime on ikkagi vaid 37%. Siirdesoodes, kus naerugaasivoog on märkimisväärselt kõrgem, on olulised turbasammalde katvus, minimaalne veetase ja mulla orgaanilise aine sisaldus ($R^2_{adj} = 0,52$; $p < 0,001$), aga ka ainuüksi turbasammalde katvus kirjeldab siirdesoodes 43% N_2O voost. Tulemuste põhjal võib väita, et turbasammalde katvus, aerolaserskanneerimise teel määratav puurinde kõrgus ja katvus on olulised indikaatorid, mis aitavad hinnata turbaalade KHG voogusid.

Kaugseire lahendusena saab ökosüsteemihingamise indikaatorina kasutada ka satelliidilt mõõdetavat maapinna temperatuuri (Publikatsioon IV). Meetod on töökindlam homogeensematel kuivendatud ($R^2 = 0,65$) ja kaevandatud ($R^2 = 0,69$) aladel ning raskemini rakendatav mikroreljeefilt ja keskkonnatingimustelt mitmekesisemates looduslikes rabades ($R^2 = 0,26$).

Doktoritöö toob esile, et inimtekkelise kuivenduse mõju on ulatuslik CO_2, CH_4 ja N_2O voogudele ja seda sõltumata turbaala tüübist, klimaatilisest vööndist või uuritavast kontinendist. Lisaks on KHG voogusid mõjutavad peamised tegurid nagu veetase või mullaniiskus, mulla temperatuur, mulla süsiniku ja lämmastiku sisaldus sarnaselt olulised nii lokaalsel kui ka globaalsel tasandil.

Turbaalade majandamisel tuleb olla ettevaatlik, et ei rikutaks seal aastatuhandete jooksul väljakujunenud erinevate keskkonnatingimuste vahelist tasakaalu, mis säilitab soid pikaajalise süsinikureservuaarina. Looduslikult funktsioneerivad sood on igasugusele hüdroloogilise režiimi muutusele äärmiselt tundlikud ning seetõttu tuleks turbaaladel majandades leida alternatiivseid viise kuivendamise vältimiseks.

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