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Master Thesis

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Methane and carbon dioxide emissions from a rewetted pond in Gribskov, Denmark

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Topic description:	Methane and carbon dioxide fluxes from a pond in a rewetted peatland located in Gribskov, a Danish Beech Forest Ecosystem. The fluxes are assessed through the usage of two different methods: automatic chambers and bubble traps. A literature review is carried out to compare field study data with existing data.
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Abstract

Natural peatlands contain the largest share of terrestrial carbon storage in the world but have suffered severe drainage due to anthropogenic uses. Greenhouse gas (GHG) emissions associated with the drainage of peatlands are attempted combatted by rewetting projects. However, GHG emissions from these new ecosystems are rarely quantified, and available studies indicate a large variety in reported emissions. This thesis investigates carbon dioxide (CO_2) and methane (CH_4) emissions from ponds emerging in rewetted peatlands.

CO_2 and CH_4 emissions are studied via two traces: a literature review and a field study. The literature review encompasses 44 studies on either CO_2 and/or CH_4 fluxes from wetlands with similar characteristics to the possible outcomes of restoring peatlands. The field study was situated at Mårumhus Pond in Northern Zealand, Denmark, a rewetted pond in a former drained peatland. The study was conducted from the 19th of September to the 18th of October 2023, including two methods for measuring emissions: automatic chambers (AC) and bubble traps (BT).

The mean and standard deviation of CO_2 fluxes reported in the literature review and measured with the AC-setup were 10,766 ± 12,440 mg CO_2 m^{-2} d^{-1} and 6,912 ± 5,905 mg CO_2 m^{-2} d^{-1}, respectively. The CO_2 fluxes measured by BT were excluded during the data processing, as the fluxes were remarkably low. The mean and standard deviation of CH_4 fluxes reported from the three different methods were highly varying, with means of 128 ± 187 mg CH_4 m^{-2} d^{-1}, 557 ± 714mg CH_4 m^{-2} d^{-1}, and 61 ± 34 mg CH_4 m^{-2} d^{-1} from the literature review, AC, and BT respectively.

The reported and measured emissions indicate responses to spatial attributes such as waterbody and ecosystem types and are highly influenced by temporal factors such as water and air temperatures, weather events, and stratification.

AC and BT showed large variations in measured fluxes, with AC exhibiting nearly 10 times higher CH_4 fluxes than BT, while BT failed to measure valid CO_2 emissions. The ranges observed in CO_2 and CH_4 fluxes are substantial across all three methods, suggesting variations between spatiotemporal attributes. Additionally, more studies on GHG emissions from ponds in rewetted peatlands as well as studies comparing measuring methods are needed.

Acknowledgements

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1 Introduction

Natural peatlands serve as the largest terrestrial long-term net carbon storage in the world (Joosten et al., 2016). According to Cooper et al. (2014) peatlands in the Northern Hemisphere store 473-621 Pg C corresponding to 40% of global soil carbon, even though peatlands cover only 3% of the Earth's terrestrial surface. This makes peatlands count as a larger storage of carbon than all forest biomass in the world (Tanneberger et al., 2020).

Globally, Europe has suffered the greatest loss of peatlands throughout the last centuries, with a total loss of 52% (Joosten & Clarke, 2002). A large part of this loss is due to anthropogenic drainage for agriculture and forestry (Cooper et al., 2014). It is estimated that 3% of total global anthropogenic greenhouse gas (GHG) emissions (2014 numbers) stem from drained peatlands amounting to 1.15 Gt carbon dioxide equivalents (CO_2-eq) $year^{-1}$ (Joosten et al., 2016). In Denmark, drained peatlands emit around 4.8 million tons CO_2-eq $year^{-1}$, and account for 1/3 of the GHG emissions from the agricultural sector (Landbrugsstyrelsen, 2023b), even though they only occupy 7% of the total agricultural area (the Danish Climate Council, 2020).

To reduce the GHG emissions associated with drained peatlands, rewetting has received increasing political awareness throughout the last decades (Cooper et al., 2014). The Paris Agreement (Paris Agreement, 2015) suggests that all drained peatlands must be rewetted to reach net zero CO_2 emission by 2050 (Kreyling et al., 2021; Tanneberger et al., 2020). Rewetting as a tool for reducing GHG emissions has been used by the Danish government since 2016 (Miljøministeriet, 2016) (latest update: 08/02/2022). Additionally, in 2021 the Danish parliament made a political agreement called '*A Green Transition of the Danish Agriculture*', aiming at rewetting 100,000 ha of peatlands by 2030 (Regeringen et al., 2021). Approximately 187 ha of drained peatlands have been rewetted between 2021 and November 2023, while 38,441 ha are under preliminary investigation (Landbrugsstyrelsen, 2023a), and around 170,000 ha of the Danish peatlands remain drained (the Danish Climate Council, 2020).

Peatland rewetting holds a large potential for preventing GHG emissions from drained peatlands (Cooper et al., 2014; Tanneberger et al., 2020). According to the Intergovernmental Panel on Climate Change (IPCC) rewetting of peatlands, especially in the boreal and temperate zone, will commonly cause GHG emission reductions (Couwenberg & Fritz, 2012). Furthermore, the Danish Climate Council (2020) promotes rewetting as a socio-economically cost-effective measure with the potential of achieving a fifth of the Danish goal of a 70 % reduction in total GHG emissions if all peat soils are rewetted.

In contrast, other studies have found that GHG emissions after rewetting exceed the emissions associated with drainage due to increased methane (CH_4) emissions, with a global warming potential (GWP) 28 times stronger than CO_2 (Cooper et al., 2014; Couwenberg & Fritz, 2012; Myhre et al., 2014). Peacock et al. (2021b) have found that emissions from artificial waterbodies are four times

larger than those from natural waterbodies and that waterbodies situated on peat soils emit significantly more GHG emissions compared to those on mineral soils.

The complexity of studied emissions from rewetted peatlands may be associated with different outcomes from rewetting projects. Draining peatlands leads to land subsidence alongside peat compression which can lead to severely decreased hydraulic conductivity of the peat (Craft, 2016; Kotowski et al., 2016). Depending on the degradation of the peat, rewetting of formerly drained peatlands may result in different hydrological states (Kreyling et al., 2021).

Kreyling et al. (2021) have found that rewetted peatlands lead to highly different ecosystems compared to natural peatlands differing in biodiversity, ecosystem functioning, and land cover characteristics even many years after rewetting. The rewetting of peatlands creates novel ecosystems where the knowledge from natural systems cannot be applied (Kreyling et al., 2021).

A rewetted peatland typically contains wetter and dryer surface types across a larger area (Franz et al., 2016). However, rewetting can also result in two extreme outcomes categorized as (1) a restored vegetated peatland with a fluctuating water table depth (WTD) below ground level or (2) the creation of a shallow pond with a open water surface staying above ground level and with sparse vegetation (Christiansen, 2019; Franz et al., 2016; Kreyling et al., 2021).

The creation of shallow ponds (WD < 3.2 m (*Section 5.1 Literature review*)) after rewetting has rarely been investigated in terms of GHG dynamics but has been linked to considerable amounts of GHG emissions (Franz et al., 2016). This thesis aims to investigate carbon dioxide (CO_2) and methane (CH_4) emissions from ponds emerging from rewetting projects.

2 Research question and sub-questions

To achieve the objective concerning the investigation of CO_2 and CH_4 emissions from ponds emerging from rewetting projects, this thesis seeks to answer an overall research question and corresponding sub-questions:

What is the magnitude of CO_2 and CH_4 emissions for ponds in rewetted landscapes?

1. How much do CO_2 and CH_4 contribute to the total flux respectively?
2. How do CO_2 and CH_4 fluxes relate to spatial factors, such as water depth and waterbody and ecosystem type?
3. How do CO_2 and CH_4 fluxes relate to environmental factors, such as wind speed, precipitation, air pressure, and temperature?
4. How do CO_2 and CH_4 fluxes evolve on a daily pattern?
5. How much does ebullition contribute to the total CH_4 flux?

A literature review will investigate CO_2 and CH_4 fluxes in ponds in rewetted landscapes as well as wetlands with similar characteristics to the possible outcomes of restoring peatlands. Additionally, a field study was conducted at Mårumhus Pond in Gribskov, Denmark, from September 19th to October 18th, 2023, to quantify these fluxes using automatic chambers (AC) and bubble traps (BT).

3 Theory

3.1 Climate effect of natural peatlands

Peatlands are characterized as wetland ecosystems with a peat layer exceeding 30 cm in thickness and consisting of a minimum of 30% organic matter (measured in dry mass) (Joosten & Clarke, 2002). Historically, peatlands have been categorized into two types: bogs and fens (Joosten & Clarke, 2002). Bogs are acidic, elevated above the surrounding landscape, and only fed from precipitation (ombrotrophic). In contrast, fens are slightly alkaline or neutral, situated in depressions in the landscape, and are fed from both precipitation, surface- and groundwater (minerotrophic) (Joosten & Clarke, 2002; Thormann & Bayley, 1997).
In general, natural peatlands function as a net GHG sink, as CO_2 stored in the peat exceed the emissions of CH_4 and CO_2, resulting in a net atmospheric cooling (*Figure 1*).

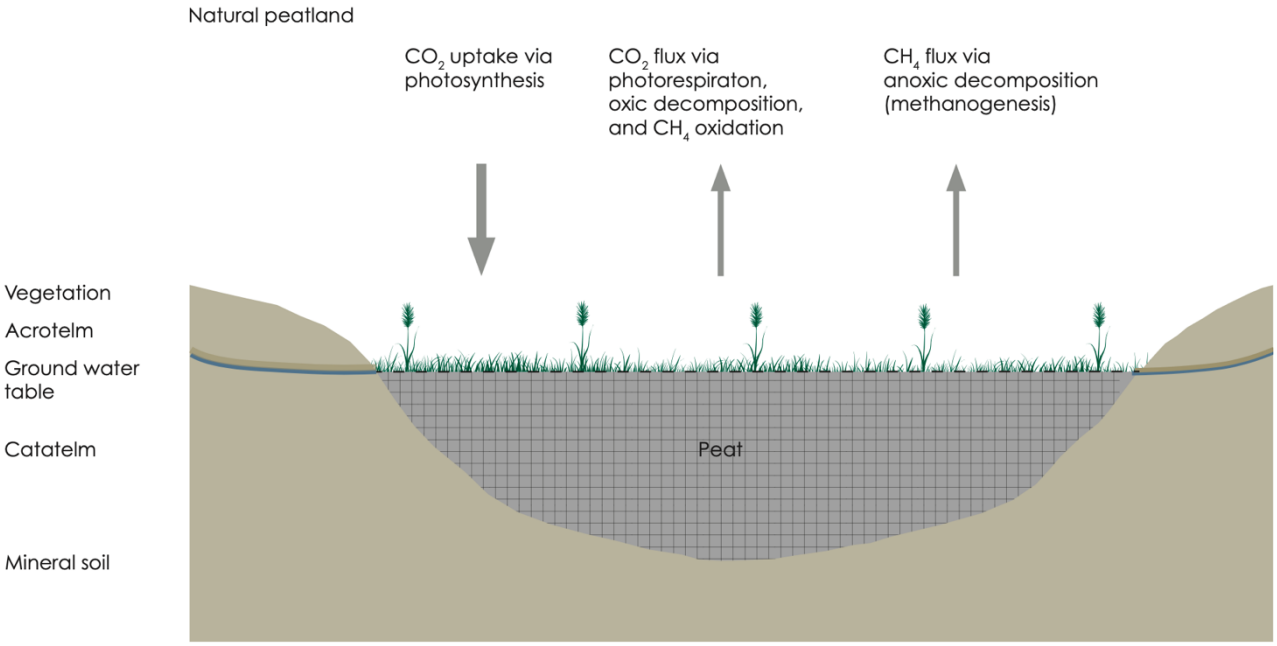


Figure 1: Natural peatlands act as net GHG sinks. The size of the arrows visualizes a larger CO_2 uptake than CO_2 and CH_4 emissions together. The text to the left represents the different layers in a natural peatland. The groundwater table is indicated by the blue line. The figure is made with inspiration from (Christiansen, 2019; Joosten et al., 2016).

3.1.1 CO_2 balance in natural peatlands

The CO_2 balance in peatlands is determined by plant photosynthesis, rate of added organic matter, and rate of decay (Chapin et al., 2011; Dean et al., 2018). Plants store CO_2 as organic carbon in the plant material by photosynthesis ($CO_2 + H_2O \rightarrow CH_2O + O_2$). When plants die, they are decomposed via respiration by bacteria and the presence of oxygen (O_2), resulting in the release of CO_2 ($CH_2O + O_2 \rightarrow CO_2 + H_2O$) (Chapin et al., 2011). Peatlands consist of two distinct layers (1) the acrotelm: an oxic (presence of O_2) upper layer with a high decay rate and (2) the catotelm: an anoxic (absence of O_2) lower layer with a slower decay rate (*Figure 1*) (Joosten & Clarke, 2002). The waterlogged, anoxic conditions in the catotelm slow down the decomposition of plant materials, leading to the

accumulation of peat (Cooper et al., 2014; Thormann & Bayley, 1997). In the process of organic matter continually accumulating, the groundwater table experiences a simultaneous rise, causing the lower part of the acrotelm to progressively transform into the catotelm (*Figure 1*). The anaerobic decomposition is estimated to be approximately 10 times slower than aerobic decomposition (rate of decay) (Petersen et al., 2023). Since the rate of organic matter added to the ecosystem exceeds the rate of decay, a carbon surplus is accumulated in the peat, making peatlands function as sinks of atmospheric CO_2 (Dean et al., 2018; Joosten et al., 2016; Joosten & Clarke, 2002).

Peatland vegetation is characterized by vascular plants and peat-forming species thriving in nutrient poor conditions such as sphagnum mosses that regulate the vertical growth of the peat (Price et al., 2016; Vasander & Kettunen, 2006). Sphagnum mosses have a slow decomposition rate, adding to the high rate of peat accumulation and as such a greater CO_2 sink function in peatlands (Bengtsson et al., 2016; Purre & Ilomets, 2021).

3.1.2 CH_4 balance in natural peatlands

Besides the CO_2 uptake, peatlands emit CH_4 as a result of the biological breakdown of biomass (Petersen et al., 2023). The production of CH_4 (methanogenesis) involves methane-producing archaea existing in anoxic conditions (*Figure 1*). The anaerobic processes are primarily controlled by the presence of labile organic material and the temperature of the sediments (Holgerson, 2015; Wik, 2016). The temperature is important for the determination of decomposition of organic matter in the sediments and is proven to correlate positively with CH_4 production (DelSontro et al., 2010; Eugster et al., 2011).

CH_4 is produced in two ways, (1) reduction of acetate through fermentation ($CH_3COOH \rightarrow CH_4 + CO_2$, called acetoclastic methanogenesis), or (2) oxidation of CO_2 into CH_4 with the presence of hydrogen ($CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O$, called hydrogenotrophic methanogenesis) (Vasander & Kettunen, 2006; Wik, 2016). Once CH_4 is produced within the catotelm of the peatland, it gradually diffuses upwards through the peat. Eventually, it reaches the acrotelm of the peatland, where microbial oxidation by CH_4-oxidizing bacteria (methanotrophs) can occur before it is released into the atmosphere, resulting in the production of CO_2 ($CH_4 + 2O_2 \rightarrow CO_2 + 2\ H_2O$) (Wik, 2016). As such, the net CH_4 concentration at the surface of the peatland is determined by the balance between methanogenesis and methanotrophs. The release of CH_4 from the peatland is determined by the gradient in CH_4 concentration between the peat and the atmosphere, this flux is defined as diffusion.

Besides diffusion, CH_4 emissions to the atmosphere can occur by the formation of gas bubbles (ebullition) (Vasander & Kettunen, 2006). Ebullition fluxes are released from the sediments directly to the atmosphere, making the bubbles less sensitive to oxidation by methanotrophs (Bastviken et al., 2004). It is triggered by changes in the pressure between the gas in the sediments and the total pressure from the water column and atmosphere, which may change due to climatic fluctuations and/or changes in the WTD (Bastviken et al., 2004; Wik et al., 2013). Ebullition might also increase due to increases in wind speed or temperature, generating turbulence in the sediment (DelSontro et al.,

2016). However, such increase is constrained temporally, as the wind induces water column mixing, adding O_2 into the water and thereby leading to oxidation of CH_4. Ebullition events may differ between strong, episodic gas bubbles and steady ebullitions of microbubbles (Goodrich et al., 2011; Hoffmann et al., 2017; Prairie & Del Giorgio, 2013). Microbubbles are smaller methane gas bubbles situated within the acrotelm but trapped in anoxic microenvironments surrounding them (Prairie & Del Giorgio, 2013). Some of these microenvironments hold the capacity to sustain anoxic in the oxic acrotelm for a sufficient period, allowing them to achieve locally high CH_4 partial pressures. Consequently, this enables the microbubbles to reach the water surface and ultimately be released into the atmosphere (Prairie & Del Giorgio, 2013). Ebullition events, either as microbubbles or strong, episodic gas bubbles can almost entirely consist of CH_4 but more often they are a mix of N_2, N_2O, CO_2, and H_2 (Wik et al., 2013).

Ebullition events are sporadic which makes it complex to measure and quantify the contribution to the total methane budget (Baron et al., 2022; Wik, 2016; Wik et al., 2013, 2014). Furthermore, microbubbles are challenging to differentiate from diffusive flux in real-world field conditions, since they are frequently emitted during steady ebullition events resulting in similar measured patterns of gas increase as diffusive fluxes (Prairie & Del Giorgio, 2013).

Additionally, CH_4 can be emitted through plant-mediated transport by vascular plants within gas-conductive plant tissue called aerenchyma (Dean et al., 2018). These plants are able to transport CH_4 directly from the anoxic layers to the atmosphere, without exposing the gas to methanotrophs in the acrotelm (Dean et al., 2018).

3.2 Climate effect of drained peatlands

Peatland drainage is executed through the implementation of ditches and drainage pipes, thereby reducing the WTD and exposing the peat to O_2. This leads to subsidence of the land surface which occurs rapidly in the first years and slower over time.
In general, drained peatlands act as a net GHG source, as CO_2 emissions are greater than the potential CH_4 uptake resulting in net atmospheric warming (*Figure 2)*.

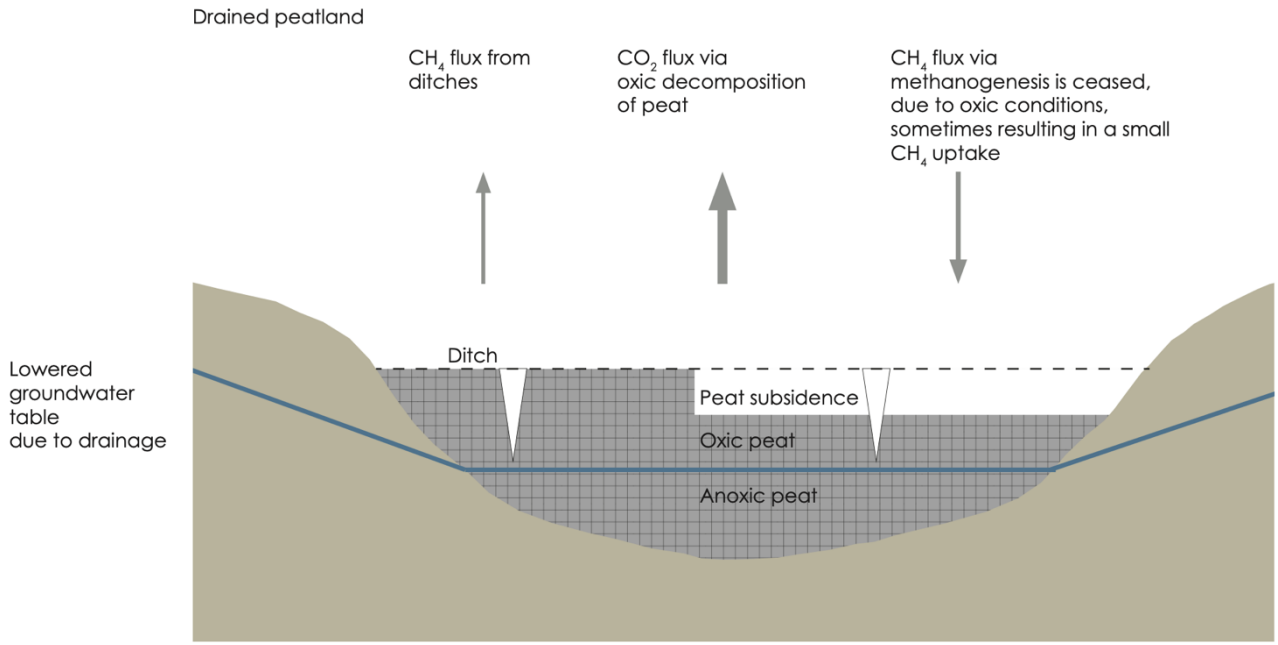


Figure 2: Drained peatlands act as net GHG sources, shown by the size of the arrows visualizing a larger CO_2 flux than CH_4 uptake. The lowered groundwater table is illustrated as a blue line. The figure is made with inspiration from (Christiansen, 2019; Joosten et al., 2016).

3.2.1 CO_2 balance in drained peatlands

The exposure of peat above the water table activates microbial communities in the sediment, oxidizing organic carbon to CO_2 when decomposing the peat and gradually continuing the land subsidence (*Figure 2*) (Pronger et al., 2014). The emissions of CO_2 from the peat continue as long as the area is drained or until all the peat is completely oxidized (Joosten et al., 2016). Land subsidence increases with further drainage or with the introduction of trees absorbing water and thereby leading to lowering the water table even more (Joosten et al., 2016). As such, drained peatlands act as net CO_2 sources.

3.2.2 CH_4 balance in drained peatlands

It is assumed that drained peat soils do not emit CH_4 but might instead act as atmospheric net CH_4 sinks (Peacock et al., 2021b). As described, methanogenesis is a strictly anaerobic decomposition process, why draining the peatlands result in oxidization of CH_4 by methanotrophs in the oxic soil, and as a result ending the CH_4 emissions (Joosten et al., 2016; Petersen et al., 2023). A net uptake of CH_4 in drained organic agricultural soils has even been documented (Petersen et al., 2023). As such, drained peatlands do not emit CH_4 but might act as net CH_4 sinks.

Even though the CH_4 emissions are reduced by draining peat soils, the drainage is often provided by establishing ditches, which carry water and dissolved organic carbon (DOC) out of the peatland resulting in increased emissions of CH_4 (and CO_2) to the atmosphere from the wet ditches (*Figure 2*) (Hiraishi et al., 2014; Joosten et al., 2016).

3.3 Climate effect of rewetted peatlands

Rewetting peatlands constitutes raising the groundwater table to the pre-drainage stage, which can be done by blocking or filling the established ditches, disabling pumping facilities, or building dikes to retain the water (Craft, 2016; Hiraishi et al., 2014). The aim is to reestablish the previous peatland ecosystem including the hydrological processes, biogeochemical processes, and vegetation, but rewetted peatlands rarely resemble the original ecosystem (Hiraishi et al., 2014; Kreyling et al., 2021). An extreme outcome from the rewetting of formerly drained peatlands is the creation of shallow ponds with sparse vegetation (*Section 1 Introduction* (Franz et al., 2016)). As these ecosystems are the scope of this thesis, the following will present processes related to CO_2 and CH_4 fluxes in shallow ponds (*Section 2 Research question and sub-questions*). 2

With the creation of a pond the groundwater table is established above ground due to peat subsidence. The common reference to the depth of the water column when addressing open water systems is water depth (WD) as opposed to water table depth (WTD) in peatlands. For simplicity the thesis is addressing both WD/WTD as negative when being above ground level and positive when below.

A complete account of the climate effect of ponds in rewetted peatlands does not exist, as it is rarely studied (Franz et al., 2016). However, the processes described in the following indicate that these ecosystems both generate an uptake and emissions of CO_2 and CH_4 (*Figure 3*).

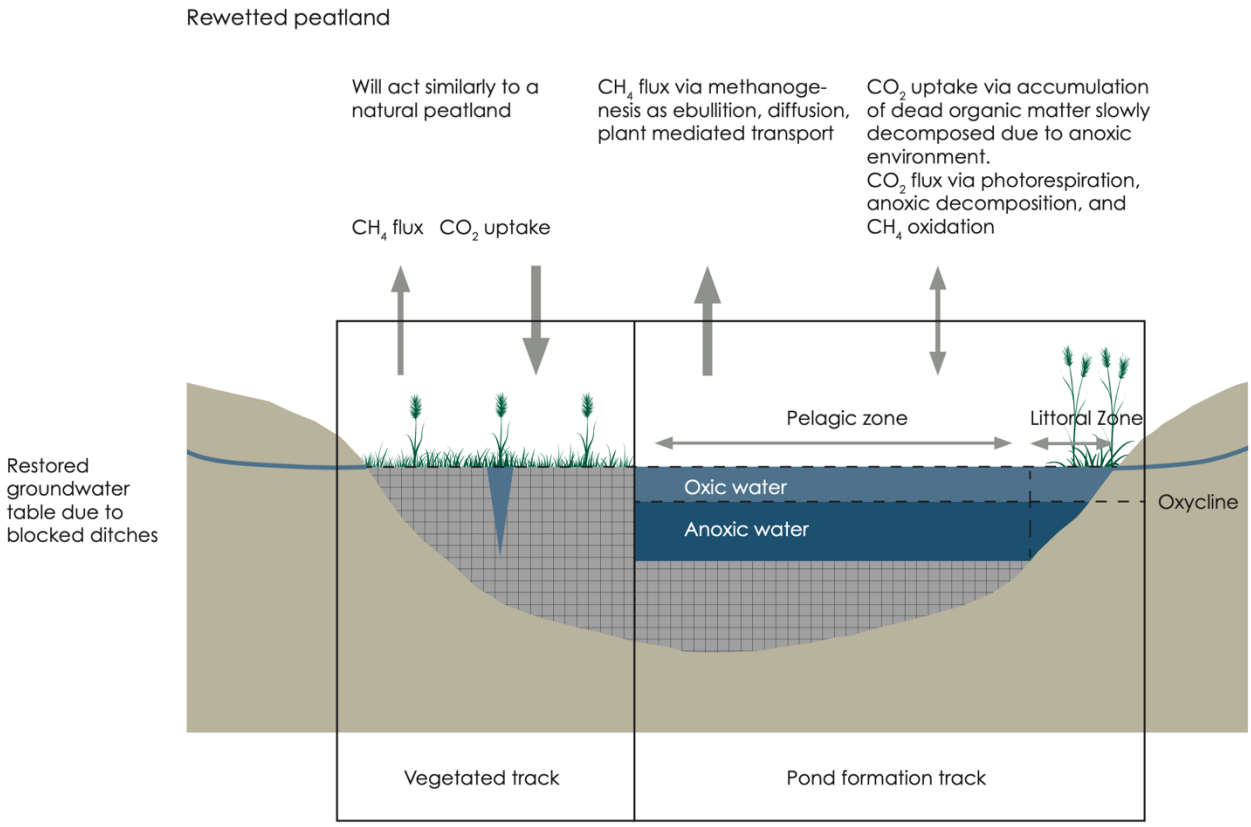


Figure 3: Shows two possible outcomes from peatland rewetting: a vegetated track and a pond formation track. It is not clear whether ponds on rewetted peatlands act as net GHG sinks or sources. The arrows visualize CO_2 and CH_4 emissions and uptake. The restored groundwater table is illustrated as a blue line. The figure is made with inspiration from (Christiansen, 2019; Joosten et al., 2016).

3.3.1 CO_2 balance in rewetted peatlands (pond formation)

The CO_2 balance in rewetted ponds situated on peat soils depends on the properties of the rewetted areas. As explained (*Section 3.1.1 CO_2 balance in natural peatlands*), the accumulation of peat, and consequently the sequestration of CO_2, rely on properties ensuring that the addition of organic matter exceeds decomposition (Dean et al., 2018; Joosten & Clarke, 2002). This balance is determined by the return of water as well as the vegetation cover. When the area is flooded after rewetting and forms an open water surface, the existing plants die off, stopping the uptake of CO_2 by photosynthesis temporarily (Purre & Ilomets, 2021). Since the area is transformed into a pond ecosystem, only sparse vegetation is expected to return. This limited vegetation regrowth leads to reduced organic matter input compared to a natural peatland, where a substantial amount of vegetation continuously contributes to peat accumulation. This will limit the CO_2 uptake in the pond ecosystem. Additionally, the anaerobic decomposition of the recent dead organic matter as well as the existing peat layer emits CO_2 by CH_4 oxidation to the atmosphere (Joosten & Clarke, 2002; Kelly et al., 1997; St. Louis et al., 2000). Generally, the decomposition rate exceeds the input of organic matter in most lakes due to the sparce vegetation (Tranvik et al., 2009). (*Figure 3*)

Holgerson & Raymond (2016) find that small, shallow ponds generally emit more CO_2 than larger lakes, which may be due to a proportionally large littoral zone (*Figure 3*) resulting in (1) increased temperature in the sediments which is driving respiration, (2) larger loads of terrestrial litter relative to water volume subsequently increasing respiration, (3) larger influence by environmental fluctuations, e.g. heavy rain and wind events driving episodic CO_2 emissions (Holgerson & Raymond, 2016; Spafford & Risk, 2018).

Petersen et al. (2023) state that emissions from the littoral zone are four times higher than that of the pelagic zone (the deeper part of the lake). Contrary small ponds with a large littoral zone are more influenced by carbon uptake through litter (Michmerhuizen et al., 1996; Spafford & Risk, 2018; Wik et al., 2013). As such, the properties of the pond and the surrounding vegetation impact whether it will act as a CO_2 sink or source.

3.3.2 CH_4 balance in rewetted peatlands (pond formation)

When flooding drained peatlands, the CH_4 flux increases due to the anaerobic decomposition of organic matter and a decrease in the consumption of CH_4 by methanotrophs (*Figure 3*) (Conrad, 1996; St. Louis et al., 2000). Another result of flooding peatlands is the change in conditions for plant growth compared to the natural peatland. The altered water table dynamics and geochemistry including nutrient enrichment leads to a shift in vegetation towards tall helophytes, suppressing peatland species such as low vascular plants and mosses by light competition. This can lead to an increase in plant-mediated CH_4 fluxes since helophytes are highly conductive for gases (Kreyling et al., 2021).

Shallow ponds also favor CH_4 emissions in many ways: (1) The shallow WD reduces oxidation before atmospheric CH_4 release and favors ebullition (*Figure 3*) (Gorsky et al., 2019; Holgerson, 2015;

Peacock et al., 2021b), (2) Shallow ponds are generally warmer due to a greater impact by solar heating, enhancing methanogenesis by temperature (Bastviken et al., 2004; Hoffmann et al., 2017; Holgerson, 2015), (3) Wind also has a greater impact on CH_4 emissions in shallow ponds, as it generates turbulence in the sediment, (DelSontro et al., 2016) though only short-term as continuous wind results in a continuous mixing of the water column leading to increased levels of O_2 and as a result decrease CH_4 diffusion. Furthermore, precipitation may decrease CH_4 fluxes as carbon is diluted (Holgerson, 2015).

Methanogenesis is also stimulated by nutrient availability, making rewetting on nutrient-enhanced peat soils result in temporarily larger CH_4 emissions (Joosten et al., 2016).

The biological processes as well as the pathways of CH_4 emissions are similar to those of natural peatlands (*Section 3.1.2 CH_4 balance in natural peatlands*) but vary depending on the stratification of the pond (*Figure 4*). For ponds without clear stratification, CH_4 is only produced in the anoxic sediments underlying the oxic water. In contrast, CH_4 can be produced in the anoxic water in stratified ponds (Eugster et al., 2011). An additional CH4 emission pathway for stratified ponds is through storage fluxes, meaning that CH_4 stored in the anoxic water can be emitted rapidly as diffusive fluxes in the spring- and fall turnover (*Figure 4*) (Bastviken et al., 2004; Michmerhuizen et al., 1996; Riera et al., 1999). In colder climates a great release of dissolved CH_4 is emitted into the atmosphere due to the ice cover melting during spring (Michmerhuizen et al., 1996) whereas in the fall the dissolved CH_4 loss is caused by a higher wind speed and the breakdown of the stratification which have been built up and accumulating CH_4 in the anoxic water during the summer (Riera et al., 1999; Wik, 2016). These periods of CH_4 loss to the atmosphere during spring and fall are immediately followed by periods of CH_4 buildup during summer and under ice cover during winter (Bastviken et al., 2004; Michmerhuizen et al., 1996; Riera et al., 1999).

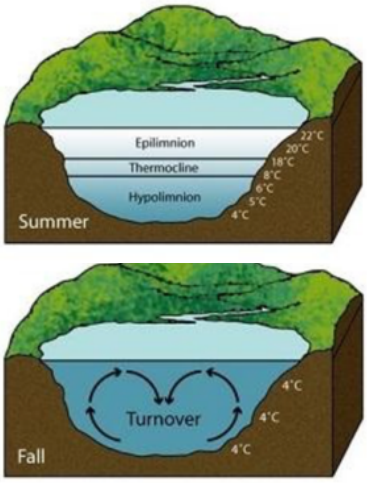


Figure 4: The process of fall turnover, resulting in a breakdown of the stratification which have been build up during the summer (Thalmann, n.d.)

4 Materials and methods

4.1 Literature review

To gain an overview of existing studies on the topic of this thesis, a literature review encompassing three distinct phases was undertaken: Phase 1a and 1b, a two-step preliminary literature search; Phase 2, a systematic review; and Phase 3, data extraction and synthesis.

To locate all papers reporting on CH_4 emissions from restored peatlands, located in the temperate or boreal climate zone, a preliminary literature search was conducted by searching through 'all available databases' at Web of Science™ (Web Of Science, n.d.) (accessed in September 2023). To include a proper number of papers that was doable within the timeframe of the thesis, different combinations of keywords were examined, giving different number of hits (*Table 1*). The search was conducted in a two-step preliminary search: In the initial search Phase 1a, the search started broad and was refined through stepwise iterations. This approach made it clear, that no former paper had examined the exact same problem as in this thesis; CH_4 emission (primarily as ebullitive fluxes) from rewetted peatlands located in the temperate climate zone (last search in search Phase 1a, *Table 1*). Based on the number of papers occurring in search Phase 1a, every paper in the search including the keywords (ebullition or ebullitive or "methane bubbl*" or "CH_4 bubbl*") and (artificial* or rewet* or restor* or dich* or flood*) (n=170) were systemically reviewed.

After reviewing the title, keywords, and abstract of all the papers, it became evident that the initial search had unintentionally led to the inclusion of studies concerning rice paddies and extensive artificial lakes like hydropower reservoirs. In order to make up for this the search focus was redirected toward artificial as well as natural shallow lakes and ponds, which have similar characteristics to ponds in rewetted peatlands. Additionally, all CH_4 pathways were included by replacing the word 'ebullition' with 'methane'. Consequently, relevant keywords associated with shallow lake/pond formation were incorporated in the second search Phase 1b (n=48) (*search Phase 1b, Table 1*).

In total, the preliminary literature search, including both steps in Phase 1a and 1b, led to a total of 218 papers, which were then assessed through a systematic review.

Table 1. The preliminary literature search was carried out in two stages, Phase 1a and 1b, including different combinations of keywords resulting in different number of hist. Keyword combinations in bold were used in Phase 2, the systemic literature review (Appendix 1).

Search Phase	Keywords	# of searches occurring
1a	(methane or CH4) and (wetland* or water bod* or pond* or lake* or peat* or aquatic* or fen* or bog* or ditch*)	27,889
	(ebullition or ebullitive or bubbl*) and (wetland* or water bod* or pond* or lake* or peat* or aquatic* or fen* or bog* or ditch*)	26,616
	(GHG or "greenhouse gas*") and (artificial* or rewet* or restor* or dich* or flood*)	8,635
	(ebullition or ebullitive or bubbl*) and (wetland* or water bod* or pond* or lake* or peat* or aquatic* or fen* or bog* or ditch*) and (rewet* or restor*	254
	(ebullition or ebullitive or "methane bubbl*" or "CH4 bubbl*") and (artificial* or rewet* or restor* or dich* or flood*)	**170**
	(ebullition or ebullitive or bubbl*) and (peat* or wetland*) and (rewet* or restor*)	28
	(ebullition or ebullitive or bubbl*) and (peat* or wetland*) and (temperate* or boreal*) and (rewet* or restor*)	5
	(ebullition or ebullitive or bubbl*) and (peat* or wetland*) and forest* and (rewet* or restor*)	2
	(ebullition or ebullitive or bubbl*) and (peat* or wetland*) and temperate* and forest and (rewet* or restor*)	0
1b	**(methane) and (rewet* or restor*) and temperate and forest**	**39**
	(methane) and ("shallow lake" or pond) and temperate and forest	**9**

It should be noted that CO_2 emissions were not included in the initial literature review, as the scope of the thesis changed during the data processing (further discussed in Section 6.6.1 Literature review).

In the attempt to gain a complete coverage, references from the evaluated papers, along with papers discovered during an in-depth exploration of the subject matter, were also incorporated in the systematic review.

In the 2nd Phase of the literature review, the 218 papers and the additional references were reviewed based on title, keywords and abstract. Studies that were conducted in a different climatic zone than the temperate or boreal, or containing waterbodies deeper than 3.5 m, were excluded.

The systematic review resulted in a total of 44 relevant studies that reported findings on CO_2 and CH_4 emissions (either as diffusive and/or ebullitive fluxes). All the studies were carried out in the temperate (93%) or boreal (7%) zone and included measures from more than 180 waterbodies in total. Among the 44 relevant studies, one meta-study was also included in the synthesis.

For the last stage of the literature review, Phase 3, all CO_2 and CH_4 emissions were standardized to a common unit, specifically, *mg CO_2 or CH_4 m^{-2} d^{-1}*. In cases where studies reported data spanning multiple time periods, sites, depths, surface area, and temperatures, averages were used, without account for possible variation between the parameters. Furthermore, for studies that conducted measurements across various types of wetlands, such as ditches and lakes, emission factors were recorded separately to facilitate individual data analysis if possible.

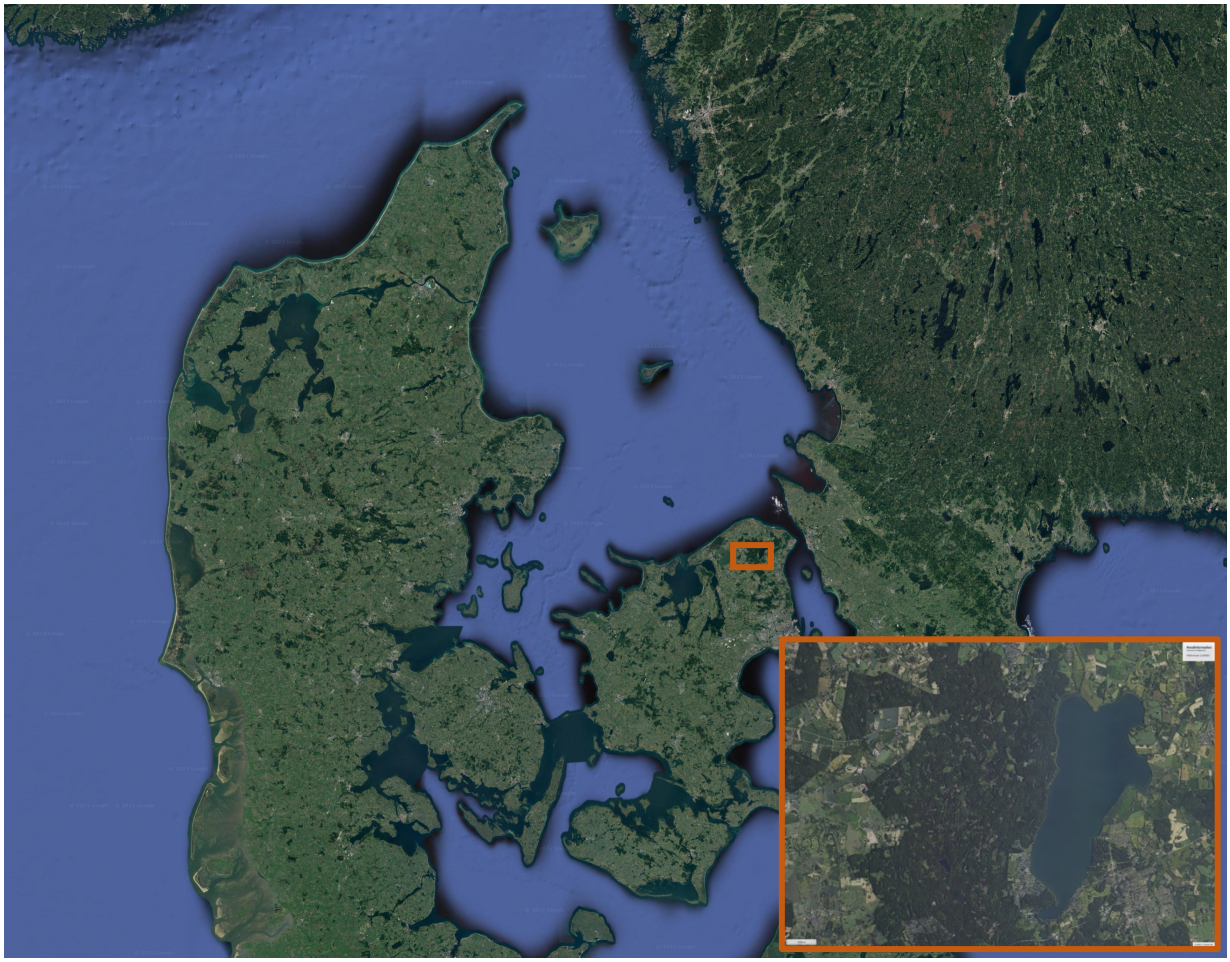
The studies included different methods for measuring CH_4 fluxes. The predominant approach involved the use of floating chambers, either manually or automatically operated, while a few used Eddy covariance flux towers. Additionally, some studies used bubble traps either independently or in combination with other measurement techniques. It is worth noting that there might be a difference in these diverse methodologies ability to assess CH_4 fluxes.

4.2 Field study

4.2.1 Study site description

4.2.1.1 Gribskov

The field study was conducted within a rewetted forest with peaty areas in depressions situated in Gribskov, in the northern region of Zealand, Denmark. (*Map 1*). Gribskov is a state-owned forest and with its 5,795 ha it is the largest old forest in Denmark. (Naturstyrelsen, 2021, n.d.a).

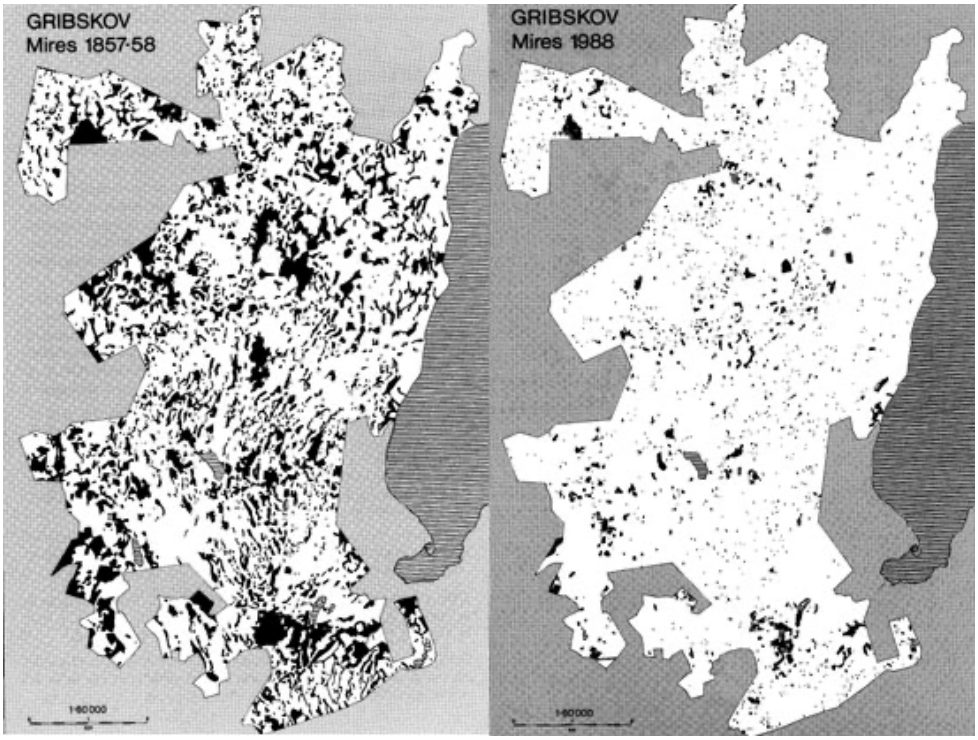


Map 1: Map of Denmark, with the location of Gribskov highlighted within the orange box. Reproduced from Google Earth and Danmarks Arealinformation (Danmarks Miljøportal, n.d.; Google, n.d.).

Gribskov originate from the last ice age, where the area was characterized by wetlands gradually overgrown by peat mosses and eventually developing into a dense forest including tree species such as downy birch (*Betula pubescens*), scots pine (*Pinus sylvestris*) and hazel (*Corylus avellana*). Approximately 3,000-3,500 years ago, beech trees followed and eventually became the dominant species in the forest. In the 1800s, the process of draining began with the purpose of making space for more effective forestry started, resulting in a total of 564 km of drainage ditches emerging through the century. The process of drainage continued, and by the 1980s wetlands and lakes covered only

3.25% of the area, i.e. a significant decline from the 20.8% coverage observed in the mid-1800s (*Map 2*). (Naturstyrelsen, n.d.c)

Currently, Gribskov largely consists of deciduous broadleaved trees, primarily beech (*Fagus sylvatica*) as well as oak (*Quercus sp.*), birch (*Betula sp.*), ash (*Fraxinus excelsior*), and various other species. Broadleaved trees cover 54% of the forest, while non-native conifers, mainly spruce (Picea abies), cover 36% of the area (Overballe-Petersen et al., 2014). Gribskov is generally characterized by many generations of monoculture forestry although ongoing transformations are gradually converting most of the forest into pristine woodland, a process set to continue until 2026 (Naturstyrelsen, 2021, n.d.c). In this context, there has been a focus since 1992 on returning the lakes and wetlands within Gribskov to their natural, hydrological condition (Naturstyrelsen, n.d.c). Since then, approximately 235 hectares of wetland areas, equivalent to 7% of the entire forest area, have been restored, which is expected to increase to 12 % (Naturstyrelsen, n.d.b).



Map 2: Map of wetlands in Gribskov in 1858 and 1988 (Naturstyrelsen, n.d.d).

4.2.1.2 Mårumhus Pond

One of the aforementioned restored wetlands is Mårumhus Pond, situated in the northwestern region of Gribskov, near the town of Mårum (*Map 3*). The pond serves as the location for the field study conducted in this thesis.

Map 3: Map of Gribskov, with the location of Mårumhus Pond marked in the orange box. Reproduced from Danmarks Arealinformation (Danmarks Miljøportal, n.d.).

The area was rewetted by filling in the pre-existing ditches in 2008 (*Appendix 2* (Naturstyrelsen, n.d.d)). It now appears as a shallow pond, with a mean WD of 0.8 m, based on measured depths (*Section 4.2.2 Field design*), and cover a surface area of 7,200 m^2, based on aerial photos in Danmarks Arealinformation (Danmarks Miljøportal, n.d.). The pond is surrounded by smaller rewetted wetlands and is located within a mature beech stand. The pond is characterized by a large open water surface surrounded by typical wetland vegetation, including cattail (*Typha latifolia*) (*Picture 1*).

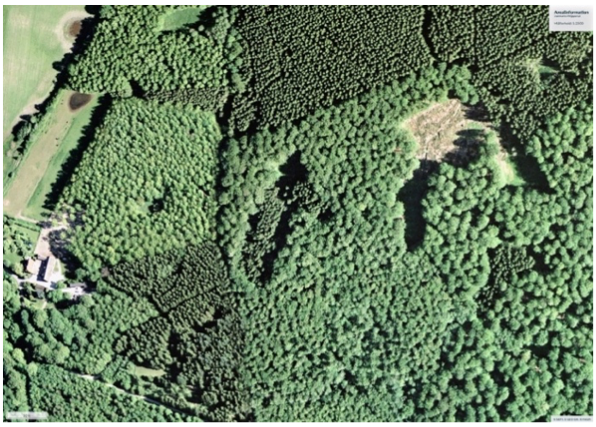
Picture 1: Pictures from the study site. Left: show the entire pond as seen from the South. Right: show the northern part of the pond, where the field study was conducted as seen from the East.

The soil around the pond is characterized by meltwater sand, peat, and in some sections, nutrient-rich moraine clay (Naturstyrelsen, n.d.b).

Historical topographic maps from 1901-1971 (Danmarks Miljøportal, n.d.) show substantial drainage activities and afforestation in the area (*Map 4a*), with drainage ditches crossing all around the area of the current pond. Orthophotos from 1954-2008 show varying vegetation cover of the area over time *(Map 4b, c, and d)*. The drainage was ended in 2008 and the following orthophoto from 2010 and the most recent from 2022 clearly demonstrate a restored open water table *(Map 4e and f)*.

Map 4a: Topographic map from approx. 1901-1971. Drainage canals marked with blue lines, cloud-like symbols marks broadleaved tree stands and stars mark conifer stands. Reproduced from Danmarks Arealinformation (Danmarks Miljøportal, n.d.).

Map 4b: Orthophoto from 1954. The area of the pond is planted with what could resemble a young stand. Reproduced from Danmarks Arealinformation (Danmarks Miljøportal, n.d.).



Map 4c: Orthophoto from 2004. The area of the pond is overgrown by conifer trees species. Reproduced from Danmarks Arealinformation (Danmarks Miljøportal, n.d.).

Map 4d: Orthophoto from 2008. The area of the pond is bare compared to the surrounding stand that comprise of tall beech trees. Reproduced from Danmarks Arealinformation (Danmarks Miljøportal, n.d.).

Map 4e: Orthophoto from 2010. The drainage ditches were filled in 2008 after which the water table increased, and Mårumhus Pond emerged. Reproduced from Danmarks Arealinformation (Danmarks Miljøportal, n.d.).

Map 4f: Orthophoto from 2022. The area as it appears today. Still with an open water surface. Reproduced from Danmarks Arealinformation (Danmarks Miljøportal, n.d.)

4.2.2 Field design

The field study at Mårumhus Pond was conducted from September 19th to October 18th, 2023, using two methods, Automatic Chambers (AC) and Bubble Traps (BT) to measure and quantify CO_2 and CH_4 fluxes. Each method consisted of 8 AC or BT placed on a transect across the pond. The study aimed at estimating the short-term changes in fluxes of these gases during different weather conditions, WDs, and daily patterns. The combination of methods allowed for comparisons between flux measurements.

4.2.2.1 Environmental parameters

To monitor temperature gradients in the water, five light and temperature loggers (HOBO loggers, Onset Corp., USA) were placed between AC no. 5 and 6. The loggers were put up on the 22nd of September until the end of the study period and were placed on a pole 20 cm apart from each other, at 0 cm, 20 cm, 40 cm, 60 cm, and 80 cm above the sediment, i.e. the 80 cm logger was placed 10 cm above the water surface. It is worth noting that the logger above the water surface was placed with no shading, as such temperatures were also affected by heating from sunlight. The loggers were installed to measure every 30 min. The logger at 40 cm was mistakenly set to measure every second and therefore only contained data from the first measuring day, it was therefore excluded from the following data processing (*Appendix 3*). Additionally, it is worth noting that the pole with the loggers was hammered into the sediment, disturbing the pond bed remarkably.

The temperature was used to identify whether the pond was thermally stratified. Following the criteria established by Esposito et al. (2023), thermal stratification is defined as at least 1°C difference between the surface and bottom waters in ponds with a WD below 1 meter. As such, the temperature measurements between the loggers placed 20 cm and 60 cm above the sediment were compared to determine if the pond was thermally stratified. It is noteworthy that thermal stratification serves as a

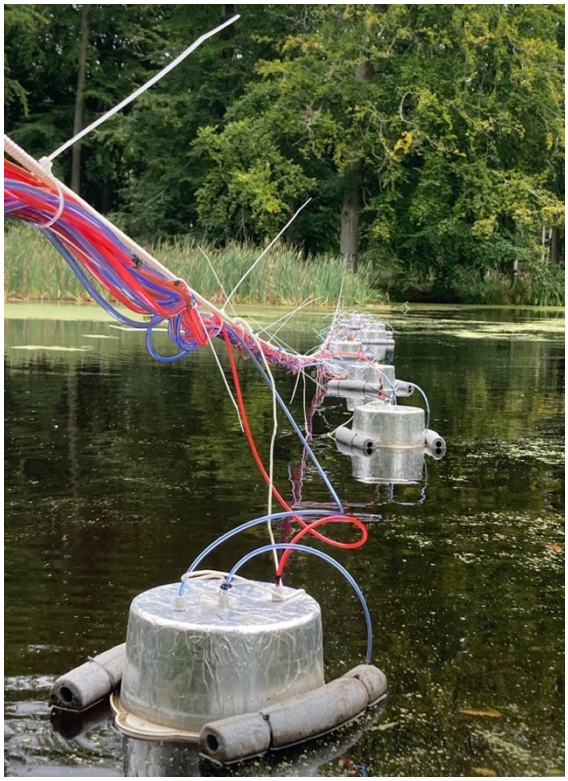
proxy for stratification in O_2 concentration, which is a more accurate way of assessing stratification. Esposito et al. (2023) conclude that in all cases of thermal stratification, the bottom water was anoxic. However, many ponds examined in the study exhibited prolonged periods of thermal stratification as opposed to stratification in O_2 concentration. This raises the possibility of an overestimation of stratification when relying solely on temperature differences (Esposito et al., 2023).

Meteorological data including mean and maximum windspeed, precipitation and air pressure are derived from the nearest land-based weather station no. 6188 called 'Sjælsmark' located approximately 25 km south of Mårumhus (*Appendix 3*) (DMI, n.d.). The distance between the study site and weather station implies that the exact weather conditions at the study site may differ, but the weather station data can be used to indicate the overall climatic conditions of the region at the given time. It is likely that weather events are similar e.g., high wind speeds in Sjælsmark indicate high wind speeds in Gribskov.

Initial WD measurements were recorded for each AC and BT upon installation (*Appendix 4 and 5*).

The original intent was to extract a soil sample using a soil auger; however, branches, algae, and compact soil, prevented a successful penetration of the soil auger into the sediments of the pond.

4.2.2.2 Automatic Chambers



Picture 2: Picture from the field study. Eight AC attached to a rope tied between two trees across the pond.

Eight AC were installed on Mårumhus Pond at a transverse gradient extending from the shoreline to the pond's midpoint, making it possible to measure the flux from the water surface (*Picture 2*). Thus, plant mediated fluxes and carbon uptake from the pond is not included in this study.

The cylindrical AC had a diameter of 34.5 cm, height of 13.8 cm and a total volume of 12.9 L. To ensure the AC stayed afloat two pieces of pipe insulation were attached on the sides of the AC with cable ties. To mitigate solar heating, the ACs were covered in reflective silver foil, as described by Bastviken et al. (2015).

The ACs were tied to a rope bound between two trees at each side of the pond, to ensure they stayed in roughly the same place. The rope was minimum 40 cm above the water surface at all positions, to enable the AC to float freely and avoid collision with the rope. All tubes connected to the AC were also tied to the rope, to avoid getting into the water.

A closed-loop system connected the AC and three instruments located at the shore: a LI-8150 multiplexer (LICOR8150), a LI-8100 CO_2 analyzer (LICOR8100) (Li-COR, Lincoln, Nebraska, USA) and a Los Gatos Research Ultra-Portable Greenhouse Gas Analyzer (UGGA) (ABB Los Gatos, Canada) (*Figure 5*). All inlet and outlet tubes were attached to the LICOR8150. The LICOR8150 was linked to the LICOR8100, which not only controlled the LICOR8150's valves but also measured the concentrations of CO_2 and H_2O. The UGGA was connected in a parallel loop on the outlet side of the LICOR8100, measuring concentrations of CO_2, H_2O and CH_4.

The use of the LICOR8150 enabled the opening and closing of valves between the eight AC and the LICOR8100 and the UGGA. With a total of 16 double valves available on the LICOR8150, the eight AC could be measured consecutively in a setup using two double valves for each AC. Each AC was connected to a set of two tubes: one for gas inlet, one for outlet, and one for ventilation with atmospheric air. The inlet and outlet tubes were each connected to a valve consecutively representing channel 1 and channel 2 (*Figure 5)*. During the 10 min measurements, only channel 1 was open (*Figure 5)*, leading the gas in a loop between the LICOR8100 and UGGA. In contrast, during purging, only channel 2 was open. This allowed the intake of atmospheric air, which was pumped into the AC to reset the air to atmospheric levels. Simultaneously, gas within the AC should be released through the vent tube (*Figure 5*).

The vent tube consisted of an approximately 40 cm long section of tubing, with the purpose of releasing gas during purging and to ensure that the pressure inside the AC remains in equilibrium with the atmospheric air pressure at all times (Davidson et al., 2002). It became evident during the measurements, that the vent tubing alone was not sufficient as the AC instead lifted slightly due to high pressure from the purging, causing gas to escape from below rather than through the vent tube.

The LICOR8150, LICOR8100 and UGGA were connected to a power outlet, making it possible to measure throughout the entire study period.

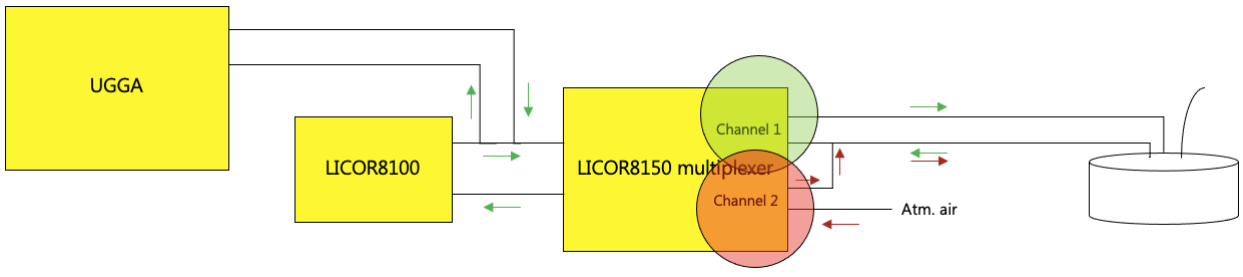


Figure 5: Technical illustration of the setup connecting the eight AC to the LICOR8100, LICOR8150 and UGGA. Green circle and arrows: Channel 1 is open when measuring, closed when purging. Red circle and arrows: Channel 2 is closed when measuring, open when purging.

Each AC underwent a 20-minute purging period, followed by a 10-minute measurement period, with one AC being processed at a time. This made it possible to measure all eight AC six times a day, enabling estimation of a daily pattern. When assessing the data, it was clear that it was almost impossible to accomplish true atmospheric concentration in the AC after purging, as they remain in

constant contact with the water surface, resulting in ongoing diffusive fluxes into the AC as well as the potential for ebullition events during the purging phase. Extending the purging phase would therefore not necessarily have improved the results. To adjust for these possible overestimated initial levels of CO_2 and CH_4, the starting point of the measuring period was instead corrected to the level of CO_2 and CH_4 flux just before the second ebullition during data processing (*Section 4.2.3.1 Automatic chambers)*. *4.2.3.1*Automatic chambers

It is worth noting that high wind speeds occasionally disrupted the AC measurements, causing three AC to turn over: AC no. 5 and 7 (19.09.2023 – 20.09.2023) and AC no. 3 (03.10.2023 – 09.10.2023). Furthermore, one tube was misconnected to AC no. 1 on the first three days of measuring (19.09.2023 – 22.09.2023). As a result, data collected during such events were excluded from the data processing phase. Additionally, CH_4 and CO_2 data from 55 measurements were excluded since they showed no values for CO_2 which is interpreted as incorrect data. In total, 91 measurements were excluded leaving a total of 1.264 measurements for further data processing.

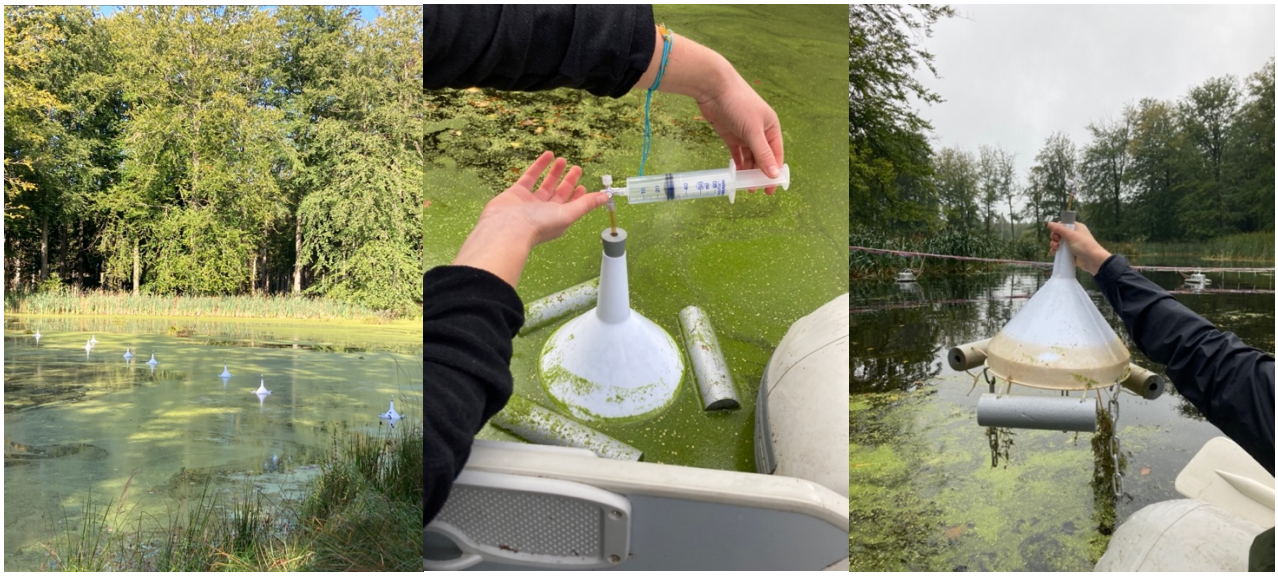
4.2.2.3 Bubble traps

Eight BT were installed in the same manner as the AC, at a transverse gradient extending from the shoreline to the pond's midpoint *(Picture 3).* The BT were placed at a distance greater than 5 meters from the AC-setup to avoid collision. The BT were made from plastic funnels that were inverted and partly submerged, similar to the set-up by Wik (2016) and Petersen et al. (2023). The funnels had a diameter of 30 cm narrowing to 2.5 cm with a total volume of 3.72 L. Three pieces of pipe insulation were attached to the funnels to ensure them staying afloat, while metal chains gave weight to secure the semi-submergence of the BT edges. Each BT was further attached to an anchor (cobblestone) to secure their fixed location while providing room to float with a distance up to 2 m. The stems of the BT were sealed gas tight with a rubber plug and a gas valve enabling gas sampling *(Picture 3).*

The funnels were made of thin white plastic, which made it possible to see through. When installing the BT on the pond the edge was submerged until a marked point on the side of the funnel indicating a total air volume of 1.83 L. When gas enters the BT from below, water is replaced by gas, increasing the gas volume, resulting in an uplift in the BT. The volume of the funnels during measurements was calculated as a relationship between the gas volume and corresponding water level height within the funnel (*Appendix 5*).

The procedure for gas sampling went as follows: First, the water level within the funnel was measured, to estimate the accumulated gas volume. Second, to avoid stratification of the accumulated gasses, the BT were shaken carefully before taking the gas sample. For each sample, 50 mL of gas was extracted using a plastic syringe *(Picture 3).* The gas extracted was transferred to gas-proof bags filled with 500 mL nitrogen (N_2). N_2 was used as it does not react with CH_4 or CO_2, making it possible to attribute all the CH_4 or CO_2 to the gas sampled from the BT. Third, after sample extraction, the funnels were elevated above the water, to release the gasses and 'reset' the system to atmospheric gas

concentrations. Gas samples were undertaken from a small inflatable boat, to minimize disturbance of the sediment and minimize the risk of inducing ebullition.



Picture 3: Pictures from the field study. Left: The eight BT located on a transect. Middle: Gas sampling from a BT. Right: BT during 'reset' after gas sampling.

The BTs were sampled four times (27.09.2023, 02.10.2023, 09.10.2023 and 18.10.2023) during the study period.

4.2.3 Data processing

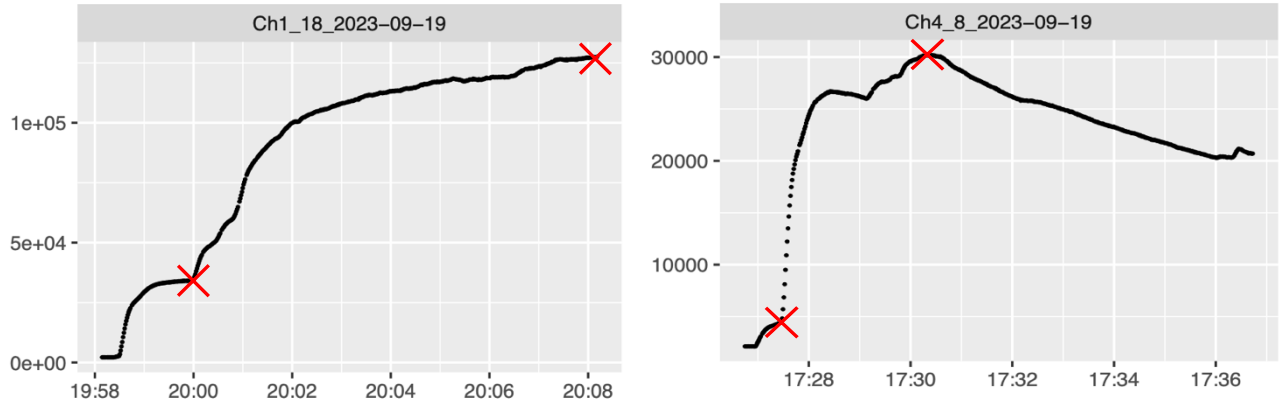
4.2.3.1 Automatic chambers

The first phase of the data processing was executed in R-Studio (R Core Team, 2023), using R-scripts authored by Klaus Steenberg Larsen and Ph.d. Annelie Skov Nielsen.
Firstly, data on CO_2 and CH_4 fluxes measured by the LICOR8100 and UGGA were calibrated using an R script authored by Klaus Steenberg Larsen (*Appendix 6*). The data was calibrated by identifying related fluxes of CO_2 in the LICOR8100 data and UGGA data, to ensure that measures of all fluxes were temporally aligned. This was done since the time frame for the LICOR8100 and UGGA machines differed. The calibration also ensured that the measured fluxes were alike, and that the set-up was working as intended.

After achieving a consistent data format for all flux data, a second pre-established R-script, developed by Ph.D. stipend Annelie Skov Nielsen was used (*Appendix 7*). The script enabled the identification of total CO_2 and CH_4 concentrations from each 10 min measuring period in each AC for the entire study period. This identification was achieved by manually marking the minimum and the maximum gas concentration for each measuring period of CO_2 and CH_4, respectively. Due to a potentially insufficient purging period (*Section 4.2.2.2 Automatic chambers*), min gas concentration was selected

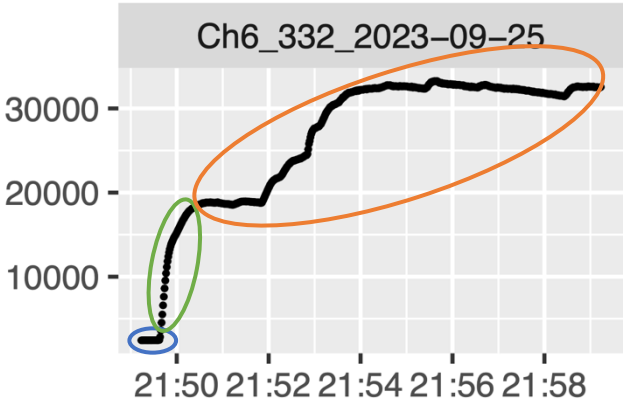
from just before the second ebullition (*Graph 1*). This approach is considered conservative since diffusion and ebullition events occurring just when the measurements began were excluded.

Initially, the scope of this thesis was to quantify ebullitive and diffusive fluxes. In this phase of the data processing, it became clear that it was not possible to separate ebullitive and diffusive concentrations, and as such the total CO_2 and CH_4 fluxes were assessed.



Graph 1: Graphs of CH_4 gas concentration measured through 10 min. from two AC. AC 1 on the 19th of September 2023 from 19.58 till 20.08 and AC 4 from the 19th of September 2023 from 17.26 till 17.36, respectively. The red x's mark where the methane flux is measured from and to. More graphs are available in Appendix 8. X-axis: minutes. Y-axis: CH_4 gas concentration in ppb.

The small tail observed in the flux curve (depicted by the blue circle, *Graph 2*) represents the brief period required for the LICOR8150 to transport gas samples from the AC to the shore where the LICOR8150 is stationed. This slight delay arises from the configuration wherein the outlet tube for purging is shared with the inlet tube for measurement periods. Additionally, the sudden peak occurring just after the intake of gases by the LICOR8150 (indicated by the green circle, *Graph 2*) signifies the gas concentration accumulated in the AC towards the end of the purging phase/just before measuring and may indicate insufficient purging. Consequently, this segment is not considered in the total flux measurement, as described above. The remainder of the curve (orange circle, *Graph 2*) represents the flux concentration measured by the LICOR8150 over the 10-minute measuring period.



Graph 2: Graphs of CH_4 gas concentration measured through 10 min. Blue circle: the brief period required for the LICOR8150 to transport gas samples from the AC to the shore. Green circle: start concentration due to insufficient purging (excluded). Orange circle: the flux concentration. X-axis: minutes. Y-axis: CH_4 concentration in ppb.

When selecting min and max concentrations for all measuring periods some were dismissed as the flux developed in an unusual manner. *Appendix 9* show some dismissed fluxes while *Appendix 8* shows all graphs for fluxes selected in this process.

Calculations of emissions

After selecting min and max concentrations, the output displayed the corresponding values for CO_2 and CH_4 in parts per million and parts per billion, respectively. Min concentrations were subtracted from max concentrations to gain the total concentration for every 10 min measurement. The ideal gas law was used to convert the measured concentrations from ppm and ppb to μmol CO_2 and nmol CH_4 in the following equation (*Appendix 7*):

$$n = dC * V * p * (1 - [H_2O]) / (R * T) \quad (1)$$

n = Amount of CO_2 and CH_4 in μmol and nmol, respectively.
dC = Change in concentration of CO_2 and CH_4 in ppm and ppb, respectively.
V = Volume in L (12.9 L) (*Section 4.2.2.2 Automatic chambers)*.
p = Pressure in ATM. In order to calculate the flux in dry air, the pressure is corrected for the water pressure by multiplying with (1 – [H_2O]).
R = The ideal gas constant (0.082057 L atm K^{-1} mol^{-1}).
T = Temperature in Kelvin. For all samples during the study period, a constant of 15 °C = 288.15 K was assumed.

The concentrations were then converted to fluxes in mg CO_2 m^{-2} d^{-1} and mg CH_4 m^{-2} d^{-1} (*Appendix 4*):

$$F\ (CO_2) = n * 10^{-3} * M /(dt * A) \quad (2)$$
$$F\ (CH_4) = n * 10^{-6} * M /(dt * A) \quad (3)$$

F = Fluxes of CO_2 and CH_4 in mg m^{-2} d^{-1}.
n = Amount of CO_2 and CH_4 in μmol and nmol, respectively.
M = Molar weight of CO_2 (44.01 g/mol) and CH_4 (16.043 g/mol).
dt = Time in days. As every measured concentration represents a 10 min measuring period, dt is equal to `1/(6*24)`.
A = Surface area of the AC in m^2 (0.0934 m^2)

4.2.3.2 Bubble traps

The gas samples from the gas-proof bags were analyzed by using an UGGA machine. The UGGA reported gas concentration output in ppm which was converted in Microsoft Excel, facilitating the calculation of mean CH_4 and CO_2 fluxes from each BT for the four samplings (*Appendix 5*).

Firstly, the initial atmospheric concentration of CO_2 and CH_4 inside the BT at placement was calculated, using the ideal gas law for each gas individually (as for the AC):

$$n = dC * V * p /(R * T) \quad (4)$$

n = Atmospheric amount of CO_2 and CH_4, in µmol.
dC = Atmospheric concentration, in ppm (CO_2: 420 ppm; CH_4; 2 ppm).
V = Air volume inside the BT at placement, in L (1.83 L) (*Section 4.2.2.3 Bubble traps*).
p = Atmospheric pressure (1 atm).
R = 0.082057 L atm K^{-1} mol^{-1}.
T = Temperature in K. For all samples during the study period, a constant of 15 °C = 288.15 K was assumed.

Second, the sampled concentrations are converted from ppm to µmol using the ideal gas law:

$$n = dC * V * p /(R * T) \quad (5)$$

n = The sampled amount of CO_2 and CH_4, in µmol.
dC = The sampled concentration of CO_2 and CH_4, in ppm. The gas sample was diluted by a factor of 11, as a 50 mL gas sample was added to 500 mL N_2 (*Section 4.2.2.3 Bubble traps*). This is corrected by multiplying the sampled concentrations by 11.
V = Gas volume in the BT when sampling, in L (*Section 4.2.2.3 Bubble traps)*.
p = Atmospheric pressure (1 atm). No change in pressure inside the BT is assumed.
R = 0.082057 L atm K^{-1} mol^{-1}.
T = 288.15 K.

Third, the initial atmospheric concentration of CO_2 and CH_4 were subtracted from the sampled concentrations to calculate the fluxes emitted from the pond between samplings.

The fluxes were then converted into mg CO_2 m^{-2} d^{-1} and mg CH_4 m^{-2} d^{-1}:

$$F = n * 10^{-3} * M /(dt * A) \quad (6)$$

F = Fluxes of CO_2 and CH_4 in mg m^{-2} d^{-1}.
n = Amount of CO_2 and CH_4, in µmol.
M = Molar weight, in g/mol (CO_2: 44.01 g/mol; CH_4: 16.043 g/mol).
A = surface area of the BT, in m^2 (0.071 m^2)
dt = Time since the last sample, in days.

The data processing is shown in *Appendix 5*.

4.2.3.3 Global warming potential

CH_4 is a very potent GHG, with a GWP 28 times stronger than CO_2 over a century and even 84 times stronger at a 20-year time period (Pachauri et al., 2015), which is suggested as a more appropriate timeframe (Abernethy & Jackson, 2022; Gorsky et al., 2019; Purre & Ilomets, 2021). The time

horizon considered is important since the atmospheric lifetime of CH_4 is short - approximately 12.4 years (Myhre et al., 2014). During this thesis a conservative GWP at 28 is used for CH_4.
CO_2 has a GWP of 1 regardless of the time period (Myhre et al., 2014; Petersen et al., 2023).

5 Results

The reported and measured fluxes in the following chapter are only expressions of net fluxes (emissions) from the surface to the atmosphere. Therefore, the stated fluxes are not an expression of the carbon balance. The words fluxes and emissions are used interchangeably.

5.1 Literature review

5.1.1 Mean CO_2 and CH_4 fluxes

To answer the research question: *What is the magnitude of CO_2 and CH_4 emissions for ponds in rewetted landscapes?* and the corresponding sub-questions, findings from existing studies are assessed through a literature review. The review is based on 44 studies, including one meta-study, resulting in 58 reported fluxes from different ecosystems (*Table 2*).
Based on the results gathered in the review, it is possible to estimate a mean CO_2 and CH_4 flux from wetlands located in the temperate and boreal zones. The reported CO_2 emissions range from 614 to 33,100 mg CO_2 m^{-2} d^{-1}, with a mean and standard deviation of 10,766 ± 12,440 mg CO_2 m^{-2} d^{-1}. Three measured CO_2 fluxes are identified as outliers with measured emissions ranging from 26,300 to 33,100 mg CO_2 m^{-2} d^{-1}, all deriving from the same paper on measured fluxes in rewetted peatlands. When excluding these outliers, the mean CO_2 flux is 3,728 ± 3,194 mg CO_2 m^{-2} d^{-1}.

The overall mean CH_4 emission is 128 ± 187 mg CH_4 m^{-2} d^{-1} ranging from 1.8 to 1,673 mg CH_4 m^{-2} d^{-1}. When separating the CH_4 fluxes into ebullitive and diffusive fluxes the means, and standard deviations are 186 ± 246 mg CH_4 m^{-2} d^{-1} and 64 ±73 mg CH_4 m^{-2} d^{-1}, respectively. The ebullitive CH_4 flux ranges from 2.01 to 1,061 mg CH_4 m^{-2} d^{-1}, while the diffusive fluxes range from 1.8 to 275 mg CH_4 m^{-2} d^{-1}. The registered value of 1,061 mg CH_4 m^{-2} d^{-1} for CH_4 ebullition fluxes is identified as an outlier, and when excluded the mean CH_4 ebullition flux is 148 ± 163 mg CH_4 m^{-2} d^{-1} and the overall mean CH_4 flux is 115 ± 132 mg CH_4 m^{-2} d^{-1}. In total, this makes ebullition contribute almost 2.5 times more to the total CH_4 flux than diffusion.

Combining CO_2 and CH_4 fluxes, the total mean flux in CO_2-eq is 14,408 mg CO_2-eq m^{-2} d^{-1}, with CO_2 accounting for 25% of the flux and CH_4 accounting for 75%.

As the review is based on a limited number of studies all reported fluxes, including the outliers, are integrated in the following results and analyses. Further discussed in *Section 6.5 How much does ebullition contribute to the total CH_4 flux?*

5.1.2 Waterbody and ecosystem types

The reported fluxes are based on studies of different waterbody and ecosystem types with different properties. Waterbody types cover lakes, ponds, ditches, and beaver ponds while terrestrial ecosystem types cover natural and restored peatlands.

The review comprises 8 studies on lakes. Lakes are categorized as natural waterbodies with a shallow, open water surface, and sparse vegetation. Ponds on the other hand, though similar in visual characteristics, are defined as artificial waterbodies. These are often created in the process of rewetting formerly drained peatlands (*Section 3.3 Climate effect of rewetted peatlands*) and therefore some are characterized by a carbon-rich sediment from the former peat accumulation. In total, the review includes 17 studies on CO_2 and CH_4 fluxes in ponds, where 9 are ponds on peat soils (*marked with * in Table 2*). The review includes 7 studies on fluxes measured in ditches. These are noticeable artificial but vary in surrounding land cover characteristics and therefore also in soil characteristics (*Table 2*). The ditches included in the review are predominantly wet ecosystems but a few of the studies include ditches with a WD at surface or just below surface level. Beaver ponds are characterized as artificial waterbodies because the flooding of another terrestrial ecosystem can be similar to the outcome of rewetting projects. The creation of beaver ponds is often associated with the flooding of forests and wetlands creating shallow and carbon-rich ponds (Wik et al., 2016). The review includes 8 studies on beaver ponds. For terrestrial ecosystems the review comprised 3 studies on natural peatlands and 15 studies on restored peatlands. What characterizes these ecosystems is that the WTD is typically fluctuating around or below ground surface level, as indicated in this literature review where all mean WTD are below surface level. As such, the measured flux in these ecosystems is measured from the soil surface and not a water surface as for the waterbody types.

The waterbody and ecosystem types ranged in depths, with mean WD/WTDs ranging from 10 cm to -320 cm with an overall mean and standard deviation of -74 ± 81 cm, and a mean maximum WD/WTD of -143 ± 101 cm (*Table 2*). Shallow waterbodies are defined as no deeper than 3.2 m.

*Table 2: CO_2 and CH_4 fluxes (both in mg m^{-2} d^{-1}) from different ecosystems in the temperate and boreal zone. Ponds are defined as artificially made water bodies, while lakes are defined as naturally occurring water bodies. Negative values related to the WD are an expression of the water table being above the ground surface. Values in bold are the considered outliers of CO_2 and CH_4 fluxes. *Ponds on peat soils (Appendix 1).*

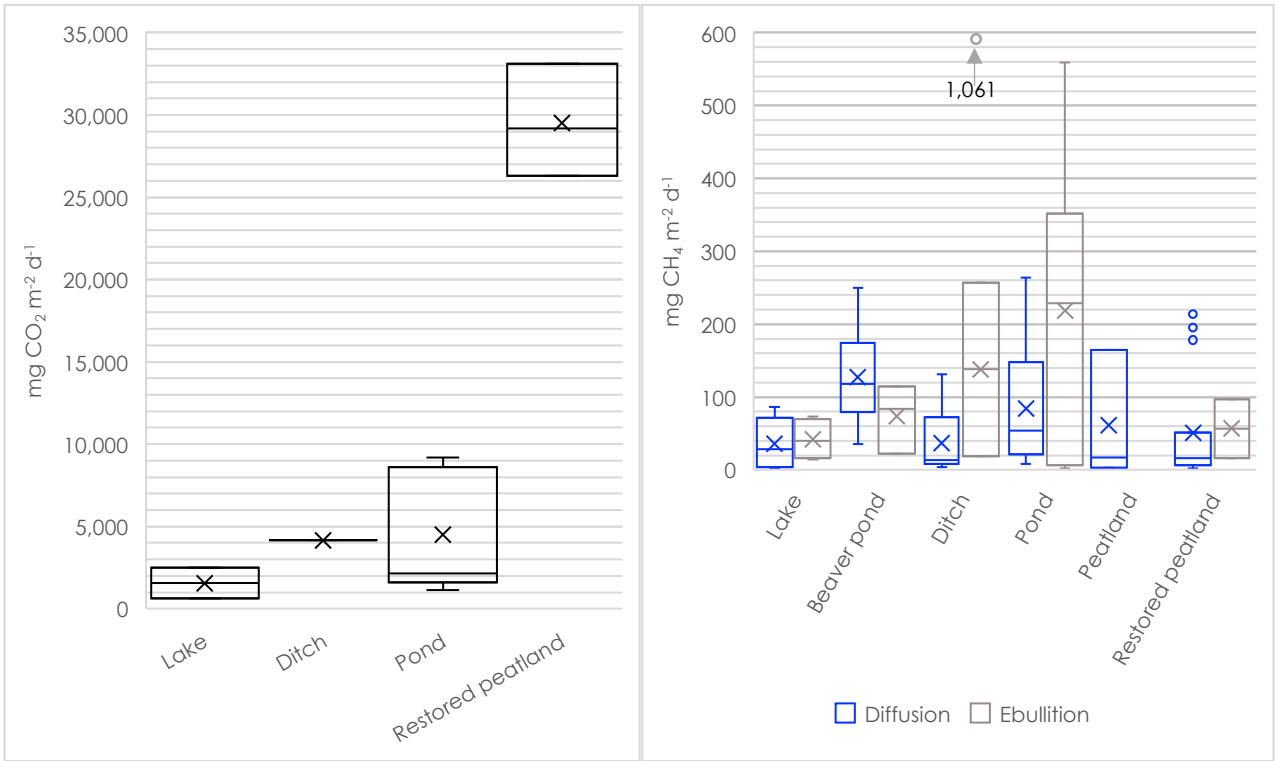
Source	Land cover	Waterbody/ ecosystem type	Water depth *mean, (max.) [cm]*	CO_2 *(std. dev. or std. error or range or quartile) [mg m^{-2} d^{-1}]*	Diffusive CH_4 *(std. dev. or std. error or range or quartile) [mg m^{-2} d^{-1}]*	Ebullitive CH_4 *(std. dev. or std. error or range or quartile) [mg m^{-2} d^{-1}]*
Yavitt et al., 1990	Forest	Beaver pond	-100		275	
Dove et al. 1999. (Read in Baron et al., 2022)	Forest	Beaver pond	(-230)			114
Weyhenmeyer, 1999. (Read in Baron et al., 2022)	Forest	Beaver pond	-140			23 (± 16.64 SD)
Wik et al. 2016 (Review)	Peatland	Beaver pond	(-300)		117 (1-147 Q1-Q3)	84
Yavitt et al, 1992 (Read in Hondula et al., 2014)	Peatland	Beaver pond	(-200)		150	
Roulet et all, 1997 (Read in Hondula et al., 2014)	Wetland	Beaver pond	(-220)		94	
Naimann et al. 1991. (Read in Franz et al., 2016)		Beaver pond	-125		36	
Bubier et al. 1993. (Read in Franz et al., 2016)		Beaver pond	(-150)		121	
Peacock et al., 2021		Ditch	-9 (-100)		72 (-1.3-1,390)	**1061 (3-3,880)**
Peacock et al., 2021 (Cook)	Forest and rural landscape	Ditch	(-78)	4170 (723-12,112)	19 (0.27-70)	257 (± 103 SE)
Köhn et al., 2022	Fen	Ditch	-71		4.3	19
Cooper et al., 2014	Heathland	Ditch			10 (± 2.7 SD)	
Cooper et al., 2014	Heathland	Ditch			14 (± 7.5 SD)	
Hendriks et al. 2007. (Read in Franz et al., 2016)		Ditch			132	

Waddington and Day, 2007. (Read in Franz et al., 2016)		Ditch			7.95	
DelSontro et al., 2016	Forest	Lake	-76 (-2)	2508	67	74 (± 63.84 SD)
Hondula et al., 2021	Forest	Lake	-49.5 (-109)		22 (± 44.81 SD)	
Bastviken et al, 2004		Lake			2.5	
Bastviken et al, 2004	Nature area	Lake			4.6	14
Wik et al. 2013	Peatland	Lake	-70 (-130)			22 (0-1122)
Wik et al. 2016 (Review)	Peatland	Lake	(-320)		86 (26-122 Q1-Q3)	59
Casper et al., 2000. (Read in Franz et al., 2016)		Lake	-230		36	
Ducharme-Riel et al. 2015. (Read in Franz et al., 2016)		Lake	-320	614		
Yavitt et al., 1990	Forest	Peatland			3.33	
Cooper et al., 2014	Heathland	Peatland			17 (± 2.5 SD)	
Kelly et al., 1992	Forest	Peatland			164	
Peacock et al., 2021b	Forest and rural landscape	Pond	(-78)	1121 (99-3,118)	27 (0.27-121)	257 (± 103 SE)
Baron et al., 2022	Forest and rural landscape	Pond	(-200)			347 (± 246.4 SD)
Baron et al., 2022	Forest and rural landscape	Pond	(-200)			200 (± 222.4 SD)
Baron et al., 2022	Forest and rural landscape	Pond	(-200)			2.3 (± 5.31 SD)
Baker-Blocker et al. 1977. (Read in Baron et al., 2022)	Agricultural land	Pond	-100			352 (± 192 SD)
Baker-Blocker et al. 1977. (Read in Baron et al., 2022)	Agricultural land	Pond	-100			432 (± 204 SD)
Baker-Blocker et al. 1977. (Read in Baron et al., 2022)	Agricultural land	Pond	-100			352 (± 192 SD)
Kifner et al, 2018 (Read in Hondula et al., 2014)	Forest	Pond	(-65)		173 (± 296 SD)	
Petersen et al., 2023	Agricultural land	Pond*	-100	7991	17 (0.67-52)	559 (269-2,366)
Vermaat et al., 2011	Agricultural land and nature r	Pond*	-85 (-230)	2064 (± 21 SE)	264 (± 2 SE)	
Scott et al., 1999	Forest	Pond*	(-154)		64 (± 9.3 SE)	
McNicol et al., 2017	Peatland	Pond*	(-150)	9192 (± 954 SE)	8.6 (± 1.5 SE)	2.01 (0.11 - 4.97)
Hoffmann et al. 2017	River valley	Pond*	(-35)		122 (± 137 SD)	101 (± 156 SD)
Franz et al., 2016	River valley	Pond*	(-36)	2167	28	
Kelly et al., 1997	Forest	Pond*			54	
Huttunen et al. 2003. (Read in Baron et al., 2022)	Forest	Pond*	(-320)			(3.5–7.5)
Männistö et al. 2019. (Read in Baron et al., 2022)	Peatland	Pond*	-100			11 (0–252.8)
Bieniada and Strack, 2021	Peat extraction site	Restored peatland		**26300 (± 21,400 SD**	195 (± 181 SD)	16 (± 33 SD)
Bieniada and Strack, 2021	Peat extraction site	Restored peatland		**33100 (± 20,300 SD**	7.7 (± 15 SD)	
Bieniada and Strack, 2021	Peat extraction site	Restored peatland		**29200 (± 19,600 SI**	178 (± 537 SD)	97 (± 628 SD)
Cooper et al., 2014	Heathland	Restored peatland			24 (± 8.1 SD)	
Cooper et al., 2014	Heathland	Restored peatland			16 (± 5.6 SD)	
Morse et al, 2012 (Read in Hondula et al., 2014)	Agricultural land	Restored peatland	Dry		6.5 (-0.48-13.47)	
Beetz et al., 2013. (Read in Abdalla et. al, 2016)	Peatland	Restored peatland	Dry		1.8	
Beyer and Höper, 2015. (Read in Abdalla et. al, 2016)	Peatland	Restored peatland	Dry		2.7	
Juottonen et al., 2012. (Read in Abdalla et. al, 2016)	Peatland	Restored peatland	Dry		3.3	
Komulainen et al., 1998. (Read in Abdalla et. al, 2016)	Peatland	Restored peatland	Dry		9.3	
McNamara, et al.,2008. (Read in Abdalla et. al, 2016)	Peatland	Restored peatland	Dry		18	
Urbanova, Picek et al., 2013. (Read in Abdalla et. al, 2016)	Peatland	Restored peatland	Dry		13	
Vanselow-Algan et al., 2015. (Read in Abdalla et. al, 2016)	Peatland	Restored peatland	Dry		214	
Wilson et al., 2013. (Read in Abdalla et. al, 2016)	Peatland	Restored peatland	Dry		28	
Yli-Petäys, et al., 2007. (Read in Abdalla et. al, 2016)	Peatland	Restored peatland	Dry		52	

When comparing waterbody and ecosystem types and the related CO_2 and CH_4 fluxes it becomes evident that restored peatlands emit the largest fraction of CO_2 compared to the other waterbody types included (*Graph 3*). Examining the release of CH_4 through diffusion the reported emissions are roughly in the same range across the different waterbody and ecosystem types, with beaver ponds and ponds emitting slightly more diffusive CH_4 fluxes than the rest. For CH_4 ebullition the largest fluxes are reported for ditches and ponds (*Graph 3*).

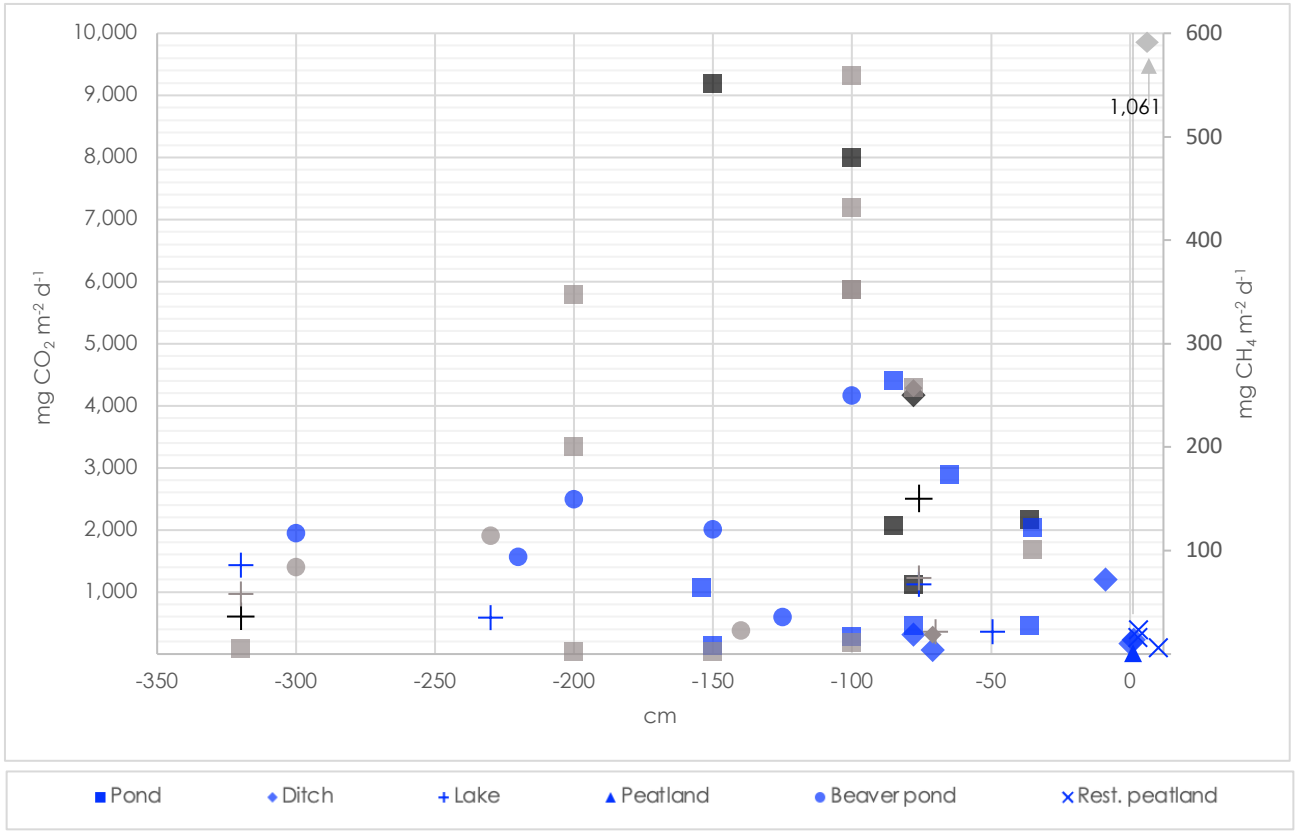
Examining the combined GHG emissions across ponds and restored peatlands the reported fluxes indicate that ponds emit 10,071 mg CO_2-eq m^{-2} d^{-1}, while restored peatlands emit 31,180 mg CO_2-eq m^{-2} d^{-1}. For ponds, 45% of the total GHG emissions are attributed to CO_2 while 55% is emitted as CH_4. In contrast, for restored peatlands 95% is attributed CO_2 while 5% is emitted as CH_4. It shall be noted that the reported CO_2 fluxes for restored peatlands are only based on three sites in one study. For lakes, the total GHG emissions add up to 3,104 mg CO_2-eq m^{-2} d^{-1} with CO_2 and CH_4 contributing equally.

It shall be noted that most studies only report on one flux and often even a single pathway. As such, when comparing fluxes to waterbody and ecosystem types, the reported fluxes represent a very limited range. For example, CO_2 fluxes in restored peatlands comprise three reported fluxes, CO_2 fluxes in lakes are only represented by two study, and for ditches, the mean CO_2 flux is based on just one study.



Graph 3: Boxplot of the different waterbody and ecosystem types (x-axis) and corresponding CO_2 and CH_4 emissions presented as both ebullitive and diffusive fluxes (y-axis). The boxes encompass the interquartile range, the vertical line the median, X indicate the mean, and the whiskers indicate the maximum and minimum of the measured fluxes, circles indicate outliers. There were no reported CO_2 emissions from beaver ponds and natural peatlands, and no reported CH_4 emissions through ebullition in natural peatlands.

The deviation in CO_2 and CH_4 fluxes across waterbody types might furthermore be attributed to the corresponding WD/WTD. The grouping 'restored peatlands' and 'natural peatlands' only include WTDs below ground surface, while ponds and lakes with a WD up to -3.2 m are included in this review. The reported fluxes illustrate a decline in fluxes at a WD deeper than 2 m, while the magnitude of reported fluxes is also declining (*Graph 4*). In general, there is no clear relationship between WD/WTD and fluxes or pathways.



Graph 4: Plot of WD/WDT, CO_2 and CH_4 fluxes, and waterbody / ecosystem type. Colors indicate flux pathways: blue = diffusive CH_4 fluxes, grey = ebullition CH_4 fluxes, black = CO_2 fluxes. The symbols indicate waterbody/ecosystem type. X-axis: WD/WTD in cm (maximum WD/WTD is used when mean WD/WTD is not available). Y-axis: flux in mg CO_2 and CH_4 m^{-2} d^{-1}.

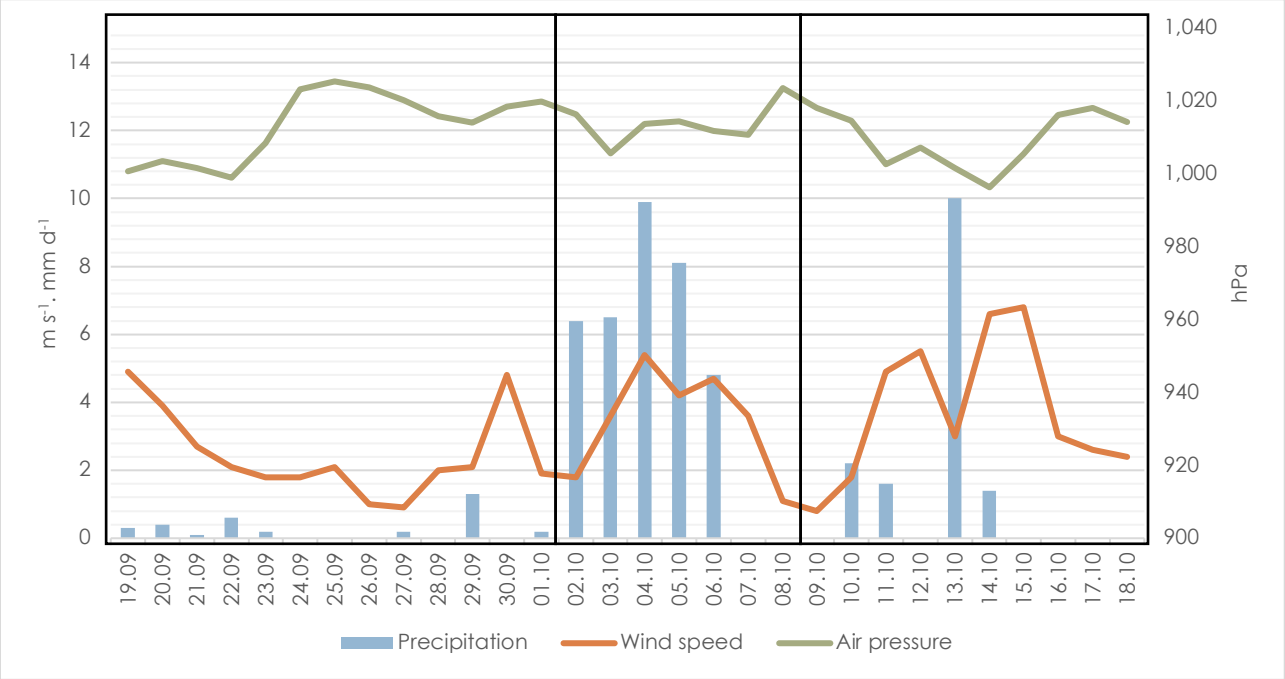
5.2 Field study

5.2.1 Environmental parameters

During the study period, the weather evolves in what can be separated into three distinct periods (*Graph 5*). (1) The first period spans from the 19th of September to the 1st of October and is characterized by calm weather with little to no rain and wind. (2) After the first period, the weather changes with increased precipitation, including total daily precipitation up to 10 mm d^{-1} and increased wind speeds. This is defined as the second period and runs from the 2nd of October to the 8th of October. (3) From the 9th of October until the end of the study period on the 18th of October, the weather has reached a third distinct period. In this period the weather is characterized by sporadic

increases in wind speeds and precipitation. Throughout the study period, air pressure fluctuates from 996 – 1025 hPa showing an inverse relationship with precipitation and wind speed (*Graph 5*).

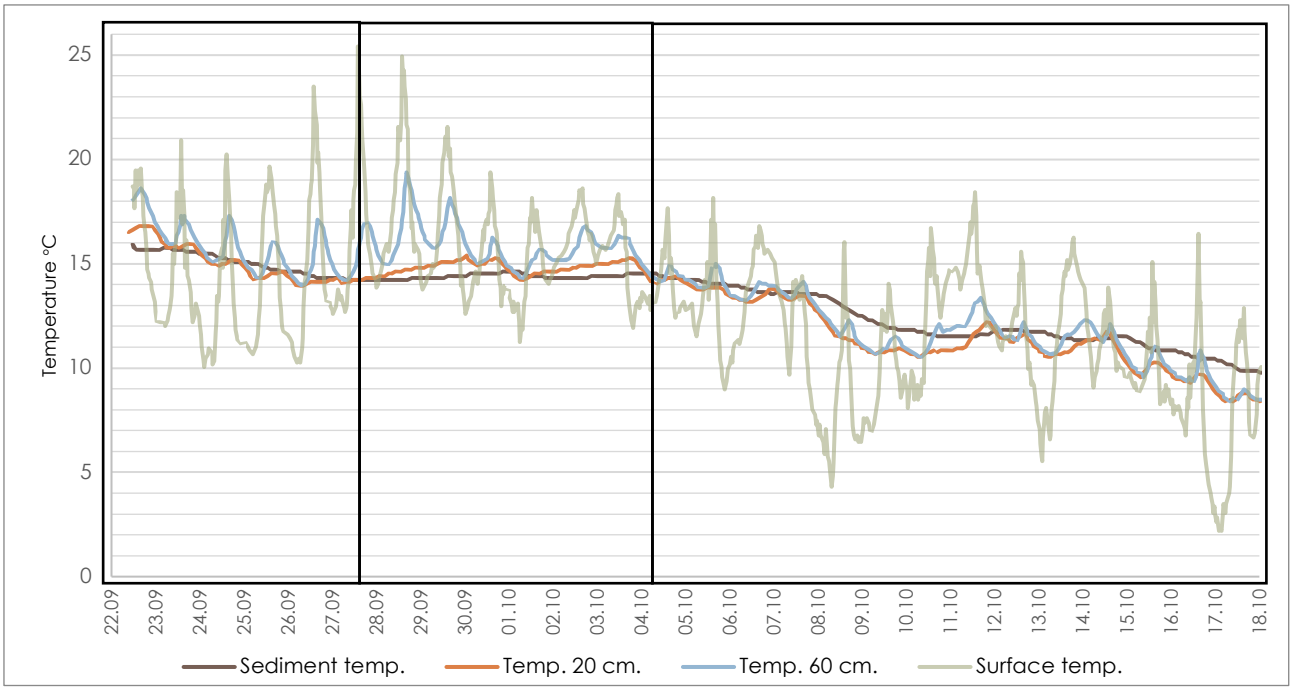
The wind curve represents the daily mean wind speed, which is smoothing out the curve, and leaving out the visualization of the hourly peaks in wind speed. Despite this, it is important to mention that hourly wind speeds at 10-13 m s^{-1} have been measured during the study period e.g. during the 4th, 6th, and 7th of October.



Graph 5: The development in weather during the study period. X-axis: time in days. 1. y-axis: mean daily wind speed in m s^{-1} and summed daily precipitation in mm d^{-1}. 2. Y-axis: mean daily air pressure in hPa. The three black boxes indicate the three different weather periods: (1) calm weather, (2) change in weather, and (3) windy and rainy weather.

The measured sediment, water, and air temperatures show an overall decline from the 22nd – 27th of September followed by a period of steadier temperatures until the 4th of October (*Graph 6*). The temperature for the rest of the study period is declining.

The temperature logger placed 10 cm above the water surface shows the largest temperature variations throughout the entire study period ranging from 2–25°C, with a max daily fluctuation of 14°C. The temperature fluctuations are partly attributed to the fact that the logger was placed with no shade, exposing it to temperature increases affected by direct sunlight (*Section 4.2.2.1 Environmental parameters*). The temperature logger placed 60 cm above the sediments has a max daily fluctuation of 4°C while the temperature loggers placed 20 cm above and, in the sediment, have a max daily fluctuation of 1°C and 0.5°C, respectively. These measurements reveal that the upper layers of the water column (60 cm above the sediment) are highly responsive to daily air temperature fluctuations, while the lower layers situated only 20 cm above the sediment are less influenced by changes in air temperature indicating that the pond has the potential to be thermally stratified (*Graph 6)*.



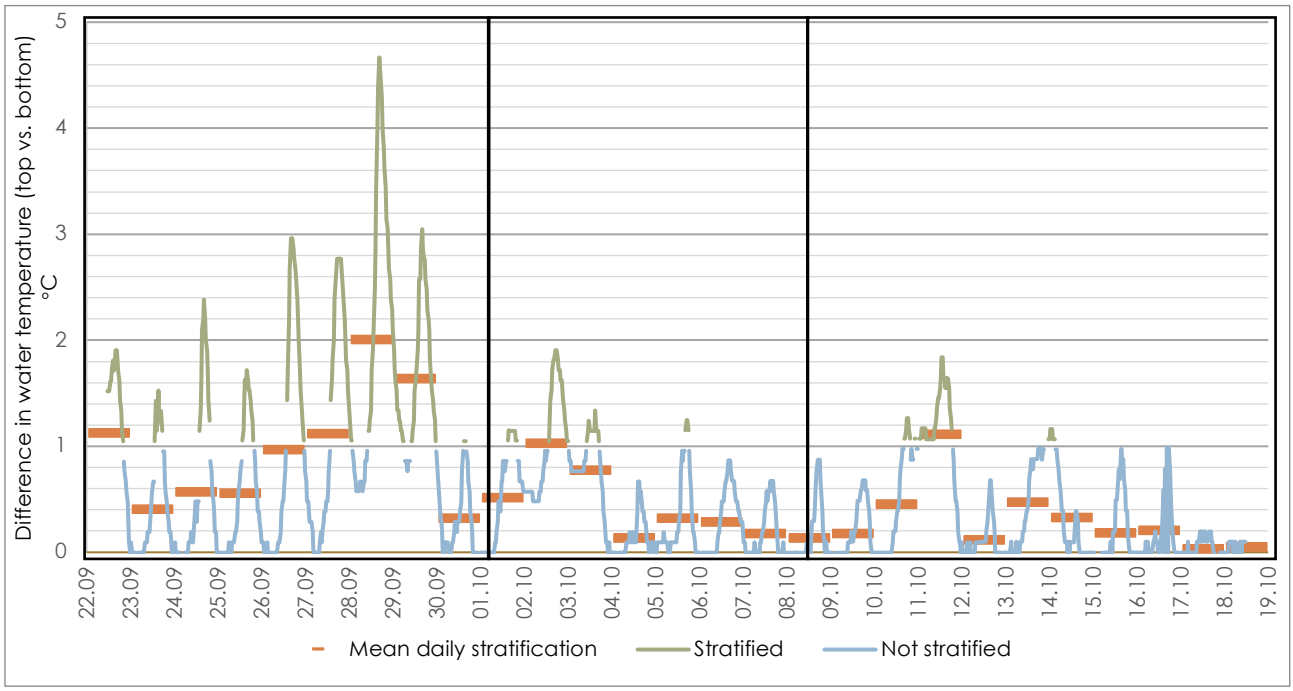
Graph 6: development in temperature during the study period. X-axis: time in days. Y-axis: temperature measurements in ℃ every 30. Min. Brown: sediment temperature. Orange: temperature 20 cm above sediments. Blue: temperature 60 cm above sediments. Green: surface temperature, around 10 cm above water surface. The three black boxes indicate the three different temperature periods: (1) decreasing temperature, (2) steady temperature, and (3) decreasing temperature.

Thermal stratification of the water column is determined by the difference in temperature between the upper and lower layers of the water column (*Section 4.2.2.1 Environmental parameters*). When calculating the difference between the water temperatures at 20 and 60 cm above the sediments it is possible to determine whether Mårumhus Pond is thermally stratified on a 30 minute and a daily basis (*Graph 7*). Thermal stratification serves solely as a proxy for stratification in O_2 concentration (*Section 4.2.2.1 Environmental parameters*). Examination of 30-minute water temperature data from Mårumhus Pond suggests that thermal stratification builds up during daytime, while the stratification seems to break down during nighttime. However, given the relatively slow water mixing rate, this daily fluctuation is mostly attributed to the rather imprecise method, more than it is an indication of daily buildup and breakdown in stratification.

By the 4th of October, there is an observed convergence in temperatures (*Graph 6*), indicating a breakdown in the thermal stratification. In addition, *Graph 7* shows only a few smaller stratification events after the 4th of October. When comparing wind speed, precipitation, and air pressure with the measured temperatures, it seems that the breakdown in stratification is related to an increase in wind speed and precipitation resulting in the mixing of the water column. This indicates a potential fall turnover event.

The three identified weather periods can when comparing to the stratification, be categorized as (1) a stratified period from the 22nd of September to the 1st of October, (2) a turnover period from the 2nd

of October to the 8th of October, and (3) a mixing period from the 9th of October to the 18th of October (black boxes in *Graph 7*).



Graph 7: Stratification of Mårumhus Pond during the study period. X-axis: time in days. Y-axis: Difference in water temperature (top vs. bottom) in ℃. Mean daily stratification values below 1℃ indicate that the 30 min stratification levels above 1℃ may just be expressions of warming of the surface water from sunlight, more than true thermal stratification. The three black boxes indicate the three different stratification periods: (1) a stratified period, (2) a turnover period, and (3) a mixing period.

5.2.2 Automatic Chambers

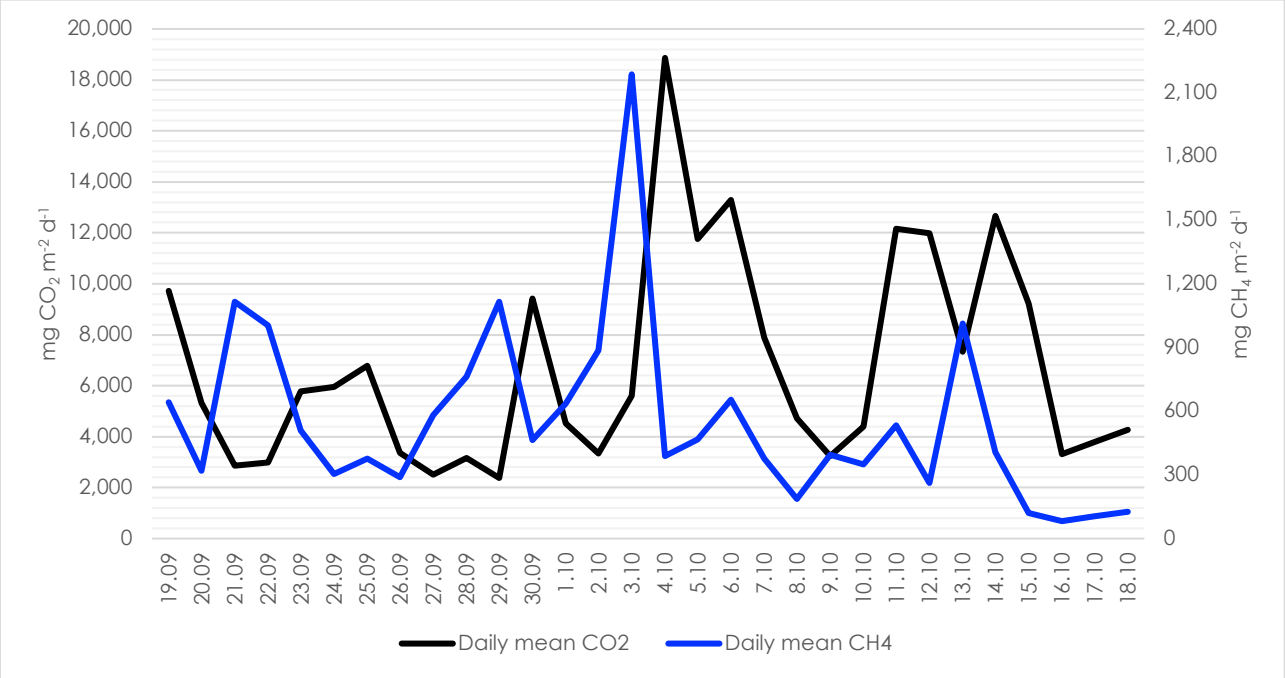
Throughout the entire study period, the eight ACs record an overall mean and corresponding standard deviation of CO_2 and CH_4 at 6,912 ± 5,905 mg CO_2 m^{-2} d^{-1} and 557 ± 714 mg CH_4 m^{-2} d^{-1}, respectively. The large standard deviations are caused by the large daily and even hourly variations in fluxes. The ranges are between 9 – 35,716 mg CO_2 m^{-2} d^{-1} and 0.5 – 7,789 mg CH_4 m^{-2} d^{-1}.

When converting the fluxes into CO_2-eq, a total of 22,510 ± 21,136 mg CO_2-eq m^{-2} d^{-1} are emitted. 31% of the overall mean flux is emitted as CO_2 while 69% is attributed to CH_4 (6,912 ± 5,905mg CO_2 m^{-2} d^{-1}; 15,597 ± 19,998 mg CH_4 m^{-2} d^{-1}). The distribution of fluxes is shown in *Appendix 4*.

5.2.2.1 Temporal variations

In general, CO_2 fluxes exhibit a shift during the study period, having lower fluxes from the 19th of September to the 1st of October with a mean of 4,709 mg CO_2 m^{-2} d^{-1} ranging from 0 – 21,892 mg CO_2 m^{-2} d^{-1}, while having a mean of 8,224 mg CO_2 m^{-2} d^{-1} and ranging from 0 – 35,716 mg CO_2 m^{-2} d^{-1} from the 2nd to the 18th of October (*Graph 8*).
Four periods of peaks are present in the CO_2 flux curve: the 23rd –25th, and 30th of September, and 4th – 6th, and 11th –14th of October.

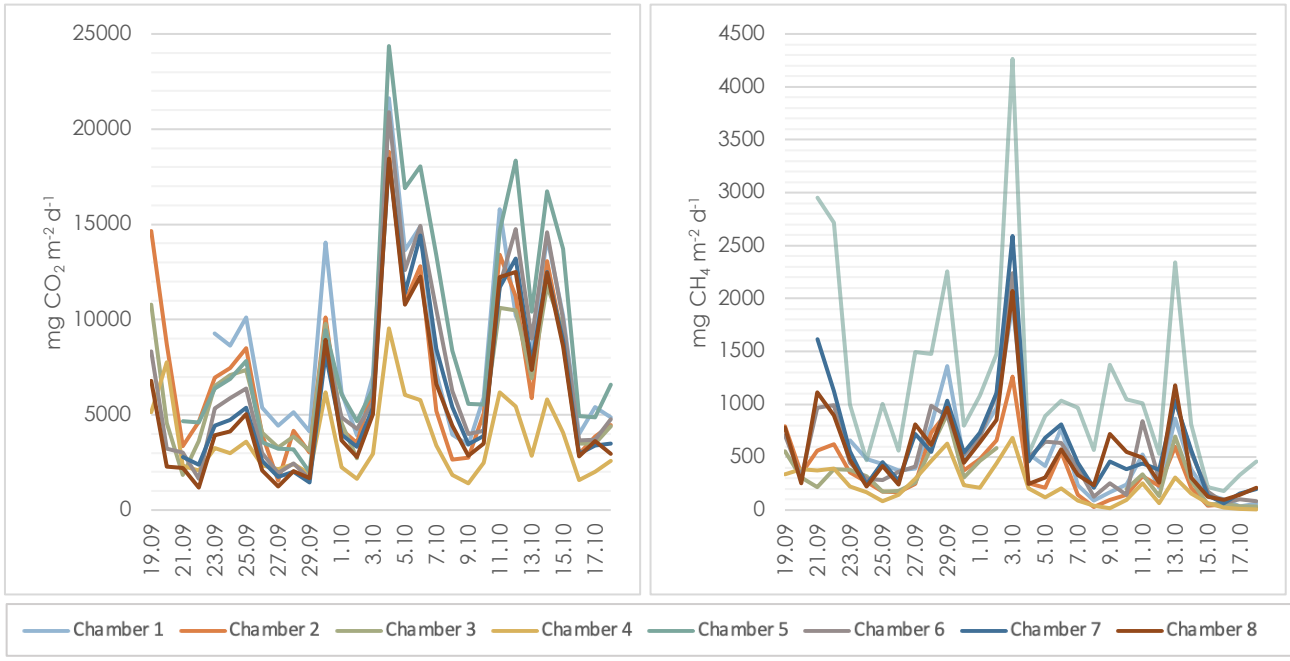
In contrast, CH_4 is steadier with slightly decreasing fluxes during the study period. From the 19th of September to the 1st of October, the mean is 625 mg CH_4 m^{-2} d^{-1} ranging from 12 – 5,919 mg CH_4 m^{-2} d^{-1}. From the 2nd to 18th of October) the mean is 506 mg CH_4 m^{-2} d^{-1} ranging from 1 – 7,789 mg CH_4 m^{-2} d^{-1}. The CH_4 flux curve contains four remarkable peaks on the 21st, 29th of September and the 3rd and the 13th of October (*Graph 8*).



Graph 8: Development in mean daily fluxes during the study period. X-axis: time in days. 1. y-axis: Mean daily CO_2 fluxes in mg CO_2 m^{-2} d^{-1}. 2. Y-axis: Mean daily CH_4 fluxes in mg CH_4 m^{-2} d^{-1}. Appendix 4.

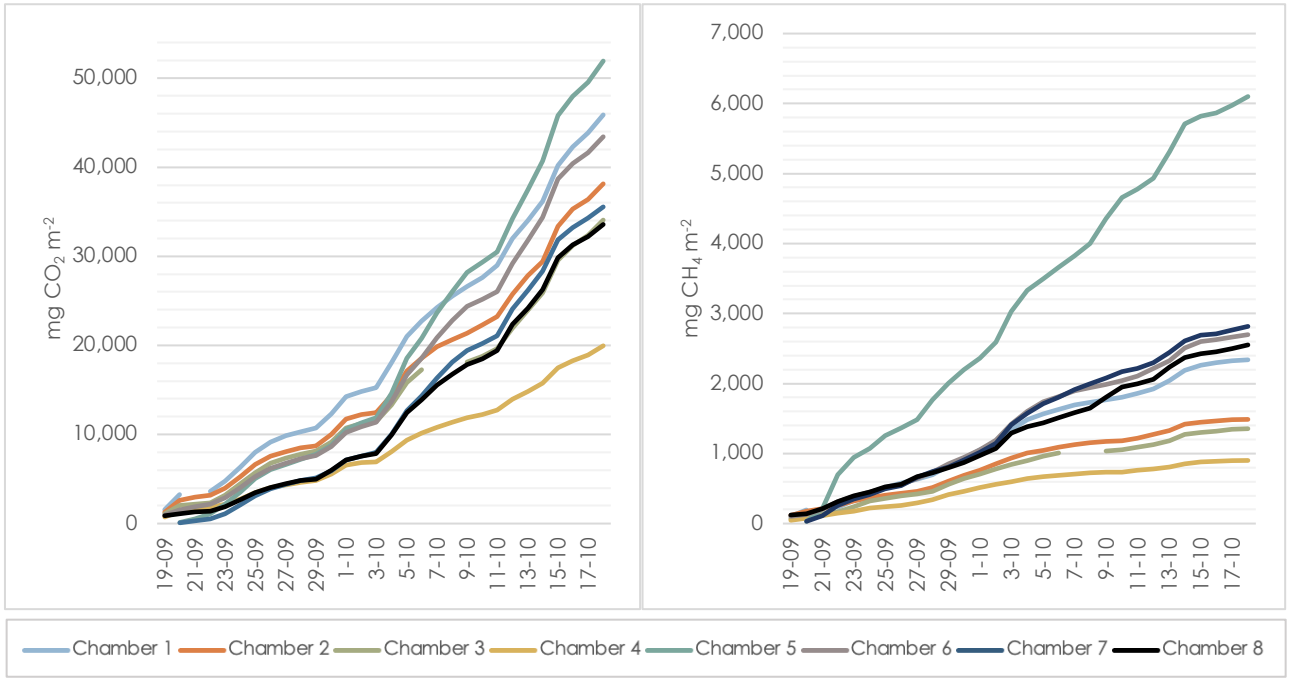
When examining the mean daily fluxes across the eight ACs (*Graph 9, left*) it becomes evident, that the pattern of the mean daily CO_2 fluxes is relatively equal across all ACs with the exception of AC no. 4 which generally shows lower measured fluxes, especially in the flux peaks from the 3rd to the 6th of October, while still following the same flux pattern as the other ACs.

For CH_4, it becomes evident, that the pattern of the mean daily fluxes (*Graph 9, right*) is impacted greatly by the flux pattern of ACs no. 5, 6, 7, and 8 while ACs no. 2 and 4 do not show the same extreme peaks (*Graph 9, right*).



Graph 9: Development in mean daily fluxes for AC 1-8 during the study period. X-axis: time in days. Left: CO_2 fluxes, Y-axis: Mean daily CO2 fluxes in mg CO2 m^{-2} d^{-1}. Right: CH_4 fluxes, Y-axis: Mean daily CH_4 fluxes in mg CH_4 m^{-2} d^{-1}. Appendix 4.

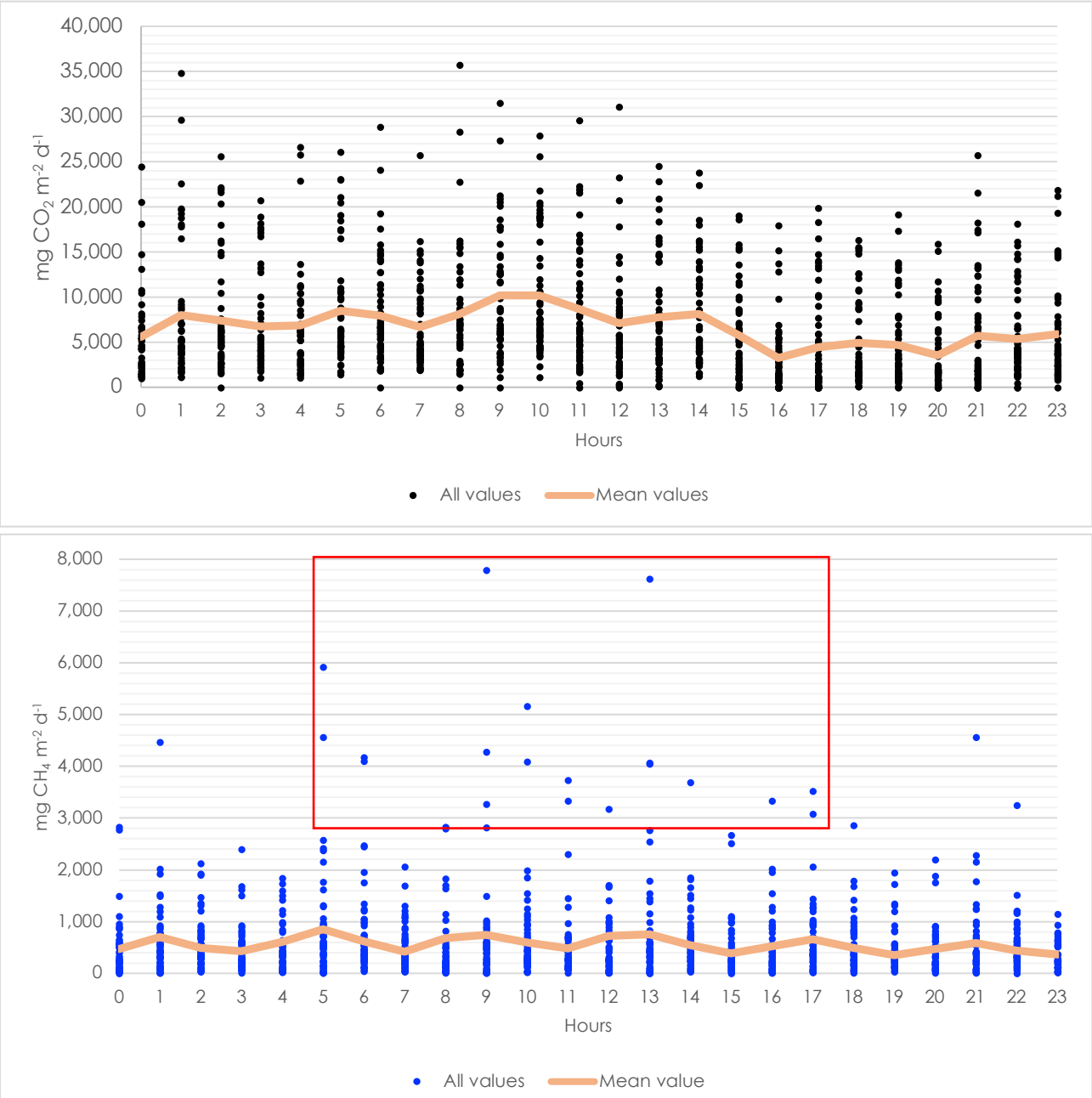
The accumulated CO_2 and CH_4 fluxes for each AC during the study period ranges from 109,483 – 249,961 mg CO_2 m^{-2} and 6,729 – 33,643 mg CH_4 m^{-2}, respectively (*Graph 10*). The mean and std. of the accumulated CO_2 flux is 186,898 ± 46,303 mg CO_2 m^{-2} and for CH_4 fluxes 15,504 ± 8,514 mg CH_4 m^{-2}. AC no. 5 reaches the highest fluxes while AC no. 4 reaches the lowest fluxes for both CO_2 and CH_4 (*Graph 10*). In fact, AC no. 4 only comprises 44% and 20% of the accumulated CO_2 and CH_4 fluxes compared to that of AC no. 5.



Graph 10: Accumulated fluxes during the study period. Left: CO_2 fluxes. X-axis: Time in days. Y-axis: CO_2 fluxes in mg CO_2 m^{-2}. Right: CH_4 fluxes. X-axis: Time in days. Y-axis: CH_4 fluxes in mg CH_4 m^{-2}. Appendix 4.

Regarding CO_2 fluxes, a daily pattern is evident showing higher mean hourly fluxes at 8,798 mg CO_2 m^{-2} d^{-1} from 07:00 - 10:00 and lower mean hourly flux of 4,169 mg CO_2 m^{-2} d^{-1} from 16:00-20:00 (*Graph 11, top*).

There is no clear daily pattern in measured CH_4, when examining the mean hourly flux which ranges from 352 – 853 mg CH_4 m^{-2} d^{-1} (*Graph 11, bottom*). Looking at the span of measured fluxes, there is a tendency for the largest fluxes to be scattered in a timespan from 05:00 - 17:00, indicating that the largest fluxes are emitted during daytime.

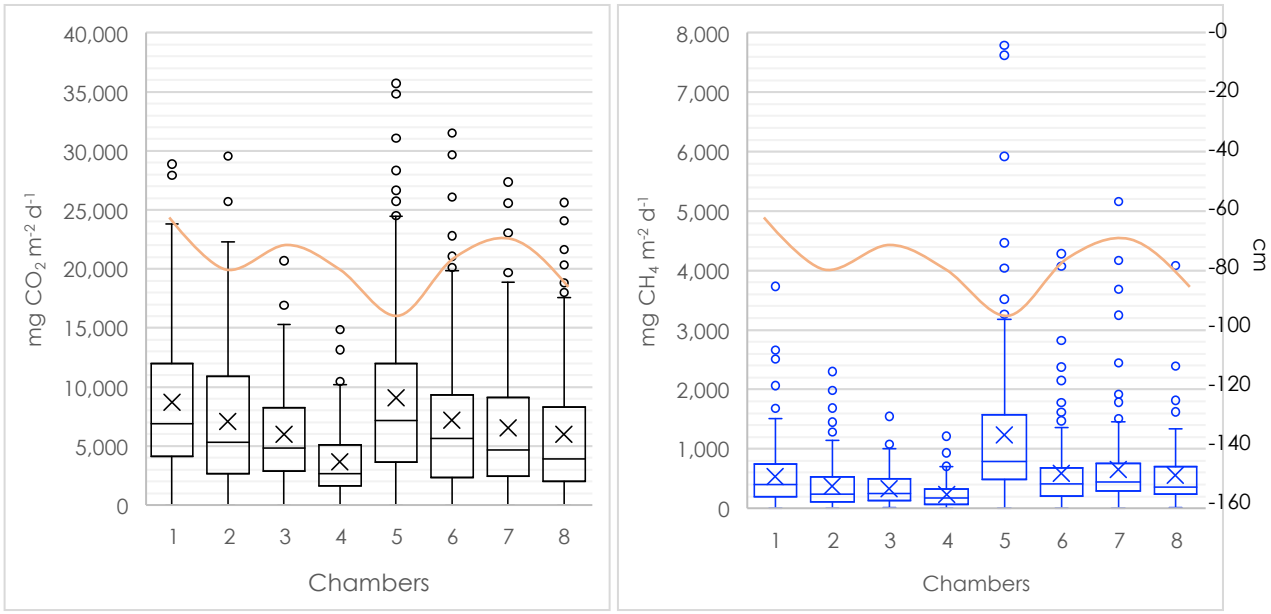


Graph 11: Development in daily pattern. Top: CO_2 fluxes. X-axis: time in hours. Y-axis: CO_2 fluxes in mg CO_2 m^{-2}. Bottom: CH_4 fluxes. X-axis: time in hours. Y-axis: CH4 fluxes in mg CH_4 m^{-2}. The red boxes indicate when most of the high CH_4 fluxes are collected. Appendix 4.

5.2.2.2 *Spatial variations*

Upon examining the CO_2 and CH_4 fluxes across the eight AC (*Graph 12*), it is obvious that fluxes for CO_2 and CH_4 across AC generally show the same pattern, meaning that high fluxes of CO_2 can also indicate high fluxes of CH_4, and vice versa, across AC. AC no. 5 captures the largest fluxes of both CO_2 and CH_4 compared to all the other ACs, while AC no. 4 captures the lowest fluxes of both CO_2 and CH_4. Examining the correlation between measured CO_2 and CH_4 levels within each individual measurement does not indicate any relationship (*Appendix 4*).

Since the ACs were situated along a transverse gradient spanning from the shoreline to the pond's midpoint, the AC number is an indicator of the distance to the shore. Having that in mind, there is no correlation between fluxes and distance to the shore. Since the pond bed does not exhibit a u-shaped configuration (*Graph 12*), the influence of the WD of the pond must be considered. Testing the relationship between fluxes and the corresponding WD indicates no correlation between WD and emissions (*Appendix 4*). Additionally, the WD corresponding to each AC is visually represented in *Graph 12,* which is also not showing a clear relationship between fluxes and WD. For instance, AC no. 5, positioned at the greatest depth, exhibits the highest fluxes, whereas AC no. 4, situated at a depth of 80 cm, displays the lowest measured fluxes. In contrast, AC no. 1, located at the lowest point, captures fluxes higher than AC no. 4 but lower than AC no. 5.



Graph 12: Boxplot showing CO_2 and CH_4 fluxes for each og the eight AC. Orange line indicate the depts below the corresponding AC. Left: CO_2 fluxes (2. Y-axis). X-axis: The eight AC. Y-axis: CO_2 fluxes in mg CO_2 m^{-2}. Right: CH_4 fluxes. X-axis: the eight AC. Y-axis: CH_4 fluxes in mg CH_4 m^{-2}. The box encompasses the interquartile range, the vertical line the median, X indicate the mean, and the whiskers indicate maximum and minimum of the measured fluxes, circles indicate outliers. The WTD corresponding to each AC is shown by the orange curve, with WTD measured in cm on the 2. y-axis. Appendix 4.

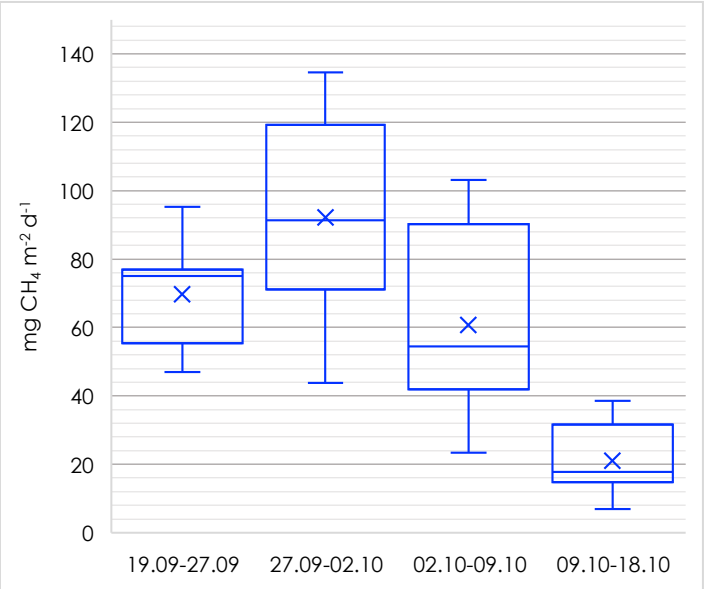
5.2.3 Bubble traps

The CO_2 and CH_4 fluxes measured in the eight BT during the study period range from 0.42 to 1,226 mg CO_2 m^{-2} d^{-1} and 7 to 135 mg CH_4 m^{-2} d^{-1}, respectively. The means and standard deviations of the two fluxes are 55 ± 216 mg CO_2 m^{-2} d^{-1} and 61 ± 34 mg CH_4 m^{-2} d^{-1}.

Comparing the mean emissions of CO_2 measured in the BT with the mean emissions from the AC, it is evident that the measured CO_2 fluxes in AC (mean: 6,912 mg CO_2 m^{-2} d^{-1}) are around 125 times larger than those from the BT. The low CO_2 emissions measured in the BTs may be caused by several factors. Due to great uncertainties connected to the BT CO_2 flux measurements, these results are not further analyzed (further discussed in *Section 6.6.3 Bubble trap setup*).

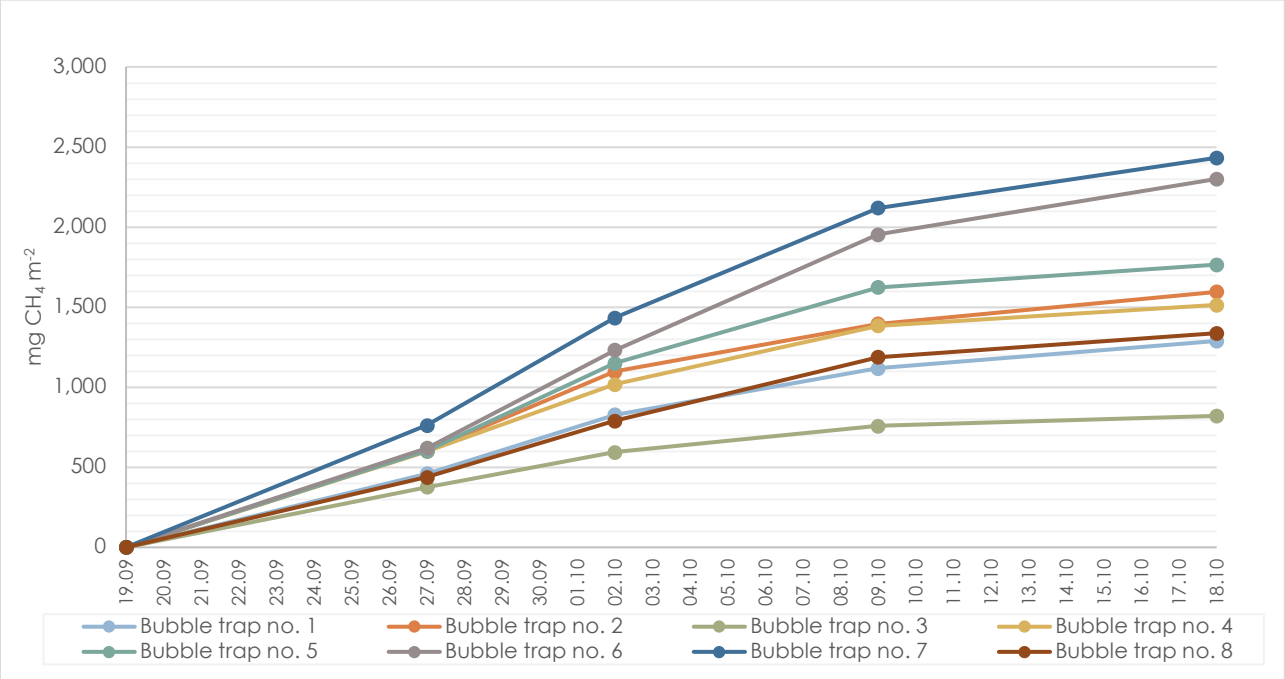
5.2.3.1 Temporal variations

The sampled CH_4 fluxes show an increase between the first sampling on the 27th of September and the second sampling on the 2nd of October. For the rest of the study period the measured fluxes are decreasing, with the last sampling showing substantially lower fluxes of CH_4 across BT than the three previous sampling periods (*Graph 13*).



Graph 13: Boxplot for measured CH_4 flux on the four different sampling days. X-axis: time periods between the four sampling days. Y-axis: CH_4 fluxes in mg CH_4 m^{-2} d^{-1}. The box encompasses the interquartile range, the vertical line the median, X indicate the mean, and the whiskers indicate maximum and minimum of the measured fluxes. Appendix 5.

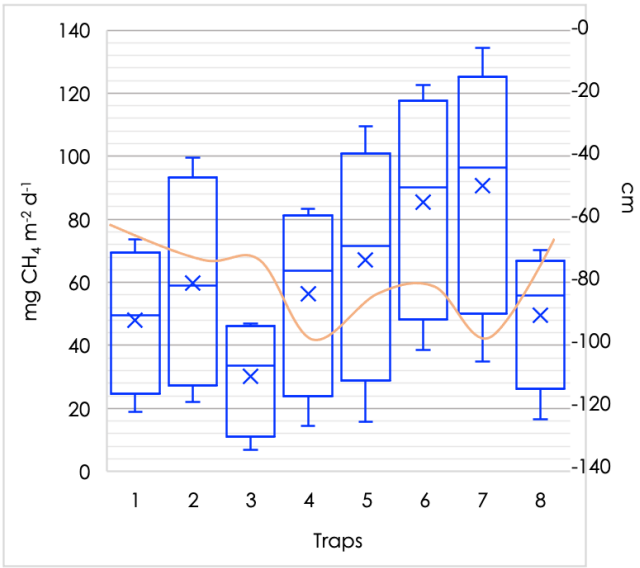
The accumulation of CH_4 sampled from the BT during the study period shows a similar pattern in measured CH_4 fluxes across the eight BTs (*Graph 14*). It is evident that BT no. 7 and BT no. 6 accumulate the largest amount of CH_4 fluxes throughout the study period with total accumulated CH_4 emissions of 2,432 and 2,300 mg CH_4 m^{-2}, respectively. On the opposite end of the scale BT no. 3 accumulates less than half of BT no. 7 with total accumulated CH_4 emissions of 821 mg CH_4 m^{-2}.



Graph 14: Accumulated CH_4 flux for the eight BT during the study period. X-axis: time in days. Y-axis: CH_4 fluxes in mg CH_4 m^{-2} d^{-1}. The four sampling days are marked with round symbols on the curves. Appendix 5.

5.2.3.2 Spatial variations

The differences in measured CH_4 fluxes might be related to differences in WD and/or distance to shore. Examining the range of fluxes across the eight BTs (*Graph 15*) it is evident that some BTs contain less CH_4 than others. BT no. 6 and no. 7 have the highest measured CH_4 fluxes of all the BTs (mean: 85 and 91 mg CH_4 m^{-2} d^{-1}) while BT no. 3 has the lowest measured CH_4 fluxes (mean: 30 mg CH_4 m^{-2} d^{-1}). However, the fluxes do not seem to be related to the WD as they are placed at depths of 80 cm, 95 cm, and 77 cm respectively, not encompassing maximums nor minimums of the WD. Therefore, there is no relationship between WD and fluxes (*Graph 15 and Appendix 5*).



Graph 15: Boxplots of CH_4 fluxes for the eight BT and the corresponding depths. X-axis: the eight BT. 1. Y-axis: CH_4 fluxes in mg CH_4 m^{-2} d^{-1}. 2. Y-axis: the depths of each BT in cm. The box encompasses the interquartile range, the vertical line the median, X indicate the mean, and the whiskers indicate maximum and minimum of the measured fluxes. Appendix 5.

5.3 Summary

In conclusion, the reported mean CO_2 and CH_4 fluxes in the literature review and the mean fluxes measured in the AC and BT are stated in *Table 3*. In addition, IPCCs emission estimates for rewetted organic soils in the temperate climate zone (Hiraishi et al., 2014) are presented in the table, as they are used in the following discussion (*6.1.1 Comparing measured and reported fluxes to IPCC emission factors*).

Table 3: The table shows a summary of reported and measured CO_2 and CH_4 fluxes as presented in the results section above. Included are also estimated emissions for rewetted organic soils in the temperate zone (nutrient status: rich) as reported by IPCC (Hiraishi et al., 2014) (recalculations in Appendix 10). A GWP of 28 for CH_4 is used.

	CO_2 [mg CO_2 m^{-2} d^{-1}]	CH_4 [mg CH_4 m^{-2} d^{-1}]	Total [mg CO_2-eq m^{-2} d^{-1}]	CO_2 share [%]	CH_4 share [%]
Literature review	10,766 ± 12,440	128 ±187	14,408	25	75
Automatic chamber	6,912 ± 5,905	557 ± 714	22,510 ± 21,136	31	69
Bubble trap		61 ± 34	1,708 ± 952		
IPCC	502	79	2,714	18	82

6 Discussion

6.1 How much do CO_2 and CH_4 contribute to the total flux respectively?

The mean CO_2 fluxes reported in the literature and measured during the field study have similar extents and ranges (*Table 3*).
Comparing mean CH_4 fluxes between the literature review and measured fluxes in the field study indicate larger differences. The largest fluxes are measured with AC, while CH_4 fluxes reported in the literature review show almost four times smaller mean fluxes, and fluxes measured with BT are even 9 times smaller than measured fluxes with AC (*Table 3*). It is likely that the reported CH_4 fluxes from the literature review are underestimated, as the total CH_4 flux involves multiple pathways, including diffusive and ebullitive fluxes. The majority of studies only report on one of these pathways, and as a result, the total flux is underestimated. The variance between measured fluxes has different implications in relation to the used methods which are further discussed in *Section 6.6 Discussion of methods*.

When comparing total GHG fluxes across literature review and the field study they present similar shares with CO_2 fluxes contributing to 25% and 31%, and CH_4 accounting for 75% and 69% in literature and AC measurements, respectively. Consequently, the measured fluxes during the field study surpass the estimates found in the existing literature by 56% in GHG emissions (*Table 3*).

6.1.1 Comparing measured and reported fluxes to IPCC emission factors

Comparing IPCCs emissions estimates to the mean fluxes reported in the literature review and measured in the field study, it is evident that the CO_2 emissions might be significantly underestimated by IPCC, by reporting 10-20 times lower emissions than what is reported in the literature review and measured during the field study (*Table 3*).
In contrast, the emission estimate reported by IPCC for CH_4 is in the same range as reported in the literature review and measured with BT (*Table 3*). However, the CH_4 fluxes measured with AC (*Table 3*) show 7 times higher fluxes than IPCC's emission estimates.

The possible underestimation of CO_2 and CH_4 fluxes reported by IPCC may lead to a general impression that rewetting organic soils has a positive, net cooling impact, which might not be the case. As mentioned, it is important to note that this thesis has not reported on carbon uptake. When comparing to other studies reporting on organic carbon (OC) uptake in artificial ponds, four recent studies have found an average OC burial rate between 67 and 216 g OC m^{-2} yr^{-1} (or 246 and 792 g CO_2-eq m^{-2} yr^{-1}) (Goeckner et al., 2022; Holgerson et al., 2023; Rogers et al., 2022; Taylor et al.,

2019). Comparing this to a yearly accumulated flux from Mårumhus Pond at 3,726 g CO_2-eq m^{-2} y^{-1} by AC[1], indicates that the fluxes are 5-15 times larger than the uptake in artificial ponds.
This comparison is too simplistic for drawing definitive conclusions, but it indicates significantly higher fluxes than the burial rate, stressing the importance of evaluating net carbon balance budgets for these man-made ponds situated in rewetted peatlands.

6.1.2 Variance in measured and reported fluxes

To explain the variation in CO_2 as well as CH_4 fluxes, it is important to consider temporal as well as spatial parameters that may vary between different studies. As an example, parameters such as (1) seasons, (2) O_2 concentration, (3) pH level, and (4) eutrophication were not included in this thesis. These factors might explain the variance in reported and measured CO_2 and CH_4 fluxes, respectively, as described below:

(1) Both CO_2 and CH_4 are highly responsive to temperature, precipitation, and wind speed (*Sections 3.1.2, 3.3.1, and 3.3.2 in Section 3 Theory*), which all vary vastly by season. Additionally, increases in both CO_2 and CH_4 have been reported at turnover events in spring and fall (*Section 3.3.2 CH_4 balance in rewetted peatlands*) (Riera et al., 1999). By leaving out the seasonal variations, the measured CO_2 and CH_4 fluxes cannot without further consideration be scaled up to yearly fluxes, as they might be considered rather high, in comparison to fluxes measured during the ice covered winter and early spring months. This consideration also holds true for emissions documented in the literature review, where seasonal considerations have not been accounted for.

(2) O_2 concentrations in the water column have a positive contribution on CO_2 fluxes, while being negatively related to CH_4 (Peacock et al., 2021b) (*Section 3.1.2 CH_4 balance in natural peatlands*). The inclusion of O_2 will contribute to illuminating the relation between O_2 concentration and CO_2 and CH_4 fluxes. Additionally, a more precise picture of stratification can be drawn.

(3) According to Peacock et al. (2021b), CO_2 fluxes and pH are inversely related and at pH levels above 8, CO_2 is almost absent. This is explained by the dominance of carbonates (H_2CO_3) over CO_2 in alkaline environments (Peacock et al., 2021b). Therefore, pH-level will determine especially the CO_2 fluxes, and can explain the variance between fluxes in different studies.

(4) A last explanation of the difference in reported CH_4 fluxes can be related to eutrophication. It is reported by Peacock et al. (2021b) and DelSontro et al. (2016) that eutrophication will alter CH_4 production. As eutrophic systems improve the circumstances for aquatic plants, eutrophic systems tend to accumulate more dead organic matter and thereby organic carbon in the sediments. In addition,

[1] *Mean monthly flux during the year is calculated by = [measured mean monthly accumulated CO_2 during the study period: 186,898 mg CO_2 m^{-2}] + [measured mean monthly accumulated CH_4 during the study period: 15,504*(28GWP) mg CH_4 m^{-2}] * [12 months] * [0.5 to acknowledge that September and October could potentially represent months during which the highest fluxes may be measured] (Appendix 10). According to Spafford and Risk (2018), mean fluxes during August, September and October are 100% higher than for May, June, and July. Therefore, a mean yearly flux during the six months is expected to be 50% lower than what is measured during autumn (August-October).*

aquatic plants will decrease the concentration of O_2 used for growing and thereby make the water become even more anoxic, decreasing the possibility of CH_4 oxidation. Furthermore, this leads to increased CH_4 ebullition, as diffusion cannot balance out the CH_4 concentration between the water and the atmosphere alone (DelSontro et al., 2016).

By not considering parameters like the above it can be difficult to compare CO_2 and CH_4 fluxes across studies, as the effects of the different parameters may lead to significantly different fluxes. However, different studies also indicate contrary findings on the influence of the same parameters. Bastviken et al. (2004) find that surface area together with phosphorous and DOC concentrations are the parameters best indicating CH_4 emissions, while Wik et al. (2016) finds no strong correlation between surface area and CH_4 emissions and instead suggest that the assumed effect from surface area is actually an effect of sediment type and WD.

6.1.3 Sub-conclusion

To answer sub-question *1. How much do CO_2 and CH_4 contribute to the total flux respectively?* The total GHG emissions based on CO_2 and CH_4 alone seem to be mostly influenced by CH_4 emissions contributing to approximately 75% of the total emissions. It seems that fluxes are highly influenced by many parameters, and it can be difficult to encompass all in one study. However, limiting the number of parameters included can limit the understanding of influences on CO_2 and CH_4 fluxes. Additionally, when comparing the measured fluxes to emission estimates presented by IPCC it appears that IPCC might be greatly underestimating the impact of rewetting on GHG emissions.

6.2 How do CO_2 and CH_4 fluxes relate to spatial factors?

6.2.1 The influence of WD and WTD on CO_2 and CH_4 fluxes

It is expected that emissions of both CO_2 and CH_4 increase with a lower WD. This increase is attributed to the reduction of CH_4 oxidation, increased organic matter accumulation in the littoral zone, and greater impact of sediment to environmental fluctuations, such as temperature, precipitation, and wind. These factors collectively impact the production and release of CO_2 and CH_4 (*Section 3.3 Climate effect of rewetted peatlands*).

However, the measured fluxes in both AC and BT do not reveal any correlation between WD and the observed levels of CO_2 and CH_4 (*Sections 5.2.2 Automatic chambers and 5.2.3 Bubble traps*). This lack of correlation may be attributed to the narrow range of WD (62 to 98 cm) measured at Mårumhus Pond, where the eight AC and BT were situated, making it challenging to detect variations.

Moreover, the literature review fails to establish a definitive relation between WD and the corresponding fluxes of CO_2 and CH_4 in shallow waterbodies. Although reported fluxes generally diminish at WDs deeper than 2 m, the limited amount of data in this range raises uncertainties about this pattern.

In a study by Wik et al. (2013), CH_4 emissions through ebullition are found to be nearly two to four times higher in shallow zones (0–2 m) compared to deeper zones (2–4 m and 4–7 m). Similarly, Bastviken et al. (2004) reports ebullition in 25–80% of AC at WDs no deeper than 4 m. These findings suggest that there is a larger potential for greater GHG emissions from shallow ponds compared to deeper lakes. However, indicating no influence of WD on GHG emissions in shallow waterbodies.

6.2.2 The influence of waterbody and ecosystem type on CO_2 and CH_4

The highest reported CO_2 fluxes based on the literature review are evident in restored peatlands, exhibiting a mean CO_2 flux of 29,533 mg CO_2 m^{-2} d^{-1}, while ponds exhibit a mean CO_2 flux of 4,507 mg CO_2 m^{-2} d^{-1}. Additionally, the mean measured fluxes by AC are 6,912 mg CO_2 m^{-2} d^{-1}. Both ecosystems are expected to contain lots of organic matter in the sediment. Comparing the emissions from restored peatland to ponds, the difference is expected to arise from their distinct water levels. Specifically, restored peatlands have a WTD below the ground surface, whereas ponds feature an open water surface. This distinction results in peat being more available for oxidization by O_2 in restored peatlands than in ponds (*Section 3.2.1 CO_2 balance in drained peatlands*).

Following up on the CH_4 ebullition fluxes reported in the literature review, artificial ponds exhibit the highest mean fluxes at 229 mg CH_4 m^{-2} d^{-1}, surpassing lakes and restored peatlands having 42 and 57 mg CH_4 m^{-2} d^{-1}, respectively. The distinctions observed between ponds and lakes and restored peatlands are caused by different factors. Lakes are assumed to contain less organic matter on the lakebed, leading to less decomposition and thereby fewer CH_4 emissions. In contrast, restored peatlands characterized by WTDs below ground surface, create an environment that is less anoxic than ponds, leading to lower CH_4 emissions.

6.2.3 The influence of shading on CO_2 and CH_4 fluxes

Shading of the pond caused by the surrounding vegetation is not included in this thesis, though it is assumed to contributed to variations in the measured emissions between the different AC and BT. The temperatures just above the water surface and in the top layer of the water column are fluctuating in a daily pattern related to the movement of the sun (*Graph 6*). Based on this, it is reasonable to assume that shading may influence temperatures within the water column. Research indicates that increased temperatures can increase the production of both CO_2 and CH_4 in the sediment (*3.3 Climate effect of rewetted peatlands*). Consequently, AC and BT located in shaded areas of the pond may exhibit lower CO_2 and CH_4 fluxes compared to those exposed to full sunlight.

Although the field study does not quantify the impact of shading, it has been observed that the canopies of the surrounding beech stand cast shade on the pond. To account for the potential influence of shading on temperatures in the water column, it could have been beneficial to install temperature loggers at each AC and BT. This will enable a more precise adjustment for the effect of shading on the measured temperatures and subsequently on the recorded CO_2 and CH_4 fluxes.

6.2.4 The influence of point sources on CH_4 fluxes

CH_4 emissions through ebullition are highly sporadic and challenging to quantify (*Sections 3.1.2 CH_4 balance in natural peatlands, and 6.5 How much does ebullition contribute to the total CH_4 flux?*). This sporadicity may contribute to the variation in measured CH_4 levels across ACs and BTs. However, the measured fluxes reveal a consistent pattern where specific ACs, and BTs consistently record the highest CH_4 fluxes (*Graph 12 and 15*).

The constant high measurements in certain ACs and BTs suggest a potential connection to what is known as point sources, as discussed by Wik et al. (2013). Point sources refer to the release of CH_4 through ebullition at fixed locations within the lake, produced deeper in the sediment (Walter et al., 2007). However, it shall be noted that point sources generally have been linked to lakes with deeper gas-rich structures or carbon-rich deposits (Wik et al., 2013). But maybe there are characteristics in Mårumhus Pond that can explain the release of CH_4 through point sources, and as such the spatial variation in fluxes. This phenomenon can explain why particular ACs and BTs consistently capture the largest fluxes throughout the study period, highlighting the importance of considering localized CH_4 production mechanisms.

6.2.5 Sub-conclusion

Answering sub-question *2. How do CO_2 and CH_4 fluxes relate to spatial factors such as water depth, waterbody, and ecosystem type?* It is evident that the variations in CO_2 and CH_4 fluxes may not be directly related to fluctuations in WD for shallow waterbodies. Other influential factors, including waterbody and ecosystem type, shading and its effects on temperatures at distinct locations within the pond, as well as the presence of point source contributing to CH_4 fluxes through ebullition, may instead be accountable for the spatial heterogeneity observed in CO_2 and CH_4 emissions.

6.3 How do CO_2 and CH_4 fluxes relate to environmental factors?

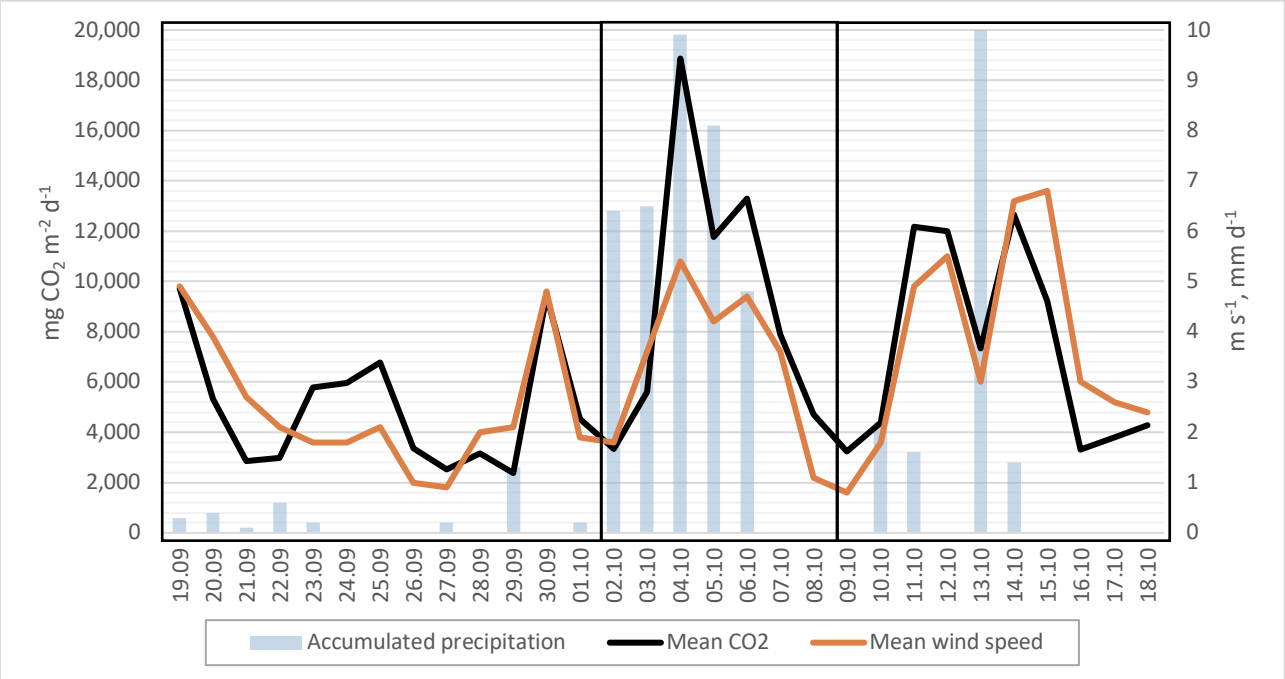
As mentioned, the weather conditions during the study period can be categorized into three distinct phases. Specifically, period (1) is identified as a stratified period, period (2) as a turnover period, and period (3) as a mixing period (*Section 5.2.1 Environmental parameters*). Upon comparing these distinct periods with the measured fluxes during the field study, it is obvious that fluctuations in both CO_2 and CH_4 fluxes can be explained by the occurrence of weather events and the dynamics of thermal stratification.

6.3.1 CO_2 response to wind speed and precipitation

According to Spafford and Risk (2018), increase in wind speed enhances CO_2 exchange across the lake-atmosphere interface, particularly in shallow lakes. The measured CO_2 flux and wind speed follow the same pattern, demonstrating a clear response for CO_2 fluxes with changing wind speeds (*Graph 16*).

Four distinct peaks in wind speed are identified, each corresponding to immediate responses in the measured CO_2 fluxes. The peak spanning from the 2nd to the 6th of October, aligning with the turnover period (2), while the peaks occurring on the 11th–12th and the 14th–15th of October, corresponding to the mixing period (3) (*Graph 16*).

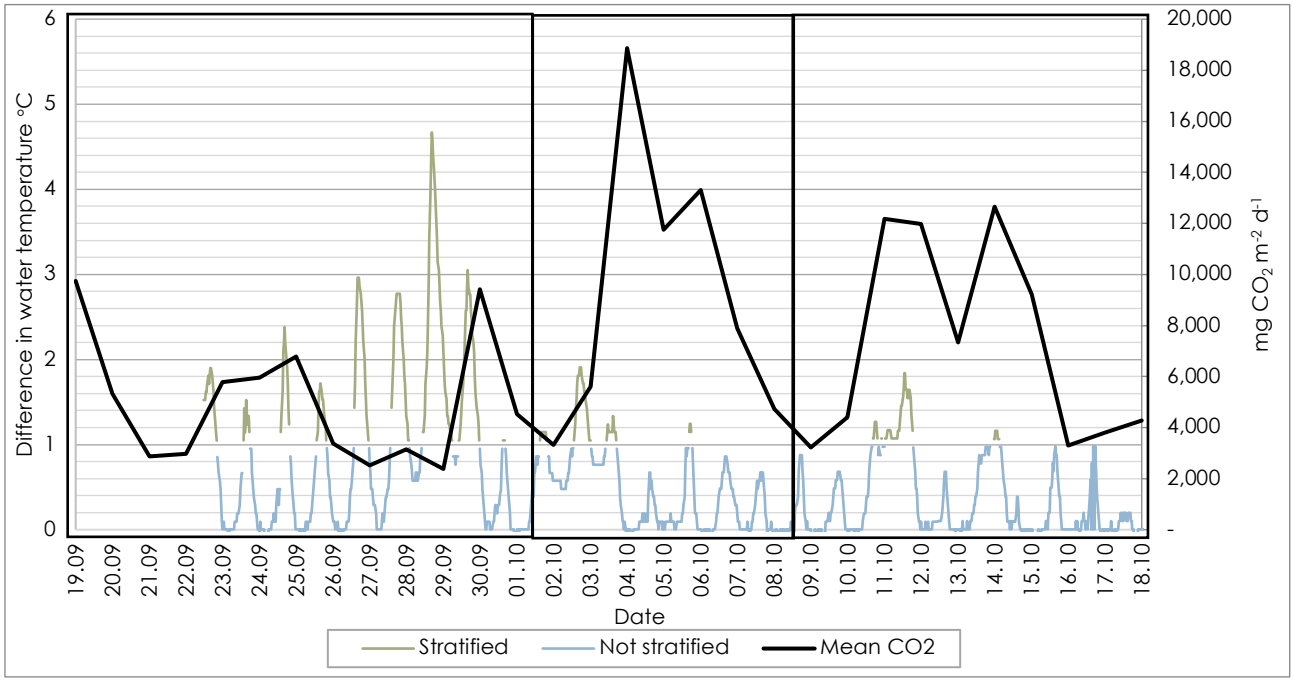
Additionally, episodic releases of CO_2 are associated with heavy rain events that activate organic substrates in the sediment (Spafford & Risk, 2018). The relation between CO_2 and precipitation is somewhat less distinct compared to wind speed (*Graph 16*). This becomes particularly evident during the mixing period (3), spanning from the 12th to the 15th of October, where a peak in wind speed and CO_2 is followed by a peak in precipitation. However, concurrently, there is a decline in both wind speed and CO_2 flux.



Graph 16: development of CO_2 fluxes, wind speed, and precipitation during the study period. X-axis: time in days. 1. Y-axis: CO_2 fluxes in mg CO_2 m^{-2} d^{-1}. 2. Y-axis: wind speed in m s^{-1} and accumulated precipitation in mm d^{-1}. The three black boxes indicate the three different weather periods: (1) calm weather, (2) change in weather, and (3) windy and rainy weather.

6.3.2 CO_2 response to fall turnover

Examining the influence of thermal stratification on CO_2 dynamics it becomes obvious that CO_2 fluxes increase when stratification breaks down during the turnover period (2) and mixing period (3) (*Graph 17*). The disruption of stratification is closely connected to increased wind speeds and precipitation, as these factors influence the mixing of the water column, resulting in uniform temperatures and O_2 levels throughout the water column (*further elaborated in Section 6.3.5 CH_4 response to fall turnover*). This can explain the increase in CO_2 levels, as CO_2 stored in the lower layers of the water column during stratification is rapidly released (Riera et al., 1999), and additionally, O_2 becomes available for the oxidation of CH_4 into CO_2.



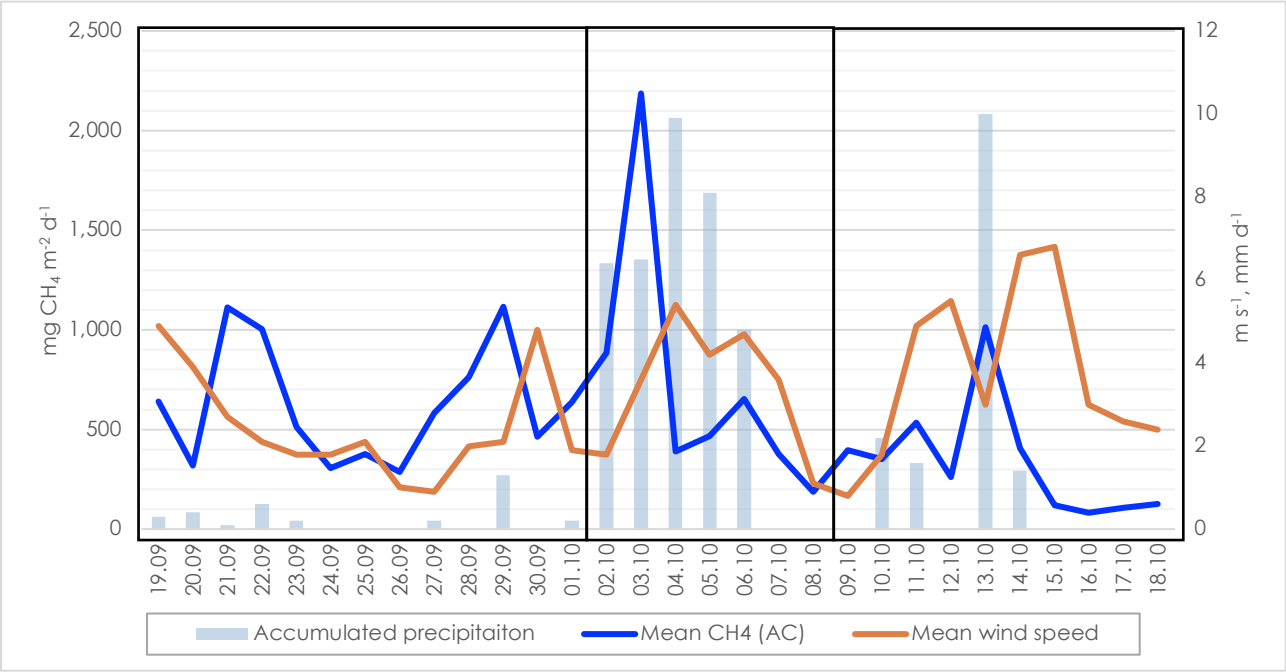
Graph 17: development of CO_2 fluxes and stratification during the study period. X-axis: time in days. 1. Y-axis: Difference in water temperature in °C. Y-axis: CO_2 fluxes in mg CO_2 m^{-2} d^{-1}. 2. The three black boxes indicate the three different stratification periods: (1) stratified period, (2) turnover period, and (3) mixing period.

6.3.3 CH_4 response to wind speed and precipitation

The trend in CH_4 flux development does not exhibit as clear a response pattern to weather events as in the case of CO_2.

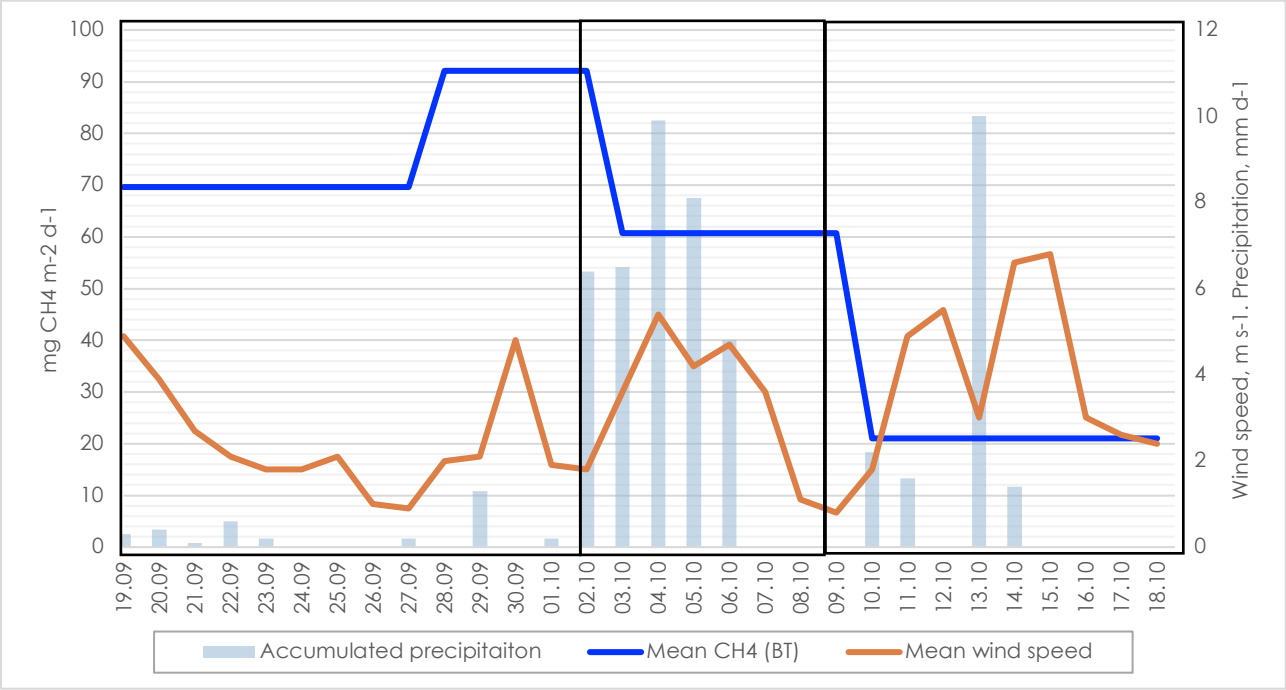
Short-term increases in wind speed can result in brief increments in CH_4 fluxes by inducing turbulence in the sediment (*Section 3.3.2 CH_4 balance in rewetted peatlands*). Conversely, long-term increases in wind speed are associated with reduced CH_4 fluxes, as continuous mixing of the water column elevates the O_2 levels. Additionally, precipitation may lead to decreasing CH_4 fluxes due to the dilution of carbon.

The CH_4 flux curve displays four notable peaks on the 21st and 29th of September, as well as the 3rd and 13th of October (*Graph 18*). The third and fourth peak appear to correspond to periods characterized by increasing wind speed and precipitation, such as the turnover period (2) and mixing period (3). However, the observed peaks in measured CH_4 exhibit a momentary nature, as they tend to revert to the original level of CH_4 fluxes within a day or two. This pattern can indicate that the initial turbulence in the water column enhances CH_4 fluxes while continuous activity has the opposite effect (*Graph 18*).



Graph 18: development of CH_4 fluxes, wind speed, and precipitation during the study period. X-axis: time in days. 1. Y-axis: CH_4 fluxes in mg CH_4 m^{-2} d^{-1}. 2. Y-axis: wind speed in m s^{-1} and accumulated precipitation in mm d^{-1}. The three black boxes indicate the three different weather periods: (1) calm weather, (2) change in weather, and (3) windy and rainy weather.

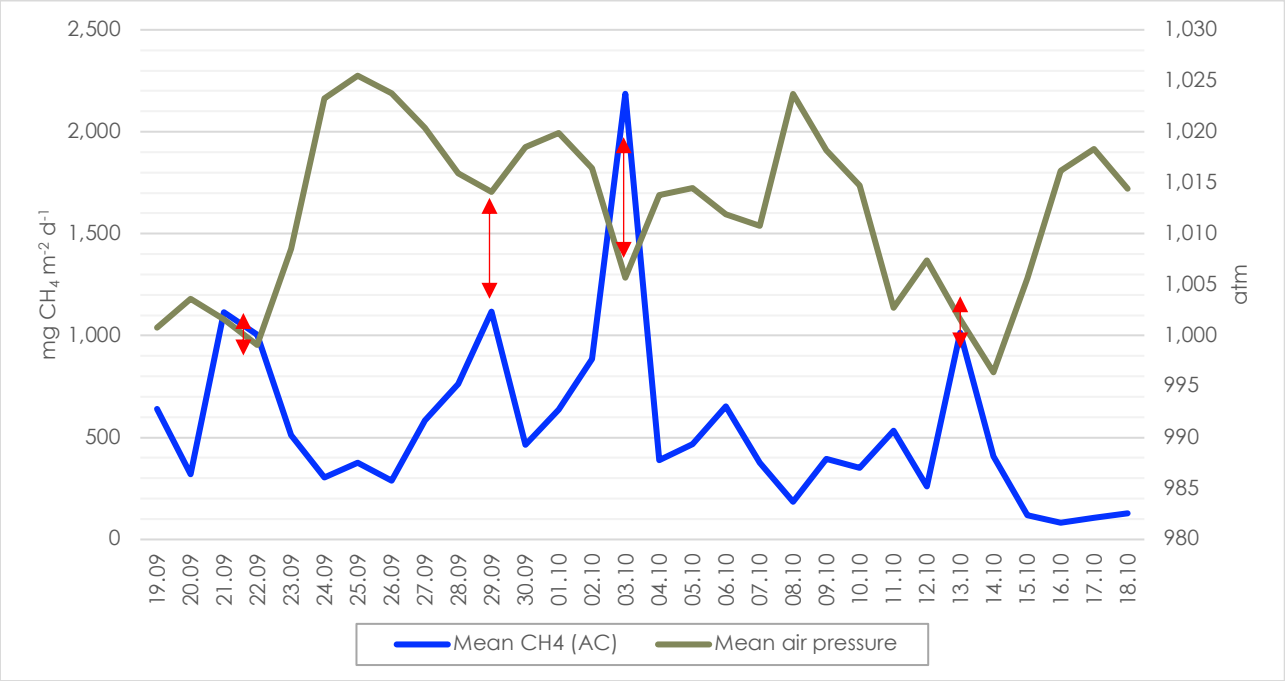
The measured CH_4 fluxes in the BT are declining through the last two sampling periods from the 3rd to the 18th of October corresponding to the turnover period (2) and mixing period (3) and thereby seems to respond to the continues increase in wind speed and precipitation (*Graph 19*).



Graph 19: development of CH_4 fluxes, wind speed, and precipitation during the study period. X-axis: time in days. 1. Y-axis: CH_4 fluxes in mg CH_4 m^{-2} d^{-1}. 2. Y-axis: wind speed in m s^{-1} and accumulated precipitation in mm d^{-1}. The three black boxes indicate the three different weather periods: (1) calm weather, (2) change in weather, and (3) windy and rainy weather.

6.3.4 CH_4 response to air pressure

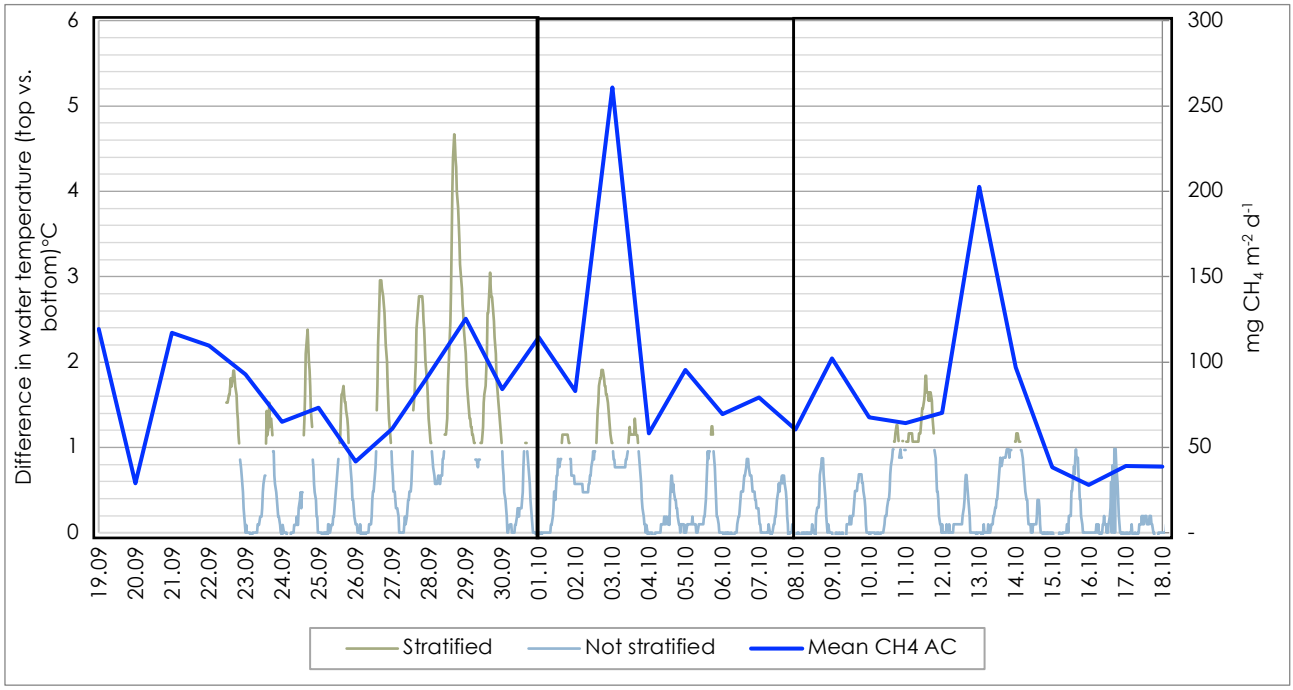
CH_4 emissions through ebullition have been linked to changes in air pressure (*Section 3.1.2 CH_4 balance in natural peatlands*). Furthermore, Wik et al., 2013 found that steep drops in atmospheric pressure leads to days with sudden increases in ebullition events. Examining the correlation between air pressure and CH_4 fluxes indicates an inverted relation, showing a pattern of simultaneous drops in air pressure and peaks in CH_4 (*Graph 20*).



Graph 20: development of CH_4 fluxes and air pressure. X-axis: time in days. 1. Y-axis: CH_4 fluxes in mg CH_4 m^{-2} d^{-1}. 2. Y-axis: air pressure in atm[1]. The three black boxes indicate the three different weather periods: (1) calm weather, (2) change in weather, and (3) windy and rainy weather. The red arrows indicate the drop in air pressure and simultaneous increase in CH_4 fluxes.

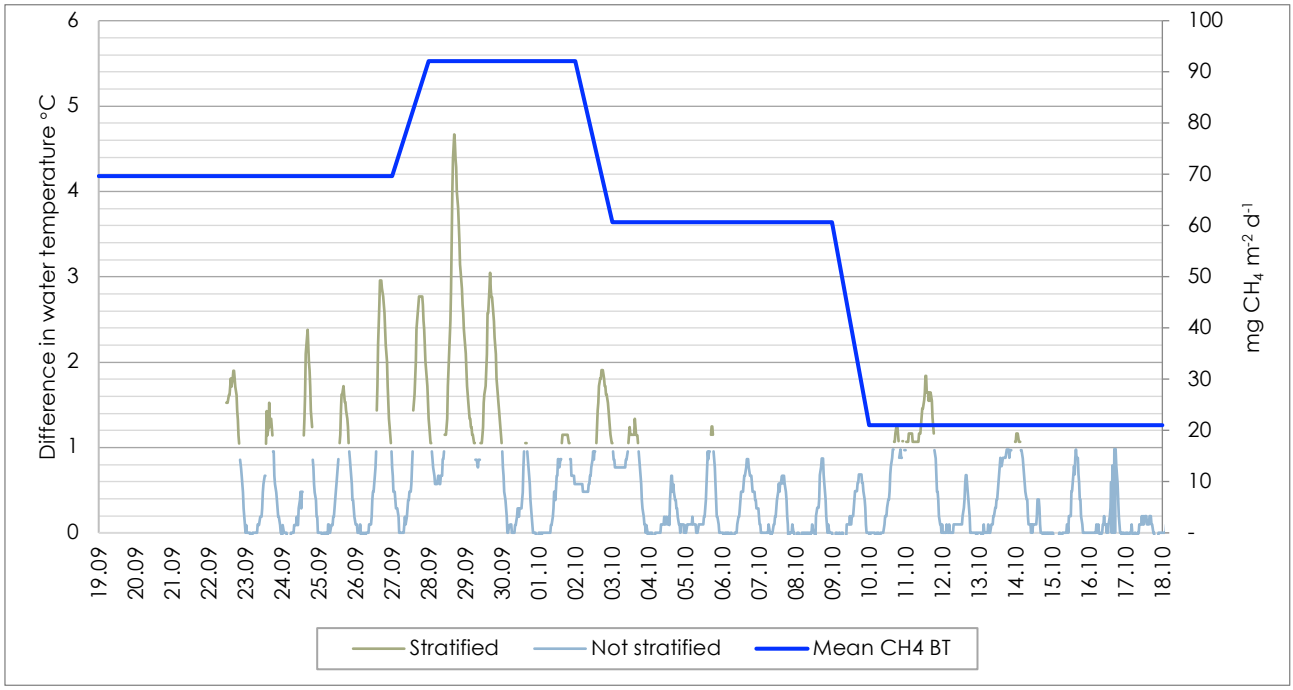
6.3.5 CH_4 response to fall turnover

Analyzing the response of CH_4 to thermal stratification suggests a potential correlation between the breakdown of thermal stratification and momentary increases in CH_4 flux (*Graph 21*). The two peaks in CH_4 flux, on the 3rd and the 13th of October, occur shortly after periods of thermal stratification. Therefore, the peaks in CH_4 fluxes indicate a response on a fall turnover event (*Section 3.3.2 CH_4 balance in rewetted peatlands*) characterized by a temporary increase in diffusive fluxes of CH_4 after a breakdown in stratification. The increase in CH_4 is only temporary, as continues breakdown in stratification increases the O_2 concentration, leading to CH_4 oxidation.



Graph 21: development in CH_4 fluxes and stratification during the study period. X-axis: time in days. 1. Y-axis: Difference in water temperature in °C. Y-axis: CO_2 fluxes in mg CH_4 m^{-2} d^{-1}. 2. The three black boxes indicate the three different stratification periods: (1) stratified period, (2) turnover period, and (3) mixing period.

Examining the sampled CH_4 from the BT, the fluxes are generally declining from the 3rd until the 18th of October, which can be related to a general decrease in CH_4 as O_2 levels increase throughout the water column with increased wind speeds and breakdown in stratification (*Graph 22*).



Graph 22: development in CH_4 fluxes and stratification during the study period. X-axis: time in days. 1. Y-axis: Difference in water temperature in °C. Y-axis: CO_2 fluxes in mg CH_4 m^{-2} d^{-1}. 2. The three black boxes indicate the three different stratification periods: (1) stratified period, (2) turnover period, and (3) mixing period.

The decrease of CH_4 fluxes in both AC and BT might also be related to the general decrease in temperatures throughout the study period (*Section 5.2.1 Environmental parameters*). The decomposition of organic matter is highly regulated by temperature in the sediment, where colder temperatures diminish the decomposition rate and subsequently the production of CH_4 as well as CO_2 (DelSontro et al., 2016; Wik et al., 2016).

6.3.6 Sub-conclusion

When answering sub-question *3. How do CO_2 and CH_4 fluxes relate to environmental factors, such as wind speed, precipitation, air pressure, and temperature?* It is evident that CO_2 and CH_4 fluxes and weather events are interdependent. Notably, changes in wind speed closely correlate with variations in CO_2 fluxes, whereas drops in air pressure appear to have a stronger association with CH_4 fluxes. Signs of a fall turnover event are apparent when examining the thermal stratification of the pond. Both CO_2 and CH_4 exhibit responses to the breakdown in stratification, yet with contrasting effects; CO_2 fluxes increase with the breakdown in stratification, while CH_4 fluxes increase briefly and then seem to decrease. This correlation aligns closely with the availability of O_2 in the water column, which enhances the production of CO_2 through the oxidation of CH_4, among other processes.

6.4 How do CO_2 and CH_4 fluxes evolve on a daily pattern?

6.4.1 CO_2 daily pattern

The daily fluctuation in CO_2 fluxes across all ACs reveal a rise during nighttime, reaching its peak between 07:00-10:00, and a subsequent decrease during the afternoon, with the lowest flux observed from 16:00-20:00 (*Graph 11*).

Spafford and Risk (2018) have documented a daily pattern of CO_2 fluxes occurring from late night to early morning. This release is associated with increased respiration during nighttime (darkness) and improved photosynthesis during daytime (presence of sunlight) (Spafford & Risk, 2018). Light data from Mårumhus Pond indicates that the sun rises at 7:00, with sunlight first reaching the lower water layers around 9:00 (*Appendix 3*). This observation aligns with the measured flux pattern, where the CO_2 flux increases during the night and peaks between 07:00-10:00. As such, the highest measured fluxes are present in the early morning when the sun is rising and not the darkest time of day, correlating to respiration. However, the pattern of daily CO_2 fluxes indicates a build-up in CO_2 production during the night being emitted in the last hours of darkness (*Graph 11*).

Additionally, CO_2 flux increasing during nighttime can be due to stratification breakdown, where CO_2 trapped in the anoxic layer during the day is released suddenly during the night (Riera et al., 1999; Spafford & Risk, 2018). Mårumhus Pond generally exhibits stratification in the afternoon/evening, which may contribute to the increase in CO_2 during nighttime/early morning together with the breakdown in stratification (*Section 5.2.1 Environmental parameters*).

Consequently, accounting for the daily variation in CO_2 fluxes is crucial, as solely measuring CO_2 fluxes during the daytime would lead to an underestimation of the lake-atmosphere CO_2 exchange.

6.4.2 CH_4 daily pattern

The daily fluxes of CH_4 do not show as clearly of a pattern as for CO_2 but indicate that the largest CH_4 fluxes are emitted between 05:00 – 17:00 (*Graph 11*). This is similar to the findings of Bastviken et al. (2004) presenting a substantial daily variation in CH_4 fluxes, reporting 9–158% greater emissions during daytime.

In contrast, Hoffmann et al. (2017) measure CH_4 emissions in a flooded former peatland in northeastern Germany, with maximum fluxes between 00.00 and 09.00 in September. The study suggests daily thermal stratification to be part of the reason for seeing this pattern, as the warmer upper layer of the water column during daytime prevents the diffusion of CH_4 through the water column (Hoffmann et al., 2017).

The daily thermal stratification at Mårumhus Pond indicates a build-up in stratification in the afternoon until evening and a breakdown during the night. As such, thermal stratification does not seem to explain the indication of a daily pattern in CH_4 fluxes.

6.4.3 Sub-conclusion

In conclusion to sub-question *4. How do CO_2 and CH_4 fluxes evolve on a daily pattern?* CO_2 fluxes proved a daily pattern emitting the highest fluxes during nighttime (increased respiration) and the lowest in the afternoon (increased photosynthesis). In contrast, CH_4 did not show a clear daily pattern, even though there are some indications of emitting the largest fluxes during daytime (between 05:00 – 17:00).

6.5 How much does ebullition contribute to the total CH_4 flux?

Although the process of CH_4 emission as ebullition is well documented in the literature (Bastviken et al., 2004; Wik et al., 2013) and various methods for measuring bubble events are recognized (Maeck et al., 2014; Petersen et al., 2023; Thanh Duc et al., 2020; Wik et al., 2013), ebullition is rarely quantified (Cooper et al., 2014; Taoka et al., 2020; Wik, 2016) and sometimes even excluded from the data, since they are 'disturbing' in the data processing (Bohdálková et al., 2013; Christiansen et al., 2012; Evans et al., 2021; Peacock et al., 2021b; Rigney et al., 2018). According to Bastviken et al. (2011) ebullition events are most likely underestimated in the existing research, and Peacock et al. (2017) stresses the need for more measurements on ebullition events.

Several studies estimate that CH_4 fluxes from ebullition events account for a larger share of the total CH_4 emission than diffusive fluxes (Bastviken et al., 2011; Taoka et al., 2020). Bastviken et al. (2004;2011) find that ebullition from lakes and ponds contributes to 50% or sometimes even >90% of the total CH_4 flux. A recent Danish study by Petersen et al. (2023) reveals that CH_4 emissions from

ebullition are 100 times higher than diffusive fluxes at a restored wetland on Fyn measured during summer. Additionally, the literature review carried out in this thesis proves that mean CH_4 ebullition fluxes are 290% higher than mean CH_4 diffusive fluxes.

It was not possible to differentiate between ebullitive and diffusive fluxes in the measured fluxes from the field study, even though it is plausible to assume that CH_4 ebullition events were captured during the field study (*Section 4.2.3 Data processing*).

If the CH_4 fluxes measured with the AC and BT primarily reflect CH_4 ebullition, then the fluxes measured using BT are relatively low, with mean fluxes of 61 ± 34 mg CH_4 m^{-2} d^{-1} compared to mean ebullition fluxes of 186 ± 246 mg CH_4 m^{-2} d^{-1} in the literature review.

Considering the study by Peacock et al. (2021b), reporting the highest CH_4 ebullition fluxes included in the literature review, with a mean of 1,061 mg CH_4 m^{-2} d^{-1} ranging from 3-3,880 mg CH_4 m^{-2} d^{-1}, it is evident that there is considerable variation in measured fluxes within this study alone. The same variation in fluxes is observed during the field study with the AC-setup reporting a mean of 557 mg CH_4 m^{-2} d^{-1}, ranging between 0.5–7,789 mg CH_4 m^{-2} d^{-1}. As such, the high flux measures from Peacock et al. (2021b) are within the same range of measured fluxes in the field study and should not be considered outliers. This emphasizes the need for more studies on ebullitive fluxes to conclude whether these high fluxes are actually outliers or expected outcomes.

6.5.1 Sub-conclusion

To answer sub-question *5. How much does ebullition contribute to the total CH_4 flux?* It becomes evident that it is not possible to differentiate between diffusive and ebullitive fluxes in the field study and it is therefore not possible to allocate a specific share of the total CH_4 fluxes to ebullition. Despite this, CH_4 fluxes measured by AC showed very high values, up to 7,789 mg CH_4 m^{-2} d^{-1}, making it plausible to assume that CH_4 ebullition events were captured during the field study. Based on results from the literature review, 75% of the total CH_4 emission is assigned to CH_4 ebullition.

6.6 Discussion of methods

6.6.1 Literature review

During the literature review, adjustments were made to the keywords, as the literature found in Phase 1a unintentionally led to the inclusion of studies assessing flooded areas in other climatic zones than the temperate and boreal zones (like rice paddies in subtropical climatic conditions) and very deep and large extensive artificial lakes (like hydropower reservoirs). Additionally, the search unintentionally excluded studies exclusively assessing diffusive CH_4 fluxes, and thereby only included studies concerning CH_4 ebullition fluxes.

The scope of the thesis evolved during the data processing of the field study results, particularly as it became challenging to distinguish between ebullition and diffusive CH_4 fluxes (*Section 4.2.3.1*

Automatic Chambers). To address this limitation, modifications were introduced in Phase 1b of the preliminary search in the literature review. This involved incorporating keywords associated with shallow lake/pond formation and using the general term "methane" instead of specific terms like CH_4 ebullition or bubbles. Additionally, CO_2 was not included in the keywords during the preliminary search, but only encompassed if studies reported on this along with CH_4 fluxes. This only led to the inclusion of few studies reporting on CO_2 fluxes. As CO_2 fluxes constitute a relevant share of the total GHG flux, this should have been included in the preliminary search.

Conducting the perfect literature search is a demanding task as the number of qualified hits is determined by the choice of wording. For example, a large range of words associated with different wetlands, or terms like 'restored' or 'rewetted' which are not frequently used, make it challenging to conduct specific searches to capture all relevant articles. This theoretical source of error could lead to a risk of overlooking central reports in the literature review.

6.6.2 Automatic chamber setup

The installation of the AC-setup in the field was a manageable process, lasting two days. Additionally, extracting data during the study period was straightforward, facilitated by the LICOR8150 storing data continuously. Though, it should be noted that much, expensive equipment, and a power source is needed, which makes it challenging to replicate to other sites.

It was considered to leave a field desktop and a router in the aluminum boxes with the instrumentation during the study period, allowing real-time monitoring of the AC measurements. While this would enhance accessibility and enable early detection of AC turnover, the true gain in the context of frequent site visits, necessary for tapping the BT, was deemed relatively small.

One significant drawback of the AC-setup is its dependency on a power outlet or a large battery for continuous measurements throughout the study period. This requirement is considered challenging in locations where rewetting projects are often situated, such as rural landscapes or forest areas like for Mårumhus Pond.

Another weakness of the AC-setup is the difficulty or near impossibility of separating CH_4 fluxes into ebullition and diffusion using the current method in R-studio. The 'flux-pattern' of ebullition and diffusive fluxes cannot be divided into two distinct groups when visually assessing their patterns. Distinguishing diffusive fluxes from steady ebullition events, particularly microbubble events, is challenging, making manual examination of numerous graphs (1,355 for the entire study period) time-consuming and uncertain.

Furthermore, the ACs continuous presence at the water surface entails challenges in purging sufficiently as gases entering the AC during purging resulted in highly variable start concentrations. Therefore, manual interpretation of all graphs corresponding to each measurement period is required. Even though a strategy for interpreting graphs was made beforehand, the process remained time-

consuming and somehow uncertain. Complete purging of the AC was not achievable, leading to an expectation of underestimated fluxes, as emissions occurring at the start of the measurement period were excluded (*Sections 4.2.2.2 and 4.2.3.1 in Section 4.2 Field study*).

6.6.3 Bubble trap setup

The installation of BT in the field was much simpler than that of the AC. Fastening them to the anchor and placing them in the pond transect required minimal effort. Extracting data during the study period was straightforward, although more time-consuming compared to the AC due to the weekly tapping requirement. Nonetheless, the method is easily replicable at other sites.

CO_2 and CH_4 fluxes measured by BT were notably low compared to CO_2 and CH_4 fluxes measured by AC (*Section 5.3 Summary*). In this context, four central uncertainties are identified; (1) CH_4 fluxes may undergo oxidation by methanotrophs within the BT, given that atmospheric air is present. However, the risk of oxidation is considered smaller as the methane is in gaseous form, making it less available for oxidation compared to dissolved methane (Baker-Blocker et al., 1997), (2) the risk of CO_2 uptake within the BT from plants and algae which can be enhanced by heat and sun coming through the thin plastic, in contrast, the ACs are coated with foil, reflecting the sun away from the ACs, (3) that BT may be somewhat leaky, and (4) the fact that some fluxes might diffuse back into the pond to reach equilibrium in gas concentrations between the water and air.

The risk of gasses diffusing back into the water is less pronounced for AC measurements, given their shorter 10-minute measurement periods. Shorter intervals between tapping the BT may reduce the risk of CO_2 and CH_4 diffusion back into the water, although no literature has reported on this according to our knowledge. A recent study by Petersen et al. (2023) used monthly tapping intervals for BT, reporting commonly measured CH_4 ebullition fluxes ranging from 270-2,360 mg CH_4 m^{-2} d^{-1}. In contrast, Wik et al. (2013) reported daily samplings of BT, with corresponding fluxes ranging from 0 – 1,122 mg CH_4 m^{-2} d^{-1}. This suggests a potential need for further testing of the method, especially in conjunction with other methods to verify the data.

6.6.4 Sub-conclusion

In summary, BT measured considerably lower CO_2 and CH_4 fluxes during the field study compared to the AC-setup. The CO_2 fluxes measured by AC align with the range identified in the literature review. On the other hand, CH_4 fluxes measured by BT are closer to the range of fluxes found in the literature review. It is crucial to note that this does not imply that AC measures incorrectly high CH_4 fluxes; rather, it reflects a need for further research on CO_2 and CH_4 fluxes from ponds in rewetted peatlands and especially the validation of the AC and BT-setup.

7 Conclusion

This thesis explored the field of peatland rewetting with specific focus on shallow ponds and its implications in terms of CO_2 and CH_4 emissions. The work indicates that rewetting of formerly drained peatlands can have large implications for GHG emissions relevant in the field of nature restoration and climate change mitigation projects in the form of rewetting, that have gained increased political awareness.

Answering the research question of the thesis: *What is the magnitude of CO_2 and CH_4 emissions for ponds in rewetted landscapes?* The reported and measured emissions presented in this thesis are highly variable suggesting that emissions of CO_2 and CH_4 are highly affected by spatiotemporal attributes as well as methodology.

The total mean fluxes found in this thesis are for CO_2: 10,766 ± 12,440 mg CO_2 m^{-2} d^{-1} and 6,912 ± 5,905 mg CO_2 m^{-2} d^{-1} found in the literature review and AC measurements, respectively. For CH_4 the fluxes are 128 ±187 mg CH_4 m^{-2} d^{-1}, 557 ±714 mg CH_4 m^{-2} d^{-1}, and 61 ± 34 mg CH_4 m^{-2} d^{-1} from the literature review, AC and BT measurements, respectively.
In CO_2-eq the total mean flux from the literature review is 14,408 mg CO_2-eq m^{-2} d^{-1} and 22,510 mg CO_2-eq m^{-2} d^{-1} from AC measurements (with a GWP of 28 for CH_4). Comparing total GHG fluxes across the literature review and the field study they present similar shares with CO_2 fluxes contributing to 25% and 31%, and CH_4 accounting for 75% and 69%, respectively.

This thesis has addressed spatial attributes such as WD/WTD, waterbody and ecosystem types, and environmental parameters such as wind speed, precipitation, air pressure, and water and air temperature. The results indicate that WD does not have an effect on CO_2 or CH_4 fluxes when assessing shallow depths, no deeper than 3.2 m. In contrast, ecosystems with WTDs below ground level, indicate significantly higher CO_2 fluxes than for ecosystems with an open water surface above ground level.
AC and BT measurements indicate that spatial variability is present between the eight AC and BT measurements, resulting in the same AC and BT continuously measuring the highest fluxes. This variability cannot be explained by the studied attributes but might be explained by other factors such as shade and ebullitive point sources.
Fluxes reported in the literature review and measured in the field study indicate that ponds in rewetted peatlands are large contributors of both CO_2 and CH_4 emissions which is attributed to the carbon-rich soils.
Weather events seem to be the main factor contributing to temporal increases in fluxes. CO_2 fluxes highly correlate with increased wind speed and breakdown in stratification. CH_4 emissions show an inverted relationship to drops in air pressure, while breakdown in stratification leads to temporary peaks in CH_4 fluxes. Wind speed and precipitation are found to influence CH_4 fluxes less than CO_2.
Measurements from the AC-setup did not show any clear indication of a daily pattern, except for CO_2 showing slightly higher fluxes during night and early morning.

The three methods have provided a way for comparing fluxes. The review shows that CO_2 and CH_4 in predominantly wet ecosystems are rarely quantified and highly variable. The AC and BT methods indicate large variations in measured CO_2 and CH_4 fluxes even in the same field study. In conclusion, more studies are needed on the subject, both in terms of reporting fluxes on CO_2 and CH_4 from ponds in rewetted peatlands and in terms of comparing methods for flux quantification.

8 Perspectives

- **Nature-Based Solutions:** Ponds and pondscapes have been suggested to provide substantial benefits for humans in terms of climate change mitigation, biodiversity conservation, and ensuring human well-being (Cuenca-Cambronero et al., 2023). Based on the fluxes reported from this thesis it must though be considered whether the creation of ponds on peat soils actually contribute positively to climate change mitigation.
- **Rewetting as a tool in climate mitigation:** This thesis has provided emissions of CO_2 and CH_4 from a pond in a rewetted peatland, suggesting large emissions when compared to other studies across different waterbody and ecosystem types. This could indicate that the creation of ponds on former drained peatlands should be avoided, but further research is needed. This study does not encompass the relation of measured and reported fluxes in terms of global GHG budgets and whether these emissions are extremes.

  Further, the study does not compare fluxes to those from the drained state of the ecosystem, as such it is not possible to state whether pond-creation on peat soils is better or worse compared to emissions related to the drained site.

  Additionally, the study has not addressed the possible uptake of carbon within the pond and as such the presented emissions do not present the net climate effect of such ecosystems.

  Complete knowledge about net emissions/uptake from drained and rewetted peatlands are important components, in the context of reaching Denmark's climate goals for 2030 and 2050.
- **Global warming potential:** When comparing CO_2 and CH_4 emissions imply the consideration of the lifetime of the gases. CH_4 is a potent GHG compared to CO_2 and as such increased emissions from rewetting should be considered. But CH_4 is also a short-lived GHG relative to CO_2.

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10 Appendix

All appendices are attached separately as Excel files, R-studio files or as PDF files.