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1 Introduction

1.1 Peat characteristics and formation

Peatland is an area covered with partially degraded plant material (peat), which is formed in areas with naturally high water table, that suppress microbial degradation. Water saturated soils are generally referred to as wetlands, but only peat covered areas are peatlands, hence peatlands are a type of wetlands. Wetlands cover up to 8% (Mitsch and Gosselink, 2007) of land surface globally, of which 3% (Joosten and Clarke, 2002) are peatlands. An area is defined as peatland if it is covered with material containing at least 30% organic matter of its dry mass. Another criteria to define a peatland is to measure thickness of the accumulated plant matter. If an area has at least 30 cm deep peat cover, it is classified as peatland. There are 266,65 km^2 (Petz, 1999) peatlands in Austria, covering about 0.3% (see Figure 1) of the territory. Greater part of it has been subjected to mining and land use change, therefore today only one-tenth (Joosten and Clarke, 2002) of Austrian peatlands are in pristine condition.

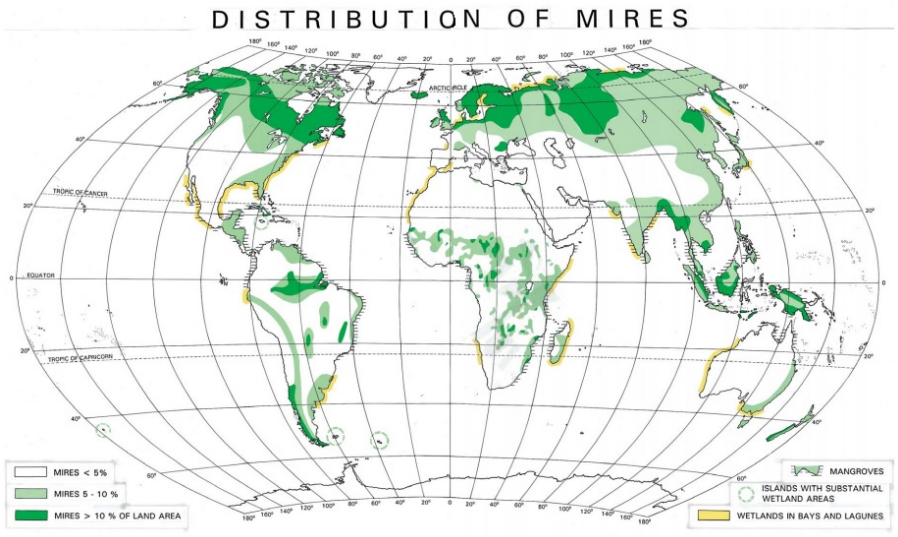

Figure 1: Global peatland distribution. White color represents areas, where peatlands are less than 5% of the total surface area. Green colored areas display locations with medium and high prevalence. From *Eino Lappalainen (ed.) "Global Peat Resources", International Peat Society Jyskä, 1996.*

Peat formation has been occurring throughout the history of Earth, however peat formed prior to Holocene has been transformed into other carbon-rich materials, such as lignin and coal (Martini et al., 2007). Lignin-rich peat is abundant in tropical regions, which have not been covered with ice during last glacial period in late-Pleistocene. In present day, peatlands are most abundant in polar and boreal regions in northern hemisphere.

Temperate and cold regions have high precipitation level, that secures constant and high water table which, in turn, eliminates oxygen. Anoxic conditions along with low air temperature, suppress plant material decomposition rate to such an extent, that vegetation litter begins to accumulate. Furthermore, glacial landforms are favorable settings for peat development. Consequently, peat development largely depends on two factors: climate and local geology.

Peatlands, most often, are classified by their genesis and nutrient content:

- fens are peatlands formed in geological depressions, such as poorly drained lowland meadows, shallow ponds and lakes. Fens are fed by nutrient rich groundwater, therefore they are minerotrophic. In terms of pH, fens are neutral to slightly acidic;
- bogs are elevated and rise above the surroundings, but can also be flat. Unlike fens, bogs receive water solely from precipitation. Precipitation does not contain high concentrations of nutrients, therefore bogs are nutrient poor (ombrotrophic) and has lower pH than fens. In addition, plant material is decomposed weaker than in fens.

Bogs and fens are often grouped together in a more general term mires. Moreover, some countries have additional peatland categories, such as, marshes, swamps (Warner and Rubec, 1997) and moors. However, peatlands most commonly are characterized by their nutritional status, thus most prevalent terms are bogs and fens.

Peatlands have micro-topography of empty cavities, called hollows, that normally are saturated with water and ridges made of living plant biomass, called hummocks, that are elevated above peatland and become saturated after heavy rainfall. Peatlands not only have spatial differences, but also vertical heterogeneity. Peatlands unlike mineral soils do not form distinctive horizons, therefore peatlands have been proposed to be differentiated depending on their hydrology. Peat, like any other soil can be either water saturated or unsaturated. This property is used to describe peatlands and is referred to as diplotelmic model (Ingram, 1978). Surface is covered with vegetation and underlying rooting zone, that upon death accumulates and forms fresh peat. This top layer, called acrotelm, has varying water saturation due to seasonal water table flux. Consequently, conditions are temporarily oxic or anoxic. Acrotelm, on average, extends down to half a meter depth. Below lies peat layer that is always saturated with water – catotelm. Persistent anoxic conditions are not favorable to microorganisms, therefore organic matter decomposition in deep peat is dominated by anoxic microorganisms, which decompose plat litter at slower rate than in acrotelm. Catotelm, depending on the age of peatland, can be several meters to several decameters deep (Clymo, 1984, Morris et al., 2011).

1.1.1 Distribution

Peatland formation occurs in areas where due to geologic and climatic properties dead plant biomass decomposition is inhibited and over long term organic matter is accumulated. Microorganisms and fungi, that decompose organic matter in biomass thrive under certain temperature, soil moisture, substrate availability and other environmental properties. Areas with high annual precipitation have high water table and low air filled pore space, hence oxygen availability is limited. Microorganisms participating organic matter decomposition require oxic conditions, therefore water saturated soils have reduced organic matter decomposition potential. Highest annual precipitation is observed in temperate and tropical climate zones (IPCC, 2014) suggesting that biomass decomposition in these areas is a slow process. In tropical regions high air temperature causes rapid evapotranspiration, therefore microbial activity is sufficient and organic matter is produced and degraded equally. On the contrary are temperate and polar regions, where air temperature is so low, that biomass production exceeds decomposition rate and favors peat formation, therefore most peatlands are found in cold climate regions. Globally majority of peatlands are situated in Eurasia and North, South America (see Table 1).

Table 1: Peatland area by region. After Joosten, Clarke, 2002. *Wise use of mires and peatlands.*

Region	**Area in km^2**
America (north, south & central)	2 050 746
Asia	1 523 287
Europe	617 492
Africa	58 534
Australia, New Zealand	8009

1.1.2 Vegetation

Peatlands as type of wetlands are seasonally or permanently saturated with water, reducing oxygen availability for roots, therefore only adapted plant species can successfully inhabit this ecosystem. Plant litter serves as parent material for peat formation influencing peat structure, physical and chemical properties. On the other hand, different types of peatlands have a characteristic botanic community, that in turn, determines peat composition.

Vegetation pattern of an area essentially depends on climatic and soil properties: temperature, water, nutrient availability, etc. Pristine mires, despite of high organic matter

content are less fertile than other ecosystems (Martini et al., 2007). Peatland surface is primarily covered with bryophytes and miscellaneous vascular plants. Nutrient poor bogs have a larger portion of moss, while fens, richer in nutrients, also supports vascular plant growth.

Moss (group *Bryophyta*, class *Sphagnopsida*, family *Sphagnaceae*, genus *Sphagnum*). *Sphagnum* mosses are non-vascular plants, which lack tissues that provide water and nutrient transport throughout the plant. These mosses have a dense central stem and branches covered with narrow leaves, topped with a star-shaped head (Smith and Smith, 2004). *Sphagnum spp.* color varies between yellow and dark brown, but most commonly it is yellow to light green or different shades of red and brown. Due to lack of specialized water transporting tissue, mosses absorb and store water in all cells, hence they are not dependent on soil moisture uptake by roots. Mosses utilize rainwater, therefore flourish in hydric habitats, where precipitation exceeds evaporation. *Sphagnum* has high cation exchange capacity, meaning they take up ions from water and release hydrogen (protons) back into soil solution. Cation exchange coupled with humic acid formation due to moss litter breakdown establishes pH as low as 4.0 (Haslam, 2004). *Sphagnum* species identification with a naked eye is difficult (see Figure 2), therefore various macroscopic (habitat, size, color, growth form) and microscopic (shape of individual cells) properties (Walker, 2015) must be investigated. *Sphangum* mosses are predominant plants in mires, covering the entire surface and creating a sort of moss carpet.

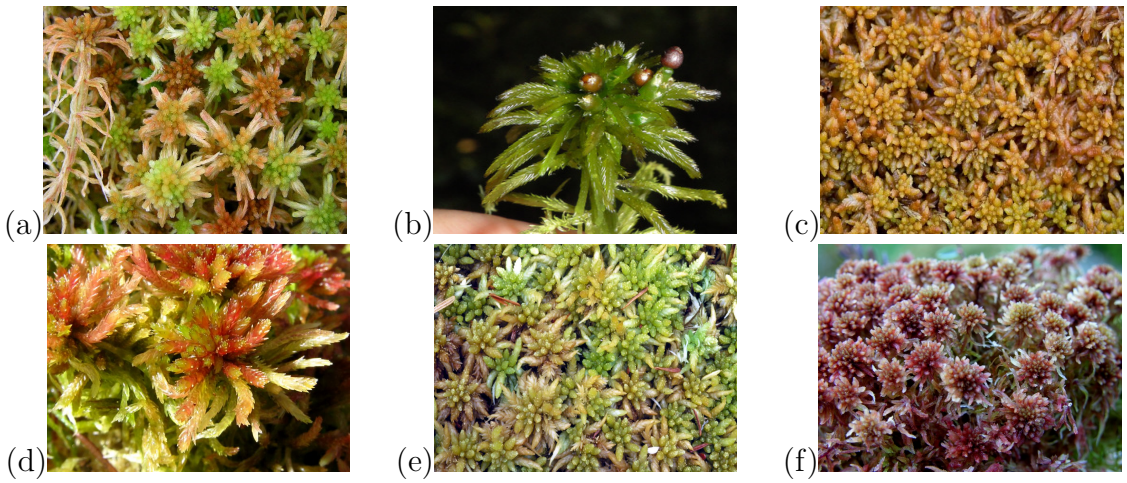

Figure 2: Main peat forming moss species. (a) *S. angustifolium.* (b) *S. cuspidatum.* (c) *S. fuscum.* (d) *S. magellanicum.* (e) *S. papillosum.* (f) *S. rubellum.*

***Ericaceae* family** (group *Angiosperms*, clade *Eudicots*, order *Ericales*). *Ericaceae* family is a group of flowering plants with 145 genera and more than a thousand species (Plant

List, 2010). *Ericaceae* family are perennial deciduous or evergreen woody shrubs and small trees. *Ericaceae* plant anatomy greatly varies depending on genus: stem can be self-supported or climbing, They are widespread in bogs due to preference of acidic soils. Most common plants in bogs are edible berries of *Vaccinium* family and heather (see Figure 3).

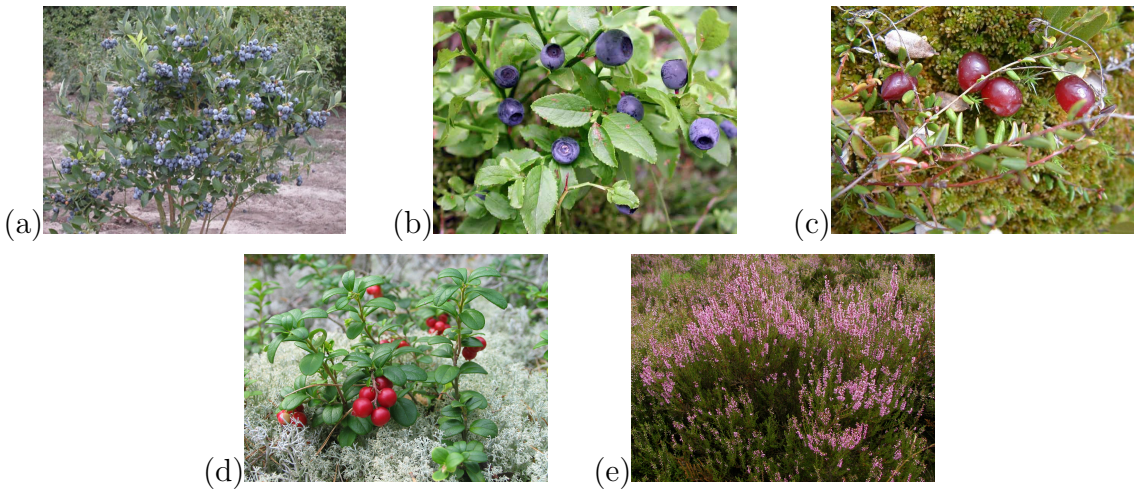

Figure 3: Common *Ericaceae* family species in peat bogs. (a) Blueberry, *Vaccinium corymbosum*. (b) Bilberry, *Vaccinium myrtillus*. (c) Cranberry, *Vaccinium oxycoccos*. (d) Lingonberry, *Vaccinium vitis-idaea*. (e) Common heather, *Calluna vulgaris*. Heather is a low-growing shrub with strongly fragrant pale purple colored flowers, that bloom during late summer. Main stem forms numerous upright branches with small, evergreen leaves. Prefers acid soils with high organic matter content, but low overall fertility (University of Connecticut, N/A).

Sedges (group *Angiosperms*, clade *Commelinids*, order *Poales*, family *Cyperaceae*). Sedges are flowering grasses growing on nutrient-poor soils with tens of genera. Sedges in peat bogs are represented by *Carex* and *Eriphorium* family species. *Carex*, commonly called simply 'sedge' are perennial grasses with sharp and thin blade-like leaves (Bugg et al., 2013). Common sedges in peatlands are bog sedge (*C. oligosperma*), mud sedge (*C. limosa*) and slender sedge (*C. lasiocarpa*). Another common bog sedge is cottongrass (genus *Eriophorum*). Cottongrass has similar anatomic properties as other sedges: blade-like leaves and, spikelet flowers, however its' head has hairy, dense top resembling a ball of cotton, hence common name cottongrass emerged. Peat bogs are inhabited by hare's-tail cottongrass *E. vaginatum* and common cottongrass *E. angustifolium*.

1.2 Peatland biogeochemistry

1.2.1 Carbon cycle

Carbon, along oxygen, nitrogen and hydrogen, is the main cellular constituent (Reece et al., 2011) and generates biomass of all life domains, including plants. When plants die their necromass is utilized by microorganisms in topsoil, which use carbon for metabolic processes. Nevertheless, the organic matter is not decomposed fully and certain fraction remains in the soil. Residual organic matter can stay in the soil for centuries, because microorganisms prefer to use freshly produced organic matter (Lützow et al., 2006). Soil organic matter is not only important source of energy for microorganisms, but also improves basic soil properties and agricultural yield, because carbon species can have other elements (sodium, potassium, phosphorous) incorporated into their structure, which are essential for plant growth.

Global carbon cycle is driven by processes that cause carbon exchange between atmosphere, lithosphere and hydrosphere. Plant litter, for the most part, is decomposed and returned to the atmosphere as carbon dioxide within few days. The remainder depicts long-term carbon repository, meaning that soil carbon storage represents the balance between carbon input and output. Soil carbon input main source is plant litter, which in topsoil is actively degraded by microorganisms and via respiration is released back into atmosphere as gaseous carbon species. Thereupon soil carbon can be further reduced by leaching into groundwater and erosion. Remaining carbon accumulates into large clusters together with other organic compound molecules, forming soil organic matter, which is most commonly defined as heterogeneous mixture of composed and partly composed plant matter (Kumada, 1987). Carbon cycle in soils is similar in all soil types, including organic soils (see Figure 4).

Organic matter decomposition occurs also in the subsurface and even deeper layers of water saturated peat, where there is little or no oxygen available. During OM decomposition, when larger molecules are split by enzymes into smaller ones, not all of them are consumed and part of it is transported into soil water, which transports and distributes them vertically. As these molecules are further decomposed, acetic acid is being formed, which is a preferred electron acceptor for anoxic organisms. As a result methane formation can take place subsurface and deep peat. Another way methane (CH_4) can be formed is by reduction of carbon dioxide using hydrogen. Biological methane is produced by miscellaneous Archaea and Proteobacteria, that together are referred to as methanogens. Methane as a gaseous substance may be transported upwards through soil pores and es-

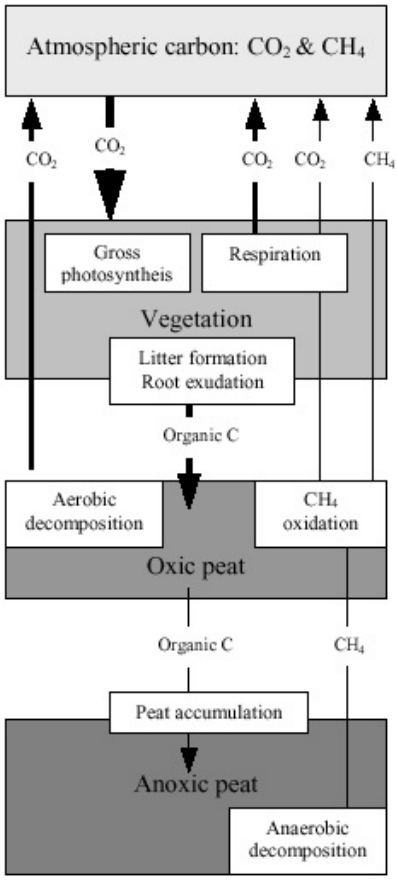

Figure 4: Pristine peatland carbon cycle. Peat is formed by accumulation of plant residues and root exudes. It is rapidly respired and returned to the atmosphere as carbon dioxide from the oxic top layer and also to less extent from the anoxic layer. Methane is the main microbial respiration product from the deeper, anoxic peat, but also can escape to atmosphere by diffusion from top peat. Vegetation also plays a major role, because photosynthesis is the main process, that transports CO_2 to soil. A part of it is rapidly respired and returned to atmosphere, while the rest is incorporated into plant biomass, which upon plant death serves as a source of organic matter and thus peat formation.
Source: Eeva-Stiina Tuittila, University of Helsinki dissertation *Restoring vegetation and carbon dynamics in a cut-away peatland*, 2000.

cape into atmosphere or may be oxidized and transported to atmosphere as carbon dioxide (see Figure 4).

Peatlands have naturally high water table, ensuring low oxygen availability, which suppresses microorganism ability to decompose plant litter. Peatlands gain more carbon carbon, than they export by decomposition, therefore in long term they serve as carbon storage. Peatlands store about one third of global terrestrial carbon or about 500 Pg (1 Pg = 10^{15} g) in boreal peatlands (Yu, 2012). Annual peat accumulation rate lies between 20 g m^{-2} $year^{-1}$ (Robinson and Moore, 1999, Turunen et al., 2002) and 50 g m^{-2} $year^{-1}$ (Gorham et al., 2003), adding an average of 1 mm peat $year^{-1}$ (Feton, 1980, Borren et al., 2004).

1.2.2 Carbon cycle under changing environmental conditions

Contemporary carbon cycle examines carbon exchange mechanisms and processes between segments or pools of carbon. Ocean is the largest carbon pool followed by soil. Atmosphere as the smallest carbon pool (Ciais and Sabine, 2014) has high sensitivity to changes,

therefore even slightest modifications in carbon balance between soil and atmosphere can have tremendous effects. IPCC (2012) showed, that mean annual temperature in Central Europe is increasing, but no significant changes in precipitation level are expected (IPCC, 2014). IPCC has also stated that precipitation distribution is expected to change, which will lead to more frequent droughts. Lower water table, increases oxic peat layer and thus advances CO_2 emissions (Mettrop et al., 2014).

Carbon dioxide is formation in wetlands occurs at higher extent than methane, because aerobic respiration (forms CO_2) yields far more energy by transporting ATP molecules compared to anaerobic respiration (forms CH_4). Aerobic respiration produces 19x more ATP molecules, but despite of recent hints of lower aerobic:anaerobic ATP production ratio of 15:1 (Rich, 2003), aerobic respiration is still far more productive and thus aerobic microorganism growth and activity dominates biological carbon cycling in soils.

CO_2 and CH_4 are both important players in atmospheric chemistry. They absorb long-wave solar radiation, which with increasing amount of these gases has a positive feedback: they trap more radiation, causing atmospheric temperature to rise. This effect is known as greenhouse effect. Even though CO_2 and CH_4 constitutes less than 1% of the Earth's atmosphere, they have a significant capability to take in solar radiation. IPCC has introduced a measure to compare various gases in form of greenhouse warming potential (GWP. According to which CH_4 contributes to global warming 28x more than CO_2 (IPCC, 2014).

Methanogenesis as a less productive form of respiration, yields on average of 40 g m^{-2} $year^{-1}$ (Yu, 2012), while CO_2 annual flux greatly varies. Silvola et al. (1996) reported that natural peatlands discharge 79 ... 347 g CO_2-C m^{-2} $year^{-1}$ (Lafleur et al., 2001), which is 2 - 8x higher than average methane production, while Kim and Verma (1992) reported CO_2 flux during growth season of 1300 g m^{-2}, exceeding average methane flux 30 times. Even though methane has higher GWP, carbon dioxide emissions from peatlands are much greater, therefore it is an important player of the global warming. Assuming IPCC climate models predicting more frequent droughts hold true, CO_2 emissions from natural peatlands are expected to rise as the water table decreases. In addition, peatlands are subjected to drainage due to land use change (e.g. growing agricultural crops) and extraction for economic profit, which has shown to accelerate microbial decomposition and thus CO_2 emissions (Waddington et al., 2002). Escalated carbon mineralization in natural and cut-over peatlands due to lowered water table has been studied on a global scale in different ecosystems, yet there is little knowledge about the fate of high-latitude mountainous peatlands under changing climate.

1.2.3 Different peat type degradation

Rate of decomposition of plant residues greatly depends on water level, but litter composition is of an equal importance. Despite moss being main botanical peat constituent, peatlands, even in pristine condition, are not monoculture ecosystems and inhabit a significant portion of sedges and grasses. Shrubs and trees are prevalent in forested peatlands, but their smaller counterparts, dwarf shrubs and trees and commonplace in natural peatlands in locales with low water table. Large portion of shrubs and trees can affect peatland moss growth. Mosses under too much shading tend to grow taller and are exposed to water deficiency, because of lack of water transporting tissues (Malmer et al., 1994), which can impede moss growth, especially in fens, where groundwater is the main source of water.

Trees and other vascular plants decompose quicker and produce more carbon concentrated litter than mosses (Reader and Stewart, 1972), moreover recurrent droughts and drainage may spread tree and shrub cover deeper into peatlands. Laiho et al. (2003) observing drained and forested peatbog in Finland concluded that 55 years after drainage litterfall from trees and shrubs increased from 20% to 68% of the total plant litter. Dwarf shrubs and trees produce more biomass than mosses and upon death add more litter in form of roots, which is larger than their above ground litter production from leaves, therefore vascular plants litter is mostly subsurface (Malmer et al., 1994). Shrubs and trees are decomposed at larger extent than mosses. Reader and Stewart (1972) studying forested peatland found that up to 38% of vascular plant litter mass is decomposed, while only 20% of *Sphangum* moss litter mass was decomposed. This indicates that forested or shrub and tree covered natural peatlands may accumulate less biomass and potentially shift their position from carbon sink to source.

There is some degree of uncertainty to whether tree litter in pristine, undrained peatlands will degrade at the same rate as in drained and/or forested ones. Minkkinen et al. (1999) comparing cellulose breakdown as a measure of OM decomposition potential in a drained and undrained part of a peat bog treed with Scots pine (*Pinus sylvestris* L.) concluded that tree litter in drained bog decomposes faster than in undrained. Meanwhile Laiho et al. (2004) conducting experiment with Scots pine litter found that it decomposes slightly faster than in drained peatsites, contradicting idea that water level drawdown enhances oxic decomposition. Ultimately, vegetation determines litter decomposition rate, because of their constituents. Mosses are thought to decompose slowly, because they contain phenolic compounds, that are toxic to microbes and animals at fairly low concentrations, while trees and shrubs do not contain phenolics, therefore decompose quicker. Dwarf tree and shrub expansion in peatlands is a step of succession as a result of drought or

drainage. Increase in tree coverage will lead to higher tree litter formation, followed by rapid degradation and CO_2 release, which will impede peat formation and potentially reduce peat carbon stock.

1.2.4 Nitrogen cycle

Nitrogen is a vital macro-nutrient to all life forms and is a building block of DNA, amino acids, cytoplasm and other essential cell components (Schooley, 1996, Bidlack et al., 2011), therefore nitrogen availability determines cell growth rate. Nitrogen comprises around 80% of the atmosphere in form of N_2 (Aneja et al., 2001) and to less extent is present in rocks and soil organic matter (Galloway, 1998). Rock and mineral erosion is a slow process in comparison to plant and animal lifespan, therefore nitrogen availability in soil depends on biological nitrogen fixation. Microorganisms assimilate atmospheric nitrogen, forming organic nitrogen species, that is later released back into soil as plant litter. Plants can utilize only inorganic nitrogen as nitrate $NO_3{}^-$ and some organic species (e.g. ammonia NH_3, urea), therefore they rely on microbial nitrogen mineralization. Nevertheless, only 3% of annual nitrogen fixation is transformed to plant available reactive nitrogen (Harper, 1984), therefore plant growth usually is regarded as nitrogen limited. Lightning can also produce reactive nitrogen, but is usually considered as a secondary source, because strikes occur sporadically and produce only a fraction of the global reactive nitrogen (Holland et al., 1999).

Biological nitrogen fixation forms up to 130 Tg N $year^{-1}$ (Galloway, 1998), of which most is returned back to atmosphere via respiration. Nitrogen is also returned to atmosphere by microbial mineralization as nitrous oxide N_2O and dinitrogen gas N_2. Biological nitrogen uptake forms ammonia (NH_3), which upon release returns back to atmosphere by diffusing through air filled soil pore spaces. Terrestrial nitrogen can also be transported to and deposited in marine ecosystems (see Figure 5).

Nitrogen deposition in peatlands occur in two ways: dry and wet deposition. Dry deposition is gaseous species (NO, NO_2, NH_3) and aerosol input to biosphere by biological fixation (Hargreaves et al., 1992). The magnitude of dry deposition depends on the nutritional status of the peatland (minerotrophic vs. ombrotrophic) and vegetation. Gaseous nitrogen uptake mainly depends on plant community nutrient demand. Bryophytes have lower nutrient demand, therefore they are not a significant source of nitrogen fixation, whereas vascular plants, like sedges can fix up to 69 kg N ha^{-1} $year^{-1}$ (Pitcairn et al., 1995). However this claim might overestimate the deposition rate and other studies have reported lower values of 16.8 kg N ha^{-1} $year^{-1}$ (Skiba et al., 1992) and 1.3 kg N ha^{-1}

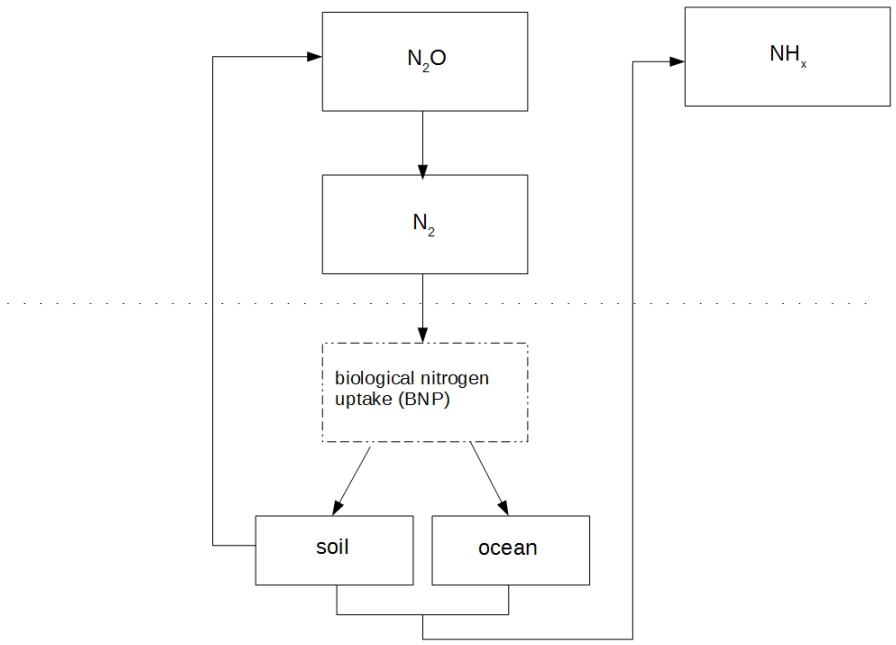

Figure 5: Nitrogen cycling between atmosphere and lithosphere. After Galloway, et al., (1995). Nitrogen fixation: Anthropogenic enhancement-environmental response. *Global Biogeochemical cycles, vol. 9.*

year^{-1} (Rosswall and Granhall, 1980), hence the true magnitude of dry deposition is highly variable and depends on various factors. Wet deposition is nitrogen accretion by precipitation containing nitrate (NO_3^-) and ammonium (NH_4^+) ions. Wet deposition is the main nitrogen acquisition process in natural rain fed bogs. Nitrogen deposition by rainfall, snow and fog contributes to 5 ... 10 kg N ha^{-1} year^{-1} globally (Bowden, 1987). In Europe it adds about 3 ... 4 kg N ha^{-1} year^{-1} (Simpson et al., 2011).

Nitrogen in peatlands is subjected to decomposition, which depends on the balance of input (accumulation) and output (mineralization and leaching). Mechanical leaching is water-soluble substance removal from live and dead plant matter, while mineralization is microbial decomposition, resulting in inorganic nitrogen species, which are either taken up by plants or transported to other ecosystems. Peatland nitrogen budget is a balance of immobilization by microorganisms/plants and decomposition. Soil organic matter decomposition rate depends reciprocally on carbon and nitrogen content. Microbes successfully decompose organic matter, when there is at least one part on nitrogen for 30 parts of carbon available (Nannipieri et al., 1978). If C-to-N ratio is higher than 30:1, microorganisms require more nitrogen successfully to utilize carbon, which in peat bogs is scarce, therefore addition of nitrogen is expected to advance rate of net mineralization (Williams,

1972, Henriksen and Breland, 1999, Bragazza et al., 2006, van Beek, 2007, Finn et al., 2015), as the number of microorganisms and their enzyme activity is increasing.

1.2.5 Nitrogen cycle under changing environmental conditions

Microbial decomposition of OM depends on physio-chemical properties of the environment, such as temperature, moisture, nutrient and carbon availability. As discussed in paragraph 1.2.2, peat bog water table level and its' changes is a fundamental factor in comprehension of OM decomposition. Nitrogen, as a nutrient, availability is another key factor controlling microbial activity.

Agricultural high-yield crop growth is dependent on additional fertilizers, which include high concentration nitrogen. Agricultural fertilizer supplies more nutrients compared to microbial fixation, however not all of the nitrogen is taken up by plants, therefore can leach into surrounding ecosystems or vaporize and return to soil with precipitation. Anthropogenic nitrogen brings additional 150 Tg (1 TG = 10^{12} g) N to the surface of the Earth every year (Schlesinger, 2009).

Anthropogenic nitrogen can have direct and indirect effects on nutrient cycling in peatlands. Additional nitrogen supply in peatlands can have indirect and direct effects. Indirect effects are mainly changes in plant community. *Sphagnum* mosses adapted to low nutrient conditions are not capable utilize all of the incoming nitrogen and does not alter their growth. Vascular plants, on the other hand, have shown to increase biomass and spread into new areas when more nutrients become available (Berendse et al., 2001). Increase of vascular plant cover can increase CO_2 emissions, because vascular plants decompose faster than mosses (Lang et al., 2009). *Sphagnum* mosses contain phenolic compounds, that are toxic to microorganisms (Verhoeven and Toth, 1995), which inhibit microbial activity, however fertilization seems to counteract it and mosses have shown to increase decomposition rate when more nutrients are bio-available (Limpens and Berendse, 2003). Direct effects of nitrogen input are associated with changes in soil chemistry. Inorganic nitrogen addition lowers pH and may reduce methane emissions due to emergence of inter species competition with nitrate reducing organisms, however this effect may disappear when the nitrate has been consumed. Lower pH can also negatively affect CO_2 emissions by suppressing enzyme activity, if the pH becomes too low. Nonetheless, nitrogen availability is not an independent factor. Oxygen availability is the key factor determining decomposition and even at lowered pH it can increase under dry conditions. Plethora of studies have dealt with moisture and nutrient level effect on plant litter decomposition, however none has attempted to look at the effects of these two key factors

simultaneously.

Fertilization does not only impact carbon cycling, but also determines nitrogen cycling. Microorganisms take up nitrogen and transform it into gaseous species (N_2O, NO, N_2) by nitrification and denitrification emitting 110 Tg N $year^{-1}$ (Schlesinger, 2009) from wetland ecosystems. Nitrification is biological $NH_{3/}NH_4^+$ oxidation to NO_3^-/NO_2^- and forms nitrous oxide (N_2O) as a sub-product. Denitrification is the same process, except it occurs in the lowest, water saturated horizons, where strict anoxic microbes utilize NO_3^-/NO_2^- and form N_2. Denitrification may also emit N_2O as a byproduct. Nitrous oxide is a colorless, non-flammable gas, that upon escape into atmosphere has a potential to trap long-wave solar radiation. Over a hundred year period, it contributes 234x more strongly to global warming than the same amount of CO_2 (IPCC, 2014). Anthropogenic nitrogen promotes N_2O loss from peatland ecosystems, especially under dry conditions (Regina et al., 1996), hence similarly to CO_2, N_2O release potential is linked to two factors: water table height and nutrient availability.

1.3 Aim of the study

Peat carbon dynamics has been studied in both laboratory and field conditions at wide extent. Current peatland research has focused mainly on temperate climate zones as they predominantly occur in cooler regions as discussed in chapter 1.1.1. In recent years, role of tropical peat in carbon dynamics is receiving alike attention, as the awareness of climate change and its effects are rising across the world. Nevertheless, peatland studies have largely focused on exploration of single factors. This study aims to observe peat degradation with respect to two individual factors and their interaction. The aims of the study can be summarized as following:

1. setup, maintain and sample oxic incubations of Pürgschachen Moor bog peat, supplemented with various degree of water and high-nitrogen fertilizer during the course of four weeks to determine effects on microbial respiration;
2. compare and analyze outcome among samples to determine role of nutrients and water level on peatland degradation with respect to peat origin (moss vs. pine);
3. compare results with similar studies and discuss differences in outcome with respect to sampling location to determine peatland degradation sensitivity in European Alpine peatlands.

This study aims to examine interaction of water level and nitrogen availability in control-

lable laboratory environment to minimize other environmental factor effect on microbial activity. Previous studies have found that pristine (predominantly moss) peat degrades faster than peat with large proportion of vascular plants. In addition, waterlogged conditions are not favorable to oxic microbial growth. Lower water table and additional nutrient supply is expected to advance peat degradation rate in pristine peat, however natural systems are more complex and are affected by individual factors and their interactions, which are not straightforward. This study aims to investigate peatland carbon dynamics in changing environmental conditions in connection with water table and fertilizer availability in high altitude peat bog.

2 Materials and Methods

2.1 Site description

Peat bogs in Austria have largely formed after the last glacial period, where melted ice water created lakes and streams, flooding valleys, which slowed down plant litter decomposition. Majority of Austrian peatlands are high altitude peat bogs lying between 0 and 1000 m above sea level (Steiner, 1982) and cover 0.15% of Austrian alpine territory (Petz, 1999). Even though high altitude peatlands are less common than lowland mires in northern Europe and north America, mountain regions are sensitive to climate change and thus Alpine peatlands can be subjected to changes in element storage and cycling as strongly as boreal peatlands (Bohdalkova et al., 2014), therefore additional research to understand carbon dynamics should be conducted.

Pürgschachen Moor (47° 34' 50" N and 14° 20' 40" E) peat bog is situated in the federal state of Styria, Austria 632 m above the sea level . It is located in river Enns catchment area and within the municipality district Liezen (pop. 6800), east of Liezen town and just south-west of Ardning village (pop. 648 (2013)) (Gemeinde Ardning, 2013). The total area of the bog is 62 hectares or 0,62 km^2 (Turk, 2006).

Figure 6: Location of Pürgschachen Moor bog in the central Austria (47° 34' 50" N, 14° 20' 40" E).

Austria was situated on the southern edge of the great ice cap during the last glacial

period. In addition, there was a smaller ice sheet covering Alps, which upon retreating created a lake. The lake has disappeared, leaving behind a flooded plain, initiating rapid plant material decomposition between 10 000 and 5000 years ago (Gemeinde Ardning, N/A). Hence, Pürgschachen Moor like numerous other bogs in Europe is of post-glacial origin. Over time the bog has accumulated peat with a depth of 6 meters.

The nearest weather station lies in the city of Admont, around to the 10 km east of Pürgschachen Moor. The city is occupying the same altitude as peat bog, therefore despite the distance, climate properties are comparable. According to long-term climatic data (1971 - 2000), mean average annual temperature is +6.6° C. Mean lowest air temperature occurs in January -4.2° C, and the highest - in July +16.3° C. Annual precipitation level is 1400 mm. The area is located in temperate climate zone, therefore there is snow coverage during winter months with the highest amount of fresh snow on December, but the biggest snow cover in January with total number of days with snow cover of 79. Mean annual wind speed is 2 m/s, blowing from E, S-E during summer and and W, N-W during winter months (ZAMG, 2002).

Pürgschachen Moor bog is a private property owned by monastery of Admont, but is leased to Pürgschachen Moor Protection Association (Moorschutzverein Pürgschachen) indefinitely. The association administers the bog territory, to repserve its' pristine and natural conditions, therefore no human activities, except research, are allowed within the central area of this bog. However, to promote understanding of ecological role of the bog and local tourism, a walking trail around the bog has been constructed in the early 2000's (Gemeinde Ardning, N/A). Since 1991, the bog is a participant site of an international wetland treaty – Ramsar. The bog is subjected to use restrictions to preserve natural alpine peatlands and to sustain it as a habitat for numerous endangered flora and fauna species included the National Red List. The bog is also included in European Union valuable and threatened habitat list NATURA2000. Under this treaty, a management plan for preservation and restoration has been developed and pursued. The interest of Pürgschachen Moor bog preservation lies not only because of the role as a habitat, but also due to the fact, that inter-Alpine bogs have been subjected to mining and undisturbed ones have become scarce.

2.2 Sampling

For this study peat samples were collected on May 16 - 17, 2016 from Pürgschachen Moor peat bog in central Austria from areas with different vegetation pattern. Peat cores were

collected using Eijkelkamp© peat sampler down to 50 cm depth to ensure that only peat from the acrotelm horizon was collected. The bog does not have significant number of hummocks, therefore peat collection was executed on a flat terrain disregarding differences of local relief. Four cores were collected from each sampling site to ensure reproducibility and randomness of the sample.

(a)  (b) 

Figure 7: Photos of Pürgschachen Moor bog taken during sampling in May, 2016. (a) sampling site with moss and heather vegetative cover. (b) sampling site with alpine pine trees in addition to mosses.

Two sampling sites with different vegetation pattern located in different area of the peat bog (see Figure 8) were selected. One site was dominated by *Sphagnum spp.* and *Calluna vulgaris*, while the second one in addition to moss and sedges was covered by low growing *Pinus mugo* (see Figure 7). Peat collected in the first sampling site was lightly decomposed in different shades of dark green to dark brown. Poorly composed *C. vulgaris* roots were found throughout the whole length of the peat core. Peat forming plant matter was largely identifiable with some amorphous, dark brown material. Peat throughout the whole core was fairly uniform without any significant changes in texture and color. On the scale of humification established by a Swedish scientist von Post in 1922, where 1 indicates undecomposed plant material and 10 marks fully decomposed plant litter, sampling site one as a lightly decomposed peat, was assigned index of H3 - H4.

The second sampling site located south of the first sampling site (see picture blah blah) was vegetated by mosses (*Sphangum spp.*), cottongrass (*Calluna vulgaris*), moutaineous pine trees (*Pinus mugo*), lingoberry (*Vaccinium vitis-idea*) and bilberry (*Vaccinium myrtillus L.*). Second sampling site peat was less uniform and had some visual differences throughout the top 50 cm used for the study. Top layer was covered with light brown, lightly decomposed with distinguishable root material, followed by darker, strongly decomposed material. 0 - 15 cm deep peat was dark brown to black, strongly decomposed with a pasty texture, corresponding to sapric H8 - H9 peat according to von Post scale EKONO (1981).

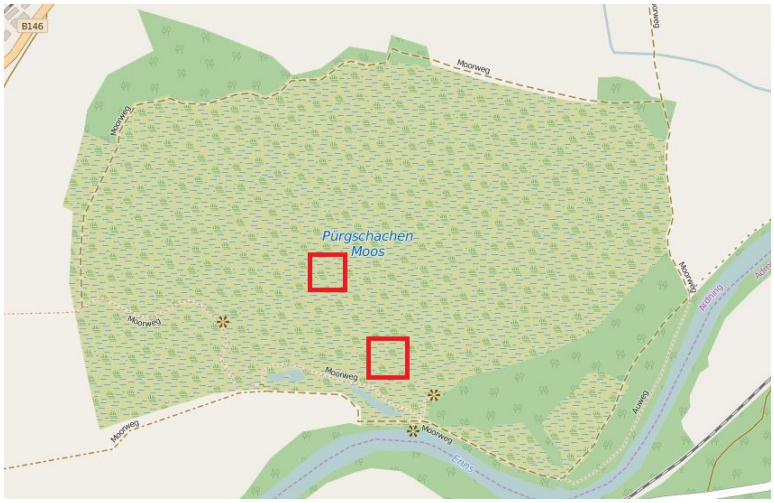

Figure 8: Sampling sites at Pürgschachen Moor. Red squares indicate approximate location, where peat core samples were extracted. Upper sampling plot marked by a red square is vegetated by mosses and sedges is located central area of the bog, while pine tree dominated samples were taken from lower plot in the south of the peat bog. Source: © OpenStreetMap.

Each peat core selected for the study was placed into individual 50x5 cm plastic pipe and covered with plastic wrap to prevent water loss. The specimen were immediately placed and stored in a cooling box with ice-packs for 24 to 48 hours before transporting to soil geography laboratory at University of Vienna, where they were placed into a refrigerator with a constant temperature of +4°C until beginning of the experiment.

2.3 Laboratory analysis

Decomposition rate in laboratory is usually tested by soil incubation (Carter, 1993). Even though laboratory does not provide natural conditions, it allows to control environment and test individual and multiple parameter effect on soil respiration. Soil respiration rate measured in laboratory conditions, however is relative and does not represent the real world scenario, rather it helps to determine relative respiration changes to make predictions, model future and seek land-use planning with regard to greenhouse gas emissions.

Peat cores were cut into 5 cm long segments. In order to study the control mechanisms of microbial decomposition, peat of 0 - 10 cm depth was neglected to eliminate respiration from living plant biomass and roots. Segments of peat between 10 and 35 cm were selected and homogenized by hand.

Homogenized peat mass from each core was split into 9 parts to test effect of three different levels of moisture and nitrogen availability (see Figure 9). Each incubated sample was replicated 4 times to ensure reproducibility (n = 4), yielding overall sample number of 72.

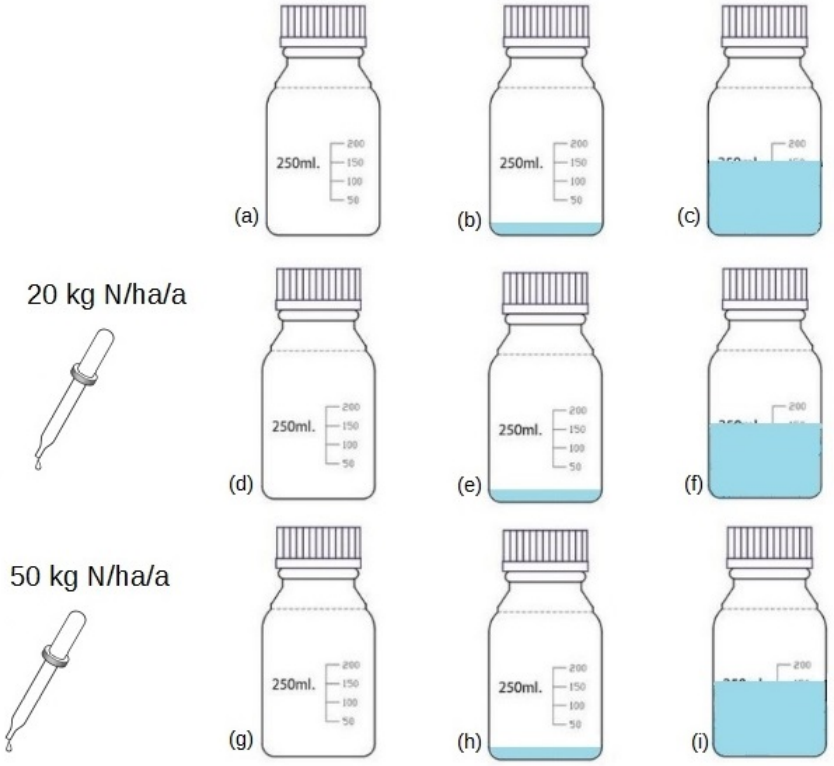

Figure 9: Laboratory incubation setup. (a) Dry peat with no added nitrogen. (b) Moist peat with no added nitrogen. (c) Water saturated peat with no added nitrogen. (d) Dry peat with added nitrogen load of 20 kg/ha/year. (e) Moist peat with added nitrogen load of 20 kg/ha/year. (f) Water saturated peat with added nitrogen load of 20 kg/ha/year. (g) Dry peat with added nitrogen load of 50 kg/ha/year. (h) Moist peat with added nitrogen load of 50 kg/ha/year. (i) Water saturated peat with added nitrogen load of 50 kg/ha/year.

For the experiment, approximately 10 grams of fresh peat was weighed into 250 ml transparent glass reagent bottles. Dry samples were adjusted to 50% of field capacity, while wet samples received 100% of their field capacity. To authors knowledge, there is no specific method to determine peat moisture, therefore field capacity was measured following soil moisture laboratory method, namely using pressure plates. Metal cylinders were filled up with fresh peat and one side was covered with cloth to keep the soil intact. The cylinder was soaked into water for 24 hours to make sure it has absorbed maximum amount of water. After 24 hours, the cylinders were removed from the water and placed onto clay and sand mixture filled funnel attached to a pressure adjusting device and covered with plastic wrap to eliminate evaporation. -0.33 bar suction pressure was applied to the peat for 4 hours, which after the peat samples were removed and weighted. Then peat filled cylinders were oven dried at 105° C for 24 hours and weighted again. The weight difference between moist peat subjected to drainage and oven dried peat represents the

amount of water, peat can hold and is equal to its' field capacity. Samples selected for water saturated conditions were topped up with double distilled water to 100 ml mark.

Fertilization was simulated by addition of nitrogen as ammonium nitrate (NH_4NO_3), a common high-nitrogen fertilizer. Samples received 3 different levels of fertilizer equivalent to 0, 20 and 50 kg N^{-1} year $^{-1}$. Ammonium nitrate concentration was calculated as

$$c_{NH_4NO_3} = \frac{m_{NH_4NO_3} * A * 1000}{MW_{NH_4NO_3}} \quad [mM], \tag{1}$$

where $m_NH_4NO_3$ is mass of ammonium nitrate (g), A represents surface area of a peat core sample (m^2) and MW_{NH4NO3} is ammonium nitrate molar mass (g/mol), yielding to 0.55 mM ammonium nitrate solution for 50 kg/N/ha/a and 0.14 mM solution for 20 kg/N/ha/a. Samples were fertilized by a plastic syringe with 1 mL of corresponding fertilizer solution, except samples with no added nitrogen, where samples received 1 mL of double distilled water.

Samples were incubated in a climate chamber at temperature of 20° C starting from day 2. At day 1 samples were kept in a laboratory room, but due to temperature fluctuations throughout the day, they were moved to strictly controlled temperature environment. Samples were measured for CO_2, CH_4 and N_2O emissions over 30 day period. Measurements were taken on days 1, 4, 8, 11, 14, 17, 22, 25 and 30. At the beginning of each measurement jars were briefly flushed with nitrogen and closed with tightly fitting lid with an opening in the middle covered with a soft rubber stopper (see Figure 10). Gases were allowed to evolve for 3 to 4 hours. At the beginning and end of each measurement 20 mL of headspace was sampled by a syringe and injected into a glass sample vial. Gases were analyzed using flame ionization detector (FID) for CO_2 and CH_4 and N_2O using electron capture detector (ECD) on a gas chromatograph (Agilent 7890B) with automatic headspace sampler (Agilent 7697A).

For pH determination 20 g of peat was mixed with 50 g of double distilled water, resulting in 1:2.5 soil-to-water ratio. Soil and water slurry was shaken into rotary shaker for 2 hours and measured using pH meter (©InoLab WTW 720).

2.4 Data analysis

CO_2, CH_4 and N_2O concentrations were calculated as concentration difference in headspace between beginning and the end of measurement and adjusted to volume, temperature and

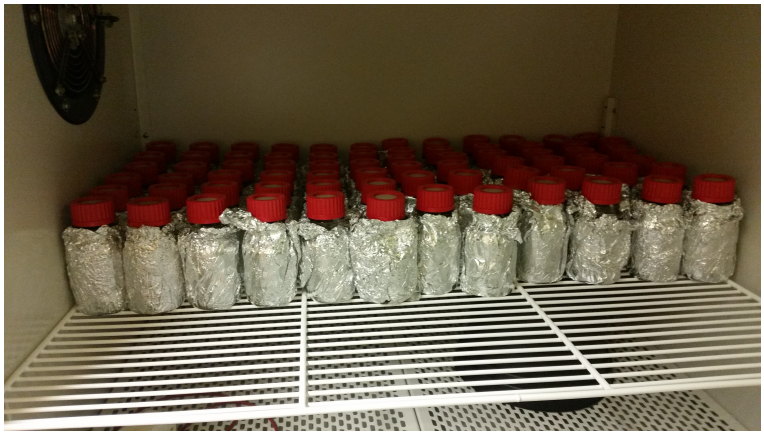

Figure 10: A photo of incubated peat samples. Jars containing peat samples with various water and fertilizer treatments in a chamber set to maintain 20° C. Jars in the photo are closed with lids, indicating that gas emission measurement is in process.

pressure. Data was corrected to express gas headspace change over time adopting ideal gas law and Liu et al. (2016):

$$\mathrm{R} = (\frac{(\mathrm{PPM_{end}} - \mathrm{PPM_{start}}) * \mathrm{MW}}{22.4\,\mathrm{mol/l}}) * (\frac{273.15K}{\mathrm{t_{actual}}}) * (\frac{\mathrm{p_{actual}}}{\mathrm{p_{STD}}}) * (\frac{\mathrm{V}_h}{\mathrm{m_{peat}}}), \tag{2}$$

where R is carbon dioxide production rate in $\mathrm{mg/g_{drypeat}/day}$, $\mathrm{PPM_{start}}$ and $\mathrm{PPM_{end}}$ are gas concentration in jar headspace at the beginning and the end of the experiment, MW is molar mass (CO_2 44.01 g/mol), 22.4 mol/l is a value, that shows how much volume one mole of ideal gas occupies at standard conditions (STP), 273.15 K is temperature at STP, $\mathrm{t_{actual}}$ is the temperate at which the experiment was conducted (in Kelvins), $\mathrm{p_{actual}}$ is estimated pressure during the experiment, taking place under normal conditions, $\mathrm{p_{STD}}$ is the suggested standard pressure value by the International Union of Applied and Pure chemistry (IUAPC) as 1 bar or 10^5 Pa (Cox, 1982), V_h is jar headspace volume without peat and water and $\mathrm{m_{peat}}$ is mass of oven dried peat. The resulting values were interpolated between sampling days to represent gas flux in a period of one day (24 hours), assuming gas formations occurs linearly.

Statistical data analysis was conducted using SPSS Statistics 23.0 for Windows (IBM, 2016) to compare means of the treatments with regard to sampling location, moisture and fertilization as independent factors (fixed factors) using ANOVA, which compares means of the treatment groups to detect differences. Graphs were created using OriginPro 8.0.

3 Results

3.1 Carbon dioxide production

Carbon dioxide production rate in all samples sharply increased and reached maximum at day 4 of the incubation. On incubation day 8 production rate sharply decreased. From day 11 through the end of experiment carbon dioxide production continued to gradually decline (see figure 11). CO_2 production rate differed between moss and pine dominated peat bog areas. Peat formed predominantly from *Sphagnum spp.* produced up to 0.69 mg CO_2 $g_{dw}{}^{-1}$ day^{-1}, whereas *P. mugo* maximum production was 0.47 mg CO_2 $g_{dw}{}^{-1}$ day^{-1}, which is 38% lower than moss dominated peat. Overall CO_2 production throughout the study in individual sample replicates ranged from $3 * 10^{-5}$ to 1.06 mg g $CO_2{}_{dw}{}^{-1}$ day^{-1}.

Table 3: Statistical analysis of the effects of sampling location, moisture and fertilization treatments (confidence level 95%). Sig. column indicates p-value. If p-value > 0.05, null hypothesis, which states that all treatment group means are equal cannot be rejected and differences in CO_2 production rate are not affected by treatments.

Independent variable	*df*	Mean Square	*F*	*Sig.*	*eta*
moisture	2	139.253	214.472	.000	.888
fertilization	2	2.265	3.489	.038	.114
location	1	75.293	115.963	.000	.682
moisture*fertilization	4	2.466	3.797	.009	.220
moisture*location	2	8.186	12.608	.000	.318
fertilization*location	2	1.060	1.632	.205	.057
moisture*fertilization* location	4	.412	.634	.640	.045

R squared = .904 (Adjusted R Squared = .874)

Cumulative carbon dioxide production after 30 days showed different outcome among the various treatments (see table 5). Cumulative production in all samples ranged between 1.17 and 10.22 mg CO_2 $g_{dw}{}^{-1}$ day^{-1}. Effects of water content and high nitrogen concentration fertilizer using analysis of variance shows that water saturation statistically has a strong effect on the outcome ($p = 0.000$) and fertilization had a less strong ($p = 0.038$), but still significant effect of the CO_2 emissions (see table 3). The combined effect of moisture*fertilization had a statistically significant effect ($p = 0.009$), but combined effect of treatments and sampling location altogether cannot be accounted for the differences in carbon dioxide production. Overall, the strongest effect on CO_2 production had water content, sampling location and their combination. ANOVA reveals only whether there exist statistically significant differences among all of the treatment groups, therefore

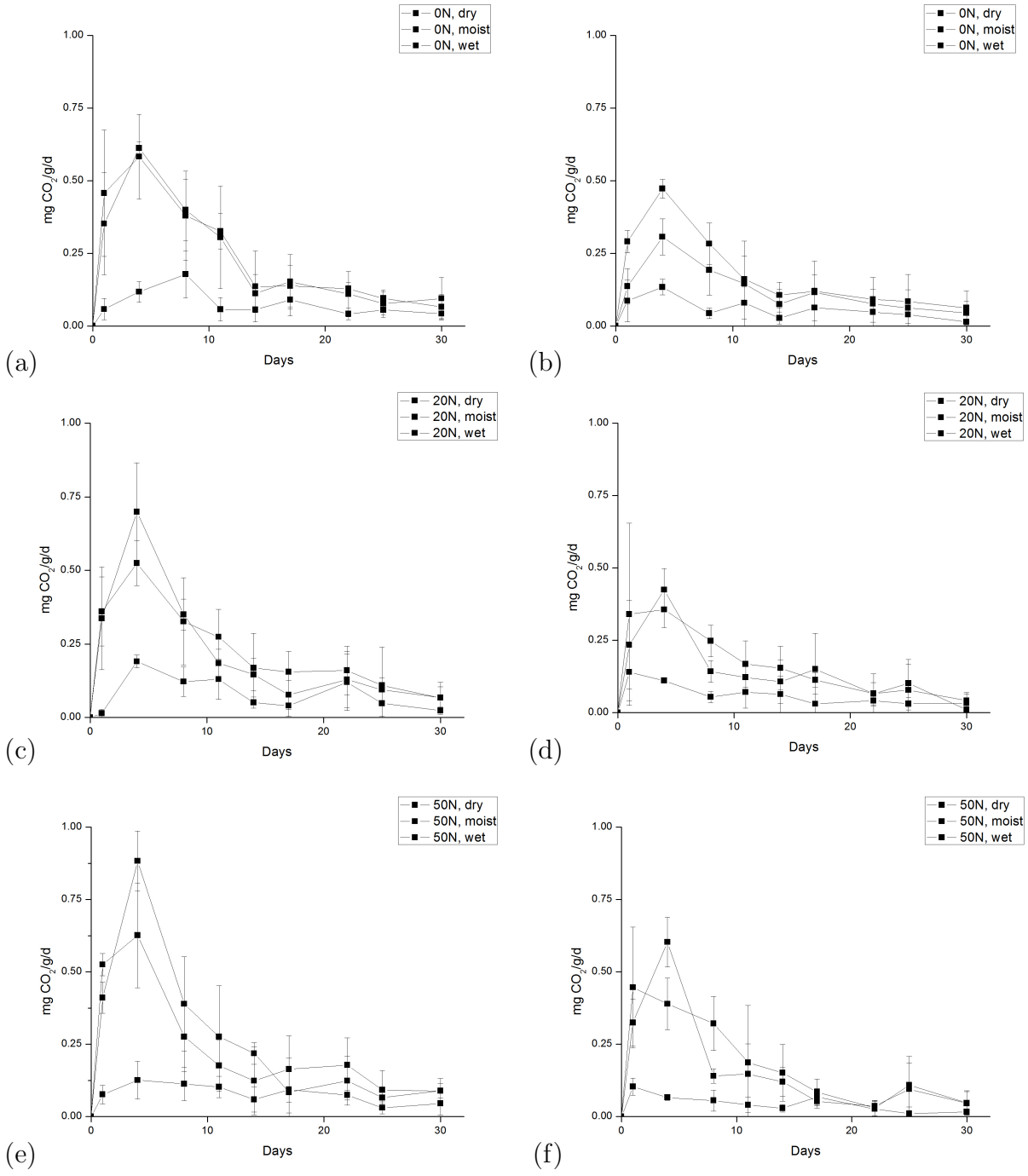

Figure 11: Measured CO_2 production rate in peat with varying degree of moisture and fertilization. Mean carbon dioxide production rate in samples with varying degree of moisture and fertilization (n=4, ±1 standard deviation) in $mg_{dw}{}^{-1}$ day^{-1}. (a) CO_2 production rate in unfertilized moss peat. (b) CO_2 production in unfertilized pine tree peat. (c) CO_2 production rate in lightly fertilized (20kg N ha^{-1} y^{-1}) moss peat. (d) CO_2 production in lightly fertilized (20kg N ha^{-1} y^{-1}) moss peat. (e) CO_2 production in lightly heavily fertilized (50kg N ha^{-1} y^{-1}) moss peat. (f) CO_2 production in heavily fertilized (50kg N ha^{-1} y^{-1}) pine tree peat.

patterns of differences were examined using LSD post hoc test. It revealed that significant difference exists only between samples without fertilization and samples with heavy fertilization of 50 kg N ha^{-1} year $^{-1}$ (p = 0.029).

Table 5: Cumulative carbon dioxide production during 30 day incubation.

Site	Nutrients	Moisture	CO_2 (mg)
Moss	atmospheric deposition	dry	7.31 ± 1.21
		moist	7.13 ± 0.92
		wet	2.36 ± 0.25
	20 kg N/ha/a	dry	6.92 ± 1.36
		moist	6.86 ± 0.44
		wet	2.57 ± 0.59
	50 kg N/ha/a	dry	7.63 ± 1.04
		moist	9.17 ± 0.71
		wet	2.40 ± 0.58
Mountain pine	atmospheric deposition	dry	5.39 ± 0.81
		moist	3.83 ± 1.02
		wet	1.75 ± 0.23
	20 kg N/ha/a	dry	4.98 ± 0.48
		moist	4.62 ± 1.16
		wet	1.69 ± 0.27
	50 kg N/ha/a	dry	5.23 ± 0.86
		moist	5.14 ± 0.88
		wet	1.31 ± 0.15

Post hoc test also revealed, that there is no statistical significance between the different CO_2 emissions from dry vs. moist samples (p = 0.614). Dry in respect to wet samples and moist in respect to wet sample difference proved to be significant (p = 0.000).

pH values dropped in all samples after 30 day incubation period. The highest decrease happened in dry samples with light fertilization of 20 kg N ha^{-1} $year^{-1}$. In *Sphagnum*-peat pH dropped for 1.37 units. Least changes in pH occurred in *wet* water-flooded samples.

3.2 Methane production

Methane production similarly to carbon dioxide production peaked after 4 days after which it dropped and stayed low until the end of experiment (see Figure 12). Measured methane production varied between 0.01 and 1.49 μg CH_4 in *Sphagnum* peat and 0.01 ... 0.71 $\mu g_{dw}{}^{-1}$ day^{-1} from mountain pine peat.

Table 7: pH values of bog peat samples before and after incubation.

	Nutrients	Water content	moss site	pine site
Before incubation			5.17	5.23
After incubation	atmospheric	dry	4.02	4.15
	deposition	moist	4.37	4.11
		wet	4.40	4.70
	20 kg N/ha/a	dry	3.80	3.89
		moist	4.30	4.23
		wet	4.94	4.70
	50 kg N/ha/a	dry	4.36	3.76
		moist	4.42	4.02
		wet	4.49	4.87

Cumulatively in a 30 day period most methane was produced in *wet* samples. Moss peat produced 3.3 times more CH_4 and pine tree peat – 3.4 times more methane, averaging 3 times higher methane production under water saturated conditions compared to dry samples (see table 9). Moist peat produced more methane than dry, but 1.8 times less in *Sphagnum*-peat and 1.6 times less in pine-peat in comparison to wet peat.

Table 9: Cumulative methane production during 30 day incubation.

Site	Nutrients	Moisture	CH_4 (μg)
Moss	atmospheric	dry	4.93 ± 2.11
	deposition	moist	9.19 ± 0.41
		wet	16.22 ± 1.49
	20 kg N/ha/a	dry	4.75 ± 2.04
		moist	9.91 ± 2.29
		wet	13.99 ± 2.95
	50 kg N/ha/a	dry	5.28 ± 0.69
		moist	9.74 ± 1.75
		wet	15.48 ± 2.89
Mountain	atmospheric	dry	2.19 ± 0.38
pine	deposition	moist	4.68 ± 0.59
		wet	7.40 ± 0.83
	20 kg N/ha/a	dry	2.84 ± 0.33
		moist	5.59 ± 1.69
		wet	5.81 ± 0.99
	50 kg N/ha/a	dry	2.09 ± 0.38
		moist	4.92 ± 0.94
		wet	7.68 ± 1.17

Methane production statistical significance was very strong, indicating causality between treatments and the outcome with moisture, location and moisture*location combined effect (p = .000). Fertilization combination with moisture and location had no statistical significance on the outcome (see table 3). Further testing of treatments using least significant difference (LSD) revealed that significant differences exist between dry, moist and wet samples (p = .000), however none of them occurred due to fertilization.

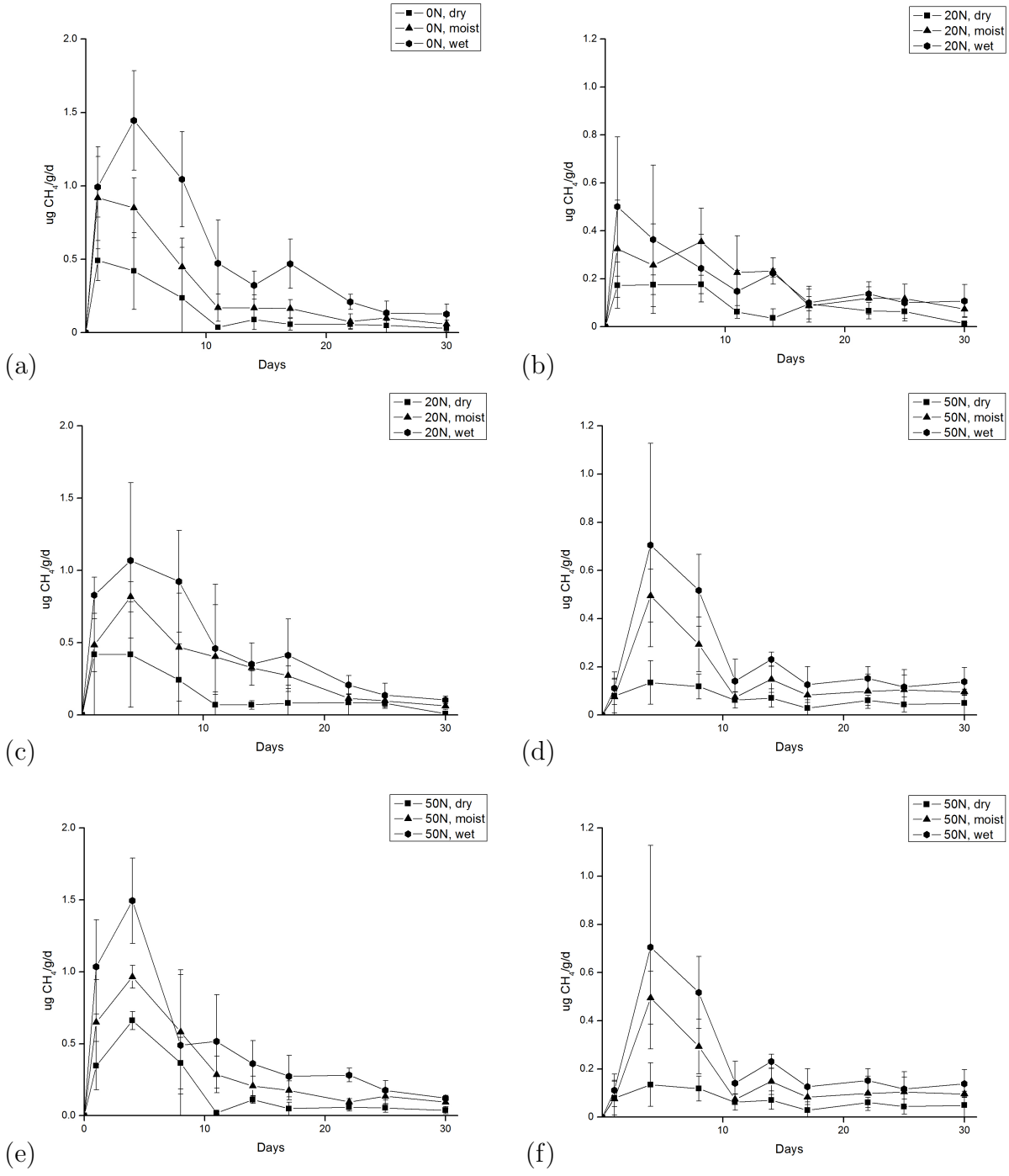

Figure 12: Measured CH_4 production rate in peat with varying degree of moisture and fertilization. Mean methane production rate in samples with varying degree of moisture and fertilization (n=4, ±1 standard deviation) in μg ${g_{dw}}^{-1}$ day^{-1}. (a) CH_4 production rate in unfertilized moss peat. (b) CH_4 production in unfertilized pine tree peat. (c) CH_4 production rate in lightly fertilized (20kg N ha^{-1} y^{-1}) moss peat. (d) CH_4 production in lightly fertilized (20kg N ha^{-1} y^{-1}) moss peat. (e) CH_4 production in lightly heavily fertilized (50kg N ha^{-1} y^{-1}) moss peat. (f) CH_4 production in heavily fertilized (50kg N ha^{-1} y^{-1}) pine tree peat.

Table 11: Outcome of the statistical significance testing for methane production using analysis of variance (α = 95%.) Sig. column indicates p-value, which exceeding 0.05 indicates that sample means cannot be explained by experimental treatments.

Independent variable	*df*	Mean Square	*F*	*Sig.*	*eta*
moisture	2	330.331	133.843	.000	.832
fertilization	2	.936	.379	.686	.014
location	1	476.272	192.974	.000	.781
moisture*fertilization	4	4.602	1.865	.130	.121
moisture*location	2	49.580	20.089	.000	.427
fertilization*location	2	.531	.215	.807	.008
moisture*fertilization* location	4	.492	.200	.938	.015

R squared = .904 (Adjusted R Squared = .874)

3.3 Nitrous oxide production

N_2O production rate reached maximum after 4 days of incubation. From day 8 of the experiment rate decreased and did not raise again, however the production rate during the second half of the experiment was fluctuating and standard deviation was higher in comparison to CO_2 and CH_4 production rates (see Figure 13). High standard deviations were experienced, especially in *Sphagnum*-peat, because individual sample replicates had high variety in the outcome and in some occasions N_2O production rate was negative, meaning nitrous oxide was consumed instead of being produced.

Table 13: Statistical analysis of the effects of sampling location, moisture and fertilization treatments (confidence level 95%) on the N_2O emissions. Sig. column indicates p-value. If p-value > 0.05, null hypothesis, which states that all treatment group means are equal cannot be rejected and differences in CO_2 production rate are not affected by treatments.

Independent variable	*df*	Mean Square	*F*	*Sig.*	*eta*
moisture	2	20964289.53	14.017	.000	.919
fertilization	2	93995.532	.063	.000	.342
location	1	46608517.62	31.163	.000	.366
moisture*fertilization	4	2407859.676	1.610	.185	.107
moisture*location	2	12549906.43	8.391	.001	.237
fertilization*location	2	10965898.50	7.332	.002	.214
moisture*fertilization* location	4	769509.781	.514	.725	.037

R squared = .648 (Adjusted R Squared = .537)

Nitrous oxide production differences are strongly related to moisture, fertilization level, location (p = .000), moisture*location (p = .001) and fertilization*location (p = .002). Moisture*fertilization and moisture*fertilization*location combined effects had no statistical significance (see table 13). LSD post hoc test unveiled that statistical significance in regard to moisture poses samples that are *dry* and *wet* as well there are significant differences between *moist* and *wet* samples. N_2O emission differences between *dry* and *moist* peat are not statistically significant and thus the different outcome is not due to the water content they were exposed to experimentally.

Nitrous oxide production was higher in *Sphagnum*-peat than in pine-peat. In dry conditions *Sphagnum*-peat emitted 2.5 times more than pine-peat. In lightly fertilized dry peat *Sphagnum* released 2.2 times more, but in heavily fertilized – 1.3 times more than mountain pine peat (see table 15).

Overall more N_2O was emitted from dry peat regardless of its origin. *Sphangum*-peat emitted 2.35 times more N_2O from dry than wet peat and dry pine-peat released 1.52 times more compared to wet conditions. Moist peat had the maximum N_2O production in both types of peat, still moss peat emitted more N_2O averaging 2.4 times more than from pine tree peat.

Table 15: Cumulative nitrous oxide production during 30 day incubation.

Site	Nutrients	Moisture	N_2O (ng)
Moss	atmospheric	dry	6014 ± 1742
	deposition	moist	6106 ± 1020
		wet	2547 ± 370
	20 kg N/ha/a	dry	5059 ± 1883
		moist	6421 ± 2937
		wet	2273 ± 333
	50 kg N/ha/a	dry	4144 ± 510
		moist	4146 ± 712
		wet	2768 ± 1043
Mountain	atmospheric	dry	2440 ± 70
pine	deposition	moist	2561 ± 310
		wet	1596 ± 233
	20 kg N/ha/a	dry	2289 ± 521
		moist	2869 ± 1661
		wet	2347 ± 1033
	50 kg N/ha/a	dry	3155 ± 879
		moist	3913 ± 1174
		wet	3738 ± 1541

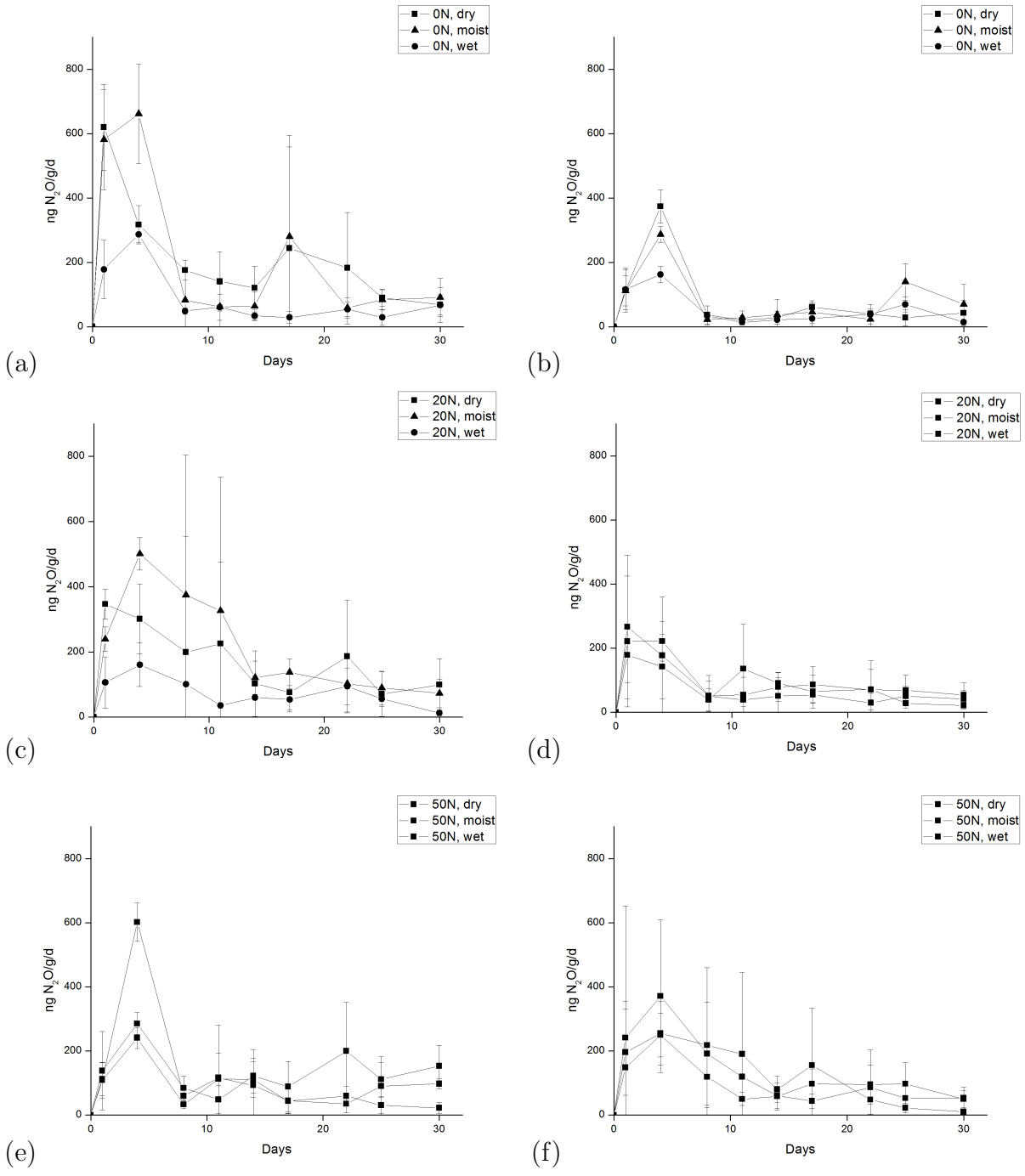

Figure 13: Measured N_2O production rate in peat with varying degree of moisture and fertilization. Mean nitrous oxide production rate in samples with varying degree of moisture and fertilization (n=4, ±1 standard deviation) in ng $g_{dw}{}^{-1}$ day^{-1}. (a) N_2O production rate in unfertilized moss peat. (b) N_2O production in unfertilized pine tree peat. (c) N_2O production rate in lightly fertilized (20kg N ha^{-1} y^{-1}) moss peat. (d) N_2O production in lightly fertilized (20kg N ha^{-1} y^{-1}) moss peat. (e) N_2O production in lightly heavily fertilized (50kg N ha^{-1} y^{-1}) moss peat. (f) N_2O production in heavily fertilized (50kg N ha^{-1} y^{-1}) pine tree peat.

4 Discussion

4.1 Aerobic respiration

Carbon dioxide cumulative production in this study was 7.13 ± 0.92 mg CO_2 $g_{dw}{}^{-1}$ day^{-1} from moss peat and 3.83 ± 1.02 mg CO_2 $g_{dw}{}^{-1}$ day^{-1} from mountain pine peat in moist conditions (FC 100%), which is lower than reported in similar studies. Aerts and Toet (1997) conducted a similar study and resulted in average of 1.25 mg CO_2 per day over the course of incubation, but this study 0.24 mg per day in bryophytic and 0.13 mg per day in ligneous peat. The higher production of their study could have occurred because of two reasons. This study neglected the uppermost 10 cm of peat and 35 cm, while they studied 2 – 10 cm deep peat cores. Peat closer to the surface is newly formed and contains fresh and easily available organic material, which microorganisms prefer, therefore the uppermost peat has the greatest organic matter mineralization potential. Secondly, this study used sedge-peat, which has been suggested to have the highest potential of carbon dioxide release in comparison moss and shrub/tree (ligneous) originated peat (Moore and Dalva, 1997, Wright et al., 2011).

Nilsson and Bohlin (1993) in a similar study resulted in average CO_2 emission rate up to 0.29 mg per day, which is slightly higher, yet comparable with peat CO_2 emissions in this study. On the other hand, analyzed samples were extracted at 0.5 – 1.0m depth, therefore findings represent emission potential from deep peat. In another study Waddington et al. (2001) compared *ex situ* emissions from peat with different land use. CO_2 outgassing from natural peat varied greatly at different depth, observing almost 3 times lower CO_2 production at 20 cm depth compared to 10 cm deep peat. Expressing their findings in weight, peat from 20 cm depth produced on average 0.24 mg CO_2 day^{-1}, which falls within the range of this study.

Nonetheless, peat respiration rate depends on depth. (Estop-Aragonés and Blodau, 2012) argues that at least half of microbial respiration occurs in the top 5 cm of peat and reduces with increasing depth resulting in 1 – 10% of their total microbial respiration to take place in peat 15 – 50 cm below the surface. Yavitt et al. (1987) showed that CO_2 emissions from peat gathered from subsurface (10 - 15 cm) are 4 times higher than in deep peat (30 - 35 cm).

Carbon dioxide is a product of aerobic heterotrophic respiration and oxygen is a vital component for this process. The extent of respiration in peat depends on depth because water table height controls oxygen availability, therefore aerobic respiration and peat

depth has an inverse relationship (Moore and Knowles, 1989, Chimner and Cooper, 2003, Berglund and Berglund, 2011, Juszczak et al., 2013). It is difficult to speculate whether the aerobic respiration rate in this study was considerably different than in similar studies, suggesting inconsistencies in laboratory methodology, as these values are relative and are used to compare response of a treatment within the same study rather than to directly compare it to other work. This suggests, that carbon dioxide emissions form peat cannot be solely explained by oxygen availability and other factors have to be taken into account equally.

Natural peatlands are most commonly covered with three types of vegetation – bryophytic (mosses), herbaceous (sedges and grasses) and ligneous (shrubs and trees), which dead biomass residuals under water logged conditions form peat. These three plant communities are different in respect to physiology, reproduction, nutrient/water requirements and other biological properties making their remnants decompose at different pace Williams and Yavitt (2003), however peat decomposition indices are not widely used in peatland carbon emission studies as they have not been found to correlate with microbial respiration Moore and Dalva (1997) or they do not correlate strongly (Nilsson and Bohlin, 1993). Nowadays state of peat decomposition is used in paleogeology to compare historic organic matter degradation in peatlands in order to reconstruct climate and environmental conditions of the past Swindles et al. (2012).

Carbon dioxide emissions in all samples analyzed in this study did not have significant correlation with von Post decomposition index (see Figure 14). Linear regression model fitted to sample means could explain only 19% of carbon dioxide emissions, therefore state of decomposition confirmed previous studies indicating weak relationship. Degree of decomposition in peat and organic soils is determined to specify how much plant litter has been degraded which can help to draw conclusions about OM content, quality and potential degradation, which in turn can help to determine potential carbon emissions in future. Degree of decomposition is a useful tool to describe how much specific peatland has been decomposed, which helps to reflect on carbon storage and potential carbon emissions, however, it is determined visually, therefore personal bias and inconsistent assignment of indices across studies cannot be dismissed. Degree of decomposition used as texture and degradation characteristic is a practical, because it is quick, straightforward and cheap method, however it has not proven to interact with rate of respiration. To authors knowledge there is no substantial study, which compares degree of decomposition among geographically different peatlands and its relation to respiration, therefore its role in peatland carbon flux could be overlooked.

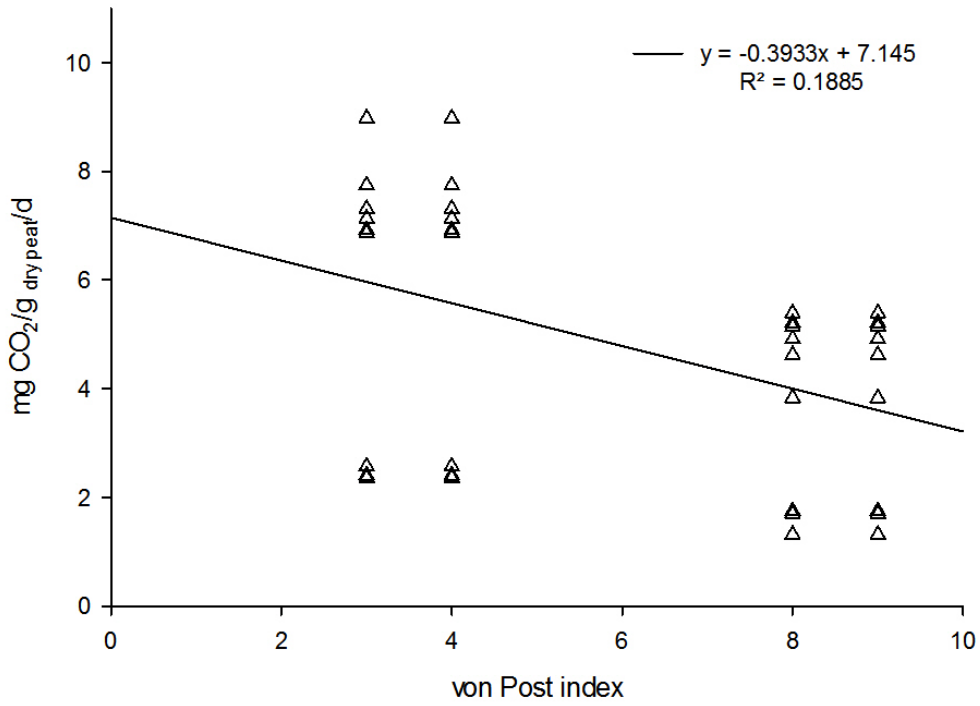

Figure 14: Relation between aerobic respiration and state of decomposition. On Y axis are mean values of all CO_2 emission samples from moss (bryophytic) and tree (ligneous) origin peat. X axis represent values associated with peat decomposition index according to classification by von Post, where 1 indicates plant litter with no signs of degradation and 10 is fully decomposed peat with no distinct texture.

Peat botanical origin is often linked to soil respiration. Previous studies have shown that peat made of vascular plants degrade faster and thus bring forth higher CO_2 emissions than mosses (Walker et al., 2016), decompose slower than mosses (Williams and Yavitt, 2003) or show no relationship with respiration whatsoever (Brouns et al., 2016). Borga et al. (1994) suggests that respiration rate differences in peat occur due to diversity of bacterial community. Respiration rate is not only affected by microbial community decomposition efficiency, but also the litter properties. Different plants form litter with different botanical and chemical properties, which affect microorganism ability to degrade it. Litter quality, which depends on plant type determines C:N ratio indicating *Sphagnum*-peat having high C:N ratio (Williams and Yavitt, 2003, Biester et al., 2013) and thus possibly high respiration if supplied with additional nutrients. Nutrient stoichiometry is an important factor to predict respiration potential, however litter chemistry has been acknowledged as well. Scheffer et al. (2001) compared litter from a peatland with *Sphagnum* dominated cover and a peatland covered with a mixture of sedges and *Sphagnum*. It was found, that sedge litter contains 8 – 29 times more water soluble

phenolics, but approx. 1.3 times less phenolic compounds in the solid litter fraction representing non-soluble phenolics. Coupling higher soluble and less non-soluble phenolics entails, that microorganisms decompose sedges quicker and easier as there is less toxic phenolics present. This study found that at first *Sphagnum* litter was decomposed more rapidly, but after the initial phase decomposition slowed down and remained very low, while sedges were decomposed at equal rate throughout the whole experiment.

Maximum respiration rate in this study was reached in moist peat. Moist bryophytic moss peat emitted 1.86 times more CO_2 than pine tree peat. Similarly dry moss peat produced more carbon dioxide emitting 1.36 times more than mountain pine peat. Litter quality and composition has not been in the center of attention in this study. It is possible that *Sphagnum*-peat had nutrient stoichiometry that fits microbial respiration preferences more than in ligneous peat or trees in this peatland contain even more phenolics than mosses. If phenolic compound effect is ruled out, carbon availability might the key factor controlling CO_2 outgassing. Previously discussed degree of decomposition might be relevant, as it represents the state of decomposition. Pine peat was attributed a higher index than moss peat implying that it has been degraded at larger extent. This suggests that there have been higher CO_2 emissions from mountain pine peat in the past, depleting majority of labile carbon pool leaving behind carbon forms, that are not recognized or preferred by microorganisms. *Sphagnum* litter being more resistant to degradation is still providing microbes with easily degradable carbon source, hence at the time of experiment respiration from moss peat was higher than from pine tree peat. This theory does not reflect any laboratory or field measurements, as this study focused on two other factors, that influence respiration – soil moisture and nutrient availability.

Moisture has long been recognized as a crucial factor influencing aerobic respiration in peatlands (Peterson et al., 1984, Moore and Knowles, 1989, Hogg et al., 1992, Waddington et al., 2002, Vien et al., 2010). Litter decomposition in peatlands is very slow in comparison to other ecosystems because of permanently high water table, which upon drought or draining is lowered and oxygen is introduced into new horizons advancing microbial activity. Ganie et al. (2016) reviewing soil respiration concluded that lower water table and thus less moisture initially will decrease microbial activity and reduce their variation as they are accustomed to drier conditions and will temporarily reduce soil respiration. On the other hand, microbes are capable of adaptation, therefore long-term drought might promote CO_2 emissions once soil microbes will be adapted (changes in community structure) and soil oxic area will be larger (expansion of habitat).

Peat with different moisture level emitted distinctly different level of carbon dioxide (see

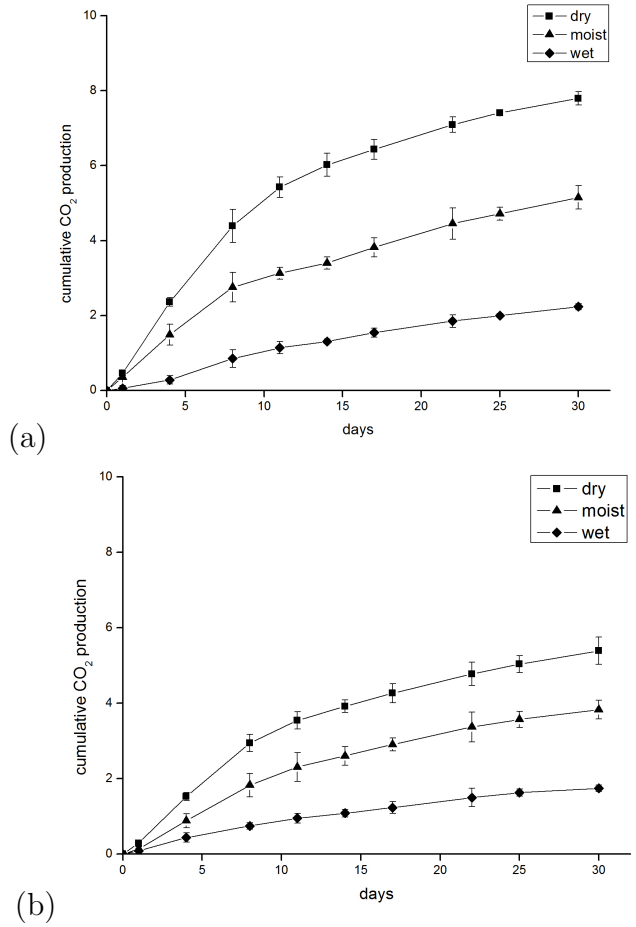

Figure 15: Cumulative carbon dioxide production rate. Y axis shows cumulative production rate in mg CO_2 $g_{\text{dry peat}}{}^{-1}$ day^{-1} (n = 4) ± SD. Dotted lines show carbon dioxide production rate at different moisture level, where dry is peat with FC 50%, moist peat is exposed to FC 100% and wet samples are over-saturated with water. (a) shows CO_2 production rate in *Sphagnum*-peat and (b) shows production rate in pine-peat.

Figure 15). In both peat types dryness increased aerobic respiration, agreeing with previous work indicating that dryer soil emits more CO_2 as more oxygen becomes available. More frequent and longer drought periods hold a great significance as climate change might caused recurring droughts and human induced land change drains natural peatland water table. It is unclear to what extent droughts will affect peat decomposition in Pürgschachen Moor because differences in water content between samples with 50% and 100% field capacity showed to be statistically indifferent. In a broader look, picturing the two dryer peat samples together they emitted around 3 times more than wet *Sphagnum*-peat and 2 to 3 times more in tree-peat. The results of this study confirm previously stated claims, that lower moisture boosts microbial activity and thus CO_2 emissions increase, however due to analytic inconsistencies it is difficult to estimate the exact moisture level at which peat bog would likely become source to the atmosphere.

As argued in chapter 1.2.5 peat bogs are nutrient poor, therefore additional nitrogen is believed to augment CO_2 emissions directly as a nutritional supplement to microbes and indirectly by causing shift in vegetation in favor of vascular plants, that are rapidly decomposed.

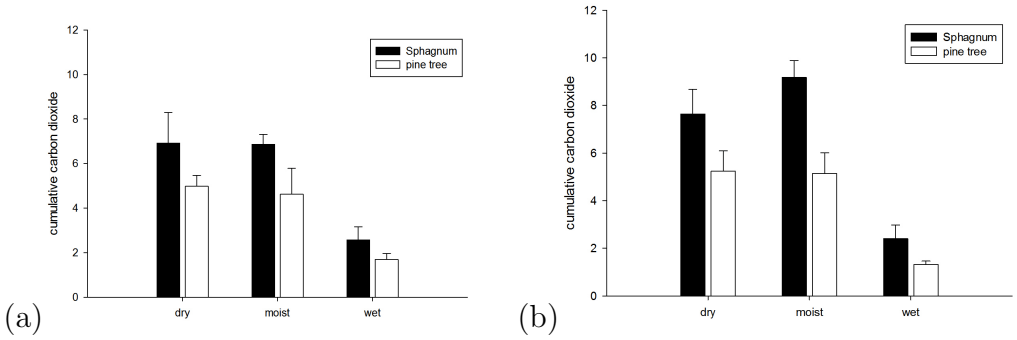

Figure 16: Comparison of carbon dioxide production at various levels of fertilization. Y axis shows cumulative CO_2 production over the whole incubation period in mg CO_2 g $ce_{dry\ peat}{}^{-1}$ day^{-1} and X axis compares CO_2 emission by water moisture. Black bars are *Sphagnum*-peat and white bars are pine tree-peat ± SD. Graph (a) shows cumulative CO_2 in lightly fertilized peat equivalent of 20 kg N ha^{-1} $year^{-1}$ and graph (b) shows heavy fertilized peat receiving 50 kg N ha^{-1} $year^{-1}$.

Fertilized peat produced more CO_2 than peat that received distilled water. CO_2 emissions also differed among samples with different water content. Peat adjusted to 100% field capacity resulted in maximum emission, but only in heavily fertilized samples (50 kg N ha^{-1} $year^{-1}$). Dryer peat did not show significant difference between lightly (20 kg N ha^{-1} $year^{-1}$) fertilized and non-fertilized samples. Lightly fertilized peat of both types emitted slightly less carbon dioxide, than control samples. *Sphagnum*-peat produced 5% less and tree-peat – 14% less CO_2. It is a relatively small difference, especially in the case of moss-peat, which implies that microbial activity will not increase even when the ombrotrophic peat bog will receive twice as much nutrients (atmospheric nitrogen deposition is approximately equivalent to lightly fertilized samples).

Heavily fertilized peat escalated microbial activity in moss peat, but did not have significant impact in mountain pine peat CO_2 emissions. Moss peat CO_2 emission in dry samples under heavy fertilization increased by 5%. This suggests, that respiration in moss peat was water, not nutrient driven. (Waddington et al., 2001) measured CO_2 emissions in a wide range of water saturation and concluded, that maximum production occurred at saturation of 79%, which exceeding microbial activity reduced. Consequentially, CO_2 emissions during drought would reduce despite of additional nutrient supply. CO_2 production in wet samples was overall very low, as was expected due to lower oxygen availability. However, these samples were not strictly anoxic and were exposed to laboratory air, therefore some CO_2 production took place in these samples as well. Ultimately peat during

drought (*dry* samples) and flooded (*wet* samples) conditions reduce aerobic respiration. Saturated *moist peat* increased microbial activity, but only when supplied with additional nutrients. Heavy fertilization increased respiration by 25%, therefore natural peatlands that do not undergo extreme water table changes are expected to increase subsurface peat when more nutrients become available.

Both peat types followed similar carbon dioxide emission pattern. Mountain pine peat also suppressed CO_2 emissions when lightly fertilized and increased again when fertilized heavily, however overall fertilization did not cause dramatic changes in respiration. Overall emissions of tree-peat were lower in all samples. Maximum CO_2 production in moss peat took place in water saturated and heavily fertilized samples and exceeded tree-peat emissions 1.8 times. Nutrient availability effect on microbial activity is uncertain, because different scenarios have been reported in other studies. Fertilization have caused increase (Zhang and Zak, 1998, Bragazza et al., 2006, Wang et al., 2015), decrease (Aerts and Toet, 1997, Aerts and de Caluwe, 1999, Brouns et al., 2016) or no effect (Lee and Jose, 2003). Reduced CO_2 production is usually explained by lowering of pH to a level, that is unfavorable for microbial enzyme activity. In this study fertilization did lead to a decrease of pH, consequentially suppressing CO_2 production in lightly fertilized peat, but did not play any role in heavily fertilized samples, which might have developed a different microbial community. However, statistically non-fertilized and lightly fertilized samples were indifferent, therefore overall heavy fertilization lead to increase in CO_2 production. Heavily fertilized samples might have altered C:N ratio, making more nitrogen available (Aerts et al., 1992). In long term it has the potential to reverse the role of peatlands in carbon cycling and become a contributor to atmospheric GHG concentration.

4.2 Anaerobic respiration

Anaerobic respiration in peat is detected by its product – methane. Anaerobic respiration results of this study depict net methane emissions, which is balance between methane production and methane consumption. Methane consumption can occur biologically by methane-oxidizing bacteria (methanotrophs) or by chemical oxidation when a methane molecule comes in contact with atmospheric oxygen.

Anaerobic respiration products are emitted at lower extent in comparison to aerobic respiration and is usually three orders of magnitude lower. Anaerobic to aerobic respiration ratio in this study yielded 1:1.9, which comes in good agreement with other studies. Anaerobic-to-aerobic respiration ratio has been reported from 1:1.2 (Williams, 1974) to

1:2.5 (Moore and Dalva, 1997).

Methane production was highest in water flooded (*wet*) samples, which agrees with previous work, because methane production requires minimal or even anoxic conditions, which in soil are achieved by saturating soil pores with water, hence methanogenesis in peat depends on water table height (Moore and Knowles, 1989, Kettunen et al., 1999, Strack et al., 2004, Dinsmore et al., 2009, Bhullar et al., 2009). Methane production was between 1.7 and 18.7 μg CH_4 g_{dw}^{-1} day^{-1} with cumulative means of 16.23 $\pm$1.49 μg and 7.40 $\pm$ 0.83 μg CH_4 g_{dw}^{-1} day^{-1} in moss and pine tree peat accordingly (see Figure 17). Dry samples emitted the least CH_4 in this experiment, producing 4.93 $\pm$2.12 μg and 2.19 $\pm$ 0.38 μg CH_4 g_{dw}^{-1} day^{-1}. In both peat types dry samples produced 3.3 times less CH_4 than in wet samples (see Figure 17).

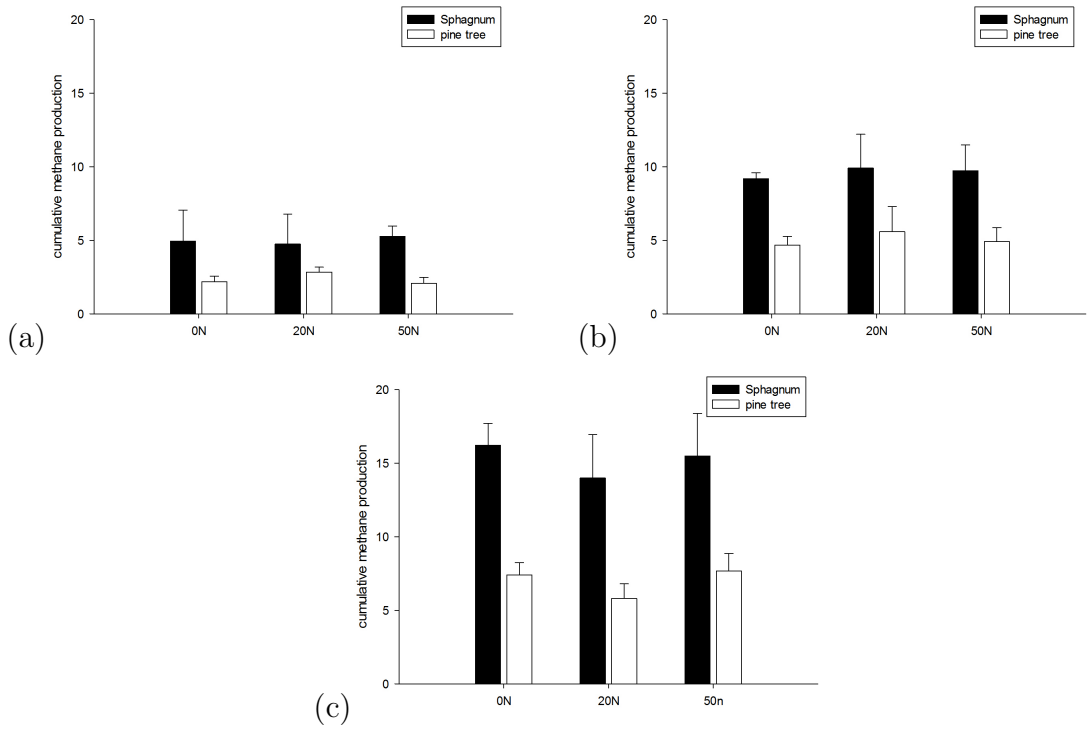

Figure 17: Cumulative methane production at various water saturation levels. Y axis shows cumulative mean methane emission in μg CH_4 g_{dw}^{-1} day^{-1} $\pm$ SD. X axis depicts methane emissions at different nutrient availability. (a) shows methane production in dry conditions, (b) shows methane cumulative production in moist conditions and (c) shows methane cumulative production in water flooded conditions.

Methane production cumulative rate in laboratory studies has been reported withing a great range varying from zero to several hundreds of micrograms. (Treat et al., 2014) reported laboratory methane production of 1.5 – 6.3 μg. It is slightly lower than in this study, however this experiment used permafrost peat, which due to low temperature is thought to have lower methanogen activity. In Williams and Crawford (1984) experiment

average methane production at 30 cm depth was 98 μg, which exceeds findings of this study. In a different study (Moore and Dalva, 1997) estimated methane emissions from a Canadian peatland resulting in average CH_4 production of 3.1 μg day^{-1}. Average daily methane emission in this study was 0.54 μg day^{-1}. Overall CH_4 emissions in this study were lower than in similar work. Lower emissions could have occurred due to the fact that this study examined acrotelmic peat layer, which is more likely to be oxygen, rather than water saturated, and inhibits methanogen activity. Moore and Dalva (1997) in their study also tested methanogenesis at different depth and found that CH_4 production in natural bog hummocks dropped by 2 orders of magnitude from surface peat to 35 cm depth in aerobic conditions. 35 cm bog peat in this study emitted only 0.01 μg CH_4 day^{-1}, however other studies report greater anaerobic respiration rates (Mettrop et al., 2014, Liu et al., 2016). Methane activity depends on water saturation so strongly, that the maximum methane production is considered to occur at water table height, decreasing above and below it. Methanogen activity also strongly correlates with temperature. (Hogg et al., 1992) incubated peat at three different temperatures 8° C, 16° C and 24° C. At the lowest temperature 8° C methane very low and almost non detectable, whereas peat incubated in the highest temperature emitted up to 0.8 μg CH_4 day^{-1}. Peat substrate has shown correlation with its botanical origin (Moore and Dalva, 1997). In this study *Sphagnum*-peat emitted 2.2 times more CH_4 than tree-peat.

4.3 Nitrogen mineralization

This study measured net nitrogen mineralization as net emission of nitrous oxide. Nitrogen mineralization is a microbial process and the rate depends on environmental conditions. Nitrogen mineralization increases with higher temperature, oxygen availability, ammonium and OM availability. C:N ration is of great importance. If nitrogen for microbial growth has been satisfied and more of it becomes available, microbes are expected to mineralize the excess nitrogen. Water availability also plays a crucial role. Maximum N_2O production occurs when water filled pore space is between 60% (Davidson, 1993) and 80% (Khalil et al., 2002). If WFPS is higher net N_2O emissions are inhibited because nitrogen consumption is more likely to take place and thus nitrous oxide emissions from saturated soils are very low or even negative. In this study all net nitrous oxide emission were positive, but production in dry samples was the highest, at times exceeding individual water flooded sample measurements by 1 to two orders of magnitude.

Nitrous oxide production ranged from 0.2 to 1017 ng day^{-1} with average cumulative production 1596 ... 6420 ng N_2O ${g_{dw}}^{-1}$ day^{-1}. Maximum production occurred in moist

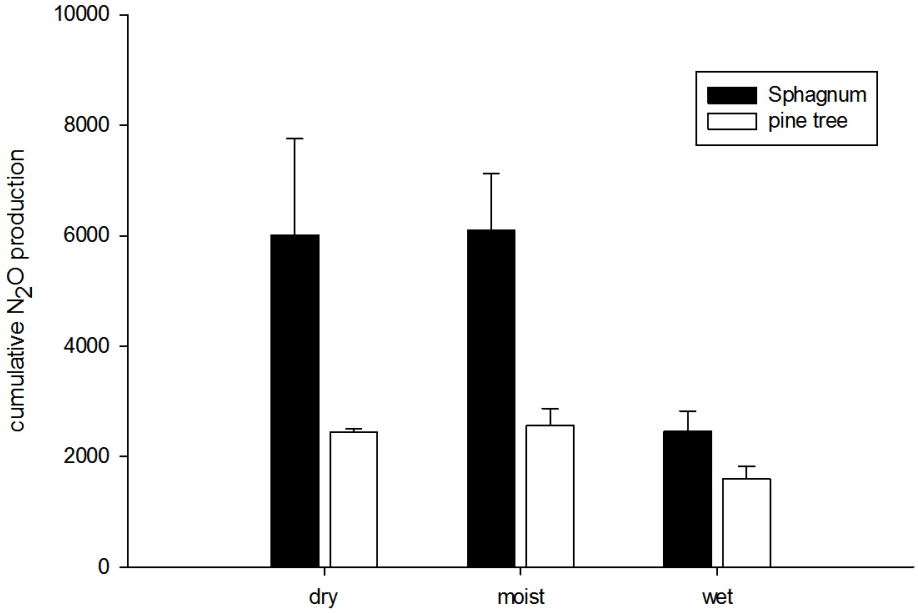

Figure 18: Cumulative nitrous oxide production at various water availability. Y axis shows cumulative nitrous oxide production in ng N_2O $g_{dw}{}^{-1}$ day^{-1} ±SD. X axis shows N_2O emissions in unfertilized peat at different water content and botanical origin.

conditions, where *Sphagnum*-peat produced 6106 ± 1021 and pine tree peat 3192 ± 1174 ng N_2O $g_{dw}{}^{-1}$ day^{-1} cumulatively (see Figure 18). N_2O emissions in this experiment are in good agreement with similar studies.

In this experiment nitrous oxide in dry and moist peat produced 1.5 to 2.5 times more N_2O than in water flooded samples regardless of origin, demonstrating that nitrogen mineralization preferably takes place in oxygenated conditions. Maximum N_2O production in this study occurred in moist samples at field capacity of 100%. FC represents amount of water soil can maintain after drainage, whereas WFPS most often used to study nitrous oxide flux in soils, shows proportion of soil pore that are filled with water and thus deals with volume, unlike field capacity, which is expressed as weight. This study used measure of field capacity to determine and adjust water content in samples, therefore direct conversion to WFPS is not possible due to lack of knowledge about particle density and porosity. Form of nitrogen also plays role in its mineralization potential. This study supplied samples with nitrogen in form of ammonium nitrate (NH_4NO_3), consisting of organic $NH_4{}^+$ and inorganic $NO_3{}^-$ nitrogen species of which ammonium ($NH_4{}^+$) is preferably taken up by microbes advancing denitrification (Jauhiainen et al., 1998). Nitrate in natural peatlands is in low abundance and microbes are not accustomed to utilize it, therefore nitrate is not expected to alter microbial decomposition rate. Nitrate, however, is pre-

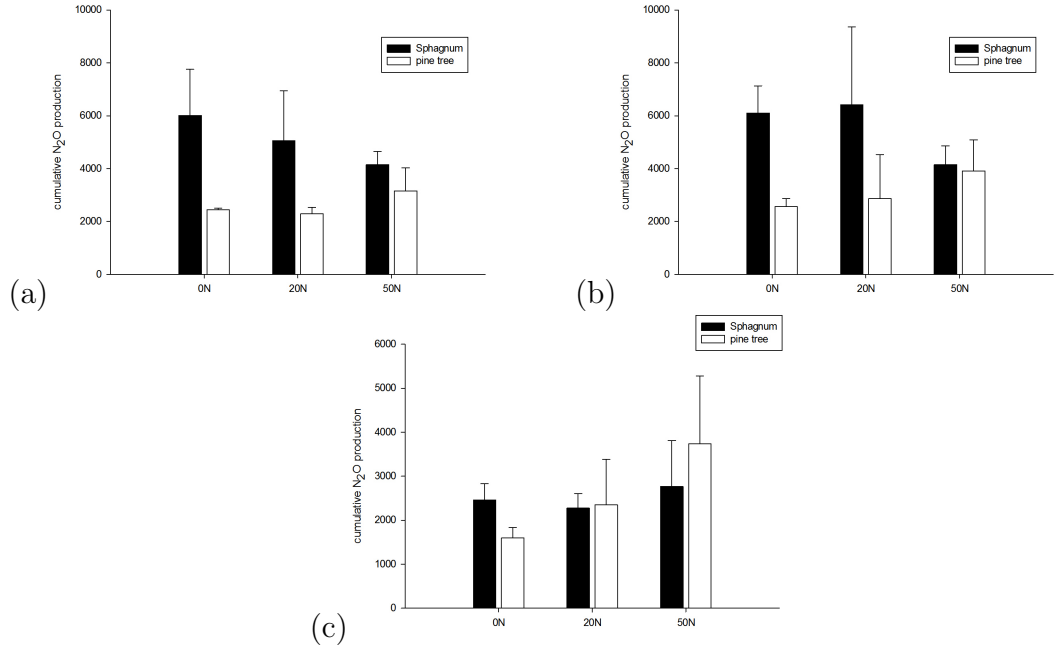

Figure 19: Cumulative nitrous oxide production at various water saturation levels. Y axis shows cumulative mean methane emission in ng N_2O ${g_{dw}}^{-1}$ ${day}^{-1}$ $\pm$ SD. X axis depicts methane emissions at different nutrient availability. (a) shows N_2O production in dry peat, (b) shows N_2O emissions from moist peat and (c) shows N_2O emissions from wet, water-flooded peat.

ferred form of nitrogen for plants, therefore fertilization with ammonium nitrate in field conditions might enhance both microbial and vascular plant biomass, increasing overall carbon mineralization.

Overall GHG emissions in this study were the highest in heavily fertilized, moist moss-peat, in both actual measured GHG emission values and when adjusted to global warming potential (GWP) CO_2 equivalents. Water flooded samples had 2 – 3 times lower overall emissions compared to dryer samples, suggesting that during short-term floods help to maintain plant litter decomposition in pristine peatlands low and preserve their role as carbon sink. On the other hand, high water table increases methane emissions, which have 28 times stronger greenhouse potential than CO_2. In *Sphagnum*-peat methane emissions were 11 – 13% of the total emissions and in pine-peat methane contributed to 6 – 8% of total emissions. Dy and moist peat methane emissions composed only up to 2% of total emissions. On the other hand, dryer samples emitted more nitrous oxide, which has the strongest GWP effect of analyzed gases, therefore adjusted to match CO_2 equivalent under all circumstances has larger GWP than methane (see Table 17). Overall pristine bogs appear to produce more microbial GHGs under dryness, however microbial processes have an optimal range of soil moisture, therefore extreme dryness might inhibit respiration, which in this study appeared as reduction in GHG emissions from samples at FC 50%.

Fertilization did not appear to have strong effect when ammonium nitrate at rate of 2 g N $m^{2\ -1}$ $year^{-1}$ was applied, but increased when heavily fertilized. Fertilization in this study appeared to increase respiration rates, but not linearly, suggesting microbial respiration in peatlands is a more complex process, that depends on various factors and their interactions and different microbial communities might exist and develop differently in laboratory studies, which could be examined more closely.

Table 17: Emissions from peat converted to CO_2 equivalents in 100 year time scale. CO_2, CH_4, N_2O columns show equivalent in CO_2 using equivalence factors from IPCC 5th report (2014). Values show relative GHG emissions as g CO_2 equivalent per kg soil.

Site	Nutrients	Water content	CO_2	CH_4	N_2O	Total
Moss	atmospheric	dry	7.31	0.13	1.59	9.04
	deposition	moist	7.13	0.26	1.62	9.01
		wet	2.36	0.45	0.65	3.46
	20 kg N/ha/a	dry	6.92	0.13	1.34	8.39
		moist	6.86	0.28	1.70	8.84
		wet	2.57	0.39	0.60	3.56
	50 kg N/ha/a	dry	7.63	0.15	1.09	8.88
		moist	9.17	0.27	1.09	10.54
		wet	2.40	0.43	0.73	3.57
Mountain	atmospheric	dry	5.39	0.06	0.65	6.09
pine	deposition	moist	3.83	0.13	0.68	4.63
		wet	1.75	0.21	0.42	2.38
	20 kg N/ha/a	dry	4.98	0.08	0.61	5.66
		moist	4.62	0.16	0.76	5.54
		wet	1.69	0.16	0.62	2.48
	50 kg N/ha/a	dry	5.23	0.06	0.84	6.13
		moist	5.14	0.06	1.04	6.31
		wet	1.31	0.06	0.99	2.52

5 Conclusions and Outlook

This study represents undisturbed peat soil microbial activity in laboratory conditions and as such it represents relative carbon mineralization rates as a response to different environmental conditions. Outcome of such studies are potential production rates rather than estimation of exact real world field fluxes and are not directly applicable and comparable to field flux studies, however it is a stepping stone in understanding peat soil microbial activity changes in diverse and changing conditions and can be benefited from to interpret gas flux measurements *in situ* or to study isolated soil microbial activity interactions with various factors disregarding plant and rhizome respiration.

Main attention of this study was observation of peat microbial decomposition. Microorganisms thrive in certain physical and chemical conditions. Main factors determining microbial growth are temperature, availability of carbon rich organic matter for energy production, water and nitrogen availability. This study investigated how changes of the latter two may affect nutrient exchanges between terrestrial ecosystems and atmosphere. It was found that drying of organic matter rich peat enhances carbon dioxide and nitrous oxide emissions into atmosphere, which both have been linked to global warming, especially the latter – nitrous oxide, capturing long wave solar radiation 298 times more efficient than carbon dioxide. Methane, absorbing 28 times more solar radiation that carbon dioxide, production takes over when water level in peatlands is high. Despite outgassing of CO_2, CH_4 and N_2O from terrestrial systems, peatlands store more plant litter carbon than return to atmosphere. However, drying and additional microbial growth stimulating nitrogen addition due to human activities in this study have shown to have a potential to increase GHG emissions. Even though drying alone did not intensify CO_2, N_2O emissions, GHG emissions from water saturated peat were 1.4 ... 1.9x lower. Fertilization enhanced emissions even further, indicating overall more rapid peatland degradation and GHG emissions upon fertilization. Moreover, GHG emissions differed, depending on peat botanical origin. Predominantly less degraded moss derived peat emitted more overall GHGs than more degraded mountain pine, however no correlation between level of peat degradation and GHGs emissions was detected.

Laboratory studies are suitable to determine environmental controls on microbial activity and can be upscaled to modeling and field studies. Under changing climate and human activities, droughts and land use transformation might reduce natural peatlands all over the world. As significant terrestrial carbon store, preservation of peatsites in their pristine condition is one of the key factors in slowing down the pace of atmospheric temperature rise. Peatland response to changing environmental conditions with respect to carbon

and nitrogen have been studied before, however examination of multiple factors together can be perplexing. This study shows, that not only physio-chemical factors determine peatland degradation, but also plant litter botanical composition may affect rate of GHG emissions. Laboratory studies are a measure of understanding basic responses of microbial activity to changing environmental factors, however GHG production rate determined at these type of studies are relative and to model a more realistic gas flux model, field studies and additional laboratory experiments are a mandatory next step.

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Abstract

Despite covering only 3% of landmass, peatlands, store about a third of total terrestrial carbon (Baird et al., 2013). Droughts, drainage, and land use change accelerates peat degradation and releases greenhouse gases into atmosphere. Peat sensitivity to changing conditions is complex and depends on interactions of environmental controls, such as temperature, organic matter, water level, and nitrogen availability of which the latter two are analyzed in this study.

In this paper, short-term microbial bog peat degradation was studied in a laboratory environment to compare greenhouse gas contribution of two different botanical compositions, i.e. *Sphagnum spp.* and *Pinus mugo*, under varying moisture content and fertilizer exposure scenarios.

Carbon dioxide release increased in peat subjected to drought like conditions by 55% in vascular peat and only 2.5% in moss peat suggesting that microbial community and not litter quality might play role in decomposition. Fertilization intensity did not increase carbon dioxide production linearly, however, under heavy nitrogen addition it increased 1.28 times in moss peat and 1.34 times in vascular plant peat. On the other hand, when methane and nitrous oxide production are taken into account, the highest greenhouse gas production in this study took place in heavily fertilized moss peat. Converted greenhouse gas emission to carbon dioxide equivalents showed, that moss peat produced 50% more than pine peat site, suggesting that pristine, moss dominated peatlands are highly sensitive to changes in environmental controls.

Abstract

Obwohl Torfgebiete nur 3% der Landmasse bedecken, lagern sie etwa ein Drittel des gesamten terrestrischen Kohlenstoffs (Baird et al., 2013). Dürren, Entwässerung und Landnutzungsänderungen beschleunigen den Torfabbau und setzen Treibhausgase in die Atmosphäre frei. Die Empfindlichkeit von Torf gegenüber sich ändernden Bedingungen ist komplex und hängt von den Wechselwirkungen der Umweltkontrollen wie Temperatur, organischer Substanz, Wasserstand und Stickstoffverfügbarkeit ab, von denen die beiden letzteren in dieser Studie analysiert werden.

In dieser Arbeit wurde der kurzzeitige mikrobielle Moortorfabbau in einer Laborumgebung untersucht, um den Treibhausgasbeitrag von zwei verschiedenen botanischen Zusammensetzungen, d. H. *Sphagnum spp.*, Zu vergleichen. und *Pinus mugo* unter verschiedenen Feuchtigkeits- und Düngemittel-Expositionsszenarien.

Die Kohlendioxidfreisetzung erhöhte sich bei trockenem Torf um 55% und bei Moostorf nur um 2,5%, was darauf hindeutet, dass die mikrobielle Gemeinschaft und nicht die Qualität der Abfälle bei der Zersetzung eine Rolle spielen könnten. Die Düngungsintensität erhöhte die Kohlendioxidproduktion nicht linear, stieg jedoch unter starker Stickstoffzugabe bei Moostorf um das 1,28-fache und bei Gefäßpflanzentorf um das 1,34-fache. Unter Berücksichtigung der Methan- und Lachgasproduktion wurde in dieser Studie die höchste Treibhausgasproduktion in stark gedüngtem Moostorf erzielt. Die umgerechneten Treibhausgasemissionen in Kohlendioxidäquivalente zeigten, dass Moos-Torf 50% mehr produzierte als Kiefern-Torf.