

Dynamics of soil carbon and nitrogen in changing boreal environments

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*The soil is the great connector of lives, the source and destination of
all.*

*It is the healer and restorer and resurrector, by which disease passes
into health, age into youth, death into life.*

*Without proper care for it we can have no community, because without
proper care for it we can have no life.*

–Wendell Berry–

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LIST OF ORIGINAL ARTICLES

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

- I. Laine, M., Strömmer, R. and Arvola, L. 2013: Nitrogen Release in Pristine and Drained Peat Profiles in Response to Water Table Fluctuations: A Mesocosm Experiment. *Applied and Environmental Soil Science* 2013: 1–7. Article ID 694368.
- II. Laine, M., Strömmer, R. and Arvola, L. 2014: DOC and CO₂-C Releases in Pristine and Drained Peat Soils in Response to Water Table Fluctuations: A Mesocosm Experiment. *Applied and Environmental Soil Science* 2014: 1–10. Article ID 912816.
- III. Laine, M., Rütting, T., Alakukku, L., Palojärvi, A. and Strömmer, R. Process rates of nitrogen cycle after harvesting in no-till and ploughed agricultural clay soils. *Manuscript*.

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THE AUTHOR'S CONTRIBUTIONS

Merjo Laine (ML) is the corresponding author in all articles of this thesis and wrote the articles with contributions from the other authors. Concerning all three papers, ML was responsible for the samplings, and for taking care of the experimental setups. ML personally performed a considerable part of the samplings and laboratory analyses. Part of the laboratory analyses were performed by the laboratory staff, and part were performed by the guidance of ML. ML performed all the statistical analyses for all three papers. ML also contributed to the individual studies as follows:

- I. and II. ML participated in establishing the experiment together with Rauni Strömmer (RS) and Lauri Arvola.
- III. ML participated establishing the experiment together with Tobias Rütting and RS. ML performed the data modeling.

ABSTRACT

Anthropogenic actions and climate change greatly affect e.g. carbon (C) and nitrogen (N) cycles in soils. The consequences can differ in various soil types. The changes in soil C and N cycles may also have an effect on adjacent aquatic systems and in the atmosphere. Dissolved organic carbon (DOC), N, and phosphorus (P) loads to aquatic ecosystems in general have caused concern. In this thesis, I mostly discuss soil C and N cycles. To a minor extent I also cover C and N in aquatic ecosystems and in the atmosphere, as they are connected to soil cycles. My study involved two separate experimental areas, a peatland and an agroecosystem. In my peat soil mesocosm experiment, I studied peat profiles taken from a complex with a pristine and a forestry-drained peatland. My focus was on changes in DOC, dissolved organic nitrogen (DON), and ammonium (NH_4^+) concentrations in soil water as a response to hydrological manipulation. In my *in situ* mineral soil agroecosystem experiment I quantified gross N transformation process rates in no-till and moldboard-ploughed soils after harvesting.

Hydrology remarkably impacted the element concentrations in pristine peat soil water. An increase in DON and NH_4^+ concentrations was seen as a response to hydrology, while DOC concentrations were not affected in comparison to control concentrations. However, DOC production in pristine peat, followed by its release into water, was also high enough to compensate the dilution caused by water additions to the mesocosms. These compounds were produced during the drought in the aerated soil layer and released to the added water by physicochemical processes when the mesocosms were rewetted. In drained peat mesocosms, the hydrological manipulation decreased the DOC concentrations, and the DON and NH_4^+ concentrations did not change significantly.

My agricultural experiment results give some environmental support for no-till over ploughing. NH_4^+ is a substrate for nitrification, and nitrate (NO_3^-) can easily leach into aquatic ecosystems, where it may cause eutrophication. Therefore, the observed higher gross immobilization rate, lower nitrification rate, and lower NO_3^- loss flux rate in no-till supported this practice when assessing the post-harvest leaching risk. In addition, a lower nitrification / immobilization ratio in the no-till indicated a decreased NO_3^- leaching risk. Higher post-harvest immobilization rate further supports no-till because it may be beneficial for crop growth during the following growing season.

TIIVISTELMÄ

Ihmistoiminnot ja ilmastonmuutos vaikuttavat suuresti esimerkiksi hiilen ja typen kiertoihin maaperässä. Seuraukset voivat olla erilaisia eri maatyypeillä. Maaperän hiilen ja typen kiertojen muutokset vaikuttavat myös vesistöihin ja ilmakehään. Liuenneen orgaanisen hiilen (DOC), typen sekä fosforin vesistökuormitus ylipäättään on aiheuttanut huolta. Käsittelen väitöskirjassani etupäässä maaperän hiiltä ja typpeä. Lisäksi sivuan vesistöjen ja ilmakehän hiiltä ja typpeä, sillä ne ovat yhteydessä maaperän ainekiertoihin. Tutkimukseni sisälsi kaksi erillistä aluetta, turvemaan ja maatalousmaan. Turvemaan malliekosysteemikokeessa tutkin turveprofilleja, jotka haettiin suokompleksilta, jossa on luonnontilainen ja ojitettu osa. Keskityin DOC:n, liuenneen orgaanisen typen (DON), sekä ammoniumin (NH_4^+) pitoisuuksien muutoksiin suovedessä vasteena hydrologiselle manipulaatiolle. Maatalousmaakokeessani (mineraalimaa) kvantifioin *in situ* typen kierron prosessien bruttonopeuksia suorakylvetyllä ja kynnetyllä viljapelloilla sadonkorjuun jälkeen.

Hydrologialla oli suuri vaikutus aineiden pitoisuuksiin luonnontilaisen turpeen vedessä. Havaitsin DON:n ja NH_4^+ :n pitoisuuksien kasvua vasteena hydrologiselle manipulaatiolle. DOC:n pitoisuuksiin sillä ei ollut vaikutusta kontroleihin verrattuna. Kuitenkin myös DOC:n tuotanto luonnontilaisessa turpeessa oli riittävän suurta kompensoidakseen laimenemisvaikutuksen, joka aiheutui vesililäyksistä malliekosysteemeihin. Näiden yhdisteiden tuotanto lisääntyi kuivan kauden aikana aerobisessa maakerroksessa, ja ne vapautuivat malliekosysteemeihin myöhemmin lisättyyn veteen. Ojitetun turpeen uudelleen vettyessä DOC pitoisuus laski, eivätkä DON:n ja NH_4^+ :n pitoisuudet muuttuneet merkitsevästi.

Maatalousmaakokeen tulokset tukivat suorakylvöä kynnön sijaan ympäristönäkökulmasta tarkasteltuna. NH_4^+ toimii substraattina nitrifikaatiolle, ja nitraatti (NO_3^-) huuhtoutuu helposti vesistöihin, joissa se saattaa aiheuttaa rehevöitymistä. Täten havaitsemani suorakylvön korkeampi immobilisaatio-nopeus (brutto) sekä alhaisemmat nitrifikaatio- ja NO_3^- :n poistumavirtausnopeudet antoivat tukea suorakylvölle sadonkorjuun jälkeisen ravinnehuuhtoutumisen näkökulmasta tarkasteltuna. Lisäksi pienempi nitrifikaatio/immobilisaatio -suhde suorakylvöllä indikoi alhaisempaa NO_3^- :n huuhtoutumisriskiä. Nopeampi sadonkorjuun jälkeinen immobilisaatio tukee suorakylvöä myös siksi, että se saattaa hyödyttää viljelykasveja seuraavana kasvukautena.

ABBREVIATIONS

C	Carbon
CH ₄	Methane
CO ₂	Carbon dioxide
DOC	Dissolved organic carbon
DOP	Dissolved organic phosphorus
DON	Dissolved organic nitrogen
DrCtrl	Drained, control
DrFluc	Drained, fluctuating water table level
GH(No.)	Gas sampling, high water table level, calendar week number in parentheses
GL(No.)	Gas sampling, low water table level, calendar week number in parentheses
H ₂ O	Water
H ₂ CO ₃	Carbonic acid
N	Nitrogen
¹⁴ N	Isotope 14 of nitrogen
¹⁵ N	Isotope 15 of nitrogen
N ₂	Nitrogen gas
NH ₃	Ammonia
NH ₄ ⁺	Ammonium
¹⁵ NH ₄ NO ₃	¹⁵ N-labeled ammonium nitrate, label in NH ₄
NH ₄ ¹⁵ NO ₃	¹⁵ N-labeled ammonium nitrate, label in NO ₃
NO ₂ ⁻	Nitrite
NO ₃ ⁻	Nitrate
N ₂ O	Nitrous oxide
OM	Organic matter
P	Phosphorus
PO ₄ ³⁻	Phosphate
PrCtrl	Pristine, control
PrFluc	Pristine, fluctuating water table level
TOC	Total organic carbon
TON	Total organic nitrogen
WH(No.)	Water sampling, high water table level, calendar week number in parentheses

1. INTRODUCTION

1.1 Biogeochemical element cycles

The most abundant elements in living cells are hydrogen (H), carbon (C), nitrogen (N), oxygen (O), phosphorus (P), and sulfur (S) (Alberts *et al.* 2002). Element cycles in ecosystems are strongly linked to each other (Xu *et al.* 2011) through the hydrological cycle, micro-organisms, and other organisms and their metabolism. Micro-organisms are crucial in the element cycles of C (Bradford 2013), N (Xu *et al.* 2011), S (Muyzer and Stams 2008), and also in the P cycle (Tamburini *et al.* 2012), although the P cycle is largely based on chemical processes. While P availability is limiting for algal growth in aquatic systems, it is N availability that often limits plant growth in boreal terrestrial ecosystems. A schematic presentation of the C, N, and P cycles is given in Fig. 1.

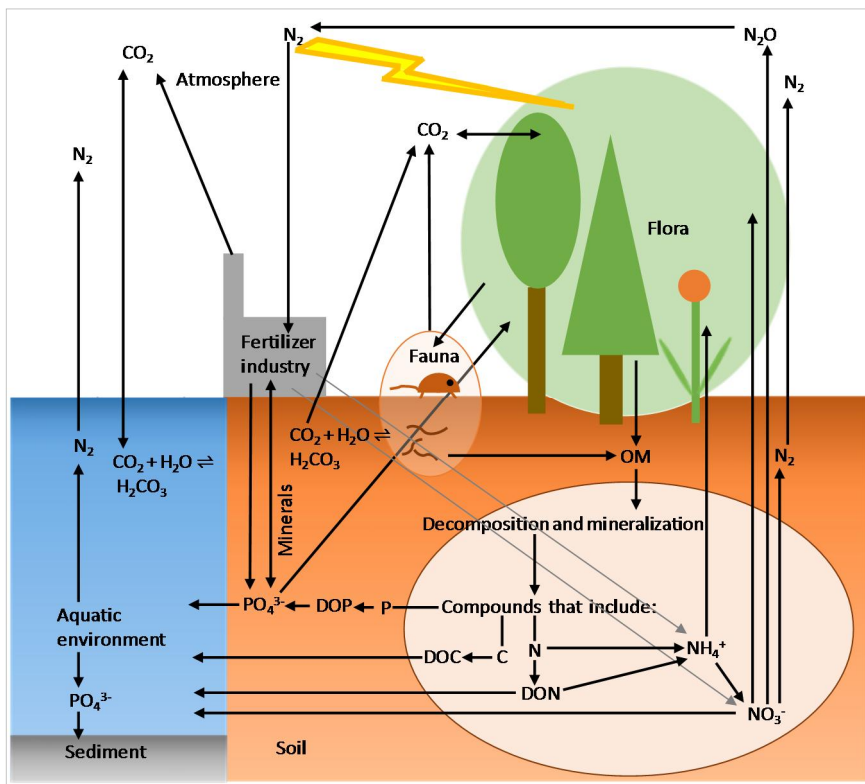


Figure 1. Schematic presentation of the C, N, and P cycles. The main focus of this thesis is presented within the pale circle in the lower right corner. Abbreviations are given in the “Abbreviations” chapter. Drawing by M. Laine.

In general, more knowledge is needed concerning the impacts of a changing environment on element cycling and biogeochemistry. Anthropogenic actions alter the atmospheric, terrestrial, and aquatic environment in many ways. An example of an untargeted environmental change is the globally changing climate that causes increased emissions of carbon dioxide (CO₂), a byproduct of industrial burning reactions. On the other hand, changes in land use are often performed intentionally to gain more land area for targeted purposes such as agriculture. However, the consequences of land use intensification can be desirable, e.g. improved soil quality for crop growth, but also undesired, e.g. accelerated soil erosion because of soil tillage. Also, it is interesting whether climate factors, such as a fluctuating water table level, have a more significant effect on human-impacted soils or more natural soils. My thesis uses a versatile approach to this theme as I am dealing with the impacts on soil processes by both land use intensification and climate change. It is important to understand nutrient cycle mechanisms in an all-round manner to act in a reasonable way e.g. when making restoration decisions or finding an explanation e.g. for the deterioration of an ecosystem.

The type of land use and changes in soil properties due to climatic factors affect element cycles and the release of elements from soil to soil water, and further to aquatic systems. For assessing the processes and factors involved in element cycles in soils, it is necessary to understand the risk of nutrient release from soil to the hydrosphere and atmosphere. As C and N in the soil are mostly in organic matter (OM), changes in the decomposition rate of OM will have an impact on nutrient loading to water systems through run-off. An indicated precipitation increase in the Baltic areas through climate change would cause increasing land run-off of allochthonous, i.e. terrestrially produced, OM and nutrients affecting the Baltic Sea ecosystem (Meier *et al.* 2012; Andersson *et al.* 2015). This would cause large changes in the export and flows of elements from the soil to aquatic systems in the Baltic area.

Anthropogenic actions have strongly altered the N cycle, leading to many global and local environmental changes such as eutrophication and climate change (Shibata *et al.* 2014). Anthropogenic impacts on the N cycle in various ecosystems reflect back to human society e.g. as changes in ecosystem services. The effects of different soil management practices on the nutrient cycle are important to recognize, as N load into aquatic systems has increased in recent decades (Vitousek *et al.* 1997; Galloway *et al.* 2003), and because N availability is crucial for plant growth. Peatland coverage is an

important factor explaining for catchment dissolved organic carbon (DOC) (Mattsson *et al.* 2005; Kortelainen *et al.* 2006), total organic nitrogen (TON) and ammonium (NH₄⁺) exports (Kortelainen *et al.* 2006). Mattsson *et al.* (2005) reported also that agricultural area is important in explaining TON export on a catchment level.

1.1.1 The carbon cycle

Only a very minor percentage of the atmosphere consists of C compounds. CO₂ is by far the most common one, with current CO₂ concentrations being approximately 380 ppm. It is an essential gas for life, as photosynthesizing organisms, i.e. plants and algae, require CO₂ for producing oxygen gas (O₂) and carbohydrates, a component of OM. CO₂ also dissolves and is stored in water (chemical equilibrium: CO₂ + H₂O ⇌ H₂CO₃). CO₂ from the soil is returned to the atmosphere mostly through the cellular respiration of organisms and by natural and anthropogenic burning.

Globally, soils contain more C than the vegetation and atmosphere combined together (Swift 2001). Soil OM stocks (soil C sequestration) depend on the balance between net primary productivity and litterfall (including below ground) on the one hand and decomposition (OM quality) on the other. Both net primary productivity and decomposition depend on climate and soil aeration status (Swift 2001). In anaerobic conditions the decomposition of OM is slow, and anaerobic soils may therefore form long-term C storages. Peatlands are globally significant C storages (Gorham 1991), so they have an important role in the global C cycle. Over two thirds of the C reservoir of Finnish ecosystems is in peat (Kauppi *et al.* 1997). In improved aerobic conditions, where decomposition is faster, CO₂ is returned to the atmosphere. A great deal of seasonal variation may occur in the importance of various C compound fluxes e.g. in Finnish peatlands, as the net ecosystem exchange of CO₂ in particular varies from negative to positive during a year (Gažovič *et al.* 2013). C loss is also an important topic in agriculture, as the conversion from natural to agricultural ecosystems (Lal 1999) or from perennial to annual crops (Heikkinen *et al.* 2013) cause C loss from soil.

Nearly all C in living organisms and OM originates from atmospheric CO₂. DOC consists of organic molecules of varying size (but technically <45 μm) and complexity. DOC export is an important part of the C output from peatlands (Jager *et al.* 2009; Olefeldt *et al.* 2013). DOC is leached from soil

and transported to aquatic systems with run-off. On a landscape level, peatland coverage is an important factor determining the total organic carbon (TOC) of aquatic ecosystems in Finland (Kortelainen 1993; Kortelainen *et al.* 2006). A drastically higher annual TOC load has been observed in southern Finland during a rainy year in comparison to a dry year (Einola *et al.* 2011). Ditch maintenance in Finnish peatland forests is observed to decrease ditch-water DOC at least for the first few years after drainage (Joensuu *et al.* 2002).

Terrestrial DOC is an important C source in boreal aquatic ecosystems. An average of 94% of the TOC in Finnish rivers has been observed to be in the form of DOC (Mattsson *et al.* 2005). Allochthonous DOC is an important energy source for bacteria in planktonic food chains (Jones 1992; Drakare *et al.* 2002). Allochthonous C input is always higher than autochthonous C input in humic lakes (Tulonen 2004). Lake bacterial production (Drakare *et al.* 2002; Lennon and Pfaff 2005; Berggren *et al.* 2007) and bacterial respiration (Drakare *et al.* 2002) have been shown to be affected by DOC inputs and DOC concentrations. Tulonen *et al.* (1992) observed a strong correlation between DOC concentration and the bacterial growth rate in a highly humic lake in southern Finland. Nutrient cycle rates in lakes are thus affected by allochthonous DOC, which may even increase the risk of eutrophication (Räsänen *et al.* 2014).

1.1.2 The nitrogen cycle

Nitrogen gas (N_2) constitutes 78% of the atmosphere by volume. N is a common element in organic compounds. For example, amino acids, nucleic acids and chlorophylls all contain N. Although N is abundant in the air, N_2 gas is relatively non-reactive and organisms cannot use it as such. N_2 must therefore be transformed into biologically available forms such as NH_4^+ and nitrate (NO_3^-) ions. This transformation is performed by micro-organisms, lightning, or industrial processes (fertilizer production). Wet and dry depositions of N from the air are important sources of N for soils. NH_4^+ , NO_3^- , and organic N can settle on the ground from the atmosphere as either wet (Jickells *et al.* 2013; Sickles and Shadwick 2015) or dry deposition (Russell *et al.* 2003).

Biologically available N has an important role in primary production (Gruber and Galloway 2008). In soil, ammonia (NH_3) is produced from atmospheric N_2 by symbiotic or free-living soil N-fixing microbes or from the

decomposition of OM (mineralization) by ammonifying microbes. NH_3 dissolves in water to produce NH_4^+ ions. Depending on species, plants can use N as NH_4^+ , NO_3^- , or in organic form. Nitrifying microbes oxidize NH_4^+ into nitrite (NO_2^-) ions and further into NO_3^- ions (nitrification). N immobilization is the opposite process to mineralization, i.e. mineral N is converted to organic compounds by micro-organisms or by plants. In addition to mineralization, nitrification, and immobilization, several other soil processes are involved in the N cycle. Some of these processes can be further separated by the substance that is changed in the process, such as the immobilization of NH_4^+ or immobilization of NO_3^- into recalcitrant organic N (Müller *et al.* 2014). N_2 is returned to the atmosphere through the process of denitrification, in which NO_3^- ions are converted to nitrous oxide (N_2O) and further into N_2 by denitrifying bacteria.

Many plant species obtain part of the N in organic form with the aid of mycorrhizal fungi, i.e. by bypassing mineralization (Näsholm *et al.* 1998; He *et al.* 2003). Mycorrhizal fungi also take up mineral ions from the soil (He *et al.* 2003; Read and Perez-Moreno 2003; Govindarajulu *et al.* 2005). He *et al.* (2003) stated that arbuscular mycorrhizal fungi transfer N from one plant to another, but Govindarajulu *et al.* (2005) have challenged this concept. Instead, they suggested that organic substances are broken down into inorganic N within the fungus, and transferred to the host plants.

Certain archaea are also involved in the N cycle. Archaea that are able to oxidize NH_3 are found in both aquatic and terrestrial ecosystems (Nicol and Schleper 2006; Prosser and Nicol 2008). The relative importance of NH_3 -oxidizing archaea and NH_3 -oxidizing bacteria depend on the physical and chemical properties of soil (Levičnik-Höfferle *et al.* 2012; de Gannes *et al.* 2014; Muema *et al.* 2015), and the relative importance of these microbes may vary in different soils (Levičnik-Höfferle *et al.* 2012).

1.1.3 Linkages between element cycles

Linkages between element cycles in the soil are complicated, and the importance of various processes are dependent on the abiotic environment and biotic community, and on their interactions with positive and negative feedbacks. For example, mineralization rate can be higher in one soil type than in another because of differing microbial communities, and this may further affect the importance of other processes. Biologically available N,

atmospheric CO₂ concentrations, and land use change all impact primary production, which in turn impacts atmospheric CO₂ concentrations (Gruber and Galloway 2008) and C sequestration (Zhao *et al.* 2011; Huang and Deng 2016).

The dynamics of soil C and N are strongly linked to each other because they are both important constituents of OM. Peat is nearly entirely organic plant debris and in agriculture the soil OM content is strongly related to crop growth and production. Soil C content and the C/N ratio affect soil N cycle process rates (Rochester *et al.* 1992; Barrett and Burke 2000; Romero *et al.* 2015) because available organic C enhances microbial growth (Romero *et al.* 2015) and N cycle processes are dependent on soil micro-organisms. Furthermore, N enrichment affects microbial community composition (Farrer *et al.* 2013) and N transformation processes affect concentrations of available N.

Available N and P promote the growth of plants and micro-organisms, which then have an effect on the decomposition of OM, and therefore on C sequestration. However, increasing N availability does not necessarily alter the decomposition dynamics of OM even if soil microbial activity was initially N-limited (Weintraub and Schimel 2003). Also, an increase in N availability may have little impact on primary production if the ecosystem is P-limited (Matson *et al.* 1999).

1.2 Environmental impacts of peatland drainage and agricultural practices

Soil management intensity varies greatly across the globe but, in general, the natural resources for human needs are often acquired at the expense of degrading environmental conditions (Foley *et al.* 2005). Globally, the land used for crop, pasture, and urban development has expanded in recent decades, while large areas have been deforested (DeFries *et al.* 2004). However, the land area used for agriculture has diminished in recent decades in the northern Europe, while forested area has increased (FAO 2014). Many peatlands in Finland, Russia, UK, Ireland, and the Netherlands have been drained for forestry, agricultural use, and peat extraction (Holden *et al.* 2004). The type of land use, such as agriculture or forestry management practices, have an impact on biogeochemical cycling, and soil and water quality.

1.2.1 Peatland drainage

Peat is accumulated when the production rate of plants exceeds the decomposition rate (Clymo 1984). Although both processes decrease with poor aeration (Geurts *et al.* 2010), the decomposition reduction is greater resulting in a net accumulation of peat. Peatlands are globally important C storages, but anthropogenic actions have diminished them. The actual peat C storage in Finland has decreased by an estimated 73 Tg of peat during years 1950–2000. However, the situation is opposite when looking at Finnish peatlands as a whole (including peat, tree stand, other vegetation and detritus) (Turunen 2008). Peatland forestry thus plays an important role in peatland C storages.

Drainage improves aerobic conditions, and the decomposition of peat is therefore increased and long-term C storages are reduced. In Finland, 29% (30.4 Mha) of the land area is peatland, 53% of which has been drained (Ylitalo 2010). In the southern and central parts of the country, 75% of peatlands have been drained (Ylitalo 2010). Although very little of the remaining Finnish peatland is currently drained for forestry purposes, considerable maintenance of the old ditch systems still occurs. A yearly 100 000-ha ditch maintenance goal was set for 2008–2016 in Finland’s “National Forest Programme 2015” (Ylitalo 2010). The cultivated peat soil area has increased in Finland during the ongoing century. This has caused an increase in annual CO₂ and N₂O emissions (Regina *et al.* 2015).

Peatland drainage increases downstream flooding (Holden *et al.* 2004) and destroys the original ecosystem. Because of the accelerated decomposition, the drainage may turn a peatland from a C sink into a C source (Vanselow-Algan *et al.* 2015). However, it is well known that the conversion from C sink to C source may not always happen (Minkkinen and Laine 1998; Lohila *et al.* 2011; Ojanen *et al.* 2014). It is estimated that the C loss as DOC output from Finnish forestry-drained peatlands totaled 24.5 Tg in the latter half of the previous century (Turunen 2008). Evans *et al.* (2014) reported generally higher DOC concentrations in undrained than in drained Finnish peatlands, while Strack *et al.* (2008) observed higher DOC concentrations in Canadian drained peatlands than in fens with no water table drawdown in a hummock-hollow complex. In comparison to C loss as CO₂, the DOC loss from wetlands can be minor, as shown by Clair *et al.* (2002).

The possible loss of the C sink function of drained boreal peatlands can potentially be restored within a few years after restoration (Vasander *et al.* 2003). At present, little restoration of Finnish drained peatlands has taken place, but interest in restoring formerly drained peatlands has increased in the Nordic countries (Maljanen *et al.* 2010). However, peatland restoration can also cause direct environmental problems such as increasing methane (CH₄) emissions (Komulainen *et al.* 1998) or P leaching (Vasander *et al.* 2003). Kieckbusch and Schrautzer (2007) stated that nutrient outputs from rewetted peatlands can be high during the first years after restoration, but hydrochemical conditions become more stable over time. Also, the new climate policy involves greenhouse gas mitigation on agricultural peat soils (Regina *et al.* 2015). Raised water tables in cultivated organic soils in Finland are projected to decrease the CO₂ emissions from these soils.

1.2.2 Agricultural soil practices

Soil fertility, crop production, and the ecosystem services of soils are generally related to soil C content. Conversion of natural ecosystems to agricultural lands can deplete the soil organic C pool of temperate ecosystems by 50% in approximately 50 years (Lal 1999). However, losses can be reduced and agricultural soils even turned into C sinks by adopting agricultural practices such as cover crops and reduced till (Lal 2004). Preventing or reducing C loss has been one motive for adopting no-till cultivation. The historic loss of C on cultivated lands has been estimated at over 50 Pg globally (Paustian *et al.* 1998; Lal 1999 and 2004; IPCC 2007) while the loss caused by soil degradation and accelerated erosion between 1850 and 1998 is estimated at 25 Pg (Lal 2004).

Crop residues left after harvesting are traditionally mixed into the mineral soil. Due to the mixing, microbes break down OM faster, weeds are controlled more effectively, and aeration of the mineral soil is improved. In no-till farming harvesting residues are left on the surface and the new crop is sown on the stubble. No-till or reduced soil tillage practices leave protective surface residues that prevent erosion and sediment discharge (Stonehouse 1997; Rasmussen 1999; Baylis *et al.* 2002; Matisoff *et al.* 2002; Montgomery 2007) and decrease N (Rasmussen 1999) and P leaching and loadings (Stonehouse 1997) into aquatic systems in comparison to the traditional method of soil mixing. No-till practice increases surface soil C contents (Franzluebbers 2008; Dong *et al.* 2012; Gómez-Rey *et al.* 2012; Virto *et al.*

2012), although the soil total C pool might not increase when deeper soil layers are taken into account (Powelson *et al.* 2014; Huang *et al.* 2015; Valboa *et al.* 2015). No-till farming is increasingly practiced due to these no-till benefits, or because of the corresponding negative effects of soil mixing. However, depending on soil conditions, soil type, and cultivated plant species, no-till farming might not always be the most appropriate cultivation technique. For example, no-till farming may increase N₂O emissions, especially in clayey soils (Six *et al.* 2004; Gregorich *et al.* 2005; Sheehy *et al.* 2013).

1.2.3 Climate change

Atmospheric temperature and precipitation, which are also the primary definers of soil temperature and moisture, are major drivers of biogeochemical cycles. Natural ecosystems are expected to confront the strongest and most comprehensive impacts of climate change (IPCC 2014a). Depending on which scenario is used, global mean surface temperatures are predicted to rise 0.3–4.8°C by the end of the century compared to values from 1986–2005 (IPCC 2014b), but the Arctic region will warm more rapidly (IPCC 2014b). An increase in annual mean precipitation is also expected at high latitudes (IPCC 2014b). The expected annual precipitation increase for Finland may be as high as 30% by the end of the century (in comparison to 1986–2005) (IPCC 2014c). Furthermore, the frequency and intensity of heavy precipitation events have increased in Europe and North America (IPCC 2014a).

Since 1750 the emissions of the greenhouse gases CO₂, CH₄, and N₂O have respectively increased by 40%, 150%, and 20% (IPCC 2014a). The present atmospheric CO₂, CH₄ and N₂O concentrations are approximately 380 ppm, nearly 1800 ppb, and over 320 ppb, respectively (IPCC 2014b). Estimations of atmospheric CO₂ concentrations by the end of the century vary considerably due to uncertainty in the estimations of future atmospheric greenhouse gas emissions, which range from <430 ppm up to >1000 ppm (IPCC 2014d).

In response to climate change, water table levels can change e.g. in boreal and subarctic peatlands (Pastor *et al.* 2003). Such changes are likely to affect peatland greenhouse gas emissions (Martikainen *et al.* 1993; Silvola *et al.* 1996; Lai 2009), and the amount and seasonality of leaching from the soil.

This is because any changes in soil hydrological conditions will influence decomposition and mineralization processes (Naden and McDonald 1989; Fenner *et al.* 2001; Clark *et al.* 2005) along with transport.

2. OBJECTIVES

C and N in soils changed by human actions either directly (soil management practices) or indirectly (climate change) are the main themes of this thesis. Both the management practices and climate change may ultimately affect the same processes in soils. Peat and agricultural mineral soils are included in this study. As the water table level can drastically vary in boreal peat soils, I investigated whether the effects of water table fluctuation for soil C and N were more significant in pristine or in forestry-drained peat soils. The focus of the agricultural soil experiment was on the N cycle in no-till and ploughed soils. To a minor extent I also discuss how C and N in the soil are linked to the hydrosphere and atmosphere. Two specific study objectives were investigated:

- 1) The DOC, dissolved organic nitrogen (DON), and inorganic N concentrations in peat soil water and the CO₂ fluxes to the atmosphere from pristine and drained peat soils in response to fluctuating water table level, and
- 2) Process-specific gross N transformation rates in agricultural clay soils under no-till and moldboard-ploughing practices after harvesting.

3. MATERIALS AND METHODS

3.1 Study sites

Two types of ecosystems were chosen for the study: a peatland complex in Lammi in the municipality of Hämeenlinna (I, II) and an experimental agricultural field belonging to Natural Resources Institute Finland (Luke) in Jokioinen (III) in southern Finland.

For the peatland study, a greenhouse experiment was carried out with peat profiles from pristine (Pr) and forestry-drained (Dr) parts of the peatland. Profile samples were collected into plastic containers, i.e. mesocosms were founded (I, II). The mesocosms were placed in an unheated, large greenhouse with a high roof equipped with roof hatches. Manipulation of the mesocosms imitated extreme yet realistic precipitation conditions.

For the agricultural clay soil study, a ^{15}N (isotope 15 of N) label experiment was carried out in the field after harvesting but before autumn ploughing. The experimental area (160 m * 60 m) included four no-till and four moldboard-ploughed 25 m * 10 m plots that had been cultivated with barley (*Hordeum vulgare* var. *annabel*).

3.2 Peat soil experiment (I, II)

Peat profiles with original vegetation cover were placed into 20 plastic cylinders (height 60 cm, Ø 24 cm). Control (Ctrl) and fluctuating water table level (Fluc) mesocosms differed in their temporal-magnitude variation of water table level, imitating drought and a very rainy period in nature. Conditions were identical except for the differing peat soils (pristine and drained) and for the water table level manipulation. A total of four mesocosm groups were used: pristine and drained peat mesocosms with control conditions and fluctuating water table level (PrCtrl, PrFluc, DrCtrl, and DrFluc mesocosms).

The day after the peat cores had been collected, the water table depths were measured in the boreholes left after coring. These depths were considered to be control hydrologic conditions and the water table of the Ctrl mesocosms were kept at this level for the duration of the experiment by adding spring groundwater on a regular basis. As water table level defines the aerobic/anaerobic conditions of the soil, certain levels were applied instead of standard water volume additions to the mesocosms, and, as a consequence, the amount of the added water varied among the mesocosms.

For Fluc mesocosms, two successive water table level manipulations were performed imitating drought and rainy periods. During the first period (low water table level) in summertime, only a small amount of spring water was added to these mesocosms. The second period (high water table level) was performed in autumn. The second period included four sampling times, WH

(40, 42, 43, and 44), where W = water sampling and H = high water table level, and the number in parentheses refers to the week number that sampling was carried out on. These four sampling times followed a substantial water addition. The same above mentioned sampling codes are also used for the control mesocosms to indicate the sampling times. The watering and sampling timetable is presented in II. A schematic presentation of the water table levels in different mesocosms and different sampling times is presented in I and II. Note that these published schematic presentations contain an error: one dot representing sampling depth is marked incorrectly in Figure 1 in I and in Figure 1 in II. The uppermost marked dot in the DrCtrl plot is 10 cm too high, and should be at -20 cm. The description of the sampling depths is written correctly in I in chapter 2.3.

Water samples were collected from three different depths (I, II) of the containers on four occasions during the high water table level period. Water from three depths was pooled for a composite sample and DOC (mg C L^{-1}), DON ($\mu\text{g N L}^{-1}$), NH_4^+ ($\mu\text{g N L}^{-1}$), and $\text{NO}_2^- + \text{NO}_3^-$ ($\mu\text{g N L}^{-1}$) concentrations of filtered water were measured. DOC concentrations were analyzed from the borehole water in peatlands, where the peat profiles were taken from, and DOC, DON, and NH_4^+ concentrations were analyzed from the spring water used for the water additions. CO_2 fluxes ($\text{g C m}^{-2} \text{d}^{-1}$) were analyzed on four occasions during the low water table level period, GL(32, 34, 36, 37), where G = gas sampling and L = low water table level, and the number in parentheses refers to week number, and three times during the high water table level period GH(40, 42, 43) = gas sampling, high water table level (calendar week numbers) (II). Although the terms DOC and DON mean dissolved organic carbon/nitrogen, the organic C and N of those compounds actually exist also in dry matter. In this study, the word “release” into soil water refers to these particles dissolving or mixing into free water that was added to the mesocosms.

For this thesis, the relative quantity of DOC, DON, and NH_4^+ concentrations in Fluc compared to Ctrl mesocosms were calculated, each sampling time separately. The relative CO_2 fluxes were calculated similarly for both high and low water table periods. The equation for each sampling time was:

$$100(\%) * \frac{\text{mean concentration or mean CO}_2 \text{ flux [PrFluc or DrFluc]}}{\text{mean concentration or mean CO}_2 \text{ flux [PrCtrl or DrCtrl]}}$$

where 100% represents the Ctrl (Pr or Dr) reference value. The result of each calculation is the relative amount in comparison to Ctrl, i.e. for example at result of 110% in PrFluc means that it is 10% higher than in PrCtrl at the same time. Results are shown graphically.

Parameter differences between the experimental groups and sampling times were analyzed statistically with the “Proc mixed with repeated statement” method by using SAS 9.2 (Anonymous 2008). The statistical methods are described more thoroughly in I and II.

3.3 Agricultural soil experiment (III)

The agricultural field experiment was performed in autumn after harvesting, but before ploughing was completed for that autumn. A ^{15}N pool dilution and tracing technique was used to quantify several gross process rates involved in the N cycle. The experiment was carried out *in situ* using a Virtual Soil Core approach (Rütting *et al.* 2011; Staelens *et al.* 2012) with five incubation times (0, 1, 2, 5, and 9 days). Before beginning the ^{15}N labeling, preliminary analyses of soil properties were performed to estimate the concentration of inorganic N to be added as the label solution. The intention was not to dramatically exceed the existing inorganic N content in soil.

Labeling of no-till and moldboard-ploughed clay soils was performed using ^{15}N -labeled ammonium nitrate solutions ($^{15}\text{NH}_4\text{NO}_3$ and $\text{NH}_4^{15}\text{NO}_3$) to a depth of 0–5 cm. Labeling details are described in III. A label was added between sowing lines, but some loose straw remained between the lines. These straw remains were removed beforehand from the labeling points, but they were replaced immediately after labeling to prevent abnormal soil drying. At days 0, 1, 2, 5, and 9, the soil was sampled to a depth of 5 cm using an auger. The samples were sieved and homogenized in a laboratory. A certain mass of the samples was extracted with 2 molar potassium chloride (KCl) and filtered. NH_4^+ and NO_3^- concentrations and ^{15}N enrichment in NH_4^+ and NO_3^- were analyzed from the extractions. The rest of each homogenized and sieved soil sample was used for the bulk soil analyses. The bulk soil samples were analyzed for total C and total N contents and for bulk- ^{15}N abundances. Soil water content and dry bulk density were also analyzed (III).

Process-specific gross N transformation rates ($\mu\text{g N (1g of dry soil)}^{-1} \text{ d}^{-1}$) in no-till and ploughed soils were quantified with a numerical tracing model (Müller *et al.* 2007) with a general mathematical notation specified by Müller *et al.* (2004). The model was modified by Rütting *et al.* (2010) to include twelve N transformation processes. I used this 12-process model to analyze my data. I reached the best model fit by marking five of the processes as non-existing in the experimental soils. Thus, the final model setup I used for data analysis contained seven N transformation processes (III), but only four of them are discussed in this thesis. These four processes are the mineralization of organic N to NH_4^+ (referred to in this text as mineralization), immobilization of NH_4^+ to organic N (referred to in this text as immobilization), oxidation of NH_4^+ to NO_3^- (referred to in this text as nitrification), and NO_3^- loss flux, which includes NO_3^- leaching, lateral diffusion, and any N gas losses. The model is based on altering $^{15}\text{N}/^{14}\text{N}$ ratios along with N cycle processes, described more thoroughly in III. The model gives one final result per process and per soil (e.g. one gross mineralization rate result for no-till). Thus, there is no statistical testing for the gross rates in this study.

One-way repeated measures ANOVA was performed for the data to analyze differences in soil properties (except one-way ANOVA for NH_4^+ -N concentrations) between treatments (no-till and ploughing) by using IBM SPSS Statistics 22 (Anonymous 2013). The statistical analyses are described more thoroughly in III. Thus, the statistical test results should not be confused with the modeled gross rate results. The same chemical analysis results of NH_4^+ and NO_3^- concentrations were used in the statistical analysis and the modeling, but they are only part of the data included in the modeling (see III; “Data Analyses”).

4. RESULTS

4.1 Peat soil carbon and nitrogen in changed hydrological conditions

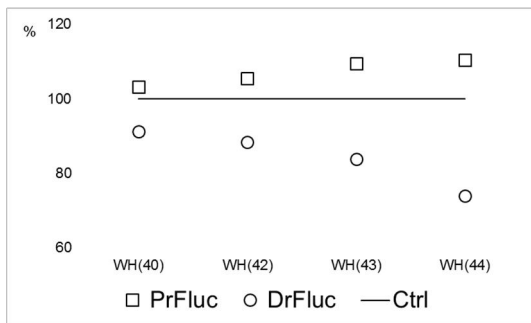
4.1.1 Releases to soil water

DOC and DON concentrations in the control mesocosms were higher in the forestry-drained peat than in pristine peat mesocosms (I, II). The opposite situation was true for the NH_4^+ concentrations (I). Differences in DOC

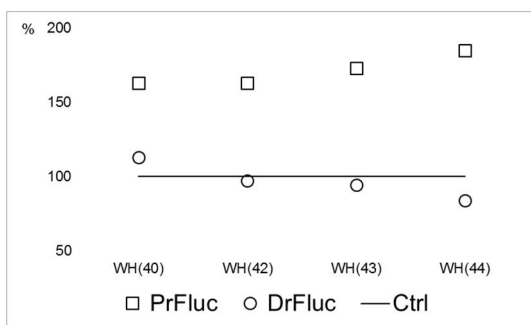
concentrations in the pristine and drained control mesocosms were roughly the same magnitude as those found between pristine and drained peatland water taken from the boreholes, from where the cores had been collected.

When Ctrl and Fluc mesocosms were compared within one peat soil type (pristine or drained), DOC concentrations were not observed to differ between PrCtrl and PrFluc. Instead, they were higher in DrCtrl than in DrFluc (Fig. 2a and II). Importantly, DOC concentrations decreased with time during the high water table period, i.e. along with the spring water additions in the DrFluc mesocosms (Fig. 2a and II).

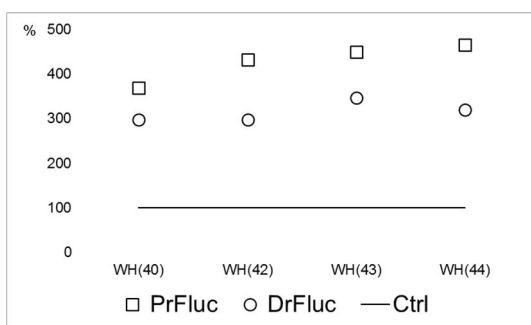
DON and NH_4^+ concentrations were clearly higher in the PrFluc mesocosms than in the PrCtrl mesocosms already at the beginning of the high water table period (Fig. 2b, 2c and I). It is important to note that significant differences were found despite the substantial water additions (i.e. dilution) to the Fluc mesocosms. Despite no statistical difference occurred in DON or NH_4^+ concentrations when the DrCtrl and DrFluc mesocosms were compared (I), it is noteworthy that NH_4^+ concentrations were approximately three times higher in the DrFluc than in the DrCtrl mesocosms at the beginning of the high water table period (Fig. 2c). No significant differences in $\text{NO}_2^- + \text{NO}_3^-$ concentrations were observed between the experimental groups.



a)



b)



c)

Figure 2. Relative amount of a) DOC, b) DON, and c) NH_4^+ in the Fluc mesocosms at sampling times WH(40–44) in comparison to Ctrl mesocosms.

Pr = pristine

Dr = drained

Ctrl = control

Fluc = fluctuating water table level

W = water sampling

H = high water table level

(No.) = calendar week number

Compared to the spring water that was added to the mesocosms, DOC, DON (except for two individual samples), and NH_4^+ concentrations were higher in the mesocosm water samples during the entire experiment. This difference was substantial especially with DOC and NH_4^+ concentrations. Thus, the added spring water diluted the soil water concentrations.

4.1.2 Carbon dioxide fluxes

CO₂ fluxes differed only between the PrCtrl and DrCtrl mesocosms and only during the low water table period (II). It is noteworthy that the CO₂ fluxes during the dry period GL(32–37) and at sampling time GH(40) were more substantial (though not statistically) from the PrFluc mesocosms than from the PrCtrl mesocosms, but the trend was opposite in drained peat, i.e. smaller (not statistically) from the DrFluc than from DrCtrl mesocosms (Fig. 3 and II). The CO₂ fluxes did not reach the control level in the Dr mesocosms within the four weeks after initiating the substantial water additions.

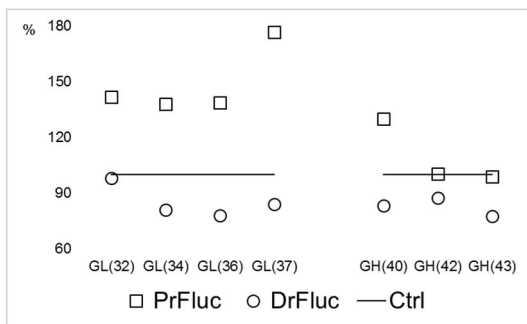


Figure 3. Relative amount of CO₂ fluxes in Fluc mesocosms at sampling times GL(32)–GH(43) in comparison to Ctrl mesocosms. Pr = pristine, Dr = drained, Ctrl = control, Fluc = fluctuating water table level, G = gas sampling, L = low water table level, H = high water table level, (No.) = calendar week number.

4.2 Nitrogen transformations and nutrient status in no-till and ploughed soils

4.2.1 Process-specific gross nitrogen transformation rates

The highest gross N transformation process rate was mineralization in both no-till and ploughed soils, and it was 14% higher in the no-till soil (Fig. 4a). The gross immobilization rate was clearly higher in the no-till treatment than with ploughing (Fig. 4a), the difference being 64%. In comparison to mineralization and immobilization rates, the gross nitrification rate was low in both soil management types (Fig. 4b).

However, it was approximately twelve times higher in ploughed than in no-till soil. The gross NO_3^- loss flux rate was also much higher in the ploughed soil (Fig. 4b), being almost 16-fold that of no-till soil. The ratios for gross nitrification rate / gross immobilization rate for no-till and ploughed soils were 0.009 and 0.183, respectively.

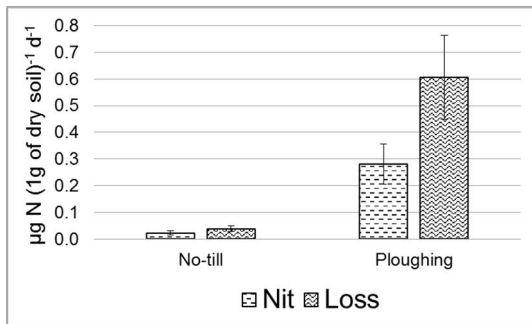


Figure 4. Quantified gross N transformation process rates (± 1 SD) of

a) mineralization (Min) and immobilization (Imm) and

b) nitrification (Nit) and NO_3^- loss flux (Loss).

a)



b)

4.2.2 Soil nutrient status

KCl-extractable NH_4^+ concentrations were mostly higher in the no-till soil in comparison to the ploughed soil, but no difference was found in the NO_3^- concentrations between the soil management types (III). Total C and N contents were higher in no-till soil. Soil bulk density was slightly, but significantly, higher in ploughed soil. The mean values and standard deviations of these soil quality parameters and the statistical test results are presented in III.

5. DISCUSSION

5.1 Carbon and nitrogen in various soil conditions

Land use is likely to be a major factor of environmental change at high latitudes, potentially causing significant alterations in soil C and N cycles (Grünzweig *et al.* 2003). The main focus of my thesis was on soil C and N releases in pristine and drained peat soils, and on the N cycle in no-till and moldboard-ploughed agricultural soils. Land use intensification and changes in soil quality due to climatic factors can cause either desired (e.g. improved aerobic conditions in drained peatlands) or undesired (e.g. diminishing soil C storages due to drainage) consequences, and they affect not only the soil but also adjacent aquatic systems (e.g. eutrophication) and the atmosphere (e.g. CO₂ concentrations).

Soil moisture conditions can change due to climatic factors (extreme events or long-term climate change) or through land use intensification-related disturbances. Moisture affects biogeochemical cycling and element releases e.g. by changes in soil aerobic conditions and associated soil biological activity, or by changes in the flow of water and associated transport of solutes. Responses to microbial communities and activities vary in different peat soils (Jaatinen *et al.* 2007). This, of course, may further impact soil nutrient status – with different impacts in various soils. In trying to determine the dynamics of soil C and N, and their responses to changing environmental conditions, it is necessary to take into account both biological and abiotic processes and factors. The mechanisms behind increasing nutrient loads to aquatic ecosystems are largely straightforward, as point sources can be identified and diffuse sources are roughly understood. However, factors controlling the more obvious reasons are complex.

5.2 Carbon and nitrogen dynamics in peat soils in response to hydrology

Most microbial processes involved in decomposition and mineralization intensify with the degree of soil aeration. Peatland drainage results in the long-term lowering of the water table and promotion of aerobic conditions that favor soil microbiological activity and the decomposition of OM (Jaatinen *et al.* 2007; Kiikkilä *et al.* 2014). The result is a long-term change in peat quality. The response e.g. in the soil C pool for forestry-drainage may differ greatly in peat soils with various nutrient status or already in relatively

slightly differing climates, such as northern and southern Finland (Laiho 2006).

In extremely dry soil conditions the produced substances are not easily dissolved into free water, but this becomes possible with an increase in water content. The dilution effect is crucial in interpreting my peatland experiment results. Respectively, the rewetted pristine and drained mesocosms were given approximately 33% and 26% more water than their controls (Table 1 in I), i.e., the overall dilution was more remarkable in the rewetted mesocosms. This means that when the concentration of an element was higher in the rewetted mesocosms than in their controls, the release of the element (fundamentally decomposition or mineralization during drought followed by dissolving or mixing into added water) exceeded the dilution effect.

When the dilution effect is considered between the first and the fourth rewetting period sampling time, the situation is different. The water table level in the rewetted mesocosms was raised to 0 cm just before the first flood sampling time (in between the time that the considerable water additions began and the first sampling time of the rewetting period). The water table of the pristine control mesocosms was approximately at this same level during the entire experiment. Thus, there were no great differences in the added water volumes between the PrCtrl and PrFluc mesocosm after the first flood sampling time, and the dilution effect after this moment was similar in both mesocosm groups (PrCtrl and PrFluc). In other words, in pristine mesocosms the effect of the dilution for the results (PrCtrl vs. PrFluc) occurred by the time of the first flood sampling. Instead, there was a greater difference in the added water volumes between the control and rewetted drained peat mesocosms after the first flood sampling time (more water to DrFluc than to DrCtrl). That is because more water evaporated from the DrFluc mesocosms, where the water level was kept at 0 cm during rewetting. The water table in the DrCtrl was at -20 cm. The C and N releases during rewetting observed in my experiment should be considered as a short-term response to an extreme period rather than a permanent long-term response.

5.2.1 Peat soil carbon

Increasing DOC or TOC concentrations in surface waters have been reported across Europe and North America (Skjelkvåle *et al.* 2003; Evans *et al.* 2005; Vuorenmaa *et al.* 2006; Monteith *et al.* 2007; Worrall and Burt 2007), and

wetlands are important sources of DOC export to aquatic ecosystems (Xenopoulos *et al.* 2003; Jager *et al.* 2009; Huotari *et al.* 2013; Pumpanen *et al.* 2014). Several possible reasons exist for the increasing DOC, e.g. changes in hydrology (Tranvik and Jansson 2002). Some disagreement exists whether summer-time precipitation will decrease or increase in Finland during the following decades, but autumn precipitation will likely increase (Jylhä *et al.* 2004). Increase in autumn precipitation has already been observed in the mid- and high latitudes in the northern hemisphere (Dore 2005; Schmidli and Frei 2005). Dry summers followed by very rainy periods may become more common in Finland in the future.

Precipitation is an important determinant for DOC flux in boreal catchments (Pumpanen *et al.* 2014), and extreme climatic events may substantially affect the quantity and quality of DOC leaching into surface waters (Hinton *et al.* 1998). This gave some support by my pristine peat soil results concerning the DOC concentrations. DOC production during drought, followed by its release into the added water, was high enough to compensate the dilution effect in the hydrology manipulated pristine mesocosms. In other words, the DOC concentrations in pristine rewetted mesocosms remained close to DOC concentrations in pristine control mesocosms (no significant difference between PrCtrl and PrFluc). Jager *et al.* (2009) studied peatland DOC run-off during dry and wet years in eastern Finland. They observed that DOC export was higher during the wetter year in comparison to the dry one, and importantly, they found that DOC run-off was highest during peak flow events during both a dry and a wet year. DOC concentrations dropped to the magnitude of the run-off events, and were ~25% higher under drought conditions.

In contrast to pristine peat mesocosms, the DOC concentrations in the rewetted drained mesocosms decreased with the water additions between the first and the last rewetting sampling time, a phenomenon related to more remarkable dilution after the first flood sampling time (see chapter 5.2). Because the element load is a product of concentration and volume, a temporary drop in the DOC concentrations in natural soil water may only have a minor influence for total export especially in springtime and after heavy precipitation events in summer and autumn when water volumes are large (Arvola *et al.* 2006). In accordance, Jager *et al.* (2009) and Sarkkola *et al.* (2009) reported limited impact of DOC concentrations on DOC and TOC export. Similarly and with the same reasoning, if conditions are very dry, a

possible increase in DOC concentrations is not necessarily seen as an increase in DOC export to the adjacent water systems in the short-term simply because water-related transportation does not occur much.

The gas fluxes from the soil are also important when considering the environmental impacts of land use. For example, the increased decomposition in drained peatlands leads to increased CO₂ release, but pristine peatlands are greater sources of CH₄ emissions, a more powerful atmospheric warming gas than CO₂ (Martikainen *et al.* 1995; Minkkinen *et al.* 2002). Minkkinen *et al.* (1999) reviewed higher annual TOC leaching rates and higher annual CO₂ emissions from drained than from undrained Finnish peatlands. In accordance with this, during drought CO₂ release obviously occurred more from the manipulated pristine mesocosms than from their controls, although the difference was not significant (Fig. 6 in II). Part of the DOC produced during a drought may have been further processed into CO₂ (before the rewetting period) and emitted to the atmosphere.

The positive response to drought on CO₂ fluxes was more evident in pristine than in drained peat mesocosms, which was as expected. Actually, in the case of drained peat, the drought period CO₂ fluxes were even slightly greater from the controls than from the manipulated mesocosms. The main factors defining OM decomposition are the environmental conditions, decomposers, substrate quality, and nutrient availability (Laiho 2006). Less easily decomposable OM was left in the drained peat because of the long-term improved aeration in the drained peatland. In addition to the abundance of the easily decomposable OM, one partial reason for the differences in pristine and drained peat mesocosms during drought could lie in the peat moisture. Although the free water table level was equal in pristine and drained peat mesocosms during the dry period, the surface peat was still drier in the drained peat, which possibly created unfavorable conditions for microbes to function properly during this period. This explanation is in line with Grünzweig *et al.* (2003), who stated that soil respiration may be limited by low soil water content.

Higher soil CO₂ fluxes were observed from the drained control than from the pristine control mesocosms during the first period (low water table level in Fluc mesocosms), which was a consequence of the 20 cm deeper aerobic layer in the drained peat control mesocosms, promoting aerobic respiration (Martikainen *et al.* 1995). The air temperature dropped before the last two

gas sampling times (II). The CO₂ fluxes on the last two sampling times were roughly equivalent in all mesocosms despite the treatments, i.e. the decline in CO₂ production to a similar rate in all mesocosms was probably largely based on the decreased air temperature that had cooled down the peat soil.

5.2.2 Peat soil nitrogen

N loads to boreal aquatic ecosystems are predicted to increase substantially during the next few decades due to changes in temperature and hydrology (Holmberg *et al.* 2006; Moore *et al.* 2010). My N compound results, together with the expected increase in extreme hydrological events, are in agreement with this prediction. Rewetting after a prolonged drought in pristine peat soil caused a sharp increase especially in soil water in NH₄⁺ concentrations and also in DON concentrations (in comparison to controls). The NH₄⁺ concentrations obviously also increased in the drained peat, but the difference was not statistically significant. DON release in drained peat was high enough to compensate dilution. Importantly, these concentration increases were obvious already at the beginning of the rewetting period, i.e. right after the prolonged drought (Fig. 2 in I). Thus, the increase was largely based on accelerated production, i.e. decomposition and mineralization, in the aerated soil layer during the drought.

When the considerable water additions began (rewetting), the produced substances that were not transformed into other substances previously could be quickly released (through dissolving and other physicochemical processes) into the added water. Kane *et al.* (2010) performed a test at a 20 cm soil depth in Alaska, USA during autumn, to observe which peat soil treatment leads to the highest and lowest total dissolved N and DOC concentrations, control, lowered, or raised water table. In accordance to my conclusions, the lowered water table treatment increased the production of these substances. However, they observed the lowest concentrations in the raised water table treatments, but unlike in my experiment, their lowered and raised water table treatments were not connected to each other. Kortelainen *et al.* (2006) reported that a high TOC export predicts high TON and NH₄⁺ exports. This is in agreement with to my conclusions in pristine peat mesocosms, that the hydrological manipulation increased the release of all three elements, DOC, DON and NH₄⁺, although in my experiment it can't be stated if organic C was a predictor.

In accordance to my NH_4^+ concentration results, Arvola *et al.* (2006) reported that the NH_4^+ and NO_3^- loads correlated positively with the precipitation amount in the same study catchment where my peat core samples were taken from. Cooper *et al.* (2007) indicated that a run-off magnitude in autumn is an important increasing factor for DON release to run-off waters. Kieckbusch and Schrautzer (2007) observed a high NH_4^+ release from a eutrophic fen after heavy autumn rainfall.

NH_4^+ concentrations were higher in the pristine manipulated mesocosms in comparison to their controls also at the end of the dry period, but the increase was much clearer after the substantial water additions had begun (I). All the NH_4^+ produced was thus not in the very meager free water available in the mesocosms during the dry period. Instead, the NH_4^+ ions were probably bound in ion-exchange sites during drought. The substantial water additions caused an NH_4^+ concentration increase in the free water.

Unlike NO_3^- , NH_4^+ is highly immobile (Vogeler *et al.* 2011), so its leaching risk is usually low. However, NH_4^+ exports from peatlands can be considerable (Kortelainen *et al.* 2006; Hynninen *et al.* 2011) as the water content of peat soil is often very high. NH_4^+ is a potential source for nitrification, so an increase in NH_4^+ concentration could also increase the NO_3^- concentration. However, my experiment revealed no differences in soil water $\text{NO}_2^- + \text{NO}_3^-$ concentrations between the experimental treatments. Some NO_3^- might have been transformed to N gases and emitted to the atmosphere. Some of the NO_3^- might also have been taken by plants. Regina *et al.* (1996) observed that nitrification was enhanced after lowering the water table in minerotrophic peat, but not in ombrotrophic peat. My experimental pristine peatland is oligotrophic.

As a whole, it appears that a rewetting event after a prolonged drought causes more evident release peaks of DOC, DON, and NH_4^+ in pristine than in drained peat soil water. These compounds were produced more efficiently in pristine peat during the drought, as there is more easily decomposable OM left in the surface layer in comparison to drained peat. When water was added, these elements were quickly dissolved or mixed into the additional water. In nature in wet conditions they could be easily transported to the adjacent environment with the natural surface water flow. I did not measure the N_2O fluxes from the mesocosms, but also they are an interesting environmental aspect of peatland dynamics. N_2O emissions often increase

due to drainage because OM degradation is increased (Maljanen *et al.* 2003). However, N₂O flux results are not always straightforward (Martikainen *et al.* 1993, 1995; Pearson *et al.* 2015), as several factors affect N₂O production.

5.2.3 Response to hydrology versus the overall situation

The response to hydrology was more evident in pristine peat than in drained peat, indicating that extreme events in nature could cause more severe release peaks of elements into soil water and possibly to the adjacent environment. However, it is important to consider the concentrations themselves, and not only responses to hydrology.

DOC concentrations were clearly higher in the drained control and the rewetted mesocosms than in any of the pristine peat mesocosms. The original DOC concentrations of the drained peatland were thus more of an important determinant for DOC concentrations than the response to the climate factor of rewetting after the drought. This is in agreement with Huotari *et al.* (2013) who observed that drainage density correlated positively with stream DOC export. With DON, the positive response to hydrology was more evident in pristine peat, but the concentrations still did not exceed drained peat DON concentrations. In the case of NH₄⁺, the situation was more straightforward. NH₄⁺ concentrations were higher in pristine control than in pristine rewetted mesocosms, and, the response to hydrology was more evident in pristine peat.

5.2.4 Limitations of the peat soil study

Mesocosms can never perfectly replicate natural conditions. However, the peat cores in the mesocosms were relatively large, intact, and undisturbed. From the practical point of view, a mesocosm experiment is an appropriate way to investigate the response of an ecosystem to a change in a particular environmental factor, such as water level changes in my case, in an otherwise controlled environment and over a reasonable period of time. The maintenance of the water table level at the soil surface in both pristine and drained peat mesocosms for four weeks during the rewetting period of the experiment was performed to imitate a very rainy period in nature. In reality, however, such a high water table for such an extended time would not occur in drained peatlands, since the ditches, unless blocked, would allow rapid run-off of surface water. Thus, the water additions to the drained peat mesocosms during the high water table period were excessive.

The problem is mitigated with the fact that the observed differences between the mesocosms can be seen in all of the bar graphics in I and II already at the first sampling time of the high water table period, when the water table level in DrFluc had been excessive only for a short period of time. However, it cannot be stated whether the concentrations in rewetted drained peat mesocosms were excessive or too low in comparison to reality. As discussed in chapter 5.2.2, the observed differences were largely based on processes observed during drought, after which the excessive amount of water was added. In this sense, the concentrations in the drained peat rewetted mesocosms were too low, as the added amount of the low-concentrated spring water was excessive. On the other hand, the produced DOC, DON, NH_4^+ , and $\text{NO}_2^- + \text{NO}_3^-$, were potentially dissolved or mixed into added free water also in the uppermost soil layer, which was now saturated with water. This could have caused excessively high concentrations especially if the production of those compounds was highest in the uppermost soil layer. It is also possible that the excessive water additions caused some false insignificant results for comparisons between DrCtrl and DrFluc, in relation to drained peatlands in nature.

5.3 Carbon and nitrogen in agricultural mineral soils

Agricultural soils are generally aerobic and disturbed, making them much more dynamic than peatlands. Agricultural soils therefore represent short-term C storages in comparison to peatlands. The C content of agricultural soils is an important factor affecting e.g. soil microbial activity (Scotti *et al.* 2015), N cycle processes (Wang *et al.* 2015), and N_2O fluxes (Regina and Alakukku 2010). OM and associated N losses from cultivated agricultural land are important both from the view point of crop growth and production, and because of the potential eutrophication of adjacent aquatic ecosystems. Cultivating the soil notably changes its N dynamics. Plant uptake is minimized after harvesting; the soil is bare and subject to autumn rainfall resulting in an increased risk of NO_3^- leaching (Porporato *et al.* 2003). As NH_4^+ is a substrate for nitrification, an increase in soil NH_4^+ can also increase the risk for NO_3^- leaching.

In agriculture, management practices have an impact on biogeochemical cycling and element releases from the soil. Currently there is increasing interest in no-till or reduced tillage farming. One reason for this is that

intensification of agriculture can result in a loss of soil OM (Matson *et al.* 1997). Significant differences in soil aeration, moisture and temperature, element contents and their vertical distribution, microbial activity, and nutrient cycling between no-till and ploughed soils have been shown (Soane *et al.* 2012; Sheehy *et al.* 2013; Singh *et al.* 2015; Nugis *et al.* 2016). In my field-based experiment carried out on a clayey agricultural field, I observed higher soil C, N, and NH_4^+ contents, along with differing process gross rates of the N cycle in the no-till management compared to the moldboard-ploughed management.

The N cycle is a complex combination of interacting processes (Rütting *et al.* 2010). In my field-based tillage experiment I focused on gross N transformation rates in the soil and the differences between no-till and ploughed (more disturbed) soil. Studies specifically comparing the simultaneous multiple process-specific gross N transformation rates in no-till and ploughed agricultural soils in humid conditions are scarce, let alone in a post-harvest situation with similar crops. No such studies carried out in boreal agroecosystems appear to exist prior to mine. To my knowledge, no other studies exist where the exact same method (^{15}N labeling and the same mathematical model) would have been used in no-till and ploughed agricultural soils. It is therefore difficult to compare my gross transformation results to other studies. More research into gross N transformation processes in agroecosystems is needed to determine whether my gross rate results are typical or representative in similar climatic conditions and in similar agroecosystems.

5.3.1 Nitrogen cycle processes in agricultural mineral soils

No-till and tilled soils in non-boreal conditions have been observed to differ in gross N transformation process rates (Muruganandam *et al.* 2010; Dong *et al.* 2012; Gómez-Rey *et al.* 2012; Hu *et al.* 2013). These studies, respectively to the reference list, were performed in North Carolina in the USA, in eastern China close to the Bohai Sea, in northwest Spain, and in southeast China. The post-harvest gross mineralization rate in my study was higher in the no-till treatment in comparison to the ploughed treatment (0–5 cm depth). However, the difference in gross mineralization rate between no-till and ploughed soil was reasonably small, especially considering that the differences between the soil management types in other transformation processes were relatively higher, even though their rates in general were lower than the mineralization

rate. The mineralization rate result was in line with other studies carried out in other humid regions (Muruganandam *et al.* 2010; Gómez-Rey *et al.* 2012; Neugschwandtner *et al.* 2014), but the opposite to those reported by Dong *et al.* (2012) for a drier area in China. Muruganandam *et al.* (2010) and Gómez-Rey *et al.* (2012) observed that mineralization in no-till soil declined rapidly after incubation commenced. N uptake by soil microbes has been observed to occur within minutes after N addition (Jones *et al.* 2013).

The N immobilization rate in my experiment was clearly higher in the no-till soil than in the ploughed soil, a result which is consistent to Muruganandam *et al.* (2010). The sampling depth of Muruganandam *et al.* (2010) was 0–10 cm, so deeper to my experiment, and in general their results for the gross mineralization rates were lower than mine, $0.9\text{--}1.9 \mu\text{g N (1g of dry soil)}^{-1} \text{d}^{-1}$ in no-till soil and $0.34\text{--}0.37 \mu\text{g N (1g of dry soil)}^{-1} \text{d}^{-1}$ in moldboard-ploughed soil, depending on the aggregate size. In my study, the immobilization rate was high in both management treatments compared to e.g. nitrification rate. Congruent immobilization results to my study, i.e. higher microbial immobilization rate in no-till than in conventional till soils have also been reported in South-America (Vargas *et al.* 2005).

No-till management has been shown to increase surface soil C content (Franzluebbers 2008; Dong *et al.* 2012; Gómez-Rey *et al.* 2012; Virto *et al.* 2012), which would also mean an increase in N content. Gross N mineralization rates have been observed to correlate positively with soil total C and total N contents, and with soil microbial biomass (Booth *et al.* 2005), and also a positive correlation of gross N immobilization rate with soil N content have been reported (Gómez-Rey *et al.* 2012). My results are compatible with these findings, as I found that the no-till soil (0–5 cm) in my experiment had significantly higher C and N contents and greater mineralization and immobilization rates in comparison to ploughed soil.

In addition to higher surface soil C and N contents, Sipilä *et al.* (2012) also observed higher microbial biomass in no-till soil than in moldboard-ploughed soil in an experiment carried out partly in the same experimental area as my study. Higher microbial biomass is probably an important factor explaining the higher gross mineralization rates I found in no-till soil. White and Rice (2009) also found a greater microbial biomass in a no-till (0–5 cm) soil than in tilled soil. Dong *et al.* (2012), whose gross mineralization result rates were lower in the no-till soil than in ploughed soil found that the microbial

biomass was also lower in the no-till soil. Soil C and N contents, microbial biomass, and mineralization rates are therefore clearly related to each other.

I observed higher nitrification rate in the ploughed soil than in the no-till soil. The results of Gómez-Rey *et al.* (2012) are in line with both the gross mineralization and nitrification rate results of my study, so they observed higher mineralization rate but lower nitrification rate in conservation till soil than in ploughed soil. However, contrasting results, i.e. higher nitrification rates in no-till in comparison to conventional till surface soils, have been observed by Muruganandam *et al.* (2010) and Hu *et al.* (2013), although nitrification rates in no-till began to decline fairly rapidly in the study by Muruganandam *et al.* (2010). Both my results and the results of Gómez-Rey *et al.* (2012) showed a higher immobilization rate in no-till/conservation till. The higher immobilization rate thus largely explains the lower nitrification rate in no-till soil in comparison to ploughed soil. Interestingly, heterotrophic bacteria are known to be better competitors for NH_4^+ than chemolithotrophic nitrifiers (Verhagen and Laanbroek 1991).

Observing the transformation rates is interesting also from the view point of the surrounding environment, as the ratio of gross nitrification rate / gross immobilization rate correlates positively with NO_3^- leaching in arable soils (Stockdale *et al.* 2002). My results showed higher ratio in the ploughed soil than in the no-till soil, indicating a higher NO_3^- leaching risk from ploughed soil. This was supported by the clearly higher NO_3^- loss flux rate from the ploughed soil, which includes NO_3^- leaching along with lateral diffusion and N gas losses.

5.3.2 Environmental impacts of no-till and ploughing practices

My gross N transformation results indicated a reduced risk of N leaching after harvesting when practicing no-till instead of ploughing. The higher immobilization rate and lower nitrification and NO_3^- loss flux rates, and lower nitrification/immobilization ratio in the no-till soil all supported this management practice. Mineralization rate was lower in the ploughed soil, which would mean lower substrate production for subsequent nitrification. However, the relative difference between the treatments was only 14%, which was not very high in comparison to other process gross rates. Thus, it appears that the higher mineralization rate did not offset the benefits of increased immobilization in the no-till soil.

In the case of agricultural soils, bulk density is usually higher in no-till than in tilled soils (Regina and Alakukku 2010; Sipilä *et al.* 2012), and therefore the aerobic conditions are usually poorer in no-till soils. Higher N₂O fluxes from no-till than from tilled soils have been observed in Jokioinen (Regina and Alakukku 2010; Sheehy *et al.* 2013) and in several studies carried out in various parts of the world (Six *et al.* 2004; Oorts *et al.* 2007; Dong *et al.* 2012; Hu *et al.* 2013).

Both Regina and Alakukku (2010) and Sheehy *et al.* (2013) suggested that the main reasons for higher N₂O emissions from no-till were the higher bulk density and more favorable water-filled pore space. However, the situation may be reversed at least in humid climates when no-till is practiced for a long time (Six *et al.* 2004). My gross N transformation rate results give some interesting supplementary information relating to the N₂O emissions. NO₃⁻ is a source for N₂O production (Maier 2009), and I found higher nitrification rates in the ploughed soil than in no-till soil. However, I also observed a much higher NO₃⁻ loss flux rate from ploughed soils. This loss flux includes NO₃⁻ leaching, so one partial reason for the lower N₂O fluxes from the ploughed soils at the Jokioinen experimental site could be that more of the NO₃⁻ may have been leached.

5.3.3 Limitations of the agricultural soil study

The NH₄⁺ concentrations and especially the NO₃⁻ concentrations in my agricultural field experiment were low. A small absolute change in concentration can be relatively quite high and can go clearly up and down within the concentration range of the measurements within a short period of time. This was the case in my study. These concentrations were included in the model in addition to the ¹⁵N analysis results. The purpose of the model is to predict, and it cannot perfectly predict a very irregular fluctuation. Thus, model fit cannot be perfect with data like mine. This causes some uncertainty in the results. However, the final modeled parameter values followed a normal distribution very well, making the results reliable.

6. CONCLUSIONS

When evaluating the environmental impacts of soil processes and properties, there is a risk of making too simplified conclusions. Soil is affected by multiple simultaneous and interacting processes and other factors that control the production or content of differing forms of C and N substances. These include microbiological processes (e.g. decomposition, mineralization, nitrification, denitrification, uptake, and immobilization), physicochemical processes (e.g. dissolution, adsorption, desorption, and ion exchange), and simply changes in soil water content (drought resulting in concentration and wetting resulting in dilution). Environmental change, be it related to land use or climate change, affects soil processes and thereby the release of substances to the environment.

In my peat soil mesocosm experiment, an increase in DON and NH_4^+ concentrations in pristine peat soil water was seen very soon after initiating a rewetting period after a prolonged drought. DOC release in pristine peat was high enough to compensate for the dilution effect of the water additions to the mesocosms. These compounds were produced in the aerated soil layer during the drought, and released to the added water through dissolving and other physicochemical processes when the mesocosms were rewetted. In drained peat mesocosms the releases of DON and NH_4^+ were also high enough to at least compensate for the dilution effect, but the release of DOC did not. Thus, the loading of C and N to adjacent water ecosystems during peak rainfall events maybe more evident from pristine peatlands than from drained peatlands. However, because the DOC and DON concentrations were higher in drained than in the pristine peat from the beginning, the overall longer-term situation can still be worse from the drained peat in the cases of DOC and DON. That is, DOC concentrations were always lower in pristine peat despite the hydrological events, and DON concentrations in pristine peat did not exceed those in the drained peat despite the higher response to hydrology.

In my agricultural post-harvest experiment I observed higher gross rates of NH_4^+ mineralization and NH_4^+ immobilization, but lower gross rates of nitrification and NO_3^- loss flux in the no-till soil than in ploughed soil. Contents of NH_4^+ were generally greater in the no-till soil. Despite NH_4^+ concentrations and its mineralization rate being higher, nitrification rate was lower in the no-till soil. This is explained by higher NH_4^+ immobilization rate in no-till. Thus, the substrate for NO_3^- production was immobilized

efficiently in no-till. This results in a reduced risk of NO_3^- leaching in no-till, as also indicated by the lower nitrification/immobilization ratio in no-till. Eutrophication therefore favors the use of no-till farming.

The data interpretation of nutrient cycles, and especially that of the N cycle, is complicated. To mitigate and adapt to the expected changes in the environment due to land use intensification and climate change, more knowledge of soil responses is needed, and preferably knowledge connecting the responses in soil to the adjacent environment.

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