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EFFECTS OF SITE PREPARATION FOR AFFORESTATION ON SOIL PROPERTIES AND GREENHOUSE GAS EMISSION

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DECLARATION

I composed this thesis, the work is my own. No part of this thesis has been submitted for any other degree or qualification.

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ABSTRACT

Forest plantations in the UK are often established on seasonally waterlogged peaty gley soils which often require site preparation (drainage and mounding) to lower the water table and prepare planting positions. Substantial changes in the physical environment of peaty gley soils can accompany site preparation including fluctuations in soil temperature and soil moisture. These and other changes can all affect soil properties and decomposition processes and well as the dynamics of CO_2, CH_4 and N_2O. A field experiment was established at Harwood Forest (NE England) to investigate the effects of three site management practices (drainage, mounding and fertilisation) frequently used for afforestation and replanting on peaty gley soils in the UK on soil properties and various of C and N, environmental variables (soil temperature, water content and water table height) and emissions of CO_2, CH_4 and N_2O. The relationship between GHG emissions and environmental variables was also examined. The experiment was laid out in a factorial split-plot design. Drainage decreased C and N concentration in the 10 cm soil layer. The soil bulk density in the 0 to 20 cm soil layer was increased by mounding. Drainage and fertilisation increased soil CO_2 efflux, whereas mounding did not affect soil CO_2 efflux. All three practices affected soil CH_4 fluxes with drainage reducing the fluxes and mounding and fertilisation increasing the fluxes. Nitrous oxide emissions were significantly affected by mounding and fertilisation, with mounding decreasing emissions and fertilisation increasing emissions. Soil temperature was the main environmental factor controlling soil respiration in this site.

Over the two years study drainage and fertilisation increased the total greenhouse budget by 13.1% and 97.9%, while mounding caused a reduction of−17.6%. Drainage plus mounding reduced the total greenhouse budget by 6.9%, while drainage plus mounding plus fertilisation increased the total greenhouse budget by 101.8%. There is a potential for up-scaling GHG emissions from newly drained peaty gley for inclusion in the UK Land Use Land-Use Change and Forestry (LULUCF) Greenhouse Gas Inventory. However up-scaling and evaluation of the net emissions requires high quality data from different sites newly drained for afforestation. More studies are needed if net fluxes from newly drained sites are to the included in the LULUCF Greenhouse Gas Inventory.

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CHAPTER 1

INTRODUCTION

1.1 GENERAL BACKGROUND

Soil is an important natural resource that provides several important ecosystem functions, such as a medium for plant growth, and plays a key role in the water, carbon (C) and nutrient cycles of terrestrial ecosystems (Bauhus et al. 2002; Hopmans et al. 2005). The Earth's soils are a major reservoir of C in terrestrial ecosystems and contain three times as much C as plants (Post et al. 1982, 1990; Eswaran et al. 1993) and twice as much C as the atmosphere as carbon dioxide (CO_2) (Post et al. 1982; Davidson et al. 2000). World's soils have the capacity to sequester CO_2 emitted from anthropogenic activities if they are able to accumulate large quantities of soil organic matter (Post et al. 1982; Tian et al. 1998; 1999). The accumulation of organic matter in soils provides energy, substrate and the biological diversity that sustain numerous functions performed by soils (Franzluebbers 2002).

The world's forests are estimated to cover an area of 4.1×10^9 hectares (ha) (Dixon et al. 1994; Dixon and Wesniewski 1995) of which 187×10^6 ha are forest plantations (c.f., Zerva and Mencuccini 2005a). Forests of the world play an important role in the global C, nitrogen (N) and sulphur (S) cycles and sequester and conserve more C than all other terrestrial ecosystems (Dixon and Wesniewski 1995). Annually, forests account for some 90% of the C flux between the atmosphere and terrestrial ecosystems through their photosynthetic uptake and respiratory release of CO_2 (Dixon and Wesniewski 1995). Forest ecosystems may be major sources of atmospheric CO_2 if they are clearfelled and converted to other land uses such as agriculture or following timber harvesting, fires etc. (Ewel et al. 1987a; Gordon et al. 1987; Weber 1990).

Brumme et al (2005) suggested that forests may store large quantities of C in soil and vegetation, thus mitigate increasing atmospheric CO_2 and associated global climate change (Heath et al. 2005; Lal 2005). The amount of C stored in forest ecosystems depends on their location, type and the age of trees (Madeira et al. 2002). Post et al. (1982) summarised 2696 soil profiles from most terrestrial biomes and estimated that forest soils contain 34% of the global organic C. According to Dixon et al (1994), the world's forests contain 50% of the total terrestrial ecosystem C of 1146 Pg, of which 787 Pg are stored in soils and 359 Pg in vegetation. Forest plantations have the potential to be managed to sequester atmospheric CO_2, thus mitigate global climate change (Vitousek 1991; Madeira et al. 2002). Several studies have shown that intensively managed forest plantations have the potential to increase C sequestration by increasing net primary productivity (NPP) compared to degraded natural forests if they are composed of fast-growing trees growing on fertile sites (Delcourt and Hariss 1980; Gladstone and Ledig 1990; Lal et al. 1998; Liski et al. 2002; Madeira et al. 2002).

The majority of the world's forest plantations are located in the boreal and temperate region (Post et al. 1982; Jobbágy and Jackson 2000). The boreal and temperate planting sites are characterised by large stocks of organic matter, water-saturated soils, low temperatures and poor fertility (Bubier et al. 1998; Strom and Christensen 2007). The growth of trees in these soils is impossible without drainage and soil preparation to improve aeration and planting conditions (Payandeh 1973; Armstrong et al. 1976; Lieffers and Rothwell 1986; Macdonald and Lieffers 1990). Waterlogging limits the supply of oxygen to plant roots which inhibits aerobic respiration, plant growth and survival (Saglio et al. 1983). Several studies demonstrated that trees growing in saturated soils are stunted in growth (Coutts and Philipson 1978; Sena-Gomes and Kozlowski 1980a; Lieffers and Rothwell 1986). The lack of oxygen supply in saturated soils may limit deep rooting and lead to trees with shallow roots that are susceptible to windthrow (Coutts and Philipson 1978; Sena-Gomes and Kozlowski 1980a; Lieffers and Rothwell 1986). Most studies have concluded that stress caused by long-term waterlogging cause root decay and plant death (Pereira and Kozlowski 1977; Kozlowski and Pallardy 1979; Kozlowski 1984).

Despite the widespread nature of waterlogging, peatlands around the world support large areas of productive forest plantations that have been established following intensive site preparation (Sutton 1993; Roy et al. 1999). According to Armentano and Menges (1986) draining peatlands for agriculture and forestry has been the most significant land use change in Europe in the last century. About 15 × 10^6 ha of peatland have been drained to increase plant biomass production in forests, particularly in Fennoscandia and the former USSR (Armentano and Menges 1986; Paavilainen and Päivänen 1995). Large areas of deep peat and peaty gley soils have also been drained for forestry in the UK (Cannell et al. 1993; Byrne and Farrell 2005). Although current policy discourages further planting on peatland, a large estate of public and private forests now exists on organic soils in the UK and elsewhere. The NPP and biomass of the vegetation increase after drainage because of increased supply of oxygen in the root zone and the stimulation of mineralisation which increases the availability of nutrients (Dang and Lieffers 1989; Von Arnold et al. 2005; Minkkinen et al. 2008). The increase in tree growth following drainage depends on site fertility. Several studies demonstrated that trees planted on drained fertile peat soils grow faster than those planted on nutrient-poor soils (Laine et al. 1995a; Minkkinen et al. 1999; Laiho et al. 2003).

Water-saturated soils, low temperatures and poor aeration in peatlands have resulted in accumulation of organic matter as peat (Bubier et al. 1998; Borren et al. 2004; Ström and Christensen 2007; Ali et al. 2008). These ecosystems are net sinks for terrestrial C stocks (Alm et al. 1997; Vitt et al. 2000; Maljanen et al. 2001a; Wieder 2001; Turunen et al. 2002; Roehm and Roulet 2003). The carbon balance of a wetland is sensitive to changes in soil water table depth. During dry summers natural wetlands can act temporarily as sources of C to the atmosphere (Alm et al. 1999). Water saturated organic soils may produce insignificant amounts of N_2O (Martikainen et al. 1993a, Nykänen et al. 1995, 2002; Regina et al. 1996, Liikanen et al. 2002). On the other hand, natural organic soils in the boreal and temperate region are significant sources of atmospheric methane (CH_4) (Bartlett and Harriss 1993; Nykänen et al. 1998; Bohn et al. 2007). However, when drained for forestry and agriculture, peatland soils may revert from C sinks to sources (Martikainen et al. 1995; Nykänen et al. 1995, Silvola et al. 1996a; Minkkinen et al. 2007). Similarly, when the soil water table depth is lowered permanently, fertile wetlands may become sources of N_2O (Martikainen et al. 1993a; Aerts and Ludwig 1997; Kasimir-Klemedtsson et al. 1997; Regina et al. 1996; 1999). In contrast, lowering the soil water table by drainage decreases CH_4 emissions in peatlands due to decreased production with increased oxygen availability and increased consumption of methane in the aerobic soil layers by CH_4 oxidising microorganisms (Moore and Knowles 1989; Roulet et al. 1993; Martikainen et al. 1995). A well-drained wetland may even be converted into a net sink for atmospheric CH_4 when methane consumption exceeds production (Freeman et al. 1993; Martikainen et al. 1995; Nykänen et al. 1998).

1.2 JUSTIFICATION

The British Isles has about 2.11×10^6 ha of deep peats (>45 cm depth), excluding lowland fens and an unknown area of shallow (<45 cm depth) peaty gley and iron pans (Cannell et al. 1993). Peatlands excluding fens are estimated to contain 3000×10^6 tonnes (t) of C, 76% of which is located in deep peats (Harrison et al. 1995). Forest ecosystems cover some 3×10^6 ha or 12% of Britain's land area (Forest Statistics 2008, http://www.forestry.gov.uk/website/forstats2008). Some 190 000 ha of deep peats and 315 000 ha of grassland on peaty gley soils are estimated to have been afforested with conifer species for commercial forestry between the 1950s and 1980s (Cannell et al. 1993; Hargreaves et al. 2003; Byrne and Farrell 2005).

Drainage followed by mounding is a common site preparation practice carried out prior to afforestation and replanting clearfelled plantations in upland peaty gley soils with seasonally high water table in the UK (Ball et al. 2007, Minkkinen et al. 2008). An increase in the growth rate of trees planted on drained peaty gley soils can be expected because of increased soil temperatures (e.g., Peterjohn et al. 1994; Kirschbaum 1995; Kätterer et al. 1998; Davidson et al. 1998) and increased oxygen availability (e.g., Hillman 1992; Peterjohn et al. 1994; Prevost et al. 1997) which may stimulate tree root growth and increase the rate of mineralisation of organic matter and nutrient cycling. Further management practices such as mounding and fertilisation carried out after drainage of peaty gley soils may further increase nutrient availability and improve the growth of planted trees. In turn, drainage, mounding and fertilisation may also alter the C and greenhouse gas (GHG) dynamics of peaty gley soils. Greenhouse gases (CO_2, CH_4 and N_2O) are needed to prevent heat escaping from the surface of the earth.

Most studies on soil organic C and GHG dynamics of drained peatlands across Europe pertain to boreal and continental peatlands (e.g., Martikainen et al. 1995; Minkkinen et al 1997; Minkkinen and Laine 1998a; Von Arnold et al. 2005). Although few studies have investigated the C and GHG dynamics of forested and clearfelled forests on deep peat and peaty gleys soil in the UK (Hargreaves et al. 2003; Zerva and Mencuccini 2005a, 2005b; Zerva et al. 2005; Ball et al. 2007), the effects of site preparation for afforestation on soil properties and GHG fluxes have not been experimentally tested in peaty gley soils under temperate maritime conditions such as in the UK.

1.3 AIMS OF STUDY

The aims of this study were to evaluate:

i) The effects of site preparation for afforestation on soil properties
ii) The effects of site preparation for afforestation on environmental variables
iii) The effects of site preparation on emissions of GHG (CO_2, CH_4 and N_2O)
iv) The relationships between GHG fluxes and environmental variables.

1.4 THESIS STRUCTURE

This thesis examines the effects of site preparation for afforestation on soil properties and GHG fluxes. The thesis consists of six chapters.

Chapter 1: This chapter introduces the importance of soils on GHG fluxes and forests on C sequestration. It also provides justification and objectives of the study. It provides a review of literature on the importance of soils on organic C storage. It also gives a review on the effect of land use changes and fertilisation on soil C storage GHG fluxes (CO_2 CH_4 and N_2O). Environmental factors affecting C storage and GHG fluxes are also reviewed in this chapter. The study site is also described in this chapter.

Chapter 2: This chapter investigates the effects of forest management practices (drainage, mounding and fertilisation) on soil properties and various pools of C and N. The experimental design layout and preparation are described here.

Chapter 3: The effects drainage, mounding and fertilisation on soil temperature, water content and water table depth are investigated here. This chapter also examines the effect of these practices on soil CO_2 efflux as well as relationships between soil CO_2 fluxes and environmental variables.

Chapter 4: In this chapter, the effects of drainage, mounding and fertilisation on soil CH_4 fluxes of peaty gley soils in upland Britain are examined. The dependence of CH_4 emissions on soil temperature, water content and water table depth is also examined here.

Chapter 5: This chapter represents the effect of drainage, mounding and fertilisation on N_2O emissions and the relationship between fluxes and environmental variables are also presented in this chapter.

Chapter 6: This chapter gives a summary of the main conclusions drawn from the integration of the entire body of work. The total greenhouse gas budget of the three practices in isolation or in combination is also summarised here.

1.5 LITERATURE REVIEW

1.5.1 The importance of soil in C storage

World's soils hold the second largest C pool in the global C cycle after the deep ocean (Schimel 1995). Soils contain 1500 Pg (1 Pg= 1 Gt = 10^{15} g) of C in the top one metre (Adams et al. 1990; Eswaran et al. 1993) and about 900 Pg of C in the depth between one and two metres (Batjes 1996). The amount of organic C stored in soils varies with ecosystems, land use and management practices (Jobbágy and Jackson 2000). About 49 % of the world's organic C stocks are stored in boreal forests, 37% in tropical forests and 14% in temperate forests (Post et al. 1982). Northern peatlands are net C sinks because their saturated soils and low temperatures limit microbial decomposition and lead to accumulation of organic matter (Gorham 1991; Borren et al. 2004; Ali et al. 2008).

1.5.2 The importance of soil organic matter

Soils play a significant role in the formation, maintenance and turn over of organic matter. Soil organic matter determines the biological, physical and chemical properties of soils (Merino et al. 2004; Grace et al. 2006). The accumulation and storage of organic matter in soils depend on addition of fauna and flora and/or loss of C from the soil through heterotrophic and autotrophic respiration and physical losses through natural and human disturbances (Baldock and Skjemstad 2000; Johnson et al. 2006). Maintenance of soil organic matter is particularly important in sustaining the productivity of many ecosystems (Henderson 1995; Hirschel et al. 1997; Kirchmann et al. 2004). In many ecosystems, the decomposition of organic matter is the only available source of plant nutrients (Damman 1978; Van Cleve et al. 1983). The amount and quality of organic matter stored in soils depend on vegetation type (Wieder and Yavitt 1991; Finer 1996), the soil physical and chemical environment, the chemical composition of the litter and its physical accessibility to soil microorganisms (Melillo et al. 1982; Taylor et al. 1989; Kemp et al. 1994; Jastrow and Miller 1997; Baldock and Skjemstad 2000; Gleixner et al. 2001).

The storage of organic matter in forest ecosystems depends on forest type and site fertility. The decomposition rates of organic matter vary with plant type (Collins et al. 1992; Henriksen and Breland 1999; Kirchmann et al. 2004). For instance, litter from a coniferous forest may decompose slowly because of its high lignin and low soluble carbohydrate contents (Melillo et al. 1989; Hobbie 1996). In contrast, litter from a deciduous forest may decompose rapidly due to its high N and low lignin contents (Jensen 1974; Millar 1974; Moore 1984; Hobbie 1996). Therefore, litter in a coniferous forest may resist microbial decomposition, thus accumulate and stay longer on the forest floor than litter in a deciduous forest. In forest ecosystems, organic matter is added to the soil as above-ground and below-ground litter (Tate 1987; Cardon 1996; Högberg et al. 2001) and some of it may be lost through microbial decomposition (Post et al. 1982; Tate 1987; Gaudinski et al. 2000). Microbial decomposition can affect ecosystem C balance by releasing CO_2 from soil organic matter that ranges in ages from recent (e.g., fine roots turnover) to years and millennia (e.g., litter and humified soil C) (Trumbore 2000).

1.5.3 Effects of forest management practices on soil C storage

Over the last century land use changes, such as deforestation in the tropics, have decreased organic C stocks (Schimel 1995; Houghton 2003). Globally, soils have lost between 40 and 90 Pg C through cultivation and other disturbances (Schimel 1995; Houghton 1999; Houghton et al. 1999; Lal 1999). The drainage of peatland soils for arable agriculture and forestry has also contributed to C losses (Minkkinen and Laine 1998b). Site preparation practices used for planting tree seedlings and for controlling competing vegetation in peatland soils, mostly involves some form of mounding (Sutton 1993). Mounds are created by disking or excavation (Trettin et al. 1995). Mounds created by disking mix surface soil horizons into the centre of the planting bed, while mounds created by excavation do not mix the surface soils (Attiwill et al. 1985; Sutton 1993). The degree of soil mixing after site preparation for afforestation or for replanting clearfelled stands greatly affects organic matter decomposition (Ross and Malcolm 1988). An increase in organic matter decomposition is expected in peat soils following drainage when soil temperature is increased and aeration is improved (Mann 1986; Post and Mann 1990; Hogg et al. 1992). The same is true to some degree for mounding which has been reported to improve aeration and to increase soil warming (Bridgham et al. 1991; Sutton 1993).

There has been some evidence in recent years that drainage and afforestation can decrease soil organic C also in peaty gley soils in the UK (Zerva and Mencuccini 2005a, Zerva et al. 2005) presumably because of increased oxygen availability that increase microbial decomposition (e.g., Brake et al. 1999). For example, Zerva et al. (2005) measured 140 ± 15 t C ha^{-1} in a drained 40-year-old Sitka spruce stand on a peaty gley soil and 274 ± 54 t C ha^{-1} in an unplanted grassland at Harwood Forest. A meta-analysis of literature on the effect of land use change on soil organic C stocks found that on average soil organic C can decrease by 42% when forests are converted to agriculture (Guo and Gifford 2002). Establishing a coniferous forest on land previously used for grazing may decrease soil organic C by 12% (Guo and Gifford 2002). According to Post and Mann (1990) and Davidson and Ackerman (1993), converting forests or grasslands to arable agriculture may decrease soil organic C stocks by 20 to 50%. Dias–Filho et al. (2001) estimated that converting forests to pasture may decrease above-ground biomass C by 100–200 tonnes ha^{-1} yr^{-1}. Land use changes and other disturbances such as fires may lead to larges losses of organic C that have accumulated in soil and vegetation over decades or thousands of years (Pypker and Fredeen 2002; Law et al. 2003; Wang et al. 2003; Kolari et al. 2004). Forest management practices such as drainage (Martikainen et al. 1995; Nykänen et al. 1995; Silvola et al. 1996a) and timber harvesting (Fernandez et al 1993; Ewel et al. 1987a; Zerva and Mencuccini 2005a) may cause loss of C to the atmosphere as CO_2. For example, Silvola et al. (1996a) found that lowering the soil water table by 12 to 40 cm in Finland doubled CO_2 emissions. Ewel et al. (1987a) also found that soil respiration was higher in a clearfelled stand than uncut forest. Musselman and Fox (1991) reported that clearfelling on all soils in the USA decreased soil C pools by 25 to 50%. According to Johnson and Curtis (2001) and Guo and Gifford (2002) about 6 to 13% of soil organic C may be lost from clearfelled forest plantations if they are left to regenerate naturally or replanted with more productive tree species. Soil water table depth greatly affects soil aeration; hence changes in the water table depth following clearfelling would have an impact on organic matter decomposition. Larger amounts of organic matter have been shown to accumulate in clearfelled stands on wet soils than in their counterparts on dry soils (Mattson and Swank 1989; Mroz et al. 1985; Johnson and Todd 1998) probably because anaerobic conditions induced by the high soil water table limit the rate of organic matter decomposition (Zerva and Mencuccini 2005a). According to Johnson and Curtis (2001) and Crimczik et al (2005), clearfelling has little or no effect on organic C in the top mineral soil if site preparation used for replanting is not intensive. According to Cannell et al. (1993) and Minkkinen et al. (2001), forestry practices such as drainage and thinning may increase soil organic C stock by 6 to 12 kg C m^{-2} in forest stands in the first rotation (60 to 100 years) depending on site type and climate. The loss of C due to land use change and other disturbances may persist for several years after they occurred (Schulze et al. 1999; Rannik et al. 2002; Kowalski et al. 2003; Zerva et al. 2005).

1.5.4 Effect of forestry machinery on soil C stocks and effluxes

In comparison with tree planting in areas with well-drained soils, site preparation carried out prior to afforestation and replanting clearfelled stands in temperate and boreal forest sites is highly mechanised. Site preparation involves the use of heavy equipment which can cause considerable disturbance to soil and lead to loss of organic C. Mechanised site preparation may cause soil compaction and alter a number of soil properties such as bulk density, air-filled pore space and hydraulic conductivity (McNabb et al. 2001; Xu et al. 2002). Losses of soil organic matter and nutrients have been reported after site preparation (Farrish et al. 1993; Rab 1994; Turner and Lambert 2000). Soil erosion (Bormann and Likens 1979) and leaching (Johansson 1994) may also lead to loss of organic matter after site preparation. Changes in the physical and chemical properties of soils following site preparation in some cases can affect site productivity (Brown and Lugo 1990; Jurgensen et al. 1997) and the survival and the early growth of planted trees (Smith et al. 1994; Huang et al. 1996; Rab 1996).

1.5.5 Carbon dioxide emissions from soil

The soil CO_2 efflux or soil respiration is a major C flux from terrestrial ecosystems to the atmosphere and is an important component of the global C cycle (Schimel 1995; Houghton 1995; Raich and Tufekcioglu 2000). According to Raich and Schlesinger (1992) and Wagai et al (1998), soil respiration accounts for 25% of the C exchange between the biosphere and the atmosphere. Globally, soil respiration varies among biomes (Raich and Schlesinger 1992; Rustad et al. 2000), vegetation types (Raich and Schlesinger 1992; Wagai et al. 1998; Raich and Tufekcioglu 2000) and seasons (Raich and Tufekcioglu 2000; La Scala et al. 2000). Raich and Tufekcioglu (2000) found that grassland soils respire more than forest soils. A review of literature by Raich and Schlesinger (1992) found that soil respiration was higher in tropical lowland forests (1092 g m^{-2} yr^{-1}) and temperate forests (662 g m^{-2} yr^{-1}). The review also revealed that soil respiration in cultivated land (544 g m^{-2} yr^{-1}) and in the boreal region (322 m^{-2} yr^{-1}) was intermediate and low in the tundra (60 m^{-2} yr^{-1}), swamps and marshes (200 m^{-2} yr^{-1}) and scrub desert vegetation.

Globally, soil respiration is estimated to release 68–75 Pg C to the atmosphere annually (Raich and Schlesinger 1992; Raich and Potter 1995; Mosier 1998; Schlesinger and Andrews 2000). Soil respiration is a composite flux that includes respiration of soil organisms and plant roots and organic matter decomposition, and the subsequent release of CO_2 at the soil surface (Raich and Nadelhoffer 1989; Malhi et al. 1999; Buchmann 2000; Hanson et al. 2000; Savage and Davidson 2001; Law et al. 2003; Wan and Luo 2003; Doff et al. 2004; Maljanen et al. 2006; Subke et al. 2006). According to Hanson et al (2000) and Raich and Tufekcioglu (2000), root respiration contributes more CO_2 to total soil respiration than all other processes. Several studies estimated that root respiration contributes 30–70% to the total soil respiration in temperate forests (Boone et al. 1998; Andrews et al. 1999; Buchmann 2000). Root respiration is estimated to contribute 50–93% to the total soil respiration in the Artic tundra (Raich and Tufekcioglu 2000). Dugas et al. (1999) estimated that root respiration in grasslands contribute 17–60% to total soil respiration.

1.5.6 Factors affecting soil respiration

Soil respiration is affected by soil temperature (Mallik and Hu 1997; Buchmann 2000; Lafleur et al. 2005; Mäkiranta et al. 2007; Minkkinen et al. 2007) and soil moisture (Singh and Gupta 1977; Schlentner and Van Cleve 1985, Davidson et al. 1998; Buchmann 2000). According to Singh and Gupta (1977) and Raich and Schlesinger (1992), the interaction between soil temperature and soil moisture influences soil organic matter decomposition and productivity in many ecosystems. Several studies reported a positive correlation between soil temperature and soil respiration in various ecosystems and soil types (Buchmann 2000; Davidson et al 2000; Fang and Moncrieff 2001; Wiseman and Seiler 2004). According to Ohashi et al. (1999), Buchmann (2000) and Pumpanen et al. (2003), soil temperature is the main driver of soil respiration processes in ecosystems when soil moisture and other factors are not limiting. In many ecosystems, soil respiration follows seasonal temperature patterns (Anderson 1973; Rochette et al. 1991). Trettin et al (1995) found that maximum soil respiration in many ecosystems occur in the summer when temperatures are at maximum. Akinremi et al (1999) and Parkin and Kaspar (2003) found that soil respiration was low in the early morning, rises as the day progresses and reaches maximum values in the mid-afternoon. Soil respiration is high in tropical forests because soil temperature and moisture are not limiting NPP and organic matter decomposition (Rhoades et al. 2000; Raich et al. 2002).

Various components of soil respiration (e.g. root respiration, oxidation of plant detritus and organic matter heterotrophs) exhibit different sensitivities to temperature (Kirschbaum 1995; 2000; Trumbore et al. 1996). It is currently assumed that the decomposition of labile carbon is sensitive to variation in temperature, while the recalcitrant carbon is insensitive (Liski et al. 1999; Giardina and Ryan 2000; Thornley and Cannell 2001; Lenton and Huntingford 2003). This may suggest that soil warming will have a greater effect on the rates of CO_2 emitted from soils in the boreal and tundra region (Niklinska et al. 1999) because these soils have the largest stock of labile carbon (Schlesinger and Andrews 2000). The residence times of different soil organic fractions are controlled by external factors such climate etc. and its decomposability (Vanhala et a. 2008). The temperature sensitive of soil organic matter decomposition is commonly modelled as a Q_{10} function (e.g., Kirschbaum 1995; Rustad et al. 2000; Fang et al. 2005). The Q_{10} for a reaction rate is defined as the factor by which the rate increases with a 10°C rise in temperature (Parkin and Kaspar 2003; Davidson and Janssens 2006). The most commonly reported Q_{10} values range between 2 and 4 (Kirschbaum 1995; Swanson and Flanagan 2001). It has been found that the Q_{10} varies with soil temperature (Howard and Howard 1993; Lloyd and Taylor 1994; Kirschbaum et al. 1995, 2000, 2006; Tjoelker et al. 2001; Leifield and Fuhrer 2005) and the depth at which temperature is measured (Kirschbaum 1995; Swanson and Flanagan 2001). Howard and Howard (1993) subjected different soils to a wide range of temperatures and observed Q_{10} values ranging between 2.01 and 2.83. De Boois (1974) found a Q_{10} value of 3 in the upper litter layer and 2 in the humus layer at temperatures ranging from 5–20°C in a forest soil profile. They concluded that the response of soil microbes to temperature decreased down the soil profile as organic matter was exhausted.

Soil moisture content also influences soil respiration and in general dry soils respire less than their wet counterparts (Davidson et al. 2000; Maier and Kress 2000; Mielnick and Dugas 2000; Pangle and Seiler 2002; Reichstein et al. 2002). Low soil moisture suppresses the soil microbial population (Schimel et al. 1999) and activity (Skopp et al. 1990) and therefore limits soil respiration. Similarly, saturated soils may limit heterotrophic and autotrophic respiration. Rochette et al. (1991) found that soil respiration increased by 90% following rainfall which fell after a long dry period.

Davidson et al. (2000) measured soil respiration in a primary forest, a secondary forest, an active cattle pasture and a degraded cattle range in eastern Amazonia. They found that soil respiration decreased from wet to dry season in all land uses. Besides soil temperature and moisture, soil respiration may be affected by other factors such as NPP, root biomass, microbial population and activity; root N concentrations, soil texture, amount and quality of substrate (Boone et al. 1998; Buchmann 2000; Dilustro et al. 2005).

1.5.7 Effect of fertilisation on soil organic C and effluxes

The growth of trees in boreal and temperate forests is limited primarily by N (Vitousek and Matson 1985; Munson et al. 1993, Elser et al. 2007; LeBauer and Treseder 2008). Nitrogen in boreal and temperate ecosystems is bound in soil organic matter (McLaughlin et al. 2000; Saari et al. 2004) and a very small fraction (~1%) is available in the form that can be utilised by plants (Saari et al. 2004). For forest plantations established on drained organic soils, drainage may increase the rate of mineralisation of organic matter (Mann 1986; Davidson and Ackermann 1993) and increase nutrient availability (McLaughlin et al. 2000). Fertilisation is a forest management practice commonly used to increase the initial growth of forest stands established on infertile soils (Allen et al. 1990; Ohtonen et al. 1992). The addition of fertilisers to infertile soils, particularly N, may increase C storage in forests by increasing litter production and also by reducing the decomposition of organic matter (Mäkipää 1995; Hobbie et al. 2002; Franklin et al. 2003; Bowden et al. 2004; De Vries et al. 2006; Crane et al. 2007; Magnani et al. 2007; Hyvönen et al. 2008). The effect of N fertiliser on organic matter decomposition may be significant, although the mechanism behind the reduction is poorly understood (Franklin et al. 2003; Grandy et al. 2008; Reay et al. 2008). According to Ågren et al (2001), the decrease in organic matter decomposition in forest soils following fertilisation may be caused by increases in the microbial production-to-assimilation ratio, a decrease in litter quality, and/or a decrease in microbial population. Aber et al (1993) found that the addition of N at 50 kg N ha^{-1} yr^{-1} increased soil organic C storage by 20–40% in boreal and temperate forests.

Recent studies conducted in boreal and temperate forests of Western Europe and the United States found that the net forest C sequestration is driven by N deposition (Sitaula and Bakken 1993; Lovett 1994; Högberg, 2007; Magnani et al. 2007; Pregitzer et al. 2008). Lovett (1994) estimated that N deposition may add up to 50 kg N ha yr^{-1} to forests in Eastern and North America and Central and Western Europe. Townsend et al. (1996) estimated that N deposition may increase organic C storage by 0.44–0.74 Pg C yr^{-1} in forest vegetation worldwide. In a recent meta-analysis, Knorr et al. (2005) found that litter decomposition was stimulated at sites with low N deposition (<5 kg ha^{-1} yr^{-1}) and litter with low lignin, whereas organic matter decomposition was reduced at sites with moderate levels of N deposition (5 to 10 kg ha^{-1} yr^{-1}) or litter which have high lignin levels. It has been suggested that root respiration in forests may increase with increasing above-ground biomass production following fertilisation (Nadelhoffer et al. 1985; Hanson et al. 2000). Other studies reported that root respiration in forest trees may decrease if C allocation to roots decreases in response to increased soil fertility caused by fertilisation (Cannell and Dewar 1994; Giardina et al. 2003).

1.5.8 Measurement of soil respiration and other trace gases

Because of its large temporal and spatial variability and dependence on many environmental variables and other factors, soil respiration is difficult to measure at ecosystem level (Norman et al. 1997; Lund et al. 1999). The early studies of soil respiration were performed under laboratory (Lundegardh 1927) or under agronomic conditions (Kucera and Kirkham, 1971). Efforts have been made in recent years to achieve a complete understanding of heterotrophic and autotrophic processes that occur in situ. The first soil respiration studies were conducted in forest ecosystems (Leith and Ouellettee 1962; Witkamp 1966; Schulze, 1967; Reiners 1968). Several laboratory and field studies on soil respiration have been conducted in many ecosystems in recent years (e.g., Trumbore et al. 1996; Liski et al. 1999; Kirschbaum 2000; Luo et al. 2001).

Several methods have been developed for measuring soil respiration in the field. Soil respiration can be measured by closed static chambers (Beyer 1991; Grahammer et al.

1991), open dynamic chambers (Hanson et al. 1993; Vose et al. 1995) and closed dynamic chambers (Rochette et al. 1991; Kim et al. 1992; Norman et al. 1997; Bremer et al. 1998). Comparisons between different chamber types have shown significant differences in soil respiration rates (Jarvis and Rayment 1997; Norman et al. 1997; Fang and Moncrief 1998; Janssens and Ceulemans 1998; Janssens et al. 2000; Rayment 2000; Pumpanen et al. 2003; 2004a; Zerva et al. 2005). Each method has its advantages and disadvantages (Nakayama 1990; Norman et al. 1992). In the past, different chambers were compared against unknown soil respiration rates, until Pumpanen et al. (2004a) compared them against known CO_2 fluxes ranging from 0.32 to 10.01 μmol CO_2 m^{-2} s^{-1}. They found that static chambers underestimated CO_2 fluxes by up to 35%. The closed static chamber has been recommended for measuring soil respiration and other trace gases (e.g., CH_4 and N_2O) because of its affordability and flexibility (Holland et al. 1999). The closed static chamber differs with an open top chamber because it has to be installed permanently in the field or days before measurements are conducted to avoid disturbing the soil. An open top chamber can be left in one position to monitor soil respiration continuously (Fang and Moncrief 1998) while fluxes collected using the closed static chamber requires a gas chromatography for analysis.

1.5.9 The role of soil on CH_4 emissions

Methane is a GHG. Its concentration in the atmosphere has increased from the pre-industrial level of 0.75 μmol mol^{-1} to the current level of 1.75 μmol mol^{-1} (Lelieveld et al. 1998; Schimel 2000; Smith et al. 2003). The global warming potential (GWP) of CH_4 for a time horizon of 100 years is 25, which means that 1 kg of atmospheric CH_4 is 25 times as effective in absorbing radiation as 1 kg of atmospheric CO_2 (IPCC 2007). Methane is estimated to contribute 20% to anthropogenic global warming (Dalal and Allen 2008). The concentration of CH_4 in the atmosphere has been increasing at an average global concentration of about 0.5 to 1% annually during the past several decades (Steele et al. 1992; IPCC 2007). The increase has been attributed to anthropogenic activities such as fossil fuel exploitation, biomass burning, rice production, digestive processes from ruminants, sewage treatment plants and landfill use (Crutzen 1991; Lelieveld et al. 1998). Globally, total fluxes from human and natural sources are estimated to contribute 600 Tg CH_4 yr^{-1} (Lelieveld et al. 1998: Smith 2005; Denman et al. 2007).

Methane emitted from soil to the atmosphere is the balance between production and consumption by methanogenic and methanotrophic bacteria, respectively (Sundh et al. 1994; 1995; Dutaur and Verchot 2007; Chen et al. 2009). The production of CH_4 by methanogens occurs during the decomposition of organic matter in anaerobic environments (Alm et al. 1997; Carrol and Crill 1997; Lloyd et al. 1998; Nykänen et al. 1998; Hou et al. 2000; Yavitt and Williams 2000; Le Mer and Roger 2001).

The reaction between CH_4 and hydroxyl radicals (OH) in the atmosphere is the major sink for atmospheric CH_4 (Wang and Ineson 2003). The reaction is estimated to remove 490±85 Tg CH_4 yr^{-1} or equivalent to 80 to 90% of CH_4 from the atmosphere (Crutzen 1991; Lelieveld et al. 1998). Microbial oxidation of atmospheric CH_4 in well-drained soils is an important biological sink for CH_4 in terrestrial ecosystems (Adamsen and King 1993; Sundh et al. 1994; Castro et al. 1995; Sitaula et al. 1995; Butterbach-Bahl et al. 1998; Roura-Carol and Freeman 1999; Smith et al. 2000). According to Singh and Tate (2007), microbial oxidation of CH_4 in well-drained soils is performed by ubiquitous, aerobic and Gram-negative methanotrophic bacteria. The oxidation of CH_4 in well-drained soils is estimated to remove 22–100 Tg from the atmosphere annually (Smith et al. 2000; Castaldi et al. 2006; Dutaur and Verchot 2007). Steudler et al (1989) estimated that temperate and tropical forest soils contribute to the removal of 37% of atmospheric CH_4 consumed in terrestrial ecosystems. Several studies estimated that boreal forest soils consume 0.5–9 kg CH_4 ha^{-1} yr^{-1} (Ambus and Christensen 1995; Kasimir-Klemedtsson and Klemedtsson 1997).

1.5.10 Effects of environmental variables on CH_4 emissions

Various factors affect the production and consumption of CH_4 in soils. They include soil water table depth (Granberg et al. 1997; Liblik et al. 1997; Tuittila et al. 2000 Frenzel and Karofeld 2000; Yang et al. 2006; Ding and Cai 2007), soil temperature (Castro et al. 1995; Granberg et al. 1997; Daulat and Clymo 1998; Saarnio et al.

1998; Ding and Cai 2007), soil moisture content (Granberg et al. 1997; Liblik et al. 1997; Hargreaves and Fowler 1998), amount and quality of organic substrate (Czepiel et al. 1995; Bossio et al. 1999; Joabsson et al. 1999; Ström et al. 2003), N content and pH (Hütsch et al 1994; Sitaula et al. 1995; MacDonald et al. 1997; Hütsch 1998; Singh et al. 1999). Liblik et al. (1997) studied CH_4 fluxes in a Canadian peatland and found that changes in water table depth explained 62% of the variability observed in CH_4 fluxes. Several studies demonstrated that CH_4 fluxes increase with temperature between 2 and 39°C (Dunfield et al. 1993; Kotsyurbenko et al. 1993; Castro et al. 1995). According to Dise (1992), soil CH_4 emitted from terrestrial wetlands during the winter can contribute 4 to 21% of the annual emissions. Hütsch et al (1994) studied CH_4 fluxes in grassland soils. They found that soils with pH varying between 4.8 and 5.1 were a source of atmospheric CH_4, while oxidation occurred in similar soils limed to increase pH to 6.8. Hütsch (1998) found that a silty loam soil cultivated to lower soil pH from 8 to 7.1 emitted more CH_4 than before liming. Microbial population size and substrate availability also affect CH_4 emissions (Yavitt et al. 1987).

1.5.11 Effects of land use change and management on CH_4 fluxes

In general, land use changes such as converting forests and grasslands to arable land decrease CH_4 consumption (Mosier et al. 1991; Dobbie et al. 1996; Smith et al. 2000; Ball et al. 2002; Maljanen et al. 2003a; Merino et al. 2004; Tate et al. 2007). For example, Dobbie and Smith (1996) measured soil CH_4 fluxes in Scotland and found that a mixed deciduous forest soil consumed more CH_4 (2.19 to 2.97 mg m^{-2} d^{-1}) than the adjacent cultivated land used for arable agriculture (0.82 mg CH_4 m^{-2} d^{-1}). The decrease in CH_4 consumption after land use change results from the disturbance effect on the population and activity of CH_4 oxidising soil microorganisms (Knief et al. 2003; Seghers et al. 2003; Tate et al. 2007). Soils that have been out of cultivation for many years have been shown to consume CH_4 ten times faster than recently cultivated soils (Willison et al. 1995).

Drainage can decrease CH_4 emissions in peatlands due to decreased production and increased CH_4 oxidation caused by improved aeration on the peat surface (Roulet et al. 1993; Glen et al. 1993; Martikainen et al. 1995; Nykänen et al. 1995). However, draining nutrient-rich organic soils may decrease CH_4 oxidation because the enhanced decomposition of organic matter increases ammonium (NH_4^+) availability (Bradford et al. 2000; Castro et al. 2000), mimicking the effects of fertilisation (see next section).

1.5.12 Effect of fertilisation on CH_4 fluxes

Studies conducted under field and laboratory conditions have shown that N fertilisation decreases soil CH_4 oxidation in many ecosystems (Steudler et al. 1989; Mosier et al. 1991, 1996; Adamsen and King 1993; Castro et al. 1995; Boeckx and Cleemput 1996; Hütsch 2001). The most reported effect of N is that of NH_4^+, which decreased CH_4 consumption by up to 70% in several soils (Adamsen and King 1993; Castro et al. 1994; Crill et al. 1994; Gulledge et al. 1997; Powlson et al. 1997; Saari et al. 1997). The decrease in soil CH_4 oxidation in N fertilised soils may be caused by the effect of N on soil CH_4-consuming microbes and on the biochemical processes involved CH_4 oxidation in soil. According to Powlson et al. (1997), soils that had received ammonium nitrate at 144 kg ha^{-1} y^{-1} for over 150 years decreased CH_4 oxidation by 50% compared with unfertilised soils. Castro et al. (1994) found that CH_4 oxidation in a 26 year old slash pine (*Pinus elliottii* var. *elliottii* Englem.) site in Florida was 5–20 times lower in the fertilised plots than their unfertilised counterparts.

1.5.13 The role of soil in N_2O flux

Nitrous oxide is a powerful GHG in the troposphere which contributes to ozone depletion in the stratosphere (Cicerone 1987). Since the beginning of the industrial revolution the atmospheric concentration of N_2O has increased from 0.275 µmol $mol^{-1}$to the current level of 0.320 µmol mol^{-1} (Smith and Conen 2004) and has been increasing at an average global concentration of 0.2–0.30% per year (Flessa et al. 1995; Conrad 1996; Mosier et al. 1998a). A kilogram of atmospheric N_2O is 298 times as effective in absorbing radiation as 1 kg of atmospheric CO_2 over a time horizon of 100 years (IPCC 2007). The average atmospheric lifetime of N_2O is estimated at 150 years (IAEA 1992).

According to Kroeze et al. (1999), global N_2O emissions were 17.7 Tg N in 1994, of which 8.1 Tg were contributed by anthropogenic activities. Globally, soils are the most important source of atmospheric N_2O (Williams et al. 1992; Bouwman et al. 1993). Soil N_2O oxide is produced by microbial processes of denitrification and nitrification (Regina et al. 1996; Bremner 1997; Wrage et al. 2001; Machefert et al. 2002; Mosier et al. 2004; Koponen et al. 2006). The two processes are controlled by the availability of oxygen, which depends on soil water content (Robertson and Tiedje 1987). Denitrification is the reduction of NO_3^- to N_2 (Groffman et al. 1999; Smith et al. 2003). Denitrification is an anoxic process that is important in N_2O production and emission in saturated organic soils (Smith et al. 1998; Dobbie et al. 1999; Ruser et al. 2001). Nitrification is the oxidation of NH_4^+ or ammonia (NH_3) to nitrate via nitrite (Bollmann and Conrad 1998; McLain and Martens 2005) and is an oxic process which is important in N_2O fluxes in aerobic soils (Wrange et al. 2001).

Agriculture is estimated to contribute 80% to the global anthropogenic N_2O emissions, more than half of which is released directly from cultivated and fertilised agricultural soils (Mosier et al. 1998b; Kroeze et al. 1999; Gödde and Conrad 2000). Crutzen et al. (2008) estimated that agriculture contribute 4.3 to 5.8 Tg $N_2O–N\ yr^{-1}$ to the N_2O growth rate. Nitrous oxide is also emitted from agricultural soils indirectly through drainage streams, ground water, rivers and estuaries (Groffman et al. 1998; McMahon and Dennehy 1999; Reay et al. 2004). The increase in anthropogenic N_2O emissions is primarily caused by increased N input into agricultural soils (Mosier et al. 1998b). Recent studies have suggested that the increasing use of biofuels to reduce dependence on imported fuels to achieve “carbon neutrality” will increase atmospheric N_2O further because of emissions associated with N fertilisation (Crutzen et al. 2008). Other sources of N_2O are oceans, biomass burning and emissions from industrial processes and automobiles (Bange 2000).

1.5.14 Effect of environmental factors on soil N_2O flux

In soils, the production of N_2O is regulated by the interaction between soil properties, climatic factors and management practices (Granli and Bockman 1994) that affect soil microbial processes (denitrification and nitrification). Factors that affect N_2O production and emission from soils include soil properties (Velthof and Oenema 1995; Pihlatie et al. 2004; Syväsalo et al. 2004) soil moisture content (Goodroad and Keeney 1984; Pihlatie et al. 2004; Del Prado et al. 2006), water filled pore space (WFPS) (Dobbie et al. 1999; Dobbie and Smith 2001, 2003, 2006), soil temperature (Skiba et al. 1998; Smith et al. 1998; Koponen et al. 2006), availability of degradable organic substrate (Ineson et al. 1998), available N (NH_4^+ and NO_3^-)(Velthof et al. 1997), soil pH (Daum and Schenk 1998; Mogge et al. 1999) and plant species (Hénault et al. 1998; Ineson et al. 1998; Syväsalo et al. 2004; Niklaus et al. 2006).

Soil temperature controls many biological processes in soils and in case of N_2O production; it may affect microbial processes by stimulating N_2O-producing soil microorganisms. According to Skiba et al. (1998) and Smith et al. (1998), soil temperature exerts more control over soil N_2O production when other factors such as soil moisture content and substrate availability are not limiting. Other studies have demonstrated that the rates of denitrification and nitrification increase with increasing temperature (Goodroad and Keeney 1984; Weir and Gilliam 1986; Smolander et al. 1998). Goodroad and Keeney (1984) incubated soils in a laboratory and found that N_2O production increased when soil moisture was increased from 0.1 to 0.3 $cm^3\ cm^{-3}$. In contrast, Bowden et al. (1990) and Zerva and Mencuccini (2005a) found no relationship between soil moisture content and N_2O flux. Several studies demonstrated that nitrification is an important process in N_2O production at WFPS from 50 to 60% (Skiba and Ball 2002; Dobbie et al. 1999). According to Dobbie et al. (1999), denitrification starts just above 60% WFPS and reaches optimum rates between 80 and 85% WFPS. The WFPS depends on the balance between the amount of water entering the soil from precipitation or irrigation and the combined effect of evapotranspiration and drainage (Dobbie and Smith 2003, 2006).

Plants may affect N_2O production by modifying soil physical conditions and inputs of degradable organic substrate and thus influence soil microbial processes (Ineson et al. 1998). Klemedtsson et al. (1987) found that plant root growth increases the turnover of root material and thus increased denitrification by providing a substrate to denitrifying microorganisms.

1.5.15 Effects of fertilisation on soil N_2O flux

Nitrous oxide emissions from soils have been shown to increase after application of N fertiliser to agriculture and forest soils (Klemedtsson et al. 1997; Flessa et al. 1998; Kaiser et al. 1998; Dobbie et al. 1999; Kroeze et al. 1999; Baggs et al. 2000; 2003; Weitz et al. 2001; Erickson et al. 2001; Ruser et al. 2006). Other studies found that atmospheric N deposition also increases N_2O emissions (Brumme and Beese 1992; Butterbach-Bahl et al 1998; Gundersen et al. 1998; Skiba and Smith 2000). The addition of N fertiliser to soil provides a substrate for N_2O production by denitrification and nitrification (Hénault et al. 1998; Dobbie et al. 1999). Butterbach-Bahl et al. (1998) found that N_2O emissions from forests that have received large quantities of N through atmospheric deposition in the temperate zone of Europe were 2 to 5 times higher than those which have received low deposition. Brumme and Beese (1992) found that N_2O emissions from a beech forest that have received N at a rate of 35 kg N ha^{-1} yr^{-1} through atmospheric deposition in Germany were 5.6 kg N_2O-N ha^{-1} yr^{-1}. Other studies (e.g., Minami and Fukushi 1983) have demonstrated that non-N fertilisers such as calcium carbonate and phosphorus may increase N_2O production indirectly by increasing soil pH.

1.5.16 Effects of forest management practices on soil N_2O flux

Saturated soils of the boreal and temperate region are sinks or insignificant sources for atmospheric N_2O (Martikainen et al. 1993a; Regina et al. 1996; Johansson et al 2003) because water logging and low temperatures limit soil microbial processes. Field and laboratory studies have shown that drainage increases N_2O emissions of fertile peat soils (Kliewer and Gilliam 1995; Regina et al. 1998; Liikainen et al. 2002) by increasing the rate of mineralisation of organic matter (Updegraff et al. 1995) which increases N substrate for N_2O formation and emission. However, the effect of drainage on N_2O production and emission is more significant in nutrient-rich organic soils than in similar soils that are infertile.

Soil cultivation, including mounding, may increase the formation and emission of N_2O because of the fertilisation affect. Mounding generally mixes or buries the litter and the organic layer beneath the mineral layers (Saari et al. 2004). This may increase organic matter decomposition rates (Mann 1986, Davidson and Ackerman 1993) and increase N availability (Vitousek and Matson 1985; Fox et al. 1986; Vitousek et al. 1992) which may enhance the formation and emission of N_2O in fertile soils.

1.5.17 The study site

The field investigation was carried out in an experiment established on unimproved grassland located between two second rotation Sitka spruce stands at Harwood Forest. Harwood is located in NE England (55° 10' N, 2° 3'W), 30 km inland of the North Sea Coast. Harwood Forest was a site for the European CARBO-AGE project (Kowalski et al. 2004; Zerva et al. 2005; Zerva and Mencuccini 2005a and b) and is currently one of the core sites of the Centre for Terrestrial Carbon Dynamics (http://ctcd.group.shef.ac.uk/ctcd.html). The elevation varies from 200 to 400 m above sea level (Zerva and Mencuccini 2005a; Ball et al. 2007). The mean annual temperature and precipitation in the area are 7.6°C and 950 mm respectively (Conen et al 2005). The dominant soil type is a seasonally waterlogged peaty gley with a black-coloured organic-rich layer of depth varying from 15 to 40 cm (Zerva and Mencuccini 2005a; Zerva et al. 2005; Ball et al. 2007).

The forest covers 4000 ha and is dominated by Sitka spruce stands established on moorland grassland which had been used for grazing domestic livestock. The forest was originally established in the 1930s and further planting took place between the 1950s and the 1980s and most stands are now in their second rotation. Site preparation used for planting varies, with old stands planted on linear ridges made with single furrow ploughs and sites often have open ditches spaced at 20–30 m. Mounding has replaced ploughing as a method of soil cultivation prior to planting in the last three decades (Ball et al. 2007; Minkkinen et al. 2008). Mounding consists of mechanically excavating the soil to a depth of 30–40 cm and heaping it upside down next to the pit. The mounds are commonly spaced at 2 × 2 m intervals (i.e., 2500 pits/ha) to plant conifers throughout upland Britain. The study site is dominated by *Calluna vulgaris*, *Festuca ovina*, *Eriophorum vaginatum* and *Deschampsia flexuosa* and had been used for grazing domestic stock up to the year before the experiment started.

CHAPTER 2

EFFECT OF SITE PREPARATION FOR AFFORESTATION ON SOIL PROPERTIES AND VARIOUS POOLS OF CARBON AND NITROGEN

2.1 INTRODUCTION

The world's soils store more carbon than is present in the biomass and in the atmosphere (Jobbágy and Jackson 2000). They also contain more than 90% of the nitrogen of the terrestrial biosphere (Schlesinger 1986) and play a major role in the global carbon and nitrogen cycles (Neil et al. 1997). Terrestrial carbon stocks are sensitive to changes in land management (i.e., conversion of forests and grassland to cropland), climate and soil disturbances (Johnson 1992; Johnson et al. 2006). Forest plantations in the UK are often established on former grasslands on peaty gley soils that require drainage and mounding to lower the water table and prepare planting spots. These forest management practices may change physical, chemical and biological properties of soils (Jurgensen et al. 1997; Merino et al. 1998) and affect the amount and quality as well as the distribution of organic matter (Paul et al. 2002). It has been demonstrated that forest management practices used to prepare forest sites for afforestation and replanting clearfelled plantations decrease carbon and nitrogen at the soil surface (Tuttle et al. 1985; Tiessen et al. 1994; Munson et al. 1993; Zerva et al. 2005) as result of changes in soil temperature and soil moisture (Armentano and Menges 1986; Trettin et al. 1995) which increases the rate of organic matter mineralisation (Vitousek and Matson 1985; Fox et al. 1986; Smethurst and Nambiar 1990; Vitousek et al. 1992).

In some sites, disturbances caused by site preparation carried out prior to afforestation and replanting may affect the survival and early growth of trees as well as the productivity of subsequent plantations (Smith et al. 1994; Henderson 1995, Rab 1996). The storage of soil organic matter has a significant influence on soil properties such as bulk density, water holding capacity and cation exchange, which are important for the stability of ecosystems. The effects of site preparation for afforestation and replanting on soil carbon and nitrogen are important not only because they (C and N) are often the major factors determining soil quality but also because soils act as a source or sink of carbon on a global scale (Johnson and Curtis 2001). Disturbances caused by forest management practices (e.g., site preparation and timber harvesting) may turn soils to sources of carbon to the atmosphere (Detwiler and Hall 1988). Site preparation has been an integral part of forest management in Britain for many years (Avery 1990; Cannell et al. 1993). The main site preparation for afforestation and replanting clearfelled plantations in upland Britain involves mechanically lowering the soil water table depth by open drainage ditches. Ploughing has also played a crucial role in the expansion of forest plantations in the UK (Patterson and Masson 1999), although in more recent years it was replaced by mounding (Ball et al. 2007; Minkkinen et al. 2008). Although changes in soil properties and carbon stocks after afforestation have been quantified in different parts of the world (e.g. Paul et al. 2002; Zinn et al. 2002; Conen et al. 2005; Zerva and Mencuccini 2005b; Zerva et al. 2005; Wang et al. 2006), information on the effect of site preparation carried out prior to afforestation on soil properties and various pools of carbon and nitrogen in peaty gley soils is lacking.

The aim of this study was to examine changes in soil properties and various pools of carbon and nitrogen on a peaty gley soil following site preparation before trees are planted. We hypothesised that site preparation would change the soil bulk density and pH and increase C and N losses.

2.2 MATERIALS AND METHODS

2.2.1 Site description

A full description of the study site is outlined in Chapter 1. Briefly, the experiment was conducted at Harwood Forest (Northumberland, NE England). The experiment was established on an unplanted site located between two second rotation Sitka spruce stands (Fig. 2.1).

Figure 2.1. The experimental site at Harwood Forest before site preparation.

2.2.2 Experimental design and establishment

The experiment has a full factorial split-plot design with six plots measuring 30 × 8 m established in May 2006 (Fig. 2.2). Three plots were selected at random and were drained by cutting open drainage ditches placed 1.5 m from the plot edges and excavated to a depth of 65 to 70 cm (Fig. 2.3). Drained and undrained plots were isolated by 10 m wide buffer strips. Within each plot, four subplots measuring 8 × 6 m were established and two of them were chosen at random to be mounded. Spot mounds were spaced at 2 × 2 m (Fig. 2.4). Subplots were isolated by 2 m wide buffer strips. Spot mounds were made by turning the soil upside down adjacent to the dug pit (depth 30–40 cm, width 40 cm), thus burying the litter layer and organic horizons of the original soil beneath the mineral layer of mounds. Mounds were about 40 cm wide and 15 cm high. One random mounded, and one random unmounded subplot in each plot were given a compound fertiliser supplying 81 kg N ha^{-1}, 72 kg P ha^{-1} and 35 kg K ha^{-1} as recommended by Taylor (1991). Each treatment was replicated three times. The fertiliser was applied once on 11 June 2006. Hence, the main plots allowed testing for drainage effects, whereas the subplots allowed testing for fertilisation and mounding, in isolation or combined.

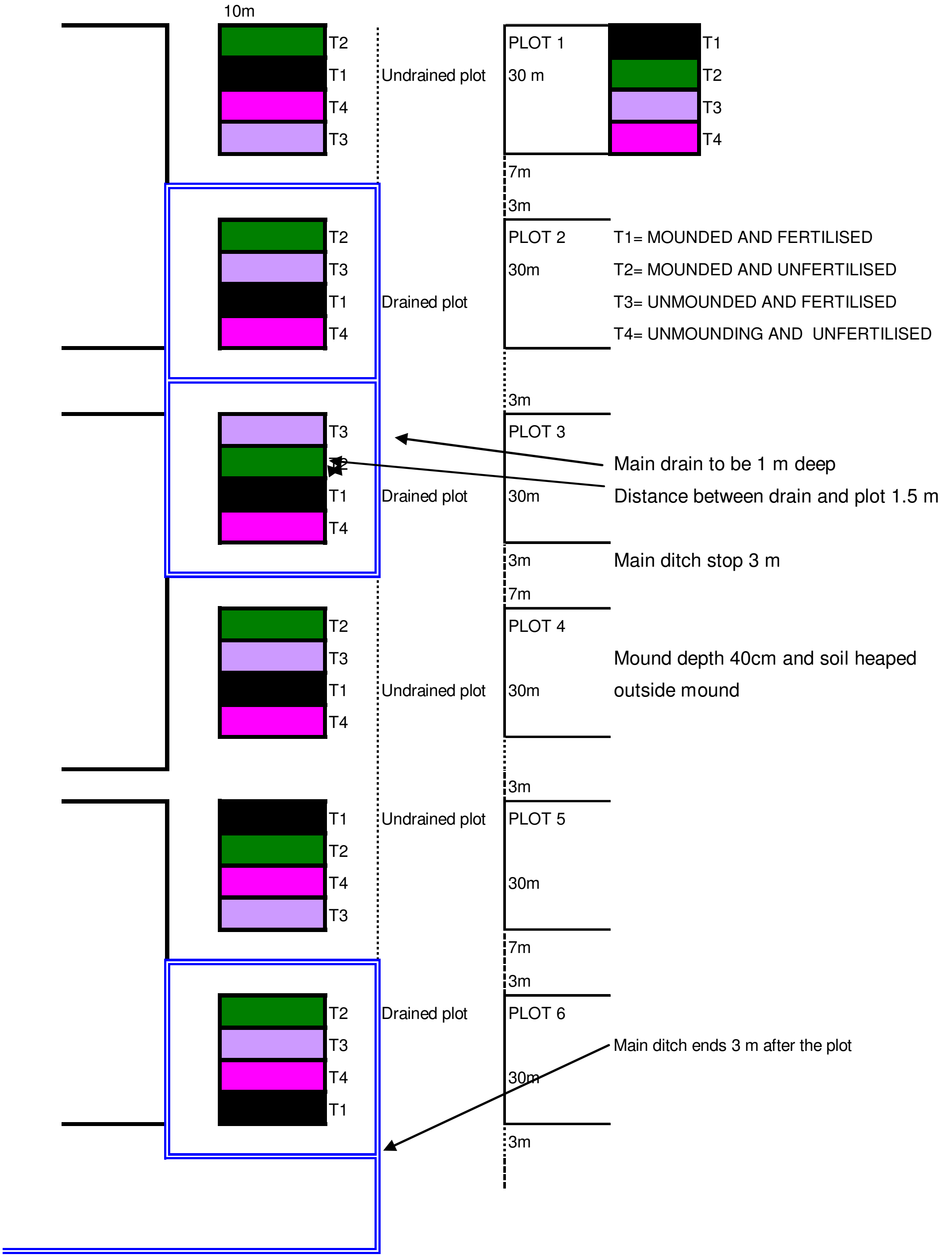

Figure 2.2. The layout of the experiment and Harwood Forest (the figure is not drawn to scale).

Figure 2.3. Open drainage ditches at the experimental site

Figure 2.4. Spot mounds were spaced at 2 × 2 m at the experimental site.

2.2.3 Soil sampling

Soil samples were collected in May 2006 (pre-samples) November 2006, February and August 2007, February and June 2008. All samples were collected to a depth of 20 cm, except in August 2007 and June 2008. Soil samples for August 2007 and June 2008 were taken at 0–10, 10–20 and 20–30 depth. In all sampling occasions, soils were collected from four randomly selected points in each subplot using a square (5 × 5 cm) corer. Soils were bulked by subplot and layer to make composite samples and bagged in black polythene bags for transportation to the laboratory in a cold box. Soil samples were kept in a cold room (3˚C) overnight awaiting preparation and analysis the following day.

2.2.4 Measurement for the determination of total C and N

Soil samples were passed through a 4 mm sieve and then oven dried at 60°C to constant weight. After drying, soil samples were passed through a 2 mm sieve using hand applied pressure before grinding in a Ball Mill (Glen, Creston, Sheffield, UK) to pass through a 0.5 mm sieve. Samples were then kept in high scintillation glass vials with linerless screw caps awaiting analysis. The samples were tested for water content before analysis. Sub-samples of about 5 grams were oven dried at 105°C for three hours (Allen 1989) for % soil moisture estimation. The soil moisture content was found to be between 1–3%, which is acceptable for the estimation of carbon.

Soil sub-samples of 4 mg for soil samples collected in the 0–10 and the 10–20 cm depth and 10 mg from samples collected in the 20–30 cm depth were weighed on a precision microbalance down to 3 decimals and a put in aluminium tin capsules. The capsules were moulded into small pallets and placed into an autosampler for loading into a Carlo Erba, NA 2500 C/N analyser for direct analysis by combustion. A combustion run contained 50 samples of soil and 7 samples of standards of known concentration. The C/N analyser gave the carbon and nitrogen totals in percentage. The mass of C and N in the core sample were calculated from:

$M_c = M_d \times C$ or N (%)/100

where: M_c is the total mass of C or N in the sample, M_d is the dry mass of the sample, and C or N (%) is the percentage obtained from the C/N analyser. The carbon and nitrogen concentration in soil samples was expressed in g kg^{-1}. The C/N analysers are considered to be more accurate and reliable that the loss on ignition method (LOI) (Sollins et al. 1999). .

2.2.5 Soil bulk density

The soil bulk density was determined for soil samples collected in May 2006 (pre-samples) and in August 2007 and June 2008. Pre-samples were collected to a depth of 20 cm without separating soil layers. The bulk density for samples collected in August 2007 and June 2008 was determined from soil cores collected at depths of 0–10, 10–20 and 20–30 cm depths. Soil samples were oven dried to constant weight at 105°C. Bulk density (P_b) was calculated by dividing the weight of dried samples by their fresh volume (Elliot et al. 1999)

$$P_b = M/V$$

where, P_b is the bulk density (g cm^{-3}), M is the dry mass of a given soil sample (g) and V is its fresh volume (cm^3).

2.2.6 Soil pH

Soil pH was measured from a soil-water suspension. About 25 ml of distilled-deionised water were added to 10 g of field moist soil (Wall and Hytönen 2005). The mixture was shaken in an orbital shaker for 10 minutes and the slurry was allowed to settle for 30 minutes before soil pH was measured with a glass electrode and a pH meter calibrated using two buffer solutions with values of 4 and 7.

2.2.7 Soil microbial biomass carbon

Soil microbial biomass carbon was determined on soil samples collected at a depth of 0 to 20 cm in November 2006 and February and August 2007 using the chloroform fumigation extraction method (e.g., Brookes et al. 1985; Vance et al. 1987). Twenty grams of soil were fumigated with ethanol-free chloroform for 24 hours in a vacuum oven (7627F Gallenkamp, Loughborough, England) containing a vial with soda lime. After fumigation, fumigated and control (non-fumigated) samples were extracted with 80 ml of 0.5 M K_2SO_4 on a reciprocal shaker (5B-6736B, Gallenkamp, Loughborough, England) set at 100 revolutions per minute (rpm) for 1 hour. After extraction, the solutions were transferred into 50 ml centrifuge tubes and centrifuged at 4000 rpm for about 10 minutes. The supernatant was then transferred into 20 ml plastic vials and filtered through 0.45 μm Millipore filters. Inorganic carbon was removed from the supernatant by acidifying to pH 2 using a concentrated phosphoric acid and purging with N_2 to degas samples. Organic carbon in the extracts was determined using an automated total organic carbon analyser (DC–80, Sartec Ltd., Kent, England) with a UV-persulphate oxidation and IR detector (Wu et al. 1990). Microbial biomass carbon was calculated as follows:

$$MB_{\mathrm{C}} = E_{\mathrm{C}}/K_{\mathrm{EC}}$$

where, E_{C} is the (organic carbon extracted from fumigated soils)–(organic carbon extracted from non-fumigated soils) and K_{EC} is the extractable component of microbial biomass carbon which is estimated to be 0.45 (Wu et al. 1990; Joergensen, 1996).

2.2.8 Inorganic N

Inorganic N (NH_4^+ and NO_3^-) was measured in samples collected in August 2007 and February 2008. Soil samples for the determination of available N were collected to a depth of 20 cm. Samples were prepared by sieving fresh soil samples through a 2.0 mm aperture sieve (Endecott Ltd., London). About 5 g of sieved fresh soil were weighed into glass bottles and 100 ml of 1 M KCl was added in each sample and sealed. The solution was thoroughly mixed on an orbital shaker (Gallenkamp, Loughborough, UK) set at 150 rpm for 1 hour. After shaking the mixed solution was filtered through a filter paper, Ashless Paper 2, (Whatman International Ltd, Maidstone, England). The extracts were kept in tightly capped vials at 3°C until analysed for NH_4^+ and NO_3^- by a continuous flow Series 3 Auto analyser system (Brann and Luebbe).

2.2.9 Above-ground plant biomass

Aboveground vegetation biomass was measured once in June 2007 from two 1 x 1 m quadrats which were established at random in each subplot. All plants within the quadrat were clipped at ground level using pruning shears, bagged in black polythene bags and transported to the laboratory. In the laboratory, samples were transferred into paper bags and oven dried to constant weight at 80°C. The biomass was expressed in tonnes per hectares of dry mass (t ha^{-1} DM).

2.2.10 Statistical analysis

All data were checked for normality and log-transformed when required. Analyses were carried out for each sampling date. The general linear model (GLM) was used for analysis of variance. The general linear model tested for effects of three main factors (drainage, mounding and fertilisation) entered as fixed factors and plot entered as random factor nested within drainage. The initial GLM included all possible second and third-order interactions. If interactions were not found to be significant, they were excluded and the model was run again without them to confirm the significance of the main factors. In case of significant interactions, the dataset was split and separate analyses were run for each combination. All analyses were run in Minitab 15 using the GLM procedure and the significance level was set at 0.05. Tukey's pairwise comparison test was applied to determine significant differences between treated and untreated plots/subplots.

2.3 RESULTS

2.3.1 Carbon and nitrogen concentrations

Mean soil organic carbon and total nitrogen concentrations are given in Table 2.1. The carbon concentration varied with depth from both sampling occasions (P=0.0001, Table 2.1), with the 20–30 cm (mineral layer) depth having the lowest soil organic carbon. Drainage decreased the soil organic carbon concentration in the 0–10 cm layer (P=0.03) at the end of the first year. Fertilisation also increased the soil organic carbon in the 0–10 cm layer (P=0.02) at the end of the first year but not in the 10–20 and the 20–30 cm soil layers (all P=0.5). Mounding had no effect on the soil organic carbon concentration in the 0–10, 10–20 and the 20–30 cm soil layers at the end of year 1 (all P=0.7). Drainage significantly decreased the soil organic carbon concentration at the end of study (June 2008) in the 0–10 cm layer (P=0.04) but not in the 10–20 cm (P=0.5) and the 20–30 cm (P=0.7) soil layers. In none of the measured layers was the soil organic carbon concentration affected by mounding or fertilisation at the end of study (Table 2.1).

The total N concentration in this site varied with soil depth (P=0.0001, Table 2.1) and was higher in the 0–10 cm and the 10–20 cm layer than in the 20–30 cm layer. The total N concentration was increased by drainage in the 0–10 cm layer in the first year of study (P=0.05) but not in the 10–20 cm (P=0.9) and the 20–30 cm (P=1.0) soil layers. The concentration of the total N was not affected by mounding or fertilisation in the first year of study (Table 2.1). In the second year of study, drainage significantly decreased the total N concentration in the 0–10 cm layer (P=0.04) but not in the 10–20 (P=0.5) and 20–30 cm (P=0.7) soil layers. Fertilisation or mounding did not affect the concentration of total N in either soil layer (Table 2.1).

The C/N ratio varied with soil depth (P=0.0001). The C/N ratio in the 0–10 cm soil layer was significantly increased by fertilisation in the first year of study (P=0.01) but not in the 10–20 (P=0.6) and the 20–30 cm (P=0.3) soil layer. Drainage and mounding did not affect the C/N ratio in the first year of study (Table 2.1). In the second year of study, the C/N ratio was not affected by drainage, mounding or fertilisation in either soil depth (Table 2.1).

Table 2.1. Effects of drainage, mounding and fertilisation on carbon and nitrogen concentrations, C/N ratio. Number (n) of sample is 12.

	Treatment			Treatment			Treatment		
Parameter/depth August 2007	Drained	Undrained	P–value	Mounded	Unmounded	P–value	Fertilised	Unfertilised	P–value
Total C (g kg^{-1})									
0–10	**419.2±13.37a**	**458.19±5.64a**	**0.03**	436.77±13.66a	440.69±9.60a	0.7	**451.16±7.48a**	**426.30±13.98a**	**0.02**
10–20	448.28±23.87a	452.01±15.53a	0.7	447.61±26.27a	452.69±10.98a	0.7	459.87±13.72a	440.43±24.62a	0.5
20–30	37.54±4.38a	37.22±4.57a	1.0	39.65±4.94a	35.11±3.83a	0.7	36.75±3.57a	38.01±5.22a	0.5
Total N(g kg^{-1})									
0–10	**18.31±0.79a**	**20.80±0.32a**	**0.05**	19.51±0.71a	19.60±0.71a	0.9	19.40±0.52a	19.71±0.86a	0.9
10–20	15.32±0.74a	15.35±0.72a	0.9	15.68±0.90a	14.99±0.49a	0.7	15.85±0.64a	14.82±0.78a	0.4
20–30	1.26±0.15a	1.23±0.14a	1.0	1.37±0.17a	1.12±0.10a	0.7	1.20±0.11a	1.30±0.17a	0.9
C/N ratio									
0–10	23.09±0.60a	22.09±0.47a	0.5	22.50±0.57a	22.68±0.55a	0.7	**23.37±0.49a**	**21.81±0.53b**	**0.005**
10–20	29.16±0.76a	29.68±0.56a	0.7	**28.52±0.70a**	**30.32±0.51b**	**0.04**	29.19±0.58a	29.65±0.74a	0.6
20–30	30.13±1.20a	29.97±1.10a	1.0	29.04±1.16a	31.06±1.06a	0.07	30.73±1.31a	29.37±0.92a	0.3
June 2008	Treatment			Treatment			Treatment		
Total C (g kg^{-1})	Drained	Undrained	P–value	Mounded	Unmounded	P–value	Fertilised	Unfertilised	P–value
0–10	**429.43±8.06a**	**460.92±5.02b**	**0.04**	446.17±8.13a	444.18±8.30a	0.8	449.04±7.86a	441.30±8.41a	0.4
10–20	444.17±16.54a	451.98±12.23a	0.5	452.23±17.72a	443.92±10.43a	0.8	447.20±15.03a	448.95±14.14a	0.9
20–30	35.81±3.82a	41.85±7.11a	0.7	39.90±4.43a	37.76±6.86a	0.5	35.89±3.26a	41.77±7.39a	0.7
Total N (g kg^{-1}))									
0–10	**17.12±0.33a**	**18.41±0.21b**	**0.04**	17.80±0.33a	17.72±0.34a	0.8	17.92±0.32a	17.61±0.34a	0.4
10–20	17.72±0.68a	18.04±0.50a	0.5	18.05±0.73a	17.71±0.43a	0.8	17.85±0.62a	17.92±0.58a	0.9
20–30	0.98±0.16a	1.23±0.29a	0.7	1.15±0.18a	1.06±0.28a	0.5	0.98±0.13a	1.22±0.30a	0.8
C/N ratio									
0–10	25.09±0.01a	25.04±0.01a	0.9	25.06±0.01a	25.06±0.01a	0.8	25.06±0.01a	25.07±0.01a	0.4
10–20	25.08±0.03a	25.06±0.02a	0.4	25.06±0.03a	25.07±0.02a	1.0	25.07±0.03a	25.06±0.03a	0.9
20–30	40.09±2.39a	38.64±2.21a	0.7	38.66±2.63a	40.06±1.92a	0.5	38.87±1.76a	39.86±2.75a	0.8

The "±" represent the standard error of mean. Different letters in bold following values within lines denote a statistically significant difference between the drained vs. undrained, mounded vs. unmounded and fertilised vs. unfertilised treatment ($P < 0.05$).

2.3.2 Soil bulk density and pH

Soil samples collected before the site was drained and mounded did not vary significantly in bulk density. The mean bulk density was 0.16±0.01 g cm^{-3} in the pre-drained and undrained plots, 0.15±0.01 and 0.16±0.01 g cm^{-3} in the pre-mounded and unmounded subplots. In the pre-fertilised and unfertilised subplots the mean soil bulk density was 0.17±0.01 g cm^{-3} and 0.16±0.01 g cm^{-3}, respectively. Figure 2.4 shows the bulk density of samples collected in August 2007 and June 2008. The soil bulk density increased with soil depth across treatments (P=0.0001, Fig. 2.4). Soil bulk density in the 0–10 cm layer ranged from 0.12 to 0.15 g cm^{-3}. In the 10–20 cm and the 20–30 cm layers, the soil bulk density ranged from 0.18 to 0.23 g cm^{-3} and 1.02 to 1.13 g cm^{-3} across treatments, respectively. Soil bulk density from both sampling occasions was not affected by drainage or fertilisation (Fig. 2.4). Mounding increased soil bulk density in the 0–10 cm layer in the first year (P=0.05) and second year of study (P=0.001), respectively. Mounding also increased the soil bulk density in the 10–20 cm layer in both dates (P=0.001, Fig. 2.4)

Soil pH in all treatments varied with soil depth (P=0.0001, Fig. 2.5). Soil pH varied from 3.9 to 4.0 in the 0–10 cm, 3.8 to 3.9 in the 10–20 cm and 4.1 to 4.2 in the 20 to 30 cm layer. Soil pH was not affected by drainage, mounding or fertilisation (Fig. 2.5).

2.3.3 Soil microbial biomass C

Mean values for soil microbial biomass carbon for all three sampling occasions are shown in Table 2.2. Soil microbial biomass carbon ranged from 1.54±0.28 to 3.50±0.07 mg g^{-1} and from 1.98±0.21 to 2.85±0.23 mg g^{-1} in the drained and undrained plots. Soil microbial biomass carbon was marginally increased by drainage in February 2007 (P=0.06). In the mounded and unmounded treatment, the soil microbial biomass carbon ranged from 1.92±0.25 to 3.27±0.20 and from 1.77±0.22 to 3.08±0.19 mg g^{-1}. The microbial biomass was slightly higher in the unfertilised subplots compared to their fertilised counterparts (Table 2.2). The soil microbial biomass carbon ranged from 1.74±0.22 to 3.06±0.17 and from 1.95±0.25 to 3.29±0.21

mg g^{-1} in the fertilised and unfertilised subplots, respectively. The three sampling dates differed significant in soil microbial biomass (P=0.0001). The microbial biomass in all treatments was lower in November 2006 and higher in August 2007.

2.3.4 Inorganic N

The mean values of inorganic N (ammonium and nitrate) in the 0–20 cm soil depth for all treatments are given in Table 2.3. Ammonium was not affected by drainage in August 2007 (P=0.18) and in February 2008 (P=0.3). Ammonium was significantly increased by fertilisation and mounding in August 2007 (all P=0.01), but not in February 2008 (Table 2.3). In none of the sampling occasions was nitrate affected by drainage, mounding or fertilisation (Table 2.3).

2.3.5 Aboveground plant biomass

Standing above-ground biomass measured as dry mass for all treatments is shown in Table 2.4. After a year of establishment, aboveground plant biomass in the drained plots (7.76 ±0.35 t ha^{-1} DM) was significantly higher than (P=0.04) in the undrained plots (6.03±0.30 t ha^{-1} DM). Aboveground plant biomass in the mounded subplots (6.84±0.36 t ha^{-1} DM) was not significantly different (P=0.9) from the unmounded subplots (6.95±0.47t h^{-1} DM). Aboveground plant biomass in the fertilised subplots (7.53±0.37 t ha^{-1} DM) was significantly higher than (P=0.002) in their unfertilised counterparts (6.26±0.37 t ha^{-1} DM).

Table 2.2. Effects of drainage, mounding and fertilisation on soil microbial biomass carbon (mg g^{-1}) based on a general linear model. Mean values ±standard error of means.

	Treatment			Treatment			Treatment		
Date	Drained	Undrained	P–value	Mounded	Unmounded	P–value	Fertilised	Unfertilised	P–value
November 2006	1.54±0.28a	2.14±0.13a	0.18	1.92±0.25a	1.77±0.22a	0.92	1.74±0.22a	1.95±0.25a	0.85
February 2007	2.69±0.30a	1.98±0.21a	0.06	2.57±0.30a	2.11±0.24a	0.17	2.30±0.34a	2.38±0.21a	0.89
August 2007	3.50±0.07a	2.85±0.23a	0.21	3.27±0.20a	3.08±0.19a	0.25	3.06±0.17a	3.29±0.21a	0.19

Different letters in bold following values within lines denote a statistically significant difference between the drained vs. undrained, mounded vs. unmounded and fertilised vs. unfertilised treatment (P< 0.05) (n=12)

Table 2.3. Effects of drainage, mounding and fertilisation on the concentration of inorganic N (mg g^{-1}) based on a general linear model. Mean values ±standard error of means.

	Treatment			Treatment			Treatment		
Variable & date	Drained	Undrained	P–value	Mounded	Unmounded	P–value	Fertilised	Unfertilised	P–value
August 2007									
NH_4^+	0.09±0.02a	0.06±0.02a	0.18	**0.10±0.02a**	**0.06±0.02b**	**0.008**	**0.11±0.02a**	**0.05±0.01b**	**0.008**
NO_3^-	0.02±0.01a	0.04±0.01a	0.32	0.03±0.01a	0.03±0.01a	0.96	0.02±0.01a	0.04±0.01a	0.23
February 2008									
NH_4^+	0.02±0.00a	0.03±0.01a	0.28	0.03±0.01a	0.02±0.00a	0.38	0.02±0.00a	0.03±0.01a	0.40
NO_3^-	0.02±0.00a	0.02±0.00a	0.86	0.02±0.00a	0.02±0.00a	0.99	0.02±0.00a	0.02±0.00a	0.40

Different letters in bold following values within lines denote a statistically significant difference between the drained vs. undrained, mounded vs. unmounded and fertilised vs. unfertilised treatment (P< 0.05) (n=12).

Table 2.4 Effects of drainage, mounding and fertilisation on standing above-ground biomass determined as dry mass (t ha^{-1} DM) based on a general linear model. Mean values± standard error of means.

Treatment			Treatment			Treatment		
Drained	Undrained	*P*–value	Mounded	Unmounded	*P*–value	Fertilised	Unfertilised	*P*–value
7.76±0.35a	**6.03±0.30**	**0.036**	6.84±0.36	6.95±0.47	0.88	**7.53±0.37a**	**6.26±0.37b**	**0.002**

Different letters in bold following values within lines denote a statistically significant difference between the drained vs. undrained, mounded vs. unmounded and fertilised vs. unfertilised treatment (P< 0.05). (n=6).

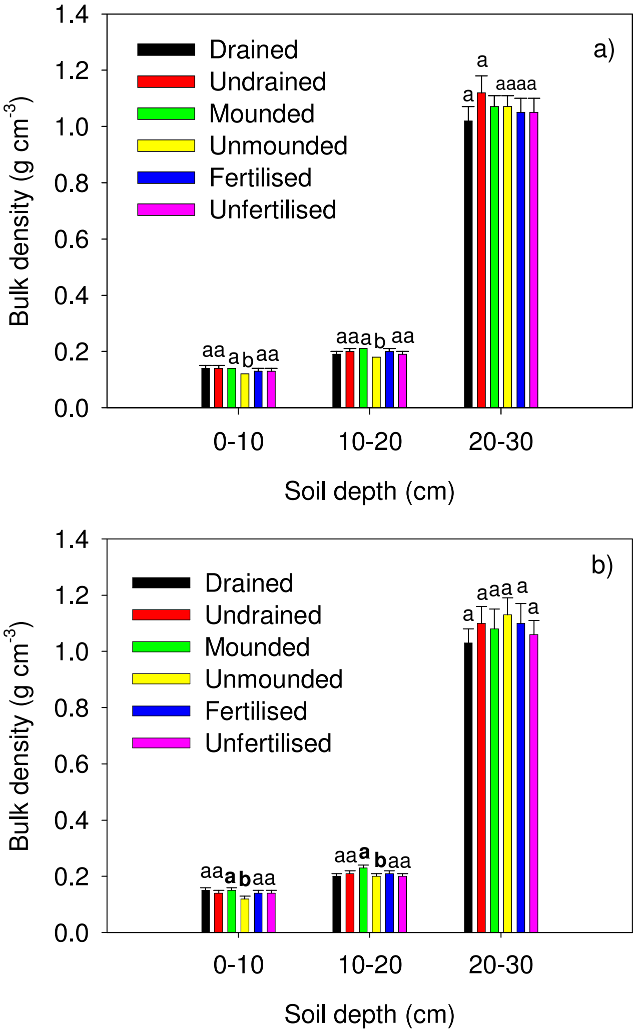

Figure. 2.4. Bulk density across treatments in a) August 2007 and b) June 2008. The vertical bars represent standard error of means (n=12). Different letters in bold indicate a significant difference in soil bulk density between the drained vs. undrained, mounded vs. unmounded and fertilised vs. unfertilised treatment ($P<0.05$).

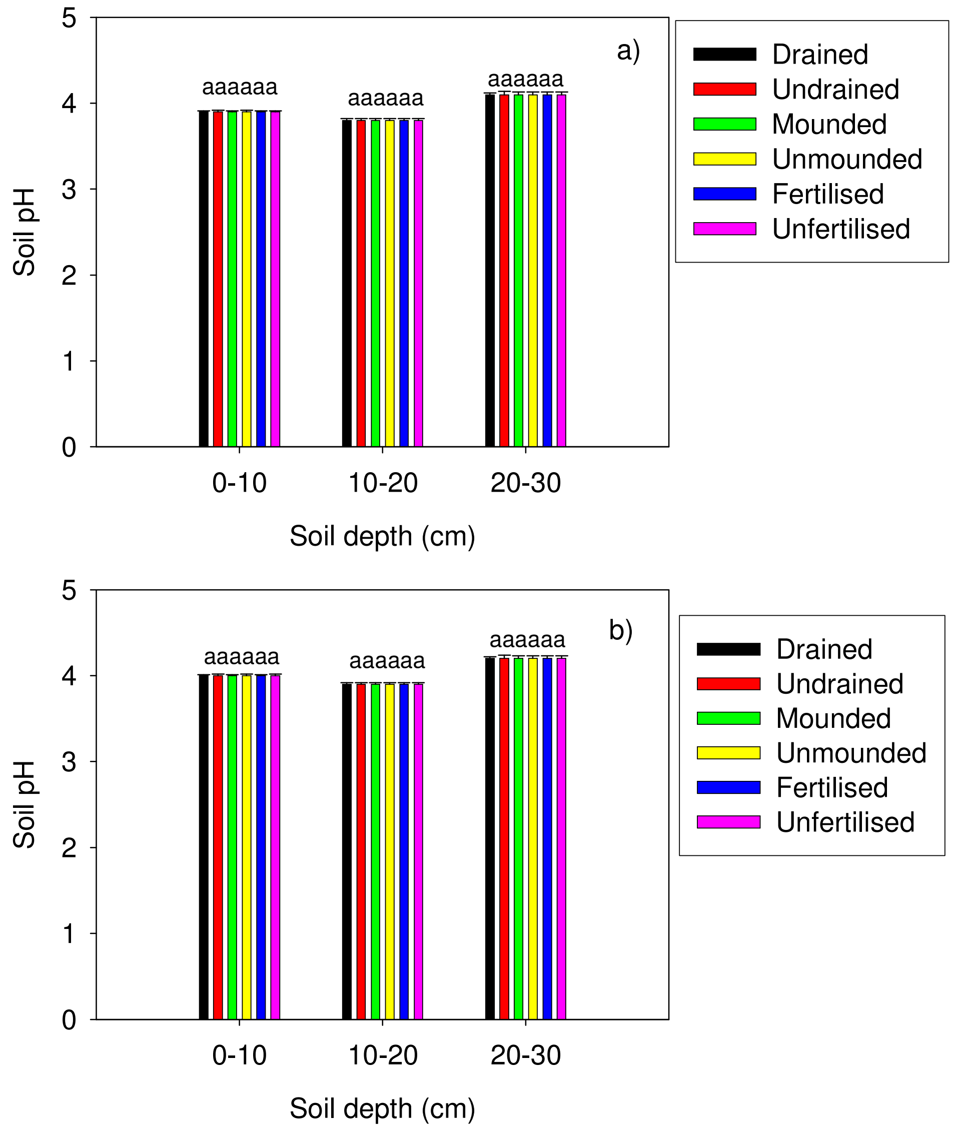

Figure 2.5. Soil pH across treatments in a) August 2007 and June 2008. The vertical bars represent standard error of means (n=12). Different letters indicate a significant difference soil bulk density between the drained vs. undrained, mounded vs. unmounded and fertilised vs. unfertilised treatment (P<0.05).

2.4 DISCUSSION

2.4.1 Carbon and nitrogen concentrations

Soils are the major reservoir of carbon in terrestrial ecosystems (Henderson 1995). Soil carbon dynamics are significantly affected by land use changes and soil management practices (Batjes 1996; Ross et al. 1999; Post and Kwon 2000; Prentice et al. 2000). Silvicultural practices such as drainage and ploughing used for afforestation and replanting clearfelled plantations have been shown to decrease soil carbon contents of peaty gley soils (Zerva and Mencuccini 2005b; Zerva et al. 2005) because the practices may create aerobic conditions favourable for decomposition (Armentano and Menges 1986; Trettin et al. 1995). The results of the present study showed that the soil carbon concentration varied with soil depth and was higher in the 0–20 cm soil layer. Drainage decreased soil organic C concentration in the 0–10 cm layer by 7 to 9%, similar to patterns reported in experiments conducted elsewhere (e.g., Ryan et al. 1992; Rab 1994). The decrease in soil organic carbon contents in this study was lower compared to values reported for these others studies, probably because of different climates, soil types, the intensity of disturbance and the time since disturbances occurred. The soil organic carbon concentration in the present study decreased with increasing depth, suggesting that drainage caused disturbance near the soil surface (e.g., Schmidt et al. 1996).

The decomposition of organic matter in peat soils is controlled by temperature, soil moisture, fertility and the quality of organic matter (Oades 1988). Drainage lowered the soil water table depth in this study and increased soil temperature and decreased soil moisture. These changes and disturbances of soil properties such as soil structure and the breakage of soil aggregates caused by drainage may have made the soil to be more aerobic and favourable for decomposition (e.g., Smith et al. 1994; Olson et al. 1996; Merino et al. 1998; Zerva et al. 2005; Tate et al. 2006; Minkkinen et al. 2008). Several studies demonstrated that soil organic carbon is lost from drained organic soils through both oxidation and enhanced soil respiration linked to improved aeration and increased temperature (Raich and Schleisnger 1992; Rey et al. 2002; Euskirchen son et al. 2003; Saiz et al. 2006). The concentration of total N in the top soils was decreased by drainage, probably because the increase in soil temperature and aeration created aerobic conditions that favoured the mineralisation of N in the organic matter and increased N uptake by plants or N losses as N_2 and N_2O emissions or as dissolved nitrate. Several studies demonstrated that site preparation practices increase leaching in upland and boreal forest soils (Nieminen 1998; Mannerkoski et al. 2005; Piirainen et al. 2007). The decrease in the concentration of total N in the present study is supported by results of studies conducted elsewhere, which reported that forest management practices such as timber harvesting and mechanical site preparation decrease total N concentration (Smith et al. 1994; Merino et al. 1998). Fertilisation increased the soil organic carbon in the 0–10 cm layer in the first year of study, similar to the effects observed in boreal and temperate forest soils (e.g., Berg and Matzner 1997; Franklin et al. 2003; Foereid et al. 2004; Olsson et al. 2005; Jandl et al. 2007). The increase in soil organic carbon following fertilisation has been attributed to the suppression of ligninolytic enzymes of soil microbes and by chemical stabilisation (Arnebrant et al. 1996; Jandl et al. 2007). The effect of fertilisation on soil organic carbon was not evident at the end of this study, probably because N uptake by plants, losses through leaching and as oxides of nitrogen depleted the pool of additional N in the second year of study.

2.4.2 Bulk density

As demonstrated in previous studies (e.g., Merino et al. 1998; Zerva 2004), mechanical site preparation increased the soil bulk density in mounded subplots. Mounding increased the bulk density from 0.12±0.00 to 0.14±0.01 g cm^{-3} in the 0–10cm layer and from 0.18±0.00 to 0.21±0.01g cm^{-3} in the 10–20 cm layer at the end of year 1 of study. At the end of the study, the bulk density in the 0 to 10 cm layer was 0.12±0.01 and 0.15±0.01 g cm^{-3} in the unmounded and mounded subplots, while in the 10–20 cm layer, the bulk density was 0.21±0.01 and 0.23±0.01 g cm^{-3} in the unmounded and mounded subplots, respectively. The increase in bulk density in the mounded subplots may suggest that equipment used for mounding caused soil compaction. Similar results have been reported in compacted forest soils following timber harvesting and site preparation (Cullen et al. 1991; Johnson et al. 1991; Merino et al. 1998; McNabb et al. 2001). The increase in soil bulk density in the mounded subplots was lower compared to values reported in these other studies, probably due to different climates, soil types and the intensity of compaction. For example, Cullen et al (1991) found that soil the bulk density increased by 21–76% in severely compacted sites at 15 cm soil depth following timber harvesting in Western Montana. Merino et al. (1998) found that different timber management practices in a hilly area in Northern Spain changed the soil bulk density by 17%.

The soil bulk density is closely related to other soil physical properties such as texture, structure, porosity, aeration and hydraulic conductivity which may be changed by compaction (Henderson 1995; McNabb et al. 2001; Block et al. 2002). These other soil properties were not evaluated, but some have been reported to change after timber harvesting and site preparation (e.g., Staaf and Olson 1991; Farrish et al 1993). Soil compaction may reduce site productivity and affect regeneration and growth of planted seedlings (e.g., Smith et al. 1994; Miller et al. 1996).

The soil bulk density in this site increased with soil depth, similar to the effects reported in other studies (e.g., Tamminen and Starr 1994; Zerva 2004). For example, Tamminen and Starr (1994) investigated the relationship between the soil bulk density and organic matter content, textural properties and depth and found that the soil bulk density increased with soil depth and remained uniform at soil depth greater than 20 cm. Tamminen and Starr (1994), Henderson (1995) and Prevost (2004) reported that the soil bulk density can be altered directly by compaction or indirectly through loss of soil organic matter.

2.4.3 Soil microbial biomass C

Soil microorganisms regulate soil nutrients through mineralisation of soil organic matter and solubilisation of soil minerals (Mazzarino et al. 1993; Franzluebbers et al. 1994; Fritze et al. 1994; Blazier et al. 2005) especially in infertile natural and agricultural systems (Yao et al. 2000). The soil microbial biomass has been estimated to represent 2–5% carbon (Jenkinson and Ladd 1981; Smith and Paul 1990) and 3–5% of soil nitrogen (Paul et al. 1999). Soils that are cultivated continuously may contain fewer microorganisms than their uncultivated counterparts (McGill et al. 1986). The soil microbial population and activity is influenced by numerous factors such as climate, soil carbon, nutrient availability and pH (Hossain et al. 1995).

Infertile saturated soils with a cold substrate (e.g., Lieffers and Rothwell 1986; Macdonald and Lieffers 1990) may reduce the growth and activity of soil microorganisms. However, drainage carried out to lower the soil water table for forestry and agriculture has been shown to increase substrate temperature and oxygen availability as well as nutrients (e.g., Lieffers and Rothwell 1987; Lieffers 1988) and may favour soil microbes and their activity. Soil microbes are expected to respond to changes in the soil environment caused by drainage (Schnürer et al. 1985).

The activity of soil microorganisms measured as microbial biomass carbon was not affected by drainage, mounding or fertilisation. Drainage increased biomass carbon slightly in February and August 2007. It seems that the improvement in the soil environment caused by drainage slightly stimulated soil microorganisms. Low soil pH and poor soil fertility in this site may have affected the soil microbial population and activity. The microbial biomass carbon was low in November 2006 and February 2007, suggesting that the ecosystem activity was low during the winter months and nutrients required by soil microorganisms and plants was probably met by the background nutrient levels of soils (e.g., Insam et al. 1989). In the late summer (August 2007) the soil microbial biomass in all treatments was higher when temperatures were higher. It seems, therefore that treatment effects may have been masked by seasonal effects. This is consistent with finding of previous studies which found that fluctuations in soil temperature affected soil microbial biomass (Lynch and Panting 1982; Sarathchandra et al. 1988; 1989).

The microbial biomass carbon in the fertilised subplots was slightly lower than in the unfertilised subplots. Studies by Yates et al. (1997) and Bardgett and Cook (1998) also demonstrated that the soil microbial biomass carbon was lower in fertilised than in unfertilised grasslands. The inhibitory effect of inorganic N on soil microbes has been attributed to the direct effect of fertilisers on soil microorganisms (Lovell et al. 1995), resulting from lowering of soil pH, inhibition of fungal ligninolytic enzyme production and a decreased production of enzymes that degrade nitrogen-containing organic matter (Smolander et al. 1994; Ettema et al. 1999; Vance and Chapin 2001). Overall, this result is comparable to previous studies which failed to detect significant effects of fertilisation on soil microbial biomass carbon (e.g., Castro et al. 1994; Vose et al. 1995; Sarathchandra et al. 2001). In contrast, increased microbial biomass carbon has been reported in fertilised forest (Hobbie 2000; Vestgarden 2001) and agricultural soils (Lych and Panting 1982; Hesebe et al. 1985). Other studies found that fertilisation decreased labile C in coniferous forest soils (Thirukkumaran and Parkinson 2000; Homann et al. 2001) in grassland as well as in pasture soils (Christie and Beattie 1989; Bristow and Jarvis 1991). Inconsistencies in these studies has been attributed to different rates and formulations of fertilisers used, site productivity and the duration of each study (Thirukkumaran and Parkinson 2000).

Plant species composition was not assessed in this study; it is also possible that changes in plant species composition caused by fertilisation (e.g., Bardgett et al. 1999; Wardle et al. 1999) and drainage may have indirectly affected soil microbial population and activity. Plants roots are a significant source of carbon (Wheatley et al. 1990) and supply soil microorganisms with highly labile substances such as carbohydrates, amino acids and fatty acids, which may stimulate microbial growth and division (Qualls 2000; Blazier et al. 2005). Norton and Firestone (1991) found that 57% of bacterial cells in Ponderosa pine seedlings were close to the roots, while only 41% were active in the bulk soil. Bardgett et al. (1999) observed that temporal changes in the productivity of different grass species affected the soil microbial community. Hence, it was surprising to find no differences in microbial biomass carbon across treatments in this study given that different levels of aboveground plant biomass productivity was found in fertilised and drained plots. Ecosystems with fewer species like the present study site, may not sustain a large community of microorganisms because of less diversity in roots exudates.

Soils in this site are acidic, with pH ranging from 3.8 to 4.2 and the low pH may have hindered the composition, multiplication and growth of soil microorganisms (e.g., Shah et al. 1990; Nodar et al. 1992). Priha et al. (2001) and Smolander and Kittunen (2002) found that soil microbial biomass carbon was higher in hardwood forest soils than in acidic coniferous soils. They attributed the difference to either root distribution and activity or the chemical composition of the organic matter. Meharg and Killham (1990) observed that the amount of carbon lost from *Lolium perenne* increased from 12.3 to 30.6% when soil pH increased from 4.3 to 6.0. They suggested that changes in the microbial biomass and plant nitrogen limitation could have resulted in increased exudation with increasing pH. In general, acidic soils favour the growth of the fungal population relative to bacterial populations (Nodar et al. 1992).

2.4.4 Inorganic N and aboveground plant biomass

Drainage of peat soils increases N mineralisation and subsequent nutrient availability to plants (Grootjans et al 1985; Cabrera 1993; Updegraff et al.1995; Bridgham et al. 1998). In this study, drainage did not affect inorganic N (NH_4^+ and NO_3^-), probably because increased N mineralisation was compensated by increased losses. Ammonium (NH_4^+) availability was increased by mounding, probably due to the increase in net mineralisation of the organic matter on the top of mounds. This was not surprising, because changes in the microtopography related to mounds and hollows after mounding modify a number of important environmental variables (e.g., Liechty et al. 1997) such as soil temperature and moisture (e.g., Nohrstedt 2000) as well as the thickness and distribution of organic and mineral soil layers (e.g., Beatty and Stone 1986; Schaetzl et al. 1990) which may lead to N mineralisation. Mounding buried the soil organic layers beneath the mineral soil of mounds and increased soil temperature and aeration. These changes may have favoured microbial activity on the top of mounds and accelerated N mineralisation. Nitrate was not affected by mounding in this study.

The fertilisation increased NH_4^+ in the first year, but at the end of this study no significant difference was detected between fertilised and unfertilised subplots. The reduction in NH_4^+in the fertilised subplots at the end of study was probably caused by increased N uptake by plants or losses as oxides of nitrogen in the first year of study. Nitrate was not affected by fertilisation. It is possible that $NO_3^-$was leached to deeper layers or lost through drainage water. Nitrate leaching from soil has been reported in experiments conducted in sites with different climate, soils types and plants (Baker and Johnson 1981; Bergstrom and Brink 1986).

Standing above-ground plant biomass production measured a year after site preparation was increased by drainage. The increase probably resulted from increased soil temperature and improvement in aeration of the root zone which favoured root growth and increased nutrient availability. The lack of oxygen in water-saturated soils has been shown to have negative effect on plant growth by limiting root and shoot growth (e.g., Sena Gomes and Kozlowski 1980b; Lieffers and Rothwell 1986;

Grossnickle 1987; Huang et al. 1994; McDonald et al. 2001; Malik et al. 2001, 2002). In addition, increased substrate temperature and oxygen availability in the drained plots may have favoured microbial activity, and thus enhanced the mineralisation of organic matter, which increased the availability of nutrients, since decomposition is enhanced under aerobic conditions (e.g., Clymo 1984).

Fertilisation also increased above-ground plant biomass production due to availability of inorganic N from the applied fertiliser. The uptake of potassium and phosphorus by plants from the applied fertiliser may also have enhanced their growth. Other studies conducted in *Calluna* dominated vegetation on heathland also found that N fertilisation increased plant growth (e.g., Aerts et al. 1991; Caporn et al. 1995; Uren et al. 1997; Carroll et al. 1999). The increase in aboveground plant biomass in the fertilised subplots may suggest that our study site is poor in nutrient.

2.5 CONCLUSION

Drainage decreased the soil organic carbon and nitrogen concentration in the top 10 cm layer of peaty gleys soils. Mounding increased the soil bulk density in the 0 to 20 cm layer of peaty gley soils. Site preparation practices did not change soil pH and microbial biomass carbon. The availability of ammonium was increased by mounding and fertilisation in the first year of study, it increased plant growth during the first year of study, which was also found on drained plots.

CHAPTER 3

EFFECT OF SITE PREPARATION FOR AFFORESTATION ON CO_2 EMISSIONS FROM A PEATY GLEY SOIL

3.1 INTRODUCTION

The current estimate of soil C pools averages 1500 Pg C, which is about 3 times the total C stored in vegetation (Eswaran et al. 1993) and almost twice the amount in the atmosphere (Schimel 1995). The soil surface CO_2 efflux or soil respiration arise from respiration by plant roots and living organisms, and from the mineralisation of organic matter, dead microorganisms, soil animals and dead plants (Gough et al. 2005; Kuzyakov 2006; Subke et al. 2006; Ding et al. 2007; Oleszczuk et al. 2008). Soil respiration constitutes an important source of CO_2 in terrestrial ecosystems and a key component of the global C cycle (Raich and Schlesinger 1992; Rustad et al. 2000). The atmospheric concentration of CO_2 has increased from 280 ppm (parts per million) since the beginning of the industrial revolution to the current level of 380 ppm (Falkowski et al. 2000). This increase has been attributed largely to natural disturbances (e.g., fire, windthrow), anthropogenic land use changes (e.g., deforestation, agricultural cultivation, afforestation) and burning of fossil fuels (Rastogi et al. 2002; Heath et al. 2005; Watson and Noble, 2005). Increased CO_2 concentrations contribute to increasing atmospheric temperatures by trapping longer wavelength of radiation in the atmosphere (Nowak and Crane 2002; Rastogi et al. 2002).

Soil respiration rates are influenced by interactions among several factors. Soil temperature and soil moisture are the most important factors controlling soil respiration rates (Davidson et al. 1998; Bowden et al. 1998; 2004; Maier and Kress 2000; Fisk and Fahey 2001; Rustad et al. 2001, Kiese and Butterbach-Bahl 2002; Liu et al. 2002; Pangle and Seiler 2002; Hashimoto et al. 2004). Soil properties and availability of C and N (Grant and Rochette 1994; Randerson et al. 1996; Boone et al. 1998; Pregitzer et al. 1998), current photosynthetic rates (Högberg et al. 2001), land use and management practices (Paustian et al. 1997; Lal 2002) have also been shown to affect on soil respiration. Soils in areas where temperature and moisture are not limiting (e.g., tropical forests) respire more than those located in cold and dry areas such as the tundra, northern bogs and deserts (Rhoades et al. 2000; Raich et al. 2002). Soil respiration releases 10–15 times more CO_2 to the atmosphere than fossil fuel burning (Andres et al. 1996; Verburg et al. 1998).

Peatlands have been a net sink of C for many centuries (Christensen et al. 1999; Oechel et al. 2000; Roehm and Roulet 2003). Saturated soils and low temperatures in these C rich ecosystems limit microbial decomposition and lead to accumulation of soil organic matter as peat (Bubier et al. 1998; Ström and Christensen 2007 Turunen and Turunen 2002). In recent years, evidence has been accumulating that globally the release of C in peatlands now exceeds the C uptake (e.g., Turetsky et al. 2002; Roulet et al. 2007) due to changes in land use and drainage.

Drainage and cultivation of peat soils for agriculture and forestry uses has been one of the most important forms of human disturbances in boreal and temperate peatlands in recent decades (Armentano and Menges 1986; Gorham 1991; Laine and Minkkinen 1996). Drainage and cultivation can change peatland soils from net sinks to net sources of atmospheric CO_2. For example, several studies have reported that changes in organic matter production and decomposition processes after drainage and cultivation increase CO_2 emissions from peatland soils (Maljanen et al. 2001b; Minkkinen et al. 2002; Lohila et al. 2003). Lowered soil water table depth increases soil temperature and the volume of aerated organic matter, which increases decomposition rates and soil respiration in peatland soils (Trettin et al. 1996; Minkkinen et al. 2008). The growth of trees planted on infertile soils is often stimulated by fertilisation (Elser et al. 2007; LeBauer and Treseder 2008). The impact of fertilisation, particularly N, on soil C is not well understood (Grandy et al. 2008; Minkkinen et al. 2008; Reay et al. 2008). Forest fertilisation may increase (Minkkinen et al. 2008) or decrease (Jandl et al. 2007) organic matter decomposition and soil respiration.

Most studies on the effect of drainage on C fluxes have been conducted in boreal and continental peatland sites (e.g., Lafleur et al. 2005; Mäkiranta et al. 2007; Minkkinen et al. 2007). The effects of site preparation (drainage and mounding) carried out prior to afforestation on soil respiration in peaty gley soils under UK conditions have not received sufficient attention. We hypothesised that (a) drainage, mounding and fertilisation would increase soil CO_2 fluxes (b) drainage and mounding would affect soil temperature, soil moisture content and water table depth and (c) soil CO_2 fluxes are controlled by soil temperature, moisture content and water table depth. The objectives of this study were to (i) evaluate the effect of drainage; mounding and fertilisation on soil CO_2 efflux over a period of two years, (ii) assess the effect of the three practices on soil temperature, moisture content and water table depth and (iii) evaluate relationships between soil CO_2 fluxes and soil temperature, soil moisture and soil water table.

3.2 MATERIALS AND METHODS

3.2.1 The study site description

The study site has been described in Chapter 1. Briefly, the study was established on unimproved grassland between two second rotation Sitka spruce stands at Harwood Forest located in NE England (55° 10’ N, 2° 3’W).

3.2.2 Experimental design and preparation

The experiment is described in Chapter 2. Briefly, the experiment has a full factorial split–plot design with six plots measuring 30 × 8 m established in May 2006. Three plots were selected at random and mechanically drained by cutting open ditches placed 1.5 m from plot edges and excavated to depth of 65–70 cm. Mounds were made by excavating the soil and by turning it upside down adjacent to the dug pit (depth 30–40 cm, width 40 cm), thus burying the litter and organic layers of the original soil beneath the mineral layer of mounds.

3.2.3 The measurement of CO_2 efflux from soil

Gas samples for soil CO_2 fluxes were collected from 5 July 2006 to 7 May 2008 with manual closed static chambers (e.g., Smith et al. 1995). Gas samples were collected weekly in July, bi–weekly between August and September 2006 and finally at approximately monthly intervals from October 2006 to May 2008. The soil CO_2 efflux measured here includes heterotrophic and autotrophic respiration, although the autotrophic respiration by above ground plant parts was excluded by clipping them at the base, whenever new shoots appeared inside collars. Samples were collected between 0900 and 1430 hrs to minimise changes in soil respiration associated with diurnal cycles (Davidson et al. 1998). A total of sixty collars (inside diameter 40 cm, height 10 cm) were inserted into the soil to a depth of 5 cm, three weeks prior to the start of measurements and left permanently in the field. The collars were positioned randomly in groups of either two or three for each subplot depending on whether the subplot had been mounded. If so, the sampling was stratified in such a way that one collar was placed on top of a mound, one inside a hollow and one on undisturbed ground. For unmounded subplots instead, only two collars were used, randomly placed on undisturbed ground. Mounds were estimated to cover about 8% of the total surface area of mounded subplots, and the hollows from which the peat had been excavated, estimated to occupy a similar proportion.

A total of twenty chambers (inside diameter 40, height 20 cm) were used to collect air samples by placing them on top of the already positioned collars, and then rotating them until all sixty had been measured. A seal between the chamber and the collar was obtained by circular elastic rubber bands on the chamber and collar outer surface. Chambers were sealed with aluminium lids with foam rubbers on the underside and a sampling port fitted with a three-way stopcock. Air samples were collected from the headspace of chambers with 60 ml polypropylene syringes and transferred into gas-tight bags (Cali-5-bond, Calibrated Instruments Inc. USA). The chambers were sealed for 30–40 minutes and a linearity check showed that linear interpolation of two points taken at the start and at the end of the closure gives a good approximation of the true CO_2 efflux (Zerva and Mencuccini 2005, unpubl. data). Linear accumulation or depletion of CO_2 was also checked every two months during the first year of the present study. For the vast majority of the checks conducted, linearity was ensured by linear regression coefficients higher than R^2=0.99. Ambient air samples taken randomly at the height of chambers gave the initial concentration of CO_2. Air samples were transferred to the laboratory and analysed by gas chromatography, using a

Perkin Elmer Model 310 GC (Perkin Elmer Ltd, Beaconsfield, Bucks, UK) with a thermal conductivity detector (TCD). External standards of 390, 1093, 5262 and 10100 ppm were used for calibration. The concentration of CO_2 in gas samples collected from chambers was calculated with the following equation (Zerva 2004):

$$F_s = d \times \frac{V}{A} \times \frac{(C_t - C_0)}{t}$$

Where: F is the CO_2 efflux (g m^{-2} d^{-1}), d (g m^{-3}) is the gas density calculated on the assumption that 1 mole of gas (1 mole of CO_2 is 44 g) occupies 22.4×10^{-3} m^3 of volume at 273 K, V is the volume of chamber (m^3), C_t is the concentration of gas (µmol mol^{-1}) inside chamber after closure time t (d), C_0 is the initial concentration of the gas (µmol mol^{-1}), A is the area of the chamber (m^2) and t is the time of chamber closure.

3.2.4 Measurement of environmental variables

Soil temperatures at 1, 5 and 10 cm depth (from now on T_1, T_5 and T_{10}) were measured simultaneously with soil CO_2 efflux with a digital temperature probe (Fisher Scientific) close to each chamber on each date. In addition, to digital temperature measurements, a system of temperature probes was installed to continuously monitor temperatures at 5 and 10 cm depth in one drained and one undrained plot. Soil temperature was logged automatically every hour by a datalogger (Campbell Scientific Ltd, Loughborough, UK).

Soil moisture was measured as volumetric content (m^3 m^{-3}) close to each chamber at 5 cm depth using a Theta probe (KT1-Basic, Delta-T Devices Ltd, Cambridge, UK). An effort was made to insert the probe in a similar position in order to minimise disturbance of the soil from its frequent insertion. In addition to Theta probe measurements, a system of time domain reflectometry (TDR) probes was installed in one drained and one undrained plot to monitor soil moisture content continuously in the top 15 cm of the soil profile. Soil moisture was recorded by data logger (Campbell Scientific) every 24 hours. Calibration was carried out against soil moisture contents measured at the study site and determined gravimetrically after oven drying over night at 105°C and corrected to volumetric values using the measured bulk density.

The water table depth (cm from soil surface) was measured from dip wells. One dip well was established in each subplot by removing a soil core (diameter 5 cm) to 80 cm depth. PVC pipes (length 1 m, diameter 5 cm) with several small holes (diameter 0.5 cm) drilled laterally were inserted inside to act as liners. Dip wells were always sealed to prevent water entering them from above.

3.2.5 Statistical analysis

All data were checked for normality and log-transformed when required. Analyses were carried out both on averaged monthly fluxes as well as on seasonal and yearly totals. The general linear model (GLM) was used for analysis of variance. The general linear model tested for effects of three main factors (drainage, mounding and fertilisation) entered as fixed factors and plot entered as random factor nested within drainage. In case of monthly measurements, month was also entered as a repeated measures factor. The initial GLM included all possible second and third-order interactions. If interactions were not found to be significant, they were excluded and the model was run again without them to confirm the significance of the main factors. In case of significant interactions, the dataset was split and separate analyses were run for each combination. All analyses were run in Minitab 15 using the GLM procedure and the significance level was set at 0.05. Tukey's pairwise comparison test was applied to determine significant differences between treated and untreated plots/subplots. For all analyses, values from individual chambers were averaged within each subplot. For the mounded subplots, weighted averaging was done by weighing each flux by the respective area covered by mound hills, mound hollows and undisturbed ground. Correlation analyses were performed in Minitab to determine relationships between environmental variables. If the variables were found to be correlated, a forward stepwise multiple linear regression analysis was performed in Minitab 15 to determine their relationship with soil CO_2 fluxes.

3.3 RESULTS

3.3.1 Effect of site preparation on environmental variables

Soil temperature (T_1, T_5 and T_{10}) are shown in Fig. 3.1a–f and 3.2a–c. Soil temperature varied from season to season. High soil temperatures were recorded in the summer to early autumn, while low temperatures occurred in the winter months. The month to month variability in soil temperature at all depths was highly significant (P=0.0001, Table 3.1). There were no significant differences in soil temperature among plots (P=0.06). Soil temperature at all measured depths was significantly increased by drainage (P=0.0001 for T_1, P=0.05 for T_5 and P=0.01 for T_{10}, Table 3.1). Mounding increased soil temperature at 1 and 5 cm depth (P=0.0001 for T_1 and P=0.01 for T_5) but not at 10 cm depth (P=0.6) (Table 3.1). Soil temperatures were more variable at 1 cm depth than at 5 and 10 cm depth. Soil temperature at all depths was not affected by fertilisation (Table 3.1). Figure 3.3 shows soil temperature recorded continuously by datalogger at 5 and 10 cm depth in a drained and an undrained plot. A very similar seasonal course was reported here.

There was no significant difference in water table depth among plots (P=0.9, Table 3.1). The water table depth varied from month to month (P=0.0001, Table 3.1) and was significantly decreased by drainage (P=0.0001, Fig. 3.4a–c). Mounding (P=0.7) or fertilisation (P=0.9) did not affect the water table depth (Table 3.1). There was no significant difference in soil moisture content among plots (P=0.7, Table 3.1). The month to month variability in moisture content was highly significant (P=0.0001, Fig. 3.4d–f). The soil moisture content was significantly decreased by mounding (P=0.0001) and drainage (P=0.01). Fertilisation did not affect the soil moisture content (P=0.3, Table 3.1). Figure 3.5 shows the soil moisture content measured continuous at the top 15 cm of the soil profile by TDR probes and recorded by datalogger in a drained and an undrained plot. A very similar seasonal pattern was observed here.

Table 3.1 Summary result of the general linear model of environmental variables.

Parameter	T_1°C	T_5°C	T_{10} °C	$W(m^3\ m^{-3})$	WT(cm)
Drainage	**<0.0001**	**0.05**	**<0.01**	**<0.009**	**<0.0001**
Plot	0.06	0.3	0.5	0.07	0.9
Mounding	**<0.0001**	**0.01**	0.6	**<0.0001**	0.7
Fertilisation	0.7	0.5	0.4	0.3	0.9
Month	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**

P values in bold are statistically significant at $P<0.05$.

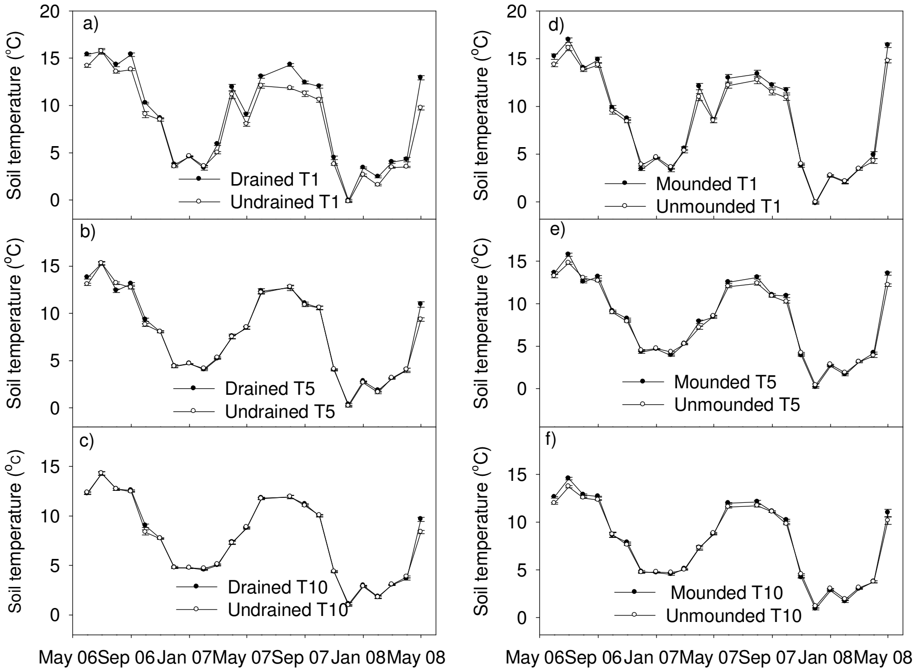

Figure 3.1. Averaged monthly soil temperature for all depths (T_1, T_5 and T_{10}) for: a–c drained and undrained plots and d–f mounded and unmounded subplots (n=34). The vertical bars are standard errors of mean.

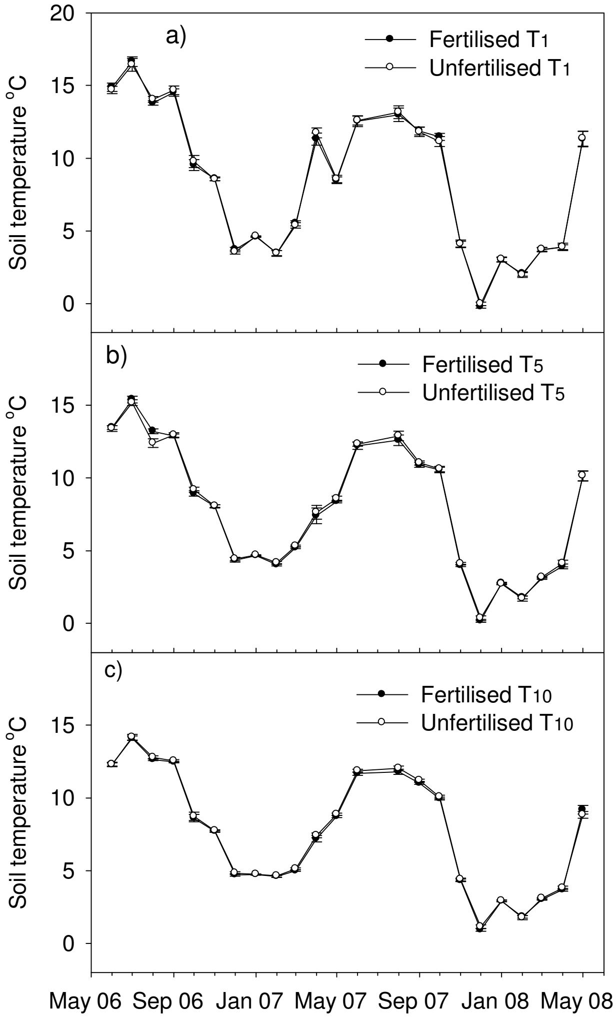

Figure 3.2 Averaged monthly soil temperature for all depths (T_1, T_5 and T_{10}) for the fertilised and unfertilised subplots (n=34). The vertical bars represent the standard error of mean.

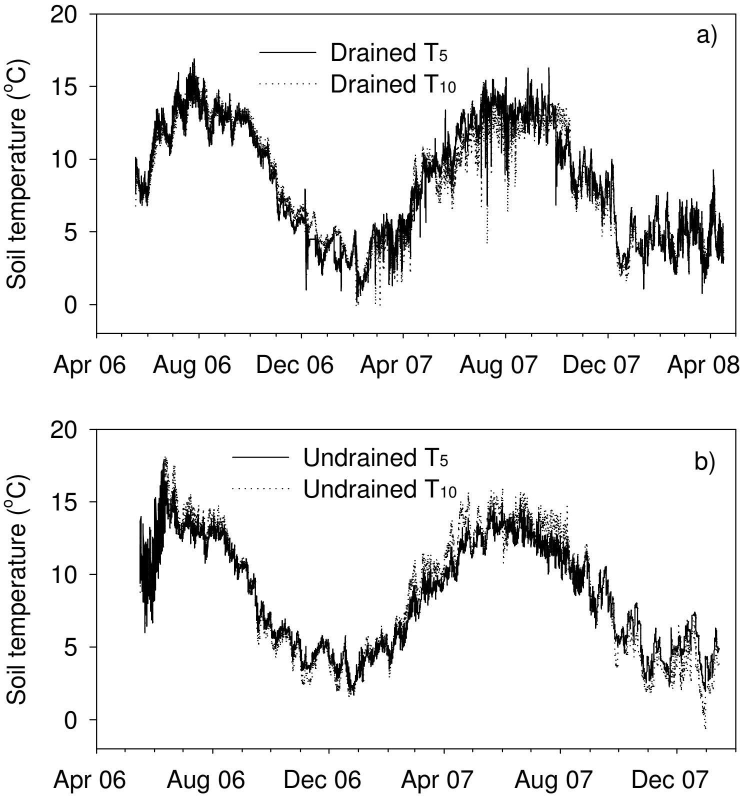

Figure 3.3. Soil temperature at 5 and 10 cm depth recorded continuous by datalogger at a) drained plot from 17 May 2006 to 16 April 2008 and b) undrained plot from 17 May 2006 to 14 January 2008.

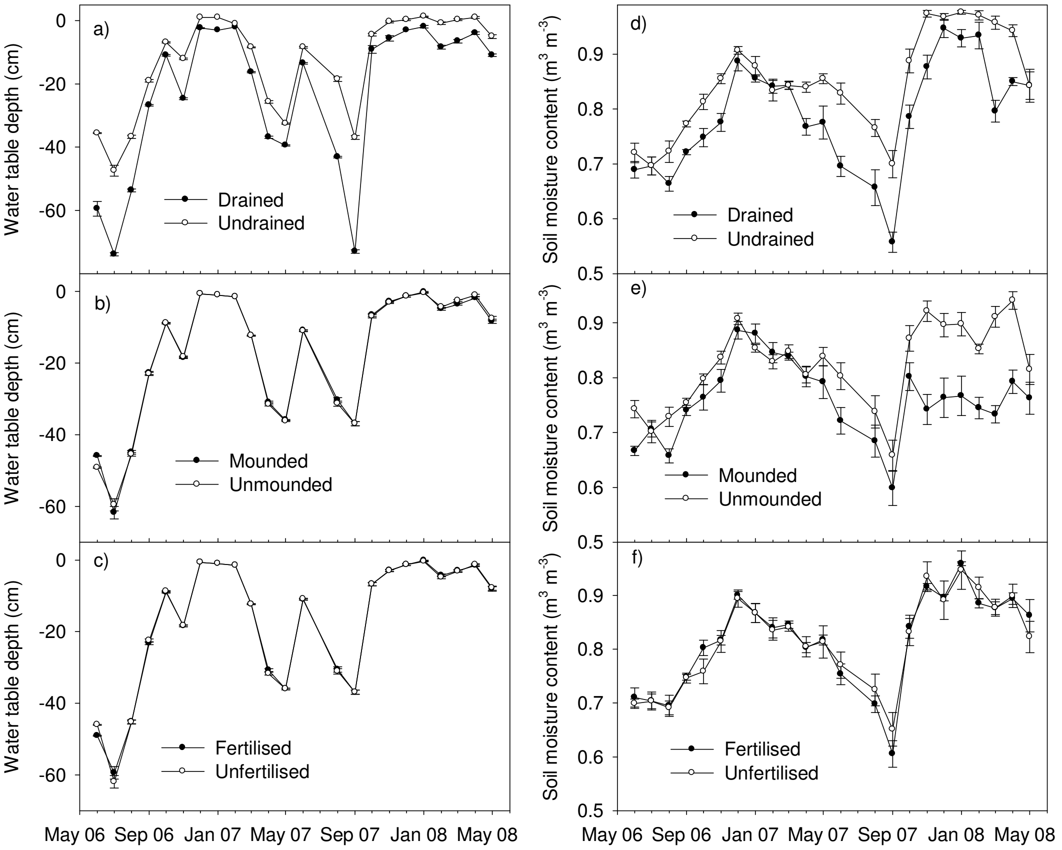

Figure 3.4. Averaged monthly water table depth (a–c) and soil moisture content (d–f) for all treatments (n=34). The vertical bars indicate the standard error of mean.

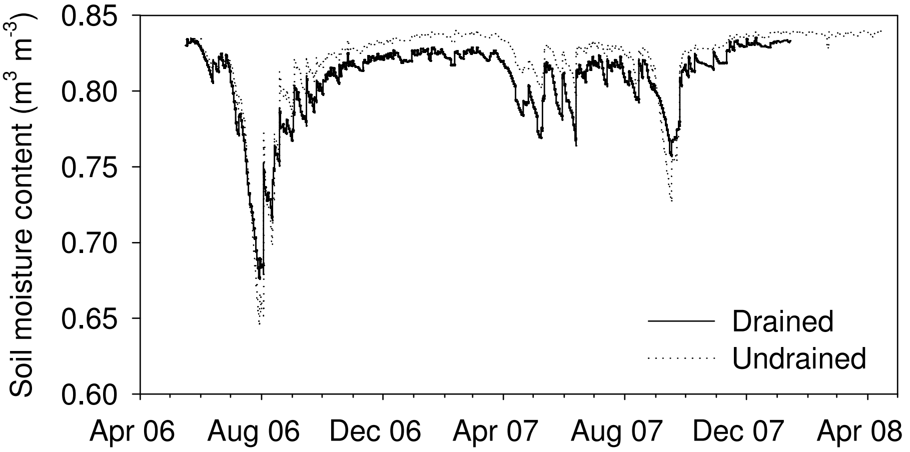

Figure 3.5. Soil moisture of the top 15 cm of the soil profile recorded continuous by TDR probes and datalogger in a drained (17 May 2007 to April 2008) and undrained plot (17 May 2006 to 14 January 2008).

3.3.2 Effects of site preparation of soil CO_2 efflux

There were significant differences in CO_2 fluxes among plots (P=0.0001, Table 3.2). There was no significant interaction between the three practices, except when the factors month and plot were also included (Table 3.2).

Table 3.2.Summary result of the general linear model of soil CO_2 fluxes for all the practices.

Sources of variation	F	P
Drainage	71.85	**<0.001**
Plot	2.56	**0.038**
Mounding	3.21	0.07
Fertilisation	17.10	**<0.0001**
Month	249.09	**<0.0001**
Drainage×Fertilisation	0.01	0.96
Drainage×Mounding	3.83	0.12
Mounding×fertilisation	0.01	0.98
Drainage×Month	1.71	**0.05**
Mounding×Month	1.40	0.15
Fertilisation×Month	2.08	**0.01**
Mounding×Plot	0.91	0.54
Fertilisation×Plot	0.37	0.82
Month×Plot	3.61	**0.0001**
Drainage×Fertilisation×Mounding	0.92	0.29
Drainage×Mounding×Month	2.25	**0.04**
Drainage×Fertilisation×Month	0.94	0.54
Mounding×Fertilisation×Month	1.78	**0.04**
Mounding×Fertilisation×Plot	3.85	**0.007**
Mounding×Month×Plot	1.02	0.47
Fertilisation×Month×Plot	0.88	0.712
Drainage×Mounding×Fertilisation×Month	1.23	0.252

P Values in bold are statistically significant (P<0.05).

3.3.2.1 Effects of drainage on soil CO_2 efflux

The soil CO_2 efflux was significantly increased by drainage (P=0.001, Table 3.2). Drainage increased soil respiration in 2006–07 (P=0.001) and 2007–08 (P=0.001). Averaged daily soil CO_2 emissions are shown in Fig. 3.6a. Averaged CO_2 fluxes were higher in 2006–07 than in 2007–08 (P=0.01). The average soil CO_2 efflux over the two years of study was 5.02±0.19 and 3.96±0.19 g m^{-2} d^{-1} in the drained and undrained plots respectively. Averaged monthly soil CO_2 fluxes from the drained plots and their undrained counterparts are summarised in Fig. 3.7a. Fluxes varied from month to month (P=0.0001, Table 3.2). The mean monthly soil CO_2 fluxes varied from a sink (–0.34±0.08 g m^{-2} d^{-1}) to a net source for atmospheric CO_2 (8.91±0.17g m^{-2} d^{-1}) in the drained plots and a sink (–0.51±0.11 g m^{-2} d^{-1}) to a net source for atmospheric CO_2 (7.37±0.29 g m^{-2} d^{-1}) in their undrained counterparts. Soil respiration in the drained and undrained plots showed a strong seasonal trend, with higher fluxes during the summer and lower fluxes during the winter (Fig. 3.7a).

Averaged seasonal soil CO_2 fluxes are shown in Fig. 3.8a. Drainage significantly increased the fluxes in the summer of 2006 (P=0.01) and 2007 *(*P=0.01). Soil CO_2 emissions were significantly increased by drainage in the autumn of 2006 (P=0.02) and 2007 (P=0.001). The soil in the drained and undrained plots respired more in the autumn of 2006 than in 2007 (P=0.0001). Drainage increased soil respiration in the winter of 2006–07 (P=0.05) but not in 2007–08 (P=0.08). The winter seasons differed significantly in soil respiration (P=0.03). Soils in the drained and undrained plots emitted CO_2 in the winter of 2006–2007 but were a net CO_2 sink in the winter of 2007–08. Drainage increased soil respiration in the spring of 2007 (P=0.05) and 2008 (P=0.01). The drained and undrained plots respired more in the spring of 2007 than in 2008 (P=0.02). Annual soil CO_2 fluxes are shown in Table 3.3. Annual fluxes were greater in 2006–07 than 2007–08. Over the two years of study, the drained and undrained plots released 18.41±0.71 and 14.53±0.62 t CO_2 ha^{-1} yr^{-1} to the atmosphere.

Table 3.3 Annual soil fluxes (t CO_2 ha^{-1} yr^{-1}) for all treatments. The number in the brackets indicates the annual C efflux (t C ha^{-1} yr^{-1}).

			Sampling year			
Treatment	2006–2007a	n	2007–2008	n	2006–2008	N
Drained	**22.16±0.75a** (6.04±0.21)	15	**15.00±1.09a** (4.09±0.30)	11	**18.41±0.71a** (5.02±0.19)	26
Undrained	**18.08±0.0.70b** (4.93±0.19)		**11.31±0.92b** (3.08±0.25)	11	**14.53±0.62b** (3.96±0.17)	26
Mounded	19.57±0.69a (5.34±0.19)	15	13.16±1.02a (3.59±0.49)	11	16.22±0.66a (4.42±0.18)	26
Unmounded	20.67±0.81a (5.64±0.22)	15	13.14±1.02a (3.58±0.28)	11	16.73±0.70a (4.56.±0.19)	26
Fertilised	**20.96±0.72a** (5.72±0.20)	15	13.51±1.04a (3.69±0.28)	11	**17.06±0.68a** (4.65±0.19)	26
Unfertilised	**19.28±0.77b** (5.26±0.21)	15	12.79±1.00a (3.49±0.27	11	**15.88±0.67b** (4.33±0.18)	26

The '±'indicate the standard error of the mean. Different letters in bold indicate significant difference in emissions ($P<0.05$) between the drained vs. undrained, mounded vs. mounded and fertilised vs. unfertilised treatment in each sampling year.

Table 3.4: Summary of the general linear model of soil CO_2 emissions in the mounded plots. The general linear model tested the significance of drainage, fertilisation and position (mounds, hollow, undisturbed ground) across sampling dates (plot nested within drainage and date entered as repeated measure).

Parameter	F	P
Drainage	1.41	0.301
Plot	11.85	**<0.0001**
Fertilisation	21.14	**<0.0001**
Position	52.04	**<0.0001**
Drainage×Fertilisation	0.01	0.99
Drainage×Position	1.06	0.35
Fertilisation×Position	6.93	**<0.001**
Drainage×Fertilisation×Position	1.43	0.24
Sampling date	37.24	**<0.0001**
Drainage×Sampling date	2.27	**<0.0001**
Position×Sampling date	3.78	**<0.0001**
Fertilisation×Sampling date	0.81	0.73
Drainage×Position×Sampling date	1.07	0.36
Drainage×Fertilisation×Sampling date	0.47	0.99
Fertilisation×Position× Sampling date	0.63	0.98
Drainage×Fertilisation×Position×Sampling date	0.86	0.74

P values in bold are statistically significant ($P< 0.05$).

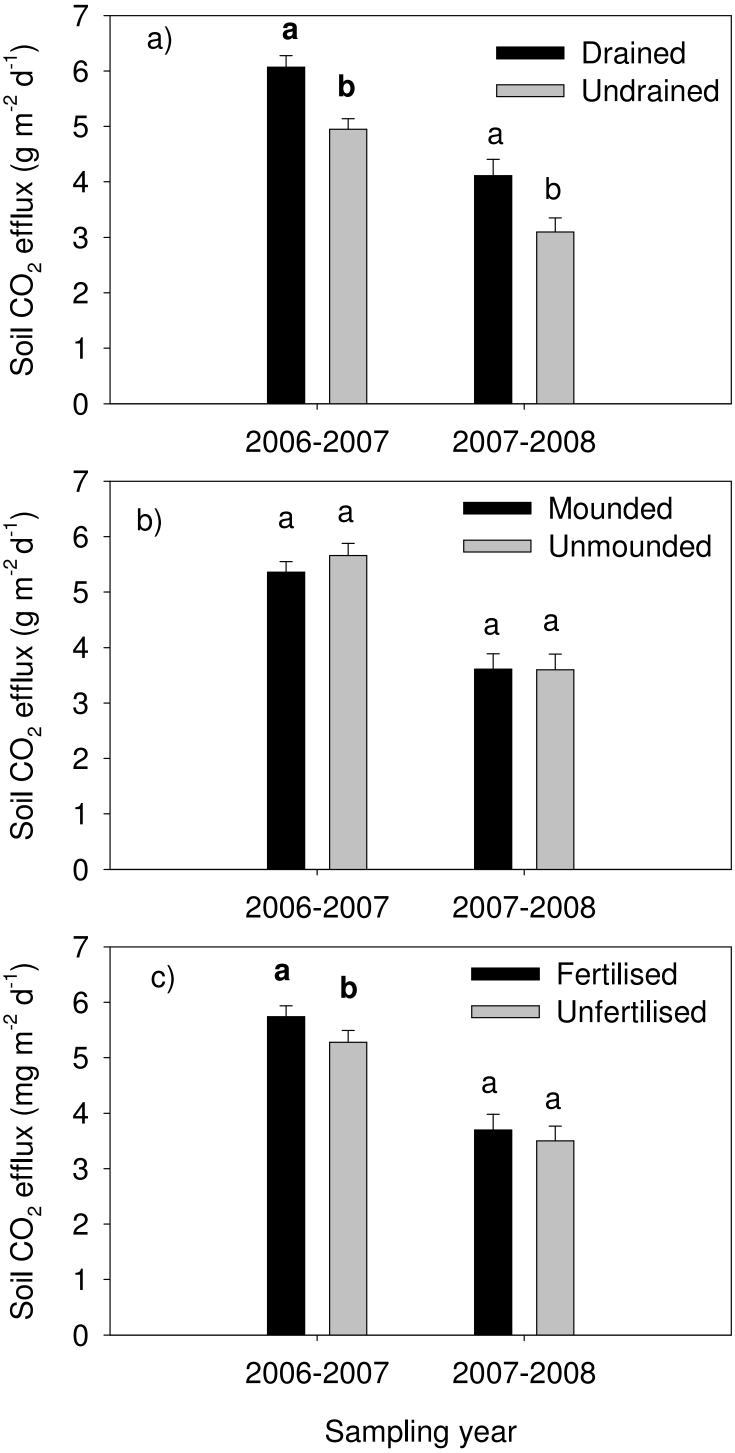

Figure 3.6. Averaged daily CO_2 emissions: a) drained and undrained plots, b) mounded and unmounded subplots and c) fertilised and unfertilised subplots. The vertical bars indicate standard errors of mean. Different letters in bold indicate significant difference in emissions ($P<0.05$) between the drained vs. undrained, mounded vs. unmounded and fertilised vs. unfertilised treatment (n=15 and 11 for 2006–07 and 2007–08)

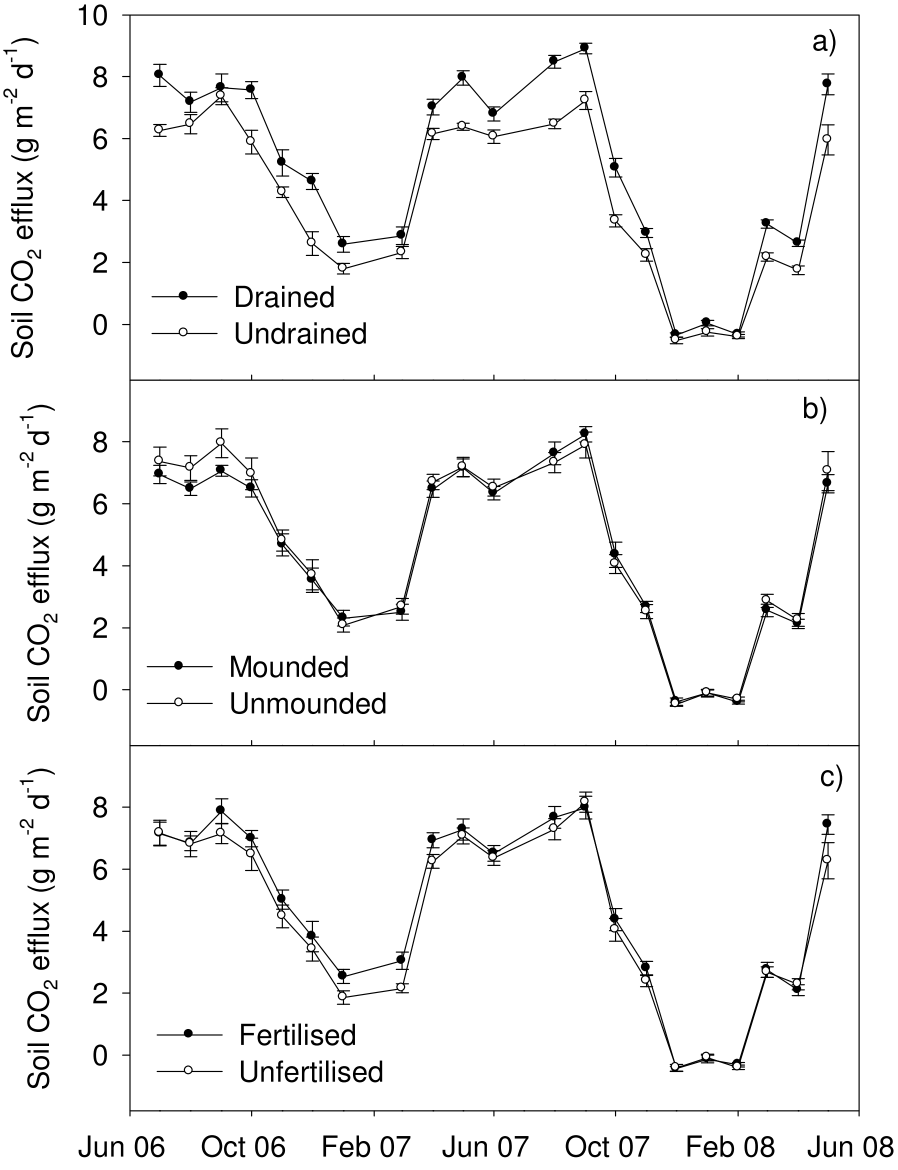

Figure 3.7. Averaged monthly soil CO_2 emissions in; a) the drained and undrained, b) mounded and unmounded and c) fertilised and unfertilised treatment. The vertical bars indicate standard errors of mean (n=26).

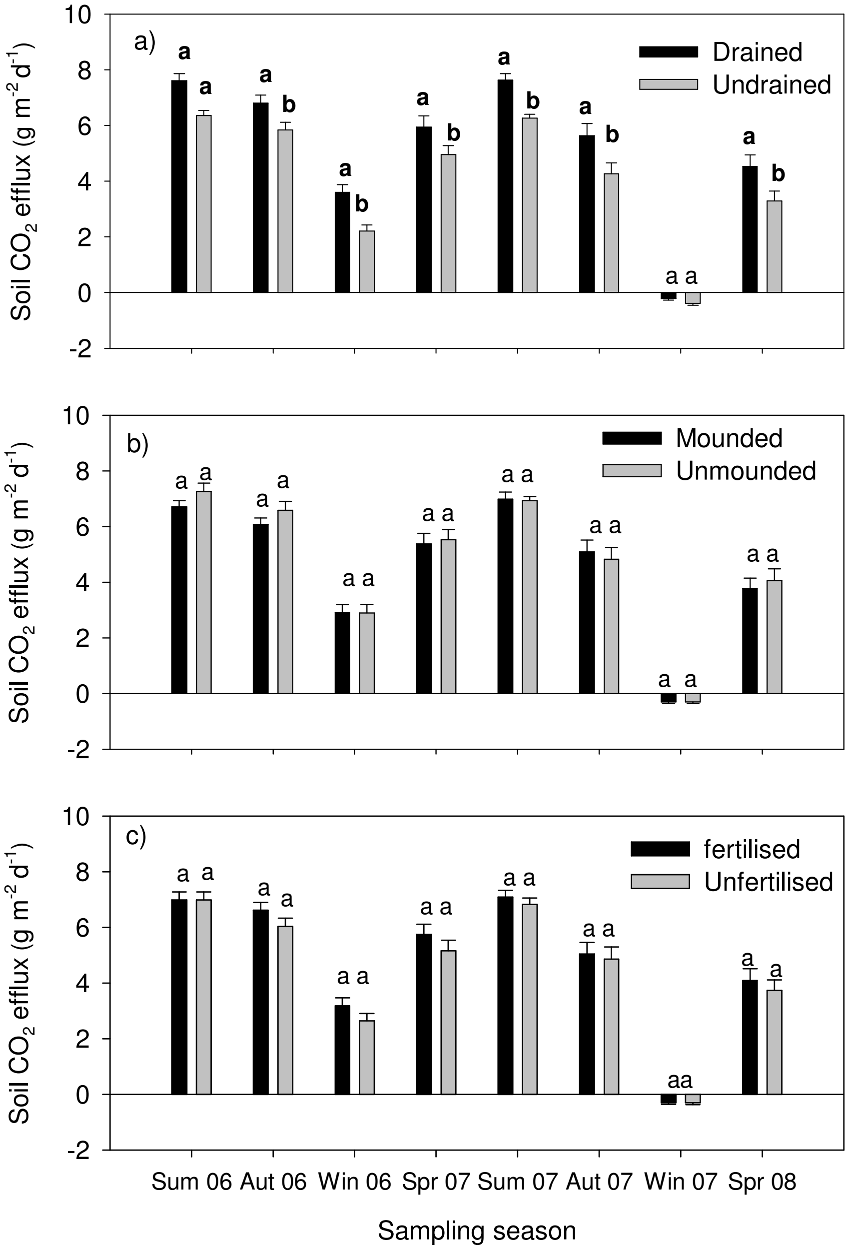

Figure 3.8. Averaged seasonal soil CO_2 emissions (sum=summer, aut=autumn, win=winter and spr=spring) from summer 2006 to spring 2007, a) drained and undrained; b) mounded and unmounded and c) fertilised and unfertilised treatment. The vertical bars represent standard error of means. Different letters in bold indicate significant differences in fluxes ($P<0.05$) between the drained vs. undrained, mounded vs. unmounded and fertilised vs. unfertilised treatment in each season.

3.3.2.2 Effects of mounding on soil CO_2 efflux

Soil respiration was not affected by mounding (P=0.07, Table 3.2). Average daily soil CO_2 fluxes are summarised in Fig. 3.6b. The average soil CO_2 efflux over the two year of study was 4.44±0.18 and 4.58±0.19 g m^{-2} d^{-1} in the mounded and unmounded subplot respectively. Soil respiration in the mounded and unmounded treatment varied from month to month ($P=0.0001$, Table 3.2). The mean monthly fluxes varied from a sink (–0.40±0.06 g m^{-2} d^{-1}) to a net source for atmospheric CO_2 (8.24±0.25 g m^{-2} d^{-1}) in the mounded subplots and from a sink (–0.47±0.07 g m^{-2} d^{-1}) to a net source of atmospheric CO_2 (7.95±0.47g m^{-2} d^{-1} g m^{-2} d^{-1}) in the unmounded treatment (Figure 3.7b). Soil respiration in the mounded and unmounded subplots showed a strong seasonal trend, with higher fluxes during the summer and lower fluxes during the winter (Fig. 3.7b). There were no significant differences in fluxes between the mounded and unmounded treatment in any season (Fig. 3.8b). However, the mounded and mounded subplots were net sinks for atmospheric CO_2 in the winter of 2007–08. Annual soil CO_2 fluxes are shown in Table 3.3. Over the two years of study, the mounded and unmounded subplots released 16.22±0.66 and 16.73±0.70 t CO_2 ha^{-1} yr^{-1} to the atmosphere.

Soil respiration measured in the mounded subplots was significantly affected by the position of chambers ($P=0.0001$) and fertilisation ($P=0.0001$, Table 3.4). There was a significant interaction between fertilisation and the position of chambers ($P=0.001$). Soil respiration differed significantly between the hollows, mounds and undisturbed ground in 2006–07 ($P=0.0001$) but not in 2007–08 ($P=0.3$). The average soil respiration in the hollows, mounds and undisturbed soil was 3.49±0.26, 4.23±0.32 and 6.25±0.57 g m^{-2} d^{-1} in 2006–07 and 3.17±0.37, 3.36±0.38 and 3.67±0.29 g m^{-2} d^{-1} in 2007–08. Soil respiration in the hollows, mounds and undisturbed ground showed a strong seasonal trend, with higher fluxes during the summer and lower fluxes during the winter (Fig 3.9). Over the two years of study, the annual CO_2 fluxes in the mounded subplots were on average 12.25±0.74, 14.09±0.72 and 18.83±0.63 t CO_2 ha^{-1} yr^{-1} in the hollows, mounds and undisturbed ground (Table 3.5).

Table 3.5. Annual fluxes (t CO_2 ha^{-1} yr^{-1}) for hollows, mounds and undisturbed ground (UDG). Values are mean±standard error. The number in the brackets indicates the annual C efflux (t C ha^{-1} yr^{-1}).

Sampling year	Subsite					
2006–2007	Hollows	n	Mounds	n	UDG	N
CO_2 (t $ha^{-1}yr^{-1}$)	12.76±0.83	15	15.43±0.72	11	22.81±0.62	26
[C] (t ha^{-1} yr^{-1})	3.48±0.23		4.20±0.20		6.22±0.17	
2007–2008						
CO_2 (t $ha^{-1}yr^{-1}$)	11.55±1.34	15	12.26±1.38	11	13.40±1.06	26
[C] (t ha^{-1} yr^{-1})	3.15±0.37		3.34±0.38		3.65±0.29	
2006–2008						
CO_2 (t $ha^{-1}yr^{-1}$)	12.25±0.74	15	14.09±0.72	11	18.83±0.63	26
[C] (t ha^{-1} yr^{-1})	3.34±0.20		3.84±0.20		5.13±0.17	

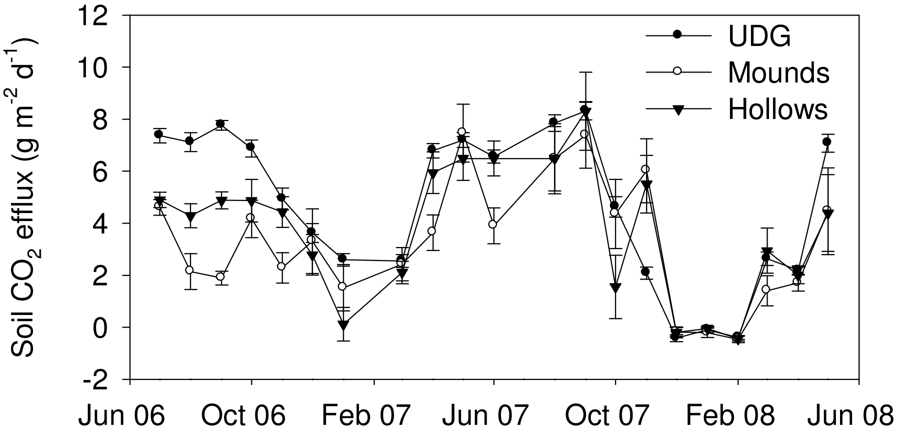

Figure 3.9 Seasonal variations in soil CO_2 efflux rates in undisturbed ground (UDG), mounds and hollows. The vertical bars indicate the standard error of the mean (n=26).

3.3.2.3 Effects of fertilisation on soil CO_2 efflux

The results of this study showed that soil respiration was overall increased by fertilisation (P=0.04, Table 3.2). Fertilisation increased soil respiration in 2006–07 (P=0.01) but not in 2007–08 (P=0.6). Averaged daily soil CO_2 fluxes are shown in Fig. 3.6c. The average soil respiration over the two years of study was 4.67±0.19 and 4.35±0.18 g m^{-2} d^{-1} in the fertilised and unfertilised treatment. Averaged monthly fluxes are shown in Fig 3.7c. Fluxes in the fertilised and unfertilised treatment varied from month to month (P=0.0001, Table 3.2). The monthly fluxes varied from a net sink (–0.44±0.07 g m^{-2} d^{-1}) to a net source of atmospheric CO_2 (7.98±0.37 g m^{-2} d^{-1}) in the fertilised subplots and 0.42±0.11 to 8.15±0.33 g m^{-2} d^{-1} in the unfertilised subplots. Soil respiration in the fertilised and unfertilised subplots showed a strong seasonal trend, with higher fluxes during the summer and lower fluxes during the winter (Fig 3.7c). The fertilised and unfertilised subplots were net sinks for atmospheric CO_2 in the winter of 2007–08. Annual soil CO_2 fluxes were higher in 2006–07 than in 2007–08 (Table 3.3). Over the two years of study, the fertilised and unfertilised subplots released 17.06±0.68 and 15.88±0.67 t CO ha^{-1} yr^{-1} to the atmosphere.

3.3.3 Relationships between soil CO_2 efflux and environmental variables

Exponential regressions (Boone et al. 1998; Burton et al. 1998; Buchmann 2000; Ball et al. 2007; Han et al 2007), linear regressions (Raich and Schlesinger 1992; Ben-Asher et al. 1994) and Arrhenius equations (Lloyd and Taylor 1994; Thierron and Laudelout 1996) have been used to describe the relationship between soil temperature and soil respiration. In present study site, exponential relationships between soil respiration and soil temperature exhibited year to year variation. Soil temperature at all depths explained 53 to 66% of the variability in soil respiration observed in all treatments in 2006–07 (Table 3.6). A strong exponential relationship between soil respiration and soil temperature (R^2 between 0.77 and 0.88) was observed in 2007–08 (Table 3.6). However, when the averaged monthly data for the whole measurement period (2006–2008) were pooled together, linear regression equations described the relationship between soil respiration and soil temperature better than the exponential function, R^2 between 0.79 and 0.84, Fig. 3.10 for the linear regressions, compared to R^2 between 0.67 and 0.75 for the exponential equation Table 3.7).

Soil respiration in this site was significantly related to soil moisture content (linear relationship R^2 between 0.21 and 0.75) when soil moisture was treated as an independent variable. Similarly, a linear relationship was also observed between soil respiration and soil water table depth, when the latter was treated as an independent variable (R^2 between 0.35 and 0.57). Correlation analysis (not shown) showed a significant correlation between all measured environmental variables in this site.

However, in view of the fact that environmental variables were not independent of each other, a forward stepwise multiple regression analysis was employed. This showed that soil temperature (T_1) (Fig. 3.10a) and soil moisture content (Fig. 3.11) were the most important variables in the drained and undrained plots. Soil temperature alone explained 80 and 84% of the variability observed in soil respiration in the drained and undrained plots. The variance increased to 84 and 88% when soil moisture was included in the model. The relationship between soil respiration and the soil water table in the drained and undrained plots appeared to be purely a result of the latter's relationship with soil temperature and soil moisture. A forward stepwise multiple regression analysis showed that soil temperature (T_1) was the most important factor controlling soil respiration in the fertilised subplots accounting for 82% of the variance (Fig. 3.10b). In the unfertilised subplots, T_1, T_5 and T_{10} were the most important factors driving soil respiration; T_1 alone explained 83% (Fig.3.10b) of the variance which increased to 85% and 88% when T_5 and T_{10} were included. Soil temperature (T_{10}) was the most important factor controlling soil respiration in the mounded and unmounded subplots, accounting for 82 and 83% of the variance (Fig 3.12). The relation between soil respiration and soil moisture and soil water table in this site was purely a result of their relationship with soil temperature.

The exponential equation was used to calculate Q_{10} values for each sampling year (Table 3.6). The Q_{10} values calculated across treatments were lower (1.8–2.3) in 2006–07 than in 2007–08 (2.3–6.5). On average the Q_{10} values varied between 2.4 and 3.4 across treatments (Table 3.7).

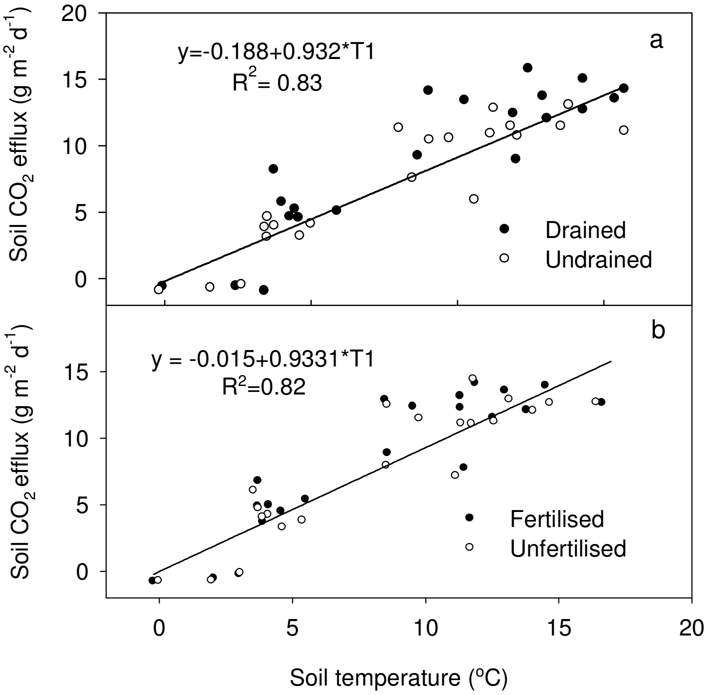

Figure 3.10. Linear relationship between soil CO_2 efflux and soil temperature (T_1) (P=0.0001) for a) drained and undrained, b) fertilised and unfertilised (n=21). Data are treatment means for each month.

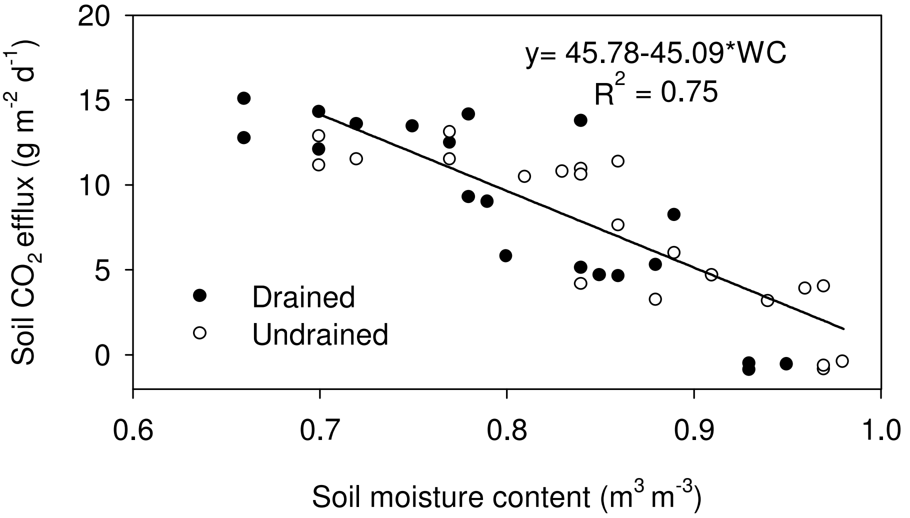

Figure 3.11. Linear relationship (P=0.0001) between soil CO_2 efflux and soil moisture content in the drained and undrained plots (n=21). Data are treatment means for each month from July 2006 to May 2008.

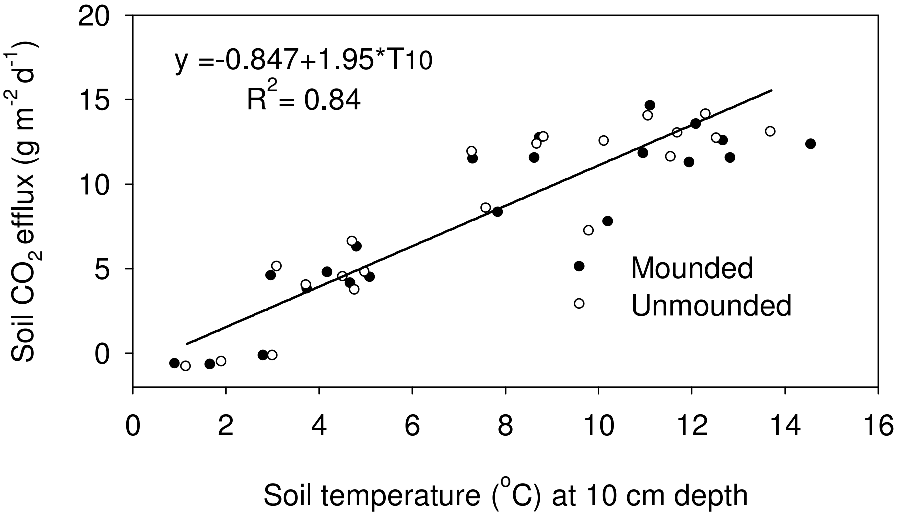

Figure 3.12. Linear relationship between soil CO_2 efflux and soil temperature (T_{10}) (P=0.0001) in the mounded and unmounded subplots (n=21). Data are treatment means for each month from July 2006 to May 2008.

Table 3.6. Annual exponential relationships between soil respiration rates (RS) with soil temperature (T), goodness of fit (R^2), Q10, number of samples and P value of exponential relationship. Q_{10} presented here are based on sampling year, i.e. July 2006–May 2007 and June 2007–May 2008. The numbers under (n) are monthly averaged values from all chambers for each treatment. Soil temperatures were measured at 1, 5 and 10 cm respectively.

Treatment	Exponential function $RS = \alpha e^{\beta T}$												
2006–2007	α	B	R^2	Q_{10}	n	P	2007–2008	α	B	R^2	Q_{10}	n	P
Drained							Drained						
1 cm	5.75	0.061	0.62	1.8	10	0.01	1 cm	1.81	0.152	0.87	4.6	11	<0.0001
5 cm	6.05	0.063	0.56	1.9	10	0.01	5 cm	2.02	0.160	0.84	5.0	11	<0.0001
10 cm	5.77	0.069	0.58	2.0	10	0.01	10 cm	1.92	0.273	0.84	5.7	11	<0.0001
Undrained							Undrained						
1 cm	4.29	0.073	0.66	2.0	10	0.01	1 cm	1.43	0.176	0.85	5.8	11	<0.0001
5 cm	4.53	0.071	0.56	2.0	10	0.01	5 cm	1.65	0.159	0.81	4.9	11	<0.0001
10 cm	4.24	0.081	0.59	2.2	10	0.01	10 cm	1.52	0.174	0.81	5.7	11	<0.0001
Mounded							Mounded						
1 cm	5.21	0.058	0.61	1.8	10	0.01	1 cm	2.455	0.112	0.77	3.1	11	<0.0001
5 cm	5.35	0.062	0.54	1.8	10	0.02	5 cm	1.943	0.145	0.83	4.2	11	<0.0001
10 cm	5.11	0.068	0.56	2.0	10	0.01	10 cm	1.52	0.181	0.88	6.1	11	<0.0001
Unmounded							Unmounded						
1 cm	5.092	0.068	0.66	2.0	10	0.01	1 cm	2.129	0.133	0.83	3.4	11	<0.0001
5 cm	5.19	0.07	0.59	2.0	10	0.01	5 cm	1.64	0.169	0.87	5.4	11	<0.0001
10 cm	4.78	0.083	0.64	2.3	10	0.01	10 cm	1.50	0.187	0.88	6.5	11	<0.0001
Fertilised							Fertilised						
1 cm	5.71	0.057	0.62	1.8	10	0.01	1 cm	1.69	0.163	0.86	5.1	11	<0.0001
5 cm	5.92	0.059	0.53	1.8	10	0.02	5 cm	1.95	0.158	0.83	4.5	11	<0.0001
10 cm	5.53	0.068	0.58	2.0	10	0.01	10 cm	1.84	0.172	0.83	5.6	11	<0.0001
Unfertilised							Unfertilised						
1 cm	4.62	0.069	0.65	2.0	10	0.01	1 cm	1.59	0.163	0.87	5.1	11	<0.0001
5 cm	4.66	0.075	0.60	2.1	10	0.01	5 cm	1.83	0.157	0.83	4.8	11	<0.0001
10 cm	4.40	0.083	0.62	2.3	10	0.01	10 cm	1.73	0.170	0.84	5.5	11	<0.0001

Table 3.7. The exponential relationships between soil respiration rates (RS) with soil temperature (T) measured at 1, 5 and 10 cm. The Q_{10} values are for the period from July 2006 to May 2008. The numbers under (n) are average values from all chambers for each treatment in each sampling month.

Treatment	Exponential function RS = $\alpha e^{\beta T}$												
Drained	α	β	R^2	Q_{10}	n	P	Undrained	A	β	R^2	Q_{10}	n	P
1 cm	3.53	0.098	0.73	2.7	21	<0.0001	1 cm	2.94	0.105	0.72	2.8	21	<0.0001
5 cm	3.86	0.101	0.69	2.7	21	<0.0001	5 cm	3.00	0.105	0.68	2.9	21	<0.0001
10 cm	3.67	0.109	0.70	3.0	21	<0.0001	10 cm	2.79	0.117	0.70	3.2	21	<0.0001
Mounded							Unmounded						
1 cm	3.52	0.086	0.69	2.4	21	<0.0001	1 cm	3.36	0.097	0.74	2.6	21	<0.0001
5 cm	3.47	0.096	0.69	2.6	21	<0.0001	5 cm	3.304	0.108	0.72	3.0	21	<0.0001
10 cm	3.23	0.109	0.71	3.0	21	<0.0001	10 cm	3.03	0.122	0.75	3.4	21	<0.0001
Fertilised							Unfertilised						
1 cm	3.67	0.092	0.70	2.5	21	<0.0001	1 cm	3.17	0.099	0.73	2.7	21	<0.0001
5 cm	3.77	0.098	0.67	2.6	21	<0.0001	5 cm	3.16	0.108	0.71	2.9	21	<0.0001
10 cm	3.47	0.110	0.69	3.0	21	<0.0001	10 cm	2.98	0.117	0.72	3.2	21	<0.0001

3.4 DISCUSSION

3.4.1 Effects of site preparation on environmental factors

In general, drainage lowered the soil water table, improved aeration and altered the soil thermal properties and energy fluxes near the soil surface. In this study, soil temperature was increased by drainage. The soil temperature was higher in the drained plots than the undrained plots because well-drained soils warm faster than wet soils. In absolute values, the difference in soil temperature between the drained plots and their undrained counterparts varied between 0.2 and 1.2°C at all depths. These results are consistent with results of drainage studies conducted in peatland sites which also found that peat temperature increased after drainage (e.g., Grootjans et al. 1985; Lieffers and Rothwell 1987; Lieffers 1988; Macdonald and Lieffers 1990; Van Cleve et al. 1990; Prévost et al. 1997). For example, Prévost et al. (1997) found that seasonal maximum temperature at 10 cm depth increased by 3.5°C after drainage (5 m away from ditches) and by 1.5° C (at distances greater than 5 m). Mounding increased soil temperature at 1 and 5 cm depth, probably because mounding exposed the mineral soil on top of mounds which absorbed more heat than the soil surface covered with litter or vegetation. In absolute values, the difference in soil temperature between mounded and unmounded subplots was 0.2 and 1.3°C. This finding is consistent with results of mounding experiments conducted in organic soils which found that soil temperature was higher on the top of mounds that on undisturbed ground (e.g., Sutton 1993; DeLong et al. 1997; Saari et al. 2004).

Soil moisture content at the present study site was decreased by drainage between May and October. This may have occurred because drainage improves aeration and increases soil temperature and evapotranspiration, resulting in a drier soil surface. Soil moisture was also decreased by mounding between May and October and this was probably caused by improved soil aeration and increased soil temperature which probably dried the soil to a depth of at least 5 cm, where soil moisture content was measured particularly at the top and around mounds. Saari et al (2004) also observed low moisture content on the top of mounds than on undisturbed soil.

The soil water table depth was significantly lowered by drainage, similar to the effect observed in other peatland sites following drainage (e.g., Laine et al. 1995b; Martikainen et al. 1995; Regina et al. 1996; Minkkinen and Laine 1998; Nykänen et al. 1998). The average soil water table was 23.0 cm below the soil surface in the drained plots and 13.8 cm in their undrained counterparts. The soil water table depth was not affected by mounding or fertilisation.

3.4.2 Effect of drainage on soil CO_2 efflux

Afforestation of peaty gley soils in the UK usually requires lowering the water table by drainage to increase the survival and growth of planted trees. The increase in tree growth may result from more oxygenated conditions in the root zone and increase in the rate of mineralisation of organic matter which increases the availability of soil nutrients (Von Arnold et al. 2005; Minkkinen et al. 2008). In addition to increasing forest growth, drainage has been shown to increased soil respiration from organic soils in the boreal and continental peatlands (e.g., Martikainen et al.1995; Silvola et al. 1996a; Minkkinen et al. 2008). In the present study, drainage increased soil respiration by 22.6 to 32.6%. Komulainen et al. (1999) measured soil respiration in southern Finland and found that fluxes increased soon after drainage depending on the effectiveness of ditches and site fertility. The result of the present study suggests that ditches were effective in lowering the soil water table depth. Drainage may increase soil respiration by improving aeration and increasing soil temperature as well as substrate availability, all of which can control the rate at which soil organic matter is decomposed (Trettin et al. 1996). Simulation studies lowering the soil water table conducted under laboratory conditions found that low water table depth treatments released 2 to 5 times more CO_2 than saturated peatland soils (Silvola et al. 1985; Moore and Knowles 1989; Moore and Dalva 1993; Freeman et al. 1993; Funk et al. 1994). Several field reports also demonstrated that increased oxygen availability after drainage may favour decomposition and turn peat soils from CO_2 sink to source (e.g., Glenn et al. 1993; Martikainen et al. 1995; Nykänen et al. 1995; Silvola et al. 1996a; Minkkinen et al. 2008). For example, Silvola et al. (1996a) measured soil respiration in Finland and found that lowering the soil water table depth by 5 to 9 cm did not affect soil respiration, whereas in sites where drainage lowered the soil water table by 12 to 40 cm, CO_2 emissions doubled.

Soil respiration processes (heterotrophic and autotrophic) are controlled by soil temperature, soil moisture content and oxygen availability (Raich and Schlesinger 1992; Peterjohn et al. 1994; Kätterer et al. 1998; Saiz et al. 2006) which were improved by drainage in this study. Heterotrophic respiration results from metabolic activity of a wide range of soil microorganisms that decompose organic matter (Christensen et al. 1996). Apart from environmental factors, heterotrophic respiration may be affected by the population of soil microorganisms and the quantity and quality of soil organic matter. In the present study, drainage improved aeration and increased soil temperature, and may have favoured the soil microbial population and activity and enhanced decomposition and soil respiration. Furthermore, increased oxygen availability and soil temperature after drainage may have favoured plant root growth of *Calluna vulgaris*, *Festuca ovina*, etc., vegetation in this study (e.g., Silvola et al. 1996b; Verville et al. 1998; Finer and Laine 1998; Hanson et al. 2000). Finer and Laine (1998) reported that root production increased with decreasing soil water table in a Scots pine forest in Finland, indicating that the production of fine roots may play an important role in the cycling of C in well drained organic soils. In addition to direct contribution to total soil respiration, plant roots may produce highly decomposable fine roots and root exudates when exposed to aerobic conditions (e.g., Thomas et al. 1996) which, probably also occurred in the present study site after drainage. The availability of decomposable fine roots and organic exudates has been shown to favour the growth and activity of soil microorganisms as well as organic matter decomposition (Lohila et al. 2003; Kuzyakov and Cheng 2004).

Soil respiration in the drained and undrained plots showed a strong seasonal trend, with higher fluxes during the summer and lower fluxes during the winter. High soil CO_2 fluxes observed in the summer to early autumn (June to September) was perhaps due to favourable soil temperatures (12–15°C) and soil moisture (0.66–0.78 $m^3\ m^{-3}$) which increased both heterotrophic and autotrophic respiration (e.g., Weber 1985). This result is consistent with results of studies conducted in various ecosystems and soil types (Edward and Sollins 1973; Schlentner and Van Cleve 1985; Jensen et al. 1996; Silvola et 1996b; Zogg et al. 1996; Mallik and Hu 1997; Davidson et al. 1998; Lin et al. 1999; Law et al. 1999; Xu and Qi 2001; Wan and Luo 2003). For example, Edwards and Sollins (1973) found that the average soil CO_2 efflux in a deciduous forest was lower in the spring and higher in the summer. Jensen et al. (1996) measured soil respiration in arable land and reported maximum fluxes at soil temperatures between 16 and 18˚C. Schlentner and Van Cleve (1985) also measured soil respiration in a mature forest in interior Alaska and observed that optimum fluxes occurred when the soil temperature was 17°C. There was consumption of CO_2 in both the drained and undrained plots in the winter of 2007–08 when the soil was water saturated, which may have reduced the availability of oxygen for heterotrophic and autotrophic respiration (Rey et al. 2002). Soil respiration is driven by temperature (e.g., Davidson et al. 1998; Buchmann 2000; Zerva and Mencuccini 2005a), suggesting that low temperatures (–0 to 3˚C) observed in the winter of 2007–08 could have contributed to CO_2 consumption. In general, water saturated soil consume or emit insignificant amount of CO_2, but emit C and CH_4 (Alm et al. 1997; Carrol and Crill et al. 1997; Nykänen et al. 1998).

Soil respiration showed a strong year to year variation. Soil respiration was greater in 2006–07 than in 2007–08. This may have occurred because 2006–07 was warmer than 2007–08. This finding agrees with results of previous studies conducted in various ecosystems world-wide (e.g., Frank et al. 2002; Irvine and Law 2002; Melillo et al. 2002; Epron et al. 2004; King et al. 2004; Von Arnold et al. 2005; Ball et al. 2007). For example, Von Arnold et al. (2005) studied soil respiration in an undrained mire site in Sweden and reported annual fluxes of 0.8×10^4 and 1.2×10^4 kg CO_2 ha^{-1} yr^{-1} in year 1 and 2, respectively. Ball et al. (2007) measured 7.8 and 10.1 t CO_2 ha^{-1} yr^{-1} in a 20 year old conifer stand and 22.3 and 13.6 t CO_2 ha^{-1} yr^{-1} in a 30 year old conifer stand at Harwood Forest in 2001 and 2002, respectively.

Annual soil CO_2 emissions observed in the drained and undrained plots were higher than values reported for organic soils drained for pasture (Nykänen et al 1995; Langveld et al. 1997). For example Nykänen et al. (1995) measured soil respiration in a fen drained for pasture in Finland and reported fluxes of 1500 kg CO_2 ha^{-1} yr^{-1}. Langeveld et al. (1997) studied soil CO_2 fluxes in peat soils drained for pasture in The Netherlands and reported that soil respiration released 11×10^3 kg CO_2 ha^{-1} yr^{-1}. Low CO_2 fluxes from grazed grasslands may be due to grazing animals, which reduces below-ground biomass and also cause soil compaction. Soil compaction may decrease the soil macroporosity and, thus reduce air diffusion and water infiltration rates (Pritchett 1979). Changes in these soil properties may increase the soil water content and consequently create a more anaerobic soil environment and reduce soil respiration (Zerva and Mencuccini 2005a). The annual fluxes in the drained and undrained plots were lower than 6.9–7.9×103 kg CO_2–C ha^{-1} yr^{-1} reported for drained organic soils used for arable agriculture in the boreal region Maljanen et al. (2004).

3.4.3 Effect of mounding on soil CO_2 efflux

Mounding is a ground preparation method commonly used for tree planting in peaty gley soils in upland Britain. Mounding buries the litter and the organic layers beneath the mineral soil on the top of mounds (Saari et al. 2004). Increased soil temperature during day-time, more pronounced wet and dry cycles as well as the mixing of the organic matter and mineral soil are considered the most important drivers of biological processes in cultivated organic soils (Johansson 1994). Soil respiration is expected to increase on the top of mounds because increased soil temperature and oxygen availability have the potential to increase soil microbial activity and organic matter decomposition rates (e.g., Doran 1980; Kessavalou et al. 1998; Rochette and Angers 1999).

Soil respiration in this study was not affected by mounding. This result contradict results of previous studies which reported that site preparation carried out prior to afforestation enhances organic matter decomposition rates and soil respiration (e.g., Johansson 1994; McClellan et al. 1990; Millikin and Bowden. 1996; Lundmark-Thelin and Johansson 1997; Mallik and Hu 1997; Smolander et al. 1998) because mixing the organic and the mineral soil layers increases soil temperature and oxygen availability and may stimulate soil microbial activity (e.g., Armentano and Menges 1986; Winkler et al. 1996; Davidson et al. 1998). Previous studies compared soil CO_2 released from mounds and undisturbed soil without considering emissions from hollows where the soil used to make mounds was excavated from.

In the present study, soil respiration was measured from three different subsites in the mounded subplots (e.g., hollows, mounds and undisturbed ground). Mounding created three subsites with a different microclimate and organic matter distribution, hence the importance of evaluating CO_2 released from the hollows. In 2006–07, the averaged CO_2 fluxes were higher in the undisturbed ground than on the top of mounds and inside hollows. Mounding improved aeration and increased soil temperature on the top of mounds all of which may enhance decomposition, it is therefore possible that the soil organic layers and the mineral soil on the top of mounds were not well mixed in 2006–07 and restricted microbial decomposition (e.g., Maier and Kress 2000; Pangle and Seilier 2002). Gas diffusion is important in determining soil aeration in soils which may influence soil microbial activity and organic matter decomposition. It is possible that the mineral soil on the top of mounds impended oxygen diffusion and depressed microbial activity and thereby slowing the CO_2 evolution in 2006–07 (e.g., Skopp et al. 1990). Field observations showed that the vegetation previously growing beneath mounds was suffocated by mounds and root respiration may have ceased, reducing soil respiration on the top of mounds. Plant root activities may influence soil respiration (Bowden et al. 1993; Raich and Tufekgcioglu 2000) and have been shown to contribute about 50% to total soil respiration (Ewel et al. 1987b; Irvine and Law 2002).

Hollows were periodically covered with water, particularly during the winter to early spring and this may have hindered organic matter decomposition. This agrees with results of Ball et al. (2007) who reported that pools of water in a clearfelled forest and inside drainage ditches reduced soil respiration at Harwood forest. Plants roots inside hollows were removed with the soil excavated to form mounds and this may have decreased autotrophic respiration. There was no significant difference in soil respiration between the hollows, mounds and undisturbed soil in 2007–08. The increase in fluxes on the top of mounds in 2007–08 may indicate that the soil organic and mineral soil layers were well mixed and increased organic matter decomposition and soil respiration. The mixing of the soil layers may have created an environment favourable for soil microbial activity and decomposition in 2007–08. This agrees with results of earlier studies which reported that site preparation increased soil respiration (Millikin and Bowden 1996; Mallik and Hu 1997; Lytle and Cronan 1998; Londo et al. 1999; Pumpanen et al. 2004b). For example, Pumpanen et al. (2004b) reported that soil respiration on the top of mounds increased slightly after mounding compared to the soil where residues were left in a clearfelled forest in Finland. They suggested that the partial mixing of the organic matter with the mineral soil improved aeration and increased soil temperature and may have favoured soil microbial activity and decomposition. The water inside hollows evaporated between the spring and autumn of 2007 and may have caused aerobic conditions favourable for decomposition in 2007–08. Furthermore, higher soil CO_2 fluxes were observed inside the hollows and on the top mounds in the summer to early autumn, particularly after sporadic rainfall events. These high fluxes may be an indication that the soil temperature and moisture condition inside the hollows and on the top of mounds was favourable for organic matter decomposition (e.g., Winkler et al. 1996; Davidson et al. 1998). There was consumption of CO_2 in the mounded and unmounded subplots in the winter of 2007–08, probably because the soil was water saturated and temperatures were low.

3.4.4 Effect of fertilisation on soil CO_2 efflux

Previous studies demonstrated that atmospheric N deposition or fertiliser application increase biomass production in temperate and boreal forest ecosystems (Aber et al. 1998; Albaugh et al. 1998; Canary et al. 2000). The application of N to infertile boreal and temperate forest ecosystems has been shown to increase tree roots, foliage and wood biomass (Canary et al. 2000; Albaugh et al. 1998). The effect of N fertilisation on below-ground processes is not well understood. Fertilisation increase nutrient availability and may enhance tree growth and above-ground litter, but may also decrease root biomass because trees allocate more C to wood production than to below-ground processes in response to increased fertility (e.g., Eriksson et al. 1996; Kurz et al. 1996; Gunderson et al. 1998; Raich and Tufekgcioglu 2000; Helmisaari et al. 2007). Several studies demonstrated that N fertilisation has different effects on C losses from tree roots (rhizodeposition). For example, Uselman et al. (2000) reported that N increases rhizodeposition. In contrast, Bowden et al. (2004) demonstrated that N fertilisation decreases rhizodeposition in forest soils. The effect of N fertilisation on rhizodeposition depends on whether root biomass is increased or decreased. Nitrogen fertilisation and atmospheric N deposition have also been shown to have different effects on soil respiration. Several studies (e.g., Hobbie, 2000; Pregitzer et al. 2000; Vestgarden, 2001; Burton et al. 2002; Bowden et al. 2004; Minkkinen et al. 2008) demonstrated that N increases soil respiration, and suggested that the stimulatory effect of N on organic matter decomposition may decrease C storage in forest soils (e.g., Aber et al. 1993; Cao and Woodward 1998). In contrast, other reports demonstrated that N fertilisation and atmospheric N deposition decreases organic matter decomposition and, thus suppresses soil respiration and may increase organic matter storage in forest ecosystems (Magill and Aber 1998; Bowden et al. 2000; Burton et al. 2002; Franklin et al. 2003; Foereid et al. 2004; Jandl et al. 2007; Pregister et al. 2008). Other studies found no effect of N fertilisation on organic matter decomposition and soil respiration (Rochette and Gregorich 1998; Hobbie and Vitousek 2000).

Fertilisation increased soil respiration by 5.6 to 8.7%. This finding agrees with results of previous studies which found that organic matter decomposition and soil respiration increased following N fertilisation (Hobbie 2000; Pregitzer et al. 2000; Burton et al. 2002; Uselman et al. 2000; Vestgarden 2001; Bowden et al. 2004; Minkkinen et al. 2008). Minkkinen et al. (2008) found that fertilisation increased pH in peatland soils. They suggested that the increase in soil pH may improve the litter nutrient content and enhance decomposition and soil respiration. However, soil pH in this study was not affected by fertilisation. Fertilisation increased standing above-ground plant biomass by 20% in the first year of this study. Although root biomass was not evaluated, it is possible that it also increased with standing above-ground plant biomass in the first year following fertilisation. Root respiration in the present study site is expected to increase following fertilisation because grasses, with virtually no C allocation to wood production like trees do, may have more photosynthate available to allocate to belowground than trees (Raich and Tufekgcioglu 2000). In contrast, several studies found that N fertilisation decrease soil respiration in forest soils (Haynes and Gower 1995; Knapp et al. 1998; Magill and Aber 1998; Maier and Kress 2000; Cardon et al. 200; Ågren et al. 2001). The mechanism which decreases organic matter decomposition in fertilised forests soils is not well understood. Foereid et al. (2004) and Jandl et al (2007) suggested that N may decrease decomposition rates of old organic matter by suppressing ligninolytic enzymes of soil microorganisms and by chemical stabilisation. A review of literature by Fog (1988) concluded that (i) N changes the composition of the microbial community through competition; (ii) ammonia suppresses the production of enzymes required for degradation of lignin and other recalcitrant compounds; and (iii) ammonia and amino compounds react with organic matter to form recalcitrant material.

There was no significant difference in soil respiration between the fertilised subplots and their unfertilised counterpart in 2007–08, although the fertilised subplots respired more than the unfertilised subplots. The leaching of NO_3^- (e.g., Vitousek and Matson 1985; Smith et al. 1994), N uptake by plants (e.g., Vitousek and Matson 1985; Emmett et al. 1991) and losses as gaseous N_2O and N_2 caused by nitrification and denitrification (e.g., Robertson et al. 1987; Brumme 1995; Sitaula et al. 1995) in 2006–07, probably depleted the pool of additional N for root growth and maintenance in 2007–08. Net CO_2 uptake occurred in the fertilised and unfertilised subplots in the winter of 2007–08, probably due to observed water saturated soils and low temperatures.

3.4.5 Soil CO_2 emissions in relation to environmental variables

Soil temperature explained the largest fraction of the variability in soil respiration rates observed across treatments in this site. This result agrees with results of previous studies (Davidson et al. 1998; Buchmann 2000; Maier and Kress 2000; Janssens and Pilegaard 2003; Pangle and Seiler 2002; Zerva and Mencuccini 2005a; Han et al. 2007). Soil respiration in all treatments showed a strong seasonal trend, with higher fluxes during the summer and lower fluxes during the winter. Soil temperature is the driver of most biological processes in soils. In the case of soil respiration, increased soil temperature may stimulate the soil microbial population and activity and accelerate organic matter decomposition. In addition, increasing soil temperature may favour autotrophic respiration by increasing plant photosynthesis and photosynthate translocation from above-ground plant parts (Han et al. 2007). Davidson et al. (1998) studied soil respiration in a temperate mixed hardwood forest in central Massachusetts, USA. They reported that soil temperature explained 80% of the variability observed in soil respiration. Zerva and Mencuccini (2005a) reported that soil temperature explained 79 to 90% of the variation in soil CO_2 fluxes in a mature Sitka spruce forest at Harwood Forest. Buchmann (2000) used an exponential equation to describe the relationship between soil respiration and soil temperature in the O_f layer in a Norway spruce stands in Germany and found that soil temperature explained 75 to 81% of the variance. Han et al (2007) also used an exponential regression equation to fit a relationship between soil respiration and soil temperature in a maize (*Zea mays* L.) field and found that soil temperature explained 80 to 97% of the variance.

Soil organic matter decomposition increases with temperature (Kirschbaum 2000) and labile C is sensitive to temperature variation than recalcitrant carbon (Thornley and Cannell 2001; Lenton and Huntingford 2003). Incubation studies conducted by Niklinska et al. (1999) have estimated that soil warming will increase decomposition rates and CO_2 emitted from soils in the boreal and tundra region. This is because soils in the boreal and tundra region have the largest stock of labile organic carbon, and are predicted to experience the greatest increase in temperature (Schlesinger and Andrews 2000). Field warming experiments have demonstrated that warming either failed to enhance soil respiration significantly or the increase was limited to a very short initial time period (Luo et al. 2001; Melillo et al. 2002; Wan and Luo 2003; Gu et al. 2004). This phenomenon has been attributed to rapid depletion of labile carbon pools which are more sensitive temperature. In contrast, incubation studies conducted by Townsend et al. (1997) demonstrated that recalcitrant soil organic carbon is as sensitive to temperature as labile soil organic carbon.

Q_{10} values varied between 2.4 and 3.4 over the two years of study. These values are within the range of 2.0 to 3.9 reported for temperate soils (Schleser 1982; Baldocchi et al. 1986; Raich and Schlesinger 1992; Hanson et al. 1993; Howard and Howard 1993; Davidson et al. 1998; Kätterer et al. 1998; Buchmann 2000). Schleser (1982) suggested that the wide variation in Q_{10} values for soils depends on natural conditions such as fertility because organic matter decomposition rates are influenced by its physical-chemical conditions and quality. Q_{10} values across treatments were lower in 2006–07 than in 2007–08, probably because differences labile substrate availability (e.g., Winkler et al. 1996) and soil temperatures between the two sampling years. Winkler et al. (1996) reported Q_{10} values between 1.7 and 1.9 at temperatures between 4 and 28°C in the A horizon of forest soils. Zerva (2004) reported Q_{10} values between 1.9 and 2.0 in a clearfelled stand at Harwood forest. They attributed the low values to the absence of roots and autotrophic respiration. Boone et al. (1998) reported a Q_{10} value of 4.6 for root respiration and 3.5 for bulk soil respiration. Overall, the Q_{10} values in this site increased with soil depth and were consistently higher at 10 cm depth. This is consistent with results of previous studies which observed that that the Q_{10} value increased with soil depth (Russell and Voroney 1998; Zerva 2004).

Soil moisture content explained 21 to 75% of the variability observed in soil respiration in this site when it was treated independently. Applying a forward stepwise multiple regression analysis showed that soil moisture content interacted with soil temperature to influence soil respiration in the drained and undrained plots. The relationship between soil respiration and soil temperature involves complex interaction of soil temperature and soil moisture. This result agrees with results of previous studies which demonstrated soil temperature interacts with soil moisture to influence soil respiration in many ecosystems (Howard and Howard 1993; Euskirchen et al. 2003; Maestre and Cortina 2003). High soil moisture levels may affect soil respiration by decreasing oxygen availability, while low moisture levels may stress soil microbial community and plant roots (Rey et al. 2002) and limit soil respiration (Maier and Kress 2000; Pangle and Seiler 2002). Soil moisture levels in the present study site were never extremely high or low to limit soil respiration.

3.5 CONCLUSION

Our study shows that drainage and mounding carried out at afforestation on peaty gley soil can cause significant changes in environmental variables. Soil temperature was increased by drainage and mounding. The two practices also caused a decrease in soil moisture content. Drainage also lowered the water table depth. Peaty soils are major sinks for atmospheric CO_2 but can be major sources following disturbances. Our study shows that drainage can increase CO_2 effluxes from a peaty gley soil under temperate and maritime condition similarly to what has already been observed in deep peaty soils both in the UK and in the boreal region. Drainage increased soil CO_2 effluxes by 22–32% because improved aeration and soil temperature may have resulted in optimal conditions for microbial and autotrophic respiration. Fertilisation increased soil CO_2 effluxes by 5–9%. The increase may be attributed to increased fertility following fertilisation which may have increased root biomass and microbial activity. Mounding had no effect on soil CO_2 effluxes. Soil temperature was the main factor influencing soil CO_2 efflux in the present study.

CHAPTER 4

THE EFFECT OF SITE PREPARATION FOR AFFORESTATION ON METHANE FLUXES

4.1 INTRODUCTION

Methane is the second most important anthropogenic greenhouse gas after CO_2 (Schimel and Gulledge 1998; van den Pol-van Dasselaar et al. 1999) and contributes 20% to anthropogenic global warming (Hütsch 2001; Dalal and Allen 2008). The atmospheric concentration of CH_4 has tripled since pre-industrial times (Lelieveld et al. 1998; IPCC 2001). The increase has been attributed to anthropogenic activities, such as fuel exploitation, biomass burning, rice production, animal husbandry of ruminants, sewage treatment plants and landfill use (Lelieveld et al. 1998).

Soils are the most important biological sources and sinks for atmospheric CH_4 (Le Mer and Roger 2001; Dutaur and Verchot 2007). Globally, most CH_4 is produced by methanogenic bacteria during anaerobic decomposition processes in terrestrial wetlands (Nykänen et al. 1998; Yavitt and Williams 2000). Methane flux from soil to the atmosphere is the result of two microbial processes, methanogenesis (microbial production) and methanotrophy (microbial consumption) (Chan and Parkin 2001; Dutaur and Verchot 2007; Chen et al. 2009). Methane can also be produced in aerobic organic soils inside soil aggregates where anaerobic microsites occur (Dataur and Verchot 2007).

Globally, an estimated 600 Tg CH_4 are released to the atmosphere annually (Lelieveld et al. 1998; Smith 2005; Denman et al. 2007), with over 70% originating from biogenic sources, such as wetland soils, rice paddies and ruminants (Denman et al. 2007). In well-drained upland soils, CH_4 oxidation by methanotrophic microorganisms is the dominant process (Sundh et al. 1994; Roura-Carroll and Freeman 1999; Le Mer and Roger 2001). Methanotrophy can also occur in wetland soils in the anaerobic/aerobic interface of the soil before CH_4 is emitted into the atmosphere (Ding et al. 2003; Dutaur and Verchot

2007). Well drained soils are estimated to provide a global sink of 22–100 Tg CH_4 yr^{-1} (Smith et al. 2000; Castaldi et al. 2006; Dutaur and Verchot 2007).

Methane fluxes from soils are influenced by soil water table depth (Ding and Cai 2007), moisture content (Hargreaves and Fowler 1998), soil temperature (Saarnio et al. 1998), quantity and quality of organic substrate (Bossio et al. 1999), pH (MacDonald et al. 1997) and N levels (Singh et al. 1999). In wetlands, vascular plants play a key role in CH_4 dynamics by providing a major pathway for CH_4 fluxes (Frenzel and Rudolph 1998; Bellisario et al. 1999; Joabsson and Christensen 2001), and possibly also in periodically flooded forests (e.g., Rusch and Rennenberg 1998; Terazawa et al. 2007).

Land use changes such as cultivation of natural soils strongly reduce the strength of the soil CH_4 sink (Smith et al. 2000; Castaldi et al. 2006). For example, Smith et al. (2000) estimated that the conversion of forests to agriculture may decrease the strength of soil CH_4 sink by up to 60%. It is known that the addition of N fertilisers to soils can enhance CH_4 emission (Ball et al. 1997; Bodelier and Laanbroek 2004). The effect of N on CH_4 fluxes has been attributed to NH_4^+ which inhibits CH_4 oxidising bacteria (King and Schnell 1994; Gulledge et al. 1997; Bodelier et al. 2000). The effect of land use and soil N on atmospheric CH_4 fluxes may persist for several years after the practices were carried out (Sitaula et al. 1995), especially in organic soils (Maljanen et al. 2001a).

Large areas of boreal and temperate peatlands across the world have been drained and developed for forestry (Laine et al. 1995b). The productivity of forest stands established on peaty gley soils in the UK is increased through drainage, mounding and fertilisation. Substantial changes in the physical environment of peaty gley soils may occur after drainage and mounding including fluctuations in soil temperature (e.g., Davidson et al. 1998; Londo et al. 1999) and soil moisture content (e.g., Cortina and Vallejo 1994). Potential reductions in CH_4 emissions in peaty gley soils after drainage and mounding may arise from improved soil aeration (Hillman 1992) and increased peat temperature (Kirschbaum 1995). These and other changes (Zerva and Mencuccini 2005a) caused by drainage and mounding may increase aeration and reduce methanogenesis, thus enhance

CH_4 oxidation in the aerobic part of the soil by methane oxidising microbes (Whalen and Reeburgh 1990; van den Pol-van Dasselaar et al. 1998).

Saturated soils in the boreal and temperate region are important sources of CH_4 (Nykänen et al. 1998, Huttunen et al. 2003a; Laine et al. 2007). Several studies demonstrated that drainage and afforestation reduces CH_4 fluxes of peatland soils (Martikainen et al. 1995; Minkkinen et al. 2002; Von Arnold et al. 2005). Much research on CH_4 emissions has been conducted in peatland soils in the boreal and continental climatic zones (e.g., Bubier et al. 1993; Huttunen et al. 2003a). Studies on the effect of site preparation carried out prior to afforestation on CH_4 fluxes of peaty gleys soils under UK conditions are lacking. We hypothesised that site preparation practises would alter CH_4 emissions from grassland on peaty gley soils. The objectives of the experiment described here were to (i) assess the effects of drainage; mounding and fertilisation on CH_4 fluxes of peaty gley soil (ii) investigate the relationship between CH_4 fluxes and environmental variables.

4.2 MATERIALS AND METHODS

4.2.1 Study site description

The study site was described in Chapter 1. Briefly, the study site was established on unimproved grassland between two second rotation Sitka spruce stands at Harwood Forest. Harwood Forest is located in northeast England (55° 10’ N, 2° 3’W).

4.2.2 Experimental design and site preparation

The description of the experiment is given in Chapter 2. The experiment has a full factorial split–plot design with six plots measuring 30 × 8 m established in May 2006. Three plots were selected at random and were mechanically drained by cutting open ditches placed 1.5 m from plot edges and excavated to a depth of 65 to 70 cm. Mounds were made by excavating the soil and turning it upside down adjacent to the dug pit (depth 30–40 cm, width 40 cm), thus burying the litter layer and organic horizons of the original soil beneath the mineral layer of mounds.

4.2.3 Measurement and analysis of soil CH_4 emissions

Methane flux was measured 31 times commencing on 17 June 2006 to 7 May 2008 as described in Chapter 3. Air samples to determine CH_4 fluxes were collected after every fourth day in June 2006, weekly in July, bi-weekly between August and September 2006 and finally at approximately monthly intervals from October 2006 to May 2008. Briefly, chambers were sealed with aluminium lids with foam rubbers on the underside and a sampling port fitted with a three-way stopcock. Air samples were collected from the headspace of chambers with 60 ml polypropylene syringes and transferred into gas-tight bags (Cali-5-bond, Calibrated Instruments Inc. USA). Each measurement cycle lasted for 30–40 minutes and a linearity check showed that linear interpolation of two points taken at the start and at the end of the closure gives a good approximation of the true flux (Zerva and Mencuccini 2005, unpubl. data). Linear accumulation or depletion of CO_2 was also checked every two months during the first year of the present study. For the vast majority of the checks conducted, linearity was ensured by linear regression coefficients higher than $R^2=0.99$. Ambient air samples taken randomly at the height of chambers gave the initial concentration of CH_4. The amount of CH_4 in air samples was determined on a Hewlett Packard 5890 GC (Hewlett Packard Ltd, Stockport, Cheshire, UK) Gas Chromatograph (GC) equipped with a flame ionisation detector (FID) and a digital integrator. External standards of 1, 3 and 10 μmol mol^{-1} CH_4 were used for calibration.

4.2.4 Calculation of soil CH_4 emissions

Methane flux rates (mg m^{-2} d^{-1}) were calculated by the following equation (Zerva 2004):

$$F = d \times \frac{V}{A} \times \frac{(C_t - C_0)}{t}$$

F = CH_4 flux (mg m^{-2} d^{-1})

d (mg m^{-3}) = gas density calculated assuming that a mole of gas (16 g CH_4) occupy 22.4 x 10^{-3} m^3 of volume at 273 K.

V = volume of the chamber (m^3)

A = area of the chamber (m^2)

C_t = concentration of CH_4 (µmol mol^{-1}) in the chamber after closure time

C_0 = initial concentration of CH_4 (µm mol^{-1})

t = closure time of chamber in minutes.

4.2.5 Statistical analysis

All data were checked for normality and log-transformed when required. Analyses were carried out both on averaged monthly fluxes as well as on seasonal and yearly totals. The general linear model (GLM) was used for analysis of variance. The general linear model tested for effects of three main factors (drainage, mounding and fertilisation) entered as fixed factors and plot entered as random factor nested within drainage. In case of monthly measurements, month was also entered as a repeated measures factor. The initial GLM included all possible second and third-order interactions. If interactions were not found to be significant, they were excluded and the model was run again without them to confirm the significance of the main factors. In case of significant interactions, the dataset was split and separate analyses were run for each combination. All analyses were run in Minitab 15 using the GLM procedure and the significance level was set at 0.05. Tukey's pairwise comparison test was applied to determine significant differences between treated and untreated plots/subplots. For all analyses, values from individual chambers were averaged within each subplot. For the mounded subplots, weighted averaging was done by weighing each flux by the respective area covered by mound hills, mound hollows and undisturbed ground.

4.3 RESULTS

4.3.1 Effect of site preparation on soil CH_4 emissions

Methane emissions were affected by all three practices (Table 4.1). Plots differed significantly in CH_4 emissions over the two years of study (P=0.0001). There were no significant interactions between the practices, except when the factors month and plot were also included (Table 4.1).

Table 4.1 Summary of the general linear model of soil CH_4 emissions over the two years of study (2006–2008).

Sources of variation	F	P
Drainage	31.37	**0.005**
Plot	7.53	**<0.0001**
Mounding	22.30	**<0.0001**
Fertilisation	29.98	**<0.0001**
Month	14.29	**<0.0001**
Drainage×Fertilisation	3.13	0.13
Drainage×Mounding	3.69	0.13
Mounding×fertilisation	0.07	0.80
Drainage×Month	3.23	0.13
Mounding×Month	3.23	**<0.0001**
Fertilisation×Month	3.41	**<0.0001**
Mounding×Plot	0.69	0.64
Fertilisation×Plot	0.08	0.99
Month×Plot	3.41	**<0.0001**
Drainage×Fertilisation×Mounding	0.01	0.95
Drainage×Mounding×Month	2.45	**0.002**
Drainage×Fertilisation×Month	1.30	0.19
Mounding×Fertilisation×Month	2.20	**0.005**
Mounding×Fertilisation×Plot	5.74	**<0.0001**
Mounding×Month×Plot	1.01	0.47
Fertilisation×Month×Plot	0.90	0.69
Drainage×Mounding×Fertilisation×Month	1.26	0.22

P values in bold are statistically significant ($P < 0.05$).

Table 4.2. Average daily CH_4 emissions (mg m^{-2} d^{-1}) for all treatments (June 2006–May 2007, and June 2007–May 2008).

			Sampling year			
Treatment	2006–07	n	2007–08	n	2006–08	N
Drained	**2.48±0.15a**	20	**0.89±0.12a**	11	**1.72±0.11a**	31
Undrained	**5.80±0.36b**	20	**3.71±0.25b**	11	**4.80±0.23b**	31
Mounded	**4.74±0.34a**	20	**2.81±0.26a**	11	**3.81±0.22a**	31
Unmounded	**3.55±0.26b**	20	**1.79±0.19b**	11	**2.71±0.17b**	31
Fertilised	**5.08±0.34a**	20	2.53±0.26a	11	**3.86±0.23a**	31
Unfertilised	**3.20±0.24b**	20	2.07±0.20a	11	**2.66±0.16b**	31

Values in bold followed by different letters indicate a significant difference between the drained vs. undrained, mounded vs. unmounded and fertilised vs. unfertilised treatment ($P<0.05$). The "±" indicate the standard error of the mean.

Table 4.3. Mean seasonal CH_4 flux (mg m^{-2} d^{-1}) by treatment and season.

				CH_4 flux (mg m^{-2} d^{-1})				
Treatment	June–Aug		Sept–Nov		Dec–Feb		March–May	
2006–2007	Summer	n	Autumn	n	Winter	n	Spring	n
Drained	**2.88±0.42a**	10	**2.39±0.24a**	5	**1.71±0.15a**	3	**2.94±0.30a**	3
Undrained	**7.34±0.85b**	10	**5.66±0.50b**	5	**3.60±0.69b**	3	**6.60±0.63b**	3
Mounded	5.39±0.75a	10	**4.78±0.51a**	5	**3.37±0.70a**	3	5.40±0.72a	3
Unmounded	4.83±0.79a	10	**3.27±0.42b**	5	**1.94±0.18b**	3	4.14±0.36a	3
Fertilised	**6.93±0.77a**	10	**4.94±0.50a**	5	3.19±0.70a	3	5.27±0.62a	3
Unfertilised	**3.29±0.64b**	10	**3.11±0.40b**	5	2.12±0.20a	3	4.28±0.53a	3
2007–2008								
Drained	–0.08±0.48a	10	**1.66±0.23a**	5	**1.16±0.15a**	3	**0.48±0.14a**	
Undrained	2.98±0.85a	10	**3.96±0.39b**	5	**4.45±0.53b**	3	**3.18±0.53b**	3
Mounded	1.82±1.01a	10	**3.25±0.37a**	5	3.17±0.52a	3	**2.82±0.57a**	3
Unmounded	1.07±0.56a	10	**2.37±0.37b**	5	2.26±0.45a	3	**0.84±0.17b**	3
Fertilised	1.36±0.77a	10	**3.36±0.41a**	5	3.05±0.50a	3	1.95±0.54a	3
Unfertilised	1.53±0.87a	10	**2.26±0.31b**	5	2.75±0.47a	3	1.71±0.33a	3

Values in bold followed by different letters indicate a significant difference between the drained vs. undrained, mounded vs. unmounded and fertilised vs. unfertilised treatment (P<0.05). The "±" indicate standard error of the mean.

4.3.2 Effects of drainage on soil CH_4 emissions

Methane emissions in the drained plots were significantly lower than in the undrained plots (P=0.01, Table 4.1). The average soil CH_4 flux over the whole study period was 1.72±0.11 and 4.80±0.23 mg m^{-2} d^{-1} in the drained and undrained plots, respectively (Table 4.2). Drainage decreased CH_4 emissions in 2006–07 and in 2007–08 (all P=0.01). The two years of study differed significantly in CH_4 fluxes (P=0.0001), with 2006–07 having more emissions than 2007–08. Averaged monthly emissions are shown in Fig. 4.1a. The month to month variability in CH_4 emissions was highly significant (P=0.0001, Table 4.1). Average monthly CH_4 emissions varied from a net sink (–0.31±0.25 mg m^{-2} d^{-1}) to a net source (5.17±0.52 mg m^{-2} d^{-1}) in the drained plots, whereas in the undrained plots fluxes varied from 1.66±0.50 to 11.06±1.21 mg m^{-2} d^{-1}. Methane emissions in the drained and undrained plots showed a seasonal trend (Fig. 4.1a). Methane fluxes were significantly decreased by drainage in the summer of 2006 (P=0.04) but not in 2007(P=0.5, Table 4.3). The two summers differed significantly in fluxes (P=0.0001) with more fluxes in the summer of 2006–07 than in 2007–08. There was consumption of CH_4 in the drained plots during the summer of 2007–08. Drainage decreased CH_4 fluxes in the autumn of 2006 (P=0.03) and 2007 (P=0.01). More CH_4 was emitted in the autumn of 2006 than in 2007 (P=0.0001). Methane emitted from drained plots was significantly lower than in the undrained plots in the winter of 2006–07 and 2007–08 (all P=0.01). Drainage decreased CH_4 fluxes in the spring of 2007 (P=0.001) and 2008 (P=0.01). The two spring seasons differed significantly in fluxes (P=0.0001) with more CH_4 emitted in the spring of 2007 than in 2008. Annual CH_4 emissions are summarised in Fig. 4.2. Annual CH_4 fluxes in the drained plots varied from 3.23±0.43 to 9.06±0.55 kg ha^{-1} yr^{-1} and were on average 6.27±0.39 kg ha^{-1} yr^{-1}. Annual CH_4 fluxes in the undrained plots varied from 13.53±0.92 to 21.18±1.38 kg ha^{-1} yr^{-1} and were on average 17.52±0.84 kg ha^{-1} yr^{-1}.

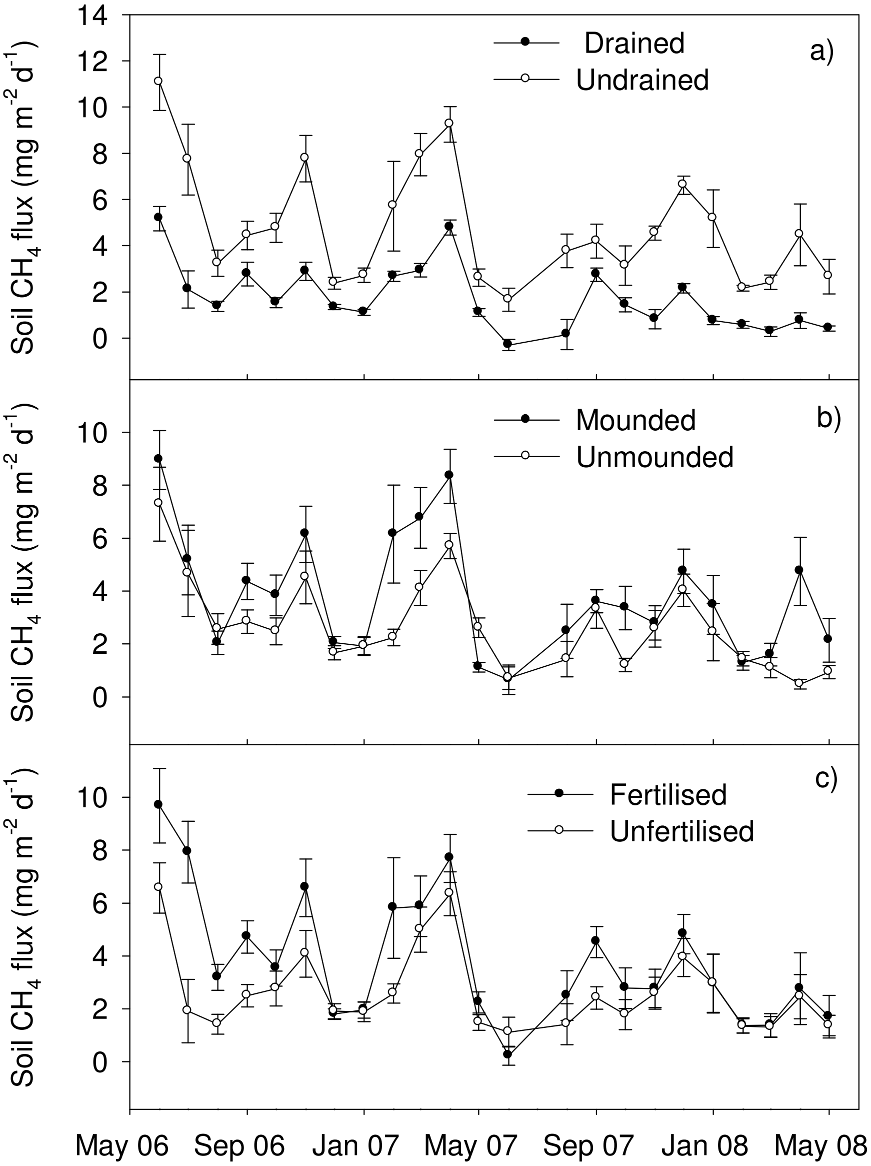

Fig. 4.1 Averaged monthly CH_4 fluxes in (a) the drained vs. undrained (b) mounded vs. unmounded and (c) fertilised vs. unfertilised treatment (n=31). Dots represent means of all chambers at each month and the vertical bars represent standard error of the mean.

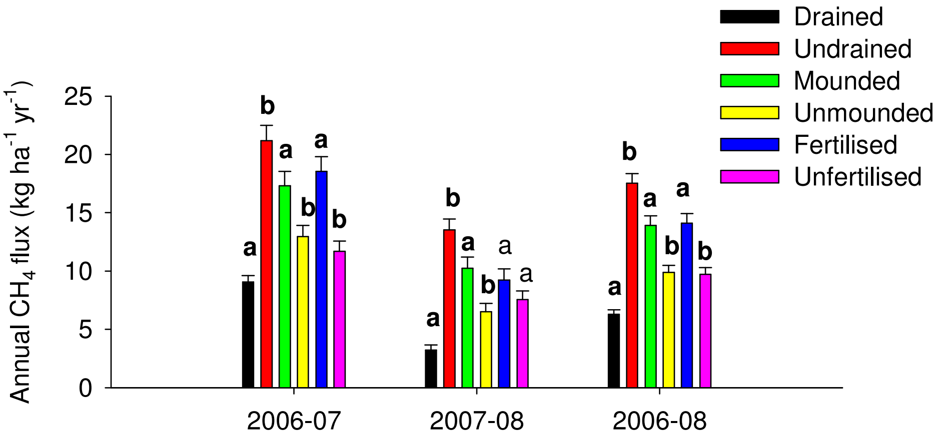

Figure 4.2 Annual CH_4 fluxes for all treatments by sampling year. The vertical bars represent standard error of mean. Different letters in bold indicate a significant difference ($P<0.05$) between the drained vs. undrained, mounded vs. unmounded and fertilised vs. unfertilised in each sampling year (n=20, 11 and 31 for 2006–07, 2007–08 and 2006–08, respectively).

4.3.3 Effects of mounding on CH_4 fluxes

Over the two years of study, CH_4 emissions were significantly affected by mounding. Mounding significantly increased CH_4 fluxes ($P=0.0001$, Table 4.1). The average soil CH_4 flux over the two years of study was 3.81±0.22 and 2.71±0.17 mg m^{-2} d^{-1}, in the mounded and unmounded treatment (Table 4.2). Mounding increased soil CH_4 fluxes in 2006–07 ($P=0.0001$) and 2007–08 ($P=0.0001$). More CH_4 was emitted in 2006–07 than in 2007–08 ($P=0.0001$). Averaged monthly soil CH_4 fluxes are shown in Fig. 4.1b. Soil CH_4 fluxes varied from month to month ($P=0.0001$). Monthly soil CH_4 fluxes ranged from 0.64±0.56 to 8.95±1.11 and 0.47±0.18 to 7.28±1.40 mg m^{-2} d^{-1} in the mounded and unmounded treatment subplots.

Methane fluxes in the mounded and unmounded treatment showed a seasonal trend (Fig. 4.1b). Mounding had no significant effect on CH_4 emissions in the summer of 2006 and 2007 (all $P=0.3$). However, more CH_4 was emitted in the summer of 2006 than in 2007 ($P=0.001$, Table 4.3). Mounding increased CH_4 fluxes in the autumn of 2006 ($P=0.0001$) and 2007 ($P=0.03$), respectively. More CH_4 was emitted in the autumn of 2006 than in 2007 ($P=0.0001$). Methane emissions were significantly increased by mounding in the winter of 2006–07 ($P=0.02$) but not in the winter of

2007–08 (P=0.3). Mounding increased fluxes in the spring of 2008 (P=0.0001) but not in the spring of 2007 (P=0.3). Annual CH_4 fluxes in the mounded treatment varied from 10.24±0.95 to 17.29±1.25 kg ha^{-1} yr^{-1} and were on average 13.92±0.82 kg ha^{-1} yr^{-1} (Fig. 4.2). Annual soil CH_4 fluxes in the unmounded treatment varied from 6.52±0.70 to 12.95±0.94 kg ha^{-1} yr^{-1} and were on average 9.87±0.62 kg ha^{-1} yr^{-1}.

Methane emitted from the mounded subplots was significant affected by the position of chambers (P=0.0001, Table 4.4). The average CH_4 emission over the two years of study was 0.67±0.17, 3.82±0.40 and 18.51±1.39 mg m^{-2} d^{-1} from the mounds, undisturbed ground and hollows, respectively. The consumption of methane occasionally took place inside the dry hollows, the top of mounds and undisturbed ground. Soil CH_4 emitted from the hollows was more variable than fluxes from the undisturbed ground and mounds (Fig. 4.3). Methane emitted from the mounded subplots was affected by fertilisation (P=0.001) and drainage (P=0.05, Table 4.4). There was a significant interaction between drainage and fertilisation (P=0.0001), drainage and the position of chambers (P=0.0001), fertilisation and the position of chambers (P=0.0001) and drainage, fertilisation and the position of chambers (P=0.03, Table 4.4). Annual CH_4 fluxes varied from 63.84±8.58 to 69.69±6.35, 2.19±0.62 to 2.60±0.94 and 5.91±0.80 to 18.62±2.19 kg ha^{-1} yr^{-1} in the hollows, mounds and undisturbed ground (Fig. 4.4). On average annual soil CH_4 fluxes were 67.55±5.10, 2.45±0.64 and 13.96±1.46 kg ha^{-1} yr^{-1} in the hollows, mounds and undisturbed ground.

Table 4.4. Summary of the general linear model of soil CH_4 emissions in the mounded plots. The general linear model tested the significance of drainage, fertilisation and position (mounds, hollow, undisturbed ground) across sampling dates (plot nested within drainage and date enter as repeated measure).

Parameter	*F*	*P*
Drainage	7.72	**0.05**
Plot	21.43	**<0.0001**
Fertilisation	15.94	**<0.0001**
Drainage×Fertilisation	0.92	0.339
Position	470.43	**<0.0001**
Drainage×Position	19.40	**<0.0001**
Fertilisation×Position	7.71	**<0.0001**
Drainage×Fertilisation×Position	3.56	**0.029**
Sampling date	24.15	**<0.0001**
Drainage×Sampling date	1.70	**0.013**
Position×Sampling date	9.13	**<0.001**
Fertilisation×Sampling date	1.49	**0.048**
Drainage×Position×Sampling date	2.04	**<0.0001**
Drainage×Fertilisation×Sampling date	0.66	0.916
Fertilisation×Position× Sampling date	1.40	**0.030**
Drainage×Fertilisation×Position×Sampling date	0.64	0.984

P values in bold are statistically significant ($P < 0.05$).

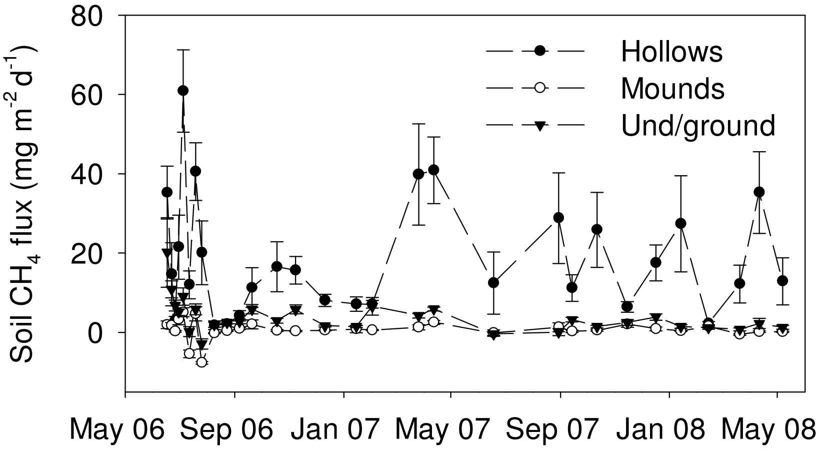

Figure 4.3. Methane fluxes from the mounded subplots (showing mean values and standard errors for fluxes from the hollows, mounds and undisturbed ground (Und/ground) for each all sampling dates (n=31).

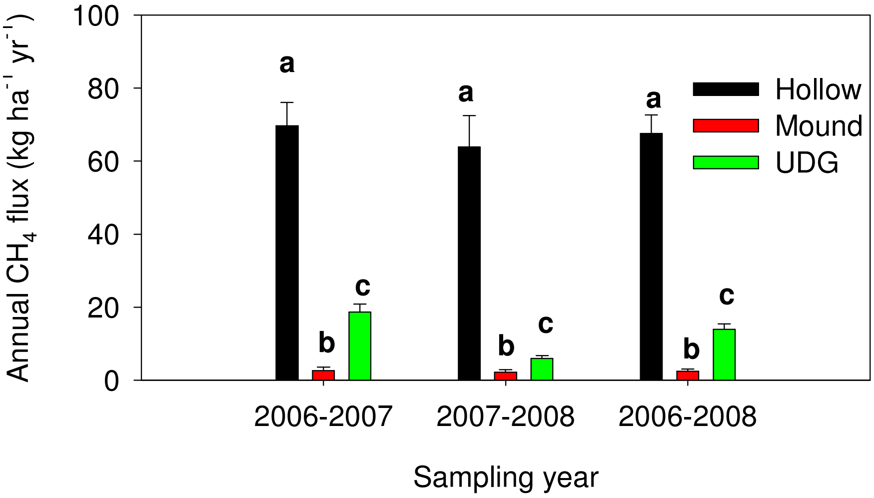

Figure 4.4. Annual CH_4 fluxes (kg CH_4 ha^{-1} yr^{-1}) from the hollows, on the top of mounds and UDG (undisturbed ground) in the mounded subplots. Different letters in bold indicate a significant difference in CH_4 fluxes ($P=0.05$) between the hollows, mounds and undisturbed ground (n=20, 11 and 31 for 2006–07, 2007–08 and 2006–08).

4.3.4 Effect of fertilisation on CH_4 emissions

Methane emissions were significantly increased by fertilisation ($P=0.0001$, Table 4.1). The average daily soil CH_4 emission rate over the two years of study was 3.86±0.23 and 2.66±0.16 mg m^{-2} d^{-1} in the fertilised and unfertilised subplots (Table 4.2). Fertilisation increased CH_4 emissions in 2006–07 ($P=0.0001$) but not in 2007–08 ($P=0.2$, Table 4.2). Methane emissions were significantly higher in 2006–07 than in 2007–08 ($P=0.0001$). Methane emissions from the fertilised and unfertilised subplots varied from month to month ($P=0.0001$, Table 4.1). Averaged monthly soil CH_4 emissions in the fertilised treatment varied from 0.23±0.36 to 9.67±1.41 mg m^{-2} d^{-1}, while in their unfertilised counterparts emissions varied from 1.12±0.56 to 6.56±0.95 mg m^{-2} d^{-1} (Fig 4.1c).

Methane emissions exhibited a seasonal trend (Fig. 4.1c). Averaged seasonal CH_4 fluxes from the summer of 2006 to the spring of 2008 are shown in Table 4.3. Fertilisation increased fluxes in the summer of 2006 (P=0.0001) but not in 2007 (P=0.7). The two summers differed significantly in fluxes with more CH_4 emitted in 2006 than in 2007 (P=0.0001). Fertilisation increased CH_4 fluxes in the autumn of 2006 (P<0.0001) and in 2007 (P=0.03). Fertilisation did not affect soil CH_4 emissions in the winter of 2006 (P=0.2) or 2007 (P=0.6) as well the spring of 2007 (P=0.2) or 2008 (P=0.9). Annual soil CH_4 fluxes in the fertilised treatment varied from 9.22±0.94 to 18.55±1.25 kg ha^{-1} yr^{-1} and were on average 14.09±0.84 kg ha^{-1} yr^{-1} (Fig. 4.2). Annual CH_4 fluxes in the unfertilised treatment varied from 7.54±0.73 to 11.68±0.88 kg ha^{-1} yr^{-1} and were on average 9.70±0.59 kg ha^{-1} yr^{-1}.

4.3.5 Relationships between CH_4 emissions and environmental variables

Methane emissions in the drainage (drained and undrained plots) and mounding (mounded and unmounded subplots) treatments were not correlated with measured environmental variables. Methane emissions from fertilised subplots were correlated with soil temperature at all depths (exponential relationship, R^2= 0.31, P=0.03, for all depths, Fig. 4.5). A weak and non significant relationship was detected between soil moisture content and CH_4 emissions in the fertilised subplots (R^2=0.14, P=0.09). Methane emitted from fertilised subplots was significantly correlated with soil water table depth (R^2=0.29, P=0.01, Fig. 4.6).

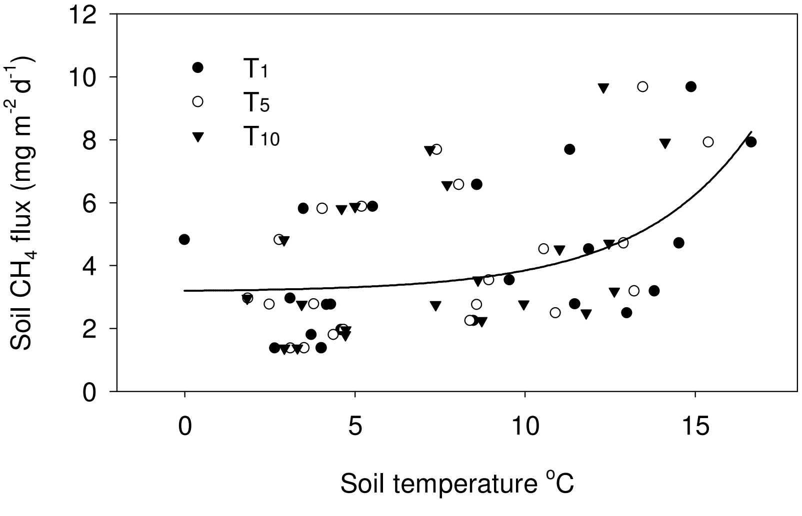

Figure. 4.5. The exponential relationship between soil CH_4 emissions and soil temperatures measured at all depths in the fertilised subplots (n=23).

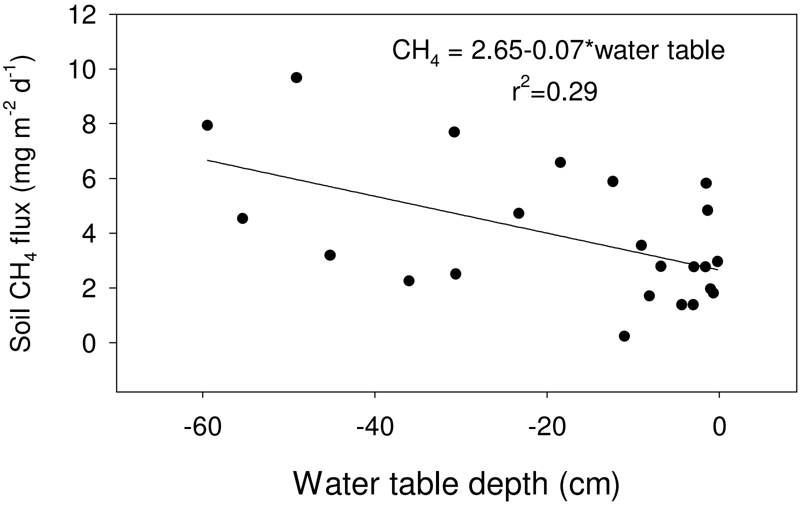

Figure 4.6. The linear relationship (P=0.01) between the CH_4 emissions and the water table depth, in the fertilised subplots (n=23).

4.4 DISCUSSION

4.4.1 Effects of drainage on CH_4 emissions

In general, drainage of peatland soils has been shown to decrease CH_4 emissions due to lower CH_4 production with increased aeration and increased oxidation of the CH_4 in the aerobic part of the soil by methane oxidising microbes (e.g., Glenn et al. 1993; Roulet and Moore 1995; Nykänen et al. 1998). The result of the present study showed that drainage decreased soil CH_4 fluxes. Annual CH_4 fluxes in the drained plots (3.23 to 6.27 ha^{-1} yr^{-1}) were higher than fluxes of 1.2–2.6 kg CH_4 ha^{-1} yr^{-1} reported for an unplanted grassland site at Harwood Forest not far from the present study site (Ball et al. 2007) and 1–2 kg CH_4 ha^{-1} yr^{-1} to uptake rates of about 1.8 kg ha^{-1} yr^{-1} measured in a drained upland bog used for forestry in Scotland (MacDonald et al. 1996; 1997). The difference could due to the variability in the water table depth between this site and other sites where the other experiments were conducted.

Annual soil CH_4 emissions in the undrained plots varied from 13.53±0.92 to 21.18±1.38 kg ha^{-1} yr^{-1} and are in the lower range of 1.10–110 kg CH_4 ha^{-1} yr^{-1}

reported for wet forest soils in Canada (Castro et al. 1993; Yavitt et al. 1995) and 7.9–204 kg CH_4 ha^{-1} yr^{-1} reported for fertilised grasslands on wet peaty soils in the Netherlands (Van Den Pol-van Dasselaar et al. 1999). The difference could be due to variations in water table depth between this site and areas where the other experiments were conducted. The soils in the other areas are deeper and may be waterlogged throughout the year, whereas peaty gley soils in the present study are seasonally waterlogged and their water table depth can drop significantly during the summer. The drop in water table depth may increase aeration and CH_4 oxidation decreasing CH_4 production and a result of reduced methanogenesis

Drainage decreased CH_4 fluxes by 57 to 76%. This finding agrees with results of previous studies conducted under field and laboratory conditions in temperate and boreal peatland sites (Freeman et al. 1993; Roulet et al. 1993; Martikainen et al. 1995; Minkkinen et al. 2002). For example Minkkinen et al (2002) reported that drainage decreased CH_4 by 50% in peaty soils in the boreal region. They attributed low fluxes from drained sites to decreased methanogenesis with increased aeration and increased oxidation of CH_4 in the aerobic part of the soil by methane oxidising microbes. Freeman et al. (1993) collected intact soil cores from a Welsh peatland to study the potential effect of climate change on CH_4 fluxes. They manipulated the water table depth within intact peat monoliths in the laboratory, thereby simulating water table lowering imposed by climate change. They found that the soil water table lowering treatment decreased soil moisture content, increased soil temperature and decreased CH_4 emissions by up to 80%. In natural organic soils, CH_4 is produced by methanogens in water saturated soil profiles where the old and fresh organic matter is subjected to anaerobic decomposition, whereas after drainage, the organic substrate is oxidised in the aerated layers before reaching saturated profiles and thus decrease the organic substrate for CH_4 production (e.g., Nykänen et al. 1998).

In the present study, CH_4 emissions in the undrained and drained plots were 35 and 65% lower in 2007–08 than in 2006–07. The summer and autumn of 2006 were warmer than in 2007 and this may have reduced fluxes in 2007–08. The decrease may also have been caused by lack of variation in CH_4 emissions or infrequent sampling in 2007–08. In 2007–08, CH_4 fluxes were measured 11 times at monthly interval, compared to 20 times in 2006–07. It is possible that episodic emissions caused by periodic rains (e.g., Kettunen et al. 1996) or rapid changes in water table depth and temperature (e.g., Windsor et al. 1992; Kettunen et al. 1996) were missed due to infrequent sampling in 2007–08.

There was consumption of CH_4 in the drained plots in the summer of 2007, which could be attributed to increased aeration due to lowering of the water which may have created a thick aerobic soil layer where CH_4 oxidation exceeded CH_4 produced in the anaerobic layer beneath (e.g., Glen et al. 1993; Nykänen et al. 1998; van den Pol-van Dasselaar et al. 1998). Several studies have shown that well-drained organic forest, natural forest and grassland soils can be a net sink for atmospheric CH_4 (Steudler et al. 1989; Mosier et al. 1991; 1997; Castro et al. 1993; Lessard et al. 1994; Ambus and Christensen 1995; Nykänen et al. 1995; Dobbie et al. 1996; Priemé and Christensen 1997; Maljanen et al. 2003a). The observed net CH_4 uptake (0.08 mg m^{-2} d^{-1}) is closed to 0–1 mg m^{-2} d^{-1} reported for temperate grassland soils (Mosier et al. 1991; Dobbie et al. 1996).

4.4.2 Effects of mounding on CH_4 emissions

Mounding buries the litter and the organic layers beneath the mineral soil layers (Smolander et al. 2000; Saari et al. 2004) and may alter environmental variables controlling soil CH_4 fluxes. To our knowledge few studies have attempted to evaluate the effect of mounding on soil CH_4 fluxes. Mounding increased soil CH_4 emissions by 34 to 59%. The effect of mounding on CH_4 emissions was more pronounced in 2006–07 than in 2007–08. A study conducted in south-eastern Finland found that mounding decreased CH_4 emissions by 33% in the first year, but increased them in subsequent years (Saari et al. 2004). They suggested that the initial decrease in CH_4 fluxes after mounding occurred as a result of CH_4 oxidation in the mineral soil on the top of mounds. They also suggested that the double organic horizons may have reduced diffusion of atmospheric CH_4 into the mineral soil beneath mounds and inhibited consumption in that layer during subsequent years. However, Saari et al. (2004) did not evaluate CH_4 fluxes from hollows where the soil used to make mounds was excavated from. High CH_4 fluxes observed in the mounded subplots in the present study can be attributed to large emissions measured from hollows which were periodically covered with stagnant water inhabited by green algae (e.g., Schiller and Hastie 1996).

Methane emissions in the mounded subplots were higher in the hollows than mounds and undisturbed ground. This finding agrees with results of previous studies which reported large fluxes from anaerobic environments, such as peatlands (Crill et al. 1988; Roulet et al. 1992; Lloyd et al. 1998; Yavitt and Williams 2000; Laine et al. 2007), rice paddies (Hou et al. 2007) and in periodically flooded forests (e.g., Rusch and Rennenberg 1998; Terazawa et al. 2007). The presence of green algae in stagnant water in the hollows may have provided a substrate for CH_4 production by methanogens (e.g., Schiller and Hastie 1996). Large CH_4 fluxes have also been reported in studies conducted in forest drainage ditches in boreal and peatland sites (Roulet and Moore 1995; Schiller and Hastie 1996; Minkkinen et al. 1997; Von Arnold et al. 2005; Minkkinen and Laine 2006) because ditch bottoms are often colonised by vascular plants, which decreases the flow of water, especially in bogs (Minkkinen et al. 2008). Vascular plants may enhance CH_4 emissions in wetlands by providing a substrate for methanogens and a major pathway for emissions (Frenzel and Rudolph 1998; Bellisario et al. 1999; Joabsson and Christensen 2001). The soil bulk density in the mounded subplots was increased by the excavator used for mounding. It is possible that soil compaction caused anaerobic conditions in the mounded subplots may have increased CH_4 production and fluxes (e.g., Ruser et al. 1998; Smith et al. 2000; Flessa et al. 2002). However, there was no relationship between soil CH_4 fluxes and bulk density.

Annual fluxes (67.55±5.10 kg CH_4 ha^{-1} yr^{-1}) observed in the hollows are similar to emissions (62 kg CH_4 ha^{-1} yr^{-1}) reported for hummocks and hollows in an Irish lowland blanket bog (Laine et al. 2007). Annual CH_4 observed in the mounds (1.95±0.71 to 2.60±0.94 kg CH_4 ha^{-1} yr^{-1}) was within the range (0.2–2.8 kg CH_4 ha^{-1} yr^{-1}) reported for an unplanted grassland site at Harwood Forest (Ball et al. 2007). Occasionally, methane was consumed in the hollows after the water had evaporated, which was also observed on the top of mounds. It seems that dry conditions in both the hollows and mounds increased aeration and increased oxidation of CH_4 which exceeded CH_4 production (e.g., Sundh et al. 1994, 1995; Nykänen et al. 1998; Maljanen et al. 2003a; Melling et al. 2005) due to reduced methanogenesis.

4.4.3 Effects of fertilisation on CH_4 emissions

Fertilisation increased soil CH_4 emissions by 20–59%. This finding agrees with results of studies conducted elsewhere which found that N increase CH_4 fluxes (e.g., Steudler et al. 1989; Castro et al. 1994; King and Schnell 1994, 1998; Hutsch 1996; Sitaula et al. 1995; Gulledge et al. 1997; Saari et al. 1997; Wang and Ineson 2003; Bodelier and Laanbroek 2004; Suwanwaree and Robertson 2005). The effect of N fertiliser on CH_4 fluxes has been attributed to NH_4^+ ions that inhibit CH_4 oxidation by competitive inhibition of the enzyme mono-oxygenase and by decreasing pH when NH_4^+ is applied to soil (Hutch 1998). In addition to NH_4^+, the activity of CH_4 oxidising bacteria is also reduced by nitrite (NO_2^-) and nitrate (NO_3^-) ions (Reay and Nedwell 2004). Increased CH_4 fluxes observed in the fertilised treatment in this study may also have been caused by the soil environment which was suitable for CH_4 formation and emission, especially in the summer/autumn of 2006. Suwanwaree and Robertson (2005) studied soil CH_4 fluxes in a forest in southwest Michigan, USA and found that adding N at 100 kg N ha^{-1} increased CH_4 emissions by 60%. Steudler et al. (1989) found that adding NH_4NO_3 at a rate of 120 kg N ha^{-1} yr^{-1} increased CH_4 fluxes by 33% in temperate forest soils. Sitaula et al (1995) demonstrated that adding N at a rate of 90 kg N ha^{-1} yr^{-1} over a period of two years increased CH_4 emissions by 38% in a Scots pine forest in Norway. Powlson et al. (1997) found that CH_4 emissions from soils that had received ammonium nitrate at 144 kg ha^{-1} yr^{-1} for over 150 years increased by 50%. Castro et al. (1994) found that soil CH_4 emitted from 26 year old fertilised slash pine (*Pinus elliottii* var. *elliottii* Englem.) plots in Florida was 5 to 20 times higher than in their unfertilised counterparts. In contrast, Whalen and Reeburgh (2000), Bradford et al. (2001) and Steinkamp et al. (2001) found that N fertiliser had no effect on CH_4 fluxes in forest soils.

There were no significant differences in fluxes between the fertilised and unfertilised treatment in 2007–08. The decline in CH_4 fluxes in the fertilised treatment in 2007–08 may indicate that N uptake by plants, losses through leaching and as gaseous

N_2O and N_2 depleted the pool of N applied in this site in 2007–08. Robertson et al. (2000) found that soils that had been out of arable agriculture for less than 10 years emitted less CH_4 than those still fertilised and used for arable agriculture. They suggested that the recovery starts soon after cultivation and fertilisation has ceased, but may persist for decades to centuries (e.g., Ojima et al. 1993; Flessa et al. 1995; Dobbie and Smith 1996; Kruse and Iverson 1995; Priemé et al. 1997; Suwanwaree and Robertson 2005).

4.4.4 Effects of environmental factors on CH_4 emissions

The lack of a relationship between CH_4 emissions in the drained and undrained plots and soil water table is contrary to results of several studies conducted in natural peatlands which collectively found that soil water table was the main controller of CH_4 emissions (e.g., Liblik et al. 1997; Nykänen et al. 1998; Frenzel and Karofeld 2000; Strom and Christensen 2007). No relationship between CH_4 fluxes and soil temperature was found for the drainage and mounding treatments. Ball et al (2007) also found no relationship between soil temperature and CH_4 fluxes in a forest close by. Soil CH_4 flux in this site was not correlated with soil water content in either of the treatments. A lack of a significant relationship between water content and soil CH_4 fluxes has been reported in other studies (e.g. Sjögersten and Wookey 2002; Maljanen et al. 2003a; Zerva and Mencuccini 2005a). The lack of a relationship between environmental variables and CH_4 emitted from the drainage and mounding treatment may suggest that substrate availability was probably more important in controlling emissions than abiotic factors at this site.

Methane emissions in the fertilised subplots were exponentially related to soil temperature at all depth (T_1, T_5 and T_{10}) and were high when the soil temperature was above 12°C. Substrate availability and the effect of temperature may have induced changes in microbial activity and increased CH_4 production and emission in the fertilised subplots. The relationship between soil CH_4 emissions and soil temperature has been commonly reported in field and laboratory studies (e.g., Dunfield et al. 1993; Castro et al. 1995; Komulainen et al. 1998; Zerva and Mencuccini 2005a). Dunfield et al. (1993) investigated the effect of soil temperature on CH_4 fluxes from a peaty soil from Ontario and Quebec in the laboratory. They found that the production rate was controlled by soil temperature with optimal values in the 20–25°C temperature range and extremely lower production rates in the 0 to 15°C temperature range. Castro et al. (1995) measured soil CH_4 consumption at Harvard Forest and found that it was strongly controlled by soil temperature between −5 and 10°C and became independent of soil temperature between 10 and 20˚C.

4.5 CONCLUSION

Our study shows that drainage carried out at afforestation can decrease CH_4 fluxes from a peaty gley soil under the temperate and maritime conditions prevalent in the UK, similarly to what has already been found in deeper peaty soils both in the UK and elsewhere. Methane fluxes in the drainage and mounding treatment did not respond to abiotic factors, suggesting that the availability of organic substrate was more important than abiotic factors. Soil CH_4 efflux in the fertilised treatment was exponentially related with soil temperature and water table depth indicating substrate availability and high temperate induced soil microbial activity.

CHAPTER 5

NITROUS OXIDE EMISSIONS FROM A PEATY GLEY SOIL: EFFECT OF DRAINAGE, MOUNDING AND FERTILISATION

5.1 INTRODUCTION

Nitrous oxide is an important trace gas that contributes to the depletion of the stratospheric ozone (Crutzen and Enhalt 1997; Ginting and Eghball 2005). Nitrous oxide is estimated to contribute 6% to global warming (Bowman 1998; Dalal and Allen 2008). Its atmospheric concentration has increased by 17% since the start of the industrial revolution (Rockmann et al. 2003; Smith and Conen 2004) and is currently increasing at a rate of 0.2 to 0.3% per year (Flessa et al. 1995; Mosier et al. 1998).

Soils are the most important source of atmospheric N_2O contributing about 57% (9 Tg yr^{-1}) to the total N_2O global budget (Kroeze et al. 1999). N_2O emitted from soils is produced by microbial processes of nitrification and denitrification (Machefert et al. 2002; Koponen et al. 2006). Increasing N inputs in agricultural and natural soils have increased N_2O emissions in the past few decades (Brumme and Beese 1992; Flessa et al. 1998). The deposition of atmospheric N onto forests and other natural soils has also contributed to increased global N_2O emissions (Gundersen et al. 1998; Skiba and Smith 2000). Agricultural soils are the main sources of anthropogenic N_2O emissions (Kasimir-Klemedtsson et al. 1997; Perez et al. 2001). Fertilised agricultural soils and grasslands are estimated to contribute 3.3 Tg N_2O–N yr^{-1} and 0.8 Tg N_2O–N yr^{-1} to the atmosphere, respectively (Stehfest and Bouwman 2006). According to Crutzen et al (2008) the agricultural contribute 4.3 –5.8 Tg N_2O–N yr^{-1}. Drained fertile soils used for agriculture in peatland sites are also sources for atmospheric N_2O (Velthof and Oenema 1995). The production and emission of N_2O is primarily controlled by the availability of mineral N, soil temperature, soil water content, and the availability of labile organic compounds (Granli and Bockman 1994). Nitrous oxide emissions from water-saturated peatland soils are insignificant (Regina et al. 1996). However, N_2O emissions from fertile soils in peatland sites generally increase after drainage (Merbach et al. 1996; Regina et al. 1996 Augustin et al. 1998b).

Site preparation for afforestation and replanting in upland Britain usually involves drainage and ploughing although in more recent years, ploughing was replaced by mounding using excavators (Ball et al. 2007; Minkkinen et al. 2008). Drainage and mounding may affect the dynamics of N_2O in peaty gley soils by changing soil temperature (Davidson et al. 1998; Kätter et al. 1998), soil moisture content (Schlentner and Van Cleve 1985; Carlyle and Than 1988) and substrate availability (Smolander et al. 1998; Ruser et al. 2006). These changes may accelerate organic matter decomposition in peaty gley soils (Zerva et al. 2005) and provide a substrate for N_2O formation and emission. Forests are commonly fertilised to increase tree growth. The addition of N compounds to increase the amount of mineral N in soils also increases the probability for gaseous N losses including the formation and emissions of N_2O (Minkkinen et al. 2008). Nitrous oxide emissions are not expected to increase in planting sites where fertilisation is restricted to phosphorus and potassium (PK) to repair nutrient balances (Minkkinen et al. 2008).

Drainage and soil cultivation as well as fertilisation have increased N_2O emissions in agricultural and forest soils in peatland sites (Kasimir-Klemedtsson et al. 1997; Flessa et al. 1998; Huttunen et al. 2003b). The effects of drainage, mounding and fertilisation on N_2O fluxes of peaty gleys soils under UK conditions have not been reported. We hypothesised that: (a) drainage, mounding and fertilisation would increase soil N_2O fluxes and (b) environmental variables would affect N_2O emissions from a peaty gley soil. The objectives of the study were to: i) measure nitrous oxide over a period of two years following drainage, mounding and fertilisation ii) investigate the relationships between N_2O and soil temperature, soil moisture content and water table depth.

5.2 MATERIALS AND METHODS

5.2.1 The study site

The study site was described in Chapter 1 and will not be repeated here.

5.2.6 Experimental design

The experimental design and site preparation were described in Chapter 2.

5.2.3 Nitrous oxide measurements and analysis

Air samples to determine N_2O were collected as described for CO_2 in Chapter 3. N_2O samples were collected 34 times over a period of 2 years from 12 June 2006 to 7 May 2008. Air samples were collected for 3 consecutive days commencing a day after fertilisation on 12 June 2006 and then every fourth day in June 2006. Samples were collected weekly in July 2006, biweekly in August and September 2006 and then monthly for the duration of the study. Measurement cycles lasted between 30 and 40 minutes and linearity checks had shown that interpolation of two points taken at the start and at the end of the closure give a good approximation of the true flux (Zerva and Mencuccini 2005, unpublished data). Linear accumulation of N_2O was also checked every two months during the first year of the present study. For the vast majority of the checks conducted, linearity was ensured by linear regression coefficients higher than $R^2=0.99$. Ambient air samples were also collected to measure the initial concentration of N_2O in the field. Air Samples were transported to the laboratory. In the laboratory, air samples were transferred from gas-tight bags to 20 ml crimp–top glass vials with rubber seals which have been evacuated repeatedly using a hand operated pump. Vials were filled with samples from bags to ensure effective flushing prior to analysis. The concentration of N_2O in samples was analysed by Agilent 6890 Series (Agilent Ltd, Stockport, Cheshire, UK) gas chromatograph system fitted with an electron capture detector (ECD) and an automated injector system. Peak integration (and autosampler control) was done with a Peak Simple system. The gas chromatograph was calibrated with N_2O standards of 1 and 10 μmol mol^{-1}.

5.2.4 Calculation of N_2O fluxes

N_2O flux rates (mg m^{-2} d^{-1}) were calculated by the following equation (Zerva, 2004):

$$F = d \times \frac{V}{A} \times \frac{(C_t - C_0)}{t}$$

F = N_2O flux (mg m^{-2} d^{-1})

d (mg m^{-3}) = gas density calculated assuming that a mole of gas (44 g N_2O) occupy 22.4 x 10^{-3} m^3 of volume at 273 K.

V = volume of the chamber (m^3)

A = area of the chamber (m^2)

C_t = concentration of N_2O (µm mol^{-1}) in the chamber after closure time

C_0 = initial concentration of N_2O (µm mol^{-1})

t = closure time of chamber in minutes.

The N_2O emission factor (the amount of N_2O emitted per year as percentage of the total N fertiliser applied) was calculated as follows; E_f= (fluxes N_2O–N kg/N applied Kg)/100%.

5.2.5 Statistical analysis

All data were checked for normality and log-transformed when required. Analyses were carried out both on averaged monthly fluxes as well as on seasonal and yearly totals. The general linear model (GLM) was used for analysis of variance. The general linear model tested for effects of three main factors (drainage, mounding and fertilisation) entered as fixed factors and plot entered as random factor nested within drainage. In case of monthly measurements, month was also entered as a repeated measures factor. The initial GLM included all possible second and third-order interactions. If interactions were not found to be significant, they were excluded and the model was run again without them to confirm the significance of the main factors. In case of significant interactions, the dataset was split and separate analyses were run for each combination. All analyses were run in Minitab 15 using the GLM procedure and the significance level was set at 0.05. Tukey's pairwise comparison test was applied to determine significant differences between treated and untreated plots/subplots. For all analyses, values from individual chambers were averaged within each subplot. For the mounded subplots, weighted averaging was done by weighing each flux by the respective area covered by mound hills, mound hollows and undisturbed ground. The relationships between N_2O fluxes and environmental factors were determined by linear regression.

5.3 RESULTS

5.3.1 The effects of site preparation on soil N_2O fluxes

The soil N_2O flux was significantly affected by fertilisation and mounding but not by drainage (Table 5.1). There were significant differences in N_2O fluxes among plots (P=0.0001, Table 5.1). There was a significant interaction between fertilisation and mounding (P=0.02).

Table 5.1 Summary of the general linear model showing the effect of drainage, fertilisation, mounding and sampling interval (month) on soil N_2O fluxes.

Sources of variation	F	P
Drainage	0.32	0.601
Plot	5.49	**<0.0001**
Mounding	137.91	**<0.0001**
Fertilisation	4.37	**0.006**
Month	23.14	**<0.0001**
Drainage×Fertilisation	0.12	0.74
Drainage×Mounding	0.02	0.89
Mounding×fertilisation	16.10	**0.02**
Drainage×Month	0.55	0.95
Mounding×Month	2.90	**<0.0001**
Fertilisation×Month	47.45	**<0.0001**
Mounding×Plot	2.90	**<0.0001**
Fertilisation×Plot	47.45	**<0.0001**
Month×Plot	1.21	0.22
Drainage×Fertilisation×Mounding	1.31	0.32
Drainage×Mounding×Month	0.85	0.66
Drainage×Fertilisation×Month	0.24	1.0
Mounding×Fertilisation×Month	2.47	**0.002**
Mounding×Fertilisation×Plot	1.28	**0.28**
Mounding×Month×Plot	0.88	0.73
Fertilisation×Month×Plot	2.12	**<0.0001**
Drainage×Mounding×Fertilisation×Month	0.86	0.65

P values in bold are statistically significant ($P < 0.05$).

5.3.2 Effects of drainage on soil N_2O fluxes

There were no significant differences in soil N_2O fluxes between the drained and undrained plots (P=0.6, Table 5.1). Large N_2O fluxes which coincided with fertilisation were observed in both the drained and undrained plots between June and

August 2006 (Fig. 5.1a). They declined sharply in September 2006 and remained consistently lower throughout the study period. The mean N_2O flux from 2006 to 2008 was 6.90±1.40 and 7.43±1.42 mg m^{-2} d^{-1} in the drained and undrained plots, respectively. In 2006–07, the mean N_2O flux was 11.81±2.62 and 12.53±2.65 mg m^{-2} d^{-1} in the drained and undrained plots, respectively. The mean N_2O flux in 2007–08 was 1.55±0.23 and 1.88±0.30 mg m^{-2} d^{-1}in the drained and undrained plots, respectively.

Averaged monthly N_2O fluxes in the drained and undrained plots are shown in Fig. 5.1a. N_2O flux rates varied from month to month ($P=0.0001$). Mean monthly flux rates varied from a sink (−1.13±0.18 mg m^{-2} d^{-1}) for atmospheric N_2O to a net source (49.20±16.60 mg m^{-2} d^{-1}) in the drained plots and from 0.89±0.27 to 48.34±15.07 mg m^{-2} d^{-1} in the undrained plots. N_2O fluxes in the drained and undrained plots showed seasonal emissions patterns (Fig. 5.2a). N_2O fluxes in the drained and undrained plots were higher in the summer of 2006 which coincided with fertilisation. Large N_2O emissions observed in the drained and undrained plots declined sharply in the autumn. There was consumption of atmospheric N_2O in both the drained and undrained plots in the winter of 2006–07. N_2O emissions started to rise again in the spring of 2007 and were higher in the summer/autumn of 2007, but fluxes were not as high as in the previous year. The fluxes were low again in the spring/winter of 2007–08. There was no significant difference in N_2O emissions between the drained and undrained plots in any season (Fig. 5.2a). Annual N_2O fluxes are shown in Fig. 5.3. The annual fluxes were higher in 2006–07 than in 2007–08 ($P=0.0001$). Annual soil N_2O fluxes varied between 1.87±0.27 and 13.71±3.04 kg N_2O–N ha^{-1} yr^{-1} in the drained plots and were on average 8.02±163 kg N_2O–N ha^{-1} yr^{-1}. In the undrained plots, annual fluxes ranged between 2.18±0.35 and 14.55±3.21 kg N ha^{-1} yr^{-1} and were on average 8.63±1.65 kg N_2O–N ha^{-1} yr^{-1}.

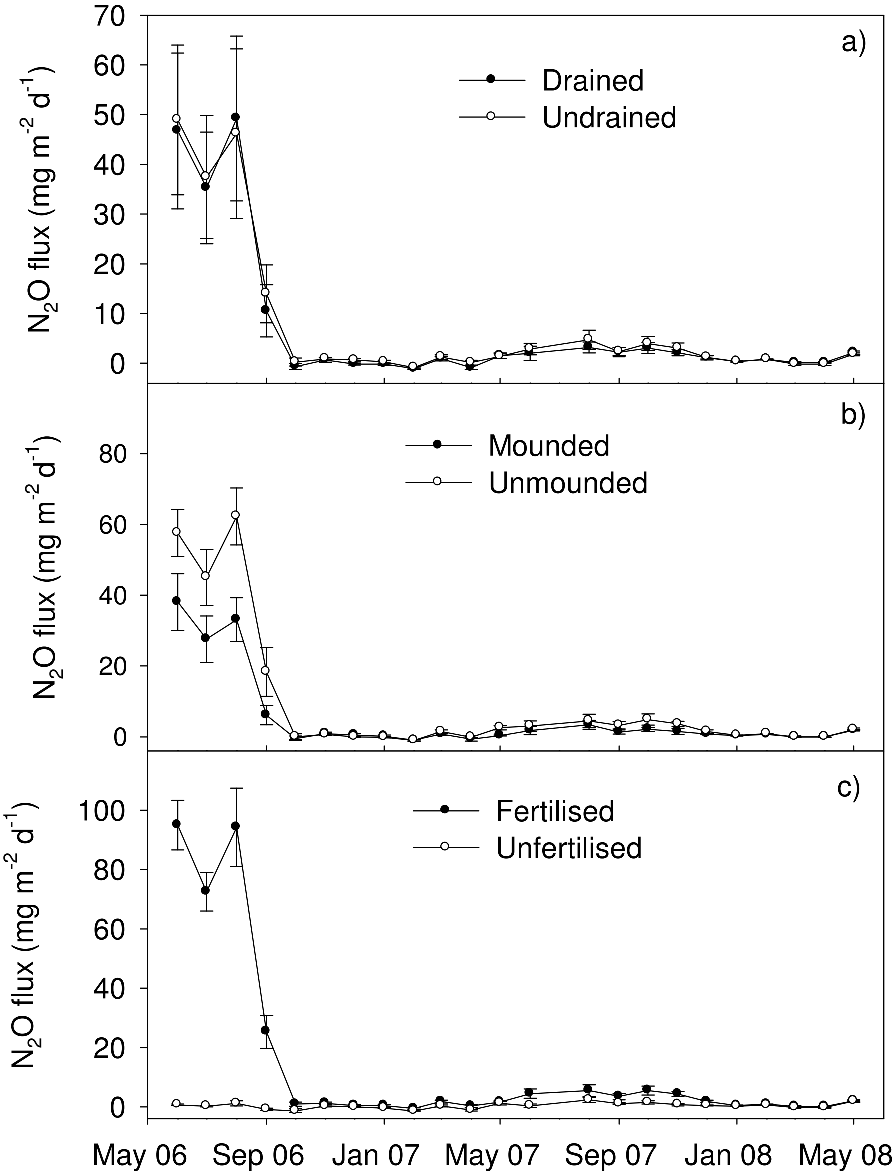

Figure 5.1 Averaged monthly N_2O flux in: a) the drained and undrained, b) mounded and unmounded and c) fertilised and unfertilised from June 2006–May 2008 (showing mean values and standard error for each month) (n=34).

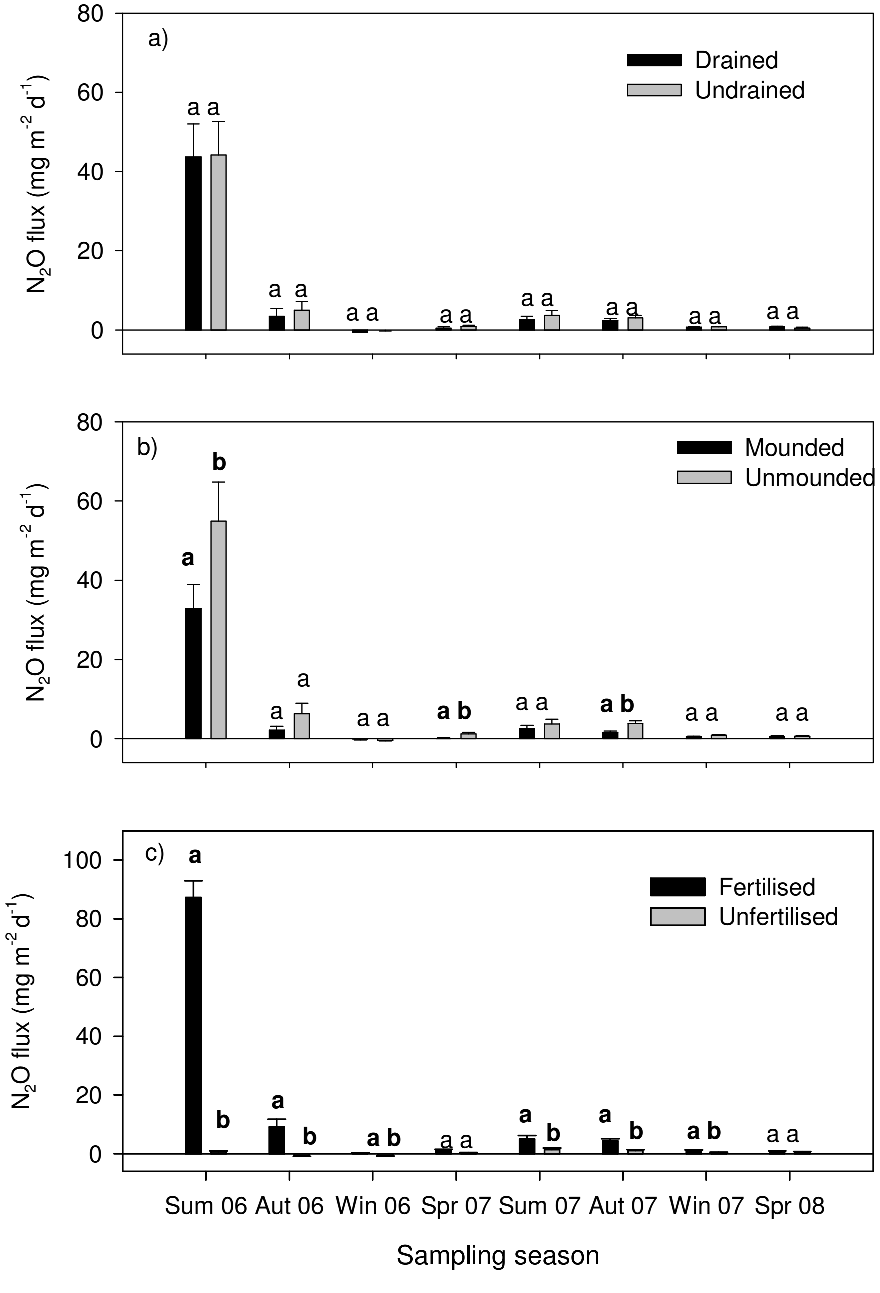

Figure 5.2. Seasonal soil N_2O flux for all treatments. The vertical bars represent standard error of mean. Different letters in bold indicate significant differences ($P<0.05$) between the drained vs. undrained, mounded vs. unmounded and fertilised vs. unfertilised treatment (sum = summer, aut =autumn, win =winter and spr= spring).

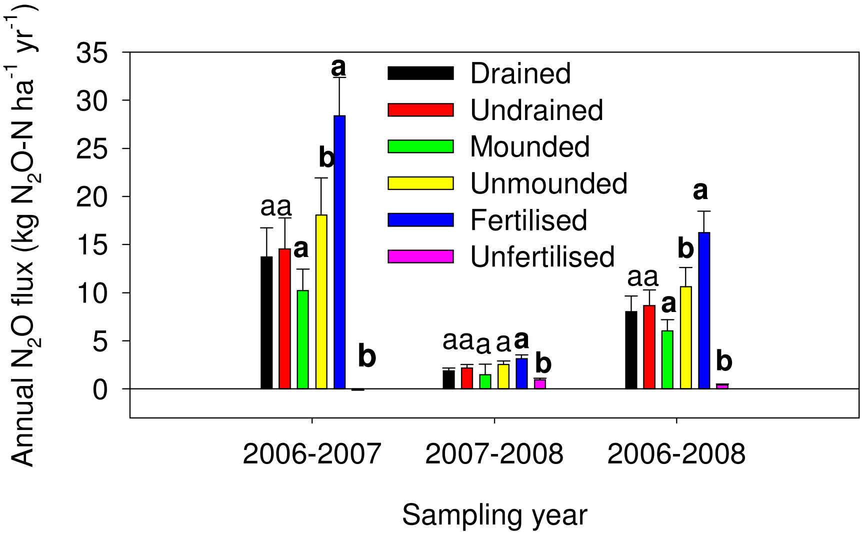

Figure 5.3 Annual N_2O flux from all treatments. The vertical error bars represent standard error of mean. Different letters in bold indicate significant differences between the drained vs. undrained, mounded vs. unmounded and fertilised vs. fertilised treatment in each sampling year ($P<0.05$) (n=23, 11 and 34 for 2006–07, 2007–08 and 2006–08, respectively).

5.3.3 Effects of mounding on N_2O flux

The N_2O fluxes in this site were decreased by mounding (P=0.0001, Table 5.1). Fluxes were significantly higher in 2006–07 than in 2007–08 (P=0.04). The mean flux was 8.78±1.92 mg m^{-2} d^{-1} in the mounded subplots and 15.55±3.17 mg m^{-2} d^{-1} in their unmounded counterparts in 2006–07, while in 2007–08 the mean flux was 1.24±0.21 mg m^{-2} d^{-1} in mounded subplots and 2.18±0.32 mg m^{-2} d^{-1} in their unmounded counterparts. Mean monthly N_2O flux rates are shown in Fig. 5.1b. Fluxes varied from month to month (P=0.0001). The mean monthly fluxes varied from a sink (–0.97±0.25 mg m^{-2} d^{-1}) for atmospheric N_2O to a net source (38.07±8.03 mg m^{-2} d^{-1}) in the mounded treatment and from a sink (−1.05±0.21 mg m^{-2} d^{-1}) for atmospheric N_2O to a net source (62.29±8.09 mg m^{-2} d^{-1}) in the unmounded treatment. Large N_2O fluxes were observed in the mounded and unmounded subplots in the summer of 2006, which coincided with fertilisation (Fig. 5.2b). Fluxes declined sharply in the autumn. There was consumption of atmospheric N_2O in the mounded and unmounded subplots in the winter of 2006–07. The fluxes in both the mounded and unmounded treatment increased steadily from the spring of 2007, reaching high values in the summer/autumn of 2007. However, the fluxes were not as high as in the previous year. Mounded and unmounded subplots differed significantly in N_2O fluxes in the summer of 2006 ($P=0.001$), the spring of 2007 ($P=0.006$) and the autumn of 2007 ($P=0.0001$, Fig. 5.2b). Annual emissions from the mounded and unmounded subplots are shown in Fig. 5.3. Annual N_2O emissions in the mounded and unmounded subplots were higher in 2006–07 than in 2007–08 ($P=0.0001$). Annual soil N_2O fluxes in the mounded subplots varied from 1.44±1.5 to 10.20±2.22 kg N_2O–N ha^{-1} yr^{-1} and were on average 6.01±1.19 kg N_2O–N ha^{-1} yr^{-1}. In the unmounded subplots, annual N_2O fluxes varied from 2.53±0.37 to 18.06±3.84 N_2O–N ha^{-1} yr^{-1} and were on average 10.63±1.98 kg N_2O–N ha^{-1} yr^{-1}.

Nitrous oxide emitted from mounded subplots was significantly affected by the position of chambers and fertilisation (all $P=0.0001$, Table 5.2). Mounds, hollows and undisturbed ground differed significantly in soil N_2O flux rates ($P=0.0001$). There was a significant difference in N_2O fluxes among mounded subplots ($P=0.0001$). There was a significant interaction between drainage and the position of chambers ($P=0.05$), fertilisation and the position of chambers ($P=0.0001$) and drainage, fertilisation and the position of chambers ($P=0.004$). Nitrous oxide emitted from hollows varied from 0.64±0.14 to 11.71±0.72 mg m^{-2} d^{-1} and was on average 8.02±1.03 mg m^{-2} d^{-1}. Nitrous oxide emitted from mounds varied from 1.53±0.23 to 8.61±0.53 mg m^{-2} d^{-1} and was on average 6.26±0.77 mg m^{-2} d^{-1}. Nitrous oxide fluxes from the undisturbed ground varied from 1.60±0.24 to 22.60±1.39 mg m^{-2} d^{-1} and were on average 15.60±1.77 mg m^{-2} d^{-1}. Annual fluxes in the hollows, mounds and undisturbed ground are shown in Fig. 5.4. Annual fluxes were higher in 2006–07 than in 2007–08 ($P=0.0001$). Annual soil N_2O fluxes in the hollows, mounds and undisturbed ground were on average 9.32±1.20, 7.28±0.90 and 18.13±2.06 kg N_2O–N ha^{-1} yr^{-1}, respectively.

Table 5.2. Summary of the general linear model of soil N_2O flux in the mounded plots. The model included the effects of drainage, fertilisation, chamber position and sampling date. Plot was entered as random factor nested within drainage. The date of sampling was also entered as a repeated measures factor.

Parameter	*F*	*P*
Drainage	0.01	0.99
Plot	29.84	**<0.0001**
Fertilisation	1447.50	**<0.0001**
Position	57.62	**<0.0001**
Drainage×Fertilisation	0.07	0.92
Drainage×Position	2.92	**0.05**
Fertilisation×Position	83.96	**<0.0001**
Drainage×Fertilisation×Position	5.57	**0.004**
Sampling date	70.65	**0.0001**
Drainage×Sampling date	0.95	0.54
Position×Sampling date	5.71	**<0.0001**
Fertilisation×Sampling date	61.70	**<0.0001**
Drainage×Position×Sampling date	0.77	0.91
Drainage×Fertilisation×Sampling date	0.61	0.96
Fertilisation×Position× Sampling date	4.63	**<0.0001**
Drainage×Fertilisation×Position×Sampling date	0.97	0.54

P values in bold are statistically significant (P< 0.05).

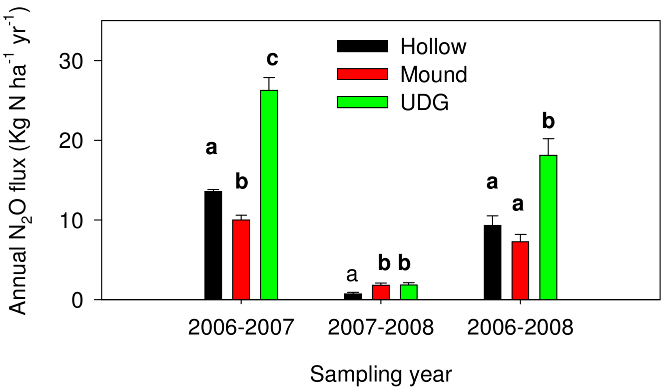

Fig.5.4 Annual N_2O flux from hollows, mounds and undisturbed ground (UDG) in the mounded subplots. Vertical bars indicate standard error of means. Different letters in bold indicate a significant difference in N_2O flux between the hollows, mounds and undisturbed ground in each sampling year (P<0.05) (n=23, 11 and 34 for 2006–07, 2007–08 and 2006–08, respectively).

5.3.6 The effect of fertilisation on soil N_2O flux

Nitrous oxide emissions were significantly increased by fertilisation ($P=0.0001$, Table 5.1). The average N_2O flux over the two years of study was 13.99±1.91 mg m^{-2} d^{-1} in the fertilised subplots and 0.34±0.10 mg m^{-2} d^{-1} in their fertilised counterparts. The two years of study differed significantly in N_2O fluxes ($P=0.0001$). The average N_2O flux in the fertilised and unfertilised subplots was 24.42±3.42 and −0.09±0.14 mg m^{-2} d^{-1} in 2006–07. In 2007–08, the average N_2O flux was 2.62±0.34 and 0.81±0.14 mg m^{-2} d^{-1} in fertilised and unfertilised subplots, respectively.

Nitrous oxide fluxes in the fertilised subplots increased soon after fertilisation in June 2006. Large fluxes in the fertilised subplots were concentrated in a relatively short period between June and August 2006 (Fig. 5.1c). The fluxes in the fertilised subplots declined sharply in September 2006 and remained close to the background emissions throughout the study. Averaged monthly N_2O fluxes are shown in Fig.5.1c. N_2O fluxes varied from month to month ($P=0.0001$). They varied from a sink (−0.61±0.27 mg m^{-2} d^{-1}) for atmospheric N_2O to net source (94.96±8.36 mg m^{-2} d^{-1}) in the fertilised subplots and from a sink (−1.41±0.06 mg m^{-2} d^{-1}) for atmospheric N_2O to net source (2.37±0.82 mg m^{-2} d^{-1}) in their unfertilised counterparts. Averaged seasonal N_2O fluxes from the summer of 2006 to the spring of 2008 are summarised in Fig. 5.2c. Fluxes in the fertilised subplots were higher in the summer of 2006 after fertilisation and declined in the autumn reaching low values in the winter of 2006–07. Fluxes increased steadily in the spring of 2007 and reached high values in the summer/autumn of 2007, but fluxes were not as high as in the previous year. The fertilised subplots consumed atmospheric N_2O in the winter of 2006–07. Fertilisation increased N_2O fluxes in all seasons (Fig. 5.2c). There was consumption of atmospheric N_2O in the unfertilised subplots in the autumn and winter of 2006.

Annual soil N_2O fluxes are summarised in Fig. 5.3. Annual fluxes varied greatly between the fertilised and unfertilised subplots. Annual soil N_2O fluxes in the fertilised subplots were higher in 2006–07 than in 2007–08. The opposite occurred in the unfertilised subplots. Annual soil N_2O fluxes in the fertilised treatment varied from 3.12±0.40 to 28.37±3.98 kg N_2O–N ha^{-1} yr^{-1} and were on average 16.25±2.22

kg N_2O–N ha^{-1} yr^{-1}, whereas in the unfertilised treatment annual fluxes varied from a sink (−0.11±0.16) for atmospheric N_2O to a net source (0.94±0.16 kg N_2O–N), and were on average 0.39±0.13 kg N_2O–N ha^{-1} yr^{-1}. The N_2O emission factor was 34.9% in 2006–07 and 2.7% in 2007–08.

5.3.5 Relationships between N_2O flux and environmental variables

N_2O fluxes were negatively correlated with the soil water table depth in the drained plots (R^2= 0.49, P=0.0001, Fig. 5.6) and undrained plots (R^2= 0.64, P=0.0001, Fig. 5.5). No relationship was observed with soil moisture content in either of the treatments in this site. Nitrous oxide fluxes in the drained plots were significantly correlated with soil temperature at all measured depths (exponential relationships R^2 =0.27, P=0.002 for T_1, R^2=0.31, P=0.001 for T_5 and R^2=0.36, P=0.001 for T_{10}). An exponential relationship between N_2O fluxes and soil temperature was also detected in the undrained plots (R^2=0.33, P=0.0.001 for T_1 and r^2=0.36 P=0.001 and R^2=0.36, P<0.001 for T_5, R^2=0.33, P=0.001 for T_{10}). N_2O fluxes in the mounded (R^2=0.23, P=0.004 for T_1, R^2=0.25, P=0.002 for T_5 and R^2=0.33, P=0.0004 for T_{10}) and unmounded (R^2=0.31, P=0.001 for T_1, R^2=0.34, P=0.0001 for T_5 and R^2=0.37, P=0.0001 for T_{10}) subplots were also exponentially correlated to soil temperature. Figure 5.6 shows the exponential relationship between soil temperature at 10 cm depth in the drained and undrained plots as well as mounded, unmounded and fertilised subplots.

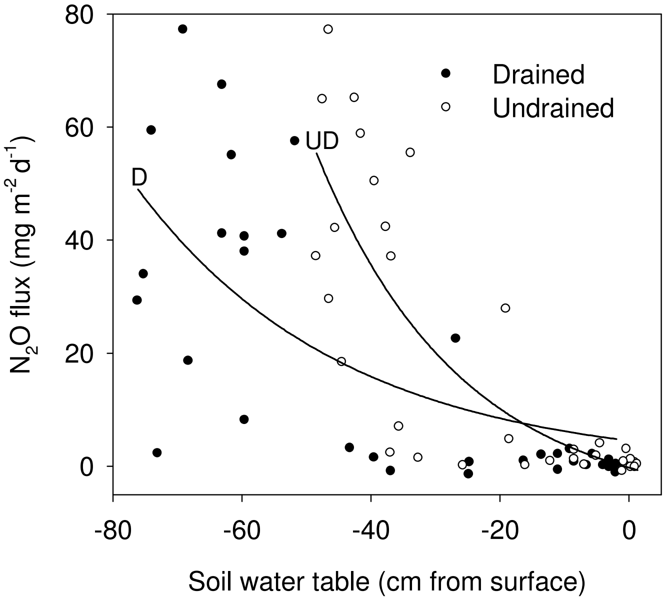

Figure 5.5 Relationships between N_2O flux and soil water table depth for the drained (D) and undrained (UD) plots. Dots represent means of all chambers at each sampling data. Fitted lines are exponential relationships for data for each treatment where $P<0.05$. Drained (D) $y=4.56^{-0.031x}$ $R^2= 0.49$, $P=0.0001$; Undrained (UD) $y=2.85^{-0.062x}$, $R^2= 0.64$, $P=0.0001$ ($n=34$).

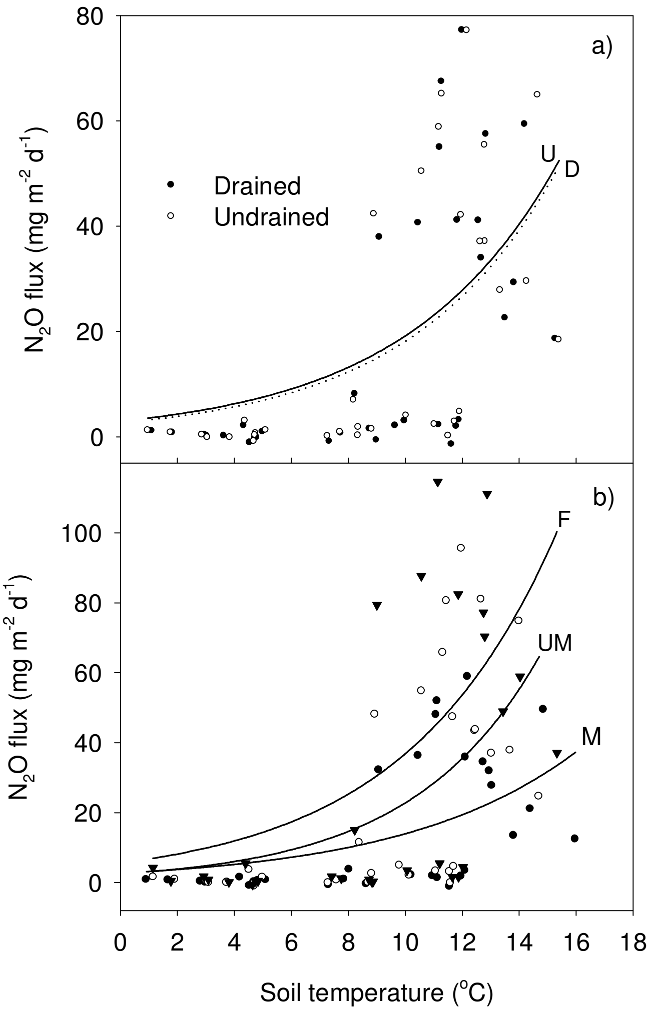

Figure 5.6 Relationships between N_2O flux and soil temperature at 10 cm depth. Dots represent means of all chambers at each sampling date. Fitted lines are exponential relationships for data for each treatment where $P<0.05$. a) Drained (D) $y=2.64^{0.193x}$ $R^2=0.33$, $P<0.001$; Undrained (U) $y=2.96^{0.186x}$, $R^2=0.35$, $P=0.0001$; b) Fertilised (F) $y=5.62^{0.188x}$ $R^2=0.33$, $P=0.001$, Mounded (M) $y=2.72^{0.164x}$ $R^2=0.28$, $P=0.001$; Unmounded (UM) $y= 2.49^{0.221x}$ $R^2=0.38$, $P=0.0001$ (n=34).

5.4 DISCUSSION

5.4.1 Effect of drainage on N_2O flux

Nitrous oxide emission in the present study was not affected by drainage. Increased aeration in the drained plots was expected to favour decomposition and release more N for nitrification. In nitrogen rich peatlands large emissions of N_2O can occur after drainage, when the increased aeration allows nitrification to take place (Freeman et al. 1993; Martikainen et al. 1993a; Kliewer and Gilliam 1995; Schiller and Hastie 1996; Regina et al. 1996, 1999; Kasimir-Klemedtsson et al. 1997; Dowrick et al. 1999; Booth et al 2006; Neill et al. 2006). Soil microbes face drastic changes in the substrate supply and soil conditions caused by drainage and associated soil surface disturbances (e.g., Saari et al. 2009). In well-drained fertile peat soils, increased soil temperature and improved aeration after drainage may favour organic matter decomposition and release N through mineralisation (King et al. 1986; Updegraff et al. 1995; Saari et al. 2009) which provide a substrate for N_2O producing microbes (Freeman et al. 1996; Smolander et al. 1998). It is possible that drainage was not drastic enough in this site because the soil water table depth rose to the soil surface in the winter. The seasonal fluctuation in the soil water table depth may have masked the effects on drainage on N_2O emissions. Soils in the present study are low in pH and inorganic N, a factor which may also have limited nitrification (e.g., Martikainen et al. 1993b; Priha and Smolander 1995; Paavolainen and Smolander 1998; Weslien et al. 2009). Prior studies conducted in peatland soils demonstrated that nitrification can be very low (e.g., Rangeley and Knowles 1988; Verhoeven 1996; Regina et al. 1996) unless the water table depth is lowered permanently by drainage (Martikainen et al. 1995; Nykänen et al. 1995; Regina et al. 1996 Aerts and Ludwig 1997). For example Regina et al. (1996) measured nitrification in peatland soils and found higher activity in soil samples collected from the most fertile drained sites. They found very little nitrification in soil samples collected from both the drained and undrained infertile peat soils. Several studies have suggested that for high N_2O emissions to occur from drained peatland soils, the optimal water table depth and soil moisture need to be combined with high temperature and a low C/N ratio (Jungkunst et al. 2004; Klemedtsson et al. 2005; Ball et al. 2007; Ernfors et al. 2007; Saari et al. 2009).

Fluxes in the drained and undrained plots were higher (13.71 and 14.55 kg N_2O–N ha^{-1} yr^{-1}) in 2006–07 than in 2007–08 (1.87 and 2.18 kg N_2O–N ha^{-1} yr^{-1}). Large emissions occurred at the beginning of the study, which coincided with fertilisation, even though no interaction was observed between drainage and fertilisation. It could be possible that fertilisation provided N for nitrification and denitrification. Nitrification and denitrification may go on simultaneously within soils with microsites of differing soil moisture levels (Davidson 1991; Renault and Stengel 1994), but denitrification relies on the nitrate produced during nitrification (Firestone and Davidson 1989). Annual N_2O fluxes (8.02 and 8.6 kg N_2O–N ha^{-1} yr^{-1}) measured in the drained and undrained plots over the two years of this study are similar to fluxes reported for organic soils used for agriculture (Nykänen et al. 1995; Flessa et al. 1998, Maljanen et al. 2003b; Regina et al. 2004; Mäkiranta et al. 2007) probably because of the effect of fertilisation in 2006–07. For example, Nykänen et al. (1995) found that a fen drained for pasture in Finland emitted 8–9 kg N_2O–N ha^{-1} yr^{-1}. Maljanen et al. (2003b) measured N_2O fluxes from boreal organic soils under different land-uses. They found that annual emissions from the cultivated soils varied from 8.3 to 11.0 kg N_2O–N ha^{-1} yr^{-1} which were twice times the N_2O (4.2 kg N_2O–N ha^{-1} yr^{-1}) emitted from the adjacent forest site. Soil under the drained and undrained plots became net sinks for atmospheric N_2O in the winter of 2006–07 (Fig. 5.2b) when the ground water level was close or above the soil surface. According to Schiller and Hastie (1994) and Regina et al. (1996), water saturated soils may consume N_2O when denitrifiers reduce atmospheric N_2O to N_2.

Nitrous oxide emissions observed in the drained and undrained plots in the 2007–08 were lower than in 2006–07 and are close to fluxes reported for boreal and temperate forest soils (Zerva and Mencuccini 2005a; Von Arnold et al 2005; Ball et al 2007). The reduction in fluxes in 2007–08 could be an indication that high fluxes in 2006–07 were probably due to the effect of fertilisation. Von Arnold et al. (2005) reported emissions of 0.3–0.9 kg N_2O ha^{-1} yr^{-1} from drained organic coniferous forest soils in Sweden. Ball et al (2007) measured fluxes of 0.2–4.7 kg N_2O ha^{-1} yr^{-1} from several Sitka spruce stands of different ages at Harwood Forest close to the present site.

5.4.2 Effect of mounding on N_2O flux

Mounds provide elevated planting positions which mitigate against high soil water table and improve the survival as well as the growth of planted seedlings. In making the mounds, the litter and the organic soil layers are buried beneath the mineral soil layers (Smolander et al. 2000; Saari et al. 2004). It has generally been found that soil cultivation improves oxygen availability (Del Prado et al. 2006) and increases soil temperature (Skiba et al. 1998; Smith et al. 1998) which may increase organic matter decomposition rates. The mixing of the organic layers with the mineral soils during soil cultivation may increase N mineralisation (e.g., Brown et al. 2000; Maggiotto et al. 2000) and enhance nitrification (e.g., Freeman et al. 1996). However, effects of mounding on N_2O fluxes are poorly understood.

Nitrous oxide emissions were reduced by mounding in the present study. Mounding created three different subsites (e.g. hollows, mounds and undisturbed ground) with a different microclimate and organic matter distribution. Nitrous oxide fluxes in the undisturbed ground were higher than those from top of the mounds and hollows. Large N_2O fluxes measured in the undisturbed ground could be related to soil moisture conditions, temperature and availability of N which favoured nitrification and denitrification (June to August 2006) (e.g., Robertson et al. 1987). There was an interaction between mounding and fertilisation, suggesting that fertilisation increased fluxes in the undisturbed ground. The mounds and the hollows had low fluxes which contributed to low N_2O emissions observed in the mounded subplots. The hollows were periodically (November to April) flooded with water, resulting in atmospheric N_2O consumption. Low N_2O emissions to net atmospheric N_2O uptake has been reported in studies conducted in water saturated soils elsewhere (e.g., Regina et al. 1996; Schiller and Hastie 1996; Jacinthe and Dick 1997; Verchot et al. 1999; Johansson et al. 2003; Saari et al. 2009). For example Schiller and Hastie (1996) measured a net N_2O uptake (-0.18 mg m^{-2} d^{-1}) from ditches located in drained boreal forests sites in northern Ontario. Nitrous oxide consumption in water saturated soils may occur through denitrifiers, but probably also through nitrifiers (Robertson and Tiedje 1987; Chapuis-Lardy et al. 2007) since denitrification in the dominant process in anaerobic conditions. The two processes are closely related to each other and to mineralisation, since nitrifiers use NH_4^+ derived from mineralisation and denitrifiers use NO_3^- produced by nitrification. The consumption of N_2O in the flooded hollows may have occurred in denitrification when atmospheric N_2O was reduced N_2 (Blackmer and Bremner 1976; Schiller and Hastie 1994, 1996; Regina et al. 1996; Johansson et al. 2003). This study also showed that there was an increase in N_2O fluxes in the hollows when water inside them evaporated in the period May to September 2007, probably because of increased nitrification activity (e.g., Regina et al. 1996). Increased nitrification in aerated peat soils may lead to nitrate leaching to the anaerobic layer beneath, thereby enhancing denitrification (e.g., Firestone and Davidson 1989; Regina et al. 1996). Occasionally, consumption of N_2O took place on the top of mounds, probably because of dry conditions caused by the mineral soil on the top of mounds or lack of substrate which inhibited nitrifiers. The mineral soil on the top of mounds may also have impended oxygen diffusion and depressed soil microbial activity and thereby reducing a substrate for both nitrification and denitrification.

Annual N_2O emissions from undisturbed ground varied from 1.86±0.17 to 26.26±16 kg N_2O–N ha^{-1} yr^{-1}. Annual N_2O emissions from hollows varied from 0.75±0.17 to 13.59±0.23 kg N_2O–N ha^{-1} yr^{-1}, while emissions from the top of mounds varied from 1.82±0.27 to 10.01±0.61 kg N_2O–N ha^{-1} yr^{-1}. Nitrous oxide emitted from the hollows, mounds and undisturbed ground was higher in 2006–07 than in 2007–08. Mounding interacted with fertilisation and the low fluxes observed in 2007–08 may be related to reduction in N due to loss as both N_2O and N_2 in 2006–07. Some of the added N may have been taken by plants as NO_3^-, a factor which may limit denitrification (Duxbury et al. 1982; Wagner-Riddle et al. 1996; Simojoki and Jaakkola 2000). In general, plants may lower N_2O production and emission by using the mineral nitrogen pool. There is also a possibility that N was also leached and washed away into drainage ditches, similar to the effects observed in upland forest sites (Mannerkoski et al. 2005; Piirainen et al. 2007) and boreal peatland forests (Nieminen 1998) after site preparation.

5.4.3 Effect of fertilisation on N_2O flux

Fertilisation increased N_2O emissions in the present study. This finding agrees with results of fertilisation studies conducted under field and laboratory conditions which reported high N_2O emissions from fertilised soils under different land uses (e.g., Brumme and Beese 1992; Matson et al. 1992; Castro et al. 1994; McTaggart et al. 1994; Neff et al. 1994; Dunfield et al. 1995; Koops et al. 1996; Velthof and Oenema 1997; Augustin et al. 1998a; Regina et al. 1998; Nykänen et al. 2002; Webb et al. 2004; Bremer 2006). However, others found no change in fluxes and consumption of N_2O has also been observed in some studies after N fertilization (Maggiotto et al, 2000; Webb et al. 2004; Bremer 2006). Nitrous oxide emissions in the present study increased soon after fertilisation because N enhanced denitrification and nitrification (e.g., Hénault et al. 1998; Kaiser et al. 1998; Dobbie et al. 1999). Nitrous oxide emitted from the fertilised subplots varied from 3–28 kg N_2O–N ha^{-1} yr^{-1}. These fluxes are similar to N_2O emissions of 1.7–27.6 N_2O–N ha^{-1} yr^{-1} measured from N-fertilised ungrazed grassland and arable land at sites distributed across Great Britain (Dobbie and Smith 2003). The emissions are also comparable with N_2O fluxes of 10–20 kg N_2O–N ha^{-1} yr^{-1} reported for fertilised grasslands in Western Europe (McTaggart et al. 1994; Velthof and Oenema 1997) and slightly higher than 5.2–9.5 kg N_2O–N ha^{-1} yr^{-1} reported for fertilised boreal forest soils (Regina et al. 1998). Dobbie et al. (1999) measured soil N_2O emissions from intensively managed agricultural fields in Scotland over three years and found that annual emissions varied widely (0.3–18.4 kg N_2O–N ha^{-1}) due to the degree of coincidence of fertilizer application and major rainfall events. Maljanen et al (2004) studied soil N_2O dynamics in boreal organic agricultural soils with different soil characteristics. Soils under barley had higher net N_2O emissions (8.48 kg N_2O–N ha^{-1} yr^{-1}) than those under grass (2.75 kg N_2O– N ha^{-1} yr^{-1}). They observed the highest N_2O fluxes (23.50 kg N_2O–N m^{-2} yr^{-1}) in bare soils. The observed net emission fluxes in the unfertilised treatment varied from a sink (–0.11±0.62 kg N_2O–N ha^{-1} yr) of N_2O to a net source (0.94±0.16 kg N_2O–N ha^{-1} yr^{-}). The fluxes are close to (0.06–1.50 kg N_2O–N ha^{-1} yr^{-1}) reported for a Sitka spruce plantation and unplanted grassland on peaty gley soil at Harwood forest (Ball et al. 2007). The soil under the unfertilised treatment consumed N_2O in the autumn to the winter of 2006, which also occurred in the fertilised treatment in the winter. The consumption of N_2O occurred when soils under the fertilised and unfertilised subplots were water saturated. As discussed in the previous sections, some denitrifiers can gain energy by using atmospheric N_2O as a substrate and, therefore, consumption of atmospheric N_2O can occur in water saturated soils.

Large N_2O emissions from the fertilised subplots were concentrated in a relatively short period fluxes fell back to near-background emissions after about 12 weeks and followed the seasonal time course in course in temperature. N_2O fluxes emitted during these period contributed largely to annual emissions in year of this study. Dobbie et al. (1999) and Dobbie and Smith (2003) observed that on average, 77% of the annual N_2O fluxes from fertilised grassland cut for conservation are emitted within four weeks of fertilisation. A similar emission pattern has been reported for fertilised grasslands in the tropics (Veldkamp et al. 1998), temperate Western Europe (McTaggart et al. 1994; Velthof et al. 1996, Clayton et al. 1997; Velthof and Oenema 1997; Williams et al. 1998; Dobbie et al. 1999), in arable cropland (Jacinthe and Dick 1997, Liu et al. 2005) and in forest soils (Regina et al. 1998). For example, Clayton et al. (1997) observed large N_2O fluxes from fertilised grassland three weeks after fertilisation. Liu et al. (2005) measured N_2O fluxes from a fertilised maize field in northeastern Colorado and found that large N_2O emissions occurred in the first two weeks after fertilisation and declined to near-background emissions after twelve weeks. Bremer (2006) measured soil N_2O emitted from a perennial rye grassland soil near Manhattan, Kansas. The study reported that fluxes from the fertilised treatment were 15 times higher than in the control three days after fertilisation. Flessa et al. (1996) found that N_2O fluxes from urine and dung patches in a pasture increased after 10 to 15 days following deposition of urine and dung to soils due to rapid mineralisation of organic N in urine and dung to NH_4^+.

The N_2O emission factor was very high (35%) in the first year of fertilisation. It was higher than values (0.4−5.8%) reported for fertilised grassland (Clayton et al. 1997; Smith et al. 1998a, b; Dobbie et al. 1999). Large emissions which occurred in the first twelve weeks of fertilisation contributed the largest emissions observed in the first year of study. The warmer summer of 2006 and available N coupled with favourable moisture conditions may have accelerated nitrification and denitrification resulting in large N_2O soon after fertilisation. Nitrous oxide emissions decreased in year 2 and resulted in an emissions factor of 2.7% which is within the range of values reported in many studies (Clayton et al. 1997; Smith et al. 1998a, b; Dobbie et al. 1999; Dobbie and Smith 2003) and above the Intergovernmental Panel on Climate Change (IPCC) default factor of 1.25%.

5.4.4 Dependence of N_2O flux on environmental variables

The exponential relationship between N_2O fluxes and soil temperature observed in this study has also been found in forest soils (Brumme 1995, Schindlbacher et al. 2004; Ball et al. 2007). According to Skiba et al. (1998) and Smith et al. (1998) N_2O fluxes increase with increasing soil temperature in site a where soil moisture and substrate are not liming. The increase in soil temperature coupled with available N probably enhanced microbial activity and increased N_2O production in this site.

Although soil moisture content is an important determinant of soil microbial population and activity (e.g., Linn et al.1984; Skopp et al 1990; Verchot et al. 1999), it was not related to N_2O fluxes in this site. The lack of dependence of soil N_2O fluxes on soil water has also been noted for forests soils (Bowden et al. 1990; Henrich and Haselwandter 1997; Zerva and Mencuccini 2005a). Fertilisation caused large N_2O fluxes which declined after twelve weeks and remained close to the level of background emissions throughout the study. This may indicate that low N availability is limiting nitrification and denitrification in this site. Soil moisture content is important in regulating N_2O emissions but its significance is more apparent in ecosystems where N is cycled rapidly to provide a substrate for nitrification and denitrification (e.g., Verchot et al. 1999). An exponential relationship was observed between soil water table depth and fluxes in the drained and undrained plots, suggesting that increased soil temperature enhanced microbial processes. The increased N_2O fluxes observed at lower table depth in this study confirm similar observations following drainage of flooded peaty soils (Terry et al. 1981, Freeman et al. 1993).

5.5 CONCLUSION

This study showed that drainage did not have a significant effect on N_2O emissions probably because of low soil pH and N availability. It also is possible that drainage was not drastic enough to alter enhance N cycling in this site. Mounding decreased fluxes and N_2O uptake occurred in periodically flooded hollows which may have been caused the reduction of N_2O to N_2 by denitrifiers. Similarly low fluxes occurred on the top of mounds, where N_2O uptake also occurred. It is possible that the mineral soil on the top of mounds impended oxygen diffusion and prevented nitrification. Fertilisation increased N_2O emissions in this site, which were concentrated over a relatively short period. Large emissions observed across treatments coincided with fertilisation indicating that fertilisation was the sources for N_2O in this site. The lack of relationship between N_2O fluxes in the fertilised subplots and soil water content and soil water table depth suggest that fluxes in this site were probably dependent on substrate (N) availability more than on abiotic factors and soil temperature induced microbial activity.

Chapter 6

6.1 SUMMARY OF MAIN RESULTS

After two years, drainage had decreased the total C and N concentration in the top 10 cm soil layer of peaty gley soil, probably because increased soil temperature and improved aeration induced soil organic matter decomposition. Fertilisation increased the total C concentration in the top 10 cm layer in year 1 of study. The total C and N concentration was not affected by mounding. The soil bulk density in the 0–20 cm layer was increased by mounding, probably because of soil compaction caused by the excavator used to make mounds. Ammonium (NH_4^+) was increased by mounding and fertilisation in 2006–07. This may have increased plant growth in the fertilised subplots in 2006–07, which was also found in the drained plots. In none of the sampling occasions was nitrate (NO_3^-), pH or microbial biomass C affected by drainage, mounding or fertilisation.

Drainage and mounding increased day-time soil temperature and decreased the soil moisture content between May and October. Drainage also lowered the soil water table depth. Drainage increased soil CO_2 emissions in 2006–07 and 2007–08. Improved aeration and increased temperature in the drained plots may have made the soil aerobic and favourable for microbial and autotrophic respiration. Fertilisation also increased soil CO_2 fluxes. Soil respiration was not affected by mounding. Soil CH_4 emissions were affected by the three practices, with drainage reducing the fluxes, mounding and fertilisation increasing the fluxes. Methane emissions in this site were controlled by the water table depth. The changes in soil temperature and moisture content observed in the drained and mounded treatment did not affect CH_4 fluxes in this site. Methane fluxes in the fertilised subplots were related with seasonal changes in soil temperature and water table depth, indicating that temperature did not stimulate soil microbial activity unless sufficient substrate was made available for CH_4-producing soil microorganisms.

Soil N_2O emissions were not affected by drainage, perhaps drainage was not drastic enough to stimulate nitrification. It is also possible that nitrification in this site was not stimulated because of low soil pH and inorganic N. Mounding reduced fluxes and N_2O uptake was observed periodically inside flooded hollows. Nitrous oxide uptake may have occurred during the reduction of N_2O to N_2 by denitrifiers, since the high soil moisture levels in soils favours denitrification. N_2O uptake also occurred in the mineral soil on the top of mounds, perhaps due to dry conditions and lack of substrate which affected nitrification. The mineral soil on the top of mounds may also have impended oxygen diffusion and depressed the microbial activity and thereby slowing nitrification. Fertilisation interacted with mounding and may have increased fluxes in the undisturbed ground. Nitrous oxide fluxes were increased by fertilisation. Large fluxes occurred soon after fertilisation and were concentrated over a relatively short period. Large fluxes which coincided with fertilisation occurred across treatments at the start of sampling, indicating that fertilisation was the source of N_2O in this site. The lack of correlation between N_2O emissions and soil moisture and water table depth in the fertilised subplots may suggest that N was more important than abiotic factors and soil temperature induced soil microbial processes.

6.2 SUMMARY OF MAIN RESULTS BY PRACTICE

Over the two years of study, drainage significantly increased; a) above-ground plant biomass, b) soil temperature (T_1, T_5 and T_{10}), c) soil CO_2 fluxes but reduced d) CH_4 fluxes, e) C, N and organic matter concentrations, f) soil water table depth and g) soil moisture content(m^3 m^{-3}). Mounding significantly increased; a) soil bulk density, b) NH_4^+ availability , c) soil temperature (T_1 and T_5) and d) soil CH_4 emissions, but reduced; e) soil moisture content(m^3 m^{-3}) and f) N_2O fluxes. Fertilisation increased; a) C in year 1 and soil organic matter concentrations, b) soil CO_2 fluxes, c) soil CH_4 fluxes, d) N_2O fluxes, e) NH_4^+ in year 1 and f) above-ground plant biomass.

6.3 TOTAL GREENHOUSE GAS BUDGET OF THE PRACTICES, IN ISOLATION OR COMBINED

The total greenhouse gas budget of CO_2, CH_4 and N_2O (t CO_2–eq. ha^{-1} yr^{-1}) was evaluated over the two years of study. The emission of 1 kg of N_2O to the atmosphere is 298 times more effective than 1 kg of CO_2, while 1 kg of CH_4 is 25 times more effective than an equal mass of CO_2 (IPCC 2007). Drainage increased soil CO_2 fluxes by 22.6 and 32.6% in 2006–07 and 2007–08, respectively (Table 6.1). Drainage reduced the CH_4–CO_2 equivalent emissions by 57.2 and 76.1% 2006–07 and 2007–08, respectively (Table 6.1). Drainage reduced the N_2O–CO_2 equivalent emissions by 5.7 and 14% in 2006–07 and 2007–08, respectively (Table 6.1). Drainage increased the CO_2 equivalent emissions by 9.3 and 23.0% in 2006–07 and 2007–08 respectively.

Mounding reduced CO_2 fluxes by 5.3% in 2006–07, but caused an increase of 0.15% in 2007–08 (Table 6.1). Mounding decreased the N_2O–CO_2 equivalent emissions by 43.5 and 43.3% in 2006–07 and 2007–08, respectively. Mounding increased the CH_4–CO_2 equivalent emissions by 33.5 and 57.1% in 2006–07 and 2007–08, respectively. Mounding decreased the total greenhouse gas budget by 22.0 and 5.8% in 2006–07 and 2007–08, respectively. Fertilisation increased soil CO_2 fluxes by 8.7 and 5.6% in 2006–07 and 2007–08, respectively. Fertilisation also increased the N_2O–CO_2 equivalent emissions by 26713.8 and 232.2% in 2006–07 and 2007–08, while the CH_4–CO_2 equivalent emissions were increased by 58.8 and 22.3%, respectively. When fertiliser was added in 2006–07, N_2O fluxes contributed more to the total greenhouse budget that CO_2. Fertilisation increased the total greenhouse budget by 146.5% in 2006–07 and 20.3% in 2007–08. Over the two years of study, drainage and fertilisation increased the total greenhouse gas budget by 13.1 and 97.9%, while mounding caused a reduction of–17.6%.

In the UK site preparation carried out prior to afforestation usually involves drainage and mounding using excavators to overcome adverse site conditions and provide a favourable environment to increase the survival and early the growth of trees (Paterson and Mason 1999; Minkkinen et al. 2008). Nitrogen and phosphorus fertilisers are often applied on nutrient poor sites such as moorland and heathland soils to ensure crop viability (e.g., Taylor 1991; Minkkinen et al. 2008).

The combined effect of drainage plus mounding and drainage plus mounding plus fertilisation on the total greenhouse gas budget was calculated over the two years of study. These are the only combinations carried out in practice in upland Britain. To our knowledge information on the effect of these combinations on the total greenhouse gas budget is lacking. Drainage plus mounding increased soil CO_2 fluxes by 16.4 and 33.8% in 2006–07 and 2007–08, respectively (Table 6.2). Over the two years of study drainage plus mounding increased soil CO_2 fluxes by 23.3%. Drainage plus mounding increased soil temperature and improved aeration both of which have the potential to increase the litter turnover and decomposition (e.g., MacDonald et al. 1999; Niklinska et al. 1999). It seems, therefore, that the effect of increased soil temperature and aeration after drainage plus mounding on organic matter and plant roots may have increased CO_2 emissions. Carbon dioxide fluxes in the drained plus mounded plots were higher in 2007–08 than in 2006–07, probably the ditches were more effective in 2007–08 and favoured both heterotrophic and autotrophic respiration. It is also possible that the organic layers and the mineral soil on the top of mounds were mixed in 2007–08 compared to 2006–07 and may have increased decomposition due to the fertilisation effect (e.g., Minkkinen et al. 2008). Plant roots are an important sources of C in many soils (e.g., Balesdent and Balabane 1996; Puget and Drinkwater 2001), and increases in soil temperature and improvement in aeration after drainage may have enhanced root growth and autotrophic respiration in this study.

Drainage plus mounding reduced the N_2O–CO_2 equivalent emissions by 46.9 and 52.3% in 2006–07 and 2007–08, respectively. Over the two years of study, drainage plus mounding reduced the N_2O–CO_2 equivalent fluxes by 47.5%. Drainage plus mounding increased soil temperature and improved aeration, but failed to stimulate N_2O emissions as observed in fertile peatland soils (e.g., Martikainen et al. 1993a; Merbach et al. 1996; Augustin et al. 1998b), probably because low pH and inorganic N hindered nitrification in the present study site. Drainage plus mounding decreased the CH_4–CO_2 equivalent fluxes by 44 and 63% in 2006–07 and 2007–08, respectively. Over the two years of study drainage plus mounding reduced CH_4–CO_2 equivalent fluxes by 50.7%. This was expected because drainage plus mounding decrease CH_4 production in waterlogged peat and increase CH_4 oxidation in the aerated surface peat (e.g., Smith et al. 2000; Oleszczuk et al. 2008; Minkkinen et al. 2008). Aerobic conditions caused by drainage and mounding may have increased CH_4 oxidation in site (Smith et al 2000). Nykänen et al. (1998) studied CH_4 fluxes in peatland soils in Finland and reported that drainage reduced CH_4 fluxes by 30 to over 100%. Overall drainage plus mounding decreased the total greenhouse gas budget by 14.9 % in 2006–07, but caused an increase of 15.6% in 2007–08. On average, drainage plus mounding reduced the total greenhouse gas budget by 6.9%.

Drainage plus mounding plus fertilisation increased soil CO_2 fluxes by 29.6 and 46.5% in 2006–07 and 2007–08 respectively (Table 6.3). Drainage plus mounding plus fertilisation increased soil CO_2 fluxes by 36.4%. When organic soils are drained and mounded for afforestation the accumulated organic matter becomes available for aerobic decomposition, due to increased temperature and aeration, leading to higher soil CO_2 release rates (Silvola et al. 1996a; Von Arnold et al. 2005). The application of N fertiliser in drained and mounded sites may stimulate soil microbial activity and increase decomposition and CO_2 release rates further. Drainage plus mounding plus fertilisation increased the N_2O–CO_2 equivalent emissions by 5797.6 and 52.7% in 2006–07 and 2007–08 respectively. On average, drainage plus mounding plus fertilisation increased N_2O–CO_2 equivalent emissions by 1335.6%. Several studies demonstrated that drainage and soil cultivation increases N_2O fluxes of fertile organic soils (Martikainen et al. 1993a, Nykänen et al. 1995; Regina et al. 1996; Kasimir-Klemedtsson et al. 1997; von Arnold et al. 2005). Drainage had no effect on N_2O fluxes, while mounding caused a reduction. This result may suggest fertilisation was the major source of N_2O–CO_2 equivalent emissions in the drainage pus mounding plus fertilisation (e.g., Ambus et al. 2001; Nykänen et al. 2002; Venterea et al. 2003; Wallenstein et al. 2006; Zhang et al. 2008) because N provide a substrate for nitrification and denitrification (e.g., Sitaula et al. 1995; Hénault et al. 1998; Kaiser et al. 1998; Dobbie et al. 1999).

Drainage plus mounding plus fertilisation reduced CH_4–CO_2 equivalent fluxes by 12.3% and 58.5% in 2006–07 and 2007–08, respectively. However, the reduction was lower in 2006–07 compared to 2007–08. The effect of drainage and mounding on soil CH_4–CO_2 equivalent emissions in 2006–07 may have been counteracted by N fertilisation which has been found to reduce the activity of CH_4 oxidizing soil microbes in many ecosystems (e.g., Flessa et al. 1995; Dobbie and Smith 1996; Hütsch 1998; Tlustos et al. 1998; Hilger et al. 2000; Reay and Nedwell 2004; Suwanwaree and Robertson 2005). Drainage plus mounding plus fertilisation increased the total greenhouse budget by 133.4 and 45.0% in 2006–07 and 2007–08, respectively. On average, drainage plus mounding plus fertilisation increased the total greenhouse budget by 101.8%.

Peatland soils play an important role in the balance of GHGs between soils and the atmosphere. In their natural state, peatlands are net C sink (Hargreaves et al. 2003). Drainage and afforestation has been one the main land management pressures on peatlands. If the water table depth of peatland soils is lowered permanently by drainage and afforestation, the accumulated organic matter becomes available for decomposition, increasing soil CO_2 release rates (Silvola et al. 1996; Lindroth et al. 1998; Broadmeadow and Matthews 2003; von Arnold et al., 2005). Hargreaves et al. (2003) found that a newly drained peatland (2 to 4 years after ploughing) emitted 2–4 t C ha^{-1} yr^{-1}, to the atmosphere. However, as forest stands develop, the NPP of trees and forest ground vegetation may compensate for the soil CO_2 emissions and a forest may become as sink of ~3 t C ha^{-1} yr^{-1} at about 4 to 8 years after afforestation (Hargreaves et al. 2003), although other sites may be net sources (Lindroth et al. 1998). Over the two years of the present study, drainage had no effect on N_2O emissions. Nitrous oxide emissions would be expected to increase when trees are planted and draw water from the soil increasing oxidation of organic matter which may release N to N_2O producing microbes. Several studies have found drained organic forest soils to be significant sources of atmospheric N_2O depending on fertility (Martikainen et al. 1993a; von Arnold et al. 2005). Furthermore, drainage and afforestation may lower CH_4 fluxes or even turn forest soils to net sinks for atmospheric CH_4 (Nykänen et al. 1998; .Maljanen et al.2003a; von Arnold et al. 2005).

In practice forest fertilisation in the UK is not recommended unless it is absolutely important (DEFRA 2008), but it is still an important silvicultural operation used for first rotation (afforestation) forest stands planted on some very nutrient demanding sites such as upland organic soils. The application of N (150 kg N ha^{-1}) is based on the Forestry Commission fertilisation guidelines (Taylor 1991). Although the fertiliser was applied soon after drainage and mounding in the present study, in practices the guidelines recommend fertilisation on a three years cycle. Fertilisation commences 6 years after planting until canopy closer at about 10–15 years (Hibberd 1991). Nitrous oxide emissions from fertilised forest soils are expected to be high soon after fertilisation but at the same magnitude observed in this study. The presence of shading from the closing tree canopy may reduce the volatilisation of nitrogen to the atmosphere, resulting in efficient use of N by trees and the forest floor vegetation. Fertilised plantations tend to grow faster and trees which may lead to C sequestration in plant biomass and soil (Aber et al. 1998).

Site preparation for afforestation may affect GHG emissions in many ways as observed in the present study. Drainage may initially increase CO_2 emission rates of peaty gleys soils, although in the long-term drainage and afforestation could be beneficial in terms of GHG budget. As the forest mature C will accumulate in trees and the forest floor vegetation as well as in the forest soil and can compensate for CO_2 emissions. In some sites the forest may even become a C sink. In addition, the forest soil can reduce or even become a net sink for atmospheric CH_4 a more potent gas than CO_2. On the other hand the release of N in from the organic matter may increase N_2O emission the forest.

6.4 CONCLUSION

Overall, drainage increased the total greenhouse budget, while mounding caused a decrease. Fertilisation was the largest contributor to the total greenhouse gas budget. Fertilisation was carried out soon after drainage and mounding, although in practices it is tending operation (post-planting) carried out after the trees have established themselves in their new environment. Therefore caution is needed in interpreting these results because the effect of fertilisation may make site preparation practices seem like emitting a lot of GHG. This study could probably have yielded high quality information if the site was not fertilised. The UK’s Land Use Land-Use Change and Forestry (LULUCF) Greenhouse Gas Inventory requires information on GHG fluxes arising in the transition between different land-uses (Milne and Cannell 2005) which are published in each year as National Inventory Reports (NIRs). Peaty gley soils hold substantial quantities of the terrestrial carbon stocks in the UK and large losses occur when they are drained for afforestation. Therefore this result shows that there is potential to include GHG fluxes from newly drained and mounded sites in the NIRs. However, this may require high quality data from different newly drained planting sites for up-scaling. Up-scaling may not only provide an exact amount of GHG lost from newly drained sites, but may also help us obtain an estimate based on the current knowledge. Therefore more studies on GHG emission and organic matter decomposition are needed from newly drained sites. Total CO_2 emitted from soil to the atmosphere include respiration from soil organisms, plant roots and organic matter decomposition. Decomposition studies may give an indication about the amount of C lost from newly drained planting sites from organic matter decomposition induced by drainage and mounding. However, if similar studies are conducted in the future fertilisation should be avoided since it is tending operation

Table 6.1 Annual total greenhouse gas budget (CO_2-eq. t ha^{-1} yr^{-1}) of the drainage mounding and fertilisation and the change (%) caused each practice.

Treatment/year	CO_2 (t ha^{-1} yr^{-1})	Change (%)	N_2O–CO_2 eq. (t ha^{-1} yr^{-1})	Change (%)	CH_4–CO_2- eq. (t ha^{-1} yr^{-1})	Change (%)	Total GHG budget (CO_2-eq. t ha^{-1} yr^{-1})	Change (%)
2006–2007								
Drained	22.16±0.75		12.84±2.84		0.23±0.01		35.23±3.61	
Undrained	18.08±0.70	22.6	13.62±3.01	–5.7	0.53±0.03	–57.2	32.23±3.74	9.3
Mounded	19.57±0.69		9.55±2.08		0.43±0.03		29.55±2.80	
Unmounded	20.67±0.81	–5.3	16.91±3.60	–43.5	0.32±0.02	33.5	37.91±4.43	–22.0
Fertilised	20.96±0.72		26.57±3.73		0.46±0.03		47.99±4.48	
Unfertilised	19.28±0.77	8.7	–0.10±0.15	26713.8	0.29±0.02	58.8	19.47±0.95	146.5
2007–2008								
Drained	15.00±1.09		1.75±0.25		0.08±0.01		16.84±1.35	
Undrained	11.31±0.92	32.6	2.04±0.33	–14.0	0.34±0.02	–76.1	13.69±1.27	23.0
Mounded	13.16±1.02		1.35±0.21		0.26±0.02		14.76±1.26	
Unmounded	13.14±1.02	0.15	2.37±0.34	–43.3	0.16±0.02	57.1	15.68±1.38	–5.8
Fertilised	13.51±1.04		2.93±0.38		0.23±0.02		16.67±1.44	
Unfertilised	12.79±1.00	5.6	0.88±0.15	232.2	0.19±0.02	22.3	13.86±1.17	20.3
2006–2008								
Drained	18.41±0.71		7.51±1.52		0.16±0.01		26.07±2.24	
Undrained	14.54±0.62	26.6	8.08±1.55	–7.2	0.44±0.02	–64.2	23.06±2.19	13.1
Mounded	16.22±0.66		5.63±1.12		0.35±0.02		22.20±1.80	
Unmounded	16.73±0.70	–3.0	9.96±1.85	–43.5	0.25±0.02	41.0	26.94±2.57	-17.6
Fertilised	17.06±0.68		15.22±2.08		0.35±0.02		32.63±2.78	
Unfertilised	15.88±0.67	7.4	0.37±0.11	4023.1	0.24±0.01	45.2	16.49±0.79	97.9

The ‘±’indicate the standard error of the mean. Negative values indicate a decrease in the total greenhouse gas budget.

Table. 6.2. Annual total greenhouse gas budget (CO_2-eq. t ha^{-1} yr^{-1}) caused by drainage plus mounding and the change (%) caused by the combined treatments.

Treatment/year	CO_2 (t ha^{-1} yr^{-1})	Change (%)	N_2O–CO_2-eq. (t ha^{-1} yr^{-1})	Change (%)	CH_4–CO_2 eq. (t ha^{-1} yr^{-1})	Change (%)	Total GHG budget (CO_2-eq. t ha^{-1} yr^{-1})	Change (%)
2006–2007								
Drained+Mounded	21.14±0.96		9.23±2.98		0.25±0.02		30.62±3.96	
Undrained+Unmounded	18.16±1.04	16.4	17.38±4.94	–46.9	0.44±0.04	–44.0	35.98±6.02	–14.9
2007–2008								
Drained+Mounded	14.68±0.51		1.26±0.28		0.10±0.02		16.03±0.81	
Undrained+Unmounded	10.98±1.26	33.8	2.63±0.54	–52.3	0.26±0.03	–63.0	13.87±1.83	15.6
2006–2008								
Drained+Mounded	17.76±0.96		5.42±1.59		0.18±0.01		23.35±2.56	
Undrained+Unmounded	14.40±0.88	23.3	10.33±2.66	–47.5	0.36±0.03	–50.7	25.08.6±3.57	–6.9

The ‘±’indicate the standard error of the mean. Negative values indicate a decrease in the total greenhouse gas budget.

Table 6.3. Annual total greenhouse gas budget (CO_2-eq. t ha^{-1} yr^{-1}) caused by drainage plus mounding plus fertilisation the change (%) caused by the combined treatments.

Treatment/year	CO_2 (t ha^{-1} yr^{-1})	Change (%)	N_2O–CO_2-eq. (t ha^{-1} yr^{-1})	Change (%)	CH_4–CO_2-eq. (t ha^{-1} yr^{-1})	Change (%)	Total GHG budget (t CO_2-equiv. ha^{-1} yr^{-1})	Change (%)
2006–2007								
Drained+Mounded+ Fertilised	22.12±1.34		18.97±5.53		0.30±0.03		41.39±6.90	
Undrained+Unmounded +Unfertilised	17.07±1.57	29.6	0.32±0.29	5797.6	0.34±0.04	−12.3	17.73±1.90	133.4
2007–2008								
Undrained+Mounded+ Fertilised	15.28±2.21		1.86±0.51		0.10±0.02		17.25±2.74	
Undrained+Unmounded + Unfertilised	10.43±1.75	46.5	1.22±0.33	52.7	0.24±0.04	−58.5	11.90±2.11	45.0
2006–2008								
Drained+Mounded+ Fertilised	18.54±1.38		10.79±3.06		0.20±0.02		29.53±4.46	
Undrained+Unmounded +Unfertilised	13.59±1.25	36.4	0.75±0.22	1335.6	0.29±0.03	−30.6	14.64±1.50	101.8

The ‘±’indicate the standard error of the mean. Negative values indicate a decrease in the total greenhouse gas budget.

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APPENDIX

Figure 1. Cutting open drainage ditches by an excavator.

Figure 2. Dry drainage ditches.

Figure 3. Drainage ditches flooded with water.

Figure 4. A hollow flooded with stagnant water.

Figure 5. A cylindrical PVC collar inserted on undisturbed ground.

Figure 6. A cylindrical PVC collar inserted in a hollow.

Figure 7. Collar and a cylindrical chamber on undistributed ground; a circular rubber was used to make a tight seal around collar and chamber.

Figure 8 . Collar and a cylindrical chamber in a hollow; a circular rubber was used to make a tight seal around collar and chamber.

Figure 9. Collar and a cylindrical chamber on a mound hollow; a circular rubber was used to make a tight seal around collar and chamber.

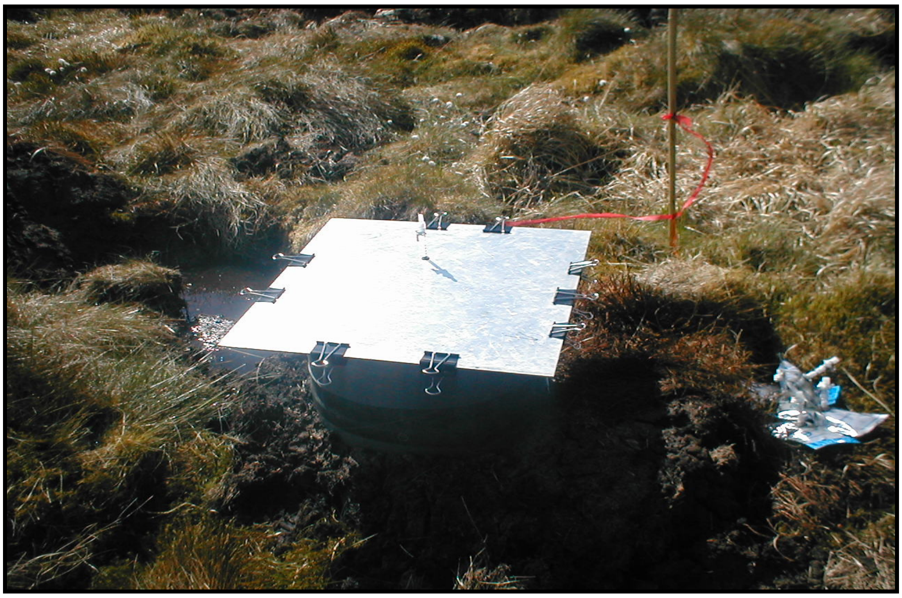

Figure 10. Collar and a cylindrical chamber and a removable aluminum lid fitted with a 3-way stop cock. The lid was tightly clipped to chambers using large clips.