ASSESSMENT OF THE APPLICATION OF A COVER CROP AND CONSERVATION TILLAGE ON SOIL AND WATER PROPERTIES AND ON DISSOLVED NITROUS OXIDE IN AN ARABLE SYSTEM

A thesis submitted to the School of Environmental Sciences of the University of East Anglia in partial fulfilment of the degree of Doctor of Philosophy

BY

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ABSTRACT

Agriculture is a major contributor to environmental pollution. About quarter of water bodies in England are classified as being good ecological and chemical status. To tackle agricultural pollution, a range of on-farm mitigation measures are recommended. The overall aim of this study was to assess the effectiveness of cover cropping and reduced cultivation methods as in-field mitigation measures to reduce diffuse water pollution, improve soil quality and reduce nitrous oxide greenhouse gas emissions. These mitigation measures were applied to seven fields within the intensive arable River Wensum catchment, eastern England, with a further two fields kept under conventional cultivation as a control. Soil and water chemistry, principally water discharging from subsurface agricultural field drains, were regularly sampled and analysed from these fields over a two-year period.

The results revealed the mitigation measures had no positive impact on soil quality. The soil chemical condition, including soil organic carbon, phosphorus, magnesium and sulphate concentrations were not improved by the use of a cover crop or reduced cultivation, whilst soil physical condition deteriorated through increased compaction, as highlighted by increased bulk density, penetration resistance and lower infiltration rates. Conversely, field drain water quality improved markedly. The presence of a winter cover crop significantly reduced mean dissolved nitrate concentrations from 13.9 mg N L^{-1} to 2.5 mg N L^{-1}, an 82% reduction. Different inversion intensity of the soil tended to have no effect on nitrate concentrations. Regarding dissolved N_2O, a slightly higher N_2O concentration was recorded in field drains under a cover crop than without cover crop. This finding suggests that whilst the use of a winter cover crop is highly effective at reducing soil nitrate losses to rivers, it does not represent an effective strategy for reducing N_2O emissions. Indirect nitrous oxide emission factors (EF_{5g} and EF_{5r}) were calculated using two approaches (IPCC 2006 and the $N_2O\text{-}N/NO_3^-\text{-}N$ ratio) for both field drain and stream water samples. Values for these two EFs obtained were found to be below the IPCC default value of 0.0025. If the IPCC were to revise EF_5 values in future then, regardless of soil type, crop type, and land use practices, a value of 0.0009 (about one third of the current value) for EF_{5g} and a value of 0.0002 (one order of magnitude lower than the current value) for EF_{5r} may be a more reasonable estimates. Such radical downward revision would at least halve the current estimates of N_2O emissions associated with N leaching and runoff from agriculture for both the UK and globally.

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Chapter 1: INTRODUCTION

1.1 Background to the study

Environmental pollution is a major global concern. When sources of water, soil and air pollution are enumerated, agriculture is listed as a major contributor. To meet rising demand from a growing population, intensification of agriculture and food production has led to the export of a range of pollutants to both the atmosphere and adjacent freshwater in farming landscapes. The pollutants include nutrients (nitrogen and phosphorus), sediment, microbes, and both pesticides and herbicides. In England, for instance only 27% of water bodies are currently classified as being of good status under new standards set down by the European Union Water Framework Directive (WFD) (Defra, 2015).

Pollution such as sewage and industrial effluent is normally easy to monitor and reduce as it generally arises from a single source. Diffuse pollution, however, arises from many sources when potential polluting substances leach into surface waters and groundwater as a result of rainfall, soil infiltration and surface runoff. Typical examples of diffuse pollution include the use of fertiliser, pesticides and atmospheric deposition. Thus, agriculture is a major source of diffuse pollution. Nitrogen and phosphorus are the two important nutrients associated with diffuse pollution. Diffuse agricultural pollution is estimated to account for approximately 25% of phosphorus, 60% of nitrate and 70% of sediment input into UK rivers nationally (Edwards and Withers, 1998; Cardenas et al., 2011).

Agriculture is not only blamed for degrading water and soil quality, but also for being one of the major contributors to greenhouse gas emissions. According to Gilbert (2012), agriculture is responsible for up to one-third of all human-caused greenhouse gas emissions. One of these gases is nitrous oxide (N_2O) which is a potent greenhouse gas in the atmosphere with 300 times more global warming potential than CO_2 and accounts for about 5% of the total greenhouse effect (Omonode et al., 2011). N_2O molecules also participate in photochemical reactions in the stratosphere which may lead to destruction of the Earth-protecting ozone (O_3) layer (Jacinthe and Dick, 1997). Agriculture alone accounts for about 60% of the total N_2O anthropogenic emissions and global agricultural N_2O emissions have increased by nearly 17% from 1990 to 2005 (Smith et al., 2007).

N_2O emissions from agriculture include direct emissions from agricultural soils due to the application of animal manure and fertiliser nitrogen in arable farming and manure production in grasslands. Also, indirect emissions result from the subsequent leaching of nitrogen to groundwater and surface waters. The Intergovernmental Panel on Climate Change (IPCC) provides guidelines on calculating national inventories of N_2O emissions associated with agriculture. Indirect N_2O emission factors (EF_s) are a way for calculating N_2O emissions from a water body as a fraction of the original N flux into the system (Well et al., 2005b). The IPCC (2006) lowered the default value for the emission factor assigned to indirect emissions, collectively known as EF_5, from 0.025 in the 1997 IPCC report to 0.0075 in the 2006 IPCC report (Outram and Hiscock, 2012). The specific emission factor for groundwater (EF_{5g}) was also decreased from 0.015 to 0.0025. Few studies, however, have examined EF_5. Data from only six studies (published between 1979 and 1993) were used by the IPCC for the determination of the default value, causing large uncertainty (Sawamoto et al., 2005). Therefore, the EF_{5g} has been calculated regardless of different topography, climate, land use, and seasonal change.

In an attempt to protect and enhance aquatic ecosystems, the Water Framework Directive (WFD) was introduced in 2000 by the European Union (Cherry et al., 2008). Each member state is required to improve its all water bodies to reach ‘good’ and non-deteriorating status by 2015. Surface waters must reach good ecological and chemical condition while groundwater must achieve a good chemical standard and pose no risk to the status of surface water into which they may flow (Cherry et al., 2008). In order to meet the WFD water body thresholds for good ecological and chemical condition in the UK, it will be necessary to reduce the losses of nitrogen, phosphorus and sediment to water bodies in places where agricultural land management is responsible for a significant contribution of diffuse pollution and where the magnitude of such losses from farming pose a challenge to meeting compliance targets. Achieving this reduction in agricultural pollutant losses will involve a combination of some changes to the way that farming is practiced and the implementation of mitigation measures to tackle the principle reason for failure.

Newell Price et al. (2011) provided summary information on a range of recommended mitigation measures in the UK as a user guide to reduce diffuse water pollution, air pollution and greenhouse gas emissions. These measures involve changes in soil, livestock, manure and fertiliser management, farm infrastructure and farm practices. They target nutrient availability (source methods), the timing of agriculture practices (timing methods) and the delivery of nutrients from sources to receiving water bodies (transport methods) (Cherry et al., 2008). Plot-scale experimental research has helped researchers to understand the mobilisation of pollutants, and to test the mitigation measure effectiveness (Hutchins et al., 2009; Cardenas et al., 2011). However, these small scale studies are frequently highly generalised and uncertain, especially when applied at farm or catchment scale. Thus, on-farm rather than plot-scale mitigation measures are highly recommended to assess the real effectiveness of a particular measure in reducing pollutants.

Cover crops and reduced tillage mitigation measures were implemented in this study. They were chosen because they are widely promoted as being useful for controlling diffuse pollution and greenhouse gases (Dabney et al., 2001; Lacas et al., 2005; Kaspar et al., 2012). One of the challenges, however, involving mitigation measures is pollution swapping which can occur when a mitigation option or best management practice is introduced to reduce the loss of one pollutant, but in doing so unintentionally leads to an increase in another pollutant, in effect one pollutant is swapped for another. Although pollution swapping has been identified for a number of years, there have been very few studies to examine the potential for pollution swapping across a range of diffuse pollution mitigation options in agricultural systems (Stevens and Quinton, 2009). Because N_2O is produced from nitrification and denitrification processes in soils (Mosier et al., 1998), any change in nitrate concentration due to implementing mitigation measures may change N_2O concentrations. While the majority of studies found that nitrate concentrations in water and soil are reduced significantly by these measures, particularly by cover crops (Staver and Brinsfield, 1998; Justes et al., 1999; Constantin et al., 2010; Singer et al., 2011), studies on the effects of these measures on N_2O produced contradictory results (Petersen et al., 2011; Abdalla et al., 2014; Sanz-Cobena et al., 2014).

1.2 Conservation agriculture

A wide range of environmental impacts, including soil degradation, soil compaction, decrease in organic matter content, water and wind erosion and eutrophication are caused by conventional farming practices. One way of minimising these negative impacts on the agricultural environment is the promotion of conservation agriculture, defined by Jones et al. (2006) as an approach to growing crops which aims to maintain high and sustainable productivity for economic viability, while conserving the environment, in particular soil and water. This focus on improving natural biological processes above and below ground, mainly through minimising tillage, maintaining soil cover throughout the year and practicing effective crop rotation, aims to minimise inputs and loss of agrochemicals and fertilisers (Figure 1.1).

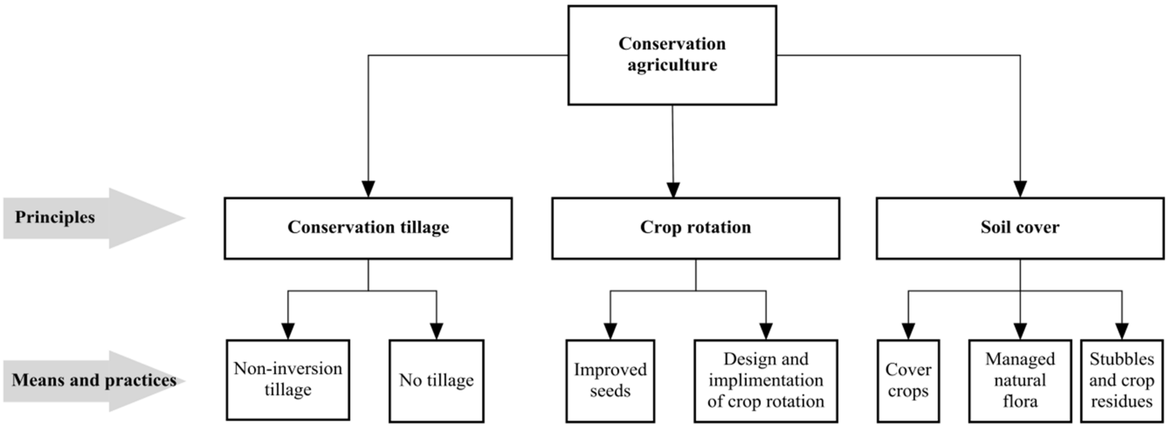

Figure 1.1 The three principles of conservation agriculture and the main practices and means needed to achieve each principle (Stagnari et al., 2010)**.**

Conservation agriculture has been practiced for three decades and has spread widely. Kassam et al. (2009) estimated that there are now some 106 million ha of arable permanent crops grown without tillage in conservation agriculture systems, accounting for an annual rate of increase globally since 1900 of 5.3 million ha. According to the FAO (2014), 150,000 ha of land were under conservation agriculture in 2011 in the United Kingdom, corresponding to 2.4 % of arable land area. The figure for the United States is over 35 million ha in 2009, representing 23 % of arable land.

1.2.1 Cover crops

Cover crops, one of the main adoptions of conservation agriculture, are crops that are grown to provide soil cover during the winter season and fallow period in annual cropping systems and also to protect the soil from erosion and loss of plant nutrients through leaching and runoff. Cover crops were first mentioned over 3000 years ago from the Chou dynasty in China (Burket et al., 1997; Holland, 2004) and their soil improvement benefits are consistently addressed by agricultural literature. They have been used for thousands of years but their reduction in use comes after utilising inorganic N fertiliser during the past 60 years. More recently, there has been renewed interest in the use of cover crops in the application of mulches for organic farming (Mosjidis and Owsley, 2002).

Traditionally, cover crops were turned-over and incorporated before planting of the cash crop; however, recent emphasis on residue management as a measure for reducing soil erosion has led to greater use of cover crops in conservation tillage systems. Some cover crops (e.g. small grain winter crop covers) can serve as a dual cash crop and cover crop (Reeves, 1994). In the past, in addition to winter cover crops, summer annuals like cowpea and soybean were grown for soil improvement, however, currently only winter season cover crops are used in temperate and subtropical zones.

There are two main types of cover crops: non-legume and legumes. Non-legume cover crops include annual or perennial forage grasses such as rye grass, warm-season grasses like sorghum Sudan grass, and brassicas and mustards. Their main purposes are for scavenging nutrients, especially N, left over from a previous crop, reducing or preventing erosion, producing large amounts of residue and adding organic matter to the soil and suppressing weeds. The higher carbon content in non-leguminous cover crops makes for slower break down, resulting in longer lasting residue. This has two results; the higher carbon residue is harder for soil microbes to break down, so the process takes longer, and the nutrients contained in the cover crops residue are usually less available to the next crop (Clark, 2008). On the other hand, leguminous cover crops include winter annuals, such as crimson clover, hairy vetch, field peas and subterranean clover; perennials such as red clover, white clover; biennials such as sweet clover; and summer annuals. Their main purposes are to fix atmospheric nitrogen for use by subsequent crops, reduce or prevent erosion, produce biomass and add organic matter to the soil, and attract beneficial insects. Winter season leguminous cover crops are essential elements of crop management practice in both conservation tillage and organic farming (Mosjidis and Owsley, 2002). Legumes are generally lower in carbon and higher in nitrogen than grass, and this lower C: N ratio causes faster breakdown of legume residue. Hence, N and other nutrients contained in legume residue are usually released faster than grasses (Clark, 2008).

1.2.1.1 Oilseed radish

Oilseed radish, a winter annual brassica cover crop, is a distinctive cover crop that farmers are planting to enhance their soil quality for economic crop production (Stivers-Young, 1998). It has the capacity to recycle soil nutrients, suppress weeds and pathogens, break up compaction, reduce soil erosion, and produce large amounts of biomass. It establishes and grows quickly during cool weather and can be planted early in the autumn to provide fast cover and a green manure crop for cash crops planted in March. Also, it has a thick, deep root that can help break up compacted soil layers and scavenge nitrate that has leached beyond the rooting zone of other crops (Williams and Weil, 2004).

Oilseed radish grows quickly and produces a large amount of biomass in a relatively short time whether it is planted in spring, late summer or early autumn. Four oilseed radish cultivars

(Adagio, Arena, Rimbo and Common), seeded in August, were tested in Michigan (Ngouajio and Mutch, 2004) over two years and produced similar amounts of dry biomass, of total biomass generally exceeding 9000 kg ha^{-1}. The thick taproot of oilseed radish can penetrate compacted layers better than other cover crops such as rye (Williams and Weil, 2004). Deep and large holes left in the soil after the decomposition of roots in the spring facilitate water and air penetration, and primary crop roots can penetrate the soil in the summer better when the soil is dry and hard. Planting oilseed radish can be a no-till alternative to deep tilling or mechanical ripping (Williams and Weil, 2004). Because of its remarkably deep root system, rapid growth and high N uptake, oilseed radish cover crops can take up most of the soluble N left in the soil profile after summer crops have ceased their uptake. Many cover crop species are nitrogen scavengers, but the roots of oilseed radish are able to absorb nitrogen at greater depths, preventing it from leaching into groundwater with the ability of absorbing of between 112 and 167 kg ha^{-1} of nitrogen (Weil et al., 2006). Oilseed radish takes up N from both the topsoil and from deep soil layers, storing the N in tissues near the soil surface. This trapped nitrogen becomes available to the next crop when the plant decomposes in the spring. Growing oilseed radish, therefore, can act as a fertiliser for the next crop in the rotation by recycling nitrogen that would otherwise be lost through leaching (Kristensen and Thorup-Kristensen, 2004).

When oilseed radish is planted in late summer or early autumn, a good stand of cover crop can provide full canopy closure in three to four weeks. This canopy intercepts raindrops, preventing soil erosion. Even after oilseed radish is killed by a hard frost, a layer of decomposing residue remains on the soil surface through the winter and into the early spring providing protection from soil erosion. After surface residues have fully decomposed in spring, runoff and erosion are reduced because of the many holes left behind from the large taproots. Rainwater rapidly infiltrates into these holes, reducing runoff (Williams and Weil, 2004). Oilseed radish grows soon after planting and provides quick ground cover that restricts weeds. When planted in the autumn, oilseed radish prevents weed germination and, consequently, seed production. This action produces a virtually weed-free seedbed in early spring (Stivers-Young, 1998). The near-complete weed suppression can be expected to last until early April, but does not extend into the summer cropping season.

1.2.2 Conservation tillage

Tillage has been practiced by farmers since the move from hunter gatherers to more settled food production systems ten thousand years ago. Tillage is the act of disturbing the soil through use of an implement powered manually or by animals or tractors. Other names for tillage include ploughing and cultivation. Hobbs et al. (2008) listed some reasons for adopting tillage. Tillage was used to provide good uniform seed germination through softening the soil and preparing the seedbed to allow seeds to be positioned easily at a suitable depth in moist soil. Weeds grow alongside crops and compete for light, water and nutrients, so every gram of resource taken up by weeds is one less gram for the crop. Tillage, therefore, enables farmers to shift the advantage from the weed to the crop and allow the crop to grow without competition. Soil nutrients needed for crop growth are released by tillage through mineralisation and oxidation after exposure of soil organic matter to air. Crop residues from the previous year and soil amendments (fertilisers, organic or inorganic) are incorporated into the soil. Roots benefit more if soil amendments and their nutrients are incorporated into the soil; if nitrogen fertilisers are not incorporated they are lost to the atmosphere. Tillage provides temporary relief from compaction and it was found to be important management practice for controlling soil borne diseases and some insects.

On the other hand, tillage also has some negative effects on both the environment and farm business as listed by Hobbs (2007). Greenhouse gas emissions from the combustion of diesel fuel add to global warming. Soil organic matter is oxidised and decreased when it is exposed to the air by tillage. It also causes disruption of the pores left by roots and microbial activity and it causes compaction of the soil below the tractor's wheel surface. The bare surface exposed after tillage is vulnerable to breakdown of soil aggregates as the energy from raindrops is dissipated (Schuller et al., 2007). This process clogs soil pores and reduces infiltration of water and runoff, which subsequently leads to soil erosion. The bare surface exposed after the tillage is susceptible to wind erosion.

Currently, tillage systems in the United Kingdom can be divided into two broad categories: inversion tillage, known as conventional tillage; and non-inversion tillage known, as conservation tillage. Table 1.1 compares conventional and conservation tillage systems. Conventional tillage systems typically comprise both primary and secondary cultivations. Primary tillage is the early main operation that involves inverting the soil using a mouldboard plough. Secondary cultivation creates a seedbed by using a single or double pass cultivator (Morris et al., 2010). Thus, this tillage system disturbs the soil completely through a sequence of operations to incorporate crop residues and additional cultivation to create a seedbed. Conservation tillage, on other hand, has been defined as any tillage that retains at least 30% of the soil surface covered by residues (Lal, 1997). This tillage involves soil management practices that minimise the disruption of the soil's structure, composition and natural biodiversity, thereby minimising erosion and degradation and also water contamination (Holland, 2004).

Table 1.1 Summary of cultivation systems (Morris, 2009)

System	System category	Typical field operation	Advantages	Disadvantages
Conventional tillage	Inversion	Primary cultivation with mouldboard plough. Secondary cultivation with power harrow disc or one or two passes of a heavy press. Cultivator drill roll	Suited for poorly drained and true sand soil. Excellent of incorporation of crop residues. Good control of weeds. Well-tilled seed bed	Higher soil erosion risk. High soil moisture loss. Timeliness consideration. Potentially higher fuel and labour costs. Low work rates
Minimum tillage	Non-Inversion	Cultivation to create stale seedbed. Spray off weeds in stale seedbed. Shallow tine or disc, press. Cultivator drill roll	Some erosion control. Well adapted to light or medium soil. Good residue incorporation	Risk of soil moisture loss with deeper cultivation. Soil compaction concerns
Strip tillage	Non-Inversion	Autumn to spring cultivation to create strips. Drill into strips. Post emergence spray as needed.	Clears residue from within row to allow pre-drilling soil warming and drying. Can allow injection of nutrients in strip area. Well suited for poorly drained soil	Strip may dry too much, crust or erode without straw residue.
Direct drilling	Non-Inversion	Spray of weeds in stubble. Drill into undisturbed surface. Post emergence spray as needed	Maximum erosion control. Soil moisture conservation. Few passes. Minimum fuel and labour costs	No incorporation of residues. Increased dependence on herbicides. Slow soil warming. Soil compaction concerns

Direct drilling, one of the non-inversion methods, refers to the sowing of crops directly into the previous crop stubble with no cultivation taking place since harvesting the previous crop, and with all crop residues left on the soil surface (Cunningham et al., 2004). In this tillage system, a narrow band of soil is cultivated that creates an environment suitable for the seed which is then placed behind the coulter and firmed by a rear roller. No significant change occurs in the soil profile from year to year and dead and decaying crop residues cover the soil surface that change the microenvironment influencing crop growth patterns (Sprague and Triplett, 1986). In this method of tillage, maximum soil erosion control and conservation of soil moisture are achieved. However, some drawbacks of this system include the incomplete incorporation of crop residues and increased dependence on herbicide due to increased volunteer weed growth.

Currently, it has been estimated that approximately 46% of arable land in the UK is under some degree of conservation tillage management (Jones et al., 2006) . It is projected that by the year 2020, conservation tillage may be adopted on 75% of cropland in the USA, 50% in other developed countries and 25% in developing countries. The change in tillage practice has been the readily researched and reported of the conservation agriculture principles, to the extent that frequently the terms conservation tillage and conservation agriculture appear interchangeable (Lal, 1997).

1.3 Aim and objectives

The overall aim of this study was to assess the effectiveness of cover cropping and reduced cultivation methods as in-field mitigation measures to reduce diffuse water pollution, nitrous oxide greenhouse gas emissions and to improve soil quality. The specific objectives of this study can be summarized as follows:

- To evaluate the effect of cover cropping, reduced tillage and direct drilling on soil properties including soil mineral nitrogen, soil macronutrients and on soil physical properties.
- To examine the effectiveness of cover cropping and reduced cultivation methods to decrease nitrate concentrations and losses in drainage water.
- To assess the impact of direct drilling, reduced tillage and cover cropping on nitrous oxide concentrations in drainage water.
- To calculate the indirect nitrous oxide emission factor (EF_5) for river and drain water based on a (long) two-year dataset and compare this value with the current IPCC (2006) default value.

1.4 Thesis structure

This thesis is divided into seven chapters. The current Chapter 1 discusses the background of the study. Conservation tillage and cover cropping as mitigation measures and independent variables in this study are described and their advantages and challenges are also presented. Readers can also find the aim and objectives of this research.

Following this introduction, Chapter 2 describes the study area and methods of sampling and procedures used in this study. It describes the geology, hydrogeology, climate and topography of the study area. Methods and locations of collecting samples for nitrous oxide measurements, and water samples from drains and porous pots and soils are presented. The analytical procedures and statistical analysis methods are also presented in this chapter.

Chapter 3 assesses the effects of cover cropping and reduced tillage on soil chemical and physical characteristics. It shows the effects of cover cropping and reduced tillage on chemical characteristics of soil including soil mineral nitrogen, organic matter, phosphorus, calcium and magnesium. In addition to chemical parameters, the effects of mitigation measures on soil physical characteristics including soil temperature, soil moisture, bulk density, penetration resistance and infiltration rate are also presented.

Chapter 4 evaluates the effects of cover cropping and reduced tillage on water quality. Rainfall variation throughout the study period is presented and the response of nitrate concentrations in the experimental measures area to major rainfall events is discussed. Spatial and temporal variations of nitrate concentration in the field drains are illustrated. Effects of cover cropping and conservation tillage on nitrate concentrations in field drains are presented and are supported by porous pot data. By taking the area of field drainage and flow rates into account, fluxes of nitrate are presented. Also, an attempt is made to calculate approximate nitrogen losses compared to nitrogen input in the soil. The summary at the end of this chapter concludes if the measures are effective in improving water quality.

Chapter 5 presents the results of dissolved nitrous oxide in field drainage and stream water samples. The production mechanisms of nitrous oxide and its different aquatic sources are introduced. Seasonal changes in the nitrous oxide concentrations of field drain and stream water samples are presented. Comparisons between nitrous oxide concentrations in stream and drain water are also shown. Spatial changes in nitrous oxide concentration along the length of the studied water course are presented. The approximate contribution of drains to the total nitrous oxide budget is determined. Finally, the effects of mitigation measures on dissolved nitrous oxide are discussed.

Chapter 6 introduces the calculation of the indirect nitrous oxide emission factor. Theoretical and applicable equations to calculate this factor are presented. Emission factors for stream and drain samples are calculated by using two approaches and they are compared to other studies, particularly to the IPCC (2006). Finally, indirect nitrous oxide emissions from both the UK and across the globe are estimated with the emission factors calculated from this study.

Finally, Chapter 7 concludes the main findings of this research, together with recommendations for future work.

Chapter 2: DESCRIPTION OF STUDY AREA AND METHODS

This chapter introduces the description of the study area including the geology, hydrogeology, climate, topography, soil and land use. The experimental design is presented together with field sampling and laboratory analytical methods. Statistical analysis methods for the treatment of data are introduced.

2.1 Demonstration Test Catchment

The UK Demonstration Test Catchment (DTC) programe is a Department for Environment, Food and Rural Affairs (DEFRA) funded initiative to evaluate the extent to which on-farm mitigation measures can cost-effectively reduce the impacts of diffuse agricultural pollution on river ecology whilst maintaining food production capacity (Wensum Alliance, 2015). Three DTC where established around the UK in 2010, each of which covers different landscape characteristics and farming systems. These catchments were the arable River Wensum in Norfolk, the livestock dominated River Eden in Cumbria, and the mixed farming system River Avon in Hampshire (Figure 2.1). The research presented here focused exclusively on the River Wensum Demonstration Test Catchment.

Figure 2.1 Location map of the UK showing the three Demonstration Test Catchments: the Hampshire Avon, Wensum and Eden (Outram et al., 2014).

2.1.1 Wensum Demonstration Test Catchment

The River Wensum is a low gradient groundwater (chalk aquifer) dominated river which flows for approximately 78 km through the country of Norfolk, from its source in South Raynham to its confluence with the River Yare in Norwich (Figure 2.2). It was selected by Natural England as one of the 31 rivers in England to be designated as a Site of Special Scientific Interest (SSSI) and it was also selected as a Special Area of Conservation (SAC) under the European Habitats Directive and as a part of the European Union’s Nature 2000 network (Sear et al., 2006). Although the Wensum is recognised as of importance, a condition assessment carried out by Natural England in 2002 showed that the river was in unfavourable ecological condition mainly due to water quality issues and physical modification which limit its ecological and hydrological potential to support a chalk river habitat (Dils et al., 2009). Cooper et al. (2014) mentioned that 99.4% of the River Wensum protected habitat is in a declining state, primarily due to excessive sediment and nutrient loadings from agriculture.

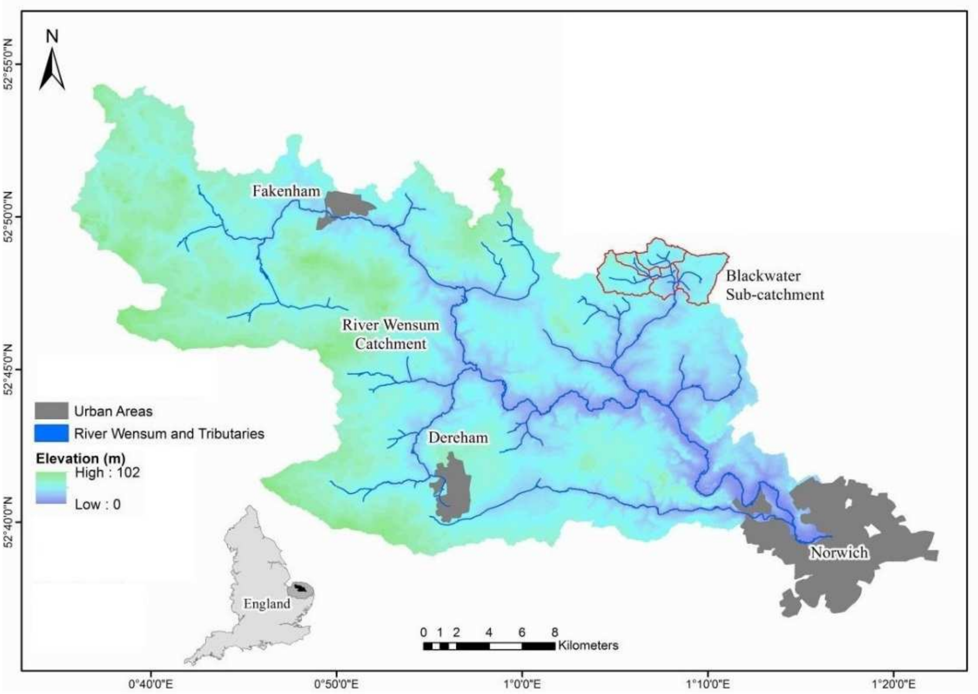

Figure 2.2 The River Wensum catchment , Norfolk.

2.1.1.1 Geology

The Wensum catchment is underlain by Cretaceous Chalk, the dominant solid geology of Norfolk. The Chalk is a white, fine grained, fissured limestone of very high carbonate fraction, deposited during the Upper Cretaceous. To the east of the catchment the Chalk is overlain by the Pleistocene Wroxham Crag formation of sands and gravels (Figure 2.3). A complex sequence of Quaternary strata over much of the catchment is formed of glacial tills, sands, gravel, alluvium, peat and river terrace deposits. Low-permeability tills in excess of 15 m in interfluve areas restrict infiltration to the underlying Chalk aquifer (Hiscock et al., 1996; Lewis, 2011; Outram et al., 2014). The Chalk is separated unconformably from Lower Cretaceous and Jurassic deposits by Carstone, a ferruginous sandstone, overlain in south Norfolk by the Gault Formation comprising grey mudstone (Arthurton et al., 1994).

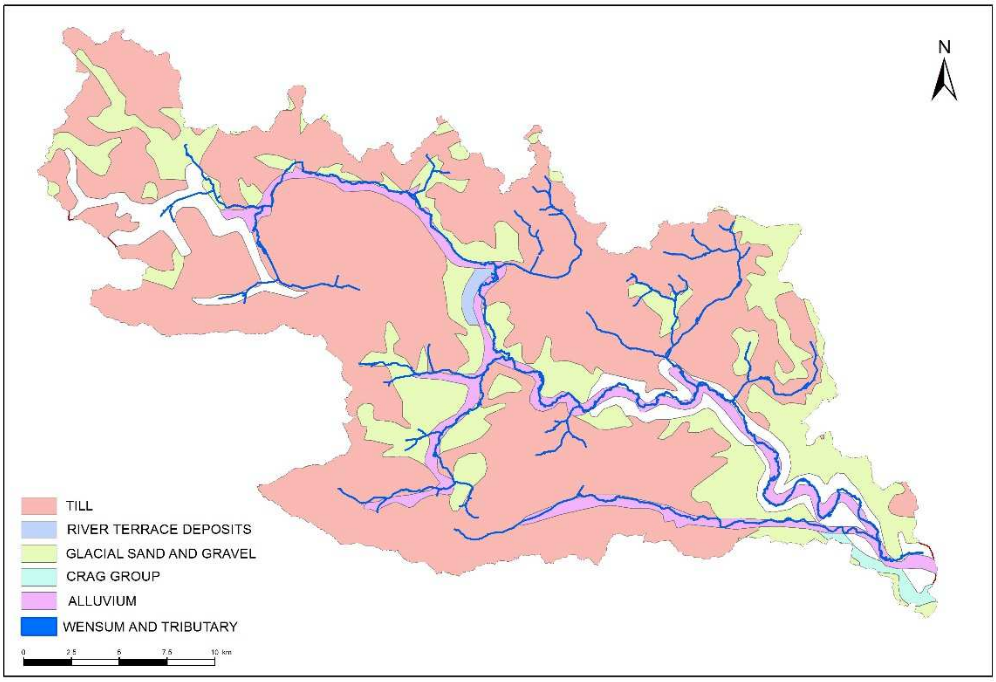

Figure 2.3 Superficial geology of River Wensum catchment (BGS, 2016).

2.1.1.2 Hydrogeology

The regional hydrogeology of Norfolk is dominated by the White Chalk sub-group that forms a major aquifer and provides a valuable groundwater resource due to its high porosity and permeability. Groundwater arising from the chalk aquifer dominates the flow regime of the River Wensum with a high baseflow index (BFI) of 0.82 at Fakenham (Ceh, 2016). The Chalk bedrock in the western part of the catchment is close to the surface, so the flow is derived primarily by groundwater discharge from the chalk aquifer. As the river progresses towards the east in the catchment the thickness of glacial sediments overlying the chalk increases and the contribution of surface water to the river increases with a baseflow index of 0.75 at Costessey Mill Norwich (Dils et al., 2009; Ceh, 2016). There is high spatial variation in transmissivity and storage capacity of the Chalk resulting from the distribution of overlying Pleistocene deposits (Hiscock et al., 1996). The transmissivity is less than 100 m^2 day^{-1} when the Chalk is confined under the till due to poor fissure development, whereas fissuring is more prevalent in the valleys where the Chalk outcrops and transmissivities can reach 2000 m^2/day. A very recent study by Outram et al. (2016) on the same catchment for the monitoring site F (Figure 2.5) summarized some farming and hydrological data for three years (Table 2.1). Total discharge volume into groundwater is between 134-234 mm with mean groundwater level (39.5-40.4 m) above sea level.

Table 2.1 Annual summary of fertiliser, hydrological, metrological nitrate and total phosphorus data for three hydrological years (2012, 2013, and 2014) for monitoring site F (see Figure 2.5). Standard deviation in parentheses (Outram et al., 2016)

	2012	2013	2014
Total nitrogen fertiliser application (kg N)	2.16×10^5	2.05×10^5	1.70×10^5
Total phosphorus fertiliser application (kg P)	2.48×10^4	3.63×10^4	2.16×10^4
Mean nitrate concentration (mg N L^{-1})	5.9 (1.6)	6.2 (1.3)	6.1 (1.7)
Total riverine nitrate-N load (kg N)	1.46×10^4	2.98×10^4	2.20×10^4
Mean total phosphorus concentration (mg P L^{-1})	0.09 (0.04)	0.09 (0.03)	0.09 (0.04)
Total riverine total phosphorus load (kg P)	2.29×10^2	4.10×10^2	3.33×10
Nitrate export coefficient	0.07	0.15	0.13
total phosphorus export coefficient	0.01	0.01	0.02
Mean total rainfall (mm)	683	633	706
Total discharge volume (m^3)	2.64×10^6	4.61×10^6	3.44×10^6
Total discharge volume (mm)	134	234	175
Annual runoff coefficient	0.2	0.37	0.25
Baseflow volume (m^3)	1.95×10^6	3.19×10^6	2.49×10^6
Baseflow index (BFI)	0.74	0.69	0.72
Mean groundwater level above ground surface	1.8 (0.3)	1.7 (0.5)	1.8 (0.3)

2.1.1.3 Climate

East Anglia is one of the driest regions in Britain. The Wensum catchment has a temperate maritime climate with a mean annual temperature of 10.1 ˚C during 1981-2010 and a mean annual total precipitation of 674 mm in Coltishall weather station near Reepham. Over the same period, mean monthly precipitation totals were highest during October and November (68.2 mm) and lowest during February (41.7 mm) (Meteorological Office, 2014), and moderate rainfall is maintained through the summer months with (55.5 mm) in July. Mean monthly minimum temperatures are lowest in February (1.3 ˚C) and August has the highest mean monthly maximum temperature (21.4 ˚C).

2.1.1.4 Topography

The topography of the Wensum catchment is relatively low with a maximum elevation above sea level of 95 m. The north and west of the catchment have the highest elevations. The majority of the catchment (78%) has a slope of 0-3 degrees, with only a small area of land area with slopes of 3-7 degrees and greater (Dils et al., 2009). These low slope angles mean that the River Wensum possesses relatively low energy, typical of rivers in East Anglia.

2.1.1.5 Soil

The Wensum catchment is characterised by rich loams, silts and sandy peats, which have a high potential for cultivation (Sear et al., 2006). Loam soil offers ideal moisture retention and drainage, preventing waterlogging because of the presence of a good balance between sand, silt and clay fractions in the soil. In the middle part of the catchment, coarse loams overlie clay, whereas lighter sandy loam soil is present in the western catchment. Soil texture has great spatial variability: the soil can change from medium clay loam to light sandy loam within a few metres. In the north west of the catchment, the soil comprises freely draining loams, with an increasing clay fraction and lower permeability in the central catchment and interfluves. Areas of well drained sandy soil occur in the lower catchment. The upper river valley soils are free draining loam with sand, giving way to peat soils. Figure 2.4 shows the soil series in the catchment. The Adventurers series forms earthy peat soils with amorphous and semi-fibrous peat soils formed mainly in reed and sedge peat, often with wood fragments from carr. The Isleham series forms humic sandy gley soils which consist of sandy and peaty soils on low-lying land affected by groundwater. The Gresham series is a stoneless slowly permeable seasonally waterlogged coarse loamy and silty over clayey soil. The Newmarket series forms a coarse loamy soil over chalk or chalk drift. The Wick series which is predominant in the Blackwater sub catchment (Figure 2.2) is a deep well drained coarse loamy typical brown earth, intermixed with gleyic brown earth (Barrow series) and typical brown sands.

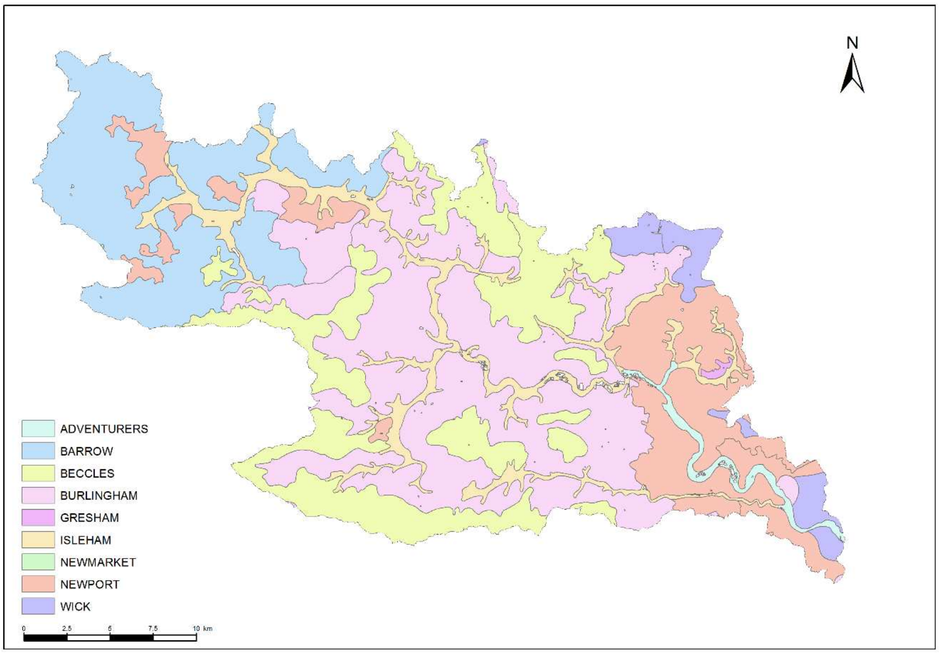

Figure 2.4 Soil series of the Wensum catchment (NATMAP, 2009).

2.1.1.6 Land use

East Anglia contains some of England's best agricultural land due to its loam soil and gentle topography. Over 85% of the land in Norfolk is used for farming, and of this 66% is used for growing crops or is left fallow. The largest land area, as it shown in Figure 2.5, is used for cereal crop production, followed by sugar beet and oilseed rape, horticulture and potatoes, whereas livestock, dairy, pig and poultry operations together account for 22% of farms (Defra, 2006). There are few cities and towns in the catchment. The largest is Norwich with its urban population of 214,000, followed by Dereham with a population of approximately 19,000 and Fakenham with a population of over 7,000 (Office for National Statistics, 2012).

2.1.1.7 Experimental Area

Of the 20 sub-catchments in the Wensum, the lowland Blackwater sub-catchment in the north east of the catchment covers an area of 60 km^2. The upper 20 km^2 of this sub-catchment represents the area intensively monitored as a part of the River Wensum Demonstration Test Catchment project. In this experimental area, the Blackwater sub-catchment is divided into six mini-catchments A to F (Figure 2.5), each of which has a bankside kiosk at its outlet monitoring parameters including pH, turbidity, temperature, ammonium, dissolved oxygen and electrical conductivity at 30 minute resolution using ion selective electrode sensors on a YSI 6600 V2 sonde. Each monitoring site also has a pressure transducer located in a stilling well which records river stage every 30 minute. Two kiosks (E and F) are additionally equipped with Hach Lange nutrient analysers including a Nitratax Plus SC probe which measures nitrate concentrations via an optical sensor and Phosphax Sigma wet chemistry analyser in combination with a Sigmatax SC sampling and homogenisation unit to measure phosphorus (Outram et al., 2014). Each kiosk, including a seventh kiosk at site M nested within mini-catchment A, also encompass an ISCO automatic water sampler (Teledyne ISCO, Lincoln, NE) containing 24, 1 L polypropylene sample bottles. Weather stations at A and D record precipitation at 15 min intervals via tipping-bucking rain gauges, alongside measurements of temperature, wind speed, humidity and solar radiation. Mini-catchment A, the focus area of this study, is ~40 m above sea level and covers an area of 5.4 km^2 with a gentle slope of ~0.37˚. This arable area is under intensive farming with 92% for cropping (with wheat, barley, sugar beet, oilseed rape and spring beans in rotation), with 5% grassland, 2% woodland and 1% urban area.

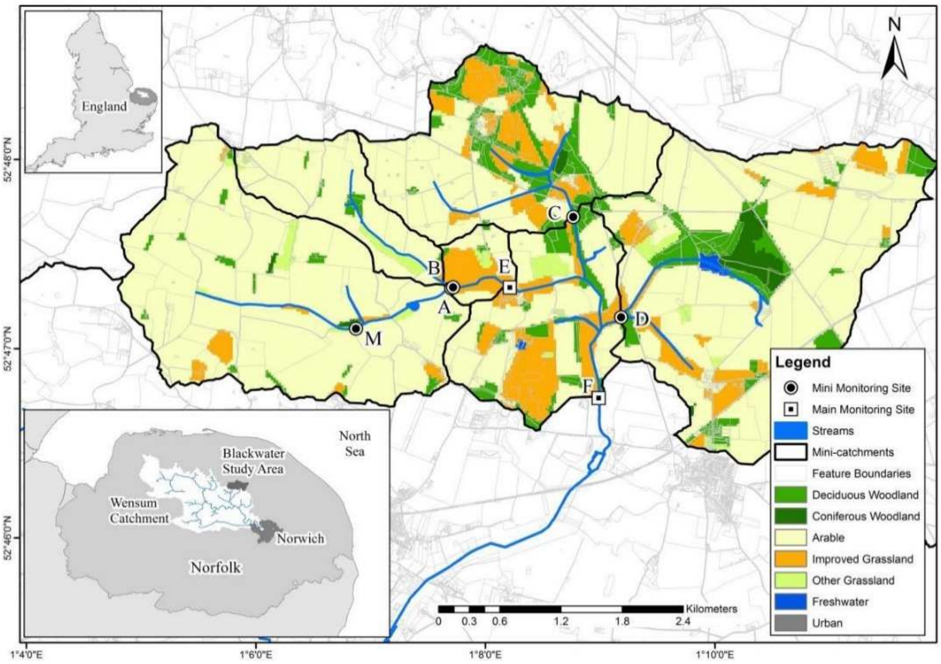

Figure 2.5 Land use across catchments the River Blackwater. It also shows the locations of mini kiosk (A B, D, and M) and kiosk (E and F).

2.2 Experimental design

In 2013, nine fields covering 143 ha of arable land were identified for trialling of winter cover crop and reduced tillage practices aimed at reducing diffuse nutrient losses into the River Blackwater (Figure 2.6, Table 2.2). These nine fields were divided into three mitigation measures treatments, with each treatment sown with the same crop and same fertiliser application rate during the 2013/2014 (spring beans; 0 kg N ha^{-1}, 30 kg P ha^{-1}, 55 kg K ha^{-1}) and 2014/2015 (winter wheat; 220 kg N ha^{-1}, 22 kg P ha^{-1}, 85 kg K ha^{-1}) farm year (September to August). Two fields were kept as control and were cultivated by mouldboard ploughing to a 25 cm depth prior to sowing. Oilseed radish (*Raphanus sativus*) cover crop (seed density = 18 kg ha^{-1}) was sown in treatment (cover crop & reduced tillage) and treatment (cover crop & direct drill) using a Lemken cultivator in late-August 2013. treatment (cover crop & reduced tillage) was then underwent reduced tillage to a depth of 10 cm using Väderstad Carrier and topdown cultivator prior to sowing with Rapid drill and treatment (cover crop & direct drill) underwent direct drilling with no inversion using Väderstad Seed Hawk. A winter crop (winter wheat) was grown in the second year (2014/2015) so cover crop was not sown in this year but reduced tillage and direct drilling practices continued in (cover crop & reduced tillage) and (cover crop & direct drill) treatments, respectively.

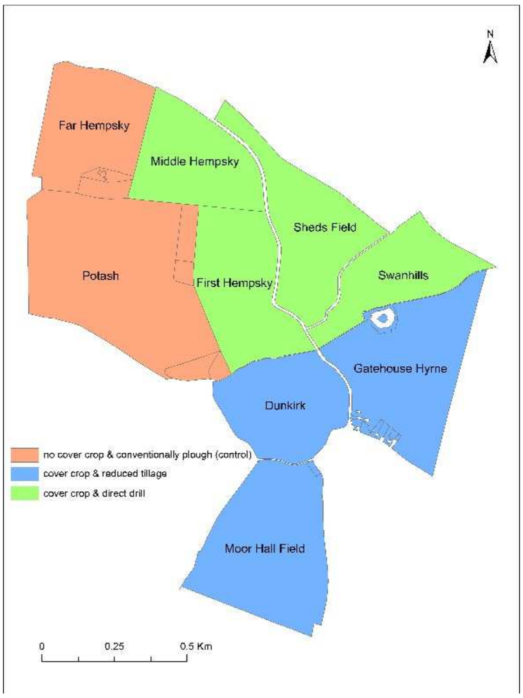

Figure 2.6 Map of the study area and the experimental treatments.

Table 2.2 Description of the experimental treatments

Treatment	Field name	Area (ha)	2013/2014					
			Cover crop	Tillage	Crop	N kg N ha^{-1}	Yield t ha^{-1}	Cultivation systems
No cover crop & conventional plough (control)	Far Hempsky	13.8	No	Plough	SB	0	5.49	Conventional agriculture
	Potash	26.8	No	Plough	SB	7	5.40	
Cover crop & reduced tillage	Gatehouse Hyrne	17.3	Yes	Reduced	SB	38	6.44	Conservation agriculture
	Dunkirk	12.9	Yes	Reduced	SB	30	5.86	
	Moor Hall Field	20.4	Yes	Reduced	SB	0	6.97	
Cover crop & direct drill	Swanhills	10.4	Yes	DD	SB	26	5.19	Conservation agriculture
	Sheds Field	14.9	Yes	DD	SB	28	6.24	
	First Hempsky	14.1	Yes	DD	SB	34	5.99	
	Middle Hempsky	11.8	Yes	DD	SB	7	6.55	

Treatment	Field name	Area (ha)	2014/2015					
			Cover crop	Tillage	Crop	N kg N ha^{-1}	Yield t ha^{-1}	Cultivation systems
No cover crop & conventional plough (control)	Far Hempsky	13.8	No	Plough	WW	226	13.5	Conventional agriculture
	Potash	26.8	No	Plough	WW	228	13.1	
Cover crop & reduced tillage	Gatehouse Hyrne	17.3	No	Reduced	WW	221	11.5	Conservation agriculture
	Dunkirk	12.9	No	Reduced	WW	219	12.8	
	Moor Hall Field	20.4	No	Reduced	WW	229	13.5	
Cover crop & direct drill	Swanhills	10.4	No	DD	WW	219	10.5	Conservation agriculture
	Sheds Field	14.9	No	DD	WW	227	10.4	
	First Hempsky	14.1	No	DD	WW	229	12.9	
	Middle Hempsky	11.8	No	DD	WW	222	12.2	

Note: SB: spring beans, WW: winter wheat

2.3 Methods

2.3.1 Field techniques

2.3.1.1 Soil sampling

Soil sampling was carried out several times throughout this study. The first soil sampling was carried out in the nine fields from 13th to 22nd May 2013 by a team of UEA students (Table 2.3). Soil samples were collected by Dutch auger at three soil depths (0-30 cm, 30-60 cm, 60-90 cm) at 12 sites in each field in a ‘W’ layout as recommended by (Pennock et al., 2007). The soil samples were not bulked because the aim was to understand the variability of soils in the nine experimental fields. In total, 324 soil samples were collected (i.e. 9 fields x 12 sites x 3 depths = 324) (Figure 2.7). Moist, well-mixed samples were placed into sealed plastic bags and transferred back to the university. The samples were then placed in a drying rack and allowed to air dry for 7 days. After removal of course stones, portions of each homogenised and dried sample was then put in sealed plastic bags for archiving, whilst the rest was gently ground with pestle and mortar and passed through a U.S. No.10 (2 mm opening) sieve (Gelderman and

Mallarino, 2012). These sieved soil samples were used for determining soil texture and soil organic matter.

Table 2.3 Summary of soil samplings date and measurements during this study

Time of sampling	May 2013	Sep. 2013	Feb. 2014	May & Jul. 2014	Feb. 2015	Jun. & Jul. 2015
Who conducted	UEA Team	ADAS	ADAS	UEA team	UEA team	UEA team
SMN (NO_3, NH_4, Available N)		4 sites/field* 3 depths	4 sites/field* 3 depths	4 sites/field	4 sites/field	4 sites/field
Top soil (pH, P, K, Mg, OM)		4 sites/field	4 sites/field		4 sites/field	4 sites/field
Soil texture	12 sites/ field* 3 depths	4 sites/field				
Physical properties (BD, IR, PR)	12 sites/field			4 sites/field		4 sites/field

NOTE: SMN: Soil Mineral Nitrogen, OM: Organic Matter, BD: Bulk density, IR: Infiltration rate, PR: Penetration resistance

A second round of soil sampling was carried out by ADAS after completion of the harvest between 27th August and 3rd September 2013. Using the farmer’s expertise, four different soil types were identified in each field. Samples were collected from each soil type in each field. The selected sampling points within each soil type were located in the same location as earlier soil sampling wherever possible. Samples were taken in two different ways according to the measurement of interest. Samples for pH, K, Mg, P, SO_4 and organic matter measurement were taken in the topsoil (0-15 cm) by hand auger in 12 locations within 2 metres around each site. These 12 samples were then bulked together to provide one representative sample for a given site (i.e. 9 fields x 4 sites = 36 samples). Samples for soil mineral nitrogen (SMN) and soil texture were collected with a Hydrocare powered auger in two concentric circles at 12 locations within 10 metres around each site. The soil cores taken from the Hydrocare drill were divided into three depths: 0-30 cm, 30-60 cm and 60-90 cm. The 12 samples for a certain depth were also bulked to provide one representative sample at a given site (i.e. 9 fields x 4 sites x 3 depths = 108 samples for SMN and soil texture measurement).

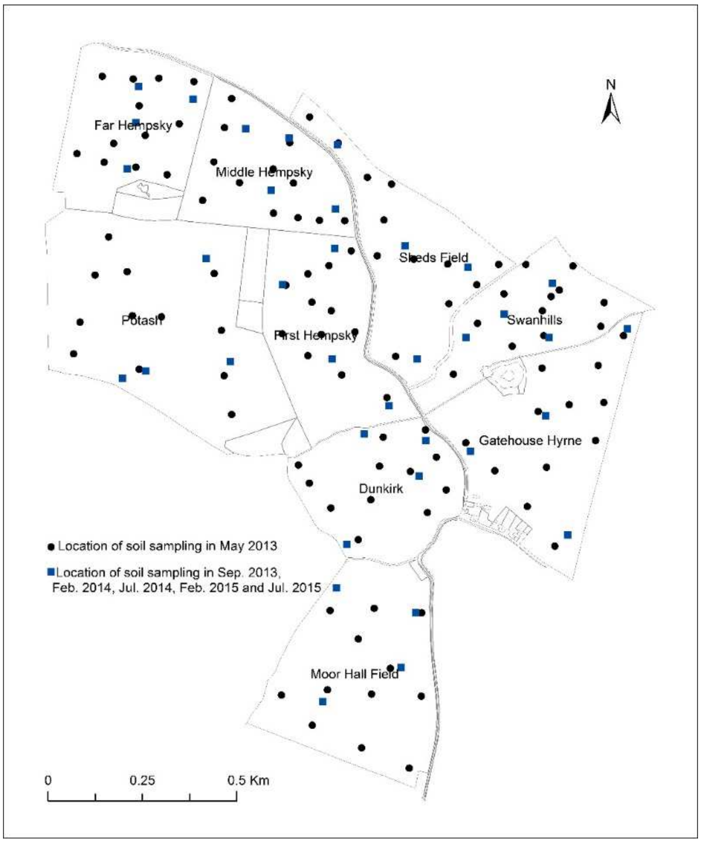

Figure 2.7 Locations and dates of soil sampling that carried out during this study.

The collected samples were placed in sealed plastic bags and sent to Natural Resource Management (NRM) laboratories for analysis. The total number of soil samples collected in the nine fields during both sampling rounds combined was 432 samples (i.e. 9 fields x 16 sites x 3 depths = 432 samples). On the 4th and 5th February 2014, soil samples were again collected by ADAS at four sites per field using the method described above. A further three soil sampling campaigns were carried out by a team of students from UEA in July 2014, February 2015 and July 2015, with these soil samples collected from the same four sites per field in topsoil (0-15 cm) only.

2.3.1.2 Field drain and stream water sampling

Like most of the arable land in the Blackwater sub-catchment, the nine measures fields are extensively under-drained by a dense network of clay and plastic agricultural field drains (or tile drains) installed at depths of 100-150 cm during numerous phases of land drainage over the past 60-70 years. Water samples from 13 of these field drains as well as from four in-stream locations (A, B, E and M) (Figure 2.8) were collected at weekly intervals for water quality analysis (i.e. nutrients and major ions) between April 2013 and April 2015. The exception to this was between April 2014 and September 2014 when drain samples were collected every two weeks and stream samples collected every month. Samples were collected in one litre polypropylene bottles after the bottles had been flushed repeatedly with water from the sampling site. Field drain water flow was measured in triplicate at each drain sampling site using a measuring cylinder and stopwatch. Rainfall data were obtained from a tipping bucket rain-gauge installed in the study area next to drain D05R.

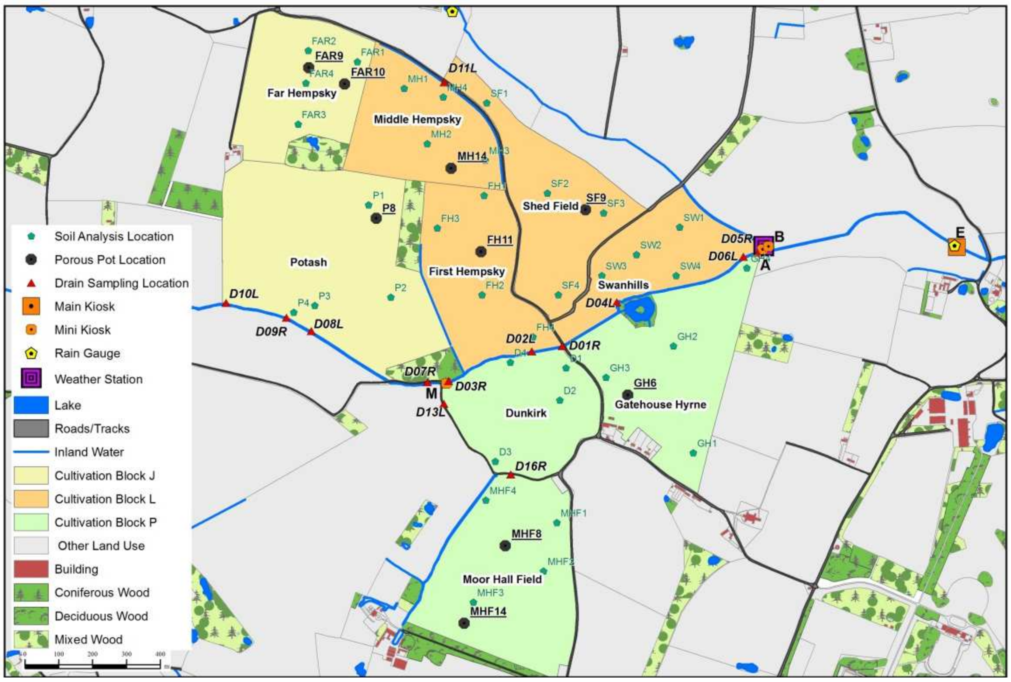

Figure 2.8 Location map of water samples for N species and dissolved N_2O analysis.

2.3.1.3 Porous pot sampling

Based on information gained from both the soil survey (144 sampling sites in the experimental area, i.e. 16 sampling sites per field) and from an electrical conductivity scan (Appendix A2) of the study area, nine locations were chosen for installing porous pots to capture soil water. Three locations of different soil textures including sandy loam, sandy clay loam and clay loam in each treatment were selected to install porous pots (Figure 2.9, Table 2.4). Locations were chosen on the basis that soil texture did not change along the profile (90 cm) and high clay content soil should be located in a high electrical conductivity zone. This is because clay soils have numerous small water-filled pores that are quite continuous and usually conduct electricity better than sandier soils. Ten porous pots, all located on a straight line and one metre distance apart from each other, were installed in mid-December 2013 at each location at a depth of 90 cm and at a 45 degree angle by ADAS using the hydrocare drill. Thus, 90 porous pots were installed in total for soil water sampling.

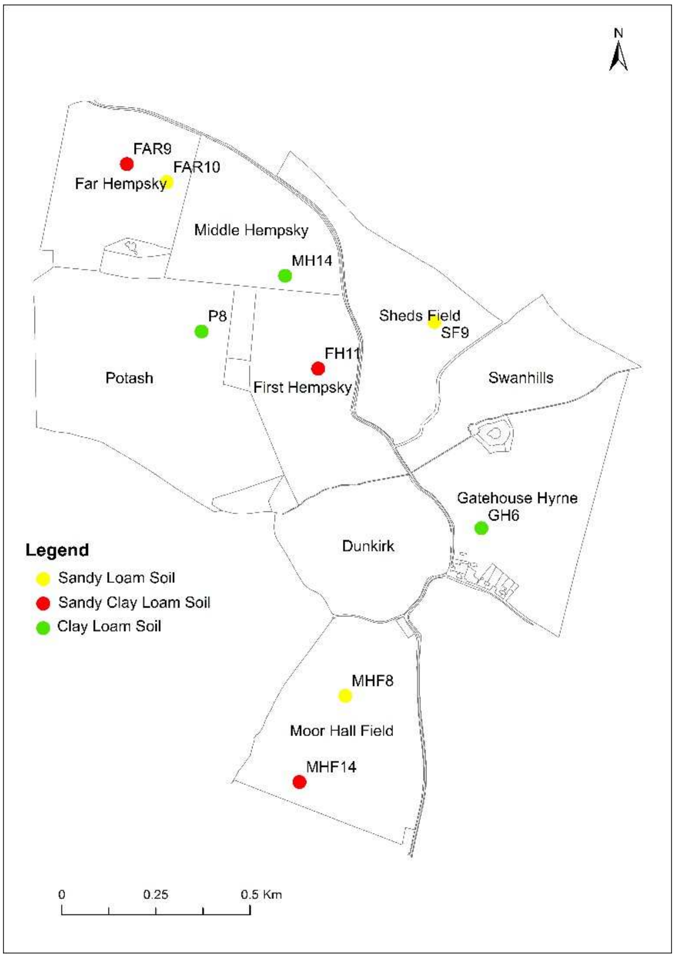

Figure 2.9 Locations of installed porous pots. Each location had 10 porous pot installed in December 2013.

Table 2.4 Soil particle size distribution at three depths for locations of porous pots

Sample name	Depth (cm)	Sand %	Silt %	Clay %	Texture
FAR10	0-30	58	33	9	Sandy Loam
	30-60	73	21	6	Sandy Loam
	60-90	78	12	10	Sandy Loam
SF9	0-30	59	26	15	Sandy Loam
	30-60	60	23	17	Sandy Loam
	60-90	61	23	16	Sandy Loam
MHF8	0-30	54	31	15	Sandy Loam
	30-60	58	27	15	Sandy Loam
	60-90	61	22	17	Sandy Loam
FAR9	0-30	45	39	16	Sandy Silt Loam
	30-60	49	34	17	Sandy Silt Loam
	60-90	54	19	27	Sandy Clay Loam
FH11	0-30	54	26	20	Sandy Clay Loam
	30-60	49	20	31	Sandy Clay
	60-90	51	19	30	Sandy Clay Loam
MHF14	0-30	52	26	22	Sandy Clay Loam
	30-60	56	22	22	Sandy Clay Loam
	60-90	53	24	23	Sandy Clay Loam
P8	0-30	44	31	25	Clay Loam
	30-60	43	30	27	Clay Loam
	60-90	41	29	30	Clay Loam
MH14	0-30	46	30	24	Clay Loam
	30-60	44	28	28	Clay Loam
	60-90	40	27	33	Clay Loam
GH6	0-30	36	34	30	Clay Loam
	30-60	26	42	32	Clay Loam
	60-90	28	39	33	Clay Loam

Note: The soil textures were classified according to the UK Soil Survey of England and Wales texture triangle.

The flexible sampling tubes of the porous pots were left exposed above the ground surface from mid-December 2013 to end of January 2014. After field capacity was reached at the end of January, the first sample batches from the porous pots were collected on the 4th & 5th February 2014. In order to obtain a fresh water sample, the pots were first flushed out with a vacuum pump. They were then left under vacuum for a few hours to drawn in water from the surrounding soil (Figure 2.10). To maximise the water volume provided by pots, each pot was pumped three times. However, some pots provided extremely low amounts of water insufficient for analysing all parameters. Second round of sampling took place on the 28th & 29th April 2014. After sampling, the flexible tubes were buried to 50 cm depth to avoid damage by farm machinery during harvesting and cultivation. One final round of porous pots samples were collected on the 25th & 26th February 2015. Thus, porous pots were sampled three times during this study.

Figure 2.10 Photographs of porous pot sampling on 4th February 2014 at site FH11.

2.3.1.4 N_2O sampling

Water samples for N_2O analysis were collected weekly between April 2013 and April 2015 from the same 13 field drains and four river locations (A, B, E and M) described in section 2.3.1.2. Samples were obtained using 20 ml glass syringes (SAMCO) with a three-way stopcock attached to the syringes by Luer-Lock fittings (Figure 2.11). Syringes were flushed three times with water from the sampling point and any air bubbles contained in the syringes were expelled before the final sample was taken. No preserving mercury (II) chloride ($HgCl_2$) was added to the sample, since it has potential to change the original N_2O content of the water through the chemical reaction. Samples were returned to the laboratory and stored at 4°C prior to analysis. Samples were generally analysed within 48 hours of sampling, thus minimising the risk of sample degradation.

Figure 2.11 Sampling for N_2O measurement from field drains (A) and stream (B).

2.3.2 Lab measurement

2.3.2.1 Soil Analysis

Particle size distribution

Particle size distributions for all 432 soil samples were determined by laser diffraction by NRM laboratories. The soil samples were suspended in water and passed through a flow cell. The flow cell is positioned in the path of a laser beam and the particles of soil passing through the cell causes the laser light to be diffracted. The amount of light that is diffracted is dependent upon the size of the particle in its path. Small particles cause greater diffraction than large particles. By measuring the diffraction pattern of the laser beam it is possible to predict the size and relative population of particles in the sample. Principles and details of determining particle size distribution by laser diffraction method can be found in (Ma et al., 2000; Wedd, 2003).

Soil mineral nitrogen (SMN)

The SMN content was also measured by NRM laboratories. The soil was chopped and mixed to obtain a homogeneous sample and stones were removed. Soil nitrate concentrations were determined calorimetrically after shaking a fresh portion of each samples with 2 mol potassium chloride (KCl) to extract the mineral N fractions and reacting with sulphanilamide ($C_6H_8N_2O_2S$) and n-(1-Naphthyl)ethylenediamine ($C_{12}H_{14}N_2$). Once in solution the nitrate-N, nitrite-N and ammonium-N can be measured calorimetrically (NRM, 2013).

Macronutrients

Soil organic carbon was determined for the first round of samples (324 samples) by a CHN instrument in the UEA laboratories. The soil sample was acidified with sulphurous acid to remove all inorganic carbon. After drying, a tiny portion (4 mg) was placed in a capsule and measured. Soil organic matter of subsequent samples was determined by loss-on-ignition (LOI) by NRM laboratories. The organic matter was destroyed by dry combustion at 430 °C and the loss in weight of the sample was reported as the percentage organic matter content. Organic matter can be determined by using the Van Bemmelen factor (1.724, which is based on the assumption that organic matter contains 58 percent organic carbon), so organic matter =organic carbon x 1.724 (Buol et al., 2011).

Available phosphorus was determined by shaking the soil with 0.5 M of sodium bicarbonate solution at pH 8.5 and 20°C. Available potassium was determined by shaking the soil with 1 M ammonium nitrate at 20 °C for 30 minutes. After filtration, the concentration of potassium in the extract is determined by flame photometry. Available magnesium was extracted from the soil by shaking with 1 M ammonium nitrate at 20 °C for 30 minutes. After filtration, the concentration of magnesium in the extract is determined by atomic absorption spectroscopy. The available sulphate was extracted from the soil under controlled conditions using a phosphate buffer extracting solution at a 1:2 ratio. The filtered extract of the sample was analysed by Inductively Coupled Plasma Emission Spectroscopy (ICP-ES) (NRM, 2013).

Physical properties

For soil bulk density measurement a core sampler was used to collect undisturbed cylindrical soil samples (7.6 cm long x 4.27 cm in diameter) from each soil sampling site at the ground surface. Bulk density (ρ_b) was measured as the mass of oven dried soil (105°C for 24 h) per volume of core (g cm^{-3}) (Ward and Trimble, 2003). Each measurement was replicated three times. A Minidisk tension infiltrometer with a disk radius of 2.22 cm (Decagon, 2007) was used to measure the infiltration rate under unsaturated conditions created using a pressure head of -2 cm. Measurements were performed at the soil surface. The soil surface was carefully levelled before the tension disk infiltrometer tests. Detail of the procedure can be found in (Decagon, 2007). A hand-pushing penetrometer (Eijkelkamp) with a cone diameter of 11.28 mm (cone number 1) was used for the measurement of soil compaction. The area of the cone base was 1 cm^2 and the shaft diameter was 8 mm. The penetration resistance measurements were made by pushing the penetrometer vertically into the soil at an approximate speed of 2 cm s^{-1}. Penetration resistance measurement was replicated three times. Soil temperature and soil moisture were both measured by the soil moisture probes AddIT series 4 (ADCON) that were installed in January 2013 close to the locations where porous pots installed (Figure 2.9). The probes record soil moisture every 15 minutes in nine soil depths (10-90 cm) and soil temperature every 15 minutes in three soil depths (15,45 and 75 cm).

2.3.2.2 Soil solution analysis

Field drain and porous pot nitrate concentration was determined by ion chromatography using a Dionex ISC 2000 instrument. A sodium nitrate ($NaNO_3$) standard (0.50-7.50 mg L^{-1}) was used for calibration. Instrument accuracy (±0.2 mg L^{-1}) was determined by analysing a certified reference material (NO_3^- = 214 µmol L^{-1}) with each sample batch with detection limit of 0.01 mg N L^{-1}. A continuous flow analyser (Skalar San++) was used for measuring ammonium with a detection limit and accuracy of 5.98 µg N L^{-1} and ± 4.57 µg N L^{-1}, respectively. It was also used for measuring nitrite with detection limit and accuracy of 0.96 µg N L^{-1} and ± 1.52 µg N L^{-1}, respectively.

2.3.2.3 N_2O measurement

Figure 2.12 shows a schematic diagram of the gas chromatogram (GC) and sample preparation line used for N_2O analysis. Samples are directly injected into a 100 cm^3 glass purging column which is flushed with helium CP grade flowing at a rate 30 ml min^{-1} (Table 2.5). The purging gas stream is passed through various traps connected by ¼ inch stainless steel tubing and stainless steel fittings. Water vapour is removed from the purging gas by passing through a reverse-flow Nafion dryer (in-line after the purge tower). The drying gas, which flows around the membrane containing the purge gas flow and in the opposite direction (not shown in the schematic diagram), should be approximately 2-3 times the flow rate of the purge gas. This dryer was found to be extremely effective. Additionally, a 2.8 cm^3 glass tube was filled with magnesium perchlorate $Mg(ClO_4)_2$ granules to trap any moisture that might pass through the reverse-flow Nafion dryer. Carbon dioxide (CO_2) is taken out of the gas stream in a 2.8 cm^3 glass tube with Carbosorb. Both, water and CO_2 are electronegative compounds and can potentially interfere with the nitrous oxide GC analysis, hence the need for them to be removed prior to sample injection. Detail of this method can also be found in (Mühlherr, 1997; Mühlherr and Hiscock, 1998; Outram and Hiscock, 2012).

The water samples were purged for 20 minutes to ensure a nitrous oxide recovery rate of >99%. Nitrous oxide is captured on glass beads packed into a 1/8 inch stainless steel loop in a trap held above liquid nitrogen at -190°C. The N_2O is then remobilised by submerging the loop in a hot bath at approximately 95°C. The sequence of events for analysing a sample once it becomes trapped inside the sample loop is as follows: 1- Start data capture, 2- Throw nupro (from on - off) valve (in line before sample valve), 3- Throw 6-port Valco sample valve, 4- Replace liquid N_2 flask with hot bath (turn off liquid N_2 heater), 5- Re-throw nupro valve (from off – on), 6- Leave for 30 secs, 7- Re-throw Valco sample valve.

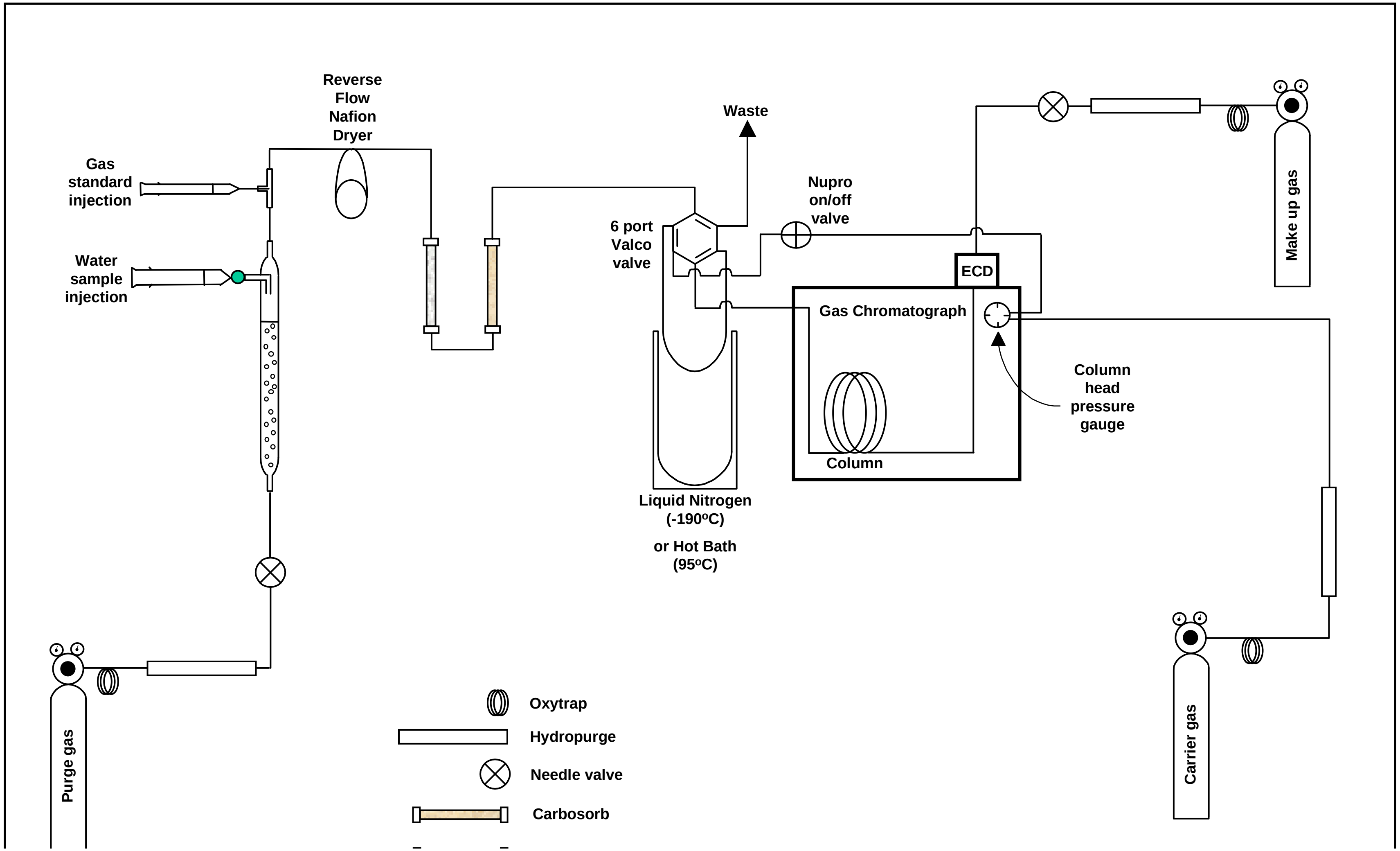

Figure 2.12 Schematic diagram of the gas chromatogram sample preparation line used for nitrous oxide (N_2O) analysis.

Table 2.5 A summary of the type of gas, the regulator pressure and the approximate flow rate

	Gas type	Regulator pressure (psi)	Flow rate (mL min^{-1})
Carrier gas	5% CH_4/bal Ar	70	15
Make-up gas	Research Grade N_2	3	20
Purge gas	CP He	10	30
Dryer gas	Zero N_2	1	65
Isotope storage	CP He	1	n/a

The volume of the analysed sample was determined gravimetrically. Syringes were weighed before and after sample injection with the difference in weight (g) being the volume of the analysed sample in ml. To flush the purging tube and clean it after each sample, an initial run of ~4 ml of sample water was injected into the purging tube and left for 4 minutes before being poured out.

Sample chromatogram

The chromatogram below (Figure 2.13) shows the typical the N_2O peaks obtained from standard. The black, blue and green peaks are for 20 ml, 60 ml and 120 ml of a 1 ppm N_2O standard, respectively. The fluctuation in the baseline between 1 and 2 minutes is due to the throwing of the sample valve.

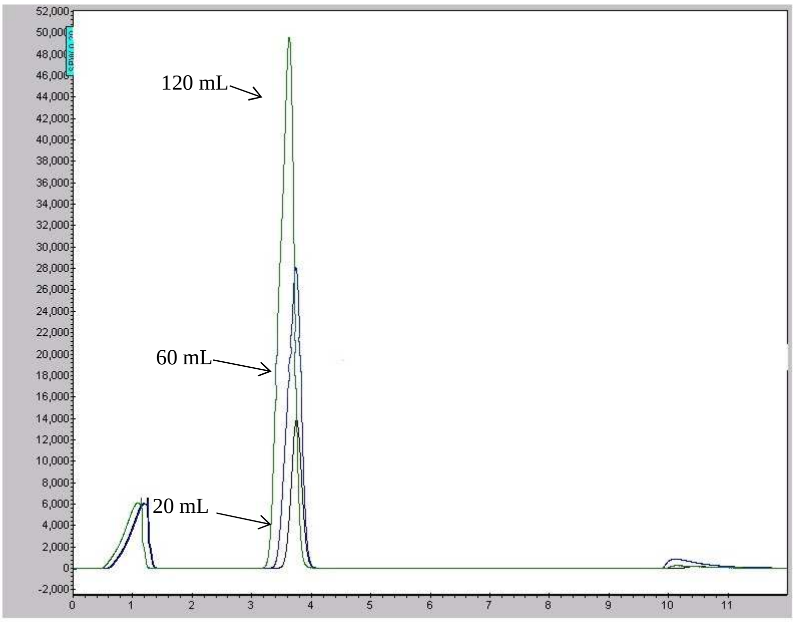

Figure 2.13 Typical chromatograms for three different volumes of N_2O standard.

Calibration

A nitrous oxide standard of 1 ppm (AIR LIQUIDE) was used to calibrate the analytical system. An aliquot of the standard (10 to 100 mL) was injected with a 10 ml airtight glass syringe (SGE) into the preparation line through a septum (Figure 2.14) prior to being trapped and then desorbed into the GC. The quality of the standard was checked by injecting about 10 mL of ambient air on several occasions. Nitrous oxide concentrations for these air samples ranged from 300 to 350 ppb according to the calibration line, which showed that the standard was of reliable quality. In each calibration, prior to injecting the standard, several blanks were analysed to ensure that there was no contamination or gas leaking in the preparation line in the GC prior to sample analysis.

Table 2.6 shows how different volumes of the N_2O standards can be used to generate a calibration model. Concentrations of nitrous oxide in the field drain samples changed over time, with generally lower N_2O concentrations in the summer compared with autumn and winter. Hence, different volumes of standard were injected according to season.

Table 2.6 An example of different standard volumes used to generate an N_2O calibration

Date	Run no.	(P) Air pressure (Pascal)	Purge Time (min)	Standard conc. (ppm)	(T) Lab temp (ºC)	Volume of standard injected (ml)	(V) Volume of standard m^3	Number of molar injected (PV/RT) (M)	Number of Injected (nM)	Area (Arb. unit)	Area-Blank (Arb. unit)
16.01.14	2814	101325	10	blank	18	0	0	0	0	27.9	0
16.01.14	2815	101325	10	blank	18	0	0	0	0	21.5	0
16.01.14	2816	101325	10	blank	18	0	0	0	0	26.1	0
16.01.14	2817	101325	10	1	18	20	2E-11	8.37134E-10	0.837134	2399.8	2374.633
16.01.14	2818	101325	10	1	18	40	4E-11	1.67427E-09	1.674269	4622.5	4597.333
16.01.14	2819	101325	10	1	18	60	6E-11	2.5114E-09	2.511403	6909.7	6884.533
16.01.14	2820	101325	10	1	18	10	1E-11	4.18567E-10	0.418567	1358.2	1333.033
16.01.14	2821	101325	10	1	18	80	8E-11	3.34854E-09	3.348537	9199.6	9174.433
16.01.14	2822	101325	10	1	18	100	1E-10	4.18567E-09	4.185672	11168.5	11143.33
16.01.14	2823	101325	10	1	18	10	1E-11	4.18567E-10	0.418567	1246.5	1221.333
16.01.14	2824	101325	10	1	18	10	1E-11	4.18567E-10	0.418567	1391.9	1366.733

R = gas constant = 8.314472 Jmol^{-1}K^{-1}, Temperature (kelvins) =Temperature (celsius+273.15)

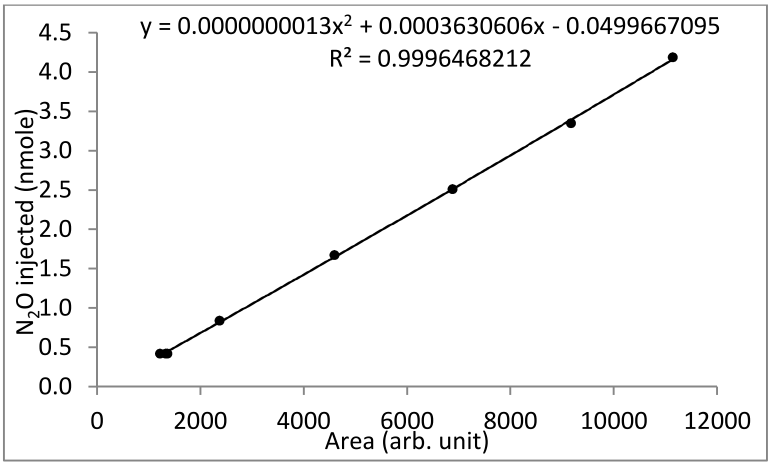

Figure 2.14 Example of calibration line for the data in Table 2.6.

Precision, accuracy and limit of detection

In each calibration, a given volume of standard, normally 10 ml, or water sample was injected three times to obtain the instrument precision. The precision of standards ranged from 0.2% to 5.8% with an average variance of 1.2% (determined from repeat analysis of 3 constant volumes of standard). The precision of N_2O concentration in the water samples was found to vary between 0.3% and 3.0% with an average variance of about 1.5% (determined from repeat analysis of 8 water samples taken at the same time and place). This is very comparable to the precision (2.5%) calculated by Outram and Hiscock (2012) who used the same instrument. Instrument accuracy was measured by calculating the differences between measured N_2O values (calculated from the calibration equation) and actual N_2O content of a standard divided by the actual N_2O content of the standard. The accuracy calculated here for the entire study was ±3%.

(Miller and Miller, 1993) discuss the definition of detection limit. The definition used here is that the limit of detection is an analyte concentration that gives a signal equal to the blank signal plus three standard deviations of the blank. Calculations using a number of blanks (and calibrations to convert the blank signal to a corresponding N_2O concentration) suggest that the limit of detection was about 0.13 nM (0.004 µg N L^{-1}). Mühlherr (1997) who utilised the same instrument quoted a limit of detection of 0.01nM. This is approximately 13 times lower than the value calculated in this study, but there is no definition of the limit of detection Mühlherr used.

2.3.3 Statistics analysis

The independent-sample t-test was used to examine the significant difference between two groups of data. A combination of one-way analysis of variance (ANOVA) with the assumptions of approximately normally distributed data and post-hoc tests (LSD) were used to compare the levels of significance among several treatments (three or more groups of data). For these tests, a significance level (p-value) of 0.05 was used. All statistical analyses were performed using SPSS for Windows® (version 20.0). Variations in data are given as standard errors. Microsoft Excel was used to plot the data and to calculate the fitted regression lines and their associations.

Chapter 3: EFFECTS OF COVER CROP AND REDUCED CULTIVATION ON SOIL QUALITY

3.1 Introduction

The need for sustainable management approaches to improve soil quality and increase agricultural production has been stressed by many studies in light of an increasing world population and climate change (Komatsuzaki and Ohta, 2007; Abdollahi and Munkholm, 2014). Soil quality is defined by Karlen et al. (1997) as the capacity of a specific kind of soil to function, within natural managed ecosystem boundaries to sustain plant and animal productivity, maintain or enhance water quality, and support human health and habitation. Inappropriate use sometimes causes soil degradation which is one of the most critical threats facing mankind which not only lessens the productive capacity of an ecosystem but also effects overall climate. It is estimated that soil degradation has already affected more than two billion hectares of land globally at a rate of 8-9 million ha y^{-1} (Alam, 2014). Mechanisms that cause soil degradation include physical, chemical, and biological processes. Major physical processes are declines in soil structure leading to crusting, compaction, erosion, desertification, environmental pollution and unsustainable use of natural resources. Important chemical processes include acidification, leaching, salinisation and reduced fertility. The most important biological processes include a reduction in total biomass carbon and declines in land biodiversity (Eswaran et al., 2001). Thus, agriculture contributes significantly in deteriorating soil quality. However, nowadays, people have come to understand that agriculture should not only be high yielding, but also sustainable and thereby new approaches such as conservation agriculture are developed.

Elements of conservation agriculture include conservation tillage, crop rotation and cover crops. Several studies have assessed the effects of different conservation agriculture elements on soil quality individually, but few studies have assessed the impacts of conservation tillage combined with cover crops. Therefore, the novel aspect of this study was to assess the effects of alternative cultivation regimes combined with cover crop on different parameters of soil quality. Conservation tillage, such as direct drill, is globally accepted as an effective approach for protecting the soil from structural degradation and erosion (Abdollahi and Munkholm, 2014). It has been commonly observed that different tillage systems have affected soil properties. Conservation tillage has been shown to increase aggregate stability, organic matter content, biological activity and soil strength (Du Preez et al., 2001; Six et al., 2002; Mijangos et al., 2006). On the other hand, increased bulk density and increased accumulation of phosphorus (P) and acidity near the soil surface has been reported as disadvantages of direct drill (Andraski et al., 2003; Dolan et al., 2006; Gál et al., 2007).

In addition to conservation tillage, the inclusion of winter cover crops in spring crop rotations may deliver a range of important ecosystem benefits. Winter cover crops scavenge soil nutrients and reduce nutrient leaching into the surrounding aquatic environment (Jackson et al., 1993; Fageria et al., 2005; Gómez et al., 2009). Improved soil quality and carbon sequestration have also been observed in the presence of cover crops (Mutegi et al., 2013). The presence of cover crops during autumn and winter when heavy rainfall events frequently occur also helps to prevent erosion and compaction by reducing rain-splash dispersion of the soil surface. They also help to reduce crusting by intercepting the large rain droplets before they can strike and compact the soil surface (Stagnari et al., 2010). Moreover, the significant effects of cover crops on nitrate loss has led some countries to include cover crops as a key element in their national strategy to reduce nitrate leaching (Munkholm and Hansen, 2012). No major disadvantages of cover crops on soil quality are known. However, concern has been raised over the potential negative effects of cover crops to the following crop yield. Cover crops may create N deficiency for the next crop if too much N is immobilised and not released in a timely manner (Fageria et al., 2005). Therefore, it was hypothesised in this study that alternative cultivation regime such as minimum tillage and direct drill combined with a cover crop would improve the overall soil quality.

3.2 Results and discussion

3.2.1 Particle size distribution

The particle size distributions as a percentage of sand, silt and clay for the 144 soil samples collected at three soil depths are shown in Figure 3.1. Samples from topsoil (0-30 cm) are concentrated in a relatively small area in the ternary diagram and most of them have clay loam, sandy loam, sandy silt loam and sandy clay loam soil texture, with only one clay soil texture sample and no sand soil texture. Conversely, samples collected from the deeper soil (60-90 cm) are scattered in a relatively large area of the triangle, with many clay and sand texture samples. This means that the texture of topsoil in the nine fields is more homogeneous than the deep soil layer. This might be due to cultivating the top soil layers that homogenised the soil over years. The greatest variations occur in the clay and sand fractions, particularly in the deeper soil layer, with very little variation in the proportion of silt sized material.

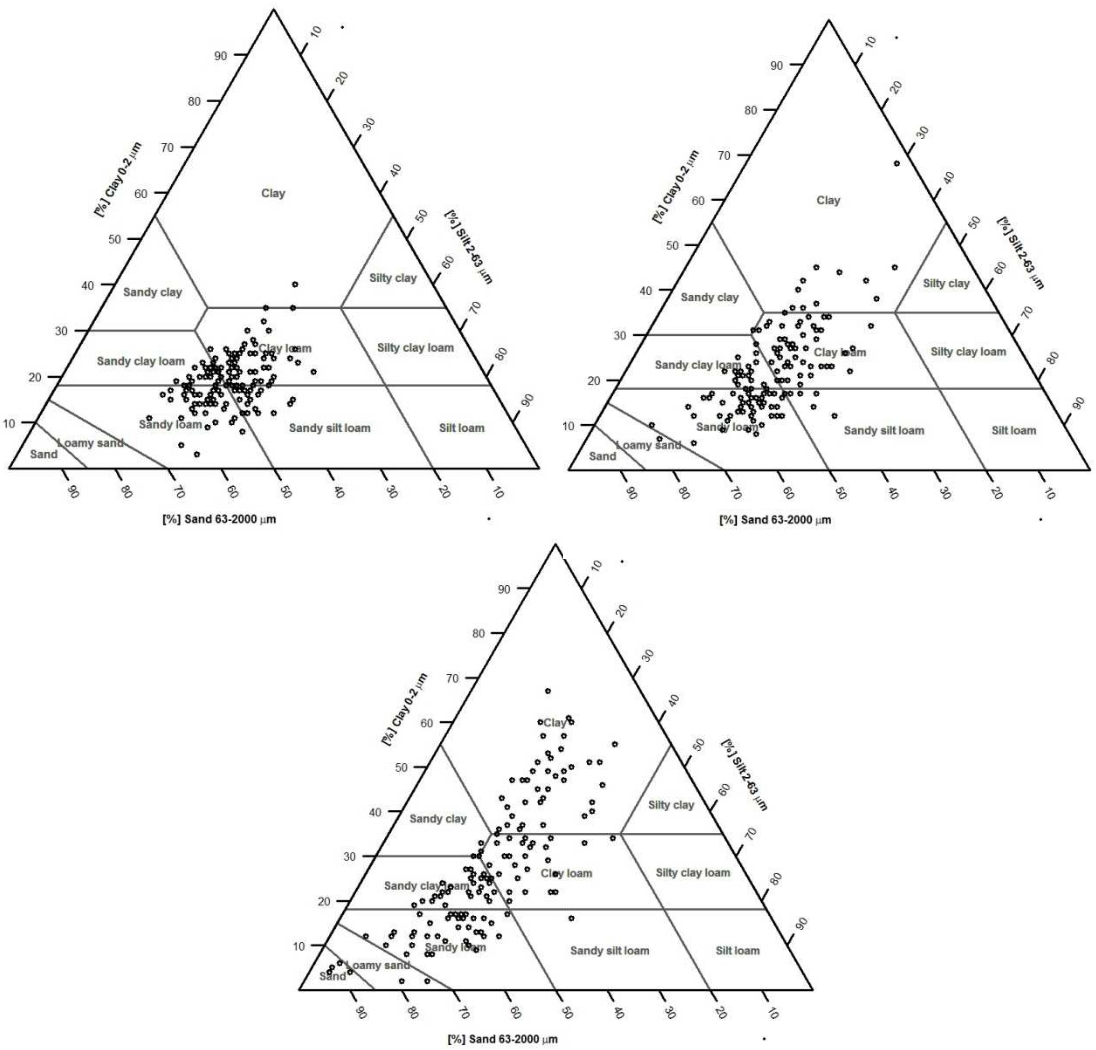

Figure 3.1 Particle size distribution of 0-30 cm depth (top left), 30-60 cm depth (top right), and 60-90 cm depth (bottom).

Spatial variation of soil fractions in the topsoil of the study area was visualised by using the data of all three soil fractions. The data of each fraction were interpolated via kriging in ArcGIS10.1 and then the interpolated maps were composite banded. The distribution of soil fractions in the topsoil (0-30 cm) are shown in Figure 3.2. Red colour indicates sand-dominated soil texture and blue colour represents soil with high clay content. Sheds Field and the southern part of Moor Hall Field show a reddish colour indicating high sand content. The eastern part of Swanhills and north-eastern part of GateHouse Hyrne fields possess relatively high silt content. On the other hand, Potash and the western part of GateHouse Hyrne fields have bluish colours indicating a soil with high clay content.

This result corresponds well with the electrical conductivity scan of these same fields (Appendix A2). The electrical conductivity of soils changes depending upon the porosity and the amount of moisture held by soil particles. Sands have low conductivity, silts have medium conductivity and clays have high conductivity. Consequently, electrical conductivity correlates strongly to soil particle size and texture (Grisso et al., 2009). Low electrical conductivity zones located in areas with high sand content and high electrical conductivity contours coincided with areas of high clay content.

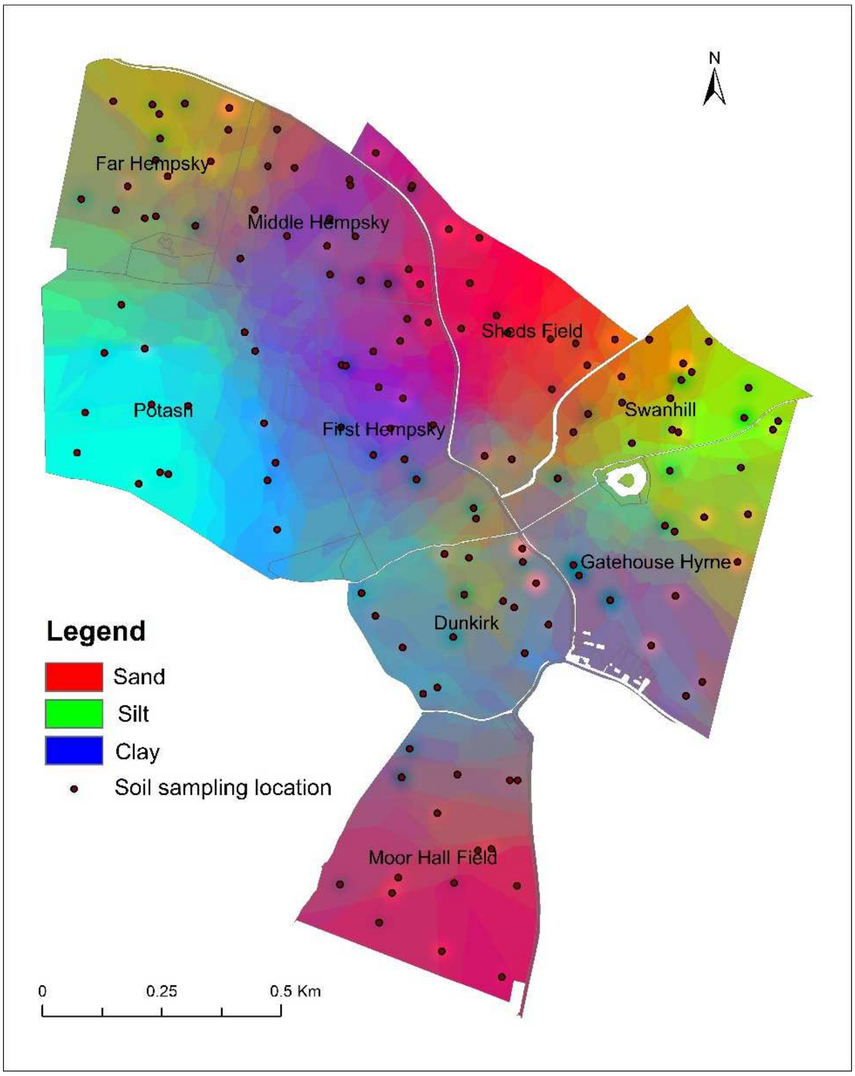

Figure 3.2 Distribution of soil fractions in the 0-30 cm soil depth.

3.2.2 Soil mineral nitrogen (SMN)

3.2.2.1 Soil nitrate

Results for soil extractable nitrogen (reported as NO_3^--N) in the topsoil are shown in Figure 3.3 and Figure 3.4. Soil nitrate concentrations were not significantly ($p > 0.05$) different among the three treatments for the same sampling date. This means that in the topsoil, the cover crops and conservation tillage did not significantly reduce soil nitrate compared to the no cover crop and conventional tillage fields. After the winter wheat harvest and before cover crop planting in September 2013, soils in all three treatments had the highest nitrate concentration (>5 mg N kg^{-1}). This was because some nitrate was left over from the previous crop and had not been leached. Soil nitrate concentration decreased substantially from September 2013 to February 2014 in all three treatments from values of >5 mg N kg^{-1} to values of <3 mg N kg^{-1}. The decrease in soil nitrate in bare fields over autumn may have been due to leaching losses as a large amount of rainfall occurred during the autumn (Figure 4.1). Additionally, the decline of soil nitrate in the two cover crop treatments over autumn may have been due to the uptake of potentially leachable soil nitrate by the growing cover crop. Evidence for this is provided by leaf and root nutrient analysis of the cover cops which showed that the average N content contained in both leaf and root was 76 kg N ha^{-1} (Table 3.1). Other studies have found a significant reduction in soil nitrate by radish cover crops compared to bare soil in the topsoil layer. Jackson et al.(1993) found that soil under a radish cover crop had a soil nitrate value of only 2 mg N kg^{-1},whereas this figure was approximately 28 mg N kg^{-1} for bare soils at 0-15 cm depth.

Table 3.1 Oilseed radish leaf and root analysis (January 2014)

N content Leaf (kg N ha^{-1})	N content Root (kg N ha^{-1})	N content Total (leaf & root) (kg N ha^{-1})	Dry matter yield Leaf (t ha^{-1})	Dry matter yield Root (t ha^{-1})	Dry matter yield Total (leaf & root) (t ha^{-1})
64	12	76	2.11	0.61	2.72

The data presented here showed that there was no significant difference in soil nitrate concentrations between cover crop fields and bare soil fields after the cover crop was incorporated into the soil (i.e. results from July 2014 sampling). Soils from cover crop & direct drill treatment surprisingly had a lower mean nitrate level after cover crop incorporation compared to the control (no cover crop & conventional plough) treatment. The expected increase of soil nitrate after cover crop incorporation is because the trapped nitrogen stays in the soil and becomes available to the next crop when the plant decomposes in the spring. Contrary to the findings in this study, many studies found that soil nitrate increased in cover cropped fields compared to bare soil after incorporation. Jackson et al. (1993), for instance, noted that soil nitrate concentrations in cover crop plots were significantly higher than in bare soil by day 37 of the post-incorporation date. Knott (1996) also found that in all three experiments, levels of soil mineral nitrogen (SMN) after peas were sown were consistently higher after cover crop incorporation than after bare stubble. Justes et al. (1999) also noted that after its incorporation, mineralisation of nitrogen within the radish cover crop increased SMN content. The absence of a significant increase in soil nitrate concentrations after cover crop incorporation in this study may be due to the fact that soil samples were collected four months after incorporation and soil nitrate might have been taken up by the following crop during that time.

Again, there were no statistically significant differences in soil nitrate among treatments for both soil sampling dates (i.e. February 2015 and July 2015) in the second year. This confirms the results from the previous year that conservation tillage did not significantly affect soil nitrate levels compared to conventional tillage. Generally, soil nitrate was lower in the second farming year than the first farming year in both February and July. This might have been due to the type of crops planted. In the first farming year, when leguminous spring beans were grown, soil nitrate levels at both sampling dates were high, potentially due to the ability of spring beans to fix atmospheric nitrogen. Consequently, soil nitrate is unutilised by spring beans. Conversely, in the second farming year when non-leguminous winter wheat was grown, soil nitrate in both sampling dates were relatively low, potentially due to the ability of wheat to scavenge nitrate from soil profile.

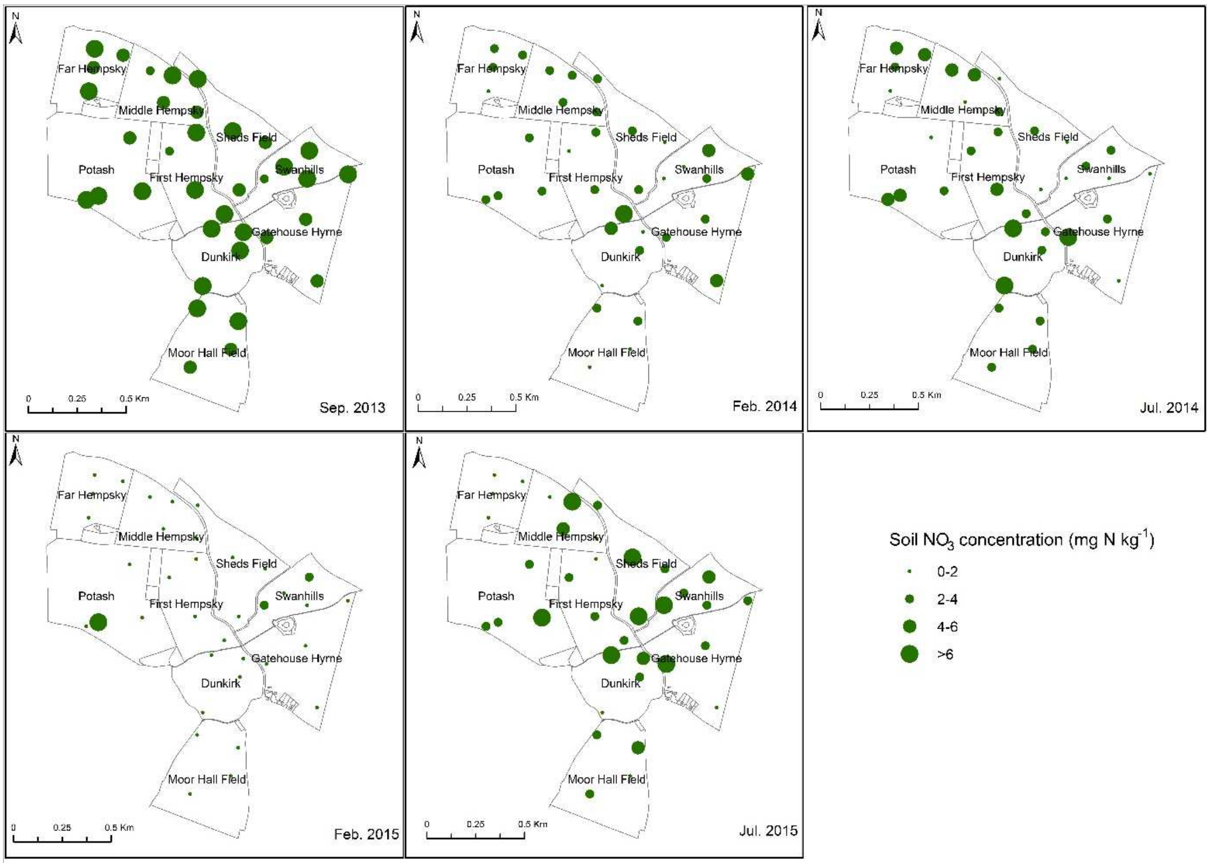

Figure 3.3: Spatial variation of soil NO_3^- concentration for the five soil sampling occasions.

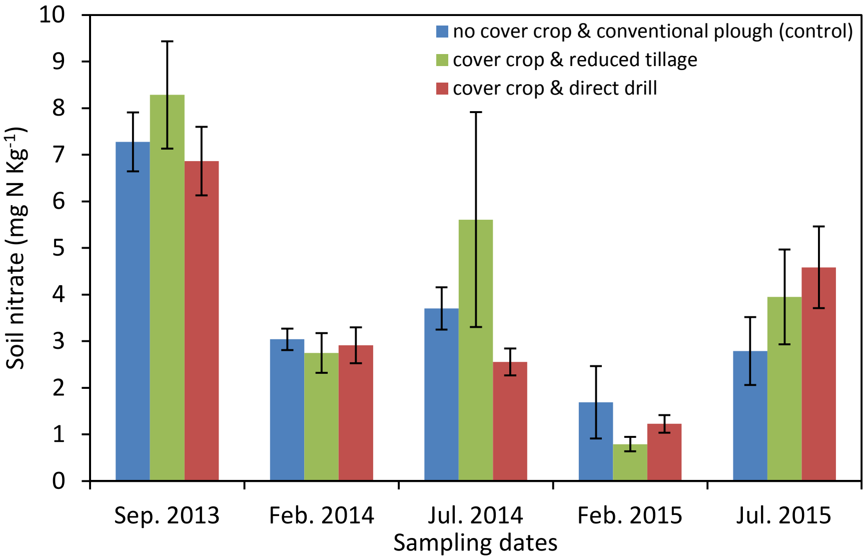

Figure 3.4 Soil nitrate concentrations at 0-30 cm depth recorded in the three mitigation treatments on five soil sampling occasions. Error bars represent the standard error.

Even though the reduction of topsoil soil nitrate by cover crops was not significant ($p > 0.05$) compared to bare soil, the cover crop did significantly ($p < 0.05$) deplete soil nitrate in the deeper soil layers (i.e. 30-60 cm and 60-90 cm) compared to bare soil (Figure 3.5). Soil from three depths was sampled in February 2014 when cover crops were present in the seven fields and the other two fields were bare. Cover crops reduced soil nitrate in the topsoil (2.83 N mg N kg^{-1}) compared to bare soil fields (3.04 mg N kg^{-1}), but the difference was not statistically significance ($p > 0.05$) (Figure 3.3 and Figure 3.4). However, the significant effect of the cover crop started appearing in the two deeper soil profiles in which soil nitrate was reduced from (3.76 to 2.18 mg N kg^{-1}) at 30-60 cm depth and from (3.54 to 0.76 mg N kg^{-1}) at 60-90 depth. Thus, compared to bare soil, cover crops reduced soil nitrate by 7%, 42%, and 79% in 0-30 cm, 30-60cm and 60-90 cm soil depths, respectively. This significant decline of soil nitrate in the deeper soils is related to the type of cover crop planted here. Oilseed radish has a thick, deep root that can help break up compacted soil layers and scavenge nitrate that has leached beyond the rooting zone of other crops. This ability of scavenging nitrogen from deep in the soil profile offers radish cover crops an extra advantage compared to other cover crop varieties because the nitrate in deeper soil layers is barely taken up by crops, so eventually it is more likely to leach into groundwater or field drains.

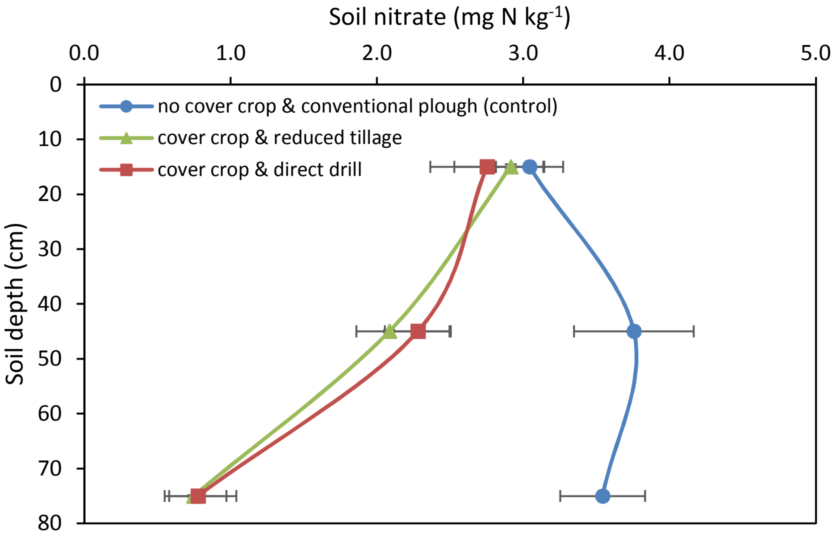

Figure 3.5 Mean of soil nitrate in three soil depthsfor soil samples collected in February 2014 (i.e. during cover crop growth) in the three treatments. Error bar represented by standard error. Note: the cover crop planted in both treatments substantially depleted soil nitrate in the deeper soil layers relative to the control treatment.

3.2.2.2 Soil available nitrogen

Available total nitrogen in the 30 cm soil profile is shown in Figure 3.6. Available nitrogen was not significantly ($p > 0.05$) different among the three treatments for the same sampling date, which again illustrates that the implemented mitigation measures did not have a substantial impact upon available nitrogen. The temporal trends in available nitrogen were very similar to that observed for soil nitrate. Soil available nitrogen post-harvest (September 2013) was relatively high, with mean values for three fields of 33 kg N ha^{-1}. Levels then declined in all experimental fields to a mean value of 17 kg N ha^{-1}. Like soil nitrate, this decrease in the bare control treatment (no cover crop & conventional plough) may be due to leaching and in the cover cropped treatments due to scavenging by the cover crop itself (Stivers-Young, 1998). Thus, the amount of soil nitrogen that was taken up by the cover crop in the 30 cm soil profile was approximately 17 kg N ha^{-1}. Oilseed radish does not fix nitrogen, so all of its accumulated nitrogen content was obtained from the soil. Therefore, of the total nitrogen content of cover crop of 76 kg N ha^{-1} (Table 3.1), 17 kg N ha^{-1} were scavenged from the 30 cm soil profile and the remaining amount of 60 kg N ha^{-1} came from deeper soil profile. This emphasises the effectiveness of scavenging N from deep soil layers by oilseed radish, thus making it a highly effective cover crop for preventing N leaching. Such reductions in soil mineral nitrogen by cover crops has also been noted in other studies. Justes et al (1999) found that the presence of a radish cover crop markedly depleted the mineral nitrogen content of the soil compared to bare soil, particularly in the upper two layers where the roots are present. They found that radish cover crops decreased soil mineral nitrogen in the 0-120 cm depth from 70 kg N ha^{-1} in September to ~30 kg N ha^{-1} in March, whereas this figure stayed relatively stable within that period for bare soils.

Soil nitrogen did not significantly increase after cover crop incorporation (February 2014 to July 2014). Again, this may be related to the soil sampling occurring four months after cover crop incorporation. Similar to the reduction in soil available nitrogen from September 2013 to February 2014, soil available nitrogen over autumn reduced notably from July 2014 to February 2015 in all three treatments. This loss of nitrogen from the soil profile throughout autumn/winter is presumably due to leaching losses as a great amount of rainfall occurred during that period (Figure 4.1). A noticeable increase was observed from February 2015 to July 2015 in all treatments perhaps due to the applied N fertilisers. Similar to soil nitrate, soil available nitrogen was generally lower in the second farming year than the first farming year in both February and July. As discussed above, this might be due to the type of crops grown in that year.

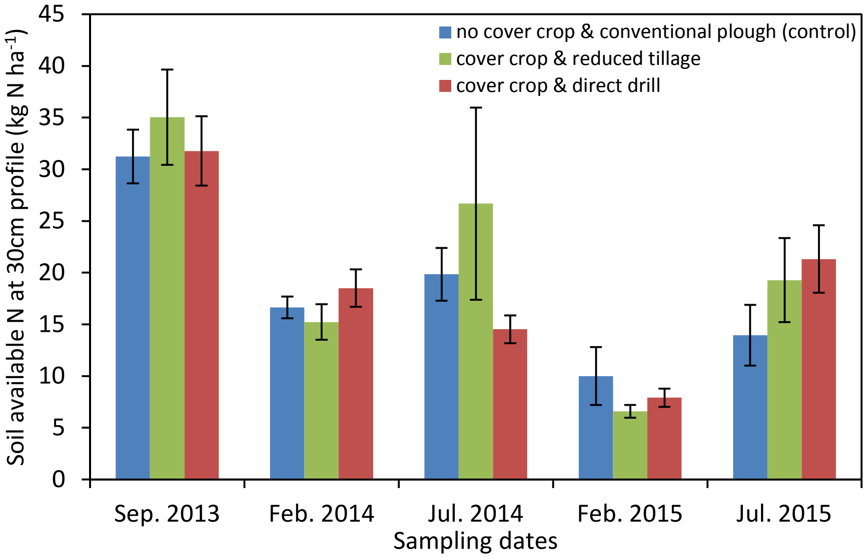

Figure 3.6 Soil available nitrogen at 0-30 cm depth in the three mitigation treatments during five sampling dates. Error bars represent the standard error.

3.2.3 Soil macronutrients

3.2.3.1 Soil Organic Carbon

For the 144 topsoil samples collected prior to the experiment (May and September 2013), the soil organic carbon (SOC) was analysed along with soil texture to understand the overall condition of the study area. The SOC values were relatively low (median 1.28%) when compared to a median for the East Anglian region in general (2.24 %) based on analyses of 2,858 topsoils samples collected between 2002 and 2005 (Rawlins, 2011). In their paper, Loveland and Webb (2003) cited 2% as a threshold below which many soil scientists believe a potentially serious decline in soil quality will occur. In this study, 86% of sampled soils on the nine fields had SOC values below 2%, indicating a potential threat to long-term soil quality.

The measurements of soil organic carbon in 144 samples from topsoil (0-30) were interpolated by ArcGIS10.1 and treatment kriging to illustrate spatial variation (Figure 3.7). It is observed that zones of low SOC (green zones) generally coincided with soil of high sand content (Figure 3.2), whilst zones of higher SOC generally matched with areas of high clay and silt content. Soil organic matter and thus SOC, tend to increase with increasing clay content. This increase depends on two mechanisms. First, bonds between the surface of clay particles and organic matter delay the decomposition process. Second, the potential for aggregate formation increases with higher clay content. Thus, macro-aggregates physically protect organic matter molecules from further mineralisation caused by microbial activity (Rice, 2002; Bot and Benites, 2005). Power and Prasad (1997) mentioned that under similar climate conditions, the organic matter content in fine textured (clayey) soils is two to four times that of coarse textured (sandy) soils.

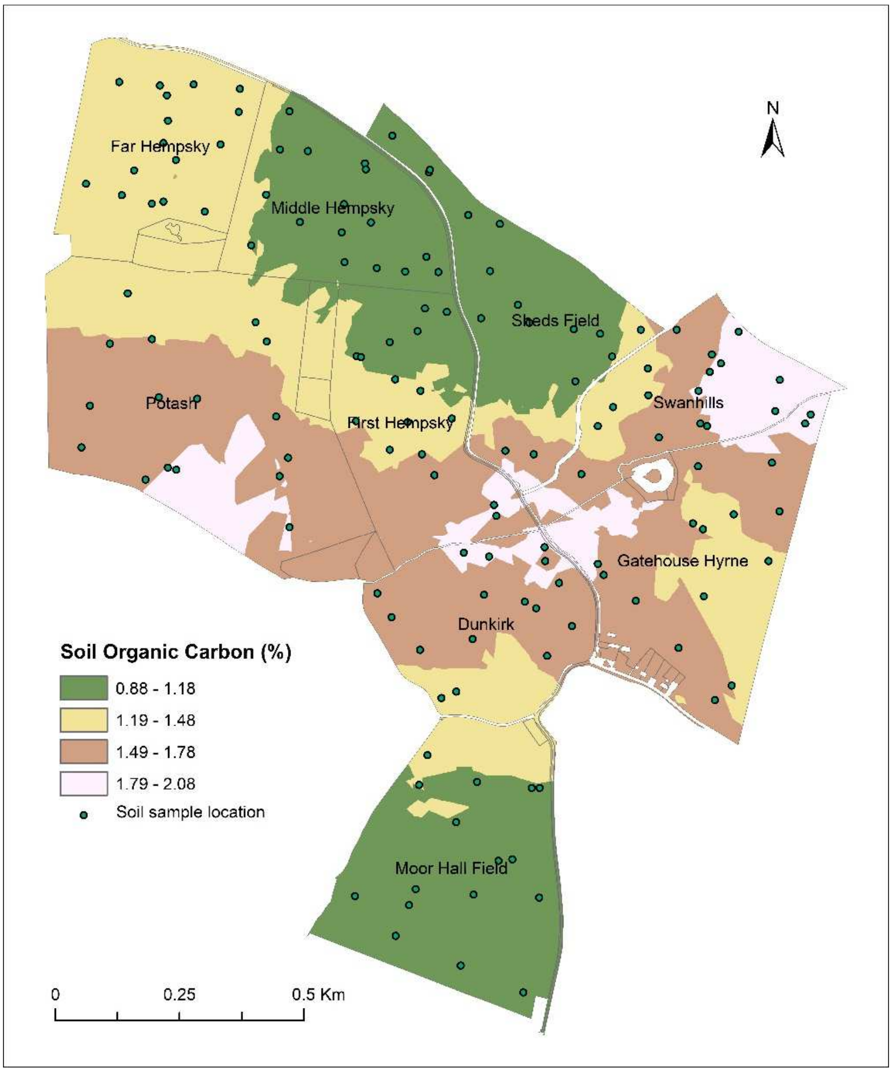

Figure 3.7 Soil organic carbon distribution in the topsoil (0-30 cm) for 144 soil samples collected in May and September 2013.

The percentage SOC can be converted into mega-grams (i.e. ton) of carbon per hectare (Mg ha^{-1}), which is commonly used for presenting SOC data, by taking the bulk density and depth of soil sampling into consideration. During the experiment, no significant ($p > 0.05$) difference in SOC was observed in the treatments for the same sampling dates (Figure 3.8). Among the three treatments in September 2013, control treatment (no cover crop & conventional plough) had the highest SOC of 89.1 Mg ha^{-1}, followed by treatment (cover crop & reduced tillage) with a value of 81.8 Mg ha^{-1} and treatment (cover crop & direct drilling) had the lowest SOC value of 76.2 Mg ha^{-1}. Regardless of the differences in farming practices, this order of SOC content of the three treatments was repeated in the following three sampling rounds. This is probably because soils in control treatment (no cover crop & conventional plough) are predominately clayey soils and soils in treatment (cover crop & direct drilling) are largely sandy (Figure 3.2) and as mentioned earlier, clayey soils tend to contain a higher percentage of SOC than sandy soils.

Over time, there was a general trend for a slight increase in SOC content of each treatment. The SOC in treatment (cover crop & direct drilling), for instance, increased from 76.2 Mg ha^{-1} in September 2013 to 80.2 Mg ha^{-1} in February 2014, 83.8 Mg ha^{-1} in February 2015 and finally 88.6 Mg ha^{-1} in July 2015. Thus, the mitigation measures practiced (i.e. cover crop and direct drill) increased SOC by 12.4 Mg ha^{-1} within two years of experimentation (September 2013 to July 2015). This may be partially due to the incorporation of cover crops into the soils as approximately 2.7 Mg ha^{-1} of biomass was added into the soil and the biomass left undisturbed on the soil surface (Table 3.1). SOC also increased in treatment (cover crop & reduced tillage) from 81.8 Mg ha^{-1} in September 2013 to 90.8 Mg ha^{-1} by July 2015, an increase of 9.0 Mg ha^{-1}. Thus, the mitigation measures practiced in this treatment (i.e. cover crops and minimum tillage) perhaps generated this increase in SOC. Surprisingly, SOC also increased in control treatment (no cover & conventional plough) within that two years of the experiment by 6.8 Mg ha^{-1}. Therefore, cover cropping, minimum and direct tillage may not be the only cause for the observed slight increase in SOC, as this increase also occurred in the control treatment. However, it is worth noting here that as the degree of soil disturbance decreased, the increase in SOC became greater.

The results presented here are in agreement with the inconsistent results of the effects of cover crops and conservation tillage on SOC in the literature. Govaerts et al. (2009) conducted a literature review to identify the influence of the different components comprising conservation agriculture on SOC. In 7 out of 78 cases, the SOC content was lower in zero tillage compared to conventional tillage; in 40 cases it was higher and in 31 of the cases there was no significant difference. The mechanisms that govern the balance between increased, similar, or lower SOC after conversion to zero tillage are not yet fully understood. Six et al. (2002) found a greater accumulation of SOC in the topsoil with no-tillage compared to conventional tillage. They stated that this greater accumulation of SOC in zero or reduced tillage can be related to the lack of soil disturbance and a better preservation of aggregates in reduced tillage compared to conventional tillage. It was concluded in a long-term (19 years) comparative study of conventional tillage to no-tillage and fallow to several winter cover crops by Calegari et al. (2008) that no-tillage resulted in a 6.84 Mg ha^{-1} increase in SOC in the upper soil layer (0-10 cm), which represented 64% more than conventional tillage. Also, fallow treatment (regardless of tillage types) resulted in the lowest SOC stocks at 40 cm depth compared to all other winter cover crops. The higher SOC in no-tillage was potentially a result of organic residues left every year on the soil surface, no soil disturbance and decreased contact with soil microorganisms.

Although the increase in SOC observed in this two-year study through implementing cover crops and reduced tillage/direct drill was small or insignificant, perhaps the real increase will be observed after a few years of continuing the experiment. Lal et al. (1998), supported by the results from Franzluebbers and Arshad (1996a, b), observed that there may be little to no increase in SOC in the first 2-5 years after a change in management practice, but will be followed by a larger increase in the next 5-10 years. West and Post (2002) concluded that SOC was generally increased by no-tillage management, but observed a delayed response, with a substantial increase observed in years 5-10. Campbell et al. (2000) found that a measureable gain in SOC could be detected in 6 years or less when weather condition were favourable.

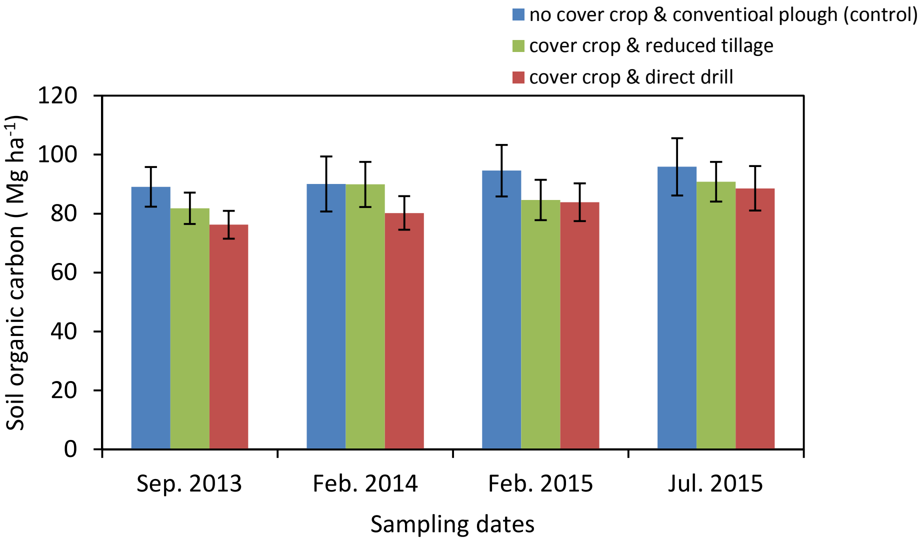

Figure 3.8 Soil organic carbon (SOC) content at 0-30 cm depth in the three treatments on four sampling occasions. Error bars represent the standard error.

3.2.3.2 Soil phosphorus

Extractable phosphorus (P) concentrations for topsoil (0-15 cm) are presented in Figure 3.9. According to Horneck et al. (2011), who classified soils based on P concentration into four categories (low, medium, high and excessive), most of the soils in the study area are in the medium range (10-25 mg L^{-1}) and in the lower end of the high range (25-50 mg L^{-1}). Very few soils were in the excessive (>50 mg L^{-1}) range and no soil was in the low (<10 mg L^{-1}) range of P level. No major increase or decrease in P level was observed in individual soil sampling locations over time. Soils with medium P concentration, such as First and Far Hempsky fields, remained relatively constant over time and soil with high P levels, such as Sheds Field and Moor Hall Field, remained relatively high over time.

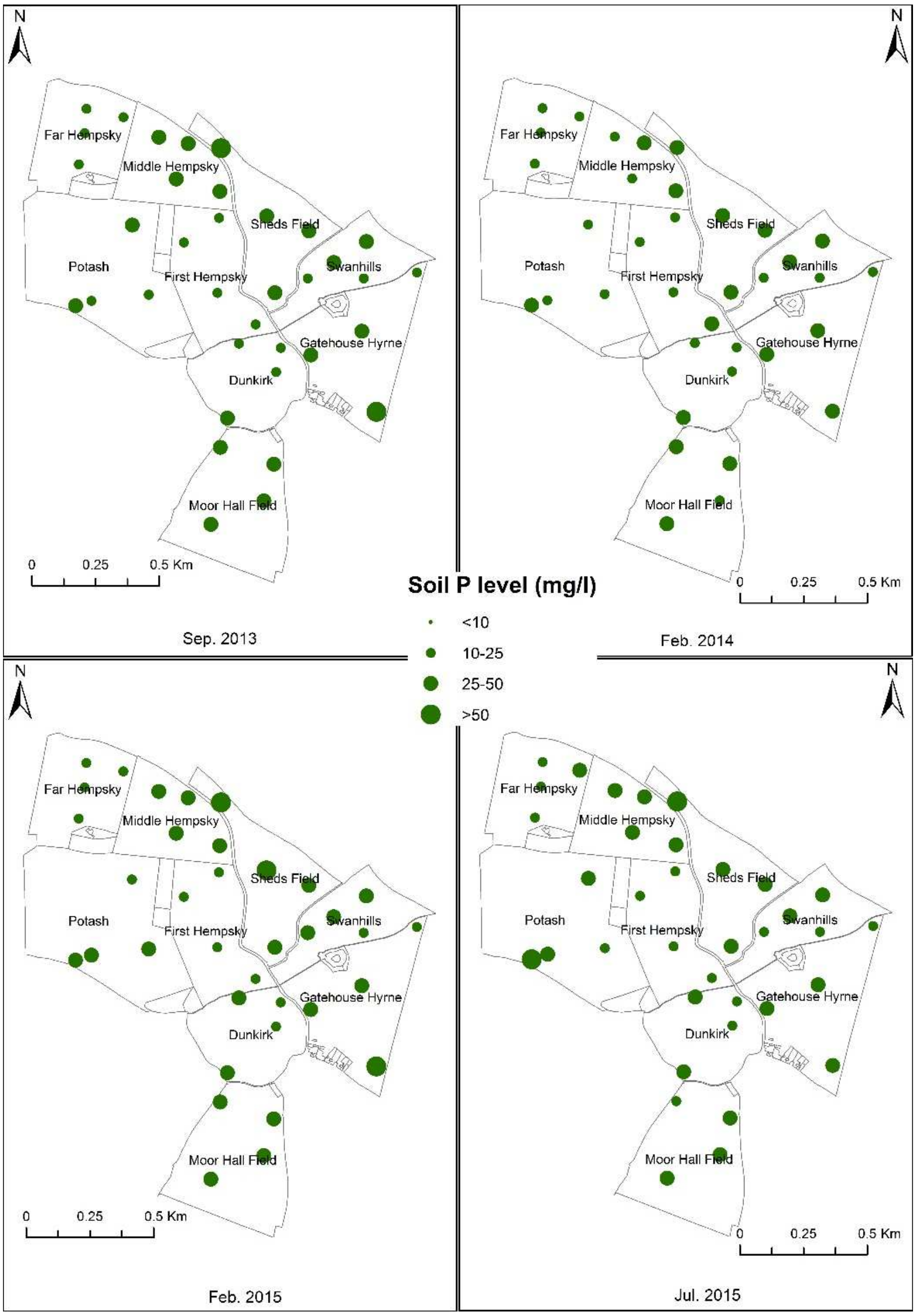

Figure 3.9. Spatial variation of soil P concentration on four sampling occasions.

Accumulation of P in surface soil was greater in treatment (cover crop & direct drill) and treatment (cover crop & reduced tillage) than in control treatment (no cover crop & conventional plough) on all four soil sampling dates (Table 3.2). During the experiment when cover crops were present in the fields (i.e. February 2014), P concentrations were significantly ($p < 0.05$) higher in treatment (cover crop & reduced tillage) (27.4 mg L^{-1}) and treatment (cover crop & direct drill) (26.2 mg L^{-1}) than in control treatment (18.0 mg L^{-1}). The lower value of P in the control treatment may be due to erosion and leaching as the soil was left bare during the winter, whereas cover crops in the measures fields protected the soil from the erosion and retained P in the soil. This higher P concentration in treatment (cover crop & direct drill) and treatment (cover crop & reduced tillage) was also observed in both February 2015 and July 2015. Moreover, no significant difference in P concentration between treatment (cover crop & reduced tillage) and treatment (cover crop & direct drill) was recorded for any of the sampling dates. The results here show that the cover crop and reduced tillage retained the soil P level in topsoil compared to conventional practices. It is also apparent that different types of reduced tillage (minimum tillage and direct drill) exerted only minor differences on topsoil P levels.

Several studies have reported higher extractable P levels in reduced tillage soils compared with tilled systems, largely due to reduced mixing of phosphate fertiliser within the soil, which causes lower P- fixation. This is an advantage when P is a limiting nutrient of crop growth, but may be a danger when P is an environmental problem due to the possibility of soluble P losses in surface runoff (Duiker and Beegle, 2006). In a 10-year study, Matowo et al. (1999) found significantly higher extractable-P concentrations in no-tillage compared to tilled soil in the top 5 cm. Andraski et al (2003) also found that long-term zero-tillage produced significantly higher concentrations of soil P in the soil surface (0-5 cm), whereas P levels were decreased at the 5-15 cm depth compared to conventional tillage. Zibilske et al. (2002) stated that while P levels in conventional tillage were lower in top 8 cm than under conservation tillage, it was higher in the lower depths than in corresponding depths of conservation tillage. This suggests a redistribution of P with time toward the surface in conservation tillage.

Cover crops can present several potential opportunities and challenges for P management in agricultural systems, including remediation of excessively high soil P, increased concentration of P at the soil surface and improved fertility of low P soil. If agricultural land is excessively high in P, then P transport from these areas to natural waters is one of the primary causes of eutrophication (Boesch et al., 2001). The concentration of P in soil can be depleted with time by reducing the use of P containing fertilisers while continuing to remove P from the soil through harvested cover crops and main crops. On the other hand, cover crops are allowed to decompose at the soil surface in no-tillage systems which may lead to an accumulation of P at the soil surface where it is susceptible to loss by surface runoff and erosion (Brown, 2006; White and Weil, 2011). The effects of cover crops on P in soil is less documented compared to the tillage effect on P. White and Weil (2011) noticed that P concentrations were lower in soil without cover crops than in soils under a cover crop. They recommended further studies be conducted to determine if cover crops could increase P removal rates in excessively high P soils or increase P availability in low P soils.

Table 3.2 Mean soil phosphorus (P) concentrations at 0-15 cm depth in the three mitigation treatments for four soil sampling dates. Means followed by different superscripted letters are significantly different ($p < 0.05$) for the same sampling date. Means followed by different superscripted numbers are significantly different ($p < 0.05$) for the same treatment. SE is standard error

	Sep. 2013		Feb. 2014		Feb. 2015		Jul. 2015	
Treatments	P (mg L^{-1})	SE	P (mg L^{-1})	SE	P (mg L^{-1})	SE	P (mg L^{-1})	SE
No cover crop & Conventional plough (control)	21.4^{a1}	2	18.0^{a1}	2	22.9^{a2}	3	26.9^{a1}	4
Cover crop & reduced tillage	29.3^{a1}	3	27.4^{b1}	3	30.0^{a1}	4	29.2^{a1}	3
Cover crop & direct drill	30.3^{a1}	3	26.2^{b1}	2	33.0^{a1}	3	34.4^{a1}	3

3.2.3.3 Soil potassium

Topsoil (0-15 cm) extractable potassium (K) concentrations for the three field treatments are shown in Table 3.3. According to (Horneck et al., 2011), who classified soils based on K concentration into four categories (low, medium, high and excessive), the majority of soils have low (<150 mg L^{-1}) K levels, a few are in the medium (150-250 mg L^{-1}) range, whilst there are no soils with either high (250-800 mg L^{-1}) or excessive (>800 mg L^{-1}) K concentration.

Before the experiment (i.e. September 2013), soils in treatment (cover crop & reduced tillage) had the highest K concentration (128 mg L^{-1}) compared to the other two treatments, whilst treatment (cover crop & direct drill) had lower K concentration (103 mg L^{-1}) than control treatment (111 mg L^{-1}). However, five months later when the cover crops were growing (i.e. February 2014) this order of treatments changed. K levels were significantly ($p < 0.05$) higher in both (cover crop & reduced tillage) (144 mg L^{-1}) and (cover crop & direct drill) (129 mg L^{-1}) treatments than in control treatment (95 mg L^{-1}). Similar to P concentrations , this lower value of K in the control treatment may be due to erosion and leaching given that the soil was left bare over winter, whereas the cover crop in the measures fields protected the soil from erosion and retained K in the soil. This higher K concentration treatment (cover crop & direct drill) and treatment (cover crop & reduced tillage) compared to control treatment was also observed in both February 2015 and July 2015.

A minor difference in K concentration between treatment (cover crop & reduced tillage) and treatment (cover crop & direct drill) was recorded in all four sampling rounds, meaning that different types of reduced tillage (reduced tillage and direct drill) potentially created minor differences in K level in the topsoil. Moreover, it is also observed that before the experiment, treatment (cover crop & direct drill) had the lowest K levels among the treatments with 103 mg L^{-1}, but that over time with implemented field measures in this treatment (i.e. cover crop and direct drill) this treatment became the highest regarding K concentration (130 mg L^{-1}). It is apparent from the comparison between control treatment and treatment (cover crop & direct drill) from the first to the last sampling date, that K levels in the topsoil decreased by 15 mg L^{-1} within two years in control treatment which may be due to erosion. Conversely, in treatment (cover crop & direct drill), K concentration increased by 27 mg L^{-1} which is likely due to the prevention of leaching and erosion by the cover crop and non-disturbance through direct drilling. Therefore, the data here indicate that cover crops and reduced tillage retained higher soil K levels in topsoil compared to conventional practices. It is also observed that direct drill practices tended to retain a greater amount of K in the topsoil than reduced tillage.

Many studies have observed higher extractable K concentration in reduced tillage systems than under conventional tillage. Minimum tillage retains and increases the availability of nutrients, such as K, near the soil surface where crop roots proliferate (Franzluebbers and Hons, 1996). Moreno et al. (2006) also observed that accumulations of K in the surface soil (0-10 cm) were greater in conservation tillage than under conventional tillage. Other studies have found higher extractable K levels at the soil surface as tillage intensity decreases (Lal et al., 1990; Ismail et al., 1994). Shallow incorporation of crop residues in reduced tillage may have resulted in higher amounts of K at the soil surface compared to conventional tillage. Du Preez et al. (2001) observed an increased concentration of K in no-tillage systems compared to conventional tillage, but this was less pronounced with depth. Some other authors have found surface accumulation of available K regardless of the tillage practice employed (Duiker and Beegle, 2006), while Matowo et al. (1999) observed no notable effect of tillage on available K concentration. In agreement with the findings here, Obi (1999) noted that K levels in soils (0-15 cm) under five different cover crops were appreciably improved compared to bare soil in a five-year cover cropping experiment. Abdollahi and Munkholm (2014) also found that cover crops significantly increased K content at the 0-10 cm soil depth and similar trends were also observed at 10-20 cm depth. In contrast to the results found in this study, Fourie et al. (2007)

found that cover cropping had no significant effect on extractable K in a study of sixteen treatments in comparison with no cover crop.

Table 3.3 Mean soil potassium (K) concentrations at 0-15 cm depth in the three treatments for four soil sampling dates. Means followed by different superscripted letters are significantly different ($p < 0.05$) for the same sampling date. Means followed by different superscripted numbers are significantly different ($p < 0.05$) for the same treatment. SE is standard error

	Sep. 2013		Feb. 2014		Feb. 2015		Jul. 2015	
Treatments	K (mg L^{-1})	SE	K (mg L^{-1})	SE	K (mg L^{-1})	SE	K (mg L^{-1})	SE
No cover crop & Conventional plough (control)	111^{ab1}	8	95^{a1}	9	120^{a1}	13	96^{a1}	13
Cover crop & reduced tillage	128^{b1}	10	144^{b2}	8	147^{a2}	8	120^{a1}	7
Cover crop & direct drill	103^{a1}	5	129^{b12}	8	157^{b2}	12	130^{a12}	14

3.2.3.4 Soil magnesium

The data of soil magnesium (Mg) concentrations are presented in Table 3.4. According to (Horneck et al., 2011), who classified soils based on Mg concentration into three categories (low, medium and high), the majority of the soils have low (<60 mg L^{-1}) Mg levels, a few are located at the lower end of the medium (60-300 mg L^{-1}) range and no soil samples have high (>300 mg L^{-1}) Mg levels. Before the experiment in September 2013, control treatment (no cover crop & conventional plough) had the highest mean Mg levels at 58.3 mg L^{-1}, followed by treatment (cover crop & reduced tillage) (53.4 mg L^{-1}) and treatment (cover crop & direct drill) (48.9 mg L^{-1}). During the cover crop trials (i.e. February 2014), no major change occurred in mean Mg levels in any of the treatments, with control treatment still having the highest concentrations and treatment (cover crop & direct drill) the lowest. The same was true for the third and fourth sampling dates.

The results here show that cover cropping and reduced tillage did not increase Mg concentration in the topsoil. A high level of Mg in a certain treatment remained high regardless of different agricultural practices, whilst a low Mg level in a certain treatment stayed low irrespective of cover crops and tillage operations. The results also show that unlike P and K which eroded and leached in the bare soil treatment during the winter, Mg concentrations were retained in a bare soil and were not reduced by erosion or leaching.

The findings in this study are in agreement with findings from a majority of other studies on the effects of tillage and cover crops on soil Mg content. Verhulst et al. (2010) in an extensive report stated that most research has shown that tillage method does not affect extractable Mg levels in the soil. Duiker and Beegle (2006) also observed no significant tillage effect on Mg content nor on its vertical or horizontal stratification. Franzluebbers and Hons (1996) found no significant effect of tillage on extractable Mg. A lack of tillage effect on Mg in soil was also observed by several others (Lal et al., 1990; Hulugalle and Entwistle, 1997; Govaerts et al., 2007). Matowo et al. (1999), however, observed higher extractable Mg concentration with conventional tillage than with zero tillage, although they could not explain this finding.

Table 3.4 Mean soil magnesium (Mg) concentrations at 0-15 cm depth in the three mitigation treatments for four soil sampling dates. Means followed by different superscripted letters are significantly different ($p < 0.05$) for the same sampling date. Means followed by different superscripted numbers are significantly different ($p < 0.05$) for the same treatment. SE is standard error

	Sep. 2013		Feb. 2014		Feb. 2015		Jul. 2015	
Treatments	Mg (mg L^{-1})	SE	Mg (mg L^{-1})	SE	Mg (mg L^{-1})	SE	Mg (mg L^{-1})	SE
No cover crop & Conventional plough (control)	58.3^{a1}	5	63.7^{a1}	6	65.7^{a1}	7	57.7^{a1}	5
Cover crop & reduced tillage	53.4^{a1}	3	53.0^{a1}	4	52.2^{b1}	4	51.2^{a1}	3
Cover crop & direct drill	48.9^{a1}	1	52.6^{a1}	3	50.9^{b1}	4	49^{a1}	3

3.2.3.5 Soil sulphate

Results for soil extractable sulphate (reported as SO_4-S) concentration are presented in Table 3.5. According to (Horneck et al., 2011), who classified soils based on SO_4 concentration into four categories (very low, low, medium and high), the majority of soils in the study had a medium (5-20 mg L^{-1}) concentration, a few were within the low (2-5 mg L^{-1}) range and no samples had either high (>20 mg L^{-1}) or very low (<2 mg L^{-1}) SO_4 concentrations.

Soil sulphate concentrations were not significantly ($p > 0.05$) different among the three treatments at the same sampling date. This means that cover crops and conservation tillage did not significantly elevate soil sulphate levels in the topsoil compared to no cover crop and conventional tillage. After the winter wheat harvest and before cover crop planting (September 2013), soils in all three treatments had the highest sulphate concentration (>8 mg L^{-1}). Soil sulphate concentrations then decreased between September 2013 and February 2014 (i.e. during autumn/winter) in all three treatments. The decrease in bare fields over autumn may have been due to leaching losses as large amounts of rainfall occurred at this time. Additionally, the sulphate decline in the two cover cropped treatments may have been due to the uptake of potentially leachable soil sulphate by the growing cover crop.

In 2015, there was again no statistically significant ($p > 0.05$) differences in soil sulphate among the treatments for both soil sampling dates (i.e. February 2015 and July 2015). This supports the results from the previous year that conservation tillage did not significantly affect soil sulphate levels compared to conventional tillage. Similar to the result here, Franzluebbers and Hons (1996) observed no change in soil sulphate under zero tillage compared to conventional tillage. Szulc et al. (2004), however, recorded the highest soil sulphate concentrations in deeply ploughed soil, whilst the smallest concentrations were observed under direct drill. The effects of different tillage systems and cover crops on soil sulphate level have been less intensively studied compared to other soil nutrients, perhaps due to the reason that it is one of the micronutrient in the soil instead of a key macronutrients for plant growth.

Table 3.5 Mean soil sulphate (SO_4) concentrations at 0-15 cm depth in the three mitigation treatments for four soil sampling dates. Means followed by different superscripted letters are significantly different ($p < 0.05$) for the same sampling date. Means followed by different superscripted numbers are significantly different ($p < 0.05$) for the same treatment. SE is standard error

	Sep. 2013		Feb. 2014		Feb. 2015		Jul. 2015	
Treatments	SO_4-S (mg L^{-1})	SE	SO_4-S (mg L^{-1})	SE	SO_4-S (mg L^{-1})	SE	SO_4-S (mg L^{-1})	SE
No cover crop & Conventional plough (control)	9.9^{a1}	1.3	6.1^{a1}	0.5	4.2^{a1}	0.2	7.4^{a1}	0.6
Cover crop & reduced tillage	8.2^{a1}	0.3	7.4^{a1}	0.8	3.7^{a1}	0.1	7.1^{a1}	0.3
Cover crop & direct drill	8.5^{a1}	0.3	7.4^{a1}	0.8	3.4^{a1}	0.3	8.2^{a1}	0.5

3.2.4 Soil physical properties

3.2.4.1 Soil temperature

Soil temperature data at three depths for the three treatments are presented in Figure 3.10. Data from temperature and moisture probes installed in clay loam soils were selected here as these provided the most complete data record. Probe number 50513 in control treatment (no cover crop & conventional plough), probe number 50369 in treatment (cover crop & reduced tillage) and probe number 50399 in treatment (cover crop & direct drill) were chosen. A maximum soil temperature of 25.1 °C was recorded in July 2014 at 15 cm depth in control treatment and a minimum soil temperature of 2.1 °C was recorded in February 2014 at 15 cm depth, also in control treatment. In all treatments, no major differences in soil temperature between the three depths were observed and it was noted that soil temperatures at the three depths exhibited the same trends over time. As summer 2015 was approaching, for instance, soil temperatures gradually increased at all depths corresponding to the gradual increase in air temperature. It is also observed that soil temperature tended to be slightly lower in the deeper soil layer (75 cm) than the upper soil layer (15 cm). This is very clear for summer 2014 and summer 2015 when soil temperatures were consistently lower at 75 cm than 15 cm, but for winter 2014-2015 soil temperature was higher at 75 cm depth than at 15 cm. Liu et al. (2013) mentioned that it is commonly observed that soil temperature is strongly correlated with air temperature and decreases with increasing depth below the soil surface. The strong correlation between air and soil temperatures was also likely responsible for the remarkable fluctuations seen in the temperature of the upper soil layers compared to the deep soil layers. The correlation between air and soil temperature is not only recorded in different soil layers but also observed at different times. Air temperatures are lower in winter than in summer and soil was also found to have lower temperatures in winter than in summer. For example, the mean value of soil temperature in the top soil layer (15 cm) in control treatment in winter 2013-2014 was 5.5 °C, whilst this figure was 19.0 °C for summer 2014.

No consistent significant differences in soil temperature between different mitigation treatments were observed. The mean values of soil temperature at 15 cm depth over the study period were 10.5 °C, 10.5 °C and 12.3 °C for control, (cover crop & reduced tillage) and (cover crop & direct drill) treatment, respectively. When a cover crop was present (i.e. winter 2013-2014) on the two treatments, the bare soil treatment had a soil temperature of 5.5 °C which was not significantly different from treatment (cover crop & reduced tillage) with a soil temperature of 5.6 °C. In summer 2014, soil temperatures at 15 cm depth in control treatment (19.0 °C) were significantly ($p < 0.05$) higher than in treatment (cover crop & reduced tillage) (15.2 °C) and in treatment (cover crop & direct drill) (15.0 °C). However, in the following summer, no significant differences in soil temperature were noted among the treatments with mean values of 11.0 °C, 10.9 °C and 11.4 °C for the control, (cover crop & reduced tillage) and (cover crop & direct drill) treatments, respectively. Therefore, overall, cover crops and different soil tillage methods had little effect on soil temperature.

Several studies have been conducted to assess the effect of different soil management approaches on soil temperature. (Fabrizzi et al., 2005) observed that mean soil temperatures were lower under no-tillage than under minimum tillage systems. They believed that the differences in soil temperature between tillage systems were due to differences in residue accumulation on the soil surface. The high solar reflectivity and low thermal conductivity of the crop residues prevent an increase of soil temperature under no-tillage. Maximum soil temperature under minimum tillage was higher than under no-tillage, but minimum soil temperature was similar for both tillage systems. Drury et al. (1999) studied soil temperatures and soil water content under no-tillage and conventional tillage in corn fields in Ontario with and without cover crop. They found that no-tillage with and without cover crop increased soil water content by 2-5% and reduced soil temperature by 1-2 °C compared to conventional tillage. These cooler soil temperatures and higher soil moisture contents in no till soils can lead to poorer seedbed conditions which may result in lower germination success. Licht and Al-Kaisi (2005) found that soil temperatures in the top 5 cm under strip tillage were higher (1.2-1.4 °C) than that under no-tillage and they stayed close to the chisel plough soil temperature. This increase in soil temperature contributed to an improvement in plant emergence rate index under strip-tillage compared with no-tillage. Other studies (Moroizumi and Horino, 2002; Chen et al., 2011; Liu et al., 2013) recorded higher soil temperature under conventional tillage than under conservation tillage.

It was hypothesised in this study that cover crop and reduced tillage would improve soil quality. Increasing (not decreasing) soil temperature is generally favourable for improving soil quality and crop productivity (Drury et al., 1999). Thus, increasing soil temperature was expected to occur as a result of having cover crops and reduced tillage. However, this study revealed that soil temperature was not significantly influenced by either cover crops or reduced tillage.

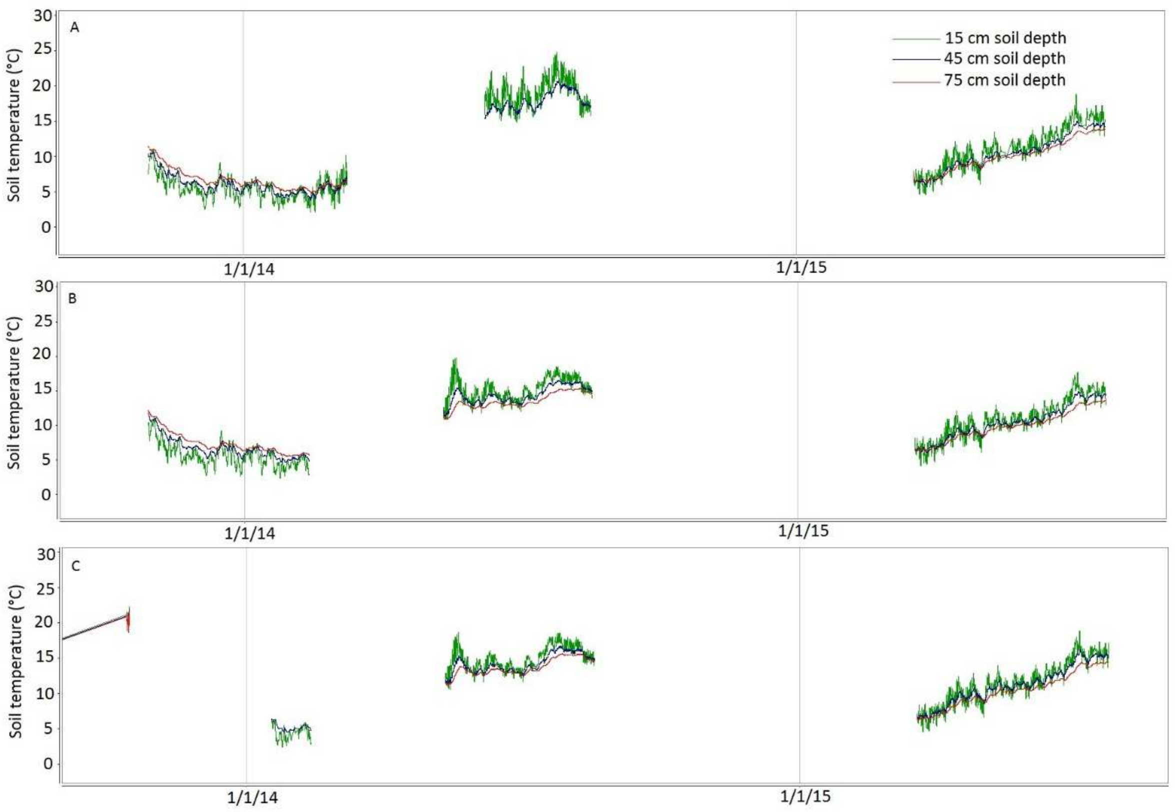

Figure 3.10 Soil temperature at three depths in (A) control treatment (no cover crop with conventional ploughing), (B) treatment (cover crop with reduced tillage) and (C) treatment (cover crop with direct drill). Note: the gap in the data are due to absence of the probes in fields because of field operations.

3.2.4.2 Soil moisture

Soil moisture data at three depths for the three treatments are presented in Figure 3.11. The soil moisture data were collected using soil moisture probes which recorded soil moisture at nine soil depths from 10 cm to 90 cm. However, here data from only three depths (10 cm, 50 cm and 90 cm) were selected to provide a clear graph. Unlike soil temperature which had no substantial differences at the three depths, it was observed that soil moisture showed significant differences between soil depths. Soil moisture increased with increasing soil depth. It was consistently higher at 90 cm than at 10 cm. For example, in treatment 9cover crop & reduced tillage) during summer 2014, mean soil moisture contents were 18%, 48% and 59% for the 10 cm, 50 cm and 90 cm soil depths, respectively. Much greater fluctuations were observed in the upper soil layers compared to deep soil layers, likely corresponding to instant changes in weather conditions with the upper layers having direct contact to the atmosphere.

There was no significant difference observed in soil moisture between the treatments. The mean values of soil moisture at 10 cm soil depth were 40%, 23% and 39% for control, (cover crop & reduce tillage) and (cover crop & direct drill) treatments, respectively, over the study period. At 90 cm depth these figures were 53%, 58% and 58% for control, (cover crop & reduce tillage) and (cover crop & direct drill) treatments, respectively. When cover crops were present (i.e. winter 2013-2014) on the two cover cropped treatments, the bare soil treatment had a mean soil moisture of 46% at 10 cm depth which was considerably higher than treatment (cover crop & reduce tillage) (18%) and treatment (cover crop & direct drill) (24%). This illustrates that soil moisture tended to be lower under cover crops compared with bare soils. In summer 2014, mean soil moisture contents at 50 cm depth were 58%, 48% and 47% for control, (cover crop & reduce tillage) and (cover crop & direct drill) treatments, respectively, which again had no significant difference. Like soil temperature, it appeared that cover crops and different soil tillage systems had little effect on soil moisture during the summer.

Verhulst et al. (2010) believed that conservation tillage may increase infiltration and reduce runoff and evaporation compared to conventional tillage, thereby soil moisture is conserved and more water is available for crop uptake. A study by De Vita et al. (2007) who assessed the effects of different tillage methods on soil moisture in wheat fields in Mediterranean soils observed that soil moisture was consistently higher under no-tillage compared to conventional tillage. The study showed the importance of saving soil moisture through a reduced tillage system, particularly in a semi-arid environment characterised by low annual rainfall and high evapotranspiration. Analysis of the water condition in the soil profiles suggested that for both studied soils, Fluvisol and Cambisol, during the entire three year experimental period, the moisture conditions were better at all depths of the soil profile under reduced tillage than under conventional tillage. Although the differences were not always statistically significant, soils under reduced tillage generally had higher soil moisture compared to conventional tillage (Sławiński et al., 2012). However, some other studies, similar to the finding of this study, found no difference. In a study to record the differences in soil moisture between conventional and different modes of conservation tillage in a long-term field trial in a temperate climate, Gruber et al. (2011) observed that different soil tillage methods had little effect on soil moisture. There was no temporal trend in soil moisture during the experiment and soil moisture decreased with increasing soil depth in most of the treatments. They suggested that significant differences between treatments were difficult to detect because of high variances between years and their interaction effects. However, slightly higher moisture contents were observed in no-till compared to all other treatments and indicates the effect of preserving soil moisture by reducing soil disturbance.

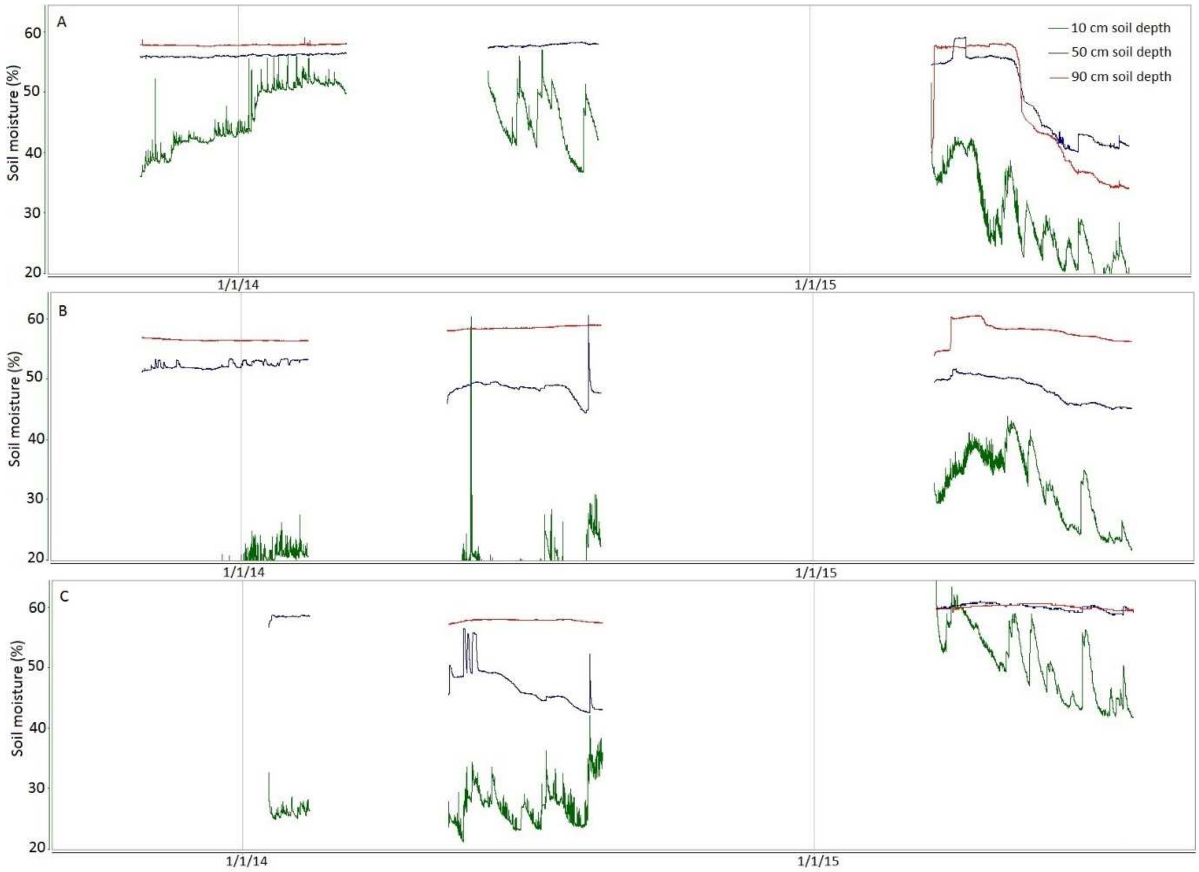

Figure 3.11 Soil moisture at three soil depths in (A) control treatment (no cover crop with conventional ploughing), (B) treatment (cover crop with reduced tillage) and (C) treatment (cover crop with direct drill). Note: the gap in the data are due to absence of the probes in fields because of field operations.

3.2.4.3 Bulk density (BD)

Although soil bulk density (BD) varies seasonally, snapshot measurements provide an indication of the status of the soil structure. There is a broad range of BD values for topsoil across the catchments (min = 1.15 g cm^{-3}; max = 1.89 g cm^{-3}) which are within the typical range of values observed for other topsoils across England and Wales (Hall et al., 1977; Rawlins, 2011).

Before the experiments (i.e. May 2013), BD measurements were the same for both (no cover crop & conventional plough) and (cover crop & reduced tillage) treatments with value of 1.50 g cm^{-3}, whilst treatment (cover crop with direct drill) had significantly ($p < 0.05$) higher BD with a value of 1.57 g cm^{-3} (Table 3.6). One year after the experiment (i.e. May 2014) soils under reduced cultivation treatment had a significantly ($p < 0.05$) higher BD (1.59 g cm^{-3}) than the soils from control (1.49 g cm^{-3}) which was conventionally ploughed. Soils from the direct drilled had significantly ($p < 0.05$) higher BD (1.67 g cm^{-3}) than both the other two treatments.

The same pattern was observed during the subsequent sampling round (i.e. June 2015), with BD significantly higher in both direct drill (1.50 g cm^{-3}) and minimum tillage (1.39 g cm^{-3}) than conventional tillage (1.34 g cm^{-3})

The results here show that BD was significantly greater under both reduced and no-tillage operations than under conventional tillage. Verhulst et al. (2010) believed that the effect of tillage on soil BD is mainly confined to the topsoil. In deeper soil layers, BD is usually similar in zero and conventional tillage systems. A plough pan may be created by tillage immediately beneath the tilled soils, causing higher BD in this horizon in conventionally tilled fields. Jabro et al. (2011) observed that there were no significant difference in soil BD in the first and second year of an experiment between conventional and strip tillage in a two-year study carried out on sandy loam soils in North Dakota. They concluded that reduced tillage reduced soil compaction overall. Contrary to this, Hernanz et al. (2002) determined BD in a loam textured soil in the semi-arid conditions of central Spain after a 13-year experiment. They observed significantly higher BD under no-tillage than under conventional tillage at 0-10 cm depth. Differences in BD between tillage methods over longer time periods (>15 years) have been somewhat more consistent. Soil bulk densities were higher in the surface layer of zero tillage than conventional tillage after 23 years on a silt loam soil with a maize-soybean rotation in Minnesota (Dolan et al., 2006). Similarly, Gál et al. (2007) reported higher (10-17% higher) BD in the 0-30 cm layer under zero tillage compared with conventional tillage on a silty clay loam soil in Indiana after 28 years, but no difference at the 30-100 cm layer.

Hence, most of the studies have observed higher BD under minimum and zero tillage than under conventional tillage. Gál et al. (2007) believed that this higher value of BD under reduced compared to conventional tillage is because the soil becomes denser over time due to the effect of consolidation when it remains untilled. Even in temperate climate, the loosening effects of annual freezing and thawing cycles, wetting and drying, and soil organism activities are not enough to prevent this increase in soil BD.

Table 3.6 Mean soil bulk density (BD) measurements in the three mitigation treatments for three soil sampling dates. Means followed by different superscripted letters are significantly different ($p < 0.05$) for the same sampling date. Means followed by different superscripted numbers are significantly different ($p < 0.05$) for the same treatment. SE is standard error

	May 2013		May 2014		June 2015	
Treatments	BD (g cm^{-3})	SE	BD (g cm^{-3})	SE	BD (g cm^{-3})	SE
No cover crop & Conventional plough (control)	1.50^{a1}	0.02	1.49^{a1}	0.03	1.34^{a2}	0.04
Cover crop & reduced tillage	1.50^{a1}	0.02	1.59^{b1}	0.04	1.39^{b12}	0.03
Cover crop & direct drill	1.57^{b1}	0.02	1.67^{c1}	0.03	1.50^{b12}	0.02

3.2.4.4 Penetration Resistance (PR)

Soil penetration resistance (PR) data are presented in Table 3.7. Before the experiment (i.e. May 2013) soils in all three treatments had similar PR values ranging from 430 to 460 N cm^{-2}, with no significant ($p > 0.05$) differences observed. The recorded values in all three treatments decreased after one year of the experiment (i.e. May 2014) to a range of 233-310 N cm^{-2}. The PR measurements were not made by the same person in all three years, so this change maybe because different pressures were applied on the penetrometer in one year compared to another. Nevertheless, comparison among the treatments for the same sampling date would still show differences in the soil properties as the same parson carried out the measurements for the same year. Soils under directly drilled had a mean PR value of 310 N cm^{-2} which was significantly ($p < 0.05$) higher than that recorded for the conventionally ploughed (245 N cm^{-2}) and the reduced tilled (233 N cm^{-2}). In the following year, significantly ($p < 0.05$) higher PR (426 N cm^{-2}) was observed for reduced tillage compared to conventionally ploughed (375 N cm^{-2}) and directly drilled (344 N cm^{-2}).

Although the result of soil PR under different treatments here are not consistent over years, it shows that PR is generally higher under reduced tillage than under conventional tillage. Abdollahi and Munkholm (2014) observed that there was a significant effect of tillage on PR at the 18-23 cm and 55-60 cm soil depths. At 18-23 cm, PR was significantly lower under conventional tillage (71 N cm^{-2}) than direct drill (90 N cm^{-2}). At 55-60 cm depth, the results were reversed with conventional tillage (218 N cm^{-2}) having significantly higher PR than direct drill (214 N cm^{-2}). Of the three tillage systems, conventional ploughing compared to minimum tillage or no-tillage resulted in a better soil fertility through producing the smallest mean weight diameter and lowest PR. Cover crops tended to reduce PR in the soil profile. This suggests that cover crops have the potential to alleviate soil compaction in the subsoil due to bio-pore formation and the stimulation of natural soil structure formation. The PR profile showed that the greatest differences among tillage treatments were in the upper layers of the un-trafficked inter-row, with no tillage systems having higher values compared to conventional tillage (da Veiga et al., 2007). Similarly, Thierfelder et al. (2005) concluded that measurements of PR and infiltration rate showed that practices of soil conservation tillage, such as reduced tillage and crop rotation, improved physical condition and prevented the development of soil crusting.

Table 3.7 Mean soil penetration resistance (PR) in the three mitigation treatments for three soil sampling dates. Means followed by different superscripted letters are significantly different ($p < 0.05$) for the same sampling date. Means followed by different superscripted numbers are significantly different ($p < 0.05$) for the same treatment. SE is standard error

	May 2013		May 2014		June 2015	
Treatments	PR (N cm^{-2})	SE	PR (N cm^{-2})	SE	PR (N cm^{-2})	SE
No cover crop & Conventional plough (control)	430^{a1}	14	245^{a2}	21	375^{a1}	14
Cover crop & reduced tillage	461^{a1}	11	233^{a2}	10	426^{b1}	14
Cover crop & direct drill	436^{a1}	9	310^{b1}	12	344^{a1}	14

3.2.4.5 Infiltration Rate (IR)

An example of a typical infiltration rate curve is presented in Figure 3.12 and the mean infiltration rate (IR) data for the three treatments over the study period is presented in Table 3.8. There were significant differences ($p < 0.05$) in IR among the treatments before the experiment started (i.e. May 2013), with control treatment (no cover crop & conventional plough) having the lowest IR at 44 mm hr^{-1}, followed by treatment (cover crop & direct drill) at 63 mm hr^{-1} and treatment (cover crop & reduced tillage) at 77 mm hr^{-1}. These differences may be partially due to differences in soil texture over the study area. Compared to other two treatments, control treatment has more clay rich soils and it is known that water moves more slowly through the small pore spaces in a clayey soil than it does through the large pores of a sandy soils. In May 2014, appreciable change occurred in the IR of the three treatments. Treatments (cover crop & reduced tillage) and (cover crop & direct drill), which had significantly higher IR values than control treatment in the previous year, now both had significantly ($p < 0.05$) lower values of 38 and 34 mm hr^{-1}, respectively, than control treatment. These two treatments were under reduced tillage and direct drill. This illustrates that IR under conservation tillage is lower than under conventional tillage. In the third year, the IR order of the treatments was same as the first year, with control treatment having the lowest IR value (3 mm hr^{-1}), treatment (cover crop & reduced tillage) the highest (8 mm hr^{-1}) and treatment (cover crop & direct drill) in between with an IR value of 4 mm hr^{-1}. However, the third year of IR data can be omitted in the comparison between different tillage systems because of a significant decline (one order of magnitude) in all the measurements which raised uncertainty about the accuracy of these data.

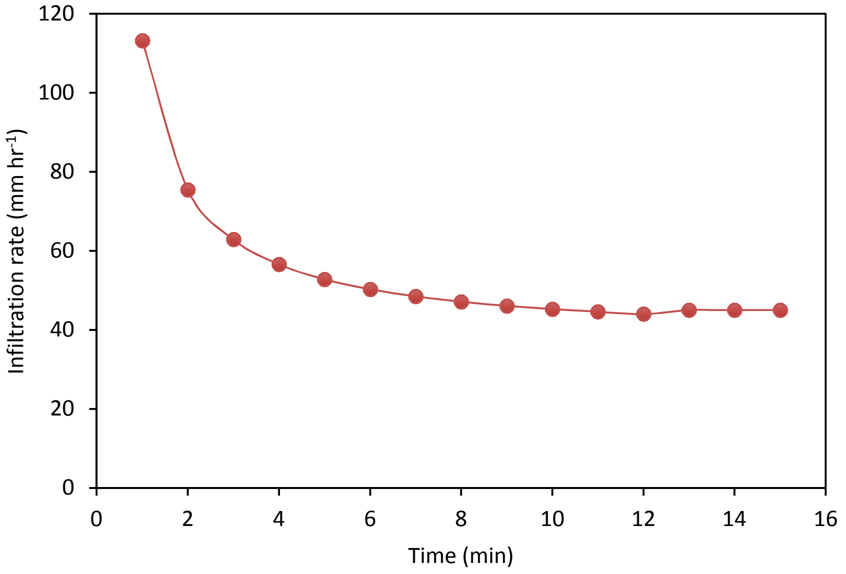

Figure 3.12 An example of an infiltration rate curve (soil location P2 in May 2014 sampling) with steady state infiltration rate (IR) value of 45 mm hr^{-1}.

The IR data in this study supported the BD and PR data and suggested that IR is lower for soils under reduced and no-tillage systems than under conventional tillage. Reynolds et al. (2002) highlighted that there are relatively few studies on the effects of various tillage systems on infiltration rate, despite its major role on water flow and chemical transport in the soil. In a review on tillage effects on infiltration rate, Strudley et al. (2008) reported that most tillage practices have pronounced effects on the infiltration rate of soil immediately following tillage, but these effects can diminish quickly. Long-term effects on the order of a decade or more tended to be less pronounced and are sometimes impossible to separate from traditional management. Infiltration rate is generally a poor indicator of management practice because of the high spatial and temporal variability. They concluded that both seasonal/annual measurements and long-term investigations are required to clarify the effects of management on infiltration rate. Lipiec et al. (2006) recorded that cumulative infiltration was highest under conventional tillage and was reduced by 61% under a no-tillage system. They concluded that the soil pore system under conventional tillage, with its higher contribution of large flow-active pores compared to minimum and no-tillage treatments, enhanced infiltration rate and water storage capacity.

However, some other studies (Arshad et al., 1999; McGarry et al., 2000; Vogeler et al., 2009) observed that conservation tillage resulted in a higher pore connectivity and higher infiltration rate than conventional tillage. Jabro et al. (2009) found that the IR at the soil surface (0-10 cm) did not differ significantly between conventional tillage and strip tillage. But, the effect of tillage on the IR at 10-15 cm depth was well pronounced. The values were 23% and 138% greater for strip tillage than for conventional tillage at the Nesson and EARC sites, respectively. As a result of repeating tillage, plough pans (compact layers) are created at shallow depths (18-25 cm) which reduce infiltration and saturated soil conditions develop above the pan leading to increased surface runoff. Plough pans also limit the growth of roots and thus reduce the amount of water available to the plant. It was concluded from their study that strip tillage can disrupt the plough pans and thus reduce surface runoff and erosion whilst increasing infiltration and transpiration rates (Temesgen et al., 2012). Miller et al. (1999) suggested that the greater infiltration potential under reduced or no-tillage systems is linked to increased surface residues, a greater bio-porosity caused largely by earthworm burrows, and soil pores that are more continuous because they are not disrupted by tillage. Conversely, other studies observed no significant change in IR under these two treatments. Topaloğlu (1999) found that tillage practices had no appreciable effect on infiltration rates in sandy clay loam soils. Kennedy and Schillinger (2006) showed that site, slope position nor tillage practice (traditional tillage and no-tillage) had any impact upon ponded infiltration rate. Karlen et al. (1994) reported IR values for no-tillage system, chisel and plough systems of 22 mm hr^{-1}, 10 mm hr^{-1} and 9 mm hr^{-1}, which were not significantly different.

Table 3.8 Mean infiltration rate (IR) of the soil in the three mitigation treatments for three soil sampling dates. Means followed by different superscripted letters are significantly different ($p < 0.05$) for the same sampling date. Means followed by different superscripted numbers are significantly different ($p < 0.05$) for the same treatment. SE is standard error

	May 2013		May 2014		June 2015	
Treatments	IR (mm hr^{-1})	SE	IR (mm hr^{-1})	SE	IR (mm hr^{-1})	SE
No cover crop & Conventional plough (control)	44^{a}	3.4	56^{a}	7.7	3^{a}	1.8
Cover crop & reduced tillage	77^{b}	4.9	38^{b}	4.6	8^{b}	0.9
Cover crop & direct drill	63^{c}	3.7	34^{b}	4.8	4^{a}	1.0

3.3 Summary

The results presented in this chapter indicate that cover crops and reduced cultivation systems did not significantly reduce soil nitrate concentrations in topsoils across the study area. However, the effectiveness of cover crops in reducing soil nitrate increased with increasing soil depth. The mean soil nitrate concentration under cover crops at 15 cm depth was 2.83 mg kg^{-1} which was not significantly lower ($p > 0.05$) than that under non-cover crop fields (3.04 mg kg^{-1}). But in deeper soil layers (i.e. 45 cm and 75 cm), soil nitrate was significantly lower in the cover crop treatments (2.18 mg kg^{-1} and 0.76 mg kg^{-1}) than the non-cover treatment (3.76 mg kg^{-1} and 3.54 mg kg^{-1}) for the two soil depths, respectively. This means that compared to bare soils, cover crops depleted soil nitrate concentrations by 7%, 42% and 79% at 15 cm, 45 cm and 75 cm depths, respectively. This substantial reduction in soil nitrate concentration in the deeper soil layers is due to the type of cover crop planted here. Oilseed radish has a thick, deep root that can help break up compacted soil layers and scavenge nitrate that has leached beyond the rooting zone of other crops. This capability of scavenging nitrogen from deep within the soil profile offers radish cover crops an extra advantage compared to other shallower rooting cover crop varieties because it is able to capture this deeper soil nitrate that would otherwise leach into groundwater or rivers via the field drain network.

Surface soil organic carbon (SOC) contents for the measures fields were relatively low (median 1.28%) compared to the median for the East Anglian region as a whole (2.24 %) based on a national survey of topsoils between 2002 and 2005. Thus, in this study, elevating the SOC content was part of the overall package of improvements in soil quality that were hoped to be achieved through trialling cover crops and reduced tillage practices. However, the results revealed that SOC contents were not significantly improved by cover crops or reduced tillage because control treatment (no cover crop & conventional plough) had consistently higher SOC concentrations than both (cover crop & reduced tillage) and (cover crop & direct drill) treatments. Regardless of the differences in farming practices, the order of SOC contents for the treatments remained the same on each of the four sampling dates. This is probably because soils in control treatment are dominated by clays which generally have a higher SOC content than the sandy soils in the other two treatments. Whilst no significant improvement in SOC content was observed during this two-year study, it is possible larger increases could occur over a longer time period (5-10 years).

As far as other soil macronutrients (P, K, Mg and SO_4) are concerned, the overall quality of soil in the study area is not in an ideal condition. Soil phosphorus concentrations were elevated, thus posing a surface water contamination risk, whilst the other macronutrients were generally below normal the range. Therefore, lowering of soil phosphorus levels and enhancing soil potassium, magnesium and sulphate concentrations were sought through the implementation of cover crops and reduced cultivation systems. The results presented in this chapter show that cover crops and reduced tillage retained soil P levels in the topsoil compared to conventional practices. This is a benefit when P is a limiting nutrient, but in the study area where P is an environmental problem, this could potentially have deleterious impacts on the river by increasing the likelihood of soluble P losses in surface runoff events. Regarding soil potassium, it was consistently found that cover crops and reduced tillage helped to maintain potassium concentrations in the topsoil compared to conventional practices. During the two year study, potassium concentrations under the cover crop and direct drilled fields increased by 27 mg L^{-1} compared to a 15 mg L^{-1} decline under conventional practices. This may in part be due to the cover crop minimising leaching and erosion, whilst direct drilling reduced soil structural disturbance. It was also observed that direct drilling tended to retain greater amounts of potassium in the topsoil than minimum tillage. No significant increases in soil magnesium and sulphate concentrations were observed in response to the mitigation measures.

It was hypothesised that the implemented measures would improve the physical quality of the soil by increasing soil temperature, moisture content and infiltration rate, whilst decreasing bulk density and penetration resistance. The results revealed that soil temperature and soil moisture did not change under different treatments. Bulk density was consistently and significantly higher for soils under direct drilling and reduced tillage than conventional ploughing. This maybe because the soil becomes denser over time due to the effect of consolidation and compaction when it remains untilled. Although the soil penetration resistance results under different treatment were not consistent, it appeared that penetration resistance was generally higher under reduced tillage than under conventional tillage practices, corresponding to higher bulk density. The infiltration rate data supported the bulk density and penetration resistance data and suggested that infiltration is lower for soils under reduced and no-tillage systems than under conventional tillage. Overall, it is concluded that implementing these mitigation measures has, to date, not substantially improved soil quality across the study area.

Chapter 4: EFFECTS OF COVER CROP AND REDUCED CULTIVATION ON SOIL NITRATE LEACHING

4.1 Introduction

Agriculture is a major contributor to environmental problems such as water pollution. Water pollution from nitrogen (N) is a serious problem in both Europe and developed countries (Ramos et al., 2002). The increased use of N fertilisers to meet increasing demands for food has increased the amount of residual inorganic N remaining in the crop root zone after harvest, which is then susceptible to winter leaching, especially in humid region soils (Drury et al., 2014). In this chapter, the main focus is on nitrates (NO_3^-)because a number of leaching and drainage studies have consistently found that NO_3^- is the dominant form of N present in soil water (Jacinthe et al., 1999; Jaynes et al., 2001; Dinnes et al., 2002; Macdonald et al., 2005).

Nitrate leaching occurs when there is an accumulation of NO_3^- in the soil profile that coincides with, or is followed by, a period of high drainage and heavy rainfall. Because nitrate and most soils in temperate regions are negatively charged, nitrate is not retained by the soil and is thus the dominant form of N leaching (Di and Cameron, 2002). A high concentration of nitrate in drinking water is considered harmful to human health, particularly for infants less than one year old. It can interfere with the transport of oxygen in the blood leading to so called 'blue baby syndrome' (Knobeloch et al., 2000) . To protect human health, world and national health organisations have established drinking water standards limiting NO_3^- concentrations to maximum of 11.3 mg NO_3^--N L^{-1} (Fewtrell, 2004). Furthermore, high nitrate concentrations in surface water bodies can cause deterioration in water quality, resulting in eutrophication, algal blooms and fish poisoning.

The main factors influencing the amount of nitrate leached from a particular land use are soil type, climate condition and management practices. Nitrate leaching rates are usually lower from fine-textured soils than from coarse-textured soils, because of slower drainage and greater potential for denitrification (Di and Cameron, 2002). The depth of soil above the groundwater table is also an important factor, with nitrate reaching the groundwater quicker in shallow soil than in deep soil. Earthworm channels, root pore spaces and large cracks can accelerate solute transport and nitrate leaching (Silva et al., 2000). In addition to soil characteristics, climate and season can affect nitrate leaching, with the amount of drainage being one of the determining factors. Nitrate leaching usually increases in seasons with a high soil moisture content and throughflow, for example during autumn and winter when evapotranspiration is low and rainfall totals are high. Summer weather conditions can also have an effect on nitrate leaching in the following winter. Long periods of hot summer weather were identified to cause more nitrate leaching over the subsequent winter than did cool wet summers. This is perhaps due to the lower N uptake in pasture land, lower denitrification loss and greater mineralisation upon rewetting following a long, dry and hot summer (Scholefield et al., 1993; Di and Cameron, 2002). The amount of applied N fertiliser also has an impact on nitrate leaching. If N fertiliser is applied at a quantities substantially above that required by crops for growth, then the surplus N will likely accumulate in the soil profile and readily leach into ground and surface water bodies.

The increased nitrate leaching loss not only creates environmental problems by increasing nitrate contamination of groundwater and surface waters, but also causes agronomic and economic losses by decreasing N use efficiency. Thus, reducing nitrate leaching from fields into ground and surface water is a key management priority. Several strategies have been proposed and studied to reduce nitrate leaching including reducing overall N fertiliser application loads, growing a cover crop to capture excess nutrients over winter, limiting N application rates prior to or during high leaching seasons, using buffer zones between fields and streams and using nitrification inhibitors (Di and Cameron, 2002). It has also been suggested that minimum tillage may cause less nitrate leaching than conventional tillage, but research so far has demonstrated mixed results with both higher and lower nitrate leaching under reduced tillage compared with conventional tillage (Meek et al., 1995; Di and Cameron, 2002). However, significant reductions in nitrate leaching may be achieved through a combination of several mitigation measures rather than a single measure. The objective of this chapter is to assess the effects of the implemented mitigation measures including cover crop, reduced tillage and direct drill on nitrate concentrations and fluxes. Although the focus is on nitrate leaching, the changes in two other forms of nitrogen (i.e. ammonium and nitrite) under different treatments are also discussed.

4.2 Results and discussion

4.2.1 Rainfall

The annual precipitation total for the hydrological year October 2013 to September 2014 was 786 mm in weather station (A) (Figure 2.5), which is higher than the 1981-2010 mean annual total precipitation of 674 mm reported for the East Anglia region in Reepham weather station

(Meteorological Office, 2014). Within the study period, summer 2013 (106 mm) was drier than summer 2014 (194 mm), whereas winter 2013-2014 (193 mm) was wetter than winter 2014-2015 (161 mm) (Figure 4.1). The overall mean monthly precipitation was 58 mm, with the highest monthly total occurring in May 2014 (129 mm) and the lowest in July 2013 (14 mm). This lowest monthly rainfall is far lower than the mean lowest monthly precipitation of 41 mm reported for period of 1981-2010, making summer 2013 one of the drier summers in recent years. Since water sampling was carried out weekly, the rainfall data here are presented at weekly intervals so that any changes in water quality parameters can be directly related to rainfall. The three highest weekly rainfall events during the study period are highlighted in red in Figure 4.1. The highest weekly rainfall occurred in mid-October 2013 (68 mm), followed late May 2014 (61 mm) and mid-October 2014 (54 mm).

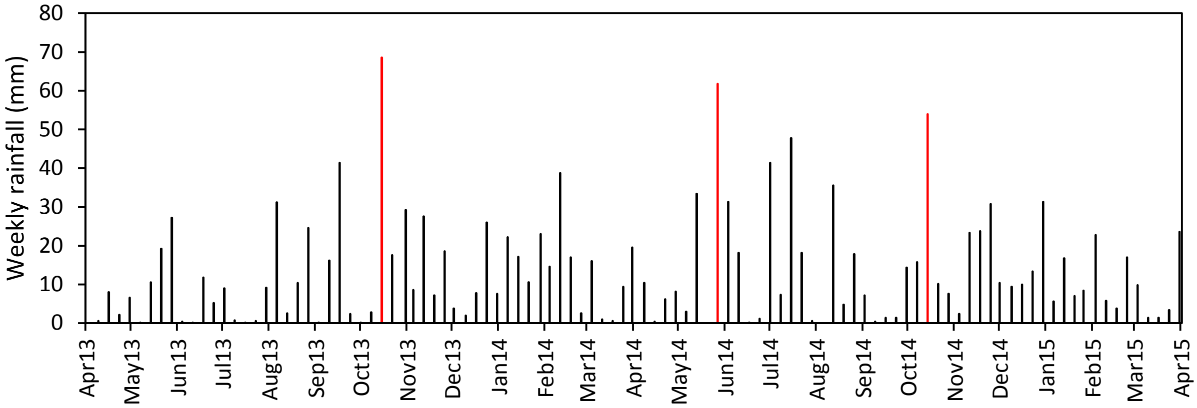

Figure 4.1 Weekly rainfall during the study period with three major rainfall events highlighted in red.

4.2.2 Nitrate

4.2.2.1 Nitrate concentration

Table 4.1 summarises the mean and range of NO_3^- concentrations measured in the field drains underlying each of the three experimental treatments between April 2013 and April 2015. Drains in the table were ordered from upstream towards downstream. In total, 500 water samples were collected during weekly sampling of the field drains during the two-year period. Some drains, for instance D2, D4 and D1 (see Figure 2.8 for location), were flowing almost continuously throughout the study period, such that a greater number of samples were collected from these drains. Conversely, other drains such as D16, D3 and D8 flowed for a shorter time and yielded fewer samples. The depths of the sampled drains were 1-1.2 m from ground surface, and groundwater table was 1.7±0.5 m and 1.8±0.3 m from ground surface for 2013 and 2014 hydrological years, respectively (see Table 2.1). The groundwater table, therefore never reached the field drainage. If groundwater level reached drains then drying of the drain flow didn't happen during summer because of continuous supply of flow from ground water. The mean nitrate concentration of all 500 samples was 5.2 mg N L^{-1}.

Water samples from drains D8 and D10 had mean NO_3^- concentrations of 9.8 mg N L^{-1} and 9.5 mg N L^{-1}, respectively, higher than any other drain. Among all the samples collected, the single highest NO_3^- concentration was measured in D10 (37.4 mg N L^{-1}) in late May 2014. The second highest value was also measured in D10 (24 mg N L^{-1}) in early June 2014, followed by the third highest value in D8 (21.7 mg N L^{-1}) collected in mid-October 2014. For most of the drains the lowest measured NO_3^- concentration was <1.5 mg N L^{-1}. The exception to this was D11, which had a minimum value of 3.3 mg N L^{-1} and a highest value of 9.0 mg N L^{-1}, giving D11 the smallest range (5.7 mg N L^{-1}) of NO_3^- concentration of any drain, whilst D10 had the longest range (36.1 mg N L^{-1}).

Table 4.1 Summary of field drain NO_3^- concentrations in the three experimental treatments for samples collected during the April 2013-April 2015 study period

Treatments	Drain ID	n	Mean of NO_3^- (mg N L^{-1})	Range of NO_3^- (mg N L^{-1})
No cover crop & conventional plough (control)	D10	59	9.5	1.4-37.4
	D8	40	9.8	0.9-21.7
Cover crop & reduced tillage	D3	41	4.4	0.5-11.4
	D16	39	4.2	1.5-10.2
	D1	77	4.8	0.9-11.1
Cover crop & direct drill	D11	20	5.9	3.3-9.0
	D2	86	2.3	0.1-12.0
	D4	80	2.8	0.2-15.9
	D6	58	6.4	1.1-14.1

Temporal variability in nitrate concentrations

The temporal variability of NO_3^- concentrations in individual field drains in each of the three treatments is shown in Figure 4.2. Gaps in the measurement are due to a lack of drain flow which usually happened each summer. As summer 2013 was approaching, field drain NO_3^- concentrations gradually decreased in all three treatments, perhaps due to both a decline in precipitation and decline of potential leachable nitrate by crops during this growth period. There was a lower mean NO_3^- concentration in summer 2013 than in summer 2014, possibly because of low rainfall in the former (106 mm) compared to the latter (194 mm). It was observed that NO_3^- concentrations tended to be lower in summer than in winter throughout the study period. In October 2013 when the cover crop was growing, significantly lower NO_3^- concentrations were measured in the field drains under cover crop fields compared to the bare control fields. No notable increase was observed in NO_3^- concentration in the drains after incorporation of the cover crop in March 2014. This is in agreement with the soil NO_3^- data (Figure 3.3) which also illustrated no change in soil NO_3^- concentration after incorporation of the cover crop.

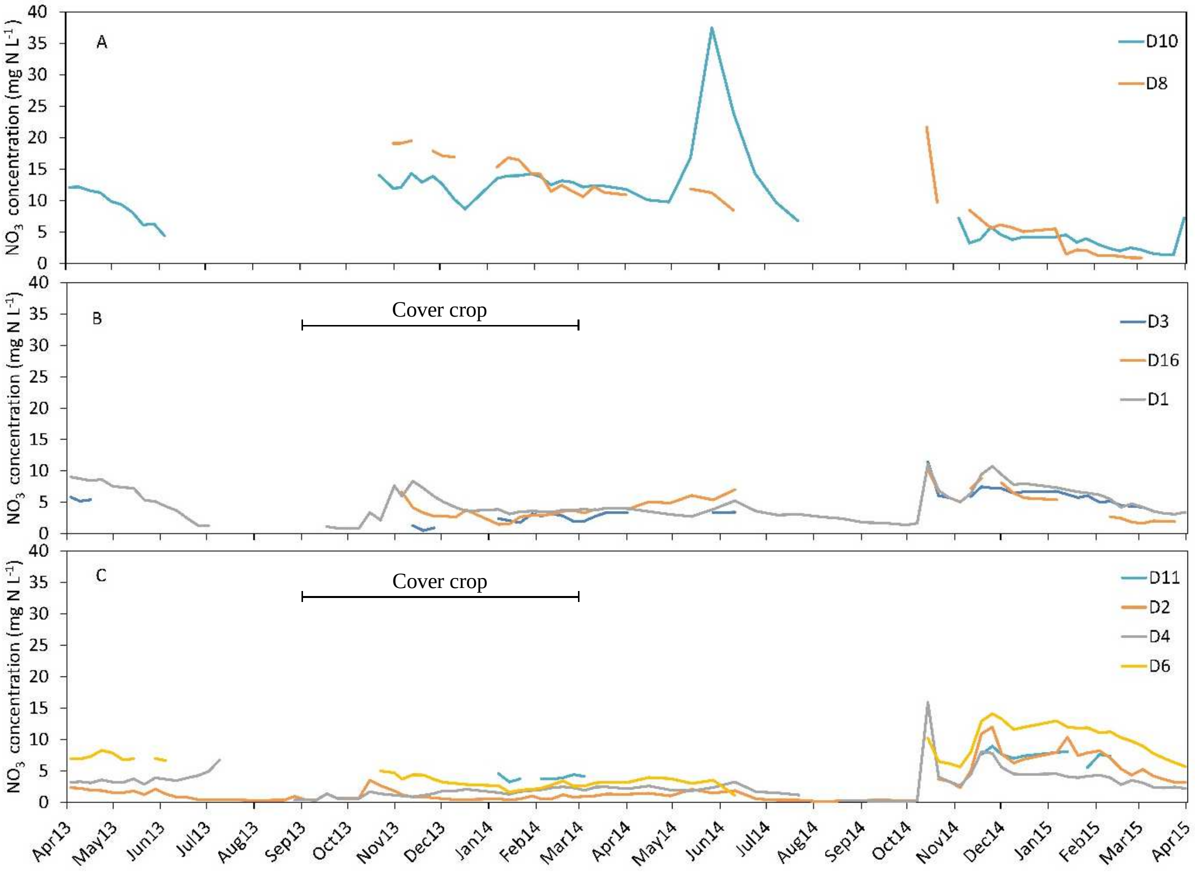

Figure 4.2 Temporal variation in field drain nitrate (NO_3^-) concentrations in (A) control treatment (no cover crop & conventional plough), (B) in treatment (cover crop & reduced tillage) and (C) in treatment (cover crop & direct drill) throughout the study period.

NO_3^- concentrations in the drains responded differently to the three main rainfall events that occurred during the study. The highest weekly rainfall total (68 mm) occurred in mid-October

2013 generated no significant change in the field drain NO_3^- concentrations. NO_3^- concentrations in drains from both (cover crop & reduced tillage) and (cover crop & direct drill) treatments were not notably affected by the second highest weekly rainfall that occurred in May 2014. However, NO_3^- concentrations in drains from control treatment, solely contributed to by D10, increased substantially with concentrations of 37.4 mg N L^{-1} recorded the highest NO_3^- concentration measured throughout the entire study. Unlike the heavy rainfall in autumn 2013, the storm event on 14th October 2014 resulted in a sharp and dramatic rise in NO_3^- concentrations in all flowing drains in each of the three treatments. About half of the sampled drains (D1, D3, D4, D8 and D16) contained the highest NO_3^- concentrations of the entire study period. For example, NO_3^- concentrations in D3 did not exceed 7.5 mg N L^{-1} at any other point over the two years of data collection, yet the collected sample during this rainfall event contained 11.4 mg N L^{-1}. This storm caused the drains from control treatment to flow with a high NO_3^- concentration of 21.7 mg N L^{-1}.

Effect of the cover crop on nitrate concentration

A summary of the effect of the oilseed radish cover crop on field drain NO_3^- concentrations is presented in Table 4.2. Field drains under the cover crop typically contained <6 mg N L^{-1} of NO_3^-, whereas samples from drains under bare soil had concentrations >8 mg N L^{-1}. The seven fields with the cover crop had NO_3^- concentrations at least one order of magnitude lower than those fields with no cover crop. The overall mean NO_3^- concentration was significantly ($p < 0.001$) lower in drain water samples under the cover crop (2.5 mg N L^{-1}) than drain water samples from bare soil (13.9 mg N L^{-1}). This represents a ~82% reduction in nitrate concentrations due to the presence of the cover crop.

This result is in good agreement with a large body of previous research investigating the effects of cover crops on reducing nitrate concentrations in water compared to bare soil. Staver and Brinsfield (1998) reported that groundwater NO_3^- concentration decreased by more than 60% in field-scale watersheds during a nine-year period as result of the use of a rye cover crop. Justes et al. (1999) observed that NO_3^- concentration in drainage water was depleted by 50%, from 20 to 10 mg N L^{-1} by the presence of an oilseed radish cover crop compared to bare soil. Kaspar et al. (2012) reported that a rye winter cover crop significantly reduced drainage water NO_3^- concentrations by 48% over five years, with a 58% reduction in the first four years. Kaspar et al. (2012) also observed that on autumn oat cover crop reduced NO_3^- concentrations by 26%.

A reduction of 21-38% in the flow weighted mean NO_3^- concentrations in tile drainage water was also reported by Drury et al. (2014) as a result of the use of a winter wheat cover crop. Premrov et al. (2014) observed that NO_3^- concentrations across three years were reduced by more than 70% using Mustard cover crop compared to no cover crop under both conventional and reduced tillage.

Table 4.2 Mean and range of NO_3^- concentration in field drains under two different treatments during cover crop trials between September 2013 and March 2014. SE is standard error

	Drain ID	n	Mean of NO_3^- (mg N L^{-1})	Range of NO_3^- (mg N L^{-1})	SE
Without cover crop	D10	19	12.8	8.7 - 14.3	0.33
	D8	16	15.3	10.6 - 19.5	0.75
Mean			13.9		
With cover crop	D3	14	2.2	0.5 - 3.2	0.22
	D16	17	3.2	1.5 - 6.6	0.28
	D1	24	3.9	0.9 - 8.4	0.41
	D11	8	3.9	3.3 – 4.6	0.15
	D2	26	1.0	0.3 – 3.6	0.14
	D4	26	1.5	0.4 – 2.5	0.12
	D6	19	3.2	1.6 – 5.0	0.22
Mean			2.5		

Alongside field drain samples, soil water was also collected intermittently from porous pots buried at 90 cm depth across the nine mitigation measures fields (see Figure 2.9 for locations) and the NO_3^- concentrations analysed (Figure 4.3). There was frequently large variation in NO_3^- concentrations between porous pots just a few metres apart. In February 2014, for instance, large variations were measured at site P8 where NO_3^- concentrations in pot numbers 6 and 9 were 3.3 and 2.2 mg N L^{-1}, respectively, whilst pots 4 and 10 had concentrations of 19.7 and 21.2 mg N L^{-1}, respectively. The greatest difference was noted in February 2015 at site FAR9 when soil water from pot 9 contained an extremely high NO_3^- concentration (48 mg N L^{-1}), approximately one order of magnitude higher than the other pots from the same site and even higher than the highest value recorded in the field drains (37 mg N L^{-1}).

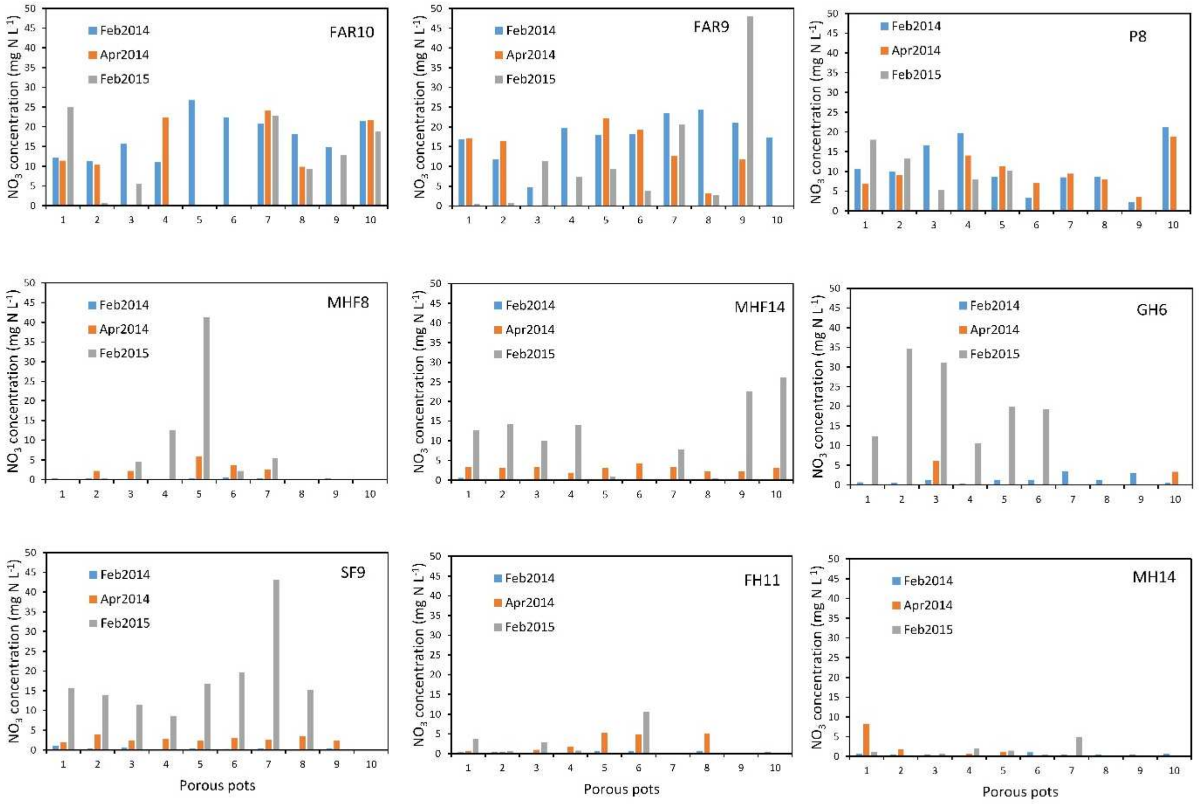

Figure 4.3 Spatial variation in soil water NO_3^- concentrations collected from porous pot samples in control treatment = no cover crop & conventional lough (top row), treatment (cover crop & reduced tillage) (middle row) and treatment (cover crop & direct drill) (bottom row). The distance between two pots is 1 metre. Locations of porous pots are shown in Figure 2.9.

The finding of a substantial reduction in soil water NO_3^- concentrations in field drains with a cover crop is also supported by the porous pot data summary in Table 4.3. In February 2014 when the cover crop was growing, porous pot NO_3^- concentrations were significantly ($p < 0.05$) lower in fields under the oilseed radish cover crop than under bare soil in all three soil types. Fields under the cover crop had mean soil water NO_3^- concentrations of 0.55 mg N L^{-1}, whereas bare soils had a mean value of 15.3 mg N L^{-1}. This represents a 96% reduction in NO_3^- concentrations by the cover crop, which is higher than the 82% reduction in NO_3^- concentrations calculated from the field drain data. This larger reduction in porous pot NO_3^- concentrations is because the porous pot samples in the cover crop fields had lower NO_3^- concentrations than the drain samples. The mean NO_3^- concentration in all porous pots samples from treatment (cover crop & direct drill) was 0.5 mg N L^{-1}, whereas this figure was 2.1 mg N L^{-1} in the drain samples for the same treatment. As well as providing evidence of a further reduction in soil NO_3^- loss to the deeper soil profile by the radish cover crop (Figure 3.5), this might also be another indication that the radish cover crop is able to scavenge NO_3^- from deeper within the soil profile.

Although most of the drains are deeper than 1m beneath the field surface, their collected water is likely to come from topsoil by preferential flow. Conversely, the water collected in the porous pots probably comes from the soil surrounding the pots (i.e. 90 cm depth) as the water is drawn into the pots under vacuum by a pump.

A slight increase in the mean porous pot NO_3^- concentration under the cover crop is observed between February 2014 (0.55 mg N L^{-1}) and April 2014 (2.7 mg N L^{-1}). This is potentially due to the incorporation of the cover crop within the soil which causes the release of contained N into the soil. However, the significant differences ($p < 0.05$) between the porous pot samples under a cover crop and no cover crop fields remained in all soil types. Fields under the cover crop had mean NO_3^- concentrations of 2.7 mg N L^{-1}, whilst bare soils had a mean concentration of 13.2 mg N L^{-1}. In contrast to the porous pot data from the cover crop fields in February 2014, which consistently showed lower NO_3^- concentrations than the drain data, porous pot samples in the cover crop fields in April 2014 (2.7 mg N L^{-1}) had very similar concentrations to the field drain samples at this time (2.9 mg N L^{-1}). This means that the collected data from field drains and porous pots were in good agreement with each other apart from the time when cover crop was growing (i.e. September to March) in which the data from the porous pot usually had lower NO_3^- concentrations.

Table 4.3 Mean of soil water NO_3^- concentrations in porous pot samples from three soil types in the control treatment (no cover crop & conventional plough), treatment (cover crop (CC) & reduced tillage (RT)), and treatment (cover crop (CC) & direct drill (DD)) collected on three sampling occasions. Means followed by different superscripted letters are significantly different ($p < 0.05$) for the same sampling date and same soil type. Means followed by different superscripted numbers are significantly different ($p < 0.05$) for the same treatment and same soil type

Soil type	Treatments	NO_3^- (mg N L^{-1})					
		n	Feb. 2014	n	Apr. 2014	n	Feb. 2015
Sandy Loam	control	10	17.5^{a1}	6	16.6^{a1}	9	10.6^{a1}
	(CC& RT)	10	0.3^{b1}	8	2.1^{b1}	8	8.3^{a2}
	(CC& DD)	8	0.5^{b1}	9	2.8^{b1}	8	18.1^{a2}
Sandy Clay Loam	control	10	17.6^{a1}	7	14.6^{a1}	9	11.6^{a1}
	(CC& RT)	9	0.2^{b1}	10	3.0^{b1}	10	10.9^{a2}
	(CC& DD)	8	0.4^{b1}	7	2.8^{b1}	6	3.2^{a1}
Clay Loam	control	10	10.9^{a1}	9	9.8^{a1}	5	10.9^{a1}
	(CC& RT)	10	1.4^{b1}	4	2.4^{b1}	6	21.3^{b2}
	(CC& DD)	10	0.5^{b1}	4	3.0^{b1}	7	1.6^{c1}

Effect of different tillage methods on nitrate concentration

During the 2014-2015 farming year, only different tillage operations (i.e. with no cover crop) were trialled as a mitigation measure in the three treatments. As such, the data collected in this period can be used to compare different tillage operations on water quality. Data from the porous pot samples collected in February 2015 showed no particular pattern among the treatments or soil types (Table 4.3). Although the presented mean values appeared to be different from each other, no significant ($p > 0.05$) differences were observed (except for the clay loam soil type) because of great variation in each batch of porous pot samples. The NO_3^- concentrations in both sandy clay loam and clay loam porous pot samples in treatment (cover crop & direct drill) were lower than that in the control treatment, but because sandy loam samples had extremely high NO_3^- concentrations (18.1 mg N L^{-1}), no notable decline in NO_3^- concentrations was observed compared to the control treatment.

It is also apparent from the field drain data (Figure 4.2) that NO_3^- concentrations during winter 2015 in (cover crop & direct drill) and (cover crop & reduced tillage) treatments were not lower than the control treatment. Results from both field drains and porous pots show that the practiced mitigation measures in 2015 (i.e. reduced tillage and direct drill) did not reduce soil water NO_3^- concentrations (Table 4.4). The mean NO_3^- concentrations for porous pot and drain samples under conventional tillage during September 2014 to March 2015 was 6.8 mg N L^{-1}, which was not statistically different from the 7.7 mg N L^{-1} and 7.1 mg N L^{-1} obtained for reduced tillage and direct drill fields, respectively.

Table 4.4 Mean NO_3^- concentrations in field drain and porous pot samples under different tillage methods from October 2014 to April 2015. Means followed by different superscripted letters are significantly different ($p > 0.05$). SE is standard error

Tillage type	n	Mean of NO_3^- (mg N L^{-1})	SE
Conventional tillage	61	6.8^a	1.0
Reduced tillage	84	7.7^a	0.8
Direct drill	102	7.1^a	0.5

Previous studies have demonstrated that the type of soil cultivation strongly influences NO_3^- concentrations, but the evidence that NO_3^- concentrations are higher for inversion compared to conservation tillage is contradictory. The results here agree with Joshi et al. (1994) who found no significant difference in mean NO_3^- concentrations between conventional and no-till systems. Similarly, Randall and Iragavarapu (1995) found insignificant differences between the average flow weighted NO_3^- concentrations of 13.4 mg N L^{-1} and 12.0 mg N L^{-1} for conventional and no-tillage corn production treatments, respectively. They concluded that the tillage system had minimal impact on NO_3^- leaching to subsurface drain flow. In a study on NO_3^- concentrations in field drainage under conventional and no tillage systems, Randall and Mulla (2001) again observed that NO_3^- concentrations were not influenced by tillage system. The insignificant difference in NO_3^- concentrations under conventional tillage and conservation tillage were also observed in other studies (Masarik et al., 2014; Premrov et al., 2014). Conversely, some studies have reported higher NO_3^- concentrations in conventional tillage than conservation tillage. Kanwar et al. (1993) monitored NO_3^- leaching beneath both continuous corn and corn-soybean rotations managed using mouldboard ploughing, chisel ploughing, ridge tillage and no-tillage practices. The three year average NO_3^- concentration in drainage water from continuous corn plots receiving mouldboard tillage was significantly greater (35.8 mg L^{-1}) than for the no-till treatment (22.2 mg L^{-1}). They believed that this lower NO_3^- concentration from no-till may have resulted from greater bypass flow, denitrification and immobilisation under non-ploughed systems.

4.2.2.2 Nitrate fluxes

The field drain area, drain flow rate and NO_3^- concentration were used to calculate field drain NO_3^- fluxes. Estimated drainage areas are presented in Figure 4.4 and Table 4.5. Map of field drain network was provided by farmers (see Appendix Figure D1). Drainage areas of the interested drains were delimited by polygon digitised using GIS with the contribution of field observations. The drainage area varied from 0.05 ha for D3 to 5.64 ha for D16, with the total area of all monitored drains equalling ~21 ha.

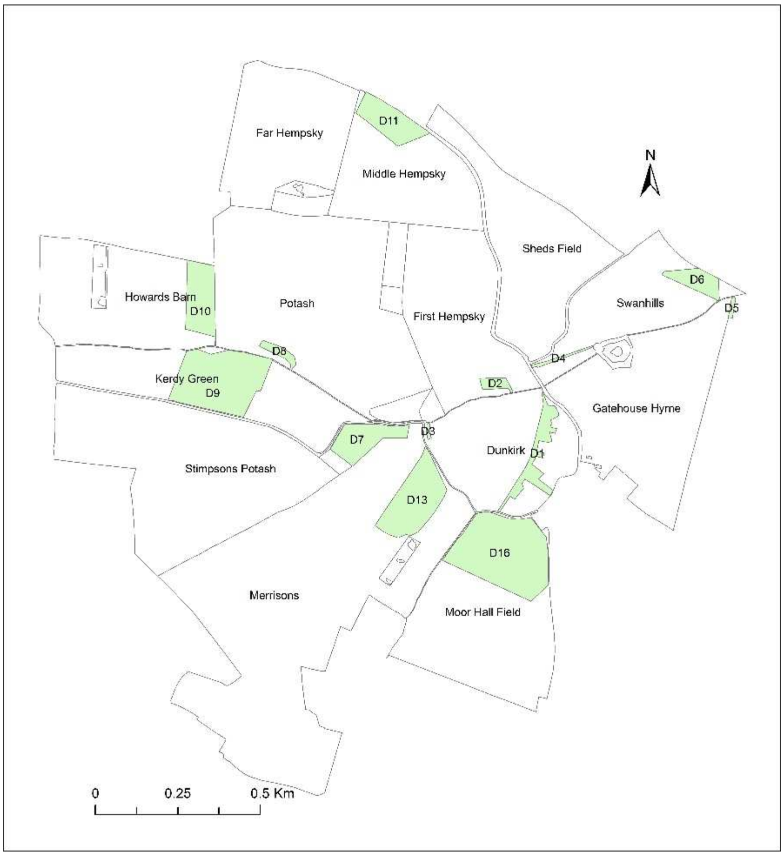

Figure 4.4 Estimated drainage area for the field drains monitored in this study.

Table 4.5 Estimated drainage area and calculated flow rate for the field drains monitored in this study

Drain ID	Estimated area (ha)	Mean flow rate (L sec^{-1})	Range of flow rate (L sec^{-1})
D1	1.25	0.30	0.00-2.35
D2	0.29	0.07	0.01-0.19
D3	0.05	0.04	0.00-0.21
D4	0.14	0.07	0.00-0.31
D5	0.08	0.05	0.00-0.17
D6	0.87	0.06	0.01-0.24
D7	1.60	0.19	0.00-1.12
D8	0.30	0.14	0.01-0.70
D9	4.49	0.26	0.01-0.85
D10	1.86	0.21	0.00-0.83
D11	1.64	0.16	0.02-0.44
D13	2.92	0.27	0.01-0.81
D16	5.65	0.14	0.01-0.78

Field drain flow rates were measured three times on each sampling occasion and a mean calculated. Drain flows in summer were lower than in winter. Generally, drains started flowing in autumn (October/November) and maintained high flow until spring (April/May) when flows gradually decreased, with most drying up during the summer (Figure 4.5). Drains D2 and D4 flowed almost continuously throughout the study period, whereas other drains (e.g. D11, D3 and D8) flowed for a shorter period of time. D13 and D16 were added to the sampling regime in November 2013. The calculated flow rates ranged from 0.001 to 2.35 L sec^{-1} with an average for all drains of 0.14 L sec^{-1}. D1 had the highest mean flow rate (0.30 L sec^{-1}), followed by D13 (0.27 L sec^{-1}), whilst D3 had the lowest mean flow rate (0.04 L sec^{-1}) (Table 5.2). Drain flow correlated (r=0.53) with drainage area as drains with larger area tended to have greater flows and drains with smaller areas had lower flows. D1 and D13 had the greatest flows and relatively large areas, whilst D3 and D5 had the lowest flows and the smallest drain areas. However, for other drains such as D11 and D16 with large area and small flow rate, this relationship did not hold true. The field drain flow rate was also correlated with rainfall. The second and the third storm events highlighted in Figure 4.1 created associated peaks in drain flow. The drain flow of D1, for example, increased significantly from 0.04 L sec^{-1} one week prior to the third storm event, to 1.5 L sec^{-1} during the storm. On the other hand, the first highlighted storm event generated no corresponding peak in drain flow, whilst smaller storm events such as the one that occurred on the 25th November 2014 generated some of the highest recorded flow rates of the entire study. Moreover, rainfall events during the summer did not generate a pulse in drain flow. Thus, whilst large increases in drain flow were produced by large storm events, not all storm events generate large increases in drain flow.

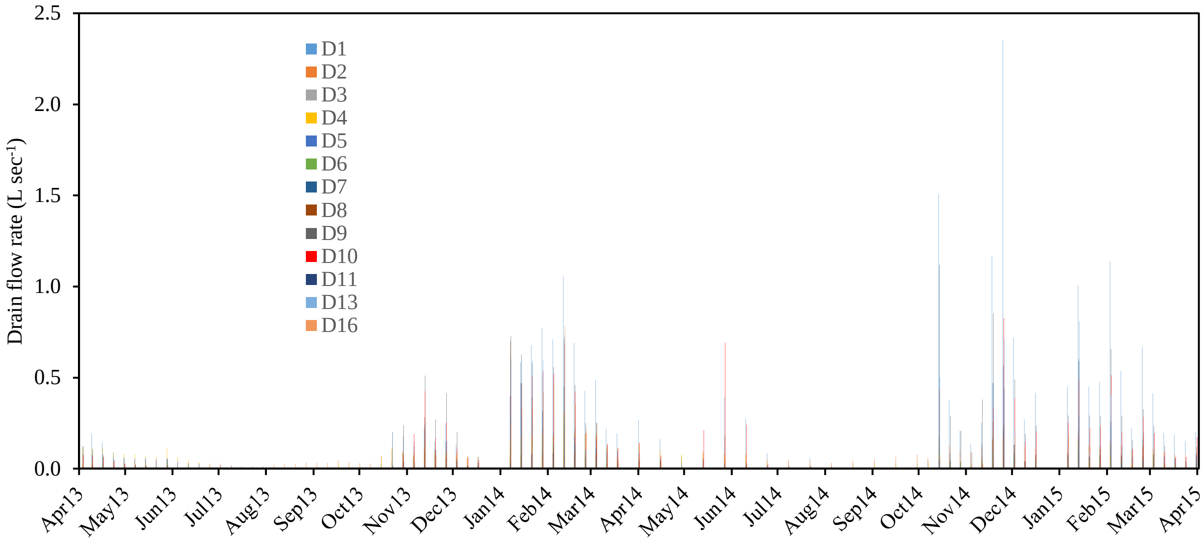

Figure 4.5 Field drain flow rates throughout the study period. Note the very low or no flow of the drains during both summers and high flow during autumn and winter.

A summary of NO_3^- fluxes in the field drains in the three experimental treatments is presented in Table 4.6. A mean NO_3^- flux of 35 kg N ha^{-1} a^{-1} was calculated for all the drains during the study, with the mean flux for most of the drains ranging from 10-40 kg N ha^{-1} a^{-1}. However, drains D11 and D16 had fluxes of <5 kg N ha^{-1} a^{-1}, whilst D8 (79 kg N ha^{-1} a^{-1}) and D3 (74 kg N ha^{-1} a^{-1}) had very high fluxes. Because of high variability in drain flow rate and drainage area, NO_3^- discharges within drain water from the agricultural soils varied considerably between the different drains. Nevertheless, the annual NO_3^- fluxes via drainage water were high in this study and were similar to the values of 10-90 kg N ha^{-1} a^{-1} measured by Rossi et al. (1991) in drainage water from arable land (with sugar beet, winter wheat or soybean) in northern Italy with an annual fertiliser input of about 150 kg N ha^{-1} a^{-1}.

Table 4.6 Mean of NO_3^- fluxes in the field drains under three treatments during April 2013 to April 2015. SE is standard error

Treatments	Drain ID	Mean of NO_3^- flux (kg N ha^{-1} a^{-1})	SE
No cover crop & conventional plough (control)	D10	26	6.1
	D8	79	21.6
Cover crop & reduced tillage	D3	74	21.6
	D16	2	0.4
	D1	43	9.6
Cover crop & direct drill	D11	4	1.2
	D2	27	4.9
	D4	41	4.5
	D6	10	2.0

Impact of the cover crop on nitrate fluxes

The impact of the cover crop on field drain NO_3^- fluxes is presented in Table 4.7. The overall mean NO_3^- flux was significantly ($p < 0.001$) lower from field drains underlying the cover crop (17 kg N ha^{-1} a^{-1}) than from drains underlying bare fields (113 kg N ha^{-1} a^{-1}). This represents a ~85% reduction in nitrate fluxes due to the presence of the cover crop. Many previous studies have found that cover crops significantly reduce NO_3^- leaching in field drains. In a field experiment on sandy loam and chalky loam soils at two sites in SE England, Macdonald et al. (2005) found that during the winter immediately after establishment, early sown cover crops (including forage rape, rye, white mustard, phacelia and ryegrass) decreased NO_3^- leaching by 29-91% compared to bare soil. Macdonald et al. (2005) concluded that cover crops are most likely to be effective when grown on freely drained sandy loam soils where NO_3^- leaching in bare fields is greatest. In a 25-year simulation using Root Zone Water Quality (RZWQ) model, Singer et al. (2011) concluded that using a winter cover crop can reduce NO_3^- fluxes by 19-28% for corn-soybean and corn-soybean-corn rotations at the watershed sub-basin scale. A number of best management practices including a cover crop, no-tillage and reduced nitrogen fertiliser were compared in field trials over 14–17 years and cover crops were found to be the most effective practices at decreasing NO_3^- leaching (36-62%) and remained efficient over longer timescales (Constantin et al., 2010).

Table 4.7 Mean and range of field drain NO_3^- fluxes under the two different cover crop treatments between September 2013 and March 2014. SE is standard error

	Drain ID	Mean NO_3^- flux (kg N ha^{-1} a^{-1})	SE
Without cover crop	D10	47	10
	D8	180	57
Mean		113	
With cover crop	D3	24	7
	D16	4	1
	D1	27	5
	D11	4	2
	D2	8	1
	D4	44	8
	D6	5	1
Mean		17	

Impacts of different tillage regimes on nitrate flux

In the 2014-2015 farming year, only different tillage regimes (i.e. with no cover crop) were trialled as a mitigation measure across the three treatments, such that data collected during this period can be used to compare different tillage practices on a particular parameter. As with NO_3^- concentrations, which showed no significant difference under different tillage practices, NO_3^- fluxes also did not change significantly ($p > 0.05$) between the three tillage types (Table 4.8), with the exception of reduced tillage. The mean NO_3^- flux from field drains under conventional tillage was 44.1 kg N ha^{-1} a^{-1}, which was slightly higher than the NO_3^- fluxes under direct drill (41.7 kg N ha^{-1} a^{-1}). Contrary to the hypothesis that reduced tillage may reduce NO_3^- leaching, higher field drain NO_3^- fluxes were recorded under reduced tillage (114 kg N ha^{-1} a^{-1}) than conventional tillage (44.1 kg N ha^{-1} a^{-1}). NO_3^- fluxes were also significantly ($p < 0.05$) higher under reduced tillage than under direct drill practices.

Similar to the results of this study, in an eight-year study to determine the potential effects of common agricultural practices on subsurface drainage water quality, Masarik et al. (2014) found that differences in the annual NO_3^- flux between conventional tillage (34 kg N ha^{-1} a^{-1})

and no-tillage (38 kg N ha^{-1} a^{-1}) corn treatments were not statistically significant. However, Masarik et al. (2014) found that NO_3^- fluxes from restored prairie land (0.07 N ha^{-1} a^{-1}) were consistently lower than both the conventional and no-tillage treatments.

Table 4.8 Mean field drain NO_3^- fluxes from different tillage systems between October 2014 and April 2015. Means followed by different superscripted letters are significantly different ($p > 0.05$). SE is standard error

Tillage type	Mean of NO_3^- flux (kg N ha^{-1} a^{-1})	SE
Conventional tillage	44.1^a	20.9
Reduced tillage	114^b	27.1
Direct drill	41.7^a	5.2

4.2.3 Ammonium

Table 4.9 summarises the field drain NH_4^+ concentrations and fluxes in the three experimental treatments during April 2013 to April 2015. In total, 376 NH_4^+ measurements were obtained from drains sampled weekly over this two year period. Some drains, for instance D2, were flowing almost continuously throughout the study period, so greater numbers of samples were collected from them. Conversely, other drains, for example D11 and D8 flowed for a shorter time and yielded fewer samples. The mean NH_4^+ concentration for all samples collected during the two years was 11.8 µg N L^{-1}. Water samples from some drains, for instance D8 and D10 had mean values of 16.6 µg N L^{-1} and 20.3 µg N L^{-1} respectively, which were higher than the mean NH_4^+ concentration of all drains together. Other drains, such as D2 and D4 had mean concentrations of 7.6 mg N L^{-1} and 8.5 µg N L^{-1}, respectively, which were lower than the mean concentration of all drains together. Among all the samples collected for the two years, the highest NH_4^+ concentration (191 µg N L^{-1}) was measured in D10 in early December 2014. The second highest was measured from D8 (181 µg N L^{-1}) in late March 2015, whilst the third highest concentration was found in D10 (174 µg N L^{-1}) collected in late May 2014 which coincided with the second largest rainfall event highlighted in Figure 4.1. Excluding D10 and D8, all other drains had maximum NH_4^+ concentrations of <100 µg N L^{-1}.

The mean NH_4^+ flux for all the drains during the study period was 65 g N ha^{-1} a^{-1}. Some drains had relatively low NH_4^+ fluxes, for instance D6 and D11 with only 15 and 36 g N ha^{-1} a^{-1} respectively, whilst others, such as D3, had a relatively high mean flux (146 g N ha^{-1} a^{-1}). Nevertheless, compared to NO_3^-, field drain NH_4^+ concentrations and fluxes were very low and typically three orders of magnitude lower than nitrate. In a previous study on N losses in subsurface drains, Jaynes et al. (2001) found that losses of NH_4^+, NO_2^- and other forms of N were negligible in comparison to losses of NO_3^-.

Table 4.9 Summary of field drain NH_4^+ concentrations and fluxes in the three experimental treatments for samples collected between April 2013 and April 2015. SE is standard error

Treatment	Drain ID	n	Mean NH_4^+ ($\mu g\ N\ L^{-1}$)	SE	Mean NH_4^+ flux ($g\ N\ ha^{-1}\ a^{-1}$)	SE
No cover crop & conventional plough (control)	D10	48	20.3	4.8	49	5.9
	D8	27	16.6	1.4	82	6.5
Cover crop & reduced tillage	D3	34	12.4	1.5	146	1.5
	D16	30	14.2	3.2	38	3.2
	D1	57	9.1	1.2	43	1.2
Cover crop & direct drill	D11	12	14.4	4.0	36	4.0
	D2	68	7.6	0.9	40	0.9
	D4	57	8.5	0.9	124	0.9
	D6	43	12.2	2.2	15	2.2

The temporal variation in field drain NH_4^+ concentration in the three treatments is shown in Figure 4.6. Gaps in the measurements are due to a lack of drain flow which usually happened during the summer. NH_4^+ concentrations in all three treatments were generally <15 $\mu g\ N\ L^{-1}$ during spring, summer and even autumn 2014. In winter, NH_4^+ concentrations showed some fluctuations, but with no specific differences among the treatments. In late May 2014, NH_4^+ concentrations showed a pronounced peak of 174 $\mu g\ N\ L^{-1}$ in control treatment (no cover crop & conventional plough) which coincided with the second storm event highlighted in Figure 4.1. Surprisingly, field drains in the (cover crop & reduced tillage) and (cover crop & direct drill) treatments did not similarly exhibit this peak. Afterwards, and also during the second farming year, NH_4^+ concentrations did not change significantly except for a sudden increase in the control treatment at the end of the monitoring period.

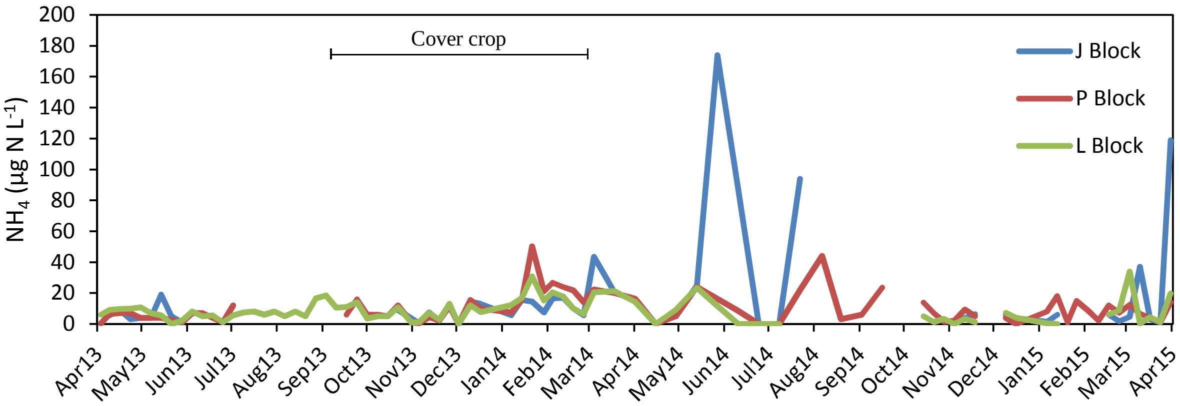

Figure 4.6 Temporal variation in the mean concentration of NH_4^+ in field drains under the three mitigation measure treatments.

Unlike NO_3^- concentrations which were significantly reduced by the presence of a cover crop during September 2013 to March 2014, field drain NH_4^+ concentrations were not dramatically lower in bare fields than in the cover crop fields. Details of NH_4^+ concentrations in field drains under the two treatments are presented in Table 4.10. The mean NH_4^+ concentration in field drains in fields with no cover crop (11.2 µg N L^{-1}) was not significantly ($p > 0.05$) lower than the mean concentration in field drains under a cover crop (13.1 µg N L^{-1}). The effect of the cover crop on NH_4^+ fluxes in field drains is also presented in Table 4.10. The overall mean NH_4^+ flux under no cover crop (89 g N ha^{-1} a^{-1}) was not significantly ($p > 0.05$) different from the mean flux with a cover crop (126 g N ha^{-1} a^{-1}).

Table 4.10 Mean and range of NH_4^+ concentrations and fluxes in the field drains under the two different treatments during cover crop growth (September 2013 to March 2014). SE is standard error

Treatments	Drain ID	n	Mean NH_4^+ (µg N L^{-1})	SE	Mean NH_4^+ flux (g N ha^{-1} a^{-1})	SE
Without cover crop	D10	18	11.8	3.6	42	12.5
	D8	15	10.5	1.7	136	46.1
Mean			11.2		89	
With cover crop	D3	13	13.8	2.7	158	49.8
	D16	16	21.6	5.2	122	40.8
	D1	22	10.7	1.7	103	32.4
	D11	7	21	5.0	120	59.3
	D2	24	10	1.4	70	12.1
	D4	23	11	1.4	283	71.0
	D6	17	11.6	1.9	25	8.1
Mean			13.1		126	

The insignificant effect of the cover crop on NH_4^+ concentrations is also shown by the porous pot data (Table 4.11). In February 2014, when cover crop was present on some fields, NH_4^+ concentrations were not significantly ($p > 0.05$) lower in cover crop fields than in bare fields, ranging from 8-46 µg N L^{-1}. The number of measurements for NH_4^+ concentration on the subsequent two porous pot sampling occasions decreased greatly, which was due to either damage to the pots or insufficient water yielded by the pots. Thus, statistical analysis could not be carried out for most of the sample measurements.

As it mentioned in the introductory of this chapter that nearly all of the studies assessed the effects of cover crops on reducing nitrogen loss from fields considered NO_3^- rather than the other two forms (NH_4^+ and NO_2^-) of nitrogen. For instance, in a study to evaluate the use of cover crops in cereal-based cropping in England, Macdonald et al. (2005) measured NH_4^+ concentrations in the water under different treatments. However, it was observed that in all experiments about 97% of the mineral N measured in the leachates was present as NO_3^- rather than NH_4^+. Consequently, leaching losses from NH_4^+ was neglected. Therefore, it was not possible to compare the results of the effects of cover crop on NH_4^+ and NO_2^- found in this study to other studies in literature.

Table 4.11 Mean of NH_4^+ concentrations in porous pot samples from three soil types in the control treatment (no cover crop & conventional plough), treatment (cover crop (CC) & reduced tillage (RT)), and treatment (cover crop (CC) & direct drill (DD)) collected on three sampling occasions. Means followed by different superscripted letters are significantly different ($p < 0.05$) for the same sampling date and same soil type.

		NH_4^+ (μg N L^{-1})					
Soil type	Treatments	n	Feb. 2014	n	Apr. 2014	n	Feb. 2015
Sandy Loam	control	6	24^{a}	0	-	3	57^{ab}
	(CC&RT	4	7^{a}	2	326^{a}	3	4^{a}
	(CC&DD)	4	32^{a}	2	14^{a}	5	293^{b}
Sandy Clay Loam	control	8	31^{a}	1	222	0	-
	(CC&RT	2	9^{a}	3	25	1	50
	(CC&DD)	5	46^{a}	1	10	2	171
Clay Loam	control	8	10^{ab}	2	20	0	-
	(CC&RT	6	33^{a}	1	12	2	72^{a}
	(CC&DD)	8	8^{b}	0	-	2	141^{a}

During the 2014-2015 farming year when only different tillage practices where trialled as a mitigation measure, the limited data obtained from the porous pot samples in February 2015 revealed no particular pattern among the treatments or soil types (Table 4.11). It seemed that NH_4^+ concentrations were higher under direct drill than under either reduced or conventional tillage. However, with only three NH_4^+ concentration measurements out of 90 sampling sites, it was not possible to conduct any statistical analysis. It is also apparent from the field drain data (Figure 4.6) that NH_4^+ concentrations in (cover crop & reduced tillage) and (cover crop & direct drill) treatments during winter 2015 were not substantially lower than the control treatment. The combined mean NH_4^+ concentration for porous pots and field drain samples under conventional tillage during October 2014 to April 2015 was 33.3 μg N L^{-1}, which was not statistically different from 11.6 μg N L^{-1} and 47.5 μg N L^{-1} obtained for reduced tillage and direct drill treatments, respectively. Thus, these combined results from both field drains and porous pots demonstrate that the reduced tillage and direct drill mitigation measures practiced in 2015 did not reduce soil water NH_4^+ concentrations compared to conventional tillage (Table 4.12).

Like NH_4^+ concentrations which showed no significant difference between conventional and conservation tillage, NH_4^+ fluxes also did not change significantly ($p > 0.05$) for the two main tillage types (Table 4.12). The mean NH_4^+ flux from field drains under conventional tillage was 88 g N ha^{-1} a^{-1}, which was slightly lower than the NH_4 flux under reduced tillage (117 g N ha^{-1} a^{-1}) and higher than the NH_4^+ flux under direct drill (17 g N ha^{-1} a^{-1}). The higher NH_4^+ flux under reduced tillage compared to conventional tillage is in agreement with the NO_3^- data (Table 4.7) which also showed higher NO_3^- fluxes in the former compared to the latter. It is noted that both NH_4^+ concentration and flux values in water samples collected under conventional tillage are in between the values obtained for reduced tillage and direct drill.

Table 4.12 Mean NH_4^+ concentrations and fluxes in both field drain and porous pot samples collected under different tillage regimes during October 2014 to April 2015. Numbers followed by different superscripted letters are significantly different ($p > 0.05$)

Tillage type	Mean of NH_4^+ (µg N L^{-1})	SE	Mean of NH_4^+ flux (g N ha^{-1} a^{-1})	SE
Conventional tillage	33.3^{ab}	12.2	88^{ab}	49.3
Reduced tillage	11.6^{a}	3.5	117^{a}	49.8
Direct drill	47.5^{b}	16	17^{b}	4.9

4.2.4 Nitrite

Table 4.13 summarises the field drain nitrite (NO_2^-) concentrations and fluxes for the three experimental treatments during April 2013 to April 2015. In total, 465 measurements were made for NO_2^- from water samples collected weekly over two years. The mean nitrite concentration for all samples collected during this period was 4.5 µg N L^{-1}, which is less than half of the mean NH_4^+ concentration (11.8 µg N L^{-1}) and three orders of magnitude lower than the mean NO_3^- concentration (5.2 mg N L^{-1}). No great variation in the mean NO_2^- concentration was observed amongst the field drains, with concentrations in each individual drain always <10 µg N L^{-1}. Among all 465 samples, the highest NO_2^- concentration of 182 µg N L^{-1} was measured in D10 in the last week of sampling (31th March 2015). The second highest concentration was measured in D8 during the same week (149 µg N L^{-1}). The mean NO_2^- flux calculated from all the drains during the study period was 25 g N ha^{-1} a^{-1}. Some drains had relatively low NO_2^- fluxes, for instance D11 and D16 with only 1-2 g N ha^{-1} a^{-1}, whilst others, such as D4 and D8, had relatively high fluxes of 46 and 48 g N ha^{-1} a^{-1}, respectively. Nevertheless, field drain NO_2^- fluxes were typically three orders of magnitude lower than NO_3^- fluxes (35 kg N ha^{-1} a^{-1}).

Table 4.13 Summary of the field drain NO_2^- concentrations and fluxes in the three experimental treatments for samples collected during April 2013-April 2015. SE is standard error

Treatment	Drain ID	n	Mean NO_2^- (µg N L^{-1})	SE	Mean NO_2^- flux (g N ha^{-1} a^{-1})	SE
No cover crop & conventional plough (control)	D10	57	7.5	3.2	19	8.4
	D8	39	7.0	3.8	48	22.5
Cover crop & reduced tillage	D3	41	4.3	0.8	51	11.8
	D16	38	4.5	0.6	2	0.5
	D1	70	3.1	0.5	28	5.7
Cover crop & direct drill	D11	20	6.8	4.1	1	0.6
	D2	73	3.2	0.5	25	3.6
	D4	70	3.3	0.5	46	7.1
	D6	57	4.1	0.6	5	0.8

The temporal variation in field drain NO_2^- concentrations across the three treatments is shown in Figure 4.7. Gaps in the measurements are due to the lack of drain flow during the summer/autumn. Throughout the study period, NO_2^- concentrations did not show a major peak in any treatment and remained relatively steady at 1-10 µg N L^{-1}. Only on a few sampling occasions did the concentrations increase significantly. NO_2^- concentrations showed a small peak of 14 µg N L^{-1} in treatment (cover crop & direct drill) in late August 2013 when drains from the other two treatments were dry. In mid-March 2014, all drains contained relatively high NO_2^- concentrations (up to 14 µg N L^{-1}) for two consecutive weeks. In late May 2014, coinciding with the second highlighted storm event in Figure 4.1, NO_2^- concentrations only increased in treatment (cover crop & direct drill) which was solely due to a concentration of 42 µg N L^{-1} in drain D10. Another concentration increase was observed in early December 2014, when all drains in (cover crop & direct drill) had a mean value of 24.4 µg N L^{-1}. Towards the end of the sampling period, NO_2^- concentrations gradually increased in all drains, with very high values recorded for D8 and D10 of 149 and 182 µg N L^{-1}, respectively. This sharp increase in the (cover crop & reduced tillage) field drain (D10) NO_2^- concentration was also replicated in the NH_4^+ (Figure 4.6) and NO_3^- concentrations (Figure 4.2). It was also observed that the third highlighted storm event in mid-October 2014 (Figure 4.1) caused a sharp increase in NO_3^- concentrations in all drains and all three treatments (Figure 4.2), but did not increase either NH_4^+ or NO_2^- concentrations.

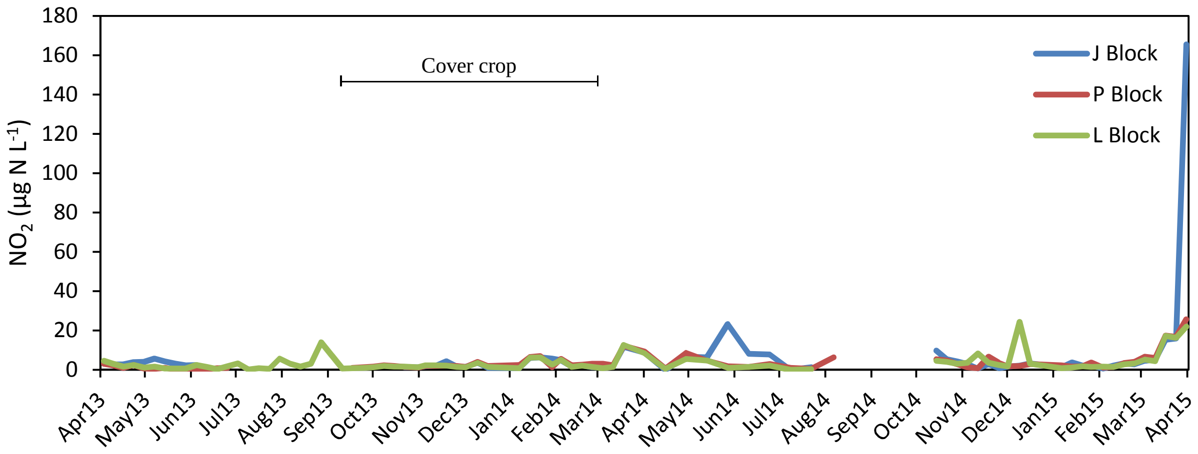

Figure 4.7 Temporal variation in the mean nitrite (NO_2^-) concentration of field drains from the three mitigation measures treatments.

The effects of the cover crop on field drain NO_2^- concentrations and fluxes are presented in Table 4.14. Unlike NO_3^- concentrations which were significantly reduced by the presence of a cover crop during September 2013 to March 2014, NO_2^- concentrations and fluxes were not significantly reduced under cover crop fields relative to bare fields, as was the also the case for NH_4^+. The mean NO_2^- concentration in field drains under cover crop fields (2.6 µg N L^{-1}) was not significantly ($p > 0.05$) different from the mean value under fields with no cover crop (2.5 µg N L^{-1}). Likewise, the mean NO_2^- fluxes were also not significantly ($p > 0.05$) different between drain samples collected under the cover crop (22 g N ha^{-1} a^{-1}) and those collected under no cover crop (25 g N ha^{-1} a^{-1}).

Table 4.14 Mean and range of field drain NO_2^- concentrations and fluxes under the two different cover crop treatments during September 2013 to March 2014. SE is standard error

	Drain ID	n	Mean NO_2^- (μg N L^{-1})	SE	Mean NO_2^- flux (g N ha^{-1} a^{-1})	SE
Without cover crop	D10	18	2.5	0.4	11	3.4
	D8	15	2.5	0.5	39	14.7
Mean			2.5		25	
With cover crop	D3	14	2.6	0.5	30	9.5
	D16	17	3.8	0.5	6	1.4
	D1	23	2.2	0.3	20	6.1
	D11	8	3.2	0.9	4	1.9
	D2	24	2.4	0.4	19	2.9
	D4	24	2.3	0.4	66	15.1
	D6	19	2.3	0.4	5	1.3
Mean			2.6		22	

NO_2^- concentrations and fluxes under different tillage are summarised in the Table 4.15. Although the mean concentration and flux values are different, these differences are not statistically significantly due to large variation. The combined mean porous pot and field drain NO_2^- concentration under conventional tillage (12.0 μg N L^{-1}) was not significantly ($p > 0.05$) higher than that under reduced tillage (5.8 μg N L^{-1}) and direct drill (6.9 μg N L^{-1}). Likewise, the mean value of 76 g N ha^{-1} a^{-1} for NO_2^- fluxes under conventional tillage was not significantly ($p > 0.05$) higher than that of 66 g N ha^{-1} a^{-1} under reduced tillage and of 26 g N ha^{-1} a^{-1} under direct drill.

Table 4.15 Combined NO_2^- concentrations and fluxes from field drain and porous pot samples collected under different tillage regimes during October 2014 to April 2015. Means followed by different superscripted letters are significantly different ($p > 0.05$). SE is standard error

Tillage type	Mean NO_2 ($\mu g\ N\ L^{-1}$)	SE	Mean of NO_2 flux ($g\ N\ ha^{-1}\ a^{-1}$)	SE
Conventional tillage	12.0^a	4.8	76^a	41.1
Reduced tillage	5.8^a	0.7	66^a	14.2
Direct drill	6.9^a	1.2	26^a	4.0

4.3 Summary

It is concluded from the data presented in this chapter that using cover crops as a mitigation measure for reducing diffuse N pollution is very effective. The mean concentration of NO_3^-, the predominant form of N in soil water, was significantly reduced from 13.9 mg N L^{-1} without a cover crop to 2.5 mg N L^{-1} with a cover crop: an 82% reduction. Likewise, NO_3^- fluxes from fields were substantially reduced from 113 kg N ha^{-1} a^{-1} in bare fields to 17 kg N ha^{-1} a^{-1} in cover crop fields: equivalent to an 85% reduction in N flux. No statistically significant change in NH_4^+ and NO_2^- concentrations and fluxes between cover crop and no cover crop fields was observed. However, these two forms of N contribute only a small proportion to total N leaching, so this does not lessen the overall effectiveness of cover crops at reducing N leaching. Conversely, different soil inversion regimes tended to have minimal effect on both the concentrations and fluxes of any form of N. Thus, the overall conclusion of this chapter is that in agricultural areas where high nitrate losses from fields into ground or surface water represents a major pollution risk, cover crops can be a highly effective mitigation measure.

Chapter 5: DISSOLVED NITROUS OXIDE

5.1 Introduction

5.1.1 Nitrous oxide

Nitrous oxide (N_2O) is a potent greenhouse gas with a present atmospheric concentration of 319 ppb (Forster et al., 2007). N_2O persists in the atmosphere and has 300 times more global warming potential than CO_2 and accounts for about 5% of the total greenhouse effect (Omonode et al., 2011). N_2O molecules also participate in photochemical reactions in the stratosphere which may lead to destruction of the earth-protecting ozone (O_3) layer (Jacinthe and Dick, 1997). N_2O is also linked to the release of nitric oxide and ammonia, which contribute to acid rain and the acidification of soils and drainage systems (Mosier and Kroeze, 1998).

The concentration of this gas in the atmosphere is increasing at an annual rate of ~0.3% (Kroeze, 1994). Since the pre-industrial era, global atmospheric N_2O concentrations have increased by about 16% from 270 to 319 ppb (Syakila and Kroeze, 2011). Agriculture is considered to be the largest source for N_2O emissions from anthropogenic activities. Agriculture accounts for about 60% of the total N_2O anthropogenic emission and global agricultural N_2O emissions increased by nearly 17% from 1990 to 2005 (Smith et al., 2007). Global agricultural N_2O emissions are projected to increase 35-60% by 2030 due to increased nitrogen fertiliser use and increased animal manure production (Bruinsma, 2003). Similarly, Scheehle et al. (2006) estimated that emissions from agricultural soils are projected to increase by more than 50% by 2020 compared to 1990. The current agricultural contribution to total global nitrogen emissions is estimated at 4.7 Tg N a^{-1} (i.e. 4.7 million tonnes of nitrogen per year), but there is great uncertainty about the magnitude of emissions because of the wide range in estimates for different agricultural sources. The non-agricultural anthropogenic sources of N_2O emissions are biomass combustion, stationary and mobile combustion, adipic and nitric acid production, solvent use, waste incineration, fugitives from oil and natural gas systems, and fugitives from solid fuels.

The total flux of N_2O into the atmosphere from all sources is currently estimated at 18.8 Tg N a^{-1}, of which natural processes account for 65% and anthropogenic activities account for 35%. The largest sources of natural N_2O emissions are soils, contributing ~35%, followed by oceans, river and estuaries. However, there is some debate as to what fraction of the emissions associated with rivers and estuaries should be considered as natural sources as these are driven primarily by anthropogenic contributions of nitrogen into water bodies. Natural sources reflect microbial processes in uncultivated soils, ocean and other aquatic systems, including wetlands (Anderson et al., 2010).

N_2O is one of the key compounds in the nitrogen cycle. It is produced naturally in the soils through the microbial process of denitrification and nitrification. A number of anthropogenic activities add nitrogen to the soils, thereby increasing the amount of nitrogen available for nitrification and denitrification, and ultimately the amount of N_2O emitted. Added nitrogen from anthropogenic activities into the soil is either directly or indirectly (Scheehle et al., 2006). Direct addition is through activities such as cropping practices, application of fertilisers, production of nitrogen-fixing crops (e.g. beans, pulses and alfalfa), incorporation of crop residues into the soil, and cultivation of high organic content soils. Indirect additions occur through volatilisation and subsequent atmospheric deposition of ammonia and oxides of nitrogen that originate from the application of fertilisers and livestock wastes onto cropland and pastureland, and subsequent surface runoff and leaching of nitrogen from these same sources (Scheehle et al., 2006).

N_2O is an obligatory intermediate of denitrification (Equation 5.1), a series of energy generating reactions during which nitrate is reduced to gaseous nitrogen compounds such as NO, N_2O or N_2. The generally accepted pathway is shown below (Bremner, 1997):

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \tag{5.1}$$

Denitrification takes places under anaerobic conditions when bacteria utilise nitrate as the terminal electron acceptor in place of oxygen. Denitrification plays a vital role in the N cycle of the atmosphere. In its absence, all biologically available N that has been released from igneous rocks of the Earth’s original crust and mantle would have been converted long ago to its more thermodynamically stable form of NO_3^- in the ocean. Denitrification also represents the only biological process for consumption of N_2O (Williams et al., 1992). It is a vital step in waste water treatment as it removes nitrate from the water and thereby helps to reduce algal blooms and eutrophication. Conversely, this process accounts for the major loss of fixed nitrogen from both soils and the oceans, with N_2O comprising >5% of the denitrification end product in soils (Nevison, 2000). It is considered as a major problem in farming as it decreases the effectiveness of fertiliser by converting nitrate into nitrogen gas. It can be seen from equation 5.1 that denitrification not only produces N_2O, but it also reduces N_2O, especially when oxygen concentrations are extremely low.

A wide range of bacteria are able to denitrify. They are facultative anaerobes and switch to NO_3^- as a terminal electron acceptor when oxygen concentrations in soil become depleted (Skiba and Smith, 2000). Types of denitrifiers include phototrophs, lithotrophs and organotrophs which generate energy for growth and regeneration from light, inorganic substances and organic substances, respectively. The latter group are the most common denitrifiers in soil and the aquatic environment. Species of *Pseudomonas, Alcaligenes, Bacillus* and *Paracoccus* are the most common denitrifers (Williams et al., 1992; Bernhard, 2012).

Early researchers assumed that denitrification was the only biological process responsible for N_2O production in soils and that essentially all of the N_2O generated from soils was produced through the reduction of nitrate by denitrifying microorganisms under anaerobic conditions. However, it is now well documented that nitrifying microorganisms contribute significantly to emission of N_2O from soils (Bremner, 1997). Nitrification is the aerobic oxidation of reduced forms of nitrogen, mainly ammonium to nitrite and nitrate (Equation 5.2). N_2O generally accounts for <1% of the nitrification end product in soil in a normal oxygenated environment (Nevison, 2000).

$$NH_4^+ \rightarrow NH_2OH \rightarrow \overset{N_2O}{\uparrow} NO_2^- \rightarrow \overset{N_2O}{\uparrow} NO_3^- \tag{5.2}$$

The process of nitrification is associated with the metabolism of chemoautotrophic bacteria of the family *Nitrobacteraceae*, as well as several species of heterotrophic microorganisms. None of these bacteria are able to oxidise ammonium to nitrate in a single step. Nitrification occurs in two steps which are carried out by two different groups of bacteria. N_2O can be a by-product of either step. In the first step, *Nitrosomonas* and *Nitrosospira* bacteria oxidise ammonium to nitrite. The subsequent oxidation of nitrite to nitrate is facilitated by a second group of bacteria which include *Nitrobacter*. Although low numbers of a few other ammonium oxidising chemoautotrophs are also present in many soils, *Nitrobacter* is the only genus known to be involved in the oxidation of nitrite (Williams et al., 1992).

Soil moisture, soil temperature, soil organic matter, nitrogen availability, availability of NH_4^+, pH, redox condition, topographic position and agricultural management practices have all been identified as main factors controlling denitrification and nitrification rates (Bouwman et al., 1993; Panek et al., 2000). Nitrification is a relatively constant process across ecosystems, whereas denitrification rates are temporally and spatially variable. In most soils, variability of NH_4^+ and oxygen are the most important factors controlling nitrification, whereas the major controls on biological denitrification include the availability of organic carbon, oxygen and nitrate or other nitrogen oxides (FAO, 2001).

Denitrification tends to produce greater N_2O emissions, whilst rates of N_2O production by nitrification tend to be smaller (Williams et al., 1992). However, conditions for nitrification in soils are much more common and thus the contribution of nitrification to total global N_2O emissions is substantial (Skiba and Smith, 2000). The balance between nitrification and denitrification as contributors of N_2O emissions will change with climate, soil condition and soil management. Generally, high rainfall, poor drainage, fine soil texture and high organic carbon content promote denitrification and associated N_2O production, whereas low rainfall, good drainage and aeration and coarse texture enhance nitrification and associated N_2O production (Groffman et al., 1991; Skiba and Smith, 2000). However, in most soils the dominant production mechanism is not static and can switch very quickly. For example, soil aeration levels can change rapidly in response to rainfall or increased oxygen demand caused by the presence of easily mineralisable organic matter (Skiba and Smith, 2000).

The oxygen and moisture content of agricultural soils depends on soil texture and drainage. Fine textured soil have more capillary pores within aggregates than sandy soils and so hold soil water more tightly. As a result, anaerobic conditions may be more easily reached and maintained for longer periods within aggregates in fine textured soils than in coarse textured soils. The water content of the soils influences N_2O emissions in all types of soil. Microbial activity generally peaks at 30-60% water-filled pore space. Nitrification and associated N_2O production also peaks at 30-60% water-filled pore space, while ideal conditions for denitrification may occur at 50-90% of water-filled pore space (FAO, 2001).

5.1.2 Nitrous oxide in aquatic systems

The large uncertainties regarding the natural and anthropogenic sources and sinks of N_2O have initiated many investigations into the N_2O found in aquatic systems. The first N_2O measurements from seawater were made by Craig and Gordon (1963). Oudot et al. (1990) reported that N_2O in the ocean is present at concentrations in excess of saturation with respect to the atmosphere at the temperatures and salinity levels observed. Consequently, the open ocean acts as a source for atmospheric N_2O. Initially, it was believed that the oceans would be the predominant source for N_2O. However, a series of studies corrected this understanding and it is now generally accepted that the ocean is a major, but not predominant, source of N_2O to the atmosphere, and contributes ~35% of the total natural sources of N_2O and ~21% of all sources (natural and anthropogenic) (Lal and Patra, 1998; Bange et al., 2001; Rao et al., 2013; Chen et al., 2014).

Several different methods have been used to estimate N_2O emissions from the ocean. Preliminary studies used measurements of dissolved concentrations in the surface water and estimated gas-transfer based on wind speeds. Estimates have also been made using observations of the correlation between dissolved N_2O and other gases. Recently, inverse modelling techniques have been applied to calculate relatively large scale emission estimates for the northern versus southern hemispheres and for land versus ocean fluxes (Anderson et al., 2010).

Based on observations of water column profiles, N_2O is thought to be generated at intermediate depths in the ocean and nitrification is believed to be the dominant production process. This is because N_2O is generally negatively correlated with oxygen and positively correlated with nitrate in subsurface and deep waters and there is a linear relationship between the excess N_2O in the water and the apparent oxygen utilisation rate (Oudot et al., 1990). N_2O production in the surface layer is believed to be small because oxygen limits denitrification, whilst sunlight inhibits nitrification. Instead, N_2O is produced in the subsurface and subsequently transported to the ocean surface. Dore et al. (1998) found that nitrification at 100-300 metres depth could produce 70-90% of the oceanic N_2O emissions based on isotopic measurements of N_2O in the oligotrophic subtropical North Pacific.

N_2O emissions from the ocean show great seasonal and spatial variation. Nevison et al. (2005) argued that N_2O concentration should not be treated as an annual constant because it possesses strong seasonality. This seasonality is caused by thermal effects during the summer and a larger mixing effect during the winter as the depth of the surface mixed layer deepens and deeper N_2O enriched waters are mixed together. Nevison et al. (2005) suggested that the mean annual N_2O emissions from ocean water may be overestimated because the majority of surface N_2O measurements have been made during summer. Furthermore, ocean N_2O emissions are not uniformly distributed over the ocean surface. Nevison et al (2003) found that N_2O concentrations are highest in the eastern tropical Pacific, moderate in the northern Pacific and Indian Ocean, and relatively low in the Southern and Atlantic Oceans. Hirsch et al (2006) also calculated high N_2O emissions from the equator, with moderate emissions from the northern hemisphere and very low emissions from the southern hemisphere. N_2O concentrations have been found at unusually high levels of supersaturation in the oceanic samples collected in the vicinity of upwelling areas (e.g. coastal Peru, Chile, Arabian Sea, western South Africa, eastern New Zealand and the California coast). Since N_2O is mainly produced in subsurface waters, upwelling provides a rapid pathway to the surface where it degases. Regions of upwelling are also areas of enhanced primary productivity, which results in higher fluxes of organic material sinking into deeper waters. Consequently, this organic input depletes oxygen levels and creates conditions favourable for denitrifers as well as nitrifers (Anderson et al., 2010). Lal and Patra (1998) discovered that the Arabian Sea contributes significantly to N_2O emissions, generating ~13-17% of net global oceanic emissions despite this area accounting for only ~1.7% of the total ocean area.

The increasing concern about N_2O in the atmosphere has also initiated numerous investigations in freshwater systems. Unlike the oceans, emissions from freshwater bodies (i.e. rivers, estuaries, lakes and groundwater) are generally classified as anthropogenic rather than natural sources of N_2O because they are highly impacted by human activities. Although they cover a relatively small area globally, they represent active sites for aquatic productivity and biogeochemical cycling which leads to relatively high emissions in comparison to the open ocean.

During transport in streams, rivers and estuaries, nitrate can be denitrified or assimilated by the biota. Within the N cycle of open water bodies, mineral N species can be produced or retained, and N_2O can be produced by nitrification as well as produced and reduced by denitrification (Well and Butterbach-Bahl, 2010). Using a global river network model, Beaulieu et al. (2011) estimated that microbial nitrogen transformations (e.g. denitrification and nitrification) convert at least 0.68 Tg N a^{-1} of anthropogenic nitrogen inputs to N_2O in river networks, equivalent to 10% of the global anthropogenic N_2O emission total. Beaulieu et al. (2011) also found that this estimate of stream and river N_2O emissions is three times greater than that estimated by the IPCC (2006). Seitzinger and Kroeze (1998) modelled the emissions of N_2O from rivers globally based on the rates of riverine nitrification and denitrification and assumptions about likely factors. They estimated that rivers could contribute as much as 25% of the global total anthropogenic N_2O release into the atmosphere. Rosamond et al. (2012) reported that global riverine N_2O emissions are currently estimated to be 0.9 Tg N a^{-1}, equivalent to 17% of anthropogenic agricultural N_2O emissions, by assuming a linear relationship between dissolved inorganic nitrogen inputs to rivers and riverine N_2O emissions. Recent studies, however, have shown that riverine N_2O emissions are the most uncertain component of the current estimate due to a combination of insufficient data and a poor conceptual understanding of the mechanisms controlling riverine N_2O dynamics spatially and temporally (Yu et al., 2013). Another study by Cole and Caraco (2001) calculated a lower N_2O contribution from rivers. Based on a modelling study, they calculated that rivers might contribute only ~1.3% of the total anthropogenic N_2O emissions in the Hudson watershed.

In a study of the eutrophic San Joaquin river in California, Hinshaw and Dahlgren (2013) established a mean dissolved N_2O concentration in surface waters of 0.91 $\mu g\ L^{-1}$ and all the samples were supersaturated. They also found that the concentrations were significantly different between months and displayed a strong seasonal pattern, with the lowest concentrations found in April and highest concentrations recorded throughout the summer months. Outram and Hiscock (2012) measured dissolved N_2O concentrations on different water compartments in the Upper Thurne river , eastern England, and found that the concentrations were highest in pumped drainage sites, followed by the drainage channels and river, whilst shallow lakes had the lowest concentrations. They found that all sites had N_2O concentrations higher than would be expected when atmospheric N_2O concentrations are in equilibrium with the water, meaning that all water bodies were acting as source of N_2O to the atmosphere.

N_2O emissions from lakes have been less thoroughly investigated compared with other freshwater bodies. Lakes are not recorded as a source of N_2O emissions in the 2007 IPCC report as they are generally considered a minor source of N_2O emissions to the atmosphere. According to Anderson et al. (2010), N_2O emissions from lakes are ~0.004-0.04 Tg N a^{-1}, equivalent to 0.02-0.2% of total global N_2O emissions. Oxygen concentrations are an important factor controlling the balance between nitrification and denitrification processes. Lake oxygen concentrations are affected by water temperature, water depth and the rate at which oxygen is consumed by biota living within the lake water and sediments. Prisu et al. (1996) argued that most studies of marine systems have concluded that N_2O is produced via nitrification in surface waters, whereas denitrification may be the source in oxygen depleted deep water. This pattern is not that simple in lakes, perhaps because of their wide range of trophic states. High N_2O concentrations observed in several eutrophic lakes apparently arise from denitrification, whereas N_2O in the water column of more oligotrophic lakes is believed to be produced by nitrification or a combination of both mechanisms. In a study of 15 Swiss lakes of different size and trophic status, Mengis et al. (1997) identified that N_2O was produced in three zones:

(1) near the surface (epilimnion) which is observed occasionally in eutrophic lakes and seems to be attributed to the presence of actively growing algae; (2) in the oxic hypolimnion (water below the thermal boundary in thermally stratified lakes) which may be due to nitrification and inhibition of N_2O reduction in the presence of oxygen; and (3) at oxic/anoxic interfaces in the hypolimnion, which is attributed to both denitrification and nitrification.

N_2O emissions from field drains have been poorly studied to date. N_2O losses in leached drainage water from agricultural land were first reported by Dowdell et al. (1979). Bowden and Bormann (1986) indicated that after N_2O is produced by microbial activity in the soil, it dissolves into soil water and leaches through the soil into streams where it rapidly degases and is released into the atmosphere. They also pointed out that these losses of N_2O from such sources, which may be important to the global atmospheric budget, have been overlooked because they may be displaced temporally and spatially from expected sites of N_2O production. Reay et al. (2003; 2004a; 2004b) conducted a study on N_2O emissions from a drainage system underlying arable fields and discharging into a ditch. They concluded that N_2O discharged by field drain outfalls was quickly released into the atmosphere during transport in the open ditch. They also found that the complexity of the process due to both spatial and temporal variability was ultimately responsible for the poor relationship between applied fertiliser N and N_2O emissions from field drains. In a study of N_2O emissions from 28 drained agricultural areas in the upper Neckar region, Germany, Hack et al. (2002) observed a wide range (0.4-60 $\mu g\ N\ l^{-1}$) of N_2O concentrations corresponding to different land use practices, which in turn corresponded to different nitrate concentrations. They concluded that the average N_2O discharge in field drain water appears to be of minor importance in comparison to the direct N_2O emissions from soils.

Groundwater systems have recently been investigated in the assessment of global N_2O sources. Numerous studies have shown that groundwater which is supersaturated with N_2O with respect to the atmosphere will release any excess N_2O into the atmosphere at its outlet, such as springs, seepages, instream and pumped wells (Bochove et al., 2001; Reay et al., 2003; Reay et al., 2004b; Minamikawa et al., 2010; Jahangir et al., 2013). McMahon et al. (2000) listed reasons for nitrogen-enriched groundwater to be a potentially important anthropogenic source of atmospheric N_2O. These were are follows: (a) dissolved N_2O in groundwater has been reported at concentrations up to three orders of magnitude larger than air-equilibrated water; (b) the areal extent of groundwater contamination from inorganic nitrogen has increased over time due to increased fertiliser usage; (c) N_2O from groundwater can be transferred to the Earth’s surface through several outlets, including well pumping, natural discharges at springs and in lakes and oceans.

In a study of N_2O in boreholes and springs in the unconfined Chalk aquifer of Cambridgeshire, Mühlherr and Hiscock (1997) suggested that groundwater may be a significant component of the global N_2O budget. This was because all the samples they collected were oversaturated with N_2O, with concentrations ranging from 13 to 320 times greater than the air equilibrium concentration. They suggested that nitrification was the main mechanism for N_2O production based on a strong positive correlation between N_2O and NO_3^- and a weak correlation between N_2O and O_2 concentrations. A study by McMahon et al. (2000) in the US central High Plains aquifer also found that N_2O may be produced primarily by nitrification as high concentrations of O_2 and NO_3^- and low concentrations of NH_4^+ and dissolved organic carbon were measured. However, they came to the conclusion that the flux of N_2O from the aquifer to the atmosphere through well pumping and groundwater discharge to streams was not a significant source of the atmospheric N_2O. Vilain et al. (2011) came to the same conclusion when they calculated annual N_2O emissions from groundwater of 0.035 kg N ha^{-1} a^{-1}, equivalent to just 1.8% of the direct N_2O flux from agricultural land.

The United Nations Framework Convention on Climate Change (UNFCC) calls for the compilation of national emission inventories for trace gases (CO_2, CH_4 and N_2O) that contribute to climate change. The Intergovernmental Panel on Climate Change IPCC (2006) has developed protocols for quantifying N_2O emissions from industry, agriculture and natural ecosystems. The total N_2O emissions from agriculture are divided between direct and indirect emission.

5.1.3 Direct N_2O emissions

N_2O emissions occurring from agricultural land use include direct emissions from the soils as well as indirect emissions caused by nitrogen flows from agricultural fields into the surrounding aquatic environment (Well and Butterbach-Bahl, 2010) as is shown in Figure 5.1. In most agricultural soils, production of N_2O is enhanced by an increase in available mineral N which consequently increases nitrification and denitrification rates. Addition of N fertiliser, therefore, directly results in extra N_2O production. Nitrogen additions to soils can be from synthetic fertilisers, animal manures, biological N-fixation through N_2-fixing crops, crop residues returned to the field after harvest and human sewage sludge applications. Most studies on N_2O emissions from agricultural soils have investigated the difference in N_2O formation between fertilised and unfertilised fields. Emissions from unfertilised fields are considered background emissions (Mosier et al., 1998). Direct N_2O emissions from agricultural soils have been well documented (Bouwman, 1996; Skiba et al., 1996; Beauchamp, 1997; Smith et al., 1998; Dobbie and Smith, 2003; Van der Hoek et al., 2007; Rochette et al., 2008b)

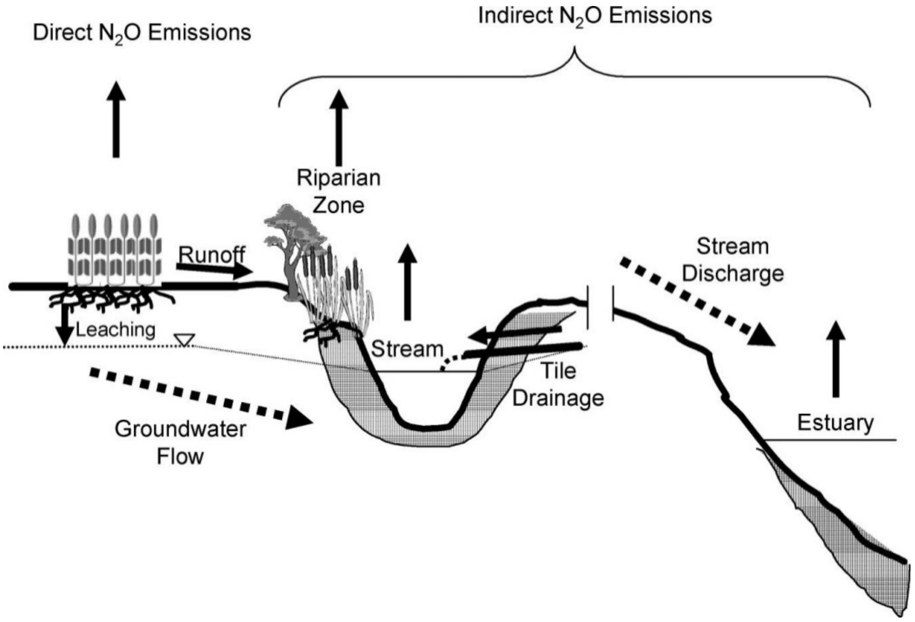

Figure 5.1 Nitrogen pathways from arable fields to adjacent environments and associated indirect N_2O emissions (Well et al., 2005b).

5.1.4 Indirect N_2O emissions

A major proportion of excess agricultural N is leached as nitrate into the surrounding aquatic environment. The IPCC (2006) considers that ~30% of applied N is lost through leaching and runoff with an uncertainty range of 10-80%. This loss of N by leaching provides enhanced conditions for nitrification and denitrification to occur and consequently accelerates N_2O production. Indirect emissions resulting from N leaching into aquatic systems, therefore, are considered a potentially important N_2O source. However, its magnitude is still under debate, with the uncertainty associated with currents estimates of almost two orders of magnitude, which is larger than the uncertainty for other N_2O sources (Well and Butterbach-Bahl, 2010). The aquatic pathway for reactive N originates in leaching and runoff from agricultural fields and ultimately ends up in the ocean after passing through a chain of connected systems (i.e. aquifers, riparian area, rivers and estuaries) as shown in Figure 5.1. N_2O produced in groundwater and unsaturated zones can be transported to the atmosphere via upwelling diffusion (Deurer et al., 2008) or groundwater discharge to wells, springs and streams, because once groundwater is discharged into surface water bodies the dissolved N_2O may partially or completely degas to the atmosphere (Reay et al., 2004b).

Indirect N_2O emissions from groundwater, streams and rivers receiving N-rich drainage water from agricultural fields have been poorly investigated to date, while direct sources of N_2O from soil have been fairly well documented (Outram and Hiscock, 2012). Furthermore, estimating indirect N_2O emissions is complicated by the fact that it is often difficult to separate the fluxes originating from agricultural land from other N sources. As an example, riparian buffers are described by Well and Butterbach-Bahl (2010) to show the complexity of natural processes. Riparian buffers between agricultural fields and streams receives N via subsurface groundwater flow, atmospheric deposition from industrial, agricultural and natural sources, biological N_2 fixation and potentially N from different sources in the stream during flooding. N_2O emitted at the soil surface is thus a mixture of groundwater-derived N_2O of mostly agricultural origin and N_2O produced in soil that originates from industrial, agricultural and natural sources.

Investigations to determine the importance of indirect N_2O emissions relative to direct N_2O emissions and to the global N_2O budget have shown inconsistent results. Studies on streams (Beaulieu et al., 2008), groundwater (Ueda et al., 1991), aquifers (Hiscock et al., 2003) and field drains (Hack and Kaupenjohann, 2002; Reay et al., 2004b) have shown that the contribution of indirect N_2O emission to the overall N_2O budget is insignificant. Conversely, other studies on different surface water bodies (Outram and Hiscock, 2012), on aquifers (Ronen et al., 1988a), and on some groundwater systems (Ueda et al., 1993) have shown that indirect N_2O emission can be significant. The latter two studies discussed that these indirect emissions cannot therefore be overlooked when constructing a global N_2O budget, especially considering the worldwide trend of increasing groundwater contamination by NO_3^-. The objective of this chapter is to investigate the spatial and temporal changes in dissolved N_2O concentrations in field drains and stream waters and the factors beyond these changes. Another objective of this chapter is to assess the effects of the mitigation measures on N_2O concentrations.

5.2 Results and Discussion

5.2.1 Spatial variation of nitrous oxide concentrations

A summary of dissolved N_2O concentrations in field drain and stream water samples collected throughout this study is presented in Table 5.1, and Figure 5.2 shows the variations in N_2O concentration for each field drain and stream site. In total, 645 water samples were collected from field drains and 308 samples were collected from stream sites, such that 953 samples in total were obtained for N_2O analysis over the duration of this study. Some drains (e.g. D1, D2 and D4) were flowing almost continuously throughout the study period, such that a greater number of samples were collected from these drains, whereas others (e.g. D11 and D13) flowed for less than one year and thus yielded fewer samples. All of the drains were found to have dissolved N_2O concentrations higher than would be expected when atmospheric N_2O concentrations at in equilibrium with water, which is ~0.35 µg N L^{-1} (Forster et al., 2007). Field drain N_2O concentrations ranged from 0.4 µg N L^{-1}, just above the atmospheric-water equilibrium, to 34.4 µg N L^{-1}, 100 times greater than atmospheric-water equilibrium. Stream water contained N_2O concentrations 1-20 times greater than the atmospheric–water equilibrium concentration. This demonstrates that all sites were acting as a source of N_2O to the atmosphere.

A mean N_2O concentration of 4.5 µg N L^{-1} was calculated for all field drain samples. Among the drains, D11 had the highest mean value (8 µg N L^{-1}) and D2 the lowest (1.9 µg N L^{-1}). For all the stream samples, a mean value of 1.4 µg N L^{-1} was calculated, with concentrations ranging from 0.35 µg N L^{-1} to 7.3 µg N L^{-1}. Among the stream sampling sites, site M had the lowest mean concentration (1.0 µg N L^{-1}) and site B the highest (1.8 µg N L^{-1}).

Table 5.1 Summary of dissolved N_2O concentrations in field drains and stream samples collected during April 2013-April 2015

Location	ID	n	Mean N_2O (µg N L^{-1})	Range of N_2O (µg N L^{-1})
Field Drain	D1	73	3.9	0.4-11
	D2	83	1.9	0.4-7.2
	D3	39	4.1	0.6-12.4
	D4	74	6.5	0.7-19.5
	D5	62	4.5	0.9-21.4
	D6	52	3.8	1.1-10.1
	D7	43	5.3	0.7-32.9
	D8	41	6.3	0.6-34.4
	D9	45	5.2	0.6-16.9
	D10	49	6.2	0.7-29.6
	D11	16	8.0	2.6-14.6
	D13	27	4.2	0.7-9.6
	D16	41	2.0	0.4-4.4
Stream	A	77	1.3	0.7-6.3
	B	77	1.8	1-4.7
	E	77	1.6	0.9-7.3
	M	77	1.0	0.35-7.1

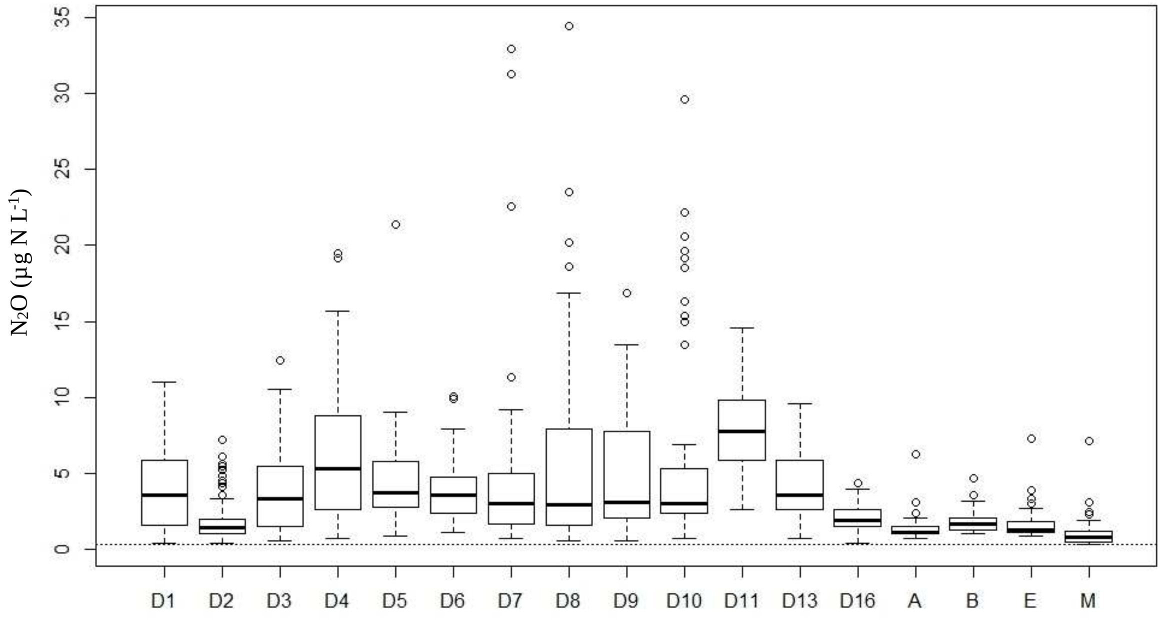

Figure 5.2 Boxplot of dissolved N_2O concentrations in field drains (D1-D16) and in stream water (A, B, E and M) for samples collected during April 2013-April 2015. The horizontal dashed line represents the atmospheric N_2O concentration when in equilibrium with water (0.35 µg N L^{-1}).

There are very few studies of dissolved N_2O concentrations in field drains. Dowdell et al. (1979), who studied dissolved N_2O in agricultural drains for the first time, found a range of 1-132 µg N L^{-1} in three different locations across southern England. In a study of N_2O discharged from 28 drained agricultural areas in the upper Neckar region, Germany, Hack et al. (2002) observed a range of 0.4-60 µg N l^{-1} in N_2O concentration corresponding to different nitrate concentrations and land use. One potential reason for the relatively wide range in reported N_2O concentrations in these two studies is that the sampled drains were located in different countries from areas of different land use. Therefore, the field drains experienced different climates, notably rainfall, and different farming practices, including amounts of applied N fertiliser, which consequently generated a wide range of N_2O concentrations. A study by Reay et al. (2004b) on one particular field drain under arable land planted with spring barely on the Bush Estate, Scotland, revealed a narrow range in N_2O concentration (2-4 µg N l^{-1}) over a 45 day sampling period. Sawamoto et al. (2010) reported that dissolved N_2O concentrations in drainage water at a depth of 1.7 m in a lysimeter that contained a brown forest soil ranged from 3.7 to 123 µg N l^{-1}. In another lysimeter experiment, Minamikawa et al. (2011) recorded an N_2O concentration range of 0.4 to 500 µg N l^{-1}. Thus, in total, a range of 0.4-132 µg N l^{-1} is reported in the literature for dissolved N_2O concentrations from field drains under different land use, which compares with 0.4-35 µg N l^{-1} recorded in this study. This is towards the lower end of the range reported in some studies, but towards the higher end of the range reported in others.

Compared to field drains, stream samples contained significantly lower N_2O concentrations. This illustrates that N_2O is rapidly degassed from field drain water once it has come into contact with the atmosphere. This degassing of supersaturated N_2O in subsurface drainage and groundwater to the atmosphere after discharge to surface water has been also reported in previous studies (Bowden and Bormann, 1986; Reay et al., 2003; Minamikawa et al., 2011; Li et al., 2013). Other studies have found similar values for dissolved N_2O in surface water. In a study on nine sites of the eutrophic San Joaquin River, California, over a 13-month period, Hinshaw and Dahlgren (2013) found a mean dissolved N_2O concentration in surface waters of 0.91 µg L^{-1} and all the samples were supersaturated. Outram and Hiscock (2012) measured dissolved N_2O concentrations on different water bodies in the Upper Thurne river, eastern England, and found that river samples contained a mean concentration of 1.7 µg L^{-1}. Outram and Hiscock (2012) also found that all the sampled sites had N_2O concentrations higher than would be expected when atmospheric N_2O concentrations are in equilibrium with water.

5.2.2 Temporal variation in nitrous oxide concentrations

The temporal variability in field drain and stream water N_2O concentration is presented in Figure 5.3. Gaps in the measurement in drain N_2O concentration are due to lack of drain flow during the summer/autumn. As summer 2013 was approaching, N_2O concentrations gradually decreased in all drains, perhaps due to both a decline in water flow rate and a decline in potentially leachable nitrate due to crop growth in this period. Drain samples contained lower N_2O concentrations in summer 2013 than summer 2014, possibly because of lower rainfall totals in 2013 (106 mm) compared with 2014 (194 mm). Although a large amount of rainfall occurred in autumn 2013 (244 mm), including the largest storm event in mid-October 2013 in which 68 mm fell in one week, N_2O concentrations in field drains remained low with no obvious peak corresponding to this storm event. The low N_2O concentrations continued through winter, spring and summer 2014 with a slight gradual increase. The low N_2O concentrations throughout this period might be due to the fact that most of the drains were draining fields under spring beans which received either no N fertiliser or only 30 kg N ha^{-1}. A pronounced increase in N_2O concentration did, however, occur in autumn 2014 when the highest values of the study period were recorded in D7 (31.3 µg N L^{-1}) and D8 (34.4 µg N L^{-1}). These higher N_2O concentrations continued throughout autumn and winter 2014-2015, such that levels were considerably higher than they had been in the previous year (2013-2014). In some drains, N_2O concentration decreased more or less continuously from winter to summer, particularly in the first sampling year, which correlated strongly with decreasing water flow rates in summer (see Figure 5.8). Temporal variations in N_2O concentration in stream samples were not as apparent as in the drain samples. Throughout the study period, N_2O concentrations were consistently highest at site B, followed by site E, site A and finally site M.

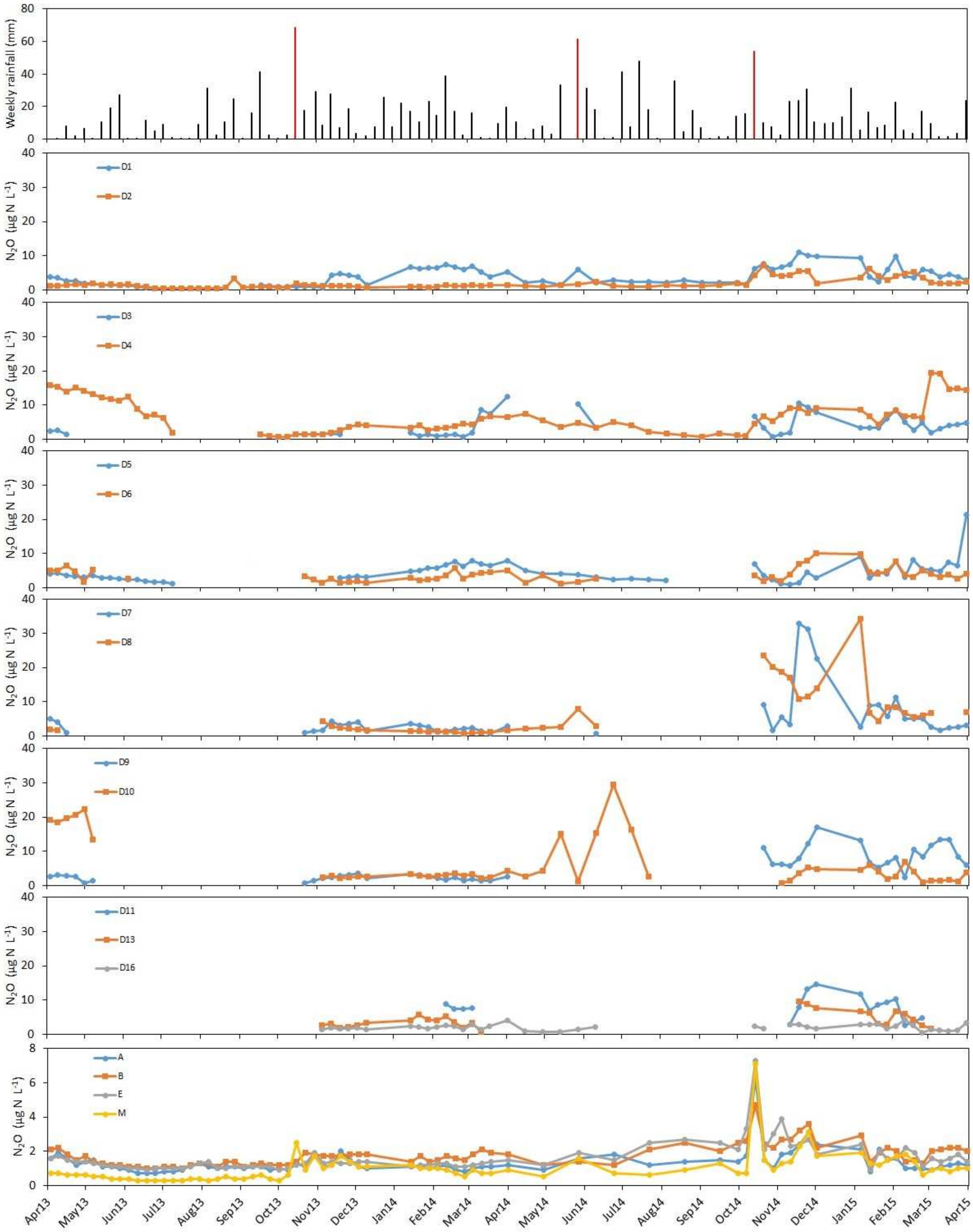

Figure 5.3 Weekly rainfall totals (top), dissolved N_2O concentrations in field drains (middle) and in stream samples (bottom) throughout study period.

N_2O concentrations in the field drains and streams responded differently to the three main rainfall events which occurred throughout the study. The highest weekly rainfall (68 mm) which occurred in mid-October 2013 created no change in N_2O concentration in the flowing drains, however, most of dry drains started flowing after this event. Also, with the exception of site M, none of the stream sites showed an increase in N_2O concentration in response to this rainfall. N_2O concentration at site M increased from 0.6 µg N L^{-1} prior the event to 2.5 µg N L^{-1} during the event. Regarding the second storm event which occurred in late May 2014, no significant increases in N_2O concentration were observed in most of the drains and stream sites, with the exception of site M which showed a slight increase (0.5 to 1.6 µg N L^{-1}) and drains D1 (1.3 to 5.9 µg N L^{-1}) and D8 (2.5 to 7.9 µg N L^{-1}) which showed modest increases. Unlike the storm event in autumn 2013, the storm event in October 2014, in which 54 mm of rainfall fell in one week, resulted in a pronounced rise in N_2O concentrations at all stream sites and flowing field drains. During this week, the maximum N_2O concentrations were recorded in all four stream locations. N_2O concentrations at site M, for example, did not exceed 2.5 µg N L^{-1} in the previous one and half years of data collection, but during this storm a concentration of 7.1 µg N L^{-1} was measured. N_2O concentrations in the field drain samples also peaked in mid-October, but this was less pronounced as most of the drains were not flowing prior to this rainfall event. Autumn rainfall, therefore, can sometimes cause a flushing event with high N_2O concentrations associated with nitrification of residual soil nitrate post-harvest. Thus, this study shows that whilst N_2O concentrations in field drains and instream are sometimes heavily impacted by storm events, on other occasions the effect is not noticeable. No clear relationship between N_2O concentration and rainfall was observed in a field drain study by Reay et al. (2004b), which they argued might be due to time lags between rainfall and the resulting impact on N_2O concentration. Such time lags are themselves likely to be extremely variable due to the spatial heterogeneity of soil N processing.

During this study, samples from field drains and steam sites were collected continuously for 24 months, such that samples were obtained twice for each season. To evaluate seasonal changes in the N_2O concentration of field drains and in stream, all samples collected in a particular season were combined (Figure 5.4) for spring (MAM), summer (JJA), autumn (SON) and winter (DJF). In all seasons, N_2O concentration was significantly lower in stream than in the field drains, which is due to the rapid degassing of N_2O from the drain water once it comes into contact with the atmosphere. N_2O concentrations were significantly ($p < 0.05$) lower during summer than any other season in both the drains and the streams, with mean concentrations of 3.02 and 1.02 µg N L^{-1}, respectively. These low summer concentrations are probably due to both a decline in water flow rate and a decline in potential leachable nitrate due to nutrient uptake by the growing crops. Lower concentrations in field drains during the summer were also observed by Hack et al. (2002). The other three seasons showed no statistically significant differences in the field drain N_2O concentrations, although values during spring (mean = 4.9 µg N L^{-1}) tended to be slightly higher than the other two seasons. Regarding stream samples, no significant difference was observed between samples collected during winter and spring, although the autumn (mean = 1.82 µg N L^{-1}) did have significantly higher ($p < 0.05$) N_2O concentrations than winter and spring. The storm event in mid-October 2014 generated extremely high N_2O concentrations at all four stream sites and this contributed to the high N_2O concentrations observed during the autumn. In a two year study investigating the effects of season on N_2O production in headwater streams in the Kalamazoo River basin, south-west Michigan, Beaulieu et al. (2008) found that there was no seasonal pattern in N_2O concentration in 2004, but in 2005 N_2O values were highest in winter and lowest during summer. Therefore, it is concluded from this study that N_2O concentrations in both the streams and field drains vary seasonally, with the lowest concentrations occurring typically during the summer.

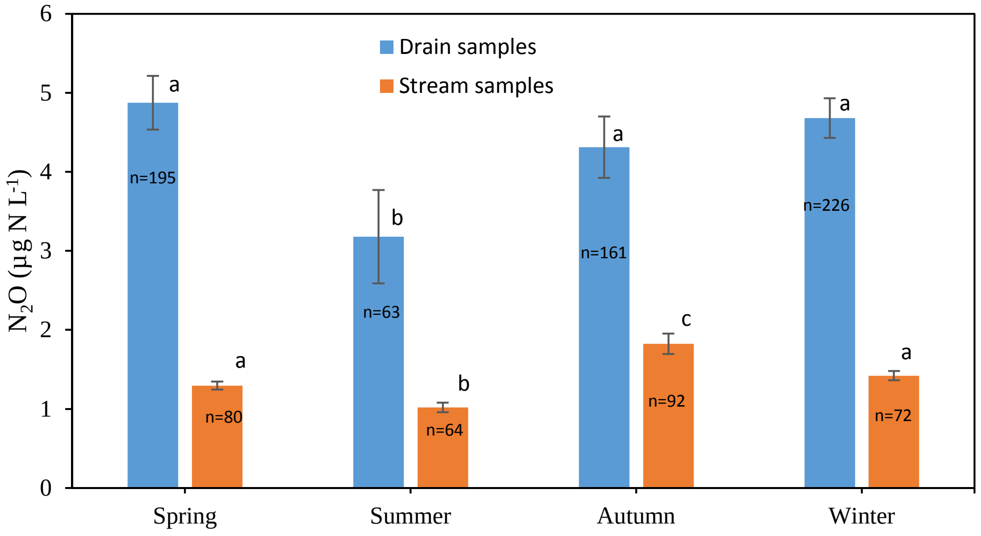

Figure 5.4 Average N_2O concentrations in field drains and in stream waters during different seasons in samples collected during April 2013-April 2015. Error bars represent the standard error. Significant differences ($p < 0.05$) are indicated by different letters for the same type of water samples.

5.2.3 Nitrous oxide fluxes

The fluxes of N_2O from field drains into the atmosphere were calculated based on the assumption that all of the dissolved N_2O, at concentrations above that of air saturation is, subsequently lost to the atmosphere (Lemon and Lemon, 1981; Bowden and Bormann, 1986; Hack and Kaupenjohann, 2002; Reay et al., 2004b). Thus, the air saturation N_2O concentration (0.35 µg N L^{-1}) was subtracted from the measured dissolved N_2O concentration, then multiplied by flow rate and divided by drain area (see section 4.2.2.2) to obtain the N_2O flux in field drains. Drain samples were taken on the drain outlets (see Figure 2.11 A), there may be losses of N_2O in the drains before samples were taken. Thus, estimates of N_2O fluxes from drains are likely to be underestimated. N_2O fluxes from stream water were calculated using the same method as Outram and Hiscock (2012). The water-air gas exchange for stream water is calculated according to Equation (5.3):

$$F = kC_w - \frac{C_a}{k'_h} \tag{5.3}$$

where: F is the flux of gas (mol cm^{-2} h^{-1}); k is the transfer velocity of N_2O across the water-air interface (cm h^{-1}); C_w is the concentration of N_2O in water (mol cm^{-3}); C_a is the concentration of N_2O in air (mol cm^{-3}); and k'_h is the Henry's law constant for N_2O which is dimensionless and obtained from literature as 1.02 (Sander, 1999). C_w is obtained from field sampling and C_a is calculated taking the ambient N_2O concentration as 319 ppb (Forster et al., 2007). A value for k was calculated according to Clark et al. (1995), using u, wind speed in (cm h^{-1}), obtained from weather station at Site A (Equation 5.4):

$$k = 2.0 + 0.24u^2 \tag{5.4}$$

The field drain and instream fluxes of N_2O are presented in Table 5.2. The mean N_2O flux for all drains during the study period was 30 g N ha^{-1} a^{-1} with a range of 0-1169 g N ha^{-1} a^{-1}. The mean N_2O flux for most drains was between 10 and 40 g N ha^{-1} a^{-1}, however, drains D16 and D11 had low mean fluxes (<5 g N ha^{-1} a^{-1}), whilst fluxes for D4 (89 g N ha^{-1} a^{-1}) and D5 (79 g N ha^{-1} a^{-1}) were high. Low flow rates are the main reason for the low fluxes in the former, whilst high flow rates explain the high fluxes in the latter drains. It is apparent that drains with high N_2O concentrations do not necessarily generate the largest fluxes of N_2O into the atmosphere, with flux depending upon both concentration and flow rate. D11, for instance has the highest mean N_2O concentration (Table 5.1) yet one of the lowest N_2O fluxes due to low flow rates per drainage area. Thus, field drain N_2O discharges from agricultural soils varied considerably between different drains because of high variability in both flow rate and drainage area. Nevertheless, the annual field drain N_2O fluxes in this study are comparable to the range observed by Hack and Kaupenjohann (2002) in which 11 out of 28 sites studied had field drain

N_2O fluxes of between 10 and 120 g N ha^{-1} a^{-1}. However, the highest value reported was only about 365 g N ha^{-1} a^{-1}, which is lower than the highest value recorded in this study (1169 g N ha^{-1} a^{-1}). Additionally, the N_2O fluxes presented here are consistent with the value of ~50 g N ha^{-1} a^{-1} recorded from agricultural drains between November and March in a study reported by Dowdell et al. (1979).

Calculated instream dissolved N_2O fluxes per hectare ranged from 0.016 to 40 kg N ha^{-1} a^{-1}, with mean value of 5.5 kg N ha^{-1} a^{-1}. This value is significantly higher than the N_2O fluxes from field drain. However, it should be noted that the stream fluxes here are calculated per area of stream surface which cover a relatively small area in the catchment (Figure 2.5 and Table 6.2). This is also comparable to the mean N_2O flux of 7.3 kg N $ha^{-1}a^{-1}$ calculated by Outram and Hiscock (2012) for the River Thurne in south-east England. Site B had the highest N_2O fluxes and site M the lowest, as was true for N_2O concentration. Also, the calculated stream N_2O fluxes here is in the range of (2-18 kg N ha^{-1} a^{-1}) estimated in a study of the eutrophic San Joaquin river in California by Hinshaw and Dahlgren (2013).

The calculated mean field drain N_2O flux of 30 g N ha^{-1} a^{-1} is three orders of magnitude lower than the mean flux of NO_3^- (35 kg N ha^{-1} a^{-1}) (see Section 4.2.2.2) and is the same order of magnitude as the mean fluxes of NH_4^+ (65 g N ha^{-1} a^{-1}) and NO_2^- (25 g N ha^{-1} a^{-1}). This again demonstrates that NO_3^- is the dominant form of N transported from field drains into the surrounding environment. Also, this emphasises that N loss in the form of N_2O is not insignificant and can be as great as or even greater than NH_4^+ and NO_2^-. Compared with direct N_2O emissions of 0.5 to 1 kg N ha^{-1} a^{-1} calculated for soils planted with winter wheat in 1996 and 1997 in the UK (Dobbie et al., 1999), these indirect emissions from field drains are ~6% of this flux. Therefore, the contribution of indirect sources of N_2O from field drains is significant.

Table 5.2 Mean and range of dissolved N_2O fluxes from field drains and stream sites during April 2013-April 2015

	ID	Mean N_2O flux ($g\ N\ ha^{-1}\ a^{-1}$)	Range of N_2O flux ($g\ N\ ha^{-1}\ a^{-1}$)
Drain	D1	42	0-572
	D2	16	0-120
	D3	67	0-1169
	D4	89	0-391
	D5	79	0-562
	D6	5	0-64
	D7	13	0-345
	D8	29	0-291
	D9	6	0-57
	D10	9	0-69
	D11	4	0-109
	D13	7	0-65
	D16	1	0-10
Stream	A	5107	1002-23000
	B	7533	1734-40967
	E	6331	1476-34977
	M	3090	16-23103

The temporal variations in field drain and instream N_2O fluxes are illustrated in Figure 5.5. The N_2O flux in all field drains was low in summer 2013 because most drains were either dry or had very low flows at this time. Fluxes from dry drains were considered to be zero as no dissolved N_2O came out of the drain. The largest weekly rainfall event (68 mm) in mid-October 2013 generated no flux increase in any drains or stream sites. However, the smaller rainfall event in mid-February 2014 (38 mm in a week) generated a sharp increase in most of drains and all the stream sites. After summer 2014, N_2O fluxes from both field drains and stream sites increased during the autumn following a flushing event in mid-October. Similar to the previous year, a less pronounced storm event in late-November 2014 (30 mm in a week) generated a sizable increase in N_2O flux from both the drains and instream, in which almost all drains (D1, D3, D6, D7, D8, D9, D10, D11 and D13) delivered their maximum N_2O fluxes into the atmosphere. Thus, it is apparent that rainfall events are important drivers behind the temporal variability in field drain and instream N_2O flux. N_2O fluxes from drains are calculated from N_2O concentration and flow rate. Field drain N_2O fluxes were significantly lower in summer compared to autumn and winter because of very low flow rates (Figure 4.5) and low concentrations (Figure 5.3). For the total annual fluxes, highest losses of N_2O into atmosphere occurred during the winter (54%), followed by autumn (25%), spring (18%) and finally summer (3%).

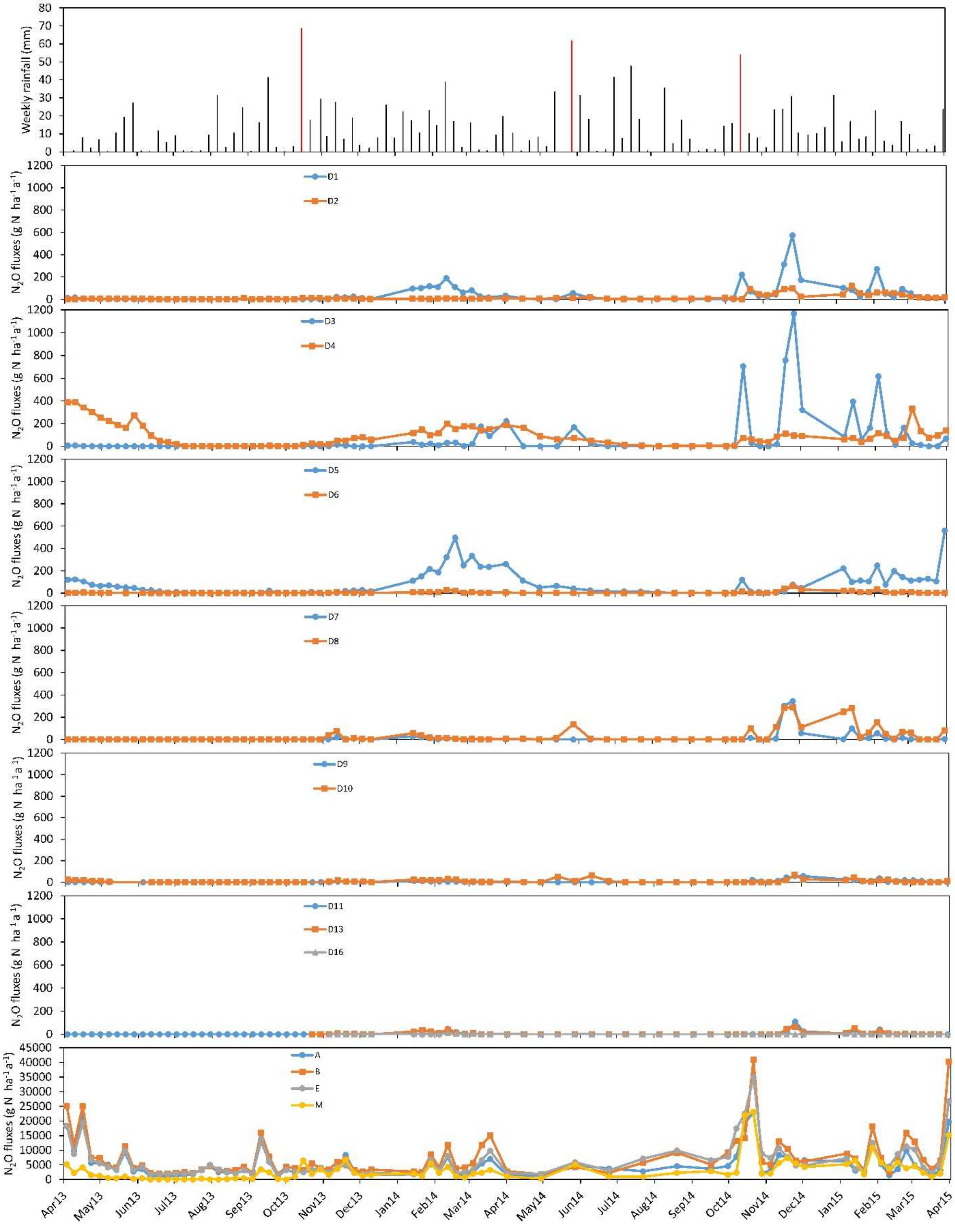

Figure 5.5 Weekly rainfall (top), dissolved N_2O fluxes in field drains (middle) and in stream (bottom) throughout study period.

5.2.4 Downstream variation in nitrous oxide concentration

Sites A, E and M are situated on the same stream with site M being 1035 m upstream of site A, which in turn is 725 m upstream of site E (Figure 2.6 and Figure 5.6). The results revealed that N_2O concentrations increased downstream and this trend was consistent across the entire study period. Out of a total of 77 sampling dates over the two-year monitoring period (Table 5.1), N_2O concentrations were higher at site A than site M on 71 occasions, whilst on 70 occasions N_2O concentration were higher at site E than site A. It was calculated that N_2O concentrations increased by an average of 0.3 µg N L^{-1} as water moved downstream from site M towards site A, and increased a further 0.3 µg N L^{-1} as water moved from site A to site E. Thus, it is calculated that average N_2O concentrations increased downstream by 0.6 µg N L^{-1} over the 1760 m distance between sites M and E.

This increase downstream may be due to two reasons. Firstly, water with high dissolved N_2O concentration enters the stream as it moves downstream. Five drains, namely D2, D1, D4, D6 and D5 enters the stream between sites M and A, and as was shown earlier all these drains had higher N_2O concentrations than stream water. Although it is assumed that drain water will lose all dissolved N_2O to the atmosphere when concentrations are above that of air saturation, some dissolved N_2O above the atmospheric-water equilibrium may remain within the water and enter the stream. Moreover, stream B had the highest N_2O concentration for almost all of the time (Table 5.1 and Figure 5.3), with a mean concentration 0.2 µg N L^{-1} higher than site E, and this entered the main stream just after site A. The joining of this high N_2O water body would likely raise the N_2O concentration downstream. The second potential cause for this increase is N_2O production within the stream itself. As N_2O is produced by both nitrification and denitrification, changing nitrate concentrations along the stream might contribute to the production of N_2O within the stream. Unlike the N_2O data, nitrate concentrations for the stream sites (see Chapter 6) illustrated no particular pattern from upstream to downstream. However, as Abbasi and Adams (2000) mentioned, it is possible that soil and water conditions may allow both nitrification and denitrification to take place simultaneously so that N_2O may be produced without an obvious change of nitrate.

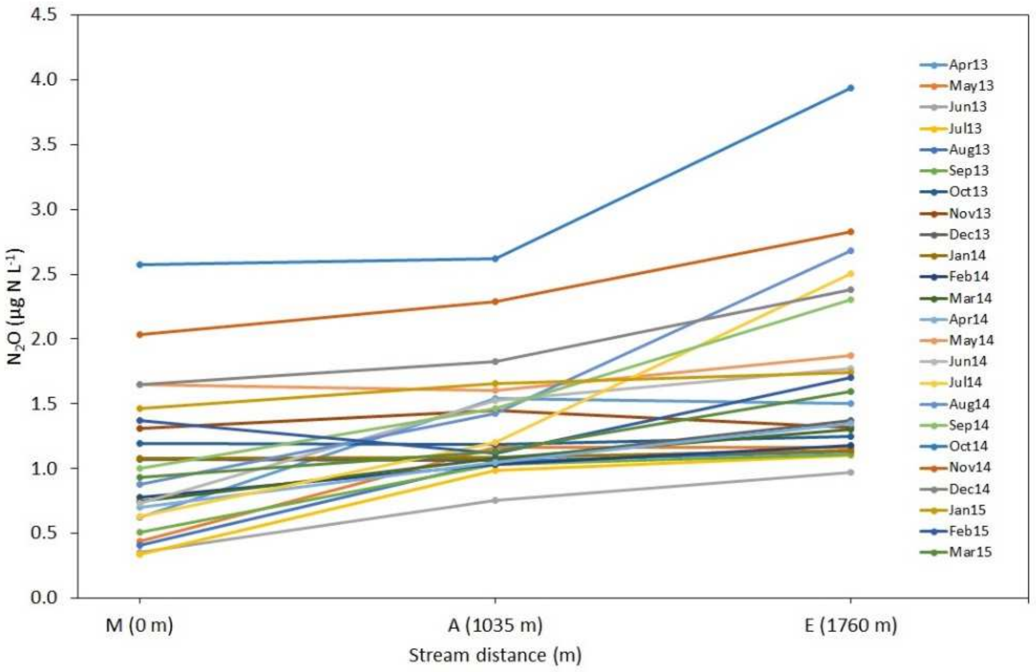

Figure 5.6 Mean monthly N_2O concentrations at three stream sampling sites ordered by distance downstream.

5.2.5 Potential factors controlling N_2O concentrations

5.2.5.1 Soil texture

In addition to the effects of rainfall and season on field drain dissolved N_2O concentrations, an attempt was also made to evaluate the effect of soil texture. Areas of the drains were divided into two dominant soil types (clay loam and sandy loam) based on the soil texture data collected in this study (Figure 3.2 and Figure 4.4). Drains D3, D7, D8, D9, D10 and D13 were within mostly clay loam soils, whereas drains D1, D2, D4, D5, D6, D11 and D16 were within mostly sandy loam soils. Although this assumption might not be completely exact because soil texture in the study area changes from one type into another within a short distance, such that no drain area has just one type of soil texture, there is a dominant soil texture for most of the drains. D8 is a good example of a mostly clay loam soil texture and D6 is a good example of a mostly sandy loam soil.

It was observed that the mean N_2O concentration from clay loam soils (5.47 µg N L^{-1}) was significantly ($p < 0.05$) higher than that from sandy loam soils (4.14 µg N L^{-1}) (Figure 5.7). This is because some drains, namely D8, D9 and D10 with clay loam soils had high mean N_2O concentrations, whilst some drains with a sandy loam soil texture, specifically D2, had low N_2O concentrations (Table 5.1). However, drains with sandy loam soils did not always have low N_2O concentrations, as can be seen for D4 and D11 which both had high N_2O

concentrations. Nevertheless, overall the data presented here suggested that drains within clay loam soils yielded higher N_2O concentrations than sandy loam soils.

Very few of the studies reported in the literature which investigated field drain N_2O concentrations considered soil texture as a potential controlling factor. Thus, it is difficult to directly compare the results here with these studies. Jahangir et al. (2013) observed that mean N_2O concentrations in the groundwater at high permeability (sandy clay loam and sandy loam) agricultural sites were significantly higher than low permeability (silty clay loam and clay loam) agricultural sites. However, numerous studies on direct N_2O emissions have assessed the effects of soil texture. Rochette et al. (2008a) stated that in fine textured soils, higher N_2O emissions are often observed as a result of reduced oxygen levels within the soil matrix due to poor drainage. In a study assessing the effects of soil texture on N_2O emissions from soils, Włodarczyk et al. (2005) emphasised that soil texture and particle size distribution significantly affected the production of N_2O. They observed that the lowest rates of N_2O production were in light textured soils developed from sand, whereas heavier textured soils developed from silt showed the highest rates of N_2O production and consumption. Even heavier textured soil developed from loams showed intermediate N_2O production. Overall, they concluded that heavier textured soils provided more favourable conditions for N_2O production than sandy soils. Hénault et al. (2012) listed a number of controlling factors of N_2O emissions from soils, which included soil texture, and stated that emissions are generally higher for fine-textured soils compared with either coarse or medium- textured soils. The higher N_2O emissions from fine textured soil maybe because of the increased frequency of anaerobic conditions associated with higher water contents in heavier soils. Therefore, assuming that dissolved N_2O concentrations are consistent with direct N_2O emissions from the soil, the findings of this study are in agreement with most other studies assessing the effects of soil texture on N_2O concentrations.

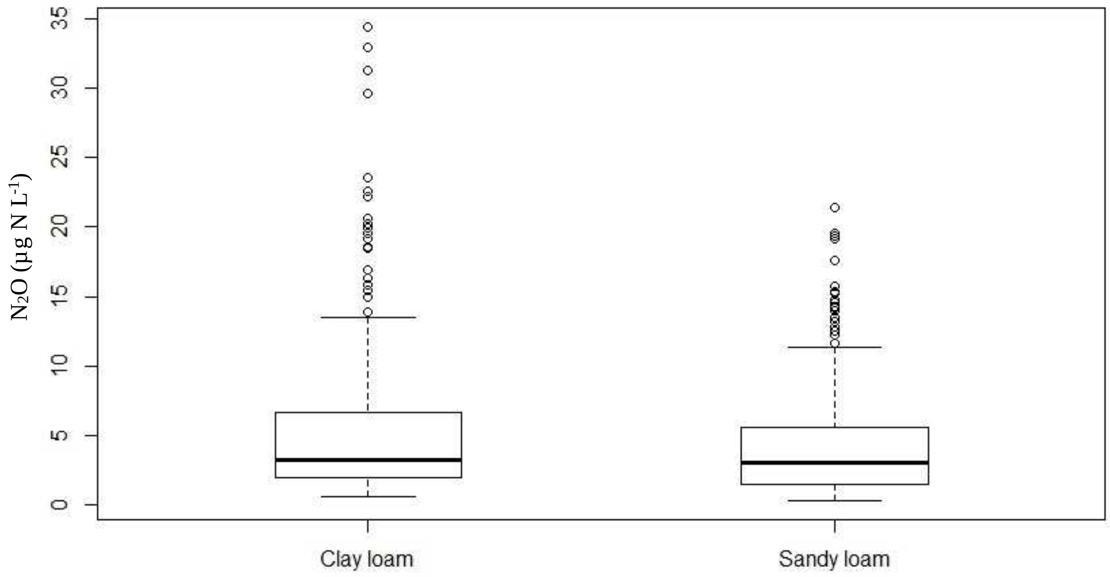

Figure 5.7 Field drain N_2O concentrations for drains underlying the two dominant soil textures in the study area: clay loam (n = 257) and sandy loam (n = 418). Samples collected during April 2013-April 2015.

5.2.5.2 Drain flow rate

N_2O is highly soluble in water and therefore field drains with high flow rates are expected to contain high dissolved N_2O concentrations. Figure 5.8 shows there was a statistically significant weak positive correlation ($r = 0.17$, $p < 0.05$) between these two variables. The correlation did, however, vary greatly amongst the drains. D2 ($r = 0.77$) and D1 ($r = 0.75$) had very strong positive correlations between N_2O concentration and flow rate, whereas D10 ($r = -0.35$), D8 ($r = -0.05$) and D4 ($r = -0.04$) had weak negative correlations. This variability is partially due to soil texture because Drains in sandy loam soils had a strong, positive correlation between the two parameters, whilst drains in clay loam soils had weak, negative correlations. This is supported by the fact that the two drains with the strongest positive correlation (i.e. D1 and D2) were in sandy loam soils and the two drains with the strongest negative correlation (i.e. D8 and D10) were in clay loam soils. However, there were exceptions to this, with drain D13 in a clay loam soil having a strong positive correlation ($r = 0.6$) and D4 in a sandy loam soil having a weak negative correlation ($r = -0.04$). Nevertheless, the data presented here demonstrate that field drain N_2O concentrations generally increase with increasing flow rate.

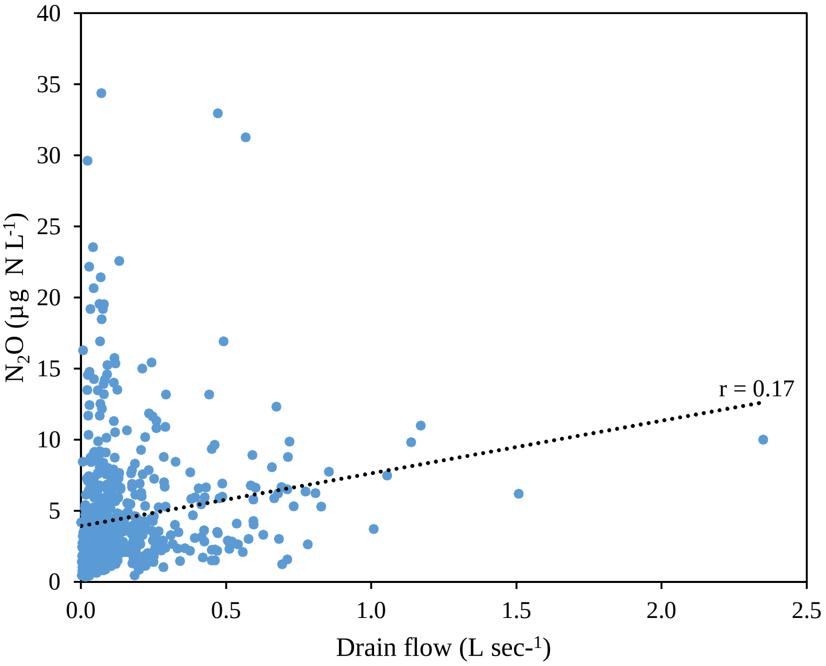

Figure 5.8 Relationship between field drain dissolved N_2O and flow rate for 617 samples collected during April 2013-April 2015. The dash line represents the trend line.

5.2.5.3 pH

The pH values of the field drains ranged from 3.7 to 8.6, with a mean value of 7.7 and a 95% confidence interval for the mean of 7.67-7.76 (Figure 5.9). This illustrates that although the pH range is wide, the majority of the samples are buffered within a very narrow range. A statistically significant negative correlation ($r = -0.19$, $p < 0.001$) was established between pH and dissolved N_2O. Hénault et al. (2012) identified pH as one of the key soil parameters which significantly influences direct N_2O emissions and that lower emission levels were observed when pH >7.3. Hénault et al. (2012) suggested that N_2O emissions from acidic soils generally exceed those from alkaline soils and this probably reflects the reported higher N_2O emissions from nitrification or higher $N_2O:N_2$ ratios at lower pH levels. Similarly, Martikainen and Boer (1993) established an inverse relationship between soil N_2O emissions and soil pH and observed that at pH 4 the production rate of N_2O was 4-8 times greater than at pH 6. Weslien et al. (2009) also observed that soil N_2O emissions were significantly and negatively ($r = -0.93$, $p < 0.05$) correlated with soil pH and suggested that this strong negative correlation represents a well-known relationship from previous studies dealing with soil condition factors on N_2O emissions in which N_2O inhibited by acidic pH, thus enhancing N_2O emissions.

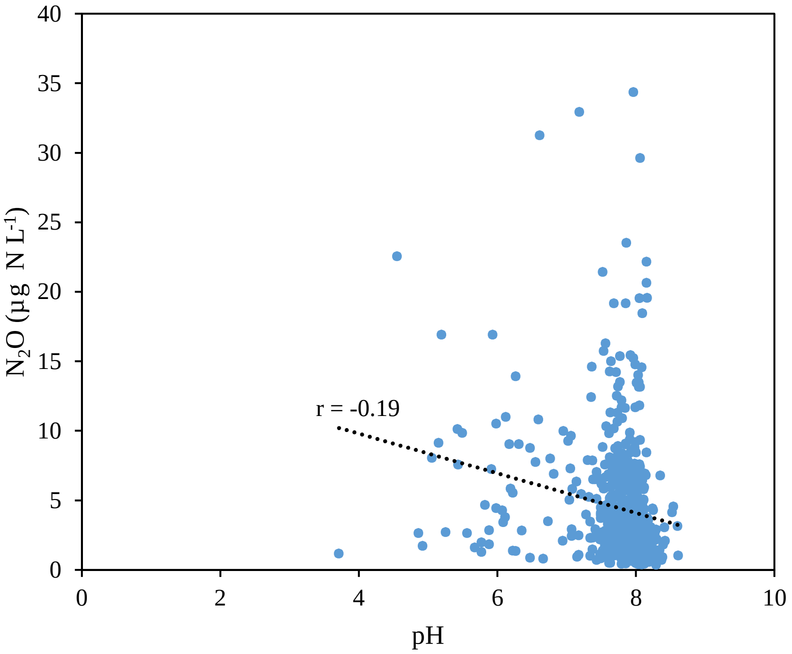

Figure 5.9 Relationship between field drain dissolved N_2O concentration and pH for 617 samples collected during April 2013-April 2015. The dash line represents the trend line.

5.2.5.4 Other nitrogen species

Figure 5.10 shows the relationship between dissolved N_2O concentration and the other three N species measured in field drains and stream water samples. N_2O concentrations were generally three orders of magnitude smaller than dissolved NO_3^-, similar to the findings of previous studies (Ueda et al., 1993; Vilain et al., 2011; Outram and Hiscock, 2012). The concentrations of N_2O and NO_3^- were positively correlated in both the field drains ($r = 0.19$, $p < 0.05$, $n = 617$) and the stream samples ($r = 0.55$, $p < 0.05$, $n = 307$). The stronger correlation in stream samples was possibly due to the greater variation in field drain N_2O concentrations and greater variability in correlation strength amongst the drains. Drains D2 ($r = 0.80$), D6 ($r = 0.67$) and D1 ($r = 0.46$) located within mainly sandy loam soils had the strongest positive correlations, whereas D8 ($r = -0.36$), D13 ($r = -0.29$) and D7 ($r = -0.15$) located within mostly clay loam soils had the strongest negative correlations. Previous studies have suggested that a positive correlation between N_2O and NO_3^- indicates that nitrification is the principle production mechanism for N_2O (Ueda et al., 1993; Hiscock et al., 2003). This is based on the fact that during nitrification of NH_4^+ to NO_3^- a small fraction (~0.1%) of the NH_4^+ is transformed to N_2O (Nevison et al., 1995). Thus, according to this, the main production mechanism for N_2O in stream water is most likely to be nitrification, whereas the dominant production mechanism in field drains might be a combination of both nitrification and denitrification, with the latter occurring mainly in clay loam soils.

In the field drain N_2O study by Hack and Kaupenjohann (2002), higher concentrations of N_2O also coincided with higher concentrations of NO_3^-. Similarly, in a study of N_2O in groundwater from the most important limestone aquifers in the UK, Mühlherr and Hiscock (1998) observed a strong positive correlation ($r = 0.89$ and $r = 0.94$) for two of the aquifers and a strong negative correlation ($r = -0.45$) for one aquifer. They suggested that both nitrification and denitrification are production mechanisms for N_2O. Surprisingly, Reay et al (2004b) observed no relationship between dissolved N_2O and NO_3^- concentration. Therefore, although the correlation between N_2O and NO_3^- observed in this study, especially in the field drain samples, is not as high as some previous studies, the results presented here do support the common understating that dissolved N_2O concentration increases with increasing NO_3^- levels. N_2O was negatively and significantly correlated with NH_4^+ in both field drain water ($r = -0.04$, $p < 0.05$, $n = 472$) and in stream samples ($r = -0.13$, $p < 0.05$, $n = 282$), and this might provide further evidence that nitrification is the main mechanism for N_2O production. N_2O concentrations were not significantly correlated with NO_2^- concentration in either the field drain ($r = 0.05$, $p > 0.05$, $n = 578$) or stream water samples ($r = 0.04$, $p > 0.05$, $n = 307$).

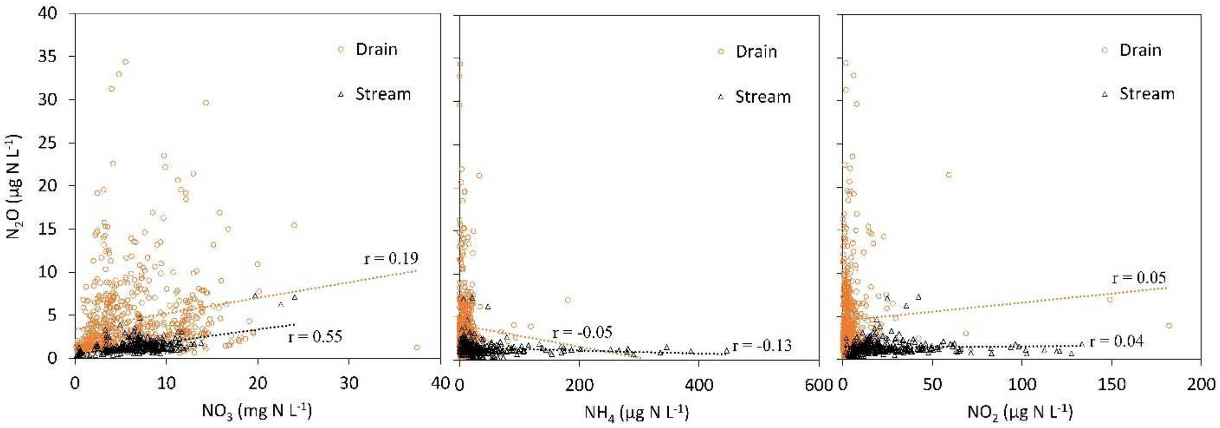

Figure 5.10 Relationship between dissolved N_2O concentration and NO_3^- (n=617, n=308), NH_4^+ (n=420, n=278) and NO_2^- (n=572, n=308) concentrations in field drain and stream water samples, respectively, collected during April 2013-April 2015. Dashed lines represent the trend line.

5.2.6 Impact of a cover crop on nitrous oxide

During autumn and winter 2013-2014, dissolved N_2O concentrations in field drains below the oilseed radish cover crop ranged from 0.6 – 8.8 µg N L^{-1}, whereas concentrations in drains underlying fields without a cover crop ranged from 0.6 - 4.3 µg N L^{-1} (Table 5.3). Mean of

N_2O concentrations in drain water under cover crop (2.61 µg N L^{-1}) was not statistically significant ($p > 0.05$) lower than that under no cover crop (2.23 µg N L^{-1}). In fact slightly higher N_2O concentration was recorded in field drains under a cover crop than without cover crop. This might be due to the accumulation of both carbon and nitrogen residues under the combined cover crop and reduced tillage management and consequently higher substrate availability for nitrification and denitrification compared to conventional management (Abdalla et al., 2012).

Table 5.3 Mean and range of N_2O concentrations in field drains under the two different cover crop treatments during September 2013 to March 2014

Treatments	ID	n	Mean N_2O (µg N L^{-1})	Range of N_2O (µg N L^{-1})
Cover crop	D1	22	3.8	0.8-7.5
	D2	24	1.1	0.6-1.8
	D3	11	2.0	0.6-8.7
	D4	22	2.8	0.7-5.9
	D6	17	2.7	1.3-5.7
	D11	4	7.8	7.3-8.8
	D16	15	2.0	1.5-2.9
Mean			2.61	
No cover crop	D8	15	1.7	0.6-4.3
	D10	15	2.8	2.1-3.6
Mean			2.23	

The primary goal of using a cover crop as a mitigation measure in agriculture is to improve soil fertility and decrease nitrate leaching rather than to reduce greenhouse gas emissions; however the latter should be not neglected when assessing the overall effectiveness of such measures. The effects of cover crops on direct nitrous oxide emissions from soil have been fairly well documented (Jarecki et al., 2009; Kallenbach et al., 2010; Dietzel et al., 2011; Abdalla et al., 2012; Sanz-Cobena et al., 2014), but to our knowledge the effects of cover crops on indirect nitrous oxide emissions from groundwater and rivers receiving N-rich drainage water from agricultural soils has not been studied to date.

The findings from studies investigating the effects of cover crops on soil N_2O emissions are not consistent. To evaluate the change of N_2O emissions following soil application of swine manure, Parkin et al. (2006) found that a rye cover crop significantly lowered cumulative N_2O emissions from fields applied with a large amount of manure compared to fields without a cover crop. However, when smaller amounts of manure were applied the decrease in N_2O emissions under the rye cover crop was not significant. In a growth chamber laboratory experiment, Jarecki et al. (2009) observed a significant reduction in N_2O emissions from soil amended with swine manure slurry in the presence of a rye cover crop. However, Jarecki et al. (2009) found that N_2O emissions were not influenced by the presence of an oat/rye cover crop in the field experiment. Abdalla et al. (2012) even found that daily N_2O emissions were significantly higher under reduced tillage and cover crop treatments compared with conventional farming. In a short study (19/06/2014 to 21/07/2014) on the same study area as this study, Garrard (2014) found that direct N_2O emissions from clay loam soil texture were consistently and significantly higher from Middle Hempsky (MH) field which was under cover crop and direct drill treatment than Potash (P) field which had no cover crop and ploughed conventionally (Figure 5.11).

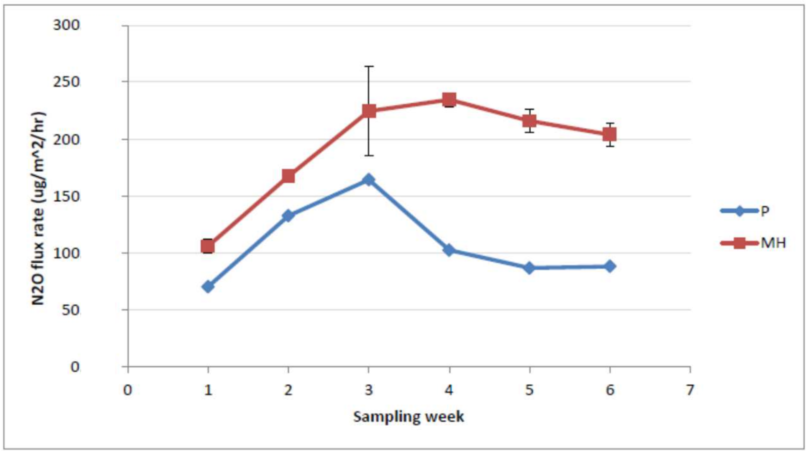

Figure 5.11 Temporal evolution of direct N_2O flux rate from Potash (P) field and Middle Hempsky (MH) over sampling period (19/06/2014 to 21/07/2014) (Garrard, 2014).

The substantial reduction in field drain NO_3^- concentrations under a cover crop is shown in Figure 5.12. The overall mean NO_3^- concentration was significantly ($p < 0.001$) lower in drains under cover crops (2.5 mg N L^{-1}) than drains underneath bare fields (13.9 mg N L^{-1}). This represent a ~82% reduction in NO_3^- concentrations due to the presence of cover crops (see Section 4.2.2.1). However, drains under cover crops contained higher N_2O concentrations than drains under no cover crops. Thus, our study suggests that the use of cover crops as an alternative farm management system to reduce N_2O emissions from agriculture would not be advisable without further research.

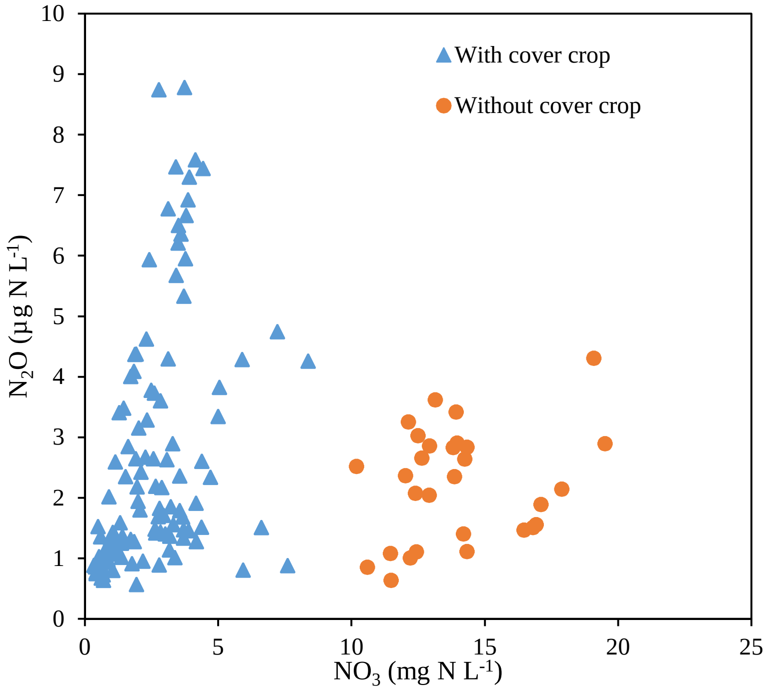

Figure 5.12 Relationship between dissolved nitrous oxide and nitrate concentrations in field drain samples collected during cover crop growth (September 2013 to March 2014) from fields with (n = 114) and without a cover crop (n = 29).

5.2.7 Impact of reduced tillage on nitrous oxide

During the 2014-2015 farming year, different tillage options only (i.e. no cover crop) were continued as a mitigation measure. N_2O concentrations from field drains under different cultivation practices are presented in Table 5.4. The mean N_2O concentration in field drains under conventional tillage (7.6 μg N L^{-1}) was not significantly ($p > 0.05$) different from that under direct drill (6.2 μg N L^{-1}). However, the mean N_2O concentration under reduced tillage (4.4 μg N L^{-1}) was significantly ($p < 0.05$) lower than that under both conventional and no tillage. Despite this finding, the lower N_2O concentrations in field drains under reduced tillage may not truly represent differences in tillage practice. That is because of the three field drains under reduced tillage, only D16 actually had significantly lower N_2O concentrations, whereas D1 and D3 showed no substantial decline in N_2O compared to the other drains (Figure 5.3). Moreover, if N_2O concentration is truly lower in drains under reduced tillage than conventional tillage, then N_2O concentration should be even lower in drains under direct drill because here the soil was not disturbed at all. However, this is not case and drains under direct drill had high

N_2O concentrations that were closer to conventional ploughing than reduced tillage. Thus, the results presented here suggest that different soil inversion methods tended to have little impact on dissolved N_2O concentrations. To our knowledge, there has been no previous study on the effects of different tillage methods on indirect N_2O emission from nitrogen leaching. Thus, it is impossible to compare the results from this study to others. However, several studies were carried out to investigate the effects of soil management on direct N_2O emissions from soil and these have shown inconsistent results, perhaps due to variability in weather and soil conditions, such as soil water content, rates and types of fertiliser application, and depths of fertiliser placement (Baggs et al., 2003; Grant et al., 2004; Venterea et al., 2005). Some researchers have reported greater N_2O emissions with conservation tillage compared to conventional tillage (MacKenzie et al., 1997; Ball et al., 1999; Baggs et al., 2003; Li et al., 2005), whilst others have found lower emissions with conservation tillage relative to conventional tillage (Civerolo and Dickerson, 1998; Grant et al., 2004; Omonode et al., 2011). Furthermore, others have observed no significant difference in N_2O emissions between these two tillage methods (Venterea et al., 2005; Grandy et al., 2006; Lee et al., 2006; Abdalla et al., 2010; Maraseni and Cockfield, 2011).

Table 5.4 Field drain N_2O concentrations under different tillage practices during October 2014 to April 2015. Numbers followed by different superscripted letters are significantly different ($p > 0.05$)

Tillage type	n	Mean N_2O ($\mu g\ N\ L^{-1}$)
Conventional tillage	35	7.6^a
Reduced tillage	62	4.4^b
Direct drill	77	6.2^a

5.3 Summary

All samples collected in this study, regardless of location and time of sampling, contained higher N_2O concentration than the water-air equilibrium, thus demonstrating that all sites were acting as a potential source of N_2O emissions to the atmosphere. This finding is in agreement with the majority of previous research which found N_2O supersaturation in water samples. It was observed that stream samples consistently contained lower N_2O concentrations than the field drains due to rapid degassing of N_2O from drain water once it comes into contact with the atmosphere. A number of factors were determined to have an impact on N_2O concentrations. Rainfall was sometimes found to be solely responsible for a change in N_2O concentration through time, although on a number of occasions the effects of heavy rainfall events were unnoticeable. Thus, whilst it is correct to state that most increases in N_2O concentration were generated by storm events, not all storm events increased N_2O concentration. Seasonally, N_2O concentrations in both field drains and stream samples changed significantly, with lower values during the summer and higher values during spring and autumn. The lowest concentrations during summer were probably due to a decline in drain flow rates and a decline in potentially leachable nitrate from fields due to crop uptake.

In addition to rainfall and season, soil texture also influenced N_2O concentration. Water samples from field drains in clay loam soils tended to contain higher N_2O concentrations than drains in sandy loam soils. However, there was considerable uncertainty associated with this result because some drains were located within a range of soil types. A strong, positive correlation was observed between N_2O concentration and field drain flow rate. This emphasises the importance of rainfall as it generally increases flow rate which in turn increases dissolved N_2O concentrations.

N_2O concentrations were generally three orders of magnitude smaller than dissolved NO_3^- concentrations. The mean N_2O flux (30 g N ha^{-1} a^{-1}) was also three orders of magnitude lower than the mean loss of N in the form of NO_3^- (35 kg N ha^{-1} a^{-1}), although was the same order of magnitude as losses of NH_4^+ (65 g N ha^{-1} a^{-1}) and NO_2^- (25 g N ha^{-1} a^{-1}). This emphasises that NO_3^- is the dominant form of N loss from field drains into the surrounding environment. Additionally, this study highlights that N loss in the form of N_2O is not insignificant and can be as high as NH_4^+ and NO_2^- losses or even higher.

One of the objectives of this study was to assess the effect of on-farm mitigation measures on dissolved N_2O concentration. The results indicated that different inversion soil practices tended to have little effect on N_2O concentrations and fluxes. It was hypothesised that a cover crop would substantially reduce dissolved N_2O concentrations because if similarly reduced NO_3^- concentration. However, higher N_2O concentrations were recorded in field drains under an oilseed radish cover crop than drains beneath fields without a cover crop. Hence, the results presented here suggest that the use of a cover crop as an alternative farm management practice to reduce N_2O emissions would not be successful.

Chapter 6: CALCULATION OF INDIRECT NITROUS OXIDE EMISSION FACTORS

6.1 Introduction

Indirect N_2O emission factors (EF_5) are a way for stating N_2O emissions from a water body as a fraction of the original N flux into the system (Well et al., 2005b). Guidelines on calculating national inventories of N_2O emissions associated with agriculture are provided by the Intergovernmental Panel on Climate Change (IPCC). The IPCC (2006) defined nitrous oxide emission factors for N leaching and runoff from managed soils in regions where leaching and runoff occur as follows:

$$EF_5 = N_2O_{(L)}\text{-}N \,/\, (Total\ N\ input \times Frac_{LEACH}) \qquad (6.1)$$

where, EF_5 is the emission factor for N_2O emissions from N leaching and runoff (kg N_2O–N (kg N)$^{-1}$) with a default of 0.0075 (range = 0.0005-0.025); $N_2O_{(L)}$–N is the annual amount of N_2O–N produced by the leaching and runoff of N additions to managed soils (kg N_2O–N a^{-1}); Total N input is the total annual amount of N added to the system, including synthetic fertilisers and animal manure, that is lost through leaching and runoff (kg N a^{-1}); $Frac_{LEACH}$ is the fraction of all N added to, or mineralised within, managed soils that is lost through leaching and runoff (kg N (kg of N additions)$^{-1}$) with a default of 30% (range = 10-80%).

Researchers calculate indirect N_2O emissions by using the default values of EF and $Frac_{LEACH}$ in equation 6.1. However, other studies such as this study which have already measured indirect N_2O emissions through dissolved N_2O concentration use this equation to calculate EF_5. The IPCC (2006) revised down the default emission factor for indirect N_2O emissions associated with N leaching and runoff (EF_5) from 0.025 kg N_2O-N (kg N)$^{-1}$ in 1997 to 0.0075 kg N_2O-N (kg N)$^{-1}$ in 2006. The EF_5 was further divided into three components according to the site of N_2O production: EF_{5g} for groundwater and surface drainage (0.0025), EF_{5r} for rivers (0.0025) and EF_{5e} for estuaries (0.0025). Thus, 0.0025 for each of the partial EFs and 0.0075 for the overall EF_5. However, EF_5 has a wide range of uncertainty (0.0005–0.025) as a result of natural variability and a lack of data to support designation. The methodology defined by the IPCC to determine EF_5 is calculated by taking the total annual flux of N_2O from a water body and dividing it by the total annual amount of N leached to the water body. This is then used to calculate national N_2O inventories. However, as most studies are often lacking in such detailed mass balance information, EF_5 values are commonly calculated by using a $N_2O/\ NO_3^-$ mass ratio derived using concentration data collected from the water body. Therefore, most studies (Hack and Kaupenjohann, 2002; Hiscock et al., 2003; Reay et al., 2004a; Höll et al., 2005; Sawamoto et al., 2005; Weymann et al., 2008; Reay et al., 2009; Outram and Hiscock, 2012; Hinshaw and Dahlgren, 2013; Minamikawa et al., 2013) on emission factors from leaching and runoff calculate EF_5 by the following method:

$$EF_5 = N_2O\text{-}N / NO_3^-\text{-}N \qquad (6.2)$$

Where, $N_2O\text{-}N$ and $NO_3^-\text{-}N$ are the concentrations of N_2O and NO_3^- measured in agricultural drainage water or groundwater. Considering the IPCC default EF_{5g} of 0.0025, for every kilogram of NO_3^--N in groundwater or drainage water, 2.5 g of N will be released as N_2O.

Reay et al. (2005) indicated that there are several areas of concern with how EF_5 is currently determined. Firstly, EF_{5g} is supposed to account for all N_2O emitted after N leaching and run-off from fields, but prior to instream processing of this N. Therefore, N_2O emissions from riparian areas are indirectly included in EF_{5g}. Such emissions are not accounted for by using the current methodology for the calculation of EF_{5g}. In agricultural areas with tile drainage that provides a direct transfer pathway for leachate into drainage ditches without interaction with the riparian zone, such riparian losses are likely to be insignificant. However, in areas where N in leachate and runoff passes through a riparian wetland or buffer strip, N processing and subsequent N_2O emissions may be important.

In addition to concerns over the potential under-estimation of riparian N_2O emissions, there are also issues with simply using the ratio of dissolved N_2O-N to NO_3^--N in drainage waters to calculate EF_{5g}. This calculation assumes no processing of the leached N occurs, either through reduction of NO_3^- or production of N_2O, between its leaching from the soil and its subsequent sampling point. In reality, it is likely that a substantial amount of N processing occurs, with the nitrate load likely being reduced relative to that initially leached and the N_2O load either increasing or decreasing. However, Nevison (2000) believed that despite some limitations on the significance of the groundwater N_2O/ NO_3^- ratio, at present, this ratio appears to be the best measurement available for linking N_2O to total leached N in groundwater.

6.2 Results and discussion

6.2.1 EF_5 for field drains and stream water

The relationships between N_2O and NO_3^- in field drain and stream water samples are presented in Figure 6.1. The EF_{5g} emission factor (ratio of N_2O-N to NO_3^--N) in drain samples largely varied between 0.00003 and 0.0106, with a mean value of 0.0012. The EF_{5r} emission factor for stream samples was between 0.00006 and 0.0028, with a mean value of 0.0003. Thus, the EF_{5g} values for field drains calculated in this study were predominantly lower than the current IPCC EF_{5g} emission factor of 0.0025 for N_2O emissions from N leaching to groundwater and agricultural drainage water (IPCC, 2006). The EF_{5g} emission factors for 90% of samples collected in this study were lower than the current IPCC EF_{5g} emission factor of 0.0025, whilst ~15% of collected samples were one order magnitude lower than this value. Similarly, the emission factors for stream samples, EF_{5r} were also always lower than the IPCC EF_{5r} emission factor of 0.0025, with a mean value of 0.0003 being an order of magnitude lower than the IPCC value. This indicates that the IPCC revised default value of 0.0025 may still be overestimating indirect N_2O emissions in systems similar to that studied here.

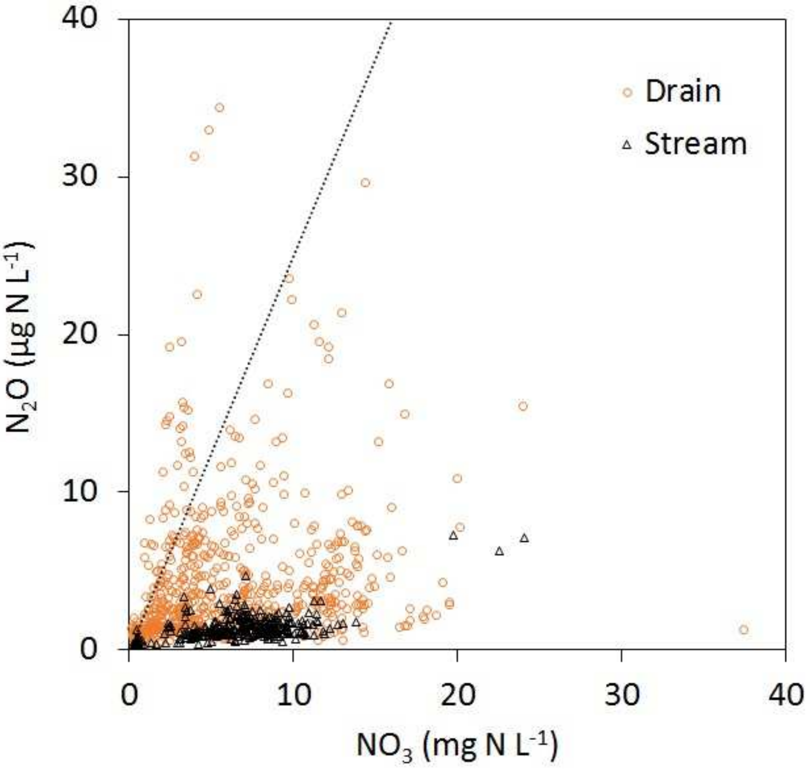

Figure 6.1 Relationship between N_2O-N and NO_3^--N in field drain (n = 617) and stream water (n = 308) samples collected during April 2013-April 2015. The dashed line represents the ratio of N_2O/ NO_3^- (EF_{5g} and EF_{5r}) of the IPCC default value (0.0025). 90% of drain samples and all stream water samples had lower N_2O/ NO_3^- ratios than the IPCC default value.

In addition to this study, a number other studies have also observed lower N_2O-N/ NO_3^--N ratios (EF_{5g}) for groundwater and river water than the IPCC default value (0.0025) (Table 6.1). Hiscock et al. (2003) calculated a value of 0.0019 for chalk groundwater in eastern England, whilst Hack and Kaupenjohann (2002) calculated a value of 0.0008 for field drain water on arable land in the upper Neckar region, Southern Germany. Studying different water bodies in eastern England, Outram and Hiscock (2012) calculated values of 0.0061 and 0.00011 for drainage channels and the River Thurne, respectively. However, few studies have calculated EF_{5g} higher than the current IPCC default value. Reay et al. (2009) derived a value of 0.003 (range 0.00008-0.036) for water samples collected from field drain outfalls in an intensively managed grazed pasture in the Ythan catchment, Aberdeenshire. Hinshaw and Dahlgren (2013) calculated a value of 0.0028 (0.0012-0.0069) for EF_{5r} in a study of the eutrophic San Joaquin River, California.

Table 6.1 Comparison of N_2O-N/ NO_3^--N ratios reported in the literature and derived from this study

Reference	Water body	Land use	N_2O-N/NO_3^--N
Hack and Kaupenjohann (2002)	Field drain (Germany)	Grassland and arable	0.0008 (0.00003-0.005)
Ueda et al. (1993)	Well and springs (US and Japan)	Forest and arable	~0.0001-0.01
Reay et al. (2004b)	Field drain (UK)	Arable	~ 0.0005-0.001
Sawamoto et al. (2003)	Subsurface drainage (Japan)	Arable	0.00076-0.0105
Dowdell et al. (1979)	Agricultural drains (UK)	Arable	~ 0.001-0.01
Ronen et al. (1988b)	Groundwater and sewage (Israel)	Arable	~0.0015-0.0067
Weller et al. (1994)	Groundwater (US)	Forest	~0.0017-0.045
Hiscock et al. (2003)	Groundwater (UK)	Arable	0.0019
Deurer et al. (2008)	Sandy aquifer (Germany)	Arable and Forest	0.002-0.042
Kim et al. (2009)	Sandy aquifer, riparian buffer (US)	Grassland	0.0022-0.0054
Vilain et al. (2011)	Groundwater (France)	Arable	0.0026
Hinshaw and Dahlgren (2013)	River (US)	Arable	0.0028 (0.0012-0.0069)
Reay et al. (2009)	Field drain (UK)	Arable	0.003 (0.00008-0.036)
Mühlherr and Hiscock (1997)	Groundwater (UK)	Arable	0.005 (0.0005-0.0025)
Well et al. (2005a)	Shallow groundwater (Germany)	Arable	0.005-0.73
Outram and Hiscock (2012)	Drainage channel (UK)	Arable	0.0061
Höll et al. (2005)	Soil solution (Germany)	Forest	0.0073
Minamikawa et al. (2010)	Subsurface drainage (Japan)	Arable	0.00820.0296
Jahangir et al. (2013)	Groundwater (Ireland)	Grassland and arable	0.0156 (0.0089-0.0223)
This study	Field drains (UK)	Arable	0.0012 (0.00003-0.0106)
This study	Stream (UK)	Arable	0.0003 (0.00006-0.0028)

Furthermore, according to the IPCC (2006), stream water and groundwater have the same emission factor of 0.0025. However, it is clear from the data presented here that stream water had significantly lower EF values (mean = 0.0003) than field drain samples (mean = 0.0012). These low stream water values are due to the low stream water N_2O content, which is a consequence of the rapid degassing of N_2O from field drains upon contact with the atmosphere prior to reaching the stream. This degassing of supersaturated N_2O in subsurface drainage and groundwater has also been reported in previous studies (Bowden and Bormann, 1986; Reay et al., 2003; Minamikawa et al., 2011; Li et al., 2013).

In this study, it was also possible to estimate EFs using the first method presented in the introduction (Equation 6.1). As mentioned earlier, unlike the N_2O-N/NO_3^--N ratio method, this method requires detailed information (Table 6.2). The total field drain area was estimated at ~21.14 ha (see Section 4.2.2.2), whilst the area of the stream is calculated as the surface area of the stream in mini-catchment A and estimated to be 0.33 ha. These areas were multiplied by the N_2O emissions to calculate the total indirect N_2O emissions annually. Total N input was calculated for the 2012-2013 and 2013-2014 farming years for mini-catchment A (Figure 6.2) using available farm business data. During 2012-2013, the majority of fields were growing spring barley malt and winter wheat feed, receiving fertiliser at a rate of 150-250 kg N ha^{-1}. Three fields contained winter oilseed rape and received 325 kg N ha^{-1} of fertiliser. During 2013-2014, some fields contained spring beans, receiving ~50 kg N ha^{-1}, some had sugar beet and received ~150 kg N ha^{-1}, and the rest had winter wheat feed. In 2013-2014, some fields with spring beans received 0 kg N ha^{-1}. The total N fertiliser applied across mini-catchment A was 67,985 kg N a^{-1} in 2012-2013 and 61,106 kg N a^{-1} in 2013-2014, thus giving an annual mean fertiliser application of 64,545 kg N. It should be noted here that these nitrogen inputs into the soils are applied nitrogen fertilisers by the farmers only. This amount does not include the amount of nitrogen fixed by any legume cops planted. For the field drain area, the total applied N fertiliser was 2,659 kg N a^{-1} for 2012-2013 and 3,080 kg N a^{-1} for 2013-2014, giving a mean total applied N fertiliser of 2,870 kg N a^{-1} (Table 6.2).

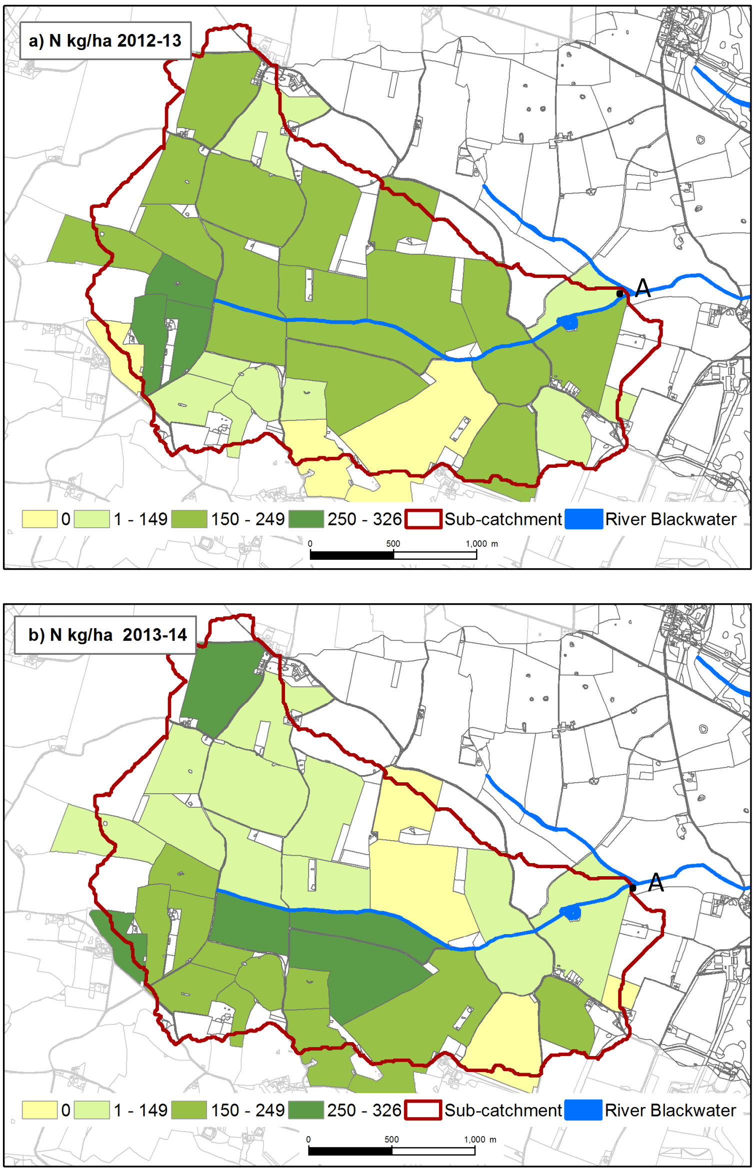

Figure 6.2 Field application of N fertiliser in mini catchment A for the 2012-2013 (top) and 2013-2014 (bottom) farming years.

Frac$_{LEACH}$, which is the fraction (given here as a %) of all applied-N lost through leaching and runoff was calculated from the leached dissolved inorganic and organic N divided by the total N input. N-leaching for the stream was calculated from the flow rate at site kiosk A (see Figure 6.2 for location), multiplied by the inorganic and organic N concentrations of stream samples collected at site A. Leaching rates for individual field drains were calculated from the drain flow rates multiplied by the inorganic and organic N concentrations obtained for each specific drain. For the stream, 15,885 kg N a^{-1} were lost through leaching from a total applied-N of 64,545 kg N a^{-1} in mini-catchment A, giving a Frac$_{LEACH}$ of 25%. For the field drains, Frac$_{LEACH}$

was calculated separately for each drain from the N leached divided by the total N applied over a certain field drain area, giving a mean estimated $Frac_{LEACH}$ value for all drains of 34%.

The mean $Frac_{LEACH}$ value of 34% for the field drains is just above the default value given by the IPCC (30%). However, a wide uncertainty range (10-80%) is given in the IPCC (2006) report and a similarly wide range of values (4-83%) was measured in the field drain samples measured in this study. The mean $Frac_{LEACH}$ value for the stream (25%) was lower than the field drains, perhaps due to a dilution by groundwater that enters the stream. Nevison (2000) criticised the default value of 30% as it was based on the general knowledge of an expert group that developed the 1996 revised IPCC methodology. The default uncertainty range (10-80%) was justified on the basis of the global scale modelling study on N loadings in rivers by Seitzinger and Kroeze (1998). Critically examining the $Frac_{LEACH}$ default value, Nevison (2000) reviewed six specific case studies in the Midwestern United States, generally in maize and/or soybean fields underlain by tile drainage and showing evidence of N leaching into rivers. The inputs and outputs of agricultural N were measured in all the cases at the watershed level over a number of years and mass balances for N were calculated. Nevison (2000) found that $Frac_{LEACH}$ values were typically ~20%. In one case, the fraction of N inputs leached ranged from 3-70%, depending on inter-annual variability in rainfall. One of the reasons that this comparatively low $Frac_{LEACH}$ value was obtained in all six of these case studies was that organic N was not considered as a component, although it may constitute a significant fraction of the total leached N (Seitzinger and Kroeze, 1998). In theory, since $Frac_{LEACH}$ includes both inorganic and organic N, so studies that only consider inorganic N may underestimate the true amount of N leaching. The calculated $Frac_{LEACH}$ values of 34% and 25% for field drains and stream water in this study, respectively, are not too dissimilar from the modelling results for the UK as a whole, in which $Frac_{LEACH}$ values of 16-24% were estimated for the Norfolk area (Cardenas et al., 2013).

Table 6.2 Emission factors Ef_{5g} and EF_{5r} calculated for field drains and stream water using two different methods and the relevant 2006 IPCC emission factors (EF_5)

	Drain	Stream
Area (ha)	21.14	0.33
Indirect N_2O emissions (kg N_2O-N ha^{-1} a^{-1})	0.03	5.11
Total indirect N_2O emissions (kg N_2O-N a^{-1})	0.63	1.67
Total N input (kg N a^{-1})	2,870	64,545
$Frac_{LEACH}$ (%)	34	25
EF_{5g}, EF_{5r} (IPCC (2006) methodology)	0.0006	0.0001
EF_{5g}, EF_{5r} (N_2O-N/NO_3^--N method)	0.0012	0.0003
EF_{5g}, EF_{5r} (IPCC default)	0.0025	0.0025

The results EF_{5g} value calculated using the IPCC methodology was similar to that calculated using the N_2O-N/NO_3^--N ratio (see Table 6.2). The EF_{5g} value was lower using the IPCC (2006) methodology than the N_2O-N/NO_3^--N ratio method for both field drain and stream water samples. This lower calculated EF_{5g} value using the IPCC methodology was expected based on previous research by Well and Butterbach-Bahl (2010). Because significant denitrification is a frequent phenomenon in near-surface groundwater, it can be expected that in many cases some of the leached NO_3^- and N_2O are denitrified before groundwater is discharged via this pathway. Whilst of the same order of magnitude, the EF_{5r} calculated for the stream using the IPCC methodology (0.0001) was three times as low as using the N_2O-N/NO_3^--N ratio (0.0003). The EF_{5g} calculated for field drains using the IPCC methodology was half of the value of EF_{5g} calculated using the N_2O-N/NO_3^--N ratio, with values of 0.0006 and 0.0012, respectively. Hence, if the EF_5 was to be revised by the IPCC, regardless of soil type, crop type and land use practices, then a value of 0.0009 (about third of the current value) for EF_{5g} and a value of 0.0002 (one order lower than the current value) for EF_{5r} may be more reasonable estimates for the types of system studied here.

Calculation of EF_{5g} values using the two different approaches does not necessarily ensure the same result. Most studies calculate EF_{5g} using the N_2O-N/NO_3^--N ratio as they often lack detailed mass balance information and few studies calculate EF_{5g} using the IPCC approach if they have the detailed mass balance information for a catchment. However, there are very few studies that have calculated EF_{5g} using both approaches. One such study is by Outram and Hiscock (2012), who calculated EF_{5g} values for different water bodies using both approaches. They found that EF_{5g} values calculated using the IPCC (2006) approach were very different from those calculated using the N_2O-N/NO_3^--N ratio. The EF_{5g} calculated for the drainage channels using the IPCC approach was an order of magnitude higher than that obtained when using the N_2O-N/NO_3^--N ratio, with values of 0.053 and 0.0061, respectively. This difference increased to two orders of magnitude difference when examining shallow lake water, with values of 0.018 and 0.0008 for the IPCC approach and N_2O-N/NO_3^--N ratio, respectively. The EF_{5r} calculated for the River Thurne using the IPCC approach was nine times as high as that calculated using the N_2O-N/NO_3^--N ratio, with values of 0.009 and 0.00011, respectively. Thus, although in this study the calculated EF_{5g} values using both approaches were not very different, large differences have been calculated in previous studies. Therefore, to achieve an accurate result and avoid miscalculation from using different approaches, the IPCC might need to propose one comprehensive and simple approach.

It is also observed in this study that regardless of the method used for calculation of EF_{5g} and EF_{5r}, the default value set by IPCC (2006) of 0.0025 might overestimate indirect N_2O emission (Table 6.2). The default value is one order of magnitude higher than the EF_{5r} for stream water calculated by either method, with a value of 0.0001 using the IPCC approach and 0.0003 using N_2O-N/NO_3^--N ratio. For the field drains, the default EF_{5g} value is four times higher than that calculated using the IPCC approach (0.0006) and more than double that calculated using the N_2O-N/NO_3^--N ratio (0.0012). Furthermore, the EF_5 values calculated using both the IPCC methodology and the N_2O-N/NO_3^--N ratio revealed that EFs are not uniform for different water bodies. This study has shown that within a single catchment different water bodies can yield different amounts of N_2O with unique EF values required for each type. Therefore, different water bodies need to be separated when emission factors are calculated, unlike the current IPCC approach which uses one EF value for all water bodies.

6.2.2 Spatial variability in EF_5 values

It was shown in Section 6.2.1 that a wide range of N_2O-N/NO_3^--N ratios existed for field drain (0.00003 to 0.0106) and stream water (0.00006 to 0.0028) samples. Other studies have also reported a wide range of values for this ratio, such as Reay et al. (2009) (0.00008-0.036) and

Hack and Kaupenjohann (2002) (0.00003-0.005) (Table 6.1). In this study, because the samples were collected from different drains, it was possible to investigate if this ratio varied spatially (Figure 6.3). None of the field drains nor the stream sampling sites had median $N_2O\text{-}N/NO_3^-\text{-}N$ ratios higher than the IPCC default value (0.0025). However, some sites yielded several samples with higher $N_2O\text{-}N/NO_3^-\text{-}N$ ratios than the IPCC default value, whilst some other locations never had any samples higher than this level. Only 10% of the total collected samples had higher $N_2O\text{-}N/NO_3^-\text{-}N$ ratios than the IPCC default value. This variation in $N_2O\text{-}N/NO_3^-\text{-}N$ ratio was generally due to N_2O concentrations measured in those sites. Sites with high N_2O concentrations, such as D4 and D11 (Figure 5.2), generally had high $N_2O\text{-}N/NO_3^-\text{-}N$ ratios, whilst sites with low N_2O concentrations, such as D5, largely had low $N_2O\text{-}N/NO_3^-\text{-}N$ ratios. However, this was not always a case as some sites, such as D2, had low N_2O concentrations but did not have a low $N_2O\text{-}N/NO_3^-\text{-}N$ ratio. The factors which influence N_2O concentrations that were described in Chapter 5, such as rainfall, soil texture, drain flow rate and land management practices, are also influencing the $N_2O\text{-}N/NO_3^-\text{-}N$ ratio.

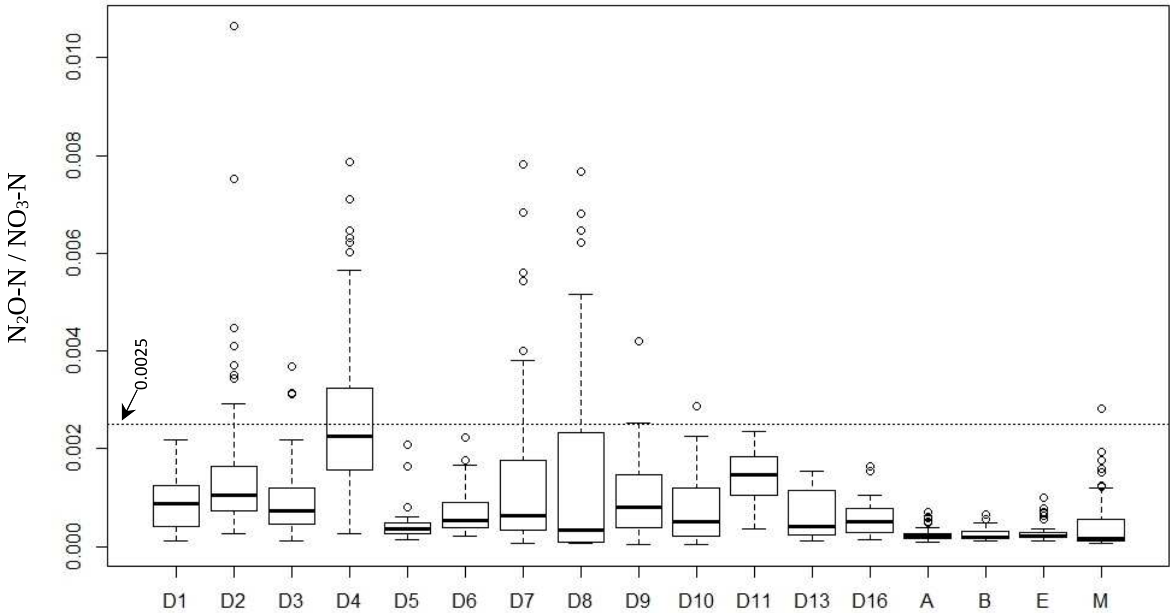

Figure 6.3 Boxplot of the $N_2O\text{-}N/NO_3^-\text{-}N$ ratio in field drains (D1-D16) and in stream (A, B, E and M) for samples collected during April 2013-April 2015. The horizontal dashed line represents the ratio of N_2O/ NO_3 (EF_5) at the IPCC default value of 0.0025.

6.2.3 Temporal variability in EF_5 values

As illustrated in Figure 6.1, the $N_2O\text{-}N/NO_3^-\text{-}N$ ratios varied substantially across field drain and stream water samples. It was possible in this study to monitor this variability temporally over the two-year sampling period. Changes in the N_2O-N/NO_3^--N ratios over time are presented in Figure 6.4. As illustrated in Figures 6.1 and 6.4, the N_2O-N/NO_3^--N ratios of stream water samples were always lower than that of the field drains throughout the entire study period by about one order of magnitude. For both field drains and stream samples, it is observed that the ratio started increasing during the summer (July) and returned to relatively stable levels in mid-October. This pattern was repeated in both sampling years. This might be due to the substantial decline in NO_3^- concentrations during the summer as the result of a decline in water flow and a decline in potentially leachable nitrate due to nutrient uptake by crops during this period.

It is also observed in Figure 6.4 that the N_2O-N/NO_3^--N ratios for stream samples were always lower than the IPCC default value (0.0025) throughout the study period. The increases which occurred during summer in both years never reached the 0.0025 level. Additionally, the N_2O-N/NO_3^--N ratio in field drain samples was always lower than the IPCC default value of 0.0025, except during the summer when the ratios exceeded 0.0025. This summer period of high ratios represents ~10% of collected samples shown in Figure 6.4. To our knowledge, there have been no previous studies on the temporal variability of EF_5 values, so a comparison with the literature cannot be made. Therefore, it is clear from the data presented here that EF values varied over time and that it might be inappropriate to have one EF value throughout a year as it suggested by the IPCC (2006).

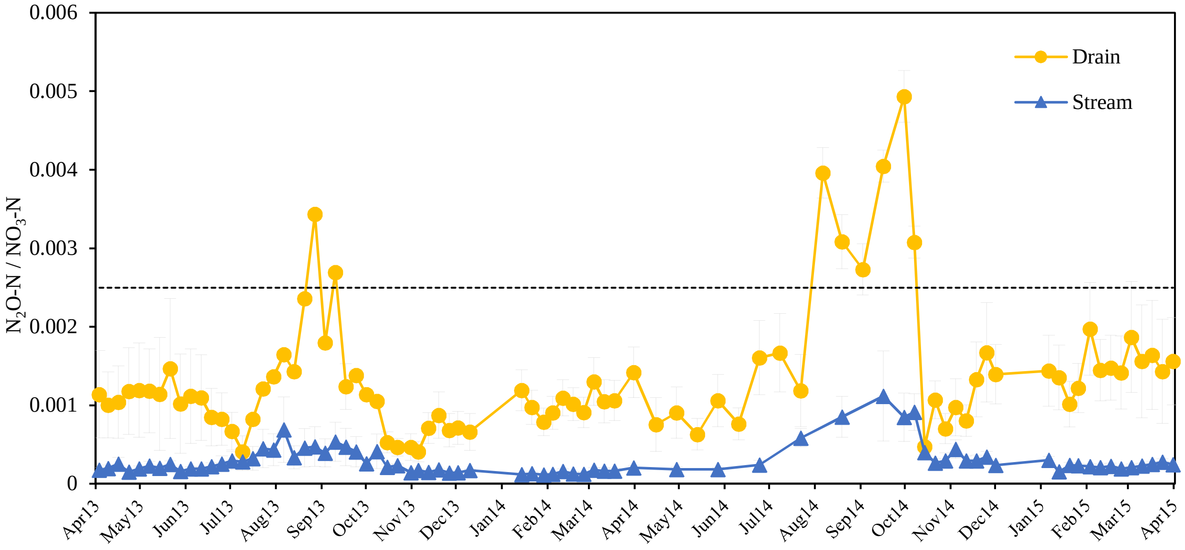

Figure 6.4 Temporal change of mean of N_2O-N/NO_3^--N in field drain and stream water samples collected during April 2013-April 2015. The dashed line represents the ratio of N_2O-N/NO_3^--N (EF_5) for the IPCC default value of 0.0025.

6.2.4 Implications of the measured EF values

From the results of this study, it appears that the indirect emissions component of the UK agricultural N_2O budget may be overestimated using the current default emission factor (EF_{5g}). Revision of this emission factor in line with the findings presented here would result in a large reduction in the estimated N_2O emissions in both the UK and globally (Table 6.3). For the UK, the indirect N_2O emissions arising from N leaching and runoff from agroecosystems total around 14.3 Gg N a^{-1}, based on the 1997 guideline default value for EF_5 of 0.025 (Reay et al., 2005). Using the revised IPCC (2006) EF_5 values of 0.0075, these indirect emissions were reduced significantly to 6.71 Gg N_2O-N a^{-1} (Syakila and Kroeze, 2011). If the calculated value for EF_5 of 0.0036 (sum of all EF_{5g}, EF_{5r} and EF_{5e}) (See Table 6.3) in this study is applied, these emissions would be further reduced to 3.22 Gg N_2O-N a^{-1}.

Similarly, the estimates of indirect N_2O emissions from leaching and runoff globally would be reduced. Firstly, these emissions were estimated to be 1.90 Tg N_2O-N a^{-1} based on the default value for EF_5 of 0.025 reported by the IPCC in 1997. However, with a revised default value of 0.0075, these estimates were substantially reduced to 0.60 Tg N_2O-N a^{-1} in 2006 (Syakila and Kroeze, 2011), and should the suggested emission factor from this study (0.0036) be applied to global indirect N_2O emissions, the estimates would be further reduced to 0.28 Tg N_2O-N a^{-1}. The above emphasises that the revision of emission factors in 2006 by the IPCC considerably lowered global estimates of indirect N_2O emissions from leaching and runoff compared to the previous estimates in 1997. However, it appears that the current IPCC inventory may still overestimate the actual emissions of N_2O. If the emission factors calculated in this study were to be applied, further significant reductions in indirect N_2O emissions similar to the level of reductions achieved in the 2006 revision, will be observed again. Several studies investigating emission factors and indirect N_2O emissions from leaching and runoff have been carried out since 2006, and so updating the IPCC guidelines and indirect N_2O estimates by incorporating these new findings from recent investigations is strongly recommended.

Table 6.3 Summary of current and previous IPCC default emission factors for indirect N leaching from agriculture, calculated emission factors from this study and implications for indirect N_2O emission estimates from the UK and globally

	IPCC 1997	IPCC 2006	This study
EF_{5g}	0.015	0.0025	0.0009
EF_{5r}	0.0075	0.0025	0.0002
EF_{5e}	0.0025	0.0025	0.0025©
EF_5	0.025	0.0075	0.0036*
Indirect UK N_2O emissions ($Gg\ N\ a^{-1}$) from N leaching and runoff	14.30+	6.71∞	3.22
Indirect global N_2O emissions ($Tg\ N\ a^{-1}$) from N leaching and runoff	1.90β	0.60β	0.28

©Assumed unchanged, because this value remained the same from IPCC (1997) to IPCC (2006).

*EF_5 is the sum of all three EFs (i.e. EF_{5g}+EF_{5r}+EF_{5e}).

+From Reay et al. (2005)

∞Total UK N_2O emissions are 59 Gg N a^{-1} for 2013 according to the National Atmospheric Emissions Inventory (2015), and from this amount, indirect emissions account for 24%, with 46% of associated with N leaching and runoff (Syakila and Kroeze, 2011).

βFrom Syakila and Kroeze (2011)

6.3 Summary

In this chapter, indirect nitrous oxide emission factors associated with agricultural nitrogen loss through leaching and runoff (EF_5) have been calculated using two approaches: the IPCC approach and the N_2O-N/NO_3^--N ratio method. Although the calculated EF_5 values were lower using the IPCC approach than the N_2O-N/NO_3^--N ratio in both field drain and stream water samples, the differences were significant. The EF_{5r} calculated for stream samples using the IPCC approach (0.0001) was three times lower than the value obtained using the N_2O-N/NO_3^--N ratio (0.0003). The EF_{5g} calculated for field drains using the IPCC approach (0.0006) was half that calculated using the N_2O-N/NO_3^--N ratio (0.0012).

The EF_5 values calculated using either method for both field drain and stream water samples were lower than the IPCC default value of 0.0025. The calculated EF_{5r} for stream samples was at least one order of magnitude lower than the IPCC default value and EF_{5g} value was at least a third lower for the field drains. This indicates that the IPCC (2006) default value may still overestimate indirect N_2O emissions. Temporal variations in the N_2O-N/NO_3^--N ratio showed that this ratio is not constant. Increases in the N_2O-N/NO_3^--N ratio were recorded during the summer in both field drain and stream samples over both monitoring years and corresponded to substantial declines in NO_3^- concentrations. Summer was the only period when that N_2O-

N/NO_3^--N ratio reached the IPCC default value for drain samples, whilst these increases never reached the default value for stream samples. It is concluded here that unlike the IPCC approach, which uses one EF value for both groundwater and stream water, different water bodies have different EF values. Thus, if the EF_5 were to be revised by the IPCC, regardless of soil type, crop type and land use practices, then a value of 0.0009 (about a third of the current value) for EF_{5g} and a value of 0.0002 (one order of magnitude lower than the current value) for EF_{5r} may be more reasonable estimates. Such a radical downward revision would more than halve the current estimates of indirect N_2O emissions associated with leaching and runoff from agriculture for the UK and globally.

Chapter 7: CONCLUSIONS AND FURTHER WORK

7.1 Conclusions

The overall aim of this study was to assess the effectiveness of cover cropping and conservation tillage as in-field mitigation measures to improve water and soil quality and nitrous oxide greenhouse gas emissions. It was hypothesised that implementing these mitigation measures would improve soil and water quality and reduce indirect emissions of nitrous oxide. However, a wide range of outcomes were obtained from the investigations carried out here, some of which were positive, some negative and others which had no discernible effect.

Apart from significant reduction in soil nitrate, the mitigation measures did not substantially improve any soil quality parameters. Cover crop depleted soil nitrate at three soil depths of 15 cm, 45 cm and 75 cm by 7%, 42% and 79%, respectively. This might be due to the distinctive properties of the oilseed radish cover crop in scavenging nitrate from deep within the soil profile. The soils were mostly below the accepted 2% soil organic carbon threshold, so it would have been a great success if the implemented measures could help elevate this problem. However, it appeared that the measures had little effect on soil organic carbon levels, although increases might be observed if the mitigation measure trials were run for several more years. Regarding other soil macro and micro nutrients, such as soil phosphorus, potassium, magnesium and sulphate, the overall quality of soil in the study area was not in good condition. Soil phosphorus concentrations were excessively high and other nutrients were generally below optimum range. Therefore, reducing soil phosphorus levels and raising the concentrations of other soil nutrients were desired through the mitigation measures. However, instead of reducing phosphorus, the measures acted to retain high soil phosphorus levels. Whilst this is an advantage for soil where phosphorus is a limiting nutrient, in this study area where phosphorus represents an environmental risk, high levels pose a potential eutrophication issue should soluble phosphorus enter the stream via surface runoff. Soil potassium levels were consistently increased by the measures, but the soil magnesium and sulphate levels were not improved. Improvements in soil physical conditions, such as increased temperature, moisture content and infiltration rate, along with decreased bulk density and penetration resistance, were sought through the measures. However, soil temperature and soil moisture did not change and bulk density and penetration resistance increased and consequently reduced the infiltration capacity.

Therefore, the mitigation measures did not improve the overall soil quality, and in some respects they actually caused a deterioration in soil quality.

Unlike soil quality, water quality was significantly improved by the mitigation measures. With the presence of a cover crop, the concentration of dissolved nitrate in soil water, the main form of N, was significantly depleted from 13.9 mg N L^{-1} under bare soils to 2.5 mg N L^{-1} under the cover crop, which represents an 82% reduction. Likewise, the nitrate leaching rate from fields were substantially reduced from 113 kg N ha^{-1} a^{-1} in bare fields to 17 kg N ha^{-1} a^{-1} in cover crop fields, equivalent to 85% reduction in N fluxes. Thus, the cover crop reduced both N concentrations and N fluxes by at least 80%. No changes in either ammonium or nitrite concentrations and fluxes in field drainage between the cover crop and no cover crop fields were observed. However, these two forms of N contributed an insignificant proportion to total N leaching and so does not lessen the great effectiveness of cover crops as a mitigation measures for reducing N leaching. On the other hand, different soil inversion intensities tended to have no effect on either concentrations or fluxes of any form of N. Overall, it was concluded that for an agricultural area where high nitrate losses from fields into groundwater or surface water is occurring, such as in the intensive arable land as Norfolk, inclusion of winter cover crops within a crop rotation is a highly recommended mitigation measure.

The only drawback of the implemented mitigation measures with respect to water chemistry was the impact they had on dissolved nitrous oxide (N_2O) concentrations. It was hypothesised that cover crops would substantially reduce dissolved N_2O concentrations because of their ability to reduce nitrate which was generally found to have a linear relationship with N_2O. However, although the differences were not statistically significant, higher N_2O concentrations were recorded in field drains under cover crops than without a cover crop. This result may suggest that the use of cover crop as an alternative farm management system to reduce N_2O emissions would not be successful. However, in the study area the implemented mitigation measures were employed to tackle high nitrate losses rather than high indirect N_2O emissions and as such were found to be effective. Overall, it is concluded from this study that the implemented mitigation measures, particularly the use of a winter cover crops, are highly recommended.

All samples collected in this study, regardless of location and time of sampling, contained higher N_2O concentration than the water-air equilibrium concentration, illustrating that all sites were acting as a source of N_2O emissions to the atmosphere. Stream samples consistently contained lower N_2O concentrations than field drain samples due to the rapid degassing of N_2O from drain water on contact with the atmosphere. Sometimes rainfall was found to be the only factor driving changes in N_2O concentration over time. However, in a few cases large storm events had negligible effects on N_2O concentration. Moreover, N_2O concentrations in both field drain and stream water samples changed significantly with season, with lowest values occurring during the summer and highest values during the spring and autumn. Although somewhat uncertain, soil texture was observed to exert an influence on N_2O concentrations, with clayey loam soils tending to contain higher N_2O concentrations than sandy loam soils. A strong, positive correlation between N_2O concentration and drain flow rate highlighted the influence of rainfall as it generally increases flow rate which in turn increases N_2O fluxes. N_2O concentrations were largely three orders of magnitude smaller than dissolved NO_3^- concentrations. Mean field drain fluxes of 30 g N ha^{-1} a^{-1} for N_2O were also three orders of magnitude lower than the mean NO_3^- loss of 35 kg N ha^{-1} a^{-1} and were comparable to the losses of NH_4^+ (65 g N ha^{-1} a^{-1}) and NO_2^- (25 g N ha^{-1} a^{-1}). This emphasised again that NO_3^- is the dominant form of N loss from field drains in the study. Also, this stressed that N loss in the form of N_2O is significant and can be as high as NH_4^+ or NO_2^- or even higher.

In this study, indirect nitrous oxide emission factors associated with agricultural nitrogen loss through leaching and runoff (EF_5) have been calculated using two approaches: the IPCC (2006) approach and the N_2O-N/NO_3^--N ratio method. For both field drain and stream water samples, the calculated EF_5 values were lower when using the IPCC approach than the N_2O-N/NO_3^--N ratio, although the differences were small. The EF_{5r} values calculated for stream water using the IPCC approach (0.0001) were three times lower than the value obtained using the N_2O-N/NO_3^--N ratio (0.0003). The EF_{5g} value calculated for field drains using the IPCC approach (0.0006) was half that calculated using the N_2O-N/NO_3^--N ratio (0.0012).

One of the objectives of this study was to compare the calculated EF_5 values based on the two years of monitoring data gathered here for field drains and streams to the IPCC default value. It was found that the calculated EF_{5g} and EF_{5r} emission factors using either method for both drain and stream samples, respectively, were lower than the IPCC default value of 0.0025. The calculated EF_{5r} for stream samples was at least one order of magnitude lower than IPCC default value and it was at least third lower in case of the field drains. This indicated that the IPCC (2006) default value, if applied, may still overestimate indirect N_2O emissions. Temporal variation in the N_2O-N/NO_3^--N ratio showed that this ratio is not constant through time. Increases in the N_2O/NO_3 ratio were recorded during the summer in both field drains and stream samples during both monitoring years. The summer period was the only time when the N_2O/NO_3 ratio reached the IPCC default value for drain samples, whilst increases in stream samples never reached the default value. It is concluded here that unlike the IPCC approach, which uses one EF value for both groundwater and stream water, different water bodies have different EFs values. Hence, if the EF_5 was to be revised again for the IPCC methodology, regardless of soil type, crop type and land use practices, then a value of 0.0009 (about a third of the current value) for EF_{5g} and a value of 0.0002 (one order lower than the current value) for EF_{5r} may be more reasonable estimates. Such a radical downward revision would more than halve the current estimates of indirect N_2O emissions associated with leaching and runoff from agriculture for the UK and globally.

7.2 Recommendations for further research

Several aspects concerning the effects of mitigation measures on soil and water quality and N_2O emissions have been addressed throughout the course of this study and the main objectives stated in the introductory chapter have been fulfilled. However, there remains room to further improve our understanding.

If possible, a long-term experiment is required to make the correct decision to accept or reject mitigation measures as part of routine agricultural practices in intensive arable areas. Thus, continuing with the implemented mitigation measures trial for the next few years with the current soil and water sampling scheme in the study area is highly recommend because, as discussed earlier, substantial changes in soil and water quality may only be observed after many years of running the trials. Moreover, to correctly assess the effectiveness of the mitigation measures, all soil and water quality parameters as well as greenhouses gases should be taken into consideration. However, emphasis should be placed on the effectiveness of such measures in tackling the predominant environmental issues in any area.

Regarding dissolved nitrous oxide measurements, some measurements of stable isotope composition of nitrous oxide in drain and stream samples might help to finally clarify the prevailing production mechanism. Indirect N_2O emissions are from aquatic systems that receive N-rich water, typically from agriculture. Dissolved N_2O measurements from other water bodies such as lakes, rivers and aquifers might help to calculate N_2O emissions more precisely as these also contribute to overall N_2O emissions. Additionally, to calculate N_2O

emissions globally and to reduce the large uncertainties in the estimated EF values, further global work is required with more sampling of groundwater and other waterbodies with variable anthropogenic influence. It would be very helpful if direct N_2O emissions from soils were measured alongside indirect N_2O emissions so that the effects of land use and weather could be observed on these major emissions and a more accurate comparison between direct and indirect N_2O losses could be made. Finally, several studies to estimate emissions factors and indirect N_2O emissions from leaching and runoff have been carried out since 2006. Therefore, updating the IPCC guidelines and indirect N_2O estimates by considering the new findings from recent investigations is recommended.

REFERENCES

Abbasi, M. & Adams, W. 2000. Gaseous N emission during simultaneous nitrification–denitrification associated with mineral N fertilization to a grassland soil under field conditions. *Soil Biology and Biochemistry* 32: 1251-1259

Abdalla, M., Hastings, A., Helmy, M., Prescher, A., Osborne, B., Lanigan, G., Forristal, D., Killi, D., Maratha, P. & Williams, M. 2014. Assessing the combined use of reduced tillage and cover crops for mitigating greenhouse gas emissions from arable ecosystem. *Geoderma* 223: 9-20

Abdalla, M., Jones, M., Ambus, P. & Williams, M. 2010. Emissions of nitrous oxide from Irish arable soils: effects of tillage and reduced N input. *Nutrient Cycling in Agroecosystems* 86: 53-65

Abdalla, M., Rueangritsarakul, K., Jones, M., Osborne, B., Helmy, M., Roth, B., Burke, J., Nolan, P., Smith, P. & Williams, M. 2012. How Effective is Reduced Tillage–Cover Crop Management in Reducing N_2O Fluxes from Arable Crop Soils? *Water, Air, & Soil Pollution* 223: 5155-5174

Abdollahi, L. & Munkholm, L. J. 2014. Tillage system and cover crop effects on soil quality: I. Chemical, mechanical, and biological properties. *Soil Science Society of America Journal* 78: 262-270

Alam, A. 2014. Soil degradation: a challenge to sustainable agriculture. *International Journal of Scientific Research in Agricultural Sciences* 1: 50-55

Anderson, B., Bartlett, K., Frolking, S., Hayhoe, K. & Jenkins, J. 2010. Methane and nitrous oxide emissions from natural sources. [Washington, D.C.]. U.S. Environmental Protection Agency, Office of Atmospheric Programs.

Andraski, T. W., Bundy, L. G. & Kilian, K. C. 2003. Manure history and long-term tillage effects on soil properties and phosphorus losses in runoff. *Journal of Environmental Quality* 32: 1782-1789

Arshad, M., Franzluebbers, A. & Azooz, R. 1999. Components of surface soil structure under conventional and no-tillage in northwestern Canada. *Soil and Tillage Research* 53: 41-47

Arthurton, R. S., Booth, S. J., Morigi, A. N., Abbott, M. A. W. & Wood, C. J. 1994. Geology of the country around Great Yarmouth British Geological Survey, HMSO: London

Baggs, E., Stevenson, M., Pihlatie, M., Regar, A., Cook, H. & Cadisch, G. 2003. Nitrous oxide emissions following application of residues and fertiliser under zero and conventional tillage. *Plant and Soil* 254: 361-370

Ball, B. C., Scott, A. & Parker, J. P. 1999. Field N_2O, CO_2 and CH_4 fluxes in relation to tillage, compaction and soil quality in Scotland. *Soil and Tillage Research* 53: 29-39

Bange, H. W., Rapsomanikis, S. & Andreae, M. O. 2001. Nitrous oxide cycling in the Arabian Sea. *Journal of Geophysical Research: Oceans (1978–2012)* 106: 1053-1065

Beauchamp, E. 1997. Nitrous oxide emission from agricultural soils. *Canadian Journal of Soil Science* 77: 113-123

Beaulieu, J. J., Arango, C. P., Hamilton, S. K. & Tank, J. L. 2008. The production and emission of nitrous oxide from headwater streams in the Midwestern United States. *Global Change Biology* 14: 878-894

Beaulieu, J. J., Tank, J. L., Hamilton, S. K., Wollheim, W. M., Hall, R. O., Mulholland, P. J., Peterson, B. J., Ashkenas, L. R., Cooper, L. W. & Dahm, C. N. 2011. Nitrous oxide emission from denitrification in stream and river networks. *Proceedings of the National Academy of Sciences* 108: 214-219

Bernhard, A. 2012. The nitrogen cycle: Processes, players, and human impact. *Nature Education Knowledge* 3: 25

BGS. 2016. Geology of Britain. 2016: http://www.bgs.ac.uk/data/mapViewers/home.html

Bochove, E. v., Theriault, G., Rochette, P., Jones, H. & Pomeroy, J. 2001. Thick ice layers in snow and frozen soil affecting gas emissions from agricultural soils during winter. *Journal of Geophysical Research: Atmospheres (1984–2012)* 106: 23061-23071

Boesch, D. F., Brinsfield, R. B. & Magnien, R. E. 2001. Chesapeake Bay Eutrophication: Scientific understanding, ecosystem restoration, and challenges for agriculture. *Journal of Environmental Quality* 30: 303-320

Bot, A. & Benites, J. 2005. The importance of soil organic matter: key to drought-resistant soil and sustained food production. Rome. FAO.

Bouwman, A. 1996. Direct emission of nitrous oxide from agricultural soils. *Nutrient Cycling in Agroecosystems* 46: 53-70

Bouwman, A., Fung, I., Matthews, E. & John, J. 1993. Global analysis of the potential for N2O production in natural soils. *Global Biogeochemical Cycles* 7: 557-597

Bowden, W. B. & Bormann, F. 1986. Transport and loss of nitrous oxide in soil water after forest clear-cutting. *Science* 233: 867-869

Bremner, J. M. 1997. Sources of nitrous oxide in soils. *Nutrient Cycling in Agroecosystems* 49: 7-16

Brown, B. D. 2006. Winter cereal–corn double crop forage production and phosphorus removal. *Soil Science Society of America Journal* 70: 1951-1956

Bruinsma, J. 2003. World agriculture: towards 2015/2030. An FAO perspective. Rome. Earthscan, London and Food and Agriculture Organization.

Buol, S. W., Southard, R. J., Graham, R. C. & McDaniel, P. A. 2011. Soil genesis and classification, sixth edition. John Wiley & Sons, 560 pp

Burket, J. Z., Hemphill, D. D. & Dick, R. P. 1997. Winter cover crops and nitrogen management in sweet corn and broccoli rotations. *HortScience* 32: 664-668

Calegari, A., Hargrove, W., Rheinheimer, D. D. S., Ralisch, R., Tessier, D., de Tourdonnet, S. & de Fatima Guimarães, M. 2008. Impact of long-term no-tillage and cropping system management on soil organic carbon in an Oxisol: A model for sustainability. *Agronomy Journal* 100: 1013-1019

Campbell, C., Zentner, R., Selles, F., Biederbeck, V., McConkey, B., Blomert, B. & Jefferson, P. 2000. Quantifying short-term effects of crop rotations on soil organic carbon in southwestern Saskatchewan. *Canadian Journal of Soil Science* 80: 193-202

Cardenas, L., Cuttle, S., Crabtree, B., Hopkins, A., Shepherd, A., Scholefield, D. & del Prado, A. 2011. Cost effectiveness of nitrate leaching mitigation measures for grassland livestock systems at locations in England and Wales. *Science Of The Total Environment* 409: 1104-1115

Cardenas, L., Gooday, R., Brown, L., Scholefield, D., Cuttle, S., Gilhespy, S., Matthews, R., Misselbrook, T., Wang, J. & Li, C. 2013. Towards an improved inventory of N_2O from agriculture: model evaluation of N_2O emission factors and N fraction leached from different sources in UK agriculture. *Atmospheric Environment* 79: 340-348

Ceh. 2016. Station number 34011 - Wensum at Fakenham. 2016: http://nrfa.ceh.ac.uk/

Chen, L., Zhang, J., Zhan, L., Li, Y. & Sun, H. 2014. Differences in nitrous oxide distribution patterns between the Bering Sea basin and Indian Sector of the Southern Ocean. *Acta Oceanologica Sinica* 33: 9-19

Chen, Y., Liu, S., Li, H., Li, X., Song, C., Cruse, R. & Zhang, X. 2011. Effects of conservation tillage on corn and soybean yield in the humid continental climate region of Northeast China. *Soil and Tillage Research* 115: 56-61

Cherry, K., Shepherd, M., Withers, P. & Mooney, S. 2008. Assessing the effectiveness of actions to mitigate nutrient loss from agriculture: A review of methods. *Science Of The Total Environment* 406: 1-23

Civerolo, K. L. & Dickerson, R. R. 1998. Nitric oxide soil emissions from tilled and untilled cornfields. *Agricultural and Forest Meteorology* 90: 307-311

Clark, A. 2008. Managing cover crops profitably. DIANE Publishing

Clark, J., Schlosser, P., Simpson, H., Stute, M., Wanninkhof, R. & Ho, D. 1995. Relationship between gas transfer velocities and wind speeds in the tidal Hudson River determined by the dual tracer technique. in Jahne B. & Monahan E.C. (Eds.), Air-water gas transfer, EON Verlag and Studio, Germany, pp: 785-800

Cole, J. J. & Caraco, N. F. 2001. Emissions of nitrous oxide (N_2O) from a tidal, freshwater river, the Hudson River, New York. *Environmental Science & Technology* 35: 991-996

Constantin, J., Mary, B., Laurent, F., Aubrion, G., Fontaine, A., Kerveillant, P. & Beaudoin, N. 2010. Effects of catch crops, no till and reduced nitrogen fertilization on nitrogen leaching and balance in three long-term experiments. *Agriculture, Ecosystems & Environment* 135: 268-278

Cooper, R. J., Krueger, T., Hiscock, K. M. & Rawlins, B. G. 2014. High-temporal resolution fluvial sediment source fingerprinting with uncertainty: a Bayesian approach. *Earth Surface Processes and Landforms* 40: 78-92

Craig, H. & Gordon, L. 1963. Nitrous oxide in the ocean and the marine atmosphere. *Geochimica et Cosmochimica Acta* 27: 949-955

Cunningham, H. M., Chaney, K., Bradbury, R. B. & Wilcox, A. 2004. Non-inversion tillage and farmland birds: a review with special reference to the UK and Europe. *Ibis* 146: 192-202

da Veiga, M., Horn, R., Reinert, D. J. & Reichert, J. M. 2007. Soil compressibility and penetrability of an Oxisol from southern Brazil, as affected by long-term tillage systems. *Soil and Tillage Research* 92: 104-113

Dabney, S., Delgado, J. & Reeves, D. 2001. Using winter cover crops to improve soil and water quality. *Communications in Soil Science and Plant Analysis* 32: 1221-1250

De Vita, P., Di Paolo, E., Fecondo, G., Di Fonzo, N. & Pisante, M. 2007. No-tillage and conventional tillage effects on durum wheat yield, grain quality and soil moisture content in southern Italy. *Soil and Tillage Research* 92: 69-78

Decagon, D. 2007. Minidisk Infiltrometer. *User's Manual, Version* 4

Defra. 2006. Final Results of the June 2005 Agricultural and Horticultural Census for England.

Defra. 2015. 2010 to 2015 government policy: water quality. 2015: https://www.gov.uk/government/publications/2010-to-2015-government-policy-water-quality/2010-to-2015-government-policy-water-quality

Deurer, M., Von der Heide, C., Böttcher, J., Duijnisveld, W., Weymann, D. & Well, R. 2008. The dynamics of N_2O near the groundwater table and the transfer of N_2O into the unsaturated zone: A case study from a sandy aquifer in Germany. *Catena* 72: 362-373

Di, H. & Cameron, K. 2002. Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. *Nutrient Cycling in Agroecosystems* 64: 237-256

Dietzel, R., Wolfe, D. & Thies, J. E. 2011. The influence of winter soil cover on spring nitrous oxide emissions from an agricultural soil. *Soil Biology and Biochemistry* 43: 1989-1991

Dils, R., Preedy, N. & Burke, S. 2009. Wensum catchment description & background research document for DEFRA-EA Demonstration Test Catchments project, October 2009.

Dinnes, D. L., Karlen, D. L., Jaynes, D. B., Kaspar, T. C., Hatfield, J. L., Colvin, T. S. & Cambardella, C. A. 2002. Nitrogen management strategies to reduce nitrate leaching in tile-drained Midwestern soils. *Agronomy Journal* 94: 153-171

Dobbie, K., McTaggart, I. & Smith, K. 1999. Nitrous oxide emissions from intensive agricultural systems: variations between crops and seasons, key driving variables, and mean emission factors. *Journal of Geophysical Research: Atmospheres (1984–2012)* 104: 26891-26899

Dobbie, K. E. & Smith, K. A. 2003. Nitrous oxide emission factors for agricultural soils in Great Britain: The impact of soil water-filled pore space and other controlling variables. *Global Change Biology* 9: 204-218

Dolan, M., Clapp, C., Allmaras, R., Baker, J. & Molina, J. 2006. Soil organic carbon and nitrogen in a Minnesota soil as related to tillage, residue and nitrogen management. *Soil and Tillage Research* 89: 221-231

Dore, J. E., Popp, B. N., Karl, D. M. & Sansone, F. J. 1998. A large source of atmospheric nitrous oxide from subtropical North Pacific surface waters. *Nature* 396: 63-66

Dowdell, R. J., Burford, J. R. & Crees, R. 1979. Losses of nitrous oxide dissolved in drainage water from agricultural land. *Nature* 278: 342-343

Drury, C., Tan, C., Welacky, T., Reynolds, W., Zhang, T., Oloya, T., McLaughlin, N. & Gaynor, J. 2014. Reducing nitrate loss in tile drainage water with cover crops and water-table management systems. *Journal of Environmental Quality* 43: 587-598

Drury, C. F., Tan, C.-S., Welacky, T. W., Oloya, T. O., Hamill, A. S. & Weaver, S. E. 1999. Red clover and tillage influence on soil temperature, water content, and corn emergence. *Agronomy Journal* 91: 101-108

Du Preez, C., Steyn, J. & Kotze, E. 2001. Long-term effects of wheat residue management on some fertility indicators of a semi-arid Plinthosol. *Soil and Tillage Research* 63: 25-33

Duiker, S. W. & Beegle, D. B. 2006. Soil fertility distributions in long-term no-till, chisel/disk and moldboard plow/disk systems. *Soil and Tillage Research* 88: 30-41

Edwards, A. & Withers, P. 1998. Soil phosphorus management and water quality: a UK perspective. *Soil Use and Management* 14: 124-130

Eswaran, H., Lal, R. & Reich, P. 2001. Land degradation: an overview. in Bridges E. M., Hannam I. D., Oldeman L. R., Penning de Vries F. W. T., Scherr S. J. & Sombatpanit S. (Eds.), Responses to Land degradation, Enfield, New Hampshire, USA, Science Publishers, pp: 20-35

Fabrizzi, K., Garcıa, F., Costa, J. & Picone, L. 2005. Soil water dynamics, physical properties and corn and wheat responses to minimum and no-tillage systems in the southern Pampas of Argentina. *Soil and Tillage Research* 81: 57-69

Fageria, N., Baligar, V. & Bailey, B. 2005. Role of cover crops in improving soil and row crop productivity. *Communications in Soil Science and Plant Analysis* 36: 2733-2757

FAO. 2001. Global Estimates of Gaseous Emissions of NH_3, NO and N_2O from Agricultural Land. Rome. International Fertilizer Industry Association ; FAO.

FAO. 2014. CA Adoption Worldwide. 2014: http://www.fao.org/nr/water/aquastat/data/query/

Fewtrell, L. 2004. Drinking-water nitrate, methemoglobinemia, and global burden of disease: a discussion. *Environmental health perspectives*: 1371-1374

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D. W., Haywood, J., Lean, J., Lowe, D. C. & Myhre, G. 2007. Changes in atmospheric constituents and in radiative forcing. *Climate Change* 20

Fourie, J., Agenbag, G. & Louw, P. 2007. Cover crop management in a Chardonnay/99 Richter vineyard in the coastal region, South Africa. 3. Effect of different cover crops and cover crop management practices on organic matter and macro-nutrient content of a medium-textured soil. *South African Journal for Enology and Viticulture* 28: 61

Franzluebbers, A. & Arshad, M. 1996a. Soil organic matter pools during early adoption of conservation tillage in northwestern Canada. *Soil Science Society of America Journal* 60: 1422-1427

Franzluebbers, A. & Arshad, M. 1996b. Soil organic matter pools with conventional and zero tillage in a cold, semiarid climate. *Soil and Tillage Research* 39: 1-11

Franzluebbers, A. & Hons, F. 1996. Soil-profile distribution of primary and secondary plant-available nutrients under conventional and no tillage. *Soil and Tillage Research* 39: 229-239

Gál, A., Vyn, T. J., Michéli, E., Kladivko, E. J. & McFee, W. W. 2007. Soil carbon and nitrogen accumulation with long-term no-till versus moldboard plowing overestimated with tilled-zone sampling depths. *Soil and Tillage Research* 96: 42-51

Garrard, N. 2014. Direct emissions of nitrous oxide from soils: A comparison of two agricultural regimes. Unpublished MSc. Desertation, University of East Anglia

Gelderman, R. H. & Mallarino, A. P. 2012. Soil Sample Preparation. *Recommended Chemical Soil Test Procedures*: 5

Gilbert, N. 2012. One-third of our greenhouse gas emissions come from agriculture. *Nature News*

Gómez, J. A., Guzmán, M. G., Giráldez, J. V. & Fereres, E. 2009. The influence of cover crops and tillage on water and sediment yield, and on nutrient, and organic matter losses in an olive orchard on a sandy loam soil. *Soil and Tillage Research* 106: 137-144

Govaerts, B., Sayre, K., Lichter, K., Dendooven, L. & Deckers, J. 2007. Influence of permanent raised bed planting and residue management on physical and chemical soil quality in rain fed maize/wheat systems. *Plant and Soil* 291: 39-54

Govaerts, B., Verhulst, N., Castellanos-Navarrete, A., Sayre, K., Dixon, J. & Dendooven, L. 2009. Conservation agriculture and soil carbon sequestration: between myth and farmer reality. *Critical Reviews in Plant Science* 28: 97-122

Grandy, A. S., Loecke, T. D., Parr, S. & Robertson, G. P. 2006. Long-term trends in nitrous oxide emissions, soil nitrogen, and crop yields of till and no-till cropping systems. *Journal of Environmental Quality* 35: 1487-1495

Grant, B., Smith, W., Desjardins, R., Lemke, R. & Li, C. 2004. Estimated N_2O and CO_2 emissions as influenced by agricultural practices in Canada. *Climatic Change* 65: 315-332

Grisso, R. D., Alley, M. M., Holshouser, D. L. & Thomason, W. E. 2009. Precision Farming Tools. Soil Electrical Conductivity. Virginia Cooperative Extension, 442-508 pp

Groffman, P. M., Rogers, J. & Whitman, W. 1991. Ecology of nitrification and denitrification in soil evaluated at scales relevant to atmospheric chemistry. in J. E. Rogers, Whitman, W.B. (Ed.), Microbial production and consumption of greenhouse gases: methane, nitrogen oxides, and halomethanes., Washington DC, American Society of Microbiology, pp: 201-217

Gruber, S., Möhring, J. & Claupein, W. 2011. On the way towards conservation tillage-soil moisture and mineral nitrogen in a long-term field experiment in Germany. *Soil and Tillage Research* 115: 80-87

Hack, J. & Kaupenjohann, M. 2002. N_2O discharge with drain water from agricultural soils of the upper Neckar region in Southern Germany. in J. Van Ham, A. P. M. R. Baede, J. G. Guicherit & F. M. Williams-Jacobse (Eds.), *Non-CO_2 greenhouse gases: scientific understanding, control options and policy aspects. Proceedings of the Third International Symposium*, , Millpress Science Publishers, Maastricht, Netherlands: pp. 185-190

Hall, D., Reeve, M., Thomasson, A. & Wright, V. 1977. Water retention, porosity and density of field soils. Soil Survey. Harpenden.

Hénault, C., Grossel, A., Mary, B., Roussel, M. & Léonard, J. 2012. Nitrous oxide emission by agricultural soils: a review of spatial and temporal variability for mitigation. *Pedosphere* 22: 426-433

Hernanz, J., López, R., Navarrete, L. & Sanchez-Giron, V. 2002. Long-term effects of tillage systems and rotations on soil structural stability and organic carbon stratification in semiarid central Spain. *Soil and Tillage Research* 66: 129-141

Hinshaw, S. E. & Dahlgren, R. A. 2013. Dissolved nitrous oxide concentrations and fluxes from the eutrophic San Joaquin River, California. *Environmental Science & Technology* 47: 1313-1322

Hirsch, A., Michalak, A., Bruhwiler, L., Peters, W., Dlugokencky, E. & Tans, P. 2006. Inverse modeling estimates of the global nitrous oxide surface flux from 1998–2001. *Global Biogeochemical Cycles* 20: GB1008

Hiscock, K., Bateman, A., Mühlherr, I., Fukada, T. & Dennis, P. 2003. Indirect emissions of nitrous oxide from regional aquifers in the United Kingdom. *Environmental Science & Technology* 37: 3507-3512

Hiscock, K., Dennis, P., Saynor, P. & Thomas, M. 1996. Hydrochemical and stable isotope evidence for the extent and nature of the effective Chalk aquifer of north Norfolk, UK. *Journal of Hydrology* 180: 79-107

Hobbs, P. R. 2007. Conservation agriculture: what is it and why is it important for future sustainable food production? *Journal of Agricultural Science-Cambridge-* 145: 127-137

Hobbs, P. R., Sayre, K. & Gupta, R. 2008. The role of conservation agriculture in sustainable agriculture. *Philosophical Transactions of the Royal Society B: Biological Sciences* 363: 543-555

Höll, B. S., Jungkunst, H. F., Fiedler, S. & Stahr, K. 2005. Indirect nitrous oxide emission from a nitrogen saturated spruce forest and general accuracy of the IPCC methodology. *Atmospheric Environment* 39: 5959-5970

Holland, J. 2004. The environmental consequences of adopting conservation tillage in Europe: reviewing the evidence. *Agriculture, Ecosystems & Environment* 103: 1-25

Horneck, D. A., Sullivan, D. M., Owen, J. S. & Hart, J. M. 2011. Soil test interpretation guide. EC 1478. Oregon State University, Extension Service.

Hulugalle, N. & Entwistle, P. 1997. Soil properties, nutrient uptake and crop growth in an irrigated Vertisol after nine years of minimum tillage. *Soil and Tillage Research* 42: 15-32

Hutchins, M., Fezzi, C., Bateman, I., Posen, P. & Deflandre-Vlandas, A. 2009. Cost-effective mitigation of diffuse pollution: Setting criteria for river basin management at multiple locations. *Environmental Management* 44: 256-267

IPCC. 2006. N_2O emissions from managed soils, and CO_2 emissions from lime and urea application. In 2006 IPCC Guidelines for National Greenhouse Gas Inventories.

Ismail, I., Blevins, R. & Frye, W. 1994. Long-term no-tillage effects on soil properties and continuous corn yields. *Soil Science Society of America Journal* 58: 193-198

Jabro, J., Stevens, W., Iversen, W. & Evans, R. 2011. Bulk density, water content and hydraulic properties of a sandy loam soil following conventional or strip tillage. *Applied Engineering in Agriclture* 27: 765-768

Jabro, J. D., Stevens, W. B., Evans, R. G. & Iversen, W. M. 2009. Tillage effects on physical properties in two soils of the Northern Great Plains. *Applied Engineering in Agriculture* 25: 377

Jacinthe, P.-A., Dick, W. & Brown, L. 1999. Bioremediation of nitrate-contaminated shallow soils using water table management techniques: Nitrate removal efficiency. *Transactions of the ASAE* 42: 1251-1259

Jacinthe, P.-A. & Dick, W. A. 1997. Soil management and nitrous oxide emissions from cultivated fields in southern Ohio. *Soil and Tillage Research* 41: 221-235

Jackson, L., Wyland, L. & Stivers, L. 1993. Winter cover crops to minimize nitrate losses in intensive lettuce production. *The Journal of Agricultural Science* 121: 55-62

Jahangir, M. M., Johnston, P., Barrett, M., Khalil, M., Groffman, P., Boeckx, P., Fenton, O., Murphy, J. & Richards, K. G. 2013. Denitrification and indirect N_2O emissions in groundwater: hydrologic and biogeochemical influences. *Journal Of Contaminant Hydrology* 152: 70-81

Jarecki, M. K., Parkin, T. B., Chan, A. S., Kaspar, T. C., Moorman, T. B., Singer, J. W., Kerr, B. J., Hatfield, J. L. & Jones, R. 2009. Cover crop effects on nitrous oxide emission from a manure-treated Mollisol. *Agriculture, Ecosystems & Environment* 134: 29-35

Jaynes, D., Colvin, T., Karlen, D., Cambardella, C. & Meek, D. 2001. Nitrate loss in subsurface drainage as affected by nitrogen fertilizer rate. *Journal of Environmental Quality* 30: 1305-1314

Jones, C., Basch, G., Baylis, A., Bazzoni, D., Biggs, J., Bradbury, R., Chaney, K., Deeks, L., Field, R., Gomez, J., Jones, R., Jordan, V., Lane, M., Leake, A., Livermore, M., Owens, P., Ritz, K., Sturny, W. & Thomas, F. 2006. Conservation agriculture in Europe: an approach to sustainable crop production by protecting soil and water? , Conservation agriculture in Europe: an approach to sustainable crop production by protecting soil and water?, Jealott's Hill, Bracknell, UK,

Joshi, J., Moncrief, J., Swan, J. & Malzer, G. 1994. Long-term conservation tillage and liquid dairy manure effects on corn. II. Nitrate concentration in soil water. *Soil and Tillage Research* 31: 225-233

Justes, E., Mary, B. & Nicolardot, B. 1999. Comparing the effectiveness of radish cover crop, oilseed rape volunteers and oilseed rape residues incorporation for reducing nitrate leaching. *Nutrient Cycling in Agroecosystems* 55: 207-220

Kallenbach, C. M., Rolston, D. E. & Horwath, W. R. 2010. Cover cropping affects soil N2O and CO2 emissions differently depending on type of irrigation. *Agriculture, Ecosystems & Environment* 137: 251-260

Kanwar, R., Stoltenberg, D., Pfeiffer, R., Karlen, D., Colvin, T. & Simpkins, W. 1993. Transport of nitrate and pesticides to shallow groundwater system as affected by tillage and crop rotation practices. *Proceedings of the National Conference on Agricultural Research to Protect Water Quality*: pp. 270–273

Karlen, D., Mausbach, M., Doran, J., Cline, R., Harris, R. & Schuman, G. 1997. Soil quality: a concept, definition, and framework for evaluation. *Soil Science Society of America Journal* 61: 4-10

Karlen, D., Wollenhaupt, N., Erbach, D., Berry, E., Swan, J., Eash, N. & Jordahl, J. 1994. Long-term tillage effects on soil quality. *Soil and Tillage Research* 32: 313-327

Kaspar, T., Jaynes, D., Parkin, T., Moorman, T. & Singer, J. 2012. Effectiveness of oat and rye cover crops in reducing nitrate losses in drainage water. *Agricultural Water Management* 110: 25-33

Kassam, A., Friedrich, T., Shaxson, F. & Pretty, J. 2009. The spread of conservation agriculture: justification, sustainability and uptake. *International Journal of Agricultural Sustainability* 7: 292-320

Kennedy, A. C. & Schillinger, W. F. 2006. Soil quality and water intake in traditional-till vs. no-till paired farms in Washington's Palouse region. *Soil Science Society of America Journal* 70: 940-949

Kim, D., Isenhart, T., Parkin, T., Schultz, R. & Loynachan, T. 2009. Nitrate and dissolved nitrous oxide in groundwater within cropped fields and riparian buffers. *Biogeosciences Discussions* 6: 651-685

Knobeloch, L., Salna, B., Hogan, A., Postle, J. & Anderson, H. 2000. Blue babies and nitrate-contaminated well water. *Environmental Health Perspectives* 108: 675

Knott, C. 1996. The effect of cover crops on the establishment and yield of vining peas and on nitrate leaching. *The Journal of Agricultural Science* 126: 471-479

Komatsuzaki, M. & Ohta, H. 2007. Soil management practices for sustainable agro-ecosystems. *Sustainability Science* 2: 103-120

Kristensen, H. L. & Thorup-Kristensen, K. 2004. Root growth and nitrate uptake of three different catch crops in deep soil layers. *Soil Science Society of America Journal* 68: 529-537

Kroeze, C. 1994. Nitrous oxide and global warming. *Science of the Total Environment* 143: 193-209

Lacas, J.-G., Voltz, M., Gouy, V., Carluer, N. & Gril, J.-J. 2005. Using grassed strips to limit pesticide transfer to surface water: a review. *Agronomy For Sustainable Development* 25: 253-266

Lal, R. 1997. Residue management, conservation tillage and soil restoration for mitigating greenhouse effect by CO_2-enrichment. *Soil and Tillage Research* 43: 81-107

Lal, R., Kimble, J. M., Follett, R. F. & Cole, C. V. 1998. The potential of US cropland to sequester carbon and mitigate the greenhouse effect. CRC Press, Ann Arbor Press, Chelsea, Michigan, USA.

Lal, R., Logan, T. & Fausey, N. 1990. Long-term tillage effects on a Mollic Ochraqualf in North-West Ohio. III. Soil nutrient profile. *Soil and Tillage Research* 15: 371-382

Lal, S. & Patra, P. K. 1998. Variabilities in the fluxes and annual emissions of nitrous oxide from the Arabian Sea. *Global Biogeochemical Cycles* 12: 321-327

Lee, J., Six, J., King, A. P., Van Kessel, C. & Rolston, D. E. 2006. Tillage and field scale controls on greenhouse gas emissions. *Journal of Environmental Quality* 35: 714-725

Lemon, E. & Lemon, D. 1981. Nitrous oxide in freshwaters of the Great Lakes Basin. *Limnology and Oceanography* 26: 867-879

Lewis, M. 2011. Borehole drilling and sampling in the Wensum Demonstration Test Catchment. *British Geological Survey Commissioned Report, CR/11/162. British Geological Survey: London* 38

Li, C., Frolking, S. & Butterbach-Bahl, K. 2005. Carbon sequestration in arable soils is likely to increase nitrous oxide emissions, offsetting reductions in climate radiative forcing. *Climatic Change* 72: 321-338

Li, X., Tang, C., Han, Z., Jingqiu, P., Yingjie, C. & Chipeng, Z. 2013. Spatial and seasonal variation of dissolved nitrous oxide in wetland groundwater. *Environment and Pollution* 3: p21

Licht, M. A. & Al-Kaisi, M. 2005. Strip-tillage effect on seedbed soil temperature and other soil physical properties. *Soil and Tillage Research* 80: 233-249

Lipiec, J., Kuś, J., Słowińska-Jurkiewicz, A. & Nosalewicz, A. 2006. Soil porosity and water infiltration as influenced by tillage methods. *Soil and Tillage Research* 89: 210-220

Liu, S., Yang, J., Zhang, X., Drury, C., Reynolds, W. & Hoogenboom, G. 2013. Modelling crop yield, soil water content and soil temperature for a soybean–maize rotation under conventional and conservation tillage systems in Northeast China. *Agricultural Water Management* 123: 32-44

Loveland, P. & Webb, J. 2003. Is there a critical level of organic matter in the agricultural soils of temperate regions: a review. *Soil and Tillage Research* 70: 1-18

Ma, Z., Merkus, H. G., de Smet, J. G., Heffels, C. & Scarlett, B. 2000. New developments in particle characterization by laser diffraction: size and shape. *Powder Technology* 111: 66-78

Macdonald, A., Poulton, P., Howe, M., Goulding, K. & Powlson, D. 2005. The use of cover crops in cereal-based cropping systems to control nitrate leaching in SE England. *Plant and Soil* 273: 355-373

MacKenzie, A., Fan, M. & Cadrin, F. 1997. Nitrous oxide emission as affected by tillage, corn-soybean-alfalfa rotations and nitrogen fertilization. *Canadian Journal of Soil Science* 77: 145-152

Maraseni, T. & Cockfield, G. 2011. Does the adoption of zero tillage reduce greenhouse gas emissions? An assessment for the grains industry in Australia. *Agricultural Systems* 104: 451-458

Martikainen, P. J. & Boer, D. W. 1993. Nitrous oxide production and nitrification in acidic soil from a Dutch coniferous forest. *Soil Biology and Biochemistry* 25: 343-347

Masarik, K. C., Norman, J. M. & Brye, K. R. 2014. Long-Term Drainage and Nitrate Leaching below Well-Drained Continuous Corn Agroecosystems and a Prairie. *Journal of Environmental Protection* 2014

Matowo, P. R., Pierzynski, G. M., Whitney, D. & Lamond, R. E. 1999. Soil chemical properties as influenced by tillage and nitrogen source, placement, and rates after 10 years of continuous sorghum. *Soil and Tillage Research* 50: 11-19

McGarry, D., Bridge, B. & Radford, B. 2000. Contrasting soil physical properties after zero and traditional tillage of an alluvial soil in the semi-arid subtropics. *Soil and Tillage Research* 53: 105-115

McMahon, P. B., Bruce, B. W., Becker, M. F., Pope, L. M. & Dennehy, K. 2000. Occurrence of nitrous oxide in the central High Plains aquifer, 1999. *Environmental Science & Technology* 34: 4873-4877

Meek, B., Carter, D., Westermann, D., Wright, J. & Peckenpaugh, R. 1995. Nitrate leaching under furrow irrigation as affected by crop sequence and tillage. *Soil Science Society of America Journal* 59: 204-210

Mengis, M., Gächter, R. & Wehrli, B. 1997. Sources and sinks of nitrous oxide (N_2O) in deep lakes. *Biogeochemistry* 38: 281-301

Meteorological Office. 2014. Climate Averages: Reepham 1981– 2010. Meteorological Office. 2014: http://www.metoffice.gov.uk/public/weather/climate/u12u1m41e

Mijangos, I., Pérez, R., Albizu, I. & Garbisu, C. 2006. Effects of fertilization and tillage on soil biological parameters. *Enzyme and Microbial Technology* 40: 100-106

Miller, J., Lamond, B., Sweetland, N. & Larney, F. 1999. Preferential leaching in large undisturbed soil blocks from conventional tillage and no-till fields in Southern Alberta. *Water Quality Research Journal of Canada* 34: 249-266

Miller, J. & Miller, J. 1993. Errors in instrumental analysis; regression and correlation. *Statistics for Analytical Chemistry* 2: 101-139

Minamikawa, K., Eguchi, S., Nishimura, S., Ihara, H., Maeda, M., Yagi, K. & Komada, M. 2013. Groundwater-induced emissions of nitrous oxide through the soil surface and from subsurface drainage in an Andosol upland field: A monolith lysimeter study. *Soil Science and Plant Nutrition* 59: 87-95

Minamikawa, K., Hayakawa, A., Nishimura, S., Akiyama, H. & Yagi, K. 2011. Comparison of indirect nitrous oxide emission through lysimeter drainage between an Andosol upland field and a Fluvisol paddy field. *Soil Science and Plant Nutrition* 57: 843-854

Minamikawa, K., Nishimura, S., Sawamoto, T., Nakajima, Y. & Yagi, K. 2010. Annual emissions of dissolved CO_2, CH_4, and N_2O in the subsurface drainage from three cropping systems. *Global Change Biology* 16: 796-809

Moreno, F., Murillo, J., Pelegrín, F. & Girón, I. 2006. Long-term impact of conservation tillage on stratification ratio of soil organic carbon and loss of total and active $CaCO_3$. *Soil and Tillage Research* 85: 86-93

Moroizumi, T. & Horino, H. 2002. The effects of tillage on soil temperature and soil water. *Soil Science* 167: 548-559

Morris, N. 2009. The adoption of conservation tillage in the United Kingdom. *Journal of the Royal Agricultural Society of England* 170: 64-70

Morris, N., Miller, P., Orson, J. & Froud-Williams, R. 2010. The adoption of non-inversion tillage systems in the United Kingdom and the agronomic impact on soil, crops and the environment—a review. *Soil and Tillage Research* 108: 1-15

Mosier, A. & Kroeze, C. 1998. A new approach to estimate emissions of nitrous oxide from agriculture and its implications to the global N_2O budget. *IGBP newsletter* 34: 8-13

Mosier, A., Kroeze, C., Nevison, C., Oenema, O., Seitzinger, S. & Van Cleemput, O. 1998. Closing the global N_2O budget: nitrous oxide emissions through the agricultural nitrogen cycle. *Nutrient Cycling in Agroecosystems* 52: 225-248

Mosjidis, J. A. & Owsley, C. M. 2002. Legume cover crop development by NRCS and Auburn University. *Making Conservation Tillage Conventional: Building a Future on 25 Years of Research*: 305

Mühlherr, I. H. 1997. Nitrous oxide in British Limestone aquifer PhD Thesis, University of East Anglia

Mühlherr, I. H. & Hiscock, K. M. 1997. A preliminary assessment of nitrous oxide in chalk groundwater in Cambridgeshire, UK. *Applied Geochemistry* 12: 797-802

Mühlherr, I. H. & Hiscock, K. M. 1998. Nitrous oxide production and consumption in British limestone aquifers. *Journal of Hydrology* 211: 126-139

Munkholm, L. J. & Hansen, E. M. 2012. Catch crop biomass production, nitrogen uptake and root development under different tillage systems. *Soil Use and Management* 28: 517-529

Mutegi, J., Petersen, B. M. & Munkholm, L. J. 2013. Carbon turnover and sequestration potential of fodder radish cover crop. *Soil Use and Management* 29: 191-198

National Atmospheric Emissions Inventory. 2015. http://naei.defra.gov.uk/

NATMAP. 2009. 20016: http://www.landis.org.uk/data/natmap.cfm

Nevison, C. 2000. Review of the IPCC methodology for estimating nitrous oxide emissions associated with agricultural leaching and runoff. *Chemosphere - Global Change Science* 2: 493-500

Nevison, C., Butler, J. H. & Elkins, J. 2003. Global distribution of N_2O and the ΔN_2O□AOU yield in the subsurface ocean. *Global Biogeochemical Cycles* 17

Nevison, C., Keeling, R., Weiss, R., Popp, B., Jin, X., Fraser, P., Porter, L. & Hess, P. 2005. Southern Ocean ventilation inferred from seasonal cycles of atmospheric N_2O and O_2/N_2 at Cape Grim, Tasmania. *Tellus B* 57: 218-229

Nevison, C. D., Weiss, R. F. & Erickson, D. J. 1995. Global oceanic emissions of nitrous oxide. *Journal of Geophysical Research: Oceans* 100: 15809-15820

Newell Price, J., Harris, D., Taylor, M., Williams, J., Anthony, S., Duethmann, D., Gooday, R., Lord, E., Chambers, B. & Chadwick, D. 2011. An inventory of mitigation methods and guide to their effects on diffuse water pollution, greenhouse gas emissions and ammonia emissions from agriculture. Report prepared as part of Defra Project WQ0106, ADAS and Rothamsted Research North Wyke.

Ngouajio, M. & Mutch, D. R. 2004. Oilseed Radish: A New Cover Crop for Michigan. *Extension Bulletin* 2907

NRM. 2013. Standard soil analytical methods. Technical information.

Obi, M. 1999. The physical and chemical responses of a degraded sandy clay loam soil to cover crops in southern Nigeria. *Plant and Soil* 211: 165-172

Office for National Statistics. 2012. Mid-2010 broad age population estimates for urban areas in E&W. 2014

Omonode, R. A., Smith, D. R., Gál, A. & Vyn, T. J. 2011. Soil nitrous oxide emissions in corn following three decades of tillage and rotation treatments. *Soil Science Society of America Journal* 75: 152-163

Oudot, C., Andrie, C. & Montel, Y. 1990. Nitrous oxide production in the tropical Atlantic Ocean. *Deep Sea Research Part A. Oceanographic Research Papers* 37: 183-202

Outram, F., Lloyd, C., Jonczyk, J., Benskin, C. M., Grant, F., Perks, M., Deasy, C., Burke, S., Collins, A. & Freer, J. 2014. High-frequency monitoring of nitrogen and phosphorus response in three rural catchments to the end of the 2011–2012 drought in England. *Hydrology and Earth System Sciences* 18: 3429-3448

Outram, F. N., Cooper, R. J., Sünnenberg, G., Hiscock, K. M. & Lovett, A. A. 2016. Antecedent conditions, hydrological connectivity and anthropogenic inputs: Factors affecting nitrate and phosphorus transfers to agricultural headwater streams. *Science of The Total Environment* 545: 184-199

Outram, F. N. & Hiscock, K. M. 2012. Indirect nitrous oxide emissions from surface water bodies in a lowland arable catchment: a significant contribution to agricultural greenhouse gas budgets? *Environmental Science & Technology* 46: 8156-8163

Panek, J., Matson, P., Ortiz-Monasterio, I. & Brooks, P. 2000. Distinguishing nitrification and denitrification sources of N_2O in a Mexican wheat system using ^{15}N. *Ecological Applications* 10: 506-514

Parkin, T., Kaspar, T. & Singer, J. 2006. Cover crop effects on the fate of N following soil application of swine manure. *Plant and Soil* 289: 141-152

Pennock, D., Yates, T. & Braidek, J. 2007. Soil Sampling Designs. in P. G.T & C. M.R (Eds.), Soil Sampling and Methods of Analysis, Second Edition, Berlin, CRC Press, pp: 1–15 (Chapter 11)

Petersen, S. O., Mutegi, J. K., Hansen, E. M. & Munkholm, L. J. 2011. Tillage effects on N_2O emissions as influenced by a winter cover crop. *Soil Biology and Biochemistry* 43: 1509-1517

Power, J. F. & Prasad, R. 1997. Soil fertility management for sustainable agriculture. CRC Press, Boca Raton, 356 pp

Premrov, A., Coxon, C. E., Hackett, R., Kirwan, L. & Richards, K. G. 2014. Effects of over-winter green cover on soil solution nitrate concentrations beneath tillage land. *Science of the Total Environment* 470: 967-974

Prisu, J. C., Downes, M. T. & McKay, C. P. 1996. Extreme supersaturation of nitrous oxide in a poorly ventilated Antarctic lake. *Limnology and Oceanography* 41: 1544-1551

Ramos, C., Agut, A. & Lidon, A. 2002. Nitrate leaching in important crops of the Valencian Community region (Spain). *Environmental Pollution* 118: 215-223

Randall, G. & Iragavarapu, T. 1995. Impact of long-term tillage systems for continuous corn on nitrate leaching to tile drainage. *Journal of Environmental Quality* 24: 360-366

Randall, G. W. & Mulla, D. J. 2001. Nitrate nitrogen in surface waters as influenced by climatic conditions and agricultural practices. *Journal of Environmental Quality* 30: 337-344

Rao, G., Rao, V. & Sarma, V. 2013. Distribution and air–sea exchange of nitrous oxide in the coastal Bay of Bengal during peak discharge period (southwest monsoon). *Marine Chemistry* 155: 1-9

Rawlins, B. 2011. A pilot study to assess soil spectroscopic methods for mapping key topsoil properties in the Blackwater sub-catchments (Wensum DTC). London. British Geological Survey Internal Report, OR/11/053.

Reay, D., Edwards, A. & Smith, K. 2004a. Determinants of nitrous oxide emission from agricultural drainage waters. *Water, Air & Soil Pollution: Focus* 4: 107-115

Reay, D., Smith, K. & Edwards, A. 2004b. Nitrous Oxide in Agricultural Drainage Waters Following Field Fertilisation. *Water, Air and Soil Pollution: Focus* 4: 437-451

Reay, D. S., Edwards, A. C. & Smith, K. A. 2009. Importance of indirect nitrous oxide emissions at the field, farm and catchment scale. *Agriculture, Ecosystems & Environment* 133: 163-169

Reay, D. S., Smith, K. A. & Edwards, A. C. 2003. Nitrous oxide emission from agricultural drainage waters. *Global Change Biology* 9: 195-203

Reay, D. S., Smith, K. A., Edwards, A. C., Hiscock, K. M., Dong, L. F. & Nedwell, D. B. 2005. Indirect nitrous oxide emissions: revised emission factors. *Environmental Sciences* 2: 153-158

Reeves, D. 1994. Cover crops and rotations. in H. J.L & S. B.A (Eds.), Advances in Soil Science: Crops Residue Management, Boca Raton, FL, USA,, Lewis Publishers, CRC Press, pp: 125-172

Reynolds, W., Elrick, D., Youngs, E., Amoozegar, A., Booltink, H. & Bouma, J. 2002. Saturated and field-saturated water flow parameters. *Methods of Soil Analysis* 4: 797-801

Rice, C. W. 2002. Organic matter and nutrient dynamics. in Lal R (Ed.), Encyclopedia of soil science, New York, Dekker, pp: 1180-1183

Rochette, P., Angers, D. A., Chantigny, M. H. & Bertrand, N. 2008a. Nitrous oxide emissions respond differently to no-till in a loam and a heavy clay soil. *Soil Science Society of America Journal* 72: 1363-1369

Rochette, P., Worth, D. E., Huffman, E. C., Brierley, J. A., McConkey, B. G., Yang, J., Hutchinson, J. J., Desjardins, R. L., Lemke, R. & Gameda, S. 2008b. Estimation of N_2O emissions from agricultural soils in Canada. II. 1990-2005 inventory. *Canadian Journal of Soil Science* 88: 655-669

Ronen, D., Magaritz, M. & Almon, E. 1988a. Contaminated aquifers are a forgotten component of the global N2O budget. *Nature* 335: 57-59

Ronen, D., Magaritz, M. & Almon, E. 1988b. Contaminated aquifers are a forgotten component of the global N_2O budget.

Rosamond, M. S., Thuss, S. J. & Schiff, S. L. 2012. Dependence of riverine nitrous oxide emissions on dissolved oxygen levels. *Nature Geoscience* 5: 715-718

Rossi, N., Ciavatta, C. & Antisari, L. V. 1991. Seasonal pattern of nitrate losses from cultivated soil with subsurface drainage. *Water, Air, and Soil Pollution* 60: 1-10

Sander, R. 1999. Compilation of Henry's law constants for inorganic and organic species of potential importance in environmental chemistry. Report of the Max Planck Institute of Chemistry, Mainz, Germany

Sanz-Cobena, A., García-Marco, S., Quemada, M., Gabriel, J., Almendros, P. & Vallejo, A. 2014. Do cover crops enhance N_2O, CO_2 or CH_4 emissions from soil in Mediterranean arable systems? *Science Of The Total Environment* 466: 164-174

Sawamoto, T., Kusa, K., Hu, R. & Hatano, R. 2003. Dissolved N_2O, CH_4, and CO_2 emissions from subsurface-drainage in a structured clay soil cultivated with onion in central Hokkaido, Japan. *Soil Science and Plant Nutrition* 49: 31-38

Sawamoto, T., Nakajima, Y., Kasuya, M., Tsuruta, H. & Yagi, K. 2005. Evaluation of emission factors for indirect N_2O emission due to nitrogen leaching in agro□ ecosystems. *Geophysical Research Letters* 32: L03403

Sawamoto, T., Nishida, T. & Matsunaka, T. 2010. Dissolved N_2O emission through drainage water from timothy (Phleum pretense L.) sward in a lysimeter experiment. *Environmental Science (Japan)*

Scheehle, E., Godwin, D. & Ottinger, D. 2006. Global Anthropogenic Non-CO_2 Greenhouse Gas Emissions: 1990–2020. Washington, DC. U.S. Environmental Protection Agency.

Scholefield, D., Tyson, K., Garwood, E., Armstrong, A., Hawkins, J. & Stone, A. 1993. Nitrate leaching from grazed grassland lysimeters: effects of fertilizer input, field drainage, age of sward and patterns of weather. *Journal of Soil Science* 44: 601-613

Schuller, P., Walling, D. E., Sepúlveda, A., Castillo, A. & Pino, I. 2007. Changes in soil erosion associated with the shift from conventional tillage to a no-tillage system, documented using 137 Cs measurements. *Soil and Tillage Research* 94: 183-192

Sear, D., Newson, J., Old, J. & Hill, C. 2006. Geomorphological Appraisal of the River Wensum Special Area of Conservation. Sheffield. Natural England.

Seitzinger, S. P. & Kroeze, C. 1998. Global distribution of nitrous oxide production and N inputs in freshwater and coastal marine ecosystems. *Global Biogeochemical Cycles* 12: 93-113

Silva, R., Cameron, K., Di, H., Smith, N. & Buchan, G. 2000. Effect of macropore flow on the transport of surface-applied cow urine through a soil profile. *Soil Research* 38: 13-24

Singer, J., Malone, R., Jaynes, D. & Ma, L. 2011. Cover crop effects on nitrogen load in tile drainage from Walnut Creek Iowa using root zone water quality (RZWQ) model. *Agricultural Water Management* 98: 1622-1628

Six, J., Feller, C., Denef, K., Ogle, S., Sa, J. C. D. M. & Albrecht, A. 2002. Soil organic matter, biota and aggregation in temperate and tropical soils-Effects of no-tillage. *Agronomie* 22: 755-775

Skiba, U., McTaggart, I. P., Smith, K. A., Hargreaves, K. J. & Fowler, D. 1996. Estimates of nitrous oxide emissions from soil in the UK. *Energy Conversion and Management* 37: 1303-1308

Skiba, U. & Smith, K. 2000. The control of nitrous oxide emissions from agricultural and natural soils. *Chemosphere-Global Change Science* 2: 379-386

Sławiński, C., Cymerman, J., Witkowska-Walczak, B. & Lamorski, K. 2012. Impact of diverse tillage on soil moisture dynamics. *International Agrophysics* 26: 301-309

Smith, K., McTaggart, I., Dobbie, K. & Conen, F. 1998. Emissions of N2O from Scottish agricultural soils, as a function of fertilizer N. *Nutrient Cycling in Agroecosystems* 52: 123-130

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O’Mara, F., Rice, C., Scholes, B. & Sirotenko, O. 2007. Agriculture. Climate Change 2007: mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK, and New York.

Sprague, M. A. & Triplett, G. B. 1986. No-tillage and surface-tillage agriculture. The tillage revolution. John Wiley & Sons, New York

Stagnari, F., Ramazzotti, S. & Pisante, M. 2010. Conservation agriculture: a different approach for crop production through sustainable soil and water management: a review. in L. E (Ed.), Organic Farming, Pest Control and Remediation of Soil Pollutants, New York, Springer Science + Business Media B.V, pp: 55-83

Staver, K. W. & Brinsfield, R. B. 1998. Using cereal grain winter cover crops to reduce groundwater nitrate contamination in the mid-Atlantic coastal plain. *Journal of Soil and Water Conservation* 53: 230-240

Stevens, C. J. & Quinton, J. N. 2009. Diffuse pollution swapping in arable agricultural systems. *Critical Reviews in Environmental Science and Technology* 39: 478-520

Stivers-Young, L. 1998. Growth, nitrogen accumulation, and weed suppression by fall cover crops following early harvest of vegetables. *HortScience* 33: 60-63

Strudley, M. W., Green, T. R. & Ascough, J. C. 2008. Tillage effects on soil hydraulic properties in space and time: State of the science. *Soil and Tillage Research* 99: 4-48

Syakila, A. & Kroeze, C. 2011. The global nitrous oxide budget revisited. *Greenhouse Gas Measurement and Management* 1: 17-26

Szulc, W., Rutkowska, B. & Labetowicz, J. 2004. The content of total sulphur and sulphate sulphur in soil profile in conditions of different soil cultivation systems. *Annales Universitatis Mariae Curie-Sklodowska. Sectio E Agricultura*

Temesgen, M., Savenije, H., Rockström, J. & Hoogmoed, W. 2012. Assessment of strip tillage systems for maize production in semi-arid Ethiopia: effects on grain yield, water balance and water productivity. *Physics and Chemistry of the Earth* 47: 156-165

Thierfelder, C., Amézquita, E. & Stahr, K. 2005. Effects of intensifying organic manuring and tillage practices on penetration resistance and infiltration rate. *Soil and Tillage Research* 82: 211-226

Topaloğlu, F. 1999. Comparing tillage techniques by using a new infiltration method. *Tropical Journal of Agriculture and Forestry* 23: 609-614

Ueda, S., Ogura, N. & Wada, E. 1991. Nitrogen stable isotope ratio of groundwater N_2O. *Geophysical Research Letters* 18: 1449-1452

Ueda, S., Ogura, N. & Yoshinari, T. 1993. Accumulation of nitrous oxide in aerobic groundwaters. *Water Research* 27: 1787-1792

Van der Hoek, K., Van Schijndel, M., Kuikman, P. & Van Schijndel, M. 2007. Direct and indirect nitrous oxide emissions from agricultural soils, 1990-2003. Background document on the calculation method for the Dutch National Inventory Report.

Venterea, R. T., Burger, M. & Spokas, K. A. 2005. Nitrogen oxide and methane emissions under varying tillage and fertilizer management. *Journal of Environmental Quality* 34: 1467-1477

Verhulst, N., Govaerts, B., Verachtert, E., Castellanos-Navarrete, A., Mezzalama, M., Wall, P., Deckers, J. & Sayre, K. 2010. Conservation agriculture, improving soil quality for sustainable production systems? in L. R & S. B. A (Eds.), Advances in soil science: food security and soil quality, Boca Raton, FL, USA, CRC Press, pp: 137-208

Vilain, G., Garnier, J., Tallec, G. & Tournebize, J. 2011. Indirect N_2O emissions from shallow groundwater in an agricultural catchment (Seine Basin, France). *Biogeochemistry* 111: 253-271

Vogeler, I., Rogasik, J., Funder, U., Panten, K. & Schnug, E. 2009. Effect of tillage systems and P-fertilization on soil physical and chemical properties, crop yield and nutrient uptake. *Soil and Tillage Research* 103: 137-143

Ward, A. D. & Trimble, S. W. 2003. Environmental hydrology, second edition. Lewis Publishers, New York

Wedd, M. W. 2003. Determination of particle size distributions using laser diffraction. *Educational Resources for Particle Technology* 4: 1-4

Weil, R. R., White, C. & Lawley, Y. 2006. Forage radish: new multi-purpose cover crop for the Mid-Atlantic. *University of Maryland Cooperative Extension Fact Sheet 824, College Park.*

Well, R. & Butterbach-Bahl, K. 2010. Indirect emissions of nitrous oxide from nitrogen deposition and leaching of agricultural nitrogen. in S. K (Ed.), Nitrous Oxide and Climate Change, Sterling, UK, Earthscan Publications Ltd, p. 162

Well, R., Flessa, H., Jaradat, F., Toyoda, S. & Yoshida, N. 2005a. Measurement of isotopomer signatures of N_2O in groundwater. *Journal of Geophysical Research: Biogeosciences (2005–2012)* 110

Well, R., Weymann, D. & Flessa, H. 2005b. Recent research progress on the significance of aquatic systems for indirect agricultural N_2O emissions. *Environmental Sciences* 2: 143-151

Weller, D. E., Correll, D. & Jordan, T. E. 1994. Denitrification in riparian forests receiving agricultural discharges. in Mitsch W.J. (Ed.), Global Wetlands; old world and new, Amsterdam, Elsevier, pp: 117-131

Wensum Alliance. 2015. River Wensum Demonstration Test Catchment Project. http://www.wensumalliance.org.uk/

Weslien, P., Kasimir Klemedtsson, Å., Börjesson, G. & Klemedtsson, L. 2009. Strong pH influence on N_2O and CH_4 fluxes from forested organic soils. *European Journal of Soil Science* 60: 311-320

West, T. O. & Post, W. M. 2002. Soil organic carbon sequestration rates by tillage and crop rotation. *Soil Science Society of America Journal* 66: 1930-1946

Weymann, D., Well, R., Flessa, H., Heide, C., Deurer, M., Meyer, K., Konrad, C. & Walther, W. 2008. Groundwater N_2O emission factors of nitrate-contaminated aquifers as derived from denitrification progress and N_2O accumulation. *Biogeosciences* 5: 1215-1226

White, C. M. & Weil, R. R. 2011. Forage radish cover crops increase soil test phosphorus surrounding radish taproot holes. *Soil Science Society of America Journal* 75: 121-130

Williams, E., Hutchinson, G. & Fehsenfeld, F. 1992. NO_x and N_2O emissions from soil. *Global Biogeochemical Cycles* 6: 351-388

Williams, S. M. & Weil, R. R. 2004. Crop cover root channels may alleviate soil compaction effects on soybean crop. *Soil Science Society of America Journal* 68: 1403-1409

Włodarczyk, T., Stępniewski, W. & Brzezińska, M. 2005. Nitrous oxide production and consumption in Calcaric Regosols as related to soil redox and texture. *International Agrophysics* 19: 263-271

Yu, Z., Deng, H., Wang, D., Ye, M., Tan, Y., Li, Y., Chen, Z. & Xu, S. 2013. Nitrous oxide emissions in the Shanghai river network: implications for the effects of urban sewage and IPCC methodology. *Global Change Biology* 19: 2999-3010

Zibilske, L., Bradford, J. & Smart, J. 2002. Conservation tillage induced changes in organic carbon, total nitrogen and available phosphorus in a semi-arid alkaline subtropical soil. *Soil and Tillage Research* 66: 153-163

APPENDIX

APPENDIX A	**Soil sampling location and soil fractions**
APPENDIX B	**Soil chemical and physical data**
APPENDIX C	**Cover crop data**
APPENDIX D	**Drain data**
APPENDIX E	**Porous pot data**
APPENDIX F	**Riverine data**
APPENDIX G	**Dissolved nitrous oxide data**
APPENDIX H	**Farm data**

APPENDIX A Soil sampling location and soil fractions data

Figure A1 Map of locations of soil samples

Table A1 coordination and percentage of soil fractions in the three soil depths

Figure A2 Electrical conductivity map for the top and bottom soil layers with soil texture

Figure A1 Map of locations of soil samples

Table A1 Percentage of soil fractions in (0-30 cm), (30-60 cm), and (60-90 cm) soil depths

Sample name	Latitude (N)	Longitude (E)	Sand 0-30	Silt 0-30	Clay 0-30	Sand 30-60	Silt 30-60	Clay 30-60	Sand 60-90	Silt 60-90	Clay 60-90
D1	52.7844873	1.119858	46	32	22	46	31	23	53	20	27
D2	52.78364387	1.11953044	48	34	18	47	36	17	35	23	42
D3	52.78209388	1.11658463	45	31	24	43	25	32	35	23	42
D4	52.78470428	1.11743469	53	34	13	55	30	15	53	27	20
D5	52.78404155	1.11481005	40	33	27	39	28	33	24	27	49
D6	52.78359362	1.1151999	47	31	22	44	31	25	49	23	28
D7	52.78297931	1.11600137	46	29	25	46	31	23	52	23	25
D8	52.78219711	1.1170388	43	31	26	44	29	27	45	25	30
D9	52.78313781	1.1175858	38	37	25	34	32	34	26	26	48
D10	52.78392873	1.11798344	40	38	22	39	38	23	36	32	32
D11	52.78461202	1.11819182	48	35	17	44	39	17	55	30	15
D12	52.78377257	1.1191906	48	32	20	48	32	20	48	26	26
D13	52.782774	1.1197886	45	37	18	46	30	24	48	30	22
D14	52.78328994	1.12057052	48	31	21	41	32	27	54	22	24
D15	52.78408153	1.12024203	62	27	11	56	29	15	55	33	12
D16	52.78474613	1.11986107	65	30	5	3	29	68	79	19	2
MH1	52.79209752	1.11331305	52	28	20	55	25	20	43	21	36
MH2	52.79059757	1.11420055	52	28	20	50	30	20	45	22	33
MH3	52.79009499	1.11671488	56	28	16	59	26	15	41	22	37
MH4	52.79183216	1.1149977	50	29	21	53	29	18	51	24	25
MH5	52.79283592	1.11280477	49	33	18	45	22	33	50	20	30
MH6	52.79214651	1.11247159	49	29	22	50	22	28	51	23	26
MH7	52.79134068	1.11201033	50	30	20	55	24	21	75	12	13
MH8	52.79043895	1.11149811	46	32	22	44	27	29	35	18	47
MH9	52.79082385	1.11298422	47	30	23	54	23	23	49	20	31
MH10	52.79111009	1.11431401	46	28	26	46	22	32	39	20	41
MH11	52.79172398	1.11501449	50	29	21	55	25	20	39	25	36
MH12	52.79076174	1.11509286	48	31	21	57	21	22	22	28	50
MH13	52.79006378	1.11425542	46	31	23	37	31	32	39	18	43
MH14	52.78992497	1.11521406	46	30	24	44	28	28	40	27	33
MH15	52.7898425	1.1160494	44	39	17	43	36	21	34	32	34
MH16	52.78980444	1.11705003	58	28	14	57	25	18	60	23	17
GH1	52.78208922	1.1252575	52	30	18	53	30	17	42	31	27
GH2	52.78494598	1.12460809	47	33	20	43	33	24	38	25	37
GH3	52.78419622	1.12157544	36	32	32	41	24	35	31	24	45
GH4	52.78693822	1.12796749	49	36	15	66	25	9	75	17	8
GH5	52.78439652	1.12141164	34	31	35	36	24	40	17	23	60
GH6	52.78370334	1.12251244	36	34	30	26	42	32	28	39	33
GH7	52.78281284	1.12372538	56	25	19	66	18	16	67	18	15
GH8	52.78184063	1.12473255	49	34	17	52	31	17	54	24	22
GH9	52.78372723	1.12454102	54	32	14	51	33	16	39	39	22
GH10	52.78506858	1.12431614	40	40	20	32	41	27	22	44	34
GH11	52.78609201	1.12454144	39	46	15	58	29	13	62	27	11
GH12	52.78519926	1.1255399	59	28	13	56	30	14	62	24	14

GH13	52.78431254	1.12650842	59	28	13	61	26	13	66	23	11
GH14	52.78521627	1.12690128	57	29	14	60	27	13	73	17	10
GH15	52.78609905	1.12674044	47	41	12	57	26	17	56	23	21
GH16	52.78678344	1.1277891	53	37	10	46	40	14	70	22	8
FH1	52.7891617	1.11659453	51	30	19	47	27	26	33	20	47
FH2	52.78652318	1.11631782	44	35	21	40	30	30	31	24	45
FH3	52.78835787	1.11448645	45	30	25	40	28	32	28	21	51
FH4	52.78535548	1.11845662	47	38	15	54	34	12	44	26	30
FH5	52.78554954	1.11839281	41	34	25	49	34	17	51	32	17
FH6	52.78613887	1.11665667	39	36	25	33	33	34	18	31	51
FH7	52.78663617	1.11536385	42	31	27	34	29	37	31	26	43
FH8	52.78718334	1.11439297	40	30	30	37	27	36	24	19	57
FH9	52.78712072	1.11593046	52	25	23	59	19	22	66	14	20
FH10	52.78714872	1.11724994	50	28	22	39	25	36	25	23	52
FH11	52.78767175	1.11636238	54	26	20	49	20	31	51	19	30
FH12	52.78790301	1.11561027	50	27	23	47	28	25	53	21	26
FH13	52.78833635	1.11462557	42	31	27	58	28	14	38	29	33
FH14	52.78858186	1.11549611	50	27	23	54	24	22	63	16	21
FH15	52.78875427	1.11634218	54	24	22	55	20	25	48	19	33
FH16	52.78908251	1.11723921	57	25	18	57	22	21	56	22	22
FAR1	52.79286178	1.11129611	52	37	11	47	36	17	51	27	22
FAR2	52.79321201	1.10916505	52	27	21	47	26	27	44	21	35
FAR3	52.79126612	1.10858467	49	25	26	52	24	24	39	22	39
FAR4	52.79235789	1.10900801	45	30	25	34	24	42	25	22	53
FAR5	52.79327589	1.11136358	63	34	3	60	32	8	58	30	12
FAR6	52.79339634	1.10997939	47	40	13	52	31	17	45	30	25
FAR7	52.79340212	1.10897985	48	38	14	50	33	17	56	28	16
FAR8	52.79349298	1.10777395	52	34	14	65	23	12	71	16	13
FAR9	52.79275454	1.10917096	45	39	16	49	34	17	54	19	27
FAR10	52.79228331	1.11070124	58	33	9	73	21	6	78	12	10
FAR11	52.79204442	1.10936156	57	31	12	58	23	19	50	25	25
FAR12	52.7918855	1.10809565	54	31	15	37	29	34	29	26	45
FAR13	52.79167838	1.10664826	43	36	21	49	34	17	60	24	16
FAR14	52.79145073	1.10769264	47	36	17	44	27	29	39	27	34
FAR15	52.79129801	1.10893509	47	37	16	56	26	18	62	21	17
FAR16	52.79108268	1.11015315	43	38	19	61	30	9	62	19	19
MHF1	52.78039675	1.11915705	51	27	22	51	23	26	30	21	49
MHF2	52.77911746	1.1184839	57	26	17	53	28	19	44	28	28
MHF3	52.77837316	1.11535349	58	24	18	60	23	17	60	18	22
MHF4	52.78107553	1.11609046	42	33	25	44	28	28	40	27	33
MHF5	52.78039441	1.11939529	50	32	18	36	33	31	64	16	20
MHF6	52.78054546	1.11753124	49	33	18	46	35	19	40	38	22
MHF7	52.78054428	1.1157992	41	40	19	38	39	23	37	34	29
MHF8	52.77983962	1.11686764	54	31	15	58	27	15	61	22	17
MHF9	52.77910335	1.11806667	55	30	15	54	29	17	58	26	16
MHF10	52.77841226	1.11924123	55	30	15	59	25	16	51	27	22

MHF11	52.77851368	1.11728976	52	31	17	47	30	23	51	25	24
MHF12	52.77865819	1.11556199	56	30	14	55	29	16	59	18	23
MHF13	52.77858166	1.1137505	49	27	24	48	21	31	32	21	47
MHF14	52.77783005	1.11490695	52	26	22	56	22	22	53	24	23
MHF15	52.7772372	1.11681249	59	22	19	66	18	16	81	7	12
MHF16	52.77670512	1.11864829	56	24	20	56	23	21	62	17	21
P1	52.78904358	1.11153692	46	33	21	45	26	29	32	26	42
P2	52.78656512	1.11232	42	40	18	43	29	28	27	24	49
P3	52.78644065	1.10896574	29	36	35	26	30	44	20	23	57
P4	52.78628121	1.10803925	40	36	24	38	33	29	17	22	61
P5	52.78530789	1.11226715	41	37	22	30	25	45	18	15	67
P6	52.78624129	1.11204076	38	38	24	37	32	31	22	24	54
P7	52.78732114	1.11201175	44	37	19	49	29	22	23	17	60
P8	52.78868931	1.11183573	44	31	25	43	30	27	41	29	30
P9	52.78770394	1.10968083	40	32	28	46	27	27	60	16	24
P10	52.78648471	1.1087258	26	34	40	15	40	45	11	34	55
P11	52.78776135	1.10854374	35	41	24	22	36	42	22	36	42
P12	52.78881605	1.10841525	46	37	17	49	31	20	49	31	20
P13	52.78966899	1.10774185	41	40	19	40	37	23	37	37	26
P14	52.78876768	1.10714891	34	43	23	22	40	38	16	33	51
P15	52.78766278	1.10647576	33	41	26	42	35	23	59	24	17
P16	52.78691646	1.10617803	32	47	21	35	43	22	23	37	40
SF1	52.7916208	1.11688446	49	32	19	49	27	24	39	29	32
SF2	52.78915513	1.11937261	54	30	16	60	26	14	54	21	25
SF3	52.78856779	1.12179354	54	31	15	63	25	12	76	12	12
SF4	52.78643488	1.11965521	53	27	20	45	26	29	25	28	47
SF5	52.78652159	1.11880872	55	31	14	56	28	16	53	26	21
SF6	52.78772464	1.12099288	54	32	14	56	29	15	61	23	16
SF7	52.7881455	1.12211086	59	29	12	59	29	12	59	28	13
SF8	52.78860801	1.12300504	62	23	15	64	25	11	74	24	2
SF9	52.78867188	1.121014	59	26	15	60	23	17	61	23	16
SF10	52.78881756	1.11967994	59	25	16	70	16	14	89	5	6
SF11	52.78893641	1.1182515	57	27	16	57	27	16	58	29	13
SF12	52.78978416	1.11858157	58	25	17	58	22	20	68	13	19
SF13	52.79062738	1.1189373	61	22	17	65	19	16	68	15	17
SF14	52.79081647	1.11800834	63	21	16	79	11	10	91	4	5
SF15	52.79166296	1.11692154	51	28	21	45	26	29	42	24	34
SF16	52.79231608	1.11583985	57	25	18	64	19	17	61	17	22
SW1	52.78810412	1.12510085	68	21	11	79	14	7	92	4	4
SW2	52.78741455	1.12315304	53	30	17	55	29	16	45	33	22
SW3	52.78689904	1.12159951	52	32	16	45	34	21	34	29	37
SW4	52.78681637	1.12485273	57	31	12	54	32	14	72	16	12
SW5	52.7860372	1.12104753	43	36	21	40	28	32	18	36	46
SW6	52.78722935	1.12207373	47	35	18	34	40	26	25	36	39
SW7	52.78789823	1.12315977	59	29	12	70	18	12	88	8	4
SW8	52.78857624	1.12407759	56	34	10	58	32	10	61	30	9

SW9	52.78745642	1.12465435	52	37	11	53	35	12	71	21	8
SW10	52.78686441	1.12466047	50	38	12	55	33	12	63	21	16
SW11	52.78793327	1.12535248	52	37	11	53	33	14	60	24	16
SW12	52.78849837	1.12593049	52	40	8	60	32	8	62	28	10
SW13	52.78779	1.1250132	40	46	14	43	45	12	39	45	16
SW14	52.786645	1.1234032	49	39	12	59	30	11	69	19	12
SW15	52.787591	1.1270905	44	44	12	54	32	14	67	21	12
SW16	52.787029	1.1269166	35	41	24	63	22	15	60	26	14

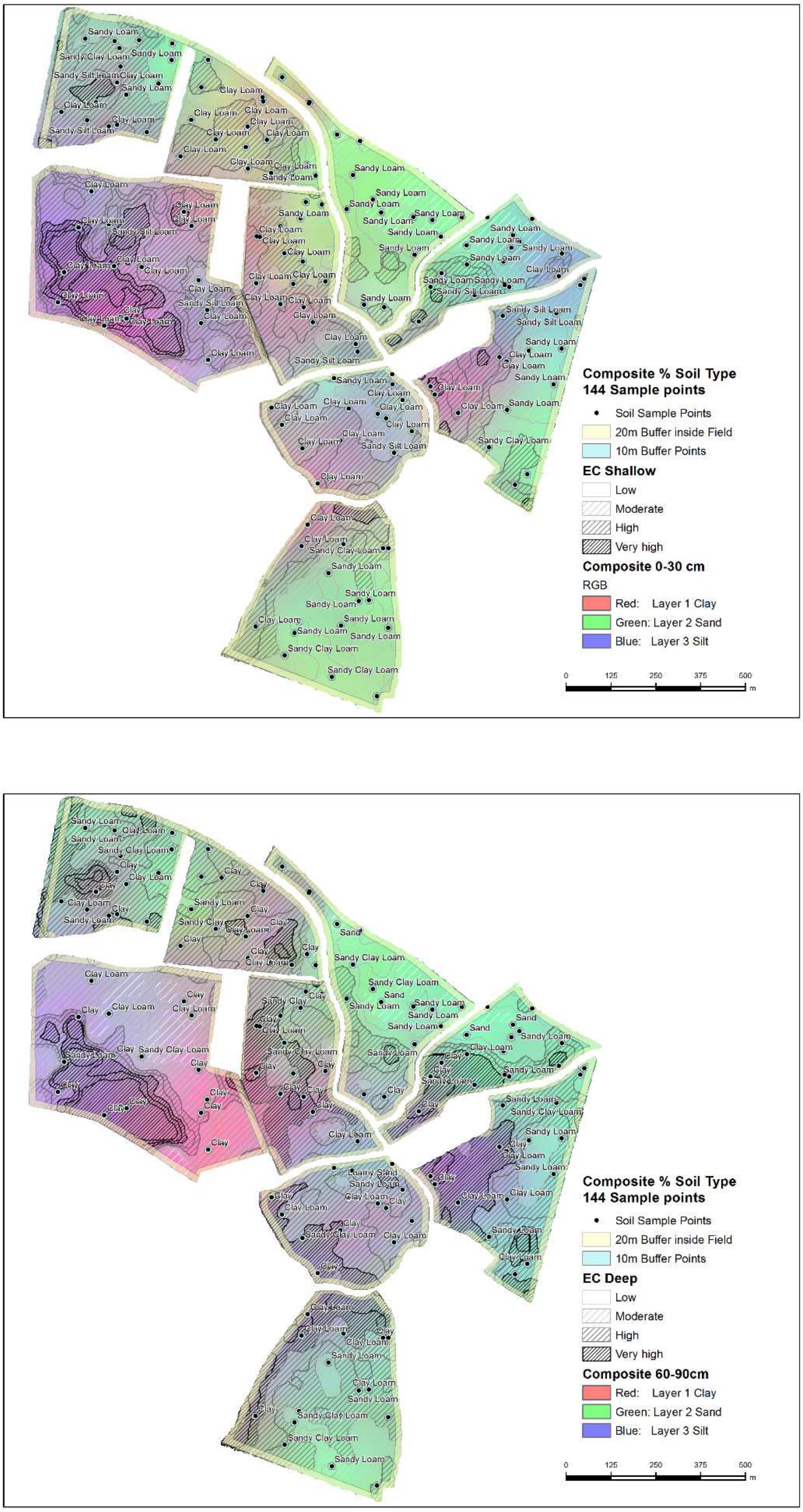

Figure A2 Electrical conductivity map for (0-30 cm) soil depth (top), and (60-90 cm) soil depth (bottom) with soil texture determined by GIS

APPENDIX B Soil chemical and physical data

Figure B1 Map of locations of soil sampling

Table B1 Soil Mineral Nitrogen (SMN) data

Table B2 Soil macronutrients data

Figure B2 Map of location of Soil Organic Carbon and soil physical measurements

Table B3 Soil Organic Carbon data

Figure B3 Photographs of field works of soil physical measurements

Table B4 Bulk density, Infiltration rate, and Penetration resistance data

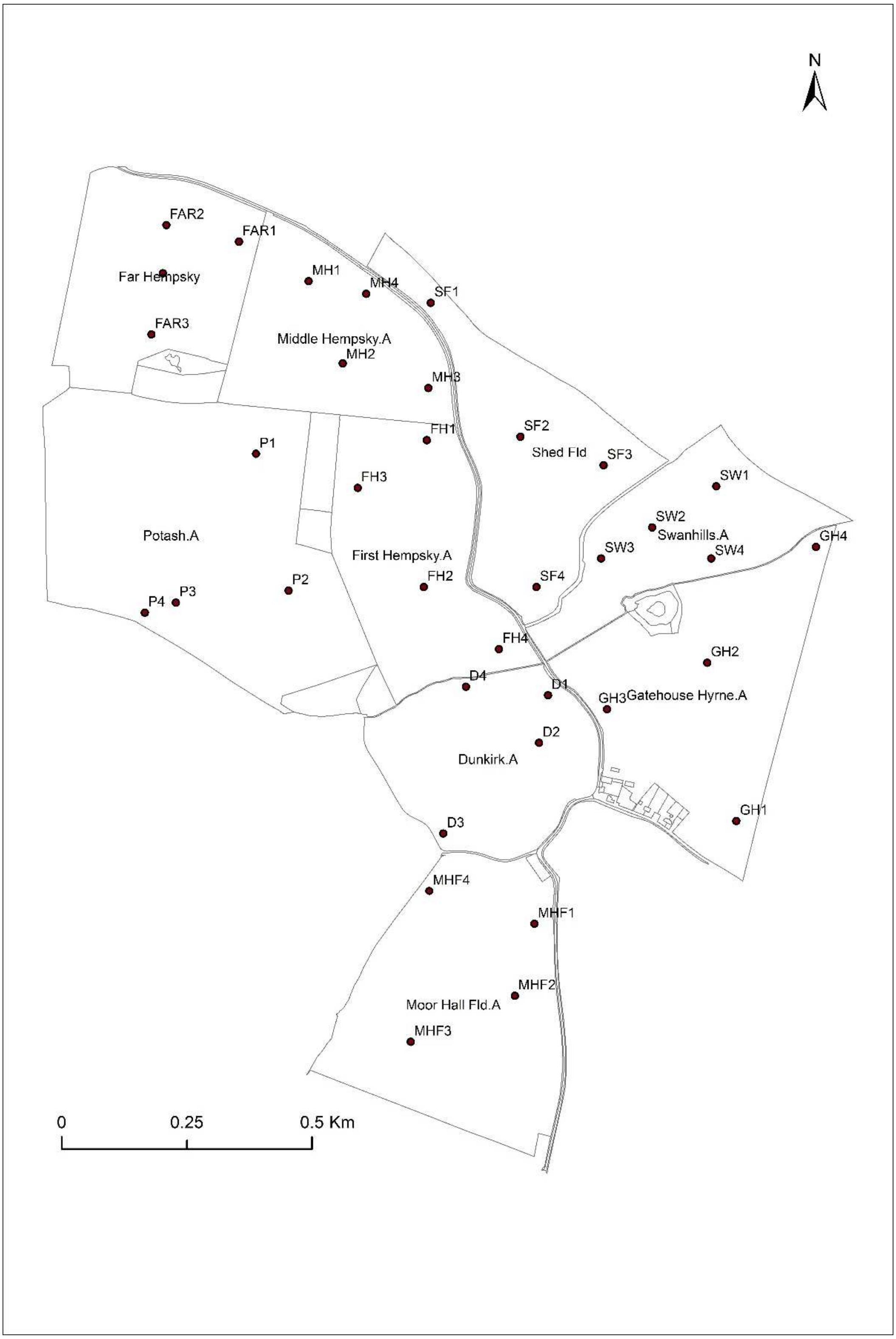

Figure B1 Map of locations of soil sampling

Table B1 Soil Mineral Nitrogen (SMN) data

Sample name	Sampling date	Depth (cm)	Dry Matter (% w/w)	Nitrate (mg N/kg)	Ammonium (mg N/kg)	Available N 30 cm profile (kg N/ha)	Depth (cm)	Dry Matter (% w/w)	Nitrate (mg N/kg)	Ammonium (mg N/kg)	Available N 30 cm profile (kg N/ha)	Depth (cm)	Dry Matter (% w/w)	Nitrate (mg N/kg)	Ammonium (mg N/kg)	Available N 30 cm profile (kg N/ha)
P1	03/09/2013	0-30	88.3	4.62	0.05	18.7	30-60	89.2	0.48	0.42	3.6	60-90	89	0.05	0.05	0.4
P2	03/09/2013	0-30	84.8	9.53	0.35	39.5	30-60	88.5	0.69	0.05	3	60-90	88.9	0.06	0.06	0.5
P3	03/09/2013	0-30	83.5	9.04	0.56	38.4	30-60	85.6	0.71	0.05	3	60-90	88	0.05	0.05	0.4
P4	03/09/2013	0-30	82	8.14	0.58	34.9	30-60	87.1	1.77	0.06	7.3	60-90	87.7	0.07	0.07	0.5
FAR1	03/09/2013	0-30	85.5	5.72	0.37	24.4	30-60	90.8	0.86	0.32	4.7	60-90	91.1	0.05	0.05	0.4
FAR2	03/09/2013	0-30	86.5	6.53	1.07	30.4	30-60	90.8	1.17	0.42	6.4	60-90	89.6	0.07	0.49	2.3
FAR3	03/09/2013	0-30	86.1	8.63	0.44	36.3	30-60	90.4	2.27	0.86	12.5	60-90	90.8	0.26	0.05	1.2
FAR4	03/09/2013	0-30	86.2	6	0.83	27.3	30-60	87.3	1.26	0.69	7.8	60-90	89.8	0.05	0.05	0.4
MH1	03/09/2013	0-30	88.2	3.74	0.05	15.2	30-60	89.7	0.56	0.36	3.7	60-90	86.8	0.05	0.05	0.4
MH2	03/09/2013	0-30	86.9	4.78	0.48	21	30-60	89.5	0.49	0.49	3.9	60-90	90.3	0.05	0.51	2.2
MH3	03/09/2013	0-30	89	5.53	0.38	23.6	30-60	94	1.32	0.28	6.4	60-90	88	0.05	0.71	3.1
MH4	03/09/2013	0-30	88.5	8.27	1.13	37.6	30-60	93.6	1.85	0.46	9.2	60-90	88.6	0.94	0.37	5.2
FH1	03/09/2013	0-30	88.6	6.66	4.31	43.9	30-60	90.1	1.73	0.45	8.7	60-90	88.1	0.06	0.34	1.6
FH2	03/09/2013	0-30	84.1	6.22	1.24	29.8	30-60	87.3	0.83	0.24	4.3	60-90	88.9	0.05	0.05	0.4
FH3	03/09/2013	0-30	85.9	3.6	0.82	17.7	30-60	86.8	0.42	0.05	1.9	60-90	89.2	0.05	0.05	0.4
FH4	03/09/2013	0-30	81.5	13.19	1.22	57.6	30-60	84.8	7.06	0.06	28.5	60-90	85.8	0.99	0.05	4.2
SF1	03/09/2013	0-30	89.1	7.17	0.84	32.1	30-60	90.6	2.21	0.34	10.2	60-90	88.1	0.69	0.05	3
SF2	03/09/2013	0-30	89.9	6.27	0.42	26.8	30-60	96.1	1.5	0.05	6.2	60-90	91.7	0.42	0.26	2.7
SF3	03/09/2013	0-30	89.6	5.35	0.46	23.2	30-60	96.1	1.94	0.62	10.2	60-90	96.1	0.2	0.49	2.7
SF4	03/09/2013	0-30	87.1	5.6	1	26.4	30-60	87.8	1.06	0.75	7.2	60-90	87.1	0.05	0.47	2.1
SW1	03/09/2013	0-30	88.6	9.46	2.32	47.1	30-60	95	6.25	0.6	27.4	60-90	97.3	0.43	0.26	2.7
SW2	03/09/2013	0-30	86.2	13.16	0.91	56.3	30-60	86.3	3.4	0.47	15.5	60-90	84	2.3	1.15	13.8
SW3	03/09/2013	0-30	85.5	3.7	0.54	16.9	30-60	85.4	2.47	0.54	12	60-90	81.7	0.67	0.34	4
SW4	03/09/2013	0-30	85.9	7.15	1.12	33.1	30-60	89.5	1.74	0.43	8.7	60-90	87.1	0.05	0.39	1.7
D1	03/09/2013	0-30	85.5	8.51	0.38	35.6	30-60	89.1	1.05	0.3	5.4	60-90	90.6	0.05	0.05	0.4
D2	03/09/2013	0-30	84.6	6.75	0.35	28.4	30-60	89.6	1.16	0.05	4.8	60-90	90.2	0.06	0.06	0.4
D3	03/09/2013	0-30	87.9	10.43	0.29	42.9	30-60	87	2.02	0.4	9.7	60-90	87.3	0.31	0.05	1.5
D4	03/09/2013	0-30	83.8	7.03	0.48	30.1	30-60	89.4	1.42	0.06	5.9	60-90	87.9	0.08	0.05	0.5
GH1	03/09/2013	0-30	87.1	5.87	0.69	26.2	30-60	92.4	1.73	0.31	8.2	60-90	90.7	0.05	0.3	1.4
GH2	03/09/2013	0-30	86.1	4.29	0.6	19.6	30-60	89.3	1.4	0.05	5.8	60-90	89.8	0.05	0.05	0.4
GH3	03/09/2013	0-30	83.7	5.19	0.35	22.2	30-60	88.3	2.83	0.29	12.5	60-90	85.6	3.11	0.39	14
GH4	03/09/2013	0-30	86	16.61	0.81	69.7	30-60	89.7	6.34	0.42	27	60-90	84.5	4.32	0.67	19.9
MHF1	03/09/2013	0-30	87.5	13.81	0.33	56.6	30-60	90.1	3.8	0.3	16.4	60-90	87.1	0.05	0.32	1.5
MHF2	03/09/2013	0-30	87.8	4.62	0.5	20.5	30-60	91.7	2.32	0.3	10.5	60-90	91.6	0.04	0.04	0.4
MHF3	03/09/2013	0-30	88.9	4.91	0.53	21.8	30-60	92.4	3.72	0.34	16.2	60-90	90.8	0.47	0.41	3.5
MHF4	03/09/2013	0-30	87.1	11.39	0.29	46.8	30-60	88.3	1.96	0.23	8.8	60-90	86.4	0.47	0.05	2.1

Sample name	Sampling date	Depth (cm)	Dry Matter (% w/w)	Nitrate (mg N/kg)	Ammonium (mg N/kg)	Available N 30 cm profile (kg N/ha)	Depth (cm)	Dry Matter (% w/w)	Nitrate (mg N/kg)	Ammonium (mg N/kg)	Available N 30 cm profile (kg N/ha)	Depth (cm)	Dry Matter (% w/w)	Nitrate (mg N/kg)	Ammonium (mg N/kg)	Available N 30 cm profile (kg N/ha)
P1	03/02/2014	0-30	83	2.65	0.68	13.3	30-60	83.4	2	0.49	10	60-90	84.2	2.56	0.43	11.9
P2	03/02/2014	0-30	77.6	3.55	1.28	19.3	30-60	81.2	5.03	0.64	22.7	60-90	84	4.17	0.52	18.7
P3	03/02/2014	0-30	78.3	2.89	0.69	14.3	30-60	80.6	3.79	0.54	17.3	60-90	83.5	3.5	0.44	15.8
P4	03/02/2014	0-30	73.8	3.27	1.19	17.9	30-60	78.4	3.58	0.69	17.1	60-90	83.1	2.99	0.39	13.5
FAR1	03/02/2014	0-30	80.1	3.48	1.21	18.8	30-60	82.9	5.01	0.88	23.6	60-90	85.2	5.01	0.93	23.8
FAR2	03/02/2014	0-30	81.8	3.32	1.49	19.3	30-60	84	3.64	0.95	18.4	60-90	85	3.02	1.16	16.7
FAR3	03/02/2014	0-30	80.1	1.65	1.34	12	30-60	84	2.27	0.77	12.2	60-90	85.2	2.95	1.15	16.4
FAR4	03/02/2014	0-30	80.2	3.53	1.05	18.3	30-60	81.8	4.73	1.48	24.9	60-90	83.1	4.14	1	20.5
MH1	03/02/2014	0-30	85.2	2.55	1.28	15.3	30-60	85.7	1.3	1.05	9.4	60-90	84.4	1.63	2.08	14.8
MH2	03/02/2014	0-30	82.7	3.39	1.03	17.7	30-60	83.7	1.42	0.6	8.1	60-90	84.6	0.56	0.56	4.4
MH3	03/02/2014	0-30	85.2	2.54	0.83	13.5	30-60	87.2	1.8	0.67	9.9	60-90	84.4	0.06	0.55	2.4
MH4	03/02/2014	0-30	84.4	3.03	5.42	33.8	30-60	86.4	1.93	0.97	11.6	60-90	84.7	0.49	0.68	4.7
FH1	03/02/2014	0-30	84.8	2.72	0.84	14.2	30-60	85.7	2.08	1.49	14.3	60-90	82.8	0.07	0.79	3.4
FH2	03/02/2014	0-30	79.8	3.13	1.43	18.2	30-60	82.9	2.26	0.63	11.6	60-90	84.6	0.71	0.51	4.9
FH3	03/02/2014	0-30	81.9	1.17	1.12	9.2	30-60	82.8	1.74	0.71	9.8	60-90	83.5	0.8	0.57	5.5
FH4	03/02/2014	0-30	76.8	6.71	1.19	31.6	30-60	79.4	4.73	0.84	22.3	60-90	83.4	1.66	0.54	8.8
SF1	03/02/2014	0-30	84.8	3.62	2.67	25.2	30-60	86.6	3.48	1	17.9	60-90	83.2	0.75	1	7
SF2	03/02/2014	0-30	86.4	2.59	0.96	14.2	30-60	88.8	1.68	1.12	11.2	60-90	89.9	0.05	0.87	3.7
SF3	03/02/2014	0-30	86.5	1.85	1.12	11.9	30-60	89	1.82	0.82	10.6	60-90	90.4	0.06	1.12	4.7
SF4	03/02/2014	0-30	83.8	2.4	0.94	13.4	30-60	83	2.02	1.15	12.7	60-90	82.8	0.59	1.18	7.1
SW1	03/02/2014	0-30	85.1	5.21	1.29	26	30-60	90.1	1.22	1.22	9.8	60-90	94.2	0.05	0.49	2.2
SW2	03/02/2014	0-30	78.1	0.07	2.88	11.8	30-60	83.3	1.37	1.07	9.8	60-90	79.7	2.99	8.76	47
SW3	03/02/2014	0-30	80.3	1.75	3.16	19.6	30-60	80.3	2.66	0.85	14	60-90	80	0.94	1	7.8
SW4	03/02/2014	0-30	82.1	3.91	1.23	20.5	30-60	84.9	1.84	0.79	10.5	60-90	85.5	0.53	0.87	5.6
D1	03/02/2014	0-30	83.5	1.03	0.54	6.3	30-60	83.2	2.1	0.53	10.5	60-90	80.4	1.1	0.84	7.8
D2	03/02/2014	0-30	81.1	3.41	1.01	17.7	30-60	83.3	2.58	0.74	13.3	60-90	85.4	0.57	0.51	4.3
D3	03/02/2014	0-30	82.8	1.57	1.08	10.6	30-60	83.6	1.32	0.66	7.9	60-90	83.9	0.69	0.49	4.7
D4	03/02/2014	0-30	78.2	4.59	1.16	23	30-60	81.2	3.47	0.61	16.3	60-90	83	0.99	0.49	5.9
GH1	03/02/2014	0-30	84.4	4.76	1.04	23.2	30-60	85.7	2.46	0.67	12.5	60-90	85.4	0.65	0.56	4.9
GH2	03/02/2014	0-30	83.7	2.07	0.92	12	30-60	84.9	1.37	0.81	8.7	60-90	85.7	0.83	0.61	5.8
GH3	03/02/2014	0-30	79.1	3.63	1.31	19.8	30-60	82.4	2.56	0.75	13.3	60-90	82.1	1.69	0.88	10.3
GH4	03/02/2014	0-30	81.7	4.91	0.9	23.2	30-60	86.4	4.14	0.78	19.7	60-90	87.1	0.7	0.65	5.4
MHF1	03/02/2014	0-30	83.5	2.33	1.16	14	30-60	85.6	1.77	0.61	9.5	60-90	83.2	0.52	0.47	4
MHF2	03/02/2014	0-30	84.3	1.86	0.93	11.1	30-60	86.4	1.82	0.69	10	60-90	86.1	0.39	0.56	3.8
MHF3	03/02/2014	0-30	83.5	0.6	1.54	8.6	30-60	86.4	1.86	0.55	9.6	60-90	86.7	0.06	0.56	2.5
MHF4	03/02/2014	0-30	83	2.26	1.06	13.3	30-60	84.3	1.9	0.55	9.8	60-90	83.7	1.12	0.49	6.4

Sample Name	Sampling date	Depth (cm)	Dry Matter % w/w	Nitrate mg N/kg	Ammonium mg N/kg	Available N 30 cm profile kg N/ha	Sampling date	Depth (cm)	Dry matter % w/w)	Nitrate N (mg/kg)	Ammonium N (mg/kg)	Available N 30 cm profile (kgN/ha)	Sampling date	Depth (cm)	Dry matter (% w/w)	Nitrate N (mg/kg)	Ammonium N (mg/kg)	Available N 30 cm profile (kgN/ha)
P1	17/07/2014	0-15	86.8	1.7	0.92	5.3	20/02/2015	0-15	83.2	0.44	0.87	4.8	21/07/2015	0-15	90.9	2.34	0.94	12.3
P2	17/07/2014	0-15	80.9	3.33	1.13	8.9	20/02/2015	0-15	79.1	0.61	1.06	6.2	21/07/2015	0-15	87.4	7.66	1.23	33.3
P3	17/07/2014	0-15	81.2	5.18	1.29	12.9	20/02/2015	0-15	80	7.04	0.77	29.2	21/07/2015	0-15	86	3.02	1.03	15.2
P4	17/07/2014	0-15	80.3	4.41	1.09	11	20/02/2015	0-15	76.2	1.01	1.22	8.4	21/07/2015	0-15	86.2	2.91	1.07	14.9
FAR1	17/07/2014	0-15	84.1	4.3	0.89	10.4	20/02/2015	0-15	80.5	1.69	1.06	10.2	21/07/2015	0-15	88.6	1.55	0.69	8.4
FAR2	17/07/2014	0-15	84.4	4.88	3.1	16	20/02/2015	0-15	83.2	0.9	0.95	7	21/07/2015	0-15	91	1.46	0.61	7.8
FAR3	17/07/2014	0-15	85.7	1.99	0.66	5.3	20/02/2015	0-15	81.5	1.13	0.92	7.8	21/07/2015	0-15	88.8	1.51	0.8	8.7
FAR4	17/07/2014	0-15	85.1	3.84	0.95	9.6	20/02/2015	0-15	81.6	0.69	1.03	6.4	21/07/2015	0-15	89	1.89	1.07	11.1
MH1	17/07/2014	0-15	87.2	5.12	2.07	14.4	20/02/2015	0-15	85.3	1.17	0.76	7.2	21/07/2015	0-15	94.3	1.65	0.99	9.9
MH2	17/07/2014	0-15	87.4	1.85	1.3	6.3	20/02/2015	0-15	84.3	1.43	1.15	9.6	21/07/2015	0-15	92.5	4.08	1.43	20.6
MH3	17/07/2014	0-15	90.6	2.49	0.95	6.9	20/02/2015	0-15	84.9	0.65	0.7	5	21/07/2015	0-15	92.5	1.61	1.13	10.3
MH4	17/07/2014	0-15	87.4	4.07	0.95	10	20/02/2015	0-15	82.8	1.39	1.88	12.2	21/07/2015	0-15	93.7	6.94	1.04	29.9
FH1	17/07/2014	0-15	89.6	2.08	1.16	6.5	20/02/2015	0-15	83.3	0.61	0.85	5.4	21/07/2015	0-15	92	1.63	1.25	10.8
FH2	17/07/2014	0-15	82.6	4.28	0.9	10.4	20/02/2015	0-15	80.9	1.64	0.78	9	21/07/2015	0-15	90.1	3.81	1.32	19.3
FH3	17/07/2014	0-15	86.7	2.15	0.8	5.9	20/02/2015	0-15	83.8	1.12	0.59	6.4	21/07/2015	0-15	90.2	2.07	0.87	11
FH4	17/07/2014	0-15	81.7	3.79	1.2	10	20/02/2015	0-15	76.7	1.17	0.88	7.6	21/07/2015	0-15	86.8	3.4	1.22	17.3
SF1	17/07/2014	0-15	89.6	1.58	0.96	5.1	20/02/2015	0-15	84.6	1.1	0.7	6.8	21/07/2015	0-15	92.4	2.96	1.54	16.9
SF2	17/07/2014	0-15	90.2	2.06	0.91	5.9	20/02/2015	0-15	85.4	0.55	1	5.8	21/07/2015	0-15	93.3	9.9	1.03	41
SF3	17/07/2014	0-15	89.7	0.95	1	3.9	20/02/2015	0-15	86.6	0.67	0.61	4.8	21/07/2015	0-15	93.2	2.73	0.57	12.4
SF4	17/07/2014	0-15	87.5	1.73	0.89	5.2	20/02/2015	0-15	84.9	0.3	0.66	3.6	21/07/2015	0-15	90.5	6.72	0.8	28.2
SW1	17/07/2014	0-15	86.5	3.03	1.13	8.3	20/02/2015	0-15	84.6	2.31	0.58	10.8	21/07/2015	0-15	92.3	4.53	1.24	21.6
SW2	17/07/2014	0-15	87	2.06	1	6.1	20/02/2015	0-15	83.9	1.69	1.01	10.2	21/07/2015	0-15	89.2	2.78	0.97	14.1
SW3	17/07/2014	0-15	84.3	1.92	0.85	5.5	20/02/2015	0-15	81.5	3.25	1.36	17.4	21/07/2015	0-15	86.2	14.65	1.03	58.8
SW4	17/07/2014	0-15	85.6	1.75	1.15	5.8	20/02/2015	0-15	81.9	0.6	0.71	4.8	21/07/2015	0-15	89.9	3.88	1.18	19
D1	17/07/2014	0-15	84.8	2.9	0.69	7.2	20/02/2015	0-15	81.4	1.23	0.9	8	21/07/2015	0-15	90.6	4.01	0.8	18.1
D2	17/07/2014	0-15	84.3	3.61	0.9	9	20/02/2015	0-15	81.1	1.03	1.03	7.8	21/07/2015	0-15	91.4	2.86	1.22	15.3
D3	17/07/2014	0-15	86.9	6.99	2.33	18.6	20/02/2015	0-15	82.2	0.06	1.19	4.6	21/07/2015	0-15	90.7	1.42	0.74	8.1
D4	17/07/2014	0-15	81.1	8.59	0.8	18.8	20/02/2015	0-15	78.6	1.49	1.06	9.6	21/07/2015	0-15	82.9	6.66	1.92	32.2
GH1	17/07/2014	0-15	87.6	1.79	0.92	5.4	20/02/2015	0-15	83.7	1.35	1.19	9.6	21/07/2015	0-15	89.9	1.74	1.32	11.5
GH2	17/07/2014	0-15	87	2.06	1.03	6.2	20/02/2015	0-15	82	0.64	0.89	5.8	21/07/2015	0-15	89.1	2.79	1.13	14.7
GH3	17/07/2014	0-15	82.3	30.02	1.08	62.2	20/02/2015	0-15	79.1	0.95	0.88	6.8	21/07/2015	0-15	83.8	14.16	1.7	59.5
GH4	17/07/2014	0-15	86.2	1.66	1.01	5.3	20/02/2015	0-15	82.1	1.36	0.71	7.8	21/07/2015	0-15	90.5	2.33	1.11	12.9
MHF1	17/07/2014	0-15	87.9	2.12	0.77	5.8	20/02/2015	0-15	84.4	0.48	0.85	5	21/07/2015	0-15	90.9	4.26	1.16	20.3
MHF2	17/07/2014	0-15	88.8	2.26	0.93	6.4	20/02/2015	0-15	84.8	0.79	1.14	7.2	21/07/2015	0-15	91.5	1.59	1.26	10.7
MHF3	17/07/2014	0-15	87.6	2.29	1.25	7.1	20/02/2015	0-15	84.6	0.06	0.95	3.8	21/07/2015	0-15	92.7	2.33	0.88	12
MHF4	17/07/2014	0-15	87.1	3.02	1.04	8.1	20/02/2015	0-15	82.2	0.06	0.79	3.2	21/07/2015	0-15	91.3	3.28	0.98	16

Table B2 Soil macronutrients data

Sample	P Sep.2013 (mg L^{-1})	K Sep.2013 (mg L^{-1})	Mg Sep.2013 (mg L^{-1})	SO_4-S Sep.2013 (mg L^{-1})	P Feb.2014 (mg L^{-1})	K Feb.2014 (mg L^{-1})	Mg Feb.2014 (mg L^{-1})	SO_4-S Feb.2014 (mg L^{-1})	P Feb.2015 (mg L^{-1})	K Feb.2015 (mg L^{-1})	Mg Feb.2015 (mg L^{-1})	SO_4-S Feb.2015 (mg L^{-1})	P Jul.2015 (mg L^{-1})	K Jul.2015 (mg L^{-1})	Mg Jul.2015 (mg L^{-1})	SO_4-S Jul.2015 (mg L^{-1})
P1	28.4	86	52	8.9	25	83.8	38.1	5.7	23	96.3	48.2	3.6	26	98.3	44.8	6.8
P2	21.4	76	57	15.4	17.6	82.2	62.35	5.1	28.2	120.8	80.5	5.0	22.6	63.6	66.7	9.1
P3	18.6	110	72	13.4	21.2	134.3	80.25	4.8	28	165.9	83.5	5.1	36	143.0	70.9	10.2
P4	31	140	86	13.9	25.4	137.5	93.15	7.4	37.6	170.8	96.4	4.7	50.2	160.0	91.6	7.8
FAR1	14.4	94	51	6.5	11.6	73.1	59.75	6.8	13	74.5	57.9	3.9	25.4	68.9	35.1	8.4
FAR2	22.8	128	48	6.9	15.6	81.1	57.3	7.6	15	85.4	48.5	3.8	18.8	71.3	44.3	5.4
FAR3	16.4	134	51	6.5	14.8	76.3	61.25	7.1	19.2	110.8	54.1	3.4	18.4	80.2	54.1	5.9
FAR4	18.4	118	49	8.0	13	95.1	57.15	4.0	19.2	132.6	56.8	4.4	17.4	81.4	53.9	5.1
MH1	31.2	82	53	8.0	24.6	91.3	41.05	5.9	37.8	165.6	49.7	3.8	39.6	79.8	33.4	6.9
MH2	25.6	87	45	7.5	23.2	113.9	29.6	6.8	28.4	129.3	44.4	3.6	49	245.0	54.5	8.7
MH3	37.6	92	41	6.4	37.8	147.2	43.4	4.5	46.6	160.4	30	3.8	43.8	109.0	30.8	7.4
MH4	31.4	122	45	8.0	34.4	172.4	46.75	6.8	38.6	232.1	45.1	3.6	39.2	141.0	34.6	7.1
FH1	20.8	131	45	7.6	22.2	125.7	43.15	8.4	19.6	140.2	41.2	3.2	23.4	73.3	32.6	7.7
FH2	16.2	109	59	7.6	15.8	128.9	63.2	8.7	15.8	124.9	63	4.0	16.2	106.0	55.5	7.9
FH3	18.6	121	57	6.5	21.4	164.4	65.4	6.1	17	203.4	86	3.5	19	133.0	62	5.1
FH4	22.8	85	57	12.1	26.4	109.1	65.5	10.2	24.6	86.6	74.1	7.0	25	58.0	60.6	12.5
SF1	51	124	46	8.3	26.2	128.4	51.2	8.4	55.2	234.4	48.3	3.1	64.8	204.0	50	11.3
SF2	41.2	107	44	7.2	33.6	109.6	48.3	6.4	53.6	206.3	40.3	3.0	44.8	138.0	40.9	11.0
SF3	41.4	91	43	6.3	36.8	184.3	43.5	7.5	46	160.4	46.8	2.5	41.2	101.0	44.6	5.7
SF4	32	86	52	7.9	28	110.7	75.3	7.2	31.8	108.0	55.8	2.4	36	107.0	74	7.1
SW1	40.6	142	45	6.7	37	167.6	51.45	7.0	38.4	199.0	51.6	2.6	38.8	242.0	42.7	7.3
SW2	39	114	50	19.8	26.6	120.3	59.25	8.5	34.6	152.7	43.6	2.8	31.6	103.0	51	8.8
SW3	23	69	48	9.6	15	123.0	56.5	7.9	25.6	117.2	55.1	3.4	20	103.0	54.7	9.0
SW4	11.8	79	53	6.3	10	73.1	58.5	7.4	13.6	87.4	39.8	2.6	18	133.0	54.5	7.6
D1	13.8	95	54	8.5	12.6	103.2	38.85	6.5	12.8	118.5	41.6	3.9	14.4	128.0	59.5	7.2
D2	14.2	126	50	8.3	11	132.7	35.5	4.3	13.4	147.1	56.7	4.0	12.4	100.0	48.8	8.2
D3	34.4	189	69	8.2	31.8	168.7	69.25	7.1	29	170.8	72.8	3.4	32	168.0	72.3	5.6
D4	21.8	68	48	8.7	22.6	97.2	48.15	5.2	34	87.4	67	4.3	36.8	94.7	58.8	8.0
GH1	52.2	138	51	7.2	43	196.1	72.15	11.0	54.6	182.9	54.9	3.3	49.6	118.0	54.5	8.4
GH2	27.4	153	64	6.7	29.6	153.6	77.15	10.8	26.8	149.4	53	3.6	25.2	102.0	65.8	7.0
GH3	34.2	170	57	8.5	33.6	175.7	68.3	10.7	43	175.3	54.3	4.2	41.6	149.0	49.5	8.6
GH4	24.6	141	44	9.1	24.6	137.5	50.95	11.5	18.2	148.3	36.3	4.0	16.8	105.0	30.6	6.9
MHF1	26.8	90	50	9.7	28.4	138.1	45.35	4.7	32.6	168.4	49.4	3.4	29.2	121.0	46.8	6.6
MHF2	34.6	138	47	7.0	25	160.6	40.05	5.2	29.8	147.7	41.1	2.6	28.6	146.0	43.5	5.8
MHF3	38.6	99	43	6.8	35.6	143.4	44.45	4.7	41	118.9	35.2	3.6	43.2	112.0	37	6.1
MHF4	29.4	133	64	9.5	31.2	122.5	46.15	7.6	25.2	146.3	64.3	3.8	21	93.5	47	6.3

Figure B2 Map of location of Soil Organic Carbon (SOC) and soil physical measurement

Table B3 Soil Organic Carbon (SOC) data

Sample	SOC (%) May2013	SOC (%) Sep.2013	SOC (%) Feb.2014	SOC (%) Feb.2015	SOC (%) Jul.2015	Sample	SOC (%) May2013	SOC (%) Sep.2013	SOC (%) Feb.2014	SOC (%) Feb.2015	SOC (%) Jul.2015	Sample	SOC (%) May2013	SOC (%) Sep.2013	SOC (%) Feb.2014	SOC (%) Feb.2015	SOC (%) Jul.2015
P1		1.508	1.508	1.508	1.566	FH1		1.508	1.218	1.334	1.392	D1		1.74	1.972	1.914	2.03
P2		2.03	2.262	2.552	2.61	FH2		1.972	2.03	2.262	2.32	D2		1.856	1.914	2.088	1.972
P3		2.378	2.378	2.494	2.436	FH3		1.914	1.682	1.682	1.914	D3		1.45	1.392	1.508	1.566
P4		2.842	3.248	3.132	3.364	FH4		2.494	2.958	3.19	3.48	D4		2.378	2.378	2.668	2.784
P5	1.88					FH5	1.47					D5	1.64				
P6	1.77					FH6	2.10					D6	1.12				
P7	2.05					FH7	1.05					D7	1.04				
P8	1.21					FH8	1.31					D8	0.97				
P9	1.35					FH9	1.05					D9	1.48				
P10	1.81					FH10	1.01					D10	1.55				
P11	1.33					FH11	0.97					D11	2.07				
P12	1.06					FH12	1.09					D12	1.49				
P13	1.08					FH13	1.19					D13	1.69				
P14	1.44					FH14	0.98					D14	1.15				
P15	1.53					FH15	0.81					D15	0.91				
P16	1.59					FH16	3.67					D16	2.08				
FAR1		1.798	1.798	1.914	1.972	SF1		1.276	1.45	1.276	1.45	GH1		1.45	2.378	1.508	1.74
FAR2		1.74	1.624	1.682	1.682	SF2		1.102	1.218	1.218	1.16	GH2		1.566	2.146	1.624	1.798
FAR3		1.798	1.624	1.798	1.74	SF3		1.044	1.218	1.16	1.102	GH3		2.32	2.9	2.61	2.726
FAR4		1.856	1.682	1.856	1.798	SF4		1.276	1.566	1.45	1.508	GH4		2.03	2.262	1.972	2.262
FAR5	1.23					SF5	1.02					GH5	1.98				
FAR6	1.18					SF6	0.82					GH6	2.11				
FAR7	0.83					SF7	0.98					GH7	1.39				
FAR8	0.80					SF8	0.81					GH8	1.92				
FAR9	1.21					SF9	0.82					GH9	0.93				
FAR10	1.10					SF10	0.82					GH10	1.59				
FAR11	1.38					SF11	0.87					GH11	1.94				
FAR12	1.20					SF12	0.83					GH12	0.78				
FAR13	1.53					SF13	0.68					GH13	0.69				
FAR14	1.27					SF14	0.85					GH14	0.80				
FAR15	1.46					SF15	0.96					GH15	1.55				
FAR16	1.49					SF16	0.92					GH16	2.48				
MH1		1.45	1.218	1.45	1.334	SW1		1.566	1.798	1.74	1.914	MHF1		1.566	1.45	1.45	1.624
MH2		1.392	1.45	1.508	1.74	SW2		1.45	1.682	1.682	1.74	MHF2		1.334	1.044	1.102	1.218
MH3		1.218	1.16	1.334	1.276	SW3		1.74	1.914	2.088	2.436	MHF3		1.102	1.16	1.16	1.334
MH4		1.218	1.276	1.508	1.45	SW4		1.74	1.798	1.914	2.088	MHF4		1.856	1.682	1.74	1.856
MH5	0.81					SW5	2.25					MHF5	1.07				
MH6	0.98					SW6	0.95					MHF6	0.93				
MH7	1.00					SW7	0.89					MHF7	1.24				
MH8	1.03					SW8	1.39					MHF8	1.00				
MH9	0.95					SW9	1.70					MHF9	0.79				
MH10	0.82					SW10	1.43					MHF10	0.78				
MH11	1.00					SW11	2.02					MHF11	0.85				
MH12	1.12					SW12	3.06					MHF12	1.02				
MH13	0.90					SW13	2.61					MHF13	1.02				
MH14	1.05					SW14	1.98					MHF14	0.96				
MH15	0.98					SW15	2.02					MHF15	0.76				
MH16	0.98					SW16	2.27					MHF16	0.78				

Figure B3 Photographs of field works to carry out soil physical measurements (bulk density, infiltration rate and penetration resistance) and soil sampling in May 2013

Table B4 Bulk density (BD), Infiltration rate (IR), and Penetration resistance (PR) data

Sample	BD May2013 (gm/cm^3)	BD Jun.2014 (gm/cm^3)	BD Jun.2015 (gm/cm^3)	IR May2013 (mm/hr)	IR Jun.2014 (mm/hr)	IR Jun.2015 (mm/hr)	PR May2013 (N cm^{-2})	PR Jun.2014 (N cm^{-2})	PR Jun.2015 (N cm^{-2})	Sample	BD May.2013 (gm/cm^3)	BD Jun.2014 (gm/cm^3)	BD Jun.2015 (gm/cm^3)	IR May2013 (mm/hr)	IR Jun.2014 (mm/hr)	IR Jun.2015 (mm/hr)	PR May2013 (N cm^{-2})	PR Jun.2014 (N cm^{-2})	PR Jun.2015 (N cm^{-2})
P1		1.51	1.52		64	0		230	357	SF1		1.72	1.59		30	0		333	333
P2		1.41	1.23		46	3		227	413	SF2		1.58	1.53		46	3		277	367
P3		1.57	1.26		91	15		210	390	SF3		1.62	1.66		51	15		250	313
P4		1.37	1.17		78	0		207	423	SF4		1.51	1.35		25	0		240	270
P5	1.45			48			347			SF5	1.66			70			387		
P6	1.31			48			427			SF6	1.73			50			407		
P7	1.36			49			480			SF7	1.57			59			427		
P8	1.44			29			373			SF8	1.63			38			420		
P9	1.45			51			493			SF9	1.60			27			453		
P10	1.26			70			393			SF10	1.58			57			360		
P11	1.50			27			533			SF11	1.51			43			400		
P12	1.62			71			393			SF12	1.62			76			400		
P13	1.48			28			533			SF13	1.67			73			440		
P14	1.50			5			420			SF14	1.66			48			373		
P15	1.59			38			413			SF15	1.59			16			453		
P16	1.33			46			587			SF16	1.64			53			533		
FAR1		1.57	1.34		58	4		250	343	MH1		1.83	1.45		29	6		347	367
FAR2		1.50	1.44		22	3		200	397	MH2		1.71	1.49		20	6		260	453
FAR3		1.48	1.47		40	3		253	300	MH3		1.68	1.53		55	6		307	420
FAR4		1.49	1.31		51	0		380	380	MH4		1.86	1.33		23	3		275	380
FAR5	1.54			48			340			MH5	1.67			95			453		
FAR6	1.57			51			360			MH6	1.60			81			460		
FAR7	1.60			38			413			MH7	1.66			62			407		
FAR8	1.66			53			493			MH8	1.64			38			540		
FAR9	1.52			70			387			MH9	1.63			87			427		
FAR10	1.51			40			373			MH10	1.52			32			413		
FAR11	1.57			21			433			MH11	1.73			79			500		
FAR12	1.48			12			427			MH12	1.67			137			340		
FAR13	1.50			47			503			MH13	1.70			88			460		
FAR14	1.57			53			460			MH14	1.50			86			460		
FAR15	1.64			56			407			MH15	1.65			95			460		
FAR16	1.56			52			320			MH16	1.58			130			493		

Sample	BD May2013 (gm/cm^3)	BD Jun.2014 (gm/cm^3)	BD Jun.2015 (gm/cm^3)	IR May2013 (mm/hr)	IR Jun.2014 (mm/hr)	IR Jun.2015 (mm/hr)	PR May2013 (N cm^{-2})	PR Jun.2014 (N cm^{-2})	PR Jun.2015 (N cm^{-2})	Sample	BD May2013 (gm/cm^3)	BD Jun.2014 (gm/cm^3)	BD Jun.2015 (gm/cm^3)	IR May2013 (mm/hr)	IR Jun.2014 (mm/hr)	IR Jun.2015 (mm/hr)	PR May2013 (N cm^{-2})	PR Jun.2014 (N cm^{-2})	PR Jun.2015 (N cm^{-2})
FH1		1.72	1.61		23	3		323	343	SW1		1.67	1.36		62	6		323	283
FH2		1.70	1.48		7	3		400	397	SW2		1.70	1.60		19	0		400	263
FH3		1.79	1.60		13	3		343	300	SW3		1.53	1.47		77	0		343	327
FH4		1.46	1.46		31	6		273	380	SW4		1.62	1.45		30	8		273	300
FH5	1.50			23			517			SW5	1.51			78			580		
FH6	1.50			61			460			SW6	1.54			75			533		
FH7	1.74			43			420			SW7	1.58			75			507		
FH8	1.47			47			493			SW8	1.50			61			420		
FH9	1.68			36			467			SW9	1.44			38			327		
FH10	1.73			78			387			SW10	1.61			45			413		
FH11	1.69			59			415			SW11	1.36			57			373		
FH12	1.69			56			400			SW12	1.31			75			320		
FH13	1.61			38			447			SW13	1.20			68			433		
FH14	1.62			58			380			SW14	1.25			59			593		
FH15	1.54			45			313			SW15	1.30			74			400		
FH16	1.67			107			460			SW16	1.40			35			387		
D1		1.67	1.39		59	11		300	453	GH1		1.59	1.43		32	6		227	407
D2		1.54	1.28		10	6		187	480	GH2		1.51	1.47		34	8		213	473
D3		1.89	1.56		21	6		227	467	GH3		1.44	1.36		57	16		207	353
D4		1.42	1.15		50	6		300	367	GH4		1.46	1.23		46	7		267	407
D5	1.27			68			533			GH5	1.36			57			533		
D6	1.51			50			473			GH6	1.44			85			493		
D7	1.45			63			560			GH7	1.54			92			380		
D8	1.62			46			540			GH8	1.53			71			393		
D9	1.39			119			500			GH9	1.57			78			433		
D10	1.39			84			507			GH10	1.61			96			467		
D11	1.40			55			360			GH11	1.38			120			370		
D12	1.38			112			577			GH12	1.56			133			413		
D13	1.39			84			473			GH13	1.55			101			427		
D14	1.52			100			543			GH14	1.62			86			400		
D15	1.61			164			580			GH15	1.49			81			427		
D16	1.39			51			507			GH16	1.33			76			373		

Sample	BD May2013 (gm/cm^3)	BD Jun.2014 (gm/cm^3)	BD Jun.2015 (gm/cm^3)	IR May.2013 (mm/hr)	IR Jun.2014 (mm/hr)	IR Jun.2015 (mm/hr)	PR May2013 (N cm^{-2})	PR Jun.2014 (N cm^{-2})	PR Jun.2015 (N cm^{-2})
MHF1		1.57	1.51		33	11		223	380
MHF2		1.68	1.45		35	8		213	387
MHF3		1.62	1.40		22	8		223	460
MHF4		1.64	1.39		59	6		213	480
MHF5	1.58			80			493		
MHF6	1.57			70			420		
MHF7	1.56			57			497		
MHF8	1.54			84			440		
MHF9	1.55			76			520		
MHF10	1.60			59			360		
MHF11	1.52			9			460		
MHF12	1.63			38			427		
MHF13	1.52			57			453		
MHF14	1.55			57			533		
MHF15	1.54			52			370		
MHF16	1.66			56			360		

APPENDIX C Location of cover crop sampling and chemical analysis

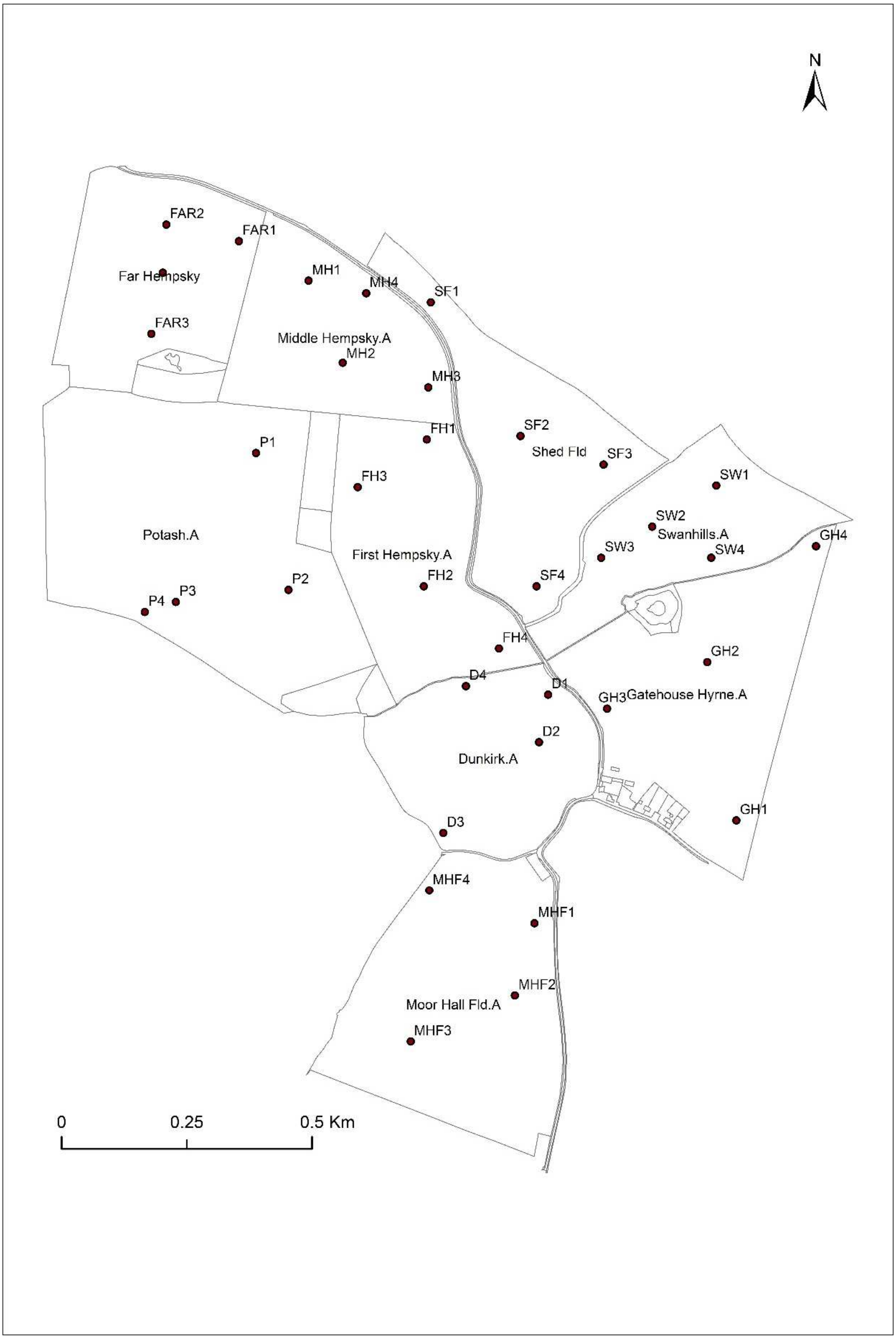

Figure C1 Map of location of sampling Radish cover crop on 22nd January 2014

Figure C2 Photographs of Oilseed radish cover crop sampling in January 2014

Table C1 Chemical analysis of Oilseed Radish cover crop (*Raphanus sativus*) sampled on 22nd January 2014 for leaf (top) and root (bottom)

Sample	Total N	Total P	Total K	Total Mg	Total S	Total C	C:N Ratio	Dry Matter	Total Dry Weight	Total Dry Weight	Total Fresh Weight	Sward height	Cover
name	% w/w	mg/kg	mg/kg	mg/kg	mg/kg	%	:1	%	g	g dry wt m^{-2}	g	(mm)	%
D1	3.61	2734	27960	957	5893	36.9	10.2	9.9	49.9	199.6	506	470	90
D2	3.62	3285	28335	1005	6116	40.4	11.2	8.6	50.5	202	587	280	75
D3	2.76	3767	36442	1110	5861	31.8	11.5	9.8	94.6	378.4	966	360	90
D4	3.17	3116	30133	1154	6179	33.3	10.5	9.2	80.7	322.8	875	315	95
FH1	2.89	3526	34471	1159	8423	39.1	13.5	9.2	47.3	189.2	519	310	80
FH2	2.25	2903	29104	1100	5308	39.8	17.7	10.4	85.6	342.4	822	540	85
FH3	2.13	2984	25979	1104	4922	32.6	15.3	11.4	69.7	278.8	609	305	70
FH4	3.28	4079	30467	1372	7492	36.4	11.1	8	24.6	98.4	307	155	30
GH1	4.11	4858	33946	1191	7013	38.2	9.3	7.3	19.6	78.4	267	175	30
GH2	2.61	2683	24905	876	6128	41.2	15.8	9.8	58	232	592	325	85
GH3	2.73	2798	22355	857	5323	35	12.8	11.2	41	164	365	230	75
GH4	2.54	2723	26507	802	5781	35.5	14	10.8	59.9	239.6	553	285	65
MH1	2.86	3145	28673	973	7096	41.7	14.6	10.4	42.5	170	410	295	30
MH2	2.68	3411	22795	1088	6523	41.5	15.5	6.8	42	168	616	385	40
MH3	3.14	3320	30777	1113	6248	41.8	13.3	9.9	63.5	254	644	335	50
MH4	3.04	3262	27903	1158	7620	39	12.8	9.2	61.6	246.4	670	305	85
MHF1	3.43	3941	30388	1017	7209	35.5	10.4	8.4	29.9	119.6	355	250	40
MHF2	3.3	3391	32130	967	6672	39.5	12	9.3	51.5	206	552	315	60
MHF3	3.25	4027	30130	908	7532	40.5	12.4	9.3	29.4	117.6	318	153	50
MHF4	4.72	3739	30506	971	7820	39.6	8.4	7.5	103	412	1369	470	95
SF1	2.74	3464	27591	879	7161	42.5	15.5	9.1	56.1	224.4	613	455	75
SF2	2.7	3189	27380	940	7162	44.4	16.4	10	51.6	206.4	516	450	40
SF3	2.5	2826	27884	832	5994	43.1	17.3	9.2	45.5	182	497	344	35
SF4	2.57	2659	24164	826	6131	42.5	16.6	9.7	64.4	257.6	666	375	75
SW1	3.52	4799	44972	1143	6566	40.2	11.4	7.5	51.3	205.2	685	365	80
SW2	3.22	3806	37991	1085	7411	41	12.7	7.8	52	208	665	370	70
SW3	2.9	3709	32600	1033	6558	42.3	14.6	7.9	28.3	113.2	360	280	50
SW4	3.5	3651	31802	1098	8542	42.3	12.1	8.8	19.6	78.4	222	240	25

SAMPLE	Total N	Total P	Total K	Total Mg	Total S	Total C	Dry Matter	Total Dry Weight	Total Dry Weight	Total Fresh Weight
NAME	% w/w	mg/kg	mg/kg	mg/kg	mg/kg	%	%	g	g dry wt m^{-2}	g
D1	3.23	3624	36217	1815	6933	39.3	8.6	9	36	105
D2	2.68	3637	27497	1603	5650	41.7	9.9	11.8	47.2	120
D3	2.67	8966	46158	1531	8106	40.5	8.7	10.2	40.8	117
D4	3.09	4680	35580	1663	7906	42.4	8.1	13.6	54.4	169
FH1	1.77	6305	32230	1382	6404	42.2	10.9	20.5	82	189
FH2	1.68	6206	36634	1637	7583	43.3	11.3	17.1	68.4	152
FH3	1.68	6640	35749	1497	7254	40.6	11.5	16.3	65.2	141
FH4	3.01	5607	33664	1805	8653	41.1	9.2	12.4	49.6	135
GH1	3.78	8183	53035	1916	8740	39.1	7.3	9.2	36.8	126
GH2	1.57	6779	32878	1514	6268	42.2	11.6	17.5	70	151
GH3	1.71	7740	34107	1772	6672	41.5	12.4	16.4	65.6	132
GH4	1.9	6698	39930	1292	6658	43	10.5	16.7	66.8	159
MH1	1.46	5950	32317	1419	6012	43.9	12.3	13.9	55.6	113
MH2	1.38	7143	26477	1635	6906	44.7	13.5	18.2	72.8	134
MH3	1.77	5907	33515	1188	6515	42.3	11	29	116	264
MH4	1.74	6183	30676	1309	6458	43.8	11.2	15.8	63.2	141
MHF1	3.05	5748	33405	1390	6608	41.2	10.9	10.5	42	96.6
MHF2	1.96	6334	38954	1560	6804	42.8	10.2	12	48	118
MHF3	2.06	8422	35556	1268	6303	42.8	11.1	14.5	58	130
MHF4	3.92	5855	51330	1842	8097	37.6	6.7	13.6	54.4	203
SF1	1.98	8189	42354	1683	8815	42.9	8.8	10.2	40.8	115
SF2	1.18	7274	30257	1387	4854	45	13.8	23.5	94	170
SF3	1.24	6117	31384	1184	4790	43.2	11.7	16.5	66	141
SF4	1.26	6572	36439	1580	6070	41.4	11.2	30	120	267
SW1	2.26	9772	43939	1523	6657	42.2	8.4	11.6	46.4	138
SW2	1.74	5799	36719	1146	7069	43.2	10.1	18.8	75.2	186
SW3	1.66	7022	34600	1535	6544	43.5	10.8	10.4	41.6	96.8
SW4	3.16	6835	42193	2004	8123	41.7	9.8	10.9	43.6	111

APPENDIX D Drain photographs and drain data

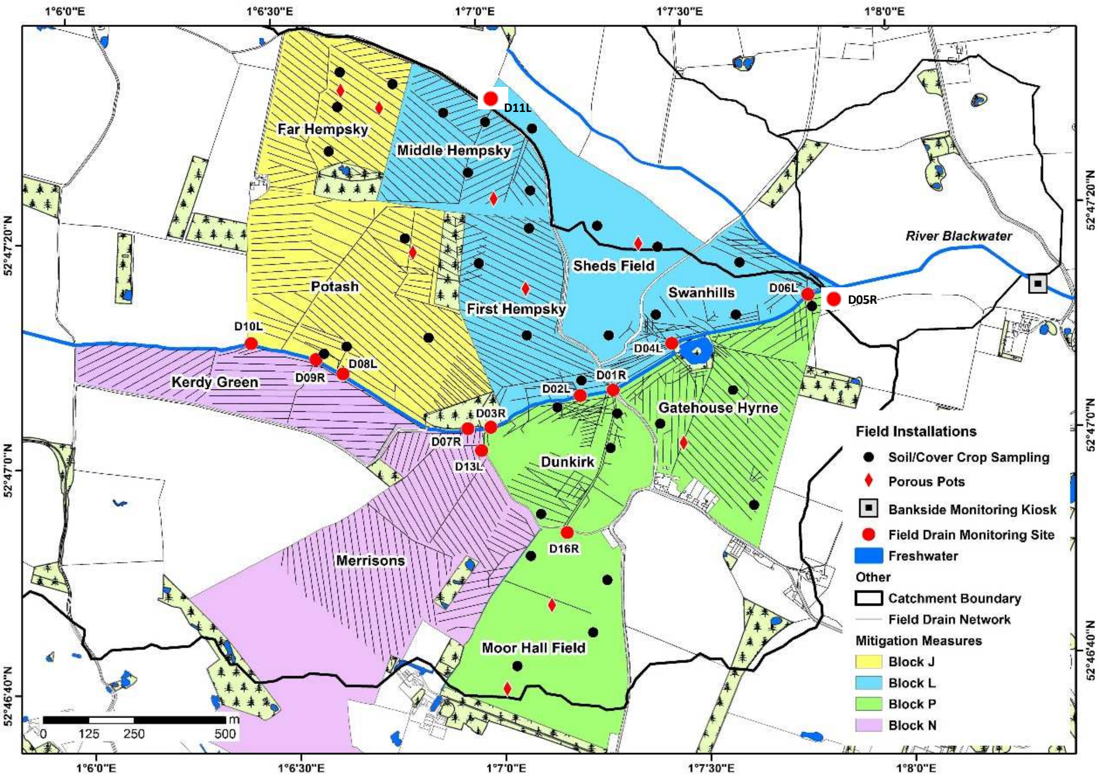

Figure D1 Map of field drain network and locations of sampled drains (indicated by red circles)

Table D1 Coordinates of drains

Drain name	Latitude (N)	Longitude (E)
D1	52.78508	1.119731
D2	52.78497	1.118388
D3	52.78428	1.114691
D4	52.78617	1.122198
D5	52.78732	1.128453
D6	52.78724	1.127815
D7	52.78427	1.113755
D8	52.78577	1.108775
D9	52.78615	1.107704
D10	52.78661	1.105097
D11	52.79221	1.115068
D13	52.78365	1.114363
D16	52.78169	1.117516

Figure D2.1 photos of field drain of D1, D2, D3, and D4

Figure D2.2 photos of field drain of D5, D6, D7, and D8

Figure D2.3 photos of field drain of D9, D10, D11, D13, and D16

Table D2 Drain flow rates data

	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	D11	D13	D16
	Flow ($L\ s^{-1}$)	Flow ($L\ s^{-1}$)	Flow ($L\ s^{-1}$)	Flow ($L\ s^{-1}$)	Flow ($L\ s^{-1}$)	Flow ($L\ s^{-1}$)	Flow ($L\ s^{-1}$)	Flow ($L\ s^{-1}$)	Flow ($L\ s^{-1}$)	Flow ($L\ s^{-1}$)	Flow ($L\ s^{-1}$)	Flow ($L\ s^{-1}$)	Flow ($L\ s^{-1}$)
03/04/2013	0.125	0.036	0.005	0.115	0.081	0.024			0.124	0.075			
09/04/2013	0.192	0.036	0.005	0.118	0.077	0.029	0.013	0.018	0.105	0.071			
16/04/2013	0.145	0.040	0.003	0.114	0.081	0.027	0.005		0.071	0.063			
23/04/2013	0.088	0.035		0.092	0.061	0.016			0.049	0.043			
30/04/2013	0.062	0.037		0.082	0.058	0.010			0.025	0.028			
07/05/2013	0.050	0.039		0.079	0.055	0.014			0.023	0.021			
14/05/2013	0.055	0.044		0.072	0.058	0.014			0.017	0.016			
21/05/2013	0.041	0.047		0.065	0.052				0.007	0.010			
28/05/2013	0.062	0.046		0.113	0.052	0.035	0.054		0.020	0.017			
04/06/2013	0.033	0.030		0.067	0.039	0.008				0.005			
11/06/2013	0.023	0.034		0.050	0.030								
18/06/2013	0.011	0.033		0.034	0.027								
25/06/2013	0.005	0.029		0.023	0.010								
02/07/2013	0.003	0.029		0.017	0.020								
09/07/2013		0.020		0.005	0.015								
16/07/2013		0.015											
23/07/2013		0.014											
30/07/2013		0.014											
06/08/2013		0.022											
13/08/2013		0.025											
20/08/2013		0.029											
27/08/2013	0.005	0.034		0.008									
03/09/2013		0.032		0.007									
10/09/2013		0.033		0.009									
17/09/2013	0.011	0.046		0.023	0.007								
24/09/2013	0.012	0.037		0.018	0.005								
01/10/2013	0.007	0.033		0.013	0.005								
08/10/2013	0.009	0.029		0.010									
15/10/2013	0.024	0.068		0.065									
22/10/2013	0.039	0.108		0.120	0.010	0.024	0.201		0.015	0.008		0.016	
29/10/2013	0.085	0.094	0.007	0.089	0.011	0.031	0.178	0.044	0.238	0.078		0.040	
05/11/2013	0.062	0.071		0.079	0.011	0.027	0.124	0.092	0.150	0.190		0.036	0.040
12/11/2013	0.217	0.101	0.018	0.137	0.022	0.075	0.241	0.283	0.511	0.426		0.260	0.193
19/11/2013	0.152	0.083	0.008	0.101	0.019	0.048	0.106		0.271	0.168		0.101	0.070
26/11/2013	0.248	0.082	0.012	0.105	0.022	0.053	0.147	0.062	0.416	0.256		0.159	0.092
03/12/2013	0.134	0.072		0.088	0.024	0.039	0.054	0.035	0.202	0.111		0.044	0.041
10/12/2013	0.059	0.060		0.070	0.015	0.015	0.009	0.017	0.066	0.057			0.016
17/12/2013	0.043	0.066		0.006	0.017	0.013	0.032		0.066	0.048			0.011
07/01/2014	0.398	0.078	0.049	0.156	0.030	0.036	0.623	0.701	0.729	0.402	0.279	0.595	0.246
14/01/2014	0.585	0.077	0.042	0.174	0.065	0.078	0.470	0.461	0.628	0.471	0.279	0.595	0.333
21/01/2014	0.679	0.077	0.039	0.183	0.082	0.100	0.393	0.342	0.578	0.506	0.279	0.594	0.376
28/01/2014	0.773	0.076	0.035	0.193	0.100	0.121	0.316	0.222	0.528	0.541		0.594	0.420
04/02/2014	0.711	0.088	0.028	0.182	0.085	0.112	0.199	0.119	0.558	0.520	0.139	0.324	0.469

	D1 Flow (L s^{-1})	D2 Flow (L s^{-1})	D3 Flow (L s^{-1})	D4 Flow (L s^{-1})	D5 Flow (L s^{-1})	D6 Flow (L s^{-1})	D7 Flow (L s^{-1})	D8 Flow (L s^{-1})	D9 Flow (L s^{-1})	D10 Flow (L s^{-1})	D11 Flow (L s^{-1})	D13 Flow (L s^{-1})	D16 Flow (L s^{-1})
11/02/2014	1.055	0.085	0.060	0.309	0.125	0.243	0.450	0.196	0.710	0.682	0.286	0.733	0.781
18/02/2014	0.691	0.082	0.048	0.199	0.173	0.119	0.221	0.104	0.462	0.424	0.130	0.267	0.358
25/02/2014	0.427	0.081	0.017	0.188	0.106	0.057	0.064	0.053	0.250	0.195	0.028	0.102	0.219
04/03/2014	0.487	0.078	0.022	0.197	0.112	0.066	0.160	0.083	0.251	0.179	0.058	0.253	0.248
11/03/2014	0.221	0.070	0.033	0.115	0.092	0.031	0.009	0.028	0.126	0.137		0.009	0.056
18/03/2014	0.192	0.071	0.020	0.111	0.097	0.027	0.005	0.032	0.064	0.115			0.052
01/04/2014	0.268	0.074	0.029	0.136	0.088	0.033	0.048	0.041	0.040	0.143			0.058
15/04/2014	0.162	0.072	0.029	0.107	0.061	0.021	0.048	0.041	0.040	0.072			0.034
29/04/2014	0.057	0.070		0.078	0.034	0.008				0.001			0.010
13/05/2014	0.080	0.093		0.088	0.045	0.013		0.055		0.212			0.012
27/05/2014	0.393	0.083	0.026	0.073	0.029	0.022		0.176		0.693			0.050
10/06/2014	0.277	0.085	0.012	0.073	0.022	0.013	0.010	0.036		0.243			0.025
24/06/2014	0.084	0.054		0.033	0.019					0.022			
08/07/2014	0.042	0.048		0.018	0.017					0.007			
22/07/2014	0.059	0.045		0.016	0.016					0.010			
05/08/2014	0.020	0.034			0.011								
19/08/2014	0.029	0.042		0.018									
02/09/2014	0.031	0.053		0.023									
16/09/2014	0.031	0.066		0.021									
30/09/2014	0.032	0.080		0.018									
07/10/2014	0.047	0.060		0.035	0.005								
14/10/2014	1.508		0.177	0.078	0.045	0.127	1.119	0.438				0.503	0.057
21/10/2014	0.377	0.124	0.016	0.042	0.010	0.032	0.085	0.042	0.291			0.050	0.012
28/10/2014	0.209	0.099	0.005	0.042	0.008	0.029	0.037		0.209				
04/11/2014	0.137	0.090		0.025	0.011	0.013			0.090				
11/11/2014	0.252	0.116	0.020	0.043	0.016	0.058	0.135	0.066	0.380	0.022			0.024
18/11/2014	1.170	0.159	0.118	0.058	0.033	0.176	0.472	0.260	0.855	0.336	0.092	0.460	0.145
25/11/2014	2.351	0.170	0.207	0.058	0.046	0.234	0.567	0.247		0.827	0.442	0.713	
02/12/2014	0.719	0.130	0.066	0.047	0.044	0.088	0.132	0.078	0.491	0.386	0.090	0.213	0.181
09/12/2014	0.271	0.117	0.019	0.024	0.038	0.042	0.042	0.034	0.190	0.149		0.028	0.039
16/12/2014	0.416	0.119	0.026	0.031	0.044	0.048	0.078	0.066	0.238	0.205	0.020	0.063	0.060
06/01/2015	0.450	0.119	0.042	0.034	0.065	0.059	0.086	0.070	0.293	0.251	0.026	0.119	0.178
13/01/2015	1.009	0.188	0.213	0.053	0.099	0.151	0.590	0.431	0.602	0.487	0.287	0.808	
20/01/2015	0.450	0.127	0.032	0.042	0.067	0.056	0.065	0.044	0.291	0.222		0.091	0.116
27/01/2015	0.477	0.124	0.046	0.043	0.073	0.061	0.109	0.077	0.288	0.230		0.133	0.118
03/02/2015	1.138	0.150	0.116	0.065	0.086	0.132	0.260	0.186	0.658	0.513	0.222	0.405	
10/02/2015	0.536	0.126	0.038	0.065	0.068	0.064	0.085	0.073	0.290	0.202	0.032	0.129	
17/02/2015	0.222	0.102	0.009	0.033	0.065	0.024		0.015	0.158	0.111			
24/02/2015	0.666	0.118	0.059	0.058	0.071	0.069	0.162	0.124	0.326	0.284	0.050	0.205	0.185
03/03/2015	0.414	0.112	0.028	0.079	0.057	0.078	0.075	0.089	0.234	0.195		0.100	0.112
10/03/2015	0.197	0.097	0.006	0.032	0.067	0.023	0.008		0.125	0.093			0.028
17/03/2015	0.184	0.079		0.024	0.046	0.016			0.059	0.065			0.016
24/03/2015	0.154	0.078		0.029	0.043	0.010			0.035	0.065			0.013
31/03/2015	0.202	0.078	0.025	0.045	0.068	0.032	0.085	0.119	0.057	0.174		0.054	

Table D3 Dissolved nitrate concentration in drains

Date	D1 NO$_3$ (mg N/L)	D2 NO$_3$ (mg N/L)	D3 NO$_3$ (mg N/L)	D4 NO$_3$ (mg N/L)	D5 NO$_3$ (mg N/L)	D6 NO$_3$ (mg N/L)	D7 NO$_3$ (mg N/L)	D8 NO$_3$ (mg N/L)	D9 NO$_3$ (mg N/L)	D10 NO$_3$ (mg N/L)	D11 NO$_3$ (mg N/L)	D13 NO$_3$ (mg N/L)	D16 NO$_3$ (mg N/L)
03/04/2013	8.99	2.33	5.79	3.22	10.71	6.99			11.81	12.12			
09/04/2013	8.75	2.26	5.18	3.31	10.87	6.91	6.91	3.43	11.61	12.15			
16/04/2013	8.44	1.96	5.38	3.11	11.20	7.32	2.32		12.30	11.59			
23/04/2013	8.61	1.88		3.59	11.67	8.28			13.27	11.26			
30/04/2013	7.59	1.58		3.26	11.39	7.89			13.05	9.88			
07/05/2013	7.34	1.56		3.16	11.20	6.78			13.04	9.36			
14/05/2013	7.24	1.81		3.75	11.34	6.93			12.85	8.13			
21/05/2013	5.34	1.22		2.89	10.32				11.68	6.13			
28/05/2013	5.10	2.13		3.92	11.10	7.01	8.29		11.95	6.31			
04/06/2013	4.35	1.35		3.68	10.04	6.62				4.39			
11/06/2013	3.66	0.86		3.48	9.35								
18/06/2013	2.43	0.80		3.92	8.81								
25/06/2013	1.32	0.46		4.27	8.65								
02/07/2013	1.28	0.48		4.96	7.78								
09/07/2013		0.48		6.76	7.25								
16/07/2013		0.44											
23/07/2013		0.37											
30/07/2013		0.30											
06/08/2013		0.30											
13/08/2013		0.36											
20/08/2013		0.35											
27/08/2013		0.97		0.40	0.50								
03/09/2013		0.42		0.47									
10/09/2013		0.33		0.41									
17/09/2013	1.12	1.37		1.40	4.07								
24/09/2013	0.87	0.62		0.65	3.61								
01/10/2013	0.88	0.68		0.62	3.22								
08/10/2013	0.87	0.70		0.61									
15/10/2013	3.37	3.55		1.72	7.90								

Date	D1 NO_3 (mg N/L)	D2 NO_3 (mg N/L)	D3 NO_3 (mg N/L)	D4 NO_3 (mg N/L)	D5 NO_3 (mg N/L)	D6 NO_3 (mg N/L)	D7 NO_3 (mg N/L)	D8 NO_3 (mg N/L)	D9 NO_3 (mg N/L)	D10 NO_3 (mg N/L)	D11 NO_3 (mg N/L)	D13 NO_3 (mg N/L)	D16 NO_3 (mg N/L)
22/10/2013	2.17	2.66		1.37	6.94	5.00	11.85		6.21	14.04		19.88	
31/10/2013	7.60	1.85	2.50	1.16	8.79	4.71	12.95	19.08	6.13	11.98		17.72	
05/11/2013	5.93	1.23		1.04	8.82	3.70	12.03	19.09	6.09	12.03		17.09	6.62
12/11/2013	8.37	0.87	1.32	0.90	9.83	4.39	12.25	19.51	4.15	14.33		19.47	4.17
19/11/2013	7.22	0.91	0.49	1.14	10.50	4.37	11.75		4.55	12.92		18.00	3.32
26/11/2013	5.90	0.70	0.94	1.45	12.26	3.66	12.51	17.89	3.96	13.87		18.66	2.75
02/12/2013	5.05	0.60		1.87	12.27	3.22	11.70	17.11	4.35	12.64		18.12	2.80
10/12/2013	4.18	0.42		1.84	12.28	3.01	10.53	16.93	4.61	10.19			2.63
17/12/2013	3.60	0.46		2.11	11.29	2.86	8.96		4.26	8.68			3.77
07/01/2014	3.85	0.58	2.40	1.57	13.98	2.65	7.48	15.34	2.32	13.52	4.57	16.33	1.48
14/01/2014	3.12	0.40	2.07	1.29	13.38	1.62	7.15	16.81	1.96	13.92	3.25	14.95	1.53
21/01/2014	3.49	0.55	1.76	1.72	12.44	1.96	6.88	16.47	1.73	13.96	3.76	13.98	2.66
29/01/2014	3.60	1.04	3.18	1.91	12.60	2.10	6.00	14.33	1.47	14.25		12.76	2.95
04/02/2014	3.51	0.53	2.79	2.02	12.80	2.27	5.12	14.20	1.13	13.82	3.76	11.15	2.88
11/02/2014	3.41	0.59	3.17	2.33	12.70	2.84	4.17	11.46	0.94	12.50	3.73	9.37	3.08
18/02/2014	3.79	1.24	2.83	2.49	14.48	3.42	3.95	12.43	0.90	13.14	3.92	8.42	3.56
25/02/2014	3.77	0.83	1.93	2.31	13.72	2.57	3.25	11.48	0.80	12.93	4.43	7.94	3.70
04/03/2014	3.87	0.96	1.99	1.92	13.87	2.61	3.17	10.60	0.43	12.13	4.14	6.88	3.29
11/03/2014	3.71	1.04	2.78	2.42	13.10	3.12	2.35	12.21	1.39	12.40		5.20	3.89
18/03/2014	4.06	1.33	3.35	2.48	13.74	3.22	1.12	11.28	1.52	12.29	4.04		
31/03/2014	3.99	1.29	3.37	2.17	13.92	3.19	2.00	10.90	2.02	11.78			3.85
15/04/2014	3.53	1.46		2.63	12.82	3.97				10.11			5.07
29/04/2014	3.06	1.03		1.99	11.64	3.79				9.77			4.87
13/05/2014	2.73	2.08		1.85	10.58	3.03		11.85		16.78			6.08
27/05/2014	3.88	1.50	3.33	2.39	9.68	3.55		11.24		37.42			5.37
10/06/2014	5.22	1.89	3.41	3.25	11.45	1.14	5.88	8.46		24.00			6.99
24/06/2014	3.57	0.56		1.70	8.42					14.34			
08/07/2014	2.97	0.35		1.51	7.63					9.69			
22/07/2014	3.08	0.35		1.19	8.01					6.78			

Date	D1 NO_3 (mg N/L)	D2 NO_3 (mg N/L)	D3 NO_3 (mg N/L)	D4 NO_3 (mg N/L)	D5 NO_3 (mg N/L)	D6 NO_3 (mg N/L)	D7 NO_3 (mg N/L)	D8 NO_3 (mg N/L)	D9 NO_3 (mg N/L)	D10 NO_3 (mg N/L)	D11 NO_3 (mg N/L)	D13 NO_3 (mg N/L)	D16 NO_3 (mg N/L)
06/08/2014	2.66	0.13			4.99								
19/08/2014	2.40	0.27		0.33									
02/09/2014	1.78	0.33		0.24									
16/09/2014	1.68	0.37		0.22									
30/09/2014	1.38	0.27		0.19									
07/10/2014	1.66	0.36		0.23	3.57								
14/10/2014	11.08		11.40	15.91	8.74	10.21	10.84	21.68				10.74	10.17
21/10/2014	6.85	3.70	6.04	4.06	7.44	6.52	6.51	9.74	19.99			8.16	6.70
28/10/2014	5.73	3.36	5.69	3.31	7.57	6.17	5.79		16.58				
04/11/2014	5.03	2.33		2.67	4.76	5.64			15.10	7.21			
11/11/2014	6.38	4.99	5.90	4.47	6.61	8.02	4.90	8.48	15.76	3.24			7.24
18/11/2014	9.43	10.91	7.46	8.10	8.84	12.93	4.82	7.07	20.13	3.82	7.65	7.25	8.81
25/11/2014	10.73	12.00	7.26	7.81	12.32	14.11	4.00	5.60		5.80	8.96	5.73	
01/12/2014	9.41	7.75	7.18	5.60	14.56	13.30	4.16	6.14	15.83	4.64	7.64	6.22	8.07
09/12/2014	7.79	6.28	6.43	4.54	14.84	11.60	3.87	5.69	15.70	3.79	7.01	6.20	6.44
16/12/2014	7.97	6.88	6.66	4.47	15.98	11.98	3.35	5.05	15.22	4.21	7.40	6.21	5.68
06/01/2015	7.33	7.95	6.73	4.55	15.99	12.97	2.70	5.52	15.14	4.20	7.94	5.77	5.38
13/01/2015	6.98	10.36	6.25	4.10	14.11	12.01	2.22	1.48	12.51	4.54	8.08	5.48	
20/01/2015	6.63	7.46	5.72	3.93	14.76	11.81	2.41	2.15	12.14	3.37		4.69	3.80
26/01/2015	6.47	7.89	6.04	4.18	14.62	11.89	2.50	2.08	12.12	3.95	5.54	3.25	
03/02/2015	6.17	8.22	5.00	4.33	14.39	11.08	2.02	1.22	10.04	3.03	7.65	5.17	
10/02/2015	5.49	7.07	5.15	3.93	14.24	11.23	2.00	1.29	9.26	2.42	7.33	4.61	2.70
17/02/2015	4.17	5.33	4.49	2.83	13.55	10.30		1.15	8.74	2.00			2.44
24/02/2015	4.75	4.33	4.35	3.52	13.85	9.73	1.63	0.91	6.99	2.49	6.97	4.54	1.82
03/03/2015	4.23	5.26	4.22	3.14	13.92	8.99	1.48	0.89	6.19	2.17		4.12	1.64
10/03/2015	3.58	4.23	3.60	2.44	13.23	7.81	1.56		6.46	1.61			2.03
17/03/2015	3.23	3.68		2.31	12.87	7.03			6.66	1.41			2.00
24/03/2015	3.11	3.22		2.46	12.30	6.32			6.72	1.36			1.89
31/03/2015	3.41	3.20	4.04	2.21	12.96	5.70	3.08	4.62	6.05	7.21		13.06	

Table D4 Ammonium concentration in drains

Date	D1 NH_4 (µg N/L)	D2 NH_4 (µg N/L)	D3 NH_4 (µg N/L)	D4 NH_4 (µg N/L)	D5 NH_4 (µg N/L)	D6 NH_4 (µg N/L)	D7 NH_4 (µg N/L)	D8 NH_4 (µg N/L)	D9 NH_4 (µg N/L)	D10 NH_4 (µg N/L)	D11 NH_4 (µg N/L)	D13 NH_4 (µg N/L)	D16 NH_4 (µg N/L)
03/04/2013	0		1	1	1	11			4				
09/04/2013	7	5	6	7	5	15	6	6	5	5			
16/04/2013	6	5	8	7	19	17	12		6	9			
23/04/2013	7	5		8	7	17			2	3			
30/04/2013	4	5		6	10	21			6	4			
07/05/2013	4	3		9	3	9			5	4			
14/05/2013	4	3		5	6	9			4	19			
21/05/2013		0		0	2				2	5			
28/05/2013	1	1		2	1	3	0		3	1			
04/06/2013	7	12		5	7	7				7			
11/06/2013	7	6		4	5								
18/06/2013	4	7		4	3								
25/06/2013	1	1		1	1								
02/07/2013	12	6		5	6								
09/07/2013		8		7	8								
16/07/2013		8											
23/07/2013		6											
30/07/2013		8											
06/08/2013		5											
13/08/2013		8											
20/08/2013		5											
27/08/2013		21		12	11								
03/09/2013		19		18									
10/09/2013		10		11									
17/09/2013	6	10		12	9								
24/09/2013	16	12		16	18								
01/10/2013	6	4		3	6								
08/10/2013	6	5		5									
15/10/2013	5	4		6	7								

Date	D1 NH₄ (µg N/L)	D2 NH₄ (µg N/L)	D3 NH₄ (µg N/L)	D4 NH₄ (µg N/L)	D5 NH₄ (µg N/L)	D6 NH₄ (µg N/L)	D7 NH₄ (µg N/L)	D8 NH₄ (µg N/L)	D9 NH₄ (µg N/L)	D10 NH₄ (µg N/L)	D11 NH₄ (µg N/L)	D13 NH₄ (µg N/L)	D16 NH₄ (µg N/L)
22/10/2013	12	10		11	9	11	11		14	9		9	
31/10/2013	2	2	0			0	1	3	4	4		0	
05/11/2013	0	0		1	4	2	3	0	2	1			1
12/11/2013	6	10	3	6	5	6	7	9	5	6		5	4
19/11/2013	2	2	1	2	2	4	4		2	2		2	3
26/11/2013	12	13	10	13	15	13	16	12	13	11		11	12
02/12/2013	0	0		0	0	0	0	0	0	0		0	0
10/12/2013	11	13		11	11	12	12	18	13	11			20
17/12/2013	7	9		5	4	9	6		9	13			13
07/01/2014			14					11		0		14	1
14/01/2014	15	16	15	20	18	16	15	16	16	15	15	15	17
21/01/2014	34	29	36	22	13	25	17	15	18	14	47	15	81
29/01/2014	19	9	16	16	10	20	9	8	10	7		6	28
04/02/2014	20	17	21	19	20	22	17	17	18	16	24	16	39
11/02/2014	16	15	21	17	17	19	17	17	17	17	21	15	35
18/02/2014	11	8	14	10	9	10	11	9	13	11	12	11	40
25/02/2014	9	6	9	7	9	8	6	4	7	7	5	5	24
04/03/2014	21	18	19	21	19	20	17	18	23	69	23	20	27
11/03/2014													
18/03/2014	20	21	20	21	21	21	21	21	21	20	22		
31/03/2014	15	15	22	14	14	14	14	15	16	15			12
15/04/2014	0	0		0	0	0				0			0
29/04/2014	4	8		7	8	13				6			6
13/05/2014	24	23		24	24	23		27		23			24
27/05/2014										174			
10/06/2014	0	0	17		1								
24/06/2014	0	0		0	0					0			
08/07/2014	0	0		0	0					0			
22/07/2014					3					94			

Date	D1 NH_4 (µg N/L)	D2 NH_4 (µg N/L)	D3 NH_4 (µg N/L)	D4 NH_4 (µg N/L)	D5 NH_4 (µg N/L)	D6 NH_4 (µg N/L)	D7 NH_4 (µg N/L)	D8 NH_4 (µg N/L)	D9 NH_4 (µg N/L)	D10 NH_4 (µg N/L)	D11 NH_4 (µg N/L)	D13 NH_4 (µg N/L)	D16 NH_4 (µg N/L)
06/08/2014	44				2								
19/08/2014	3	8											
02/09/2014	6												
16/09/2014	24	10		18									
30/09/2014													
07/10/2014													
14/10/2014	6		23	5	6	5		11				7	12
21/10/2014		1		1		2							7
28/10/2014	2	3	1	3	4	4	5		4				
04/11/2014	2	0			20					0			
11/11/2014	17	4	7	0	26	6	3		3	4			4
18/11/2014			5		3	1	0	5	0	8			
25/11/2014					6								
01/12/2014													
09/12/2014	5	5	3	18	4	3	5	6	2	191	3	4	
16/12/2014			0		9								
06/01/2015		0	16		8		0	1	0	2	1		0
13/01/2015		0	18		6			6			0	12	
20/01/2015			1										
26/01/2015			15		4								
03/02/2015													
10/02/2015			2										
17/02/2015	6	5	25	5	11	8		5	5	7			6
24/02/2015			14	7	13	18	1			2	0		1
03/03/2015	10	3	25	8	7	91	36	8	3	1		0	2
10/03/2015	14	0	3			1	11		16	37			3
17/03/2015	3	3		3	3	7			2	3			5
24/03/2015	1	2		0	6	2			0	1			1
31/03/2015	15	37	13	14	33	8	29	181	17	119		305	

Table D5 Nitrite concentration in drains

	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	D11	D13	D16
Date	NO_2	NO_2	NO_2	NO_2	NO_2	NO_2	NO_2	NO_2	NO_2	NO_2	NO_2	NO_2	NO_2
	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)
03/04/2013	0.8	0.6	5.4	0.7	0.5	12.4			0.1	3.9			
09/04/2013	1.0	0.8	3.4	0.9	0.9	7.5	1.4	2.2	1.0	3.1			
16/04/2013	0.8	0.6	1.0	0.6	0.7	3.5	2.6		1.2	2.7			
23/04/2013	2.4	1.4		2.0	1.5	3.6			0.9	3.8			
30/04/2013	0.6	0.2		0.8	0.9	2.2			0.5	4.1			
07/05/2013	0.4	2.4		0.5	0.7	2.0			0.1	5.6			
14/05/2013	0.9	0.2		0.3	0.6	1.2			1.2	4.2			
21/05/2013	0.1				1.1					3.1			
28/05/2013				0.1	0.2	1.2	0.6			2.3			
04/06/2013	0.5	0.6		6.0	9.2	0.7				2.4			
11/06/2013	0.3				42.3								
18/06/2013	0.8	0.0		0.6	0.4								
25/06/2013	0.9				0.8								
02/07/2013		2.1		4.2									
09/07/2013				0.2									
16/07/2013		0.7											
23/07/2013		0.5											
30/07/2013		5.6											
06/08/2013		3.1											
13/08/2013		1.6											
20/08/2013		3.0											
27/08/2013		18.9		9.0	7.7								
03/09/2013		7.5		7.1									
10/09/2013		0.6		0.6									
17/09/2013	1.0	0.7		1.0	0.8								
24/09/2013	1.2	0.8		1.1	1.2								
01/10/2013	1.6	1.2		1.3	1.5								
08/10/2013	2.2	2.1		1.7									
15/10/2013	1.9	1.8		1.4	2.1								

Date	D1 NO_2 (µg N/L)	D2 NO_2 (µg N/L)	D3 NO_2 (µg N/L)	D4 NO_2 (µg N/L)	D5 NO_2 (µg N/L)	D6 NO_2 (µg N/L)	D7 NO_2 (µg N/L)	D8 NO_2 (µg N/L)	D9 NO_2 (µg N/L)	D10 NO_2 (µg N/L)	D11 NO_2 (µg N/L)	D13 NO_2 (µg N/L)	D16 NO_2 (µg N/L)
22/10/2013	1.5	1.7		1.6	1.6	1.3	1.2		1.7	1.2		1.1	
31/10/2013	1.2	1.4	1.0	1.3	2.3	1.2	1.2	1.1	1.1	1.3		2.8	
05/11/2013	2.2	2.5		2.2	2.9	1.9	3.1	1.8	1.7	1.6		1.6	1.7
12/11/2013	2.4	2.4	1.5	2.0	1.5	2.1	1.7	2.5	1.4	1.6		1.5	1.9
19/11/2013	2.0	1.8	2.1	2.1	1.6	2.1	3.1		2.1	4.3		2.1	2.3
26/11/2013	2.4	1.4	1.5	1.5	1.3	1.5	1.2	1.5	1.7	1.6		1.4	1.6
02/12/2013	1.2	1.4		1.0	1.1	1.3	1.1	1.2	1.1	1.6		0.9	1.7
10/12/2013	3.5	3.3		3.7	9.5	3.4	3.7	3.6	3.4	3.5			4.7
17/12/2013	1.5	1.7		1.6	2.6	1.0	0.9		1.0	1.0			2.4
07/01/2014			2.0		0.0	0.3	0.6	1.2	0.2	0.3	1.1	2.2	2.9
14/01/2014	5.8	5.9	6.2	5.7	6.6	6.2	5.8	6.2	5.9	5.9	6.6	6.6	7.2
21/01/2014	6.1	6.3	6.6	6.3	8.5	6.2	21.0	6.2	6.4	6.5	6.5	6.7	8.0
29/01/2014	0.1		0.7			2.5			5.1				2.9
04/02/2014	5.3	4.8	5.0	4.9	4.7	4.9	4.6	4.9	4.7	4.8	5.3	5.0	6.1
11/02/2014	2.0	2.1	2.2	1.4	1.8	1.6	2.0	1.4	1.4	2.1	0.9	0.5	2.3
18/02/2014	2.3	1.9	2.5	1.9	2.1	2.2	2.3	2.2	2.3	2.3	2.5	2.3	3.1
25/02/2014	1.4	1.7	1.3	1.3	1.8	1.3	1.3	1.0	35.0	1.4	1.5	2.6	6.2
04/03/2014	0.9	0.7	0.9	0.7	0.9	0.9	2.8	0.7	10.3	1.5	1.1	4.8	7.3
11/03/2014	1.2	1.0	2.9	1.7	1.6	1.4	1.3	1.9	3.2	1.9		3.1	2.8
18/03/2014	11.1	11.1	13.1	11.9	11.0	14.0	10.7	11.0	14.9	12.4	13.3		
31/03/2014	8.0	8.1	11.7	8.8	7.6	8.2	9.5	9.3	8.3	8.2			7.9
15/04/2014	0.8	1.0		0.3	1.1	0.0				0.0			0.5
29/04/2014	10.3	4.9		5.4	4.3	6.1				6.6			7.0
13/05/2014	4.6	4.6		4.8	4.9	4.7		5.0		7.5			4.9
27/05/2014	1.7	0.8	1.4	1.0	3.0	1.1		3.7		42.8			1.9
10/06/2014	1.7	0.5	1.3	2.1	2.4	1.8	1.7	2.1		14.2			1.5
24/06/2014	2.9	2.1		2.1	2.4					7.8			
08/07/2014	1.0	0.0		0.0	1.2					0.0			
22/07/2014	0.5	0.3		0.2	1.0					1.2			

Date	D1 NO_2 (µg N/L)	D2 NO_2 (µg N/L)	D3 NO_2 (µg N/L)	D4 NO_2 (µg N/L)	D5 NO_2 (µg N/L)	D6 NO_2 (µg N/L)	D7 NO_2 (µg N/L)	D8 NO_2 (µg N/L)	D9 NO_2 (µg N/L)	D10 NO_2 (µg N/L)	D11 NO_2 (µg N/L)	D13 NO_2 (µg N/L)	D16 NO_2 (µg N/L)
06/08/2014	6.3												
19/08/2014													
02/09/2014													
16/09/2014	1.6	1.9		1.5									
30/09/2014													
07/10/2014					1.5								
14/10/2014	7.9		4.5	1.3	1.2	7.6	19.7	9.7				5.4	3.6
21/10/2014	3.1	3.5	4.0	4.4	10.4	4.2	5.1	5.4	7.2			4.3	6.8
28/10/2014	3.0	2.6	2.9	3.1	2.6	3.6	4.4		2.1				
04/11/2014	1.3	4.0		4.0	7.1	2.4			1.8	2.9			
11/11/2014	0.7		0.7	8.3	1.3		0.4	0.6					
18/11/2014	4.4	6.5	7.0	1.9	6.4	3.7	6.0	3.1	2.7	2.2	2.3	0.8	8.7
25/11/2014	3.9	2.6	3.3	2.6	4.2	4.2	1.8	0.9		0.5	0.3	1.0	
01/12/2014	1.2	1.9	3.4	1.7	3.9	2.6	1.2	1.6	7.9	1.5	1.0	1.5	0.7
09/12/2014	2.1	5.8	3.1	5.3	5.6	2.2	1.7	2.8	1.5	1.5	84.2	1.1	0.4
16/12/2014	1.8	1.5	4.2	5.4	4.5	2.9	2.3	3.2	7.3	3.2	2.4	3.8	2.5
06/01/2015	1.0	1.2	2.9	0.5	3.2	0.7	0.3	1.7	2.5	0.9	0.8	0.9	2.9
13/01/2015	1.1	0.9	2.0	1.0	2.9	1.6	1.7	6.3	1.2	1.2	0.9	13.0	
20/01/2015	0.9	2.0	2.7	1.3	2.9	1.7	1.1	1.5	0.9	2.6		0.2	1.4
26/01/2015	3.1	2.4	4.2		3.3	0.9	6.0	2.8	2.8	1.4	1.0	1.2	
03/02/2015	0.9	1.0	1.2	3.7	24.2	1.0	0.7	0.9	0.7	0.2	0.6	0.7	
10/02/2015	1.1	1.4	1.8	1.5	5.1	1.1	0.8	1.3	0.8	2.9	1.7	2.0	1.2
17/02/2015	3.7	3.1	4.0	2.4	3.6	2.6		2.8	2.5	3.5			2.5
24/02/2015	4.1	2.5	4.7	2.7	4.8	5.0	2.4	2.8	2.8	2.5	2.5	2.9	3.0
03/03/2015	5.2	4.7	7.7	4.2	10.0	6.4	4.0	5.3	4.7	3.9		4.6	6.8
10/03/2015	3.0	1.4	8.8	6.2	11.8	5.4	5.1		5.7	5.2			5.9
17/03/2015	16.5	17.1		15.9	20.0	18.4			17.0	15.3			18.1
24/03/2015	15.8	14.7		14.9	27.9	18.8			15.9	15.9			17.5
31/03/2015	22.0	21.7	29.5	23.0	59.2	21.2	68.7	149.0	24.5	182.0		67.6	

Table D6 pH values of drain samples

	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	D11	D13	D16
Date	pH	pH	pH	pH	pH	pH	pH	pH	pH	pH	pH	pH	pH
03/04/2013	7.59	7.75	7.81	7.53	7.49	7.65			7.86	7.85			
09/04/2013	7.65	7.76	7.93	7.77	7.76	7.88	7.96	7.95	8.11	8.09			
16/04/2013	8.12	8.12	8.23	8.03	7.97	8.09	8.24		8.08	8.16			
23/04/2013	7.99	7.98		7.96	7.95	8			8	8.15			
30/04/2013	7.6	7.58		7.71	7.71	7.92			7.93	8.15			
07/05/2013	7.64	7.73		7.74	7.59	7.79			7.89	8.04			
14/05/2013	7.84	7.8		7.79	7.74	7.72			8.07	8.08			
21/05/2013	7.83	7.86		7.79	7.81				8.1	8.12			
28/05/2013	7.68	7.73		7.73	7.63	7.72	7.84		8	7.99			
04/06/2013	7.73	7.69		7.72	7.77	7.94				7.95			
11/06/2013	7.81	7.6		7.52	7.59								
18/06/2013	7.84	7.79		7.74	7.85								
25/06/2013	8.12	8		7.93	7.91								
02/07/2013	8.03	7.79		7.79	7.76								
09/07/2013		8.08		8.39	3.71								
16/07/2013		8.29											
23/07/2013		7.85											
30/07/2013		8.03											
06/08/2013		7.63											
13/08/2013		7.61											
20/08/2013		8.15											
27/08/2013		8.03		7.83	7.87								
03/09/2013		8.13		8.14									
10/09/2013		8.23		8.13									
17/09/2013	8	8.01		7.94	8								
24/09/2013	7.99	7.9		7.9	8.02								
01/10/2013	8.12	8.03		8.01	8.13								
08/10/2013	8.04	7.95		7.98									
15/10/2013	7.93	7.81		7.83	7.92								
22/10/2013	7.95	7.87		7.85	7.91	8.06	8.13		8.22	8.18		8.18	
31/10/2013	7.94	7.93	8.31	7.93	7.99	8.02	8.12	7.91	8.14	8.15		8.18	
05/11/2013	8.16	8.23		8.13	8.41	8.27	8.22	8.25	8.42	8.24		8.27	8.17
12/11/2013	7.95	7.96	7.86	7.91	8.25	8.09	8.07	8.06	8.15	8.04		8.05	8.04
19/11/2013	8.09	8.12	8.11	8.12	8.25	8.19	8.6		8.11	8.16		8.11	8.09
26/11/2013	7.83	7.93	8.07	7.84	7.93	8.11	8.06	7.99	8.06	8.13		8.09	8.07
02/12/2013	7.59	7.52		7.64	7.59	7.7	7.66	7.7	7.66	7.81		7.87	7.98
10/12/2013	8.08	8		7.98	7.94	8.14	8.1	8.09	8.02	8.13			8.06
17/12/2013	8	7.91		7.89	8.09	8.04	7.75		7.94	7.9			7.91
07/01/2014	7.81	7.9	7.64	8	8.17	7.9	7.66	7.92	7.94	7.75	7.99	8.07	8.28
14/01/2014	7.75	7.78	7.84	7.74	7.66	7.68	7.74	7.86	7.71	7.92	7.87	7.92	7.66
21/01/2014	8.01	8.11	8.38	8.02	8.03	8.05	8.21	8.21	8.15	8.29	8.2	8.11	8.3
29/01/2014	7.14	6.66	6.22	5.56	7.08	7.07	7.69	7.48	5.25	4.86		6.07	4.92

	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	D11	D13	D16
Date	pH	pH	pH	pH	pH	pH	pH	pH	pH	pH	pH	pH	pH
04/02/2014	7.7	7.69	7.97	7.66	7.53	7.72	7.93	7.77	7.81	7.8	7.75	7.8	7.96
11/02/2014	7.81	7.84	7.83	7.95	8.02	8.18	8.14	7.9	8.05	7.94	7.98	7.76	8.02
18/02/2014	8.06	8.06	8.11	7.96	7.98	8	8.18	8.1	8.12	8.12	8.01	8.07	8.22
25/02/2014	7.91	7.97	8.19	7.86	7.85	8	8.26	8.07	8.14	8.13	8.06	8.14	8.26
04/03/2014	8.13	8.1	8.06	8.12	7.88	8.07	8.04	8.25	8.17	8.15	8.05	8.13	8.11
11/03/2014	7.67	7.67	7.72	7.63	7.6	7.77	7.9	8.01	7.86	8.12		8.37	8.24
18/03/2014	7.49	7.52	7.66	7.48	7.41	7.6	7.83	7.79	7.67	8	7.96		
31/03/2014	7.32	7.37	7.35	7.38	7.3	7.43	7.41	7.57	7.49	7.59			7.65
15/04/2014	6.94	7.17		7.05	7.04	7.57				7.51			7.91
29/04/2014	7.17	7.15		7.21	7.28	7.34				7.58			7.99
13/05/2014	7.53	7.69		7.79	7.84	7.87		7.64		7.64			7.81
27/05/2014	7.56	7.58	7.57	7.6	7.66	7.91		7.73		7.53			8.17
10/06/2014	8.06	7.9	7.81	7.79	8.41	7.81	8	7.84		7.92			7.88
24/06/2014	8.05	8.15		8.11	8.09					8.06			
08/07/2014	7.38	7.34		7.49	7.5					7.56			
22/07/2014	7.34	7.53		7.48	7.35					7.86			
06/08/2014	7.57	7.67			7.81								
19/08/2014	7.56	7.68		7.75									
02/09/2014	7.52	7.52		7.49									
16/09/2014	7.86	7.98		7.98									
30/09/2014	7.63	7.64		7.65									
07/10/2014	7.66	7.75		7.64	7.65								
14/10/2014	7.5		7.56	7.55	7.43	7.83	7.63	7.6				7.51	7.96
21/10/2014	7.68	7.76	7.86	7.77	7.71	7.96	7.85	7.86	7.8			7.79	7.83
28/10/2014	7.64	7.49	7.43	7.75	7.93	7.9	7.55		7.66				
04/11/2014	7.72	7.93		7.71	7.77	7.95			7.91	7.9			
11/11/2014	5.91	5.98	5.88	6.31	6.47	6.11	6.08	5.93	6.19	5.77			5.88
18/11/2014	6.12	6.22	5.98	6.17	6.26	6.81	7.18	6.59	6.55	6.73	6.76	7.06	6.35
25/11/2014	6.95	7.75	7.02	7.91	7.68	7.37	6.61	7.84		7.64	8.04	6.47	
01/12/2014	5.49	5.77	5.05	5.15	7.07	5.42	4.55	6.26	5.19	5.82	7.36	5.43	5.67
09/12/2014	6.43	6.54	5.2	6.75	5.99	5.46	6.36	5.26	7.27	6.35	5.67	5.9	5.71
16/12/2014	7.96	8.02	7.99	7.89	7.8	7.91	7.86	7.89	7.93	7.99	7.83	7.85	8.12
06/01/2015	8.06	8	8	7.94	7.85	7.91	8.15	7.96	8.06	8.04	7.99	7.97	8.17
13/01/2015	7.67	7.77	7.77	7.82	7.6	7.7	7.74	7.94	7.88	7.85	7.8	7.78	
20/01/2015	8.06	8.03	7.83	8	8.54	8.52	7.99	7.95	8.02	8.04		7.95	7.95
26/01/2015	7.76	7.77	7.9	7.77	7.7	7.9	7.86	7.79	7.85	7.93	7.91	8.12	
03/02/2015	7.61	7.71	7.7	7.74	7.55	7.73	7.63	7.82	7.82	7.87	7.68	7.8	
10/02/2015	7.89	7.97	8.09	8.35	7.98	8.06	8.11	8.02	8.11	8.1	8.15	8.12	8.24
17/02/2015	7.61	7.62	7.79	7.72	7.62	7.88		7.71	7.73	7.97			8.24
24/02/2015	7.54	7.73	7.87	7.8	7.76	7.83	7.82	7.88	7.92	8.03	7.82	7.92	8.12
03/03/2015	7.94	8.02	8.09	8.05	7.95	8.04	8.1	8.14	8.05	8.25		8.11	8.34
10/03/2015	7.75	7.74	7.95	7.68	7.65	7.99	7.97		7.77	8.01			8.29
17/03/2015	7.95	7.96		8.08	7.82	8.16			8.01	8.19			8.61
24/03/2015	7.92	7.95		7.99	7.76	8.2			8.15	8.18			8.21
31/03/2015	7.59	7.65	7.79	7.62	7.52	7.87	7.73	7.74	7.84	7.72		7.69	

Table D7 Weekly rainfall data

DATE	Weekly rainfall (mm)	DATE	Weekly rainfall (mm)	DATE	Weekly rainfall (mm)
03/04/2013	0	02/12/2013	3.8	05/08/2014	0
09/04/2013	0.6	10/12/2013	2	12/08/2014	35.6
16/04/2013	8	17/12/2013	7.8	19/08/2014	4.8
23/04/2013	2.2	24/12/2013	26	26/08/2014	17.8
30/04/2013	6.6	31/12/2013	7.6	02/09/2014	7.2
07/05/2013	0.2	07/01/2014	22.2	09/09/2014	0.4
14/05/2013	10.6	14/01/2014	17.2	16/09/2014	1.4
21/05/2013	19.2	21/01/2014	10.6	23/09/2014	1.4
28/05/2013	27.2	29/01/2014	23	30/09/2014	14.4
04/06/2013	0.4	04/02/2014	14.6	07/10/2014	15.8
11/06/2013	0.2	11/02/2014	38.8	14/10/2014	54
18/06/2013	11.8	18/02/2014	17	21/10/2014	10.2
25/06/2013	5.2	25/02/2014	2.6	28/10/2014	7.6
02/07/2013	9	04/03/2014	16	04/11/2014	2.4
09/07/2013	0.8	11/03/2014	1	11/11/2014	23.4
16/07/2013	0.2	18/03/2014	0.6	18/11/2014	23.8
23/07/2013	0.6	25/03/2014	9.4	25/11/2014	30.8
30/07/2013	9.2	31/03/2014	19.6	01/12/2014	10.4
06/08/2013	31.2	08/04/2014	10.4	09/12/2014	9.4
13/08/2013	2.6	15/04/2014	0.4	16/12/2014	10
20/08/2013	10.4	22/04/2014	6.2	23/12/2014	13.4
27/08/2013	24.6	29/04/2014	8.2	30/12/2014	31.4
03/09/2013	0.2	06/05/2014	3	06/01/2015	5.6
10/09/2013	16.2	13/05/2014	33.4	13/01/2015	16.8
17/09/2013	41.4	20/05/2014	0	20/01/2015	7
24/09/2013	2.4	27/05/2014	61.8	26/01/2015	8.4
01/10/2013	0.2	03/06/2014	31.4	03/02/2015	22.8
08/10/2013	2.8	10/06/2014	18.2	10/02/2015	5.8
15/10/2013	68.6	17/06/2014	0.2	17/02/2015	3.8
22/10/2013	17.6	24/06/2014	1.2	24/02/2015	17
31/10/2013	29.2	01/07/2014	41.4	03/03/2015	9.8
05/11/2013	8.6	08/07/2014	7.4	10/03/2015	1.4
12/11/2013	27.6	15/07/2014	47.8	17/03/2015	1.4
19/11/2013	7.2	22/07/2014	18.2	24/03/2015	3.4
26/11/2013	18.6	29/07/2014	0.6	31/03/2015	23.6

APPENDIX E Porous pot data

Figure E1 Map of locations of installed porous pots

Table E1 Coordinates of installed porous pots

Table E2 Dissolved nitrate concentration in porous pots

Table E3 Ammonium concentration in porous pots

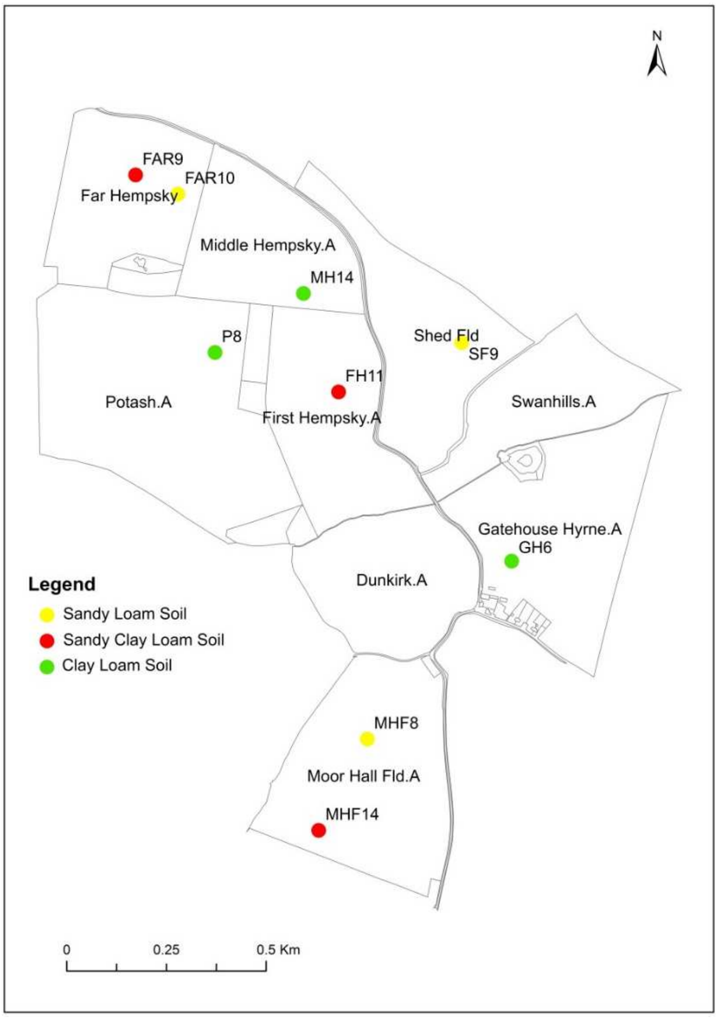

Figure E1 Map of locations of installed porous pots

Table E1 Coordinates of installed porous pots

Sample name	Latitude (N)	Longitude (E)
FH11	52.78767175	1.11636238
FAR9	52.79275454	1.10917096
MHF14	52.77783005	1.11490695
MHF8	52.77983962	1.11686764
SF9	52.78867188	1.121014
FAR10	52.79228331	1.11070124
P8	52.78868931	1.11183573
MH14	52.78992497	1.11521406
GH6	52.78370334	1.12251244

Table E2 Dissolved nitrate concentration in porous pots

Sample ID	Feb2014 NO_3 (mg N/L)	Apr2014 NO_3 (mg N/L)	Feb2015 NO_3 (mg N/L)	Sample ID	Feb2014 NO_3 (mg N/L)	Apr2014 NO_3 (mg N/L)	Feb2015 NO_3 (mg N/L)	sample ID	Feb2014 NO_3 (mg N/L)	Apr2014 NO_3 (mg N/L)	Feb2015 NO_3 (mg N/L)
FAR10-P1	12.13	11.41	24.92	FAR9-P1	16.82	17.13	0.44	P8-P1	10.71	6.95	18.02
FAR10-P2	11.36	10.40	0.68	FAR9-P2	11.68	16.37	0.78	P8-P2	9.98	9.03	13.25
FAR10-P3	15.64		5.54	FAR9-P3	4.73		11.35	P8-P3	16.66		5.25
FAR10-P4	11.03	22.27	0.16	FAR9-P4	19.82		7.35	P8-P4	19.73	14.04	7.90
FAR10-P5	26.89		0.30	FAR9-P5	18.03	22.24	9.39	P8-P5	8.64	11.37	10.10
FAR10-P6	22.29			FAR9-P6	18.16	19.27	3.88	P8-P6	3.29	7.09	
FAR10-P7	20.85	24.11	22.74	FAR9-P7	23.47	12.60	20.56	P8-P7	8.49	9.41	
FAR10-P8	18.15	9.87	9.27	FAR9-P8	24.47	3.19	2.68	P8-P8	8.61	7.90	
FAR10-P9	14.92		12.95	FAR9-P9	21.08	11.71	48.03	P8-P9	2.20	3.60	
FAR10-P10	21.44	21.68	18.76	FAR9-P10	17.41			P8-P10	21.17	18.88	
MHF8-P1	0.38		0.18	MHF14-P1	0.58	3.33	12.58	GH6-P1	0.73	0.01	12.30
MHF8-P2	0.29	2.23	0.38	MHF14-P2	0.07	3.11	14.16	GH6-P2	0.57	0.03	34.71
MHF8-P3	0.23	2.23	4.58	MHF14-P3	0.35	3.31	10.01	GH6-P3	1.26	6.17	31.13
MHF8-P4	0.04		12.60	MHF14-P4	0.24	1.90	13.98	GH6-P4	0.34		10.44
MHF8-P5	0.35	5.92	41.23	MHF14-P5		3.13	0.84	GH6-P5	1.18		19.93
MHF8-P6	0.57	3.69	2.21	MHF14-P6	0.12	4.27	0.36	GH6-P6	1.22		19.14
MHF8-P7	0.29	2.67	5.46	MHF14-P7	0.01	3.36	7.77	GH6-P7	3.45		
MHF8-P8	0.20	0.01	0.12	MHF14-P8	0.24	2.24	0.43	GH6-P8	1.26		
MHF8-P9	0.36	0.01		MHF14-P9	0.26	2.34	22.72	GH6-P9	3.04		
MHF8-P10	0.16	0.01		MHF14-P10	0.01	3.13	26.23	GH6-P10	0.63	3.34	
SF9-P1	1.05	1.90	15.74	FH11-P1	0.33	0.71	3.83	MH14-P1	0.60	8.29	1.16
SF9-P2	0.33	3.87	13.86	FH11-P2	0.36	0.38	0.72	MH14-P2	0.40	1.85	0.22
SF9-P3	0.60	2.46	11.44	FH11-P3	0.07	1.00	2.89	MH14-P3	0.51		0.72
SF9-P4		2.76	8.62	FH11-P4	0.21	1.87	0.76	MH14-P4	0.32	0.68	2.10
SF9-P5	0.50	2.36	16.89	FH11-P5	0.74	5.27	0.18	MH14-P5	0.27	1.22	1.44
SF9-P6	0.27	2.98	19.69	FH11-P6	0.66	4.85	10.68	MH14-P6	1.04		0.52
SF9-P7	0.47	2.65	43.10	FH11-P7				MH14-P7	0.42		4.89
SF9-P8	0.06	3.52	15.21	FH11-P8	0.61	5.18		MH14-P8	0.43		
SF9-P9	0.36	2.38		FH11-P9				MH14-P9	0.38		
SF9-P10				FH11-P10	0.39			MH14-P10	0.66		

Table E3 Ammonium concentration in porous pots

Sample ID	Feb2014 NH_4 (µg N/L)	Apr2014 NH_4 (µg N/L)	Feb2015 NH_4 (µg N/L)	Sample ID	Feb2014 NH_4 (µg N/L)	Apr2014 NH_4 (µg N/L)	Feb2015 NH_4 (µg N/L)	Sample ID	Feb2014 NH_4 (µg N/L)	Apr2014 NH_4 (µg N/L)	Feb2015 NH_4 (µg N/L)
FAR10-P1	2		13	FAR9-P1	15			P8-P1	38		
FAR10-P2	0			FAR9-P2	15			P8-P2	6		
FAR10-P3				FAR9-P3				P8-P3			
FAR10-P4	0			FAR9-P4	106			P8-P4	5		
FAR10-P5	33		38	FAR9-P5	36			P8-P5	4		
FAR10-P6				FAR9-P6	0			P8-P6			
FAR10-P7	19		119	FAR9-P7		222		P8-P7	5		
FAR10-P8				FAR9-P8	19			P8-P8	0	19	
FAR10-P9				FAR9-P9	39			P8-P9	21		
FAR10-P10	92			FAR9-P10	15			P8-P10	5	20	
MHF8-P1				MHF14-P1	12			GH6-P1		12	
MHF8-P2	8		2	MHF14-P2				GH6-P2	7		
MHF8-P3				MHF14-P3				GH6-P3	30		
MHF8-P4				MHF14-P4	5			GH6-P4			
MHF8-P5	6			MHF14-P5				GH6-P5	32		
MHF8-P6	8			MHF14-P6		20		GH6-P6	101		
MHF8-P7	5		4	MHF14-P7		47	50	GH6-P7	10		
MHF8-P8		645	7	MHF14-P8				GH6-P8	16		
MHF8-P9		8		MHF14-P9		7		GH6-P9			144
MHF8-P10				MHF14-P10				GH6-P10			0
SF9-P1				FH11-P1	86			MH14-P1			
SF9-P2	4		43	FH11-P2				MH14-P2	8		
SF9-P3	64		290	FH11-P3	10		301	MH14-P3			
SF9-P4				FH11-P4	49	10	41	MH14-P4	0		190
SF9-P5	37	20	154	FH11-P5				MH14-P5	16		93
SF9-P6				FH11-P6	29			MH14-P6	32		
SF9-P7	25		516	FH11-P7				MH14-P7	0		
SF9-P8				FH11-P8	55			MH14-P8	2		
SF9-P9		8	463	FH11-P9				MH14-P9	3		
SF9-P10				FH11-P10				MH14-P10	5		

APPENDIX F Riverine data

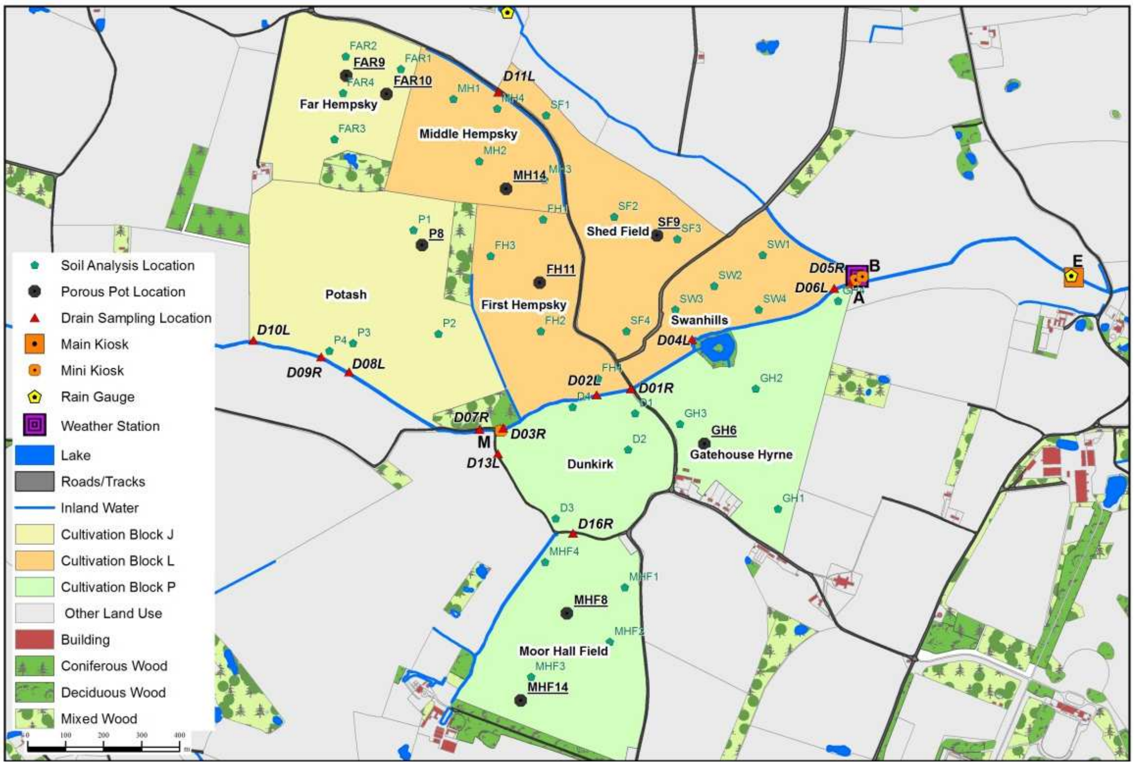

Figure F1 Location map of stream samplings (A, B, E, and M) (indicated by orange square)

Table F1 Coordinates of stream sampling sites

Sample name	Latitude (N)	Longitude (E)
A	52.78767175	1.11636238
B	52.79275454	1.10917096
E	52.77783005	1.11490695
M	52.784268	1.11686764

Table F2 Nitrate concentration in stream samples

	A	B	E	M		A	B	E	M		A	B	E	M
Date	NO_3	NO_3	NO_3	NO_3	Date	NO_3	NO_3	NO_3	NO_3	Date	NO_3	NO_3	NO_3	NO_3
	(mg N/L)	(mg N/L)	(mg N/L)	(mg N/L)		(mg N/L)	(mg N/L)	(mg N/L)	(mg N/L)		(mg N/L)	(mg N/L)	(mg N/L)	(mg N/L)
03/04/2013	8.20	9.70	7.71	8.85	24/09/2013	3.48	8.03	4.88	0.46	22/07/2014	2.52	6.49	3.71	0.74
09/04/2013	8.25	8.95	7.47	8.59	01/10/2013	3.73	8.53	5.14	0.73	19/08/2014	2.37	5.94	3.44	0.55
16/04/2013	3.43	6.98	6.94	7.04	08/10/2013	3.85	8.04	5.06	0.53	16/09/2014	2.49	6.10	3.51	0.45
23/04/2013	7.16	9.56	7.01	6.43	15/10/2013	6.47	9.54	6.77	7.96	30/09/2014	2.47	5.82	3.40	0.42
30/04/2013	5.28	9.17	6.16	4.98	22/10/2013	3.93	9.33	4.99	4.45	07/10/2014	2.37	6.16	3.34	0.49
07/05/2013	3.97	9.20	5.76	3.10	31/10/2013	10.84	13.81	11.38	13.05	14/10/2014	22.54	7.10	19.72	24.05
14/05/2013	4.23	9.31	4.94	2.89	05/11/2013	6.03	9.59	6.35	7.66	21/10/2014	8.37	5.97	6.90	9.58
21/05/2013	3.31	8.16	5.09	1.36	12/11/2013	8.79	9.92	8.49	10.49	28/10/2014	6.24	6.50	5.47	7.23
28/05/2013	4.52	8.67	5.62	4.17	19/11/2013	8.65	10.33	8.29	9.84	04/11/2014	5.26	7.07	4.92	5.25
04/06/2013	3.61	8.75	5.36	2.21	26/11/2013	10.35	11.47	9.64	12.22	11/11/2014	7.74	6.53	6.59	8.61
11/06/2013	3.11	8.30	5.08	1.62	02/12/2013	8.54	10.53	7.92	10.49	18/11/2014	10.96	6.44	9.06	11.43
18/06/2013	3.12	8.47	5.19	0.91	10/12/2013	6.54	9.74	6.74	8.01	25/11/2014	11.26	6.54	9.70	11.69
25/06/2013	3.71	9.02	5.67	0.63	14/01/2014	10.04	9.37	9.60	11.80	01/12/2014	9.56	6.91	8.32	10.62
02/07/2013	3.48	8.74	5.50	0.54	21/01/2014	9.77	8.78	8.88	10.78	06/01/2015	8.00	7.04	7.50	8.87
09/07/2013	3.95	9.00	5.91	0.46	29/01/2014	10.59	8.88	9.66	11.93	13/01/2015	7.35	6.95	6.72	7.48
16/07/2013	4.03	8.87	5.57	0.39	04/02/2014	10.65	8.84	9.61	11.25	20/01/2015	8.31	6.80	7.21	8.56
23/07/2013	4.31	8.58	5.42	0.36	11/02/2014	8.64	7.38	8.13	9.03	26/01/2015	7.51	7.29	6.99	7.98
30/07/2013	4.13	8.92	5.46	0.34	18/02/2014	9.18	7.83	8.30	10.11	03/02/2015	8.64	6.73	7.50	8.90
06/08/2013	3.12	7.84	4.63	0.18	25/02/2014	8.23	7.95	7.55	9.35	10/02/2015	8.44	6.89	7.19	8.80
13/08/2013	3.76	8.79	5.34	0.49	04/03/2014	6.73	7.32	6.43	7.46	17/02/2015	6.59	7.12	5.91	6.84
20/08/2013	3.92	8.27	5.11	0.39	11/03/2014	7.79	8.54	7.06	8.48	24/02/2015	4.54	6.37	4.23	6.46
27/08/2013	4.03	8.39	5.01	0.36	18/03/2014	7.56	8.22	6.93	8.43	03/03/2015	6.31	6.78	6.08	7.04
03/09/2013	3.77	7.87	4.89	0.46	31/03/2014	5.95	7.54	6.07	7.26	10/03/2015	5.83	7.07	5.52	6.21
10/09/2013	2.12	8.04	4.74	0.43	29/04/2014	4.20	7.08	4.74	4.75	17/03/2015	5.64	7.03	5.45	5.30
17/09/2013	3.45	8.36	5.07	0.55	27/05/2014	11.20	5.85	9.02	12.69	24/03/2015	5.10	6.83	5.18	5.28
					24/06/2014	5.25	6.99	4.58	6.23	31/03/2015	5.44	6.12	5.22	6.08

Table F3 Ammonium concentration in stream samples

Date	A NH$_4$ (µg N/L)	B NH$_4$ (µg N/L)	E NH$_4$ (µg N/L)	M NH$_4$ (µg N/L)	Date	A NH$_4$ (µg N/L)	B NH$_4$ (µg N/L)	E NH$_4$ (µg N/L)	M NH$_4$ (µg N/L)	Date	A NH$_4$ (µg N/L)	B NH$_4$ (µg N/L)	E NH$_4$ (µg N/L)	M NH$_4$ (µg N/L)
03/04/2013	91	13	56	5	24/09/2013	38	31	30	32	24/06/2014	115	18	21	21
09/04/2013	133	16	65	13	01/10/2013	21	22	25	21	22/07/2014	7		1	15
16/04/2013	173	15	87	17	08/10/2013	5	5	5	6	19/08/2014	7	4	79	11
23/04/2013	187	15	86	20	15/10/2013	202	8	48	8	16/09/2014	36	18	36	38
30/04/2013	281	20	90	19	22/10/2013	108	28	22	17	30/09/2014				6
07/05/2013	346	18	101	23	31/10/2013	46	0	8		07/10/2014	103	5	17	2
14/05/2013	446	27	60	27	05/11/2013		2		1	14/10/2014	47	13	21	7
21/05/2013	386	34	99	19	12/11/2013	71	16	24	9	21/10/2014	82	19	53	5
28/05/2013	152	11	54	16	19/11/2013	39	14	9	8	28/10/2014	66	7	7	4
04/06/2013	335	21	66	29	26/11/2013	51	23	17	18	04/11/2014	25	10	9	1
11/06/2013	290	46	48	39	02/12/2013	0	0	0	0	11/11/2014	53	19	21	4
18/06/2013	154	19	27	64	10/12/2013	79	28	23	20	18/11/2014	26	16	9	3
25/06/2013	96	19	33	73	14/01/2014	64	30	41	21	25/11/2014	25	4	7	
02/07/2013	55	19	107	58	21/01/2014	84	28	54	23	01/12/2014	26	7	4	
09/07/2013	48	21	33	56	29/01/2014	30	34	31	15	06/01/2015	68	8	39	8
16/07/2013	33	25	26	54	04/02/2014	61	33	48	23	13/01/2015	39	8	40	5
23/07/2013	32	27	28	58	11/02/2014	39	32	41	22	20/01/2015	14		12	
30/07/2013	39	36	60	40	18/02/2014	47	28	45	19	26/01/2015	41		26	
06/08/2013	35	33	50	31	25/02/2014	69	28	66	18	17/02/2015	57	13	28	11
13/08/2013	19	30	22	28	04/03/2014	71	28	54	27	24/02/2015	14	8		
20/08/2013	18	26	29	29	11/03/2014	107	11	59		03/03/2015	5	4	14	10
27/08/2013	11	21	26	21	18/03/2014	94	19	36	20	10/03/2015	79	4	13	
03/09/2013	37	39	43	38	31/03/2014	88	23	56	17	17/03/2015	144	9	58	30
10/09/2013	175	33	98	21	29/04/2014	170	19	73	22	24/03/2015	252	6	62	6
17/09/2013	19	16	24	14	27/05/2014	169	10	70	13	31/03/2015	266	6	138	280

Table F4 Nitrite concentration in stream samples

	A	B	E	M		A	B	E	M		A	B	E	M
Date	NO_2 (µg N/L)	NO_2 (µg N/L)	NO_2 (µg N/L)	NO_2 (µg N/L)	Date	NO_2 (µg N/L)	NO_2 (µg N/L)	NO_2 (µg N/L)	NO_2 (µg N/L)	Date	NO_2 (µg N/L)	NO_2 (µg N/L)	NO_2 (µg N/L)	NO_2 (µg N/L)
03/04/2013	12.5	12.9	12.9	7.2	01/10/2013	26.3	19.9	22.8	4.0	19/08/2014	14.4	1.9	13.9	1.5
09/04/2013	15.1	15.3	14.6	7.7	08/10/2013	29.5	22.5	18.1	5.8	16/09/2014	36.2	12.5	14.9	7.8
16/04/2013	20.9	17.7	21.5	11.1	15/10/2013	82.6	19.2	52.8	29.7	30/09/2014	27.5	12.9	15.6	3.7
23/04/2013	23.0	14.4	20.5	10.7	22/10/2013	112.4	24.9	44.1	15.0	07/10/2014	61.4	7.9	27.2	5.3
30/04/2013	21.1	13.0	18.4	6.5	31/10/2013	43.6	18.8	34.9	13.9	14/10/2014	35.5	19.0	42.3	24.9
07/05/2013	37.8	13.3	26.7	7.6	05/11/2013	63.1	19.7	34.6	30.8	21/10/2014	98.1	19.6	56.2	27.1
14/05/2013	36.3	11.1	27.2	5.1	12/11/2013	43.8	17.9	44.9	11.5	28/10/2014	120.9	19.6	31.4	19.5
21/05/2013	48.4	14.0	31.2	5.5	19/11/2013	40.2	19.4	35.3	11.1	04/11/2014	96.3	14.8	20.8	8.4
28/05/2013	32.5	12.7	23.7	6.9	26/11/2013	29.2	16.3	25.8	8.4	11/11/2014	62.3	15.3	33.4	9.8
04/06/2013	63.9	11.4	31.4	8.5	02/12/2013	50.4	20.2	42.5	10.9	18/11/2014	39.4	16.9	29.2	13.6
11/06/2013	83.0	10.5	35.4	6.2	10/12/2013	65.1	23.6	46.2	23.0	25/11/2014	18.6	17.8	22.3	11.0
18/06/2013	127.5	9.7	37.8	7.0	14/01/2014	18.0	18.8	20.7	9.9	01/12/2014	25.2	17.9	28.1	11.5
25/06/2013	102.6	8.6	38.4	7.7	21/01/2014	19.5	19.1	24.2	11.3	06/01/2015	13.8	17.6	15.7	6.8
02/07/2013	71.5	8.5	30.8	4.1	29/01/2014	8.1	14.6	11.1	5.6	13/01/2015	17.9	18.7	16.8	9.7
09/07/2013	47.8	7.9	21.3	3.2	04/02/2014	14.5	15.3	15.9	8.7	20/01/2015	11.9	13.1	9.8	4.1
16/07/2013	38.7	12.5	20.6	4.1	11/02/2014	9.8	9.3	10.3	5.6	26/01/2015	10.4	15.2	12.1	5.0
23/07/2013	46.8	20.7	30.1	5.6	18/02/2014	10.8	10.0	12.2	5.7	03/02/2015	5.6	8.6	6.4	5.7
30/07/2013	41.5	30.1	43.8	6.1	25/02/2014	14.2	13.3	16.8	7.8	10/02/2015	5.5	8.1	6.4	2.3
06/08/2013	44.0	28.6	49.5	7.1	04/03/2014	9.9	9.9	9.2	5.0	17/02/2015	11.4	14.4	11.5	5.8
13/08/2013	20.8	26.2	19.7	4.8	11/03/2014	12.5	11.7	14.9	5.9	24/02/2015	9.0	16.6	14.2	6.4
20/08/2013	22.4	25.7	23.8	4.2	18/03/2014	34.6	20.2	32.9	18.0	03/03/2015	15.5	13.9	16.0	7.8
27/08/2013	37.9	52.7	46.6	10.3	31/03/2014	21.3	13.9	21.7	12.5	10/03/2015	21.0	13.0	19.4	9.5
03/09/2013	35.8	32.4	38.0	11.8	29/04/2014	21.6	53.6	10.3	28.9	17/03/2015	65.8	33.7	54.5	33.9
10/09/2013	65.1	27.7	42.1	3.2	27/05/2014	49.6	6.7	46.2	10.6	24/03/2015	71.4	33.7	60.0	29.5
17/09/2013	25.2	19.3	21.9	2.4	24/06/2014	133.4	12.5	61.1	14.4	31/03/2015	102.0	31.8	93.7	119.3
24/09/2013	25.2	21.8	23.2	3.4	22/07/2014	48.8	37.0	24.2	22.6					

Table F5 Flow and inorganic and organic N fluxes at site A

Date	flow m^3 sec^{-1}	flux (kg N a^{-1} ha^{-1})	Date	flow m^3 sec^{-1}	flux (kg N a^{-1} ha^{-1})	Date	flow m^3 sec^{-1}	flux (kg N a^{-1} ha^{-1})
03/04/2013	0.0304	11571.44	15/10/2013	0.0113	2761.77	22/07/2014	0.0073	971.50
09/04/2013	0.0306	11502.82	22/10/2013	0.0094	1485.16	19/08/2014	0.0094	1245.04
16/04/2013	0.0318	12816.36	29/10/2013	0.0437	16496.13	16/09/2014	0.0161	2041.07
23/04/2013	0.0174	4230.68	05/11/2013	0.0296	6888.98	30/09/2014	0.0083	1104.58
30/04/2013	0.0198	4520.75	12/11/2013	0.0369	12707.37	07/10/2014	0.0141	2143.25
07/05/2013	0.0198	3940.04	19/11/2013	0.0339	11406.98	14/10/2014	0.0949	82929.56
14/05/2013	0.0416	6467.66	26/11/2013	0.0437	17047.38	21/10/2014	0.0384	13587.22
21/05/2013	0.0416	7372.86	02/12/2013	0.0109	3598.98	28/10/2014	0.0231	6053.68
28/05/2013	0.0341	7506.14	10/12/2013	0.022	5640.53	04/11/2014	0.0255	5371.84
04/06/2013	0.0341	6118.90	17/12/2013	0.0087	1805.31	11/11/2014	0.0387	11789.48
11/06/2013	0.0341	5344.63	07/01/2014	0.0804	32403.62	18/11/2014	0.0741	29584.11
18/06/2013	0.0341	4935.98	14/01/2014	0.0747	36396.17	25/11/2014	0.108	45843.25
25/06/2013	0.0341	5215.58	21/01/2014	0.0622	26559.24	01/12/2014	0.0511	7171.13
02/07/2013	0.0341	5323.12	27/01/2014	0.169	71949.38	09/12/2014	0.039	12311.34
09/07/2013	0.0341	5398.40	27/01/2014	0.169	72642.23	16/12/2014	0.0496	15970.33
16/07/2013	0.0424	7060.03	29/01/2014	0.169	69391.18	06/01/2015	0.0553	18154.42
23/07/2013	0.0424	6859.46	04/02/2014	0.0856	38035.70	13/01/2015	0.1172	35592.66
30/07/2013	0.0424	7340.82	11/02/2014	0.1533	56224.87	20/01/2015	0.0723	24761.37
06/08/2013	0.0424	6632.15	18/02/2014	0.091	31825.82	26/01/2015	0.0562	17581.45
13/08/2013	0.0424	6792.60	25/02/2014	0.0632	23199.40	03/02/2015	0.1297	42497.38
20/08/2013	0.0424	7086.77	04/03/2014	0.0531	14853.36	10/02/2015	0.0903	26796.86
27/08/2013	0.0181	2808.34	11/03/2014	0.0393	10398.27	17/02/2015	0.0903	24547.18
03/09/2013	0.0181	2973.88	18/03/2014	0.0252	6667.59	24/02/2015	0.0833	35279.92
10/09/2013	0.0181	2203.29	31/03/2014	0.0292	7882.49	03/03/2015	0.0556	15833.22
17/09/2013	0.0209	2965.96	29/04/2014	0.0324	5762.76	10/03/2015	0.0556	15026.65
24/09/2013	0.0428	6316.79	27/05/2014	0.04	17975.52	17/03/2015	0.0556	14220.09
01/10/2013	0.0428	6370.78	03/06/2014	0.0329	17191.95	24/03/2015	0.0556	14079.81
08/10/2013	0.0232	3255.78	24/06/2014	0.0267	5708.84	31/03/2015	0.0556	16008.56

APPENDIX G Dissolved nitrous oxide data

Table G1 Dissolved nitrous oxide concentration in drains and river samples

Table G1 Dissolved nitrous oxide (N_2O) concentration in drain (D1-D16) and river (A-M) samples

Date	D1 N_2O (µg N/L)	D2 N_2O (µg N/L)	D3 N_2O (µg N/L)	D4 N_2O (µg N/L)	D5 N_2O (µg N/L)	D6 N_2O (µg N/L)	D7 N_2O (µg N/L)	D8 N_2O (µg N/L)	D9 N_2O (µg N/L)	D10 N_2O (µg N/L)	D11 N_2O (µg N/L)	D13 N_2O (µg N/L)	D16 N_2O (µg N/L)	A N_2O (µg N/L)	B N_2O (µg N/L)	E N_2O (µg N/L)	M N_2O (µg N/L)
03/04/2013	3.83	1.05	2.42	15.75	4.08	5.05	4.96	1.76	2.70	19.18				1.63	2.10	1.62	0.71
09/04/2013	3.63	1.07	2.72	15.37	4.37	5.00	4.05	1.56	3.11	18.46				1.90	2.19	1.73	0.70
16/04/2013	2.46	1.41	1.42	14.01	3.59	6.44	1.04		2.85	19.56				1.62	1.78	1.45	0.58
23/04/2013	2.54	1.55		15.24	3.41	4.73			2.60	20.65				1.22	1.48	1.38	0.57
30/04/2013	1.97	1.47		14.22	3.15	1.73			0.64	22.17				1.35	1.66	1.35	0.57
07/05/2013	1.97	1.87		13.19	3.52	5.19			1.37	13.48				1.52	1.38	1.32	0.48
14/05/2013	1.53	1.55		12.19	2.84									1.12	1.32	1.16	0.45
21/05/2013	1.42	1.57		11.69	2.77									1.07	1.24	1.12	0.42
28/05/2013	1.29	1.49		11.31	2.58									0.95	1.20	1.07	0.41
04/06/2013	1.30	1.70		12.51	2.26	2.60								0.92	1.11	1.03	0.41
11/06/2013	0.90	1.15		8.85	2.45									0.70	1.13	0.96	0.34
18/06/2013	0.61	1.00		6.58	1.86									0.72	1.03	0.95	0.31
25/06/2013	0.53	0.47		7.15	1.71									0.68	1.00	0.96	0.35
02/07/2013	0.45	0.43		6.12	1.55									0.78	1.12	1.02	0.34
09/07/2013		0.38		1.82	1.17									0.82	1.12	1.00	0.29
16/07/2013		0.36												0.92	1.02	0.99	0.30
23/07/2013		0.45												1.17	1.18	1.14	0.42
30/07/2013		0.41												1.25	1.27	1.34	0.35
06/08/2013		0.49												1.09	1.32	1.36	0.35
13/08/2013		0.51												0.97	1.10	1.02	0.37
20/08/2013		0.82												1.06	1.43	1.03	0.46
27/08/2013		3.33												1.09	1.42	1.13	0.44
03/09/2013		0.76												1.04	1.15	1.10	0.42
10/09/2013		0.89												1.07	1.23	1.11	0.53
17/09/2013	1.29	1.02		1.37	8.46									1.10	1.32	1.12	0.64
24/09/2013	1.08	0.88		0.97										0.92	1.24	1.11	0.45
01/10/2013	0.95	0.73		0.78										0.96	1.15	0.94	0.34

Date	D1 N_2O (µg N/L)	D2 N_2O (µg N/L)	D3 N_2O (µg N/L)	D4 N_2O (µg N/L)	D5 N_2O (µg N/L)	D6 N_2O (µg N/L)	D7 N_2O (µg N/L)	D8 N_2O (µg N/L)	D9 N_2O (µg N/L)	D10 N_2O (µg N/L)	D11 N_2O (µg N/L)	D13 N_2O (µg N/L)	D16 N_2O (µg N/L)	A N_2O (µg N/L)	B N_2O (µg N/L)	E N_2O (µg N/L)	M N_2O (µg N/L)
08/10/2013	0.98	0.64		0.68										0.89	1.24	0.98	0.57
15/10/2013	1.02	1.79		1.32										1.31	1.43	1.18	2.53
22/10/2013	0.96	1.42		1.26	3.45	3.35	0.87		0.72					1.08	1.85	1.27	0.88
29/10/2013	0.88	1.28		1.35		2.35	1.30		1.41					1.70	1.81	1.86	1.67
05/11/2013	0.81	1.14		1.43		1.34	1.74	4.31	2.10	2.37		2.62	1.51	1.06	1.74	1.32	0.99
12/11/2013	4.26	1.19	1.59	2.02		2.61	4.44	2.90	2.31	2.84		3.07	1.92	1.19	1.75	1.36	1.21
19/11/2013	4.75	1.18	1.53	2.60	2.85	1.52	3.16	2.40	2.80	2.05		1.99	1.56	2.01	1.72	1.28	1.66
26/11/2013	4.29	1.08		3.49	3.08	1.69	3.57	2.15	3.15	2.36		2.24	1.69	1.55	1.78	1.32	1.37
03/12/2013	3.83	0.97		4.38	3.31	1.86	3.98	1.90	3.50	2.66		2.50	1.83	1.09	1.84	1.37	1.08
10/12/2013	1.28	0.77		4.09	3.04	1.41	1.40	1.56	2.13	2.52		3.27	1.49	1.03	1.81	1.38	1.06
14/01/2014	6.78	0.86	1.81	3.41	4.67	2.85	3.50	1.52	3.32	3.42		4.04	2.35	1.07	1.45	1.07	1.19
21/01/2014	6.22	0.99	0.92	4.01	4.96	2.18	3.09	1.47	3.01	2.91		5.79	2.20	1.11	1.65	1.22	0.97
28/01/2014	6.36	0.80	1.37	2.65	5.82	2.43	2.66	1.12	2.71	2.65		4.28	1.71	0.99	1.41	1.19	1.03
04/02/2014	6.51	1.03	0.90	3.16	5.84	2.68	1.23	1.41	2.10	2.84		4.02	2.17	1.14	1.49	1.27	0.98
11/02/2014	7.47	1.36	1.14	3.29	6.85	3.61	1.50	1.08	1.57	3.03	8.78	5.32	2.64	1.23	1.72	1.29	0.86
18/02/2014	6.67	1.09	1.44	3.78	7.63	5.68	1.87	1.11	2.27	3.62	7.30	3.56	2.37	0.91	1.62	1.09	0.74
25/02/2014	5.95	1.20	0.57	4.63	6.28	2.65	2.12	0.64	1.41	2.86	7.44	1.81	1.48	0.84	1.54	1.07	0.55
04/03/2014	6.93	1.33	1.95	4.38	7.90	3.73	2.37	0.86	1.80	3.26	7.59	3.26	2.90	1.03	1.81	1.23	0.86
11/03/2014	5.34	1.21	8.74	5.94	6.88	4.30	1.49	1.01	1.51	2.08		0.71	1.46	1.14	2.08	1.32	0.69
18/03/2014	3.72	1.37	7.28	6.56	6.50	4.47	0.81	1.25	1.42	2.37			2.38	1.07	1.89	1.36	0.68
01/04/2014	5.25	1.47	12.42	6.52	7.90	5.12	2.92	1.70	2.54	4.39			4.01	1.19	1.85	1.50	0.92
15/04/2014	2.09	1.07		7.28	5.03	1.37		2.11		2.69			0.90				
29/04/2014	2.48	0.94		5.45	3.97	3.48		2.31		4.19			0.85	0.91	1.20	1.19	0.49
13/05/2014	1.32	1.41		3.54	3.95	1.06		2.52		14.99			0.80				
27/05/2014	5.95	1.55	10.34	4.77	3.93	1.63		7.85		1.25			1.38	1.60	1.45	1.87	1.65
10/06/2014	2.20	2.32	3.39	3.40	3.07	2.56	0.73	2.91		15.44			2.09				
24/06/2014	2.92	1.08		5.01	2.37					29.62				1.77	1.21	1.53	0.73

	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	D11	D13	D16	A	B	E	M
Date	N_2O	N_2O	N_2O	N_2O	N_2O	N_2O	N_2O	N_2O	N_2O	N_2O	N_2O	N_2O	N_2O	N_2O	N_2O	N_2O	N_2O
	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)	(µg N/L)
08/07/2014	2.31	1.02		3.95	2.69					16.29							
22/07/2014	2.30	0.93		2.19	2.31					2.61				1.21	2.10	2.51	0.63
05/08/2014	2.13	1.38		1.69	2.20												
19/08/2014	2.83	1.21		1.19										1.43	2.50	2.68	0.88
02/09/2014	2.14	1.16		0.84													
16/09/2014	2.23	1.37		1.56										1.52	1.99	2.47	1.27
30/09/2014	2.22	2.03		1.07										1.41	2.49	2.14	0.74
07/10/2014	1.65	1.47		0.95										1.68	2.60	3.33	0.74
14/10/2014	6.20	4.36	6.67	4.63	7.04	3.55							2.45	6.26	4.68	7.28	7.10
21/10/2014	7.71	7.24	3.26	6.76	3.54	2.03	9.10	23.53	10.91				1.60	1.52	2.43	2.13	1.52
28/10/2014	6.01	4.50	0.71	5.23	2.39	3.24	1.74	20.22	6.26					1.00	2.19	2.99	0.92
04/11/2014	6.63	4.10	1.28	7.09	1.08	1.94	5.49	18.56	6.05	0.74				1.78	2.69	3.86	1.32
11/11/2014	7.25	4.44	1.84	9.05	0.87	3.79	3.42	16.91	5.84	1.28	2.86		2.86	1.87	2.75	2.30	1.38
18/11/2014	11.00	5.55	10.52	9.05	1.35	6.90	32.94	10.82	7.75	3.50	8.02	9.64	2.83	2.38	3.16	2.40	2.32
25/11/2014	9.99	5.48	9.28	7.67	4.55	7.86	31.26	11.65	12.33	5.29	13.17	8.78	2.22	3.13	3.57	2.74	3.10
02/12/2014	9.86	1.97	8.05	9.14	2.93	10.13	22.56	13.92	16.91	4.68	14.61	7.57	1.62	2.38	2.24	1.83	1.65
06/01/2015	9.34	3.63	3.42	8.54	9.02	9.88	2.61	34.37	13.17	4.62	11.69	6.75	2.78	2.15	2.89	2.44	1.85
13/01/2015	3.71	6.12	3.29	6.71	2.88	4.49	8.91	6.65	6.61	6.00	7.00	6.24	2.91	0.80	1.40	0.89	1.30
20/01/2015	2.26	4.14	3.30	4.34	4.55	4.14	9.19	4.31	5.29	4.07	8.59	3.10	3.05	2.14	1.86	1.99	1.25
27/01/2015	5.88	2.86	5.93	7.13	3.97	4.80	5.71	8.39	6.68	2.03	9.38	2.89	1.69	1.55	2.20	1.63	1.45
03/02/2015	9.83	4.06	8.75	8.34	7.56	7.66	11.32	8.31	8.06	2.64	10.18	6.58	2.43	1.56	2.03	1.62	1.68
10/02/2015	4.11	4.83	4.99	6.78	3.22	3.93	5.07	6.66	2.39	6.92	2.60	5.93	4.41	0.96	1.42	2.20	1.82
17/02/2015	3.64	5.16	2.70	6.84	8.09	3.13	5.00	5.41	10.64	4.07	3.64	4.30	2.55	0.99	1.52	1.93	1.41
24/02/2015	5.88	3.56	4.70	6.12	5.47	5.04	4.93	5.87	8.44	1.04	4.68	2.67	0.45	0.97	1.34	1.05	0.58
03/03/2015	5.45	2.25	1.78	19.53	5.32	4.08	2.61	6.82	11.84	1.36		1.57	1.42	0.95	1.95	1.62	0.91
10/03/2015	3.69	1.89	3.23	19.18	4.85	3.12	1.65		13.52	1.32			1.24	1.09	2.15	1.44	0.98
17/03/2015	4.62	2.02	4.00	14.56	7.36	3.75	2.35		13.47	1.67			1.04	1.19	2.23	1.64	0.84
24/03/2015	3.73	2.01	4.39	14.79	6.49	2.45	2.70		8.43	1.18			1.06	1.27	2.25	1.82	0.96
31/03/2015	2.79	2.38	4.77	14.27	21.42	4.09	3.04	6.97	5.94	3.92			3.38	1.15	2.00	1.44	0.97

APPENDIX H Farm data

Table H1 Data of fertiliser applied

Table H1 Data of fertiliser applied for 2012-2013 and 2013-2014 farm year in the mini-catchment A

Field name	CROP 2012-2013	Area (ha)	N fertilizer kg N ha^{-1}	P fertilizer kg P ha^{-1}	K fertilizer kg K ha^{-1}	S fertilizer kg S ha^{-1}	Field name	CROP 2013-2014	Area (ha)	N fertilizer kg N ha^{-1}	P fertilizer kg P ha^{-1}	K fertilizer kg K ha^{-1}	S fertilizer kg S ha^{-1}
Green Yards	Sugar Beet	11.3	128.4	0.0	132.6	22.2	Green Yards	Spring Barley Malt	11.4	150.4	24.4	92.7	11.5
Dunkirk	Winter Wheat Feed	13.1	235.3	26.6	100.6	18.8	Dunkirk	Spring Beans Dried	12.9	30.2	30.3	57.4	14.4
Gatehouse Hyrne	Spring Barley Malt	16.6	150.8	25.4	96.0	12.1	Gatehouse Hyrne	Spring Beans Dried	16.9	38.0	29.3	55.4	14.7
Swanhills	Spring Barley Malt	10.4	145.6	26.4	100.0	11.7	Swanhills	Spring Beans Dried	10.6	26.3	26.4	50.0	12.6
Far Hempsky	Spring Barley Malt	12.5	149.8	24.7	93.6	12.0	Far Hempsky	Spring Beans Dried	13.5	0.0	27.9	52.8	11.1
Church	Spring Barley Malt	3.6	146.4	25.2	95.5	11.7	Church	Spring Beans Dried	3.6	0.0	29.9	56.6	11.9
First Hempsky	Spring Barley Malt	13.8	150.4	25.6	96.7	12.0	First Hempsky	Spring Beans Dried	13.8	34.4	29.8	56.5	14.6
Moor Hall Fld	Spring Barley Malt	18.6	151.1	26.2	99.0	12.1	Moor Hall Fld	Spring Beans Dried	20.0	0.0	29.5	56.0	11.7
Georges Field B	Winter Wheat Feed	14.0	210.0	26.9	78.2	0.0	Georges Field B	Spring Beans Dried	14.0	48.3	0.0	6.0	0.0
Merrisons	Spring Beans Dried	40.7	0.0	30.6	30.6	12.1	Merrisons	Winter Wheat Feed	43.0	222.9	0.0	0.0	17.8
Sapwells	Winter Wheat Feed	12.9	236.1	0.0	0.0	18.9	Sapwells	Sugar Beet	12.7	111.6	0.0	137.3	19.7
Potash	Winter Wheat Feed	25.9	240.7	24.1	91.2	19.3	Potash	Spring Beans Dried	26.0	0.0	28.0	53.0	11.7
Glebe	Winter Wheat Feed	24.8	236.2	0.0	0.0	18.9	Glebe	Sugar Beet	25.4	108.5	0.0	133.2	19.2
Reepham Road Gardens	Spring Barley Malt	4.3	148.6	24.1	109.9	11.7	Reepham Road Gardens	Winter Barley Malt	4.3	150.9	0.0	0.0	12.1
Beggar Hall	Spring Beans Dried	13.1	0.0	0.0	0.0	0.0	Beggars Hall	Winter Wheat Feed	13.1	224.0	0.0	0.0	26.2
Reepham Road Gardens	Spring Barley Malt	4.0	148.6	24.1	109.9	11.7	Reepham Road Gardens	Winter Barley Malt	3.9	169.6	0.0	0.0	13.6
Sapwells	Winter Barley Malt	5.7	123.5	0.0	0.0	17.1	Sapwells	Winter Oilseed Rape	5.7	225.0	0.0	0.0	45.8
Reepham Road Gardens	Spring Barley Malt	9.7	148.6	24.1	109.9	11.7	Reepham Road Gardens	Winter Barley Malt	9.7	150.5	0.0	0.0	12.0
17 Acres	Winter Barley Malt	6.5	128.5	0.0	0.0	18.2	17 Acres	Winter Oilseed Rape	6.5	225.0	0.0	0.0	45.8
Newlands	Winter Oilseed Rape	8.0	326.2	87.2	98.7	50.1	Newlands	Winter Wheat Feed	8.0	241.5	0.0	0.0	19.3
Newlands	Winter Oilseed Rape	7.8	326.2	87.2	98.7	50.1	Newlands	Winter Wheat Feed	7.8	241.5	0.0	0.0	19.3
Newlands	Winter Oilseed Rape	8.7	326.2	87.2	98.7	50.1	Newlands	Winter Wheat Feed	8.7	241.5	0.0	0.0	19.3
Low Farm Ave	Winter Barley Feed	17.8	150.5	0.0	0.0	12.0	Low Farm Ave	Winter Oilseed Rape	19.4	277.3	91.1	96.5	93.9
Stimpsons Potash	Winter Barley Feed	24.3	153.2	0.0	0.0	12.3	Stimpsons Potash	Winter Oilseed Rape	24.9	299.8	96.8	102.5	100.1
Lane Field	Winter Wheat Feed	12.0	241.1	0.0	0.0	19.3	Lane Field	Sugar Beet	11.8	122.9	0.0	139.9	21.0
Cooks Cottage	Winter Wheat Feed	10.6	236.3	0.0	0.0	18.9	Cooks Cottage	Sugar Beet	10.9	109.1	0.0	132.5	19.3
Carfour	Spring Barley Malt	15.9	91.1	30.3	114.7	7.3	Carfour	Spring Beans Dried	16.2	7.5	29.3	55.6	12.2
Kerdy Green	Winter Barley Feed	13.3	150.5	0.0	0.0	12.0	Kerdy Green	Winter Oilseed Rape	13.9	289.3	95.5	101.1	98.1
Forest Field	Winter Wheat Feed	15.1	235.2	0.0	0.0	18.8	Forest Field	Sugar Beet	16.4	104.1	0.0	125.9	18.5
Howards Barn	Winter Wheat Feed	14.1	231.6	0.0	0.0	18.5	Howards Barn	Sugar Beet	14.1	113.9	0.0	136.9	20.3
Salle Old Grounds	Winter Barley Feed	12.7	151.2	0.0	0.0	12.1	Salle Old Grounds	Winter Oilseed Rape	12.6	320.3	99.6	105.4	103.8
Fronthouse		6.7	0.0	0.0	0.0	0.0	Fronthouse	Winter Wheat Feed	6.7	253.5	0.0	0.0	20.3