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N leaching to groundwater from dairy production involving grazing over the winter on a clay-loam soil

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Abstract

This study investigated concentrations of various N species in shallow groundwater (< 2.2 m below ground level) and N losses from dairy production involving grazing over the winter period on a clay loam soil with a high natural attenuation capacity in southern Ireland (52°51'N, 08°21'W) over a two year period. A dense network of shallow groundwater piezometers was installed to determine groundwater flow direction and N spatial and temporal variation. Estimated vertical travel times through the unsaturated zone (<0.5 yr, time lag) allowed the correlation of management with groundwater N within a short space of time. There was a two way interaction of the system and sampling date ($P < 0.05$) on concentrations of DON, oxidised N and NO_3^- -N. In contrast, concentrations of NH_4^+ -N and NO_2^- -N were unaffected by the dairy system. Grazing

over the winter had no effect on N losses to groundwater. Mean concentrations of DON, $\text{NH}_4^+\text{-N}$, $\text{NO}_2^-\text{-N}$ and $\text{NO}_3^-\text{-N}$ were 2.16, 0.35, 0.01 and 0.37 mg L^{-1} respectively. Soil attenuation processes such as denitrification and DNRA resulted in increased $\text{NH}_4^+\text{-N}$ levels. For this reason, DON and $\text{NH}_4^+\text{-N}$ represented the highest proportion of N losses from the site. Some of the spatial and temporal variation of N concentrations was explained by correlations with selected chemical and hydro-topographical parameters ($\text{NO}_3^-\text{-N}/\text{Cl}^-$ ratio, distance of the sampling point from the closest receptor, watertable depth, depth of sampling piezometer, DOC concentration). A high explanatory power of $\text{NO}_3^-\text{-N}/\text{Cl}^-$ ratio and the distance of the sampling point from the closest receptor indicated the influence of point sources and groundwater-surface water interactions.

1.Introduction

The European Union Water Framework Directive (EU WFD; 2000/60/EC) attempts to achieve at least ‘good ecological status’ for all waterbodies by 2015 with implementation of programmes of measures (POM) by 2012. The main POM currently in place to achieve the goals of the WFD in Ireland is the Nitrates Directive (91/676/EEC), enacted in 2006 under Statutory Instrument (SI) 378, 2006, subsequently replaced by SI 610 in 2010. For nitrate (NO_3^-), the WFD utilises a maximum admissible concentration (MAC) of 50 mg L^{-1} for groundwater set under the Groundwater Directive (GWD 2006/118/EC). In the context of the WFD, groundwater is defined as a saturated zone in fractured bedrock or in significant bodies of sand and gravel. Shallow groundwater ($< 30 \text{ m}$ below ground level (bgl)), which generally forms a perched watertable in glacial till subsoils is not considered part of this groundwater definition. Such shallow groundwater is important where there is a direct interaction between such shallow groundwater bodies and a surface waterbody (Fenton et al., 2009a). Heterogeneities in soil and subsoil anisotropies, hydrological properties and topography make transport pathways highly variable, and there is generally a time lag or delay between losses at the surface and nutrient changes in groundwater (Fenton et al., 2011). Estimation of such a time lag is important when correlating surface management with shallow groundwater quality.

Organic systems of production generally operate at low stocking densities relative to conventional systems. Low stocking densities offer the potential to extend the grazing season throughout the autumn and winter. Grass-clover swards are considered to have poor spring growth, however relatively high growth rates can be expected during late summer and autumn. Such a system requires a calving date in late April or May, rather than the typical mean calving date in late February or March for conventional spring-calving dairy system in Ireland. From a sustainability perspective, grazing during the winter, albeit at low stocking rates, carries considerable risk of nitrogen (N) losses to groundwater (Jewkes et al., 2007), despite the fact that the grass growth can take up a considerable quantity of N during the mild winter conditions experienced in Ireland (Humphreys et al., 2009).

Previous research in temperate grassland indicated that deposition of urine in the late autumn and winter leads to increased N leaching (Cuttle and Bourne, 1993; Williams and Haynes, 1994; Stout et al., 1997; Di and Cameron, 2002; Decau et al., 2004). The hydrolysis of urea following the deposition of urine also increases the pH of the soil over several units and can reach values of up to 9. (Haynes and Williams, 1993; Shand et al., 2002). This has a solubilising effect on soil organic matter and leads to higher dissolved organic N (DON) leaching losses (Wachendorf et al., 2005). Consequently there is a potential for substantial mineral and organic N leaching losses to groundwater if significant effective rainfall (ER) occurs after grazing (Francis, 1995; Hack-ten Broeke and van der Putten, 1997), which is likely during the winter in Ireland. However, there is currently little data available on the environmental impact of grazing by dairy cows at relatively low stocking density over the winter on N leaching and groundwater quality. The current study investigated the relationship between losses from grazing and N concentrations in shallow groundwater on glaciated till soils with poor drainage conditions and an inherent high natural attenuation capacity. The primary objective of the current study was to install a high density monitoring network on a dairy farm to investigate the effect of a system of dairy production involving grazing over the winter on a clay loam soil on concentrations of various N species in shallow groundwater and N losses over a two year period. The secondary objective was to identify spatial and temporal patterns in N concentrations and to elucidate groundwater flow direction. To study the level of dilution and proximity to source of contamination on site, chloride anion (Cl^-) and NO_3^- -N/ Cl^- ratios were also investigated.

2. Materials and Methods

2.1 Site description

The study was conducted at Solohead Research Farm (52 ha) in Ireland ($52^\circ 51' \text{N}$, $08^\circ 21' \text{W}$) from February 2008 to February 2010. The local climate is humid temperate oceanic with a long potential growing season from February to mid December. During the last 10 years the site received on average 1017.5 mm of annual rainfall, and this resulted in mean annual ER of 551.8 mm. Mean growing season length was approximately

305 days, based on a soil temperature above 5 °C (Parson, 1988) at 0.05 m depth. Annual grass production rates measured under high input of fertilizer N over the last 10 years period ranged between 14.6 and 16.9 t **dry matter (DM)** ha⁻¹.

The dominating soils are poorly drained Gleys (90%) and Grey Brown Podzolics (10%) with a clay loam texture and low permeability (Table 1). Soil texture exhibits very small heterogeneity down to 0.9 m. Sand and gravel lenses are also present which come to the surface in places and allow for fast infiltration (Fig.1). The mean soil organic carbon and total N content at 0 to 0.3 m depth is 4.48% and 0.48%, respectively with a corresponding soil bulk density of 1.09 g cm⁻³. Elevation on site ranges from 148.5 to 155.5 m Above Ordnance Datum (AOD). Potential point sources of pollution on the farm are indicated in Fig 2a. The depth to bedrock (Cappagh White Devonian Sandstone Formation (Archer et al., 1996)) on site is very uneven ranging from 5 to 10 m. The sandstone aquifer is a low permeability confined aquifer with saturated hydraulic conductivity (k_s) of 0.001 to 0.02 m day⁻¹, low dissolved oxygen (1 to 4 mg L⁻¹) and electrical conductivity of less than 150 **millivoltmeter** (mV) (Jahangir et al., 2012). The overlying quaternary till contains a perched watertable (depth of 0 to 2.2 m bgl; elevation of 147.3 to 155.2 m AOD, Fig. 3), which is an unconfined and poorly productive aquifer. The watertable is shallowest during the winter season. Although a number of ditches (4 m bgl) and tile drains (1.8 m bgl with spacing of 25 m) were installed between 1960 and 1995 across the farm to improve the drainage on site by lowering the watertable (Gleeson, T., 1997, personal communication). Nevertheless much of the farm remains seasonally wet, waterlogged or flooded due to impeded drainage.

2.2 Weather data collection

Weather data (soil temperature, volumetric soil moisture at 0.05 m depth and rainfall) was recorded every 30 min using the Solohead Meteorological station (Campbell scientific Ltd, Loughborough, U.K). Daily weather data (rainfall, maximum and minimum air temperature, wind speed and solar radiation) were inputted into the grassland hybrid model of Schulte et al. (2005) to estimate daily ER and soil moisture deficit (SMD) based

on poorly drained soil criterion. Water filled pore space (%) at 0.05 m depth was calculated as the ratio of the volumetric soil moisture content to total pore space (total porosity) as described by Luo et al. (2008).

2.3 Experimental design and descriptions of the dairy systems

The treatments (Fig. 2b) corresponded to three grass-clover based dairy production systems (each > 8.5 ha) involving two different grazing seasons and corresponding annual stocking densities:

ES-100N – Early spring calving with fertilizer N: mean calving date of 17th February, grazing period from February to November, annual stocking density of 2.15 cows ha⁻¹ and receiving annual fertilizer N input of 100 kg ha⁻¹.

ES-0N – Early spring calving without fertilizer N: mean calving date of 17th February, grazing period from February to November, annual stocking density of 1.6 cows ha⁻¹ and received no fertilizing N input.

LS-0N – Late spring calving without fertilizer N: mean calving date of 17th April, annual stocking density of 1.7 cows ha⁻¹ between calving and 1st September and then 1.3 cows ha⁻¹ until closing date at the end of January, and receiving no fertilizer N input.

The main characteristics of the different systems are presented in Table 2. Each farming system consisted of 18 Holstein-Friesian dairy cows that calved down during a three-month period each spring. Cows were turned out (**returned outside**) to graze immediately after the calving date and remained outside until drying off the following winter, except when the ground was too wet and cows were occasionally retained indoors to avoid damaging the grassland. The length of the grazing season for each system was determined according to a number of days at pasture as described by Humphreys et al. (2009).

During 1999, most of the area of the farm (52 ha) was divided into 8 blocks (each an area approximately 6 ha) depending on soil type and drainage status as described by Humphreys et al. (2008a). Six of these blocks were assigned into this experimental study (34 ha). Each block was divided into four paddocks (each > 1 ha),

one paddock from each block was assigned to each of the three systems in a randomized complete block design. The paddocks were predominantly used for grazing and to a lesser extent for silage. The pasture was dominated by 0.76 % perennial ryegrass (Lolium perenne L.) with 24 % white clover (Trifolium repens L.) and was allocated to livestock in a rotational grazing system. Thus, each paddock was rationally grazed on up to 12 occasions per grazing season. The rotation length varied between 21 days in spring and summer, and extending to 42 days or more in autumn/winter. During the main grazing season, the cows were outside by day and night, and were milked twice daily. The entire area of each system was grazed between turn-out and closing for first silage in late March or early April (Table 2). Following harvest for silage, the area was again used for grazing.

The slurry collected over the winter period was applied to the soil and evenly distributed between the three systems using an umbilical system in late January, late March, April, June, late September and early November.

In the ES-100N system, fertilizer N was applied in split applications to grazing areas in February, March and April as urea (0.46 g N g^{-1}) at application rates of $28.8 \text{ kg N ha}^{-1}$ each time and in May as calcium ammonium nitrate (CAN, 0.275 g N g^{-1}) with an application rate of $34.4 \text{ kg N ha}^{-1}$. Fertilizer to silage areas was split into two applications with $28.8 \text{ kg N ha}^{-1}$ applied for early grazing in February and $85.2 \text{ kg N ha}^{-1}$ applied in April. No fertilizer N was applied for either for grazing or silage after early May; reliance was replaced by biological N fixation in association with white clover to supply N in the soil for the remainder of the growing season. The white clover content of herbage DM was maintained by broadcasting (**spreading**) 5 kg ha^{-1} of clover seed (cultivars Chieftain and Crusader, 1:1) onto silage stubble after the harvest of first-cut silage. The silage area was rotated from year to year and approximately 0.2 of the farm area was over-sown with white clover seed each year.

2.4 Ground water sampling and analyses

In July 2008, twelve piezometers constructed from polyvinyl chloride pipes, were installed in each paddock; 288 piezometers in total. The slotted screen opening interval on the lower 0.2 m of the casing was covered by a filter sock. The annulus (< 0.03 m) between casing and the piezometer wall was grouted with sand and bentonite on the soil surface to prevent water channelling down the outside of the casing. In each paddock 10 piezometers were used for water quality assessment and two were used to measure watertable depth. Samples were taken fortnightly, after purging the piezometers, during the main winter drainage period and following periods of high rainfall during other times of the year. Sampling was conducted 31 times between 20th August 2008 and 2nd February 2010 (19 and 12 times during drainage seasons 2008/2009 and 2009/2010 respectively). On each occasion a 50 ml sample was taken from each piezometer and bulked into two composite samples per paddock. All samples were filtered (Whatman no.2 filter paper, $d = 0.125$ m, 0.45 μm) within an hour after sampling and then analysed for ammonium N ($\text{NH}_4^+\text{-N}$), total oxidised N and nitrite N ($\text{NO}_2^-\text{-N}$) on a Thermo Konelab analyser (Technical Lab Services, Ajax, Ontario Canada). Total dissolved N (TDN) and dissolved organic carbon (DOC) were measured by chemiluminescence after catalytic combustion at 720°C using a Shimadzu TOC-VCPH analyzer with an ASI-V autosampler. Dissolved organic N (DON) was calculated as the difference between TDN and dissolved inorganic N ($\text{DIN} = \text{total oxidised N} + \text{NH}_4^+\text{-N}$). Nitrate N ($\text{NO}_3^-\text{-N}$) was calculated as the difference between **total oxidised N** and $\text{NO}_2^-\text{-N}$. If the result was negative, the proportion of DON or $\text{NO}_3^-\text{-N}$ was assumed to be negligible, i.e. not significantly different from zero.

Chloride and dissolved reactive phosphorus were determined calorimetrically using Aquakem 600 Discrete Analyser.

Nitrogen losses of all N species (kg ha^{-1}) for each system were estimated by multiplying the mean N concentrations (mg L^{-1}) recorded on the sampling date with the volume of ER (mm) between two sampling occasions with an assumption that the N concentrations recorded in the groundwater were representative of concentrations in the ER reaching the piezometer screen. Due to high soil organic carbon, it was assumed

that the unsaturated zone had a high attenuation capacity (Humphreys et al., 2008b). Therefore N concentrations or loads found in shallow groundwater on site do not represent losses from the agricultural system, for example from LS-0N, but instead represent transformed losses from LS-0N buffered by natural attenuation during migration through the unsaturated zone.

2.5 Watertable depth and vertical travel time

Shallow watertable levels were measured fortnightly using an electric-level indicator with an acoustic signal (DIP-50 Geosense, Suffolk, U.K.) as a part of regular groundwater sampling regime. All piezometers on site ($n = 380$) were surveyed to convert watertable depths to m AOD, and a groundwater flow map was constructed using a proportion of these piezometers in January 2010. Mean nutrient contour maps were constructed in the same way.

Vertical travel times (T_t (yr)) were calculated to estimate the residence time of solute/nutrients in the unsaturated zone, or in other words, how long leached nutrients migrate to the watertable. The time lag was calculated for twelve paddocks across the farm using the methodology of Fenton et al. (2011). The variables needed were: depth of unsaturated zone (mean depth to watertable in a particular paddock for the duration of the study), ER (m) for a particular period (February 2009 to February 2010) and effective porosity (n_e). Total porosity was measured in four replicate soil samples taken at three depth sections (0 to 0.3, 0.3 to 0.6 and 0.6 to 0.9 m) at twelve locations around the farm as described by Hao et al. (2007). The effective porosity was determined as 70% of total porosity values, which coincided with values taken for this site by Fenton et al. (2009a).

2.6 Statistical analyses

Statistical analyses were performed using the SAS system, version 9.01 (SAS Institute Inc., Cary, North California, USA). Normality distribution of residuals was tested with PROC UNIVARIATE. All the N concentration variables were transformed using a natural log ($y = \log(x+1)$) and were analysed using a two

way analysis of variance (PROC MIXED). The time factor such as the sampling date was entered as a repeated measure with a compound symmetry (CS) covariance structure and the other factors (system and system $\times$ sampling date interaction) were entered as fixed effects. The chloride concentrations and C:N ratios were analysed using the same model without previous transformations.

Without taking the natural attenuation capacity of the soil into account it was attempted to correlate other parameters with N species in the shallow groundwater. Correlations between chemical and hydro-topographical variables (elevation of groundwater sampling screen opening interval, depth of sampling piezometer, distance of the sampling point from the closest receptor, watertable depth, NO_3^- -N/ Cl^- ratio and DOC) averaged for a sampling point and concentration of N species in the groundwater were determined by Pearson's correlation analyses (PROC CORR). In order to find indicators that affect N concentrations in the groundwater single and multiple stepwise linear regression analyses with all measured N species and other factors were performed using a PROC REG procedure. This procedure allowed a classification of the importance of the studied factors based on their significance in explaining the variance of measured variables. All relationships found were diagnosed for linearity, constant variance, outliers, normality and co-linearity. The annual N losses were subjected to two way analyses of variance (PROC MIXED) with a repeated measure covariance structure investigating the effect of the system, year and their interaction. Post hoc treatment comparisons were made according to Tukey and Bonferroni.

3. Results

3.1 Weather data

The annual rainfall in 2008 and 2009 was exceptionally high, 1405 and 1403 mm, respectively and exceeded ten years mean by 24% (Fig.4a). In 2008, the highest monthly rainfall occurred in August (199.4 mm) and October (183.2 mm) and the lowest was recorded from February to May (from 48.4 to 117.6 mm month⁻¹). In contrast in 2009, the highest monthly rainfall occurred in November (310.4 mm) and January (205.0 mm) and the lowest was recorded from February to June (from 21.6 to 94.8 mm month⁻¹). Uncharacteristically wet

weather conditions in 2008 and 2009 were also evident from **water filled pore space** records at 0.05 m depth indicating complete soil saturation during 14 months. Higher SMD values during the summer (16.3 to 26.9 mm day⁻¹) reflected drier soil conditions; however SMD did not exceed value of 40 mm day⁻¹. Mean estimated SMD in 2008/09 was 4.38 mm day⁻¹ and in 2009/10 was 5.95 mm day⁻¹ (Fig.4d). The daily mean soil temperature at 0.05 m depth is shown in Fig.4e. The monthly mean ranged from winter lows of 2.98 °C to summer highs of 15.87 °C. There was very little variation between the years (9.75°C in 2008 and 10.43°C in 2009). Rainfall typically exceeded evapotranspiration during seven months from July until January. Over the first groundwater sampling period (August 2008 to February 2009), site received 923 mm of rainfall, of which ER was 728 mm. Effective rainfall occurred on 108 days within this period, giving a mean recharge of 6.74 mm d⁻¹. Although the amount of rainfall was higher for the second groundwater sampling period (February 2009 to February 2010), the ER of 750 mm was only a slightly different from the previous sampling period. Effective drainage occurred during 115 days which resulted in recharge of 6.52 mm d⁻¹ (Fig.4c).

3.2 Vertical travel time, watertable depth and groundwater flow

The estimated vertical travel time through the unsaturated zone to shallow groundwater ranged between 0.21 and 0.57 years at different locations around the farm (Fig.1). The watertable depth below the surface was relatively shallow (Fig.3) and its seasonal variation could impose seasonal saturation of the topsoil. The mean watertable depth for the first and second sampling period was **0.69** (SD = 0.29) and **0.78 m bgl** (SD = 0.29), respectively. The shallowest mean watertable depth was recorded in February 2010 with minimal and maximal contour of 147.8 and 154.7 m OAD, respectively. The deepest mean watertable was recorded in November 2009 with minimal and maximal contour of 147.3 and 154.7 m OAD, respectively.

The groundwater flow map (Fig. 2a) created in January 2010 shows groundwater flow direction mirroring surface topography. The watertable depth was intercepted by two deep narrow open drains constructed along the northern and western edge of the paddocks. Except for these possible receptors, there is also the larger

river Pope bordering the farm on the south-east and east. The central area and north-west are up-gradient and hydrologically connected through shallow flow lines to down-gradient areas around the river in south-east. The minimal contour of 148.08 m AOD was located in the north-east of the farm, near the river. The maximal contour (155.17 m AOD) was observed on the top of the mound in the centre of the farm. The hydraulic gradient of the watertable between these two points is relatively high (3.5 %) and is a reflection of a land slope. Groundwater head contours show groundwater flow direction towards the river, however there are some partial flows diverting from the main flow towards the drains in the north.

3.3 Effect of dairy production systems on N concentrations in groundwater

Mean concentrations of N species recorded in shallow groundwater under each of the dairy systems are presented in Table 3. There was a significant two way interaction (from $P < 0.0001$ to $P < 0.05$) between system and sampling date on concentrations of TDN, DON, oxidised N and NO_3^- -N (Table 3, Fig. 5). In contrast, concentrations of NH_4^+ -N and NO_2^- -N were unaffected by the dairy system, however, the effect of the sampling date was highly significant ($P < 0.0001$). The mean C:N ratio of the dissolved organic matter was quite low (12.01. or 13.26) and also influenced by two way interaction ($P < 0.05$) between the dairy system and sampling date.

3.4 Spatial and temporal pattern of nutrient concentrations in groundwater

The temporal variation of N concentrations in shallow groundwater is presented in Fig. 5; their spatial variation around the farm is illustrated using groundwater nutrient contour maps (Fig. 6). Although, there was a considerable within year variation observed for all concentrations of N species in groundwater (Table 3, Fig.5), the concentrations were independent of weather variables. There were no relationships found between the concentrations and amount of rainfall, ER, SMD or soil temperature on a daily, weekly or monthly basis. While there was no clear seasonal pattern observed for concentrations of DON, DIN concentrations,

particularly NO_3^- -N exhibited sharp increases and declines during both winter seasons. This pattern was not observed for Cl^- concentrations (Fig.5).

Some of the spatial and temporal variation of N concentrations was explained by the relationships with the chemical and hydro-topographical variables (Table 4). The coefficient of multiple determination (model R^2) represents the proportion of variation explained by all independent variables. However, for each combination of independent and dependent variables, the partial correlation coefficient (partial R^2) measured the strength of the correlation between two variables when the effects of other variables were held constant.

3.4.1 DON

Dissolved organic N concentrations ranged from 0 to 50.28 mg L^{-1} , with a mean of 2.16 mg L^{-1} . The model developed as a result of multiple linear regression analysis between DON mean concentrations and some chemical and hydro-topographical variables explained 47.9% of the variation ($P < 0.001$; Table 4). The concentrations were positively influenced by $\text{NO}_3\text{-N}/\text{Cl}^-$ ratio ($R^2 = 0.213$, $P = 0.012$) and by the distance of the sampling point from the closest receptor ($R^2 = 0.266$, $P = 0.001$). Elevated concentrations of DON were observed around the farm yard migrating down gradient (August 2008 to May 2009) and in the south-west corner of the farm (August 2008 to March 2009; Fig. 6a).

3.4.2 Ammonium N

The concentrations of NH_4^+ -N were lower compared with DON, within the range of 0 to 14.11 mg L^{-1} , with a mean value of 0.35 mg L^{-1} . Multiple linear regression analysis between mean concentrations and some selected variables explained 88.7% of the variation ($P < 0.0001$; Table 4). The model indicated that the NH_4^+ -N concentrations were positively correlated to $\text{NO}_3\text{-N}/\text{Cl}^-$ ratio ($R^2 = 0.639$, $P < 0.0001$), watertable depth ($R^2 = 0.179$, $P < 0.0001$), distance of the sampling point from the closest receptor ($R^2 = 0.039$, $P = 0.015$) and elevation of the sampling screen ($R^2 = 0.031$, $P = 0.018$). Increased NH_4^+ -N concentrations were detected in the western part of the farm along an old drain (September 2009 to February 2010), around the

farm yard (March to September 2009) and in the south-west corner of the farm (September to December 2008; Fig 6b).

3.4.3 Nitrate N

Nitrate concentrations ranged from 0 to 4.24 mg L⁻¹ with a mean of 0.37 mg L⁻¹. However, increased concentrations were recorded on the top of the mound in the central part of the farm migrating down gradient (September 2008-December 2008, November 2009-February 2010); in the south-west corner of the farm (September 2008-February 2009) and on the northern farm boundary (all year round; Fig 6c). From all the selected factors the concentrations of NO₃⁻-N were positively correlated only to the depth of sampling piezometer ($R^2 = 0.210$, $P = 0.013$; Table 4).

3.4.4 Nitrite N

The levels of NO₂⁻-N were very low, lower than 0.52 mg L⁻¹. Despite this, the developed model explained the variation very well ($R^2 = 0.826$, $P < 0.0001$; Table 4). The most important controlling factor influencing NO₂⁻-N levels was NO₃-N/Cl⁻ ratio ($R^2 = 0.592$, $P < 0.0001$), DOC concentration ($R^2 = 0.110$, $P = 0.045$), water table depth ($R^2 = 0.072$, $P = 0.090$) and the distance of the sampling point from the closest receptor ($R^2 = 0.051$, $P = 0.014$). All the factors were affecting the concentrations positively.

3.4.5 Cl⁻ anion and Nitrate/Cl⁻ ratio

Exceptionally high variation was observed for Cl⁻ concentrations ranging from 2.77 to 199.02 mg L⁻¹, giving a mean of 17.45 mg L⁻¹. The distribution of Cl⁻ concentration in shallow groundwater was positively related to watertable depth ($R^2 = 0.291$, $P = 0.003$) and to the distance of the sampling point from the closest receptor ($R^2 = 0.159$, $P = 0.011$). Elevated concentrations of Cl⁻ were observed around the farm yard migrating down gradient (August 2008-December 2008; March 2009-November 2009); in the south-west corner of the farm

(August 2008 -November 2009) and on the northern farm boundary (August 2008-January 2009, September 2009 - January 2010; Fig. 2c).

Nitrate/Cl⁻ ratio, a proxy for point source proximity, occurred within the range of 0 to 0.59 with peaks recorded in the western part of the farm along the old drain (November 2009-February 2010) and in the south-west corner of the farm (September 2008-January 2009; Fig.6d).

3.4.6 Dissolved reactive phosphorus

Concentrations of **dissolved reactive phosphorus** were negligible in comparison with other N species concentrations

3.5 Annual losses of N to groundwater (after buffering effect of soil)

Annual losses of all N species from the dairy production systems to groundwater buffered by natural soil attenuation are presented in Table 5. The dairy system had no effect on the losses, except for DON. The DON losses from ES-100N were higher by 14 and 27 % in 2008/09 and 2009/10, respectively, compared with losses from systems receiving no fertilizer ($P < 0.05$). Dissolved organic N represented 61% and 84% of overall losses in the 2008/09 and 2009/10 periods, respectively. Nitrate N comprised only a small proportion (between 7.9 and 20.2 %) of the N losses to groundwater.

Although, cumulative ER caused similar recharge during both sampling periods (728 to 750 mm), the seasonal differences in concentrations led to differential losses over time (Fig.7). The recharge to groundwater occurred more gradually and uniformly during 2008/09; the seasonal pattern of N leaching was similar in all dairy systems and highly depended on the amount of ER. The discrepancy in N leaching between the systems was observed during 2009/10, when losses from ES-100N were higher between March and October compared with systems receiving no fertilizer. However, from October 2009 onwards N leaching from all the systems followed the same seasonal pattern; losses quickly rose due to increased ER and N concentrations followed by decline from November to January as a result of decreased N

concentrations, while ER was still high (Fig.7). Dissolved inorganic N losses were significantly lower ($P < 0.05$) in 2008/09 compared with 2009/10. Despite this, the overall N losses between 2008/09 and 2009/10 were not significantly different (Table 5).

4. Discussion

4.1 Weather data

The years of the study had exceptionally high rainfall compared with previous years. This resulted in a higher soil moisture content, higher annual ER and longer drainage seasons (between 108 and 115 days) in both years. High soil moisture conditions led to long periods of anoxic conditions of the soils with considerable redox potential. This promoted reduction biochemical processes such dissimilatory NO_3^- reduction to NH_4^+ (DNRA) and denitrification while mineralization rates were reduced. The watertable fluctuations across the site were consistent with recharge and also seem to be consistent across the site. Therefore biochemical processes in the soil rather than dilution during vertical migration in the unsaturated zone and mixing in shallow groundwater controlled N concentrations in groundwater.

4.2 Vertical travel time, watertable depth and groundwater flow

Depth to the watertable on site was low (therefore unsaturated zone was thin) and, despite fluctuations, remained close to the surface throughout the study. Vertical travel time to groundwater, within or just longer than a single drainage season, calculated in this study agreed with those found by Fenton et al. (2009a). Such time lag allow management and groundwater concentrations (which take a certain time to migrate vertically to groundwater and thereby affect nutrient concentrations) to be compared in an accurate way. The travel times can be justified by the high ER during the study period and the sites high n_e values ranging from 19% to 49%.

4.3 Effect of dairy production systems on N concentrations in groundwater

In the present study, the concentrations of all N species in shallow groundwater exhibited a large variation within the dairy systems (Table 4). A significant two way interaction of the system and sampling date on N concentration of TDN, DON, oxidised N and NO_3^- -N indicated that the extent of the main dairy system effect on water quality changed over time. This could be a result of a timing of single successive grazing occasions. However from the statistical analyses it is clear that the effect of the sampling date was notably more

important than the dairy system (Table 4). Hence, the winter grazing in the LS-0N system did not have a greater deleterious impact on groundwater quality than the earlier calving dairy systems.

4.4 Spatial and temporal pattern of nutrient concentrations in groundwater

Despite the fact that the paddocks within the systems had been under almost uniform management, the nutrient contour maps indicated a considerable spatial variation in the concentration of N species in shallow groundwater. **Chloride in groundwater originates from rainfall and organic sources. The natural background concentration of Cl^- in Ireland is $\sim 18 \text{ mg L}^{-1}$ (Fenton et al., 2009b).** The principal point sources of N contamination indicated by increased $\text{NO}_3^- \text{-N} / \text{Cl}^-$ ratio were located in the western part of the farm along the old drain, in the south-west corner of the farm, in the area around the farm yard and on the northern farm boundary (Fig.6d). Some of them are still present (farm yard) and the others have been removed some time ago— i. e. dirty water irrigators (western part of the farm along an old drain), however still affecting the concentrations in groundwater. Horizontal movement on site is slow and therefore the legacy from such sources may remain for some time. Apart from these sources, the recent water quality on the farm is also being influenced by the diffuse sources of contamination such as fertilization and grazing management.

The characterization of spatial and temporal variation in the concentrations provides insights into the control of the distribution of N species and their losses. A considerable temporal variation in N concentrations in shallow groundwater independent of weather variables indicated that the shallow groundwater chemistry is rather controlled by soil parameters (e.g. physical structure, heavy texture and high soil organic matter content) than by ER. However, the weather variables such as rainfall, ER and SMD control the vertical travel time to the watertable. Considering, that this study took place under humid oceanic climatic conditions with exceptionally high rainfall and ER, the vertical travel time of solutes from the soil surface to shallow groundwater (<2.2 m) was short.

On the basis of available data it was not possible to investigate the correlations between soil parameters responsible for the control over groundwater chemistry and N concentrations in groundwater in the present study. However, some hydro-topographical and chemical factors have been analysed in order to explain the variation of N species in shallow groundwater. A high explanatory power of selected variables indicated heterogeneity in hydro-topographical properties between the sampling locations and thus heterogeneity in N transport pathways which affect the spatial distribution of N in the shallow groundwater. However the present models are applicable only to shallow groundwater at Solohead Research Farm.

4.4.1 DON

Dissolved organic nitrogen concentrations on site were high (mean of 2.16 mg L^{-1}) and dominant to the other N species, which comprised only a small proportion of TDN in the shallow groundwater. In general, N in groundwater occurs almost exclusively as NO_3^- -N as ammonification and nitrification occur in the unsaturated zone and NO_3^- -N is mobile in soils to be transported to groundwater in high quantities (Bouwman et al., 2005, Kroeger et al., 2006). There has been very limited attention paid to DON leaching from agricultural systems (Van Kessel et al., 2009) in the past and thus very little is known on the role of DON in the N cycle in agricultural soils (Murphy et al., 2000). It originates from plant litter leachates, microbial and root exudates, and hydrolysis of insoluble soil organic matter (Haynes, 2005) and can be subsequently transported in a dissolved or colloidal form (Green et al., 2008). It usually consists of high molecular weight complex humic substances (polyphenol-bound N), which are resistant to microbial attack and unavailable for plant uptake (Stepanauskas et al., 2002; Marschner and Kalbitz, 2003; Neff et al., 2003; Jones et al., 2004; Dijkstra et al., 2007); consequently it turns over slowly and contributes to N leaching to groundwater. In general, DON concentrations beneath the managed grasslands vary between 0.025 mg L^{-1} and 10 mg L^{-1} (Watson et al., 2000; Siemens and Kaupenjohann, 2002; Jones et al., 2004; Vinther et al., 2006, Van Kessel et al., 2009). High DON leaching on this site indicates high DON production, which may be attributed to the presence of legumes in the swards (Oelmann et al., 2007), application of organic manures

(Murphy et al., 2000) or slow decomposition processes associated with high soil moisture content (Trasar-Cepeda et al., 2000; Aitkenhead- Peterson et al., 2002). A positive correlation with NO_3^- -N/ Cl^- ratio also indicates point source influence. Elevated concentrations around the farmyard are likely to be a result of runoff from the yard, milking shed, and slurry and dirty water collection and storage. Horizontal migration of DON with groundwater flow towards lower groundwater elevation is being caused by the differences in hydraulic gradients. In addition, a positive relationship between DON concentrations and the distance of the sampling point from the closest receptor reveals that the River Pope and local drains interact with the groundwater, cause a dilution when the watertable is very shallow and consequently reduce the concentrations of DON at the groundwater-surface water interface.

4.4.2 Ammonium N

Ammonium concentrations were also high and comprised more than 45% of DIN. In general, it is produced in the litter zone and leached down the profile to be nitrified at depth (Mian et al., 2009). The ion is positively charged and therefore can be bound to negative exchange sites of soil organic matter or clay particles (Feigenbaum et al., 1994; Stark & Richards, 2008). It is highly soluble and mobile when amination and ammonification are inhibited by low temperatures, or at very high concentrations such as beneath urine patches (Mian et al., 2009). However, elevated DON and NH_4^+ -N concentrations in conjunction with low NO_3^- concentrations across the farm and inherent properties of the soils on site are more indicative of DNRA (Kartal et al., 2007, Francis et al., 2007). This biochemical process is mainly mediated by bacteria that are obligate and facultative anaerobes with fermentative metabolism (Koike and Sorenson, 1988). Consequently the process takes place in highly anaerobic soils with high **soil organic carbon** content (Buresh and Patrick, 1978, Sorenson, 1978), in the same habitat as denitrification (Tiedje et al., 1981; Hill, 1996). The multiple linear regression model in the present study showed that the concentrations of NH_4^+ -N were also increased by point sources (western part of the farm along the old drain, around the farm yard and in the south-west corner of the farm) and diluted by the river in the same way as

DON. Furthermore, since watertable depth explained a lot of variation in $\text{NH}_4^+\text{-N}$, it is assumed that the increased thickness of unsaturated zone allows for a greater time for biochemical transformation (i.e. nitrification, ammonification, DNRA) to take place (National Research Council, 1993) and consequently results in higher concentrations in groundwater.

4.4.3 Nitrate N

The $\text{NO}_3^-\text{-N}$ concentrations were consistently low, well below the maximum allowable concentration (MAC) of $11.3 \text{ mg NO}_3^-\text{-N L}^{-1}$ for groundwater throughout the study period. Substantial and inconsistent peaks in $\text{NO}_3^-\text{-N}$ relative to Cl^- concentrations provided evidence of substantial biochemical processes taking place in the unsaturated zone. Decreases in concentration could be partially attributed to immobilization by plants (Peterjohn and Correll, 1984; Haycock and Pinay, 1993) or microbes to sustain primary production. However, considering the anaerobic nature of the soils and high **soil organic carbon** content down to 0.9 m in the soil profile (Table 1), low $\text{NO}_3^-\text{-N}$ concentrations in this study are more likely attributed to soil attenuation processes such as DNRA (Hill, 1996) or denitrification (Jacobs and Gilliam, 1985; Lowrance, 1992). The presence of these biochemical processes on site may have contributed to the absence of significant water quality responses. Humphreys et al. (2008b) reported a lack of effect of management practices at the surface on water quality at Solohead Research Farm in 2001 and 2002. Mean $\text{NO}_3^-\text{-N}$ concentrations were $< 3.0 \text{ mg L}^{-1}$, not exceeding MAC limit during the experimental period and N losses to shallow groundwater were low and largely independent of N inputs, N surpluses, deposition of excreta-N at the soil surface and residual mineral N in the soil at the start of the drainage season (Humphreys et al., 2008b). Elevated concentrations of $\text{NO}_3^-\text{-N}$ in the present study, recorded on the top of the mound in the central area of the farm are likely related to the presence of a gravel lens (Fig.2a) and the deep watertable at this location, rather than to management practice on the soil surface. Occurrence of highly permeable gravel and the deep watertable increase residence time in presumably aerobic unsaturated zone. This might enhance nitrification rather than denitrification or DNRA at this location. In addition, a positive correlation to the

depth of sampling piezometer (depth of screen opening of 0.2 m) points to vertical stratification of NO_3^- -N concentrations in groundwater.

4.5 Annual losses of N to groundwater (after buffering effect of soil)

The effects of the dairy systems on N losses through leaching were based on their effects on N concentrations in the groundwater. Higher DON losses from ES-100, due to higher DON concentrations compared with the systems receiving no fertilizer could be explained by higher stocking density of grazing animals in this system and thus by higher deposition of excreta over the treatment area. There was no evidence of higher annual N losses from the system involving grazing over winter compared with earlier spring calving dairy production systems during both experimental years. Lack of a difference is likely due to buffering capacity of the soils on site.

Since ER caused similar recharge in 2008/09 and 2009/10, the volume of rainfall draining through the root zone each year was not responsible for the differences in N losses between the years. Consequently year to year variation in the DIN and DON losses was likely led by the differences in N concentrations. As the performance of dairy systems did not change, increased DIN concentrations (NO_3^- -N and NH_4^+ -N) in 2009/10 were likely caused by enhanced mineralization.

The N leaching from all systems in the 2009/10 followed a typical pattern described in previous leaching studies. An initial increase in losses was followed by a decrease after a peak concentration had been reached at the end of the drainage season, because essentially all N had been washed out of the soil by that time (Steele et al., 1984; Heng et al., 1991; Oenema and Boest, 1998; Scholefield et al., 1993; Silva et al., 1999; Decau et al., 2003).

High DON losses from grazed grasslands have been previously reported (Dick et al., 2000), and it has been postulated that they may be an important component of N flux to groundwater (Murphy et al., 2000).

Reported losses were highly variable and ranged from 0.3 kg DON $\text{ha}^{-1}\text{yr}^{-1}$ in a grass clover system (Saarijarvi et al., 2004) to a maximum of 127 kg DON $\text{ha}^{-1}\text{yr}^{-1}$ in a pasture following the application of urine

(Wachendorf et al., 2005; Van Kessel et al., 2009). Thus, the annual DON losses reported here were within the range of values previously published for grazed grasslands. The majority of NO_3^- -N losses to groundwater were attenuated by **soil organic carbon** rich and anaerobic soils (discussed above). High losses of NH_4^+ -N were also indicative of biochemical processes (i.e. DNRA) while migrating throughout unsaturated zone. The loads of DON measured in deep multi level wells (up to 30 m bgl) on site (Jahangir et al., 2012) in 2009/10 were considerably lower ($1 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) in comparison with DON losses measured in the present study. On the other hand, the loads of NO_3^- -N were noticeably higher ($4.5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). This indicates that DON and NH_4^+ -N undergo further mineralization and nitrification while moving throughout subsoil sediments and bedrock formations. Although, the loads of DON and NH_4^+ -N are quite high, the horizontal movement to the receptor on site (the River Pope) is very slow (Fenton et al., 2009a) and therefore they are unlikely to deteriorate the surface water quality, unless the sources are close to the local drains connected directly to the river.

5. Conclusions

In this study we evaluated the impact of dairy production systems on a clay loam soil on N concentrations and N losses to groundwater over a two year period. A dense network of shallow groundwater piezometers was installed to determine groundwater flow direction and N spatial and temporal variation. Vertical travel time of approximately one drainage season due to a shallow watertable and high ER allowed correlations between nutrient losses and shallow groundwater nutrient concentrations within a small time lag period. However, the high natural attenuation capacity of the soil due to high C and anaerobic conditions ensured low concentrations of nutrients making correlations difficult; i.e. grazing over the winter period had no effect on groundwater quality on this site. The results indicate that any losses from the dairy production systems were instantaneously reduced by denitrification and DNRA, which consequently resulted in increased NH_4 -N levels. For this reason, DON and NH_4^+ -N represented the highest proportion of N losses, presently migrating slowly towards receptors on site. Some of the spatial and temporal variation of N concentrations was

explained by correlations with selected chemical and hydro-topographical parameters. Strong correlations to NO_3^- -N/ Cl^- ratio and the distance of the sampling point from the closest receptor indicated presence of point sources of contamination and groundwater-surface water interactions on site.

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Table 1 Soil properties at Solohead Research Farm at three different depths. All values are means of at least three determinations with standard deviations in parentheses.

Depth	Sand	Silt	Clay	Bulk density	Particle density	Total porosity	CEC	pH	Total N	Total organic C	Total inorganic C	Organic matter†	C:N ratio
m	%	%	%	g cm ⁻³	g cm ⁻³	%	Meq/100gDM		%	%	%	%	
0.0-0.3	36.7 (6.4)	35.0 (3.5)	28.3 (3.1)	1.09 (0.16)	2.52 (0.15)	58.7 (6.5)	25.4 (2.3)	6.60 (0.2)	0.48 (0.05)	4.48 (0.52)	0.09 (0.03)	6.55 (0.84)	9.34 (0.27)
0.3-0.6	36.0 (4.4)	36.0 (2.0)	28.0 (2.7)	1.43 (0.26)	2.59 (0.15)	44.5 (10.2)		7.00 (0.2)	0.22 (0.05)	2.02 (0.41)	0.08 (0.08)	2.74 (0.53)	9.09 (0.28)
0.6-0.9	36.0 (6.1)	35.7 (3.2)	28.3 (2.9)	1.70 (0.20)	2.67 (0.20)	37.1 (6.8)		7.65 (0.2)	0.06 (0.02)	0.49 (0.16)	0.54 (0.29)	1.63 (0.84)	7.88 (0.69)

† calculated from loss on ignition results (%) using the Ball's equation: OM = (0.476*loss on ignition) - 1.87 (Ball, 1964)

Table 2 Details of the experimental systems at Solohead Research Farm in 2008 and 2009. Each system was stocked with 18 Holstien-Friesian dairy cows. ES-100N - early spring calving with Fertiliser N. ES-0N - early spring calving without Fertiliser N. LS-0N - late spring calving without Fertiliser N.

	2008			2009		
	ES-100N	ES-0N	LS-0N	ES-100N	ES-0N	LS-0N
Mean first grazing date	14 th Feb	14 th Feb	14 th Apr	13 th Feb	13 th Feb	15 th April
Mean last grazing date	12 th Nov	12 th Nov	14 th Jan	18 th Nov	18 th Nov	26 th Jan
Mean days grazing cow ⁻¹	220	234	231	218	234	235
Total area (ha)	8.5	11.3	14.2	8.5	11.3	14.2
Proportion of area harvested for silage						
1 st cut (Apr – May)	0.5	0.3	0.5	0.4	0.2	0.5
2nd cut (Jun – Jul)	0.0	0.4	0.2	0.1	0.3	0.5
Stocking rates on grazing areas (cows ha ⁻¹)						
Feb-Mar	2.1	1.6		2.1	1.6	
Apr-May	4.2	2.3	3.4	3.4	2.0	3.2
Jun-Jul	2.1	2.8	2.3	2.4	2.4	3.3
Aug-Sep	2.1	1.6	1.7	2.1	1.6	1.7
Oct-Nov	2.1	1.6	1.3	2.1	1.6	1.3
Dec-Jan			1.3			1.3
Total stocking rate	2.1	1.6	1.5	2.1	1.6	1.5

Table 3 Mean concentrations of N species and Cl⁻ in groundwater (mg L⁻¹) under three dairy production systems with standard deviations in the brackets. ES-100N - early spring calving with fertiliser N, ES-0N - early spring calving without fertiliser N, LS-0N - late spring calving without fertiliser N. **P value <0.05 is significant.**

mg L ⁻¹	August 2008 - February 2010			P value		
	ES-100N (SD)	ES-0N (SD)	LS-0N (SD)	System	Date	System x date
Total N	3.13 (5.10)	2.79 (2.66)	2.90 (4.50)	0.079	0.0001	0.003
Dissolved organic N	2.35 (3.35)	2.06 (1.89)	2.09 (1.94)	0.077	<0.0001	<0.0001
Ammonium N	0.37 (0.95)	0.35 (1.03)	0.34 (0.80)	0.431	<0.0001	0.470
Total oxidised N	0.33 (0.37)	0.42 (0.53)	0.38 (0.49)	0.440	<0.001	0.006
Nitrite N	0.01 (0.02)	0.01 (0.04)	0.01 (0.03)	0.552	<0.0001	0.245
Nitrate N	0.33 (0.37)	0.41 (0.52)	0.37 (0.48)	0.443	<0.0001	0.004
C:N ratio	13.26 (8.61)	12.01(6.14)	12.05 (9.29)	0.441	<0.0001	0.012
Cl ⁻	18.50 (13.40)	15.97 (8.12)	17.84 (9.43)	0.221	<0.0001	0.024

Table 4 Results of single and multiple regression analyses between N and Cl⁻ concentrations in groundwater as dependent variable and some chemical and hydro-topographical factors as independent variables.

Dependent variable (mg L ⁻¹)	Chemical or hydro-topographical factor (x, z, q, w)	Unit	Partial R ²	Partial P value	Relationship	Model R ²	Model P value
Total N (y)	NO ₃ -N/Cl ⁻ ratio (x)	-	0.258	0.0049	$y = 70.251x + 0.010z + 0.836$	0.536	<0.0001
	distance from the receptor (z)	m	0.278	0.0005			
Dissolved organic N (y)	NO ₃ -N/Cl ⁻ ratio (x)		0.213	0.0119	$y = 36.709x + 0.006z + 1.018$	0.479	0.0002
	distance from the receptor (z)	m	0.267	0.0012			
Ammonium N (y)	NO ₃ -N/Cl ⁻ ratio (x)	-	0.639	<0.0001	$y = 22.881x + 0.528z + 0.001q - 0.040w + 5.528$	0.887	<0.0001
	watertable depth (z)	m bgl	0.179	<0.0001			
	distance from the receptor(q)	m	0.039	0.0150			
	elevation of the screen (w)	m AOD	0.031	0.0176			
Oxidised N (y)	depth of sampling piezometer (x)	m			$y = 0.247x + 0.016$	0.214	0.0115
Nitrite N (y)	NO ₃ -N/Cl ⁻ ratio (x)	-	0.592	<0.0001	$y = 0.514x + 0.0003z + 0.012q + 0.00003w - 0.016$	0.826	<0.0001
	dissolved organic C (z)	mg L ⁻¹	0.110	0.0045			
	watertable depth (q)	m bgl	0.072	0.0090			
	distance from the receptor (w)	m	0.051	0.0138			
Nitrate N (y)	depth of sampling piezometer (x)	m			$y = 0.238x + 0.020$	0.210	0.0125
Cl ⁻ (y)	watertable depth (z)	m bgl	0.291	0.0025	$y = 14.331x + 0.036z + 4.818$	0.449	0.0004
	distance from the receptor (z)	m	0.159	0.0111			

Table 5 Mean annual losses of N species from three dairy production systems to groundwater ($\text{kg ha}^{-1} \text{ yr}^{-1}$) with standard deviations in the brackets. ES-100N - early spring calving with Fertiliser N, ES-0N - early spring calving without Fertiliser N, LS-0N - late spring calving without Fertiliser N.

	August 2008 - February 2009			February 2009 - February 2010			P values		
	ES-100N (SD)	ES-0N (SD)	LS-0N (SD)	ES-100N (SD)	ES-0N (SD)	LS-0N (SD)	System	Year	System x Year
Total N	22.91 (7.89)	21.42 (4.65)	19.96 (4.80)	19.18 (13.08)	18.39 (11.66)	19.27 (8.95)	0.271	0.202	0.797
Dissolved organic N	19.14 (5.39)	17.20 (2.76)	16.45 (3.44)	14.06 (11.04)	10.55 (4.48)	11.66 (4.72)	0.042	<0.0001	0.809
Ammonium N	1.93 (2.76)	1.36 (0.37)	1.26 (0.34)	2.59 (2.18)	3.23 (4.51)	3.02 (2.56)	0.796	0.011	0.597
Oxidised N	1.85 (0.64)	2.86 (2.70)	2.25 (1.57)	2.82 (1.16)	3.80 (2.45)	3.60 (2.45)	0.476	0.003	0.870
Nitrite N	0.03 (0.03)	0.04 (0.04)	0.04 (0.05)	0.06 (0.05)	0.11 (0.12)	0.09 (0.10)	0.915	0.001	0.636
Nitrate N	1.82 (0.62)	2.82 (2.7)	2.21 (1.52)	2.76 (1.14)	3.71 (2.40)	3.51 (2.44)	0.465	0.004	0.865

Fig. 1 Vertical travel time (yr) to shallow groundwater in several paddocks taking **effective rainfall** of 0.75 m.

a)

- A Slurry Tank - possible point source
- B Old drain discharging to surface water - deep and interacts with groundwater
- C Historical dirty water application site - possible point source
- D Gravel lenses with increased infiltration capacity

b)

- Groundwater Flow Direction
- Head Contour (m AOD)

c)

Fig. 2 a) Groundwater flow map at Solohead Research Farm. Groundwater head contours are based on watertable data from 380 wells installed in this area. Each well is indicated by a dot on the map; b) designated experimental treatments; c) chloride spatial distribution over experimental period.

Fig. 3 Mean groundwater level (m AOD). Vertical bars represent the maximum and minimum level.

Fig. 4 Meteorological data at Solohead Research Farm from February 2008 to February 2010: a) daily rainfall (mm d^{-1}), b) daily evapotranspiration (mm d^{-1}), c) effective rainfall (mm d^{-1}), d) soil moisture deficit (mm d^{-1}) and e) daily mean of soil temperature at 0.05 m depth ($^{\circ}\text{C}$).

Fig. 5 Temporal variation of concentrations (mg L⁻¹) of a) total N (TN), b) dissolved organic N (DON), c) ammonium N, d) nitrite N, e) nitrate N and f) chloride in groundwater under three dairy systems: ES-100N - early spring calving with Fertiliser N, ES-0N - early spring calving without Fertiliser N, LS-0N - late spring calving without Fertiliser N.

a) DON

b) Ammonium N

c) Nitrate N

d) Nitrate N/Cl⁻

Fig. 6 Spatial distribution across the farm (mg L^{-1}) a) mean dissolved organic N (DON) b) ammonium N c) nitrate N and d) associated nitrate/Cl⁻ ratio in shallow groundwater over the study period

Fig. 7 Cumulative N losses (kg ha⁻¹) from three dairy production systems to groundwater in relation to cumulative effective rainfall (mm) in 2008/09: a) early spring calving with Fertiliser N, b) early spring calving without Fertiliser N, c) late spring calving without Fertiliser N and in 2009/10: d) early spring calving with Fertiliser N e) early spring calving without Fertiliser N, f) late spring calving without Fertiliser.

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