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Impact of local weather conditions and agronomic practices on groundwater quality in a karst aquifer on an intensive dairy farm in Southern Ireland

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ABSTRACT

Exploring the relationship between agricultural nitrogen (N) loading on a dairy farm and groundwater reactive N concentration such as nitrate (NO_3^-) is particularly challenging in areas underlain by thin soils and karstified limestone aquifers. The present study on such a site aims to relate changes in detailed paddock specific agronomic N-loading, hydrogeological and geological site characteristics with N occurrences in groundwater over a 10 year period from 2001 to 2011. In addition, the concept of vertical time lag from source to receptor is considered. Statistical analysis used regression with automatic variable selection. Four scenarios were proposed to describe the relationships between paddock and groundwater wells using topographic and hydrogeological assumptions. Monitored nitrate (NO_3^-) concentrations in the studied limestone aquifer showed a general decrease in the observed time period (2002 - 2011). Statistical results showed that a combination of improved agronomic practices and site specific characteristics such as thicknesses of the soil and

unsaturated zone together with hydrogeological connections of wells and local weather conditions such as rainfall, sunshine and soil moisture deficit (SMD) were important explanatory variables for NO₃ - concentrations. Statistical results suggested that the following agronomic changes improved groundwater quality over the 10 year period: reductions in inorganic fertiliser usage, improvements in timing of slurry application, the movement of a dairy soiled water (DSW) irrigator to less karstified areas of the farm and the usage of minimum cultivation reseeded on the farm. In many cases the explanatory variables of farm management practices tended to become more important after a 1 or 2 year time lag. Results indicated that the present approach can be used to elucidate the effect of farm management changes to groundwater quality and therefore the assessment of present and future legislation implementations.

Keywords: Farming practices; Farm management; Nitrate; Karst; Groundwater quality.

1. Introduction

Global population growth is predicted to increase the demand for food by up to 100% by 2050 (Godfray et al., 2010). To meet the growing worldwide need for agriculture produce, environmentally sustainable, economically viable and productive farming systems are required (Tilman et al., 2002). In Ireland, agriculture is dominated by dairy and beef cattle production from managed grassland (CSO, 2011). The European Union (EU) milk policy is due to change radically in 2015 with the abolition of farm level milk quotas and the ambitious target of a 50% increase in milk production by 2020 has been set in Ireland under the Food Harvest report (DAFF,

2010). Such targets for the agri-food sector must be achieved within current EU environmental legislation and will be further exacerbated by climate change such as an increase in precipitation during the winter time (Brouyère et al., 2004). The EU Water Framework Directive (WFD; OJEC, 2000) is a multi-part and multi-stage piece of legislation that aims, *inter alia*, to achieve at least “good” water quality status in all water bodies by 2015 with programmes of measures (POM) to achieve such a status implemented by 2012. In Ireland, the Nitrates Directive (EC, 2001) implemented since 2007 is Ireland’s agricultural POM. This Directive places restrictions on all potential N inputs into a farming system including: cattle stocking rates with a default of 170 kg N per ha⁻¹ or 250 kg N per ha⁻¹ on derogation farms (present study site), organic and inorganic fertiliser rates of use, the time of spreading and their storage. Closed periods are in place for spreading of inorganic (September to January) and some organic slurry (October to January) and farmyard manure (November to January) fertiliser. Application of DSW may occur provided there is no rain forecast within 48 hours of application and application rates must not exceed 50 m³ ha⁻¹. In general, 59% of Ireland’s rivers, over 47% of the lakes, 64% of the estuaries and 85% of the groundwater are already at “good” to “high” ecological status (EPA, 2010). For areas where the targets of the WFD will not be achieved by 2015 further legislative steps may be taken in areas of non-compliance and this could reduce farm productivity or at least add to production costs in some circumstances (Dillon and Delaby, 2009).

Leaching of N fluxes from an agricultural system to groundwater occur from point sources such as farmyard storage or from diffuse chronic sources from soil or through incidental losses during or after application of fertilisers especially when this

coincides with an episodic rainfall event (Basu et al., 2011; Brennan et al., 2012; Fenton et al., 2009a). Once anthropogenic reactive N (N_r) is lost it cascades through the environment (Galloway and Cowling, 2002) and occurs in many forms in groundwater such as NO_3^- , nitrite (NO_2^-) and ammonium (NH_4^+). Stuart et al. (2011) indicate that leached losses could increase in future decades due to predicted changes in agricultural land use and precipitation as well as an increase in temperature and evapotranspiration in the UK. The assessment of the effect of weather variation such as rainfall intensity on NO_3^- leaching is complicated by the requirement for long term datasets of groundwater chemistry, farm management practices and meteorology (Randall and Vetsch, 2005). Local weather changes can result in reduced agronomic response to fertiliser application resulting in reduced yields and greater nitrogen surpluses on farms (Derby et al. 2005). Feaga et al. (2010) linked rainfall quantity and timing with fluctuations of soil solution NO_3^- concentrations and load losses. Soil moisture stress has also been linked with NO_3^- leaching. Tyson et al. (1997) found that NO_3^- leaching over a 13-year period from the fertilized grassland was significantly correlated with the preceding summer's soil moisture deficit (SMD), with the highest losses after dry summers. Soil moisture stress can contribute to increased NO_3^- leaching through reducing crop uptake and/or organic N released from the soil microbial biomass during soil rewetting (Halverson et al., 2000). In addition, it can be assumed that climate change will play an important role to the hydrological cycle with changes to recharge, groundwater levels and flow processes including subsequent changes to groundwater quality (Brouyère et al., 2004).

Karst aquifers are an important water resource, which cover about 20% of the

the world's population (Ford and Williams, 2007). Although karst aquifers are very vulnerable in terms of water quality, the exploration, understanding and interpretation of karst aquifers is still rather challenging mainly due to fast groundwater flow velocities in the conduit systems (Goldscheider et al., 2007). Classical hydrogeological site investigations such as pumping test analysis and/or determination of groundwater isolines have a high potential for failure as the results often only reflect the specific (i.e. local) area that has been monitored and do not show the flow behavior of the entire study area (Bakalowicz, 2005). The characterisation of conduit systems has many complications such as spatial distribution of the conduits and temporally variable discharge (Goldscheider et al., 2008). To elucidate the shape and connections of shallow conduits, 2D and 3D geoelectric resistivity surveying (Hamdan et al., 2012) has been used as well as microgravity surveying in karst systems (Hickey, 2010).

Exploratory data analysis applied to groundwater NO_3^- - data is an effective means of explaining spatial and temporal trends of NO_3^- - in shallow groundwater (< 30 m) (Nas, 2009). Maximum likelihood Tobit regression analyses (sets a censored NO_3^- - concentration e.g. background level and builds a model based on the significance of explanatory variables) has been used by many to investigate elevated NO_3^- - concentrations in aquifer systems (Fenton et al., 2009b; Lichtenberg and Shapiro, 1997; Yen et al., 1996). Explanatory variables across these studies include but are not limited to: landuse around individual monitoring wells, distance of the monitoring well from potential point sources, saturated hydraulic conductivity (k_s) of screen intervals, screen interval depth, depth to top of aquifer, denitrification potential determined by groundwater di-nitrogen (N_2)/argon (Ar) ratios, redox potential, earth's dry ice-free surface and provide potable water for approximately 20-25% of

dissolved oxygen concentration and N_2 . Other techniques such as logistic regression can predict the likelihood that a certain groundwater threshold concentration will be breached (Menció et al., 2011). This can also be used to find significant explanatory variables that explain spatial and temporal patterns of groundwater NO_3^- concentrations (e.g. well depth, geology and presence of a fracture network, nitrogen fertiliser loading, soil drainage class percentages, seasonality of water table position) (Nolan, 2001; Tesoriero and Voss, 1997). Furthermore, Oenema et al. (2010) used multiple linear regression to evaluate the significance of different agricultural practices on NO_3^- groundwater occurrences in the Netherlands.

Many studies have been undertaken to help to define, develop and improve best management practices to achieve better groundwater quality worldwide (Jalali, 2005; Thorburn et al., 2003; Zhang et al., 1996;). However, exploring relationships between farm management practices and groundwater water quality is further complicated due to time lags from source to receptor via hydrological and hydrogeological pathways (Wang et al., 2012). For Ireland, it is now clear that the achievement of WFD targets by 2015 may not be possible where time lags are too long (Fenton et al., 2011a). Such time lags depend on socio-economic factors, soil/subsoil type, bedrock geology/hydrogeology and climatic factors such as rainfall (Stark and Richards, 2008) and should be estimated when attempting to relate agricultural management and groundwater quality (Meals et al., 2010). Farms present in areas of moderate to high recharge, with shallow free draining soils of low effective porosity (n_e), underlain by extremely vulnerable limestone aquifers typically have: 1) optimal conditions for intensive dairy farming and 2) the shortest vertical travel times to groundwater (1-2 years on the current study site e.g. Fenton et al., 2009a).

Therefore, such farms have the capacity to affect groundwater quality quickly through management change, but it is difficult to provide a tool for the prediction of time lag that has to be simple on the one hand and be reflective of a highly complex environment on the other.

To date there has been limited work relating long term farm management and local weather variation with NO₃ - concentrations in groundwater at farm scale, especially in highly vulnerable areas. The objective of this study is to relate changes in farm management and local weather conditions with groundwater quality over a 10 year period on an intensive dairy farm with free draining soils and a vulnerable limestone aquifer, whilst also considering time lag.

2. Materials and methods

2.1 Site description and characterisation

The intensive dairy farm study site (48.1 ha) at the Teagasc Dairy Production Centre, Fermoy, Co. Cork (8°15'W, 52°10'N) is located in a lowland limestone area in southern Ireland. The site is up-gradient of the Funshion River, close to a public water supply well and down-gradient of the large River Blackwater (Fig. 1). The perennial grassland farm is located on a limestone plateau with flat topography and negligible runoff. Two inferred groundwater divides are presented in Fig. 1, emanating from the juncture of the two rivers and intersecting the southern boundary of the site (Kelly and Motherway, 2000; Preston and Mills, 2002). The study site consists of 11 boreholes (BH 1-12, note BH 6 collapsed shortly after installation and was not suitable for this study) drilled at different stages since 2001 and are distributed across the entire farm

(Fig. 1). Three wells (BH 4, BH 11, BH 12) are 150 mm diameter open boreholes and

the remainder consist of a 50 mm diameter piezometer casing. Average drilling depth on site is 40.8 m (minimum depth of 22.0 m at BH 5 and maximum depth of 59.5 m at BH 3).

2.2 Soil and geology

The soil consists of a freely drained acid brown earth (Haplic cambisol), derived from mixed sandstone-limestone glacial till (Kramers et al., 2009). Soil thickness ranges from 0 to 4.5 m (Bartley and Johnston, 2006), which is underlain by a karstified Waulsortian limestone bedrock commonly occurring at an average of 2.5 m depth (Bartley, 2003). The A horizon of the soil consists of 53% sand, 31% silt, 16% clay with a dry bulk density of 1.1 g cm^{-3} and a total porosity of 52% (Kramers et al., 2009). This is confounded by limited preferential flow in the A-B soil horizons (Kramers et al., 2009). The Waulsortian Limestone covers 1.7% of the land area of Ireland (Ryan et al., 2006a) and is generally more karstified and less bedded than other limestone types such as the Ballysteen Formation (Fig. 1), because of the occurrence of coalescenced massive calcareous mud-mounds and the much lower content of the shale components (GSI, 2000). A land survey was carried out to determine the borehole surface elevation for comparing gathered water table depth of different boreholes with each other in m above ordinance datum (AOD). Soil thickness, thickness of the epikarst, depth of unsaturated zone, k_s and connectivity between the borehole (open or piezometer casing) were gathered for each borehole based on drilling logs and data collected from Bartley (2003).

2.3 Hydrogeology and groundwater quality

On the study site a bromide tracer test was performed on the surface around BH 7 by Bartley (2003), which indicated a hydrogeological pathway from BH 7 to BH 5, BH 9, and BH 10 (Fig. 2). Depth from surface to groundwater was measured regularly (in total 72 times) between July 2001 and September 2003, sporadically between 2004 and 2011 and more intensively in May 2011 for a shorter time period. All well elevations were surveyed and depth to groundwater converted to hydraulic heads in m AOD. Drilling logs, failed boreholes and resistivity profiles indicate the abundance of dry locations in the farm subsurface up to a depth of 50 m and these are indicated in Fig. 2. This indicates the possibility that the observed heads do not represent a true water table, but rather heads in discrete conduits and fractures. Thus, a conduit flow hypothesis is supported, in which the conduits dominate the flow with the existence of a perched watertable at discrete locations. A farm-scale hydrogeological investigation was established in 2001, which also included the measurements of monthly NO_3^- concentrations in groundwater starting in 2002. Briefly, after purging the previous day, a Grundfoss pump was used to collect 100 ml of groundwater. The samples were filtered immediately, using a $0.45\ \mu\text{m}$ micropore membrane filter, transferred to polyethylene screw top bottles, and frozen prior to chemical analysis. Analysis of groundwater quality followed the standard procedures such as described of Jahangir et al. (2012). For the present study NO_3^- , NO_2^- , NH_4^+ , TON, TN and DON were taken into account.

2.4 Geophysical survey

Surface conductivity (to 6 m depth) and resistivity (to 50 m depth) geophysical surveys (Apex Geoservices) of the farm were carried out to ascertain lateral and

vertical variations in overburden material, depth to bedrock and bedrock lithology

2.3 Hydrogeology and groundwater quality

(Fig. 2). Three electrical conductivity (EC) intervals from low (11-14 mS m⁻¹) to high (17-20 mS m⁻¹) were observed at the site (Fig. 2). High EC interpret collapse structures that are filled with finer materials such as silt or clay due to the karstified underground (Fig. 2). Using this information an elongated conduit system trending north-west to south-east is inferred in the middle of the farm. Furthermore, a resistivity profile (A-B, Fig. 2) show that bedrock is affected by karst features such as conduits, air-filled cavities and increased fracturing in the middle of the farm and general groundwater flow direction is also influenced by this conduit system (Fig. 2).

2.5 Local weather conditions

The studied local weather data consists of daily measurements of average, minimum and maximum air temperature, total solar radiation and daily rainfall, which were recorded at the experimental site during the entire study period. Daily meteorological input parameters (rainfall, temperature and sunshine hours) were inputted into the hybrid model for Irish grasslands of Schulte et al. (2005) to elucidate daily SMD, actual evapotranspiration and effective drainage (ED).

2.6 N loss

Nitrogen (N) loss (kg ha⁻¹) was determined annually by multiplying the average NO₃-N concentrations (mg L⁻¹) with ED (mm) as used in previous studies (e.g. Hooker et al., 2008; Salazar et al., 2012).

2.7 Agronomy

A total of 48.1 ha of permanent grassland containing greater than 80% of perennial ryegrass (*Lolium perenne*) and grazed exclusively by dairy cattle were used

for this study. The experimental site was used to compare diverse animal genotypes and nutritional treatments during the 10 year evaluation period (Coleman et al., 2008; Horan et al., 2004; McCarthy et al., 2012). In all experiments, a rotational grazing management system was practiced usually commencing in early February and concluding in late November each year. The frequency and intensity of grazing was recorded as the number of grazing days per hectare per month for each paddock. During the winter months between late November and early February, all animals were housed and all animal slurry was collected and stored. During periods of excessive rainfall during the grazing season, animals were occasionally housed and on-off grazing (Kennedy et al., 2009) was used as a management tool to facilitate grazing and to avoid soil structural damage. Best nutrient management practices have been applied on the farm in recent years with an increased focus since 2008 due to the implementation of the Nitrates Directive in 2007 to increase slurry use efficiency and reduce fertiliser N application to the levels stipulated under Statutory Instruments. All animal slurry generated from dairy cattle on site during winter was reapplied to the land area during the following grazing season. The total N inputs at the paddock level (weighted on the basis of paddock size) in the form of both inorganic and organic fertilisers and slurry were monthly recorded during the study, while off-takes of harvested grass for silage conservation were deducted. A centre pivot DSW irrigation system was operated on site to reapply dairy yard washings. In 2006 the area used for DSW irrigation was changed from the highly vulnerable middle area (10 ha) to the north-western area (22 ha) of the farm (Fig. 1). The total N irrigated as DSW is known for the years 2002, 2003 and 2004, because of N leaching studies of Ryan et al. (2006b) during that time where total N in DSW ranged from 20.0 to 823.0 mg L⁻¹ and the amount of N applied ranged from 0.5 to 84.7 kg ha⁻¹. Approximately, 15% of

the total farm area was reseeded annually from 2006 to 2011. In 2006 and 2007, seedbed preparation for reseeded was achieved by inversion ploughing. However, this practice was discontinued and replaced by minimum tilling cultivation techniques from 2008 onwards.

2.8 Conceptual site model

By combining the aforementioned collected data, a conceptual site model was developed and is illustrated in Fig. 3. The most vulnerable part of the site is north of the groundwater divide in the central paddocks of the farm (Fig. 2). Short unsaturated zone travel times to groundwater are driven by high ED and thin soil. Soil NO_3^- - resulting from inorganic/organic fertilisation, grazing animal urine/dung returns, soil mineralisation and atmospheric N deposition can rapidly migrate along a well-connected conduit system to down gradient receptors. Large cavities in the karstified rock are known, but connectivity to the larger conduit system can be low, which is known because of the long recovery duration of the aquifer after pumping at BH 4.

2.9 Statistical analysis

2.9.1 Descriptive analysis

Mixed models (Proc Mixed, SAS Institute, 2006) of repeated measures as used by Philibert et al. (2012) were carried out to determine the effect of month and year on climatic and groundwater quality data with month included as a repeated effect within borehole. A compound symmetry covariance structure among records within borehole provided the best fit to the data and Tukey's test was used to determine differences between treatment means.

2.9.2 Regression analysis

Relationships between groundwater quality data such as NO_3^- , TON, TN, DON, NO_2^- and NH_4^+ (Table 1) as response variables (Table 2) and the possible explanatory variables in the overall dataset (Table 3), which are related to previous sections, were explored by regression using automatic variable selection. As the agronomic inputs (e.g. slurry application in kg N ha^{-1}) and outputs (e.g. silage harvest in kg N ha^{-1}) for the paddocks were available on an annual basis, the monthly measurements and records were summarised for the analysis. The variables were modelled using Normal distribution based multiple linear regression except for the response variables NO_2^- and NH_4^+ (Table 2) as NO_2^- and NH_4^+ observations were heavily censored below the detection limit. Therefore, for NO_2^- and NH_4^+ a count of detections was used in a logistic regression based on the proportion of detections observed. To assess evidence of time lag, the explanatory variables (Table 3) were lagged from 0 to 3 years (based on the proposed time lag of 1 to 2 years for this site by Fenton et al., 2009) and the effect on variable selection was explored. The size of F and Wald statistics was used for the interpretation of relative importance in explanatory variables (Table 3) influencing groundwater quality (Table 2).

For NO_3^- , TON, NO_2^- and NH_4^+ (Table 2) two approaches were taken: 1) Differentiation between years and 2) Bulk period assumption (Fig. 4). The first approach included fitting year as a factor in the statistical analysis, effectively examining processes within year. This allowed a broad assessment of the extent to which the measured explanatory variables in the data set (Table 3) were sufficient to describe the overall processes. The second approach provided an approximation to the best description of the constant processes such as water table deviations or surface

conductivity that was possible with the information available. Without year as a factor all changes over the period of investigation have to be ‘explained’ explicitly by the statistical outcome including the first approach (differentiation between years). As a consequence of numerical difficulties with the data for TN and DON only the first approach without year as a factor was reliable and therefore used for evaluation. Goodness of fit statistics (R^2) was calculated using the reduction in residual variance between a model with intercept only and the full fitted model.

Four scenarios were proposed for the statistical analysis (Fig. 4). Each scenario defined sets of paddocks for each borehole that were likely to contribute to the observed responses. The different scenarios were based on topographic assumptions (e.g. concentric distribution of paddocks around the borehole; Fig. 1) and the hydrogeological assumptions on groundwater pathways from an on-site tracer experiment (Bartley, 2003). The difference between the four scenarios is illustrated in Figure 5 taking BH 9 as an example. In general, for scenario 2 a smaller catchment area with 25 paddocks was taken into account compared to scenario 4 where the greater catchment area included 34 paddocks. In addition to the proposed scenarios, a possible time lag of 0 to 3 years was considered (Fig. 4). Thus, 96 cases were evaluated for the statistical analysis.

3. Results

3.1 Local weather conditions

Mean monthly rainfall and ED data for 10 years (2001-2011) and the 30 year average (1981-2011) are presented in Table 4. Weather conditions varied considerably between years during the study period. Rainfall averaged 996 mm over the 10 year

period, whereas the 30 year average was 1022 mm. Over the 10 year period, the highest monthly rainfall and ED was recorded in November (115 and 89 mm, respectively) while February (61 and 38 mm, respectively) and April (61 and 18 mm, respectively) had the lowest monthly rainfall and ED. From the 30 year average data, rainfall was highest in October (112 mm) including 70 mm ED and lowest in July (64 mm) including 9 mm ED. Mean monthly ED was 39.9 mm during the study and was highest in 2002 and 2009 (63.9 and 57.0 mm, respectively) and lowest during 2010 and 2011 (27.9 and 26.6 mm, respectively).

3.2 Agronomy (2001 - 2011)

A broad characterisation of farming practices at the experimental site during the study period is outlined in Table 5. A Code of Good Agricultural Practice to Protect Waters from Pollution by Nitrates gave guidance and advice to farmers up to 2007 (Anon, 1996). This changed in 2007 when the Nitrates Directive was implemented in Ireland, which was subsequently taken as Ireland's agricultural POM under the EU WFD. Proposing a time lag effect of 1 year on the present site, management changes coupled with ED in subsequent years changed the overall NO_3^- trends on the farm (Fig. 4). During the 10 year study period, the overall dairy herd size increased from 108 to 138 dairy cows (equivalent to a stocking rate increase of 28%). Grazing season length was increased from 231 days in 2001 to highs of 295 and 306 days in 2005 and 2007, respectively. From 2002 this figure was always greater than 272 days and is presently maintained at greater than 280 since 2009. Fertiliser N was reduced to comply with the nitrates regulations introduced in Ireland in 2007, while feed N input was also reduced based upon experimental requirements. The overall reduction in fertiliser and feed N use and increased overall farm stocking

rate was achieved by increasing organic fertiliser application during spring to replace inorganic N application and by increasing grazed grass utilisation at the experimental site. As a consequence of the overall increase in herd size, both milk and milk fat plus protein production increased during the study.

3.3 Groundwater quality trends

From 2002-2011, the combined application of DSW, slurry and chemical fertiliser was relatively consistent including reductions after the implementation of the Nitrates Directive in 2007 (Fig. 6). Concentrations of NO_3^- in groundwater were highly variable throughout the study, but were typically greatest during autumn and early winter. In addition, some very high NO_3^- concentrations (up to $59 \text{ mg NO}_3\text{-N L}^{-1}$) occurred close to two boreholes (BH 7 and BH 8) in the middle of the farm in 2002 when the DSW irrigator was placed in this highly vulnerable area (Fig. 1, Fig. 2). In the present study, on average, groundwater $\text{NO}_3\text{-N}$ concentrations across the farm declined over the study from 16.0 mg L^{-1} in 2002 to 7.3 mg L^{-1} during 2010 with a low of 6.6 mg L^{-1} in 2011 (Fig. 6). The overall mean concentration of NO_3^- exceeded the maximum allowable concentration (MAC) in groundwater defined by the WFD in Ireland ($11.3 \text{ mg NO}_3\text{-N L}^{-1}$) during the first 7 years of the study and declined below the MAC for the last 3 years of the study period (2009, 2010 and 2011).

3.4 N loss

The estimated N loss decreased in the study period in total with the maximum value of 76 kg ha^{-1} in 2002 and the minimum value of 25 kg ha^{-1} in 2011 including a deviation between 2005 and 2009, when mean N loss ranged from 49 to 58 kg ha^{-1} (Fig. 4). In 2002 high N losses were related to high $\text{NO}_3\text{-N}$ concentrations in

groundwater in addition to high ED. After the decrease of N losses in 2004 and 2005, N losses increased from 2005 to 2009. The increased N losses between 2005 and 2006 referred to high NO₃-N concentrations and medium ED. In 2007 N losses were related to high NO₃-N concentrations and the lowest ED of the study period, whereas in the following two years N losses referred to medium (in 2008) and high (in 2009) ED coupled with already decreasing NO₃-N concentrations. From 2010 to 2011 ED was low and NO₃-N concentrations were under MAC ensuring lower N losses than previous years.

3.5 Statistical analysis

The scenarios created for NO₃⁻, TON, TN and DON using Normal-based regression with automatic selection had a range of R² from 0.43 to 0.79. Best results were achieved for NO₃⁻ from scenario 1 (Fig. 5). In this case the R² ranged from 0.68 to 0.75 including a slightly better result with year fitted as a factor than for selections with year not fitted (Fig. 4). For NO₂⁻ and NH₄⁺ the model selections for the logistic regressions were likely to be liberal as there were technical difficulties in fitting a repeated measures structure to the limited, year-averaged dataset.

Comparing scenarios 1 to 4 (Fig. 5) for NO₃⁻ with each other, scenario 1 seems to give the most interesting results from a climatic, geological and agronomic perspective. The F statistic indicates that the main factors influencing NO₃⁻ concentrations in groundwater were soil and rock thickness in the unsaturated zone to the top of the water table and the connections of the boreholes especially those already known by the tracer test of Bartley (2003) (BH 7, BH 8, BH 9 and BH 10). Other factors showing an effect on NO₃⁻ concentrations in groundwater were borehole

type (closed piezometer casing or open borehole), SMD, sunshine, year, the different intensity of karst features on the study site indicated by the geophysical survey, fertiliser, grazing days, silage and slurry.

For TON, the R^2 values are slightly lower than in the NO_3^- - scenarios ranging from 0.43 to 0.75 for the year fitted cases and from 0.40 to 0.74 for the year not fitted cases (Fig. 4). The results with the highest R^2 values were achieved for the year fitted cases for scenario 1, 3 and 4. Scenario 2 seemed to have less predictive potential compared to the other scenarios (Fig. 5). The statistical outcomes for TON data were similar to the NO_3^- outcome as well. The explanatory variables for local weather conditions (Table 3) such as sunshine and SMD showed a significant influence together with geological settings such as thickness of soil and epikarst in the unsaturated zone to the water table. Farm management practices appeared to be associated with a 1 and 2 year time lag. These practices included fertiliser application and grazing days if year was included, silage if not and slurry for year and year not fitted.

The R^2 values for TN and DON were around 0.5, which indicated less explanatory power than the analysis of NO_3^- - and TON. In these cases rainfall as a climatic factor had the most important influence. Nevertheless, farm management practices selected were fertiliser, slurry and grazing days. The outcome for DON in scenario 3 after a 2 year time shift proposed slurry to be the most important factor.

For NO_2^- - and NH_4^+ no single variable was most descriptive as the relative

the R^2 values reached only 0.49 as the maximum value. In addition to the factors that were already observed in the aforementioned cases, reseeding, ED and DSW appeared to be the first time. DSW appeared only for NH_4^+ and showed one of the highest influences on the NH_4^+ concentrations.

Overall the statistical results showed that geological settings such as soil and rock thickness in the unsaturated zone to the top of the water table and local weather conditions such as rainfall, sunshine and SMD consistently were important. In many cases the explanatory variables of farm management practices tended to become more important after 1 or 2 years of time lag, which concurs with those estimated by Fenton et al. (2011a).

4. Discussion

4.1 The approach taken

To comply with the Irish obligations pursuant to the EU Nitrate Directive derogation request (EC, 1991), the current study was undertaken to study the impact of local weather conditions, site specific conditions and agronomic management to groundwater quality beneath an intensive dairy production system. Several studies on grassland sites with less complex geology concur with some of the findings of this study (Levison and Novakowski, 2009). However, this study is unique as the statistical approach used herein, albeit with a high resolution 10 year dataset, allowed such an assessment to work on a more complex terrain. Such complex terrains are often avoided as they are deemed too expensive and complex to monitor. Also shorter term datasets on such terrains could result in inaccurate management decisions. It is also important to point out that such an approach is appropriate where nutrient importance of the variables was similar, as indicated by the Wald statistic. In addition,

concentrations and not fluxes are deemed to be important such as under the present restrictions of the EU WFD in which concentration thresholds and MAC are important and not fluxes. This negates the need for hydrogeological data collection such as hydraulic and physio-chemical spring responses e.g. by using environmental tracers or such as defining volume and storage capacity of the conduit system (Einsiedl, 2005).

4.2 Local weather conditions

Statistical results of the present study indicate that local weather conditions are always a factor to consider while studying groundwater quality. Given the temporal variability in weather conditions and NO_3^- concentrations over the period of this study, it was only possible to explore indicative relationships between NO_3^- concentrations in groundwater and climate, (hydro-)geological factors and surface level nutrient management (see also Fenton et al., 2011b; Fraters et al., 2005). As shown in previous hydrogeological studies in karst environments, high rainfall events coincide with major mobilisation of NO_3^- - in quick pulses through the unsaturated zone, rather than slow uniform recharge (Drew and Hötzl, 1999; Igbal and Krothe 1994). This is augmented for NO_3^- - originating from inorganic sources (Katz, 2004; Wells and Krothe, 1989). To gain a better impression of the impact of local weather conditions especially in karstified regions and to improve future management decisions, the current statistical approach would benefit from a higher resolution monitoring system such as high resolution sensors at a spring outlet or at least the collection of in-situ borehole mean nutrient concentrations over time via passive diffusion samplers.

Nitrogen fertiliser is well known as an important contributor to agricultural production (Whitehead, 1995), but the efficiency of N use within animal-based systems is often poor (Watson and Atkinson, 1999). The evaluation of the impact of grazing systems to the impact on water quality is complicated by the nature of water movement in soils, the possibility of external influences on groundwater quality and the time lag between the surface and groundwater (Baily et al., 2012; Fenton et al., 2011a).

In the current study, statistical analysis suggested that slurry and fertiliser application are closely related to NO_3^- - leaching. Organic and inorganic fertilisers can vary significantly due to their different properties and compositions. While organic waste has often less mineral N immediately available for plant uptake than inorganic fertilisers (Whitehead, 1995), inorganic fertilisers are often more likely to be affected by immediate leaching than organic wastes (Di et al., 1998; Thorburn et al., 2003). Di et al. (1998) emphasised that the application rate for organic wastes and inorganic fertiliser should be regulated differently according to their effects on NO_3^- - leaching. In addition, Lalor et al. (2011) observed that N use efficiency can be optimized by switching application of slurry from summer to spring (April instead of June because of cooler and wet weather conditions combined with strong grass growth) or changing the application method (e.g. use of the trailing shoe application method instead of the splashplate). On the present study site, improvements in slurry utilisation were achieved by choosing application times more specifically in spring and autumn, thereby reducing the requirement for inorganic fertiliser application from 2008 onwards. For example in 2001 the application of slurry was in May, whereas in 2008 the application was performed in January, February, March and April. In 2009 the

4.3 Agronomy (2001 - 2011)

application of slurry was performed only in February and March. The reduction in farm-gate fertiliser N use coupled with the increased overall stocking rate (and consequently milk production from the site) was indicative of increased N use efficiency on the research farm contributing to increased N retention within the farm system.

An earlier study on the present site by Bartley (2003) showed that groundwater NO_3^- concentrations were highest in the areas of highest organic N loading. In addition, Bartley and Lee (2007) observed a relationship between grazing intensity and average groundwater NO_3^- concentration during subsequent years on the study site. Similarly, Strebel et al. (1989) and Oenema et al. (2010) demonstrated that grazing is one of the most important factors that affects NO_3^- leaching at farm scale. In the present study the statistical results also indicate that grazing is an important factor that can have a significant effect on groundwater quality. This was notable especially after a 1 year time lag for NO_3^- and TON within scenario 1 (Fig. 5). The results of this analysis suggest that, although grazing intensity increased at the site over the study period and while nutrient management practices improved and NO_3^- concentrations decreased, increased grazing intensity should be strategically positioned on less vulnerable areas within the site (similar to DSW irrigation) to reduce risk to groundwater resources.

Division of a farm into high and low risk leaching areas could be an effective management approach for the positioning of a DSW irrigation system. Consistent with the current study, data from an EPA (2005) study point to the avoidance of DSW irrigation on vulnerable areas within the site as an effective strategy to improve

groundwater quality. By combining the DSW irrigation dataset of Ryan et al. (2006a) who monitored leaching observations taken in ceramic cups at a depth of 1 meter between 2001 and 2004 and the water quality information for BH 7 from 2002 and 2004, a relationship with increasing NO_3^- can be concluded. In 2006 the DSW irrigator was moved from the smaller high risk DWS area 1 zone to the larger lower risk zone of DSW area 2 (Fig. 1). It is noteworthy that this change coincided with a general decrease of NO_3^- concentrations on the farm including BH 7 and the boreholes affected by the DSW area 1 for which the connection is known from a bromide tracer experiment (Bartley, 2003). The statistical outcomes of this analysis indicate a relationship between DSW spreading and NH_4^+ in groundwater. However, the statistical approach adopted in this study did not find a relationship between DSW application and NO_3^- in groundwater, which could also indicate that other factors were more important for the overall NO_3^- concentration changes. In addition, assuming a time lag on a yearly basis could under predict DSW vertical travel times. This can be seen in context within the study of Gibbons et al. (2006) who observed that the duration of topsoil saturation following the application of large amounts of dairy wastewater at the same site can be very short during rainfall events.

Ploughing is well known as a contributor to soil organic N releases (Whitmore et al., 1992). Strebel et al. (1989) noted that ploughing results in a significant decrease of soil organic N content coupled with intensive NO_3^- leaching for a short time period until a new steady-state condition is achieved with less organic N content in the soil. The statistical results did not show a strong relationship between NO_3^- concentrations in groundwater and changes in management regarding the gradual transition of the start of the adoption of minimum till cultivation reseeded in 2006 and the stopping of

ploughing in 2008 although the change of this management practice coincided with a general decrease in NO_3^- concentrations. It may be the case that such a change was not a significant factor in the statistical results due to the 2 year transitional period involved or perhaps due to other factors such as a reduction in fertiliser inputs or improvement in slurry application techniques also introduced at this time. While acknowledging the difficulty of fitting a suitable correlation structure to the comparably smaller NH_4^+ concentration dataset, the statistical outcomes suggest that the used reseedling method can have an influence on NH_4^+ concentrations in groundwater.

4.4 Groundwater quality trends

On this site, statistical results indicated that the connectivity with the entire aquifer as opposed to screened intervals was a better predictor of N concentrations in the aquifer. Open boreholes are in contact with nutrients as they migrate vertically through the subsoil, through the weathered epikarst (the thickness of which was highly significant) and therefore represent a composite nutrient concentration where cross contamination from surface input to groundwater can occur. However, where fluxes are important (perhaps in the future of the EU WFD) discrete fractures or conduits may be more important (Haag and Kaupenjohann, 2001; Landig et al. 2010; Levison and Novakowski, 2009) and therefore discrete screens or packers in conjunction with open boreholes may be needed. Wells with integrated piezometers are drilled to a certain depth and the water strike may or may not have good connectivity with the aquifer. On the one hand wells with piezometer casing can give a more reliable measurement if the connectivity to the aquifer is good because the water samples that were taken are always from the same aquifer level. But on the

other hand the same type of well could give less reliable groundwater quality measurements if the connectivity is bad. In general, it needs to be taken into account if a water sample was taken from an open borehole or a borehole with piezometer especially if water samples are compared with each other.

The analysis of NO_3 - occurrence data from 2002 to 2011 showed overall a decreasing trend of mean NO_3 - concentrations on the farm (Fig. 6). The statistical results indicate that the implementation of the Nitrates Directive helped in some parts to improve the water quality on the study site. Van Grinsven et al. (2012) stated that a general, convincing decrease of NO_3 - in groundwater could not be observed in north-western countries of Europe since 2000 despite major improvements to soil N balance. As increased NO_3 - concentrations due to agriculture coupled with karst environments is not a concern for most of the north-western European countries, these countries also have to deal with longer time lags. In some areas in the UK time lags are even estimated up to several decades (Wang et al., 2012). This leads to the conclusion that the implementation of the Nitrates Directive could effectively lead to better water quality, but it may be the case that in most of the areas the improvement cannot be recognised quickly.

4.4 Statistical analysis

As indicated in the aforementioned discussion a statistical approach that incorporates a time lag effect can be used to predict future changes in water quality. It is also important to note that for a highly complex terrain such as in the present site the easiest statistical scenario (Scenario 1, Fig. 5) proved most effective. This prevents the immediate deployment of expensive hydrogeological equipment and also should

encourage researchers to attempt further investigations of equally complicated sites over a similar timeframe. Because of significant changes in farm management (i.e. appropriate slurry and fertiliser application rate and strategy including a significant reduction in fertiliser rate since 2008; avoiding ploughing; careful management of high risk zones within the farm) and the already declining NO_3^- concentrations in groundwater, it is expected that this site will be able to comply with desired water quality standards as stipulated by WFD into the future.

5. Conclusions

The statistical approach used herein is an effective method for exploring the relationships between farm management, local weather conditions and groundwater nutrient concentrations both spatially and temporally. Results can guide the expectations of farm managers and policy makers with respect to the achievement of water quality targets within certain time frames. It is especially useful for farming areas within the remit of the EU WFD as it is a nutrient concentration driven approach and not concerned with nutrient fluxes. It therefore allows the practitioner to explore complex terrains such as free draining soils underlain with karst limestone aquifers without the need for a high end hydrogeological investigation. Over the 10 year monitoring period, the results of this study indicate that a combination of site characteristics (i.e. depth of the unsaturated zone, soil/subsoil and rock thickness), local weather conditions (such as rainfall, sunshine and SMD) and agronomic practices (i.e. reduced fertiliser rate, appropriate slurry and DSW application strategy, minimum cultivation and strategic management of high risk zones) were important factors influencing NO_3^- concentrations in groundwater. Furthermore, these results indicate that improved nutrient management practices on a highly vulnerable site with

free draining soil can have relatively fast impacts (2 years) on groundwater quality and can lead to an achievement of the water quality targets set by for example the WFD.

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Captions for Figures

Fig. 1. Location and characteristics of the study site.

Fig. 2. Surface conductivity map on the study site including resistivity profile, groundwater flow direction trend and dry areas.

Fig. 3. Conceptual site model using data from 2005 as an example.

Fig. 4. Organisational chart of the regression analysis.

Fig. 5. Paddock to borehole relationship of scenario 1 to 4 as used in the regression analysis taken BH 9 as an example.

Fig. 6. Nitrate concentrations, N-loss, N application and precipitation during the study period.

Captions for Tables

Table 1. The concentration of Nitrogen in groundwater during the study period

Table 2. Response variables used for regression analysis at farm scale.

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Table 4. Cumulative monthly rainfall and effective drainage for the study site during the study period (2001 to 2011) compared to the 30 year average.

Table 5. Farm system characteristics at the experimental site (2001 to 2010).

Figures

Fig. 2. Surface conductivity map on the study site including resistivity profile, groundwater flow direction trend and dry areas. a) Electrical conductivity on the study site with information of dry areas and general groundwater flow direction. b) Resistivity profile. c) Geological interpretation of resistivity profile.

Fig. 3. Conceptual site model using data from 2005 as an example.

Fig. 4. Organisational chart of the regression analysis.

Fig. 5. Paddock to borehole relationship of scenario 1 to 4 as used in the regression analysis taking BH 9 as an example. a) One paddock associated with a borehole within this paddock. b) The assumption of paddock to borehole relationship was made using a small catchment area (25 paddocks for all boreholes) by reverting to the known hydrogeological pathways from a tracer experiment (Bartley, 2003) and the general groundwater flow direction towards northeast c) Topographic assumption using concentric relationships. d) Same assumptions were taken as in b) but with a greater catchment area (34 paddocks for all boreholes).

Fig. 6. Mean NO₃-N concentrations, mean N-loss, total N application and precipitation per year during the study period. In 2010 additional slurry application was to replace DSW applications due to a dysfunctional DSW irrigator system.

Tables

Table 1. The concentration of Nitrogen in groundwater during the study period.

	Total No. of Samples	$\pm^*$	Mean	Median	Minimum	Maximum
NO ₃ -N (mg L ⁻¹)	694	6.1	11.6	11.2	0.0	59.0
NO ₂ -N (mg L ⁻¹)	694	0.1	0.1	0.0	0.0	1.9
NH ₄ -N (mg L ⁻¹)	656	0.9	0.3	0.0	0.0	11.0
Total N (mg L ⁻¹)	172	5.7	11.3	11.7	0.4	33.8
Dissolved organic N (mg L ⁻¹)	172	2.7	1.9	1.1	0.0	19.8

*Std Dev: standard deviation.

Table 2. Response variables used for regression analysis at farm scale.

Topic	Response variable	Explanation	Range
Ground-water quality	NO ₃ -N	Nitrate nitrogen in groundwater in mg L ⁻¹	0 - 59.0
	TON	Total organic nitrogen in groundwater in mg L ⁻¹	0 - 59.0
	TN	Total nitrogen in groundwater in mg L ⁻¹	2.5 - 27.0
	DON	Dissolved organic nitrogen in mg L ⁻¹	0 - 19.8
	NH ₄ -N	Ammonium nitrogen in groundwater in mg L ⁻¹	0 - 11.0
	NO ₂ -N	Nitrite nitrogen in groundwater in mg L ⁻¹	0 - 1.9

Table 3. Explanatory variables used for regression analysis at farm scale.

Topic	Explanatory variable	Explanation	Range
Local weather conditions	rainfall	Monthly rainfall in [mm]	3.5 - 259.5
	EFFrainfrd	Free drained soil in [mm month ⁻¹]	0.0 - 100.0
	temp	Monthly mean temperature in [°C]	0.6 - 17.4
	sunshine	Monthly cumulative sunshine hours in [hr]	23.7 - 243.4
	SMDfrdTOT	Monthly cumulative soil moisture deficit in [mm]	1.5 - 2251.2
	SMDfrdAVER	Monthly average soil moisture deficit in [mm]	0.0 - 72.6
	SMDfrdMAX	Monthly maximum soil moisture deficit in [mm]	0.5 - 82.3
Agriculture	reseeded	Reseeding status during the study period (no reseeding = 0 - 2 0; ploughing and reseeding = 1; minimum-tillage reseeding = 2)	
	fertiliser	Yearly fertiliser application in [kg N ha ⁻¹] per paddock	0 - 420
	slurry	Yearly slurry application in [kg N ha ⁻¹] per paddock	0 - 282
	DSW	Yearly mean irrigation of dairy soiled water in [kg N ha ⁻¹] estimated from observations of Ryan et al. (2006b) per paddock (no irrigation = 0; irrigation = 67)	0 - 67
	silageNremov	Amount of N removed because of silage harvest on the paddock in [kg N ha ⁻¹]	0 - 210
totN		Total N application (fertiliser, slurry and dairy soiled water less silage harvest) in [kg N ha ⁻¹]	0 - 459
(Hydro-)geology	grazingd	Yearly grazing days per [ha]	5.8 - 32.4
	zAOD	End of well in [m] above ordinance datum (AOD)	-5.9 - 21.5
	zbg1	Total depth of each well below ground level in [m]	33.0 - 59.5
	grelevAOD	Ground elevation on the surface at each well in [m] AOD	52.2 - 56.0
	toprockAOD	First rock appearance in well in [m] AOD	49.6 - 54.0
	soilthick	Soil thickness at each well in [m]	2.0 - 4.0
	epikthick	Thickness of epikarst at each well in [m]	30.5 - 57.0
	soi lrockthickWT	Thickness of soil and epikarst to the water table at each well in [m]	20.6 - 28.5
	screenotopelev	Top of screen in well with piezometer casing in [m] AOD	22.2 - 27.5
	screenbottomelev	Bottom of screen in well with piezometer casing in [m] AOD	18.2 - 24.5
	piezoropen	Open well or well with piezometer casing (p = piezometer; o = open well)	
	maxwtableAOD	Maximum water table above ordinance datum in [m] taken from 72 measurements between 2001 and 2003 after Bartley (2003)	25.1 - 26.1
	minwtableAOD	Minimum water table rise above ordinance datum in [m] taken from 72 measurements between 2001 and 2003 after Bartley (2003)	29.5 - 41.1
	wtablelrange	Range of water table deviation in [m]	4.0 - 15.0
	kf	Hydraulic conductivity in [m day ⁻¹] after Bartley (2003)	0.004 - 27.0
	kfrange	Defined zones with low, medium, high hydraulic conductivity (Kf) (Kf low = 1; Kf medium = 2; Kf high = 3)	1 - 3
	geophysmSm	Surface conductivity at each well in [mS m ⁻¹]	11.5 - 16.0
	geopysKAT	Defined zones with low, medium, high surface conductivity at each well (low conductivity = 1; medium conductivity = 2; high conductivity = 3)	1 - 3

Table 4. Cumulative monthly rainfall and effective drainage for the study site during the study period (2001 to 2011) compared to the 30 year average.

Month	Rainfall (mm)		Effective Drainage (mm)	
	30 year average	2001 - 2011	30 year average	2001 - 2011
January	109	109	88	89
February	80	61	58	38
March	85	79	51	39
April	66	61	26	18
May	69	78	17	19
June	70	75	14	10
July	64	86	9	19
August	84	69	26	23
September	77	74	25	13
October	112	108	70	62
November	104	115	79	89
December	102	81	82	59
Total Annual	1,022	996	545	478

Table 5. Farm system characteristics at the study site (2001 to 2011).

Year	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Experimental cow (No.)	108	117	117	117	126	126	128	140	138	138	138
Stocking rate (cows ha ⁻¹)	2.25	2.44	2.44	2.44	2.63	2.63	2.67	2.92	2.88	2.88	2.88
Grazing season (days)	231	272	293	291	295	273	306	301	287	282	285
N inputs (kg ha ⁻¹)											
Fertiliser	294	294	289	296	331	259	313	244	248	252	249
Feed	41	41	39	35	37	40	20	23	29	36	25
Total	335	335	328	331	368	299	333	267	277	288	274
N exports (kg ha ⁻¹)											
Total	75	88	92	93	93	91	89	93	92	96	98
N balance											
Surplus (kg ha ⁻¹)	260	247	236	238	275	208	244	174	180	180	174
N-use efficiency (%)	22.4	26.2	28.0	28.1	25.4	30.5	26.7	34.8	33.9	34.9	36.0
Milk production											
Milk volume ('000 L ha ⁻¹)	12.4	14.6	15.6	15.5	15.5	15.5	14.6	14.6	14.4	15.5	15.3
Fat plus protein (tons ha ⁻¹)	0.93	1.11	1.16	1.20	1.21	1.19	1.13	1.18	1.18	1.27	1.28