

Variations in travel time for N loading to groundwaters in four case studies in Ireland: Implications for policy makers and regulators

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Abstract

Mitigation measures to protect waterbodies must be implemented by 2012 to meet the requirements of the EU Water Framework Directive. The efficacy of these measures will be assessed in 2015.

Whilst diffuse N pathways between source and receptor are generally long and complex, EU legislation does not account for differences in hydrological travel time distributions that may result in different water quality response times. The “lag time” between introducing mitigation measures and first improvements in water quality is likely to be different in different catchments; a process that should be considered by policy makers and catchment managers. Many examples of travel time variations have been quoted in the literature but no Irish specific examples are available. Lag times based on initial nutrient breakthrough at four contrasting sites were estimated to a receptor 500 m away from a source. Vertical travel times were estimated using a combination of depth of infiltration calculations based on effective rainfall and subsoil physical parameters and existing hydrological tracer data. Horizontal travel times were estimated using a combination of Darcian linear velocity calculations and existing tracer migration data. Total travel times, assuming no biogeochemical processes, ranged from months to decades between the contrasting sites; the shortest times occurred under thin soil/subsoil on karst limestone and the longest times through thick low permeability soils/subsoils over poorly productive aquifers. Policy makers should consider hydrological lag times when assessing the efficacy of mitigation measures introduced under the Water Framework Directive. This lag time reflects complete flushing of a particular nutrient from source to receptor. Further research is required to assess the potential mitigation of nitrate through denitrification along the pathway from source to receptor.

Key Index Words: Lag time, Water Framework Directive, mitigation, nitrate

Introduction

Intensification of agriculture poses a challenge to the sustainable management of soils, water resources and biodiversity. Nitrogen losses from agricultural areas can contribute to

ground and surface water pollution (Stark and Richards, 2008; Humphreys *et al.*, 2009). The Water Framework Directive (WFD; European Parliament and Council, 2000) attempts to achieve at least “good ecological status” for all

waterbodies by 2015 with programmes of measures in place by 2012. The Nitrates Directive (European Council, 1991) enacted in the Republic of Ireland (from now on referred to as Ireland) in 2006 under Statutory Instrument (SI) 101, 2009 is currently the main legislative mitigation measure in place to achieve the goals of the WFD. The Nitrates Directive sets limits on stocking rates on farms in terms of the quantity of N from livestock manure that can be applied mechanically or directly deposited by grazing livestock on agricultural land. A limit of 170 kg N ha⁻¹ year⁻¹ from livestock manure was set. However, the EU Nitrates Committee approved Ireland's application for a derogation of this limit to allow grassland-based (mostly dairy) farmers to operate at up to 250 kg N ha⁻¹ year⁻¹ from livestock manures, under the understanding that this derogation will not impinge on meeting the requirements of the Nitrates Directive. The current average stocking density on dairy farms is 1.81 livestock units (LU) ha⁻¹ (Humphreys, 2008).

Nitrate (NO₃) leaching pathways between soils, groundwaters and rivers are generally long and complex (Collins and McGonigle, 2008) and such pathways vary depending on soil/subsoil type, bedrock geology/hydrogeology and climatic factors such as rainfall. The lag time between introducing protection measures and first improvements in water quality is therefore likely to be different in different catchments comprising different soils and geologies and should be considered by policy makers and catchment managers (Kronvang *et al.*, 2008). Such hydrological travel times will be used in this study to indicate the nutrient breakthrough response times under different geologies and to categorise each site as high risk or otherwise.

The saturated zone beneath a water table is technically defined as groundwater, which is a principle receptor of water and leached nutrients from the unsaturated soil/subsoil. This definition applies to groundwater protection schemes. However, groundwater recognised as

a 'groundwater body' or aquifer, in the context of the WFD, in Ireland is defined in terms of the saturated zones in fractured bedrock or in significant bodies of sand and gravel. Therefore, groundwater in subsoils, even though it is the initial recipient of leached nutrients, is not considered part of the aquifer. In a recent study investigating groundwater and surface water contributions to stream flow in Ireland, subsurface soil and subsoil water (with the exception of sand and gravel) were termed interflow, and shallow groundwater were described as shallow bedrock groundwater where permeability is higher and fracturing is more dominant (Jennings *et al.*, 2007).

Saturated subsoils (interflow) and an underlying groundwater body (shallow and deep groundwater) may often form a hydraulic continuum, but usually do not share the same hydrogeological characteristics. This defining framework has clear implications for nutrient migration pathways and their management. It is therefore important to investigate all scenarios that could contribute to water quality status. The components of lag time for nitrates are: vertical travel time, horizontal travel time, flushing of an aquifer to below a threshold value and travel time through the hyperoic zone. In this paper, vertical and horizontal travel times will only be considered. This travel time represents initial or fastest breakthrough of nitrate to a surface waterbody. Total lag time may be much longer.

Four contrasting case studies – site characteristics

The objective of this paper was to compare the potential vertical and horizontal travel times of nitrate to a virtual surface water receptor on four contrasting study sites in Ireland. These hydrological travel times were then used to indicate the likely first hydrological response times to be expected in a virtual receptor 500 m from the site. Groundwater vulnerability categorisation is based on the thickness and permeability of sub-soil overlying bedrock aquifers and the thickness of the unsaturated

zone in sand and gravel aquifers (Misstear and Brown, 2008). Each site was assigned a vulnerability category:

Oak Park, Co. Carlow – high vulnerability (unsaturated zone sand and gravel thickness > 3 m); Moorepark, Co. Cork – extreme vulnerability (moderate permeability soils 0.5–4.0 m thick, karst); Solohead, Co. Tipperary – moderate vulnerability (low permeability soils, 5–10 m thick); Johnstown Castle, Co. Wexford – moderate vulnerability (low permeability soils, 5–10 m thick).

Sand and gravel aquifer, Oak Park, Co. Carlow

This case study was based on hydrogeological investigations performed by Premrov *et al.* (2007 and 2008), and on a bromide (Br⁻) tracer study initiated by Hooker (2005). Hydrogeological investigations were carried out on a 10 ha area, underlain by a shallow sand and gravel aquifer with inter-bedded silt/clay lenses (Premrov *et al.*, 2007 and 2008). The Br⁻ experiment was carried out on an adjacent field approximately 200 to 300 m south-east of the case study site (Hooker, 2005). Sand and gravel aquifers underlie approximately 2% of Ireland and are the only aquifers with solely intergranular permeability. Such aquifers are usually unconfined and are generally thin; typically between 5 and 15 m saturated thickness. Premrov *et al.* (2007) found mean concentrations at the start of the experiment in November 2006 in the unsaturated zone, at 0.9 m below ground level (bgl), of 59 mg NO₃-N L⁻¹. Mean concentrations of 35 mg NO₃-N L⁻¹ were observed in the saturated zone during the same period.

Karst limestone aquifer, Curtins farm, Fermoy, Co. Cork

Atypically in the EU, Ireland has predominantly unconfined bedrock aquifers with fissure permeability only. Carboniferous limestone is the primary aquifer and occurs as outcrop or shallow bedrock covering approximately 50% of Ireland. The free draining acid brown earth, which dominates the site, com-

prises 1.69% of the land area of Ireland, and soils with similar vulnerability extend to 4.6% of the country (Ryan *et al.*, 2006). The Teagasc research farm at Curtins, Moorepark is located within a lowland limestone area. The farm presently evaluates alternative management systems for spring calving pasture-based milk production. Well-drained soils are present to a depth of 4.5 m. Due to the underlying karstified limestone aquifer, groundwater flow direction is not uniform on site. Depth to bedrock approximately is 2.5 m and depth to groundwater is approximately 29 m.

Poorly productive aquifer, Solohead, Co. Tipperary.

The Teagasc research facility at Solohead is located on a poorly drained till to a depth of 5 m, overlying a poorly productive Devonian sandstone and mudstone aquifer >10 m depth. The site is undulating with occasional emergence of sand and gravel at the surface. This is a heavy textured clay loam soil (25% sand and 42% clay). Heavy textured soils classified as clay, clay loam or silty clay loam represent 32% of agricultural soils in Ireland (Humphreys *et al.*, 2008). The boundary between the till and underlying bedrock is not uniform and the sand and gravel aquifer is not wholly confined. Shallow groundwater flow direction in the till may deviate from general flow direction in the deeper confined aquifer (Daly and Teillard, 2001). Concentrations of NO₃-N in shallow groundwater (1 m bgl, below ground level) under four dairy grassland based systems from 2001–2002 were examined by Humphreys *et al.* (2008). Losses of NO₃-N to shallow (1 m bgl) groundwaters were low; mean groundwater NO₃-N concentrations were <3.0 mg L⁻¹. NO₃-N losses were 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 drainage.

Poorly productive aquifer, Johnstown Castle, Co. Wexford

The Teagasc research facility at Johnstown Castle, located on heterogeneous glacial deposits on a 4.2 ha gently sloping (2%) field, comprised six non-grazed plots on a beef farm. The overburden, morainic in nature, varies in thickness from 1–20 m. This is underlain by Pre-Cambrian greywacke, schist and massive schistose quartzites that have been subjected to low grade metamorphism (Diamond, 1988). The surface elevation increases sharply at the head of the plots (>71 m Above Ordnance Datum) forming a hill (Sandhill) comprising both sand and fine loamy till. Some of this sand may have been soliflucted, resulting in stratification between sand and underlying low permeability fine till. The Sandhill is well to excessively drained, whereas the plots are moderately to poorly drained. Shallow groundwater NO₃-N concentrations ranged from background levels <1 mg NO₃-N L⁻¹ to >11.3 mg NO₃-N L⁻¹ (Fenton *et al.*, 2009).

Materials and methods

Calculation of vertical pathway travel time in soil/subsoil

Vertical water flow and leaching of nutrients through unsaturated soils (Fig. 1) are influenced by negative pore water pressure. Flow is facilitated by matrix and gravitational potential gradients. Depending on the matrix potential, which is related to the pore size dis-

tribution or soil/subsoil type, and water content in the unsaturated zone, the hydraulic conductivity will change. For vertical travel time estimation, depth of infiltration calculations based on effective rainfall (from January 2005 - January 2008) and subsoil physical parameters versus watertable depth were used as follows for all case studies:

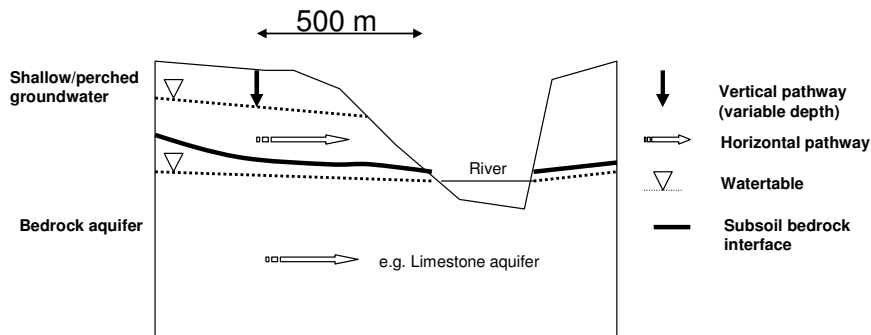
$$RF = \frac{ED}{NED} \quad [1]$$

$$PV = \frac{RF}{n} \quad [2]$$

$$TD = \frac{PV}{NED} / 1000 \quad [3]$$

where RF is recharge flux (mm day⁻¹), ED is effective rainfall (rainfall-actual evapotranspiration) for the study period (mm), NED is number of days recharge occurs in this period (days), PV is pore velocity (mm day), *n* is effective porosity and TD is total depth of infiltration during the study period (m). Depending on antecedent soil conditions, effective rainfall may or may not occur and can be determined daily by a specific model, e.g. Schulte *et al.* (2005) for grassland systems in Ireland under different drainage conditions (Moorepark, Solohead and Johnstown Castle). Premrov *et al.* (2009) has developed an extension of this model for tillage systems, based on results from the Oak Park case study site. Meteorological data was collected on sites in Oak Park, Moorepark and Johnstown Castle. Solohead weather data was compiled using site

Figure 1: Generic diagram of vertical and horizontal pathways to a virtual receptor 500 m away.



specific rainfall data and the remaining data from the Moorepark site. Where available, tracer experiment breakthrough data in shallow ceramic cups/piezometers was also used (Oak Park and Moorepark).

Calculation of horizontal travel time in shallow groundwater

Flows in the saturated zone (Fig 1.) are a function of the saturated hydraulic conductivity (K_{sat}) and the potential gradient, which in most cases is gravitational. In the saturated zone, K_{sat} remains constant at a particular location but varies spatially due to the heterogeneity of the aquifer and between different aquifers and geological units. K_{sat} may also vary due to anisotropies in the aquifer. Horizontal travel time estimation was calculated by effective Darcian linear velocity, v (m day⁻¹). This can be calculated from:

$$v = -K_{sat} \frac{1}{n_e} \frac{dh}{dx} \quad [4]$$

where K_{sat} is estimated from slug injection tests (Bouwer and Rice, 1976), hydraulic gradient is the change in the watertable head over the distance between two wells in the field (dh/dx) and n_e is effective porosity calculated from soil cores. For the Oak Park case study, K_{sat} values were taken from the literature and are indicative of Irish fluvio-glacial sand and gravel aquifers. K_{sat} values were then combined with Premrov *et al.* (2007) data to calculate the linear velocity and travel times. In the Solohead case studies K_{sat} and n_e values from the literature were used. Where these factors were not available, or in circumstances where assessment of linear velocity is very complex, as with karst or fractured bedrock aquifers (Moorepark), actual horizontal travel time using tracer breakthrough in wells was used taken from Bartley (2003). For the Johnstown Castle study, K_{sat} data (slug injection tests) from shallow groundwater piezometers (Fenton *et al.*, 2009) and n_e data (Fenton *et al.*, 2008) were used to estimate horizontal travel time.

Results

The vertical and horizontal travel time results from the individual case studies are presented below.

Sand and gravel aquifer, Oak Park, Co. Carlow

Vertical travel time

Mean precipitation during the 2005–2008 period was 857.5 mm. The estimated vertical matrix travel time of approximately 1 to 2 years corresponded with mean total effective rainfall of 350.2 mm. Effective drainage was estimated to have occurred on 89 days year⁻¹ (mean value for 2005 to 2008), which equates to an average vertical pore velocity of 15.7 mm day⁻¹ ($n_e=25\%$) through the unsaturated zone and depth of infiltration in this period of 1.4 m. In the tracer experiment initiated by Hooker in 2004, the breakthrough of Br⁻ tracer occurred in suction cups at 0.9 m depth ($n=8$) from 27 to 42 days after surface tracer application (Hooker, 2005). This gives daily vertical travel times of 0.02 to 0.03 m day⁻¹ (Hooker, 2005). In the saturated zone the time of first occurrence of the tracer was after 227 days, while the peak concentration occurred 345 days after tracer application (Premrov *et al.*, 2007). The average watertable depth in this period was 2 to 5 m bgl (Premrov *et al.*, 2007 and 2008).

Horizontal travel time

Horizontal travel time must consider vertical variations and lateral changes in the sediment sequence, from gravel to sand to clay/silt, which can occur over quite short distances, with domination of sand and gravel sediments in the area (Premrov *et al.*, 2007 and 2008). It can be seen from Table 1 that a range of horizontal travel times from 0.056 to 5.6 years is expected for a fluvio-glacial/mixed sand and gravel aquifer.

Total travel time

The total travel time of approximately 1 to 6.6 years is the combination of vertical travel time

Table 1: Oak Park case study aquifer parameters and estimated horizontal travel times.

Aquifer type	K_{sat}	dh/dx	n_e	ν	Horizontal travel time
	m day ⁻¹		%	m day ⁻¹	years
Clayey gravel	0.1-1 ^b	0.0122 ^d	25 ^g	4.9·10 ⁻³ -0.488 ^f	280.7 - 2.81 ^f
Fluvio-glacial sand and gravel	100-500 ^{a,h,c}	0.0122 ^d	25 ^g	4.88-24.4 ^f	0.28 -0.056 ^f
Mixed sand and gravel ^h	5-100 ^b	0.0122 ^d	25 ^g	0.244 -4.88 ^e	5.6 - 0.28 ^e
Homogeneous gravel ^h	100-1000 ^b	0.0122 ^d	25 ^g	4.88-48.8 ^f	0.28 -0.03 ^f

^a Based on Allen and Milenic (2003); ^b Based on Misstear *et al.* (2009); ^c Based on McConville *et al.* (2001); ^d From Premrov *et al.* (2007); ^e From Premrov *et al.* (2007) - computed values; ^f This study; ^g From Fetter (2001); ^h From Kruseman and de Ridder (1992), given for comparative purposes.

(approximately 1 year) through the unsaturated zone (according to Hooker, 2005) and the fastest horizontal travel times from Table 1 through the saturated zone to a virtual surface receptor 500 m away from the chosen site (0.056 to 5.6 years - for fluvio-glacial/mixed sand and gravel). This reflects the best case scenario in terms of lag time between changed practices and improved water quality.

Karst limestone aquifer, Curtins farm, Fermoy, Co. Cork

Vertical travel time

Mean precipitation during the 2005–2008 period was 953 mm. The vertical matrix travel time of approximately 1 to 2 years corresponded with mean total effective rainfall of 437.2 mm. Mean effective rainfall was estimated to have occurred on 158.5 days, which equates to an average vertical pore velocity of 10.03 mm day⁻¹ through the unsaturated zone and depth of infiltration in this period of 1.2 m. The depth to bedrock is approximately 1–4

m on site, which means travel time at different locations will reflect this depth. Bartley (2003) used a surface Br⁻ tracer recharge experiment to quantify the vertical unsaturated travel time to ceramic cups at 1 m bgl and to the screened section of a well at 29 to 32 m bgl in groundwater. Maximum vertical travel velocity due to preferential flow paths through the unsaturated zone was estimated as 1.66 m day⁻¹, giving a travel time of approximately 17 days. On a neighbouring farm, also located on thin soils (depth to bedrock 1 m) underlying a karstified limestone, first occurrence of tracer in a screened interval at approximately 22 m bgl using the same methodology was 34 days (Richards *et al.*, 2005). Maximum soil solution concentrations were observed 34-65 days after application, giving a vertical travel time of 34 to 65 days (Richards *et al.*, 2005).). It was concluded that matrix flow through the overburden and preferential flow through the remainder of the unsaturated zone (karst limestone) is quantitatively more important for

Table 2: Moorepark, horizontal travel time from K_{sat} range and actual horizontal travel time from a tracer experiment in the karst aquifer to receptor 500 m away from the source.

K_{sat}	Tracer breakthrough at point 4	Tracer breakthrough at point 10	Horizontal Travel time based on K_{sat}	Horizontal Travel time based on Tracer
m day ⁻¹	days	days	time	months
0.004 to 27 ^a	44	51	Days to Years	2 to 3

^a Based on Bartley (2003), slug injection tests

transport of conservative tracers to groundwater. Over two monitoring years, the mean annual farm average $\text{NO}_3\text{-N}$ concentrations were 11.9 and 15.2 $\text{NO}_3\text{-N L}^{-1}$ with considerable temporal variation observed within years (Bartley, 2003).

Horizontal travel time

The horizontal travel time in the karst aquifer was difficult to estimate as considerable data was required (n_e , fracture density, fracture truncation, fracture orientation, fracture trace length, fracture spacing, mechanical aperture, effective hydraulic aperture, and aperture opening size) to estimate K_{sat} values. A K_{sat} range was available from slug injection tests (Bartley, 2003) but was not indicative or comparable with actual horizontal flow travel times. Instead, tracer breakthrough in several wells originating from the vertical travel time study was used to calculate horizontal travel time. The horizontal travel time to the receptor 500 m away from the source following this pathway was estimated as 2 to 3 months.

Total travel time

Vertical travel time was estimated at 1 to 2 years to allow for differences in soil depth. Ceramic cup and borehole tracer data indicated faster preferential flow paths on site. Once water and nutrients pass through the soil layer, travel time increases rapidly. Horizontal travel time was estimated as 2 to 3 months. Total travel time from source to receptor following groundwater flow direction was approximately 1.5 – 2.5 years and was dependent on soil thickness. In such a complex environment a modelling exercise (e.g. global or distribution methods) at the correct scale should be undertaken.

Poorly productive aquifer, Solohead, Co.

Tipperary

Vertical travel time

Mean precipitation during the 2005–2008 period was 1059 mm. The mean vertical matrix travel time of approximately 1 year cor-

responded with mean total effective rainfall of 481 mm. Mean effective rainfall was estimated to have occurred on 91 days, which equates to an average vertical pore velocity of 52 mm day^{-1} through the unsaturated zone and depth of infiltration in this period of 4.8 m. The mean (mean watertable depth \pm standard deviation) perched shallow watertable on site during this time varied from 1.5 \pm 0.3 m bgl (summer) to 0.6 \pm 0.2 m bgl (winter). Vertical travel time to the perched groundwater is achievable within one drainage season. Infiltration may take several years to reach the boundary of the sand and gravel confined layer. During this time period effective rainfall ranged from 312 to 605 mm, which means that in some years vertical travel time in 1 year was not achievable.

Horizontal travel time

Horizontal travel times on the poorly productive aquifer were estimated to be 3424 years in the shallow perched groundwater to 152 years in the confined aquifer (Table 3). Estimates of the horizontal travel time could be improved further through investigation of K_{sat} on the site.

Total travel time

Combining vertical travel time of 3 years and the horizontal travel time of 3424 years for the shallow perched groundwater, the total travel time is estimated to be 3427 years. For zones that are connected to the confined aquifer the travel time may be shorter.

Poorly productive aquifer, Johnstown Castle, Co. Wexford

Vertical travel time

Mean precipitation during the 2005–2008 period was 1046 mm. The vertical matrix travel time of approximately 1 year corresponded with mean total effective rainfall of 553 mm. Mean effective rainfall was estimated to have occurred on 178 days, which equates to an average vertical pore velocity of 9.7 mm day^{-1} through the unsaturated zone

Table 3: Solohead, horizontal travel time in shallow and deeper groundwater to surface water virtual receptor 500 m away from the source.

Groundwater layer	K_{sat} m day ⁻¹	dh/dx	n_e %	v m day ⁻¹	Horizontal Travel time years
A	0.01 ^a	0.004	10 ^c	0.0004	3424
B	0.224 ^b	0.004	10 ^c	0.00896	152

A Shallow perched groundwater

B Deeper Confined Aquifer

^{a, c} Based on Fetter (2001)

^b Based on Daly and Teillard (2001)

Table 4: Johnstown Castle, mean horizontal travel time in six monitored plots to a surface water receptor 500 m away from the source.

Plot	K_{sat} m day ⁻¹	dh/dx	n_e ^a %	v m day ⁻¹	Horizontal Travel time Years
1	0.009 ^a	0.39	32	0.011	125
2	0.008 ^a	0.24	32	0.006	228
3	0.011 ^a	0.35	32	0.012	114
4	0.011 ^a	0.38	32	0.013	105
5	0.012 ^a	0.32	32	0.012	114
6	0.008 ^a	0.32	32	0.008	171

^a From Fenton et al. (2008)

and depth of infiltration in this period of 1.7 m (Fenton *et al.*, 2008).

The depth to the median watertable during this period was 1.01 m (Fenton *et al.*, 2009).

Horizontal travel time

Horizontal travel time ranged from 105 to 228 years (Table 4.)

Total travel time

Vertical travel time was estimated at less than 1 year and horizontal travel time was estimated from 105 to 228 years. This gives a total travel time of approximately 106 to 229 years to the receptor 500 m away from the source.

Discussion

Irish case studies

This is the first attempt at calculating nitrate lag times in Ireland and there are no direct methodologies to calculate lag time across Europe. Groundwaters in Ireland are at risk of failing to meet the water quality targets of the WFD by 2015. Slow groundwater travel times

and therefore high residence times of water in certain aquifers means that reversing this increasing trend may not be achievable within the 2015 timeframe. A comparison of total travel times from all the contrasting case studies is presented in Table 5.

For the purposes of illustration, a simple model of transport has been used to show differences in first breakthrough travel time from source to receptor in four contrasting sites. This simple methodology shows the wide variability in travel times that can be expected within catchments in Ireland (several months to years to reach a receptor 500 m away). In reality considerable heterogeneities and anisotropies in soil, sediment and bedrock hydrogeological properties make transport pathways and times highly variable. Examples of this include discrepancies between travel time based on matrix flow (depth of infiltration) or preferential flow (tracer breakthrough data). In some instances N transfer to shallow groundwater is rapid with little chance for attenuation. Zones of high conductivity asso-

Table 5: Vulnerability category, attenuation capacity, total estimated travel times and lag time issues for the four contrasting case studies.

Case study	Vulnerability category	Attenuation capacity	Total travel time	Lag time issues
Oak Park, Co Carlow	High	Moderate	Months to Decades	Indicative of initial breakthrough Contaminated shallow groundwater interaction with groundwater and surface water. Flushing utilising specific yield of the aquifer should be incorporated into lag time.
Moorepark, Co. Cork	Extreme	None	Years	N mineralisation in soils, travel time will change depending on soil N status, thickness of soil and preferential flow paths. Flushing utilising specific yield of the aquifer should be incorporated into lag time.
Solohead, Co. Tipperary	Moderate	High	Decades	Connectivity to semi confined aquifer. Phosphorus lag time to surface water and groundwater. NO ₃ unlikely to be problematic.
Johnstown Castle, Co. Wexford	Moderate	High	Decades	Contaminated shallow groundwater interaction with surface water in close proximity to excess nutrients. N mineralisation in soils.

ciated with gravel channel deposits or fracture systems, may connect a source area with a receptor so that travel times may be significantly less than estimated. In this instance a small area of land may have a disproportionate effect on water quality at the receptor.

Saturated soil and subsoil is not deemed

groundwater (with the exception of sand and gravel aquifers) under the WFD definition. Groundwater contribution to surface waters in poorly productive aquifers may be as low as 5% but can be as high as 30% when flow along the subsoil/bedrock interface is considered; by contrast, in productive aquifers such

as sand and gravel and karstified limestone, the contribution may be from 80% to 90% (Misstear *et al.*, 2009). Components of deep groundwater, intermediate (interflow and shallow groundwater) and overland flow that contribute to stream flow were assessed in seven pilot catchments and used to constrain and inform the numerical model NAM (Nedbør-Afstrømnings-Model) in Ireland (Jennings *et al.*, 2007). This model was then extended to regional catchments. The model estimated large amounts of intermediate flow contributed to stream flow, e.g. 318 mm y^{-1} in the Owenduff poorly productive aquifer. In such a catchment deep groundwater contributed 128 mm y^{-1} and overland flow contributed 1322 mm y^{-1} . In the Suck pilot karst catchment intermediate flows were 362, 171 and 124 mm y^{-1} respectively. Such information, when combined with nitrate concentrations leaving the rooting zone can be used to calculate loads of nutrients in intermediate and groundwater flow. This is important for calculating aquifer flushing times to certain EU threshold values.

For groundwater classification chemical and quantification status under the WFD several tests must be complied with. For example “no significant diminution of surface water chemistry and ecology” determines if groundwater impacts on a surface water body achieving the goals of the WFD. The groundwater threshold values used are surface water quality standards with a dilution factor. This dilution factor is determined by the shallow and deep groundwater contributions, but not the interflow components. Further differentiation of intermediate flow is needed.

Drilling records for the Oak Park site indicate that the clay/silt lenses are not continuous and may not represent a barrier for transport of water soluble pollutants in this fluvio-glacial deposit aquifer (Premrov *et al.*, 2007). Therefore, the estimated travel time of 1 to 6.6 years was shown as an indication of the fastest possible total travel time through the unsaturated and shallow groundwater through

the coarser gravel lenses present at this site

Irish bedrock aquifers are deemed to have low attenuation potential due to their fractured and karstified nature and protection is mainly provided by the overlying glacial tills (Daly and Warren, 1998). In karst environments K_{sat} measurements are only indicative of how fast groundwater flows at a specific point, which leads to vast ranges in K_{sat} e.g. In Waulsortian limestone (a lithological subgroup of Carboniferous Limestone), K_{sat} values range from 4 to 2500 m day^{-1} (Goldscheider and Drew, 2009). In the vicinity of the case study site, K_{sat} values such as a public water supply, 873 m day^{-1} and nearby water abstraction wells range from 10 to 200 m day^{-1} . From slug injection tests, bedrock K_{sat} values in the case study area ranged from 0.004 to 27 m day^{-1} (Bartley, 2003). The soils here have high total nitrogen levels and thus elevated mineralisation rates. Even if fertiliser inputs are reduced it will take a long time to prevent losses (Grimvall *et al.*, 2000) from the unsaturated zone. In this respect groundwater trend monitoring will not reflect mitigation measures in place at farm level but instead will represent a mixture of bio-geochemical processes.

In the Oak Park and Moorepark case studies, shorter travel times and less denitrification potential in the soils and aquifer put receptors at risk. In terms of maximising the effectiveness of resources applied to improve water quality, it is likely to be most successful if problem receptors are identified and mitigation measures targeted in areas that contribute to that receptor. Implementing measures on sites for which there is no receptor water quality problem or for which the travel time and N remediation potential is such that changes in management are not likely to impact water quality (a blanket approach) is not an efficient use of resources. The time it takes to flush an entire plume of nitrate through interflow, shallow groundwater and deep groundwater pathways should be considered as an important component of lag time. This of course

must also integrate degradation processes.

In Solohead high N remediation potential (e.g. due to high denitrification potential) makes improvements in N concentrations as a result of changed management unlikely. There is a low risk of nitrate transport to the receptor as highlighted by the low shallow groundwater concentrations observed by Humphreys *et al.* (2008).

Both the Solohead and Johnstown Castle case studies area have soil types representative of a large proportion of the agricultural area of Ireland and not only have long lag times between management changes and improvements in water quality, but also have large capacities for denitrification. It is important that this is taken into account by policy makers because the implementation of changed management practices may not lead to any improvement in water quality with respect to nitrate within the prescribed legislative timeframes.

Meteorology can also play an important role in the lag time between implementation of mitigation measures and water quality responses. Effective rainfall varied by 203 mm yr⁻¹ between the four sites with the lowest drainage in Oak Park and the highest in Johnstown Castle. Previous research (Schulte *et al.*, 2006) has highlighted the effect that effective rainfall can have on the mean concentration of nitrate leached, being higher under low drainage and low under high drainage. The inter-annual variation in effective rainfall must also be taken into consideration when evaluating water quality responses to mitigation measures.

Implications for policy

In catchments with mixed vulnerability such variable hydrological response times may lead to more stringent measures being implemented without allowing time for the efficacy of measures to be identified. Many non Irish specific examples show that water quality targets have been too optimistic in the past because the “inertia controlling the nutrient cycle

plays a key role in the response of water quality to the newly adapted agricultural measures” (Granlund *et al.*, 2005). To put the Irish examples into context, one needs to look to such countries as Finland and Poland where state supported agriculture ceased in the late 1980s and early 1990s, which lead to reduced fertilizer use. In 1998, the Finish Council of State issued a Decision in Principle on the water protection targets of 2005 and P and N loads should be reduced by 50% of those in the early 1990s and this was implemented through the Finish Agri-Environmental Programme in 1995. A review of water quality data from 1981 to 2000 showed that little or no reduction of riverine nutrient loads was achieved during 1995-1999 (Granlund *et al.*, 2005). A similar delayed response would be expected based on the present case studies.

A nutrient loss audit, which integrates farm scale nutrient balances, with improved knowledge in smaller scale field estimates of nutrient loss to water, during the hydrological lag period, could be used by policy makers to predict future trends in water quality. In areas where groundwater is not presently affected by N surpluses due to denitrification, such methods may not be applicable.

A clear understanding of hydrological response times will enable the efficacy of mitigation measures to be estimated first at the plot scale and then at the catchment scale. The availability of catchment scale data on soil and subsoil classification, K_{sat} , depth of subsoil, bedrock K_{sat} and artificial drainage networks will hamper evaluation of mitigation measures. Zones in close hydrological connectivity with sensitive receptors are areas that should be targeted for mitigation measure evaluation and areas where the most rapid responses would be anticipated. Integration of surface and subsurface hydrological pathways to surface waters is necessary to reduce nitrogen and phosphorus loss to water (Richards *et al.*, 2009). Policy makers and farmers must understand the high temporal and spatial variation of travel time at the catchment scale. This

means some areas of a catchment will respond rapidly whereas other areas will have much longer travel times.

Conclusions

The efficacy of mitigation measures for the control of diffuse water pollution cannot be assessed without considering lag time. Lag time issues will become increasingly important as the WFD 2015 deadline approaches and incorporating first breakthrough and flushing of a nutrient in a hydrogeological system at interflow, shallow groundwater and deep groundwater will become a priority. The case studies reported in this paper indicated that initial hydrological response times can vary greatly from months to years and, thus, they indicate the potentially long total travel time period between source farming activities and receptor response. The hydrological lag time of water quality response to the implementation of mitigation measures has been shown to be a function of effective rainfall and the hydrogeological properties of soil, subsoil and aquifers. Further biogeochemical processes during nutrient transport will further complicate the interpretation of water quality responses.

Internationally, past legislation has proved to be over optimistic leading to unfounded conclusions about the efficacy of mitigation measures (Kronvang *et al.*, 2008). This highlights the need for an Irish specific study. Additional mitigation measures and legislation should not be implemented or become more stringent until lag time is considered fully. Without doing so, the efficacy of present mitigation measures may not be fully understood.

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