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4

5 **Methodology for the location of a subsurface permeable reactive**
6 **barrier for the remediation of point source pollution on an Irish**
7 **farm.**

8

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10

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14 **Abstract**

15

16 Nutrient loss from agricultural sources to water continues to be a national challenge. Diffuse
17 pollution from agricultural sources is considered to be the primary cause of slight-to-moderate
18 water pollution in Ireland, but agricultural point sources, such as farmyards, are often not
19 considered due to their scattered spatial distribution and small areal extent. Agricultural point
20 sources tend to be small and localised hot spots of nutrients and, therefore, can be efficiently
21 treated using environmental technologies developed for wastewater and contaminated land
22 treatment. A small area associated with soiled water irrigation, on a 4.27 ha case study site at
23 Teagasc, Johnstown Environmental Research Centre, Ireland, was identified, where
24 groundwater nitrate-nitrogen (NO₃-N) concentration exceeds the maximum admissible
25 drinking water concentration of 11.3 mg N L⁻¹. A continuous, shallow permeable reactive
26 barrier may be suitable to remediate point source pollution at this site. A methodology,

1 based on site and groundwater characterisation, successfully located a site for a
2 permeable reactive barrier.

3

4 *Key index words:* Permeable reactive barrier; hydrogeological investigation; groundwater;
5 nitrate; point source pollution.

6

7 **Introduction**

8

9 The Surface Water Directive (EEC, 1975), the Groundwater Directive (EEC, 1980),
10 the Drinking Water Directive (EC, 1998) and the Nitrates Directive (EEC, 1991) has
11 focused considerable attention on the disposal of agricultural wastewaters in Ireland.
12 The Water Framework Directive (WFD) (EC, 2000) aims to achieve at least “good
13 status” in all surface and groundwaters by 2015.

14

15 The nitrate-nitrogen (NO₃-N) concentration in rivers and groundwater is a key water
16 quality indicator in Ireland. From 2004 to 2006, 25% of groundwater had NO₃-N
17 concentrations greater than the drinking water guide concentration of 5.65 mg N L⁻¹
18 and 2% exceeded the maximum admissible concentration (MAC) of 11.3 mg N L⁻¹
19 (Lucey, 2006). Agricultural activities are probably the most significant anthropogenic
20 sources of NO₃-N contamination in groundwater (Oyarzun *et al.*, 2007). Current
21 agricultural practices (application methods, dosages and storage) while achieving high
22 nutrient efficiency and nutrient management cannot avoid incidental nutrient loss to
23 surface and groundwater. In aquifers with low permeability pathways of nutrient loss
24 both historically and in the future may pose a threat to receptors for long periods of
25 time.

1 The control of phosphorus (particulate and soluble forms) before it enters a waterbody
2 and remediation of nitrate in a waterbody should be integrated. The correct siting of
3 an environmental technology (structure used to remediate or control a contaminant) to
4 intercept a pollution plume such as a permeable reactive barrier will be an important
5 step in the remediation of point sources. Such technologies may be *ex - situ*
6 (farmyard) and *in-situ* (in the field actually in the contaminant plume) (Fenton et al;
7 2007).

8
9 Point source pollution from agricultural practices can include inappropriately
10 managed agricultural soiled waters, such as dairy farmyard soiled water, leaking
11 septic tanks or storage facilities (soiled water and slurry storage, lagoons,
12 hydrocarbons) or drainage leaks from low points on the farmyard. Dairy farmyard
13 soiled water may comprise farmyard runoff, parlour washings, silage and farmyard
14 manure effluents, along with general farmyard washings. Under new legislation soiled
15 water may not contain faecal matter leading to lower nutrient concentration (EC,
16 2006). This soiled water is stored and then landspread or irrigated. Where hydraulic
17 loads exceed the carrying capacity of the soil, irrigators may be point sources of
18 pollution in the field. In poorly drained soils, surface runoff may also occur.

19
20 When nitrogen (N)-rich fertilizer applications exceed plant demands and the
21 denitrification capacity of a soil, leaching of N in the form of NO₃-N to groundwater
22 may occur. Due to its high mobility (Shamrukh *et al.*, 2001), significant amounts of
23 excess N can be transported as NO₃-N to a waterbody, potentially leading to
24 eutrophication, and episodic and persistent hypoxia, where dissolved oxygen is less
25 than 2 mg L⁻¹ (Abu – Ashour *et al*, 1994; Kung *et al*, 2000; NRC, 2000). NO₃-N

1 leaching is dependent on the hydraulic loading rate, soil water content, soil type and N
2 loading rate.

3

4 Point source pollution has a clearly identifiable point of discharge and occurs at or
5 near an agricultural waste facility and exhibits high levels of $\text{NO}_3\text{-N}$ or ammonium-
6 nitrogen ($\text{NH}_4\text{-N}$) in a limited area. The effects of point source pollution accumulate
7 over time (Schilling and Wolter, 2001). Identifying the source, the potential nutrient
8 pathway and a potential receptor (e.g. stream) is important, where remediation is
9 considered. Both $\text{NO}_3\text{-N}$ and chloride (Cl) are negative ions and do not adsorb to the
10 soil matrix. However, $\text{NO}_3\text{-N}$ concentrations are reduced by biochemical processes
11 through denitrification. Using the $\text{NO}_3\text{-N}$ to Cl ratio, the source and groundwater flow
12 pathway may be identified as Cl concentration is conservative and $\text{NO}_3\text{-N}$
13 concentration decreases relative to the distance from the source. The concentrations of
14 both parameters are also affected by diffusion, dispersion and dilution (Obenhuber
15 and Lowrance, 1991; Agriculture and Agri-Food Canada, 2002).

16

17 Conventional *in situ* methods for N removal include:

- 18 • monitored natural attenuation, wherein the source of pollution is initially
19 found, stopped and then advection, dispersion and chemical-plus biological
20 degradation of the contaminant is allowed to occur over a long period of time
21 (USEPA, 1997a);
- 22 • pump-and-reuse, wherein the pumped water is recycled for a certain purpose
23 (e.g. cooling equipment) and then treated;
- 24 • pump-and-treat, wherein treated water is used to irrigate crops;

- 1 • pump-and-waste (Bronstein, 2005), wherein contaminated water is evaporated
- 2 or injected into a saline aquifer or geological unit;
- 3 • phytoremediation (Suresh and Ravishankar, 2004).

4

5 Monitored natural attenuation depends on the denitrification capacity of a soil and the
6 distance from the receptor. Pump-and-treat may be expensive and pump-and-waste is
7 not sustainable and may cause plume migration. For remediation of contaminated
8 water generated on a farm, *ex situ* methods for N removal may be used. These include
9 continuously moving biofilm reactors (Rodgers and Burke, 2002), sequencing batch
10 biofilm reactors (Rodgers *et al.*, 2004), trickling filters (Kuai *et al.*, 1999), activated
11 sludge systems (Gao *et al.*, 2004), and fluidised-bed biofilm reactors (Rabah and
12 Dahab, 2004). These methods have shown good potential for biological N removal
13 but need to be adapted for the control of farmyard point source pollution. Successful
14 remediation prior to land application decreases the potential for groundwater
15 contamination.

16

17 An *in situ* subsurface remediation barrier, comprising a treatment zone of reactive
18 materials that degrades or immobilises contaminants as groundwater flows through it,
19 referred to as a permeable reactive barrier (PRB), may be used to attenuate the
20 movement of nutrients and other agricultural contaminants (Powell and Powell,
21 1998). PRBs comprise low-cost, low-value permeable waste products, which provide
22 a carbon (C)-rich substrate for NO₃-N removal (USEPA, 1997b). They provide
23 preferential conduits for contaminated groundwater flow, wherein wastewater flows
24 through a C-rich mixture (e.g. woodchip) to reduce NO₃-N concentration. A review of
25 remediation and control systems for the treatment of agricultural wastewaters has

1 identified PRBs as a feasible option for *in situ* NO₃-N remediation from point sources
2 on Irish farms (Fenton *et al.*, 2008). PRBs have been used extensively in the
3 remediation of chlorinated solvents, metals and inorganics, fuel hydrocarbons,
4 nutrients, radionuclides and other organic contaminants at full- and pilot-scales in
5 urban and industrial scenarios (USGS, 1999). Two traditional PRB designs are
6 commonly used (Figure 1):

7 a) Funnel-and-gate system (Starr and Cherry, 1994) consisting of an impermeable
8 funnel that directs groundwater to a reactive wall.

9 b) Shallow continuous trench (Pierzynski *et al.*, 2005), placed adjacent to
10 groundwater flow and backfilled with reactive material and soil.

11 Two other adaptations are: (1) the injection well configuration (Pierzynski *et al.*,
12 2005), where a well network is drilled perpendicular to the groundwater flow
13 direction and the reactive material is injected directly into the plume, and (2)
14 interception of the plume by a drainage system. Here, the contaminated water is
15 transported off-site to a reactive cell (Pierzynski *et al.*, 2005).

16
17 A review of existing worldwide PRB installations for inorganic and radionuclide
18 contamination emphasises that PRBs may be successfully employed with a thorough
19 site investigation, but the long-term performance of the reactive materials needs
20 further investigation (Bronstein, 2005). PRBs installed for the interception and
21 remediation of chlorinated hydrocarbon and chromium (VI) plumes in groundwater
22 suggest various alterations to more traditional PRB types such as reactive wall type,
23 excavate and fill, reaction vessel, funnel and multiple gate systems suggesting site
24 specific conditions (USEPA, 1997). Temporary, continuous trenches have been
25 installed in agricultural scenarios to investigate NO₃-N removal rates from artificial

1 recharge experiments (Robertson *et al.*, 2000; Schipper *et al.*, 2005). Fluctuations of
2 watertable height may cause alternating anaerobic and aerobic conditions in
3 continuous trenches leading to decreased denitrification rates (Schipper and Vojvodic-
4 Vukovic, 2001). The barrier porous media may be placed above the watertable only if
5 it remains tension saturated allowing anaerobic conditions to exist (Robertson, 1995).

6
7 The objective of this paper is to develop a methodology, based on site-specific
8 conditions, to locate a PRB on unconsolidated material above bedrock to intercept
9 NO₃-N contamination from an agricultural point source. The methodology developed
10 may be used to locate PRBs on other agricultural sites.

11

12 **Materials and Methods**

13

14 **Site identification**

15

16 The 4.2 ha study site was located at the Teagasc, Johnstown Castle Environmental
17 Research Centre, Co. Wexford. Baseline data established a groundwater NO₃-N plume
18 arising from point source pollution from a soiled water irrigator system spreading
19 effluent with a biochemical oxygen demand (BOD₅) concentration believed to be
20 greater than 1000 mg L⁻¹. The irrigator moved over a 4000 m² area within the 4.2 ha
21 site. However, due to the slope of the site, the irrigator was confined to a much
22 smaller area, resulting in ponding with subsequent recharge. The source was identified
23 by documenting historical management practices and locating irrigation infrastructure.
24 This site was chosen to evaluate methodologies for the implementation of a PRB.

25

1 **Site description**

2

3 Identified potential receptors on site are: a) Tenches pit stream to the west which
4 flows to a shallow lagoon; b) Tenches pit stream which connects to the Kildavin River
5 to the south; and c) groundwater (Figure 2). In 2003, six hydrologically isolated study
6 plots were established between the source and the receptor. Further isolation was
7 achieved by excavating two shallow, unlined trapezoidal drains, excavated to a depth
8 of 1 m, with bases ranging from 71.08 m AOD to 70.2 m AOD and 71.10 m AOD to
9 70.30 m AOD, respectively, along the northern edge of the plots. Flow in these drains
10 did not interact. Overland flow from each study plot was collected in a drain at the
11 lowest topographical point. Subsurface drainage was collected with a herring bone
12 subsurface drainage system (drain spacing, 1 m) located at a 1 m depth below the
13 ground surface. Subsurface flow was measured using V-notch weirs. The study plots
14 were instrumented with a total of 18 piezometers - 3 piezometers installed in each
15 plot.

16

17 **Site characterisation**

18

19 A site characterisation was carried out to identify possible point sources and receptors.
20 The contaminant NO₃-N from the point source was identified and all infrastructure
21 (subsurface pipes and connectors for irrigator) located back to the surface storage
22 area. The area was surveyed and the distance from source to receptors was measured.
23 All existing data on the site, such as soil type, thickness and texture, soil profiles,
24 drainage conditions, subsurface geology, and subsurface and surface drain location,
25 was collated.

1

2 **Water balance**

3

4 A water balance of the site was used to calculate the travel time from the source to the
5 watertable. Daily weather data, recorded at the Johnstown Castle Weather Station,
6 were used to calculate daily soil moisture deficit (SMD) using a Hybrid model for
7 Irish grasslands. The site had moderately drained soil. Potential evapotranspiration,
8 ET_0 (mm day⁻¹), was calculated using the FAO Penman-Montieth equation (Allen *et*
9 *al.*, 1998):

10

$$11 \quad ET_0 = \frac{0.408\Delta(R_n - G) + \gamma \frac{900}{T + 273} u_2 (e_s - e_a)}{\Delta + \gamma(1 + 0.34u_2)} \quad (1)$$

12

13 where R_n is the net radiation at the crop surface (m⁻² d⁻¹), T is the air temperature at a
14 2 m height (°C), u_2 is the wind speed at a 2 m height (m s⁻¹), e_s and e_a are the
15 saturation and the actual vapour pressure curves (kPa °C⁻¹), and γ is the psychrometric
16 constant (kPa °C⁻¹). ET_0 was then converted to actual evapotranspiration (Ae) using an
17 Aslyng scale recalibrated for Irish conditions (Schulte *et al.*, 2005). Effective rainfall
18 was calculated by subtracting daily actual evapotranspiration from daily rainfall
19 (assuming no overland flow losses due to the high infiltration capacity of the soil on
20 this site). Soil moisture deficit (SMD) on day one (January 1st, 2006) was set to zero
21 and effective drainage was estimated for each subsequent day. Modelling the effective
22 drainage enables the infiltration depth of water to be calculated at specific hydraulic
23 loads where the soil effective porosity is known. This infiltration depth may be

1 compared to watertable data to investigate if recharge to groundwater in that
2 particular year affects water quality.

3

4 **Groundwater characterisation**

5

6 A topographic base map with a contour interval of 2 m and a field boundary overlay
7 was generated using ArcGIS™ for data obtained on 11 July, 2006. This allowed
8 surface (topography) and subsurface features (watertable) to be compared spatially.

9 Due to the sloped profile of the site, 18 multilevel piezometers were drilled (rotary
10 drilling) prior to this study to represent specific geological units and not depths
11 (Figure 2). Two stratigraphic units, from 63 m above ordnance datum (m AOD) to 67
12 m AOD and from 67 m AOD to 70 m AOD, respectively, were drilled. Data will be
13 described using m AOD to allow comparisons of plume position eliminating
14 topographical differences. A further piezometer (FH7) was installed and surveyed on
15 the Sandhill area in 2005 (Figure 2). All piezometers had a slotted screen length of 1
16 m. Drilling logs and samples from the piezometers were used to develop a conceptual
17 model of the subsurface. The piezometers were surveyed using TOPCON AT-G4
18 equipment (TOPCON, Ireland) and the locations of the wells were recorded using
19 digital mapping software (ArcGIS™ 9.1, ESRI, Ireland). The depth to water level in
20 each monitoring well was measured using an electronic water-level indicator (Van
21 Walt Ltd, Surrey, UK) and groundwater heads were determined in m AOD. Surface
22 water features, such as streams, drains and lagoons, were also surveyed on 11th July,
23 2006. Groundwater head data was contoured (block kriging) using GW-Contour 1.0
24 software (Waterloo Hydrogeologic, Canada). The topographic base map was merged
25 with well locations and groundwater head input files. These groundwater maps were

1 used to track groundwater flow direction over time and NO₃-N concentration using
2 groundwater heads and water quality data in each well as inputs (Fenton and Hyde,
3 2006). From March 2005 to March 2007, water levels were measured weekly in each
4 monitoring well and nitrite-nitrogen (NO₂-N), total oxidized nitrogen (TON), NH₄-N,
5 ortho-phosphate (PO₄), and Cl concentrations within each well were measured every 2
6 weeks. Water samples were filtered through 0.45 µm filter paper and analysed using a
7 Thermo Konelab 20 Analyser (Technical Laboratory Services, Ontario, Canada).

8

9 Prior to the study, soil cores (n = 46) at the piezometer locations and drains were taken
10 at 1m depths and analysed for bulk density and particle density. Total porosity was
11 calculated from (Brady and Weil, 1996):

12

$$13 \quad n = 100 \left(1 - \frac{\rho_b}{\rho_d} \right) \quad (2)$$

14

15 where n is the total porosity (%), ρ_b , the bulk density (kg m⁻³), and ρ_d is the particle
16 density (kg m⁻³).

17

18 Saturated hydraulic conductivity, K_{sat} , on site was determined using falling head slug
19 tests (instantaneous injection of 1 L of water) (Horslev, 1951; Bouwer, 1976). To
20 establish a hydraulic connection between the source and potential receptors, the
21 hydraulic gradient was calculated using:

22

$$23 \quad \frac{h_A - h_B}{L} \quad (3)$$

24

1 where h_A and h_B are hydraulic heads calculated by electronically dipping a piezometer
2 and converting depth to watertable to m AOD, and L (m) is the length between these
3 two piezometers.

4

5 The quantity of water discharging from a known width of aquifer, Q ($\text{m}^3 \text{ day}^{-1}$), was
6 determined using (Darcy, 1856):

7

$$8 \quad Q = -K_{sat} A \frac{dh}{dx} \quad (4)$$

9

10 The average linear velocity, v (m day^{-1}), was calculated from:

11

$$12 \quad v = -K_{sat} n \frac{dh}{dx} \quad (5)$$

13

14 where v is equal to Q/A , K is the hydraulic conductivity (m day^{-1}), $A = bw$, where b is
15 the aquifer thickness (m), w , the width (m), and dh/dx is the hydraulic gradient.

16

17 The transmissivity, T ($\text{m}^2 \text{ day}^{-1}$), is calculated using the aquifer thickness, b :

18

$$19 \quad T = K_{sat} b \quad (6)$$

20

21 To investigate the variation in the $\text{NO}_3\text{-N:Cl}$ ratio on the site, groundwater and drain
22 samples were analysed. Groundwater temperatures were recorded in two piezometers
23 with similar piezometer total depths (2c and 5c, Figure 2) at 30 minute intervals using
24 real time electronic divers (Van Walt Ltd, Surrey, U.K.).

1

2 **Trench thickness - bench scale testing**

3

4 The kinetics of denitrification will depend on C and NO₃-N availability, pH,
5 temperature, soil texture, soil management, tillage, rainfall events, rates of microbial
6 respiration and nitrification, water filled porosity, soil mineral N content, soil type,
7 and redox conditions. A reactive material should be chosen and tested to optimise
8 contaminant residence times in the reactive barrier. On-site soil cores of fine loamy
9 brown earth, fine loamy gley and sandy brown podzolic soils were tested for
10 denitrification rate ($\mu\text{g N lost as NO}_3\text{-N g}^{-1} \text{ dry soil day}^{-1}$) using soil incubation tests.
11 The denitrification rate of the gley soil amended with woodchips (5:2 g dry weight of
12 woodchips to soil) was also examined. The retention time, t (days), needed to achieve
13 denitrification was calculated using:

14

$$15 \quad t = \frac{C_{treated}}{C_{max}} / r \quad (7)$$

16

17 where $C_{treated}$ is the desired concentration after remediation, C_{max} is the greatest
18 concentration expected, and r is denitrification rate determined from batch
19 experiments. The retention time was then multiplied by the groundwater flow velocity
20 to calculate the thickness of the trench. Based on chemical stoichiometric relations,
21 denitrification of one mole of NO₃-N will require 1.25 moles of C. This equates to a
22 mass balance of 1.07 kg of available C per 1 kg of NO₃-N. With approximately 50 %
23 of C availability in woodchip (based on bulk density) the treatment of 1 kg of NO₃-N
24 will require approximately 2 to 2.5 kg of woodchip (Fahner, 2002).

1

2

Results

3

4 **Site characterisation**

5

6 The soil texture comprises a 15 to 40 cm-deep loam (soil group, brown earth in Plots
7 1 and 2), overlying a loam-to-clay-loam (soil group, gley) subsurface soil and there
8 was a quartzite outcrop along the western side of the site. The textural change across
9 the site was responsible for differential drainage. The study area comprised two well-
10 drained plots (Plots 1 and 2 - brown earth), two imperfectly-drained plots (Plots 3 and
11 4 - gley) and two poorly-drained plots (Plots 5 and 6 - gley with higher clay content)
12 (Figure 2).

13

14 **Water balance**

15

16 Over the study period, the site received mean precipitation of 1046 mm, of which the
17 Hybrid model calculated 553 mm drainage through the root zone in a process known
18 as effective drainage. Model output showed effective drainage occurred on 178 days,
19 giving an average recharge rate of 3.11 mm d⁻¹. The mean soil total porosity was
20 32.2±4.9%. The average pore velocity was estimated to be 9.7 mm d⁻¹, giving an
21 approximate mean travel depth of 1.7 m in a moderately-drained soil over the study
22 duration. The depth to the median watertable during this period was 1.01m, which
23 equates with the base of the intersecting drains in Plots 3 and 4. Therefore, the
24 watertable intersects these drains at certain times of the year and infiltrating water
25 upslope from the drains recharges to groundwater within 1 year. The hydraulic load of

1 the soiled water irrigator varied from 10 to 50 mm year⁻¹. This would increase the
2 mean depth of travel on the irrigated site when the irrigator was in operation by 10
3 cm. Therefore, the main receptor was groundwater but with surface water receptors
4 forming boundaries to the site.

5

6 **Groundwater investigation**

7

8 Initial baseline sampling of the piezometers on-site showed NO₃-N concentrations
9 above the drinking water limit of 11.3 mg NO₃-N L⁻¹. Groundwater temperature on
10 site during the study period ranged from 9.5°C to 10.5°C in piezometers 2c and 5c
11 which is suitable for denitrification to occur at depths below 1 m.

12

13 The strike and dip of the quartzite outcrop combined with drilling log data gave an
14 estimated unconfined aquifer thickness of approximately 10 m and a saturated
15 thickness, based on mean watertable and depth to the impermeable zone, of
16 approximately 7 m. Piezometer parameters, K_{sat} , and groundwater quality parameters
17 are presented in Table 1. Hydraulic gradients, calculated using Equation (3) based on
18 median and maximum watertable heights, showed a hydraulic gradient between the
19 source and potential receptors.

20

21 A groundwater map was constructed using watertable data and surveyed surface water
22 features on July 11th, 2006. As no significant seasonal deviation occurred, a median
23 groundwater map was used to show groundwater flow direction. Groundwater
24 contours (based on groundwater heads) deviated little from topography within the six
25 isolated plots (Figure 3). Therefore, topography was used to infer the groundwater

1 flow direction on the Sandhill area where a lower piezometer density exists.
2 Groundwater flow direction was consistent throughout the study period and median
3 groundwater flow contours were used to locate a PRB parallel to watertable contours.
4 Where groundwater flow direction changes, the orientation of the PRB should be
5 based on mean conditions. Based on median and maximum hydraulic heads, a barrier
6 containing a 2 m-deep reactive zone is needed (reactive media should fill subsurface
7 from 68 m AOD to 70 m AOD). This would ensure the reactive material was covered
8 at all times by the watertable. A cross sectional conceptual model of the plume
9 positions the centroid (area with highest nutrient concentration) around 2c – 5c
10 (Figure 4). Nutrient concentration decreases outwards from the centroid. The extent of
11 the plume migration vertically is unknown. Lateral plume extent varies from 350 m
12 from 1c to 6c and extends further to 400 m at piezometer 1b. As the lateral plume
13 diameter near to the source decreases the trench needs to be less than 350 m (Figure
14 3), to capture all groundwater flow migrating to Plots 2, 3, 4 and 5 (Figure 3).

15

16 Combining the hydrogeological characterisation data, plume distance and travel times
17 were calculated (Table 2). A steep hydraulic gradient in Plot 4 resulted in
18 groundwater flow to Plots 1 and 6. A significant hydraulic gradient existed between
19 Plots 5 and 6. Average linear velocity was higher in Plots 4 and 5. Therefore, the
20 centroid was able to migrate quickly in two directions. When aquifer thickness was
21 considered, Plot 5 has highest T values indicating plume migration was quickest from
22 Plots 4 and 5. Therefore, plume migration is greatest (in a given time interval) in Plot
23 5, migrating to a potential receptor to the west. Migration from Plot 4 eastwards was
24 slower. Travel times from the centroid outwards are also similar with plume migration

1 faster in a westward direction. Therefore, two travel times must be considered in
2 groundwater remediation of the site.

3
4 Due to subsurface characteristics, a plume originating from a point source may
5 migrate to several receptors in different timescales. Remediation should concentrate
6 on the most immediate of these pressures or be located close to the pollution source.

7

8 **Source tracking**

9

10 Source tracking was used to connect the source, pathway and receptor of the nutrient
11 loss. The median $\text{NO}_3\text{-N}:\text{Cl}$ ratio in drains intersecting groundwater flow between the
12 source and the plots were 0.46 (max 0.84) and 0.38 (max 0.72). Mean watertable
13 depths in piezometers 3c and 2c during the same period are 0.52 m and 2.06 m,
14 respectively. Therefore, the watertable from the up-gradient area intersected the drain
15 adjacent to 3c and the flow in the drain was towards 2c. This means that contaminated
16 groundwater passed into the plots and was then picked up in groundwater samples in
17 the piezometers. To prevent contamination of surface water, the PRB should be
18 located upslope from these drains and attenuate groundwater before any surface water
19 groundwater interactions can take place (Figure 3).

20

21 **Trench thickness**

22

23 Using the denitrification rates in Table 3, Equation 7 was used to calculate the
24 retention time needed to remediate the highest expected $\text{NO}_3\text{-N}$ concentration
25 expected ($24.24 \text{ NO}_3\text{-N mg L}^{-1}$) to allowable levels. The retention time was then

1 multiplied by the groundwater flow velocity to give the barrier thickness. The site is
2 primarily on gley soils (95%) and the proposed trench location was on this soil type.
3 Natural attenuation on-site would take longer periods of time. Potential receptors on
4 site are approximately 200 m from the source. This would allow natural attenuation in
5 gleys within 7.35 years. The travel time from Plots 3, 4, 5 and 6 would be less than
6 this. However, natural attenuation to the east may be an option as travel times are
7 much higher and the receptors are a greater distance away.

8

9 **Discussion**

10

11 The choice of PRB will depend on the scale of the project. In this investigation, a
12 continuous trench was chosen over a funnel and gate system, as less geotechnical
13 input was needed. Both options, however, would need hydrogeological professional
14 input to locate a PRB. A site investigation of this scale may not be viable for
15 individual farmers. Contamination may be from point or non-point sources needing
16 varied amounts of site and hydrogeological characterisation. In this study, the site
17 characteristics merited a PRB for groundwater remediation. Hydraulic conductivity,
18 measured *in situ*, provides the retention times needed for denitrification to occur. This
19 may be different on other sites where retention times or migration pathways may not
20 make a PRB a viable option for remediation (unconsolidated material or bedrock).
21 The watertable on other sites may not be shallow raising the costs of PRB
22 construction. Once the pollution source has been stopped, contamination residence
23 times in free draining fluviogravels may be short due to high permeability. Therefore
24 construction of a PRB would be unjustified. Also where the groundwater body is an
25 important receptor, remediation within this waterbody may not be justified. In such

1 cases remediation of the pollution before it reaches the groundwater body is
2 preferable.

3

4 Calculation of the contaminant flux at source or along a control plane away from the
5 source may be expensive due to drilling costs. Therefore, this methodology is best
6 suited to small point sources or plumes which have already reached shallow
7 watertable interfaces at surface groundwater interaction sites. Further research into
8 less permanent, low-cost monitoring systems is needed.

9

10 For this case study the dimensions, orientation and reactive media chosen for the PRB
11 on this site are presented in Table 3. The exact location of the proposed PRB is
12 presented in Figure 3. The following methodology can be used to establish a PRB on
13 this site for point source remediation:

14 1. Thorough site characterisation using all available data relating to the site is
15 required. Data management and appropriate visual presentations such as
16 maps, graphs and diagrams should be compiled. Distance from source to
17 receptor should be calculated and topography defined.

18 2. Installation of a piezometer network between the source and potential
19 receptors. Field visual tools (e.g. VS-Fast system) for soil field assessment
20 may be a useful tool for preliminary studies, which enables *in situ* estimates
21 of soil consistency, soil structure and texture (McGarry and Sharp, 2001).
22 Other systems based on BS 5930:1999 are used in groundwater protection
23 schemes to describe sub-soils (GSI, 1999).

24 3. Groundwater analysis and soil sampling should be carried out and a
25 preliminary dataset should be compiled. Use calculated parameters to

1 calculate groundwater travel times and distances in certain timeframes.
2 Combine aquifer data with water quality data and form a three dimensional
3 conceptual model of the subsurface and identify the plume centroid. This
4 conveys what is known or suspected about contamination sources, release
5 mechanisms, and the transport and fate of the contaminant. Draw a sub-
6 surface cross section. Construct groundwater flow maps. Compile watertable
7 data (vertical position of reactive barrier).

8 4. The PRB trench thickness should be designed for specific water quality
9 targets. Batch or column experiments should be carried out to calculate the
10 reaction rate and equilibrium constant of the contaminant with the reactive
11 media.

12 5. Identify travel times to potential receptors and locate the PRB up-gradient of
13 the receptor. Compare PRB installation with monitored natural attenuation.

14
15 Before construction, the site should be evaluated to ensure design depth and width
16 may be achieved. Trial holes should be considered. The ability of emplacing the
17 reactive material without aquifer obstruction should be assessed to avoid clogging of
18 media and smearing soil walls thus decreasing permeability. During and after
19 installation, a monitoring network should be installed to investigate if denitrification
20 is occurring in the trench and to investigate groundwater flow alteration due to the
21 barrier construction. The ease of excavating the reactive media for replacement
22 purposes after a period of time should be considered. Monitored natural attenuation on
23 site should also be considered for areas further away from the source. A number of
24 wells should be drilled in such locations. Pump-and-treat and pump-and-reuse would
25 need considerable investment, drilling, discharge licences, and would need surface

1 structures and maintenance which could interrupt farming practices. Recycling of
2 water on farms is more likely to stem from soiled water remediation or rainwater
3 harvesting and reuse. Pump-and-waste would also need a disposal licence and would
4 merely export the problem elsewhere. The funnel-and-gate option is cost-prohibitive
5 and would need geotechnical and engineering input in the design phases. However, a
6 more feasible option for gate construction, such as compressed clay or another low-
7 permeability material, should be investigated. A PRB installed south of the
8 investigative plots would not capture all contaminated groundwater and could not
9 achieve surface water quality targets. The current configuration would intercept
10 contaminated groundwater before entering the six plots and before hydraulic gradients
11 at location 4c divide the plume.

12

13 **Conclusions**

14

15 A continuous, shallow PRB may be suitable for Irish conditions to remediate point
16 sources. Each site will have site-specific conditions but the methodology developed
17 for this study site, based on site and groundwater characterisation, can successfully
18 site a PRB and calculate the dimensions and orientation of the barrier. Further
19 research should be carried out on the denitrification rates of different reactive media
20 when combined with different soil groups. Higher $\text{NO}_3\text{-N}$ removal rates will
21 necessitate lower residence times and increased remediation. The longevity of the
22 reactive media needs to be investigated and a cost-benefit analysis for the remediation
23 of contaminated groundwater undertaken. A broader methodology should be
24 investigated which takes into account other site characteristics, such as unconsolidated
25 material, fractured bedrock and a deep watertable.

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1 **Captions for figures & tables**

2

3 **Figure 1:** Two types of subsurface PRBs a: funnel and gate, b: continuous trench;
4 with source, NO₃-N plume, reactive material, treated plume and receptor. Watertable
5 (WT) positioned within treatment wall containing reactive material and barrier
6 constructed adjacent to groundwater (GW) flow direction.

7 **Figure 2:** Field site layout showing plot location, irrigator source, potential receptors
8 (Tenches pit stream, lagoon, Kildavin River and connection to artificial lake system),
9 piezometer locations and drainage of the study site.

10 **Figure 3:** Groundwater contours (modelled using block kriging) based on
11 groundwater heads and topography. Flow from high to low hydraulic head contours at
12 right angles to contours. Plume centroid location (10 – 15 mg NO₃⁻ N L⁻¹) PRB
13 orientation, location and dimensions.

14 **Figure 4:** Schematic diagram showing cross sectional (1c – 6c) conceptual model of
15 the contamination plume with source on the sandhill. Highest median NO₃- N
16 concentration is within plume centroid. Watertable shows hydraulic gradient from plot
17 4 towards plots 1 and 6. Centroid vertical and horizontal thickness and dilution fronts
18 can also be seen.

19

20 **Table 1:** Piezometer parameters

21 **Table 2:** Plume distance and travel times using hydrogeological parameters

22 **Table 3:** Reactive media denitrification rate and PRB thickness to reduce NO₃- N
23 concentration from 24.24 NO₃- N mg L⁻¹ (highest concentration) to 11.3 mg NO₃- N
24 L⁻¹ (allowable concentration)

25

Table 1: Piezometer and groundwater data over the study period.

Plot	I.D	Elevation m AOD	Total depth m	Multilevel	Watertable height			Groundwater NO ₃ -N concentration mg L ⁻¹		Groundwater NH ₄ -N concentration mg L ⁻¹	
					Median m	Max m	K_{sat} m day ⁻¹	Median mg l ⁻¹	Max mg l ⁻¹	Median mg l ⁻¹	Max mg l ⁻¹
1	c	71.48	4.35	1	4.35	4.35	0.02	4.8	11.85	0.03	1.42
	b	69.91	4.13	2	2.85	4.13	0.02	12.71	22.56	0.1	2.84
	a	67.04	3.64	2	3.73	3.64	0.02	6.37	9.54	0.24	0.79
2	c	71.83	4.38	1	3.18	4.38	0.04	12.8	24.24	0.33	5.63
	b	69.52	4.13	2	3	4.02	0.18	12.81	22.3	0.38	5.72
	a	67.22	3.14	2	1.01	3.14	0.08	1.21	14.77	0.05	2.05
3	c	70.87	3.24	1	0.74	2.29	0.02	12.31	17.34	0.07	1.38
	b	69.47	2.67	1	1.09	2.59	0.18	8.99	16.83	0.02	0.31
	a	67.6	3.55	2	0.8	2.15	0.07	12.26	19.37	0.07	2.18
4	c	70.96	2.49	1	1.04	2.24	0.02	6.01	10.69	0.05	0.14
	b	68.92	2.94	2	0.69	1.41	0.13	0	6.85	0.08	0.41
	a	67.34	2.7	2	0.94	1.75	0.12	0.02	6.57	0.04	0.46
5	c	71.71	4.33	1	2.18	3.58	0.05	14.29	19.94	0.02	0.46
	b	68.88	2.87	2	0.67	1.47	0.19	9.08	18.92	0.03	0.12
	a	67.03	1.55	2	0.53	1.55	0.26	9.06	11.35	0.05	2.06
6	c	70.68	3.01	1	1.38	2.73	0.07	9.61	11.09	0.13	1.02
	b	68.09	3.18	2	0.45	1.19	0.08	4.19	8.44	0.08	0.71
FH7	a	67.24	2.95	2	0.96	1.55	0.07	3.12	14.66	0.04	2.23
		72.43	4.14	2	2.97	4.14	0.02	6.44	12.66	0.06	0.15

* K_{sat} measured *in situ* using falling head slug tests.

Table 2: Plume distance and travel times using hydrogeological parameters

Parameters	Plots					
	1	2	3	4	5	6
area (ha)	0.78	0.75	1.01	0.94	0.41	0.41
piezometers	3	3	3	3	3	3
piezometer density (piezometer/ha)	0.26	0.25	0.34	0.31	0.14	0.14
Total porosity (%)	0.32	0.32	0.32	0.32	0.32	0.32
Depth to impermeable zone (m)	10.00	10.00	10.00	10.00	10.00	10.00
Depth of saturated zone (m)	7.00	7.00	7.00	7.00	7.00	7.00
Slope (%)	0.02	0.02	0.02	0.02	0.02	0.02
width (m)	50.00	50.00	55.00	55.00	30.00	30.00
Q m ³ day ⁻¹ (mean discharge)	0.11	0.27	0.36	0.65	0.48	0.18
v m day ⁻¹ (average linear velocity) (takes porosity into account)	0.02	0.03	0.07	0.12	0.16	0.05
v m day ⁻¹ (max)	0.02	0.06	0.07	0.13	0.18	0.07
K m day ⁻¹ (mean hydraulic conductivity)	0.02	0.08	0.07	0.12	0.19	0.07
T m ² day ⁻¹	0.14	0.56	0.49	0.84	1.33	0.49
Mean hydraulic head (piezometer c)	67.13	68.65	70.13	69.92	69.53	69.30
Mean hydraulic head (piezometer a)	63.31	66.21	66.80	66.40	66.50	66.28
Hydraulic head (piezometer c) max	67.13	67.45	68.58	68.72	68.13	67.95
Hydraulic head (piezometer a) min	63.40	66.21	65.45	65.59	65.48	65.69
Mean distance (m) between source and piezometer (c)	250.00	250.00	250.00	250.00	250.00	250.00
Mean distance (m) between c and receptor (lower Tenches pit stream) (LTPS)	200.00	200.00	200.00	200.00	200.00	200.00
Plume distance (m) in 1 year (mean)	8.51	11.32	24.99	42.84	57.43	18.04
Plume distance (m) in 1 year (max)	8.71	22.27	26.59	48.18	65.67	24.11
Travel time (year) from proposed PRB to piezometer (a) (120 m)	14.10	10.61	4.80	2.80	2.09	6.65
Travel time (year) from c to receptor (LTPS)(200 m)	23.50	17.68	8.00	4.67	3.48	11.08

Table 3: Reactive media denitrification rate and PRB thickness to reduce $\text{NO}_3^- \text{N}$ concentration from $24.24 \text{ mg NO}_3^- \text{N L}^{-1}$ (highest concentration) to 11.3 mg L^{-1} (allowable concentration)

Reactive media	Denitrification rate* ($\mu\text{g L g day}^{-1}$)	Retention time (days)	PRB thickness (m)	
			mean	max
Brown earth	2.09 ± 0.01	223.04	16.61	19.91
Gley	4.34 ± 0.10	107.41	8.00	8.00
Gley + Woodchip	21.70	21.48	1.60	1.91

*adapted from Sullivan and McDermott (2007)

Table 4: PRB orientation, reactive media type and dimensions.

PRB dimensions		
Horizontal (x)	Vertical (y)	Thickness (z)
(m)	(m)	(m)
250	2	1.6 – 1.9
Orientation	Parallel to groundwater contours	
Reactive media	Woodchip and gley soil mix (ratio 5:2)	

Figure 1: Two types of subsurface PRBs a: funnel and gate, b: continuous trench; with source, NO₃-N plume, reactive material, treated plume and receptor. Watertable (WT) positioned within treatment wall containing reactive material and barrier constructed perpendicular to groundwater (GW) flow direction.

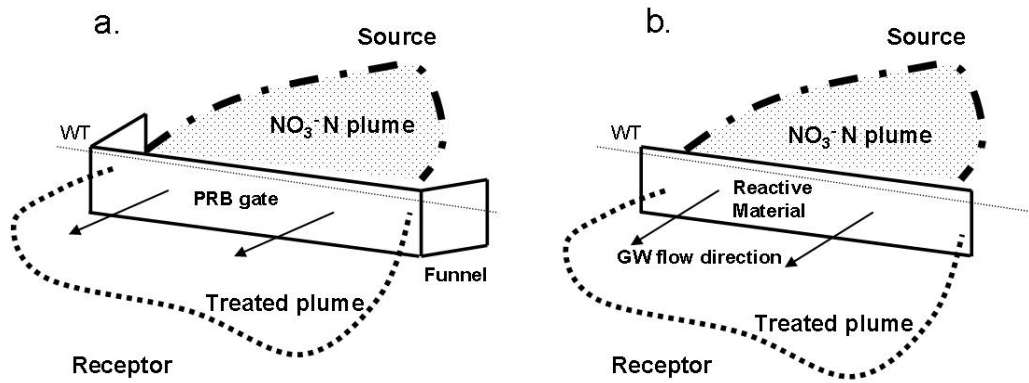


Figure 2: Field site layout showing plot location, irrigator source, potential receptors (Tenches pit stream, lagoon, Kildavin River and connection to artificial lake system), piezometer locations and drainage of the study site.

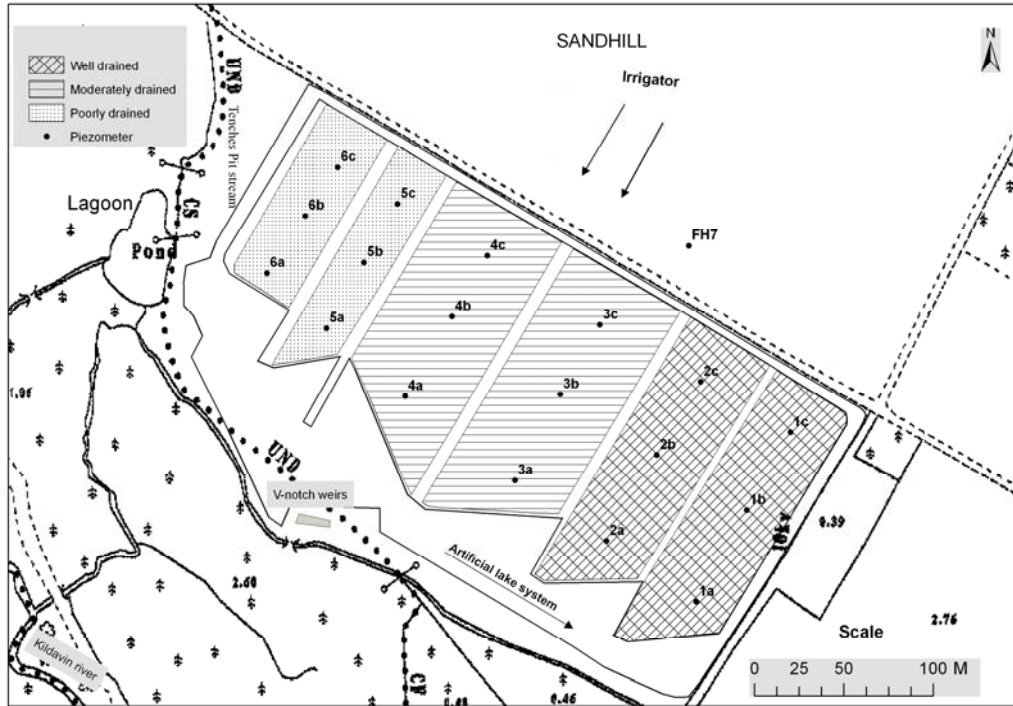


Figure 3: Groundwater contours (modelled using block kriging) based on groundwater heads and topography. Flow from high to low hydraulic head contours at right angles to contours. Plume centroid location (10 – 15 mg NO₃-N L⁻¹) PRB orientation, location and dimensions.

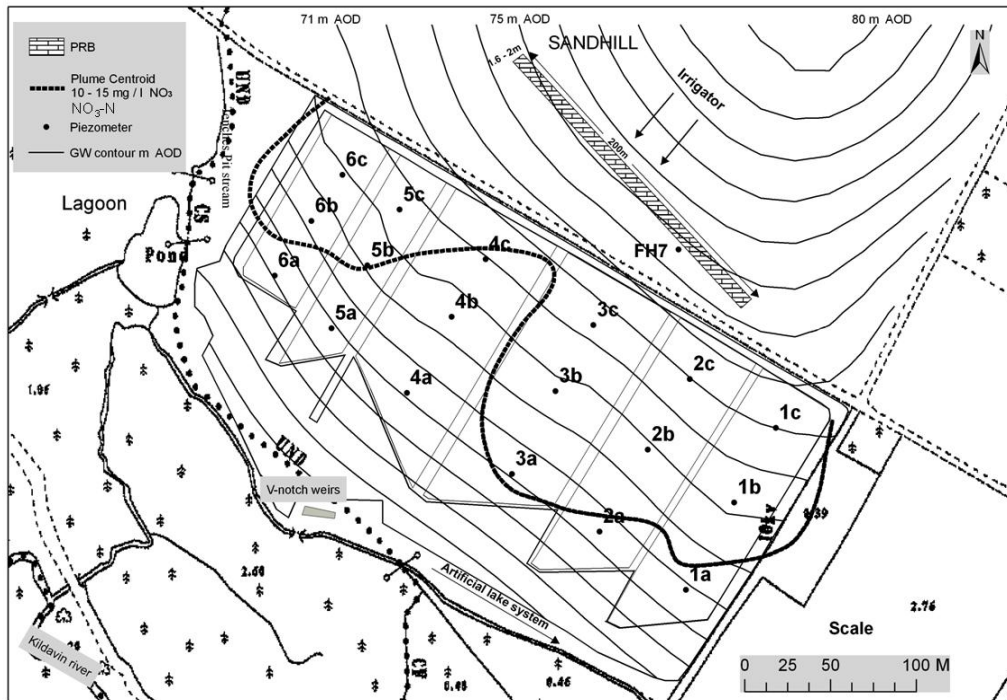


Figure 4: Schematic diagram showing cross sectional (1c – 6c) conceptual model of the contamination plume with source on the sandhill. Highest median $\text{NO}_3\text{-N}$ concentration is within plume centroid. Watertable shows hydraulic gradient from plot 4 towards plots 1 and 6. Centroid vertical and horizontal thickness and dilution fronts can also be seen.

