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

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Abstract

Nutrient loss from agricultural sources to water continues to be a national challenge. Diffuse pollution from agricultural sources is considered to be the primary cause of slight-to-moderate water pollution in Ireland, but agricultural point sources, such as farmyards, are often not considered due to their scattered spatial distribution and small areal extent. Agricultural point sources tend to be small and localised hot spots of nutrients and, therefore, can be efficiently treated using environmental technologies developed for wastewater and contaminated land treatment. A small area associated with soiled water irrigation, on a 4.27 ha case study site at Teagasc, Johnstown Environmental Research Centre, Ireland, was identified, where groundwater nitrate-nitrogen (NO_3-N) concentration exceeds the maximum admissible drinking water concentration of 11.3 mg N L^{-1}. A continuous, shallow permeable reactive barrier may be suitable to remediate point source pollution at this site. A methodology,

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

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

Introduction

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

The nitrate-nitrogen (NO_3-N) concentration in rivers and groundwater is a key water quality indicator in Ireland. From 2004 to 2006, 25% of groundwater had NO_3-N concentrations greater than the drinking water guide concentration of 5.65 mg N L^{-1} and 2% exceeded the maximum admissible concentration (MAC) of 11.3 mg N L^{-1} (Lucey, 2006). Agricultural activities are probably the most significant anthropogenic sources of NO_3-N contamination in groundwater (Oyarzun *et al*., 2007). Current agricultural practices (application methods, dosages and storage) while achieving high nutrient efficiency and nutrient management cannot avoid incidental nutrient loss to surface and groundwater. In aquifers with low permeability pathways of nutrient loss both historically and in the future may pose a threat to receptors for long periods of time.

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

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

When nitrogen (N)-rich fertilizer applications exceed plant demands and the denitrification capacity of a soil, leaching of N in the form of NO_3-N to groundwater may occur. Due to its high mobility (Shamrukh *et al*., 2001), significant amounts of excess N can be transported as NO_3-N to a waterbody, potentially leading to eutrophication, and episodic and persistent hypoxia, where dissolved oxygen is less than 2 mg L^{-1} (Abu – Ashour *et al*, 1994; Kung *et al,* 2000; NRC, 2000). NO_3-N leaching is dependent on the hydraulic loading rate, soil water content, soil type and N loading rate.

Point source pollution has a clearly identifiable point of discharge and occurs at or near an agricultural waste facility and exhibits high levels of NO_3-N or ammonium-nitrogen (NH_4-N) in a limited area. The effects of point source pollution accumulate over time (Schilling and Wolter, 2001). Identifying the source, the potential nutrient pathway and a potential receptor (e.g. stream) is important, where remediation is considered. Both NO_3-N and chloride (Cl) are negative ions and do not adsorb to the soil matrix. However, NO_3-N concentrations are reduced by biochemical processes through denitrification. Using the NO_3-N to Cl ratio, the source and groundwater flow pathway may be identified as Cl concentration is conservative and NO_3-N concentration decreases relative to the distance from the source. The concentrations of both parameters are also affected by diffusion, dispersion and dilution (Obenhuber and Lowrance, 1991; Agriculture and Agri-Food Canada, 2002).

Conventional *in situ* methods for N removal include:

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

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

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

An *in situ* subsurface remediation barrier, comprising a treatment zone of reactive materials that degrades or immobilises contaminants as groundwater flows though it, referred to as a permeable reactive barrier (PRB), may be used to attenuate the movement of nutrients and other agricultural contaminants (Powell and Powell, 1998). PRBs comprise low-cost, low-value permeable waste products, which provide a carbon (C)-rich substrate for NO_3-N removal (USEPA, 1997b). They provide preferential conduits for contaminated groundwater flow, wherein wastewater flows through a C-rich mixture (e.g. woodchip) to reduce NO_3-N concentration. A review of remediation and control systems for the treatment of agricultural wastewaters has identified PRBs as a feasible option for *in situ* NO_3-N remediation from point sources on Irish farms (Fenton *et al*., 2008). PRBs have been used extensively in the remediation of chlorinated solvents, metals and inorganics, fuel hydrocarbons, nutrients, radionuclides and other organic contaminants at full- and pilot-scales in urban and industrial scenarios (USGS, 1999). Two traditional PRB designs are commonly used (Figure 1):

a) Funnel-and-gate system (Starr and Cherry, 1994) consisting of an impermeable funnel that directs groundwater to a reactive wall.
b) Shallow continuous trench (Pierzynski *et al.,* 2005), placed adjacent to groundwater flow and backfilled with reactive material and soil.

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

A review of existing worldwide PRB installations for inorganic and radionuclide contamination emphasises that PRBs may be successfully employed with a thorough site investigation, but the long-term performance of the reactive materials needs further investigation (Bronstein, 2005). PRBs installed for the interception and remediation of chlorinated hydrocarbon and chromium (VI) plumes in groundwater suggest various alterations to more traditional PRB types such as reactive wall type, excavate and fill, reaction vessel, funnel and multiple gate systems suggesting site specific conditions (USEPA, 1997). Temporary, continuous trenches have been installed in agricultural scenarios to investigate NO_3-N removal rates from artificial recharge experiments (Robertson *et al.*, 2000: Schipper *et al.*, 2005). Fluctuations of watertable height may cause alternating anaerobic and aerobic conditions in continuous trenches leading to decreased denitrification rates (Schipper and Vojvodic-Vukovic, 2001). The barrier porous media may be placed above the watertable only if it remains tension saturated allowing anaerobic conditions to exist (Robertson, 1995).

The objective of this paper is to develop a methodology, based on site-specific conditions, to locate a PRB on unconsolidated material above bedrock to intercept NO_3-N contamination from an agricultural point source. The methodology developed may be used to locate PRBs on other agricultural sites.

Materials and Methods

Site identification

The 4.2 ha study site was located at the Teagasc, Johnstown Castle Environmental Research Centre, Co. Wexford. Baseline data established a groundwater NO_3-N plume arising from point source pollution from a soiled water irrigator system spreading effluent with a biochemical oxygen demand (BOD_5) concentration believed to be greater than 1000 mg L^{-1}. The irrigator moved over a 4000 m^2 area within the 4.2 ha site. However, due to the slope of the site, the irrigator was confined to a much smaller area, resulting in ponding with subsequent recharge. The source was identified by documenting historical management practices and locating irrigation infrastructure. This site was chosen to evaluate methodologies for the implementation of a PRB.

Site description

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

Site characterisation

A site characterisation was carried out to identify possible point sources and receptors. The contaminant NO_3-N from the point source was identified and all infrastructure (subsurface pipes and connectors for irrigator) located back to the surface storage area. The area was surveyed and the distance from source to receptors was measured. All existing data on the site, such as soil type, thickness and texture, soil profiles, drainage conditions, subsurface geology, and subsurface and surface drain location, was collated.

Water balance

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

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

where R_n is the net radiation at the crop surface (m^{-2} d^{-1}), T is the air temperature at a 2 m height (ºC), u_2 is the wind speed at a 2 m height (m s^{-1}), e_s and e_a are the saturation and the actual vapour pressure curves (kPa ºC^{-1}), and γ is the psychrometric constant (kPa ºC^{-1}). ET_0 was then converted to actual evapotranspiration (*Ae*) using an Aslyng scale recalibrated for Irish conditions (Schulte *et al.,* 2005). Effective rainfall was calculated by subtracting daily actual evapotranspiration from daily rainfall (assuming no overland flow losses due to the high infiltration capacity of the soil on this site). Soil moisture deficit (SMD) on day one (January 1st, 2006) was set to zero and effective drainage was estimated for each subsequent day. Modelling the effective drainage enables the infiltration depth of water to be calculated at specific hydraulic loads where the soil effective porosity is known. This infiltration depth may be compared to watertable data to investigate if recharge to groundwater in that particular year affects water quality.

Groundwater characterisation

A topographic base map with a contour interval of 2 m and a field boundary overlay was generated using ArcGIS^{TM} for data obtained on 11 July, 2006. This allowed surface (topography) and subsurface features (watertable) to be compared spatially. Due to the sloped profile of the site, 18 multilevel piezometers were drilled (rotary drilling) prior to this study to represent specific geological units and not depths (Figure 2). Two stratigraphic units, from 63 m above ordnance datum (m AOD) to 67 m AOD and from 67 m AOD to 70 m AOD, respectively, were drilled. Data will be described using m AOD to allow comparisons of plume position eliminating topographical differences. A further piezometer (FH7) was installed and surveyed on the Sandhill area in 2005 (Figure 2). All piezometers had a slotted screen length of 1 m. Drilling logs and samples from the piezometers were used to develop a conceptual model of the subsurface. The piezometers were surveyed using TOPCON AT-G4 equipment (TOPCON, Ireland) and the locations of the wells were recorded using digital mapping software (ArcGIS^{TM} 9.1, ESRI, Ireland). The depth to water level in each monitoring well was measured using an electronic water-level indicator (Van Walt Ltd, Surrey, UK) and groundwater heads were determined in m AOD. Surface water features, such as streams, drains and lagoons, were also surveyed on 11th July, 2006. Groundwater head data was contoured (block kriging) using GW-Contour 1.0 software (Waterloo Hydrogeologic, Canada). The topographic base map was merged with well locations and groundwater head input files. These groundwater maps were used to track groundwater flow direction over time and NO_3-N concentration using groundwater heads and water quality data in each well as inputs (Fenton and Hyde, 2006). From March 2005 to March 2007, water levels were measured weekly in each monitoring well and nitrite-nitrogen (NO_2-N), total oxidized nitrogen (TON), NH_4-N, ortho-phosphate (PO_4), and Cl concentrations within each well were measured every 2 weeks. Water samples were filtered through 0.45 μm filter paper and analysed using a Thermo Konelab 20 Analyser (Technical Laboratory Services, Ontario, Canada).

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

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

where n is the total porosity (%), ρ_b, the bulk density (kg m^{-3}), and ρ_d is the particle density (kg m^{-3}).

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

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

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

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

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

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

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

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

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

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

To investigate the variation in the NO_3-N:Cl ratio on the site, groundwater and drain samples were analysed. Groundwater temperatures were recorded in two piezometers with similar piezometer total depths (2c and 5c, Figure 2) at 30 minute intervals using real time electronic divers (Van Walt Ltd, Surrey, U.K.).

Trench thickness - bench scale testing

The kinetics of denitrification will depend on C and NO_3-N availability, pH, temperature, soil texture, soil management, tillage, rainfall events, rates of microbial respiration and nitrification, water filled porosity, soil mineral N content, soil type, and redox conditions. A reactive material should be chosen and tested to optimise contaminant residence times in the reactive barrier. On-site soil cores of fine loamy brown earth, fine loamy gley and sandy brown podzolic soils were tested for denitrification rate (μg N lost as NO_3-N g^{-1} dry soil day^{-1}) using soil incubation tests. The denitrification rate of the gley soil amended with woodchips (5:2 g dry weight of woodchips to soil) was also examined. The retention time, *t* (days), needed to achieve denitrification was calculated using:

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

where $C_{treated}$ is the desired concentration after remediation, $C_{\max}$ is the greatest concentration expected, and *r is* denitrification rate determined from batch experiments. The retention time was then multiplied by the groundwater flow velocity to calculate the thickness of the trench. Based on chemical stoichiometric relations, denitrification of one mole of NO_3-N will require 1.25 moles of C. This equates to a mass balance of 1.07 kg of available C per 1 kg of NO_3-N. With approximately 50 % of C availability in woodchip (based on bulk density) the treatment of 1 kg of NO_3-N will require approximately 2 to 2.5 kg of woodchip (Fahner, 2002).

Results

Site characterisation

The soil texture comprises a 15 to 40 cm-deep loam (soil group, brown earth in Plots 1 and 2), overlying a loam-to-clay-loam (soil group, gley) subsurface soil and there was a quartzite outcrop along the western side of the site. The textural change across the site was responsible for differential drainage. The study area comprised two well-drained plots (Plots1 and 2 - brown earth), two imperfectly-drained plots (Plots 3 and 4 - gley) and two poorly-drained plots (Plots 5 and 6 - gley with higher clay content) (Figure 2).

Water balance

Over the study period, the site received mean precipitation of 1046 mm, of which the Hybrid model calculated 553 mm drainage through the root zone in a process known as effective drainage. Model output showed effective drainage occurred on 178 days, giving an average recharge rate of 3.11 mm d^{-1}. The mean soil total porosity was 32.2±4.9%. The average pore velocity was estimated to be 9.7 mm d^{-1}, giving an approximate mean travel depth of 1.7 m in a moderately-drained soil over the study duration. The depth to the median watertable during this period was 1.01m, which equates with the base of the intersecting drains in Plots 3 and 4. Therefore, the watertable intersects these drains at certain times of the year and infiltrating water upslope from the drains recharges to groundwater within 1 year. The hydraulic load of the soiled water irrigator varied from 10 to 50 mm year^{-1}. This would increase the mean depth of travel on the irrigated site when the irrigator was in operation by 10 cm. Therefore, the main receptor was groundwater but with surface water receptors forming boundaries to the site.

Groundwater investigation

Initial baseline sampling of the piezometers on-site showed NO_3-N concentrations above the drinking water limit of 11.3 mg NO_3-N L^{-1}. Groundwater temperature on site during the study period ranged from 9.5ºC to 10.5ºC in piezometers 2c and 5c which is suitable for denitrification to occur at depths below 1 m.

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

A groundwater map was constructed using watertable data and surveyed surface water features on July 11th, 2006. As no significant seasonal deviation occurred, a median groundwater map was used to show groundwater flow direction. Groundwater contours (based on groundwater heads) deviated little from topography within the six isolated plots (Figure 3). Therefore, topography was used to infer the groundwater flow direction on the Sandhill area where a lower piezometer density exists. Groundwater flow direction was consistent throughout the study period and median groundwater flow contours were used to locate a PRB parallel to watertable contours. Where groundwater flow direction changes, the orientation of the PRB should be based on mean conditions. Based on median and maximum hydraulic heads, a barrier containing a 2 m-deep reactive zone is needed (reactive media should fill subsurface from 68 m AOD to 70 m AOD). This would ensure the reactive material was covered at all times by the watertable. A cross sectional conceptual model of the plume positions the centroid (area with highest nutrient concentration) around 2c – 5c (Figure 4). Nutrient concentration decreases outwards from the centroid. The extent of the plume migration vertically is unknown. Lateral plume extent varies from 350 m from 1c to 6c and extends further to 400 m at piezometer 1b. As the lateral plume diameter near to the source decreases the trench needs to be less than 350 m (Figure 3), to capture all groundwater flow migrating to Plots 2, 3, 4 and 5 (Figure 3).

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

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

Source tracking

Source tracking was used to connect the source, pathway and receptor of the nutrient loss. The median NO_3-N:Cl ratio in drains intersecting groundwater flow between the source and the plots were 0.46 (max 0.84) and 0.38 (max 0.72). Mean watertable depths in piezometers 3c and 2c during the same period are 0.52 m and 2.06 m, respectively. Therefore, the watertable from the up-gradient area intersected the drain adjacent to 3c and the flow in the drain was towards 2c. This means that contaminated groundwater passed into the plots and was then picked up in groundwater samples in the piezometers. To prevent contamination of surface water, the PRB should be located upslope from these drains and attenuate groundwater before any surface water groundwater interactions can take place (Figure 3).

Trench thickness

Using the denitrification rates in Table 3, Equation 7 was used to calculate the retention time needed to remediate the highest expected NO_3-N concentration expected (24.24 NO_3-N mg L^{-1}) to allowable levels. The retention time was then multiplied by the groundwater flow velocity to give the barrier thickness. The site is primarily on gley soils (95%) and the proposed trench location was on this soil type. Natural attenuation on-site would take longer periods of time. Potential receptors on site are approximately 200 m from the source. This would allow natural attenuation in gleys within 7.35 years. The travel time from Plots 3, 4, 5 and 6 would be less than this. However, natural attenuation to the east may be an option as travel times are much higher and the receptors are a greater distance away.

Discussion

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

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

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

1. Thorough site characterisation using all available data relating to the site is required. Data management and appropriate visual presentations such as maps, graphs and diagrams should be compiled. Distance from source to receptor should be calculated and topography defined.
2. Installation of a piezometer network between the source and potential receptors. Field visual tools (e.g. VS-Fast system) for soil field assessment may be a useful tool for preliminary studies, which enables *in situ* estimates of soil consistency, soil structure and texture (McGarry and Sharp, 2001). Other systems based on BS 5930:1999 are used in groundwater protection schemes to describe sub-soils (GSI, 1999).
3. Groundwater analysis and soil sampling should be carried out and a preliminary dataset should be compiled. Use calculated parameters to

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

4. The PRB trench thickness should be designed for specific water quality targets. Batch or column experiments should be carried out to calculate the reaction rate and equilibrium constant of the contaminant with the reactive media.
5. Identify travel times to potential receptors and locate the PRB up-gradient of the receptor. Compare PRB installation with monitored natural attenuation.

Before construction, the site should be evaluated to ensure design depth and width may be achieved. Trial holes should be considered. The ability of emplacing the reactive material without aquifer obstruction should be assessed to avoid clogging of media and smearing soil walls thus decreasing permeability. During and after installation, a monitoring network should be installed to investigate if denitrification is occurring in the trench and to investigate groundwater flow alteration due to the barrier construction. The ease of excavating the reactive media for replacement purposes after a period of time should be considered. Monitored natural attenuation on site should also be considered for areas further away from the source. A number of wells should be drilled in such locations. Pump-and-treat and pump-and-reuse would need considerable investment, drilling, discharge licences, and would need surface structures and maintenance which could interrupt farming practices. Recycling of water on farms is more likely to stem from soiled water remediation or rainwater harvesting and reuse. Pump-and-waste would also need a disposal licence and would merely export the problem elsewhere. The funnel-and-gate option is cost-prohibitive and would need geotechnical and engineering input in the design phases. However, a more feasible option for gate construction, such as compressed clay or another low-permeability material, should be investigated. A PRB installed south of the investigative plots would not capture all contaminated groundwater and could not achieve surface water quality targets. The current configuration would intercept contaminated groundwater before entering the six plots and before hydraulic gradients at location 4c divide the plume.

Conclusions

A continuous, shallow PRB may be suitable for Irish conditions to remediate point sources. Each site will have site-specific conditions but the methodology developed for this study site, based on site and groundwater characterisation, can successfully site a PRB and calculate the dimensions and orientation of the barrier. Further research should be carried out on the denitrification rates of different reactive media when combined with different soil groups. Higher NO_3-N removal rates will necessitate lower residence times and increased remediation. The longevity of the reactive media needs to be investigated and a cost-benefit analysis for the remediation of contaminated groundwater undertaken. A broader methodology should be investigated which takes into account other site characteristics, such as unconsolidated material, fractured bedrock and a deep watertable.

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Captions for figures & tables

Figure 1: Two types of subsurface PRBs a: funnel and gate, b: continuous trench; with source, NO_3-N plume, reactive material, treated plume and receptor. Watertable (WT) positioned within treatment wall containing reactive material and barrier constructed adjacent 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.

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Figure 4: Schematic diagram showing cross sectional (1c – 6c) conceptual model of the contamination plume with source on the sandhill. Highest median NO_3- 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.

Table 1: Piezometer parameters

Table 2: Plume distance and travel times using hydrogeological parameters

Table 3: Reactive media denitrification rate and PRB thickness to reduce NO_3- N concentration from 24.24 NO_3- N mg L^{-1} (highest concentration) to 11.3 mg NO_3- N L^{-1} (allowable concentration)

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

					Watertable height			Groundwater NO_3-N concentration mg L^{-1}		Groundwater NH_4 -N concentration mg L^{-1}	
Plot	I.D	Elevation m AOD	Total depth m	Multilevel	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
	a	67.24	2.95	2	0.96	1.55	0.07	3.12	14.66	0.04	2.23
FH7		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 NO_3^-N concentration from 24.24 mg NO_3^-N L^{-1}(highest concentration) to 11.3 mg L^{-1}(allowable concentration)

Reactive media	Denitrification rate* (µg 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) (m)	Vertical (y) (m)	Thickness (z) (m)
250	2	1.6 – 1.9
Orientation	Parallel to groundwater contours	
Reactive media	Woodchip and gley soil mix (ratio 5:2)	

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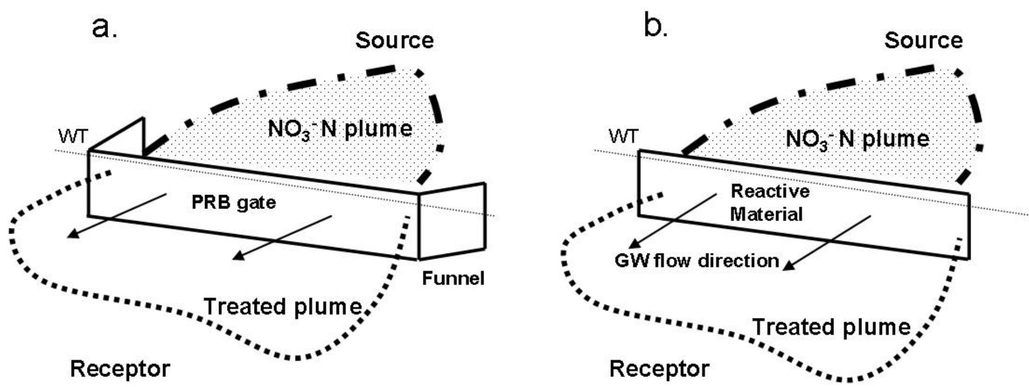

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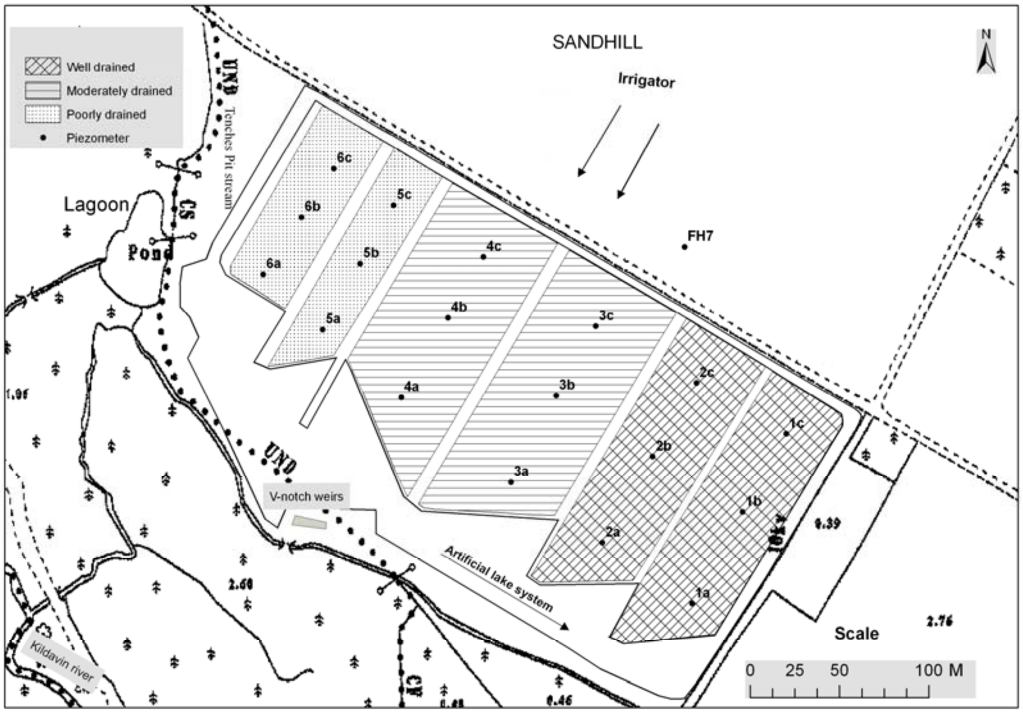

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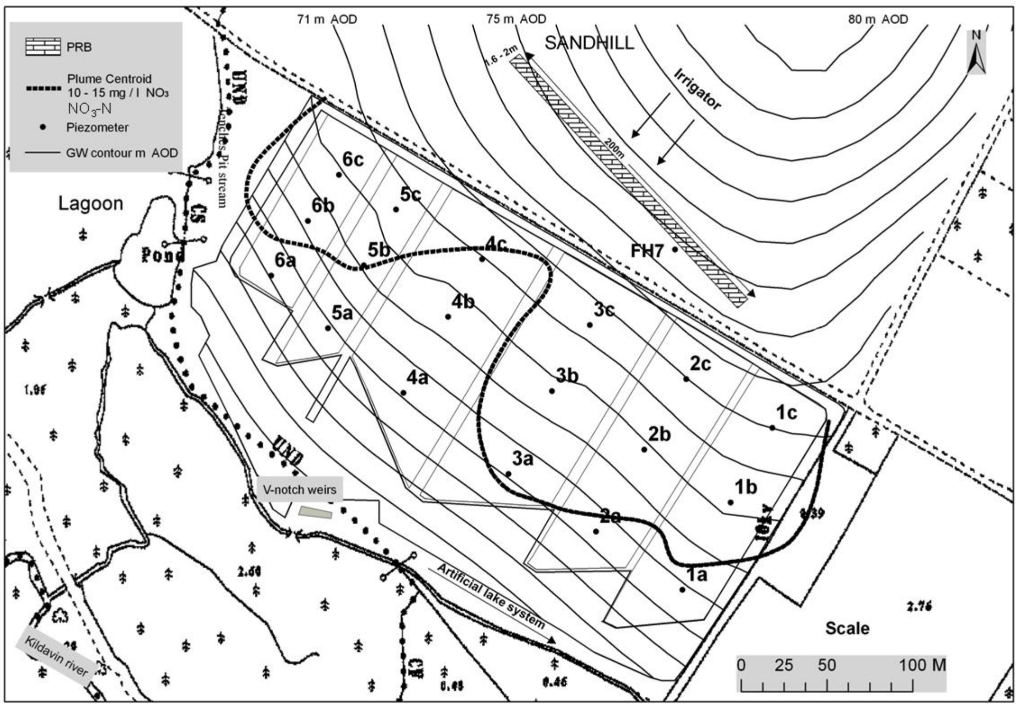

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