Title	A review of remediation and control systems for the treatment of agricultural wastewater in Ireland to satisfy the requirements of the Water Framework Directive
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Publication Date	2008
Publication Information	Fenton, Owen, Healy, Mark G., & Schulte, R.P.O. . (2008). A review of remediation and control systems for the treatment of agricultural wastewater in Ireland to satisfy the requirements of the Water Framework Directive. Biology and Environment: Proceedings of the Royal Irish Academy, 108B(2), 69-79.
Publisher	RIA
Link to publisher's version	https://www.ria.ie/biology-and-environment
Item record	http://hdl.handle.net/10379/1499

A review of remediation and control systems for the treatment of agricultural wastewater in Ireland to satisfy the requirements of the Water Framework Directive

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Abstract

In Ireland, agricultural activities have been identified as major sources of nutrient input to receiving waters and it has been estimated that these activities contribute 75.3% of the nitrogen (N) and 33.4% of the phosphorus (P) in these waters. Strategy at European level focuses on the prevention of nutrient loss by improved farm management. However, it does not focus on nutrient remediation or incidental nutrient loss of farmyard manures to surface and groundwater. This review describes the impact of agriculture on the environment in Ireland and examines emerging technologies for agricultural wastewater treatment. An integrated approach at pre-treatment and field stages for nitrate (NO_3) remediation and P control is recommended.

Keywords: Nitrogen, phosphorus, surface water, groundwater remediation.

1. INTRODUCTION

1.1 Legislative background

The Surface Water Directive, 75/440/EEC (EEC, 1975), the Groundwater Directive, 80/68/EEC (EEC, 1980), the Drinking Water Directive, 98/83/EC (EC, 1998), the Nitrates Directive, 91/676/EEC (EEC, 1991a), and the Urban Wastewater Directive, 91/271/EEC (EEC, 1991b), have focused considerable attention on the environmentally safe discharge of agricultural wastewaters in Ireland. The Water Framework Directive (WFD), 2000/60/EC (EC, 2000), came into force on 22nd December, 2000, and was transposed into Irish legislation by the European Communities (Water Policy) Regulations 2003 on 22nd December, 2003. Eight river basin districts (RBDs) have been established on the island of Ireland to facilitate the aim of achieving “good status” in all Irish water bodies by 2015.

The WFD will bring about major changes in the regulation and management of Europe's water resources that include, in general:

- a requirement for the preparation of integrated catchment management plans that identify point and non-point pollution, water abstraction and land use;
- the introduction of an EU-wide target of "good ecological status" for all surface waters and groundwaters;
- the planning and implementation of efficient and cost-effective measures to protect groundwaters and surface waters.

1.2 Agriculture and water quality in Ireland

In Ireland, agriculture is an important national industry that involves approximately 270,000 people, 6.191 million cattle, 4.257 million sheep, 1.678 million pigs and 10.7 million poultry (CSO, 2006). It utilizes 64% of Ireland's land area (Fingleton and Cushion, 1999), of which 91% is devoted to grass, silage and hay, and rough grazing (DAF, 2003). Grass-based rearing of cattle and sheep dominates the industry (EPA, 2004). In 2004, 60 million tonnes of agricultural waste were generated, of which 60.6% was from cattle manure and slurry (EPA, 2004; Table 1).

Livestock production is associated with external inputs of nitrogen (N) and phosphorus (P), which include chemical fertilisers, soiled waters and slurries. Nitrate (NO_3) leaching from wastewater irrigation is dependent on the hydraulic loading rate, soil water content and soil type (Ryan, 1998). Since both NO_3 and soil have negative electrostatic charges, NO_3 in solution tends not be taken up by the soil below the rooting depth and travels through the soil, leading to increased potential for NO_3 groundwater contamination (Abu – Ashor et al., 1994; Kung et al., 2000). The increases in dissolved P concentrations in rivers and streams have been linked - through overland flow and erosion losses - to the accumulation of excess soil P in these catchments under intensive animal production (Boesch et al., 2001). Daly et al. (2001) examined the sorption capacity and desorption dynamics in Irish grassland soils and found that high organic-matter soils have low P sorption capacities and poor P reserves compared with mineral soils, resulting in P losses from these organic soils where P amendments exceed crop needs (Daly et al., 2001).

Nutrient losses to surface and groundwater may have an adverse impact on biodiversity and ecology of aquatic agri-environment ecosystems (Schulte, 2006). Agricultural nutrient inputs are the most significant nutrient load entering receiving waters in Ireland and have been estimated to comprise 75.3% and 33.4% of the N and P loads in these waters (RBD, 2005). A survey of 1132 rivers and streams from 2001 to 2003 (Toner et al., 2005) estimated that the percentage of pollution attributed to agriculture was approximately 32% in rivers and streams that were slightly or moderately polluted, but only 15% in those that were seriously polluted. Other studies indicate that diffuse P losses from agriculture may contribute to eutrophication (Clabby et al., 1992; Bowman et al., 1996; Lucey et al., 1999; Mc Carrigle et al., 2002).

At present, the European strategy to restore the "good ecological status" of surface water and groundwater focuses on reducing further nutrient loss to these water bodies. Results from a Water4all (2005) project suggest that regulation alone may not achieve sufficient improvement in water quality in soils and groundwater aquifers in an acceptable timeframe and there may be need for more accelerated solutions (Water4all, 2005).

The objective of this paper is to examine emerging technologies for agricultural wastewater treatment in Ireland to satisfy the requirements of the WFD.

2. Current measures for the protection of waters

Traditionally, agricultural wastes are disposed of by land spreading. In land spreading, the recharge rate, the time of year of application, the hydraulic conductivity of the soil, the soil water content, the depth of soil to the water table and/or bedrock, and the concentration of nutrients and suspended sediment in the wastewater (soiled water and any discharge containing nutrients) are some of the defining parameters that determine NO_3 movement through the soil to the water table. The recommended maximum rate of application is 5 mm per hour and the quantity applied should not exceed 50 m^3 per hectare per application (ADAS, 1985).

3. Pre-treatment and *in situ* amendments

3.1 Alum and polyacryamide

Aluminium sulphate (alum) and polyacrylamide (PAM) are chemical flocculants commonly used in water treatment plants to remove P and suspended sediment. They can be used as pre-treatment and *in situ* amendments for agricultural wastewater amelioration. Alum should be applied to water or wastewater in a pH range of 5.5 - 9.0 as it has been found to be non-toxic in this range. Its final concentration in drinking water distribution systems and receiving waters should remain below 200 µg Al L^{-1} as this is the safe upper-limit concentration of aluminium for drinking water (WHO, 2003). PAM causes suspended particles to join and form aggregates which then rapidly settle out of suspension and are of filterable size, thereby removing particulate P from solution (Adin and Asano, 1998). Its soil stabilising and

flocculating properties improve runoff water quality by reducing sediments, N, dissolved reactive P (DRP) and total P (TP), chemical oxygen demand (COD), pesticides, weed seeds, and microorganisms in runoff (Sojka et al., 2007).

3.1.1 Alum and PAM for farm water treatment

The direct addition of alum or PAM to farm wastewater before land application may reduce the risk of nutrient loss to surface waters. The addition of alum sludge directly to soil prior to land spreading of wastewater may also be a viable option to control P. To date, the use of chemical amendments have mainly been investigated in poultry litter studies (Moore et al., 1999; Moore and Edwards, 2005, 2007). The US Department of Agriculture (USDA) has made the use of alum a conservation standard practice in several US states (Moore and Edwards, 2005) and, presently, about 700-800 million broilers per year are grown with alum in the US (P.A. Moore, personal communication). Limited work is investigating chemical additions to dairy wastewater (McFarland et al., 2003). In Ireland, no study has investigated the use of alum for wastewater treatment. Therefore, issues relating to Al release to surface waters need to be investigated.

Sims and Luka-McCafferty (2002) used alum as a poultry litter amendment (application rate 0.11 ± 0.01 kg alum per bird) on a farm-scale study. Alum was applied every six weeks to the litter before land spreading, after removal of each flock of broilers for processing. Alum amendment was shown to decrease litter pH (control 7.8±0.3 to amended 7.2±0.3), and the solubility of P (1475±492 to 405±192 mg kg^{-1}), inorganic arsenic (As) (19±4 to 7±3 mg kg^{-1}), copper (Cu) (272±50 to 172±45 mg kg^{-}

^{1}) and zinc (Zn) (29±7 to 15±10 mg kg^{-1}). Similar results were reported by Moore et al. (1997), who applied alum to poultry litter at a rate of 0.091 kg per bird (corresponding to 10% alum by weight of the broiler litter). In this study, reduced litter pH and decreased NH_3 volatilization from the litter resulted in atmospheric NH_3 reductions of 97% after 4 weeks in alum-amended houses.

Moore and Edwards (2005) also investigated the effects of alum addition to poultry litter on Al in runoff from 52 randomised 1.52m x 3.05m plots. Over a 10-year study, the application rates of alum-treated broiler litter were: 65, 130, 195 and 260 kg N ha^{-1}. Total and soluble Al concentrations in the runoff ranged from 0.6 to 1.6 mg Al L^{-1} and 0.1 to 0.2 mg Al L^{-1}, respectively. Udeigwe et al. (2007) also found that alum-amended litter can reduce the amount of water soluble P (WSP) in surface runoff water. Moore et al. (1999) reported WSP concentrations ranging from 15 to 40 mg P kg^{-1} in soils fertilised with unamended poultry litter at application rates of 2.24 to 8.98 Mg ha^{-1}. Alum-amended litter, applied at rates ranging from 65 to 265 kg N ha^{-1}, produced soil WSP concentrations similar to unfertilised soils - approximately 20 mg P kg^{-1} in this study.

The use of alum as a soil amendment has been shown to increase the binding potential of soils and is effective in immobilizing soluble P (McFarland et al., 2003; Zvomuya et al., 2006). McFarland et al. (2003) applied dairy wastewater (875 mg TN L^{-1}, 87 mg TP L^{-1}, 4.4 mg PO_4-P L^{-1}, 244 mg NH_3-N L^{-1}, 244 mg Al L^{-1} and pH 7.9) at 20 mm to three 2.5 m x 3 m plots: a control plot (5.4% slope), a plot amended with alum (alum dosage, 521.6 g; 6.4% slope), and a plot amended with gypsum (gypsum dosage, 576 g; 5.9% slope). Under a rainfall intensity of 76.2 mm h^{-1}, alum-amended

plots had maximum TP and PO_4-P concentrations in surface runoff of 14.3 and 0.07 mg L^{-1}; pre-application surface runoff concentrations were 13.3 and 0.66 mg L^{-1}, respectively. Post-application TP and PO_4-P concentrations from the gypsum-amended plots were 11.1 and 0.57 mg L^{-1}, respectively; pre-application surface runoff TP and PO_4-P concentrations were 12.1 and 0.54 mg L^{-1}, respectively. Al concentrations in the surface runoff water from the alum amended plot was 314 mg L^{-1} – 30% more than the pre-application Al runoff concentration of 220 mg L^{-1}. The authors did not measure soluble Al in this study, but the soils contained about 5,000 to 6,000 ppm Al before any alum was added (A.M.S. McFarland, personal communication).

Other studies using alum buffer strips have shown reductions in runoff DRP of up to 86% (Peters and Basta, 1996; Basta and Storm, 1997; Gallimore et al., 1999; Haustein et al., 2000; Dayton et al., 2003). Dayton and Basta (2005) applied poultry litter at a rate of 8.8 Mg ha^{-1} to the upper 75% area of a 0.5m-wide by 1m-long flume, inclined at a slope of 5%. In the remaining 25% of the downslope flume area, air-dried water treatment residue (WTR; Al range 1.39-165 g kg^{-1}) was applied to a buffer strip at rates of 0 (the control), 5, 10, and 20 Mg ha^{-1}. Under rainfall intensities of 70 mm hr^{-1}, applied for 30 minutes, mean DRP concentrations in the control studies were 31.1 mg L^{-1}. For WTR additions of 5, 10 and 20 Mg ha^{-1}, mean DRP in the surface runoff was reduced by 37.6, 50.5 and 86.2%, respectively.

PAM has also been used to separate solid and liquid components of swine manure. Optimum PAM dosage rates vary with the amount of SS in the liquid manure; 26 and

79 mg PAM L^{-1} for samples containing 1.5 and 4.1 g TSS L^{-1}, respectively, have achieved 90 to 94% removals (Vanotti and Hunt, 1999).

PAM greatly reduces irrigation-induced erosion on furrow irrigated fields while sediment ponds can be constructed to remove suspended sediment from irrigation runoff with seasonal application rates of 1 kg PAM ha^{-1} (furrow sub-surface irrigation) to 5 kg PAM ha^{-1} (sprinkler irrigation) (Lentz and Sojka, 1992). Application rates of 1 kg PAM ha^{-1} should be applied after first cultivation to reduce furrow irrigation-induced erosion and an additional 0.5 – 1 kg ha^{-1} for the next three irrigations. An initial dose of PAM at 10 mg L^{-1} in irrigation inflows during the furrow advance period may achieve 93% reduction in sediment loss (Lentz and Sojka, 1992).

3.1.2 Alum and PAM for surface waters

Alum and PAM may be also used to reduce the SS and nutrient concentration of surface waters. Nutrient-rich agricultural wastewater has caused eutrophication in the Salton Sea, California (Mason et al., 2005). The removal of dissolved P and P-laden sediment from this water using non-ionic PAM (2 mg L^{-1}) and alum (4 mg L^{-1}) - added to ditches receiving tributary waters - substantially reduced SS and turbidity in low energy systems (velocity gradients $< 10\ s^{-1}$) by 95%, and soluble P by 93%. Best results are obtained when PAM and alum are used in conjunction with settlement basins or low-flow regimes.

3.2 Ochre

Ochre (a ferric oxyhydroxide precipitate) deposits can occur due to acid mines, and can be ecologically devastating (Gray, 1996). The sorption capacity of ochre to sequester P ranges from 0.5 g P kg^{-1} to 2 g P kg^{-1} (Bozika, 2001) and is site-specific (Heal et al., 2005). Preliminary studies on the P-sorption capacity of ochre from the Avoca-Avonmore river catchment in the south-east of Ireland suggest that it is capable of adsorbing up to 16 g PO_4-P kg^{-1} (Fenton et al., 2007). The ochre P adsorption capacity compares very favourably with other low cost media (Table 2). The potential for ochre to reduce P from soiled water is high and, if used in conjunction with biofilters, may provide an efficient means of treating soiled water. Ochre-P pellets, developed by the University of Newcastle in the U.K (Heal et al., 2005), allow *in situ* applications of ochre at specific locations (P stripping zones) on a farm without discoloration of water. They absorb P from solution and may be used in the remediation of wastewaters from different sources, such as agricultural runoff. Exhausted pellets may then be pulverized and applied as fertilizer. As P desorption from saturated ochre is < 1% (Fenton et al., 2007), it may be used in surface water and replaced when saturated.

3.3 Relevance and applicability of alum, PAM and ochre for Ireland

The EU Sewage Sludge Directive, 86/278/EEC (EEC, 1986), specifies limit values for maximum concentrations of heavy metals in soil and sludge and limit values for maximum annual quantities of heavy metals introduced to the soil (Table 3). A Code of Good Practice for the Use of Biosolids in Agriculture (DEHLG, 1999) sets new standards for treatment of biosolids. These standards are broadly in line with the USEPA ‘Class A’ standard. Biosolids that meet such standards must have very low

pathogen content, have low metal content and the organic matter is stabilized so there is little odour or possibility of attracting pests that spread disease. Such "exceptional quality biosolids" can be used on the farm without a site permit, or can be sold to consumers for garden use. This presents new challenges for the optimisation of sludge treatment and final effluent quality. However, not all sludge is suitable for land application. In a study in the south east of Ireland, 21% of soils breached the provisions of the EU Sewage Sludge Directive for heavy metals before any sludge application (McGrath and McCormack, 1999). This, coupled with the suitability and availability of tillage lands, poses problems for sludge application.

With 90% of all sludge coming from agriculture, the addition of alum or PAM to farm wastewater before land application would reduce the risk of nutrient loss to surface waters. This could be done in various ways: direct alum or PAM application to soil, simultaneous application during land spreading or prior application to storage facilities. Another option is to apply alum and PAM in buffer strips. Ochre could be applied in conjunction with alum and PAM to sequester P after precipitation of solids has occurred. However, the possibility of heavy metal loss in surface runoff needs to be further investigated.

4. Emerging technologies for wastewater treatment

4.1 N removal

Conventional methods have been used to remediate NO_3 contamination, including: monitored natural attenuation (ASTM, 1998); pump-and-treat (USEPA, 1990),

wherein treated water is used to irrigate crops; pump-and-waste (USEPA, 1990), wherein contaminated water is evaporated or injected into a saline aquifer or geological unit; and phytoremediation (Suresh and Ravishanker, 2004). Pump-and-treat may be expensive and pump-and-waste is not sustainable and causes plume migration.

New and emerging pre-treatment remediation technologies, such as 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), fluidised-bed biofilm reactors (Rabah and Dahab, 2004) and rotating biological contractors (Ayoub and Saikaly, 2004) have shown good potential for biological N removal from domestic and agricultural wastewaters. These technologies may be used to remediate dairy parlour washings and soiled water, and may reduce storage volumes and associated costs.

4.1.1 Permeable reactive barriers

Low-cost, *in situ* treatment systems, called permeable reactive barriers (PRBs), may be used to treat groundwater. PRBs are defined as "an emplacement of reactive materials in the subsurface designed to intercept a contaminant plume, provide a flow path through the reactive media, and transform the contaminants into environmentally acceptable forms to attain remediation concentration goals down-gradient of the barrier" (Powell and Powell, 1998). *In situ* subsurface denitrification trenches, wherein wastewater flows through a carbon (C) rich mixture to reduce NO_3 concentrations, is a PRB adapted for agricultural use (Healy et al., 2006). Organic C

amendments offer low-cost surface and subsurface treatment alternatives for wastewater.

Four types of PRB exist: (1) a funnel-and-gate system used primarily for halogenated hydrocarbons, aromatic compounds and heavy metal remediation; (2) an injection well configuration where a reactive wall is generated through injection of a reactive solution; (3) passive collection with reactor cells, where contaminated water is drained to a reactive zone; and (4) a shallow continuous trench used for NO_3 remediation. Horizontal flow (Erickson et al., 1974) or vertical flow (Robertson and Cherry, 1995) denitrification trenches using a solid carbon source (woodchip) as the filter media have been used in Australia, Canada, USA and New Zealand (Foundations for Water Research, 2004).

4.1.2 Reactive materials for PRB

Denitrification may be increased in soils by the addition of an external C amendment. This C amendment could include woodchip, wheat straw, corn, vegetable oil, sawdust mulch, treated newspaper or unprocessed cotton (Volokita et al., 1996). *In situ* treatment may involve material being used separately or mixed with soil or sand. Different media have different denitrification rates (Table 4). Sawdust has higher denitrification rates due to its associated higher surface area but is prone to clogging. After barrier construction, Schipper et al. (2004) measured saturated hydraulic conductivities of 0.48 m day^{-1} and 65.4 m day^{-1} in a PRB sawdust wall and aquifer, respectively; this caused groundwater flow under - rather than through - the reactive

media. Another disadvantage of sawdust is its low durability over time (Horn et al., 2006).

Healy et al. (2006) examined the use of various wood materials as a carbon source in laboratory horizontal flow filters to denitrify NO_3 from a synthetic wastewater. The filter materials were: sawdust (*Pinus radiata*), sawdust and soil, sawdust and sand, and medium-chip woodchippings and sand. Two influent NO_3-N concentrations, 200 mg L^{-1} and 60 mg L^{-1}, loaded at 2.9 to 19.4 mg NO_3-N kg^{-1} mixture d^{-1}, were used. The horizontal flow filter with a woodchip/sand mixture, loaded at 2.9 mg NO_3-N kg^{-1} d^{-1}, performed best, yielding a 97% reduction in NO_3-N at steady-state conditions. Greenan et al. (2006) investigated four different C sources - mixed with C source to soil volume ratios of 1 - in anaerobic batch experiments, as follows: (i) 3-10 cm long wood chips (predominately *Quercus* spp.) (ii) wood chips saturated with soybean oil (48% oil by weight) (iii) dried cornstalks collected after harvest, and (iv) paper fibres from corrugated cardboard. Over a 180 day study period, denitrification rates ranged from 0.427 g N kg^{-1} substrate d^{-1} for the ground cornstalks to 0.066 g N kg^{-1} substrate d^{-1} for the wood chips

4.1.3 Implementation of PRB

In PRBs, the reactive material is placed in a trench and sealed to surface level with clay to avoid surface - subsurface cross contamination and to achieve anaerobic conditions. The reactive zone must have a higher conductivity than the surrounding soil to encourage flow into the reactive zone (Simon and Meggyes, 2000). Filter gravel should be placed at the edges of the reactive zone to stop small particles

washing and clogging the trench. Geotechnical considerations, such as subsurface soil strength and the presence of cobbles, should be considered. A temporary piezometer network or ground penetrating radar survey should be utilized to identify the location and movement of the migrating nutrient plume on-site. A trench orientated perpendicular to groundwater flow direction, taking annual deviations into consideration, should be placed at various depths, depending on average water table heights. It should also be placed at strategic positions near potential point pollution sources, soiled water installations, slurry and silage facilities, along shallow groundwater zones adjacent to riparian zones, ditches, or open water ways (Seong-Chun et al., 2005).

The time frame for site evaluation, hydrogeological study, engineering design and implementation could take from 14 to 30 weeks (Kalin, 2004). Irish farmers, under the REPS, must leave a 1.5 m-wide buffer strip of uncultivated land beside watercourses. Buffer strips may have a positive effect on P and pesticide loss, as low soil P concentration and permanent cover “trap” P. A trench placed at such a location integrates nutrient remediation and control and could potentially cut down on the design and implementation timeframe. A methodology suitable for Irish conditions for the location, construction, trench type, dimensional criteria and monitoring of a permeable reactive barrier from a point source has been devised (O. Fenton, unpublished data). The long term performance of PRBs needs to be assessed

4.2 Willow and reed plantations

Willows (*Salix* spp.) are also gaining in popularity in Ireland and elsewhere for the treatment of domestic and agricultural wastewater (Rosenqvist and Dawson, 2005; Börjesson and Berndes, 2006). A long growing season and a high nutrient retention capacity make them ideal for wastewater treatment (Dimitriou and Aronsson, 2004). The Landfill Directive, 99/31/EC (EC, 1999), forces local authorities to reduce the volume of organic waste disposed in landfills. To date, willows have been viewed as an alternative, environmentally-friendly energy source to satisfy the greenhouse gas (GHG) emission requirements of the Kyoto Protocol (Rice, 2003).

Willows assimilate nutrients into plant biomass. They remove pollutants by directly assimilating them into their tissue. Biomass production in willows is dependent on the amount of N, P and potassium (K) that is applied to the soil (Hodson et al., 1993).

Willows are normally planted at 0.75–1.5m centres between rows and at 0.5–0.6m distances along each row, and are harvested every 3-5 years (Aronsson et al., 2002). Sludge can be allowed to percolate between the willow rows through a drip irrigation system and is normally applied at a rate of 80 kg N ha^{-1} yr^{-1} (Aronsson et al., 2002). Nutrients are permanently removed from the system by annual harvesting.

Compared to conventional wastewater treatment, wastewater irrigation of willow plantations can offer great savings. Dawson (2004) estimated that a willow area of approximately 3,000 ha would be required for the disposal of all domestic sewage sludge in Ireland. Rosenqvist and Dawson (2005) calculated that savings of around €7

- €18 kg^{-1} N could be made in a willow irrigation system compared to a conventional wastewater treatment plant.

In Sweden, wastewater irrigation of willow plantations is now commonly used (Perttu, 1998; Lindoff Communications Ltd., 2004; Dimitriou and Aronsson, 2004; Dimitriou and Aronsson, 2005) and hydraulic loading rates of up to 600 mm yr^{-1}, yielding 125 kg N ha^{-1}, may be applied without the risk of N leaching to groundwater (Börjesson, 1999). In Kågeröd, southern Sweden, biologically treated wastewater from a population equivalent (PE) of 5000 was used for irrigation on an 11 ha willow plantation (Lindoff Communications Ltd., 2004). Wastewater was applied from May to October at an average rate of 4 – 5 mm d^{-1} (730 – 770 mm yr^{-1}), giving average yearly N and P application rates of 72 kg N $ha^{-1}yr^{-1}$ and 10 kg P $ha^{-1}yr^{-1}$, respectively. Average Tot-N, Tot-P and biochemical oxygen demand (BOD) concentrations were reduced by 79%, 11% and 55%, respectively. Evapotranspiration was not measured in this study.

Regular fertilization and irrigation increases the biomass and the nutrient retention within the willow shoot. In a study conducted in New York, Adegbidi et al. (2001) found that, under annual nutrient application rates of 224 kg N ha^{-1}, 112 kg P ha^{-1} and 224 kg K ha^{-1}, drip irrigated at 20-60 mm wk^{-1} during the growing season, between 2.5 Mg ha^{-1} yr^{-1} (for non-irrigated plots) and 27.5 Mg ha^{-1} yr^{-1} (for irrigated plots) of biomass was produced from willows. Biomass production rates of *Phragmites australis* (Cav.) Trin. Ex. Steudel, a plant commonly used in CWs for the treatment of wastewater, is within this range. Karunaratne et al. (2004) investigated the effects of harvesting *P. australis* in a wetland in Central Japan and found that biomass levels

rose to 1250 g m^{-2} (approximately 12.5 Mg ha^{-1}) in July. Similar figures have been found in Ireland (Healy et al., 2007). *P. australis* does appear to have a greater ability to remove N and P, however. In a CW in Ireland, Healy et al. (2007) measured maximum nutrient retentions of approximately 15.5 mg N g^{-1} dry weight (DW) and 1.6 mg P g^{-1} DW in *P. australis*. In a wetland in Northeast Italy, planted with *P. australis*, Bragato et al. (2006) measured maximum Tot-N and Tot-P concentrations of 27 mg N g^{-1} DW and 0.8 mg P g^{-1} DW in July. These values are far in excess of the measurements conducted by Adegbidi et al. (2001), where maximum nutrient retentions of 3.7 – 7.2 mg N g^{-1} DW and 0.6 – 0.7 mg P g^{-1} DW were measured in willows.

5. Remediation options for agriculture in Ireland

To meet the requirements of the Nitrates and WFD directives, groundwater and surface water remediation technologies are required to capture nutrient loss where nutrient management fails. An integrated approach is needed to address multiple simultaneous challenges of N and P losses. Consequently, *in situ* and pre-treatment of farmyard manures should integrate N remediation and P control.

Low-cost, low-management remediation technologies, such as PRBs and willows, have good potential in Ireland because they can be implemented at farm level. As woodchip and woodchip mixed with soil/sand barriers may result in NO_3 removal and, depending on hydraulic loading rate, may have a long lifespan, the growth of willow plantations to provide a C source for PRBs should be investigated. Batch and column experiments investigating the denitrification rates and required retention times

in PRBs to achieve water quality targets of different solid carbon media should be investigated. A decision support system should be developed to provide guidelines to farmers in the location of a PRB, on available and suitable reactive media, and associated costs.

Buffer strips, amended with ochre, or willow plantations may also be used to treat surface water and runoff. Mitigation measures utilising existing agricultural infrastructure such as open drains and farmyard outlets should be considered, which divert drainage and runoff water to reactive cells, then trap sediment (particulate P) and sequester soluble P.

Specifications for the implementation of these technologies on-site should be developed and future national policy needs to change to incorporate remediation technologies. Future work should compare the cost-benefit of implementing the alternative remedial technologies and estimate the economic value of such improvements on the ecology of Irish rivers.

6. Conclusions

1. Current legislation is focused on prevention of nutrient losses from agricultural sources. Remediation and control technologies are recommended to account for incidental losses.
2. Waste products, such as alum from water treatment and ochre from acid mine wastewater, should be investigated for use in P control in surface water and dirty water. The release of heavy metals from these chemical amendments

should be investigated on different soil types to address WHO drinking water guidelines.

3. Options for NO_3 removal include *in situ* denitrification trenches and willow plantations.

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Table 1 — Estimated agricultural organic managed waste generation in 2001 (EPA, 2004).

Waste category	Waste generation (Tonnes wet weight)	%
Cattle manure and slurry	36,443,603	60.6
Water (dairy only)	18,377,550	30.5
Pig slurry	2,431,819	4.0
Silage effluent	1,139,231	1.9
Poultry litter	172,435	0.3
Sheep manure	1,336,336	2.2
Spent mushroom compost	274,050	0.5
Total	60,170,025	

Table 2 — Maximum adsorption capacities (g P kg^{-1} substrate) of different media. (after Mann, 1997; Heal et al., 2005).

Amendment	Maximum adsorption capacity g P kg^{-1}
Danish sands	0.02-0.13
Gravel	0.03-0.05
Bottom ash	0.06
Steel furnace slag	0.38-1.4
Blast furnace slag	0.05-0.65
Fly ash	0.62
Shale	0.75
Laterite	0.75-1.38
Zeolite	1-2.2
Serpentinite	1
EAF steel slag	2.2
Polkemmet ochre	26
Minto ochre	30.5

Table 3 - Maximum values for concentrations of heavy metals in soil and sludge (for agricultural use)(EEC, 1986).

Parameters	**mg kg^{-1}** Soil	**mg kg^{-1}** Sludge
Cadmium	1	20
Copper	50	1000
Nickel	30	300
Lead	50	750
Zinc	150	2500
Mercury	1	16

Table 4 - Solid carbon reactive media and nitrate removal rates.

Reference	Experiment type	Influent NO_3-N concentration mg L^{-1}	Media % by volume	Residence time days	NO_3-N removal rate %
Healy et al., 2006	Lab column	60	Woodchip (50%)	-	97
Fahner, 2002	Field study	63	Sawdust (30%)	3.5-7	76
	Lab column	12	Sawdust (30%)	0.5-7	40
Carmichael, 1994	Lab column	50-87	Woodchip (100%)	1.6	72-83
Schipper and Vojvodić, 2001	Field study	5-15	Sawdust (30%)	-	95
Robertson et al., 2000	Field study	57	Waste cellulose[1] (15%)	17	80
	Field study	1.2	Waste celluslose (15%)	30	83

[1] Waste cellulose = wood mulch, sawdust, leaf compost.