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Nutrient loading on subsoils from on-site wastewater effluent, comparing septic tank and secondary treatment systems

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Abstract

The performance of six separate percolation areas was intensively monitored to ascertain the attenuation effects of unsaturated subsoils with respect to on-site wastewater effluent: three sites receiving septic tank effluent, the other three sites receiving secondary treated effluent. The development of a biomat across the percolation areas receiving secondary treated effluent was restricted on these sites compared to those sites receiving septic tank effluent and this created significant differences in terms of the potential nitrogen loading to groundwater. The average nitrogen loading per capita at 1.0 m depth of unsaturated subsoil equated to 3.9 g Total-N/d for the sites receiving secondary treated effluent, compared to 2.1 g Total-N/d for the sites receiving septic tank effluent. Relatively high nitrogen loading was, however, found on the septic tank sites discharging effluent into highly permeable subsoil that counteracted any significant denitrification. Phosphorus removal was generally very good on all of the sites although a clear relationship to the soil mineralogy was determined.

Keywords nitrogen; phosphorus; on-site wastewater; septic tank; percolation; subsoil

Changes / comments made shown in blue

1. Introduction

[All paragraphs in this section have been amended and shortened and several refs have been removed such as (Cogger and Carlilse, 1984; Wolf et al., 1998; Department of the Environment, 1999; Manassaram, et al., 2006, Swartz et al., 2003].

Groundwater is an important resource in Ireland which is at risk from the increasing numbers of decentralized houses and their respective on-site wastewater treatment systems. Domestic wastewater from over one third of the population is treated by on-site systems and, with more than 25% of all water supplies provided by groundwater (EPA, 2005), the protection of groundwater resources from contamination by domestic wastewater effluent is imperative. The unsaturated subsoil above the water table or bedrock into which on-site effluent is discharged (i.e. the percolation area) is therefore an integral part of the overall on-site treatment system, particularly since the main aquifers in Ireland occur in fissured or fractured bedrock formations overlain by subsoils of variable thickness and permeability (Misstear and Daly, 2000).

Groundwater needs to be protected from nitrogen pollution for both ecological reasons (nutrient enrichment of sensitive surface waters) and health based reasons, the latter mainly due to a perceived link between nitrate concentrations and methaemoglobinemia ("blue-baby syndrome"), increased incidence of cancer, adverse reproductive effects, and other possible effects. Indeed the limit for maximum nitrogen concentrations in groundwater is often founded on health based grounds at 50 mg-NO_3 /L as defined by the EU Drinking Water directive (98/83/EC) and WHO guideline value (2006). There is increasing scepticism, however, as to the simple link that has been made in the past between drinking nitrate in water and methemoglobinemia due to the generally poor correlation between drinking-water nitrate and blue-baby syndrome (L'hirondel and L'hirondel, 2002; Fewtrell, 2004) and that numerous other factors such as hereditary enzyme deficiencies, protein intolerance and a variety of trace chemicals are now known to cause the disorder (Avery, 1999; Knobeleoch et al., 2000; Ward et al., 2005). Elevated nitrate concentrations in groundwater however, are an indicator of wastewater or agricultural pollution and also indicate potential microbial risks if the groundwater is used as a drinking water supply.

In temperate fresh waters, dissolved phosphorus tends to be the limiting nutrient (Schindler, 1977) and excess loading onto lakes and rivers can initiate algal blooms and generate negative aesthetic and eutrophic conditions in such water bodies. In Ireland groundwater forms the baseflow of rivers, with annual average contributions ranging from less than 30% of total river flow where the catchment is underlain by poorly productive aquifers to over 80% where there are regionally important aquifers in good hydraulic connection to the river (Fitzsimons and Misstear, 2006). Therefore, groundwater must be protected from phosphorus pollution.

A septic tank acts primarily as a settlement chamber providing quiescent, anaerobic conditions that facilitate the reduction of the organic and suspended solids content of wastewater (Goldstein and Wenk, 1972; Viraraghavan, 1976; Canter and Knox, 1985). The environment within the septic tank is, however, largely ineffective in reducing the nutrient loading of the wastewater, acting only to convert the influent organic nitrogen to ammonium achieving little total nitrogen removal across the process (Lawrence, 1973; Canter and Knox, 1985; Beal et al., 2005; McCray et al., 2005). Equally, the anaerobic conditions in the tank convert most of the influent phosphorus, in the form of both organic and condensed phosphate (polyphosphate), to soluble orthophosphate which passes out in the effluent (Willhelm et al., 1994; Zanini et al., 1998; Beal et al., 2005). A secondary treatment system can be installed as an alternative to a septic tank or to provide subsequent treatment of septic tank effluent prior to discharge to subsoil. The controlled aerobic environment for the accelerated microbial degradation of organic matter and often, nitrification of ammonia does not, however, result in a significant reduction in the total nitrogen loading across the process. Equally, most package wastewater treatment systems are not generally designed to remove phosphorus, although some phosphorus reduction (around 15%) is usually achieved in bacterial assimilation, precipitation and adsorption (Metcalf and Eddy, 2003).

The soil treatment system, comprised of a series of subsurface percolation trenches, is a crucial component of the gravity flow treatment system with much research concentrating on the flow of effluent and the mechanisms of pollutant attenuation within the subsoil (Jenssen and Siegrist, 1990; Beal, et al., 2005). The biogeochemical mechanisms for purification and hydraulic performance are complex and have been shown to be highly influenced by the biomat zone which forms at the soil-gravel interface along the base and wetted sides of the percolation trenches. Reduced percolation rates through the biomat due to clogging as a result of anaerobic activity can cause the effluent to pond above the biomat but leaves unsaturated conditions below, for aerobic degradation processes to operate on percolating effluent (Bouma, 1975; Siegrist and Boyle, 1987; Beal et al., 2008). The development of a biomat takes several months but will eventually reach a steady state equilibrium (Hillel, 1980) which the Long Term Acceptance Rate (LTAR) - the basis of several design codes in Europe (CEN, 2006) and elsewhere – attempts to define. There is a clear relationship between the organic loading rate and rate and extent of biomat development and so the provision of secondary treatment before the percolation trenches reduces the rate and extent of biomat growth (Siegrist and Boyle, 1987; Jantrania and Gross, 2006). The removal efficiencies of nutrients in unsaturated subsoil can be limited (Jenssen and Siegrist, 1990; Beal et al., 2005) although nitrification of septic tank effluent however, is also commonly reported in the unsaturated zone beneath the biomat (Pell and Nyberg, 1989b; Van Cuyk et al., 2001). Other studies have shown that denitrification of percolating nitrates in soils tends to occur more readily in fine-textured soils (i.e. clays and silt/clays) compared to coarse-textured soils (silts and sands) (Tucholke et al., 2007).

The attenuation of ortho-P within the subsoil treatment system is controlled by soil adsorption and mineral precipitation and is dependent not only on its clay content, but also on the presence of Al, Fe, or Mn in acidic soils and the presence of Ca in alkaline soils (Robertson, 2003). Phosphorus precipitation is dominated by iron and aluminium when pH <6 (Erickson et al., 2007) whilst at pH levels >6 the reactions are a combination of physical adsorption to Fe and Al oxides and precipitation as sparingly soluble calcium phosphates onto calcareous sands (Arias et al., 2001). Other studies have shown that conditions of low redox potential (i.e. saturated, partially anaerobic) reduce the mineralization rate and increase the adsorption rate (Pell and Nyberg, 1989a).

Two sequential projects funded by the Irish EPA have been undertaken to test out the efficacy of on-site wastewater treatment systems on a range of sites with subsoils of different percolation characteristics receiving domestic wastewater effluent. The research objectives addressed by this paper were to investigate the nutrient loadings occurring at different depths beneath the different percolation trenches and assess the implications in terms of groundwater pollution with respect to housing development in unsewered, rural areas.

2. Methods

2.1. Site selection and construction

Six separate households in Ireland were used for the research projects on sites of varying subsoil permeability. In all sites new on-site treatment systems were constructed by the research team at the beginning of each study. Two-chamber septic tanks were installed on Sites 1, 2 and 3 while secondary treatment systems were installed on the other sites: a rotating biological contactor, RBC (Biodisc®, Klargester) on Site 4 and peat filters (Puraflo®, Bord na Mona) on Sites 5 and 6. The six sites were chosen to cover a range of subsoil conditions as characterised by the British Standards Institution's BS5930 (1999) classification system (Table 1). The percolation T-values of the different subsoils span the range 4 to 60 as determined by the onsite standardised Irish falling head percolation test, the T-test (Mulqueen and Rodgers, 2001), equivalent to field saturated hydraulic conductivities in the range 0.08 to 1.05 m/d (Elrick and Reynolds, 1986). The effluent from all six sites (three sites discharging septic tank effluent (STE) and three sites discharging secondary treated effluent (SE) from packaged plants) entered percolation trenches at 2.45 m centres built to EPA specifications (EPA, 2000) consisting, in each case, of 110 mm diameter perforated PVC pipe sitting on a bed of 250 mm of washed gravel (20 to 30 mm diameter) in a 450 mm wide trench (see Fig. 1) at a slope of 1:200. The design specification was generally one 20 m long trench per person per household. The achievement of an equal loading rate on each trench was achieved by a distribution box specifically designed for the projects.

2.2. Instrumentation and sample analysis

The flow profile to the percolation areas from the septic tanks and secondary treatment systems was measured using tipping bucket flow-gauges (Unidata, Australia) placed underneath each of the four distribution box outlets at each site. Automatic samplers (Bühler Montec) collected 24 hour composite samples of STE and SE. Suction lysimeters (Soilmoisture Equipment Corporation) were installed at the start (0 m), middle (10 m) and end (20 m) of each trench to nominal depths of 0.3, 0.6 and 1.0 m below the invert of the percolation trenches respectively (Fig. 1a). Lysimeters were installed at more frequent intervals at Sites 1 and 4 corresponding to distances 0, 5, 10 and 15 m (and 20 m for Site 1) along each trench, in order to refine the analysis of biomat spread. The careful installation of these lysimeters into the undisturbed subsoil directly beneath the trench was essential to avoid creating artificial preferential flowpaths and to ensure that they were located within the effluent plume. At each site nine tensiometers (Soil Measurement Systems) were installed at the same three sampling depths along separate trenches in order to obtain a profile of soil moisture tension across the percolation area. Meteorological variables (rainfall, temperature, wind speed, relative humidity, solar radiation and sunshine hours) on each site were recorded by a weather station (Campbell Scientific) and tipping-bucket type rain gauges (Casella).

The lysimeters were put under a suction of 50 kPa using a vacuum-pressure hand pump which was low enough to prevent the extraction of bound moisture that would otherwise be unavailable to recharge. Sampling was carried out the following day using a vacuum-pressure pump and the samples were collected in sterilised plastic sample tubes, stored in a cool box before being taken directly to the laboratory for analysis. All STE, SE and soil moisture samples were analysed for ammonium, nitrite, nitrate, chemical oxygen demand, orthophosphate and chloride using a Merck Spectroquant Nova 60® spectrophotometer and associated reagent kits. Samples were also tested for total nitrogen using a Hach Lange LT200 thermostat and spectrophotometer DR2800 to ascertain the fraction of organic and inorganic nitrogen present. Sites 2, 3, 5 and 6 were each studied for a period of 12 months, compared to Sites 1 and 4 which were studied over a longer period of 32 months in order to monitor any further biomat spread and trends in removal processes with respect to time.

Samples of the subsoils at different depths were analysed in Trinity College Dublin using X-ray diffraction analysis using a Phillips PW1720 X-ray generator with a PW1050/25 diffractometer and PW3313/20 Cu k-alpha anode tube, to reveal the respective mineral composition. Finally, at the end of the research studies on Sites 1 and 4, a specialist closed-circuit television (CCTV) company sent a camera down each percolation pipe to inspect the extent of biomat development.

3. Results and Discussion

The extent of the active percolation area (i.e. spread of the biomat) first needed to be established in order, (i) to decide which of the lysimeters were sampling percolating effluent and, (ii) to factor in the effect of any rainfall dilution on the samples due to recharge through the subsoil

3.1. Extent of active percolation area and effect of dilution by rainfall

Chloride was used initially as a tracer to identify the extent of the percolation area which was receiving wastewater effluent. The results of the laboratory analysis for Cl at the different sample positions at the three different depth planes and along each trench were analysed with respect to time.

From this a conceptual model (assuming isotropic and homogeneous soil properties) was derived for the analysis of the attenuation of the percolating effluent according to how far across the percolation area the biomat was spreading. The concentrations of all the other measured parameters were then averaged across all trenches and across all sampling positions where the effluent was known to have reached at the 0.3, 0.6 and 1.0 m depth planes.

An example of the method of calculating loads is shown in Fig. 2(a) for Site 2, where similar chloride loading rates for all planes indicate that the effluent had spread across the whole length of the percolation area. Similar analysis of Cl levels carried out on Sites 3, 4, 5 and 6 showed decreased concentrations at the 5, 10 or 15 or 20 m sample distances along the same depth planes, as shown on Fig. 2(b), suggesting that the average results across the trenches and depth planes recorded at the 0 m sampling positions only would be the more representative way to assess effluent attenuation on those sites. At Site 1 the results showed that the biomat in the trenches receiving STE had spread past the 0, 5 and 10 m sampling positions but not as far as the 15 or 20 m positions.

Loading rates of nitrogen and phosphorus compounds at the different depths were then calculated on a daily basis according to a mass balance of effluent flow plus any rainfall recharge. The extent of effluent dilution in the subsoil from effective rainfall was calculated using rainfall and evapotranspiration values obtained on site and calculations based on the Penman-Monteith method (FAO, 1998). Effective rainfall was calculated by subtracting the daily actual evapotranspiration and accumulated soil moisture deficit figures from the daily rainfall measurement.

The effect of dilution on the attenuation of the percolate was then calculated by determining the zone of contribution around each trench. The disparity in Cl concentrations between the effluent as it leaves the septic tank and where it was sampled at the three depth planes in the subsoil beneath the percolation trench was examined during both dry periods and periods of sustained effective rainfall. The reduction in Cl concentration with subsoil depth during periods of zero effective rainfall was attributed to the physical straining of colloidal matter in both the biomat and underlying subsoil. The difference, therefore, between data showing the reduction in Cl concentration between the trench influent (i.e. STE, or SE) and at the three depth planes during periods of effective rainfall compared to periods of zero effective rainfall was used to calculate the percentage reduction in Cl concentration due exclusively to dilution. Having quantified the dilution factors at each depth plane, a simple mass balance approach was adopted to estimate the zone of contribution (A_c) of effective rainfall at each depth plane. For example, on Site 1 using the average concentration of Cl measured in rainfall of 4.0 mg/L the areal zone of contribution of effective rainfall for the 0.3 m depth plane receiving STE was estimated at 23.8 m^2. Using a biomat length of 11 m (as indicated by high Cl concentrations continually measured at the 0, 5 and 10 m sample locations along the trenches and corroborated by a CCTV investigation inside the percolation pipework), this equated to a zone of contribution of effective rainfall of an extra 0.15 m on both sides of each of the 0.45 m wide trenches at the 0.3 m depth plane. Similarly, calculations carried out for the 0.6 m and 1.0 m depth planes below each percolation trench showed that the effluent plume had dispersed to an estimated 0.4 m on the sides of the trenches at the 0.6 m depth plane and 0.53 m on the sides at the 1.0 m plane. These calculations were carried out on all sites to yield accurate lengths of the biomats along the trenches, as shown on Table 2. It is evident from the data that the reduced organic loading brought about by the additional secondary treatment of wastewater prior to discharge to the percolation areas had inhibited the formation of a biomat preventing distribution of the effluent along the entire base of the trenches. The installation of the secondary treatment systems greatly reduced the organic load on the percolation areas as expected with an average 77% lower COD loads being discharged to the trenches on Sites 4, 5 and 6 compared to the sites with septic tanks. The effluent only reached up to a maximum of 4 m along the length of the SE percolation trenches with a considerably shorter biomat development for the higher permeability subsoil sites.

Examination of the soil moisture tension values from the tensiometers installed at the different sample positions (front, middle and back) along the trenches were also used to corroborate the conceptual models for each site. As can be seen in Figure 3(a), the tensiometers at the three depth planes at the 17.5 m sample location where no effluent was recorded on Site 1 responded directly to variations in effective rainfall over the total sampling period. By comparison, the tensiometers at the 2.5 m sample location (Figure 3(b)) show some response during high rainfall events but in general the readings remained more stable across the trial period as the subsoil soil moisture conditions were influenced more by the percolating effluent than by the contribution of effective rainfall. Hence, under those percolation trenches receiving STE, it was physical, chemical and biological processes rather than dilution that were the more dominant attenuation processes operating in the subsoil as the effect of rainfall dilution on the percolating effluent in the subsoil was relatively small.

3.2. Nitrogen loadings and removal

The mean daily on-site wastewater flow rates on the six sites as measured continuously over the sampling periods by the instrumentation are shown on Table 2. The effective areal hydraulic loadings rates have then been calculated on the basis of the lateral effluent spread on the biomat spread and width of effluent spread at the 1.0 m depth plane. The average total-nitrogen loads of STE and SE discharged into the percolation trenches on each site over the research periods are also shown in Table 2. When these were compared to the respective loading rates after percolating through 1 m depth in the subsoil (taking into account the dilution by effective rainfall) significant differences were found between the sites receiving STE and SE, particularly when comparing the sites of moderate permeability (Sites 2 and 3 (STE) and Sites 5 and 6 (SE)). The nitrogen in the STE, discharging mainly in ammoniacal form (with approximately 20% organic-N), was seen to nitrify in the gravel and in the upper 0.3 m of subsoil and then denitrify in the sporadically saturated conditions below the biomat, often termed anoxic microsites (Parkin, 1987), where biological oxygen demand exceeds supply. In comparison, the secondary treatment systems on Sites 4, 5 and 6 were clearly discharging at least partially nitrified effluent but while the SE underwent further slight nitrification within the subsoil, the nitrate remained largely unchanged as it percolated down through the subsoil, leaving potentially higher total-N loads to the groundwater (see Tables 2 and 3). The equivalent reduction in total-N load between the SE and after 1.0 m of unsaturated subsoil on Sites 4, 5 and 6 was only 25%, 9% and 24% respectively compared to 59%, 76% and 89% on the STE sites. Fig. 4 compares this difference between the percolating STE on Site 2 to the SE on Site 5 where total-N loads of 16.7 g/d were found at the nominal point of discharge to groundwater on Site 5 compared to 6.8 gTotal-N/d at the same depth on Site 2. The inhibition to biomat formation along the percolation trench on the sites receiving SE would result in a reduction in microbial denitrification (promoted in the biomat by the reducing conditions). In addition, even if saturated conditions existed locally within the subsoils on Sites 4, 5 and 6, the organic load of the percolating effluent may not have been sufficient to support the facultative heterotrophs required for denitrification.

An interesting correlation can be seen in Table 2 between higher total-N loads per capita at 1 m depth below the base of trench and subsoil permeability (Table 1), for both STE and SE. This is particularly evident (see Fig. 5 and Table 3) on the highly permeable subsoil Sites 1 and 4. The SE again remained largely untouched in its nitrate form en route to the groundwater as expected, but the STE on Site 1 did not appear to be denitrify as thoroughly as on Sites 2 and 3 whilst nitrification was still occurring between the 0.3 and 1.0 m depth planes. The high permeability subsoil had muted the spread of the biomat even under a high organic loading promoting a relatively high hydraulic loading on that localized area but maintaining unsaturated aerobic conditions in the subsoil beneath unsuitable for denitrification of the effluent.

The average results from the sites show that the potential total-N load to groundwater beneath the percolation areas from the secondary treatment systems was approximately twice that compared to the sites receiving just septic tank effluent, or up to three times greater if the high permeability sites (which are rare in Ireland) are ignored. When the average total-N-removed is compared between the sites, the difference is more acute with the sites receiving STE shown to be removing almost seven times (25 g/d) the average total-N than the subsoil receiving SE (3.63 g/d). If these figures are normalised with respect to population (as all the sites had different full-time residences), it is clear from Fig. 6 that there is a distinct difference with respect to nitrogen removal in the subsoil when comparing the discharge of STE and SE into the subsoil as represented by influent COD load per trench.

It should also be noted that the base of the plastic modules containing the Bord na Mona peat filters on Sites 5 and 6 had been sealed in order to collect all the percolated effluent to one outlet pipe before evenly distributing the effluent between the percolation trenches. This resulted in slightly flooded conditions in the base of the filter, conditions which promoted 51% reduction in inorganic-N loading through the peat filter on Site 5 and 25% reduction on Site 6. Such anoxic conditions in this shallow flooded zone are not typical for other types of secondary treatment systems and therefore the total-N load would normally be higher for SE, as revealed on Site 4.

3.3. Phosphorus loadings and removal

The secondary treatment systems on Sites 4, 5 and 6 had little effect on the ortho-P loadings of the STE, acting to reduce their respective influent loads by an average 12%. These package treatment systems are not usually designed to remove P, apart from the reduction that occurs as a result of biological assimilation.

The attenuation of ortho-P within the subsoil is controlled by soil adsorption and mineral precipitation, reactions that are closely related to soil pH (Robertson, 2003, 2008). A reduction in effluent pH often occurs across secondary treatment systems as a result of nitrification, evidence of which can be seen in Table 4, which could reduce the potential for ortho-P fixation in the subsoil. The pH values of the percolate after passing through 1 m of subsoil however, show an increase on all sites (with the exception of Site 3) which is not what would be expected if the pH change was attributed purely to the processes of nitrification / denitrification; denitrification should recover just under half of the alkalinity lost by the process of nitrification and so a net decrease in pH might have been expected on Sites 1, 2 and 3. Sites 1 and 2 were both on limestone-derived subsoil which would account for the pH increase compared to Site 3 which is on metamorphic and granite-derived subsoil where the expected decrease in pH was indeed measured. As discussed previously, very little denitrification occurred on Sites 4, 5 and 6 and so the large pH rise on Site 5 was probably due to the influence of the soil and subsoil mineralogy on the percolate, as the subsoil was derived from predominantly limestone-rich glacial drift deposits (Conroy et al., 1970).

The results show that the majority of the phosphate removal on Sites 1, 2, 3 and 4 were achieved within the first 0.3 m of subsoil. X-ray diffraction analysis of the subsoil in Site 1 showed that it contained quartz, feldspars, chlorite and calcite with a small amount of swelling illite-smectite clay. This means that the subsoil below the percolation trench contained soil particles of a larger specific surface area which are more conducive to phosphorus adsorption, in addition to presence of calcite under the prevalent alkaline conditions. Particle size distribution (PSD) analysis of the subsoils on Sites 2 and 3 showed that they contained a higher clay content which may account for the greater percentage removal of ortho-P above the 0.3 m depth plane. The high phosphate removal in Site 4 is more surprising perhaps, as the mineralogy of the subsoil did not suggest significant potential for phosphate load removal: the subsoil is classified as sandy gravel at the 0.2 m depth plane with an overlying thickness of sandy silt up to ground level and X-ray diffraction analysis revealed no calcite or oxides of Al, Fe and Mn. The PSD analysis did reveal 11% clay at 0.2 m depth plane, which may be partially responsible for sorption of the phosphate ions and a reduction in clay content with subsoil depth, mirroring the reduction in phosphorus fixation with depth. This could however, also be due to the fact that most of the PO_4-P compounds that could be fixed had already been bound in shallower depths. Interestingly, the high hydraulic loading on this site resulting from the limited spread of the biomat (see Table 2) also did not appear to affect the phosphate removal.

The phosphate removal through the subsoil at Site 6 was relatively low through the first 0.6 m of subsoil, particularly as the influent load was comparatively small on that site. PSD analysis of the subsoil showed it to contain a relatively low clay content, despite its low permeability (which was attributed to the high density of the matrix) and X-ray diffraction analysis revealed neither calcite nor oxides of Al, Fe and Mn. The comparatively low pH of the percolating effluent on Site 6, along with the high hydraulic loading, may explain the less efficient phosphate removal with depth.

The results at Site 5 stand out from the other sites as significant ortho-P concentrations (average 6.9 mg/l) were picked up in the effluent after percolating through 1.0 m of subsoil. More ortho-P was removed in the 0.0 to 0.3 m depth interval than in the interval between the 0.3 to 0.6 m depth planes, which was attributed to the higher clay content in the upper layer. There was then a noticeable increase in ortho-P fixation between the 0.6 m and 1.0 m depth planes (Table 4). The results of X-ray diffraction analysis at this depth showed that while the subsoil contained calcite, it was devoid of Al, Fe and Mn oxides, hydrous oxides or dissolved ions and therefore, fixation would be confined to the high pH range – consistent with the alkaline conditions that did exist at Site 5. As at Site 6, this site also had a relatively high hydraulic loading due to the limited spread of the biomat and this may have contributed to the less effective phosphorus removal.

The potential for a reduction in phosphate removal over time - due to the available sites for PO_4-P adsorption sites becoming filled - was examined. The trend on Site 1 (Figure 7) shows that although there was a continuously high removal of PO_4-P between the STE and 0.3 m plane over the duration of the project, there is some suggestion of a slight reduction in phosphorus removal towards the end of the 32 month trial period at the 0.3 and 0.6 m depth planes (which would have been more highly loaded than the deeper 1.0 m plane). A similar examination on Site 4 over 32 months, however, showed no apparent reduction in removal performance even though the subsoil was more highly loaded and also did not appear to have a suitable mineralogy for phosphate removal. The same constant phosphate removal performance was also measured through the subsoil on all the other sites, although they were only monitored for 12 month periods.

3.4 Discussion

A significant difference was found between the nitrogen loadings in the subsoil and hence potential loadings to groundwater, when the percolation areas receiving septic tank and secondary treated effluent were compared. This reduction in nitrogen removal from percolating effluent which has undergone secondary treatment has also been discussed in other literature (Kristiansen, 1981; Beal et al., 2005; Jantrania and Gross, 2006). This research has shown that the average total inorganic nitrogen load after 1.0 m depth of unsaturated subsoil, was 3.9 g-N/d per capita for the sites installed with secondary treatment systems compared to 2.1 g-N/d per capita on the sites receiving septic tank effluent. These disparities become larger if the sites with fast percolating subsoil characteristics (which are relatively rare in Ireland) are discounted, yielding a comparison of 3.4 g-N/d per capita for the sites installed with secondary treatment systems compared to 1.1 g-N/d per capita on the sites receiving just septic tank effluent.

These loading figures can be compared to the E.U. Nitrates Directive 91/676/EEC (European Commission, 1990) to give an indication as to the maximum housing density that should be permitted. The Directive specifies that the annual limit for nitrogen application to ground is 170 kg-N/ha which equates to a stocking rate of 2 cows per hectare. This application is to the ground <u>surface</u> and the application limit is based on the assumption that the majority of the nitrogen is taken up by surface vegetation (crops), with little N leaching down to the groundwater. Indeed, a recent project in Ireland (Bartley, 2003) showed that approximately 10% of total-N applied at the surface from dairy cows made it to the groundwater. This is not the case, however, with on-site wastewater effluent which is introduced into the subsoil typically at depths of 0.6 to 1.0m below the ground surface and so is not intercepted by the vegetation. Hence, if a conservative estimate is taken that the application limit stated in the Directive allows for a maximum of 10% of the total-N applied at the surface to make it past the root zone to the groundwater, a simple calculation yields that a maximum density of one 4-person household every 0.33 ha (~0.82 acre) for secondary treatment on-site systems would be at the nitrogen loading limit. The equivalent calculated density for households using septic tank systems is one household every 0.18 ha (~0.44 acre). These calculations imply that the density of houses using properly installed septic tanks is unlikely to reach that which would promote concerns with respect to Nitrates Directive whereas, a continuity of single acre plots using secondary treatment systems is perhaps a more realistic scenario in certain parts of the country. Furthermore, if the annual nitrogen loading is calculated on the percolation area alone, it equates to 284 kg-N/ha.yr for the secondary treated systems, which exceeds the nitrate application guidelines, compared to 153 kg-N/ha.yr for a household discharging to a septic tank and percolation area. It should be noted that the influence of high subsoil permeability was starting to be noticed for the septic tank effluent at Site 1 whereby the effluent was moving rapidly through the vadose zone and neither totally nitrifying and / or denitrifying. Hence, if nitrates in groundwater are a problem in a particular area then an upper limit on subsoil permeability limit with respect to the acceptability for STE discharge may need to be defined. It should be noted, that in parallel to the nutrient analysis, the microbiological quality of the percolating effluent was also analysed during the course of this project as reported in detail elsewhere. This revealed excellent removal of both viral and bacterial indicators after 1 m of unsaturated subsoil across all sites (whether receiving STE or SE) with only isolated incidences found at low concentrations at this depth.

If septic tanks are not the desired system for on-site treatment in a particular area, other methods can be employed specifically to target nitrogen removal from the secondary treated effluent. The results described above indicate that dispersal of secondary treated effluent by gravity flow results in only a small part of the percolation area used. Therefore, alternative means of dispersal such as low pressure drip irrigation or shorter trench lengths should be used to spread the effluent over a wider area. Alternatively, nitrogen removal can be specifically targeted as part of the design of the package plants although this has extra operational and maintenance implications (for example, effluent recirculation to promote denitrification). Another concept that has been researched with some success is the addition of denitrification layers (comprised of waste cellulose solids) below the trench receiving secondary treated effluent (Robertson et al., 2000; Bedessem et al., 2005). Alternatively, the secondary effluent could be discharged into the root zone of plants as they will use some of the nutrient. This has obvious implications as to the use of the land, however.

The results from the study have shown that the phosphate is largely removed by the subsoils as a result of the mineralogy of the respective soils. Although, there was no strong evidence of the removal capacity decreasing after 32 months of operation, longer term studies on P plumes beneath on-site wastewater disposal systems have been carried out (Robertson 1998, 2008). Recently evidence of secondary slow or irreversible sorption of P has also been questioned in a long term monitored site, which implies that if this P attenuation mechanism is “missing” that the P in the groundwater could remain ultimately mobile (Robertson, 2008). Hence, any P that makes it through the subsoil into the groundwater may ultimately end up discharging into rivers and other surface water bodies. The Environmental Quality Standard (EQS) for phosphorus in rivers in Ireland is set at only 0.035 mg/l and although this study has shown relatively low potential P loadings beneath the on-site wastewater percolation areas, with the exception of Site 5 perhaps, it would not take much P in groundwater baseflow to cause the P in a river to exceed such a limit. The average P concentrations on Sites 1 to 6 at the 1.0 m depth plane were 0.8, 1.0, 0.1, 1.0, 6.9 and 0.6 mg/l respectively and so, depending the dilution of the effluent by groundwater (a function of aquifer characteristics and recharge in the area) and density of on-site systems, such low potential loadings could prove significant with respect to phosphorus levels in surface water bodies. Equally, an interim classification of groundwater bodies in Ireland, undertaken in response to the implementation of the EU Water Framework Directive, suggests that more groundwater bodies will fail to achieve "Good Status" on the grounds of phosphorus loadings than nitrogen loadings (EPA and River Basin Coordinating Authorities, 2008), in particular the vulnerable karst limestone aquifers in the west of the country.

Finally, it should be emphasised that these research results have been derived from carefully installed on-site systems, where great care has been taken to ensure an even distribution of effluent across each percolation area. Unfortunately, this does not appear to be the reality for most existing on-site wastewater treatment systems in use in Ireland and elsewhere today. Therefore, poorer performance of on-site systems can often be expected than for the systems reported here.

4. Conclusions (these have been made more succinct)

- The potential nitrogen loading per person to the groundwater beneath percolation areas receiving on-site secondary treated wastewater effluent was approximately two to three times that from the equivalent septic tank percolation areas.
- The reduced biomat formation along the percolation trenches receiving secondary treated effluent resulted in more concentrated hydraulic loading of the effluent, with a lower organic content that has limited denitrification.
- Higher resultant nitrogen loadings beneath the percolation areas were found on the more highly permeable sites for both secondary treated and septic tank effluent.
- An upper limit of one house every third of a hectare might be appropriate with respect to the density of on-site systems in an area to meet the requirements of the EU Nitrates legislation.
- Phosphorus removal was linked to subsoil mineralogy and appeared to be independent of the hydraulic loading rate. However, the relatively low potential P loadings beneath the on-site wastewater percolation areas may still be significant with respect to groundwater baseflow discharging into rivers and other surface water bodies.

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Table 1. Summary of site characteristics

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Effluent type	STE	STE	STE	SE	SE	SE
No. residents	6	4	4	3	5	4
T-value	3.7	15	33	4.5	29	52
Subsoil classification†	gravely SILT	sandy CLAY	gravely, clayey SAND	sandy GRAVEL	sandy CLAY to SILT	gravely, clayey SAND
Field sat. hyd. cond. - k_{fs}, m/d #	1.05	0.28	0.13	0.84	0.15	0.08
LTAR, L/m^2d §	40	20	10	40	15	10

† see BS5930 (British Standards Institution, 1999)
‡ Normalised permeability coefficient - see CEN, 2006
# Field saturated hydraulic conductivity - see Rodgers and Mulqueen, 2006
§ Long Term Acceptance Rate - see CEN, 2006

Table 2. Site characteristics and average total nitrogen loads across research periods

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean flow per capita (L/d)	119	105	82	90	60	123
Biomat length along trenches (m)	11	20	8	0.4	3	4
COD loading per trench (g/d)	72.0	40.2	108.4	12.8	14.1	24.5
Effective hyd. load[†] ($L/m^2.d$)	24.0	11.7	22.8	373.3	41.7	54.7
Influent total-N to subsoil (g/d)	60.4	28.9	19.6	20.2	18.4	17.8
Total-N after 1m depth of subsoil (g/d)	25.0	6.8	2.1	15.2	16.7	13.6
Total-N per capita after 1m depth of subsoil (g/d)	**4.17**	**1.70**	**0.53**	**5.07**	**3.34**	**3.40**

[†]per unit area of trench

Table 3 The average breakdown of Total Inorganic-N in STE (Site 1) and SE (Site 4) across the three depth planes.

	% of Total Inorganic-N					
	NH_4-N		**NO_2-N**		**NO_3-N**	
	Site 1	*Site 4*	*Site 1*	*Site 4*	*Site 1*	*Site 4*
STE / SE	98.3	28.0	0.3	19.2	1.4	52.8
0.3 m subsoil	27.2	1.3	4.3	6.0	68.5	92.9
0.6 m subsoil	13.4	1.1	1.7	3.8	84.9	94.9
1.0 m subsoil	4.3	1.1	0.3	2.6	95.4	96.1

Table 4. Site characteristics and average ortho-phosphate loads across research periods

	Site 1	**Site 2**	**Site 3**	**Site 4**	**Site 5**	**Site 6**
Effluent	STE	STE	STE	SE	SE	SE
pH of influent	7.4	7.7	7.7	7.2	6.4	6.4
pH at after 1 m depth	8.0	8.0	7.1	7.6	8.2	6.6
Influent ortho-P (g/d)	13.2	5.9	1.2	7.0	9.5	2.0
Ortho-P at 0.3m depth (g/d)	1.2	1.2	0.1	0.8	6.9	1.7
Ortho-P at 0.6m depth (g/d)	0.8	0.7	0.0	0.2	5.8	1.3
Ortho-P at 1m depth (g/d)	0.6	0.6	0.0	0.2	2.2	0.2
Ortho-P at 1m depth per capita (g/d)	**0.10**	**0.15**	**0.00**	**0.07**	**0.44**	**0.05**

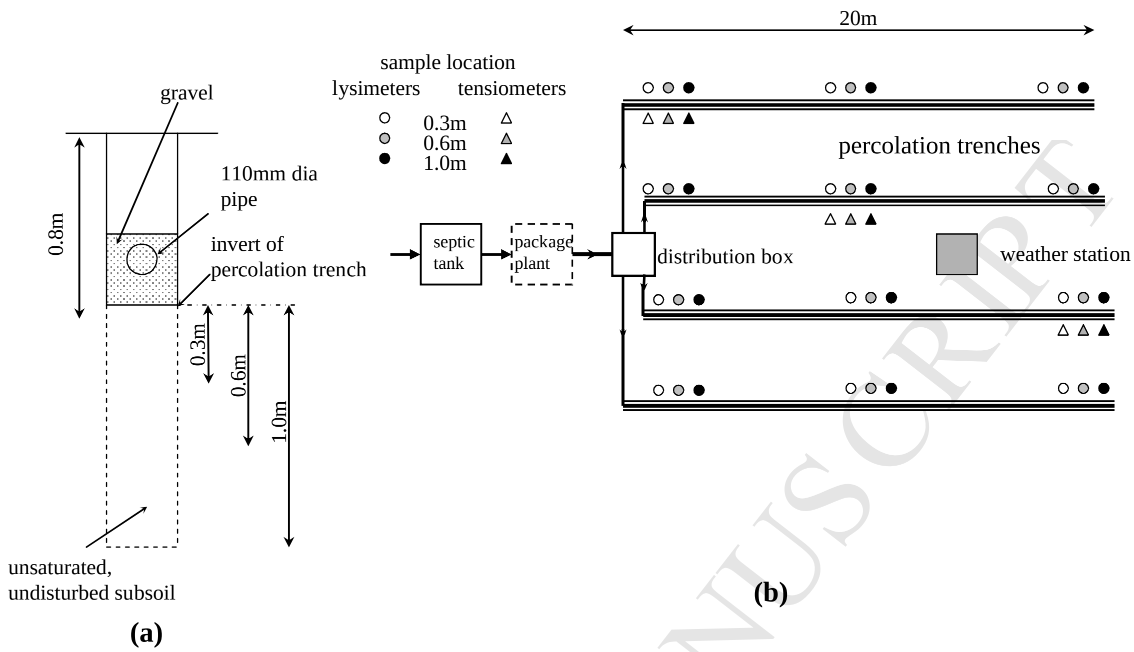


Fig. 1. Instrumentation layout: **(a)** Cross-section of trench and **(b)** Plan view of trenches.

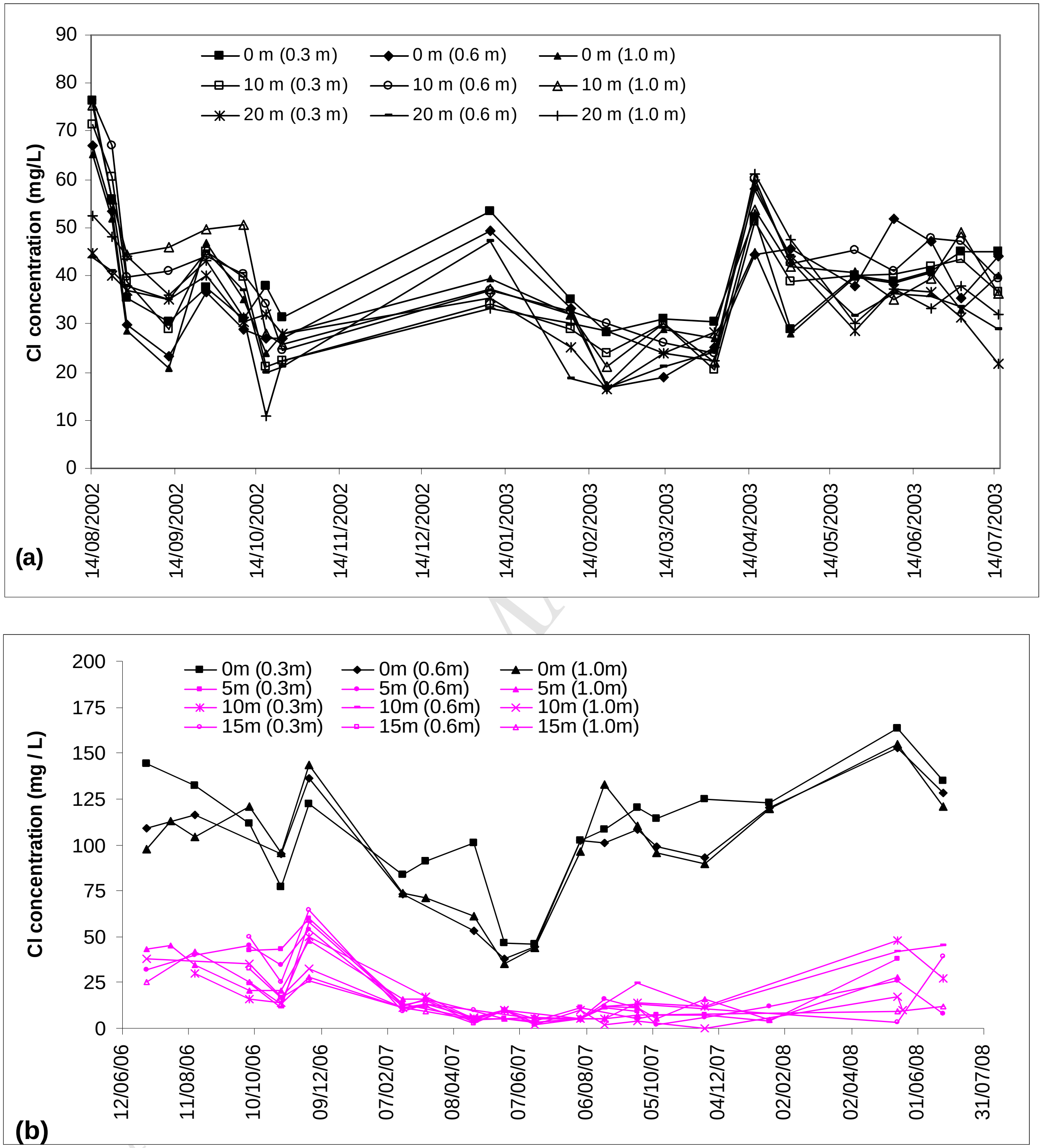
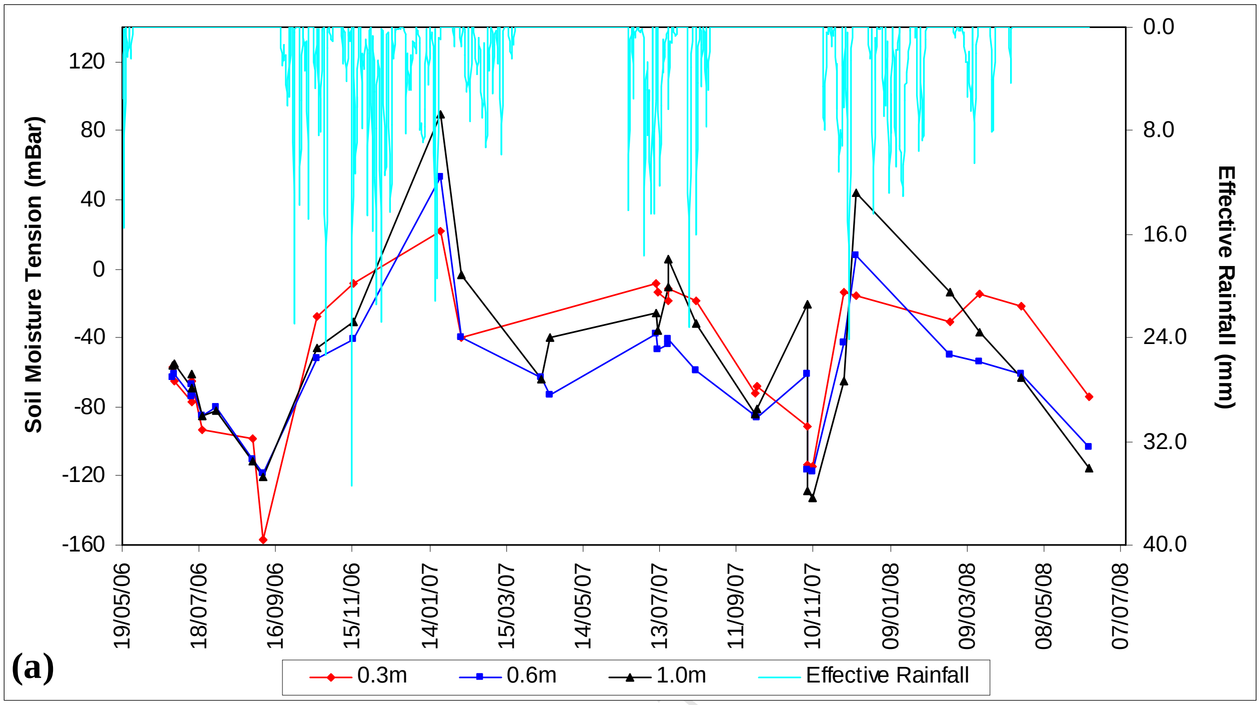


Fig. 2. Average Cl concentrations across all depth planes across the percolation area: **(a)** Site 2 and **(b)** Site 4.



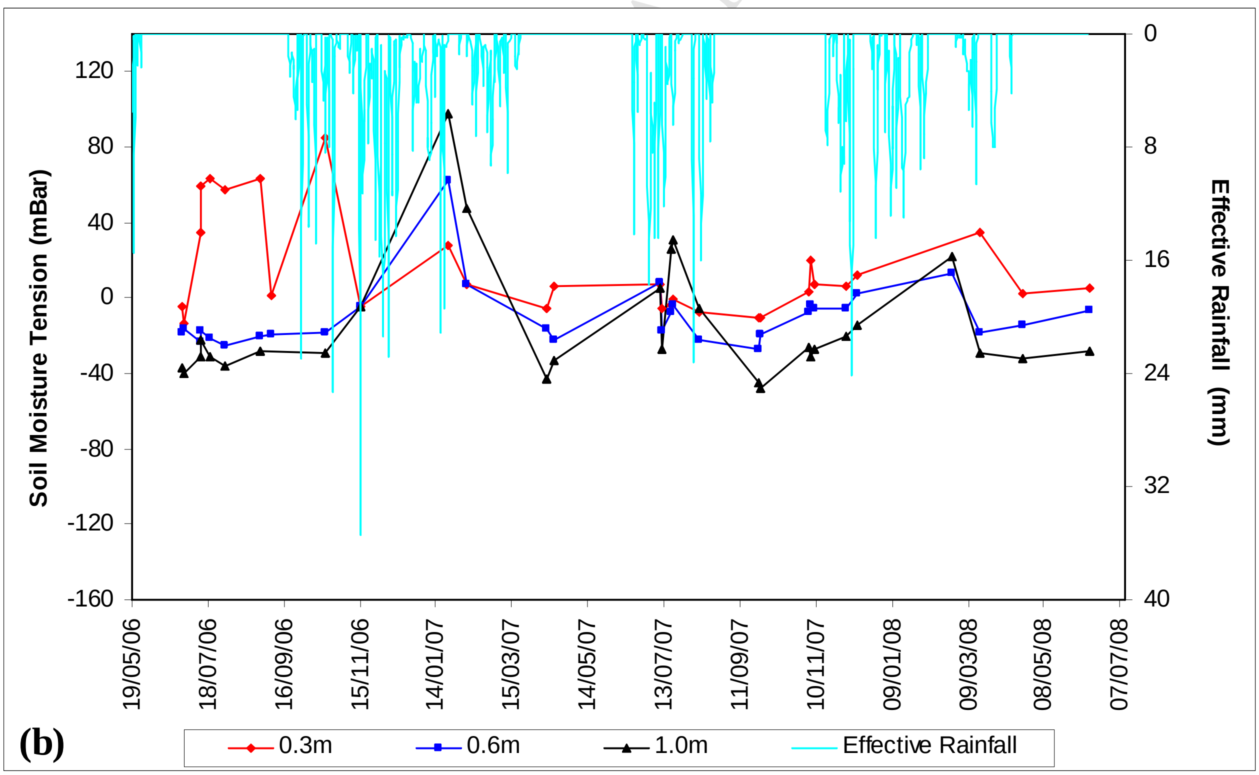


Fig. 3 Soil moisture tension plotted against effective rainfall on the percolation trenches receiving STE (Site 1) for: **(a)** the 17.5 m sample position and **(b)** the 2.5 m sample position.

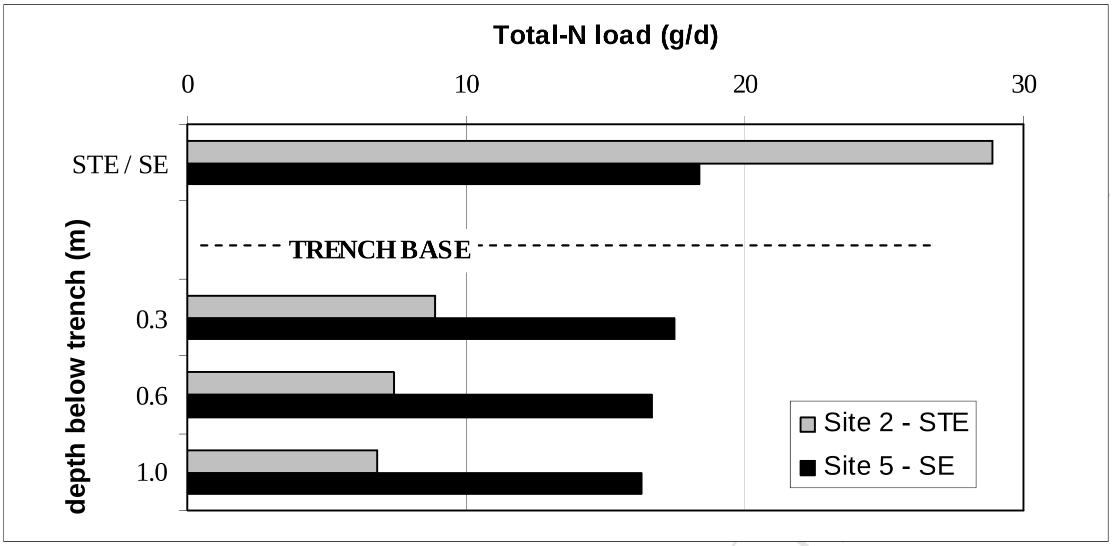


Fig. 4. Total N-loading with depth (Site 2 compared to Site 5).

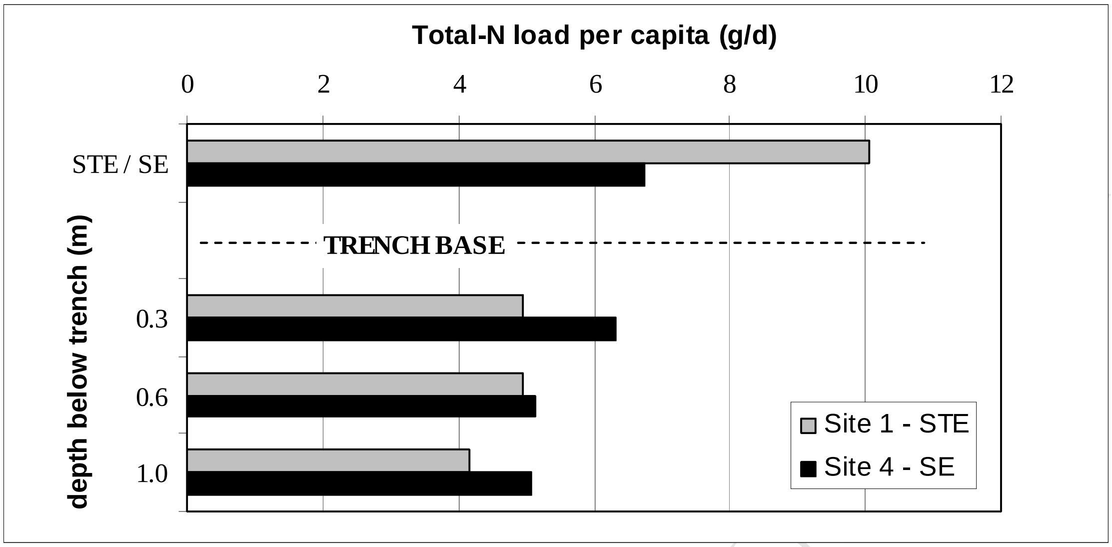


Fig. 5. Total N-loading per capita with depth through high permeability subsoil (Site 1 compared to Site 4).

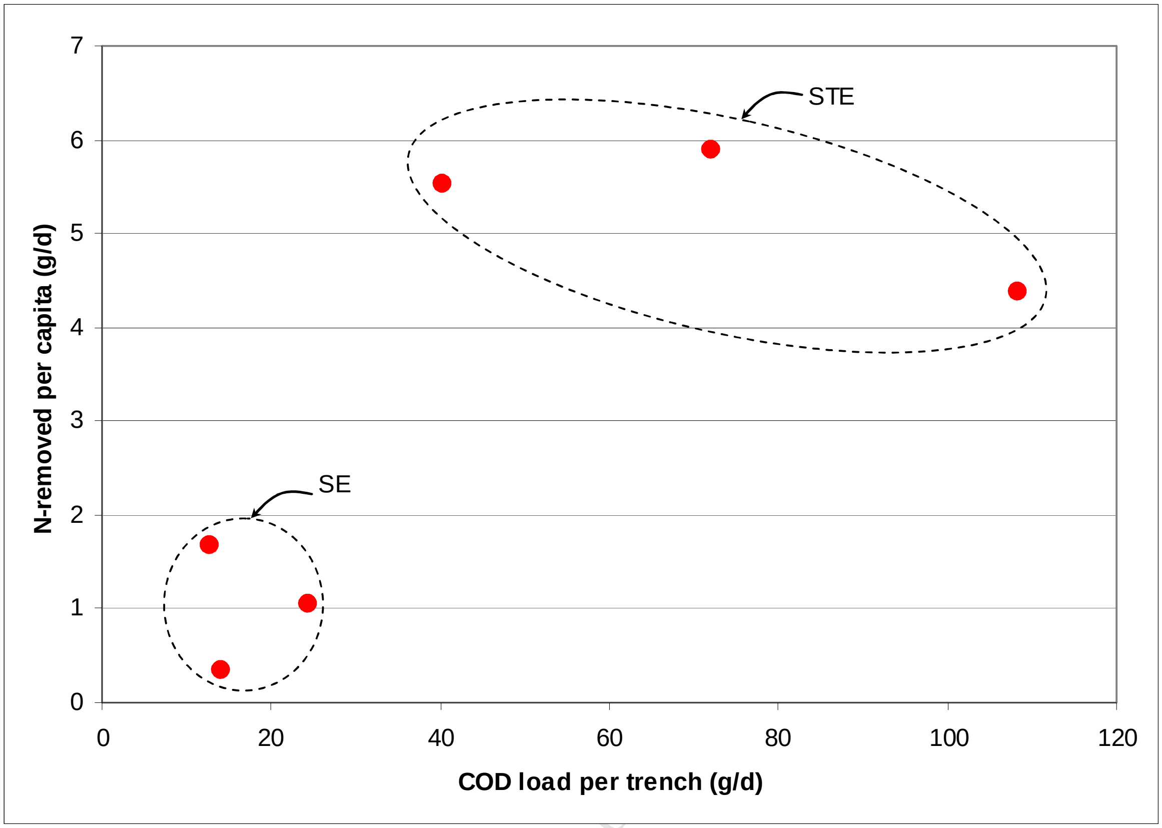


Fig. 6 Correlation between total-N removed per capita and influent COD loads per trench.

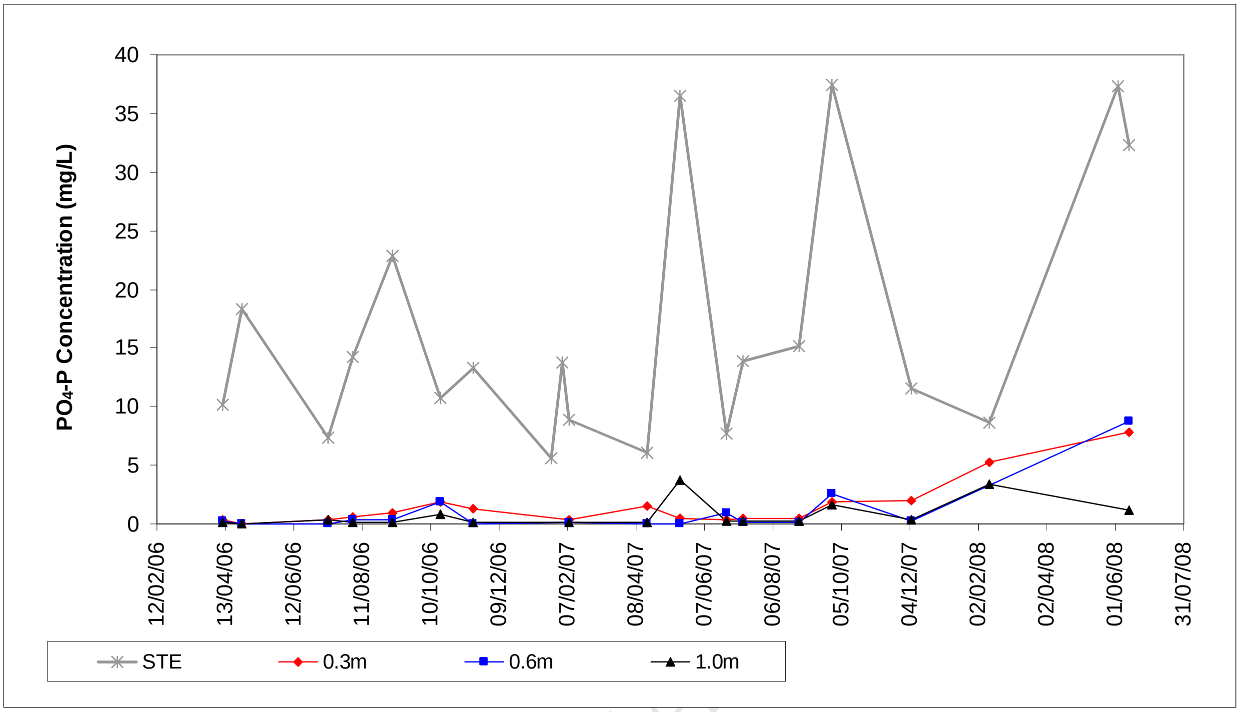


Fig. 7 Ortho-phosphate concentrations over time at Site 1.