Small headwater streams of the Auckland Region Volume 3: Nitrate and Phosphate Removal

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Small headwater streams of the Auckland Region Volume 3: Nitrate and Phosphate Removal

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1 Executive Summary

Small headwater streams can be highly vulnerable to modification from land use and management changes (e.g., urbanisation, cultivation, deforestation), and re-engineering (e.g., piping and damming). Currently, streams providing year-round habitat for fish, invertebrates or aquatic plants are given greater protection under the Proposed Auckland Regional Plan: Air, Land and Water[1] than streams that dry up for part of the year. The Auckland Regional Council (ARC) requires information on the value of small headwater streams in terms of their function and natural values, to aid development of management options.

The objective of this study was to quantify the nitrate and phosphorus processing capacity of a small headwater wetland under dry-stock grazing, a predominant land use in the Auckland Region. The wetland was located in Totara Park, a regional farm park administered by the Auckland Regional Council (ARC), and was 15m long, 5m wide at its widest point (area approximately 60m^2), and had organic matter depths between 50 and 90cm.

Two tracer experiments were conducted to assess the capacity of the wetland to remove added nitrate. The first, undertaken in May 2004, was a surface addition of nitrate solution, with lithium bromide added as a conservative tracer. The second study, undertaken in December 2004, was a subsurface addition, using a similar tracer solution although with the addition of phosphorus. Piezometers (shallow groundwater wells) were installed into the wetland to allow sampling of the groundwater.

In the subsurface addition, net removal of added nitrate was 66%, considerably more than for the surface addition in the first experiment, where attenuation at the wetland surface was neglibile and only nitrate that soaked into the wetland soils was treated. Attenuation of phosphate (as DRP) was surprisingly high, at about 97%.

The upwelling of groundwater in headwater wetlands have important influences on nutrient transformations within a wetland, by forcing nitrate-bearing water through zones of low free-oxygen and high organic matter, both necessary for denitrification to occur. These headwater wetlands are processing water that has seeped through to groundwater from the surrounding catchment. However, surface runoff that flows over the wetland is processed only if it has time to soak into the sediments.

Cattle hooves can punch through the organic anaerobic sediments, allowing a route for nutrient-laden subsurface water to bypass this processing and emerge at the surface where attenuation is negligible. Where headwater wetlands occur, best management practice would be to fence stock out and allow wetland vegetation to develop, to benefit from the processing capacity of headwater wetlands.

[1] These provisions are subject to a number of appeals and may change.

2 Introduction

Wetlands are landscape features where water, often groundwater, remains at the soil surface for sufficient periods that the vegetation naturally present must be specially adapted to having below-ground structures (roots and rhizomes) which can tolerate permanently, or semi-permanently, saturated conditions. Under these conditions, organic matter builds up in the soils, forming a thick layer of mulch, rich in carbon and low in dissolved oxygen.

In agricultural landscapes, fertilizer and urine from grazing stock contribute to nitrogen, predominantly in the form of nitrate, entering groundwater. This water may then flow towards surface waters such as streams and rivers. Nutrients such as nitrate and phosphate present in the water contributes to eutrophication of these waterbodies, and may lead to problems such as excessive weeds, algal slimes and other problems.

Wetlands often occur where groundwater arises and enters these water bodies. These wetlands may be significant because they have considerable potential to remove nitrate via microbial denitrification, that is, converting the nitrate (NO_3^-) to nitrogen gas (N_2). This process is illustrated below.

$2NO_3^- + 5H_2 + 2H^+ \rightarrow N_2 + 6H_2O$ -133.9 $\Delta G^{0'}$ (kcal/mole NO3-) (Tiedje et al., 1982)

Denitrification is an anoxic process, so it will only occur in locations where free oxygen (O_2) is absent, such as in wetlands. It can also be seen from the above formula that this process is energetically costly, and microbes can not undertake this process unless they have a ready source of energy. In wetlands, this energy source is present as carbon in the form of organic matter.

Phosphate removal is more problematic, as there is no effective gaseous removal mechanism. Attenuation within a wetland is generally restricted to filling exchangeable storage compartments such as plant matter or adsorption to soil particles. However, P flux out of groundwater is generally low unless soils have been fertilized with phosphatic fertiliser or large volumes of animal faecal wastes.

The aim of the present study was to examine the P-attenuation, denitrification capacity and the functioning of a small pasture wetland at the head of an Auckland stream with low flows during dry summer periods. The study contributes to information on the values of headwater streams so that land managers are able to make decisions on the best ways to protect or utilise them. It is a companion to McKergow et al. (2006); Small Headwater Streams of the Auckland Region Volume 2, and more information about the background to the study of headwater stream functions can be found in that volume.

3 Methods

3.1 Site Description

The research was conducted on the headwaters of the Puhinui Stream at Totara Park. Totara Park is a Manukau City Park and is grazed by drystock. Two hundred 1-2 year old heifers graze the pasture as set stock (15 per paddock). Super-phosphate is applied to the pasture each autumn at a rate of 400 kg/ha.

The HGA underlying the sites is Waitemata sandstone, the most extensive HGA in the region. The Waitemata sandstones are a series of rocks or formations that make up the Waitemata Group. These rocks underlie a large proportion of the region. The generally consist of alternating sandstones and mudstones, which are overlain by a thick weathering profile consisting of silty clays. The Waitemata Group rocks have low to moderate hydraulic conductivity, and hence have low inflows to streams.

The study wetland was the Swamp site described in McKergow et al. (2006). The wetland was 15m long, and 5m wide at its widest point (area approximately 60m^2). Organic matter depths ranged between 50 and 90cm down the centre line. The wetland stood at the head of one of the tributaries of a small stream with very low summer flows. The wetland and stream were in pasture grazed by beef cattle, and grazing of the wetland had had a significant influence on the vegetation present. Dominant species were water-tolerant pasture grasses (Yorkshire fog, *Holcus lanatus*), with only remnant native wetland species remaining, primarily the soft rush, *Juncus effusus* (Figure 1 and 2). As well as grazing out most of the native wetland vegetation, the stock had caused considerable treading damage to the wetland soils, with indications that cattle hooves were pushing through the entire depth of the organic soil to the non-organic subsoil beneath. An electric fence was installed around the lower half of the wetland at the beginning of the study to minimise the influence of cattle during the study. Outflow from the wetland entered a weir (v-notch exit) with continuous stage-monitoring to gauge flow (Figure 2).

Figure 1:

Wetland site showing adjacent grazed pastures, outlet weir (left of picture) and stage monitoring apparatus.

Figure 2:

Wetland from below outlet point, showing weir and stage monitoring equipment.

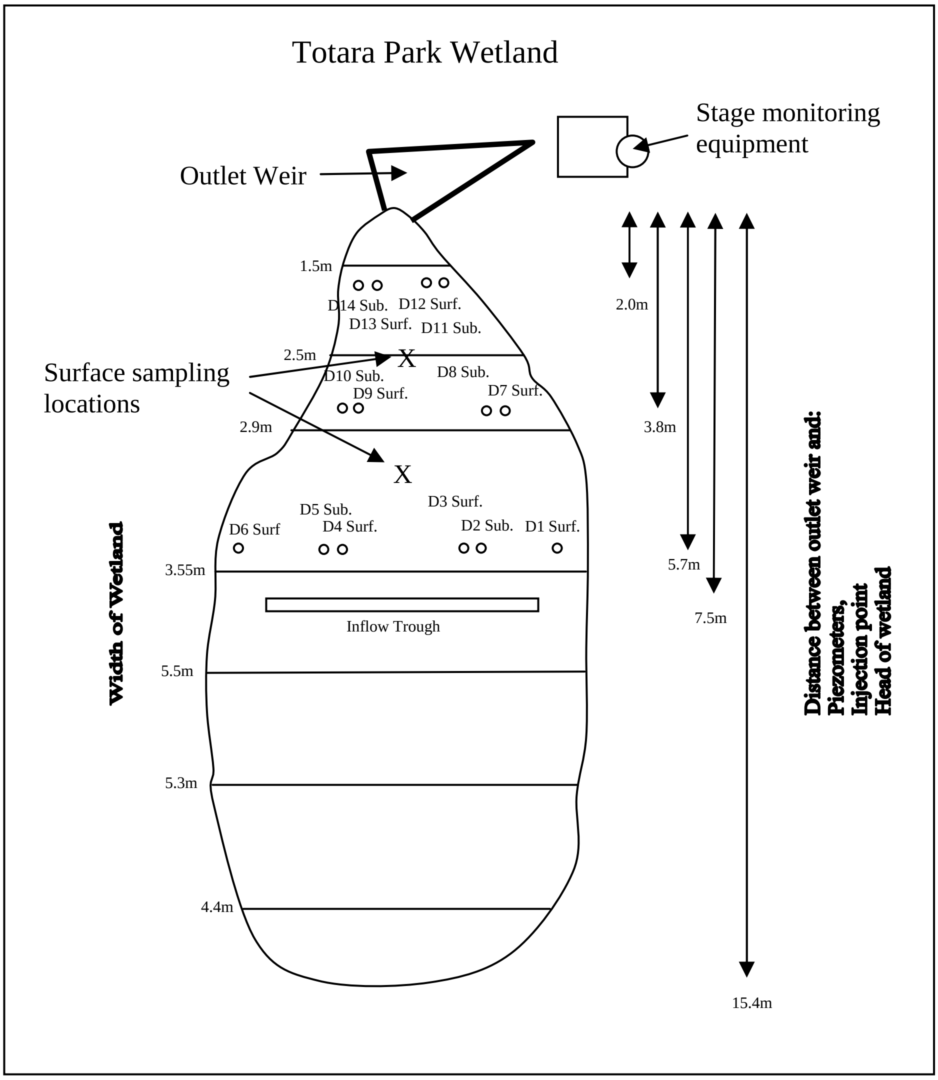
3.2 Experimental Procedures

Two studies were undertaken to assess the capacity of the wetland to remove added nitrate. The first, undertaken in May 2004, was a surface addition of nitrate solution, with lithium bromide added as a conservative tracer. The second study, undertaken in December 2004, was a subsurface addition, using a similar tracer solution (details of each study given below).

Piezometers (shallow groundwater wells) were installed into the wetland to allow sampling of the groundwater. Generally in pairs, they were inserted to two different depths to allow sampling of different layers of wetland water. The position of piezometers is shown in Figure 3. An input trough for adding tracer solution evenly across the surface of the wetland (Expt 1), was dug into the top layer of soil about half way down the wetland. In Experiment 2 this was replaced by 4 input piezometers installed to a depth of 20cm.

Figure 3:

Location of sampling piezometers in wetland (flow from bottom of page to top). The input trough for surface addition of tracer (Experiment 1) is located in the centre of the wetland. In the second experiment (subsurface addition) this was replaced by four input piezometers.



Piezometers were constructed using lengths of PVC plumbing pipe. These were joined to specially fabricated lengths of slotted pipe and sealed at the base with an end-cap. The slotted section was covered with a thin felted geotextile fabric to prevent soil particles entering the piezometer. Each piezometer was installed into the wetland soil by hand-augering a hole into the wetland soil while at the same time inserting a length of tube approximately 4 times the width of the piezometer (see Figure 3 for piezometer gear). When the required depth had been reached, the auger was removed, leaving the piezometer insertion tube in place. The base of the hole was sealed by adding drillers bentonite clay pellets (sodium modified bentonite, Rheogel® LX pellets, Omya New Zealand Ltd). Then some coarse driller's sand (2 mm silica sand, P. B. White Minerals PTY Ltd, Potts Point NSW) was dropped in the hole. The piezometer was inserted into the centre of the hole, and more sand was placed around it until the slotted section was covered. An additional layer of bentonite was added, and any depth of hole still remaining back-filled with the removed soil. The tube was then slid out of the hole, and an additional layer of bentonite was added at the surface to prevent preferential intrusion of surface water (Figure 4).

Figure 4:
Piezometer installation gear.

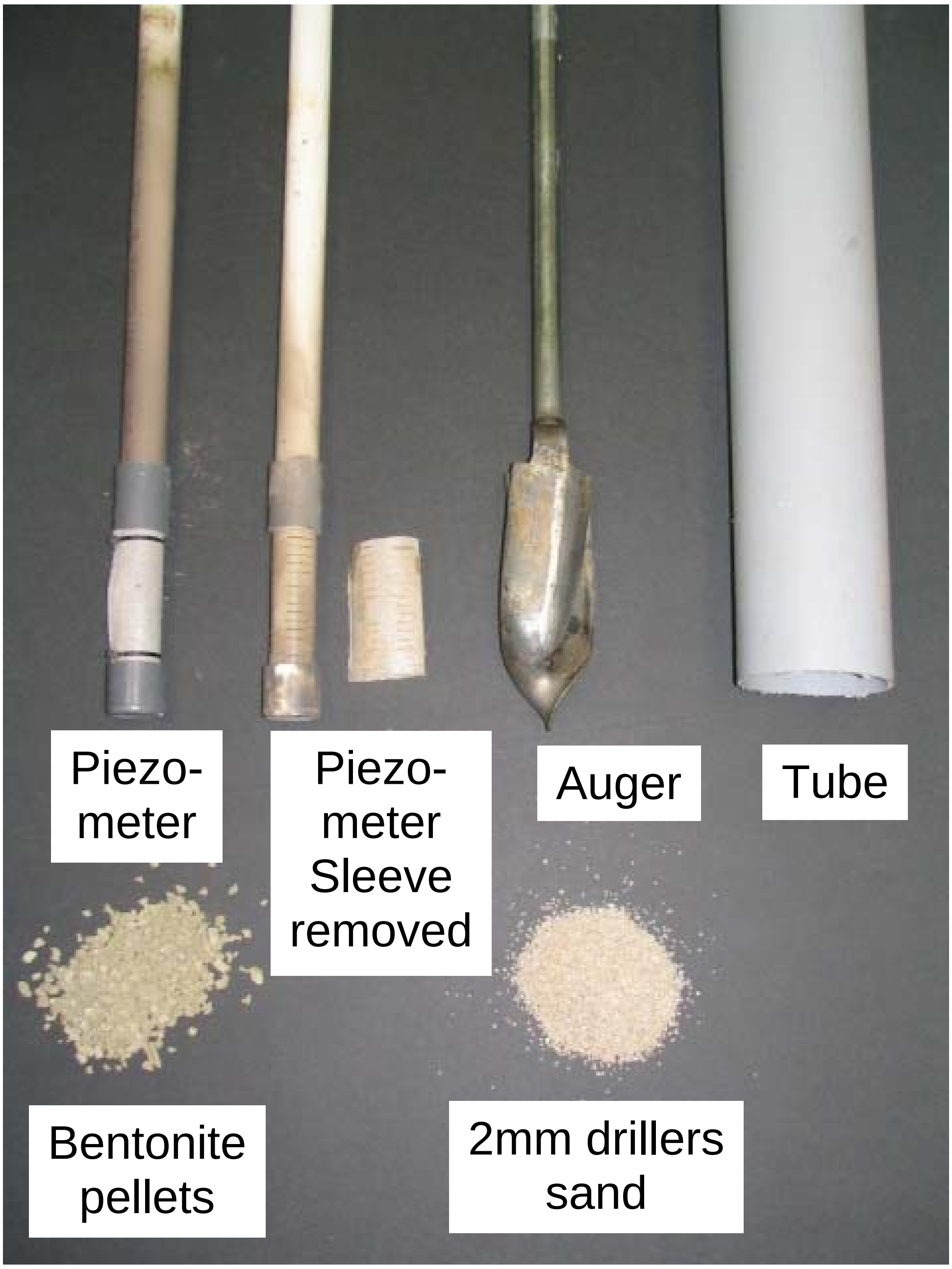
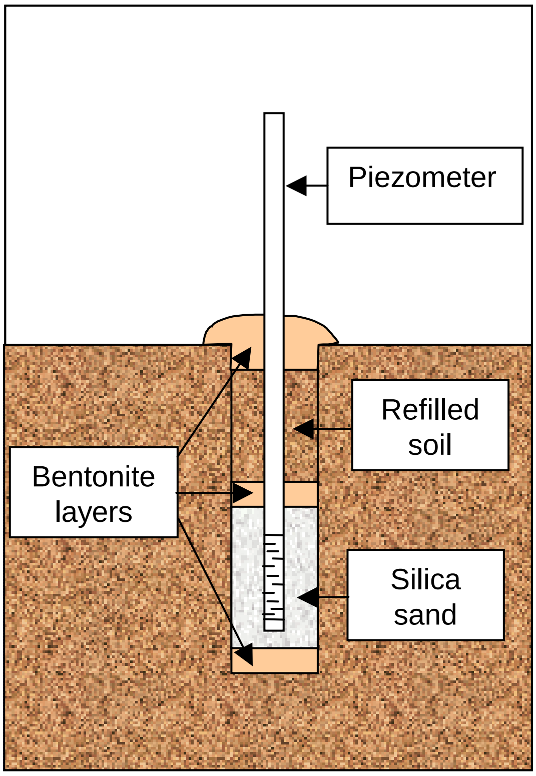


Figure 5:
Schematic of piezometer after installation.



3.3 Experiment 1. Surface addition of nitrate tracer solution

Experiment 1 was undertaken in May 2004. An input trough, which prevented tracer solution from settling directly into the wetland soil, was set into the wetland close to its widest, and extending almost over its full width (Fig. 3). A 30 L solution of 1000 ppm N as nitrate solution with lithium bromide added (final conc. 30 ppm) as a conservative tracer was dispensed into the wetland by allowing it to siphon into the trough, and tracer was progressively displaced over the leading edge of the trough by in-flowing water from further up the wetland. This allowed the tracer to flow over the full width of the wetland surface. The outflow rate from the wetland was 31 ml s^{-1} during this period, and the input of tracer took about 30 minutes. Rhodamine dye was also added to the trough to allow visual observation of the movement of the tracer solution. Samples were collected from two surface locations in the wetland at distances of 2.5m and 5.0m from the input trough, and from the outlet weir. Additionally, samples of wetland groundwater were sampled from the piezometers using syringes with aquarium tube attached. Conductivity electrodes connected to a logger were placed in several of the piezometers to assess diffusion of tracer into the sediment, and a multiparameter sonde (Hydrolab DataSonde 3, Austin, Texas) was placed in the outlet weir.

3.4 Experiment 2. Subsurface addition of nitrate tracer solution

In December 2004, a second nitrate tracer experiment was undertaken at the same site. To test the subsurface processing and allow adequate contact between the tracer solution and soil anoxic/anaerobic zones, the tracer was added to the subsurface of the wetland in the position previously occupied by the trough used in the first experiment, but using four "input" piezometers (20 cm depth) equidistant across the width of the wetland (Figure 6).

Figure 6:
Input piezometers. Tracer solution has Rhodamine dye added to allow visual observation of the progress of the tracer.

Ten litres of tracer solution (100 ppm N as nitrate, 100 ppm P as dissolved reactive phosphate, 660 ppm Br plus 20 ml Rhodamine dye for visual observation) was pumped into the input piezometers using a piston meter dosing pump (Model PM6013, Fluid Metering Inc, Syosset, NY) at 250 ml min^{-1} (Figure 7). Some of the tracer solution rose rapidly to the surface, as can be seen as a red colour around the base of the left-most piezometer in Figure 10. Flow to this piezometer was switched off. When there was any indication of upwelling from the other piezometers, the pump was turned off for a period to prevent the tracer solution from by-passing the soil. The pump was then turned on again until further signs of rapid by-pass were apparent. Only a very small amount of tracer was apparent as upwelling.

Samples were collected from the outlet weir and the piezometers by hand. Overnight sampling was undertaken using an automatic sampler from the outlet weir. A multi-parameter water quality sonde was placed in the outlet weir of the wetland to monitor environmental conditions (DO, pH, conductivity and redox). An additional sonde was placed in the weir of a nearby bush catchment (Figure 8) to allow comparison of general environmental water quality entering these streams for this and any future studies (catchments were of similar size, slope, soil and development, but contrasting riparian protection). Due to the extensive canopy in the "bush" catchment, there was no wetland development in this stream.

Figure 7:
Addition of tracer solution to the wetland.

Figure 8:

Bush weir and sonde.

4 Results and Discussion

4.1 Experiment 1. Surface addition of nitrate tracer solution

On the day of the tracer addition, the weather was fine. Tracer solution displaced from the input trough rapidly moved over the surface of the wetland, often finding preferential flow paths, and was visible at the outlet weir within 4 minutes of being added. Most of the tracer had passed through within 2-4 hours, although some rhodamine dye was visible for a further 2 hours. Persistent rain the following day was not thought to affect the tracer results, as most of the tracer had already exited the wetland.

Only selected samples were analysed from those collected. The ratio of nitrate-N to Br in each sample was compared with that in the original tracer solution to calculate nitrate removal (rather than simply dilution), and are shown in Figure 9. The tracer solution peaked in the outlet weir (distance 3) within 1 hour, and passed through almost entirely within 2 hours. No removal of nitrate was measurable from surface samples[2]. Some of the tracer solution however was able to enter the soil profile by diffusion or other processes. Again using the ratio of nitrate-N to Br in each sample compared with that in the original tracer solution, significant removal of the nitrate diffusing into the soil profile occurred over the sampling period (29 h) from both the shallow and the deep piezometers (Figure 10), averaging 78% and 80% respectively. Most piezometers showed removal over 90%, although 2–3 had removal of around 35–45%, thus lowering the average. This subsurface removal of nitrate was extremely rapid (Figure 10), and, when expressed as the amount remaining, followed a roughly log-linear response with time (Figure 11). The subsurface removal of nitrate observed in this experiment however, was small compared with the total volume of nitrate added, and appears not to have resulted in significant overall attenuation from the wetland, as this was negligible (Figure 9).

[2] Small variations in bromide results were well within normal analytical variability.

Figure 9:

Nitrate measured at 3 distances down wetland compared to amount added (calculated from bromide concentrations).

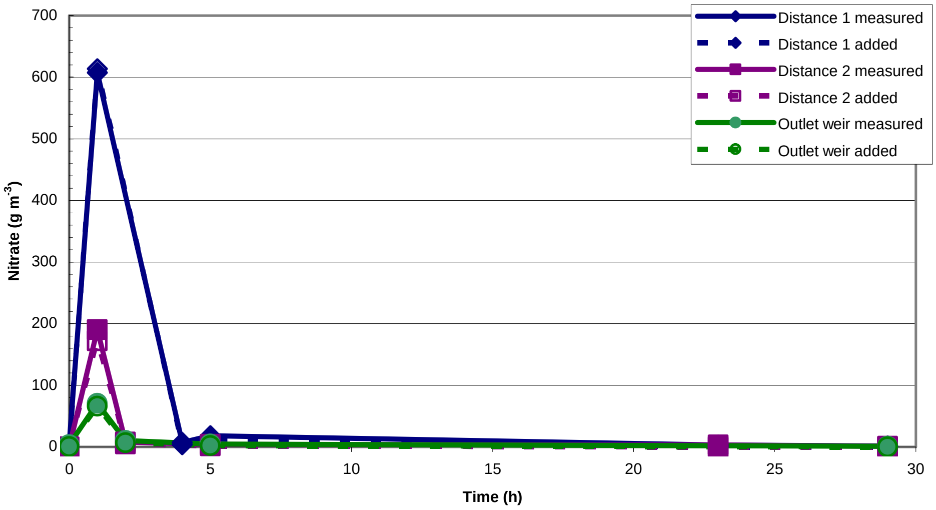


Figure 10:

Nitrate removal in a subsurface piezometer (Shallow piezometer, Upstream row).

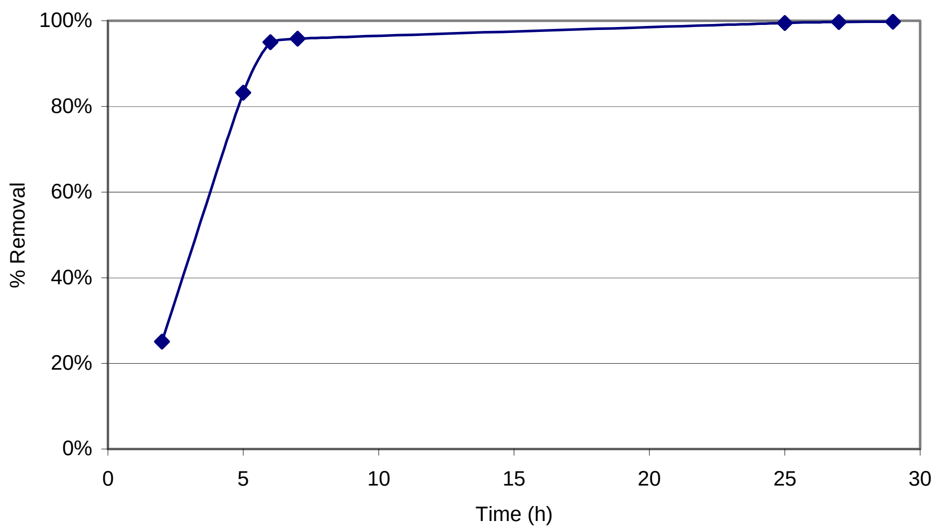
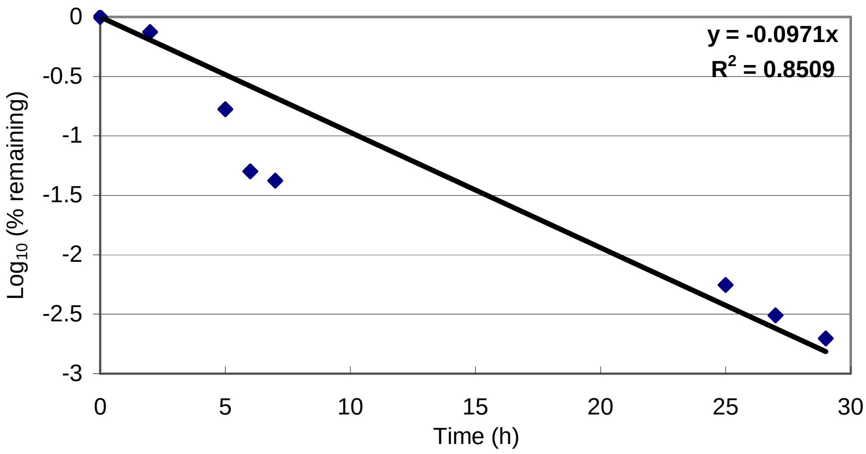


Figure 11:

Relative removal rate of nitrate based on % of nitrate remaining.



4.1.1 Discussion

Movement of water through wetland soils is sometimes extremely slow with theoretical retention times ranging from weeks to months. This enhances contact of nitrate-bearing groundwater with anoxic zones within the wetland, and allows sufficient time for denitrification to occur. Monitoring a wetland tracer experiment for this full period is both time-consuming and expensive. This initial experiment was designed with a surface addition about half way along the wetland in order to reduce the necessary monitoring period and thus costs. However, the flow paths of water through and over the wetland allowed the tracer solution to largely bypass significant contact with anoxic/anaerobic zones within the soil profile. Thus denitrification was restricted to the small amount which was able to diffuse into the soil. In addition, water flow at the surface was extremely rapid, with Rhodamine dye visible in the outlet weir within 4 minutes of addition. Preferential flow paths were obvious over the surface, and appeared to be increased by stock damage to the wetland, with water flowing from hoof-print to hoof-print.

The removal in the subsurface zones (collected from piezometers) illustrated important points in the functioning of this wetland. Firstly, a hill-slope wetland such as this one, is a point where groundwater rises to the surface, and once at the surface there is only limited exchange with the subsurface due to the rapid movement down the hill combined with the greater resistance of flow through the soil. Secondly, significant removal only occurs within the soil matrix (ignoring the plant uptake component which would be small in this wetland), although this is relatively rapid, and appears to conform to a first order rate reaction. Thirdly, stock access can alter wetland hydrology at a micro and macro scale, by grazing out natural wetland species and by altering vertical and horizontal flow paths with tread damage through the entire depth of organic wetland soil (Figure 12).

This first experiment acted as an example of how surface water is treated by wetlands, and how hoof damage to wetlands may allow subsurface water to well-up and flow over the top of the wetland and therefore bypass much of the processing that would have occurred had the water moved through the organic layers.

Figure 12:
Upwelling water in a cattle hoof print.

4.2 Experiment 2. Subsurface addition of nitrate tracer solution

Average flow out of the wetland was 25 ml s^{-1} during this study (a little lower than during the May tracer study). Temperature fluctuations (Fig. 13) were typical for this time of year. They were much higher in the pasture wetland than in the nearby bush site, due to the shading effects of trees, giving temperature stability in the bush. Unlike a pasture stream, which often displays extreme fluctuations in pH due to photosynthesis cycles of submerged aquatic plants absorbing available CO_2, thus affecting the carbonate equilibrium, the wetland showed pH stability similar to that in the bush (Fig. 13).

Figure 13:

pH and Temperature. A. Pasture Wetland and B. Bush site.

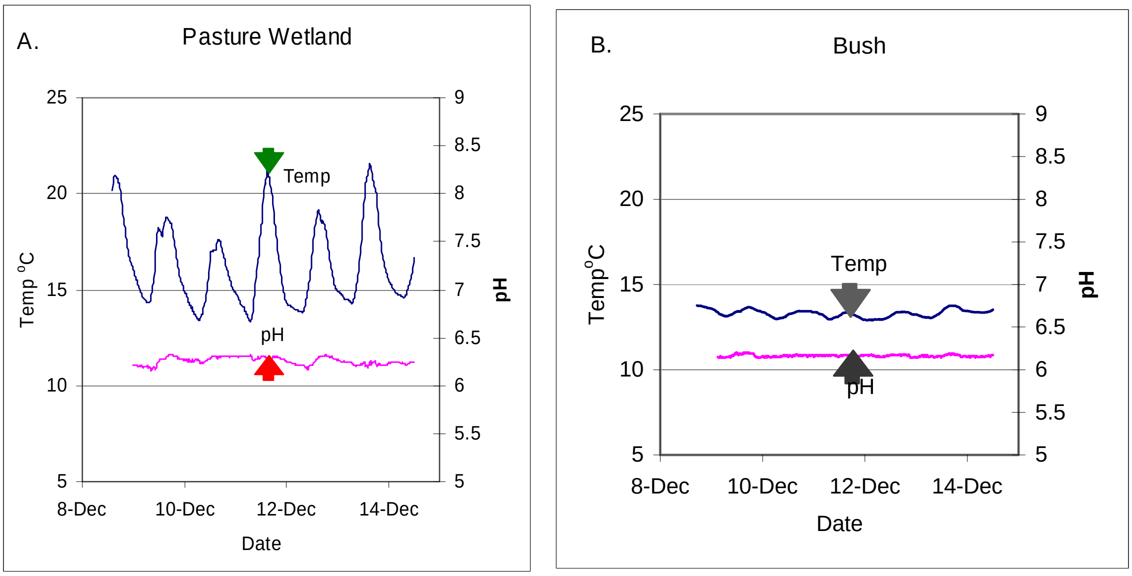
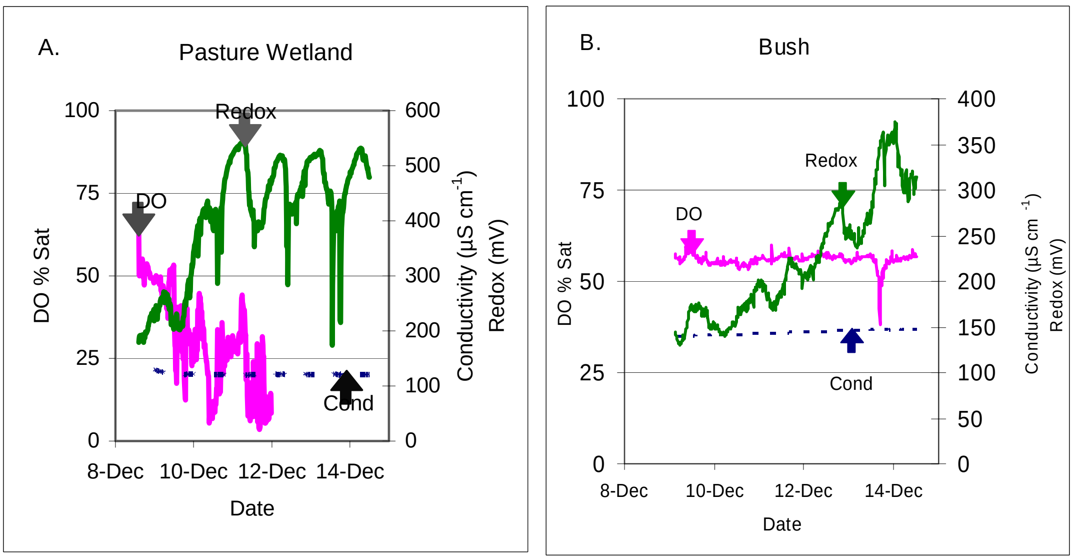


Figure 14:

Dissolved oxygen, conductivity and Redox. A. Pasture wetland. B. Bush site.



Dissolved oxygen (DO) exiting the wetland (Figure 14) was relatively stable compared with a typical pasture stream, but more variable than the Bush site. After the 12th Dec., the DO electrode became submersed in soil at the base of the weir, and data after this time have been excluded. Specific conductivity, a measure of the ability of a solution to carry an electrical current, and thus a measure of total dissolved ions, was stable at both sites throughout the period of the study. Oxidation/reduction potential (redox) mediates chemical reactions of a solution, and can be useful when oxygen saturations fall below zero. In this instance, oxyen saturations in both the wetland outflow and bush stream were not excessively low, except when the sonde in the wetland had slipped into the sediment.

Nitrate concentrations measured in the outflow and the amount (background plus added nitrate) predicted on the basis of bromide concentrations are shown in Figure 15. Net removal of added nitrate was 66%, considerably more than for the surface addition in the first experiment.

Attenuation of phosphate (as DRP) was also measured (Figure 16). Surprisingly high removal was evident, at about 97%.

Figure 15:

Nitrate removal from subsurface nitrate addition. (Samples taken from outlet weir).

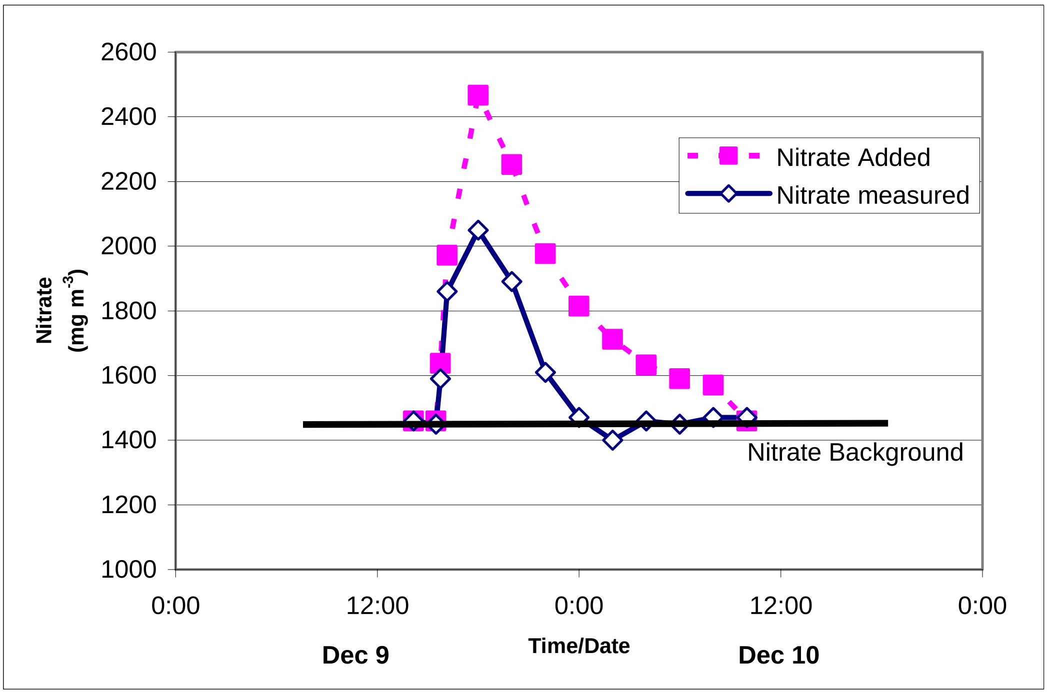
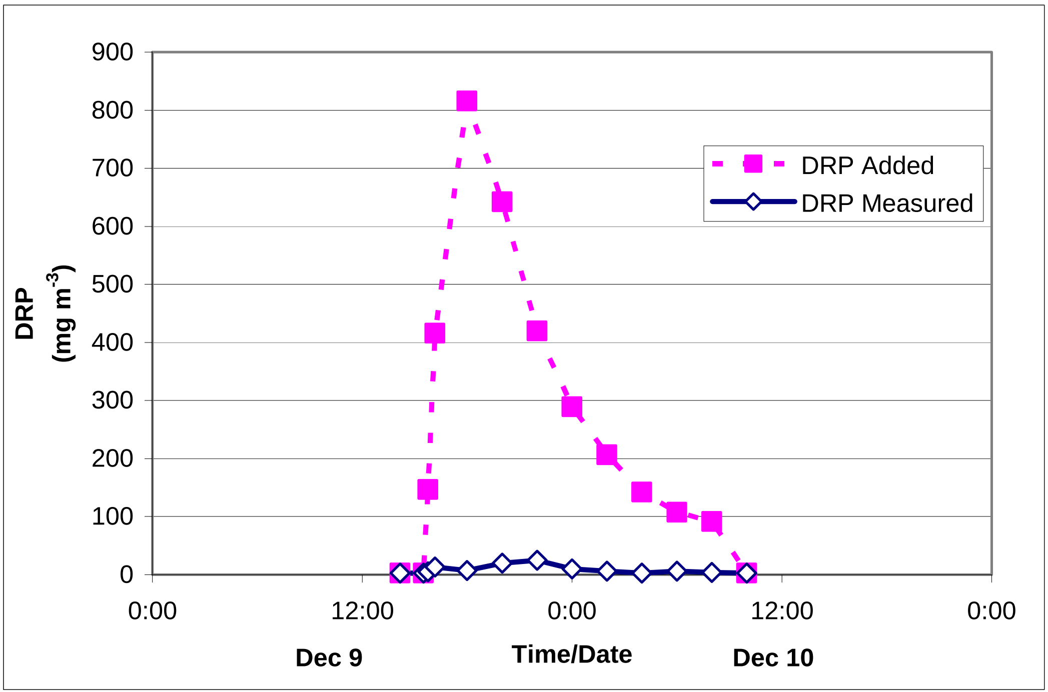


Figure 16:

DRP attenuation from subsurface addition. (Samples taken from outlet weir).



4.2.1 Discussion

The tracer solution added within the wetland subsurface soil layer was forced it to pass through anoxic/anaerobic zones within the wetland. This is much more natural than the previous surface addition experiment, and allowed denitrification processes naturally present within the wetland to operate. Considerably higher nitrate attenuation was recorded than in the first experiment (May 2004). Despite these experiments being undertaken in different seasons, it was clear that the primary reason for different rates of nitrate removal was due to the different methods of adding the tracer solution (surface addition in May, giving retention time of 2-4 hours compared with sub-surface addition in December which gave retention times of 18-20 hours). The nitrate removal rate of 66% recorded in this second trial falls within the normal range of removal recorded for similar small wetlands in pastoral areas (Burns and Nguyen, 2002; Rutherford and Nguyen, 2004; Sukias et al., 2004), despite the highly degraded nature of the wetland in question.

Attenuation of DRP was very high, at 97%. As the potential for gaseous removal of P is negligible, removal must comprise only attenuation within finite storage compartments such as within plant matter or adsorption to soil particles. Once these limited storage compartments are full, no further removal can occur, and phosphate release is not uncommon in such wetlands (Sukias et al., 2006). It is expected that the removal recorded here could not be sustained in the long term.

This experiment, and that performed earlier in May, demonstrated wetland nutrient transformation processes which were similar to that expected from other pasture-type wetlands. Outflow was not excessively low at these times, however some very low flows from this site have been recorded (7 ml s^{-1}, Pattinson pers com), and the edges of the wetland had become dry, like the adjacent pasture area. While wetlands, by definition, are almost continuously wet, the outflow during dry periods may be insufficient to sustain flow in downstream reaches. Thus the wetland itself may act as an important reservoir for repopulating stream invertebrates etc. when flows increase. In this regard, protection of these wetlands from stock access may be of greater significance than in some other environments. In addition, grazing pressure from stock as well as displacing the normal assemblage of wetland plants, may limit the build-up of organic matter in the wetland soil, which may take many decades or more to develop. This may have long-term consequences for the sustainability of small wetlands with stock access i.e. they may have limited or nil organic accretion (Author's observation: stock appear reluctant to graze deeper areas of larger wetlands, possibly due to fear of becoming "bogged").

Comparison with the bush clad site in an adjacent catchment provided some interesting perspectives, which may be of value to land managers in these catchments. Temperature fluctuations from the pasture wetland were extremely high, and were probably attributable to a variety of factors associated with high insolation, such as the high wetland surface area, thin layer of water passing over the surface, and lack of true wetland vegetation to shade the surface. The stability of pH was also of interest. The groundwater/wetland matrix would expect to show stable pH, however a stream in a pasture area would generally show extreme fluctuations associated with CO_2 uptake, causing swings in the carbonate equilibrium and associated pH changes.

Dissolved oxygen was relatively high in the bush stream, but not greatly dissimilar in the outflow from the wetland, which would be expected to have much lower DO due to: the high amounts of organic matter in wetlands which consumes DO; and the lack of true wetland vegetation which release oxygen into the soil profile from their roots. The small size of this wetland may have lessened the DO reduction which commonly occurs in wetlands of larger scale. As noted above, the gradual reduction in DO seen in Figure 14 was due to the DO electrode gradually becoming fouled by the wetland sediment.

One observation which may have significance when comparing streams with (bush site) and without riparian protection (e.g., our "pasture wetland site") was the lack of a true wetland (see definition, p.1) in the bush-clad sites (see Figure 17). It was hypothesised that the mature trees shading the stream would eventually displace any wetland plants present, and thus sustainable wetlands do not develop unless the slope of the catchment is such that soil conditions are too wet for tree roots over a widespread area, causing an opening in the overhead canopy, allowing enough light for low-growing wetland species to proliferate.

Figure 17:
Headwater area of stream in bush-clad site showing lack of “wetland” formation.

In this site, despite the extensive amount of leaf-litter, build up of a discrete organic layer, as found in wetland areas, was somewhat limited (see Figure 18).

Figure 18:
Wet riparian sediments with tree-roots exposed.

5 Summary

Nutrient attenuation, particularly nitrate, within wetlands depends on effective contact between nutrient-laden groundwater and anoxic/anaerobic zones within the wetland sediments. Nitrate attenuation in the study wetland was reasonable (66%) when nutrient-bearing water (our tracer solution) entered the organic sediments, but was negligible when applied at the surface where aerobic conditions prevailed and retention times were very short (4–10 minutes). Phosphate removal was also considerable (97%) when the tracer solution was applied via piezometers, although it should be stressed that, because P attenuation is restricted to uptake into exchangeable storage compartments, the observed rate of P removal is unlikely to be sustained in the medium-to-long term.

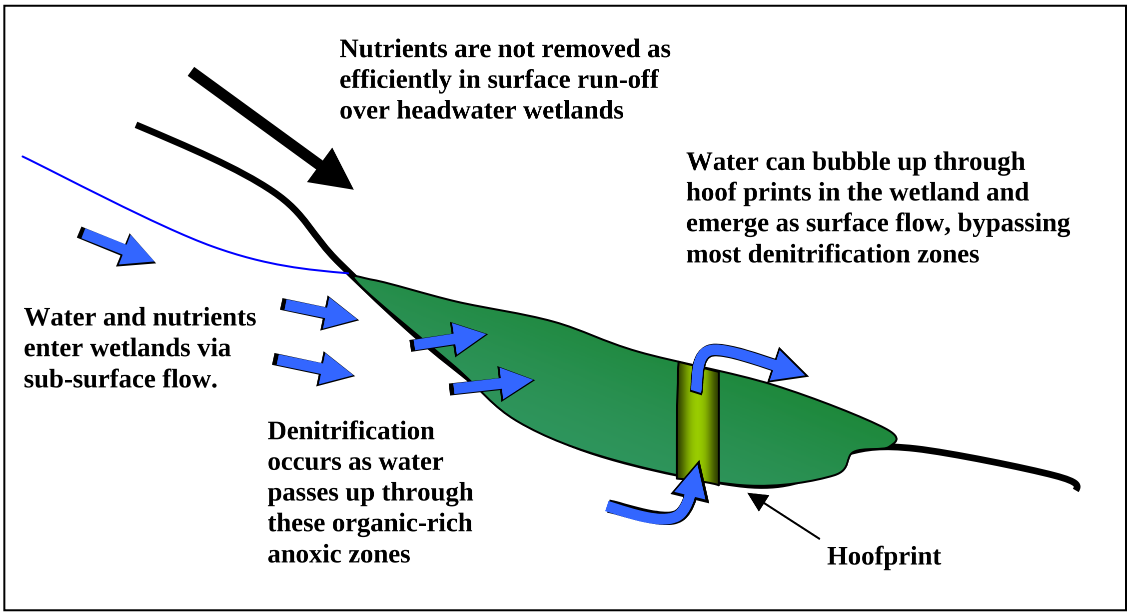
Effective nitrate removal requires thick organic sediments in a wetland. Build up of organic sediments may take place over periods of years or decades. Within wetlands in the headwaters of streams with temporary flows, where the flux of water into and out of the wetland may be very limited during dry periods, the build up of organic sediments is likely to take even longer. Unrestricted grazing of a wetland potentially reduces the effectiveness of the organic layers in removing nitrate. Firstly it allows disruption of hydraulic flow paths on a macro and micro scale, and secondly it may limit the continued replenishment and build-up of organic matter associated with the growth and senescence of plant matter within the wetland. This may be roughly analogous to the grazing of cattle beneath forest remnants, where the cattle may not kill the trees, but by preventing the growth of seedlings, ultimately cause the forest area to be destroyed.

Comparison of physico-chemical conditions of water exiting the wetland with the stream water in a nearby catchment showed both had similar and stable DO, pH and conductivity, but temperature exhibited extreme fluctuations from the wetland. This may have been exacerbated by the lack of true wetland vegetation which could shade the wetland surface.

The upwelling of groundwater in headwater wetlands observed during these two tracer experiments have important influences on nutrient transformations within a wetland, by forcing nitrate-bearing water through zones of low free-oxygen and high organic matter, both necessary for denitrification to occur. However, cattle hooves can punch through the organic anaerobic sediments, allowing a route for nutrient-laden subsurface water to bypass this processing and emerge at the surface where attenuation is negligible (Figure 19). These headwater wetlands are processing water that has seeped through to groundwater from the surrounding catchment. However surface runoff that flows over the wetland is processed only if it has time to soak into the sediments.

Figure 19:

Schematic representation of the water flow and nutrient processing in headwater wetlands.



6 Conclusions and Recommendations

This report is part of four volumes of research on the values of headwater streams and overall conclusions and recommendations are summarized below.

6.1 Implications for Management

6.1.1 Values of headwater streams

Collier (1993), in his review of the conservation of freshwater invertebrates, advocated a habitat- rather than species-based approach to conserving biodiversity. The protection of a range of rare, endangered, or representative habitats is most likely to ensure the protection of a wide range of invertebrate species, as well as maintain natural ecosystem processes.

Our research on the natural values of headwater streams has shown that there are significant biodiversity values associated with headwater habitats that dry up or contract in length for part of the year and are often not mapped as blue lines on topographic maps (Parkyn et al. 2006). For all land uses assessed, additional taxa occurred in the mud, pools, and flowing habitats that were not found in the perennial streams sampled. Therefore, protection of these habitats would enhance the overall biodiversity of stream communities.

However, our research also showed that despite the presence of additional taxa, the overall community composition and structure, and invertebrate metrics of ecosystem health were not significantly different between perennial stream habitats and the smaller headwater habitats. Mud samples were the most different from perennial samples as might be expected, but surprisingly, mud also contained communities of freshwater invertebrates. It seems likely that mud can act as a short-term refugium for some species, but other species may have adapted to exploit this habitat more permanently.

Based on the invertebrate species composition, there does not seem to be a rationale to separate Category 1 and 2 streams. However, it seems reasonable to suggest that stream reaches that are completely dry would have less value than streams with moisture, at a given point in time. In order to rank the differences between streams that all have a dry phase we would need to know the proportion of time that streams are wet and able to support aquatic life. Hydrological studies in one area of Auckland (Totara Park, Waitemata sandstones) indicated that 2 of the 4 streams ceased flowing for part of the year at the point where the weirs were placed (McKergow et al. 2006). In the smallest pasture catchment (0.7 ha) the stream stopped flowing for only 10 days in summer, while in a larger pastoral catchment (2.1 ha) stream flow dried up to occur only as storm flow between January and mid-April. Because of the influence of groundwater on these headwater areas, it is difficult to predict flows based on catchment areas. Currently we have estimates of the stream length per hectare that is intermittently flowing (or changing in length) from the Spatial Extent survey (Wilding & Parkyn 2006), but little understanding of how flow varies over time for these headwater systems.

The main differences in natural values occurred between land uses. Clearly, riparian vegetation improved the conditions of the streams towards that of native forest and allowed the existence of aquatic species associated with native forest streams. This suggests that riparian planting is a valuable method for managing headwater streams and it also shows that headwater streams with existing vegetation could be valuable sources of recolonists for stream restoration. Small, vegetated gullies are often pockets of refugia for native forest stream species within a pastoral catchment. Protecting these areas could be particularly valuable as source areas for restoration downstream and could mean that successful restoration is achieved after several years rather than several decades. For instance, if the headwater streams in a catchment were piped and filled (e.g., during urban development of a pastoral area), and only the perennial streams were restored with riparian planting, then it would take much longer for the recolonisation of stream communities to occur as there would be no upstream source of recolonists. Retaining headwater streams that already have riparian vegetation would improve the speed and success of the restoration process.

6.1.2 Recommendations for current management

Small headwater streams and wetlands are extensive in the Auckland region compared to the length of higher-order streams. Management of these areas is complex and decisions on the protection of these areas may ultimately depend upon socio-economic factors as well as ecological factors. An important question that remains unanswered is that of the cumulative effect of widespread loss or deterioration of headwater stream habitat. However, our research does provide information to help with management of rural and urban headwater streams.

Rural

There are several ways that headwater streams and wetlands could be managed under dry stock agriculture. One way is to fence all small waterways and plant them with native riparian plants, as was the case in the PR streams that we studied. This clearly has biodiversity benefits, particularly in summer, when even the remaining moist mud habitat was able to support EPT taxa. Communities of invertebrates in pastoral streams have changed from that of the original forested condition, but significant improvements in habitat and biodiversity of pastoral streams could be gained by fencing and planting riparian buffers. When there is adequate shade from riparian vegetation, water temperatures are lower and dissolved oxygen levels are higher during the summer months, creating healthier conditions for the invertebrate communities. Shade and cover from planted buffers also provides habitat for fish and koura (no koura were found in non-perennial pastoral headwater streams, but they were common in native forest).

The other important function of riparian buffers is for water quality in most stream systems. Fencing stock out of streams at Totara Park produced lower annual loads of *E.coli* than in streams open to stock (McKergow et al. 2006). However, headwater stream flow is greatly influenced by groundwater and subsurface flow. This means that the water can be carrying leached pollutants from the surrounding land use or historical land uses that have bypassed the riparian zone. Nitrogen loads in the riparian protected stream at Totara park were similar in the protected (Bush) and open (Swamp) sites.

Significant processing of nitrate and phosphorus (>90%) can occur in headwater wetlands under base flow conditions but this function can be reduced by stock access (Sukias & Nagels 2006). Hoof prints can create holes in wetlands that allow subsurface water to flow up and over the surface of the wetland where negligible denitrification occurs. Stock can also eat vegetation that would have naturally added to the organic build up in the wetland and therefore, stock reduce the processing capacity. Where headwater wetlands occur, best practice would be to fence stock out and allow wetland vegetation to develop. Planting with taller tree species is not recommended if the goal is to reduce nitrogen loads, as wetlands will revert to streams once shaded. Storm flows contribute significant amounts of pollutants and reduce the functioning of the wetlands. Efforts to extend protection or rough vegetation (e.g., encourage long grasses above wetlands by electric fencing in winter) may help to slow flood flows and give time for settling and infiltration of contaminants from the water flow.

The consequences of not managing these areas by removing stock are a continued export of sediments and faecal bacteria that will contribute to pollution and, in the case of sediment, accumulation downstream. With no riparian buffers on headwater streams, direct fertilizer additions and open access to stock, exports of nitrogen and phosphorus will remain high.

If fencing and/or planting headwater streams is not feasible then an alternative could be to construct wetlands at the base of catchments before the streams enter significant waterbodies (e.g., lakes, estuaries). However, this option would provide no biodiversity protection for the headwaters and may impede fish passage. Another alternative could be strategic protection of some of the headwater stream network. Existing tools for predicting fish assemblages could assist with this process, at least for fish biodiversity (John Leathwick, NIWA. pers. comm.). We recommend further research in order to make predictions about the placement, or the percentage, of streams that should be protected.

<u>Recommendations</u>

Based on research carried out over the last three years, we strongly recommend fencing stock out of headwater streams and wetlands for water quality improvements. For wetlands, fencing could take the form of hotwire fences that could be removed for stock grazing if the wetland dried up in summer.

For biodiversity goals in headwater streams, we recommend riparian protection with planted buffers of native trees. While shaded buffers reduce the nutrient processing capacity of headwaters, they provide multiple ecological benefits.

It may not be necessary to protect every headwater tributary to achieve improved biodiversity and water quality. We recommend further research into catchment-based approaches to assess the cumulative impacts of not managing all pastoral headwater streams and potential methods to select important or representative reaches.

Urban

When catchments are converted to urban land use there is potential for severe loss of stream function through piping and infilling (Rowe et al. 2006, Wilding 1996). Effectively all habitat values are lost and functions such as natural attenuation of contaminants, connectivity for species dispersal, food webs etc., are impaired. Urbanisation of catchments can also mean a loss of groundwater recharge from the increased impervious area. Therefore, it is likely that streams in urbanized catchments dry up for longer periods of time in summer and/or over a greater length.

Our research has shown that temporary headwater streams have similar aquatic invertebrate communities to those in perennial streams, but can also provide habitats that add additional species to the overall biodiversity of the catchment. The consequences of losing these streams will be loss of habitat values and a decline in overall biodiversity. Furthermore, urbanization that increases the duration of the dry period may decrease the biodiversity values of these headwater streams.

While intercepting nitrogen and phosphorus in urban streams may not be as necessary as it would be in pasture, it is worth noting that groundwater flow to these streams may still be carrying nutrients from historical land use, and simply piping them would transport these nutrients downstream without any instream attenuation. In addition, headwater streams may be just as important for the processing of stormwater contaminants as for rural contaminants, and incorporating natural stream functioning into urban design could make these streams important resources for treating urban runoff.

Recommendations

Our recommendation is that headwater streams be protected with riparian planting when catchments are converted to urban land use, for the sake of instream habitat, biodiversity, and ecosystem functioning – i.e., contaminant processing.

We recommend further research into the cumulative effects of the loss of headwater streams and better spatial modeling of the impact of urban development on catchment biodiversity and stream functioning.

6.2 Recommendations for future research

From the state of the science currently, we have concluded that intermittently flowing headwater streams do have values similar to that of perennial streams and their management should therefore be similar. However, we recognize that it may not be feasible for all headwater streams to be protected. Thus, there are a number of additional research areas that could allow us to differentiate between streams of higher and lower ecological value or provide a process for sustaining ecological and economic values.

Cumulative effects

Currently, the ARC has to deal with applications to alter headwater streams and wetlands on a piecemeal basis. There are no tools available to assess the cumulative effects of changing land use, or piping and damming streams. How many waterways can be lost (to infilling, piping or damming) in a catchment before this has impacts on catchment functions such as downstream water quality and quantity, or habitat provision? Conversely, is there a proportion or spatial arrangement of streams in a catchment that could be restored to enhance habitat and biodiversity, and improve water quality but still be affordable for the region?

This will be a difficult question to answer but one that is very important to consider. The first step would be to ascertain whether it is possible to assess the cumulative effects of stream loss and to consider the wide-ranging implications from species protection and habitat provision through to downstream effects on water quality and quantity and ecosystem functioning.

Variation through time

Can the length of time that headwater streams are wet be used to rank or value the headwater streams? At present, we have a widespread estimate of the amount of stream length that is intermittently flowing or changing in length (Wilding & Parkyn 2006), but no widespread estimates of the variability in flow through time of these headwater systems. Are headwater streams typically dry for a matter of days or a matter of months through the year, and how does this period differ between years? Do the streams typically dry out at the same time each year? Is this the best time of year to make a stream valuation?

These questions could be answered by incorporating monitoring of the weirs installed at Totara Park into a monitoring network and by investigating means to economically survey the temporal variation in hydrology of a wide range of headwater streams.

Urban headwaters

Traditional urban development creates large areas of impervious surfaces, which means a large proportion of rainfall can no longer infiltrate and extensive stormwater systems are required. This can have a profound impact on stream hydrology, resulting in a stream flow regime that is more flashy, has a higher risk of flooding in lowland areas. Water quality is also affected, as pollutants that accumulate on impervious surfaces enter streams more rapidly and effectively (Brydon et al. 2006). Headwater wetlands can provide water detention and water storage during rain events, and water release during dry periods. Headwater streams and swales could be managed to slow flood flows and trap contaminants to reduce downstream effects. Together with measures to reduce impervious area in urban catchments, headwater streams and wetlands could be managed as important resources to ameliorate the effects of stormwater run-off and they could also provide significant areas of natural and biodiversity values within an urban context. To further the management of headwater streams in urban areas, studies of the present values and functions of urban headwater streams are needed and, in particular, investigation of the effects of low–impact urban design on the values and functions of urban headwater streams.

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