Functions of riparian forest in urban catchments: a case study from sub-tropical Brisbane, Australia

Michael J. Newham[1], Christine S. Fellows and Fran Sheldon[2].

Australian Rivers Institute, Griffith University, Nathan, Queensland, Australia 4111

[1] Email addresses: m.newham@griffith.edu.au

[2] f.sheldon@griffith.edu.au

Keywords: urban stream, riparian forest, ecosystem function, algal biomass, denitrification, riparian revegetation.

Abstract :

Riparian forests are vital for maintaining healthy stream ecosystems; acting as buffers against nutrient and contaminant inputs, contributing energy subsidies and providing favorable instream habitat conditions. In urban catchments riparian forests are often degraded or cleared, removing the ecosystem functions the forest provides. Intact riparian forest along urban waterways, may mitigate some aspects of degradation associated with an urbanized catchment. In Bulimba Creek, an urbanized catchment in southeast Queensland, Australia, we investigated some ecosystem functions provided by riparian forest. We found that during baseflow periods a forested riparian corridor provided energy subsidies to the stream through litterfall and had a controlling influence on instream production through shading. Denitrification potential of benthic sediments increased with increasing levels of woody debris and organic matter, deposited from riparian vegetation. Denitrification was nitrate limited, indicating some potential to reduce nitrate loads in the stream. Riparian soils also showed moderate denitrification potential; which, through management strategies, could be utilized to reduce excess nitrate loads. These results suggest that riparian forests provide important functions for urban streams; highlighting the importance of conserving forest remnants in urban landscapes and the usefulness of replanting degraded riparian forest to enhance stream health and habitat condition.

Introduction

Riparian forests provide essential functions for healthy streams which regulate nutrient and contaminant inputs, maintain microclimate and low light levels, and provide energy subsidies for instream organisms and microbial processes (see Naiman and Decamps 1997 for review). The riparian zones of urban catchments are often degraded or even totally cleared of forest, often with engineered channels, removing the beneficial processes that these forests may provide. In addition, streams in urbanized catchments often suffer extensive degradation from contaminants and altered hydrology (Walsh et al. 2005). While not addressing issues of altered hydrology, the presence of riparian forest may mitigate some level of degradation in urban streams.

The capacity of riparian zones to reduce nutrient and contaminant loads to streams was established in the seminal paper by Peterjohn and Correll (1984). In the 25 years since, an ever growing body of literature supports, and further elaborates, on both the contaminant buffering capacity and nutrient removal processes which occur in riparian zones; of these special attention has been given to denitrification (Ambus and Lowrance 1991; Hill 1996; Groffman et al. 2002; Sabater et al. 2003). Denitrification is an anaerobic, microbial metabolic process that permanently removes nitrogen from stream systems by reducing oxidized forms of nitrogen to gaseous products (N_2O and N_2) which are released into the atmosphere. Riparian zones can provide conditions favorable for denitrification including moisture, soils high in organic carbon and through-flowing surface- and sub-surface water laden with nitrate. In catchments with high nitrate levels denitrification in riparian soils has been found to remove over 90% of nitrate content in sub-surface flow before it reaches the stream channel (see Hill 1996 for review). Instream denitrification can also act as a selfpurification mechanism, but mass rates are often less than that in soils and dependent upon biofilms and organic matter storage in the stream channel (McClain et al. 2003).

Riparian forests provide many functions for maintaining stream health and favorable habitat conditions (Gregory et al. 1991). The riparian forest canopy provides shade for stream channels and assists in reducing instream temperature fluctuations (Rutherford et al. 2004) and, by limiting light reaching the stream, controls instream primary production (Bunn et al. 1999; Mosisch et al. 2001). Riparian forests also provide allocthonous inputs of organic carbon to stream channels through wood and litter fall. Although this terrestrial carbon may not be a major contributor to stream foodwebs (Bunn et al. 1999), it is a basis for many microbial processes (Findlay and Sinsabaugh 1999) within stream ecosystems. When accumulated in large volumes (eg. debris dams), riparian litter (leaves, wood, bark) forms 'hot-spots' of microbial nutrient processing, particularly for denitrification (McClain et al. 2003).

Urban streams face particular degradation effects as a result of large impervious catchments, where rainfall is efficiently converted to runoff, along with the added impact of stormwater flows from these impervious areas being piped directly to stream (Paul and Meyer 2001; Walsh et al. 2005). Through the stormwater pipe network, larger stormflows and their associated contaminants are delivered directly to the stream channel, bypassing the riparian zone and its buffering capacity (Groffman et al. 2002). The efficient conversion of rainfall to runoff means even small rainfall events may generate high flows in urban streams (Walsh et al. 2005), resulting in highly incised and eroded stream channels with limited capacity to retain organic matter and form the debris dams that are important for instream nutrient processes (Groffman et al. 2005). Increased nutrient concentrations from greater catchment loads, combined with a reduced removal capacity, can lead to increased primary production, higher abundance of nuisance algae and dominance of macrophytes, particularly where riparian shading is reduced (Chessman et al. 1992; Murdock et al. 2004; Taylor et al. 2004; Meyer et al. 2005).

Previous studies have examined ecosystem processes, which may be influenced by riparian condition in urban catchments, and shown relationships with metrics of urbanization (Walsh et al. 2001, Meyer et al. 2005). However, these studies have not attempted to use "riparian condition" at the study site as a descriptive variable in their analysis of ecosystem processes (but see Roy et al. 2005). In urban streams, the general conclusion is that the presence of riparian vegetation alone is not enough to fully protect their biotic integrity, given the range of other disturbances associated with urbanization. While not disputing this, we suggest that the degree of degradation to stream processes caused by urbanization may be mitigated to some degree by the functions of the riparian forest. In this study, we attempt to quantify some instream processes and investigate the effect, if any, which the presence and condition of the surrounding riparian forest has on these processes. The processes investigated were nutrient retention (denitrification), instream production (algal biomass) and organic matter deposition (litter fall and presence of organic debris); while riparian condition was assessed as percentage riparian canopy cover and through the use of a rapid appraisal method.

Methods

Study area

Bulimba Creek is an urban stream located in the lower Brisbane River catchment, Southeast Queensland, Australia. The area has a sub-tropical climate and experiences summer dominated rainfall, with 55% of annual rainfall occurring between December and March when the monthly average can exceed 150 mm. Lowest rainfall occurs from June to September with an average rainfall of less than 70 mm per month. Therefore, streams experience an extended period of stream baseflow conditions during the cooler winter months. Over the winter period when sampling occurred the average temperature range is 11 - 22^{o}C.

The Bulimba Creek catchment covers 122 km^2, which is about 10% of the Brisbane City area. Much of the land around Bulimba Creek was initially cleared for rural development of pasture and crops 100 years ago but has been progressively converted to residential areas, which now house over 120 000 people. Today, only small pockets of active rural industry remain, although much of the eastern catchment retains a low impervious surface area with low density residential development.

Nine study sites were located along 2nd to 4th order reaches of the stream, with the most downstream site at the boundary of tidal influence (Figure 1). These sites were selected to be at least 1 km apart by stream distance and in reaches that do not experience total drying. All sampling was conducted during the winter baseflow period between May and August 2006, and occurred no less than one week following a rain event.

Stream characteristics

Total catchment imperviousness, riparian canopy cover and streamwater nutrient content were assessed as potential influences on stream ecosystem processes. Catchment imperviousness was estimated by methods based on those of Walsh et al. (2001) using satellite imagery of the catchment. Riparian canopy cover over the stream at each study site was measured by fish-eye lens hemispherical photography, using a Nikon D70 digital camera, and image analysis using WINPHOT software (Ter Steege 1994, Bunn et al. 1999). Physical habitat characteristics of the stream and riparian zone were described and classified using a modified version of the RARC (Rapid Appraisal of Riparian Condition: Jansen et al. 2004) assessment systems (Costelloe, 2005). The RARC (Jansen et al. 2004) collect data on indicators of functional features of the riparian zone, including habitat (width and continuity), cover, debris (ground and dead trees), percentage native vegetation and regeneration features. The modified version used here (Costelloe 2005) takes into account reduced riparian width within the scoring system, which we thought important given that Bulimba Creek was an urban catchment with restricted, non-natural, riparian widths. The RARC score varies between 0 and 1, with lower values indicative of poor riparian condition, and higher values better riparian condition. Conductivity (TPS Model WP-84), dissolved oxygen and temperature (TPS Model WP-82Y) measurements were made at each site to characterize water quality.

A water sample (50mL) was collected from each site for dissolved nitrogen and carbon concentrations, samples were filtered in the field using 0.45 μm syringe filters and frozen until analyzed. Analysis for nitrate plus nitrite (hereafter referred to collectively as nitrate) and ammonium was performed on a LACHAT Instruments Flow Injection Analyser (Lachat Instruments, Loveland, Colorado) using QuikChem

Method 31-107-04-1-D and a modification of QuikChem Method 10-107-06-4 which excluded the dialysis procedure. Total nitrogen (TN) was determined by a persulfate digestion followed by nitrate analysis as described above (Hosomi and Sudo 1986). Dissolved organic carbon (DOC) analysis was conducted using a Shimadzu TOC analyser (TOC-VCSH, Shimadzu Corporation, Kyoto, Japan).

Denitrification potential

Denitrification potential of shallow riparian soil and instream sediment was analyzed by collecting eight samples each of riparian soil and instream sediment from each site. Sample collection and analysis occurred between 3 and 9 May 2006. Riparian soil was collected to 30cm using a hand auger within 3 m of the stream bank. Instream sediments were collected to a depth of 3 cm in areas of sand or silt, which were common substrate types at all sites. Soil and sediment samples were homogenized prior to removing subsamples for analysis. Sample preparation began within 6 hours after the first sample was collected.

Denitrification potential was measured using the acetylene block method (Balderston et al. 1976). Two treatments were used, a control with ambient nutrient conditions and a nitrate-spiked sample with an additional 5 mg N L^{-1}. The moisture content of the soil and sediment samples was visually estimated to determine the amount of water required to produce a sediment slurry with an approximate 1:1 ratio of dry sediment to water. The actual moisture content was determined by drying a small amount of sediment at 70^{o}C overnight, which was then used in subsequent calculations. 'Dummy' slurries were prepared as for analyzed samples, but were sacrificed to get a measure of initial nitrate concentration in the prepared slurries. Treated sediment slurries were incubated in 250 mL sample bottles, with anoxic conditions achieved by flushing headspace with N_2 gas for 2 minutes, and with 10% of the headspace replaced with acetylene gas. A gas chromatograph with an electron capture detector (Agilent 6890 micro ECD; methane/argon carrier gas; HP Gas Plot Pro column) was used to measure nitrous oxide accumulation. A Bunsen coefficient of 0.6 at 295°K was used to estimate the amount of nitrous oxide dissolved in the slurry water. The total moles of nitrous oxide per bottle was plotted against time. The production rate of N_2O was calculated using linear regression and a ratio of 2:1 was used to convert N_2O-N produced to NO_3-N consumed in units of mg N kg dry soil^{-1} day^{-1}.

Benthic algal biomass

The chlorophyll-a content of benthic sediments was assessed at each study site as an indicator of the benthic algal biomass. Samples were collected in conjunction with sediment sampling for denitrification analysis. Three benthic sediment core samples were collected from random positions at each study site to a depth of 2 cm using a cut-off syringe (i.d. 29 mm). Sediment cores were stored in plastic vials and frozen until analysis. Chlorophyll-a was extracted from the sediments using 90% acetone for 24 hours at 4 °C. Samples were acidified with 2N HCl to correct for phaeophytin. Calculations of chlorophyll-a concentrations followed that of Lorenzen (1967).

Contribution of organic matter from riparian vegetation

Litter fall rate in the riparian zone immediately adjacent to the creek was used as a proxy for direct measurement over the creek. Different methods were used to measure litter fall at sites with forested riparian vegetation and dead biomass accumulation at sites dominated by grass and groundcovers. Methods used for litter collection were designed to remain inconspicuous in the urban setting to avoid unwanted attention and vandalism. This sampling was conducted in June and July of 2006.

The rate of litter fall in forested riparian zones was determined using collection quadrats. Each site had four quadrats of 0.36 m^2. However, site three had very different riparian vegetation on each bank, so three quadrats were installed on either side of the creek and were analyzed separately as sites 3a and 3b. The standing stock of litter within each quadrat was collected and returned to the lab for drying and weighing when the quadrats were installed and again five weeks later to calculate a rate of litter fall. Following each collection, leaf litter was dried at 70°C for at least 48 hours, then cooled to room temperature in a dessicator. The dry weight was recorded either as standing stock weight in the first case, or as a rate in the second case and expressed in units of g m^{-2} day^{-1}.

At sites with tall stands of grass (sites 7 and 9) a biomass accumulation method similar to that of Dugas et al. (1999) and Frank and Dugas (2001) was used. In four quadrats (each 0.09m^2) per site, grass was cut and collected along with leaf litter that had accumulated within the standing grass. In the laboratory, the leaf litter and dead grass was separated from the samples and the living grass was removed and discarded. Processing of samples followed that described above for leaf litter. Five weeks later, a second collection was undertaken using the same procedure in new quadrats located less than 2 m from those used for the first collection. The initial mean weight was used as the standing stock weight and the difference between this and the second mean weight was recorded as the rate of dead biomass accumulation (g m^{-2} day^{-1}).

Riparian litterfall and accumulated biomass were assessed for carbon and nitrogen content using a continuous flow-isotope ratio mass spectrometer (Micromass Isoprime Euro Vector EA 300, Manchester, UK) at Griffith University. From each collection quadrat a third sample of leaf litter, excluding large sticks, was collected following the litterfall investigation. At grassy sites, samples of dead grass were collected from the second quadrats which were used for rate calculation. This was returned to the lab and dried for 48 hours at 60^{o}C. The samples of litter were ground into a homogenous powder and analyzed for carbon and nitrogen content. The carbon and nitrogen content was calculated using the %C and %N contents by mass and multiplying by the total mass of litter as rate or standing stock. C:N ratios were calculated using the atomic weights of C and N.

Data analysis

Linear regression was used to investigate the relationships between ecosystem process measures and local- and catchment-scale environmental factors. Environmental variables were selected *a priori* as those expected to influence the processes. For benthic algal biomass the variables selected were canopy cover, instream nitrate concentration and instream habitat complexity. Nitrate concentrations, organic matter content of sediments and vegetation cover were chosen for denitrification potential. Instream denitrification potential was also analyzed for relationships with ‘habitat complexity’ and ‘debris’ scores (between 0 and 1) from RARC. Litterfall rates and standing stock were regressed with variables of vegetation and canopy cover. All processes were investigated with respect to catchment imperviousness and catchment area. A p value < 0.05 was considered significant for the regression analysis. Differences in denitrification potentials across the nine sites were assessed using

ANOVA performed in SAS version 9.0 (SAS Institute Inc, Cary, NC). The two-factor design used sample location (instream sediment or riparian soil) and treatment (nitrate spiked or control) as the independent variables. Mean values of denitrification potential for each location and treatment grouping within each of the nine sites were analyzed providing a total of 36 observations with four observations from each site. In the case of a significant ANOVA result, Tukey's studentized range (HSD) test was used to determine where significant differences lay.

Results

Stream and Catchment Characteristics

Canopy cover at the sites ranged from 26.7 to 79.3 % with all but two sites having greater than 50 % cover (Table 1). Although this is an urbanized catchment, the creek flowed through an almost continuous corridor of green space, which varied in width from 50 to over 200 m. On-ground observation showed that riparian forest was present at most sites and made up much of the area of the green space corridor. The riparian health indicator (RARC Score, Table 1) showed only moderate levels of riparian health at most sites, despite even high levels of riparian cover. This was due to low scores for channel stability and abundance of native vegetation.

The range of land use activities within the Bulimba Creek catchment was reflected in the pattern of catchment imperviousness (Fig. 1, Table 1). At the time of study the catchment was heavily urbanized on the western side with imperviousness of up to 80 % in some areas. The eastern catchment tended to have much less impervious surface area with less intensive development. Catchment imperviousness increased dramatically in the first 3 sites from 25 to 57 % as the drainage area first covered mainly forested headwaters, then heavily urbanized sections of the catchment. From site 4 onwards, the catchment imperviousness remained between 45 and 50 %.

Denitrification potential

Denitrification potentials differed significantly between sediment type ($F_{1,32} = 7.55$; $p < 0.01$) and nitrate treatment ($F_{1,32} = 12.78$; $p < 0.01$) while the interaction was not significant ($F_{1,32} = 0.12$; $p = 0.73$). The Tukey *post-hoc* groupings for these significant factors showed that denitrification potential of riparian soil was greater than that of instream sediments and that samples receiving added nitrate were greater than control samples (Fig. 2). Control instream sediments had the lowest denitrification potentials, with all values less than 2 mg N kg^{-1} day^{-1}. The addition of nitrate to instream sediments increased the denitrification potentials by as much as 8 mg N kg^{-1} day^{-1}, although in most cases the rate increased by 2-3 mg N kg^{-1} day^{-1}. Addition of nitrate to soil slurries also increased denitrification rates, though in most cases to a lesser extent than for stream sediments. Across both instream sediments and riparian soils, except for two cases, denitrification potential was nitrate limited.

A significant relationship was found in the control treated instream sediments between initial nitrate concentration and denitrification potential ($R^2 = 0.65$, $p < 0.01$), though the degree of variance between the replicates makes conclusions tenuous. This relationship would be expected due to the low nitrate concentrations limiting the possible rate of denitrification. Similar relationships were not found with sediments treated with additional nitrate nor in the riparian soils. However, denitrification potential of riparian soil was related to its organic matter content for both control ($R^2 = 0.81$, $p < 0.001$) and nitrate ($R^2 = 0.87$, $p < 0.001$) treatments (Fig. 3).

Regression analysis revealed that only the control treated instream sediments showed significant relationships with measures of stream condition. Instream denitrification potential had significant relationships with 'habitat complexity' ($p < 0.05$) and 'debris' ($p < 0.01$), which both indicate the degree of instream habitat heterogeneity and storage of organic matter. The 'debris' score explained 71% of the variation in instream denitrification, while 'habitat complexity' explained 53% of the variation.

Benthic algal biomass

There were no significant relationships between benthic chlorophyll-a concentration and measured environmental variables, but a trend of increasing chlorophyll-a concentration with decreased canopy cover was suggested (Fig. 4). There are two apparent outliers in the data at approximately 50 % and 80 % canopy cover (Fig. 4). These two sites had channel morphologies vastly different to other sites. The low chlorophyll-a concentration at 50 % canopy cover could be attributed to a channel depth of over 80 cm, limiting benthic light penetration. The site at 80 % canopy cover had a water depth of approximately 10 cm and channel width of 11 m, providing good light conditions for algal growth despite the high canopy cover. Other sites tended to have water depth ranges from 10-30 cm and width of 2-3 m. When the two outlying sites are removed from the regression of benthic chlorophyll-a with canopy cover the result becomes significant and explains 90 % of the variation ($R^2 = 0.90$, $p < 0.05$). Benthic chlorophyll-a concentration was not significantly correlated with streamwater nitrate concentration (which ranged from 0.34 – 0.97 mg N L^{-1}), measures of stream condition (habitat complexity and debris) or the degree of catchment imperviousness.

Contribution of organic matter from riparian vegetation

Standing stock of leaf litter and rates of litter fall in the riparian zone tended to be similar between sites, although some large variations due to vegetation management activities were observed. Most sites had standing stocks of litter between 300 and 500 g m^{-2} of litter with litter fall rates of 1 to 5 g m^{-2} day^{-1} (Fig. 5). Site 3a is a notable exception where regular mowing and maintenance of lawn in the riparian zone limited the accumulation of litter. Sites 7 and 9, where the grass collection method was employed, had litter accumulation of greater than 10 g m^{-2} day^{-1}. Standing stocks, however, were comparable to those at other sites. Assuming a constant standing stock of litter, this indicates a much faster turnover rate of litter in grassed riparian zones.

Leaf litter samples across the nine sites had a mean carbon content of 48.3% by weight with a range from 44.7 to 51.4%. Mean nitrogen content was 1.1% by weight with a range from 0.9 to 1.4%. Combined with standing stock values, this revealed that the riparian zone stores about 100 to 200 g of carbon and 2 to 6 g of nitrogen per square meter as leaf litter on the soil surface (Table 2). Accumulation rate of carbon varied between 0.5 and 7 g m^{-2} day^{-1} and nitrogen accumulated at a rate of up to 0.06 g m^{-2} day^{-1}.

Riparian litterfall did not display significant linear relationships with site characteristics that were expected to influence, or be influenced by, leaf litter dynamics, possibly indicating complex interactions beyond the scope of this study. For example, there were no significant relationships between litterfall rate, or litter standing stock, and soil organic matter content. The characteristic ‘vegetation cover’ did not display any significant relationships with litterfall standing stock or accumulation. The soil organic matter content was also not significantly related to the ‘vegetation cover’ score.

Discussion

Although highly urbanised, Bulimba Creek does not have an 'engineered' or concrete channel devoid of natural structure typical of many urban creeks; rather, it is characterised by an intact riparian forest along much of its length. This contrast to the usual view of urban creeks as being little more than drains, possibly reflects the sub-tropical climate of south-east Queensland, with high rates of vegetation growth and variable flashy flooding which prohibits construction and urbanisation too close to the channel. The presence of the intact riparian forest meant that common attributes of urban streams, such as high stream temperatures and high algal biomass were not observed in this creek, as riparian shading acts to reduce both instream temperature fluctuations and algal production (Bunn et al., 1999). The degrading influences on Bulimba Creek, therefore, may be related more to changes in hydrology through both higher runoff from upstream impervious area and piped stormwater flows containing contaminants (see Walsh et al. 2005) rather than degraded riparian conditions.

Denitrification

The relatively low rates of denitrification observed in this study, when compared to other studies in urbanized catchments, suggest that Bulimba Creek may not be severely impacted by nitrate contamination, since the denitrifying community does not appear to be as abundant as reported elsewhere (Groffman et al. 2002; Inwood et al. 2005; Inwood et al. 2007). However, across much of the Bulimba Creek catchment, stream sediments and the associated riparian soils tended to be sandy, which could also contribute to the relatively low denitrification rates, as sandy soils will limit the organic matter content of the soil and moisture retention would be lower.

This is reflected in our results where the highest riparian denitrification potentials are found with high organic matter content, which occurs at sites with finer textured soils. Denitrification potentials of Bulimba Creek, when compared to literature values, were more similar to potentials reported for less impacted streams than other urban areas. For example, Inwood et al. (2005) found along a gradient of urbanization, increasingly higher instream denitrification potentials in a gradient from forested, agricultural then urban catchments, without additional nitrate treatment. In that study, the mean denitrification potential for instream sediments from the urban catchment was 15 mg N kg^{-1} day^{-1}. This is a much higher potential than that found in our study of 0-2 mg N kg^{-1} day^{-1}. Our values are more similar to the reported values from forest 1.5 mg N kg^{-1} day^{-1} (Inwood et al. 2005). In Bulimba Creek we did observe a relationship between denitrification potential at ambient nitrate loading and measures of channel 'debris' and 'habitat complexity'. These measures are indicative of the amount of woody debris and leaf litter accumulated in the stream channel, and reflect overall riparian condition. Although we found no significant relationship between denitrification and organic matter content of instream sediment, the abundance of woody debris was a factor in promoting higher denitrification potential. Therefore, in this urban stream the riparian forest provides woody debris and organic matter to form debris dams and organic matter accumulations which provide conditions for higher denitrification potential.

Denitrification potentials of riparian soils of Bulimba Creek were also more similar to forested catchments than urbanized catchments. Groffman et al. (2002), using a denitrification enzyme activity method involving addition of nitrate and carbon, found denitrification potentials of urban riparian soils (5-30cm depth) in Baltimore, U.S.A., were greater than 10 mg N kg^{-1} day^{-1} while the forested and low density urban catchments in this study had rates of 0.7-2.3 mg N kg^{-1} day^{-1}. Comparatively, the riparian sediments at a similar depth range (0-30 cm) in Bulimba Creek had denitrification potentials between 2 and 10 mg N kg^{-1} day^{-1}, which makes then more similar to the low density – forested soils of the Baltimore study. Denitrification potential of riparian soils therefore appears to reflect both the density of urbanisation and the prevalence of available nitrate. This was further confirmed by the addition of nitrate to the Bulimba Creek riparian soils, which suggested their denitrification potential was nitrate limited, and indicates that even though the sandy sediment and soils may limit denitrification potential across much of the Bulimba Creek catchment, the system may retain a buffering capacity against increases in nitrate load.

Contribution of organic matter from riparian vegetation

Mean riparian litter fall along Bulimba Creek (1 to 5 g m^{-2} day^{-1}) is greater than litter fall within Toohey Forest, a forest reserve within the Bulimba Creek catchment (0.9 g m^{-2} day^{-1}) (Birk 1979). The greater litterfall along Bulimba Creek may be a result of greater vegetation density and production rates associated with riparian conditions (Naiman and Decamps 1997), a trend found by Catterall et al. (2001) between upland and riparian sites within Toohey forest. Assuming a constant rate throughout the year, litter fall into the Bulimba Creek channel and riparian zone equates to approximately 350 g to 1500 g m^{-2} year^{-1}, about half of which is carbon and 1% is nitrogen. On the riparian soil, assuming relatively stable standing stocks of about 300g m^{-2}, the turnover rate of litter could be from 1 year in forested sites to as little as 10 weeks in grassy areas, some of which would be incorporated as soil organic matter. The deposition rates presented here for the forested sites are likely underestimates of organic matter deposition to the stream channel. Along with direct litterfall to the channel, litterfall in the riparian zone will also move towards the channel by wind-throw and repositioning by flooding. This is the case for all forested sites, however, cleared sites had large accumulation rates of organic matter (> 12 g m^{-2} day^{-1}), which consisted mainly of grasses, and as such would not provide the same delivery of coarse material to the stream channel as forested sites. Instead of course particulate matter coming from these cleared sites there is likely to be higher rates of dissolved organic carbon in their runoff, leached from the large decomposing biomass. This precludes the formation of organic debris accumulations in these reaches which can be important for instream food resources and nutrient processing.

Benthic algal biomass

Riparian shading may be limiting benthic algal growth in Bulimba Creek. The influence of shading on instream production obscured any effects that nutrient concentration may have had, as shading seemed to limit primary production to below a level where nutrient availability would be a limiting factor. This result is in contrast to other studies of urban streams and algal production that showed strong correlation between chlorophyll-a concentration and measures of urban density or nutrient concentration (Chessman et al. 1992, Taylor et al. 2004). However, Roy et al. (2005) found that in paired sites of urban streams, those with riparian shading had reduced algal biomass compared with those without riparian forest cover. This result again reflects the fact that Bulimba Creek flows through a green space corridor that provides significant shading over much of the stream, compared to a more ‘typical’ urban stream that lacks riparian forest. For example, in urban catchments of Melbourne, Australia, which had high benthic light levels, benthic algal biomass was positively correlated with urban density and most strongly with drainage connection, which provided phosphorus to the stream (Taylor et al. 2004). Chessman (1985) found that in human impacted sections of the La Trobe River, chlorophyll-a concentrations peaked with peaks in nitrate concentration in late winter and spring, which is also the time of year of greatest runoff in that region (Lake 1995). In Bulimba Creek during baseflow sampling, nitrate concentration in the creek did not exceed 1 mg N L^{-1} and benthic chlorophyll-a did not relate to nutrient concentrations, but appeared to be limited by available light. During periods of increased runoff in summer, nutrient concentrations could increase, although it is expected that shading would be a dominant controlling factor year-round. Further investigation of chlorophyll-a concentrations and stream nutrient concentrations year-round would be required to determine the effects of streamflow and nutrients on seasonal variation of algal growth.

Riparian vegetation

The measured stream characteristics of benthic algal biomass and leaf litter fall showed ties with the nature of the riparian forest of Bulimba Creek. Benthic algal biomass was limited by the low light environment produced by shading riparian trees. The input of organic matter to the stream and riparian soil, especially during baseflow conditions when stormwater drainage to the creek is minimal, is dependent on local vegetation condition. Denitrification in riparian soils, although having a high potential, is possibly restricted by a limited interaction between groundwater and shallow soils. As a management tool, the denitrification potential of shallow riparian soils could be utilized by managing urban stream hydrology to connect nitrate-laden water with riparian soils through the construction of riparian wetlands and detention basins for surface runoff (Groffman et al. 2002). Our findings support these management options as riparian denitrification rates were found to be much higher than those of instream sediments per kilogram of soil. High stormflow events in urban catchments may engage shallow soil denitrification by raising streamwater levels to flood the surrounding riparian area and promote bank storage (Rassam et al. 2006). This process could be enhanced through the previously mentioned management options of riparian wetland construction. Deep-rooted riparian vegetation can aid in the removal of nitrate and other nutrients from groundwater by direct uptake, and can account for substantial removal in the absence of conditions that favour denitrification (Schade et al. 2005).

This urban riparian forest was found to provide important, although limited, nutrient processing functions for Bulimba Creek. Restoration of a more natural hydrograph and increasing hydrologic interaction with riparian soils is vital for improving urban stream health, yet where degraded riparian conditions exist in urban catchments, revegetation may be a useful method to return some degree of stream health to these reaches.

Acknowledgements

This work was undertaken as part of an Honours project in the Australian Rivers Institute, Griffith University, Australia, under the supervision of the late Dr Christy Fellows. We dedicate this paper to her memory and her enthusiasm for riparian zones and nutrient processes. We also thank many people for their field assistance, in particular Amanda Carter, and Pat Laceby for map making expertise. Funding for the project came from a Griffith University Research Grant (GURG) to Sheldon, Fellows and Jones. We thank an anonymous reviewer for helpful comments on this manuscript.

References

Ambus P, Lowrance R (1991) Comparison of denitrification in two riparian soils. Soil Sci Soc Am J 55:994-997.

Balderston W, Sherr B, Payne W (1976) Blockage by acetylene of nitrous oxide reduction in *Pseudomonas perfectomarinus*. Appl Environ Microbiol 31:504-508.

Birk EM (1979) Overstorey and understorey litter fall in a eucalypt forest: spatial and temporal variability. Aust J Bot 27: 145-156.

Bunn SE, Davies PM, Mosisch TD (1999) Ecosystem measures of river health and their responses to riparian and catchment degradation. Freshwat Biol 41:333-345.

Catterall CP, Piper SD, Bunn SE, Arthur JM (2001) Flora and fauna assemblages vary with local topography in a subtropical eucalypt forest. Aust Ecol 26:56-69.

Chessman BC (1985) Artificial-substratum periphyton and water quality in the lower La Trobe River, Victoria. Aust J Mar Freshwat Res 36:855-871.

Chessman BC, Hutton PE, Burch JM (1992) Limiting nutrients for periphyton growth in sub-alpine, forest, agricultural and urban streams. Freshwat Biol 28:349-361.

Costelloe J (2005) Protocols for geomorphic and riparian zone indicators of river health. Quantifying the Health of Ephemeral Rivers (QHER) project. Final Report to Land and Water Australia, Canberra.

Dugas WA, Heuer ML, Mayeux HS (1999) Carbon dioxide fluxes over bermudagrass, native prairie, and sorghum. Agric For Meteor 93:121-139.

Findlay S, Sinsabaugh RL (1999) Unravelling the sources and bioavailability of dissolved organic matter in lotic aquatic ecosystems. Mar Freshwat Res 50:781-790.

Frank AB, Dugas WA (2001) Carbon dioxide fluxes over a northern, semiarid, mixed-grass prairie. Agric For Meteor 108:317-326.

Gregory SV, Swanson FJ, McKee WA, Cummins KW (1991) An ecosystem perspective of riparian zones. Bioscience 41: 540-551.

Groffman PM, Bulware NJ, Zipperer WC, Pouyat RV, Band LE, Colosima MF (2002) Soil nitrogen cycle processes in urban riparian zones. Environ Sci Tech 36:4547-4552.

Groffman PM, Dorsey AM, Mayer PM (2005) N processing within geomorphic structures in urban streams. J N Am Benthol Soc 24:613-625.

Hill AR (1996) Nitrate removal in stream riparian zones. J Environ Qual 25:743-755.

Hosomi M, Sudo R (1986) Simultaneous determination of total nitrogen and total phosphorus in freshwater samples using persulphate digestion. Int J Environ Stud 27:267-275.

Inwood SE, Tank JL, Bernot MJ (2005) Patterns of denitrification associated with land use in 9 midwestern headwater streams. J N Am Benthol Soc 24:227-245.

Inwood SE, Tank JL, Bernot MJ (2007) Factors controlling sediment denitrification in Midwestern streams of varying land use. Microb Ecol 53:247-258.

Jansen A, Robertson A, Thompson L, Wilson A (2004) Development and application of a method for the rapid appraisal of riparian condition, River Management Technical Guideline No. 4. Land and Water Australia, Canberra.

Lake PS (1995) Of floods and droughts: river and stream ecosystems of Australia. Pages 659-694 in C. E. Cushing, K. W. Cummins, and G. W. Minshall (editors). Ecosystems of the World 22: River and Stream Ecosystems. Elsevier, Amsterdam.

Lorenzen CJ (1967) Determination of chlorophyll and pheo-pigments: spectrophotometric equations. Limnol Oceanogr 12:343-346.

McClain ME, Boyer EW, Dent CL, Gergel SE, Grimm NB, Groffman PM, Hart SC, Harvey JW, Johnston CA, Mayorga E, McDowell WH, Pinnay G (2003) Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosys 6:301-312.

Meyer JL, Paul MJ, Taulbee WK (2005) Stream ecosystem function in urbanizing landscapes. J N Am Benthol Soc 24:602-612.

Mosisch TD, Bunn SE, Davies PM (2001) The relative importance of shading and nutrients on algal production in subtropical streams. Freshwat Biol 46: 1269-1278.

Murdock J, Roelke D, Gelwick F (2004) Interactions between flow, periphyton and nutrients in a heavily impacted urban stream: implications for stream restoration effectiveness. Ecol Eng 22:197-207.

Naiman RJ, Décamps H (1997) The ecology of interfaces: riparian zones. Annu Rev Ecol Syst 28:621-658.

Paul MJ, Meyer JL (2001) Streams in the urban landscape. Annu Rev Ecol Syst 32:333-365.

Peterjohn WT, Correll DL (1894) Nutrient dynamics in an agricultural watershed: observation on the role of a riparian forest. Ecol 65:1466-1475.

Rassam DW, Fellows CS, DeHayr R, Hunter H, Bloesch P (2006) The hydrology of riparian buffer zones; two case studies in an ephemeral and a perennial stream. J Hydro 235:308-324.

Rutherfurd ID, Marsh NA, Davies PM, Bunn SE (2004) Effects of patchy shade on stream water temperature: how quickly do small streams heat and cool? Mar Freshwat Res 55: 737-748.

Rutherfurd ID, Jerie K, Marsh N (2000) A Rehabilitation Manual for Australian Streams. Cooperative Research Centre for Catchment Hydrology and Land and Water Resources Research and Development Corporation.

Sabater S, Butturini A, Clement J, Burt T, Dowrick D, Hefting M, Matre V, Pinay G, Postolache C, Rzepecki M, Sabater F (2003) Nitrogen removal by riparian buffers along a European climatic gradient: patterns and factors of variation. Ecosyst 6:20-30.

Schade JD, Welter JR, Marti E, Grimm NB (2005) Hydrologic exchange and N uptake by riparian vegetation in an arid-land stream. J N Am Benthol Soc 24:19-28.

Taylor SL, Roberts SC, Walsh CJ, Hatt BE (2004) Catchment urbanization and increased benthic algal biomass in streams: linking mechanisms to management. Freshwat Biol 49:835-851.

Ter Steege H (1994) HEMIPHOT: A programme to analyse vegetation indices, light and light quality from hemispherical photographs. Tropenbos Documents 3. Tropenbos Foundation, Wageningen.

Walsh CJ, Roy AH, Feminella JW, Cottingham PD, Groffman PM, Morgan RP (2005) The urban stream syndrome: current knowledge and the search for a cure. J No Am Benthol Soc 24:706-723.

Walsh CJ, Sharpe AK, Breen PF, Sonneman JA (2001) Effects of urbanization on streams of the Melbourne region, Victoria, Australia. I. Benthic macroinvertebrate communities. Freshwat Biol 46:535-551.

Table 1: Stream characteristics for each study site. Streamwater nitrate content was sampled at time of denitrification assay. Catchment Imp. is total catchment imperviousness. CC - canopy cover.

Site No.	Instream Sediments	Riparian Soils	Streamwater Physicochemical Parameters: Conductivity ($\mu S\ cm^{-1}$)	DO 24hr Max (ppm)	DO 24hr Min (ppm)	Water Temp 24hr Max (°C)	Water Temp 24hr Min (°C)	NO_3-N ($mg\ L^{-1}$)	CC (% ± SE)	Dominant Riparian Vegetation	RARC Score	Catchment Imp. (%)	Catchment Area (km^2)
1	Silty	Silty clay	411	4.05	3.07	18.2	16.1	0.43	53.8 ± 1.7	*Melaleuca quinquenervia, Brachiaria mutica*	0.416	25.6	3.0
2	Sandy gravel	Sandy loam	381	5.81	2.95	16.9	15.2	0.34	53.6 ± 3.0	*M. quinquenervia, Casuarina glauca*	0.432	49.5	7.2
3	Sandy gravel, some fine sand	Clay to sandy clay	423	9.70	8.78	16.7	14.9	0.44	34.8 ± 7.1	*Cynodon dactylon, Eucalyptus spp.*	0.461	57.2	16.2
4	Sandy, some exposed bedrock	Fine sand	297	9.88	8.23	16.9	15.0	0.97	65.0 ± 2.0	*M. quinquenervia, Cinnamomum camphora*	0.471	46.5	38.7
5	Sandy	Fine sand	375	8.76	6.62	16.2	15.5	0.67	77.1 ± 3.3	*C. camphora, Lomandra longifolia*	0.515	46.0	40.0
6	Sandy with areas of silt	Sandy loam	367	10.01	7.08	17.2	15.6	0.61	64.2 ± 2.0	*C. camphora, C. glauca*	0.463	48.3	44.0
7	Sandy	Sandy loam	385	10.14	7.46	16.6	14.8	0.47	64.1 ± 2.0	*C. camphora, Acacia concurrens*	0.535	47.6	46.5
8	Sandy	Sandy loam	375	9.91	6.72	16.7	15.2	0.69	79.3 ± 3.0	*C. camphora, Eucalyptus spp.*	0.481	49.4	50.9
9	Clay	Clay	405	9.84	9.21	20.6	17.3	0.65	26.7 ± 5.0	*B. mutica, C. camphora*	0.272	49.4	60.6

Table 2: Mass of carbon and nitrogen in standing stocks and accumulated leaf litter. '*' indicates sites where grass collection method was used.

Site No.	Standing Stock (g m^{-2})		Fall Rate (g m^{-2} day^{-1})		C:N molar ratio
	Carbon	Nitrogen	Carbon	Nitrogen	
1	219.0	4.94	0.9	0.02	51.7
2	192.1	3.63	1.7	0.03	61.8
3a	13.5	0.39	1.3	0.04	40.2
3b	202.9	5.95	2.1	0.06	39.8
4	95.6	2.23	1.3	0.03	50.0
5	146.6	2.73	1.1	0.02	62.6
6	193.0	3.85	0.6	0.01	58.6
7*	151.5	2.85	5.8	0.11	62.0
8	199.7	5.18	1.0	0.03	45.0
9*	128.5	4.22	5.6	0.18	35.5

Fig. 1: Bulimba Creek catchment showing location of study sites.

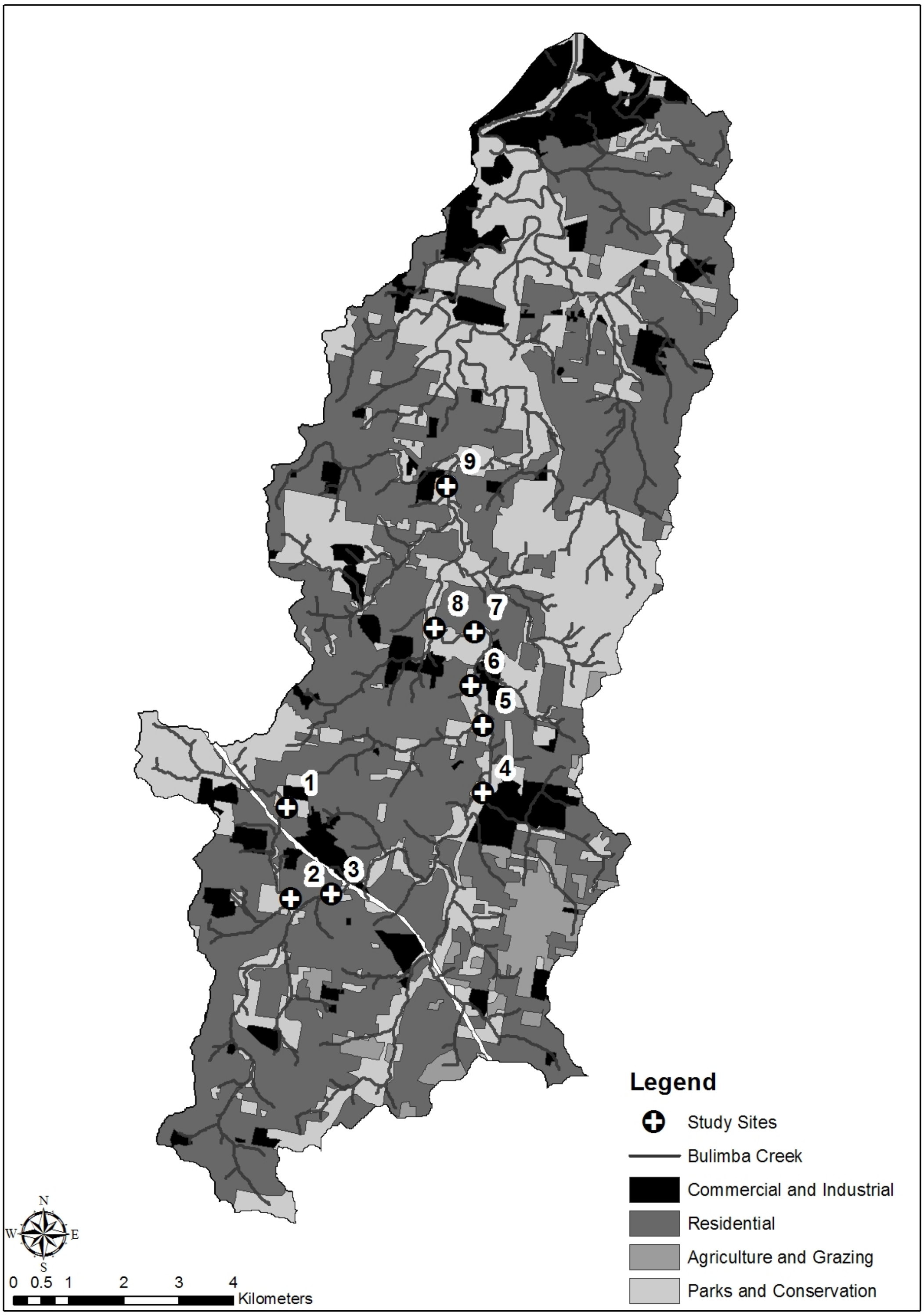
Fig. 2: Denitrification potential mean and SE for each site and nitrate treatment (n=4). Sites are arranged from 1 to 9, left to right within each treatment. Stippled bars represent control treatments while solid bars represent samples that received additional nitrate.

Fig. 3: Mean (+SE) denitrification potential (a), organic matter content (b) and slurry nitrate concentration before spike (c) of instream sediments (spotted bars) and riparian soil (solid bars) from 9 reaches of Bulimba Creek. Denitrification potential is that with additional nitrate treatment.

Fig. 4: Relationship between canopy cover and benthic chlorophyll-a concentration. Points are means ± SE (n=3).

Fig. 5: Leaf litter standing stocks in riparian zones and the accumulation rate from litter fall traps. Site 3a is managed parkland that is frequently mown. Site 3b is the opposite bank that has thick brush to 3 m with overstory trees. Bars and points are means ± SE (n=4). Calculation of accumulation rate at grassy sites (Sites 7 and 9) did not contain replication to provide standard error.

Figure 1



9
8
7
6
5
4
1
2
3
Legend
Study Sites
Bulimba Creek
Commercial and Industrial
Residential
Agriculture and Grazing
Parks and Conservation
N
W
E
S
0 0.5 1 2 3 4
Kilometers

Figure 2

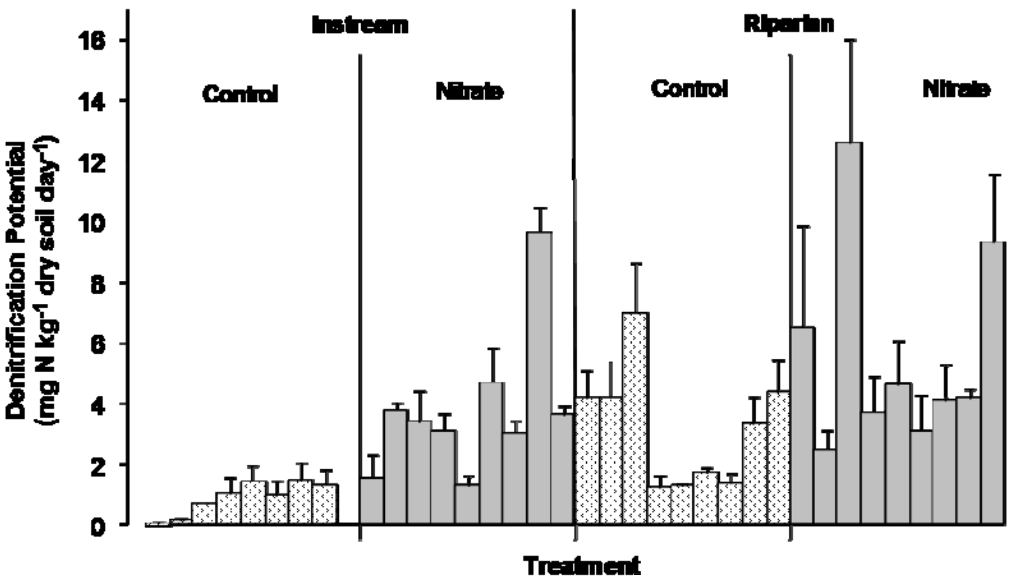
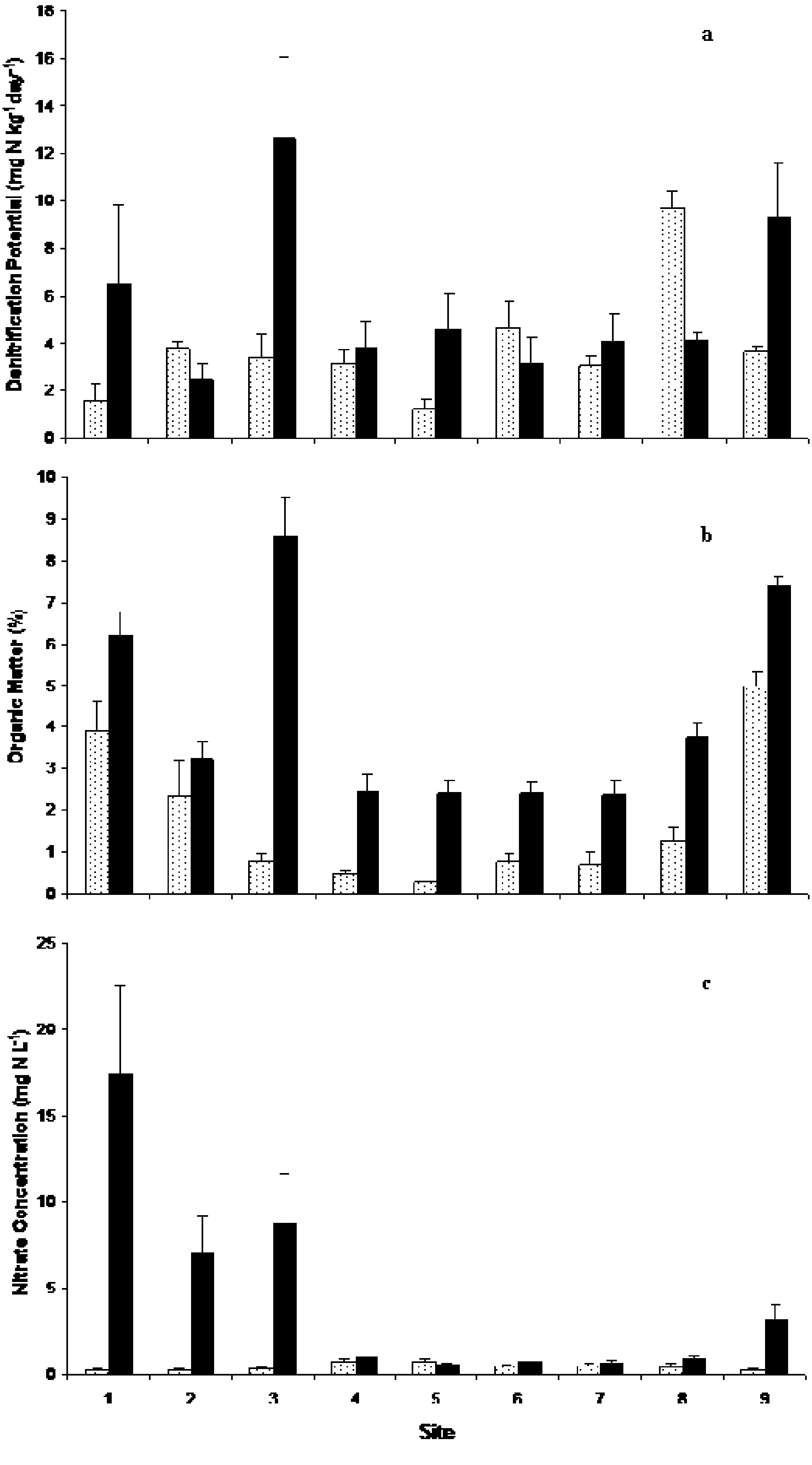
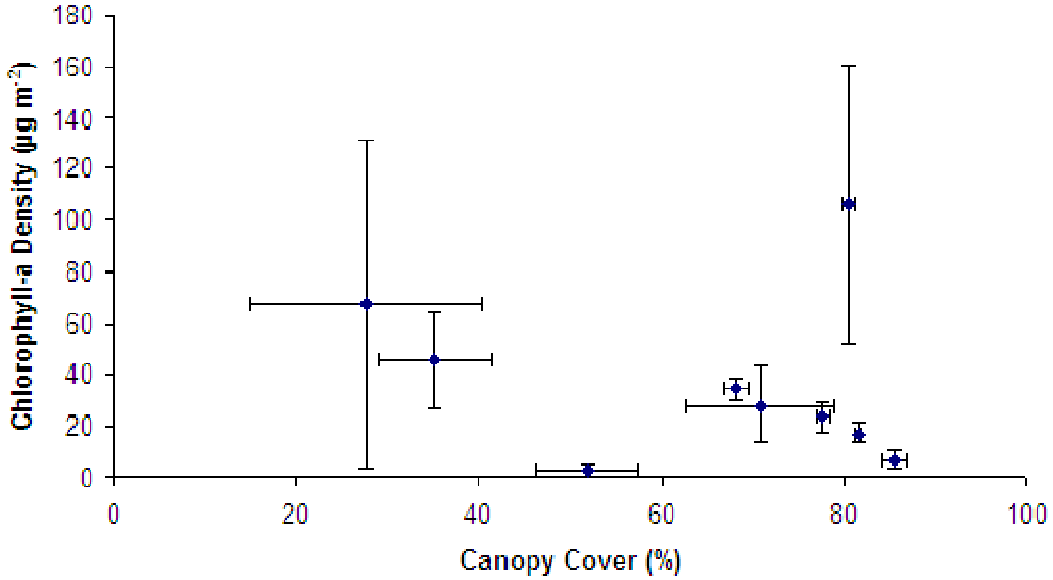

Instream
Riparian
Control
Nitrate
Control
Nitrate
Denitrification Potential
(mg N kg-1 dry soil day-1)
0
2
4
6
8
10
12
14
16
Treatment

Figure 3



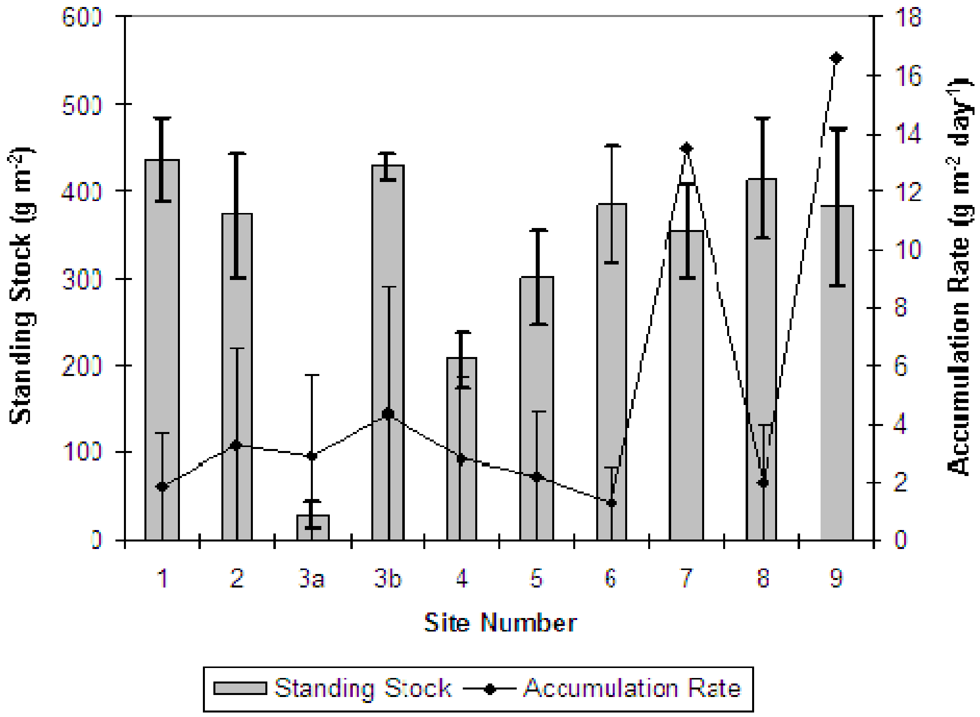
a
b
c
Denitrification Potential (mg N kg-1 day-1)
Organic Matter (%)
Nitrate Concentration (mg N L-1)
Site
1
2
3
4
5
6
7
8
9

Figure 4



Chlorophyll-a Density (µg m-2)
180
160
140
120
100
80
60
40
20
0
0
20
40
60
80
100
Canopy Cover (%)

Figure 5



600
500
400
300
200
100
0
Standing Stock (g m-2)
18
16
14
12
10
8
6
4
2
0
Accumulation Rate (g m-2 day-1)
1
2
3a
3b
4
5
6
7
8
9
Site Number
Standing Stock
Accumulation Rate