Quantifying nutrient removal by riparian vegetation in Ellen Brook

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Executive Summary

The aim of this report was to assess the effectiveness of riparian vegetation as a best management strategy to reduce nutrients from agricultural runoff before it entered waterways of the Swan Canning river system. The study focussed on riparian vegetation in flat sandy regions of the Ellen Brook Catchment.

Eutrophication of the Swan Canning estuary is a key environmental issue affecting the health and human amenity of the system (Swan River Trust, 2009). This is being driven by nitrogen and phosphorus enrichment from the catchment. Ellen Brook contributes 7% of the total flow into the Swan Canning river system, yet it contributes up to 39% of total phosphorus and 28% of total nitrogen annually from coastal catchments (Swan River Trust, 2009). Much of the Ellen Brook is characterised by poor, nutrient deficient soils (Bassendean sands) with fertiliser use required to achieve agricultural production. Unfortunately, the soils have low phosphorus retention and high leaching capacity (Barron et al. 2008), as a consequence nutrient release and loss into waterways is high where fertilisers are used. Understanding the effectiveness of riparian vegetation is essential for nutrient management in the future.

At Bingham Creek, the riparian vegetation was found to facilitate nitrogen removal through denitrification, however phosphorus removal was limited. There was little slope at Bingham Creek and, combined with high permeability of the sandy soils, this resulted in a lack of surface flow. The primary phosphorus removal pathway in riparian zones is through the interception of particulates in surface flow. Without horizontal surface flow, this primary phosphorus removal pathway was absent. The lack of slope also resulted in little horizontal subsurface flow from the paddock to the riparian zone and water movement was dominated by vertical rise and fall of groundwater over an annual cycle. Water also entered the riparian zone from the stream at the start of the wet season. The long residence time facilitated denitrification and caused the groundwater to act as a store for phosphorus. The lack of soil storage capacity at Bingham Creek resulted in high groundwater nutrient concentrations. The riparian vegetation was characterised by a native canopy, and an exotic understorey and groundcover. Exotic species took up more nutrients, but also had greater rate of turnover and nutrient release. Native riparian vegetation assimilated some nutrients but also improved soil phosphorus binding capacity through the addition of organic matter.

At Lennard Brook, riparian vegetation was found to have a higher capacity to remove phosphorus and nitrogen than at Bingham Creek. Slope and a shallow water table created surface flow, maximising phosphorus removal. Groundwater was continuously in contact with the active root zone, maximising nutrient assimilation by riparian vegetation. Soils at Lennard Brook had a higher iron and clay content and a better capacity to hold onto nutrients, restricting movement of nutrients from soil to underlying groundwater. In contrast to Bingham Creek, Lennard Brook exhibited high soil nutrient concentrations and low groundwater concentrations. The vegetation at Lennard Brook had a greater proportion of native species, providing greater long-term input of organic matter and nutrient uptake.

Phosphorus removal of riparian vegetation in flat sandy locations can be improved by:

- Improving soils through soil amendment in the riparian zone

- Lining stream beds with phosphorus-binding amendments
- Introducing or maintaining native aquatic plants (e.g. *Cycnogeton sp.*) in streams
- Planting native wet-dry tolerant sedges in a strip 5m wide along the stream bank to increase nutrient removal potential and to facilitate aeration of soil and groundwater

The riparian vegetation at Lennard Brook was functioning well and should be promoted as best management practice for nutrient removal and the associated environmental benefits riparian zones provide on streams on this soil type. Riparian vegetation provides aquatic and terrestrial habitat, energy and food, contributing to terrestrial and aquatic biodiversity; it shades and cools streams and has shown to improve instream biogeochemical processes. Riparian vegetation also improves soil quality by contributing organic matter, increasing infiltration rates and protecting from erosion. This reduces nutrients entering streams, stops sediment clogging waterways and helps maintain water clarity. Existing riparian vegetation throughout the catchment should be protected and fenced off to limit disturbance and promote these attributes.

Contents

Figures

Tables

Chapter 1 General Introduction

Use of riparian vegetation as a nutrient reduction management strategy

Eutrophication of the Swan Canning estuary is a key environmental issue affecting the health and human amenity of the system (Swan River Trust, 2009). This is being driven by nitrogen and phosphorus enrichment, resulting from human activities such as agriculture and to a lesser extent urbanisation (Swan River Trust, 2009). These high nutrient inputs have led to algal blooms, deoxygenation events, fish kills and poor water quality.

Riparian vegetation, by definition, is vegetation adjacent to streams or rivers. It is comprised of trees, shrubs, grasses and herbs, which serve to moderate environmental processes between the catchment and stream (Herron and Hairsine 1998; Naiman and Decamps 1997; Verry et al. 2004). Riparian vegetation has been used extensively world-wide as a best management strategy for nutrient reduction (Narumalani et al. 1997; Lyons et al. 2000; Brian et al. 2004). Riparian vegetation takes up a small fraction of land but has a disproportionately important role in maintaining and improving water quality (Dosskey et al. 2010). Brian et al (2004) demonstrated it could reduce stream suspended sediment loads by between 40 and 80%, phosphorus by up to 90% and nitrogen by up to 99%.

Besides improving water quality, riparian vegetation benefits both aquatic and terrestrial ecosystems through providing habitat for flora and fauna, shading streams and thereby regulating water temperature and limiting algal blooms, contributing organic matter to fuel the food web, reducing bank erosion and overland flow and providing aesthetic qualities (Brian et al. 2004; Francis 2006; Mander et al. 1997; Naiman and Decamps 1997; Tabacchi et al. 1998,2000; Verry et al. 2004).

Riparian vegetation actively contributes to nutrient reduction by plant assimilation and by altering the underlying physical and chemical conditions of riparian soils and groundwater (Dosskey et al. 2010; Naiman and Decamps 1997; Narumalani et al. 1997; Verry et al. 2004). Nitrogen is primarily removed through microbial denitrification and to a lesser extent plant uptake (Brian et al. 2004; Dhondt et al. 2006; Mander et al. 1997; Montreuil et al. 2010; Tabacchi et al. 1998; Vidon et al. 2010; Vought et al. 1994), which generally occurs in subsurface waters. By contrast, phosphorus is primarily removed from surface flows through soil sorption and to a lesser extent plant assimilation (Brian et al. 2004).

For riparian vegetation to work there must be a flow of water through the riparian zone. This flow can be across the soil surface, which allows for particulate matter to be trapped and nutrients intercepted (Vought et al. 1994; Narumalani et al. 1997; Tabacchi et al. 1998, 2000; Knight et al. 2010), or it can be subsurface flow though the root zone, which allows for nutrient assimilation and for chemical transformations in groundwater to occur (Vought et al. 1994; Narumalani et al. 1997; Tabacchi et al. 2000). As water progresses through the riparian zone nutrients are progressively stripped, whereas in reaches lacking vegetation there is limited opportunity for interception (Figure 1-1). When riparian vegetation is removed its absence can have deleterious impacts on stream processes. It can lead to the release of nutrients stored in riparian zone soils, while decreasing organic matter input to streams (Sabater et al. 2000). Furthermore, removing riparian vegetation increases light availability, which can increase the potential for algal blooms to occur.

Three components that affect the water delivery from the paddock through the riparian zone are slope, soil type and a subsurface impermeable layer (Dosskey 2001). These factors have a strong bearing on water movement through the riparian zone, particularly whether movement is dominated by surface or subsurface flow. Typically, reaches with sufficient slope, good soils and an impermeable layer result in surface movement through the riparian zone (Fig 1.1).

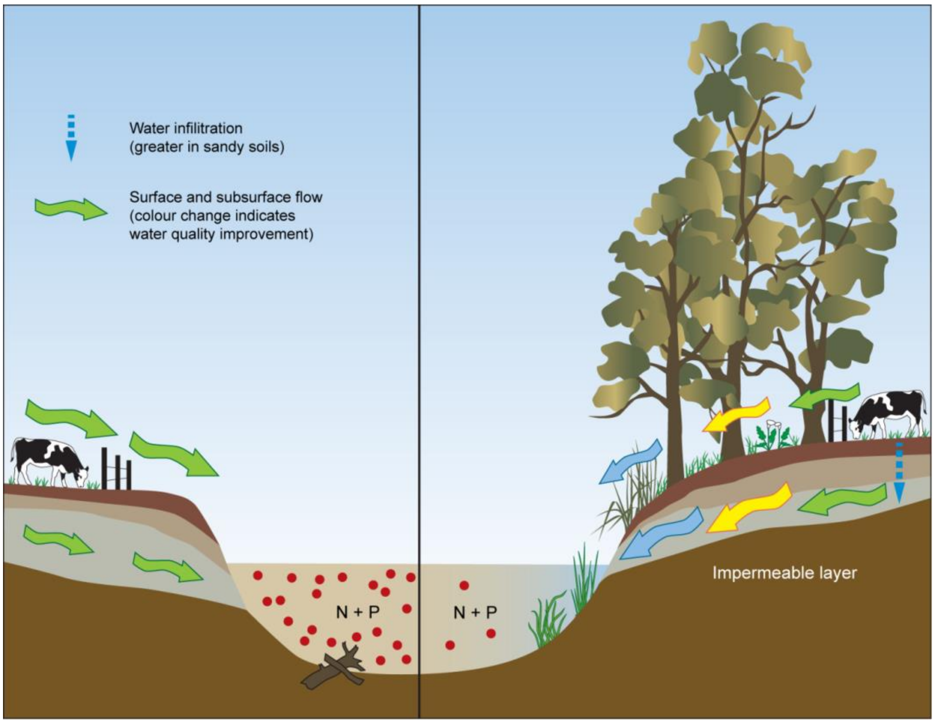


Figure 1-1. Conceptual model highlighting the importance of flow in nutrient removal for vegetated and unvegetated riparian zones

Ellen Brook catchment

Ellen Brook is a relatively large sub-catchment (716km^2) of the Swan Canning river system, north east of the city of Perth, Western Australia. The climate is Mediterranean, with hot, dry summers and mild, wet winters, with most rainfall occurring from May through to October. Streams within the catchment are mainly ephemeral, with 40% of the discharged flow from baseflow from groundwater (Barron et al. 2008). Stream flow occurs mainly in late winter.

Ellen Brook contributes 7% of the total flow into the Swan Canning estuary, yet it contributes upwards of 39% of total phosphorus and 28% of total nitrogen annually (Swan River Trust, 2009). The main nutrient sources identified in the catchment are fertilisers, animal waste and soil-bound nutrients (Barron et al. 2008). Land use in the catchment is dominated by agriculture, with extensive livestock production (cattle). Much of the Ellen Brook catchment is characterised by poor nutrient deficient soils so fertiliser is required to achieve agricultural production. Unfortunately, the soils have low phosphorus retention and high leaching capacity (Barron et al. 2008), as a consequence nutrient release and loss into waterways is high where fertilisers are used.

Nutrient export from Ellen Brook poses a serious threat to the health of the Swan Canning estuary and a number of management strategies have been implemented to reduce it (Swan River Trust, 2009). Riparian vegetation is used as a best management practice in Ellen Brook however there is limited research on the effectiveness of riparian vegetation for reducing nutrient concentrations in flat sandy-soil systems. Factors which could be affecting the functionality of riparian vegetation in Ellen Brook include low slopes, poor soils and the absence of a subsurface impermeable layer. These factors can have a strong bearing on water flow through riparian zones and ultimately affect the nutrient stripping capacity of riparian zones.

Ellen Brook is predominantly underlain by highly permeable sands of the Bassendean Dune System, although there are also regions with duplex soils (sands over clays and loams over clays). The topography of the catchment varies markedly, with the west side of the catchment being predominantly flat (mainly comprising of coastal dunes) whereas in the east it slopes down from an escarpment. Two riparian zones within the Ellen Brook catchment were chosen to provide a comparison of locations with differing slopes and soil types: Bingham Creek and Lennard Brook (Figure 1-2).

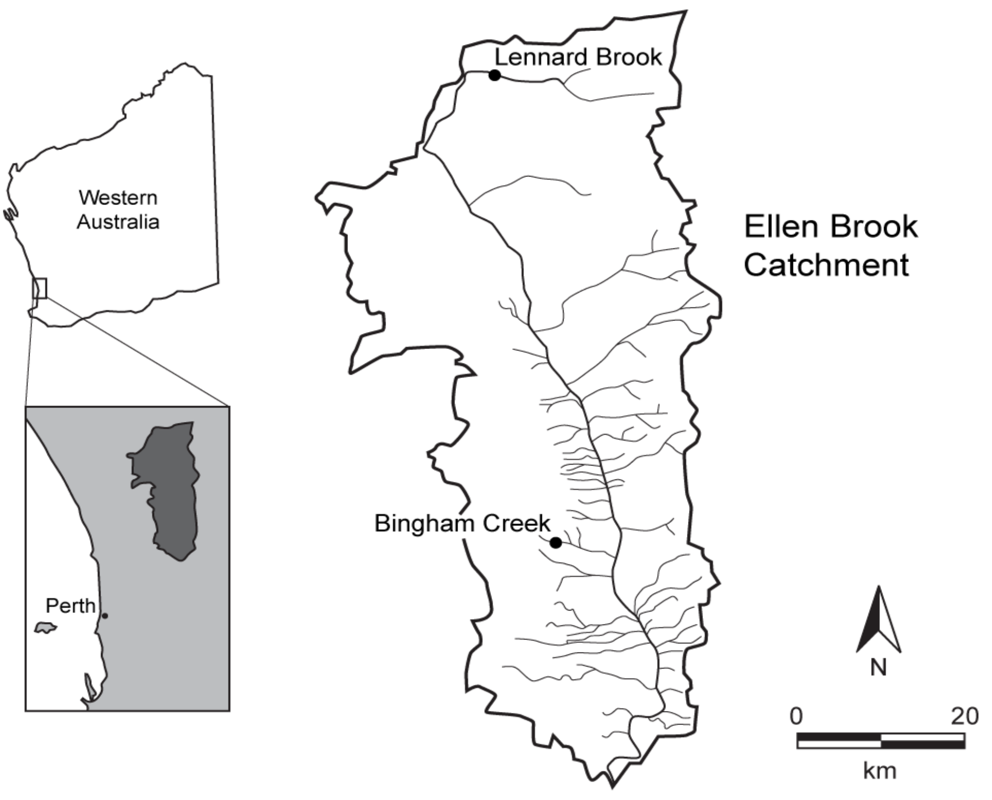


Figure 1-2. Map illustrating the location of Bingham Creek and Lennard Brook within the Ellen Brook catchment

Bingham Creek is in the south-west of the catchment (Figure 1-2). It is a small intermittent stream running from the west of the catchment and flows directly into the main channel of

Ellen Brook. Upstream of the study site is primarily used for livestock and horticulture (a strawberry farm) and riparian vegetation is patchy. The study site is flat and is underlain by deep Bassendean sands (Figure 1-3). The study site is characterised by a 30m wide strip of existing riparian vegetation on either side of the stream that is fenced off (Figure 1-4). The riparian vegetation is adjacent to a paddock used for cattle grazing but no fertiliser is used.

Figure 1-3. Pictures characterising the soil types at Bingham Creek (left) and Lennard Brook (right)

Lennard Brook is at the top of the catchment (Figure 1-2). It is a perennial stream running east towards the Ellen Brook main channel. The stream is spring fed and the study site is around 5km from the source. The upper catchment of the stream is well forested and is bordered by agricultural land. The study site is underlain by iron-rich sands (Figure 1-3). The topography is characterised by rolling hills and the study site is strongly sloped. There is an extensive strip (width) of existing riparian vegetation, which is fenced off (Figure 1-4). The riparian vegetation is bordered upslope by a paddock where sheep grazing occurs. No fertilisers were used during the study period.

Figure 1-4. 1Pictures characterising the vegetation at Bingham Creek (left) and Lennard Brook (right)

The aim of this report is to address the question:

Is riparian vegetation effective at reducing nutrients entering Ellen Brook?

Understanding the nutrient dynamics of water, soil and vegetation and how they interact with one another is imperative for answering this question. Therefore, this report is broken up into three sections covering each of these aspects.

The chapter on water focuses on the hydrology and nutrient dynamics at Bingham Creek and Lennard Brook. To do this groundwater and stream water were analysed at both locations. Groundwater was assessed using nested piezometers at three different depths (0.5, 1.5 and 2.5 m), along a transect from the stream through the riparian zone and into the paddock. From this, the hydrology of the groundwater was determined as well as physiochemical and nutrient dynamics. Groundwater quality was measured over time to determine what changes occur from baseflow through to the end of the wet season.

Soil was sampled from Bingham Creek and Lennard Brook in the stream, riparian zone and paddock. This was sampled over four depths (0.05, 0.5, 1.5 and 2.5 m) to see how the physical, chemical and nutrient dynamics of the soils differed. This was done to see how soils were affecting hydrology, to determine the capacity of soils to hold onto nutrients and to assess nutrient stores.

Finally, vegetation was sampled at both sites, which were split into the stream, riparian zone and paddock. Within the riparian zone, the vegetation was categorised into functional groups: canopy, understorey and groundcover. The vegetation was classified and the dominant species sampled and analysed for nutrients. This was done over spring and autumn to see whether seasonal differences occurred in diversity and for nutrients. A litterfall test was carried out to assess litter and nutrient contributions from the riparian vegetation to underlying soils and groundwater.

Information from each of these elements of the study was then used to create conceptual models and to compare how the riparian zone functioned at each site. From this management recommendations were developed.

Chapter 2 Water

Introduction

To be effective as a nutrient filter, riparian vegetation needs to intercept the main pollution transport pathways (Dosskey 2001; Dosskey et al. 2010). Nutrients are primarily transported in water, so the relative proportions of water flowing above and below the ground and their rate of flow determine riparian nutrient removal performance (Vought et al. 1994; Narumalani et al. 1997; Dosskey et al. 2010; Figure 2-1). Surface flow through the riparian zone facilitates the removal of particulate-bound nutrients, whereas subsurface flow through the root zone facilitates uptake of dissolved nutrients (Vought et al. 1994; Narumalani et al. 1997; Tabacchi et al. 2000).

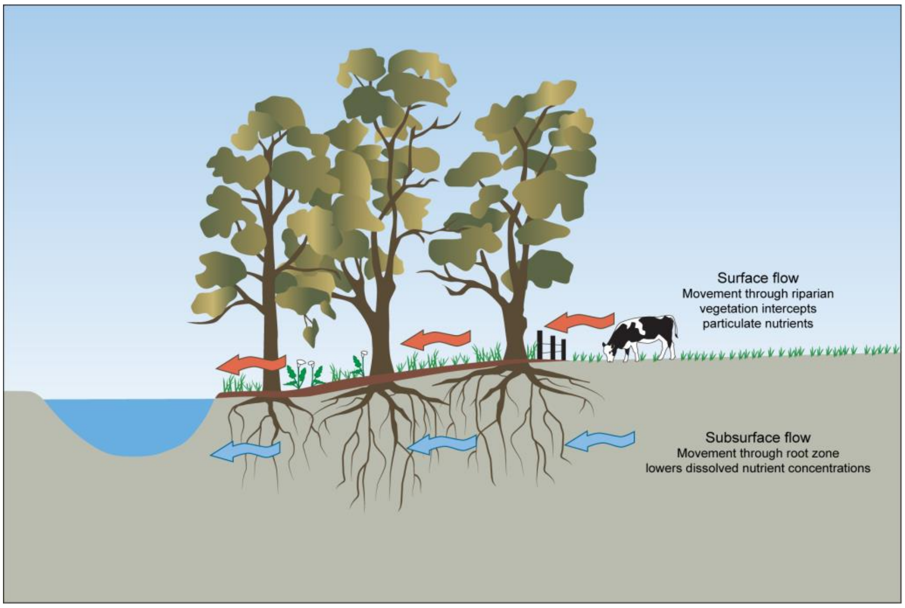


Figure 2-1. Diagram showing the importance of surface and subsurface flow through a riparian zone and how this affects nutrients

The key elements in the effectiveness of riparian zones have been shown to be slope, soil type and the presence of a subsurface impermeable layer, which optimise flow patterns and nutrient removal (Dosskey 2001).

Slope strongly influences water movement, often dictating the proportion of surface and subsurface flow from the paddock to the stream. Where there is limited slope, there can be a lack of horizontal water movement, decreasing the interaction with the riparian zone (Schoonover and Willard 2003). Weaver (2011) showed that in flat deep sand systems, water bypassed the riparian zone and entered from directly beneath the streambed from deep groundwater flow.

Soil type, particularly soil texture, affects the hydrology of soils. Soil texture describes the relative proportions of sand, silt and clay within a soil (Coyne and Thompson 2006) and influences porosity, pore size distribution, water holding capacity and permeability (Brutsaert 2005; Coyne and Thompson 2006; Ward and Robinson 2000). For example, sands have a large grain size and are usually more loosely packed than clays, and so have higher infiltration rates. This can dictate whether water moves across the surface (clay) or beneath it (sand) (Coyne and Thompson 2006). Soil type also affects rate of flow. Highly permeable sands are synonymous with rapid flow, whereas clays can be nearly impermeable leading to very slow flows (Ward and Robinson 2000; Coyne and Thompson 2006). The rate of flow, together with soil water holding capacity, determine the residence time of water in the soil. Fast flow or limited holding capacity limits the opportunity for nutrient uptake or nutrient transformation processes such as denitrification (Reddy and DeLaune 2008). The location of the water table, which is defined as the level below which the ground is saturated with water, can dictate groundwater interaction with roots of vegetation (Hill 1996).

A subsurface impermeable layer (clay, bedrock etc.) is often a facet of a duplex soil, for example where clay underlies a sand or loam. The presence of this layer restricts the downward passage of water. Being unable to penetrate (or penetrate very slowly) the lower layer, water preferentially travels horizontally through the highly permeable upper layer. This ensures water remains in the root zone of the riparian vegetation and promotes surface flow (Schoonover and Williard 2003). In soil types lacking this impermeable layer, like deep sands, water often infiltrates down the soil column beneath the riparian vegetation and rises and falls over time.

Riparian vegetation improves water quality through physical, chemical and biological processes (Dosskey et al. 2010; Naiman and Decamps 1997; Narumalani et al. 1997; Verry et al. 2004). Physically, vegetation increases hydraulic roughness decreasing surface flow and in turn sediment and nutrient deposition in the riparian zone. Bank stability is also increased which reduces erosion and sediment transport (Narumalani et al. 1997; Verry et al. 2004). Plant cover increases infiltration rates of underlying soils, as the network of roots provide preferential flow pathways into the soil, decreasing surface runoff (Greene et al. 1994, Figure 2-2). Chemically, riparian vegetation modifies redox potential and facilitates transformation of nutrients. This can result in nutrient loss (e.g. the loss of nitrogen gas through denitrification) or render nutrients less available for plant growth (eg binding of phosphates by iron in the soil) (Naiman and Decamps 1997; Narumalani et al. 1997). Biologically, riparian zones reduce nutrient concentrations through the assimilation of nutrients into plant material or through microbial immobilisation (Narumalani et al. 1997). These are however only temporary stores from which nitrogen and phosphorus may be released. Nitrogen can be permanently removed through denitrification, particularly in slow flowing subsurface water with high organic matter loads and low oxygen (Dosskey 2001; Dosskey et al. 2010; Jordan et al. 1993).

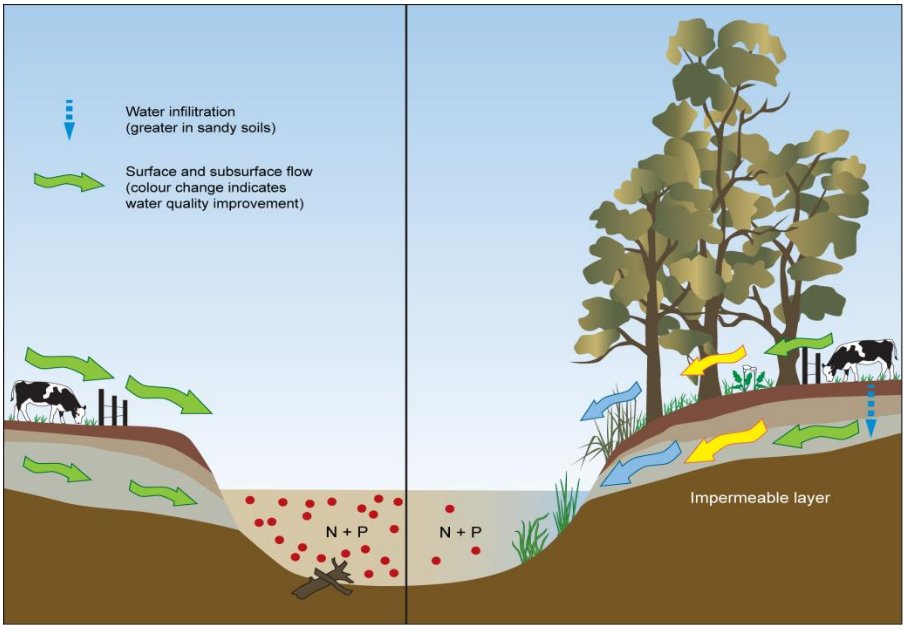


Figure 2-2. Conceptual model highlighting the benefits of riparian vegetation on surface and subsurface water quality and how it influences instream water quality

Riparian zones can be sinks or sources of nutrients. Riparian vegetation areas can adsorb nutrients in runoff through infiltration, particularly in regions with permeable soils, such as floodplains or alluvial pans (Puigdefabregas et al. 1998; Herron and Wilson 2001). This reduces delivery of nutrients to streams by filtering particulate matter and reducing surface runoff, thereby decreasing sediment and dissolved nutrient movement (Vought et al. 1994; Narumalani et al. 1997; Tabacchi et al. 1998, 2000; Knight et al. 2010). However riparian zones can be sources of nutrients when saturated soils lead to subsurface flows intercepting soil nutrients and conveying high nutrient flows to streams (Huang and Laften 1996).

This chapter determines whether the riparian zone intercepts the main transport pathway of nutrients, the passage of water from the paddock to the stream. The hydrology of two sites with different slope and soil type is compared: the Bingham Creek site on flat, deep sands and the Lennard Brook site, also on sands but with greater slope and better soils. This chapter also investigates how nutrient concentrations vary spatially and temporally in groundwater. The key questions addressed in this chapter are:

- **How does hydrology in sandy soils influence the movement of nutrients to the stream?**
- **Are nutrients moving through the groundwater and bypassing riparian buffers?**

The hypothesis that is being tested in this chapter is:

Sandy soils with no slope and which lack a deep impermeable layer will exhibit little horizontal movement of water from the adjacent paddocks through the root zone of the riparian vegetation.

Methods

Piezometers

In April 2011, 39 piezometers were installed at the Bingham Creek site, in the paddock and riparian zone, using a hand auger. Piezometers were all 50mm diameter PVC pipe, slotted through the bottom 1m. Two piezometer nests were installed (three replicates at each depth), at 1.5 and 2.5 m depth, at 1, 2, 4, 8, 16, 32 and 64m distance from the stream. At 64m the 2.5 m piezometers could not be installed due to saturation of the soil. In April 2012, an additional depth of 0.5 m was added (three replicates at each distance), together with a full nest of piezometers (0.5, 1.5, 2.5 m) at 96m. One piezometer was installed at 2.5 m depth at 96m but no others could be installed due to soil saturation. With difficulty, due to soil saturation, two piezometers (no replicates) were installed at 0.5 and 1.5 m depths in the streambed, to compare groundwater in the riparian zone and underlying the stream.

It was intended to have an equal number of piezometers installed at the Lennard Brook site, however, due to the saturated soil profile, this was not possible. Piezometers were installed in April, 2012. All three replicates of the 0.5 m depth piezometers were installed at 1, 2, 4, 8, 16, 32, 64 and 96m from the stream. Only four 1.5 m piezometers were able to installed, one replicate at 16m and three replicates at 32m due to saturated soil profile and the proliferation of laterite in the soil column (paddock only). A mechanical auger was used in June 2012 to add nine 2.5 m (three replicates at 32, 64 and 96m) and six 1.5 m piezometers (three replicates at 64 and 96m). None could be installed in the stream due to high flows.

Groundwater sampling

To ensure a representative water sample, piezometers were first purged before sampling. Purging was done by removing three casing volumes of water, using a submersible bilge pump. Before purging the depth to groundwater was measured and following purging physicochemical readings were taken using a YSI multi-parameter probe (YSI 556MPS). Parameters sampled included pH, DO (mg/l and %), temperature (C^o), redox potential, salinity and conductivity. A water sample was also collected to determine turbidity (Hach 2100P Turbidimeter). To determine the mean pH, values were returned to their hydrogen ion concentration $[H^+]$, whereupon the mean value was calculated. Values were then returned to pH by substituting the hydrogen ions into $\log_{10}[H^+]$.

Once piezometers recharged (one days after purging), depth to groundwater was measured. Grab samples were taken from piezometers using a dropper. Samples were then processed as follows: Samples for filterable reactive phosphorus (FRP), oxidised-nitrogen (NO_X-N), ammonium-nitrogen (NH_4-N) and dissolved inorganic carbon (DIC) were filtered through a 0.45µm, and for gilvin (colour) 0.2µm, millipore filters. Samples for total phosphorus (TP) and total nitrogen (TN) were not filtered. Samples were bottled, stored on ice and returned to the laboratory on the same day. To represent key hydrological events, groundwater samples were collected from both sites, four times in May, June, August and November. These times represented baseflow, first flush, wetted profile/raining and wetted profile/not raining, respectively (Table 2-1). An additional sample from Bingham Creek was taken in July, to determine whether nutrients from a fertiliser applied to the paddock were entering the groundwater. The fertiliser that was applied was CSBP Super:Potash 3:1, 6.8% P, 12.4% K, 7.9% S, 15% Ca, which was applied at 40kg/Ha.

Table 2-1. Groundwater sampling dates

Location	Date	Sampling period
Bingham	9.7.11	First flush
Bingham	18.8.11	Wetted profile raining
Bingham	13.10.11	Wetted profile not raining
Bingham	21.5.12	Baseflow
Bingham	18.6.12	First flush
Bingham	23.7.12	Fertiliser application
Bingham	23.8.12	Wetted profile raining
Bingham	19.11.12	Wetted profile not raining
Lennard	24.5.12	Baseflow
Lennard	21.6.12	First flush
Lennard	24.8.12	Wetted profile raining
Lennard	20.11.12	Wetted profile not raining

Purging physico-chemical groundwater validation

To ensure representative physico-chemical and nutrient concentration data was being collected, a purging trial was undertaken on one row of nested piezometers (1.5 and 2.5 m). This assessed an appropriate timeframe between purging and collecting groundwater for nutrient analysis. Before the piezometers were purged physico-chemical readings were taken using a YSI multi-parameter probe (YSI 556MPS). Purging was then undertaken using a submersible bilge pump, with three casing volumes of water being removed. Physico-chemical data was collected immediately after purging, the following day and then three days later.

Groundwater purging validation

The pH, conductivity and dissolved oxygen content of shallow and groundwater typically returned to their pre-purging levels after one day (Figure 2-3b-d), with similar results three days after purging. Redox potentials took three days to return to previous levels (Figure 2-3a). There was greater variability in all variables following purging at 1.5 m than at 2.5 m. From this a decision was made to take samples one days after purging.

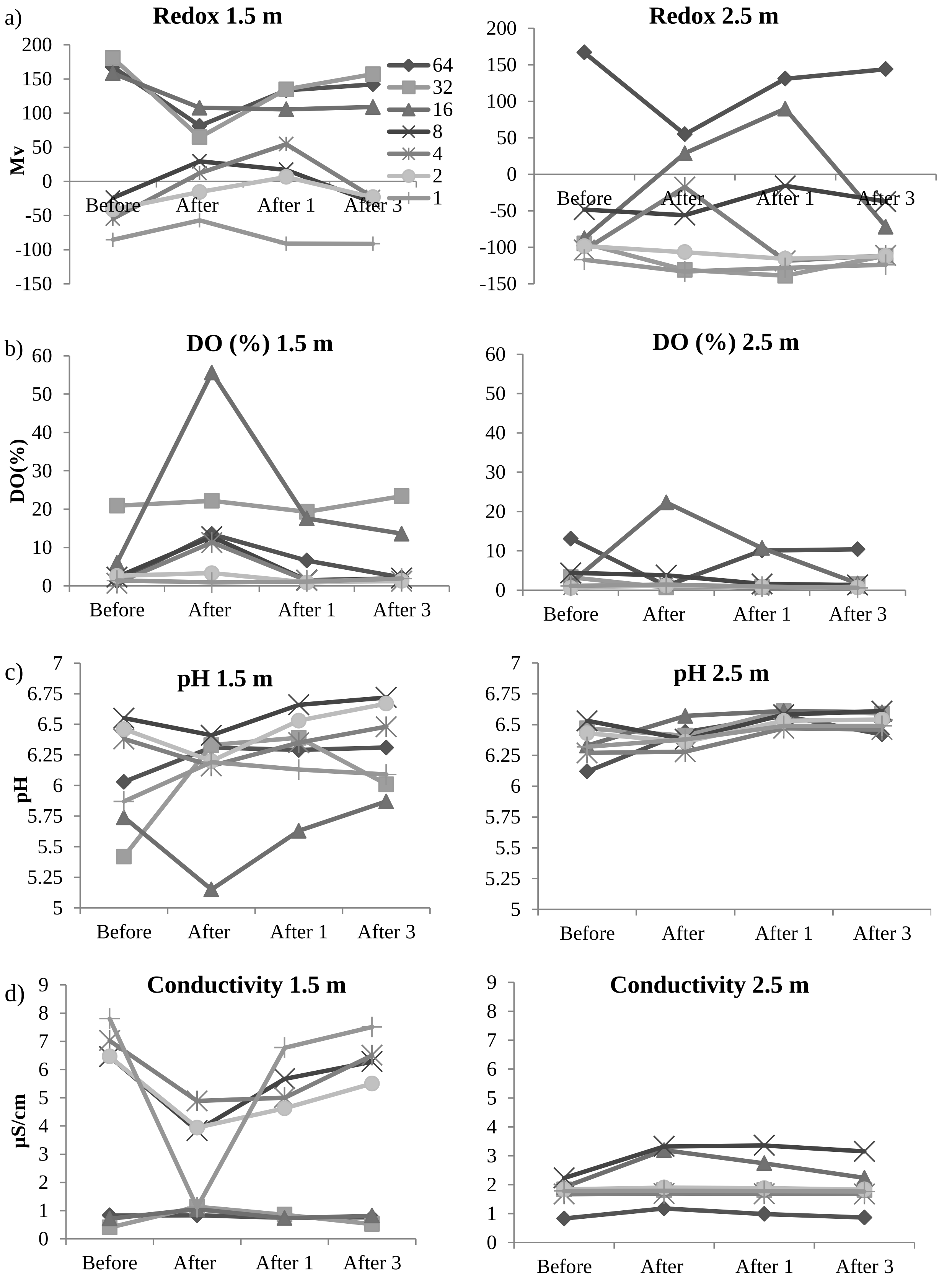


Figure 2-3. Variations in a) redox potential, b) dissolved oxygen content (%), c) pH and d) conductivity between shallow and deep groundwater during a trial, showing how parameters change after purging (after= directly after, after 1= one day, after 3= three days)

Stream water sampling

Grab samples were taken from the stream at the same times as groundwater sampling, as well as opportunistic sampling during any other sites visits, to maximise frequency of sampling while the streams were flowing. Stream profiles were determined in April 2012. Stream depth and velocity were measured each time a sample was taken to determine discharge and nutrient load. The method used was adapted from Mitchell and Stapp (1994):

$$Flow = ALC/T$$

Where A= Average cross sectional area of stream

L= Length of stream reach assessed

C= Correction factor (0.8 for rocky-bottom streams or 0.9 for muddy-bottom streams)

T= Time

Water quality analysis

All samples were frozen on arrival back to the Marine and Freshwater Research Laboratory (NATA accredited No. 10603). All samples were analysed for total phosphorus (TP), total nitrogen (TN), filterable reactive phosphorus (FRP), nitrate-nitrite-nitrogen or oxidised nitrogen (NO_x-N), ammonium-nitrogen (NH_4-N) and dissolved organic carbon (DOC) according to the methods listed in Table 2-2. Stream samples were also analysed for chlorophyll *a* on six occasions in 2011 and two in 2012 (Table 2-2). A one off sample of reactive iron was taken from Bingham Creek and Lennard Brook during baseflow groundwater sampling (May 2012).

Samples for gilvin and fulvics:humics ratio (E4:E6 ratios) were kept refrigerated and in the dark before analysis. Gilvin concentrations were analysed using a dual beam mass spectrophotometer (Novaspec©II) to determine absorbance at 440nm on filtered samples. Values were multiplied by 2.303 x100 to determine the gilvin concentration (Kirk, 1994). E4:E6 ratios compared absorbance at wavelengths of 465nm and 665nm respectively. The E4:E6 ratio can be used to distinguish the relative proportion of humic and fulvic acids. The ratio represents the extent of decomposition of humic substances, with high ratios indicating more recent degradation (i.e. a higher proportion of fulvic acids) (Wrigley et al. 1988).

Chloride sampling

Stored groundwater samples from 23/7/12 for Bingham Creek and 21/6/12 for Lennard Brook were analysed for chloride to determine the suitability for using chloride as a groundwater tracer. Results illustrated that concentrations were too variable and, at places, too high for chloride to be used effectively as a tracer.

Table 2-2. Methods for water quality analysis

Parameter	Units and DL (if applicable)	Method	Lab
Water samples			
TP	µg.P/L (<25)	Colorimetric analysis following digestion of potassium persulphate (Valderrama 1981)	MAFRL
TN	µg.N/L (<250)	Colorimetric analysis following digestion of potassium persulphate (Valderrama 1981)	MAFRL
FRP	µg.P/L (<10)	Colorimetric analysis following addition of molybdate and ascorbic acid reagents (Johnson 1982)	MAFRL
NOx	µg.N/L (<10)	Colorimetric analysis, following nitrate reduction under acidic conditions (Johnson 1983)	MAFRL
DOC	mg.C/L (<0.5)	Automated combustion followed by NDIR method (APHA 2005)	MAFRL
NH_4	µg.N/L (<15)	Colorimetric analysis following the addition of phenolate (Switala 1993)	MAFRL
Chlorophyll a	ug.L (<0.1)	Colorimetric analysis following the addition of acetone (APHA 2005)	MAFRL

Data analysis

A few water quality values fell below the detection limit and so zeroes were substituted for those values. Single factor ANOVAs and Tukey's pairwise tests were used to differentiate between groundwater depths (two or three levels: 0.5, 1.5 or 2.5 m) for concentrations of each nutrient species and stream separately, using distances and times as replicates. Statistical analyses passed the Levene's test of homogeneity.

The two sites were chosen so that sloped and flat geomorphologies could be compared, as this provides a comparison between the sloped model (Lennard Brook) and the non-traditional (flat, deep sand) model (Bingham Creek). It was planned to have an equal number of piezometers at both of the sites, however due to the hydrological nature of both sites and the slope it was not possible. Therefore the capacity for analysis between sites was limited. Due to missing replicates of piezometers at the Lennard Brook site, only the 0.5 m depth data could be analysed statistically. For the 0.5 m depth, a three factor analysis of variance (ANOVA) analysis was carried out on sites that had water present (1-4m distance). The three fixed factors were Site (two levels: Bingham Creek or Lennard Brook), Distance (three levels: 1, 2 and 4m) and Time (three levels: June, August and November 2012). All nutrients and DOC were used as dependent variables. A two factor ANOVA was carried out at the 1.5 m depth, 32m from the stream, for the factors were Site (same two levels) and Time (same three levels). All nutrients and DOC were used as dependent variables. Principal components analysis (PCA) was used to illustrate the difference between groundwater depths based on their chemical composition for each stream separately.

Results

Rainfall

Rainfall varied considerably over the two sampling years and neither followed the long-term average monthly rainfall patterns (Figure 2-4a-b). The long-term average at the Bingham Creek site is 655.5mm, while in 2011, 638.6mm of rain fell and only 544.5mm in 2012. The average yearly rainfall for the Lennard Brook site is 738.7mm but only 636.6mm fell in 2012 (Figure 2-4a-b).

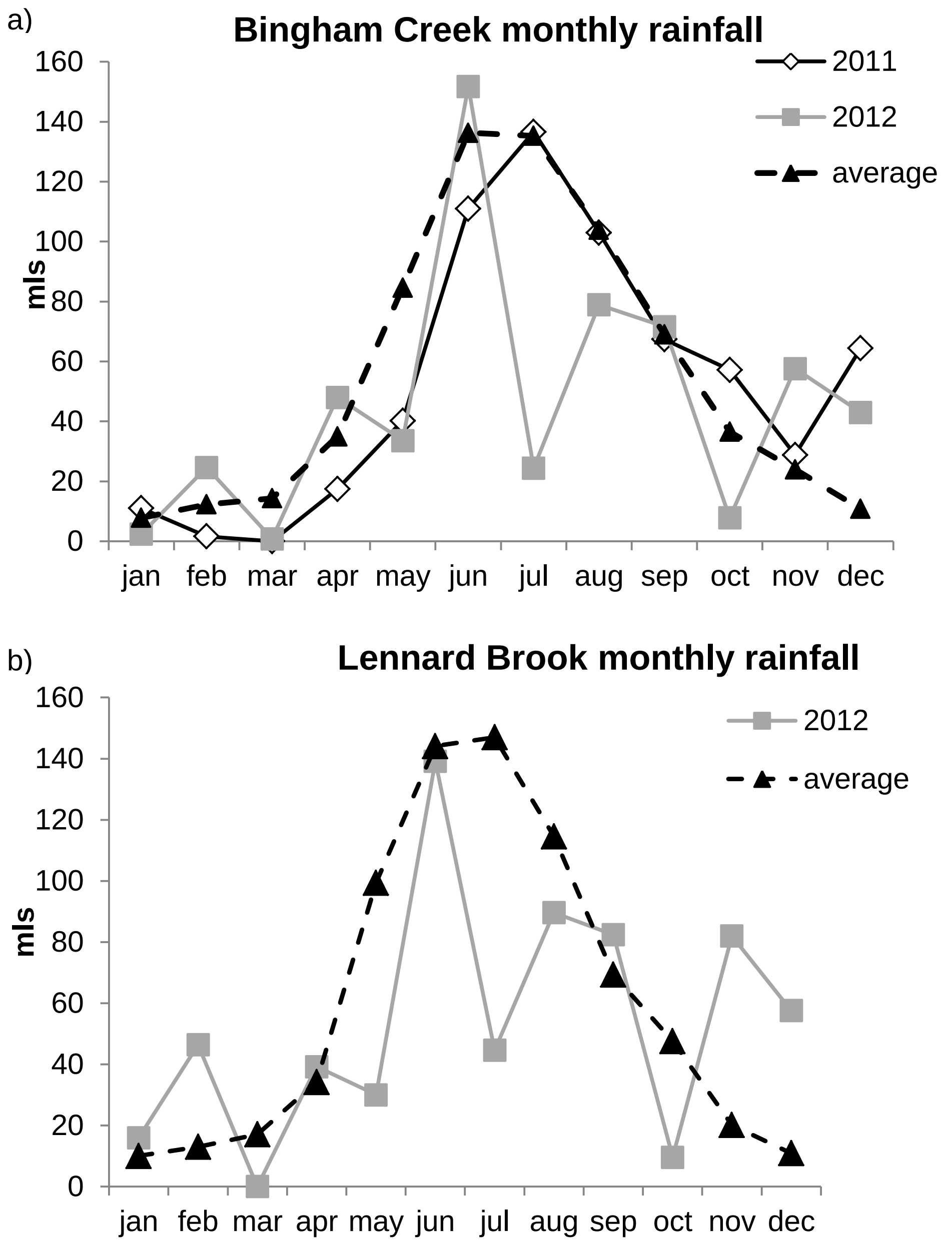


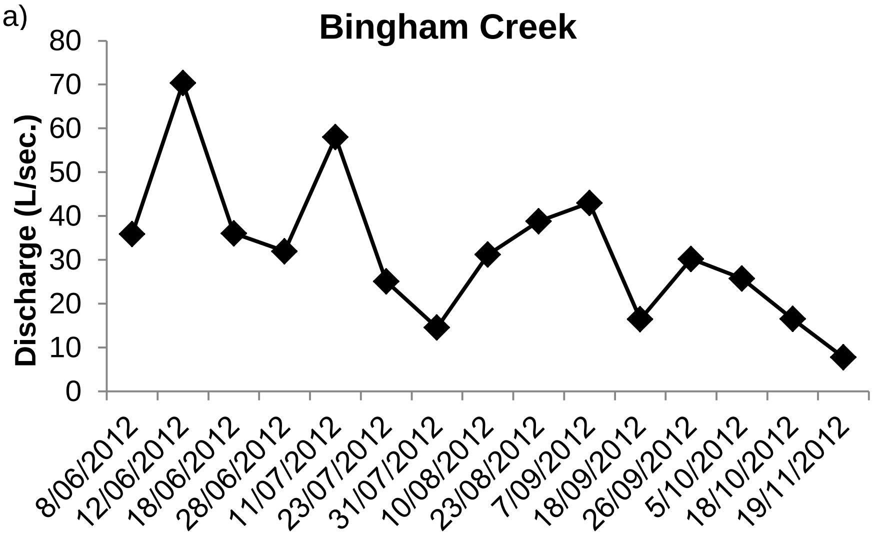
Figure 2-4. Comparison of long-term average monthly rainfall totals between a) Bingham Creek and b) Lennard Brook over the two sampling years

Nutrients and flow in the streams at Bingham Creek and Lennard Brook

Stream Flow

Discharge rates were similar for both streams, although they varied considerably over time (Figure 2-5a-b). Bingham Creek, the intermittent stream, had greater variation in discharge being strongly influenced by rainfall. Lennard Brook is perennial and discharge remained relatively high and constant over the sampling period, decreasing only when rainfall ceased in November (Figure 2-5a-b).

a)



b)

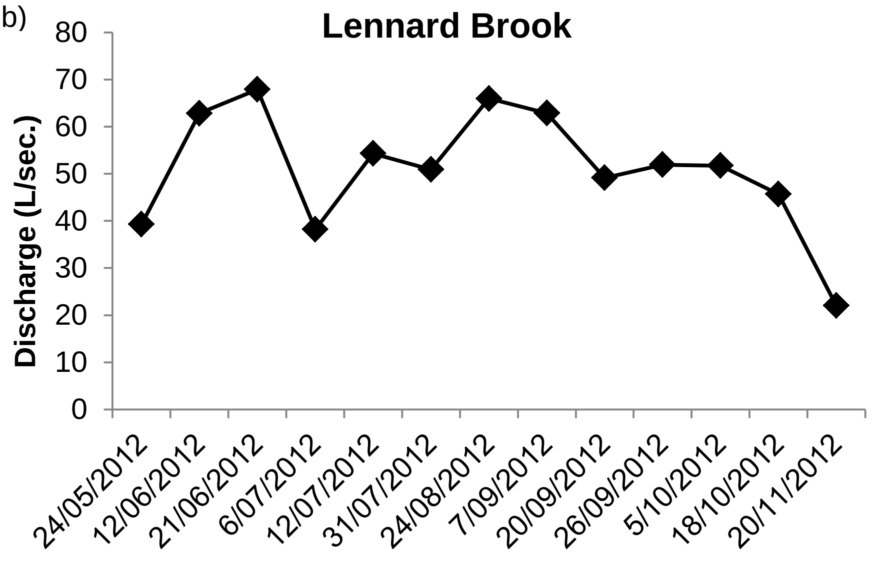


Figure 2-5. Comparison of stream discharge between a) Bingham Creek and b) Lennard Brook over an extensive sampling event

Stream nutrient concentrations

Total phosphorus (TP) and filterable reactive phosphorus (FRP) concentrations were higher and more variable at Bingham Creek. At Bingham Creek ~80% of TP consisted of FRP, which ranged from 600 to 900 µg P/L^{-1}. Whereas, at Lennard Brook, FRP only contributed ~60% and concentrations were consistently very low ranging from 10-20 µg P/L^{-1} (Figure 2-6a-b).

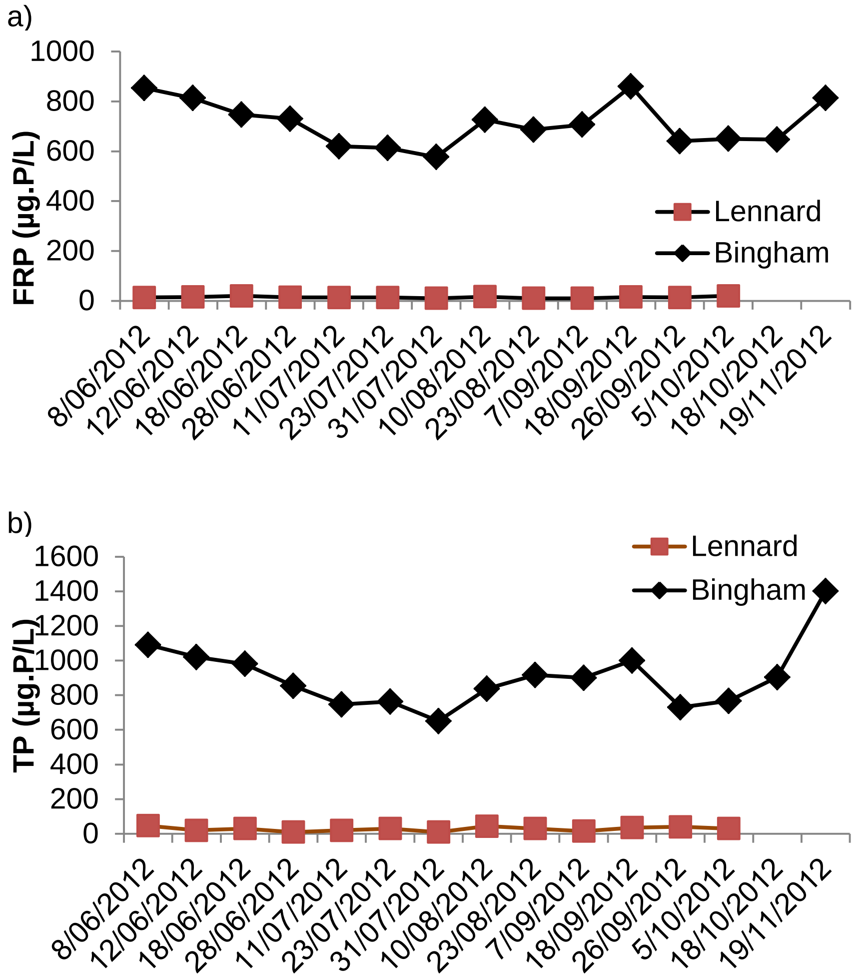


Figure 2-6. Comparison on instream a) filterable reactive phosphorus and b) total phosphorus concentrations between Bingham Creek and Lennard Brook over time

Oxidised nitrogen concentrations were fairly consistent over the sampling period, although concentrations were 40 times higher at Lennard Brook than Bingham Creek (Figure 2-7a). At Lennard Brook oxidised nitrogen concentrations exceeded ANZECC guidelines, whereas at Bingham Creek they were well below (Table 2-3). In contrast, total nitrogen and ammonium nitrogen concentrations were higher at Bingham Creek (Figure 2-7c-d), with ammonium concentration being well below ANZECC guidelines at both sites (Table 2-3).

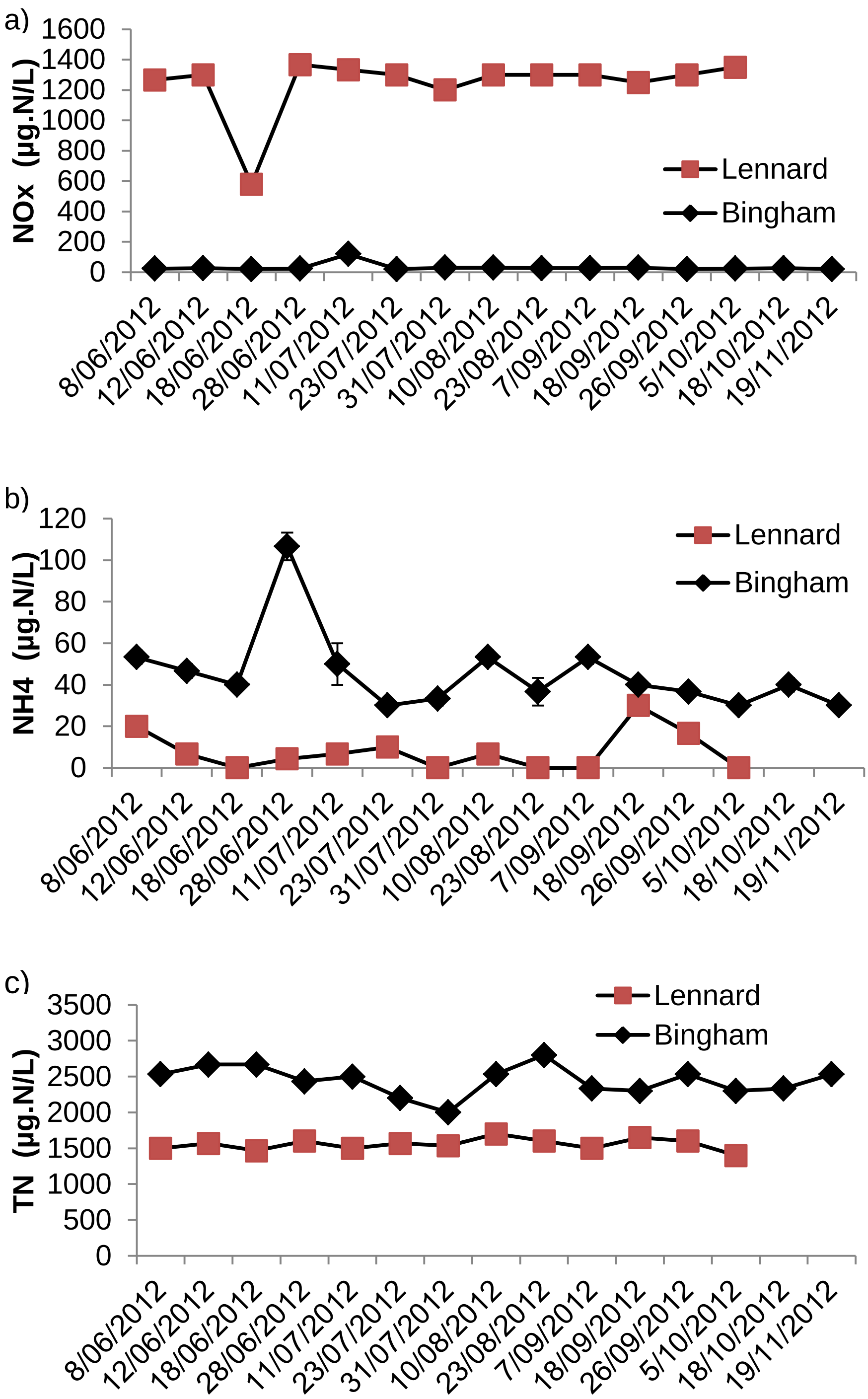


Figure 2-7. Comparison on instream a) oxidised nitrogen, b) ammonium and c) total nitrogen concentrations between Bingham Creek and Lennard Brook over time

Table 2-3. Comparison of ANZECC guidelines for lowland streams with instream nutrient concentrations for Lennard Brook and Bingham Creek. Highlighted values indicate concentrations higher than trigger values

	FRP	TP	NOx-N	NH_4-N	TN
Ecosystem type	(µg. P/L)	(µg. P/L)	(µg. N/L)	(µg. N/L)	(µg. N/L)
Lowland River	40	65	150	80	1200
Lennard Brook (Av)	14.15	27.27	1231.52	7.06	1546.97
Bingham Creek (Av)	722.05	914.36	32.05	46.9	2464.1

The factor driving stream nutrient load differed between Bingham Creek and Lennard Brook. At Bingham Creek nutrient loads were strongly correlated with flow (discharge), with the exception of oxidised nitrogen, which had a strong correlation between load and concentration (Table 2-4). At Lennard Brook, nutrient loads were not correlated with either flow or concentration for filterable reactive phosphorus, oxidised or total nitrogen. The remaining nutrients had strong positive correlations to concentration. The link between load and discharge at Bingham Creek is likely a function of its intermittent nature, where instream nutrient concentrations are consistent and load is dependent on the fluctuating flow rates. Whereas the perennial Lennard Brook has fairly stable flows and therefore nutrient concentrations were the primary influence on stream nutrient loads (Table 2-4).

Table 2-4. Comparison of the correlation of load:discharge and load:concentration between Bingham Creek and Lennard Brook.Values that are highlighted indicate a significant correlation

	Bingham Creek		Lennard Brook	
	Load:Discharge	Load:Concentration	Load:Discharge	Load:Concentration
	R^2	R^2	R^2	R^2
FRP	0.942	0.045	0.214	0.277
TP	0.916	0.004	0.0315	0.688
NOx-N	0.378	0.945	0.063	0.110
NH_4-N	0.689	0.536	0.001	0.976
TN	0.986	0.327	0.058	0.278
DOC	0.990	0.195	0.235	0.920

The hydrology of groundwater at Bingham Creek and Lennard Brook

Slope and groundwater levels over time

The slope from paddock to stream at Bingham Creek was negligible (slope = 0.016, a rise of less than 1.5 m over the 96 m transect) while Lennard Brook was relatively steep especially in the paddock (slope =0.099, ~9.5 m over the 96 m transect -Figures 2.8a, b).

There was a greater variation over time in the depth to groundwater at Bingham Creek compared to Lennard Brook where groundwater levels remained fairly constant in the riparian zone (0.1-0.3m beneath the surface) fluctuating only 0.2m at any depth over the study period. Due to the steep slope at Lennard Brook, piezometers did not intercept the water table at 96m and so no values are shown (Figure 2-8b).

At Bingham Creek, the lowest groundwater levels occurred during baseflow, with maximum depth to groundwater occurring at 16m (1.5 m) to a minimum of 0.9 m at 96 m. The elevation of the water table decreased from 0 m to 4 m distance along the transect, resulting in a negative slope away from the stream. The water table then became closer to the surface from 4 m to the end of the transect in the paddock (Figure 2-8a). After the stream flowed (first flush), groundwater levels rose towards the surface, although there was still a negative slope away from the stream up to 16m along the transect within the riparian zone. With subsequent time and rainfall, groundwater levels increased and peaked during the wet season (raining), before subsequently declining again (not raining). During this time, the slope of the water table declined from the paddock down to the stream (Figure 2-8a). In the paddock the water table was always below 0.5 m elevation (i.e. 0.5 m piezometers were consistently dry). In the riparian zone the water table had a maximum elevation of 0.23m.

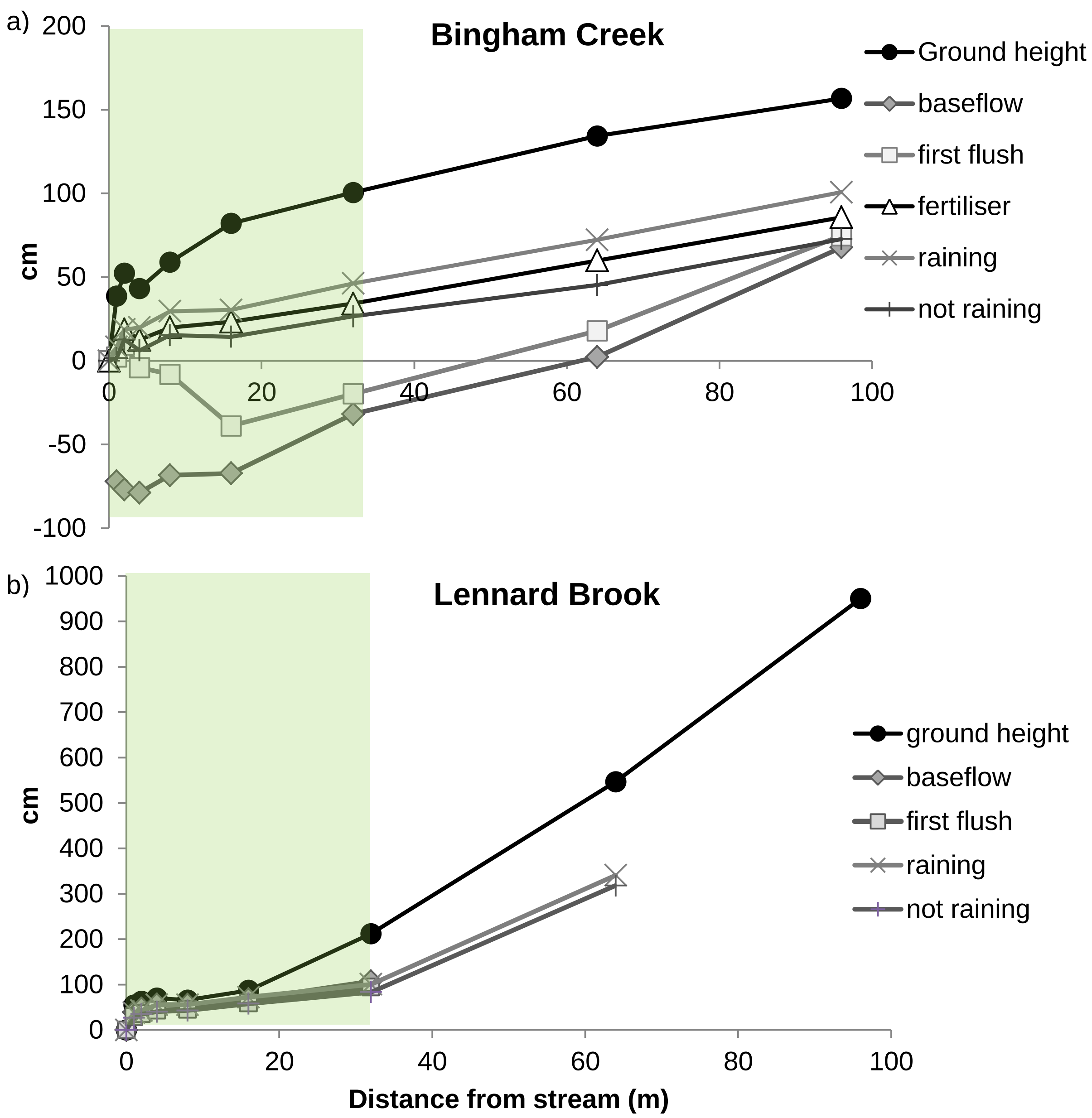


Figure 2-8. Change in depth to water in relation to ground height over five sampling periods at a) Bingham Creek and b) Lennard Brook, highlighted section shows riparian zone

Physio-chemical changes in groundwater

The redox potential of shallow and deep groundwater remained negative and relatively constant throughout the riparian zone but was higher and mostly positive in the paddock. Depth was an important factor, redox potentials decreasing with depth in the riparian zone (Figure 2-9a-b). At Lennard Brook, redox potentials were higher in the paddock than the riparian zone, but were positive throughout and substantially higher than at Bingham Creek (Figure 2-9c).

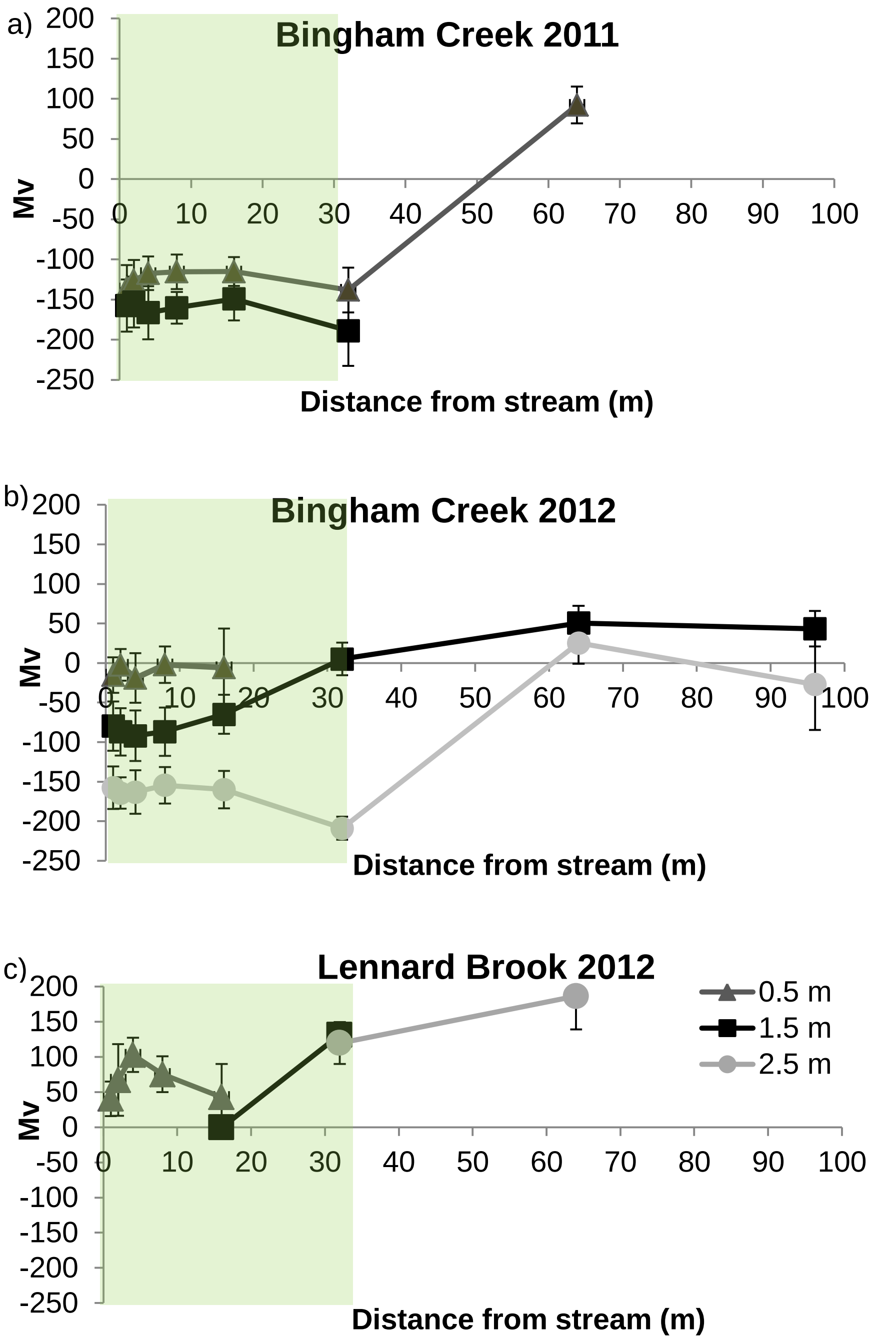


Figure 2-9. Comparison of redox potentials at a) Bingham Creek 2011, b) Bingham Creek 2012 and c) Lennard Brook along a gradient from the stream to the paddock, the shading represents the riparian zone

At Bingham Creek, the redox potential at 1.5 and 2.5 m depth varied greatly over time, but followed the same trends across the paddock and riparian zone. In 2011 it decreased after the onset of the wet season, whereas in 2012 it showed greater variability. Depth and zone were important factors with redox potentials being higher in shallower groundwater and lower in the riparian zone (Figure 2-10a-b). At Lennard Brook redox potential remained consistent and positive for most time periods and depths (Figure 2-10c). After the wet season (not raining) groundwater first appeared in the 2.5 m piezometers (only located in the paddock) and being at the water table had a positive redox potential. In contrast, water at 0.5 m was only present in the permanently flooded riparian zone and redox decreased as a result of long-term waterlogging.

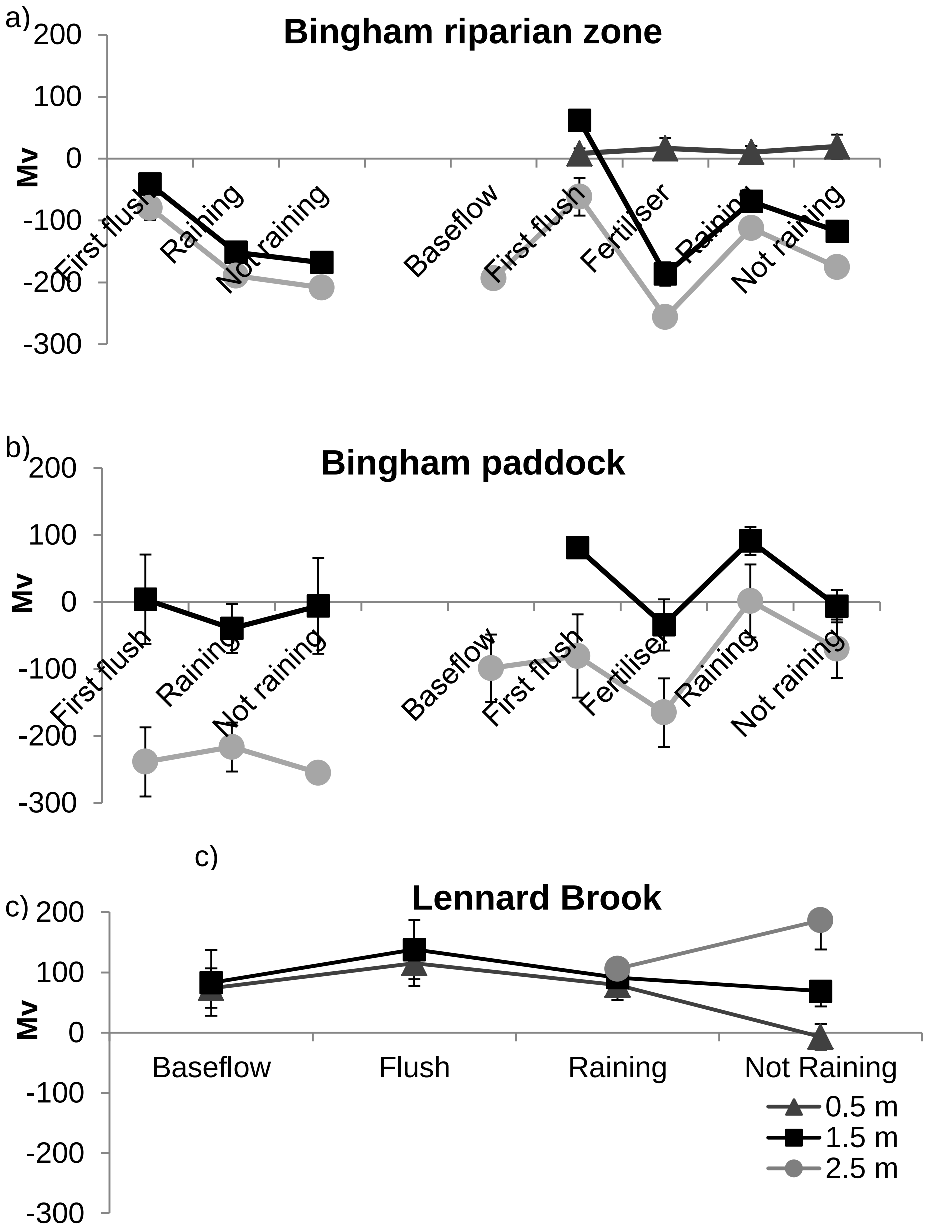


Figure 2-10. Comparison of redox potentials over time and depth at Bingham Creek a) riparian zone, b) paddock and c) Lennard Brook, note data for Bingham Creek is over two years

Dissolved oxygen concentrations at Bingham Creek were consistent across 2011 and 2012, with highest concentrations occurring in the paddock at 64 and 96 m, before decreasing through the riparian zone to the stream (Figure 2-11a-b). Depth was an important factor, with concentrations decreasing with depth. At Lennard Brook a similar pattern occurred with dissolved oxygen concentrations decreasing towards the stream, although concentrations at 0.5 m depth were more variable (Figure 2-11c).

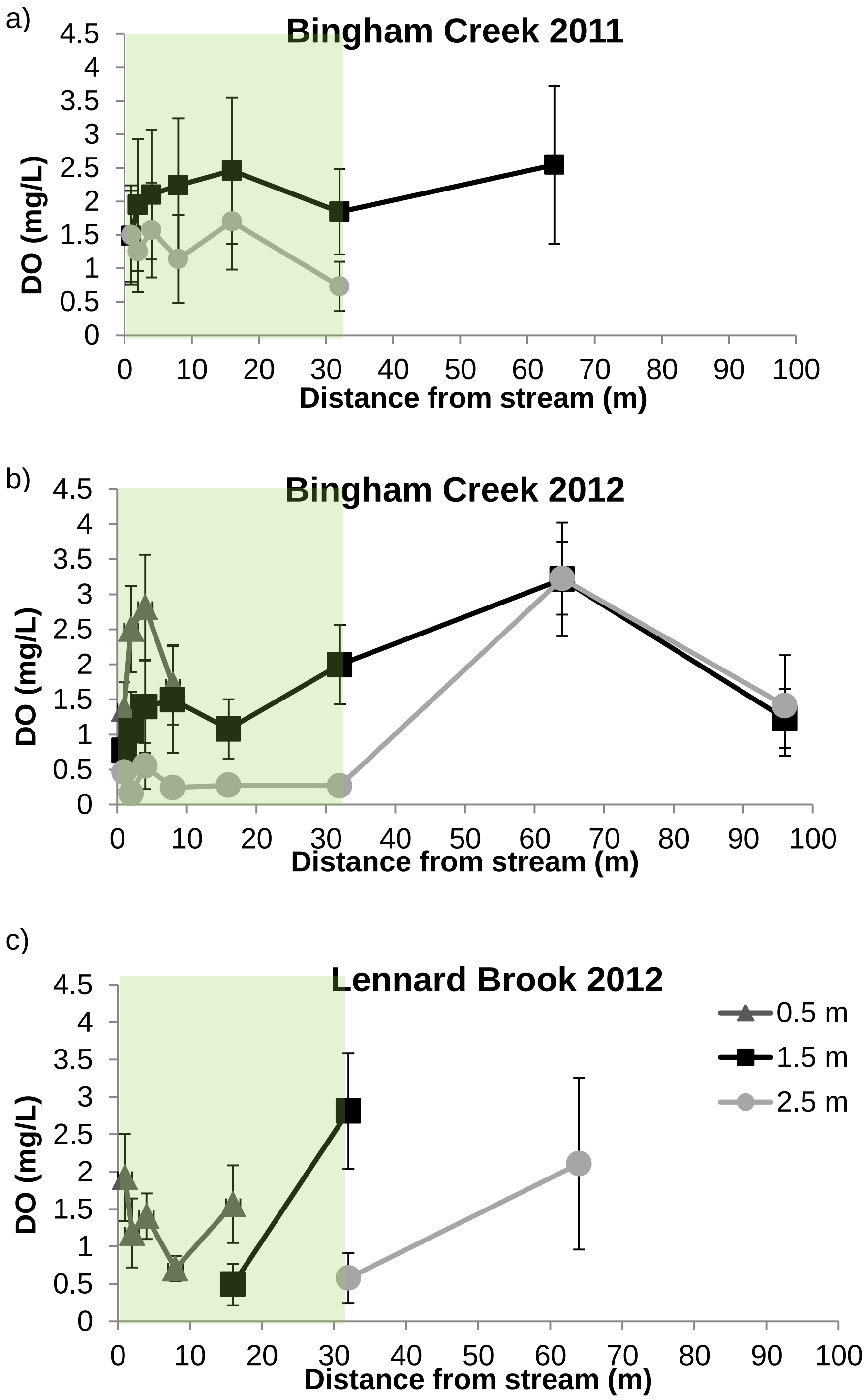


Figure 2-11. Comparison of dissolved oxygen concentrations at a) Bingham Creek 2011, b) Bingham Creek 2012 and c) Lennard Brook along a gradient from the stream to the paddock, the shading represents the riparian zone

Dissolved oxygen concentrations in shallow and deep groundwater at Bingham Creek changed markedly over time, with concentrations peaking at the first flush and then decreasing (Figure 2-12a-c). This trend was consistent over both years and occurred in both the paddock and riparian zone. Dissolved oxygen concentrations were higher in the paddock than the riparian zone and decreased with depth (Figure 2-12a-b). A similar trend occurred at Lennard Brook.

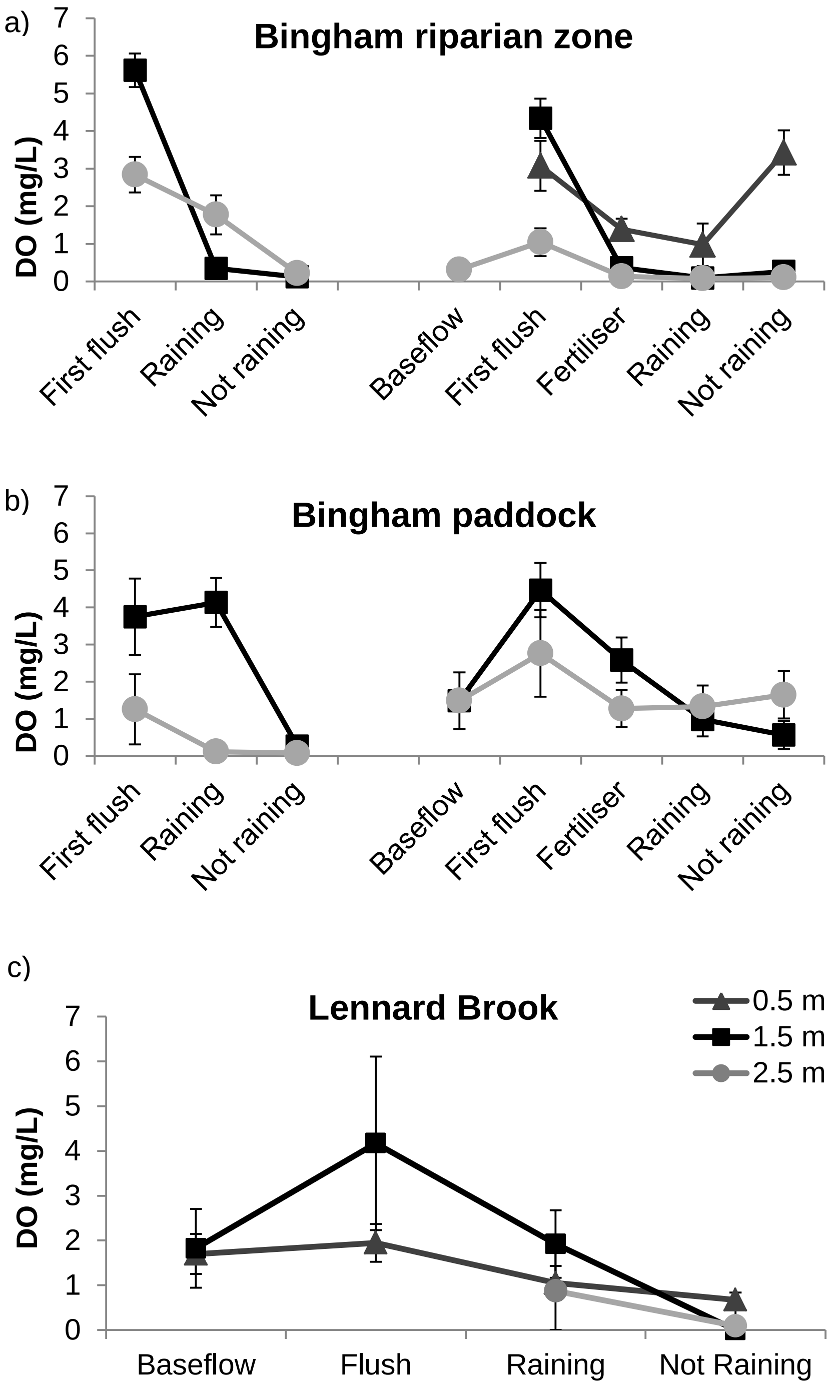


Figure 2-12. Comparison of dissolved oxygen concentrations over time and depth at Bingham Creek a) riparian zone, b) paddock and c) Lennard Brook, note data for Bingham Creek is over two years

The pH at Bingham Creek remained constant (pH 6-7) across the transect (Figure 2-13a-b), over time (Figure 2-14 a-b) at both 1.5 and 2.5 m depths in both 2011 and 2012. The values of pH changed a little more over time at 0.5 m depth and were lower, at around 6. At Lennard Brook, pH was considerably lower, between 4 and 5, and there was no pattern across the transect or change over time (Figure 2-13 and 2.14c).

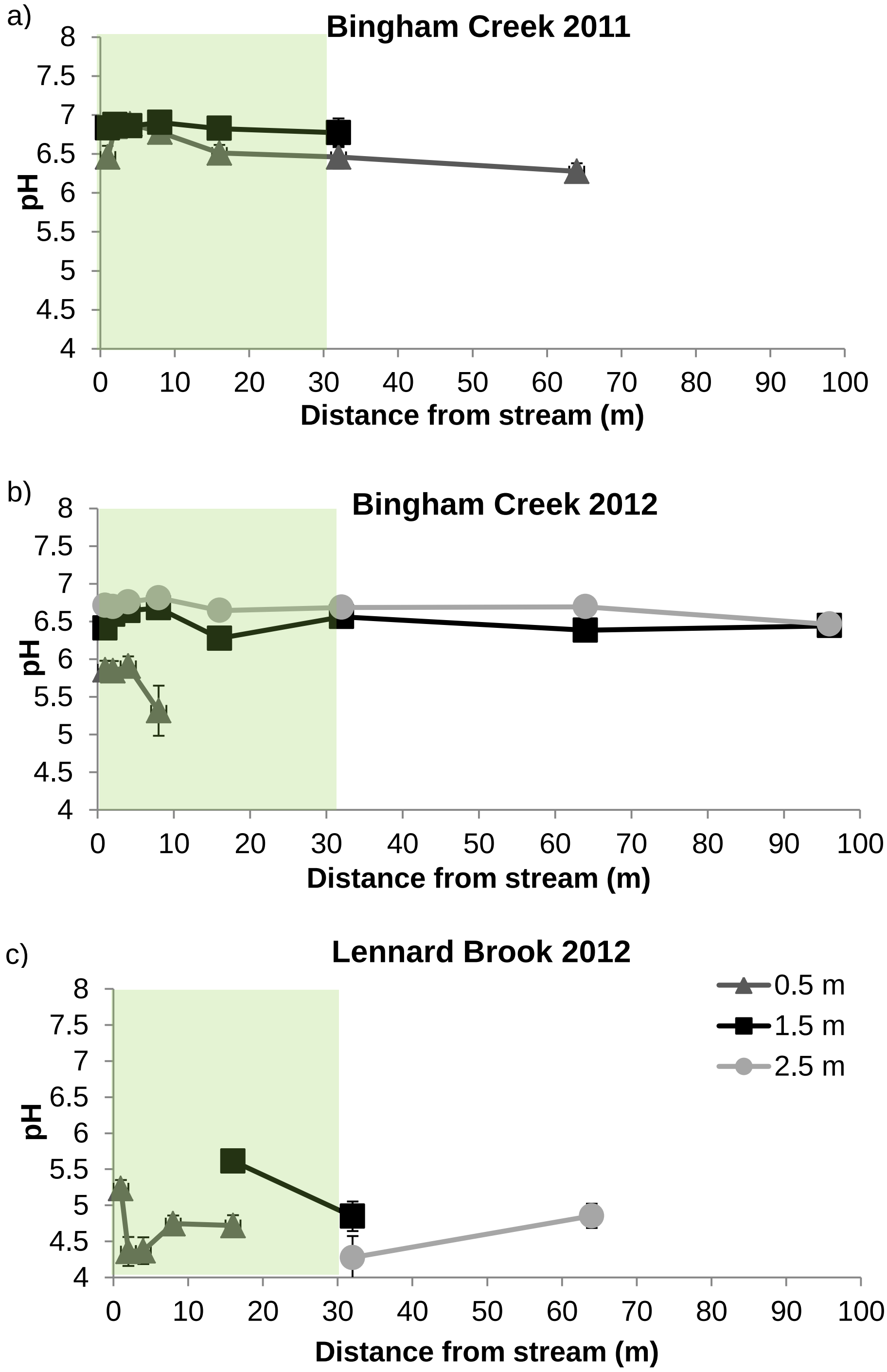


Figure 2-13. Comparison of pH at a) Bingham Creek 2011, b) Bingham Creek 2012 and c) Lennard Brook along a gradient from the stream to the paddock, the shading represents the riparian zone

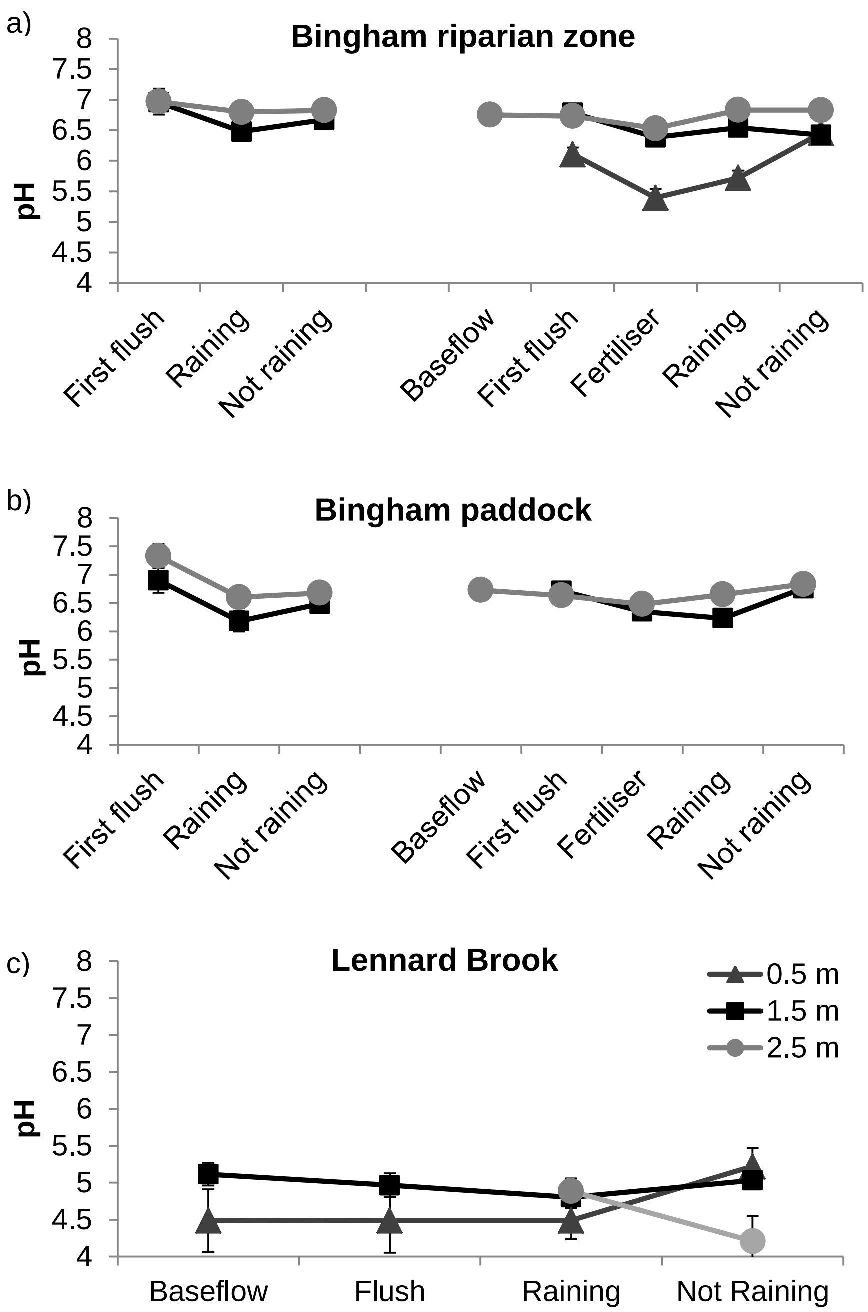


Figure 2-14. Comparison of pH over time and depth, at Bingham Creek a) riparian zone, b) paddock and c) Lennard Brook, note data for Bingham Creek is over two years

There is a consistent trend in temperature across Bingham Creek (both years) and Lennard Brook, with temperatures being highest in the paddock and then decreasing through the riparian zone towards the stream. Trends in temperature were consistent across all three depths, with the shallowest samples being the coolest (Figure 2-15a-c).

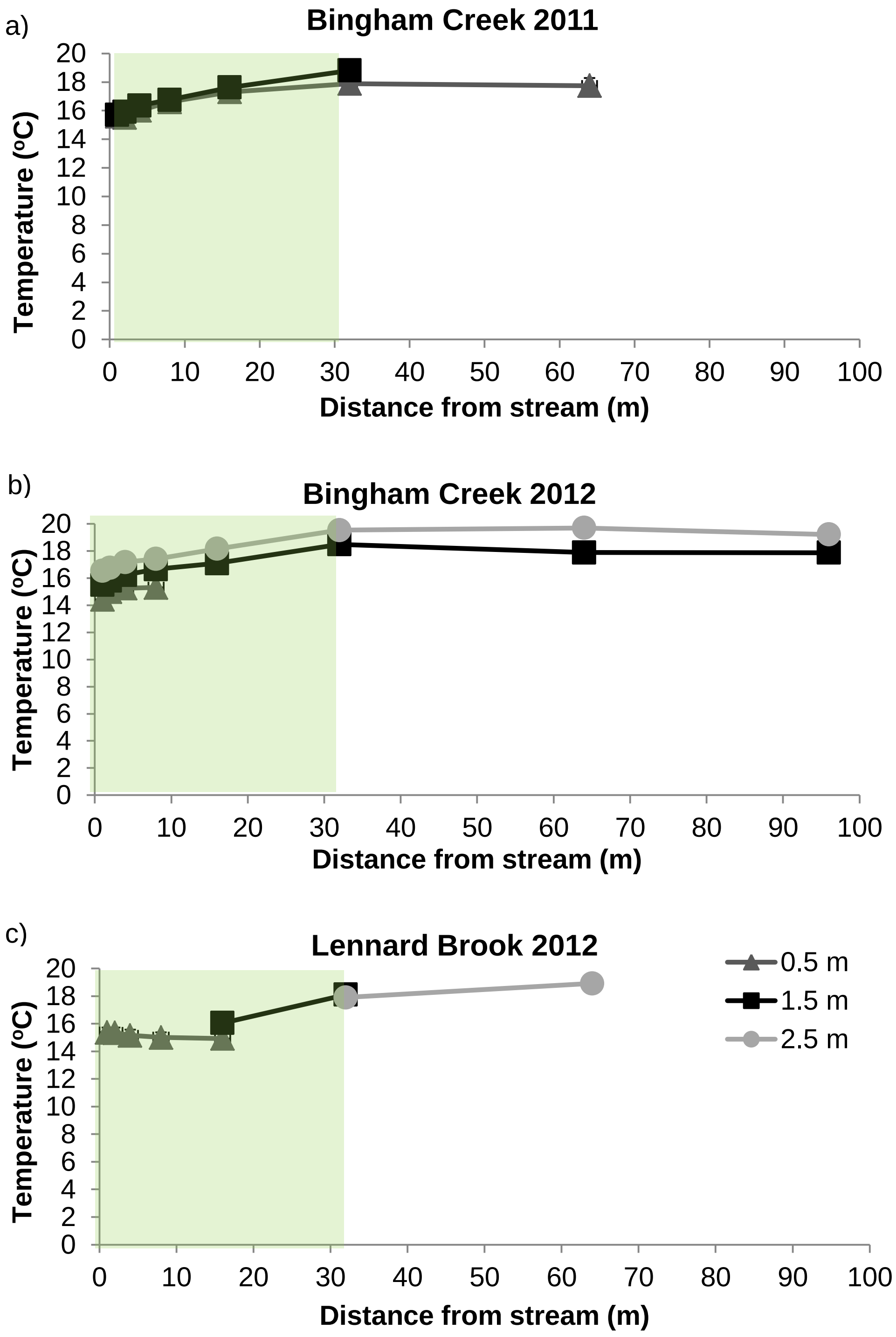


Figure 2-15. Comparison of temperature at a) Bingham Creek 2011, b) Bingham Creek 2012 and c) Lennard Brook along a gradient from the stream to the paddock, the shading represents the riparian zone

The temperature of groundwater showed a clear trend across each depth, with temperatures decreasing as the wet season progressed, before increasing again at the end of the wet season, in line with increasing air temperatures (Figure 2-16a-b). Depth was an important factor, with shallower groundwater being more readily influenced by air temperatures. Zone was also influential, with groundwater temperatures lower in the riparian zone. At Lennard Brook, a similar pattern occurred with groundwater temperatures falling then increasing with the finish of the wet season (Figure 2-16c).

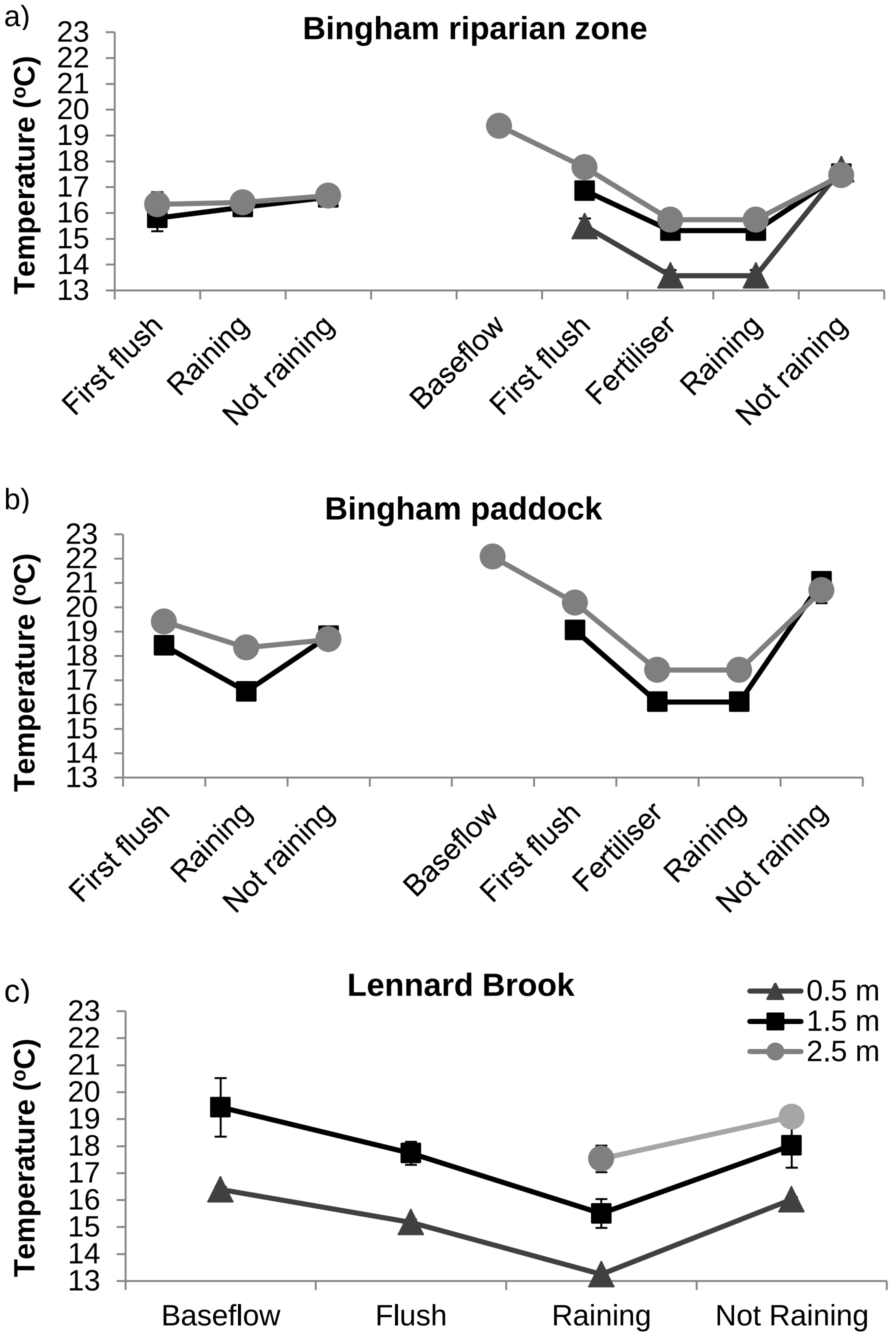


Figure 2-16. Comparison of temperature over time and depth at Bingham Creek a) riparian zone, b) paddock and c) Lennard Brook, note data for Bingham Creek is over two years

Carbon dynamics of stream and groundwater at Bingham Creek and Lennard Brook

Stream dissolved organic carbon concentrations and gilvin were more than ten times higher at Bingham Creek (DOC~60mg.C/L^{-1}) compared to Lennard Brook, which had little or no carbon (DOC~5mg.C/L^{-1}) (Figure 2-17a-b). E4:E6 ratios were consistent over time at Bingham Creek, whereas at Lennard Brook showed considerable variability. Overall the E4:E6 ratio was higher in Bingham Creek (Figure 2-17c).

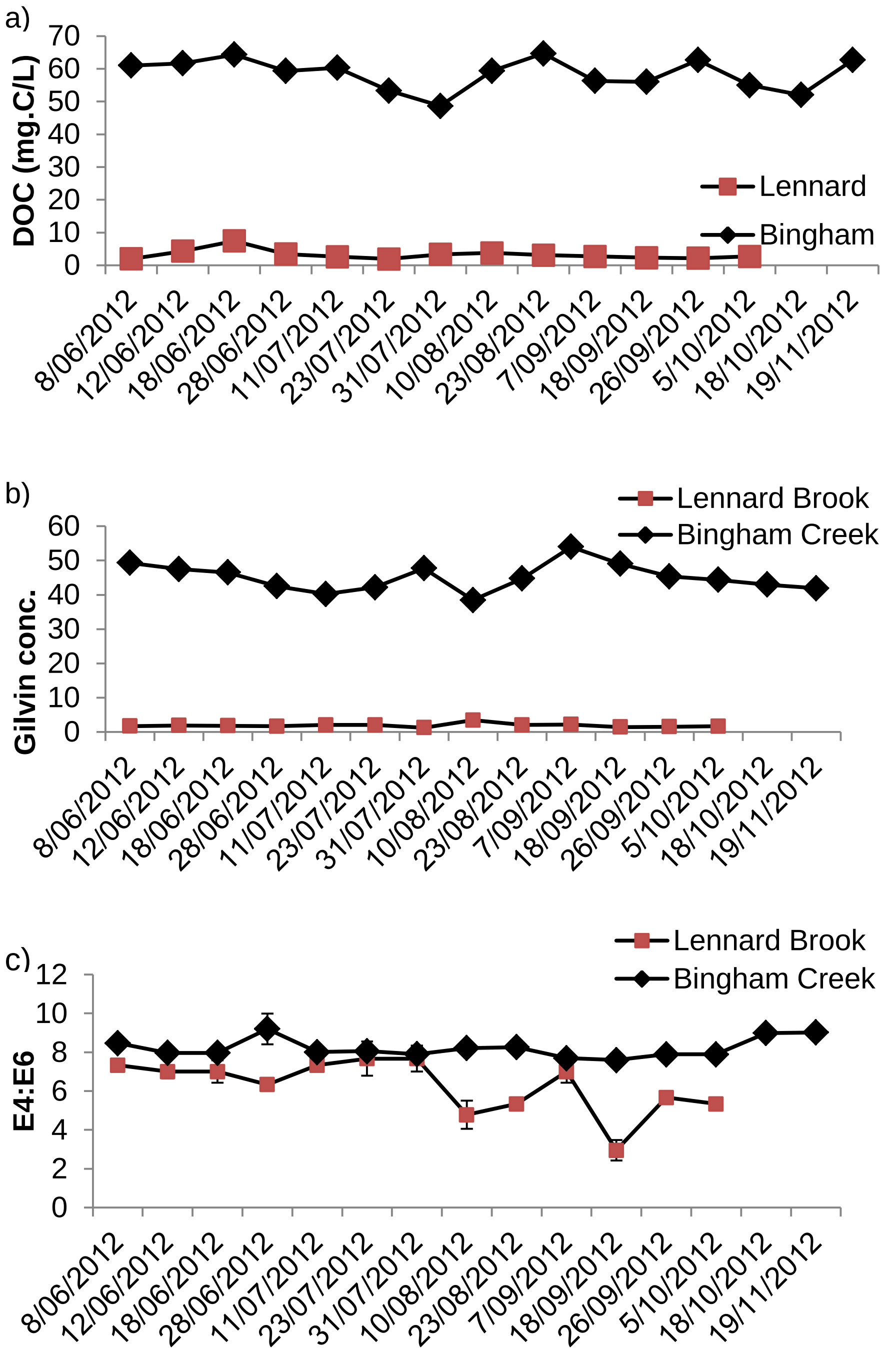


Figure 2-17. Comparison on instream a) dissolved organic carbon concentrations, b) gilvin and c) E4:E6 ratios between Bingham Creek and Lennard Brook over time

In 2011, total organic carbon concentrations were measured instead of dissolved organic carbon but these parameters were closely related (99.4% correlation). At Bingham Creek dissolved organic carbon (DOC) concentrations were low in the paddock and highest in the middle of the riparian zone, before decreasing towards the stream (Figure 2-18b). Depth was a factor, with concentrations higher in shallower samples. DOC concentrations in the stream were higher than at 2.5 m depth, and corresponded most closely to concentrations at 1.5 m depth. At Lennard Brook, DOC concentrations were highest in the riparian zone at 0.5 m depth, but were substantially lower than at Bingham Creek (Figure 2-18c).

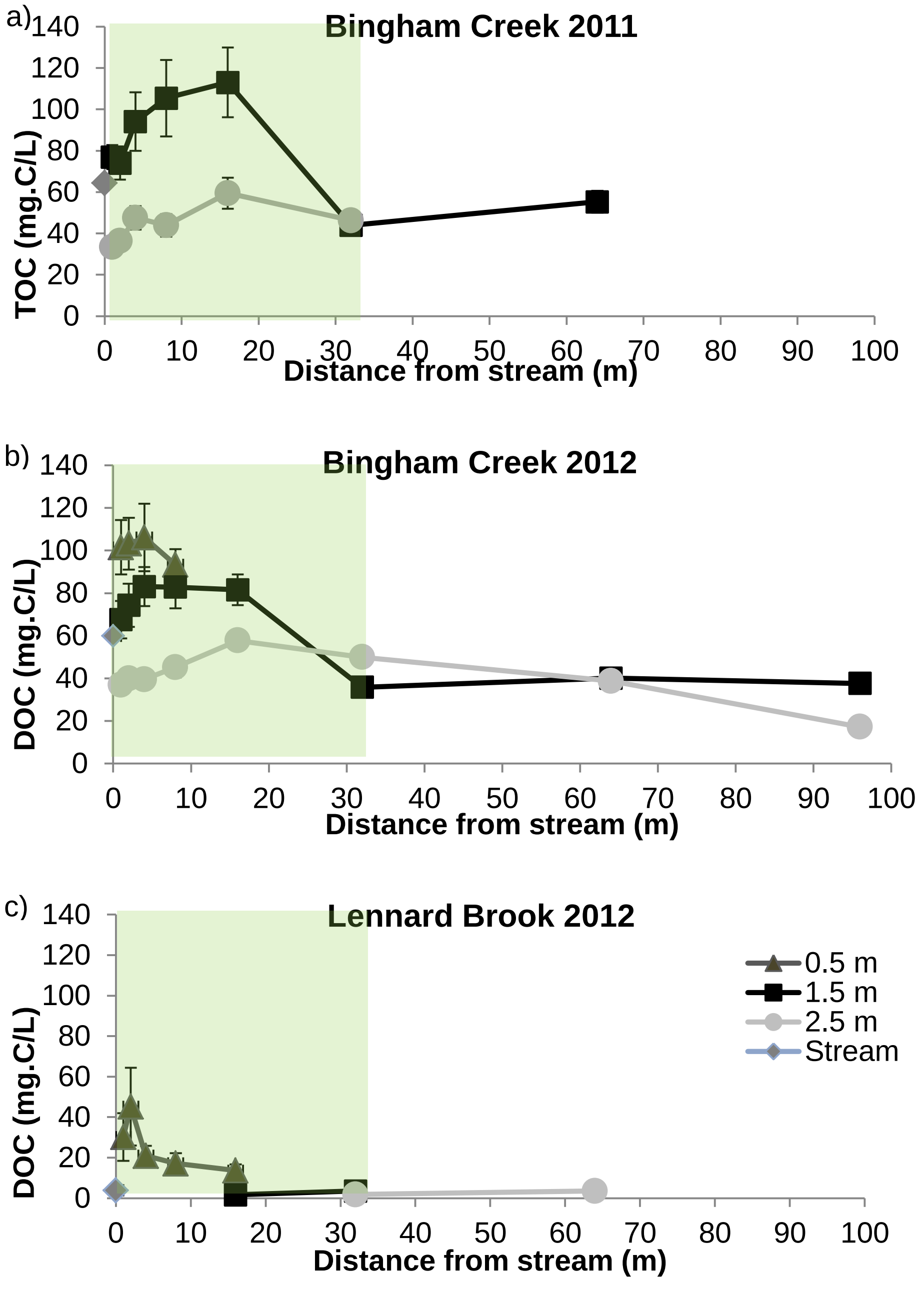


Figure 2-18. Comparison of dissolved organic carbon concentrations at a) Bingham Creek 2011, b) Bingham Creek 2012 and c) Lennard Brook along a gradient from the stream to the paddock, the shading represents the riparian zone. Note 2011 sample was for total organic carbon

At Bingham Creek, dissolved organic carbon concentrations were consistent at 1.5 and 2.5 m depth in the paddock and showed little change over time (Figure 2-19a). Within the riparian zone, dissolved organic carbon concentrations increased with time at 0.5 and 1.5 m depth before decreasing once the wet season ceased (Figure 2-19b). At Lennard Brook concentrations were substantially lower and consistent over time. At 1.5 and 2.5 m depth there was no carbon, whereas at 0.5 m depth concentrations were considerably higher (Figure 2-19c).

Groundwater gilvin concentrations showed the same pattern as DOC for depth, time and zone (Figure 2-20a-c).

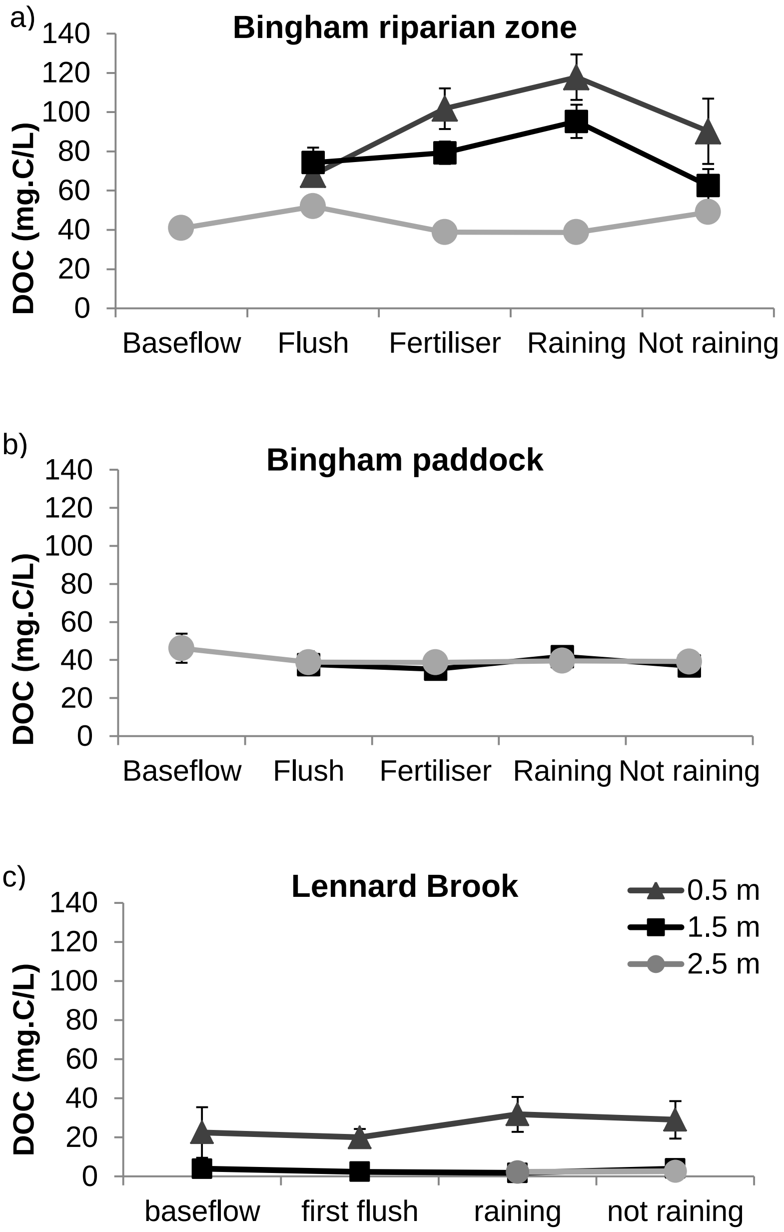


Figure 2-19. Comparison of dissolved organic carbon concentrations over time and depth at Bingham Creek a) riparian zone, b) paddock and c) Lennard Brook in 2012

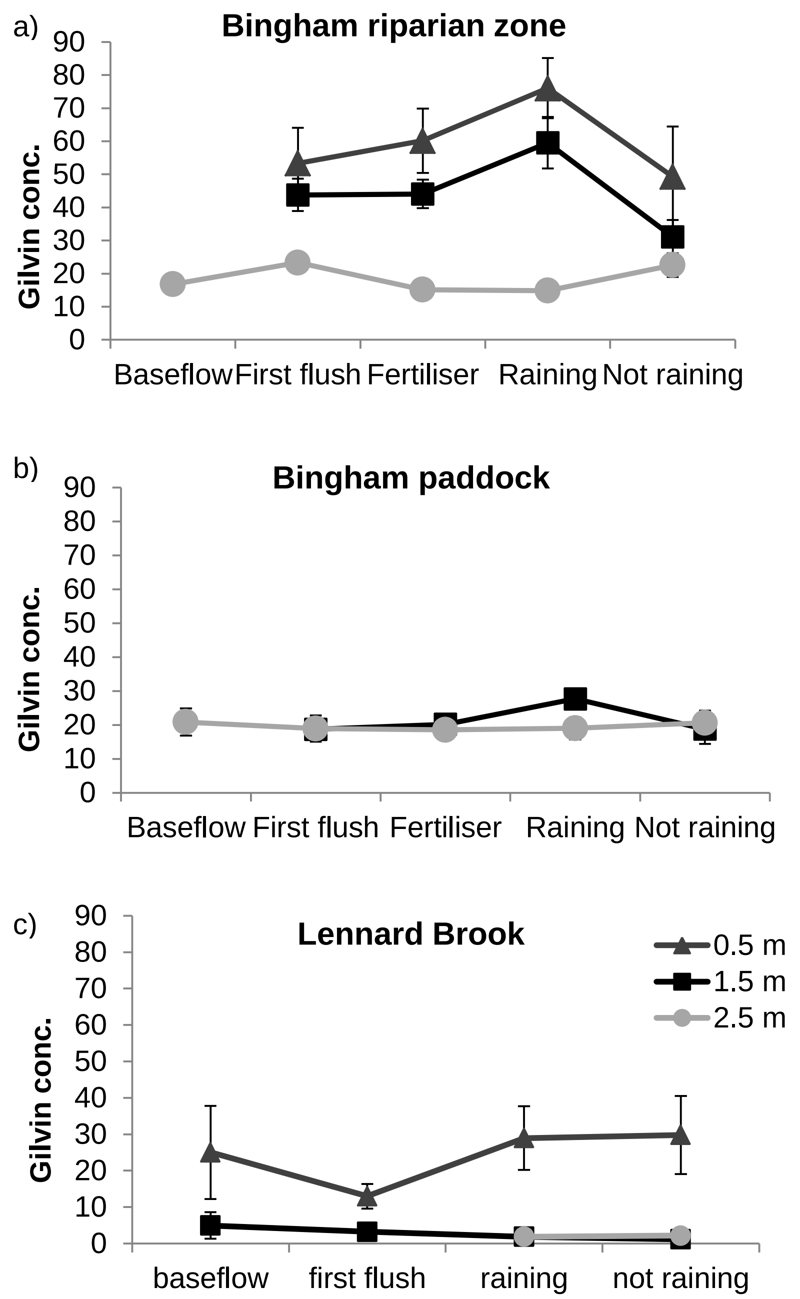


Figure 2-20. Comparison of gilvin concentrations over time and depth at at Bingham Creek a) riparian zone, b) paddock and c) Lennard Brook in 2012

Bingham Creek had a much higher E4:E6 ratio than Lennard Brook, with the minimum values at Bingham Creek being higher than maximum values at Lennard Brook (Figure 2-21a-c). Bingham Creek showed no clear patterns, however, samples from 2.5 m depth were generally lower and 1.5 m depth generally the highest. There was little difference between the paddock and riparian zone (Figure 2-21a-b). At Lennard Brook, the ratio decreased with increasing depth and there was no pattern in regards to changing E4:E6 ratios over time (Figure 2-21c).

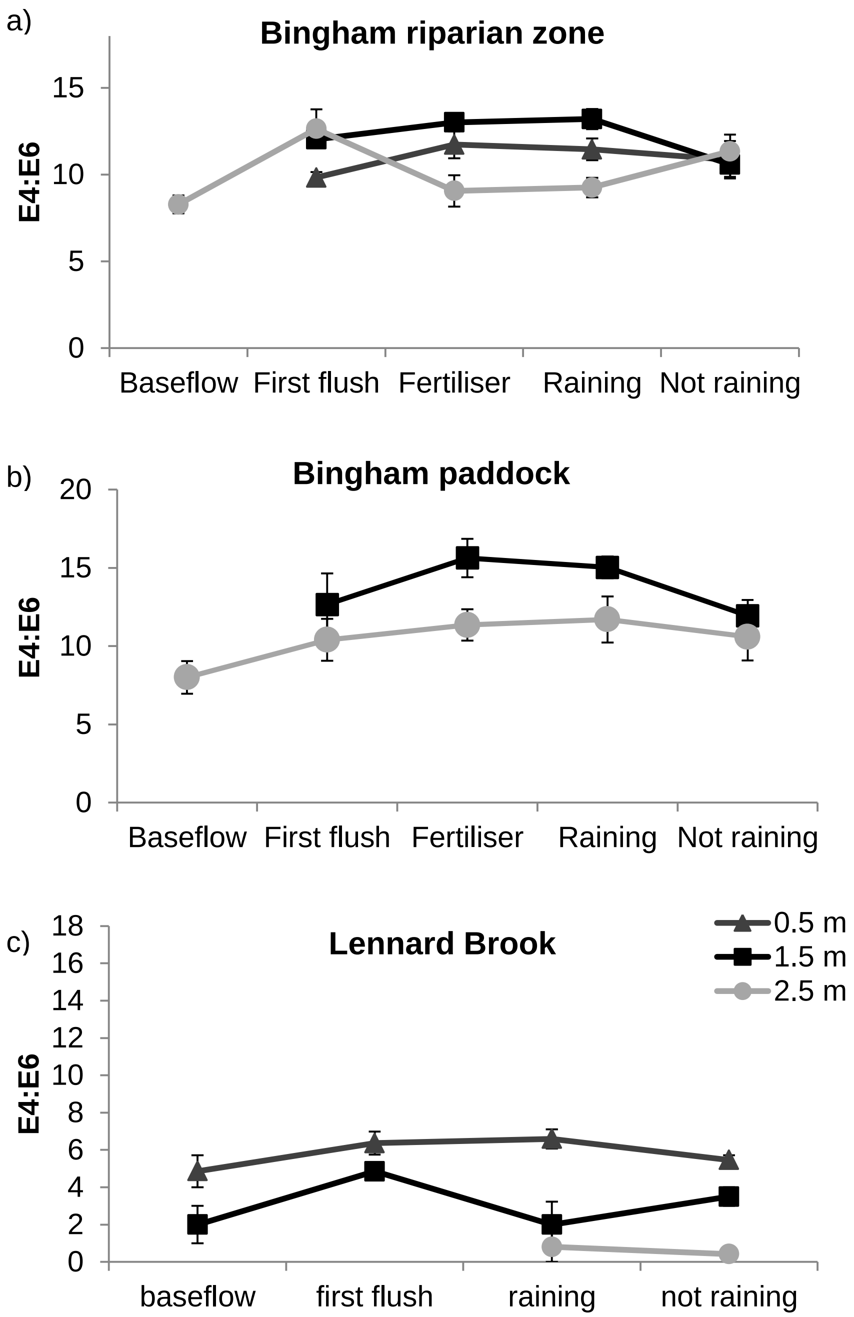


Figure 2-21. Comparison of E4:E6 ratio over time and depth at Bingham Creek a) riparian zone, b) paddock and c) Lennard Brook in 2012

Nutrients in groundwater at Bingham Creek and Lennard Brook.
At Bingham Creek, filterable reactive phosphorus (FRP) concentrations were low in the paddock at all depths, but for groundwater shallower than 1.5 m, increased through the riparian zone towards the stream. FRP concentrations in the stream were substantially higher than in the groundwater at Bingham Creek in 2011 and 2012. Lennard Brook had a slight increase in concentrations at 0.5 m depth from 16 m towards the stream (Figure 2-22c).

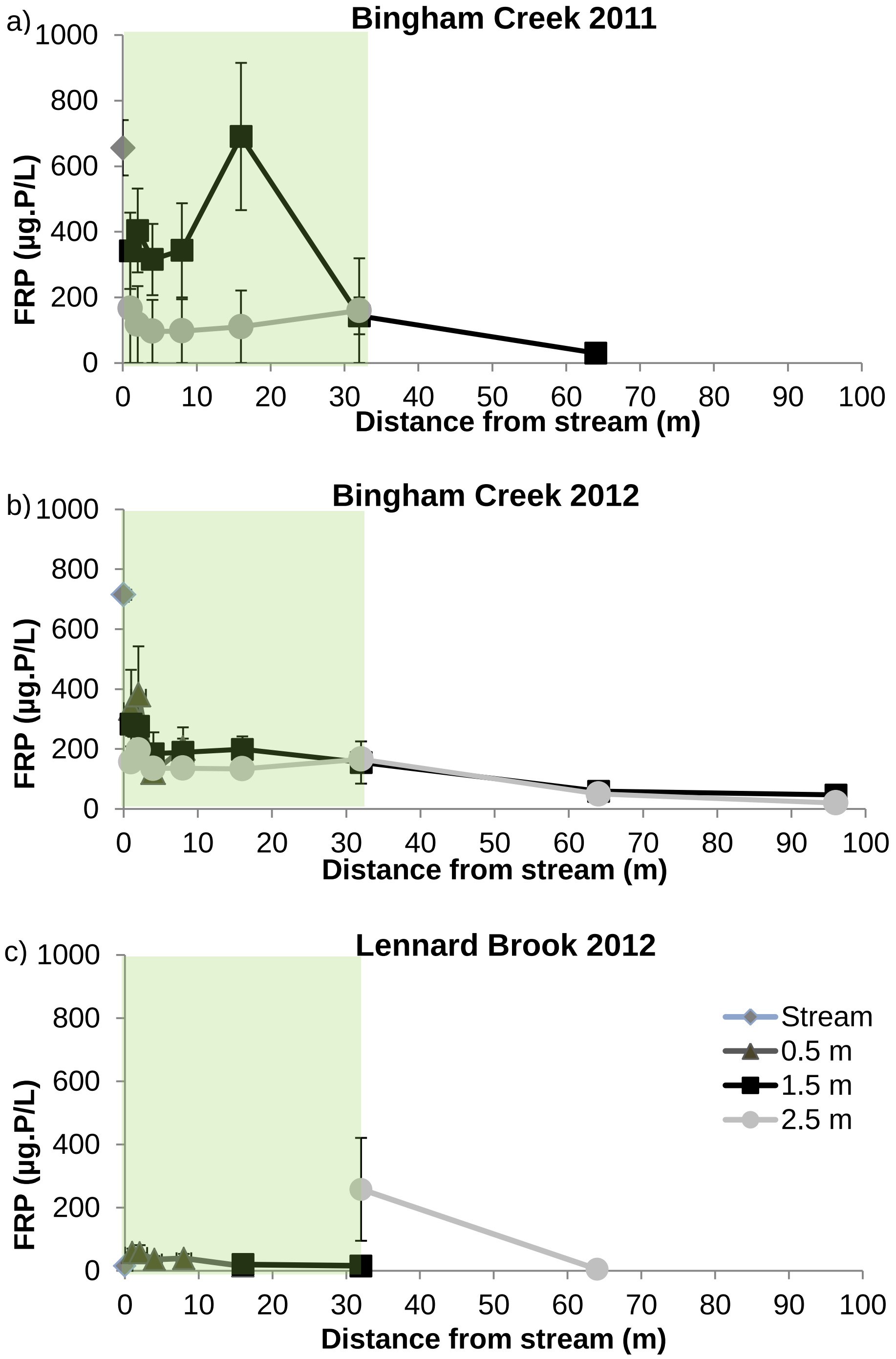


Figure 2-22. Comparison of filterable reactive phosphorus concentrations at a) Bingham Creek 2011, b) Bingham Creek 2012 and c) Lennard Brook along a gradient from the stream to the paddock, the shading represents the riparian zone

Total phosphorus concentrations in the paddock acted similarly to filterable reactive phosphorus, with concentrations increasing from the paddock towards the stream at Bingham Creek (Figure 2-23a-b). Depth was influential, with concentrations being higher in shallower groundwater. At Lennard Brook total phosphorus concentrations increased towards stream at 1.5 and 2.5 m depths, whereas at 0.5 m concentrations were highest at 16 m before decreasing towards the stream (Figure 2-23c).

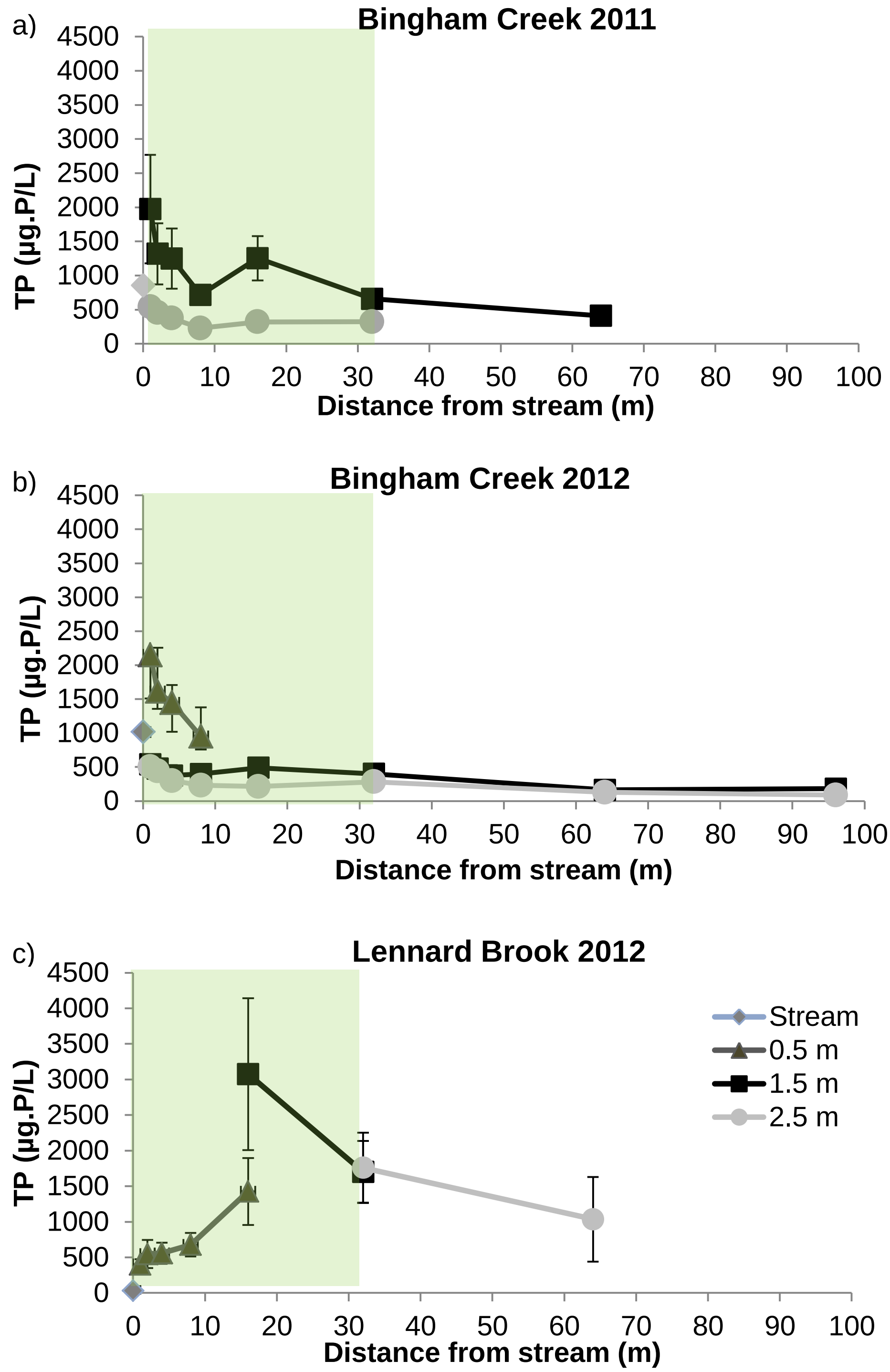


Figure 2-23. Comparison of total phosphorus concentrations at a) Bingham Creek 2011, b) Bingham Creek 2012 and c) Lennard Brook along a gradient from the stream to the paddock, the shading represents the riparian zone

There are no ANZECC guidelines for groundwater, however, considering groundwater flows into the stream, comparing nutrient concentrations with ANZECC trigger values is relevant. Given their locations groundwater concentrations at Bingham Creek can be compared to lowland river limits and Lennard Brook to upland river values (Table 2-3).

Filterable reactive phosphorus increased in the riparian zone from first flush through to the end of the wet season in both years, but it was not as high in 2012 (Figure 2-24a-b). Filterable reactive phosphorus concentrations were consistently low at 2.5 m depth across both the paddock and riparian zone (Figure 2-24a-b). While they were comparatively low FRP concentrations at 2.5 m depth were still more than double ANZECC guidelines (Table 2-3). Concentrations at 1.5 m depth were considerably lower in the paddock and did not show as clear a trend as the riparian zone, which showed a gradual increase over time. At Lennard Brook there was very little filterable reactive phosphorus and concentrations did not vary over time (Figure 2-24c). The concentrations at 2.5 m depth were exceptionally high (40 times ANZECC guidelines) and is likely an artefact due to the disturbance of clay during piezometer installation.

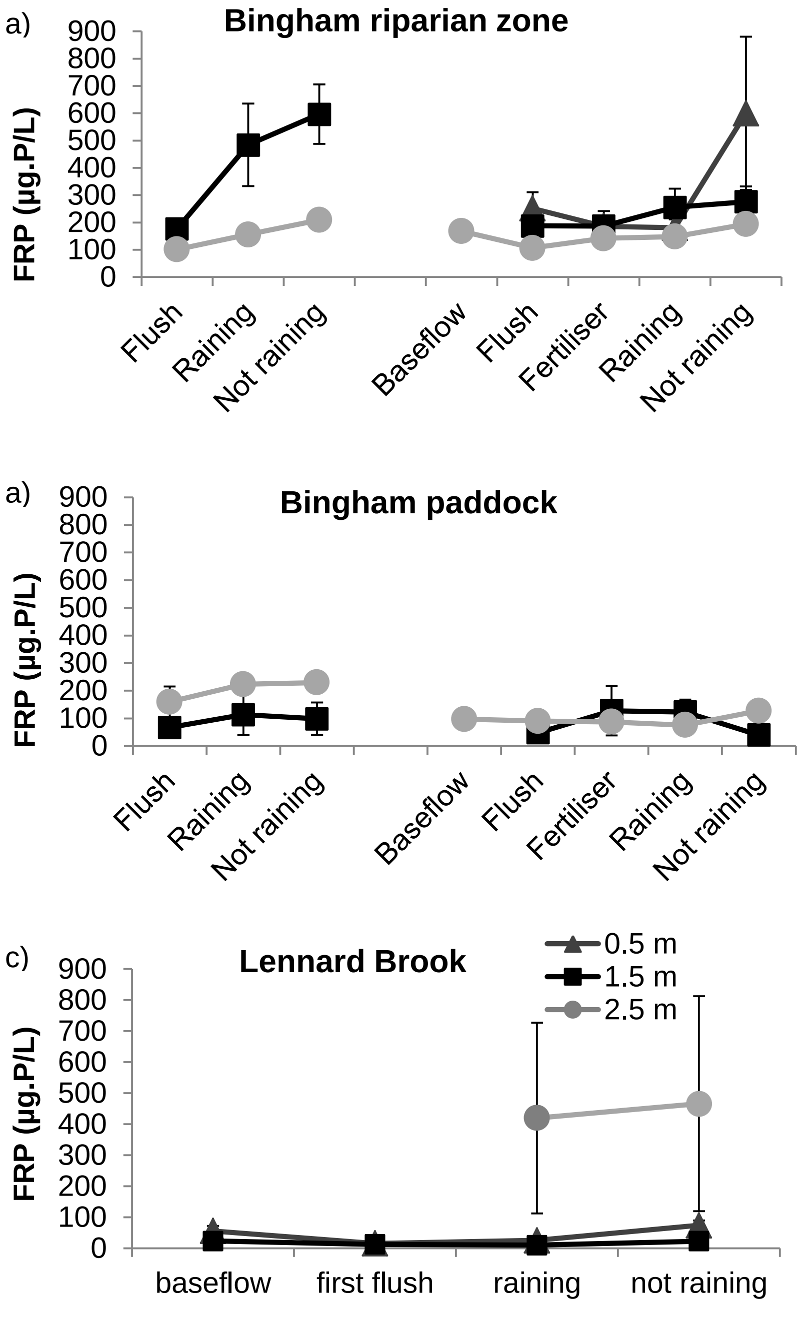


Figure 2-24. Comparison of filterable reactive phosphorus concentrations over time and depth at Bingham Creek a) riparian zone, b) paddock and c) Lennard Brook, note data for Bingham Creek is over two years

The total phosphorus concentrations at 1.5 and 2.5 m depth primarily decreased over time, but at 0.5 m depth it was considerably higher and increased over time. These trends were consistent across both zones, except concentrations were higher in the riparian zone, which all exceeded ANZECC trigger values (Figure 2-25a-b). At Lennard Brook, trends in total phosphorus concentrations were consistent at all depths, being highest at 1.5 m depth and lowest at 0.5 m (Figure 2-25c). Total phosphorus concentrations at 1.5 m depth exceeded ANZECC trigger values by between 75-130 times, whereas at 0.5 m it was 20-50 times greater (Table 2-3). Overall total phosphorus concentrations were higher at Lennard Brook than at Bingham Creek, particularly at 1.5 m.

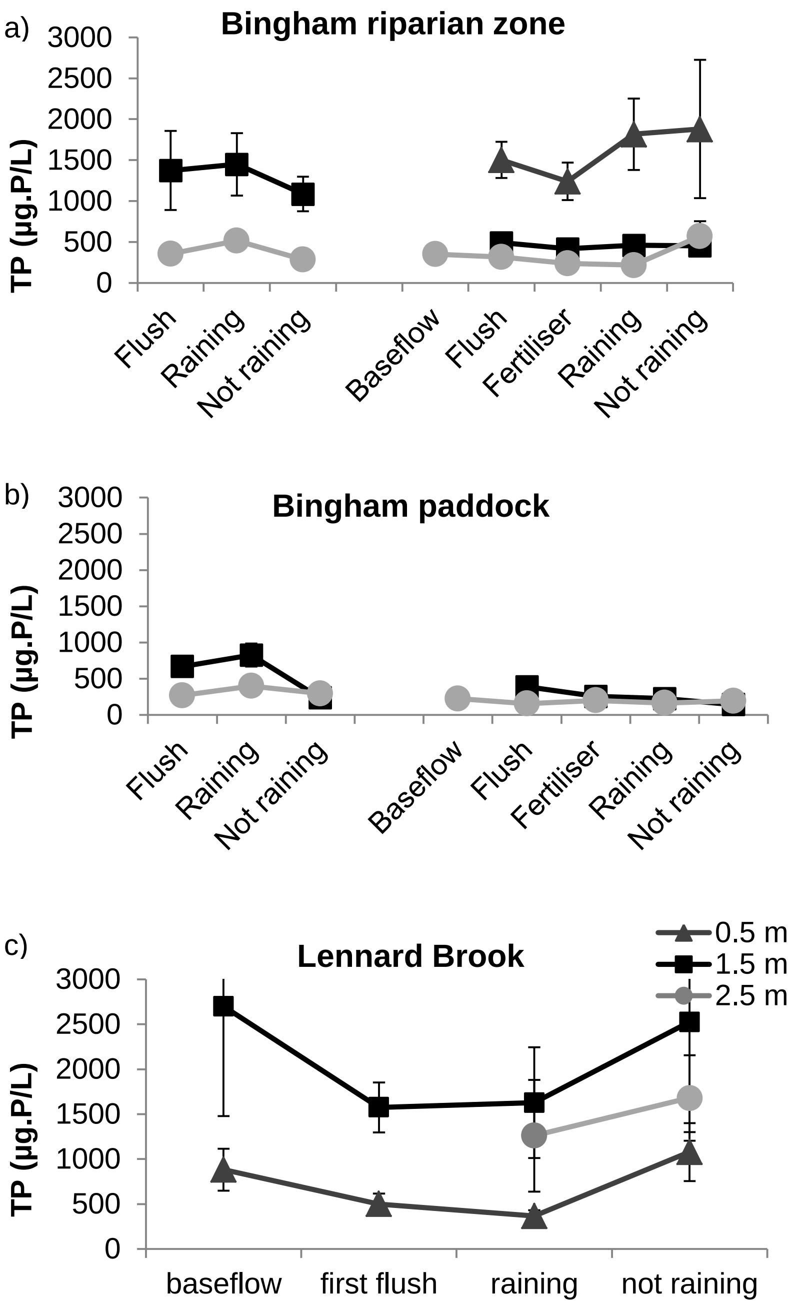


Figure 2-25. Comparison of total phosphorus concentrations over time and depth at Bingham Creek a) riparian zone, b) paddock and c) Lennard Brook, note data for Bingham Creek is over two years

Oxidised nitrogen concentrations were highest in the paddock and decreased exponentially through the riparian zone towards the stream at Bingham Creek and this trend was consistent across all depths (Figure 2-26a-b). At Lennard Brook, oxidised nitrogen concentrations were low in the paddock at all depths but peaked in the riparian zone in 0.5 m groundwater, 8m from the stream and then decreased rapidly towards the stream. Oxidised nitrogen was exceptionally high in stream water (Figure 2-26c).

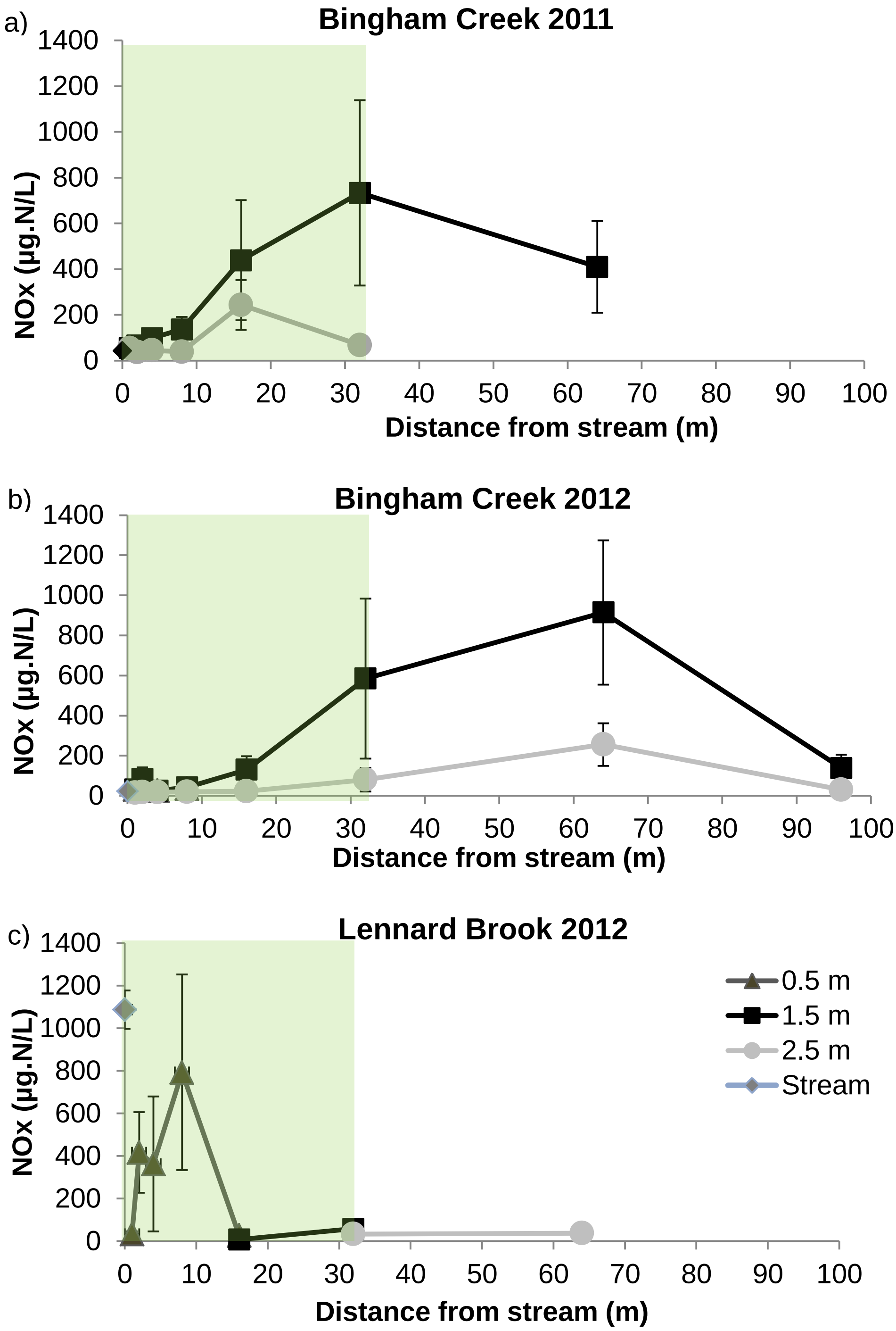


Figure 2-26. Comparison of oxidised nitrogen concentrations at a) Bingham Creek 2011, b) Bingham Creek 2012 and c) Lennard Brook along a gradient from the stream to the paddock, the shading represents the riparian zone

At Bingham Creek ammonium concentrations were constant in the paddock at 64 and 96 m. At 1.5 m depth, concentrations remained consistent, while groundwater at 2.5 m depth increased through the riparian zone, but was highly variable (Figure 2-27a-b). At Lennard Brook ammonium concentrations were lowest at 0.5 m depth and were consistent throughout the riparian zone. Concentrations were highest in groundwater at 32m from the stream (Figure 2-27c)

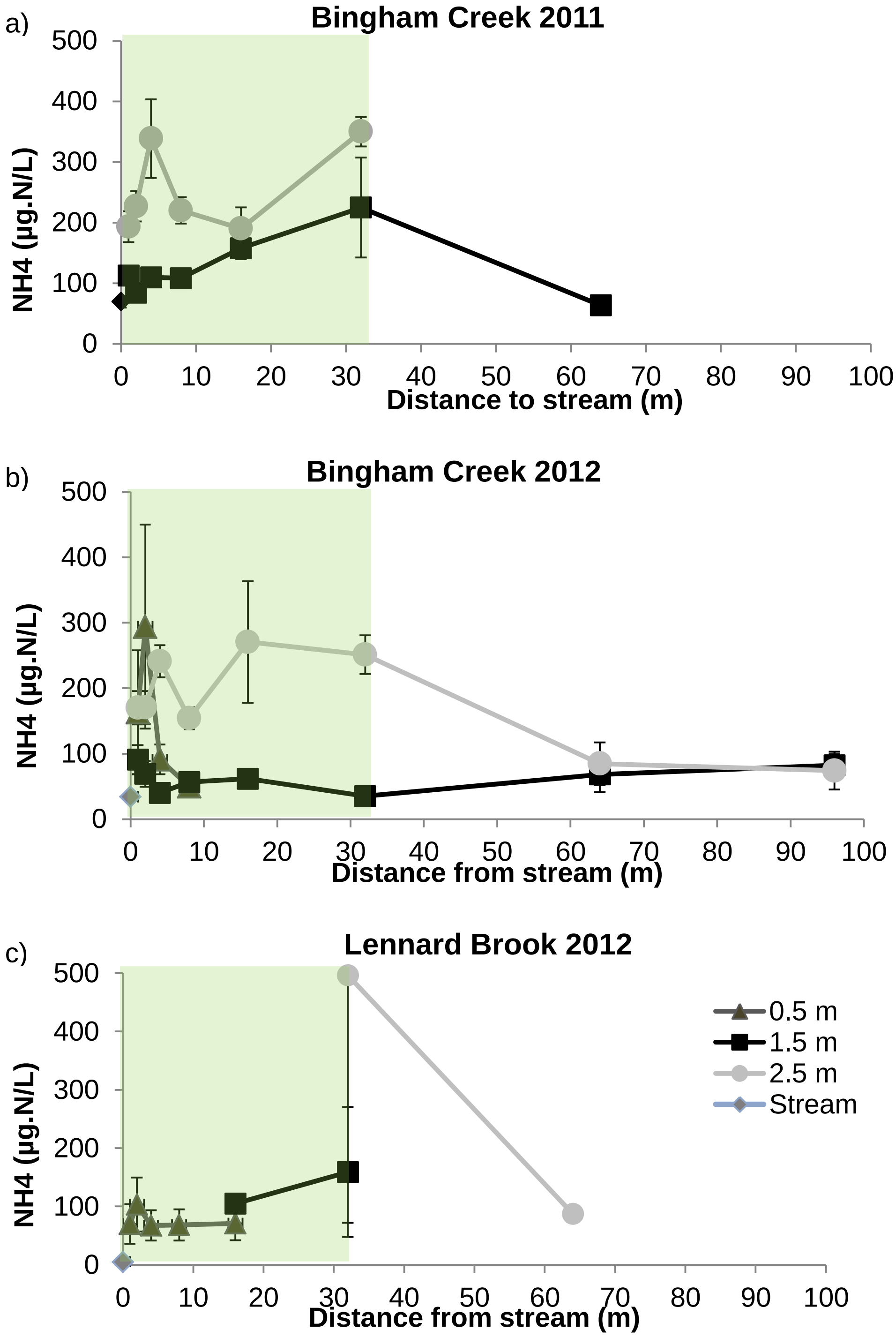


Figure 2-27. Comparison of ammonium concentrations at a) Bingham Creek 2011, b) Bingham Creek 2012 and c) Lennard Brook along a gradient from the stream to the paddock, the shading represents the riparian zone

At Bingham Creek, total nitrogen concentrations at 1.5 and 2.5 m depth decreased from the paddock to the edge of the riparian zone, increased slightly at 16m along the transect before again decreasing closer to the stream (Figure 2-28a-b). Total nitrogen concentrations were highest in the 0.5 m groundwater, which more than doubled in instream concentrations. At Lennard Brook, concentrations were highest at 0.5 m depth but concentrations decreased very close to the stream (Figure 2-28c).

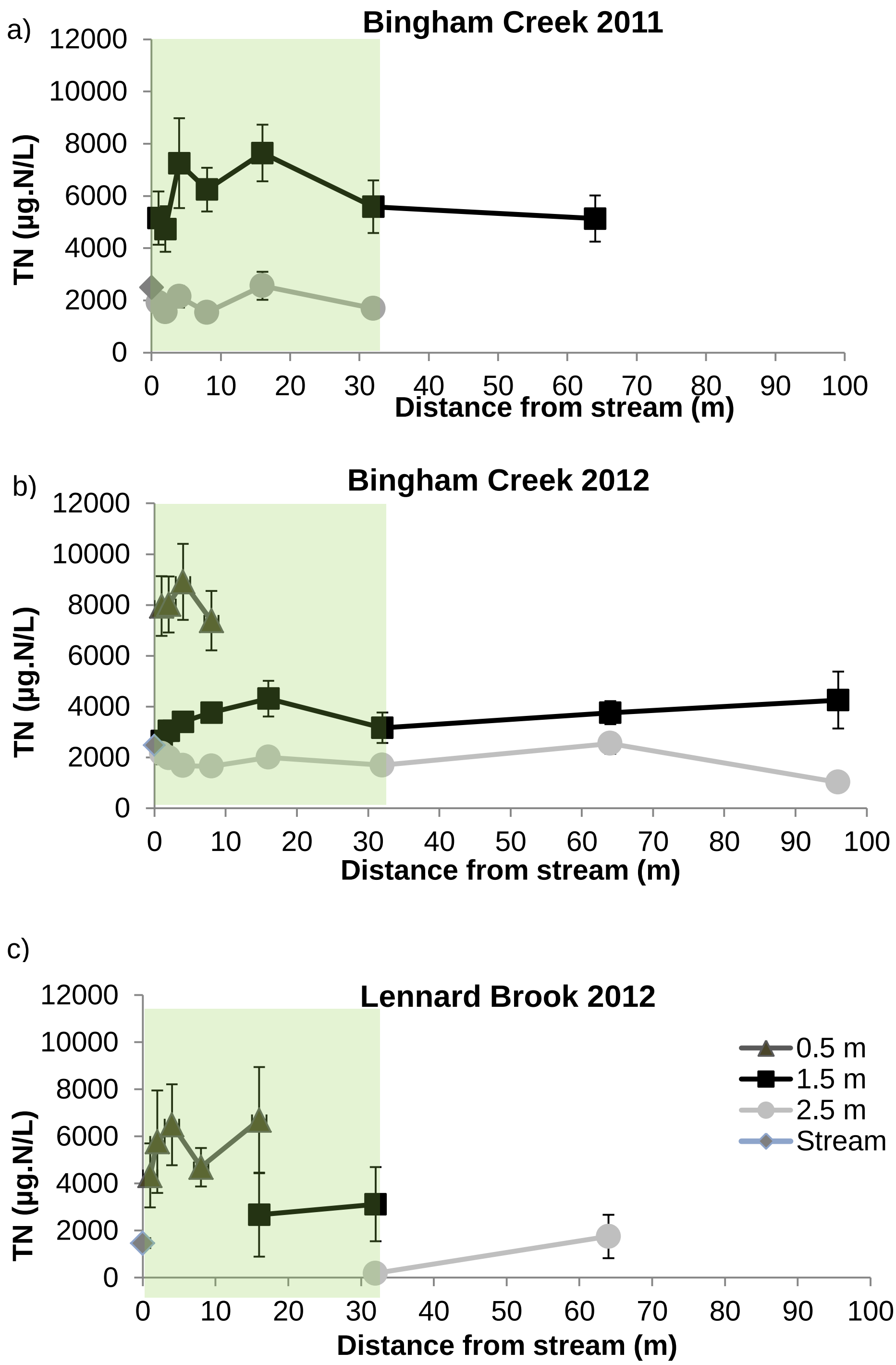


Figure 2-28. Comparison of total nitrogen concentrations at a) Bingham Creek 2011, b) Bingham Creek 2012 and c) Lennard Brook along a gradient from the stream to the paddock, the shading represents the riparian zone

At Bingham Creek, oxidised nitrogen concentrations decreased from the first flush through to the end of the wet season, however, this trend was most apparent in the paddock, where concentrations were substantially higher (Figure 2-29a-b). Only groundwater in the paddock at 1.5 m depth was greater than ANZECC guidelines (5-9 times greater). At Lennard Brook, groundwater oxidised nitrogen concentrations were consistently low and there was no apparent trend over time (Figure 2-29c).

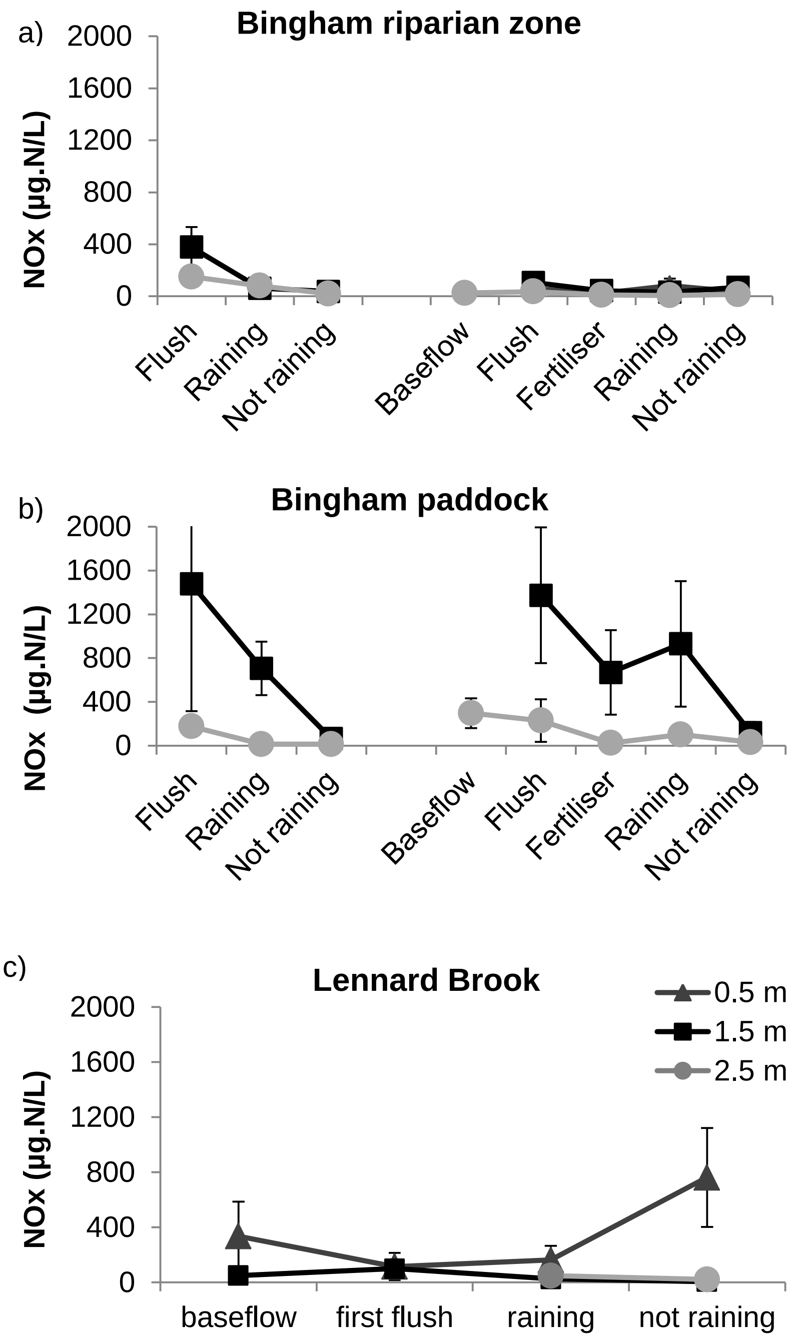


Figure 2-29. Comparison of oxidised nitrogen concentrations over time and depth at Bingham Creek a) riparian zone, b) paddock and c) Lennard Brook, note data for Bingham Creek is over two years

Ammonium concentrations at Bingham Creek remained relatively constant over time at 1.5 and 2.5 m depths, whereas at 0.5 m depth concentrations decreased over the wet season and then increased at the end of the wet season (Figure 2-30a-b). Across the riparian zone and paddock, ammonium concentrations were very similar, with concentrations at 2.5 m depth consistently higher than 1.5 m, however there was a decreasing trend in paddock concentrations over time (Figure 2-30a-b). At Lennard Brook, ammonium concentrations were lower, decreasing after first flush before increasing at the end of the wet season, consistent across all depths (Figure 2-30c).

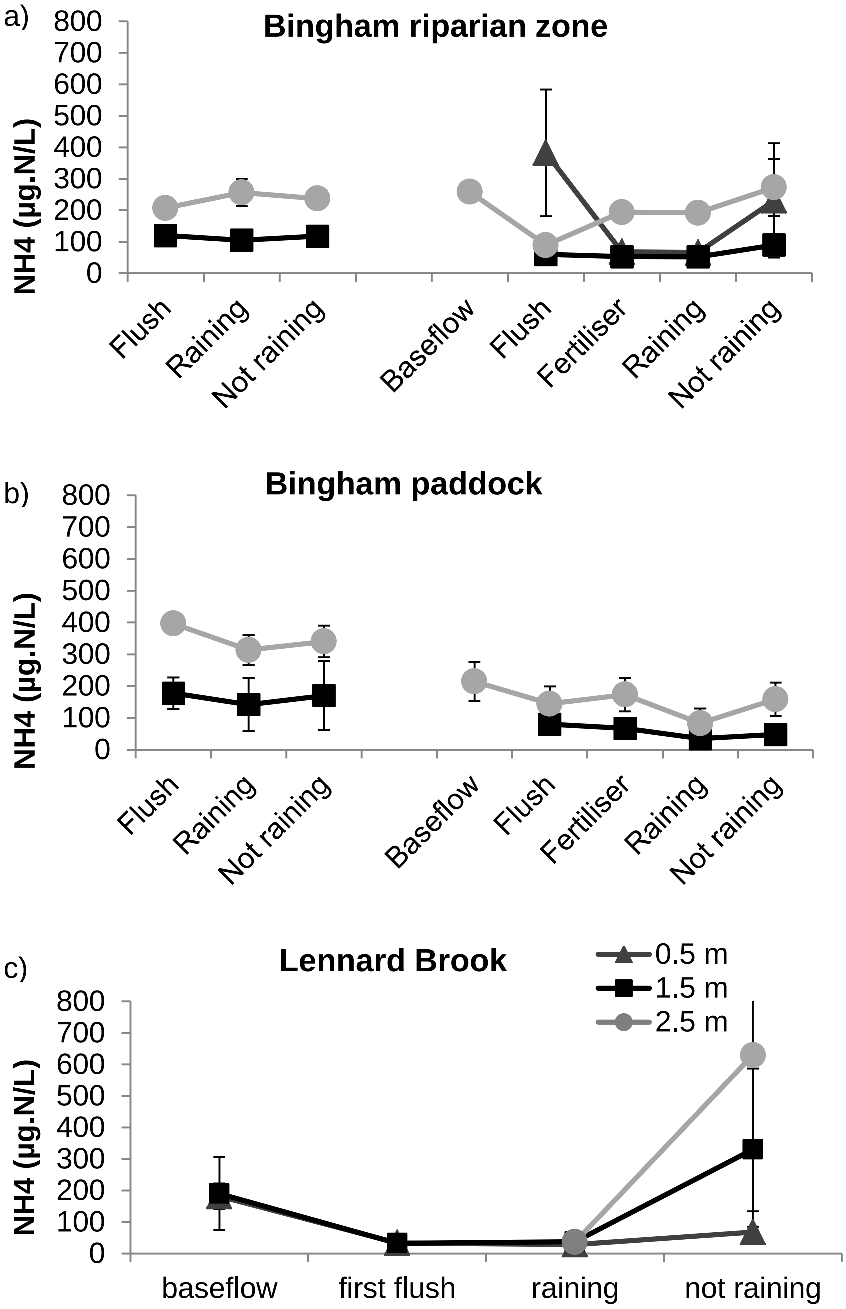


Figure 2-30. Comparison of ammonium concentrations over time and depth at Bingham Creek a) riparian zone, b) paddock and c) Lennard Brook, note data for Bingham Creek is over two years

Total nitrogen behaved similarly to total phosphorus at Bingham Creek with concentrations primarily decreasing over time, which was fairly consistent across all three depths (Figure 2-31a-b). This trend was consistent over the riparian zone and the paddock and concentrations decreased with depth. Total nitrogen concentrations at 0.5 m depth were up to eight times greater than ANZECC guidelines (Table 2-3). At Lennard Brook, total nitrogen concentrations decreased from baseflow through to the end of the wet season, with all samples acting in a similar fashion, but with greatest variability occurring at 0.5 m depth (Figure 2-31c).

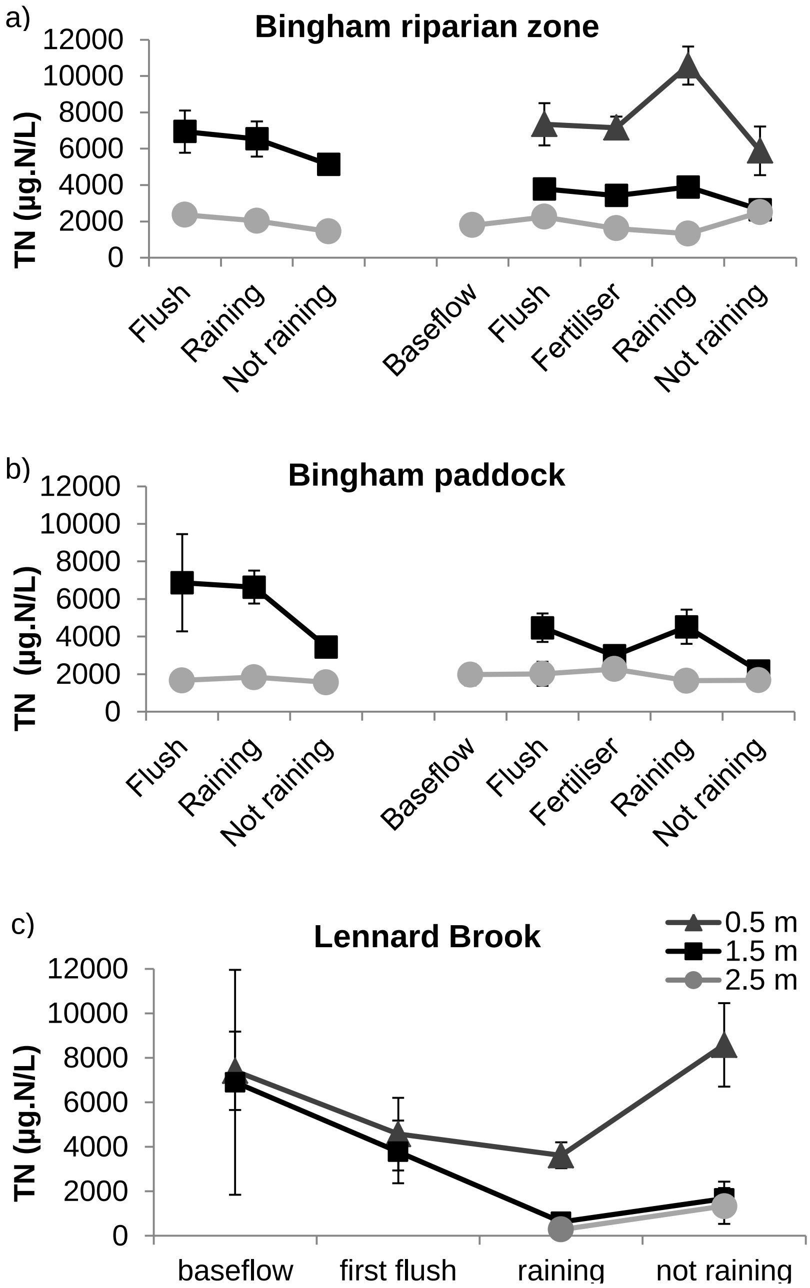


Figure 2-31. Comparison of total nitrogen concentrations over time and depth at Bingham Creek a) riparian zone, b) paddock and c) Lennard Brook, note data for Bingham Creek is over two years

Nutrient concentration changes with depth

At Bingham Creek in 2011, nutrient concentrations at 1.5 m depth had significantly higher concentrations (P<0.05) than at 2.5 m for all nutrients except ammonium (Table 2-5). Forty percent of the variation between groundwater quality at 1.5 and 2.5 m is described by ammonium, with much higher ammonium concentrations at 2.5 m. At 1.5 m the distribution of points is explained by both low ammonium and high filterable reactive phosphorus (Figure 2-32). Nutrient concentrations were also significantly different at each depth in 2012. They decreased with depth for all nutrients, except ammonium, which was highest at 2.5 m (Table 2-5).

Table 2-5. Mean nutrient concentration variations over two depths at Bingham Creek in 2011. Means are least-squares means from ANOVA, values in parentheses are standard, * = *P* < 0.05

Nutrient	Depth of groundwater	
	1.5 m	2.5 m
FRP (μg/L)*	339 (54.6)	164 (9.3)
TP (μg/L)*	1117 (166.4)	375 (33.6)
NOx-N (μg/L)*	270 (79.7)	81 (21.8)
NH_4-N (μg/L)*	126 (14.2)	253 (16.6)
TN (μg/L)*	6007 (429.1)	1898 (140.4)
TOC (mg/L)*	82 (5.4)	45 (2.3)

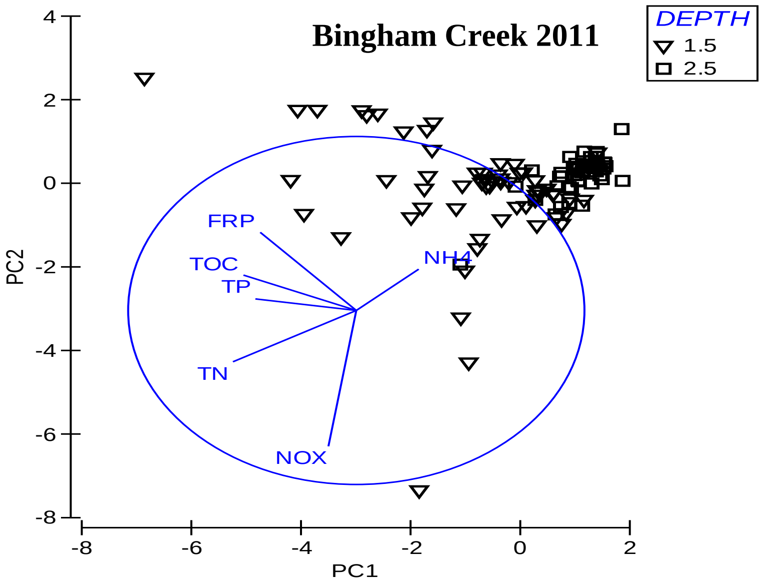


Figure 2-32. PCA ordination plot of water quality data showing the variations in nutrient concentrations with depth at Bingham Creek in 2011

A Tukey's test showed that in 2012, groundwater at 2.5 m depth had significantly lower concentrations than at 1.5 m for TN and NO_x-N but had higher concentrations of NH_4-N (Table 2-6). Concentrations were also lower for FRP, TP and TN at 2.5 m than groundwater at 0.5 m. Shallow groundwater at 0.5 m depth had significantly higher concentrations of all nutrients except NO_x-N than groundwater at 1.5 m. Dissolved organic carbon concentrations were significantly different at all three depths, decreasing with depth (Table 2-6).

Table 2-6. Mean nutrient concentrations over three depths at Bingham Creek in 2012. Means are least-squares means from ANOVA, values in parentheses are standard errors from ANOVA. * = $P < 0.05$. Letters indicate depths that were significantly different in Tukey's tests

	0.5 m	1.5 m	2.5 m
FRP (µg/L)*	278 (61.7)[a]	173 (19.9)[b]	134 (7)[b]
TP (µg/L)*	1599 (223.7)[a]	379 (38.4)[b]	291 (29.4)[c]
NOx-N (µg/L)*	50 (17.7)[a]	310 (90.9)[b]	54 (15.8)[a]
NH_4-N (µg/L)*	161 (54.2)[a]	61 (5.8)[b]	186 (16.2)[a]
TN (µg/L)*	8034 (568)[a]	3726 (252.4)[b]	2000 (145.7)[b]
DOC (mg/L)*	98 (6.3)[a]	63 (3.3)[b]	43 (1.5)[c]

The PCA shows few patterns in nutrient concentrations at Bingham Creek in 2012, but the concentration of ammonium explained most of the variation (35%). The distribution of samples at 0.5 m depth is primarily explained by high total phosphorus, filterable reactive phosphorus and ammonium. At 1.5 m there was greater variability in the data and was associated with both high oxidised nitrogen and ammonium (Figure 2-33). Again the groundwater at 2.5 m depth shows the least variability and is primarily explained by high ammonium concentrations.

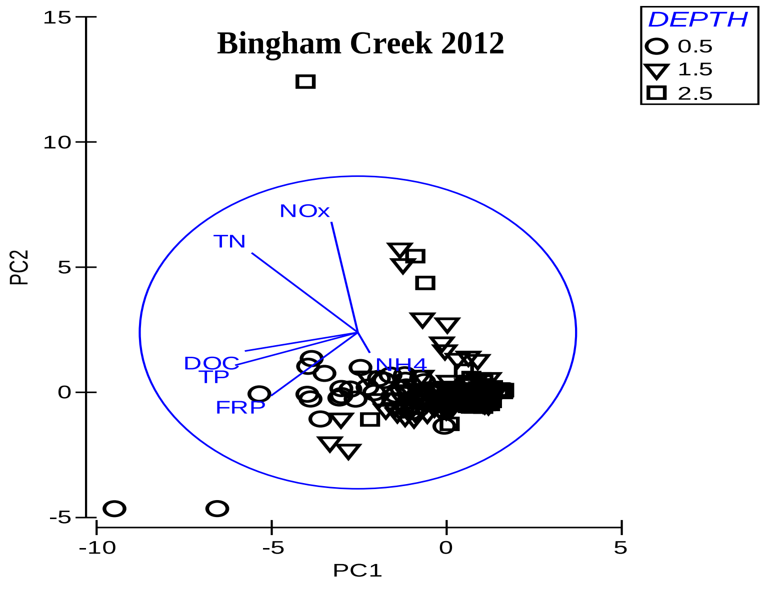


Figure 2-33. PCA ordination plot of environmental data showing the variations in nutrient concentrations over three depths at Bingham Creek in 2012

At Lennard Brook, groundwater nutrient concentrations varied with depth, but not as significantly as at Bingham Creek. There were significant differences in concentrations between FRP, TP, TN and DOC (Table 2-7). Trends in nutrient concentrations with depth showed some similarity to Bingham Creek with TN and DOC concentrations being highest at 0.5 m depth and NH_4-N concentrations at 2.5 m depth.

Table 2-7. ANOVA table representing mean nutrient concentrations over three depths at Lennard Brook in 2012. Values in parantheses are standard errors from ANOVA. . * = $P < 0.05$. Letters indicate depths that were significantly different in Tukey’s tests

	0.5 m	1.5 m	2.5 m	P value
FRP (µg/L)*	42 (6.4)[a]	17 (4.1)[a]	443 (218.6)[b]	<0.000
TP (µg/L)*	698 (106.1)[a]	2067 (415.3)[b]	1470 (375.7)[b]	<0.000
NOx-N (µg/L)	337 (113.9)	45 (16.2)	35 (10.8)	0.247
NH_4-N (µg/L)	76 (13.5)	145 (73.3)	333 (254.2)	0.053
TN (µg/L)*	5981 (791.9)[a]	2520 (1064.9)[a]	810 (429.1)[b]	0.006
DOC (mg/L)*	26 (4.6)[a]	3 (0.6)[b]	3 (0.3)[b]	0.007

At Lennard Brook, FRP was significantly higher at 2.5 m than at 0.5 or 1.5 m depth. Total nitrogen was significantly lower. Groundwater at 0.5 m was significantly higher in DOC than at both 1.5 and 2.5 m and lower in TP than at 1.5 m (Tukey's test , Table 2-9).

At Lennard Brook the PCA shows few patterns in groundwater nutrient concentrations. There was considerable spread of points for all three depths but overall was skewed towards high oxidised nitrogen concentrations (Figure 2-34). The distribution of samples at 0.5 m can be explained by high dissolved organic carbon and deeper groundwater (2.5 m) explained by high filterable reactive phosphorus.

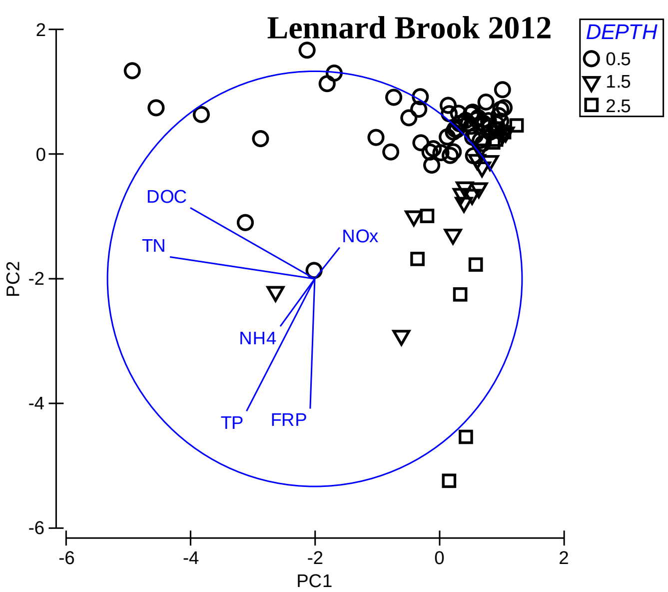


Figure 2-34. PCA ordination plot of water quality data showing the variations in nutrient concentrations over three depths at Bingham Creek in 2012

Comparison between sites

There was a significant difference between Bingham Creek and Lennard Brook for NH_4-N, FRP and TP (Table 2-8). Furthermore, there was a significant difference for DOC with the interaction term Site*Time indicating carbon concentrations varied temporally and between sites (Table 2-8). This provides an indication that groundwater nutrient concentrations were markedly different between shallow (0.5 m) groundwater between Bingham Creek and Lennard Brook. All concentrations were higher at Bingham Creek with the exception of TP.

Table 2-8. Significant factors from ANOVA comparing nutrient concentrations from 0.5 m depth groundwater samples from 1-4m between Bingham Creek and Lennard Brook

BINGHAM-LENNARD 1-4 m at 0.5 m					
Nutrient	Variable	df	Mean square	F	*P*
NH_4-N	Site	1	664446.296	5.142	0.029
FRP	Site	1	1069629.63	8.951	0.005
TP	Site	1	$2.176*10^7$	12.376	0.001
DOC	Site*Time	2	5555.839	5.116	0.011

There were significant differences in TP, TN and DOC concentrations between Bingham Creek and Lennard Brook (Table 2-9). Furthermore, within sites there was a significant difference for DOC carbon concentrations within sites. These results provide an indication that there were significant variations in groundwater nutrient concentrations between sites. At Bingham Creek, TN and DOC concentrations were higher; alternatively TP concentrations were highest at Lennard Brook.

Table 2-9. Multivariate analysis table comparing nutrient concentrations from 1.5 m depth groundwater samples from 32 m between Bingham Creek and Lennard Brook.

BINGHAM-LENNARD 32 m at 1.5 m					
Nutrient	Variable	df	Mean square	F	*P*
TP	Site	1	3690138.89	8.983	0.011
TN	Site	1	$1.422*10^7$	6.055	0.030
	Time	2	9245000	3.936	0.048
DOC	Site	1	5678.227	94.947	0.000

Discussion

Nutrients and flow in the stream at Bingham Creek and Ellen Brook

Nutrient concentrations, in particular total phosphorus (TP) in Bingham Creek, epitomise the phosphorus enrichment issue in the catchment. Instream TP concentrations exceeded ANZECC guidelines by nearly 15 times. Of this, approximately 80% was filterable reactive phosphorus (FRP), highlighting the availability of phosphorus in eastern flowing streams in the Ellen Brook catchment. The high availability of inorganic phosphorus can lead to eutrophic conditions further downstream and negatively impact the Swan Canning estuary (Swan River Trust 2009). Lennard Brook had comparatively low TP concentrations, however, they still exceeded ANZECC guidelines for upland streams. The concentrations at both streams were consistent with previous research in Ellen Brook (Sharma et al. 1996) highlighting the longevity of this issue.

Instream total nitrogen (TN) concentrations were twice ANZECC guidelines at Bingham Creek (Table 2-1). However, oxidised nitrogen only made up a small portion of TN (~1%) and concentrations were below ANZECC guidelines, which is consistent with streams at the bottom of the Ellen Brook catchment (Peters and Donohue 2001). In contrast, Lennard Brook had high TN concentrations and the oxidised nitrogen portion was very high (80%); both nutrients exceeded ANZECC guidelines. Sharma et al. (1994) identified that Lennard Brook was characterised by high oxidised nitrogen concentrations and identified that TN concentrations were equally high in streams throughout the catchment, consistent with our results.

Instream nutrient concentrations at Bingham Creek were dependent on flow. This is illustrated by the strong positive correlations between load and discharge for all nutrients analysed except oxidised nitrogen (Table 2-4). This infers that nutrients are consistently available to the stream and that rainfall and subsequent flow is driving nutrient output from Bingham Creek. Stream phosphorus loads had a strong positive correlation with discharge, which is consistent with lowland streams in Ellen Brook (Donohue et al. 2001; Peters and Donohue 2001).

Stream nutrient concentrations varied substantially between Bingham Creek and Lennard Brook and can be explained by a number of processes. Bingham Creek is a lowland stream. Upstream of Bingham Creek is an agricultural landscape with poor underlying soils (Bassendean sands). Lowland streams are typically characterised by high nutrient concentrations, due to accumulation of upstream nutrients (Wilcock et al. 1999). Upstream agricultural practices act as a point source for nutrients and this coupled with poor soils (incapable of binding nutrients) can lead to the easy mobilisation of nutrients into streams (Lyons et al. 1998). These processes are driving the high instream nutrient concentrations at Bingham Creek, whereas in Lennard Brook, comparatively low nutrient concentrations occur due to its location in the headwaters of a forested catchment with good soils. As a result there are limited sources of nutrients and those that are available can be intercepted by vegetation and soils.

The hydrology of groundwater at Bingham Creek and Lennard Brook

The hydrology of Bingham Creek is defined by the lack of slope, no impermeable subsurface layer and poor sandy soils (the latter being discussed in the next chapter). This resulted in a lack of surface flow at Bingham Creek. Rainfall that fell on the highly permeable sands rapidly infiltrated vertically into the soil, rather than moving horizontally as surface flow, as a consequence of the lack of slope. Fluctuations in groundwater never reached the soil surface in the riparian zone or paddock. Groundwater entered the root zone (top 40cm) up to 8m from the stream in the riparian zone, but not in the paddock. Due to the lack of slope, the water table fluctuated vertically with rainfall and there was limited horizontal groundwater movement. Interestingly, as a consequence of the groundwater falling well below the stream bed at the end of the dry season, and the lack of a confining soil layer, there was an occurrence of the water table having a slope away from the stream into the riparian zone (Figure 2-8). This suggests that at first flush, the stream flows into the riparian zone groundwater, contributing to the groundwater in the riparian zone up to 16m from the stream, and the rise in the watertable over the wet season (Figure 2-35). Once the soil column reaches maximum saturation, there is the potential for slow horizontal groundwater flows due to the slight slope of the water table from paddock to stream (Figure 2-35).

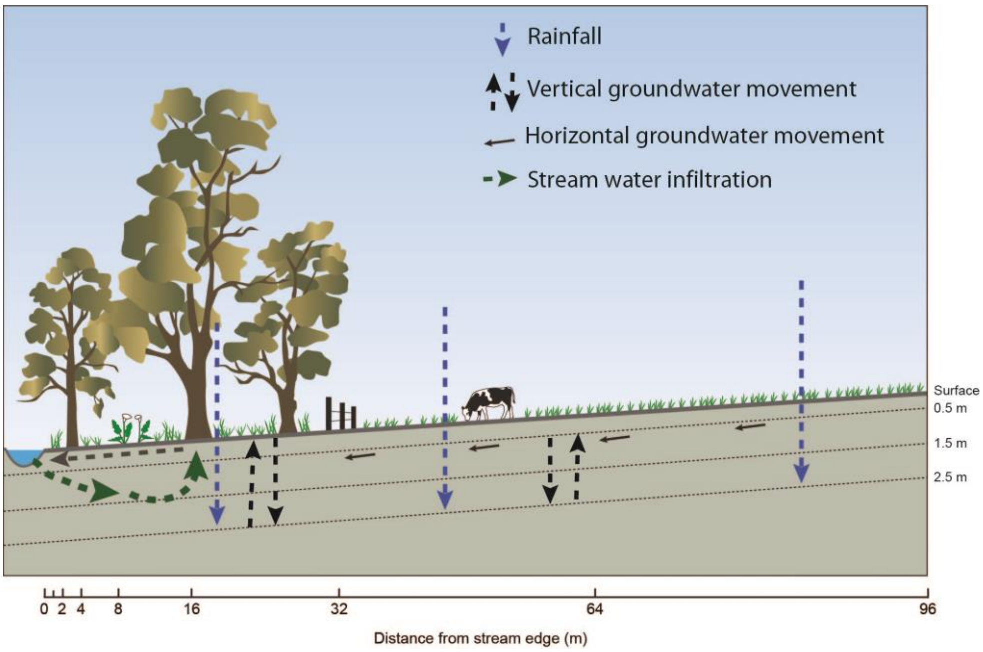


Figure 2-35. A conceptual model of the hydrology at Bingham Creek

In contrast, the hydrology of Lennard Brook is quite different. Lennard Brook is steeply sloped, has an increasing soil clay component with depth and is characterised by sandy soils. Due to the steep slope, surface flow occurs from the paddock through to the riparian zone, however, flow into the stream was not observed (Figure 2-36). Unlike Bingham Creek, Lennard Brook is a perennial system, resulting in minimal fluctuations in groundwater in the riparian zone (<0.2m over the year), which remained in the active root zone throughout the sampling period. Furthermore, the watertable intercepted the surface and pooling on the soil surface occurred in some sections of the riparian zone. However, due to the steep slope in the paddock, groundwater was not encountered within the top 2.5 m at 96m and, unlike the riparian zone, did not intercept the active root zone (Figure 2-36).

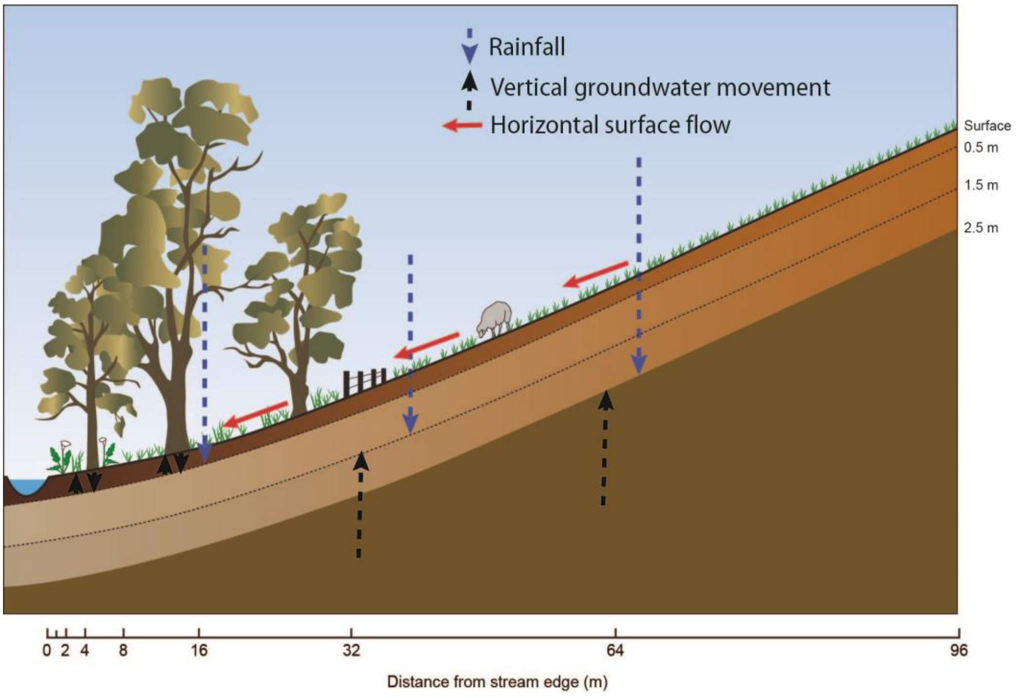


Figure 2-36. Conceptual model highlighting the key hydrological processes occurring at Lennard Brook

The effect of riparian vegetation on groundwater nutrient concentrations

Bingham Creek

The reduced horizontal movement of water through the riparian zone influenced the ability of riparian vegetation to intercept and reduce nutrient concentrations. The hydrology also affected the physico-chemical nature of the groundwater, with implications for nutrient transformation.

Dissolved organic carbon (DOC) and gilvin concentrations showed that surface water differed from the deeper groundwater (Figure 2-19-2.20). Concentrations decreased with depth, water from 2.5 m being clear with little DOC across the paddock and riparian zone throughout the sampling period, while surface water was frequently darkly tannin-stained. This suggests the surface groundwater is not mixing or in recent contact with the deeper groundwater. Further evidence of this is variation in E4:E6 ratios, which showed a higher ratio in the shallower groundwater, indicative of greater fulvic acid fraction (Wrigley et al. 1988). This suggests carbon in surface groundwater is relatively ‘new’, having been recently degraded and that the shallower water is closer to the carbon source (Petrone et al. 2009). However, E4:E6 ratios are all high indicating groundwater carbon is all recently degraded, showing carbon is actively being lost from the soil, which is consistent with Bassendean sands (Gerritse 1994). Bassendean sands have a high leaching capacity, resulting in carbon in the soil being readily lost. Groundwater DOC concentrations and E4:E6 ratios can help explain the hydrology of Bingham Creek. Concentrations are low in 2.5 m depth groundwater, as a result of limited mixing and dilution due to landscape rise in groundwater. The lower E4:E6 ratios in deepest (2.5 m) paddock groundwater can be explained by a dilution effect, with the rise in older groundwater diluting the existing fresher carbon. The higher E4:E6 ratios in groundwater shallower than 1.5 m depth provide an indication that fresh carbon is percolating from surface soils (Wrigley et al. 1988). Furthermore, DOC concentrations in the groundwater at all depths increased over the wet season indicating carbon is being mobilised from surficial organic stores (see next chapter) in the riparian zone into the groundwater through rainfall input over the wet season.

Groundwater redox potentials at Bingham Creek showed clear trends across the paddock and riparian zone and with depth. Redox potentials were negative throughout the riparian zone and decreased with depth; a result of slow groundwater movement, long residence times and low dissolved oxygen concentrations. Previous work has illustrated riparian vegetation can strongly affect underlying redox potentials (Tabacchi et al. 1998; Dwire et al. 2006). Riparian vegetation provides organic matter to underlying groundwater, fuelling microbial respiration and reducing redox potentials. In contrast, redox potentials were less reducing in the paddock most likely due to lack of organic input and the shorter periods of waterlogging.

The limited groundwater movement, lack of aeration and reducing conditions is consistent with groundwater at Bingham Creek being dominated by FRP and ammonium (NH_4-N). FRP is released from metal bonding (e.g. iron) under anaerobic conditions (Vought et al. 1994), which explains why FRP concentrations were high in groundwater at 1.5 m depth. However, FRP concentrations were not high in groundwater at 2.5 m depth, probably as a result of dilution and limited mixing with shallower phosphorus rich groundwater. The increasing redox potential of deeper groundwater can explain why NH_4-N concentrations were highest at 2.5 m depth. Under anaerobic conditions ammonium can be released in groundwater as a result of nitrification (Naiman and Decamps 1997). Furthermore, deeper groundwater is not flushed or aerated so there is no potential for NH_4-N to be released or transformed and it therefore accumulates (Duval and Hill 2007). This is supported by the lower E4:E6 ratios in deepest groundwater (2.5 m) indicating that the water is older and not being flushed.

At Bingham Creek groundwater nutrient concentrations were typically at their highest closer to the surface, particularly FRP and to a lesser extent NH_4-N (not the highest but very high), probably as a result of the proximity to the nutrient source. Riparian vegetation contributes organic matter to soil surface and as this is degraded, NH_4-N is released (Naiman and Decamps 1997), which can explain elevated concentrations closer to the surface where organic matter concentrations were also high. The lack of horizontal water movement means there is little inflow of new nutrients and for nutrient mobilisation to occur there must be rainfall. Rainfall percolates through surface organic matter and surface soils, which have high nutrient concentrations, allowing nutrients to be mobilised into the shallow groundwater.

Total phosphorus and FRP concentrations were at their highest in the riparian zone and in groundwater near the surface. There are three processes which are likely to have affected phosphorus concentrations in riparian groundwater. Firstly, it is probable that stream water was flowing into adjacent shallow groundwater (i.e. a ‘losing’ stream). The high phosphorus concentrations in Bingham Creek, which flow into the riparian zone, raise groundwater phosphorus concentrations in water shallower than 1.5 m. Secondly, the increase in phosphorus concentrations in the riparian zone can be linked to phosphorus release from riparian vegetation. Phosphorus release from riparian vegetation occurs though the breakdown of roots and riparian litter (Narumalani et al. 1997; Tabachhi et al. 2000; Qui et al. 2000). These nutrients are then mobilised through rainfall and groundwater interception. Thirdly, riparian vegetation changes the chemical nature of underlying groundwater, leading to anaerobic and highly reducing conditions, which is fuelled by organic carbon. This in turn can enhance the mobilisation of FRP (Vought et al. 1994). As a result phosphorus concentrations in the riparian zone are increasing at Bingham Creek, although as it occurs in the riparian zone it allows for interception by the riparian vegetation. Within the paddock, groundwater phosphorus concentrations are still considered high (double the ANZECC guidelines for lowland streams), indicating the presence of legacy phosphorus in groundwater. There was no pulse of phosphorus after the paddock was fertilised, indicating there was insufficient rainfall for the mobilisation of phosphorus in the paddock.

At Bingham Creek, nitrogen (NO_x-N, NH_4-N and TN) concentrations decreased over time through the riparian zone (Figure 2-26-2.31). Both NO_x-N and TN are highest in the paddock and decrease through the riparian zone, however, this trend is most apparent for NO_x-N (Figure 2-29 and 2.37). The reduction of NO_x-N and no increase in NH_4-N suggests that the nitrogen is either being assimilated by vegetation or lost through denitrification (Starr and Gillham 1993; Hanson et al. 1994). The physico-chemical nature of deeper underlying groundwater is ideal for denitrification with anaerobic conditions, readily available carbon and strongly reducing conditions (Starr and Gillham 1993; Hill and Cardaci 2004; Dwire et al. 2006). The slow water movement and long residence time of groundwater at Bingham Creek favours denitrification which has been identified in previous studies in Ellen Brook (Peters and Donohue 2001).

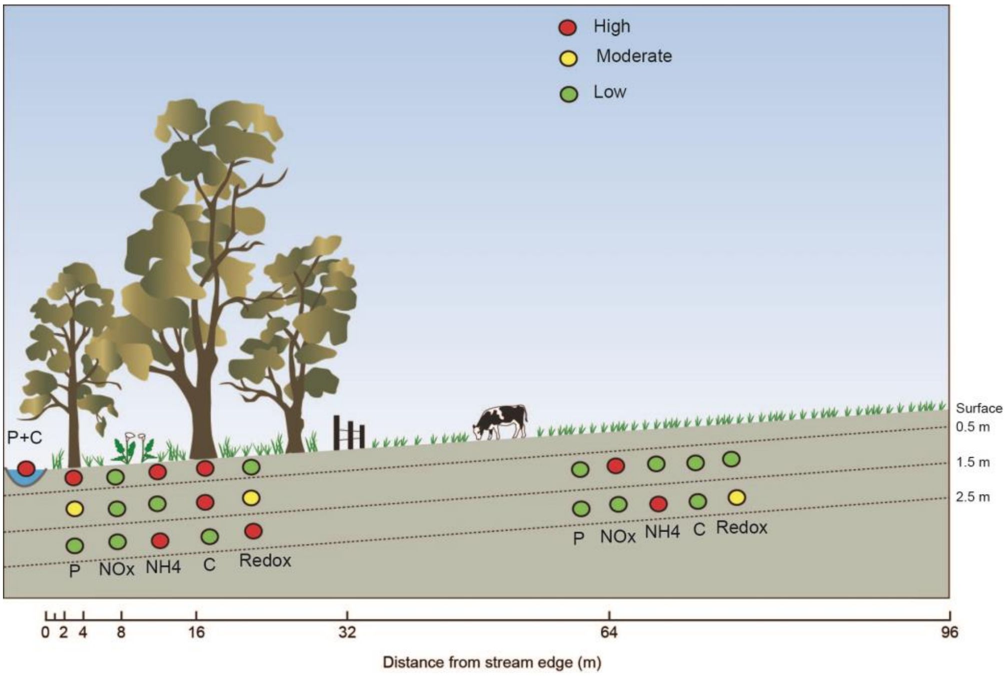


Figure 2-37. A conceptual model comparing groundwater conditions and nutrient concentrations between the paddock and riparian zone at different depths for Bingham Creek**.** High for redox represents strongly reducing conditions and low represents weakly reducing/oxidising

Lennard Brook

Unlike Bingham Creek, there was little variation in the groundwater physio-chemical conditions or nutrient concentrations at Lennard Brook.

At Lennard Brook, the stream water was clear, which was consistent with the groundwater. Groundwater DOC concentrations and gilvin were both low, with the highest concentrations at 0.5 m depth, however, these concentrations were all lower than those at Bingham Creek. DOC concentrations were highest in the riparian zone, as a result of carbon percolating from surficial carbon sources. The carbon that is available has low E4:E6 ratios, indicating groundwater DOC is old, particularly when compared to Bingham Creek (Wrigley et al. 1988). As a result the carbon available is recalcitrant and provides an indication that soil at Lennard Brook is holding onto carbon more effectively than Bingham Creek.

Groundwater redox potentials remained oxidising at Lennard Brook over the wet season, which is likely a result of the hydrology and the potential for flushing. Old water is replaced by new groundwater allowing for aeration of the groundwater to occur. Consequently, oxidising conditions can affect groundwater nutrient concentrations, particularly limiting the release of FRP and NH_4-N (Boomer and Bedford 2008). As a consequence the concentrations of these nutrients were very low at Lennard Brook (Figure 2-38).

Within the riparian zone all nutrient concentrations remained relatively low in comparison to Bingham Creek, which could be due to a number of factors. Firstly, Lennard Brook has greater flow than Bingham Creek, which could be flushing the water and nutrients out of the riparian zone. In contrast, Bingham Creek has little water movement and FRP has accumulated, as seen by the background concentrations in the paddock at Bingham Creek being greater than those in the Lennard Brook riparian zone. Secondly, soil type could be affecting groundwater nutrient concentrations in the riparian zone, with better soils holding onto more nutrients (Vought et al. 1994 and see next chapter). In saying this, riparian vegetation is increasing groundwater phosphorus concentrations, as seen by the higher groundwater concentrations in the riparian zone at both sites. Furthermore, groundwater phosphorus concentrations were higher in the 0.5 m depth groundwater in the riparian zone, compared to the stream.

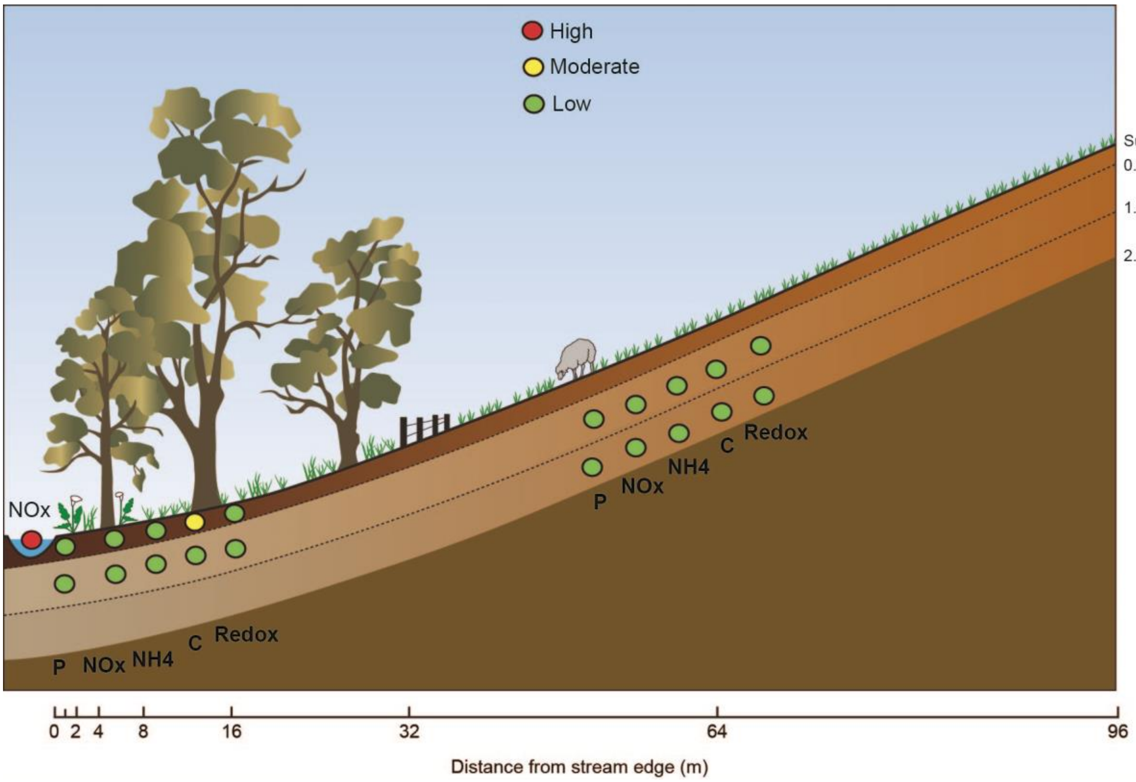


Figure 2-38. A conceptual model comparing groundwater conditions and nutrient concentrations between the paddock and riparian zone at different depths for Lennard Brook. High for redox represents strongly reducing conditions and low represents weakly reducing/oxidising

Is water flow through the riparian zone bypassing the riparian vegetation?

For riparian vegetation to facilitate the removal of nutrients, the groundwater must interact with the riparian vegetation. At Lennard Brook there is evidence of surface flow due to the steeper slope, whereas Bingham Creek is lacking horizontal flow above or below ground. Therefore, the phosphorus removal capacity at Bingham Creek is reduced. The most effective phosphorus removal mechanism occurs during surface flow, where trapping of particulate phosphorus occurs. The main phosphorus removal pathway is through surface trapping, with much of this in particulate form (Vought et al. 1994; Narumalani et al. 1997; Tabacchi et al. 1998; Brian et al. 2004; Ballantine et al. 2008; Knight et al. 2010). As a consequence, the phosphorus removal capacity at Bingham Creek is likely to be limited, and will not be as high as previous studies. Phosphorus removal by vegetation from groundwater is limited, because the phosphorus requirements of the vegetation are not high (Jackson et al. 1997; Hinsinger 2001).

At Bingham Creek water movement is primarily restricted belowground and there is a slope on the water table towards the stream. This coupled with the presence of permeable sands is resulting in the likely (but slow) horizontal movement from the paddock to the stream however, the time interval is unknown. While the groundwater is possibly flowing into the riparian zone, it is predominantly below the feeding root zone (active zone). Though it is not below the tap root zone of trees so there will be some nutrient uptake by trees (Stone and Kalisz 1991; Canadell et al. 1996). Globally, approximately 75% of plant roots are in the top 40cm of soil (Jackson et al. 1996). For nutrient uptake and removal to occur it is imperative that groundwater moves through this active zone, particularly for nitrate reduction (Dhondt et al. 2006). Considering the limited surface flow at Bingham Creek, it is essential for groundwater to be interacting with the active root zone to ensure nutrients are intercepted.

At Bingham Creek however, the groundwater is only intercepting the root zone 0-8m from the stream. This interaction occurs for the majority of the wet season, when nutrients in the riparian zone are being mobilised, however, less than half of the riparian zone is actively interacting with the groundwater and nutrients, although in wetter years the level of interaction could be greater. Besides this interaction, the water slope suggests that water is moving from the stream into the riparian zone during baseflow and first flush, then back out during the wet season, some of which occurs in the active root zone. This provides an indication that water is cycling through the riparian zone from the stream and to a lesser extent, the paddock (Figure 2-35). As a result of this cycling, long residence times and reducing conditions, there is the capacity for FRP release, however the flowpaths into the riparian zone illustrates the capacity of riparian vegetation to process water from the stream and paddock to differing extents.

At Lennard Brook there is evidence of surface flow from the paddock into the riparian zone. This allows for riparian vegetation to intercept surface flows, reducing the movement of particulate-bound phosphorus through the riparian zone to the stream. Furthermore, Lennard Brook is characterised by shallow groundwater within the riparian zone. As a result groundwater was in the active root zone for the whole wet season, maximising the nutrient uptake capacity of riparian vegetation. The interaction was occurring throughout the whole riparian zone. This highlights how slope and an increased clay content within deeper soils can affect the movement of water through the riparian zone. It results in the water coming into continuous contact with the riparian zone above and below ground maximising the nutrient removal potential of riparian zones. Furthermore, it emphasises the variability in hydrology between Bingham Creek and Lennard Brook, which has shown to affect nutrient dynamics.

Chapter 3 Soil

Introduction

Soil is often the first point of contact for incoming nutrients, both at the surface and below ground. Soil plays a pivotal role in hydrology, nutrient dynamics, plant growth and decomposition rates (Lyons et al. 1998). The movement of water through the soil, its interaction with the soil and the relative proportions of above and belowground flow can influence riparian zone nutrient removal efficiency (Fennessy and Cronk 1997).

As described in the previous chapter, soil type, particularly texture, affects the hydrology of the riparian zone. Soil type not only affects the physical nature of soils but also their chemical and microbial properties, particularly in relation to nutrient dynamics (Tabacchi et al. 2000; Fuchs et al. 2009; Kang et al. 2011; Obour et al. 2011). For example, clays typically have high phosphorus concentrations because the smaller clay particles have greater surface area and binding sites compared to sands (Ballantine et al. 2009; Coyne and Thompson 2006; Cross and Schlesinger 1995; Obour et al. 2011; Young 1997). As the Ellen Brook catchment is mostly underlain with Bassendean sands, extensive use of phosphorus fertiliser is required in agriculture to make up for the phosphorus deficient soils, however due to its poor nutrient holding capacity, much of this added phosphorus is lost to runoff during winter rains (Barron et al. 2008; Summers et al. 1999).

Phosphorus removal capacity and soil nutrient concentrations are affected by chemical composition, fertiliser use and organic matter accretion rates (Tan 2000). Previous studies have shown soil phosphorus concentrations are higher in regions where the soil is rich in aluminium, iron, potassium or calcium. The relative proportions of each of these ions is governed by soil pH (Coyne and Thompson 2006; Jones 2001; Obour et al. 2011). The ions provide adsorption sites for phosphorus to bind to and also influence the phosphorus speciation in soils. Soils that are rich in calcium have been linked to high apatite phosphorus concentrations (Ann et al. 1999). Bonding in apatite molecules is little affected by the redox potential of the soil, while bonding to metal such as iron and aluminium is strongly redox dependent (Tan 2000). Furthermore, soil pH is a controlling factor of soil apatite phosphorus concentrations, with lower pH more conducive to high phosphorus concentrations (Ann et al. 1999). Phosphorus retention (PRI) and phosphorus buffering indexes (PBI) have been developed that measure the cumulative ability of different processes within the soil to retain phosphorus. For example, soils that have a high clay and reactive iron content have a higher phosphorus retention index than sands with no clay or iron.

Microorganisms in soil also influence phosphorus availability and speciation. Microbial growth in soil can act like roots of vegetation increasing the movement of organic phosphorus (Richardson and Simpson 2011). Soil microbes mediate the solubility and mineralisation of phosphorus (Richardson et al. 2009). Soil type can affect microbial proliferation and their effectiveness in breaking down organic matter in soil (Hassink et al. 1993). Microbial activity is greater in sandy soils, as soils with greater clay fraction hold onto more organic matter (Hassink et al. 1993).

Unlike phosphorus there is limited capacity for soils to bind nitrogen. Nitrogen is mediated primarily by microbial processes and organic matter in the soil (see below). Mineralisation of soil nitrogen by microbial communities directly influences plant productivity (Vitousek and Howarth 1991; Anttonen et al. 2002). Nitrogen enters soils through fixation from the atmosphere by biological fixation, much of which is assimilated by microbes and plants (Vitousek et al. 1997; Scharenbroch and Lloyd 2004). When these organisms die the nitrogen is released. Consequently up to 90% of terrestrial nitrogen occurs in soil organic matter (Pulford 1991). Denitrification is a microbially mediated process within soil, facilitating the complete removal of nitrogen from the soil (Kadlec and Knight 1996; Craft 2001). Soil types with high water holding capacities stimulate low redox conditions, facilitating conditions ideal for denitrification where nitrate is present (Kadlec and Knight 1996; Tan 2000).

Riparian vegetation influences soils through a number of processes, which alter the soil structure and influence soil nutrient dynamics. Roots of riparian vegetation enhance the soil structure, reducing the potential for erosion to occur and limiting the release of nutrients (Vought et al. 1994; Mander et al. 1997; Lyons et al. 2000; Easson and Yarbrough 2002; Dosskey et al. 2010; Raty et al. 2010). Roots can also increase pore sizes in soil, aerating soils, encouraging the growth of microbial communities and enhancing water infiltration rates. This allows for water to be intercepted and for nutrient transformations by plants and microbes to occur. Riparian vegetation increases roughness of soil surface, reducing overland flow and facilitating the trapping of particulate nutrients (Dosskey et al. 2010).

Riparian vegetation contributes large quantities of organic matter to underlying soils, which influences soil carbon dynamics. Within soil, microorganisms are primarily responsible for the regulation of organic matter accumulation and are able to transform organic matter into useful by-products such as inorganic phosphorus and nitrogen (Craft 2001; Reddy and DeLaune 2008; Young 1997). The breakdown of organic matter in soils facilitates the release of carbon to the soil, which has a central role in denitrification, as carbon is an electron acceptor in microbial respiration (Craft 2001; Reddy and DeLaune 2008; Tan 2000). Different soil types have varying abilities to integrate carbon, those with more binding sites (e.g. clay) are able to hold onto more carbon compared to Bassendean sands, which have shown to leach carbon from the soil (Barron 2008).

In this chapter, we compare the soil type and characteristics and nutrient dynamics of two sites within Ellen Brook: Bingham Creek on Bassendean sands and Lennard Brook, which has sands that are rich in iron. This chapter investigates how soil nutrient dynamics in the paddock, riparian zone and stream differ between sites and how phosphorus retention varies spatially. The key questions being addressed are:

- Where and in what forms are nutrients being stored in the stream, riparian and paddock soils?
- Are there differences in soil nutrient concentrations between Bingham Creek and Lennard Brook?
- Is riparian vegetation affecting the nutrient dynamics of underlying soils?

Methods

Soil sampling

Initial soil samples were collected in October 2011 from Bingham Creek. The sampling area was split into three zones: the stream, riparian zone and paddock, with three randomly selected sites from each zone. Samples were taken at two depths: the top 0.05m was sampled using a trowel and samples at 0.5 m depth were taken using a hand auger. Three samples from each zone and depth were collected (with representative sample collected and frozen for nutrient analysis in the laboratory), air-dried and stored for later use.

In 2012, soil samples were collected in April and June, from Bingham Creek and Lennard Brook. The sampling area was again split into three zones: the stream, riparian and agricultural zone, with three randomly selected replicate samples from each zone. Four depths were sampled, the top 0.05, 0.5, 1.5 and 2.5 m, however samples for each depth were often not possible. At Bingham Creek all replicates were attainable, with the exception of 2.5 m samples from the stream. The saturated soil profile at Lennard Brook made collection difficult, only samples from the top 0.05m and the paddock had full replication. In the riparian zone, only one sample was collected for 1.5 m and 2.5 m. In the stream there were no samples from 0.5 m, 1.5 m and 2.5 m due to continuous stream flow.

The top 0.05m was sampled using a trowel and the 0.5 m, 1.5 m and 2.5 m were sampled using a hand auger. A mechanical auger was used to collect the 0.5 m, 1.5 m and 2.5 m soil samples from the paddock at Lennard Brook due to laterite formations within the soil making the use of a hand auger impossible.

Particle size analysis

Soil particle size analysis was carried out by CSBP Soil and Plant Laboratory, using a method adopted from Indorante et al. (1990). Proportions of four different size categories shown in Table 3-1 were provided by this method.

Table 3-1. Scale used for soil particle size analysis

Particle type	Size (μm)
Coarse sand	200-2000
Fine sand	20-200
Silt	>2-<20
Clay	<2

Soil nutrient analysis

All soil samples collected were analysed for organic matter, total kjeldahl nitrogen, total phosphorus, organic phosphorus, 1M HCL extractable phosphorus and 1M NaOH extractable phosphorus and total extractable iron according to the methods listed in Table 3-2. In this study 1M HCL extractable phosphorus represents apatite phosphorus and 1M NaOH represents non-apatite phosphorus.

Phosphorus absorbance

Phosphorus retention and phosphorus buffering indices were calculated, together with Colwell phosphorus analysis according to the methods listed in Table 3-2. The phosphorus retention index is the Western Australian standard and was developed for sandy soils of south-western Australia (Bolland et al. 2003). The phosphorus buffering index is advocated as the national standard and assesses the soil's ability to bind and release phosphorus for plant uptake. Colwell phosphorus was analysed to provide an estimate of previously adsorbed phosphorus, thereby increasing the accuracy of both indices (Bolland et al. 2003). These analyses were carried on all soil samples from 2012.

Table 3-2. Methods used for soil analysis

Parameter	Units and DL (if applicable)	Method	Lab
Organic matter	%	Automated combustion followed by NDIR method (Dean 1974)	MAFRL[a]
TP	mg.P/kg (<5)	Kjeldahl digestion with P measured by colorimetry (Aspilla et al. 1976)	MAFRL[a]
Organic P	mg.P/kg (<5)	Digestion of ashed sample using hydrochloric acid, followed by titration of phenolphthalein (Aspilla et al. 1976)	MAFRL[a]
1M HCL extractable P	mg.P/kg (<5)	Modification of Williams et al. (1976) method, extracted by hydrochloric acid	MAFRL[a]
1 M NaOH extractable P	mg.P/kg (<5)	Modification of Williams et al. (1976) method, extracted by sodium hydroxide	MAFRL[a]
TKN	mg.N/g (<0.04)	Involves the addition of copper sulphate and digestion in sulphuric acid	MAFRL[a]
Particle size analysis	__	Indorane et al. (1990) method (sand 20-2000µm, silt 2-20µm, clay <0.002-2µm)	CSBP[b]
Colwell P	mg.P/kg	Colwell (1965) method, extraction by sodium bicarbonate, P measured by colorimetry	CSBP[b]
PRI	----	Adaptation of Allen and Jeffrey (1990) method, P extracted by potassium chloride	CSBP[b]
PBI	----	Adapted from Rayment and Lyons (2011), P extraction and colorimetry analysis	CSBP[b]
Extractable Fe	mg.Fe/kg (<5)	Adapted from Standards Australia (1999), acid digestion and spectrometric analysis	MAFRL[a]

[a] Marine and Freshwater Research Laboratory

[b] CSBP Soil and Plant Analysis Laboratory, South Lake

Results

Physical soil characteristics

Soil particle size analysis
At Bingham Creek, coarse sand contributes greater than 70% of all soil material across each zone and depth. The clay component increases with depth, except in the riparian zone (Figure 3-1a-c).

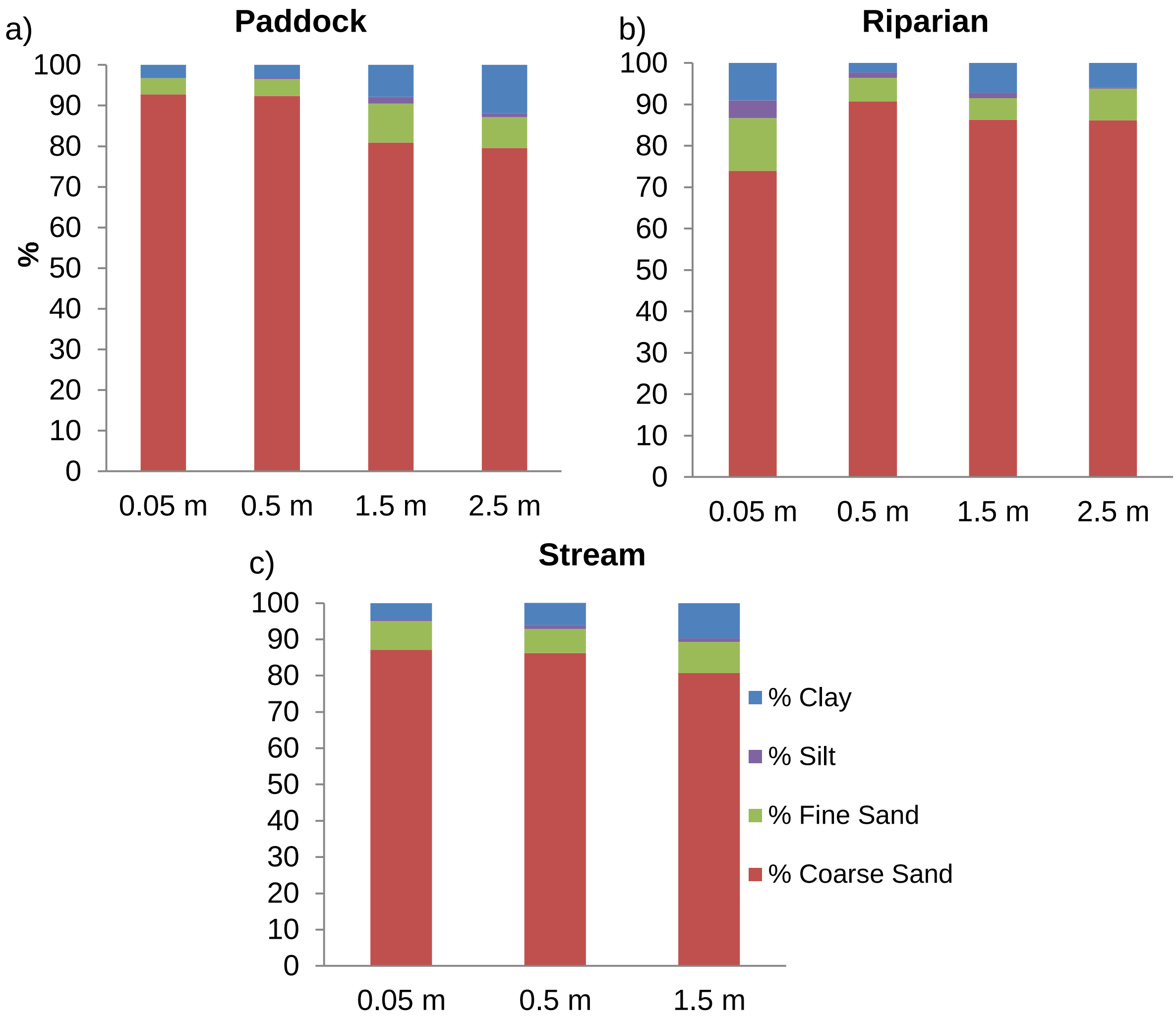


Figure 3-1. Comparison of particle size analysis across the a) paddock b) riparian zone c) stream across four different depths at Bingham Creek

Lennard Brook soils are also largely composed of coarse sand, but not to the same extent as at Bingham Creek. Within the paddock, coarse sand contributes 70%+ of soil material and decreases with depth as the clay fraction increases (Figure 3-2a). In the riparian zone there is no clear pattern, however coarse sand is the main contributor (Figure 3-2b). The streambed is underlain primarily by coarse sand (Figure 3-2c). Silt fractions are noticeably larger here than at Bingham Creek, especially in the riparian zone and paddock.

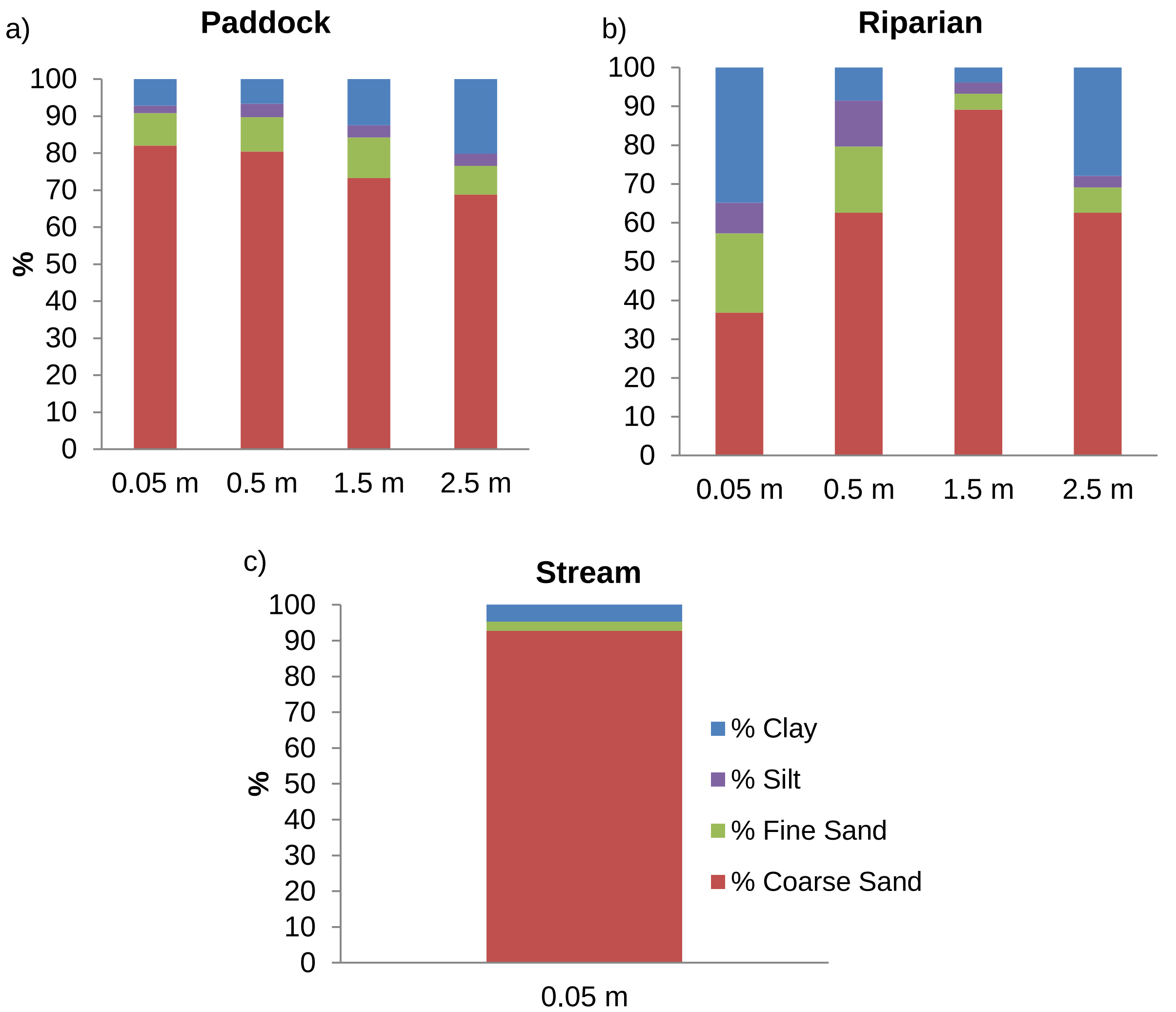


Figure 3-2. Comparison of particle size analysis across the a) paddock b) riparian zone c) stream across four different depths at Lennard Brook, note only one replicate for 1.5 m and 2.5 m samples in the riparian zone

Soil chemical composition

Lennard Brook soils have very high total extractable iron concentrations in the paddock, the highest at 1.5 m depth (~110,000mg.Fe/kg; Figure 3-3b). In comparison, Bingham Creek has little iron, with surface riparian soils having the highest concentration (~5,500mg.Fe/kg). There was practically no iron in soils at the surface or at 0.5 m depth in the paddock (Figure 3-3a).

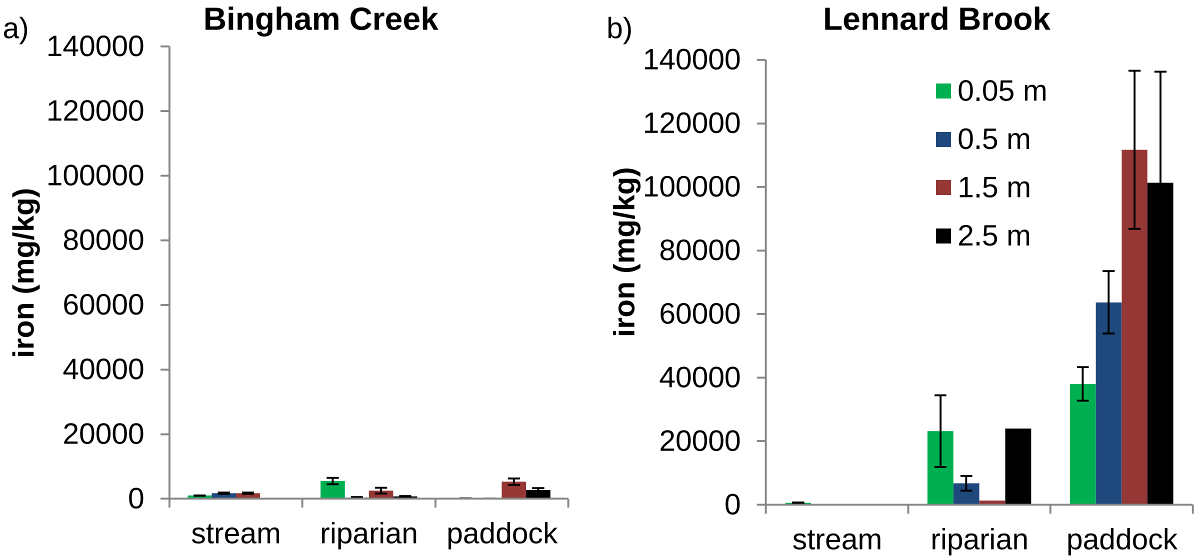


Figure 3-3. Comparison of total extractable iron concentrations between a) Bingham Creek and b) Lennard Brook across the stream, riparian zone and paddock across four depths, note only one stream sample and replicates of one for 1.5 m and 2.5 m Lennard Brook riparian soils

Soil organic matter at Bingham Creek was highest in surface soils, with little or no organic matter at any other depth (Figure 3-4a-b). At Lennard Brook the soil had a greater proportion of organic matter, with concentrations in surface riparian soils approximately 4 times greater than highest Bingham Creek concentrations (Figure 3-4c). Organic matter decreased markedly with depth.

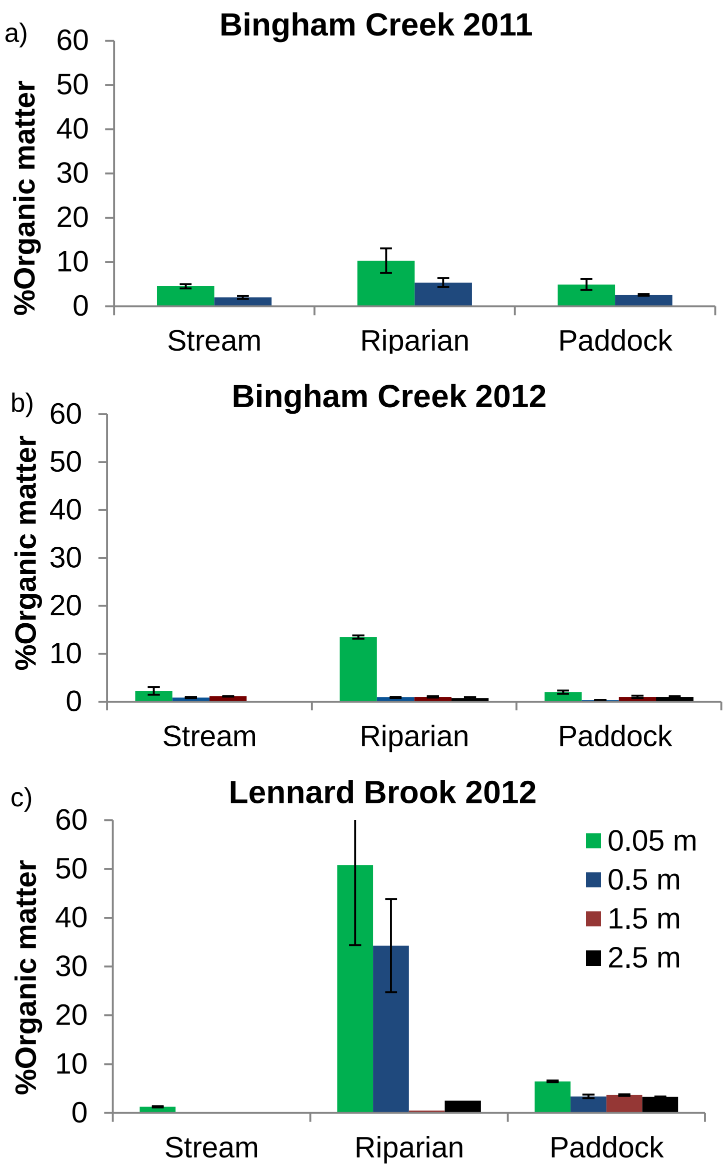


Figure 3-4. Comparison of percentage soil organic matter across the a) Bingham Creek 2011 b) Bingham Creek 2012 c) Lennard Brook at four different depths, note Lennard Brook has only one replicate for 1.5 m and 2.5 m samples in the riparian zone

Soil nutrient concentrations

Bingham Creek soils had low total phosphorus in the paddock and stream. The highest concentrations occurred in the top 0.05m of riparian soils and concentrations decreased with depth (Figure 3-5a-b). Lennard Brook soils had high total phosphorus concentrations, particularly in the paddock, where they were twice as high as maximum Bingham Creek concentrations (Figure 3-5c). Similarly, total phosphorus concentrations were highest in surface riparian soils at both sites.

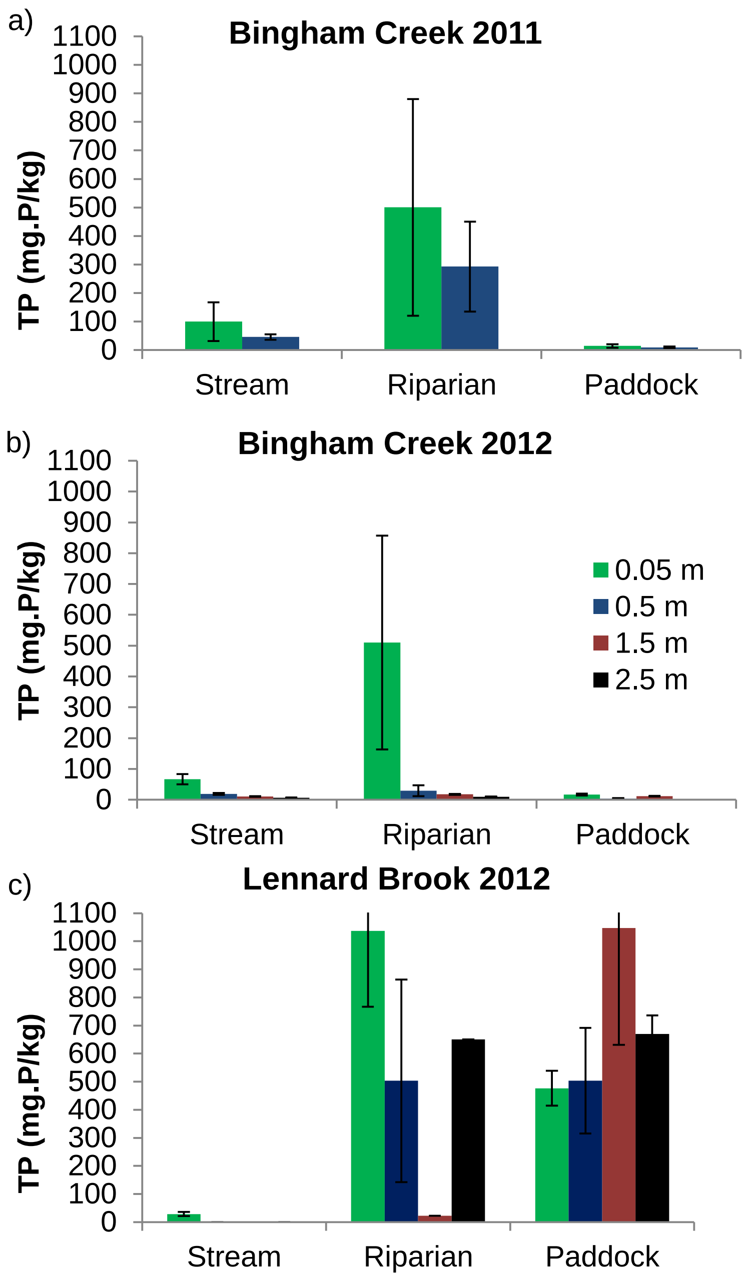


Figure 3-5. Comparison of total phosphorus concentrations across the a) Bingham Creek 2011 b) Bingham Creek 2012 c) Lennard Brook at four different depths, note Lennard Brook has only one replicate for 1.5 m and 2.5 m samples in the riparian zone

Bingham Creek soils had little or no organic phosphorus in the paddock or stream and the highest concentrations occurred in riparian surface soils (Figure 3-6a-b). Organic phosphorus made up approximately 25% of soil total phosphorus. Lennard Brook had substantially higher organic phosphorus concentrations in the riparian zone and paddock, also with highest concentrations in surface riparian soils (Figure 3-6c). Approximately 36% of total soil phosphorus was made up of organic phosphorus at Lennard Brook.

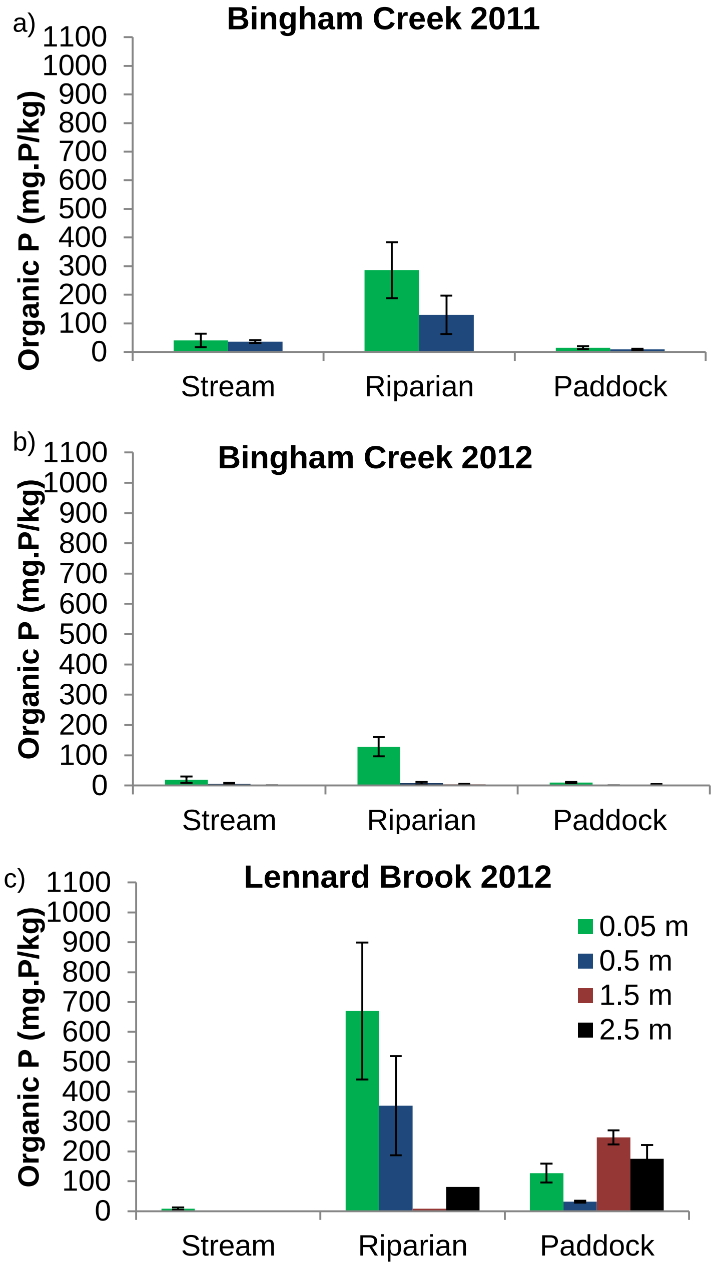


Figure 3-6. Comparison of organic phosphorus concentrations across the a) Bingham Creek 2011 b) Bingham Creek 2012 c) Lennard Brook at four different depths, note Lennard Brook has only one replicate for 1.5 m and 2.5 m samples in the riparian zone

There were large variations in 1M NaOH (non apatite) extractable phosphorus concentrations at Bingham Creek over 2011 and 2012, particularly in the riparian zone. In 2011, concentrations were twice the 2012 values (Figure 3-7a-b), however, concentrations were once again highest in surface riparian soils. Lennard Brook soils had consistent 1M NaOH extractable phosphorus concentrations in the paddock, which were comparable to surface riparian soils (Figure 3-7c).

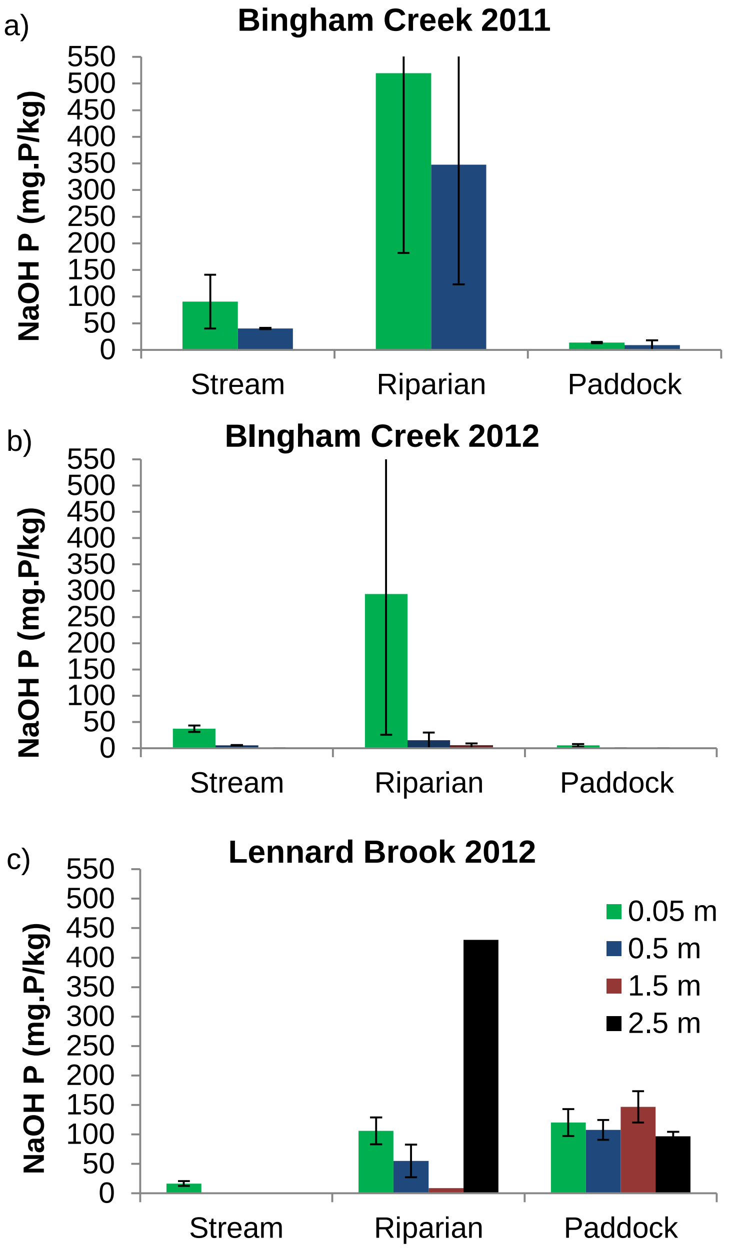


Figure 3-7. Comparison of 1M NaOH phosphorus concentrations across the a) Bingham Creek 2011 b) Bingham Creek 2012 c) Lennard Brook at four different depths, note Lennard Brook has only one replicate for 1.5 m and 2.5 m samples in the riparian zone

Similar trends occurred for 1M HCl (apatite) extractable phosphorus but concentrations were substantially lower than 1M NaOH extractable phosphorus (Figure 3-8a-c). There was a clear decrease in concentration with depth with the exception of riparian soil at 2.5 m depth at Lennard Brook. The highest concentrations for Bingham Creek and Lennard Brook occurred in surface riparian soils.

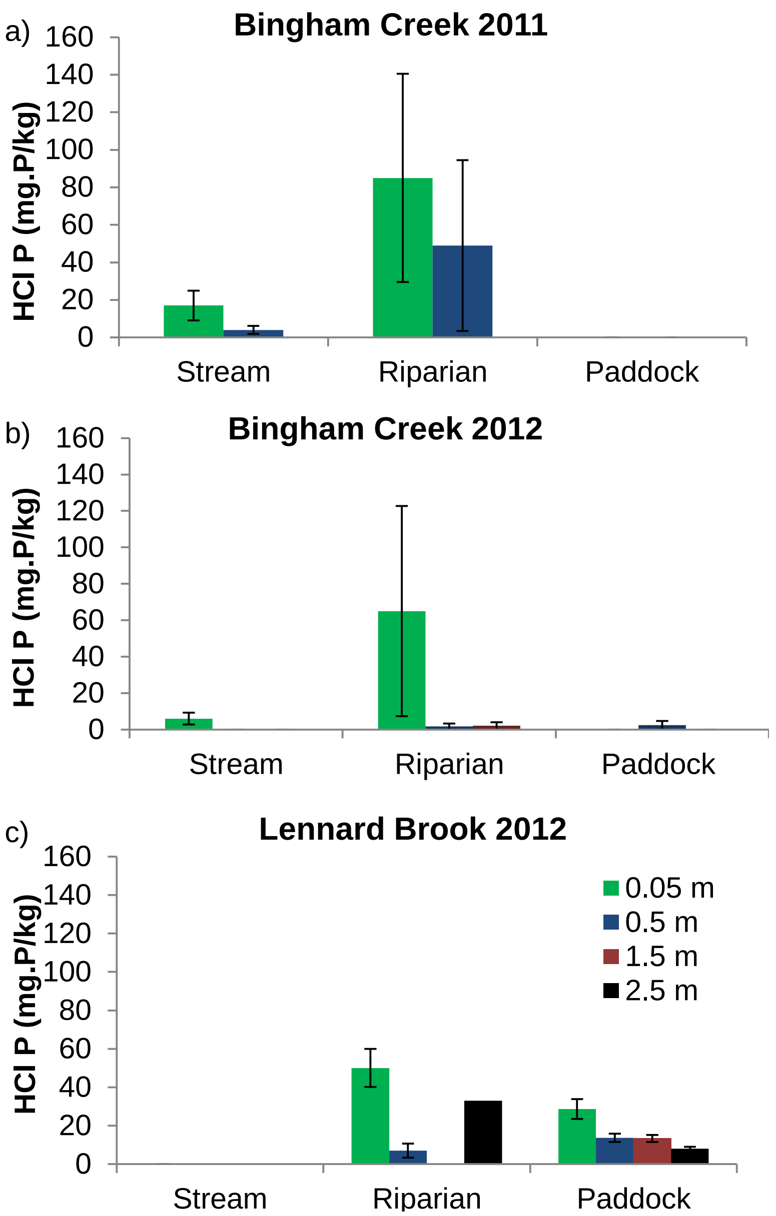


Figure 3-8. Comparison of 1M HCL phosphorus concentrations across the a) Bingham Creek 2011 b) Bingham Creek 2012 c) Lennard Brook at four different depths, note Lennard Brook has only one replicate for 1.5 m and 2.5 m samples in the riparian zone

Total Kjeldahl Nitrogen concentrations were low at Bingham Creek, with concentrations highest at the soil surface and decreased with depth, and the highest concentrations were in surface riparian soils (Figure 3-9a-b). At Lennard Brook nitrogen was only detectable at 0.5 m depth and in surface soils (Figure 3-9c). Concentrations were highest in surface riparian soils.

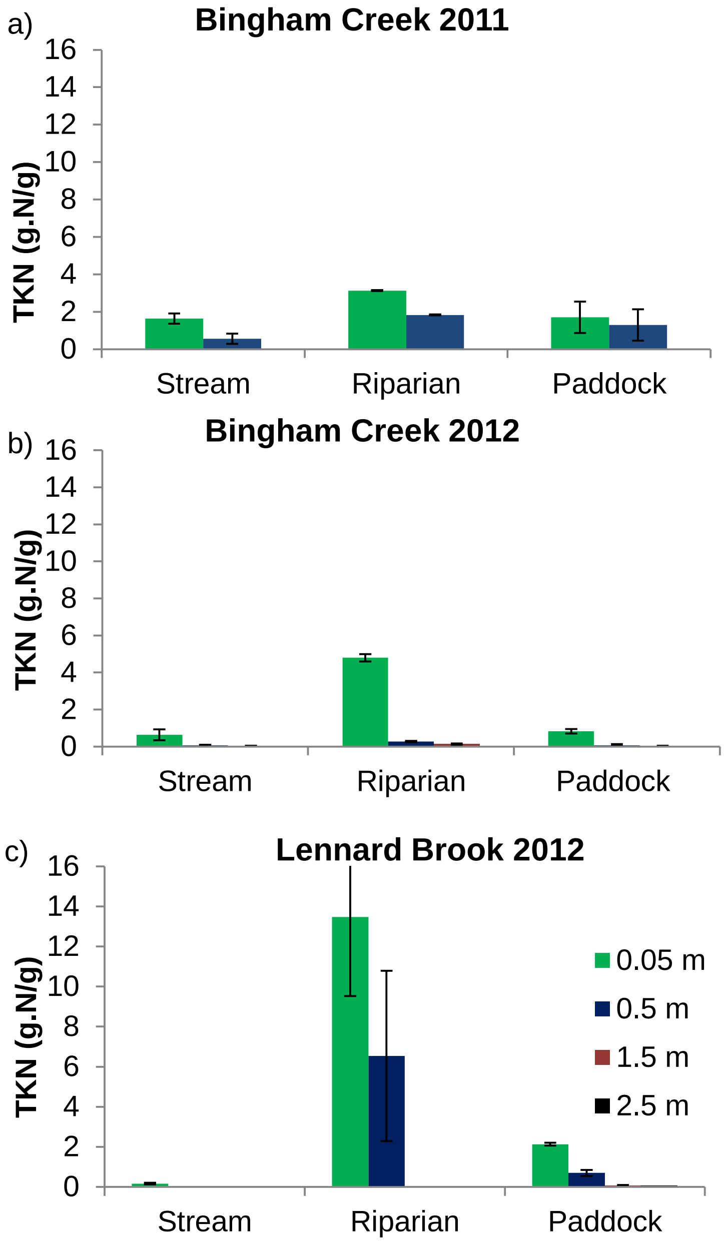


Figure 3-9. Comparison of total kjeldahl nitrogen concentrations across the a) Bingham Creek 2011 b) Bingham Creek 2012 c) Lennard Brook at four different depths, note Lennard Brook has only one replicate for 1.5 m and 2.5 m samples in the riparian zone

Soil phosphorus retention

Colwell phosphorus concentrations at Bingham Creek were highest in surface soils and decreased exponentially with depth. There was little or no Colwell phosphorus in paddock soils (Figure 3-10a). At Lennard Brook there was a similar pattern with concentrations highest in surface soils and decreasing with depth. Concentrations were highest in riparian soils, but unlike at Bingham Creek there was substantial Colwell phosphorus in paddock soils as well (Figure 3-10b).

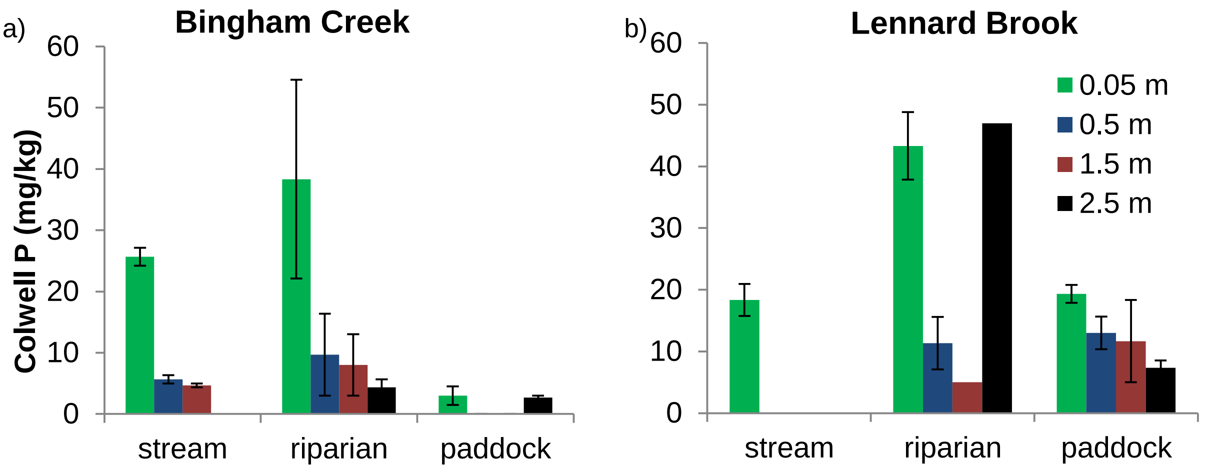


Figure 3-10. Comparison of Colwell phosphorus concentrations between a) Bingham Creek and b) Lennard Brook across the stream, riparian zone and paddock across four depths, note only one stream sample and replicates of one for 1.5 m and 2.5 m Lennard Brook riparian soils

The phosphorus retention index (PRI) at Bingham Creek was highest in surface riparian soils at approximately 5. The PRI of paddock surface and 0.5 m soils was zero (Figure 3-11a). Comparatively, Lennard Brook had exceptionally high PRI values in surface riparian soils (~580), which decreased with depth. However, in the paddock PRI increased with depth (Figure 3-11b). The only comparable points between sites were stream samples at Lennard Brook (PRI ~4) and surface riparian soil at Bingham Creek.

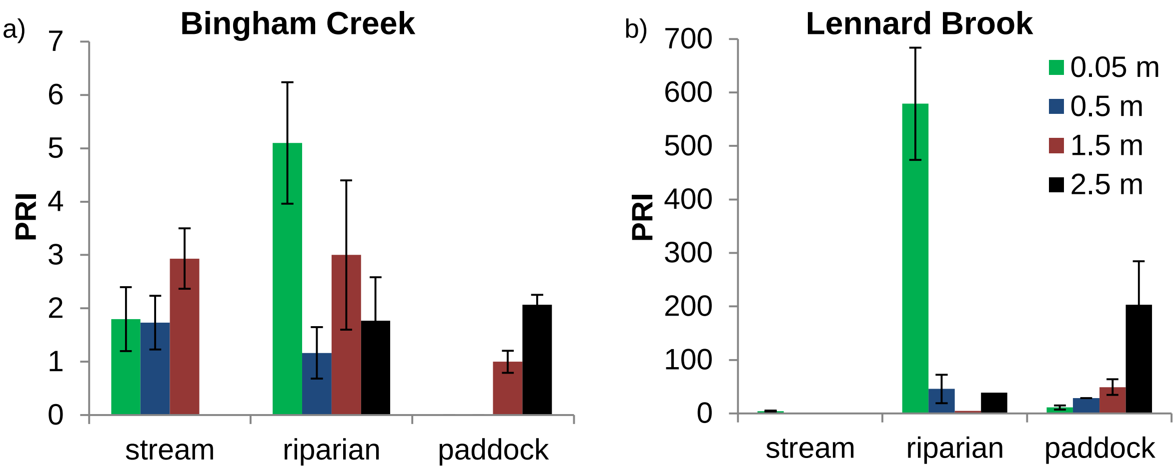


Figure 3-11. Comparison of PRI values between a) Bingham Creek and b) Lennard Brook across the stream, riparian zone and paddock across four depths, note only one stream sample and no replicates for 1.5 m and 2.5 m Lennard Brook riparian soils. Note two orders of magnitude difference in y axis values

The phosphorus buffering index (PBI) at Bingham Creek was highest in surface riparian soils and decreased with depth. In the paddock, PBI values increased with depth (Figure 3-12a). Comparatively, Lennard Brook PBI values were much higher, reaching a maximum of approximately 325 in the surface riparian soils (Figure 3-12b) and the PBI in paddock increased with depth. The only comparable points were the stream at Lennard Brook and the surface riparian soils at Bingham Creek.

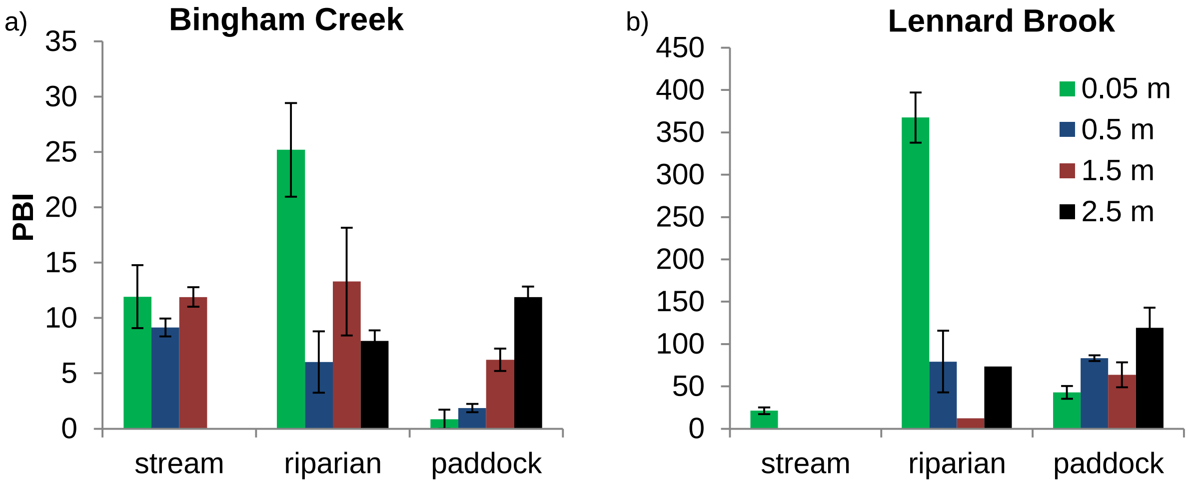


Figure 3-12. Comparison of PRI values between a) Bingham Creek and b) Lennard Brook across the stream, riparian zone and paddock across four depths, note only one stream sample and no replicates for 1.5 m and 2.5 m Lennard Brook riparian soils. Note order of magnitude difference in y axis scale

Discussion

Physical and chemical composition of soils

Although the soils at Bingham Creek were very similar to Lennard Brook, both dominated by coarse sand, the hydrology and nutrient dynamics of the soils were quite different. This can be explained by differences in soil texture and chemical composition.

Soil texture and composition have a strong influence on water movement and nutrient dynamics in riparian zones (Ward and Robinson 2000; Richardson et al. 2001; Coyne and Thompson 2006). Landscapes characterised by sand are often dominated by subsurface flow (Ward and Robinson 2000), in accord with the lack of surface flow, noted in the last chapter, at Bingham Creek. At Lennard Brook there was greater variability with increasing clay content with depth. The lower permeability of clay suggests that soil composition could create a less permeable layer at depth affecting water movement. Water that passes readily through the upper layers of coarse sand would meet greater resistance with depth and tend to flow horizontally with the slope above the impermeable layer. Furthermore, increasing clay content would slow groundwater movement at depth, due to reduced pore space (Coyne and Thompson 2006).

The physical makeup of soils not only affects hydrology, but also soil nutrient dynamics. In particular, soil texture can affect the nutrient binding capacity of soils (Vought et al. 1994). Both sites were dominated by sands, which structurally have few available binding sites for nutrients, particularly phosphorus (Tan 2000; Coyne and Thompson 2006). However, at Lennard Brook there was a greater clay content, which has the capacity to increase available binding sites (Vought et al. 1994; Lyons et al. 1998; Tan 2000). Therefore it is expected soil phosphorus concentrations would be higher in Lennard Brook soils, particularly with increasing depth.

The chemical makeup of the soils at Bingham Creek and Lennard Brook would also affect soil phosphorus and nitrogen dynamics, particularly iron and organic matter content. In soil, ions such as aluminium, potassium, calcium and iron can all affect soil phosphorus concentrations (Coyne and Thompson 2006; Jones 2001; Obour et al. 2011). These ions provide adsorption sites for phosphorus to bind to and having more of these ions can result in higher soil phosphorus concentrations (Lyons et al. 1998; Obour et al. 2011). Iron concentrations were substantially higher at Lennard Brook than Bingham Creek (Figure 3-1). This was not surprising given the orange colour of the soil at Lennard Brook and the proliferation of lateritic deposits throughout the soil column in the paddock. The high iron concentrations in the paddock could act as a store for phosphorus added from fertiliser use. Interestingly, at Lennard Brook there were high iron concentrations in surface riparian soils but little with increasing depth. Surficial iron is likely to have been delivered from the paddock to the riparian zone during overland flow. At Bingham Creek concentrations were very low with virtually no iron in shallow paddock soils. This disparity in soil iron concentration highlights how chemically deficient Bingham Creek soils are, which in turn affects their capacity to store phosphorus and their potential to uptake phosphorus from groundwater.

Soil organic matter content was primarily restricted to the top 0.5 m of soils at both sites. Organic matter is typically confined to surface soils, due to litter accretion rates and the limited physical turnover in soils (Jobbagy and Jackson 2000; Collins and Kuehl 2001). Furthermore, organic matter content was highest in surface riparian soils, assuming that vegetation is actively contributing organic matter to underlying riparian soils (discussed in the next chapter). Factors that affect soil organic matter accretion rates include vegetation type, soil type and soil moisture (Jobbagy and Jackson 2000), all of which vary between Bingham Creek and Lennard Brook. Physical makeup of soil can also affect organic matter retention rates. The greater proportion of coarse sand at Bingham Creek can explain the lower proportion of organic matter in soils. This is because the Bassendean sands have a limited retention capacity and organic matter can be stripped away due to water flow through the soils. Soil organic matter content is an important component of soils, which influences nitrogen and to a lesser extent soil phosphorus dynamics (Vought et al. 1994; Jobbagy and Jackson 2000). Typically, soil nitrogen is stored in the organic matter and there is a strong correlation between high organic matter content and elevated soil nitrogen concentrations (Collins and Kuehl 2001; Porporato et al. 2003; Brovelli et al. 2012). The outcome for this study is that elevated iron concentrations and organic matter content in the riparian soils at Lennard Brook results in a far greater phosphorus and nitrogen interception and storage capacity than at Bingham Creek.

Where and what forms of nutrients are being stored in the stream, riparian and paddock soils and are there differences between Bingham Creek and Lennard Brook?

Nutrients differ in their concentration and form between the stream, riparian zone and paddock, which was apparent at Bingham Creek and Lennard Brook, and this can be explained by differences in the physical and chemical composition of soils.

Bingham Creek

At Bingham Creek, soil phosphorus concentrations were highest in surface riparian soils (Figure 3-13). This is likely to be the product of an increased organic matter contributed by riparian vegetation through litterfall and plant death (Jobbagy and Jackson 2000; Raty et al. 2010). The organic matter improves phosphorus uptake capacity and storage as described above, but also contains phosphorus absorbed and stored by riparian vegetation (Lyons et al. 1998; Raty et al. 2010). Due to the limited turnover of riparian soils and no overland flow there is no mechanism for soil or organic matter to be moved from the riparian zone or to greater soil depths, resulting in phosphorus accumulation in surface riparian soils (Figure 3-13).

Phosphorus concentrations were low in paddock soils which could be due to a number of reasons. Firstly, Bassendean sands are classified as phosphorus deficient soils,which have very low natural phosphorus concentrations (Barron et al. 2008). Surface paddock soils had a PRI of zero (Figure 3-15a) indicating there is little or no potential for the soil to retain phosphorus as seen in previous studies (Bolland and Allen 2003). Secondly, while these soils require high fertilisation rates to promote agricultural productivity (Bolland and Allen 2003;

Barron et al. 2008) fertiliser has not been applied to Bingham Creek soils for over a decade, so little P remains.

Interestingly, the surface stream soils had the second highest phosphorus concentration compared to surface riparian soils. Bingham Creek has very high instream filterable reactive phosphorus concentrations which can be being transferred to the soil through sorption and coprecipitation, resulting in the corresponding higher concentrations (House 2003; Noll et al. 2009). Secondly, it is an intermittent stream, dominated by terrestrial plants during periods of no flow, allowing a buildup of organic matter and higher soil phosphorus during dry periods.

Phosphorus in soils can exist as either organic or inorganic and typically the inorganic fraction is greater than the organic component (Tan 2000). This was consistent with the results of this study, with the organic phosphorus component comprising approximately 25% of total phosphorus. Organic P was higher in riparian zone soils, explained by their higher organic matter content (Tan 2000). The NaOH extractable phosphorus was the next largest phosphorus component. Orthophosphate is a component of NaOH extractable phosphorus which can range from 27-90% (Doolette et al. 2011). The greater the proportion of NaOH extractable phosphorus the more available orthophosphate is for plant uptake and consequently the release into water (Doolette et al. 2011), however, it was only encountered in soils shallower than 0.5 m depth and in very low concentrations. This provides an indication that phosphorus in Bingham Creek soils has a small labile fraction, which is restricted to surface soils.

Nitrogen concentrations at Bingham Creek were also highest in surface soils, particularly in the riparian zone (Figure 3-13). There was little to no nitrogen in soils at 0.5 m depth and below, and little variability in concentrations throughout the soil profile. Nitrogen is often a limiting element for plant growth, which can explain why concentrations are lower in deeper soils, as any new nitrogen that enters these soils is rapidly assimilated by plants (Porporato et al. 2003; Adair et al. 2004). The highest concentrations are in riparian surface soils as nitrogen in soil is primarily derived from organic matter and riparian vegetation contributes large quantities of organic matter as described for phosphorus above (Figure 3-13). This trend is consistent with previous studies which highlight nitrogen accumulation in riparian soils (Adair et al.2004).

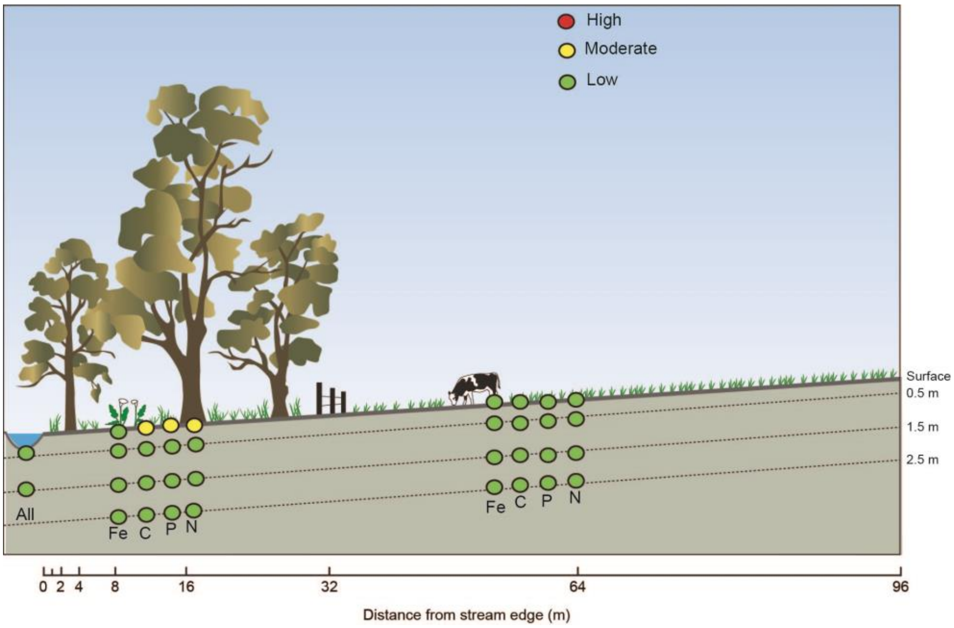


Figure 3-13. Conceptual model highlighting the chemical and nutrient makeup of soils with depth at Bingham Creek

Lennard Brook

Unlike Bingham Creek, there were greater concentrations and variation in soil phosphorus and nitrogen at Lennard Brook. The patterns seen at Lennard Brook were very different to those seen at Bingham Creek, highlighting the differences in soil quality.

Soil phosphorus concentrations were equally high in surface riparian soils and in the paddock at Lennard Brook. At 1.5 m depth, phosphorus concentrations more than double the maximum concentration at Bingham Creek. Surface riparian soils at Lennard Brook had a high clay-silt component (~40%) which allows greater phosphorus adsorption and correspondingly higher phosphorus concentrations (Figure 3-14; Vought et al. 1994; Lyons et al. 1998). The high organic matter content in shallower soils provides a store and a source (originally from the live riparian vegetation) of phosphorus to soils (Jobbagy and Jackson 2000; Raty et al. 2010) and the surface riparian soils had a high iron content which binds phosphorus. Together, these factors result in a high PRI for surface riparian soils resulting in considerable phosphorus adsorption and storage in the soil (Figure 3-14).

In contrast with Bingham Creek soil phosphorus concentrations were high in the paddock at Lennard Brook (Figure 3-5), a function of the higher clay and iron content resulting in a similarly high PRI for all Lennard Brook soils (Figure 3-15b).

The surface stream soils had very low phosphorus concentrations and were lower than the corresponding soils at Bingham Creek. This is a result of the stream soil being composed of greater than 90% coarse sand, which has limited binding sites for phosphorus (Singh and Gilkes 1991; Hassink et al. 1993; Tan 2000). Furthermore, there is little iron in the soil and no detectable instream phosphorus.

At Lennard Brook the composition of soil phosphorus varied more markedly than at Bingham Creek. Firstly, the organic phosphorus fraction was greater (~33%) and NaOH phosphorus contributed approximately 26%. Secondly, unlike Bingham Creek, organic and non-apatite phosphorus was found in soils at all depths and zones. The higher organic fraction can be linked to greater soil organic matter content (Fabre et al. 1996; Collins and Kuehl 2001). Furthermore, the greater fraction of labile phosphorus can be linked to more iron and clay in the Lennard Brook soils (Tan 2000). There were however, high phosphorus concentrations in the paddock soils. Due to high soil iron concentrations the soil was tested for strongly bound inorganic phosphorus which consequently made up approximately 70% of soil phosphorus present. This shows that most of the phosphorus in paddock soils is bound to the soil as a result of high iron concentrations. Overall soil phosphorus concentrations were higher at Lennard Brook and there was a greater fraction of labile phosphorus.

Lennard Brook has more than double the nitrogen concentration in surface riparian soils compared to Bingham Creek (Figure 3-9a-c). This can be explained by the substantially higher organic matter content in riparian soils at Lennard Brook which were five times greater than Bingham Creek. As previously seen, as soil organic matter increases, so does soil nitrogen concentration (Vought et al. 1994; Jobbagy and Jackson 2000). The higher organic matter content in riparian soils has previously been explained and is a result of the soil type and hydrology of the shallow riparian zone soils. Similarly to Bingham Creek, nutrient concentrations were relatively low in paddock and stream soils due to the limited availability of organic matter (Figure 3-14).

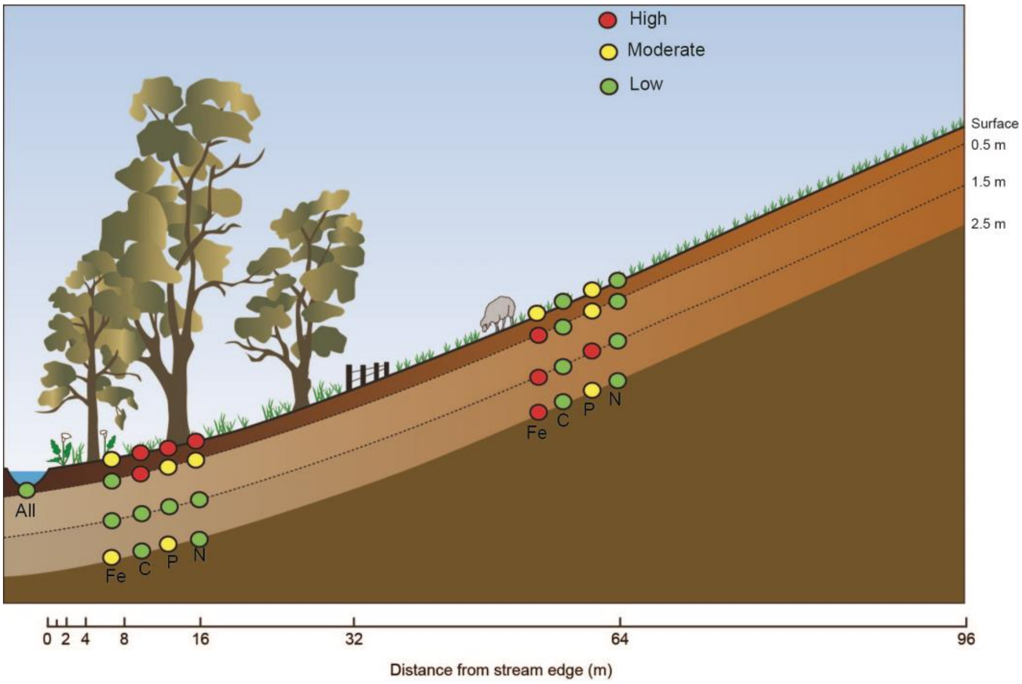


Figure 3-14. Conceptual model highlighting the chemical and nutrient makeup of soils with depth at Lennard Brook

Comparison of soil nutrient dynamics between Bingham Creek and Lennard Brook

Soil nutrient dynamics were quite different between Bingham Creek and Lennard Brook. Much of this is explained by the better soils at Lennard Brook which provide a greater trapping and absorbance capacity due to increased iron, organic matter content and higher clay fraction in Lennard Brook soils, resulting in higher phosphorus and nitrogen storage (Tan 2000). Conversely, at Bingham Creek the soils are poor. Bassendean sands have been shown to have low nutrient concentrations and limited capacity to hold onto phosphorus (Barron et al. 2008). The PRI and PBI of Bingham Creek soils were predominantly ranked as below extremely low (Figure 3-15a), highlighting the limited capacity of Bingham Creek soils to hold onto phosphorus. At Lennard Brook, the PRI and PBI's were higher, however most soils still had low scores. The exception was the surface riparian soils at Lennard Brook (Figure 3-15b), highlighting the value of the riparian zone in improving soil condition and nutrient removal capacity.

Table 3-3. Classification of soil PRI and PBI values which explain the key in Figure 3-15, adapted from Summers and Weaver 2006

Capacity of soil to sorb P	PRI	PBI
Exceedingly Low	<0.35	<5
Exceptionally Low	0.35-1	5-10
Extremely Low	1-2	10-15
Very, very low	2-9	15-35
Very low	9-28	35-70
Low	28-87	70-140
Moderate	87-275	140-280
High	275-1680	280-840

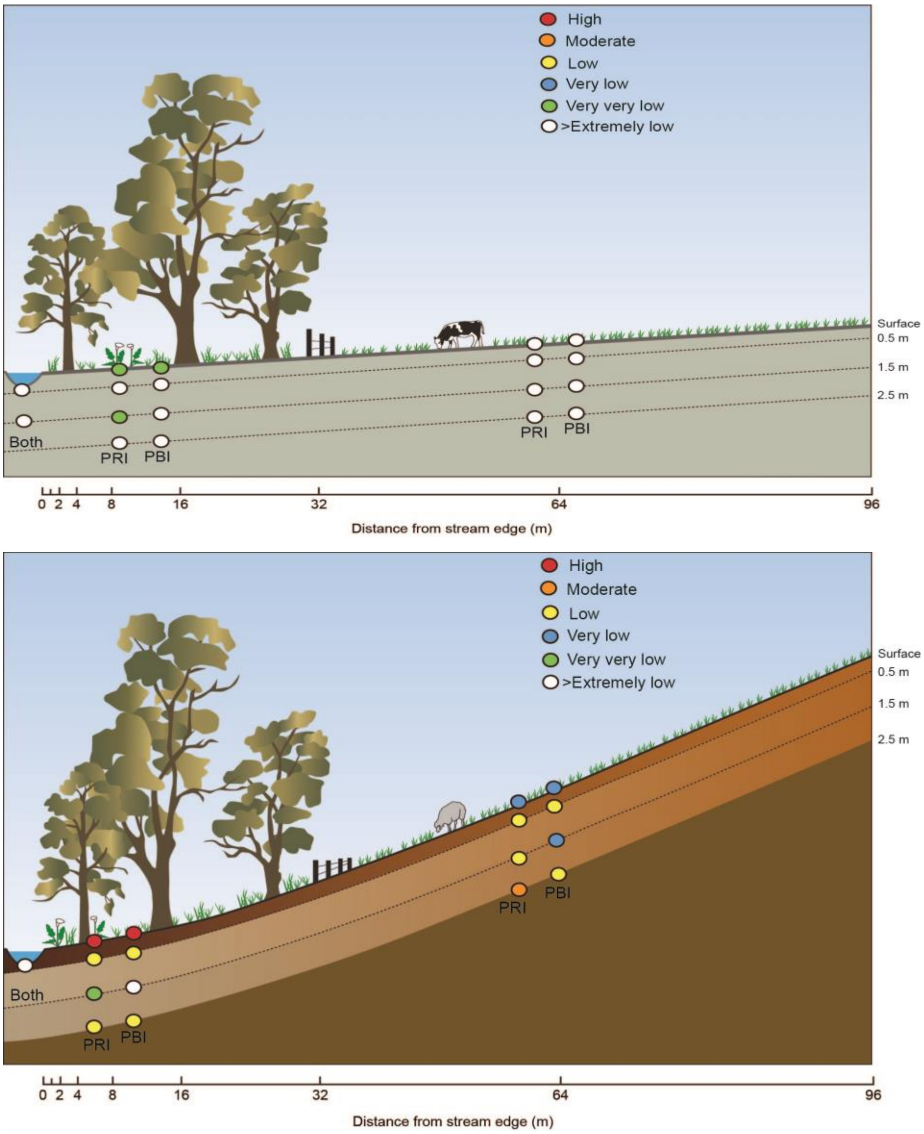


Figure 3-15. Conceptual model highlighting the PRI and PBI of soils with depth at a) Bingham Creek and b) Lennard Brook. Classification of PRI and PBI values detailed in Table 3-3

Soil nitrogen concentrations did not show as great a difference between sites, although, concentrations were higher at Lennard Brook. The main differences occurred within the riparian zone due to the greater proportion of organic matter in riparian soils at Lennard Brook. The greater proportion of silt and clay in surface riparian soils aided in the storage and retention of soil organic matter, as binding and integration into the soil matrix occurs more readily (Tan 2000), whereas at Bingham Creek, surface riparian soils were predominately composed of coarse sand and consequently had a lower organic matter content (Hassink et al. 1993).

Is riparian vegetation affecting the nutrient dynamics of underlying soils?

It is apparent that riparian vegetation is indeed affecting nutrient dynamics of underlying soils at Bingham Creek and Lennard Brook. Although from the results obtained, riparian vegetation was having a greater effect on nutrient dynamics at Lennard Brook. Both phosphorus and nitrogen concentrations were higher in surface riparian soils due to improved soil uptake and storage as described above.

It was clear that riparian vegetation was affecting soil nitrogen concentrations. Not only were concentrations highest in surface soils riparian soils, they dwarfed concentrations in the stream and paddock. Riparian vegetation is adding nitrogen to underlying soils through leaf litter and the death of vegetation, which contributes organic matter to soil. This organic matter provides nitrogen to underlying soils (Vought et al. 1994; Jobbagy and Jackson 2000; Tan 2000) but also provides a major store of nitrogen, as there is no consistent movement of organic matter to the stream or paddock soils.

Soil phosphorus concentrations, like nitrogen, were highest in surface riparian soils, providing an indication that riparian vegetation is contributing phosphorus to underlying soils. The high proportion of organic phosphorus in riparian soils provides an indication that it is derived from organic matter produced from riparian vegetation. The higher concentrations in Bingham Creek soils might also be derived from groundwater deposition. Furthermore, the physical and chemical properties of Lennard Brook soils enhanced the soil phosphorus concentrations in riparian soils.

Finally, riparian vegetation appears to be raising the PRI and PBI of Bingham Creek and Lennard Brook soils. This is most evident at Lennard Brook as seen by the high values in Figure 3-15b. It has also improved the Bingham Creek phosphorus retention capacity which is significant as the paddock PRI was zero and surface riparian PRI was five. In a management context this is important as phosphorus adsorption by Bassendean sands is very low, but has been improved by riparian zone processes.

Chapter 4 Vegetation

Introduction

Riparian vegetation is vegetation adjacent to the stream that serves to moderate environmental processes occurring between the catchment and the stream (Herron and Hairsine 1998). Riparian vegetation is a naturally occurring filter that has been shown to strip nutrients from groundwater and soil. Subsequently, riparian vegetation has been used as a nutrient reduction tool.

Riparian vegetation can influence nutrient dynamics in both soil and groundwater but may act as a nutrient source or sink to adjacent waterways (McKergow et al. 2006; Tabacchi et al. 2000). Riparian vegetation has the capacity to intercept and take up nutrients but can also contribute nutrients to streams and soils (Figure 4-1). Whether riparian vegetation functions as a source or sink greatly influences its value as a nutrient reduction tool.

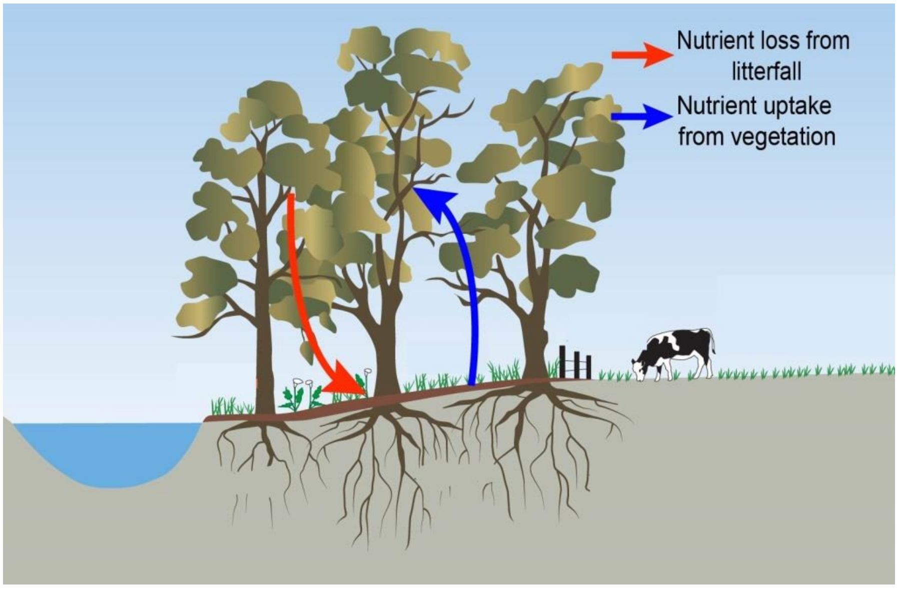


Figure 4-1. A diagram highlighting potential nutrient loss and uptake in riparian zones

Riparian vegetation acts as a nutrient sink primarily by intercepting and storing incoming nutrients (Tabacchi et al. 2000). This can occur by trapping nutrients from surface flow, altering underlying physical and chemical conditions (which have been discussed previously) and through plant assimilation (Tabacchi et al. 2000). Phosphorus and nitrogen uptake by riparian vegetation varies widely between vegetation types and between studies (Mander et al. 1997). Nitrogen uptake has shown to range from 30-220 kg.N.ha/year and phosphorus 5-50 kg.P.ha/year, illustrating the variability of riparian vegetation to act as a sink (Mander et al. 1997).

There are a number of processes affecting nutrient uptake in vegetation including seasonal variation, vegetation age and species type. Seasonal variation in nutrient uptake is well documented with nutrient uptake peaking in spring and early summer during times of maximum plant growth requiring rapid nutrient assimilation (Chapin 1980; Vitousek 1982). Vegetation age affects nutrient uptake (Dosskey et al. 2010) being greater in younger plants, particularly when comparing young trees to older forests (Mander et al. 1997). Nutrient assimilation by plants decreases as they mature, as older plants do not require excess nutrients to drive rapid growth (Osbourne and Kovacic 1993; Mander et al. 1997). However as plants mature they also store greater quantities of nutrients further bolstering the value of riparian vegetation as a nutrient sink.

The nutrient removal efficiency of riparian vegetation can vary substantially across different functional groups of vegetation. Plants vary greatly in size, growth rate, longevity and form, which all impact on their nutrient removal capacity (Dosskey et al. 2000). The structure of riparian vegetation can range from being a grass buffer, to shrubs, trees or a combination of the three (Dosskey 2001; Knight et al. 2010). Grass buffers are the preferred vegetation type for reducing surface flow and intercepting particulate nutrients, however nutrient assimilation is low (Knight et al. 2010; Lyon et al. 2000). Tree only buffers have low sediment trapping and phosphorus interception capacity when compared to grasses (Knight et al. 2010; Mckergow et al. 2006b) but are more effective at assimilating nitrogen (Lyon et al. 2000). Finally biomass accumulation and assimilation is greater in forested buffers compared to grass buffers, increasing the nutrient storage of riparian zones (Hefting et al. 2005).

Riparian vegetation can be partitioned into above and belowground biomass, where nutrient uptake and storage varies markedly. The main nutrient interceptor site for vegetation is below the surface through extensive root systems, however, once assimilated these nutrients are then distributed throughout the plant. Nutrient concentrations may be higher in belowground plant matter, however, as there is typically greater biomass above ground (Naiman and Decamps 1997), more nutrients are stored there. There has been limited work however assessing belowground productivity and nutrient dynamics in riparian zones (Naiman and Decamps 1997) and so their relative importance is not well understood.

Native and exotic riparian species have different nutrient removal capacities. Exotic species can invade disturbed riparian zones and are often capable of rapid growth (Ehrenfeld 2003; Naiman and Decamps 1997). This growth usually correlates with considerable nutrient uptake, altering nutrient uptake capacity of riparian vegetation. Exotic plants tend to have better resource use strategies and are more effective at intercepting nutrients than native species (Meisner et al. 2011). Native species, particularly in Australia, are suited to nutrient deficient soils and have limited nutrient demands (Tilman 2004). Furthermore, many native riparian species are perennial and any nutrients that are taken up are stored away for long periods of time (Meisner et al. 2011).

Riparian vegetation actively takes up nutrients as it grows and nutrients are subsequently released as plants die or through litterfall (Dhondt et al. 2006; Dosskey et al. 2010). In situations where nutrient loss through death exceeds nutrient uptake, it can lead to riparian zones becoming nutrient sources to adjacent waterways (Sabater et al. 2000). There are a number of factors that affect this, mainly plant type, which are again influenced by vegetation composition and whether it is native or exotic. Nutrient assimilation in grasses is not as high as trees, however, grasses have a higher turnover rate, leading to greater nutrient release

(Lyons et al. 2000). Exotic and native species have varied nutrient uptake capacities and consequently release nutrients at different rates. Exotic species increase nutrient uptake morphologically by increased root growth and through greater stimulation of microbial decomposers which makes nutrients more bioavailable (Meisner et al. 2011). As nutrient uptake is typically greater in exotic species, when they die more nutrients are released (Chapin 1980). Many exotic species in riparian zones are weeds which can fully mature and die in one year, leading to widespread seasonal nutrient release, turning a riparian zone from a store to a source (Ehronfeld 2003). For example English willows in Australian systems drop their leaves in a short period of time leading to rapid nutrient release (Stokes and Cunningham 2006). Natives typically are longer living and evergreen leading to greater nutrient storage and decompose more slowly (Richardson et al. 2007; Meisner et al. 2011). To counteract the nutrient loss in riparian zones periodic clearing of vegetation is suggested as a management strategy (Vought et al. 1994; Raty et al. 2010), however, this is only appropriate in riparian zones that are dominated by vegetation with short life cycles such as grasses and shrubs (Raty et al. 2010).

The effectiveness of riparian vegetation to reduce nutrients depends on whether it acts as a source or sink, or both at different times of the year. This chapter seeks to quantify nutrient storage and release at the two sites in the Ellen Brook catchment.

In the Ellen Brook catchment, much of the native vegetation, including riparian vegetation has been cleared, mainly for agriculture. How the remaining riparian vegetation in the catchment is functioning is unknown, yet planting riparian vegetation in the catchment has been undertaken as a best practice management strategy. Given the lack of information about native Australian vegetation on sandy soils it is hard to predict how riparian vegetation is interacting with nutrients in the riparian zone. This chapter aims to investigate how vegetation nutrient dynamics varies between functional groups and between native and exotic species. It will also address nutrient loss from riparian vegetation though litterfall. The key questions addressed in this chapter are:

- How does vegetation composition vary between sites and time?
- What are the concentrations of nutrients stored in the vegetation?
- How do nutrient concentrations vary between functional groups?
- How do nutrient concentrations vary between native and exotic species?
- At what rate are nutrients being recycled from riparian vegetation?

Methods

Vegetation assessment

Vegetation sampling was undertaken at Bingham Creek and Lennard Brook in September 2012 (spring) and in April 2013 (autumn). The sampling design was based on the work of Keighery (1994), McGilvray (2006) and the Swan River Trust (2008) to align with previous data collected on riparian vegetation in the Swan Canning river system.

Firstly, a site assessment was undertaken measuring riparian width, native crown death, natural regeneration of native species, health and disturbance. Riparian health was determined using the vegetation condition scale developed by Keighery (1994) which ranked vegetation health from pristine to completely degraded, going on the health of vegetation and presence of exotic species. Disturbance was ranked from none to very high and visible disturbance to the riparian zone from outside sources was also identified.

Functional groups were determined by assessing structural layers which provides a rapid overview of vegetation type. Layers were: canopy (>2m), understorey (0.2-2m) and groundcover (<0.2m), defined as canopy (trees), understorey (shrubs) and groundcover (sedges/rushes, grasses and herbs). Total projected foliage cover ("crown cover") was measured in each structural layer. Projected foliage cover is defined by Keighery (1994) as 'the total area under an imaginary line bounding the extremities of all plants in each group described', which is referred to as percentage cover.

Cover categories used by Keighery (1994) were adapted for the definition and classification of vegetation assemblages and were defined as follows:

- <2% (not considered to constitute a structural layer)
- 2-10%
- 11-30%
- 31-70%
- 71-100%

To ensure consistency and accuracy three replicate samples using standardised quadrat sizes were used for each structural layer, being canopy ($10m^2$), understorey ($4m^2$) and groundcover ($1m^2$).

The dominant species are those with the greatest foliage cover (percentage cover) in each structural layer (Keighery 1994). For this study a dominant species was considered to have a cover greater than 10%. They were identified and classified as native or exotic. Condition related attributes were noted, which included total number of species within each layer, which were divided into native and exotic to provide an exotic index.

Vegetation nutrient analysis

Nutrient analysis was undertaken using vegetation from the three structural classes in the riparian zone, and vegetation from the stream and paddock at Bingham Creek and Lennard Brook. Within each structural class, plants were split into exotic and native groups, to allow for nutrient stores to be clearly defined. Species contributing over 10% of structural layers were chosen for nutrient analysis, with exotic and native species bulk sampled, dried and analysed for total phosphorus and total kjeldahl nitrogen by Marine and Freshwater Research Laboratory (Table 4-1). There were three replicates for each sample.

Leaf litter traps

To assess litterfall contributions from riparian vegetation, litterfall traps were installed at Bingham Creek and Lennard Brook in December 2012. The traps were 25cm x 25cm with a fine mesh basket, capable of trapping the thin needles from *Melaleuca rhaphiophylla.* The traps were positioned on 0.7m high legs to ensure they remained above surrounding low growing vegetation such as *Zantedeschia aethiopica*. Six litterfall traps were randomly positioned throughout the riparian zone and canopy cover was recorded during installation.

Litterfall traps were emptied at the end of each month to determine litterfall contributions and nutrient concentration changes over time. Once collected, litter was dried and weighed and was analysed for total phosphorus and total kjeldahl nitrogen as described above. A one off sample (January 2013) was also analysed for total carbon. A loss on ignition and analysis was undertaken in the laboratory.

Groundcover litter

A groundcover litter sample was taken in December 2012 at both locations. A 25cm x 25cm quadrat was used and samples were collected from five different random locations within the riparian zone. All whole organic matter was collected, then dried and weighed and analysed for total phosphorus, total kjeldahl nitrogen, total carbon and loss on ignition by the methods described below (Table 4-1).

Table 4-1. Methods used for vegetation analysis

Parameter	Units and DL (if applicable)	Method	Lab
Organic matter	%	Automated combustion followed by NDIR method (Dean 1974)	MAFRL
Carbon content	%	Automated combustion followed by NDIR method (Dean 1974)	MAFRL
TP	mg.P/kg (<5)	Kjeldahl digestion with P measured by colorimetry (Aspilla et al. 1976)	MAFRL
TKN	mg.N/g (<0.04)	Involves the addition of copper sulphate and digestion in sulphuric acid	MAFRL

Results

Vegetation composition

There were similarities in the health and composition of riparian zones between Bingham Creek and Lennard Brook, however it appeared that Lennard Brook vegetation was healthier (Table 4-2). Both riparian zones have a similar zone width but both the canopy and understorey cover were greater at Lennard Brook. Groundcover at both sites was very similar. The exotic species index was greater at Bingham Creek. At both sites the index decreased from spring 2012 to the autumn 2013 sample.

Table 4-2. Comparison of key dynamics of riparian zone health and composition for Bingham Creek and Lennard Brook

	Bingham	**Lennard**
Riparian health	Good	Excellent
Level of disturbance	Medium	Low
Natural regeneration of native species	Occasional	Common
Width of riparian zone (m)	30m	30m
Canopy height (m)	10m	12-15m
Canopy cover (%)	31-70%	71-100%
Understorey cover (%)	11-30%	31-70%
Groundcover (%)	71-100%	71-100%
Exotic species index (2012/2013)	67% / 41%	38% / 13%

Dominant species composition differed markedly at Bingham Creek between spring and autumn, with a higher diversity in spring (Table 4-3). Variation was most noticeable in the groundcover where dominant species richness went from six in spring to two in autumn. The stream vegetation changed from aquatic vegetation (*Cycnogeton sp.*) in spring to terrestrial plants (*Pennisetum clandestinum*) in autumn. The majority of the dominant flora was exotic with the exception of canopy and stream vegetation in spring.

Table 4-3. Variation in dominant species (greater than 10% cover) at Bingham Creek between spring 2012 and autumn 2013. Note (in) represents how many quadrats species were recorded in and (N) represents native and (E) exotic

Bingham Creek spring 2012	**Bingham Creek autumn 2013**
Stream 1. *Cycnogeton sp.* (N)	**Stream** 1. *Pennisetum clandestinum* (in 3) (E)
Canopy 1. *Melaleuca rhaphiophylla* (in 3) (N) 2. *Eucalyptus rudis* (in 2) (N)	**Canopy** 1. *Melaleuca rhaphiophylla* (in 3) (N) 2. *Eucalyptus rudis* (in 3) (N)
Understorey 1. *Zantedeschia aethiopica* (in 1) (E)	**Understorey** none
Groundcover 1. *Pennisetum clandestinum* (in 3) (E) 2. *Ehrharta longiflora* (in 3) (E) 3. *Moraea flaccida* (in 2) (E) 4. *Zantedeschia aethiopica* (in 1) (E) 5. *Hypochoeris radicata* (in 1) (E) 6. *Cynodon dactylon* (in 1) (E)	**Groundcover** 1. *Pennisetum clandestinum* (in 3) (E) 2. *Cynodon dactylon* (in 1) (E)

Similar to Bingham Creek, species diversity at Lennard Brook was greater in spring compared to autumn, species richness decreasing from twelve to eight. There was a greater presence of native species within the understorey and groundcover than at Bingham Creek, particularly in autumn 2013, with all the dominant species being native (Table 4-4). Unlike Bingham Creek, the stream vegetation remained aquatic between seasons in this perennial stream.

Table 4-4. Variation in dominant species (greater than 10% cover) at Lennard Brook between spring 2012 and autumn 2013. Note (in) represents how many quadrats species were recorded in and (N) represents native and (E) exotic

Lennard Brook spring 2012	Lennard Brook autumn 2013
Stream 1. *Cycnogeton sp.* (N)	**Stream** 1.*Cycnogeton sp.*(N)
Canopy 1. *Melaleuca rhaphiophylla* (in 3) (N) 2. *Taxandria linearifolia* (in 3) (N) 3. *Eucalyptus rudis* (in 2) (N)	**Canopy** 1. *Melaleuca rhaphiophylla* (in 3) (N) 2. *Taxandria linearifolia* (in 3) (N) 3. *Eucalyptus rudis* (in 2) (N)
Understorey 1. *Zantedeschia aethiopica* (in 3) (E) 2. *Taxandria linearifolia* (in 2) (N) 3. *Pteridium esculentum* (in 1) (N)	**Understorey** 1. *Pteridium esculentum* (in 1) (N)
Groundcover 1. *Zantedeschia aethiopica* (in 3) (E) 2. *Baumea sp.* (in 2) (N) 3. *Patersonia occidentalis* (in 2) (N) 4. *Juncus sp*. (in 1) (E) 5. *Lagenophora huegelii* (in 1) (N)	**Groundcover** 1. Unidentified sedge (in 3) (N) 2. *Patersonia occidentalis* (in 3) (N) 3. *Baumea sp.* (in 1) (N)

Nutrient analysis

Phosphorus

Phosphorus concentrations for both sites were fairly similar with exotic species having higher phosphorus concentrations than most of the native species (Figure 4-2a-d). Groundcover vegetation at the Bingham Creek in autumn 2013 exhibited the highest concentrations. Within native species the stream vegetation (*Cycnogeton sp.*) showed the greatest phosphorus concentration.

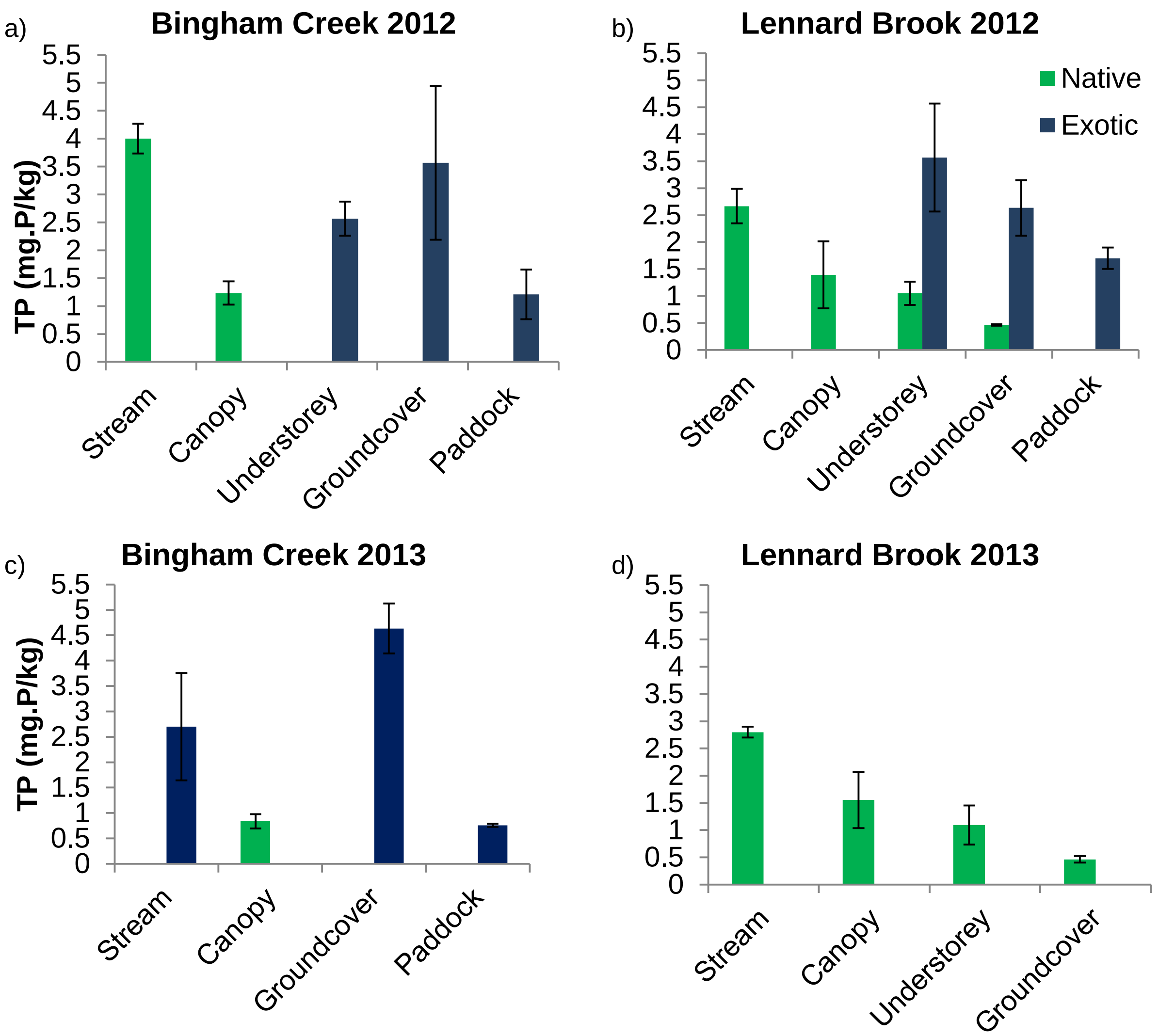


Figure 4-2. A comparison of vegetation phosphorus concentrations in native and exotic species for a) Bingham Creek spring 2012 b) Lennard Brook autumn 2013 c) Bingham Creek spring 2012 and d) Lennard Brook autumn 2013

Nitrogen

Nitrogen concentrations within the vegetation exhibited a similar pattern to that of phosphorus. Concentrations were typically higher in exotic species, with the exception of native instream vegetation which had the highest concentration (Figure 4-3a-d). This trend was apparent over both seasons.

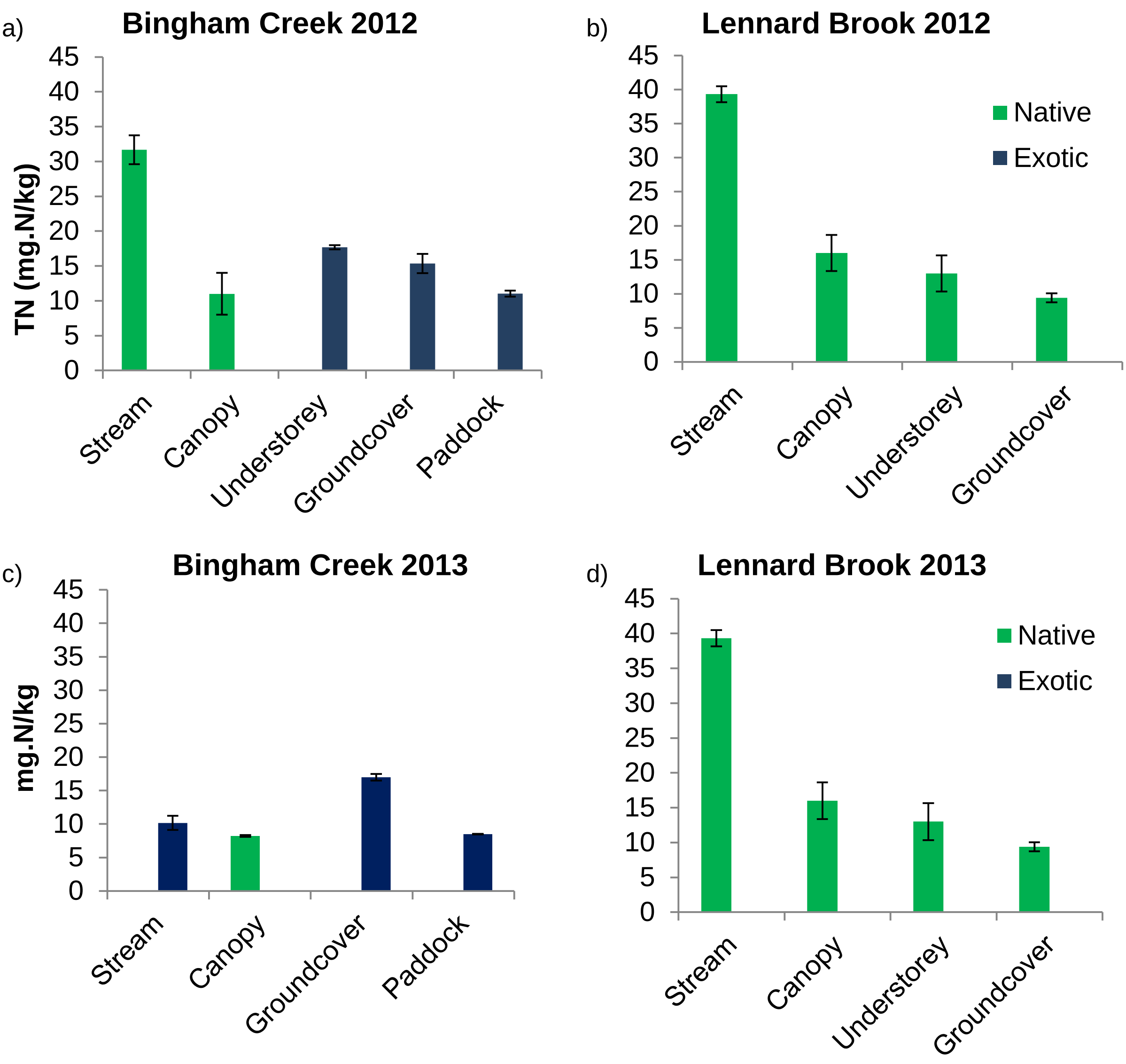


Figure 4-3. A comparison of vegetation total kjeldahl nitrogen concentrations in native and exotic species for a) Bingham Creek spring 2012 b) Lennard Brook autumn 2013 c) Bingham Creek spring 2012 and d) Lennard Brook autumn 2013

Litterfall traps

Weight

The average leaf litter mass at Bingham Creek and Lennard Brook was very similar and both showed the same trend over time. Leaf litter weights decreased from January to April, remaining low until August, whereupon weights increased and subsequently peaked in December (Figure 4-4). The total litterfall weight over the year was marginally higher at Lennard Brook (404 g/m^2) compared to Bingham Creek (369 g/m^2).

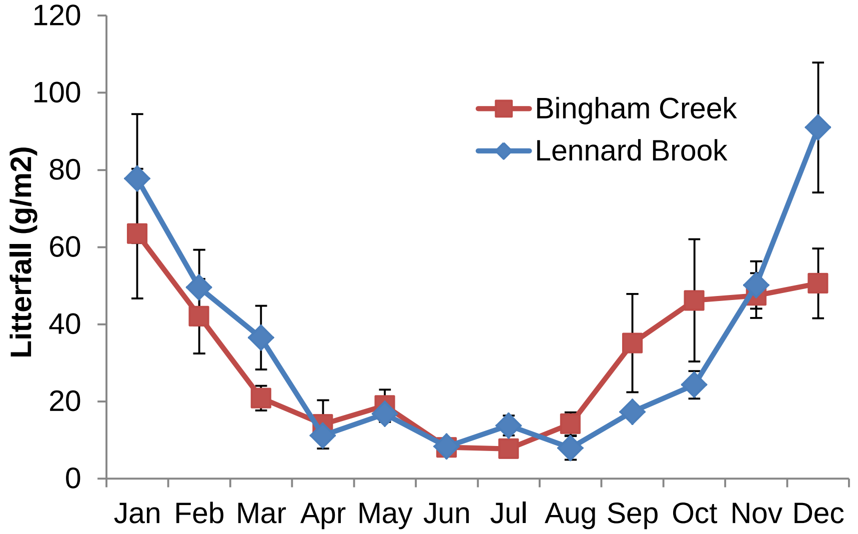


Figure 4-4. Comparison of litterfall weights over time between a) Bingham Creek and b) Lennard Brook

Nutrients

Phosphorus

Bingham Creek litterfall had higher TP concentrations (Figure 4-5a). However, at both Bingham Creek and Lennard Brook, there was no clear trend over time. Total phosphorus loads were very similar between sites and followed a similar pattern to litterfall mass, which peaked in summer and was lowest over winter (Figure 4-5).

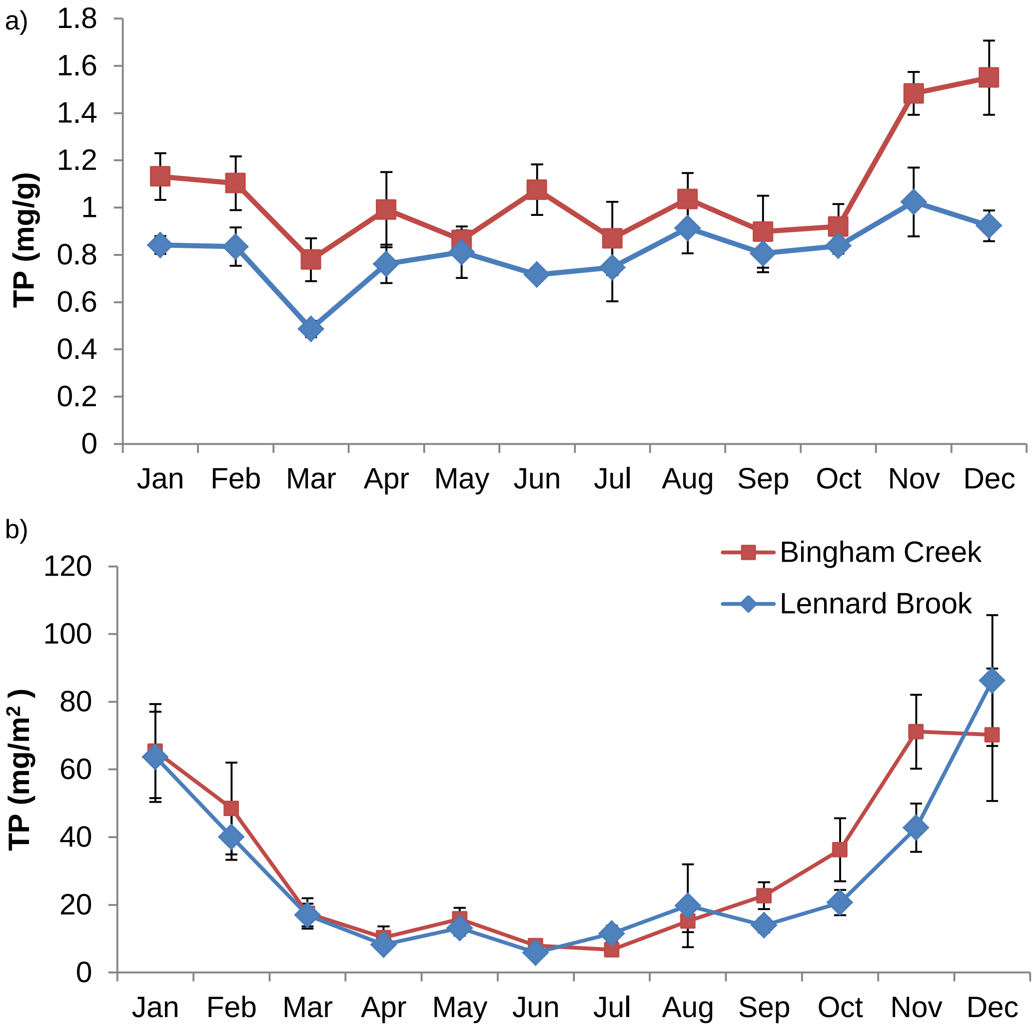


Figure 4-5. Comparison of litterfall a) total phosphorus concentrations and b) loads over time between Bingham Creek and Lennard Brook.

Nitrogen

TKN concentrations of the vegetation were very similar between sites and changes in concentration over time mirrored one another but overall were minimal (Figure 4-6a). TKN loads were also similar, with considerable overlap between sites, but values peaked in summer and were lowest during winter (Figure 4-6).

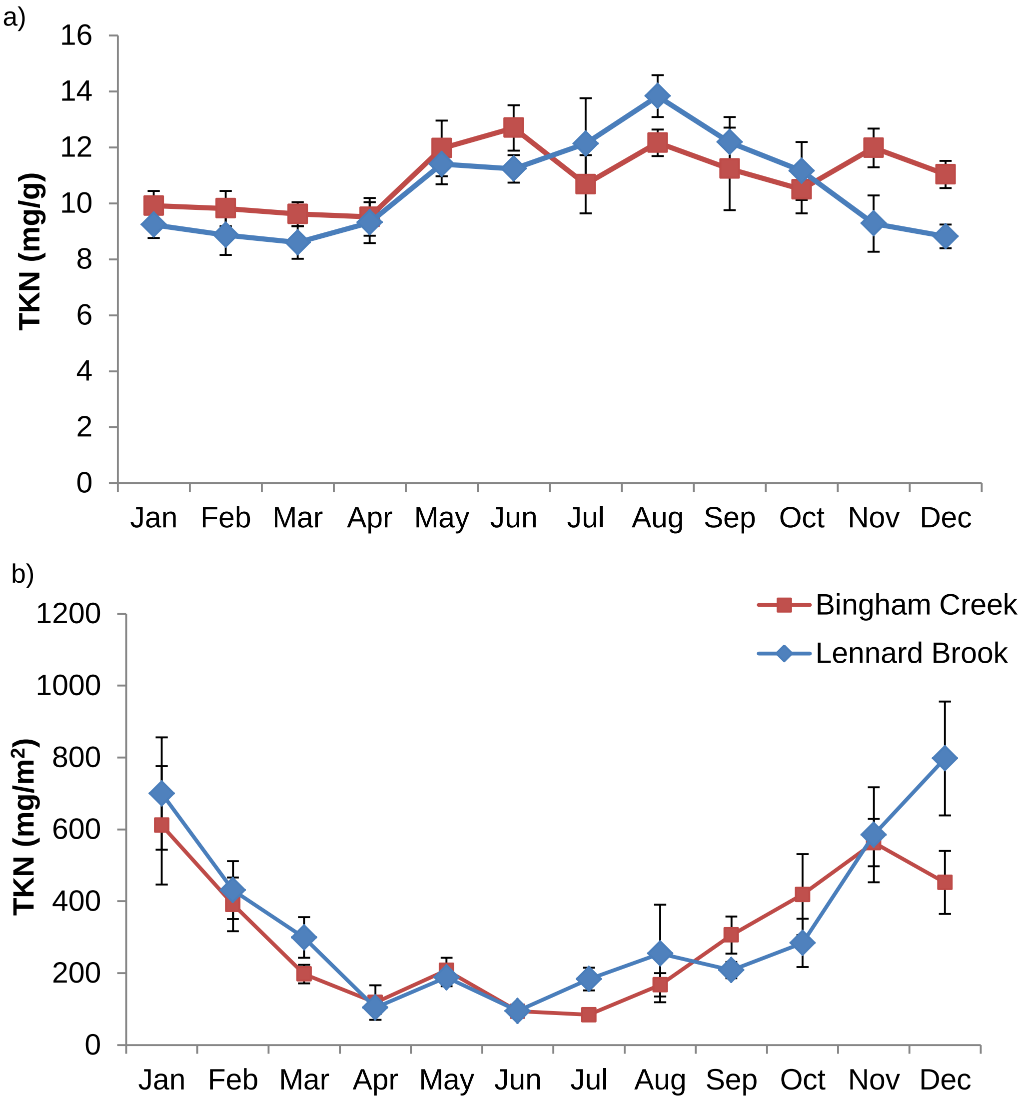


Figure 4-6. Comparison of litterfall a) total kjeldahl nitrogen concentrations and b) loads over time between Bingham Creek and Lennard.

Groundcover litter

Lennard Brook had more than double the weight of ground litter than that present at Bingham Creek (Table 4-4). TP and TKN concentrations of the ground litter were higher at Bingham Creek. However, due to the greater biomass of litter the storage of TP and TKN is greater in litter at Lennard Brook (TP- 495 mg/m^2, TKN- 7921 mg/m^2) compared to Bingham Creek (TP- 298 mg/m^2, TKN- 3717 mg/m^2). Organic matter and organic carbon content was similar, although marginally higher at Lennard Brook.

Table 4-5. Comparison of groundcover litter weights and chemical composition between Bingham Creek and Lennard Brook. Standard errors in parentheses

	Bingham Creek	**Lennard Brook**
Average weight (g/m^2)	295.224 (81.18)	769.376 (169.99)
TP concentration (mg.P/g)	1.01 (0.08)	0.644 (.04)
TKN concentration (mg.N/g)	12.58 (1.09)	10.28 (0.68)
Organic matter content (%)	91.37 (0.73)	92.90 (1.26)
Organic carbon content (% C)	45.4 (0.4)	47.8 (0.7)

Discussion

Does vegetation composition vary between sites and over seasons?

The composition of riparian vegetation can strongly influence the nutrient removal capacity of riparian zones. Superficially, the riparian vegetation at Bingham Creek and Lennard Brook appeared similar, both having a native canopy and dense groundcover, however, Lennard Brook had a denser canopy and understorey. The groundcover was similar between sites but there were more exotic species at Bingham Creek. Overall similarity, which was based on dominant species diversity between sites, was low (32% similarity). At Bingham Creek there were fewer dominant species (eight) in the riparian zone, compared to Lennard Brook (11).

Health and the level of disturbance are key factors that affect species composition (Richardson et al. 2007; Renofalt and Nilsson 2008). Both sites were fenced, however, at Bingham Creek there was evidence of cattle being in the riparian zone for extended periods. The higher level of disturbance at Bingham Creek is shown by the higher exotic species diversity and reduced regeneration of natural species (Richardson et al. 2007). Riparian zones are hotspots for exotic species due to high nutrient and water availability, the level of disturbance in riparian zones and the proximity to agricultural land which can provide a source of exotic species (Stohlgren et al. 1998). The higher level of disturbance at Bingham Creek provided greater opportunity for exotic species to flourish (Richardson et al. 2007; Renofalt and Nilsson 2008). Overall the health index of the riparian vegetation was better at Lennard Brook, likely to be linked to the lower level of disturbance.

The riparian community composition not only differed between sites, but also over time, with a reduction of species diversity and the exotic species index from spring to autumn. Growth and diversity is typically highest in spring and lowest in autumn (Chapin 1980), consistent with our results. The reduction in species is also likely to be due to drying soils (Stromberg et al. 2007) creating less favourable conditions for exotic species, such as arum lily at Lennard Brook, which require damp soils. The reduction in exotic species over time is also likely to be an effect of the shorter lifespan of annual exotic species such as cape tulips (*Moraea flaccida*) and exotic grasses. In contrast, native grasses and sedges are often perennial (Tremont and McIntyre 1994) and remained throughout autumn. The intermittent nature of Bingham Creek also resulted in a change in stream vegetation over time, from aquatic (*Cycnogeton sp.*) to terrestrial plants (*Pennisetum clandestinum*).

This further highlights the change in species composition between Bingham Creek and Lennard Brook which affects nutrient dynamics. Changes in species composition alters the quantity of potential nutrient uptake. If there is a lower cover of vegetation there is a lower capacity to take up nutrients (Hooper and Vitousek 1998), however, the reduction occurs between spring and autumn when groundwater is not in the active root zone. The greatest nutrient uptake by riparian vegetation occurs during periods of rapid growth which is often in late spring, early summer (Hooper and Vitousek 1998). Therefore the capacity for riparian vegetation to assimilate nutrients would be low due to limited contact with groundwater.

How do nutrient concentrations stored within vegetation differ?

Phosphorus and nitrogen concentrations in the vegetation changed over time, between sites and differed between zones. At Bingham Creek phosphorus concentrations were highest in the stream during spring 2012 and in the groundcover in the riparian zone in autumn 2013 (Figure 4-2a and 4.2c). The highest concentrations in stream vegetation are encouraging as they indicate that phosphorus assimilation is occurring (Mars et al. 1999). Considering the lack of phosphorus removal processes at Bingham Creek it is heartening to find an avenue for phosphorus uptake. The high groundcover phosphorus concentration in autumn 2013 can potentially be explained by high phosphorus concentrations in the surficial groundwater in the riparian zone. Once again this is an avenue for phosphorus removal. At Lennard Brook the highest phosphorus concentrations were in the understorey (spring 2012) and stream vegetation (autumn 2013). The understorey at Lennard Brook is dominated by arum lily (*Zantedeschia aethiopica*) which has a high phosphorus uptake capacity, storing it in the rhizome (Chen et al. 2009).

Lowest phosphorus concentrations occurred in the paddock over both sampling periods at Bingham Creek due to limited soil and water phosphorus concentrations there. At Lennard Brook native groundcover had the lowest phosphorus concentrations in both seasons. Native vegetation generally has a lower phosphorus demand (Hooper and Vitousek 1998; Meisner et al. 2011) and groundwater phosphorus concentrations were low at Lennard Brook compared to Bingham Creek.

The same pattern was encountered for nitrogen in vegetation at Bingham Creek. Nitrogen concentrations were highest in the stream in spring 2012 and decreased in all zones in autumn 2013, where concentrations were highest in exotic groundcover (Figure 4-3a and 4.3c). Whereas at Lennard Brook the highest concentrations were in the stream vegetation over spring and autumn, indicating that nitrogen was being assimilated from the stream.

How do nutrient concentrations vary between functional groups in the riparian zone?

Nitrogen and phosphorus concentrations were similar in the canopy at both Bingham Creek and Lennard Brook due to similar species composition. Conversely, the highest concentrations occurred in the groundcover at Bingham Creek and within the understorey at Lennard Brook. It was difficult comparing nutrient concentrations between functional groups of the riparian zone due to dissimilar vegetation composition.

The greatest variation in functional groups can be explained by the presence or absence of exotic species. Where native species were present in all functional groups (as was the case at Lennard Brook) there was a reduction in phosphorus and nitrogen concentrations from the canopy down to groundcover (Figure 4-1 and 4.2) and concentrations were lower than for exotic species. When exotic species were present, phosphorus and nitrogen concentrations were highest in groundcover at Bingham Creek and the understorey at Lennard Brook, as these were the layers in which exotic species dominated.

How do nutrient concentrations differ between native and exotic species?

Within the riparian vegetation, the majority of exotic species exhibited higher phosphorus and nitrogen concentrations compared with native vegetation. The exception to this was high phosphorus and nitrogen concentrations in the native stream vegetation. The highest nutrient concentrations in exotic vegetation occurred in the riparian zone, likely the result of the elevated groundwater nutrient concentrations in this zone. Exotic species often have higher nutrient concentrations due to physiological adaptations such as greater root mass, allowing for greater nutrient uptake (Ehrenfeld 2003; Naiman and Decamps 1997). As a result exotic species will be competitively advantaged by the nutrient enrichment at the sites.

At Bingham Creek, exotic species composed 67% and 41% of vegetation in spring 2012 and autumn 2013 respectively, whereas at Lennard Brook the exotic composition decreased from 38% to 13% over these two seasons. Exotic species are able to take up more nutrients but also have a shorter life span and release nutrients within the year, whereas native species take up less nutrients but maintain a long term store (Meisner et al. 2011). The change in species diversity and reduction in exotic species index from spring to summer illustrates that the majority of exotics are not perennial and are removing and then returning nutrients from the riparian zone annually, however, the nutrients are being transformed in this process from being highly soluble and available (inorganic) to being bound in organic form which requires decomposition and mineralisation before the nutrients would be available for algal growth in waterways (Vitousek 1982). Therefore, it is important to strike a balance in riparian vegetation, having vegetation that is capable of being able to take up high nutrient concentrations, but also has a long life span. A method that has been touted to overcome this is planting rapid growing riparian vegetation with high nutrient demands and then harvesting this vegetation periodically (Vought et al. 1994; Strauss et al. 2006; Raty et al. 2010). This promotes nutrient removal, but also limits the nutrient release through plant death and litterfall. However, this can be time consuming, expensive, and disturbance to plants and soil can result in significant particulate nutrient loss (i.e. sediment) to the stream during harvesting.

What rate are nutrients being recycled from riparian vegetation?

The riparian zones at Bingham Creek and Lennard Brook are both actively assimilating nutrients, however, riparian vegetation is also returning nutrients back to the riparian zone through plant death and litterfall. There was no understorey at Bingham Creek and litter fell only from the native tree canopy. At Lennard Brook, the understorey consisted of bracken (*Pteridium esculentum*), native shrub (*Taxandria linearifolia*) and arum lilies (*Zantedeschia aethiopica*), and while they did create litter, the lack of height precluded quantifying it in litterfall traps. Litter from this site also reflected the native canopy.

Litterfall rates decreased from January towards winter before increasing again in May. This change over time is consistent with native trees of Australia which drop their leaves over summer in periods of high temperatures and low water availability (Bell 1999). The increase in May could be related to the onset of winter winds and higher rainfall, knocking leaves down. Higher leaf litter weights at Lennard Brook are explained by the greater percentage of canopy cover, which translates to greater litterfall (Williams and Wardle 2007).

Total phosphorus concentrations in litter were higher at Bingham Creek compared to Lennard Brook, even though live tissue concentrations were higher at Lennard Brook. This can be explained by plants living in nutrient enriched environments not recycling as much phosphorus out of dying leaves (Chapin 1980; Vitousek 1982; Wright and Westoby 2003). Furthermore, there can be seasonal variations in phosphorus litterfall which can be linked to phosphorus availability and environmental constraints (no water), creating stress and leading to limited recycling of phosphorus from dying plant matter (Muune-Bosch and Leonor 2004). Litterfall nitrogen concentrations showed a similar trend to phosphorus, however the variations were not as great (Figure 5a-b). This can be explained by groundwater nitrogen concentrations being similar within the riparian zone.

Phosphorus litter loads were similar between sites, decreasing over time (Figure 4c-d), in accord with decreasing litter mass and phosphorus concentrations. The higher concentrations at Bingham Creek are counterbalanced by the higher litterfall weight at Lennard Brook, resulting in similar loads. This trend was similar for nitrogen loads between sites, however nitrogen loads far exceed phosphorus loads, which is consistent with literature (Vitousek 1982; Wright and Westoby 2007). This is due to greater nitrogen concentrations in plant matter (Vitousek 1982; Wright and Westoby 2007).

Besides directly measuring litter loss from the canopy, the litter covering the ground was assessed to compare litter loads and the corresponding nutrient dynamics. As shown in the litterfall trial, the groundcover litter weight was more than twice as high at Lennard Brook. This can be explained by two processes: firstly, there was greater plant cover at Lennard Brook which has a cumulative effect on groundcover litter and secondly, there were more exotic species at Bingham Creek which have the potential for greater decomposition rates (Ehrenfeld 2003; Ashton et al. 2005). This is due to the chemical composition of exotic species (Allison and Vitousek 2004; Liao et al. 2008) and can result in lower surface litter weights. Furthermore, the higher nutrient concentrations in surface litter at Bingham Creek can be explained by the greater proportion of exotic species in the riparian zone. As exotic species are often annuals, this leads to regular input of nutrients through seasonal mortality (Vought et al. 1994; Meisner et al. 2011). In contrast, native species are typically perennial and recycle nutrients from senescent leaves before their death and loss, leading to lower nutrient concentrations in litter (Chapin 1980).

Riparian vegetation is clearly modifying soil properties and nutrient loads particularly of surface soils as illustrated in the previous chapter. This is best highlighted by litter organic matter content, which was very similar between sites. However, the greater average litter mass at Lennard Brook, has resulted in soil organic matter content which is highest in the riparian zone at Lennard Brook.

Chapter 5 General Discussion

Is riparian vegetation effective at reducing nutrients entering Ellen Brook?

Key findings of nutrient uptake potential by riparian vegetation for flat sandy soil systems, typified by Bingham Creek.

- Riparian zones are reducing nitrogen through denitrification
- Phosphorus removal by riparian zones is limited due to:
  - Lack of surface flow which eliminates the main phosphorus removal pathway
  - Little horizontal movement of groundwater
  - Anaerobic conditions in groundwater
  - Poor soils unable to bind phosphorus
- Nutrients are primarily stored in groundwater not bound to soil
- Slow movement of groundwater results in the riparian zone being a store of phosphorus
- Riparian vegetation improves soils and phosphorus binding capacity through input of organic matter but is also a source of phosphorus
- Flow from the stream into riparian zone allows phosphorus removal from water that would otherwise pass directly to Ellen Brook

Riparian vegetation at Bingham Creek, a flat sandy soil system in the Ellen Brook catchment, was effective in reducing nitrogen concentrations but phosphorus removal capacity was limited. This is a function of the hydrology, soil type and vegetation dynamics.

The flow of water through the riparian zone is necessary to intercept nutrients on their passage from paddock to stream (Dosskey 2001; Dosskey et al. 2010). At Bingham Creek where there is no slope and highly permeable sands, the water rapidly infiltrates down into the soil profile and there is no surface flow (Figure 5-1a). The primary phosphorus removal pathway in riparian zones is through the interception of particulates in surface flow (Vought et al. 1994; Narumalani et al. 1997; Tabacchi et al. 2000). Without horizontal surface flow this primary phosphorus removal pathway is absent from Bingham Creek.

The movement of subsurface flows through soil can also directly affect nutrient removal in riparian zones (Vought et al. 1994; Narumalani et al. 1997; Dosskey et al. 2010). At Bingham Creek, there was limited horizontal subsurface flow from the paddock to the riparian zone due to lack of slope (Figure 5-1a). As a result water movement is dominated by vertical rise and fall over an annual cycle, with potentially some horizontal movement resulting from a slight gradient of the water table for most of the year from paddock to stream. Interestingly, at Bingham Creek there is an input of water into the riparian zone from the stream during the first flush of winter rains. This means that the riparian zone is taking up and modifying high nutrient water from the stream that would otherwise rapidly find its way into Ellen Brook and the Swan River. Consequently groundwater in the riparian zone is acting as a store for phosphorus. The long residence time and cycling of water within the riparian zone is aiding in nitrogen removal through denitrification and since groundwater phosphorus concentrations are lower than those in the stream, dilution and a small amount of phosphorus uptake is occurring through plant and soil uptake.

The soils at Bingham Creek are characterised by poor Bassendean sands. These soils are typified by low nutrient binding capacity (PRI) resulting in rapid leaching of added nutrients (Barron et al. 2008). As a consequence soil nutrient concentrations are low and groundwater nutrient concentrations are high (Figure 5-1c and 5.1e). Uptake of these nutrients is limited to plant uptake by the roots directly from groundwater and the minor improvement in soil binding capacity provided by the addition of organic matter to the surface soils by vegetation in the riparian zone.

Key findings of nutrient uptake potential by riparian for upland streams of Ellen Brook, typified by Lennard Brook

- Riparian vegetation has a higher capacity to remove more phosphorus and nitrogen than flat sandy regions like Bingham Creek
- Slope and shallow water table create surface flow, maximising phosphorus removal by removal of particulates
- Groundwater is continuously in contact with active root zone of riparian vegetation, allowing nutrient assimilation by riparian vegetation
- Soils have a better capacity to hold onto nutrients, restricting movement of nutrients from soil to underlying groundwater (more organic matter, clay and iron)
- Nutrients are primarily stored in soils not groundwater

In contrast Lennard Brook is a more typical of the traditional riparian vegetation paradigm with a much higher capacity to remove nitrogen and phosphorus. The slope and shallow water table at Lennard Brook provide conditions suitable for surface flow from the paddock to the riparian zone (Figure 5-1b), providing removal of phosphorus attached to particulates as a consequence of interception of surface flow by vegetation.

The passage of groundwater through shallow subsurface layers in the riparian zone at Lennard Brook means groundwater was constantly in contact with the active root zone of riparian vegetation, allowing nutrient assimilation by riparian vegetation.

Finally, while the soil at Lennard Brook was classified as a sand, the higher proportion of clay and iron in soils increased the nutrient holding capacity. As a result of better soils at Lennard Brook, incoming nutrients were stored in the soil instead of being released into water, resulting in high soil nutrient concentrations and low groundwater nutrient concentrations (Figure 5-1d and 5.1f).

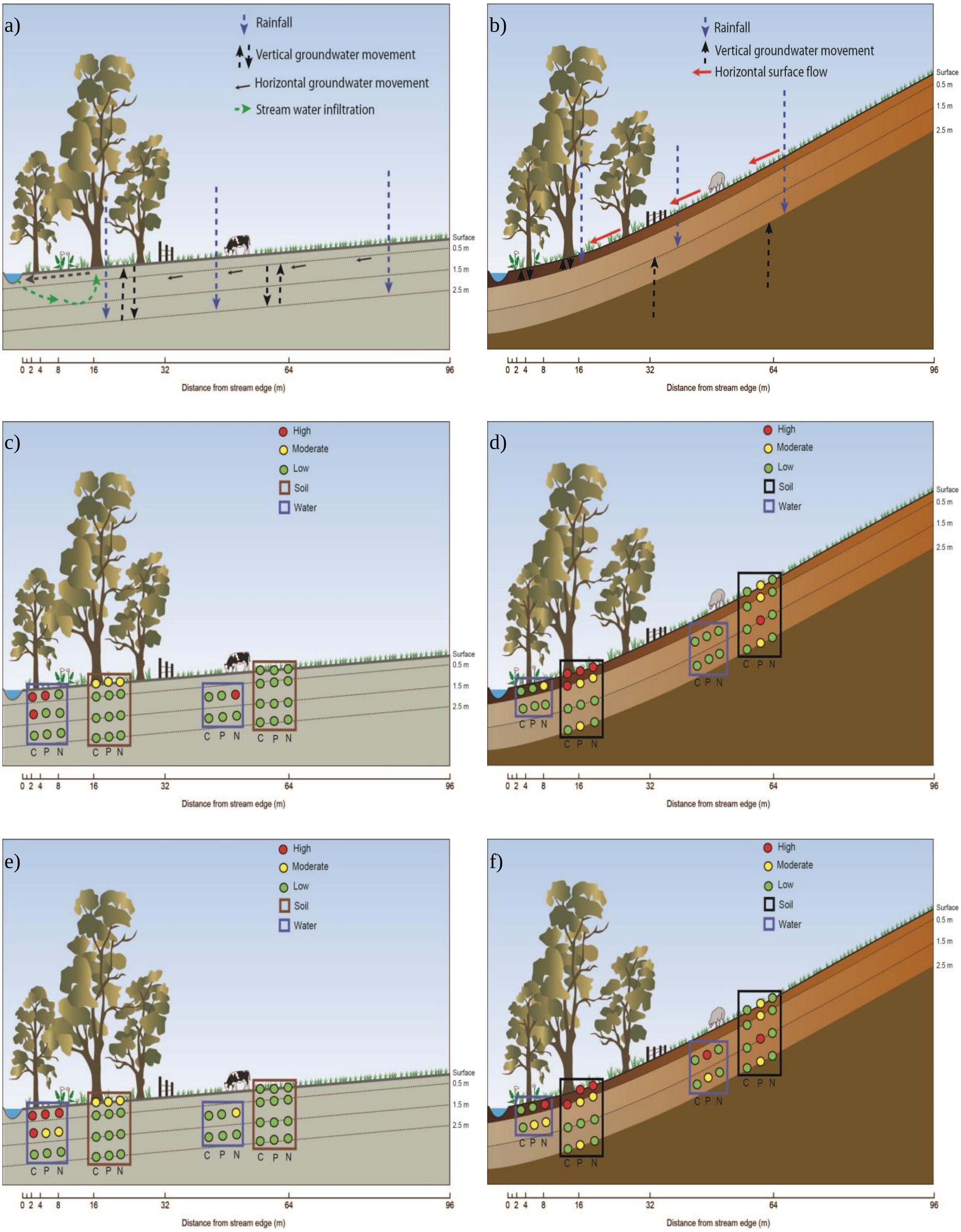


Figure 5-1. Models highlighting the hydrology at a) Bingham Creek and b) Lennard Brook and total and inorganic groundwater nutrient concentrations in relation to soil nutrients at Bingham Creek (c and e) and Lennard Brook (d and f)

Management Recommendations

Management recommendations to improve nutrient uptake by riparian vegetation for flat sandy soil systems, typified by Bingham Creek.

- To improve the phosphorus removal capacity of riparian zones and streams:
    - Improve soils through soil amendment in riparian zone
    - Line stream beds with phosphorus-binding amendments
    - Introduce or maintain native aquatic plants (e.g. *Cycnogeton sp.*) to streams
    - Plant native, wet-dry tolerant sedges in a strip 5m wide along stream bank to increase nutrient removal potential and to facilitate aeration of soil and groundwater
- Grass riparian buffers are not appropriate in flat, sandy soil systems
- Fence off existing riparian vegetation to reduce disturbance to existing riparian vegetation
- Protect existing riparian vegetation for the environmental benefits they provide

There is limited phosphorus removal capacity of riparian zones in flat sandy systems at Bingham Creek. Therefore, methods to increase the phosphorus removal capacity would be beneficial.

As previously mentioned the most effective phosphorus removal pathway in riparian zones is through surface flow (Vought et al. 1994; Narumalani et al. 1997; Dosskey et al. 2010). Due to the slope and soil type in this area, it is not possible to generate surface flow. Phosphorus removal in riparian zones can be enhanced however, by improving the phosphorus storage and removal capacity of riparian soils. Soil amendments could be used to improve near stream riparian soils before planting riparian vegetation to increase phosphorus storage, reduce leaching of soils and improve vegetation growth. This would provide a greater potential for phosphorus interception and uptake of phosphorus from groundwater flowing in from the paddock and stream. Considering the high instream phosphorus concentrations in many streams of Ellen Brook catchment, this could be an effective tool for the reduction in phosphorus flows out of the catchment.

At both Bingham Creek and Lennard Brook, native aquatic instream vegetation (*Cycnogeton sp.*) was found to actively assimilate phosphorus and nitrogen. *Cycnogeton sp.* would be an ideal species to plant into streams in this soil type as it has shown to be effective in high instream nutrient concentrations and to tolerate the seasonal drying associated with intermittent streams. This would enhance the nutrient removal capacity of riparian zones and actively reduce instream nutrient concentrations.

The type of vegetation in a riparian zone can have a strong effect on underlying physico-chemical conditions and nutrient uptake and release. At both Bingham Creek and Lennard

Brook there was a dense native canopy, poor understorey and a thick groundcover which had exotic species present. It is recommended that native shrubs and sedges should be planted with trees to create a complete riparian forest with an understorey present. This can create competition for exotic species in the groundcover, increasing nutrient uptake potential and improving soil condition. While exotic species had greater nutrient uptake, their seasonal nature meant nutrients were being taken up and then rapidly re-released after death. Exotic species generally have soft tissues that are rapidly decomposed, while native species breakdown more slowly and contribute to soil structure (Vought et al. 1994; Meisner et al. 2011). Planting native sedges has the capacity to aerate soils through their root structure (Kadlec and Knight 1996), oxidising shallow groundwater and soil, creating conditions that are conducive to FRP binding. Sedges also have an excellent potential to take up nutrients directly for growth. Species used need to tolerate a wide range of water regime being able to tolerant extending drying and flooding associated with intermittent streams.

Previous studies have highlighted the differences in nutrient removal capacities of different riparian vegetation types (Lyons et al. 2000). Planting riparian buffers with grass would not be appropriate in the flat sandy systems. This type of riparian vegetation is most effective at removing particulates from surface flow (which does not occur on these sandy soils) and the subsurface flows are not likely to intersect the shallow roots of grasses (Vought et al. 1994; Lyons et al. 2000; Brian et al. 2004).

To protect existing riparian vegetation within the catchment and maximise its nutrient removal capacity, it should be fenced. Fencing riparian zones to keep livestock out has a number of benefits. Firstly, it limits disturbance and destruction of native vegetation, allowing regeneration of native species to occur (Kauffman and Krueger 1984; Richardson et al. 2007; Renofalt and Nilsson 2008). Reduced disturbance of riparian zones has been linked to fewer exotic species. Secondly, apart from physical disturbance, livestock have been identified as a direct dispersal mechanism for exotic species (Stohlgren et al. 1998; Robertson and Rowling 2000; Shafroth et al. 2002). Finally, fencing does not allow livestock into streams, reducing re-suspension of phosphorus due to erosion or the input of nutrients through defecation (Robertson and Rowling 2000).

Management recommendations to improve nutrient uptake by riparian vegetation for upland streams of Ellen Brook, typified by Lennard Brook

- Riparian vegetation should be promoted as Best Management Practice for nutrient removal and associated environmental benefits on streams on this soil type
- Existing riparian vegetation on this soil type should be protected from disturbance, which can be done through fencing
- The introduction or maintenance of native aquatic plants (e.g. *Cycnogeton sp.*) should be promoted in streams

The extant riparian vegetation at Lennard Brook had the capacity to reduce both nitrogen and phosphorus concentrations. Therefore, the primary management recommendation is to optimise the health and protection of riparian vegetation zones on this soil type.

The existing riparian vegetation at Lennard Brook is functioning appropriately and should be left alone. Riparian vegetation should be promoted as Best Management Practice for nutrient removal on this soil type. It can be used as a demonstration to highlight how riparian vegetation can function. Existing riparian vegetation should be protected due to associated environmental benefits (discussed below) and should be fenced off to ensure the integrity of existing riparian zones.

Recommendations for prioritising sites for riparian restoration in the Ellen Brook catchment

- The nutrient removal effectiveness of riparian vegetation can be improved by prioritising the location of restoration projects, according to the hydrology and nutrient status, by:
  - Identifying slope and main water movement pathways
  - Determining nutrient storage capacity of different soil types
  - Identifying nutrient hot spots

Prioritising locations for riparian zone restoration within the Ellen Brook catchment could be improved through the classification of slope, soil type and nutrient hot spots in the catchment. This will inform appropriate riparian vegetation management for the conditions in that area.

Where sites have surface flow carrying particulate-bound phosphorus, fenced grassed riparian zones would assist nutrient removal. Grasses and shrubs have the capacity to reduce flows, trap sediment and have roots near soil surface which can actively uptake nutrients under these conditions (Vought et al. 1984; Vigiak et al. 2008; Dosskey et al. 2010). Grassed sites can also facilitate denitrification under these conditions (Lyons et al. 2000). Such sites would not have the many environmental benefits associated with native riparian forest but could assist nutrient retention. Fencing would prevent trampling and erosion of the riparian zone that would increase nutrient release.

Sites which have only subsurface flow would require native trees and sedges planted in the riparian zone. This would limit phosphorus release (creating oxidising conditions) and enhance the potential for denitrification and nitrogen removal (Mander et al. 1997; Lyons et al. 2000; Mckergow et al. 2003). The restoration of riparian forest will have associated environmental benefits wherever it is planted, however, to maximise the nutrient removal, vegetation should be planted in regions where there is high nutrient flow through the landscape.

Riparian vegetation also has a number of other benefits besides their nutrient removal capacity. Riparian vegetation provides aquatic and terrestrial habitat, contributing to terrestrial and aquatic biodiversity (Mander et al. 1997; Parkyn et al. 2003). It provides stream shading, energy and food for terrestrial and aquatic organisms and has shown to improve instream biogeochemical processes (Dosskey et al. 2010; Mander et al. 1997). Furthermore, riparian vegetation enhances underlying soils contributing organic matter, increasing infiltrations rates and provides protection from erosion (Easson and Yarbrough 2002; Mander et al. 1997; Raty et al. 2010). This further limits nutrients entering streams, stops sediment clogging waterways and helps maintain water clarity (Easson and Yarbrough 2002; Raty et al. 2010).

Future Work

To further assess the effectiveness of riparian vegetation as a BMP there is a range of work which could be undertaken. This work includes:

- Undertaking a more extensive soil sampling regime, conducted before and after planting riparian vegetation so that changes in nutrient stores and retention capacity can be assessed.
- Conduct monitoring on stream and groundwater nutrient concentrations before and after planting riparian vegetation. This will provide greater insight into the potential nutrient removal capacity of riparian vegetation.
- Undertake an assessment of the current riparian vegetation within the catchment and compare this with the predicted water movement pathway. This could be used to assess the effectiveness of nutrient removal by riparian vegetation throughout the catchment.
- Groundwater sampling could be undertaken over a longer period to see how riparian vegetation functions over years that are wetter or drier than those sampled.

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