RIPARIAN BUFFER ZONES OF THE CANTERBURY AGRICULTURAL LANDSCAPE AND THEIR EFFECTIVENESS FOR NITROGEN AND PHOSPHORUS RETENTION

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Abstract

Waterways in New Zealand have been severely degraded by agricultural activities during the past two centuries and land-use intensification is set to continue. Diffuse-source inputs are widely perceived to be a significant problem in low-lying areas, and riparian buffer zones are identified as a strategy to reduce land-use impacts. I surveyed the width and vegetative composition of riparian areas adjacent to 88 narrow farm waterways on the Canterbury Plains during the summer of 2009-2010. I investigated soil phosphorus and nitrogen availability at agriculture/riparian boundaries adjacent to high-intensity (dairy) and low-intensity (dry-stock grazing) pastures, and compared nutrient retention by fenced-off grass filter strips with that of native-planted buffers. Finally, I used growth metrics of three species commonly occurring in native buffers (*Cordyline australis, Plagianthus regius* and *Pittosporum tenuifolium)* and seasonal grass biomass accumulation, to assess growth response in relation to nutrient and moisture status of shallow (10cm) riparian soils. Riparian margins of Canterbury's agricultural waterways were commonly between 2 – 5 metres wide. Most were dominated by exotic pasture grasses and annual/perennial forbs or, exotic shrubs and trees, as opposed to native vegetation. Nutrient availability at the agriculture/riparian boundary appeared to be influenced more by seasonal effects and site-specific characteristics, than land-use. While nutrient retention was variable, it was generally higher in native-vegetated buffers than in grass buffers and also strongly influenced by season and site context. Accumulated grass biomass, and to a lesser extent native riparian species, appeared to be influenced primarily by soil moisture within shallow riparian soils, rather than nutrient status. Of the three trees assessed, the unique surface-root structures of *Cordyline australis* may confer an advantage for uptake of near-surface water and nutrients. However, on lowland plains, established native riparian trees that acquire nutrient resources from throughout the soil profile and shallow ground-waters also constitute an essential component of buffer systems to ensure optimal nutrient removal. This thesis recommends continuing use of narrow grass buffers in combination with down-slope planting with a range of native species to optimise nutrient uptake depth and storage longevity. It also suggests further study of plant/soil/nutrient interactions and, nutrient flow within riparian buffers in the context of topography to ensure that land-manager revegetation efforts live up to expectations.

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CHAPTER 1

General Introduction

Background

Agricultural non-point source pollution is one of the main sources of freshwater pollution world-wide (Zhang et al. 2010). Subsequent waterway degradation not only affects water for human use and consumption but causes stream environments to become inhospitable to freshwater species, leading to reduced in-stream biotic diversity and impaired or altered ecosystem function and loss of ecosystem services (Sabater et al. 2000, Blann et al. 2009, Lecerf and Richardson 2010, Hladyz et al. 2011, Liess et al. 2012). Agricultural activities contribute a wide range of pollutants to waterways that include chemicals from applied fertilisers, herbicides and insecticides, organic matter and faecal contaminants from animal excreta and sediment, arising from bank destabilisation, hillside erosion or tillage, that increases turbidity and loss of habitat and spawning areas (Lambert et al. 1985, Hart et al. 2004, Dabney et al. 2006, Winkworth et al. 2010). Of particular concern, substantial nitrogen (N) and phosphorus (P) entering waterways from these sources can cause nuisance algal and plant growth that blocks waterways and reduces flow or, creates toxic in-stream environments due to excessive ammonia concentrations (McDowell and Wilcock 2008). Water quality impacts often reflect the cumulative load of contaminants lost from land-use activities, therefore the continued expansion and intensification of agricultural practices, linked to the global demand for higher food production, constitutes the greatest threat to fresh water quality world-wide (McDowell and Wilcock 2008, Wilcock et al. 2011b).

In New Zealand, agricultural intensification has been rapid over recent decades as a result of farm conversions (e.g., sheep farming to dairy farming), increased pastoral stocking rates and higher grain yields which have required greater use of nitrogenous and phosphatic fertilisers and, significant use of limited freshwater resources for irrigation (Cullen et al. 2006, MacLeod and Moller 2006, Wilcock et al. 2011b). Intensification is especially prevalent in low-lying landscapes and thus low-elevation streams have been severely impacted nationwide (Larned et al. 2004). The Canterbury Plains, in the South Island, is of high significance to New Zealand's agricultural production. It constitutes approximately 20% of New Zealand's total farmland, represents over half of the irrigated land in the country and

consequently, over 75% of the region is now under agricultural production (MfE 2006). By 2010 Canterbury produced half of the nations' grain seed and fodder crop (Dynes et al. 2010) and, as a result of numerous dairy conversions, the Canterbury dairy herd increased from 5% to 15% of the national herd, between 1994 and 2009. Such rapid land-use intensification has been facilitated by increased water abstraction consents, irrigation usage and nitrogenous fertiliser application (MfE 2006). The net effect has been increased volume and concentration of nutrients, sediment and animal effluent being dispersed into water bodies (MacLeod and Moller 2006). Most lowland Canterbury waterways are already considered severely degraded and increased commitment towards remedial and preventative measures is required (ECAN 2010, Greenwood et al. 2012).

Water quality mitigation

Global concern for continued deterioration in water quality has prompted the development of environmentally sustainable land management strategies, collectively known as 'best management practices' (BMPs). These are intended to mitigate the effects of a range of agricultural land-use activities on waterways, and are widely recommended in many regions around the world (Muscutt et al. 1993, Collins et al. 2007, Chaubey et al. 2010, Vidon 2010, Lam et al. 2011). Dairying practices in New Zealand, have come under considerable scrutiny as a major contributor to waterway pollution (Cameron and Di 2004, Houlbrooke et al. 2004, Toor et al. 2004b, Wilcock et al. 2006, Monaghan et al. 2007b). Subsequent development of BMPs designed to improve dairy farming methods have been widely recommended, and adoption of many of these, such as improved management of dairy-shed effluent and sludge ponds, and constructed wetlands to treat sub-surface drainage, have resulted in significant improvements to water quality immediately affected by point-source pollution (Hart et al. 2004, Tanner et al. 2005, Monaghan et al. 2008, Wilcock et al. 2009, Lam et al. 2011).

However, non-point-source inputs of N and P from adjacent grazed pastures remain the major contributor to waterway pollution (Wilcock et al. 2009, Roygard et al. 2012). Effectively managing non-point source, or diffuse inputs, to farm water-ways is more challenging and strategies employed to address these include restricting winter grazing of adjacent pastures, revised effluent and irrigation application methods and improved nutrient management (e.g., timely & strategic fertiliser application). (MfE 2000, Maneer et al. 2004, Collins et al. 2007, Monaghan et al. 2007a, McDowell and Houlbrooke 2009, Monaghan et al. 2010, Lam et al. 2011). Management of riparian margins as 'buffer zones' aims to further minimise diffuse nutrient inputs and is regarded as the 'last line of defence' (Parkyn 2004)

Riparian zones are the interface between aquatic and terrestrial ecosystems and encompass the vegetated strip of land extending along stream, river, lake and other fresh water margins (Gregory et al. 1991, Naiman and Decamps 1997, Parkyn 2004, Richardson et al. 2007). Although in natural systems they often lack discrete boundaries, vegetated riparian margins containing trees, grasses and other herbaceous vegetation constitute complex, three-dimensional environments that provide and support essential ecological functions (Gregory et al. 1991, Mander et al. 1997)**.** These include stream-bank stabilisation, in-stream temperature regulation, provision of resource inputs to stream food webs and habitat for fish and invertebrate fauna (Parkyn 2004).

Riparian margins in most production landscapes have been highly modified and indigenous riparian vegetation removed to optimise agricultural and horticultural productivity (Correll 2005, MacLeod and Moller 2006). However, in recent decades the value of maintaining riparian vegetation as a water pollution mitigation strategy has been recognised (Barling and Moore 1994), and planting stream-side vegetation to protect and improve chemical and physical health of waterways is widely recommended by regional land and water resource managers. Furthermore, revegetating with indigenous species is promoted as a holistic approach, aimed to encourage the presence of native biodiversity, restore ecological integrity and, increase aesthetic value (Collier et al. 1995, Correll 2005, ECAN 2005(a), Ledgard and Henley 2009).

Riparian buffer effectiveness

In intensively managed landscapes the primary role of riparian buffers is to filter pollutants in surface run-off and subsurface flow to prevent them entering and degrading waterways (Stutter et al. 2012). Grass and grass-like species have been commonly used in this role however, tree and shrub buffers or, models that use mixed grass, shrub and tree vegetation, have more recently been evaluated. The use of riparian buffer zones as a pollution mitigation strategy is the subject of an extensive body of theoretical and empirical research, which report numerous influences on their ability to retain nitrogen, phosphorus and sediment from adjacent agricultural land-use activities (Dillaha et al. 1988, Phillips 1989a, Muscutt et al. 1993, Mankin et al. 2007, Yuan et al. 2009).

The combined effects of climate and topography on land-use, may initially determine the pollutant source, flux (mass, concentration & velocity), transport mechanism (sediment- or particulate-adsorbed or, dissolved**)** and flow-path (surface or sub-surface) from land-use activity to the buffer (Phillips 1989a, Polyakov et al. 2005). The relative ability of riparian

zones to then retain pollutants within the buffer is strongly linked to infiltration rate and storage capacity of buffer soils, vegetation and soil biota, (Cooper et al. 1995, Gillingham and Thorrold 2000, Lee et al. 2000, Mankin et al. 2007, Mayer et al. 2007, Zhang et al. 2010). The type and composition of buffer vegetation is crucial to buffer effectiveness due to its important influence on the three-dimensional structure of riparian zones (Polyakov et al. 2005, Zhang et al. 2010). Ground-cover vegetation (e.g. grasses, shrubs and weeds) can provide high surface roughness which physically intercepts run-off, thereby reducing flow velocity, filtering pollutants and determining soil infiltration rate (Abu-Zreig et al. 2003, Mankin et al. 2007). Plant uptake and denitrification have been identified as the two most important biological processes involved in nutrient removal in riparian buffers (Dhondt et al. 2003). Thus, nutrient residence-time in riparian buffers will vary depending on riparian hydrology, soil chemistry and the assimilative requirements of the plant and soil biota.

Unsurprisingly, the effectiveness of riparian buffers has been found to increase with increasing buffer width and decrease with increasing buffer slope (Lee et al. 2004, Correll 2005, Mayer et al. 2007, Liu et al. 2008, Zhang et al. 2010). However, while significant nutrient removal efficiency has been demonstrated from buffers 10-30 metres wide and greater, it is suggested that for those adjacent to narrow farm waterways, this may be significantly wider than land-owners can be expected to remove from active production (Hickey and Doran 2004).

Riparian Management in New Zealand

In New Zealand, researchers have been aware of the potentially adverse effects of nutrient run-off, on stream water quality, for several decades (McColl et al. 1977, McColl and Gibson 1979a, b). Earlier experiments were directed at reducing nutrient loss in run-off on sheep farming hill country (Lambert et al. 1985) and several authors suggested using stream channel vegetation or dense riparian vegetation to prevent stream eutrophication (Cooper and Cooke 1984, Smith 1987). Since then a considerable body of research has investigated farming impacts on waterways, finding strong links between intensifying land-use activity and deteriorating fresh-water quality (Quinn and Stroud 2002, Larned et al. 2004, Niyogi et al. 2007). Others have investigated strategies to reduce these impacts, including use of best management practices (Collins et al. 2007, Wilcock et al. 2009).

The Resource Management Act (1991) promotes the sustainable use of resources while "avoiding, remedying, or mitigating any adverse affects on the environment" (RMA

1991) and riparian management has been identified as an important tool to achieve this goal. In 1995 the Department of Conservation and NIWA released an informative publication, documenting theoretical concepts and guidelines to riparian management for use by regional and local authorities (Collier et al. 1995). This has since been supported by a wealth of literature accessible to land managers, on the benefits of constructing and maintaining well-managed riparian margins (MfE 2000, ECAN 2005(a))

On-farm management of riparian areas might initially begin with fencing along stream-sides to permanently exclude stock, and this has been particularly effective for reducing stream-bank erosion, decreasing sedimentation and reducing faecal contamination of waterways (Collins et al. 2007). Additionally, setting aside a rank grass, un-grazed 'buffer zone' along waterways is promoted to reduce sediment-adsorbed particles and other organic particulate matter inputs to waterways by filtering pasture run-off (ECAN 2005(a)).

A resurgence of interest in planting native stream-side vegetation to improve stream water quality and ecosystem health has resulted in many interested community groups becoming involved in revegetation projects (Ledgard and Henley 2009, Collins et al. 2013). Native plant rehabilitation is also actively promoted by regional authorities throughout New Zealand and is supported by a wealth of information on appropriate species to use, planting-out strategies and care and maintenance of riparian vegetation. Additionally, in recognition that riparian zones in differing topographical landscapes require variable approaches to their management, the Riparian Management Classification (RMC) was developed to aid riparian management strategies at local scales (Quinn 2003b, 2009). These documents describe a rapid assessment protocol for determining riparian function within the context of land-use and landform, enabling estimation of optimal grass-filter-strip widths, predicted to provide most effective control of a range of agricultural pollutants (Quinn et al. 2001).

Investigations have since been conducted to assess improvements to stream environments subsequent to riparian planting, Metrics such as habitat quality, fish and invertebrate diversity and abundance and physicochemical aspects of water conditions, such as temperature, clarity, turbidity and water chemistry have been considered. Several studies report improvement in some physicochemical aspects of the stream environment, or small shifts in stream biotic assemblages related to these, after a relatively short time (Parkyn et al. 2003, Parkyn 2004, Jowett et al. 2009, Greenwood et al. 2012, Collins et al. 2013). For example Greenwood et al, (2012), found the presence of pollution-sensitive taxa was higher at managed stream reaches than at completely unmanaged ones, although this was generally mediated through riparian food subsidies, lowered temperatures, better flow regime and,

narrower and deeper channels (Greenwood et al. 2012). However, it is generally acknowledged that further improvement in biological indices such as the return of sensitive invertebrate taxa and indigenous fish species that require established riparian habitat, may take considerably longer (e.g.10 – 20 years or more) (Parkyn et al. 2003, Parkyn 2004, Jowett et al. 2009, Greenwood et al. 2012).

While the in-stream environments of waterways on the Canterbury Plains are well-studied there is very little data on the condition of riparian areas. The scale of riparian management is reportedly limited to relatively small, disconnected patches. Importantly, the effectiveness of existing riparian buffers on the Canterbury Plains, as a water pollution mitigation strategy, has not been investigated. In addition, although native vegetation is recommended for stream-side planting, only a few studies have been conducted that investigate its performance in functional riparian roles (Czernin and Phillips 2005, Phillips and Marden 2006b, Marden et al. 2007, Phillips et al. 2009, Phillips et al. 2011).

Waterways on the Canterbury Plains exist within a matrix of highly valued and intensely-farmed land and therefore any measure to reduce waterway degradation, including riparian management, is crucial to environmentally sustainable farming. Unfortunately there are many gaps in our knowledge regarding riparian management on the Canterbury Plains and only a paucity of data seems to exist on even fundamental questions. For example although advocated, to what extent is riparian management is being implemented? How wide are typical riparian 'buffers' on the Plains and what type of vegetation is in them? There is little quantification of the level of nutrient pollution entering riparian buffers in run-off in this lowland region and, does this differ between land-uses? How well are existing grass filter strips and planted indigenous vegetation doing at removing agricultural nutrients? Lastly, could some of our native riparian species be particularly useful for removing nutrients in buffers? Research investigating these questions is required if we are to make good land management decisions.

Research Aims

This thesis investigates riparian management on the Canterbury Plains, New Zealand. I aim to a) quantify the extent to which riparian management has been adopted adjacent to small farm waterways, b) determine whether N and P exports to riparian margins in run-off are greater adjacent to high land-use intensity dairying operations, as opposed to low intensity dry-stock grazing and, c) whether grass filter strips, or planted native riparian margins are more effective for N and P removal within buffers and finally, d) investigate which, of three

commonly-planted native species, may be most effective for use in riparian margins. The study has been compiled as three stand-alone data chapters. The first, (Chapter 2) reports on a survey of riparian margins of 88 small, farm waterways within the Canterbury agricultural landscape and documents their width and vegetative composition. The second, (Chapter 3) uses in-situ ion-exchange resin bags to compare soil N and P availability at riparian/ agricultural boundaries adjacent to ten dairying operations and ten dry-stock grazing land-use activities. I then use fence-line and stream-side resin-bag N and P concentrations to compare retention by grass filter strips, with native-planted buffers. In the third, (Chapter 4) the growth of three riparian species commonly found in established, planted native buffers in Canterbury is assessed to determine whether they show a growth response to the N and P levels in which they have established. In Chapter 5, I discuss my results and suggest possible ways of maximizing riparian management for waterway protection in the context of the Canterbury Plains. Chapter 2 has been submitted to NZJFWR for publication and is formatted as a manuscript, in that journals layout.

Characterising riparian buffer zones of an agriculturally modified landscape

Abstract

Riparian buffer zones are viewed as an important management tool for waterway protection. However, little qualitative or quantitative assessment has been made of their dimensions or composition. I surveyed 88 small agricultural waterways within the Canterbury region to characterise the width and vegetative composition of riparian 'buffer' zones. Less than 20% of those surveyed were 10 metres or wider, as recommended for greatest effectiveness, and approximately 65% were 5 metres wide or less. Mean plant taxa diversity was lowest in buffers ≤ 5 metres (4 taxa) and highest in buffers 25-30 metres wide (11 taxa), although 98% of all taxa recorded in the survey occurred within 5 metres of the stream-side. Compared with native vegetation, exotic pasture grasses, exotic weeds and adventive trees and shrubs dominated riparian vegetation at all distances from the stream-side. Furthermore, buffer width and vegetation type did not differ with adjacent land-use despite decades of emphasis on increasing buffer widths and planting native vegetation, to reduce water contamination risks with more intense land-use practices.

Introduction

Riparian zones are the interface between aquatic and terrestrial ecosystems linking and influencing the ecological functioning of both (Gregory et al. 1991, Richardson et al. 2007). As a buffer between the land-use activities of a broader catchment area and a waterway, their management can have a disproportionately large effect on stream water quality relative to their size (Quinn 2003a). Riparian buffer zones are commonly viewed as 'the last line of defence' by managers for protecting waterways from degradation (Fortier et al. 2010). Among their key functions, vegetated riparian buffer zones can trap run-off contaminants, enabling deposition of suspended particles and increased infiltration of soluble contaminants within the soil profile. Furthermore, buffer zones can facilitate biological transformation of pollutants from land-use activities via uptake by plant and soil fauna and, microbial nitrification and denitrification (Lam et al. 2011).

Implementation of riparian buffer zones along agricultural waterways is widely promoted as a 'best management practice' by regional councils and is increasingly portrayed as an important environmental management tool (Parkyn 2004, Collins et al. 2007, Monaghan et al. 2008, Wilcock et al. 2009, Chaubey et al. 2010, Balestrini et al. 2011, Lam et al. 2011). Current best practice for protection and remediation of agricultural waterways includes fencing to exclude stock, and retaining vegetated stream-side margins (Cooper et al. 1995, Parkyn 2004).

Internationally, numerous studies have demonstrated the effectiveness of riparian buffer zones at reducing land-use impacts on waterways by reducing sediment (Lee et al. 2000, McKergow et al. 2003, Mankin et al. 2007, Lakel et al. 2010), nitrogen and phosphorus (Peterjohn and Correll 1984, Fennessy and Cronk 1997, Dhondt et al. 2003, Bedard-Haughn et al. 2004a, Hoffmann et al. 2009), and faecal contaminant loads to streams and rivers (Collins et al. 2004, Duchemin and Hogue 2009, Winkworth et al. 2010). Riparian vegetation can also provide shade, habitat and food resources for in-stream fauna (Parkyn 2004, Jowett et al. 2009, Wilcock et al. 2009) and valuable high-diversity habitats and migratory corridors connecting terrestrial fauna within landscape matrices (Corbacho et al. 2003). Thus, well managed riparian buffer zones provide a crucial ecological interface between terrestrial and freshwater ecosystems.

In an un-impacted landscape, riparian vegetation communities will be determined by climate, elevation, the regional pool of species, and the hydrological, geo-morphological and

disturbance regime of the region (Richardson et al. 2007). Prior to human settlement, the Canterbury Plains on the east coast of New Zealand's South Island, was a complex patchwork of fertile lowland native forest, scrubby tussock grasslands and patches of swampy wetlands nearer the coast (Meurk 2008). Extensive deforestation by early Polynesian settlers preceded further deforestation, 'straightening' of rivers, and draining of many coastal wetlands following European settlement in the 1840s (Ecroyd and Brockerhoff 2005). Although the Canterbury region has over 27,000km of river channels with diverse origins including alpine, foothills and lowland-spring-fed rivers, many naturally meandering river systems have been replaced by irrigation drains and water-races, in a landscape now dominated by pastoral farming (Sturman 2008).

Early stream-side management focussed on stream-bank engineering functions such as bank stabilisation and protection from erosion, to drain potentially productive land and also, to provide shelter-belts for stock and crops against harsh environmental conditions (Phillips and Daly 2008b). Native riparian vegetation was cleared and replaced with utility species such as willow (*Salix spp*.), poplar (*Populus spp*.), macrocarpa (*Cupressus macrocarpa*), and gorse (*Ulex europaeus).* Imported forage grass and horticultural production seeds were accompanied by an influx of exotic weed seeds and these, and many garden 'escapees' make up the diverse vegetation that now occupies Canterbury's riparian areas (Sullivan et al. 2005, Skarpaas and Shea 2007). The success of these exotic species has been partially due to similarity in climatic conditions between Canterbury and their countries of origin (Wilson et al. 1992), dispersal mechanisms that favour the Canterbury Plains environment (Williams 2006, Sullivan et al. 2009) and co-evolution with a diverse, seed-dispersing, exotic bird fauna (MacLeod et al. 2008).

While Canterbury stream systems and in-stream environments are well studied, there has been little assessment of riparian zones. In light of considerable emphasis on riparian management and initiatives by water managers to encourage planting vegetated 'buffer zones' to minimise adverse agricultural impacts (ECAN(a) 2005, ECAN 2005(b)), there is a need to measure progress towards this goal. In this study I characterise riparian buffer zones of small streams and waterways within an agricultural landscape. My primary focus was to quantify buffer widths and to broadly characterise their vegetative composition.

Methods

Study area and site location

The study was conducted in the Canterbury region on the east coast of New Zealand's South Island. Topographically the area includes steep, radial valleys, gently sloping foothills, and the alluvial floodplains of the Canterbury Plains. The Canterbury Plains and foothills have been farmed since settlement in the 1860's and traditionally, cropping (e.g., wheat, barley) and low intensity sheep and seasonal dairying have been common (Dynes et al. 2010). However, increased irrigation over the past 25 years has led to more intensive cropping and dramatic intensification of dairy farming (Wilcock et al. 2011a).

For this study, a field survey was conducted between November 2010 and January 2011 (austral late spring/early summer). Prior to this a large number of potential sites (rivers and streams accessible by road) had been identified on topographic maps (NZNS 260 series 1:50,000) and ARCview (ArcMap 9.3). However, during the survey, many of these were found to be dry and were excluded from the study, and additional waterways located in-situ. Subsequently, riparian margins adjacent to 88 permanent waterways throughout the Ellesmere and Banks Peninsula areas, the Oxford foothills and, on the coastal Plains north and south of the Waimakariri River, were surveyed during a single visit to each. These riparian margins were representative of the range of buffer widths, and the vegetative composition, typical of each area and included streams, irrigation drains and water-races (the latter two being important due to their prevalence across the Canterbury region).

Sampling methods

At each waterway, a 5 metre buffer reach was selected based on its representativeness in width and vegetation type, of the visible stream reach and, its accessibility for detailed survey methods. A transect was measured across the full buffer width, defined as the distance from the stream-edge to where agricultural activity started and land-use activity was restricted (e.g., fenced-off from grazing animals) for stream-protection purposes (Naiman and Decamps 1997). At the majority of margins this measurement was taken from the fence-line although, at mainly cropping operations, fencing was absent and land-use edge was used.

All plant taxa located within 1 m either side the transect were recorded and identified in the field, except in a limited number of cases where samples were returned to the laboratory for subsequent checking (Poole and Adams 1994a, Popay et al. 2010). For several herbaceous

exotic taxa the seasonal timing of the field survey prevented absolute identification due to indistinguishable species-specific characteristics among similar plants. Subsequently all taxa were allocated to an 'operational taxonomic unit' (OTU) which was the lowest identifiable strata of genera or family of the plant (see Supp. Table 1). For example, plants of the carrot family were allocated to Apiaceae, plants with dandelion-like flowers to Asteraceae, thistles to Carduus. It was deemed that in such cases, greater taxonomic resolution was unnecessary and would not add value to the study. Additionally, for the purposes of this study exotic pasture grasses, which included cultivars of perennial ryegrasses (*Lolium multiflorum* & *L. perenne*), tall fescue (*Festuca arundinaceae*) and cocksfoot (*Dactylis glomerata*), were allocated to a single group.

A cover score for each OTU was allocated based on a visual estimation of its percentage cover within each transect. Cover scores were; '1' = 1–5%; '2' = 6–10%; '3' = 11–20%; '4' = 20–50% and '5' > 50% cover. Cover estimations for buffer zones greater than 5m wide were carried out in 5m sections along the total transect width. In general, taxa were identified in the field, except in a limited number of cases where samples were returned to the laboratory for subsequent checking (Poole and Adams 1994a, Popay et al. 2010).

At each site, adjacent land use was recorded as that which was visible at the time of the survey and categorised within; dairying, sheep, other grazing (e.g., beef, horses, deer or alpaca) and, cropping (e.g. barley) or other land use (e.g., stock forage such as clover or grass).

Data Analysis

Taxa (OTUs) were assigned to either 'exotic' (alien to New Zealand) or 'native' (indigenous to New Zealand) according to Poole and Adams (1994) or Popay et al (2010). They were further categorised as either 'woody' (long-lived shrubs and trees with a woody stem) or 'weedy' (shorter-lived herbaceous forbs, grasses and ground-cover plants).Taxa scoring a '5' within any 5m section were recorded as 'locally dominant'. To derive a numerical indicator of relative-dominance for each OTU, within the landscape, cover scores were weighted according to their value as follows; '1' (* 0.05), '2' (* 0.05), '3' (* 0.1), '4' (* 0.3) and '5' (* 0.5). This generated a Cover Score Index (CSI) for OTU regional dominance comparisons. Data was tested for normality and heterogeneity, and despite transformation failed, requiring non-parametric analysis. Kruskal-Wallis one-way ANOVAs and Spearman Rank Order Correlation were performed (Crawley 2005)(R, version 2.14.2) to determine if other factors (land-use, waterway type or waterway width) correlated to buffer width. DCA ordination

(PCOrd, Version 4.01) was used to examine determinants of vegetative taxa distribution across the landscape.

Results

Among the 88 sites, the highest percentage (48%) of buffer zones were 2–5 m wide, 17% were < 2 m wide, and only 6% were ≥ 25 m. (Fig. 1a). A breakdown of buffer widths is presented in Appendices 1A and 1B by region and land-use type, respectively. Mean taxonomic richness was highest in 25–30 m wide buffers and lowest in ≤ 5 m wide buffers (11 and 4 taxa, respectively) (Fig. 1b). Despite the paucity of taxa in narrow buffer zones 98% of all taxa recorded in the survey occurred within 5 m of the stream-side.

A total of 108 plant species were observed during the survey, 77 were exotic and 31 native. Of these, 45 were identified to species, 37 to genera, 19 to Family or higher and 7 individual plants could not be confidently identified. The resulting data was analysed within 61 OTUs, of which 26 comprised native taxa and 35 exotic taxa. A list of these is provided in Appendix 2. Exotic taxa were present at 99% of sites and generally dominant at all distances from the stream-side, whereas native taxa only occurred at 45% of sites and were dominant at 19% of these (Fig 2, Table 1). Weedy plant (shorter-lived herbaceous forbs, grasses and ground-cover species) diversity was high, 75 species were represented by 31 taxa compared with woody (long-lived shrubs and trees with a woody stem) taxa (30) comprising 33 species. Weedy taxa percentage-cover was also much greater than that of woody taxa at 61% and 39%, respectively (Fig. 3).

Exotic pasture grasses were present at 95% of sites and locally dominant at 44% of these, while weeds, although present at 72% of sites, dominated only 5% (Table 1). Gorse and willow were the most frequently occurring single taxa (32% & 25%, respectively). Adventive trees and shrubs were more often present (at 40% of sites) than exotic plantation trees (at 19% of sites) but neither group was particularly dominant (at 7% & 8% of sites, respectively). Native sedges (e.g., *Carex spp*.) and flax *(Phormium tenax)*, were present at 34% of sites and dominant at 14% of these. By contrast, all other native trees and shrubs although present at 24% of sites, were never dominant. Not surprisingly, the cover-score index (CSI; highest possible value = 75) showed that riparian margins across the region were dominated by exotic pasture grasses, followed by willow, gorse and other mostly exotic vegetation, including broom, eucalypts, native flax, elderberry, poplar, thistles, blackberry and native sedges (Appendix 3).

Of the land use and catchment factors measured only waterway width was significantly correlated with buffer width (Spearman Rank Order Correlation Coefficient

0.296, P = 0.00524). Land-use showed no relationship with buffer width (Fig. 4a), although dairying appeared to influence plant taxa composition (Fig.4b).

Discussion

Buffer widths

The Canterbury region is representative of many areas worldwide that have been subject to large scale natural habitat conversion to farmland (Bowers and Boutin 2008). Of the 88 buffer reaches surveyed in this study, I recorded a surprisingly high percentage of buffers that were barely 5 m wide (65%), with many of these being under 2 m wide (17%). These findings were in accordance with those of Greenwood et al. (2012), who surveyed 64 streams in Canterbury and noted buffer zones were generally from < 1 m to 5 m wide. This is despite increased awareness among land managers of the benefits of riparian buffer zones for water quality protection, and their promotion by local authorities as best management practice.

Comparative data on widths of existing buffer zones is surprisingly sparse in the literature (Bedard-Haughn et al. 2004b, Lee et al. 2004, Balestrini et al. 2011). In documented guidelines of 60 jurisdictions within Canada and the United States, mean buffer widths in production landscapes are suggested to be approximately 15 to 20 metres for small permanent streams and intermittent streams, respectively. However, in experimental studies, existing buffers ranging between 3–15 metres have frequently been used to explore optimal buffer widths predicted to provide effective protection from a range of agricultural pollutants (Barfield et al. 1998, Mendez et al. 1999, Bedard-Haughn et al. 2004a, Balestrini et al. 2011). In a study of pesticide contamination of 14 low-order Danish streams adjacent to cropping, minimum buffer-strip widths were predominantly 1–6 m and average buffer strip widths were predominantly 5–10 m (Rasmussen et al. 2011).

Typically the minimum buffer width required should vary with the type of pollutant and its transportation method and rate into the buffer zone. Grass filter strips of 1–4 m can achieved 20–70% reductions in suspended sediment and total phosphorus (Wilcock et al. 2009), with maximum benefits achieved at widths of 6 m or more (Yuan et al. 2009). Similarly, Magette (1989) found that fescue grass filter strips of 9.2 m were considerably more effective than those of 4.6 m for suspended sediment, total P and total N. However, retention of dissolved, and highly mobile N and P species, in run-off and sub-surface flow, can require greater buffering width, although denitrification within the first few metres can play a dominant role in NO_3 –nitrogen depletion (Balestrini et al. 2011). Wilcock (2009) reported that grass filter strips of 1–4 m can achieve 80–95% reductions in faecal bacteria associated with dairy shed effluent (*E. coli* and *Campylobacter*), while others report that

although 5 m buffer strips reduced delivery of faecal microbes in dairy farm effluent by ≥ 94% during low flow simulations, buffers would need to exceed 5 m in order to markedly reduce their delivery to waterways along preferential flow paths and during high flow events (Collins et al. 2004). In general, buffers of 4 – 6 m may retain sediment or particulate-associated pollutants effectively but removal of dissolved N and P has been shown to require greater widths to prevent waterways being affected and importantly, landscape topography, buffer slope, vegetation type and density, soil type and flow convergence are also important considerations determining buffer success (Yuan et al. 2009).

In Canterbury and other intensively-farmed areas, regional councils advocate permanently excluding livestock by fencing, in combination with 2–3 metres of 'set-aside' dense pasture grass immediately adjacent to water-ways (ORC 2005, ECAN 2005(a)). Both are recommended as minimum protection for small agricultural waterways and a 'first step' towards managing diffuse nutrient losses in flat landscapes, with sloping or poorly drained soils requiring wider margins (ES 1999-2011, EW 2004, ECAN 2005(a), 2005(b)). For example, in the intensively-farmed Taranaki region in the North Island, a suitable minimum width of a metre of protected buffer (on each bank) for every metre of stream-bed was suggested as a 'rule of thumb' (TRC 1993).

However, as agriculture and related land-management practices continue to intensify, increasing water pollution risks, land managers should take into account the compounding pressures on riparian buffer zone functioning, and increase buffer widths accordingly. My findings indicate that at present this is not occurring. Furthermore, there appears to be no definitive guidelines in New Zealand on the width of buffer zone required for differing land use activities.

Studies suggest that wider, zoned riparian buffers of grass, herbaceous and woody vegetation can provide above- and below-ground complexity in ground-cover and rooting depths to ensure higher nutrient removal efficiency and more effective waterway protection (Correll 2005, Mankin et al. 2007). Incorporating plantings of native vegetation in mixed buffers can fulfil water protection aims while improving native stream-habitats (Collier et al. 1995). However, buffer zones of < 5 m wide are unlikely to support self-sustaining vegetation or suppress weeds due to edge effects, whereas those over 10 m wide are more effective and are recommended for minimal maintenance and greater habitat diversity (Parkyn et al. 2000, Davis and Meurk 2001a, Parkyn 2004, Reeves et al. 2006).

Buffer widths of 15–30 m (along all waterways) were recommended following an extensive review of contaminant retention studies in the United States (Wenger 1999). These

wider buffers were expected to provide good control of sediment, phosphorus, nitrate and other contaminants, plus provision of environmental conditions necessary for maintaining aquatic habitat, such as temperature control through shade, woody debris resources and wildlife corridors (Wenger 1999). Although a review of buffer widths in Canada, suggests that while buffers of 10–30 m wide may be desirable, research should focus on those up to 10 m wide, as such widths are likely to be more achievable in production landscapes (Hickey and Doran 2004).

Vegetative diversity in riparian communities

Ecological theory suggests that the assembly of biological communities is determined, in part, by environmental conditions that function as a series of filters (Booth and Swanton 2002). In riparian margins plant diversity and richness can depend on available seed source, dispersers and dispersal mechanism which may be further modified by anthropogenic mechanisms such as disturbance (Parendes and Jones 2000, Booth and Swanton 2002). It may also depend on the dominant vegetation type and environmental limitations or advantages imposed by it, such as competition for nutrients, water or light. For example, willows may reduce both richness and diversity by 'shading out' light-loving taxa, whereas open riparian margins with long grass and high light may provide ideal habitat for weed-seeds to prosper (Parendes and Jones 2000).

In the Canterbury region, I found that willows were the most prolific exotic tree and dominated riparian areas across the agricultural landscape. This is a legacy of early river boards and government agencies, which from about the 1930s onwards planted willow and poplar for river protection and soil conservation as their extensive mat-like roots provided considerable cohesion to erodible soils although with maturity, willow root structure becomes less effective (Phillips and Marden 2006a). Issues with willows and poplars, such as their prone-ness to cracking and toppling, that aided their successful colonisation and subsequent infestation of many waterways and wetlands, often negatively impacts low-energy waterways by blocking channels and reducing flow (Phillips and Daly 2008a). Furthermore, widespread damage to willows caused by the arrival of the willow sawfly (*Nematus oliospilus*), which causes defoliation and mortality among willow plantings, has led researchers to explore the use of native vegetation to full-fill stream-bank engineering functions (Czernin and Phillips 2005, Phillips and Marden 2006b, Phillips and Daly 2008a). Plant diversity within my study sites was low among willow and other tree-planted buffers, and higher along exposed grass buffers.

Riparian habitats are well known to be particularly susceptible to invasion by exotic taxa, and can act as sources, havens and corridors for weed dispersal (Stohlgren et al. 1998, Truscott et al. 2008). I found that several frequently occurring exotic taxa such as gorse, elderberry (*Sambucus nigra*), blackberry (*Rubus fruticosus*) and broom (*Cytisus scoparius*) all dominated the sites they occurred at for up to 10 m from the stream-side. The presence of some of these species is likely due to co-evolution with exotic frugivorous and omnivorous plant predators within their original geographic environment. For example, numerous introduced birds species such as blackbirds (*Turdus merula*), starlings (*Sturnis vugaris*), greenfinch (*Carduelis chloris*) and house sparrows (*Passer domesticus*) may aid dispersal of fruit- or seed-bearing adventive plants in production landscapes by using fence-posts as stepping-stones across the landscape and streamside margins as habitat corridors (Blackwell et al. 2005, Williams 2006, MacLeod et al. 2008). Similarly, studies conducted in agricultural regions of Australia found that over 60% of riparian vegetation comprised exotic invasive groundcover (Burger et al. 2010), and that many plant invasions are mediated by plant-frugivore interactions with exotic bird species (Buckley et al. 2006).

Interestingly, although native taxa were present at almost half of the sites I surveyed they contributed very little to vegetation cover either within local riparian areas or over all regional riparian margins. Despite decades of environmental agencies advocating the value of planting native species in preference to exotic species, few continuous stretches of native vegetation were found and most, in accordance with other reports, were limited to small disconnected patches or, had an extensive ground-cover of exotic grass and weeds (Greenwood et al. 2012). Quinn (2003), reported the dominant riparian vegetation of 313 Canterbury streams to be grass at 48% of sites, followed by willow (26%), low shrubs (9%), native trees at 8% and wetland plants at 2%. I found similar 'within-site' dominance for exotic grass, adventive shrubs and native trees (44%, 7% & 5% respectively) however, willow dominated much fewer sites (14%) and there were more native flax and grasses (13%). This slightly higher percentage of native vegetation in buffer zones of 3–6 metres may be partly due to the persistence of natural stream-side riparian vegetation, or alternatively, to recent (1–10 year old) restoration plantings, as native plantings become the preferred option for stream-side management. However, while local government and water agencies actively encourage planting of native riparian vegetation, surprisingly few of our reaches (12.5%) were planted as part of these initiatives.

Finally, although plant taxa composition of buffers adjacent to dairying appeared to be slightly different to that of other land-uses, this was most likely due to environmental

conditions associated with land management, such as higher soil moisture and nutrient status or, the predominantly lowland location of dairying operations.

Conclusion

The majority of riparian buffer zones in agricultural Canterbury are less than optimal in width and complexity to cope with a range of agricultural contaminants. The findings of this study indicate that in general land owners have not modified either their buffer widths, or the types of plants within riparian buffer zones in response to either changing land use activity or intensity. While some grassy buffer zones may be sufficiently wide to intercept sediment, suspended solids and faecal contaminants, they may not adequately capture other agricultural pollutants, particularly soluble nutrients in sub-surface flows and run-off, before they can enter waterways. More emphasis must be placed on increasing buffer zone widths and vegetative complexity to create buffer zones that can better protect Canterbury waterways. At present there are many unmanaged riparian margins along which sustainable native vegetation could be established without the need to shift fence-lines and some progress towards this is could be achieved. Planting native riparian margins is an integral part of a holistic approach to enhancing stream habitat, and a potentially powerful tool for mitigating damage done by decades of land transformation to agriculture as well as preventing future waterway degradation. Unfortunately, despite decades of advocacy relatively little progress appears to have been made.

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Figure Legends

Fig. 1: a) Frequency of occurrence of differing buffer widths of 88 small agricultural waterways on the Canterbury Plains, New Zealand and **b)** mean taxonomic richness at each buffer width range (1± SE; n = number of sites) .

Fig. 2: a) Taxonomic richness of exotic and native vegetative taxa at differing buffer widths of 88 small agricultural waterways on the Canterbury Plains, New Zealand and **b)** percentage cover of exotic and native taxa at each buffer width range.

Fig. 3: a) Taxonomic richness of weedy (short-lived herbaceous forbs and grasses) and woody (long-lived shrubs and trees) taxa at differing buffer widths of 88 small agricultural waterways on the Canterbury Plains, New Zealand and **b)** percentage cover of weedy and woody taxa at each buffer width range.

Fig. 4: a) Mean buffer widths of 88 waterways on the Canterbury Plains, New Zealand within four land use categories (median, 25th & 75th percentiles (box) and 10th & 90th percentiles (whiskers) are shown), and **b)** DCA of community composition of 88 riparian buffer zones in four land use categories (Solid square = dairy; Solid circle = sheep; Open triangle = other grazing; Open diamond = cropping or other land-use).

Figures
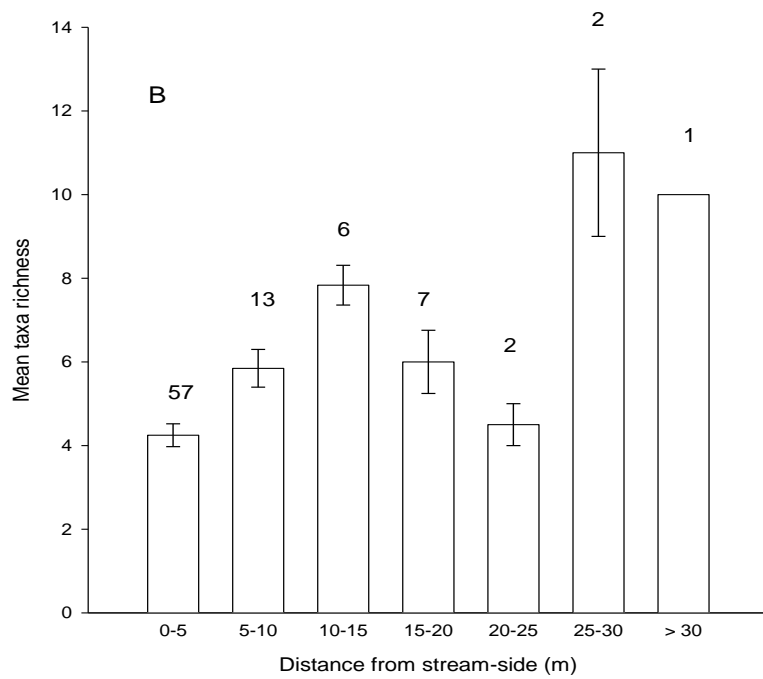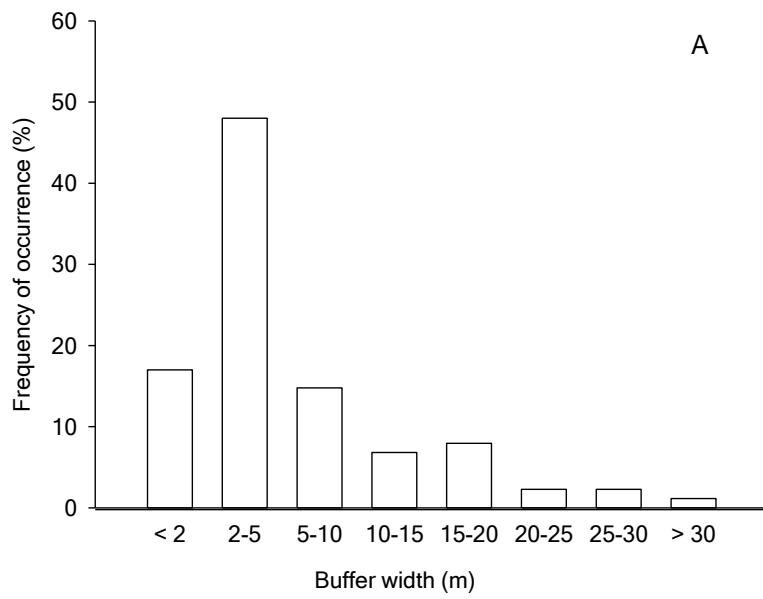
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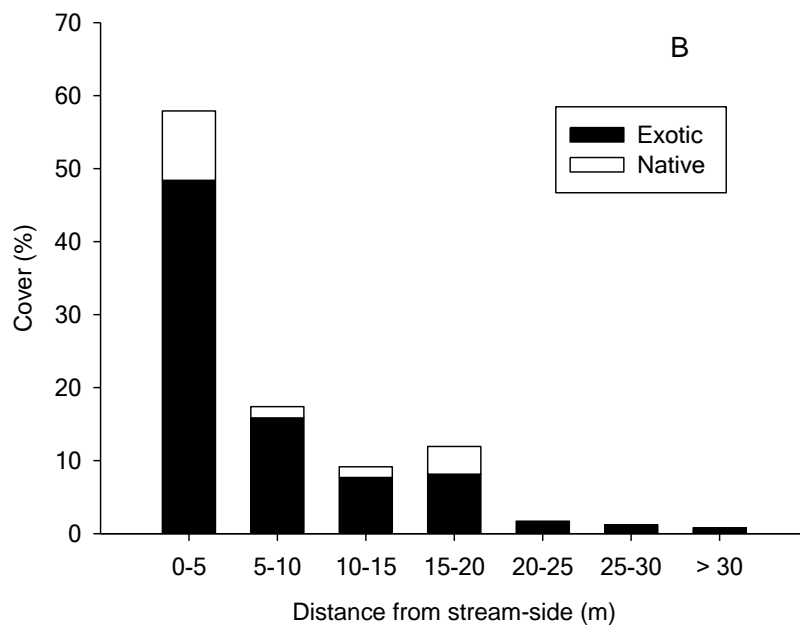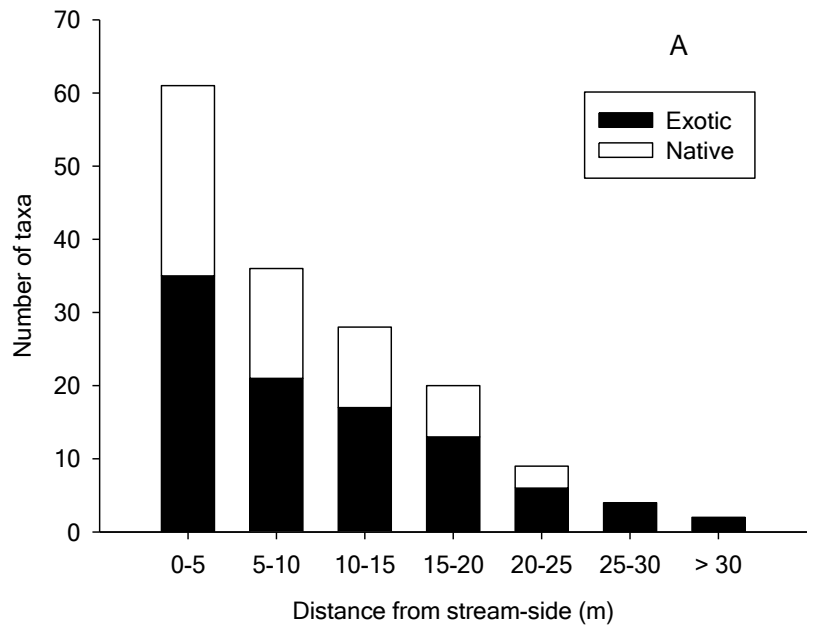
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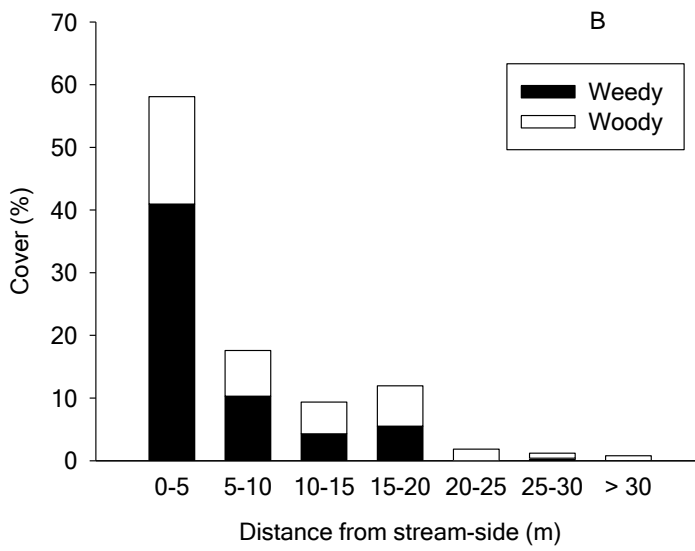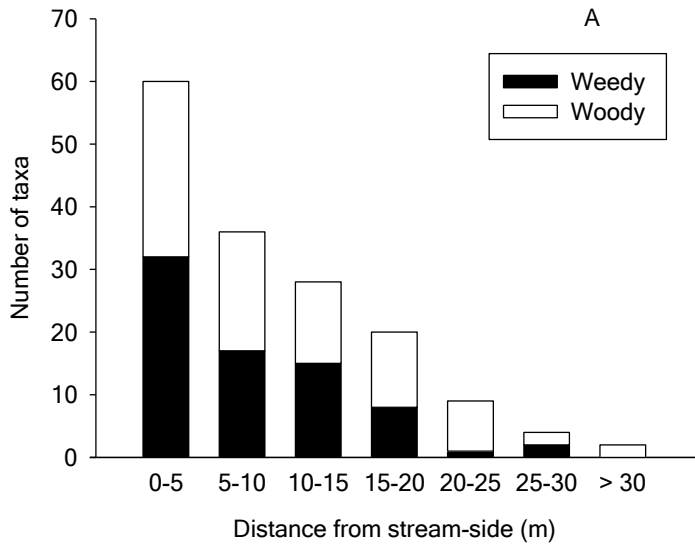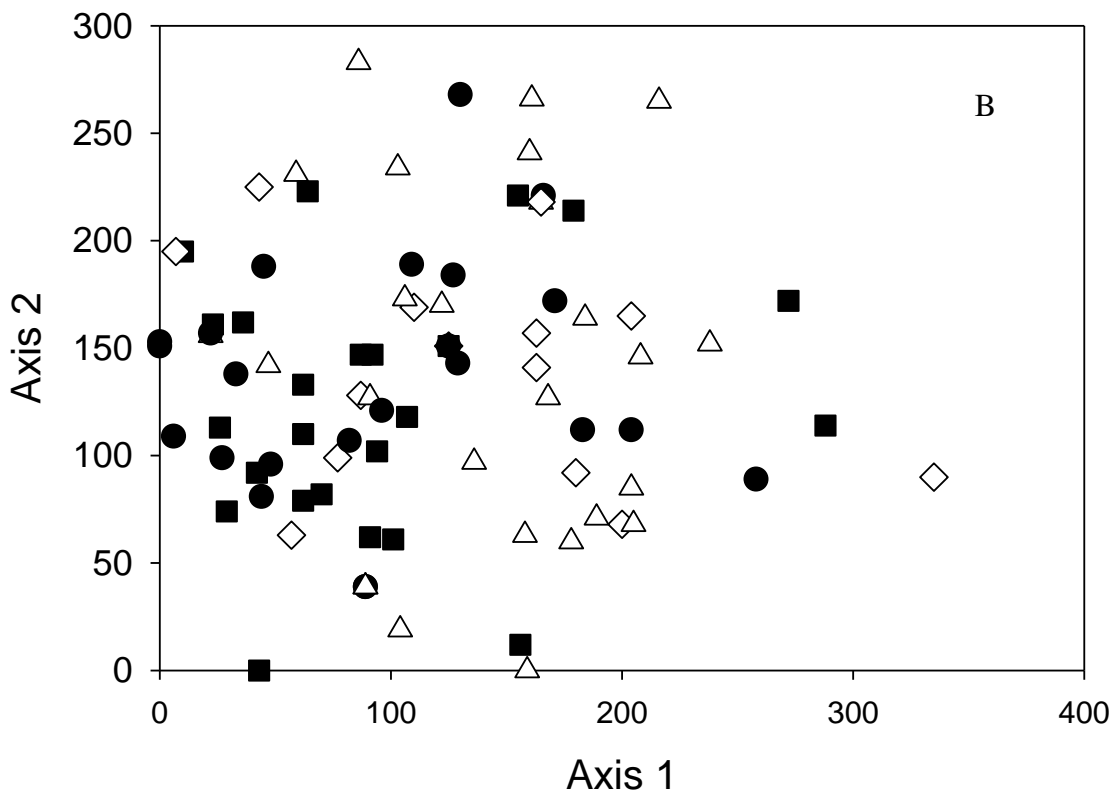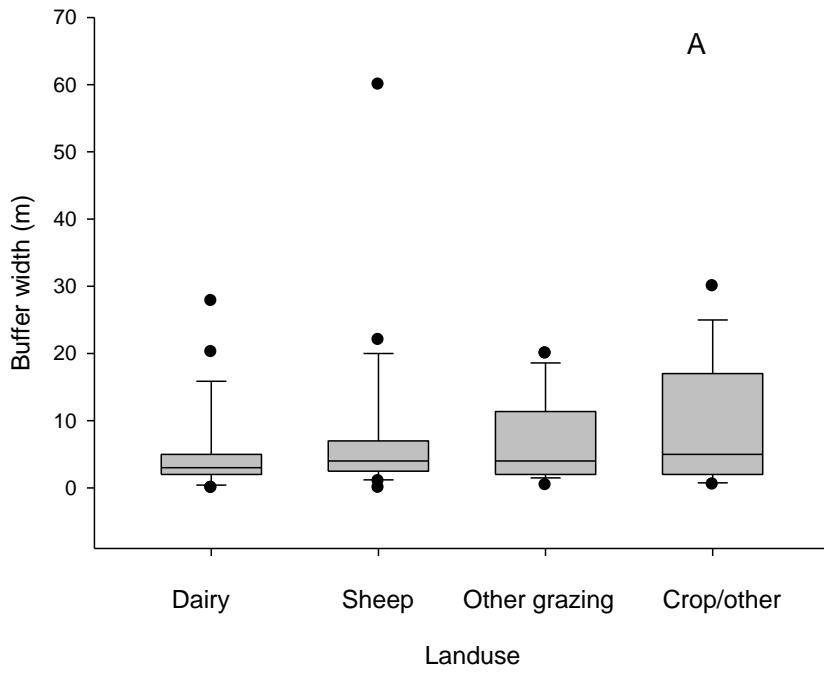
Figure 3.

Figure 4.

Tables

Table 1. Presence and dominance of grouped taxa within (local), and across all (regional), riparian margins of 88 small agricultural waterways of the Canterbury region of the South Island, New Zealand (CSI highest possible score = 75, see Methods for calculation and Appendices 2 & 3 for taxa breakdown).

Vegetation category	Present (%)	Locally dominant (%)	Regional dominance (CSI)
Grass	95	44	38
Weeds	72	5	16
Adventive shrubs &trees	40	7	8
Gorse	32	5	5
Willow	25	14	9
Planted exotic trees	19	8	11
Native flax & sedges	34	14	8
Native shrubs and trees	24	5	10

Riparian nutrient dynamics in a flat landscape - Grass filter strips, vegetated buffers and the influence of land-use

Abstract

Land-use intensification, from sheep to dairy farming in New Zealand, has been identified as a significant factor increasing agricultural pollutant loads to waterways. Implementing riparian buffer zones is a management tool available to reduce and prevent further impacts on water quality. Planting native vegetation in riparian margins has been widely promoted in New Zealand for protection of farm waterways, yet active riparian planting is still not common. To determine typical nutrient loads entering riparian buffers from dairying and sheep farming activities on the Canterbury Plains, availability of soil nitrogen (N) and phosphorus (P) at 10cm depth was compared at field-edges adjacent to 10 high-intensity (dairy) and 10 low-intensity (sheep or dry-stock) pastures. I then compared N and P removal by riparian buffers of mixed native vegetation with that of grass-only filter strips. Within shallow riparian soils of the Canterbury plains, field-edge N and P accumulation was highly variable but not significantly different between land-use activities. Overall native buffers retained 1–170% more N and P than grass strips and were 2-12% more efficient at retaining N than grass buffers although they were 1% less efficient at trapping P. Season strongly affected field-edge N and P availability and buffer nutrient retention which were further influenced by site characteristics, management context and nutrient chemistry, processing and cycling with the landscape and riparian zone. From these results, I recommend that a) an upslope, rank grass strip (of 1 – 2 m in flat landscapes) adjacent to the land-use is the most effective way of trapping of particulate-adsorbed nutrients in run-off, b) in combination, a down-slope buffer (5 – 10 m) of mixed New Zealand native shrub and tree species is likely to provide better waterway protection than grass only and c), to optimise stream buffering effort topographic setting, riparian zone hydrology and seasonal land-use management must be taken into account when planning buffer width and placement.

Introduction

Preventing and reducing the adverse impacts of agricultural productivity on fresh water systems is an issue of global concern and, continued expansion and intensification of agricultural practices places waterways at risk of further degradation. In New Zealand, agricultural intensification, from sheep/beef farming to dairy farming, has been widespread over recent decades with notably, greater exports of nutrient pollutants to waterways from dairying, than pastures grazed by dry-stock (e.g. sheep and beef cattle) (Cameron and Di 2004, MacLeod and Moller 2006, Wilcock et al. 2006, Houlbrooke et al. 2011, Wilcock et al. 2011b). To prevent the continued degradation associated with land-use intensification, implementation of 'best management practices' for farm waterway protection is advocated by land and water management authorities (TRC 1993, ECAN 2005(a)). However, while many of these practices substantially reduce point-source pollution (e.g. dairy-shed effluent), diffuse-source nutrient losses from field-edge run-off remain a major contributor to waterway pollution (Wilcock et al. 2009, Roygard et al. 2012).

The use of vegetated riparian buffer zones has been identified as an important management tool to reduce sediment, nutrients and faecal contaminants in run-off, from entering farm waterways (Quinn et al. 2001, Parkyn 2004, McKergow et al. 2006a, Collins et al. 2007, Wilcock et al. 2009). Stream-side planting of New Zealand's native riparian vegetation is widely promoted for this purpose with the benefits to native fauna, by improvement of riparian and in-stream habitat, being well-documented (Collier et al. 1995, ECAN 2005(a), Jowett et al. 2009). Despite its advocacy, this strategy has received little attention in Canterbury, the majority of riparian buffers consisting of narrow rank pasture-grass and weed strips, with only sporadic, disconnected native riparian planting in evidence (Greenwood et al. 2012) (Chap. 2). This may be directly due to an absence of empirical research documenting the ability of New Zealand native vegetation to act as riparian buffers and, land-owner caution in implementing a potentially time-consuming and costly riparian strategy that may not eventuate in better water protection outcomes, than retiring and fencing off grass strips.

International research has shown that riparian buffers are capable of removing a substantial proportion of particulate N and P from field-edge run-off and, dissolved N and P from sub-surface flow and shallow ground-waters (Dosskey et al. 2002, Abu-Zreig et al. 2003, Blanco-Canqui et al. 2004b, Balestrini et al. 2011). However, buffer effectiveness can

be dependent on a multitude of complexly interacting factors that include the pollutant source and transport mechanism (Daniels and Gilliam 1996, Mayer et al. 2007, Dosskey et al. 2010), its hydrological flow-path in relation to buffer placement within the context of landscape topography (Smith 1989, Hart et al. 2004, Polyakov 2005, Dosskey et al. 2006, Sheppard et al. 2006, Kronvang et al. 2009). Furthermore, the length and intensity of wetting events, soil type and buffer slope will influence pollutant transportation and whether it occurs as run-off or subsurface flow, and infiltrates, flows through or beneath the buffer vegetation (Smith and Monaghan 2003, McKergow et al. 2006a, Hoffmann et al. 2009). Importantly, the ultimate determinant of riparian buffer zone pollutant retention effectiveness, and thus waterway protection, is the buffer width and the type and composition of vegetation within the buffer (Magette et al. 1989, Blanco-Canqui et al. 2004a, Syversen 2005, Mankin et al. 2007, Liu et al. 2008, Zhang et al. 2010, Rasmussen et al. 2011).

Infiltration has been identified as the key physical process within riparian buffers enabling reduction and removal of agricultural pollutants (Barfield et al. 1998, Lee et al. 2000, Sheppard et al. 2006, Mankin et al. 2007). Grass buffers have proven to be particularly effective for trapping and filtering suspended sediment, sediment-adsorbed P, and particle-associated N entrained in surface run-off (Smith 1989, Mendez et al. 1999, Blanco-Canqui et al. 2004a, Duchemin and Hogue 2009). This is attributable to the high particulate-trapping efficiency of grass, a function of stalk density and percent ground-cover, which reduces run-off velocity and increases infiltration rate (Mendez et al. 1999, Abu-Zreig et al. 2003, Mankin et al. 2007). Grass buffers may also reduce dissolved nutrients in run-off, but have proven less effective than deeper-rooting vegetation for removing dissolved N and P infiltrated within the soil profile (Dillaha et al. 1988, Osborne and Kovacic 1993, Daniels and Gilliam 1996).

By comparison, buffers consisting solely of trees have proven less effective than grass filter strips for retaining particle-adsorbed nutrients in surface run-off, due to lack of surface vegetation roughness, which would otherwise be provided by high stem density of herbaceous ground-cover vegetation (Phillips 1989b). In some instances shading-out of herbaceous ground-cover by tree or shrub canopies may also cause crusting of bare surface-soil thus, reducing soil porosity and infiltration rate. For example, eucalypt buffers in Australia were less effective at reducing total N and P, filterable reactive P (FRP) and sediment loads, than grass buffers especially during high run-off events (McKergow et al. 2006b).

The key biological processes controlling nutrient removal in vegetated buffers are plant uptake, microbial immobilisation and denitrification (Mander et al. 1997, Dhondt et al. 2003, Hefting et al. 2005, Polyakov 2005). Buffers of shrub and tree species have been demonstrated to be more effective than grass at removing dissolved nutrients infiltrated within the soil profile, this being attributable to improved soil structure and increased infiltration depth which increases residence-time of pollutants within the buffer zone (Schoonover et al. 2005, McKergow et al. 2006b, Balestrini et al. 2011). Under woody vegetation pollutant retention is enhanced by the greater depth, spatial extent and complexity of shrub and tree rooting structures (Borin and Bigon 2002), as well as increased nutrient requirements of both the vegetation and associated diverse soil biota (Lee et al. 2000, Mankin et al. 2007).

Climatic and seasonal factors can have strong effects on nutrient retention in riparian zones. Soil nutrient availability and limitation are strongly influenced by seasonal plant growth cycles and fluctuations in their associated assimilatory plant and soil microbial populations (Osborne and Kovacic 1993, Syversen 2002, Dhondt et al. 2003, Sovik and Syversen 2008). Sabater (2003) however, found no influence of season on nitrogen removal by herbaceous or tree buffers across a European climatic gradient, concluding that nitrate attenuation was more closely related to nitrate load and hydraulic gradient. In contrast, Hefting (2005) found no effect of lateral nutrient loading on N removal by plant productivity or uptake in a later study of similar European riparian buffer zones. Thus, although nutrient assimilation rates can be influenced by external nutrient load this is likely to be context-dependent and determined by existing riparian soil nutrient status (i.e., limited or saturated) and soil biotic requirements (Sabater et al. 2003, Hefting et al. 2005). Nutrient uptake by plants differs with the age and growth rate of riparian species, which may affect nutrient demand and storage capacity, Mander et al (1997) finding that young forest stands were considerably more efficient at removing N and P inputs than older stands in the USA.

Although the factors influencing nutrient retention in riparian buffers are generally understood there is still a gap in our knowledge regarding which management strategies are most effective in reducing nutrient runoff from agricultural areas. Specifically, in New Zealand it is not clear whether the advocated, but time-consuming and costly, planted riparian buffers of mixed indigenous ground-cover, shrub and tree species, would be more effective than existing rank grass margins for waterway protection, particularly in the context of lowland plains areas. Furthermore, it is not clear how increased agricultural intensity

specifically, from sheep to dairy farming, which occurs mainly on the plains, will affect riparian buffer performance.

To address these questions, in a novel approach I used ion-exchange resin bags to determine the nutrient (nitrogen and phosphorus) status of shallow soils within riparian buffers adjacent to farm waterways on the Canterbury Plains. I first compared soil nutrient status at the agricultural/riparian boundary adjacent to high-intensity (dairying) and low-intensity (dry-stock grazing) land use activities. I then compared nutrient retention within riparian buffer zones of rank grass filter strips, with that of buffer zones planted with New Zealand native vegetation.

Materials and Methods

Study Area and Site Locations

This research was conducted on the Canterbury Plains on the east coast of New Zealand's South Island (latitude 42–43°, longitude 171–173°) between April 2010 and January 2011 (austral late autumn – early summer). Mean annual rainfall ranges from 600–800 mm and mean annual temperatures from 10°–12.5° C (Sturman 2008). Soils across the Plains range from deep silt loams in eastern lowland areas to stony sandy loams further west. The Canterbury Plains has been farmed for over 150 years and traditionally, cropping (e.g., wheat, barley) and low intensity sheep and seasonal dairying have been common (Pawson and Holland 2008). In recent decades, land-use intensification, mainly by sheep or beef farming conversion to dairying, has been made possible by the higher stock-carrying capacity of intensively irrigated land (MacLeod and Moller 2006, Wilcock et al. 2011b). This has resulted in highly improved pastures with rapid cycling and greater loss, of applied nutrients (MfE 2006). Canterbury waterways now have the highest mean concentration of dissolved inorganic nitrogen (> 2mg/L) in New Zealand and enriched, to excessive concentrations of dissolved reactive phosphorus (> 0.5 mg/L) with the highest concentrations occurring in spring-fed plains waterways (ECAN 2010). Small lowland streams and drains are generally most impacted due to their size, proximity to adjacent land-use and the cumulative effect of upstream land-use (ECAN 2010, Greenwood et al. 2012).

Study Design

I used a repeated measures nested design, incorporating two different riparian treatments (native vegetation and grass) within two land-uses intensity categories (high and low), over 3 approximate seasonal periods (winter, spring and summer). I located native and grass riparian buffers at reaches of 20 small farm waterways that included braided river tributaries, spring-fed creeks, lowland streams and drains. Appendix 4 shows the geographic location of these sites. Ten riparian margins were adjacent to high intensity land-use (i.e., dairy farming) and ten adjacent to low-intensity land-use (e.g., sheep) with five of each riparian treatment in each. Land managers at high intensity dairying sites generally grazed stock on a 20–30 day rotation and most irrigated paddocks with diluted dairy shed effluent. Farms at low intensity sites either ran several hundred sheep on rotation or, grazed small numbers (< 50 head) of dry-stock such as sheep, cattle or horses. These farms did not use irrigation, but did apply fertiliser bi-annually. Most native vegetation buffers contained 5-10 species of varying

growth form which included *Phormium tenax, Cordyline australis, Carex spp., Pittosporum tenuifolium, Pittosporum eugenioides, Plagianthus regius, Hoheria populnea* and Olearia and Hebe species. Rank grass buffers contained mainly of cultivars of perennial ryegrasses (*Lolium multiflorum & L. perenne*), tall fescue (*Festuca arundinaceae*) and cocksfoot (*Dactylis glomerata*). Up to 5–10% of annual/biennial weeds were also present in most buffers. In Buffer zones were all between 2.5 – 5.5 metres wide (Appendices 5 & 6 show some of the native (App. 5), and grass buffers (App. 6) used in this study).

I hypothesized that, due to sequestration of nutrients by riparian buffer vegetation, I would observe a gradient of higher-to-lower nutrient concentrations at the land-use/riparian boundary, this being the nutrient run-off source, than at the stream-side edge. The strength of the gradient in soil nutrient availability across the riparian buffer should be proportional to the amount of nutrient retention by the buffer. Nutrient retention was defined as the difference in soil nutrient concentration between the agricultural/riparian boundary (field edge) and the near-stream riparian edge.

Ion-Exchange Resin Bags

I used ion-exchange resin bags to measure soil nutrient availability. These consisted of 5 X 5 cm nylon bags filled with 5 grams (dry weight) mixed bed ion-exchange resin beads (Dowex^{TM} Marathon^{TM} MR-3 Mixed Ion Exchange Resin). Validation tests were carried out to calibrate and confirm resin absorption of PO_4^{3-}–P, NO_2^-, NO_3^- & NH_4^+ –N, and subsequent efficiency of extraction procedures. All bags were pre-leached in 0.5 M HCL prior to deployment, rinsed multiple times in DI water and stored in Ziploc bags at 4° C until required (as per Aridlands Ecology Lab Protocol).

Ion-exchange resin bags are commonly used for assessing in-situ soil nutrient status and provide a measure of the relative availability of nutrients in the soil solution, which correlate well with other in-situ soil nutrient indices (Hart and Binkley 1985, Saggar et al. 1990). The resin beads within predominantly exchange cations and anions with soil solution ions in mass flow, and to a lesser extent, by diffusion (Binkley 1984, Carlyle and Malcolm 1986). The bags act as an infinite sink for soil nutrients, so are particularly effective at indicating nutrient supply rates over extended periods (> 4 weeks) in the field (Meason and Idol 2008). The resin bag method is highly sensitive to field conditions involving transport and mobility of ions, so is particularly appropriate for this study (Binkley 1984, Meason and Idol 2008).

Field and Laboratory Methods

At each location two transects were set, one at the field edge of the buffer ("top bags") and a second near the stream-side edge of the buffer ("bottom bags"). Each transect was approximately 4 metres in length. The exact position of bag deployment was determined by site-scale topography specifically, at the lowest point in the landscape along the fence-line, to increase the chance of run-off. At 1 m intervals along each transect a trowel was used to make narrow slits in the soil to 10 cm depth. A resin bag was inserted to the bottom of each slit with nylon cord protruding disturbing as little of the surrounding soil as possible. Soil around the bag was then tamped down and the cord secured to a narrow stake pushed into adjacent soil. Resin bags were deployed at each site on three occasions from April to July, 2011 (early Winter), July to October, 2011 (early Spring) and October, 2011 to January, 2012 (early Summer), for an average of 92, 97 and 91 days, respectively. Deployment and retrieval occurred over a 3-week period and, each set of five retrieved bags were placed in Ziploc bags as a composite sample, labelled by site and position (top or bottom) and returned to the laboratory. Soil moisture recordings were also taken adjacent to each resin-bag location (soil probe name), before fresh bags were installed in the same position.

Returned bags were cleaned of adherent plant roots and soil and rinsed multiple times in ultra pure water (Milli-Q). For each composite 5-bag sample (25g dry weight resin), resin-adsorbed ionic nutrients were extracted by shaking in 200ml 2 M KCL, then filtering through Whatman™ 24 mm glass micro-fibre filters (Aridlands Ecology Lab Protocol). Colorimetric analysis was then performed on the eluants using an EasyChem® continuous flow auto-analyser (Systea, Italy) for NO_3^- and NO_2^- –N and PO_4^{3-} –P. NH_4^+ –N was extracted by the indophenol blue method (Solorzan.L 1969) and analysed by a spectrophotometric analyser (Shimadzu, Kyoto, Japan).

Data Analysis

Chemically analysed nutrient concentrations for NO_3, NO_2 & NH_4 –N, and PO_4 –P were converted to resin-adsorbed, KCL-extracted concentrations for descriptive and statistical analysis by the following equation:

$$\mu g\ nutrient\ g^{-1}\ dry\ resin = (nutrient\ (mg/L)* 0.200)* (1000/25)$$

Average N and P supply at field-edge margins (top bags) for each site, during the study period were derived from the means of the 3 season values. Datasets were dominated by

extreme values and failed normality tests (Shapiro-Wilk). Log10 or reciprocal transformation was performed and ANOVAs used to compare soil nutrient supply at high and low land-use intensity sites. Chi Square analysis was used to determine site-relatedness of extreme values. Between-season differences in accumulated field-edge nutrients were also examined using Kruskal-Wallis non-parametric tests to compare overall median values and, between land-use intensities.

Datasets for nutrient retention analysis comprised paired top-bag (T) and bottom-bag (B) nutrient concentration values, and the ratio of B/T was used to estimate net retention at each site. A ratio of < 1 indicated net retention, whereas a ratio of > 1 indicated net release. Retention ratios were highly variable and most required Log10 transformation to pass normality tests. Mean, standard deviation and 95% confidence interval was then calculated for each nested treatment group (5 sites/group), then back-transformed for graphical presentation. Average nutrient retention (for 10-month duration) at each site was derived from the mean of the 3 season values derived from.

Graphical comparisons were displayed using a log10 y-axis where required. Statistical comparison of simple datasets requiring none, or a common normalising method (Log10 or Reciprocal transformation) were analysed using ANOVA and MANOVA. Complex or larger datasets otherwise requiring different normalising methods were analysed using non-parametric Kruskal-Wallis one-way analysis of variance on ranks to test if means were similar. Percent retention was calculated on untransformed data, using the equation:

$$\%Ret = (T - B) / T$$

Median percent retention values were used in subsequent analysis, as opposed to means to avoid skewing data containing extreme negative outliers (bottom >>> top). Retention efficacy was calculated using the equation:

$$Eff = (ln(T) - ln(B)) / m$$

where ln is the natural log of top and bottom bag concentrations and m is the linear buffer width. Soil moisture readings were averaged across each top or bottom site and regressions performed to determine their correlation to nutrient concentrations. Graphical analysis is presented using Sigma Plot (Systat Software Inc, San Jose, CA, USA) and all statistical analysis was performed using R (version 14.2.2) (Crawley 2005).

Results

Field-edge nutrient availability

Field-edge (top bag) nutrient availability was highly variable within each land-use type over the 10-month study period (Figure 1). Although mean soil N and P for each site (average value of 3 x 90-day periods) was generally lower at low intensity sites, ANOVAs indicated that there was no statistical difference in nutrient availability between high- and low-intensity land-uses (Table 1). The mean NO_3 was most variable at low intensity sites (2.52–27.78 µg g-1 dry resin) which had the 3 highest mean values. High intensity sites varied most in mean NO_2 (0.16–8.62 µg NO_2 g^{-1} dry resin) and NH_4 (108–614 mg NH_4 g^{-1} dry resin). Field-edge PO_4 was lower at low intensity sites, but spanned 3 orders of magnitude within both land-use categories (2.24–110 µg PO_4 & 1.23–512 µg PO_4 g^{-1} dry resin, for low- and high-intensity, respectively).

Between-season analysis showed significant differences in median field-edge concentrations for NO_2 & NH_4 and PO_4, although NO_3 availability was unaffected by season (Table 2). Field-edge NO_3 was generally low and median values exhibited less variation overall, except for 2 sites (10 & 14) which were consistently high for the duration of the study (Fig. 2a). NO_2 peaked in winter at most sites with concentrations generally > 1 µg NO_2 g^{-1} dry resin (range 0.02–24.97 µg NO_2 g^{-1} dry resin), whereas during spring and summer, concentrations were as much as 3 orders of magnitude lower (Fig. 2b). NH_4 concentrations were consistently higher than all other N & P species by 2–3 orders of magnitude. At the majority of sites NH_4 peaked during summer (range 119–647 µg NH_4 g^{-1} dry resin), although higher values were recorded at 2 sites for spring (Fig. 2c). PO_4 peaked during winter with values ranging from 1.5–1535 µg PO_4 g^{-1} dry resin, and were 2–3 orders of magnitude higher during this time than in spring and summer (Fig. 2d).

The high variability of field-edge N and P within each land-use intensity category meant there was no significant effect of land-use and very little combined effect of land-use intensity and season (Table 3). Overall, nutrient concentration ranges were generally slightly higher at high-intensity sites than at low-intensity sites except during spring and summer for NO_3. These results are discussed within the context of site-dependent factors such as topography and, environmental and management differences, in the discussion below.

Riparian Buffer Retention

Net retention (B/T ratio) of N and P was estimated by comparing field-edge resin bag concentrations with those of near-stream resin bags. Mean nutrient retention for the 10-month period showed little evidence of the expected nutrient dilution gradient from top to bottom riparian edges, within land-use/riparian-type groups (Figure 3). Extreme values dominated the data and net retention of N and P for the period was not significantly different between the two buffer types (ANOVA, $p > 0.1$). However, median percent-retention data indicated that, over the duration of the study, native buffers performed slightly better than grass buffers by 6%, 170%, 1% and 34% for NO_3, NO_2 & NH_4 and PO_4 respectively. NO_3 was retained at 13/20 sites (7 grass & 6 native), NH_4 was retained at 13 sites (5 grass & 8 native), NO_2 at only 3 sites (2 grass & 1 native) and PO_4 at 8 sites (3 grass & 5 native).

There were no significant relationships between land-use and retention of N species, although NH_4 retention approached significance with slightly higher retention at low intensity sites (ANOVA, $F_{1,18} = 2.841$, $p = 0.109$). Land-use had a significant influence on PO_4 retention which was greater at high-intensity sites compared with low-intensity sites (ANOVA, $F_{1,18} = 5.857$, $p = 0.0263$).

Seasonal buffer retention

Between-season analysis showed that nitrogen retention was not significantly influenced by season, although within the data there was some evidence of seasonal patterns, which differed depending on the N species analysed. There was moderate-good retention of NO_3 during winter (Fig.4a), which occurred consistently across 16/20 sites. Native buffers generally performed slightly better than grass buffers, with median NO_3 retention being 54% at high intensity sites and 41% at low intensity sites, while medians were 35% and 25% for high- and low-intensity sites within grass buffers.

By contrast, NO_2 retention (Fig. 4b), was low and inconsistent in winter, occurring at only 9/20 sites. Five native buffers retained 65–94% of NO_2 and 4 grass buffers retained 53–82% NO_2. At several sites bottom resin-bag concentrations were 10–100 times higher than top resin-bag concentrations, indicating stream-side NO_2 accumulation. Of the nitrogenous species, only NH_4 retention (Fig. 4c), showed a relationship with land-use intensity but this was not statistically significant (ANOVA, $F_{1,18} = 2.841$, $p = 0.109$). Slightly higher NH_4 retention was achieved by 6/10 native buffers (17-92 *%Ret*, med. 20%), than by 5/10 grass buffers (17–71 *%Ret*, med. -26%).

Retention of PO_4 was significantly higher during winter than during any other period (ANOVA, $F_{5,54} = 4.713$, $p = 0.001$). It was also significantly higher at high intensity dairying sites than at low intensity sites (ANOVA, $F_{1,18} = 5.803$, $p = 0.027$) (Fig. 4d). This mainly occurred in grass buffers with percent-retention ranging from 59–99 % at 4 sites, and a median value of 94 % PO_4 retention over all 5 sites, compared with a median of 82% retention by native buffers at high intensity sites.

In spring ratios showed consistent and moderate retention of NO_3 & NH_4 by most buffers, although N retention by grass buffers was highly variable adjacent to high intensity land-use. NO_3 (Fig. 5a), was retained by 6/10 native buffers (range 3–82 *%Ret*, med. 10%) and also by 6/10 grass buffers (range 9–72 *%Ret*, med. 14%). Grass buffers at high-intensity sites retained significantly less NO_2 than other land-use/buffer types ($F_{3,16} = 2.384$, $p = 0.04$), although at low-intensity sites, only 1 site lost NO_2 (med. -1175 & 37 *%Ret*. over 10 high- and 10 low-intensity sites, respectively) (Fig. 5b). Native buffers retained NO_2 slightly better than grass buffers overall. NH_4 retention (Fig. 5c), was higher and more consistent in native buffers, occurring at 8/10 sites (range -81% – 87 *%Ret*, med. 41 %, n=10) compared with grass at only 5/10 sites (range -781 – 77 *%Ret,* med. 5%). Retention ratios for PO_4 during spring showed that removal and losses were generally low (Fig. 5d). PO_4 was retained by 8/10 native buffers, with a median value overall of 16% retention and by 5/10 grass buffers, with a median value of -18% (refer to refer to Appendix 7).

During summer NO_3 was consistently retained by both buffer types and losses to waterways were low (Fig. 6a). NO_3 was retained at 8/10 native buffers (range 6–97 *%Ret*, med. 19%) and 5/10 grass buffers (range 27–79 *%Ret*, med. 13%). Over half of the buffers (12/20) lost NO_2 and concentrations measured from streamside bags were frequently 10–100 times greater than upslope bags (Fig. 6b). NH_4 was retained at 15 sites overall and losses were low (Fig. 6c). Grass buffers removed 14–91% NH_4 at 8 sites (med. 26%), and native buffers removed 34–82% NH_4 at 7 sites (med 45%). Both PO_4 retention, and loss, in summer were generally low, retention occurring at only 4/10 native sites (14–70 *%Ret*) and 5/10 grass sites (6–57 *%Ret*). The median retention values for both native and grass buffers (n=10 each) was -2% and losses from the remaining 11/20 sites ranged from 1–98 % (refer to Appendix 7).

Riparian Buffer Efficiency

Buffer width was used to estimate riparian buffer efficiency taking into account nutrient retention per linear metre of buffer vegetation at each site (refer to Appendix 8 for details).

Similar patterns of either nutrient removal or nutrient loss occurred across land-use intensity and buffer-type categories with median efficiency values for each land-use/buffer type group generally corresponding with percent retained (i.e., positive N or P retention, or N or P loss). Native riparian vegetation was 1% less efficient at retaining PO_4 per linear metre of buffer, than grass set-aside buffers, but 2%, 8% and 12% more effective than grass for NO_2, NO_3 & NH_4 respectively.

The influence of soil moisture status

There was limited correlation between accumulated resin-adsorbed nutrient concentrations and soil moisture status. At field-edge (top) positions only PO_4P during winter was significantly influenced by soil moisture ($F_{1,18} = 5.128$, $p = 0.04$). There was a significant influence of soil moisture status on stream-side (bottom) nutrient concentrations during Winter for PO_4 ($F_{1,18} = 5.445$, $p = 0.03$), Spring for NO_2 ($F_{1,17} = 4.465$, $p = 0.05$) and Summer for NO_2 ($F_{1,18} = 11.65$, $p = 0.003$) and NO_3 ($F_{1,18} = 6.253$, $p = 0.02$).

Soil saturation was significantly influenced by season as expected ($F_{2,54} = 11.1$, $p < 0.0001$), and to a lessor extent by the interaction between land-use and season ($F_{5,51} = 5.273$, $p < 0.001$), although not by land-use alone ($F_{1,55} = 2.527$, $p = 0.1176$).

Discussion

Land-use effects on field-edge nutrient availability

Intensification of farming practices on New Zealand's Canterbury Plains could place already degraded waterways at further risk from increased nutrient inputs. Dairying is known to contribute larger nutrient loads to adjacent waterways than sheep or dry-stock farming systems, as a result of higher fertiliser application, effluent and irrigation water usage and, in greater animal urine volumes (Monaghan et al. 2007b, McDowell and Wilcock 2008). Although it was expected that soil nutrient levels adjacent to dairy pastures would be considerably higher than adjacent to dry-stock pastures, my results provided little evidence to support this. These findings and their probable causes are discussed below.

On the Canterbury Plains, under high- and low-intensity land-uses, field-edge nitrogen and phosphate–P (PO_4) concentrations were characterised by wide variation. At several sites, specific one-off management influences may have contributed to 'extreme' high nutrient concentrations encountered. For example, harvesting of a pine shelter-belt immediately adjacent site #1, ploughing of sheep-grazed pasture for barley-cropping adjacent to site #10, and sheep access directly to the waters edge at another site (#11) all appeared to cause substantial disturbance to field-edge PO_4 availability, and these effects became consistent for following seasonal periods. Other 'extremes' were common across all nutrient species within a site, but not across seasons. At two low-intensity sites (#14 and #17), addition of organic nitrogen (manure and grass clippings) directly over resin-bag positions, prior to 1-2 weeks snow-covering, created a temporary and concentrated nutrient supply. A number of workers also report that spatially variable urine deposition can contribute to uneven patterns in soil nutrient availability, also providing a significant source of concentrated and biologically labile NO_3 and NH_4 (Williams and Haynes 1990, Burns et al. 1995b, Moir et al. 2011).

Nitrate-N (NO_3) concentrations in run-off from grazed pastures can reportedly be highly variable (e.g.,11–4997 mg m^{-3}) (Smith 1989). In dairy pastures, the main source of NO_3 is associated with nitrification of NH_4 originating from high-density urine patches (Maneer et al. 2004, Moir et al. 2011). I recorded low and fairly constant (1.4–30 µg) field-edge soil NO_3 concentrations (at 10cm depth), adjacent to dairy pastures and this varied little between seasons. This is likely due to the topographical context of the Canterbury Plains where the majority of dairy-farming operations in this study, and indeed in New Zealand, are

located within flat landscapes of coastal plains where the groundwater table is relatively shallow. Under the mostly well-drained soils, surface flows may only occur under saturation-excess conditions resulting from irrigation and soil compaction or, as intermittent responses to excessive rainfall events, occurring in winter and early spring (Smith and Monaghan 2003, McKergow et al. 2006a, McDowell and Houlbrooke 2009). The solubility of NO_3 means that it is highly susceptible to leaching and indeed high urine volumes are more likely initiate macro-pore flow and a rapid response in leached nutrient concentrations (Magesan et al. 1996). Thus sub-surface flow, following within-paddock leaching has been identified as the dominant pathway for NO_3 losses and reportedly, these can be 5 to 10 times higher than in surface run-off (Cooke and Dons 1988, Ledgard et al. 1998, Cameron and Di 2004). Importantly, my research was conducted within shallow (10cm) soils and therefore these results, in accordance with other reports, suggest that in lowland Canterbury under high intensity management, NO_3 most likely enters waterways via sub-surface flow during the majority of the year.

Season produced the strongest effect on field-edge soil nitrite–N concentrations, regardless of land-use activity. In winter NO_2 concentrations were 10 to 100-fold higher than in spring and summer and, were also higher relative to NO_3 at this time. There are a number of explanations for these findings. NO_2 is known to have short residence time in the soil, and is usually rapidly transformed to NO_3 under aerobic conditions by nitrifying bacteria (Burns et al. 1995b). However, under winter-early spring soil saturation conditions of lowland Canterbury, nitrification rates may be slower, causing elevated concentrations of NH_4 and NO_2. This may indicate that while initial conditions were favourable for NO_2 production, further processing was inhibited. Furthermore, inhibition of NO_2 oxidisers is known to occur in the region of urine patches and excessive concentrations of free NH_4^+ ions (Burns et al. 1995a, Burns et al. 1995b)

This study recorded significantly higher field-edge NH_4 concentrations (by 10 to 100-fold), than NO_2 and NO_3. Because most soils in temperate regions have a net negative charge, NH_4^+ ions are readily adsorbed onto soil exchange surfaces which limits NH_4 mobility in the soil. This provides a greater opportunity for plant uptake, immobilisation within organic matter or association with clay particles as opposed to leaching (Di and Cameron 2007). However, during winter, the mobility of clay-sorbed NH_4^+ ions in the saturated clayey lowland soils may increase, causing high field-edge deposition in run-off. This may reflect the tendency of the mainly silty loams of the lowland dairying sites to be susceptible to structural breakdown due to the combined effects of soil saturation and compaction by dairy

animals which can reduce drainage capacity. Soil NH_4 levels peaked in summer, indicating there may be a reservoir of inorganic N held within surface soils, most likely a result of decreased NH_4 mobility under drier soil conditions and, greater rates of mineralization of organic N due to warmer temperatures.

Phosphate–P can be strongly held by the soil matrix, and is predominantly delivered to riparian areas in surface run-off, or via preferential flow-paths (surface channelling), as a result of sedimentation (Heathwaite and Dils 2000, Toor et al. 2004b). However, dissolved P can also be lost from grazed pastures via leaching, entering sub-surface flow-paths (McDowell et al. 2001, Toor et al. 2004a). In flat landscapes deposition of sediment-bound PO_4 at field-edges is entirely dependent on favourable hydrological conditions and thus, run-off volumes are generally lower in spring and summer with very little lateral movement of P occurring (McDowell et al. 2001, Sheppard et al. 2006). In this study, season was the main influence on field-edge PO_4 concentrations at all sites. During winter, field-edge PO_4 levels were 2–3 orders of magnitude higher than at any other time and appeared directly related to soil moisture conditions. It was also considerably higher at high-intensity sites (165.24μg) than at low-intensity sites (34.49 μg), most likely due to the more constant soil moisture conditions required of high intensity farming methods. I found that PO_4 accumulation at inland sites adjacent to low-intensity sheep pastures was significantly lower. This was likely due to superior drainage afforded by the stony, sandy soils which maintained a significantly higher soil-moisture deficit, promoting infiltration and absorption of rainfall and applied fertilisers within the paddock, resulting in negligible run-off (Daniels and Gilliam 1996, Sheppard et al. 2006).

Nutrient retention by riparian buffers

Riparian management along small farm waterways on the Canterbury Plains, most commonly consists of rank, pasture-grass filter strips, despite land and water management authorities advocating planting of native riparian vegetation (Chapter 2). However, the effectiveness of New Zealand native vegetation as agricultural nutrient filters for water quality protection has not yet been quantified. I compared nutrient retention effectiveness of ten rank pasture-grass filter strips, with ten riparian buffers of mixed native shrub and tree species. I found that although differences were not statistically significant, the native-vegetated buffers demonstrated slightly higher percent-retention of field-edge nutrient load and, higher per-metre efficacy overall, compared with grass buffers of similar widths. My results supported those previously noted in the literature for N and P removal by grass and mixed shrub/tree

vegetation, with both buffer types exhibiting variable effectiveness depending on the N or P species. These findings were also highly reflective of site-specific management, field conditions and stream-side topography, as well as seasonal influences on nutrient mobility and processing, within the climatic and topographic context of the Canterbury Plains.

Many studies report fundamental differences in buffer retention between particulate-associated and dissolved nutrients, between surface run-off and sub-surface flow, and retention of each by grass and herbaceous vegetation (Magette et al. 1989, Daniels and Gilliam 1996, McKergow et al. 2006b, Wang et al. 2012). Daniels and Gilliam (1996) measured up to 50% reduction of total N and P loads, and up to 80% losses of soluble N and P within 6m grass and grass/tree buffers. Osborne & Kovacic (1993) reported that forested buffer strips reduced NO_3 concentrations in shallow groundwater more effectively than grass, but were less efficient at removing total and dissolved P in surface flow. Schoonover (2004) recorded between 68% and 98% reductions of N and P in buffers of forest and groundcover species, but 100% removal by giant cane buffers, due to greater stalk density. Sovik & Syversen (2008) demonstrated on average, 40% higher retention of N and P by young aspen and alder trees, than thick grass due to enhanced infiltration.

Nitrate removal is typically attributed to denitrification, infiltration or plant uptake (Bedard-Haughn et al. 2004b) and reviews suggest that its removal by riparian buffers is often inconsistent and highly variable (Hickey and Doran 2004, Mayer et al. 2007, Zhang et al. 2010). However, in most studies, analysis and reporting of NO_3 concentrations includes NO_2, as NO_2 is known to have a short residence time in aerobic soils, due to rapid microbial transformation to NO_3 (Burns et al. 1995a). However, in this study, I analysed NO_2 and NO_3 separately and found that much of the variability in NO_3 ($+NO_2$) analyses could be explained by the species measured.

There was moderate-high retention of relatively limited, near-surface NO_3 within the majority of buffers on the Canterbury Plains. Overall, mixed native vegetation retained NO_3 slightly more effectively than grass. Exceptions were specific to sites that had undergone disturbance (e.g., drain digging to remove weeds), or had incurred additions of landowner-deposited organic waste (e.g., grass clippings, horse manure). I also found a significant seasonal component to NO_3 retention, with effectiveness highest in winter (median values 30% and 45% for grass and mixed vegetation, respectively). This was slightly higher than the 28% retention reported for 8m grass/weed buffers adjacent to intensively grazed, irrigated pastures in California (Bedard-Haughn et al. 2004b) and by Duchemin (2009), within 5m

grass, and grass/willow saplings strips in Canada, at 33% and 30%, respectively. I also found that three grass buffers adjacent to irrigation drains consistently indicated NO_3 loss during spring and summer. This may have been due to unexploited NO_3 slightly deeper than was accessible by shallow grass roots. Similarly, a study conducted adjacent to drains in Dutch lowlands found that 5m grass buffer strips were ineffective at removing NO_3 due to both site-specific hydrology and, downward seepage of run-off, which passed beneath the buffer zone (Noij et al. 2012).

However, Balestrini et al (2011) found that narrow herbaceous and herbaceous/mixed-tree buffers adjacent to irrigation ditches on intensively-farmed Italian lowlands removed almost all NO_3, attributable to high plant and microbial uptake possibly resulting from greater depth and spatial extent of roots and associated assimilative microbial communities. The results presented in this study suggest that nutrient cycling within the rooting zone of native buffers was sufficient to utilise surface NO_3 available from adjacent pastures, although it is acknowledged that higher NO_3 concentrations may have been present, deeper than was measured in this study.

Nitrite retention, unlike nitrate, was extremely variable and highly inconsistent over time within both buffer types. Each buffer type appeared capable of retaining up to 99% of incoming NO_2 although it frequently appeared to 'leak' through several of the grass buffers. In general native buffers were more effective at retaining NO_2, regardless of field-edge concentration. At several grass buffer sites, and although NO_2 levels were low at the field-edge, stream-side levels were often several orders of magnitude higher, indicating NO_2 accumulation. This may have been attributable to seasonal influences on near-stream NO_2 processing rates. For example, in winter, dense grass buffers associated with well-irrigated dairying sites consistently removed 53–84% of NO_2. This could be attributable to high grass-stalk density increasing the infiltration capacity of uncompacted and aerated riparian soils, thus improving soil NO_2 availability and allowing rapid microbial transformation to NO_3. However, conditions favouring NO_2 accumulation may start to occur as decreasing winter temperatures reduce microbial nutrient requirements, progressively wetter soils inducing a lag phase in the second stage of nitrification (Burns et al. 1995a). It is also possible for NO_2 recycling to occur by microbial-mediated reduction of NO_3 (Burns et al. 1995a) although as stream-side conditions become more similar to wetland conditions, anaerobic denitrification may be more likely to occur (Ruiz-Rueda et al. 2009). At several sites, during the final (early summer) sampling, resin-bags were often retrieved from soggy stream-side sediments, as stream levels were significantly higher than during winter and spring, following seasonal

rainfall and upstream snowmelt. Notably at such sites *P. tenax*, a species common to native wetlands, was generally present. Additionally, stream-side seepage of leached sub-surface flow into sediments would increase soil saturation and anaerobic conditions, prolonging NO_2 residence time as biological processes that consume NO_2^- become weaker than those that produce it (Ruiz-Rueda et al. 2009).

Despite high field-edge concentrations, moderate-high retention of NH_4 was achieved by the majority of buffers. Native vegetation was more effective overall, with median values at 20%, 41% and 45% for winter, spring and summer respectively, compared with grass at median values of -26%, 5% and 26% for the same periods. In wetter, winter-spring conditions, excess and highly mobile NH_4^+ in surface flows may have been transported through grass buffers to stream-side edges in run-off, accounting for some loss and reducing buffer effectiveness. As a comparison, Bedard-Haughn (2003), reported that three, 8m mixed grass/annual weed buffers decreased NH_4 by 34% during a 7 month study and similarly, using four replicated 5m buffers, 47% retention of NH_4 was recorded by grass, compared with 57% by grass/poplar saplings (Duchemin and Hogue 2009).

Notably, high winter field-edge PO_4 was effectively removed by dense grass filter strips found typically adjacent to irrigated, high-intensity dairy sites, but was not removed by low stature, less-vigorous riparian grasses associated with drier, sheep-farming activities. Infiltration within the soil profile can account for over 90% of P removal in buffers (Mankin et al. 2007). Grasses provide high, ground-surface roughness, due to high stem density which reduces run-off flow velocity, enhancing infiltration rate, increasing contact time within riparian vegetation (Abu-Zreig et al. 2003)**.** However, variation in trapping efficiency and infiltration capacity between grass species can depend on differences in above- and below-ground structure, such as species-specific stem- or root- density and length or, the presence of decaying thatch at ground surface (Abu-Zreig et al. 2003). Although considerably lower (10–100 times) field-edge PO_4 was recorded during both spring and summer, grass buffers at high intensity sites seemingly became less effective at removing it. A possible explanation is that as the growing season proceeds, grass biomass becomes increasingly concentrated in stem height (measured at > 1.5m high by summer) rather than stem density, resulting in a corresponding loss of near-surface trapping efficiency.

Many authors report substantially lower trapping efficiencies for particulate and soluble P by tree or shrub buffers, especially under high surface runoff conditions or, where bare-ground occurs as a result of herbaceous ground-cover being shaded–out and suppressed by dense canopy cover (McKergow et al. 2006a, Dosskey et al. 2010). At two of my more

established buffers (15 yrs) shading-out had occurred but for these sites my results indicated no evidence of lowered trapping efficiency. In fact, generally good P-retention by all native buffers was only confounded by site management, with losses mainly occurring at disturbed sites. During winter, most of the native buffers retained high PO_4 loads almost equally as well as grass filter strips, which may have been directly related to the mixed shrub/tree/groundcover of most of the native buffers.

Several authors report only short-term P-removal effectiveness by grass filter strips, with continued PO_4 fertiliser application eventually causing saturation of stream-side sediments (Cooper et al. 1995, Gillingham and Thorrold 2000, Laubel et al. 2003, Bedard-Haughn et al. 2004b, Sheppard et al. 2006, Stutter et al. 2009). Annual light grazing or harvesting of grass filter strips is suggested as a mechanism to reduce saturation risk (Stutter et al. 2009). However, accumulation of P in stream-banks can also contribute to waterway P-enrichment via bank-erosion (Laubel et al. 2003, McDowell et al. 2003, Kronvang et al. 2005). I found no evidence of PO_4 accumulation in my grass, or comparatively young mixed species native buffers. In mixed herbaceous and tree buffers, P-saturation risk should be reduced with improved infiltration, and increased bank stability provided by more complex root structures (Phillips and Marden 2006b, Sheppard et al. 2006, Marden et al. 2007).

In summary, I found a significant seasonal component to both field-edge nutrient availability and buffer zone nutrient retention on the Canterbury Plains. High soil moisture conditions in late-winter-spring generally increased N and P mobility, decreasing spatial variability of nutrients and increasing bio-availability. In winter, grass frequently coped better than native vegetation at filtering nutrients due to greater stalk density. Seasonally-mediated processes cycling N and P within the riparian soil profile play a vital role in nutrient removal via uptake and retention in plant biomass and, transformation by assimilatory microbial communities, which fluctuate in size with changing temperatures (Frank and Groffman 2009). During high growth periods of spring and summer soil nutrient requirements increase and at high-intensity land-use sites, mixed native vegetation generally performed more effectively than grass. This was most likely due to the increased nutrient requirements of larger stature plants, more diverse microbial populations afforded by mixed vegetation (Wardle 2002) and higher spatial distribution, and thus a wider and deeper zone of influence of plant roots and associated assimilative species compared with grass (Borin and Bigon 2002, Balestrini et al. 2011).

Implications for management

The results of this work provide an important insight into factors affecting nutrient loads entering shallow rooting zones of riparian margins on the Canterbury Plains. A key finding was the importance of seasonal effects and land-use activity for explaining nutrient concentrations in adjacent riparian soils.

This study demonstrated that narrow riparian buffers typical of lowland Canterbury are capable of substantial retention of agriculturally-derived nitrogen and phosphorus. Planted buffers of New Zealand native vegetation were found to perform slightly better than grass filter strips in most instances. However, this work supports the value of maintaining some version of high-percentage ground-cover (eg. pasture-grass) adjacent to the agricultural boundary to provide filtering of sediment-bound nutrients and increase run-off infiltration rate. Planting down-slope mixed shrub/tree vegetation towards and adjacent to waterways is recommended to increase soil infiltration of nutrients in run-off. In lowland areas, dissolved nutrients do not appear to be entering waterways in surface run-off from adjacent dairy pastures, but predominantly in sub-surface flow, following within-paddock leaching. Thus the planting of deeper rooting mixed shrub/tree native vegetation will maximise nutrient removal (via plant and microbial uptake) at greater depths within the riparian soil profile and, from shallow stream-side ground-water. Seasonal nutrient and irrigation management of adjacent fields must be considered when determining buffer width and composition requirements, as well as site-scale topography. In conclusion, it is proposed that utilising the surface-pollutant trapping capacity of rank pasture-grass alongside the spatially complex deeper-rooting systems of mixed native vegetation, may be a practical way to optimise waterway protection within riparian buffers of small farm waterways.

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Figure Legends

Fig. 1: Nutrient concentrations at field-edge riparian boundaries of 20 agricultural waterways adjacent to low intensity (n=10) and high intensity (n=10) land-use activities on the Canterbury Plains. Values represent the average of 3 sampling periods over 10 months duration for resin-adsorbed, KCL-extracted nutrient concentrations (Nutrient g^{-1} *dry resin*) for a) nitrate, b) nitrite, c) ammonium and d), phosphate.

Fig. 2: Seasonal fluctuation in field-edge soil nutrient concentration of 20 agricultural waterways of the Canterbury Plains. Values represent resin-adsorbed, KCL-extracted nutrient concentrations (Nutrient g^{-1} *dry resin*) accumulated over 3 (~ 93 day) periods, for a) nitrate, b) nitrite, c) ammonium and d) phosphate.

Fig. 3: Nutrient retention by riparian buffers adjacent to high-intensity land-use (dairying, n=10) and low-intensity land-use (dry-stock grazing, n=10), within grass and native riparian buffer zones of farm waterways on the Canterbury Plains. Values represent the ratio of stream-side nutrient concentrations to fence-line nutrient concentrations, averaged for 3 sampling periods over 10 months duration for a) nitrate, b) nitrite, c) ammonium and d) phosphate. Positive retention is indicated by a of ratio < 1.

Fig. 4: Nutrient retention during winter by riparian buffers adjacent to high-intensity land-use (dairying, n=10) and low-intensity land-use (dry-stock grazing, n=10), within grass and native riparian buffer zones of farm waterways on the Canterbury Plains (*n=5* per treatment). Values represent the ratio of stream-side nutrient concentrations to fence-line nutrient concentrations for a) nitrate, b) nitrite, c) ammonium and d) phosphate. Positive retention is indicated by a of ratio < 1.

Fig 5. Nutrient retention during spring by riparian buffers adjacent to high-intensity land-use (dairying, n=10) and low-intensity land-use (dry-stock grazing, n=10), within grass and native riparian buffer zones of farm waterways on the Canterbury Plains (*n=5* per treatment). Values represent the ratio of stream-side nutrient concentrations to fence-line nutrient concentrations for a) nitrate, b) nitrite, c) ammonium and d) phosphate. Positive retention is indicated by a of ratio < 1.

Fig 6. Nutrient retention during summer by riparian buffers adjacent to high-intensity land-use (dairying, n=10) and low-intensity land-use (dry-stock grazing, n=10), within grass and native riparian buffer zones of farm waterways on the Canterbury Plains (*n=5* per treatment). Values represent the ratio of stream-side nutrient concentrations to fence-line nutrient concentrations for a) nitrate, b) nitrite, c) ammonium and d) phosphate. Positive retention is indicated by a of ratio < 1.
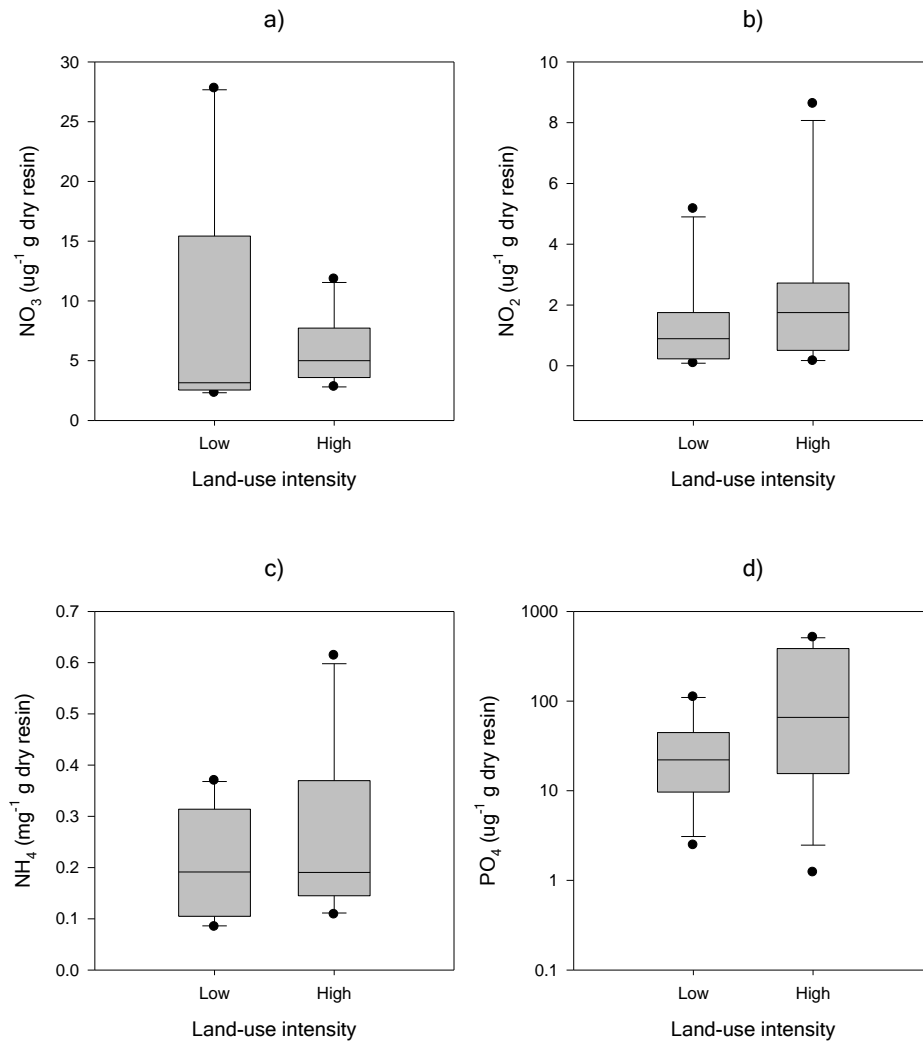
Figures

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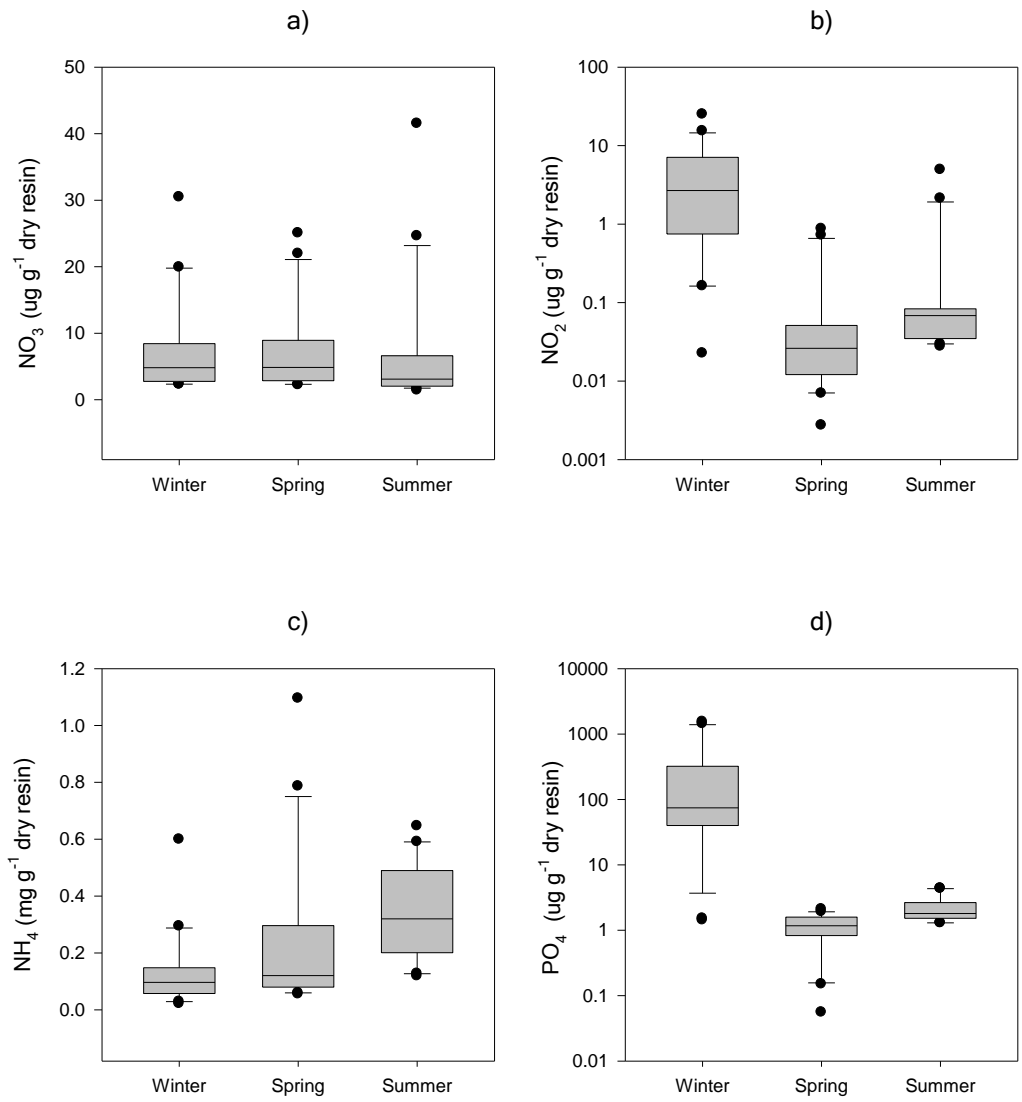
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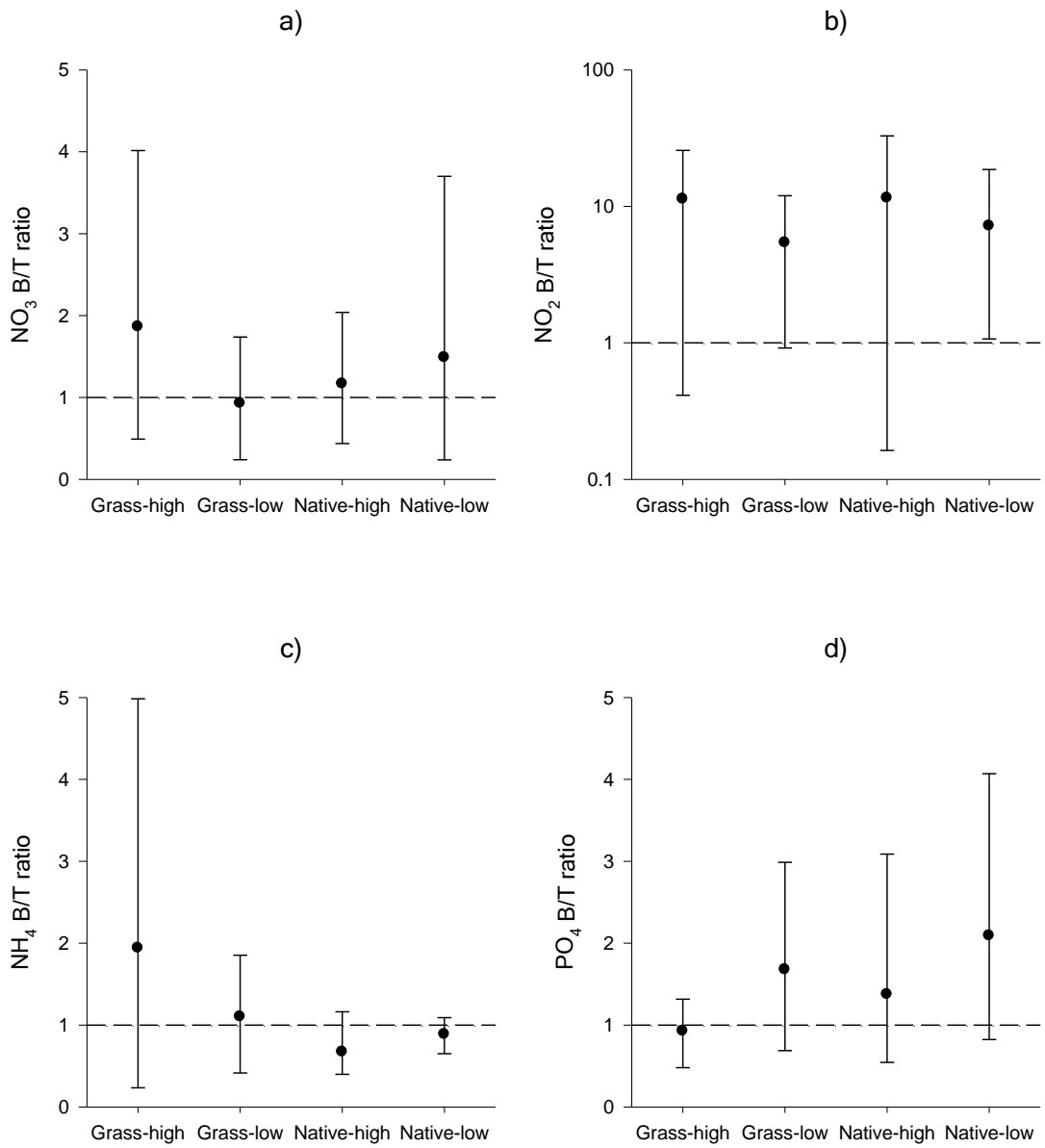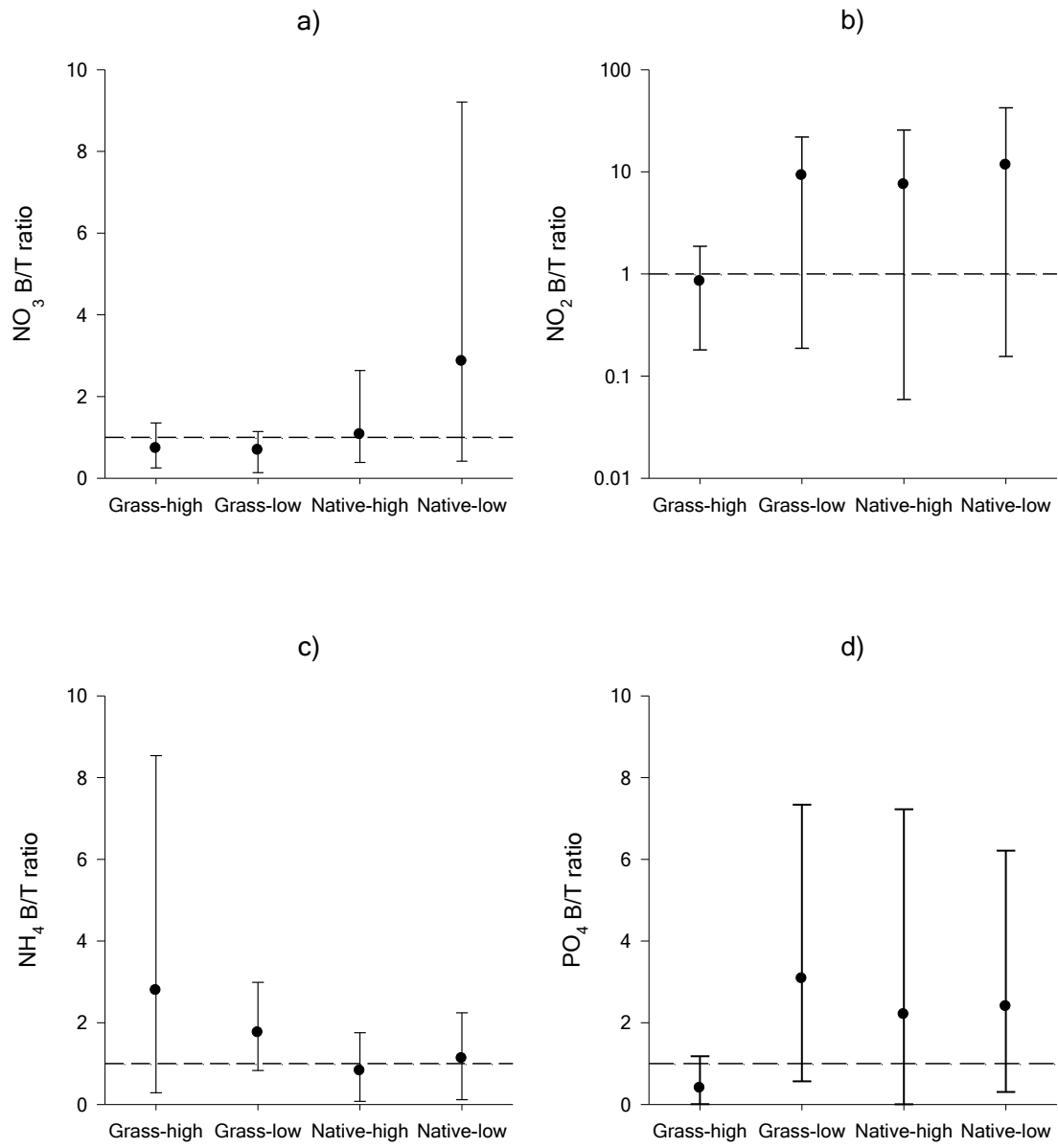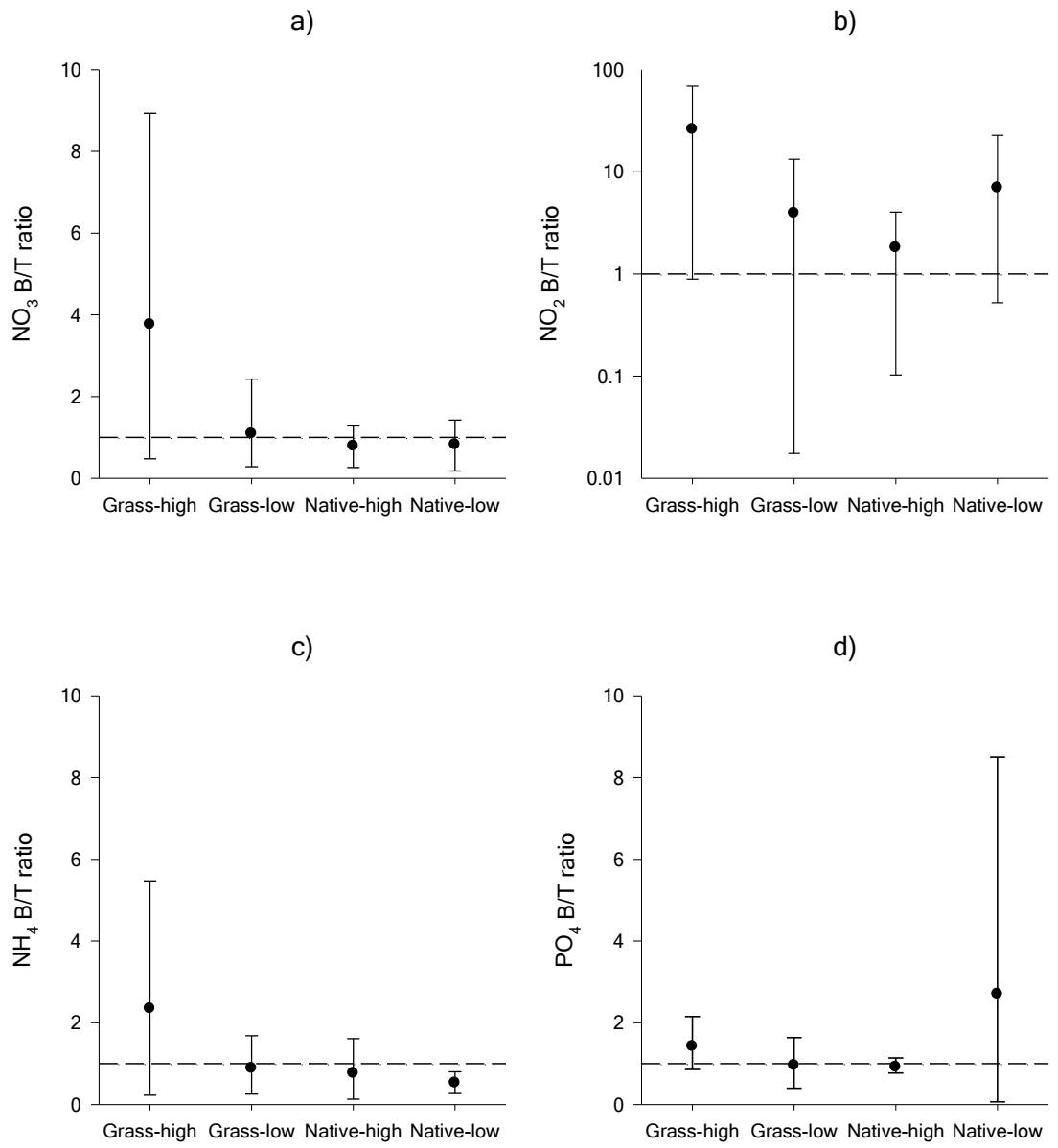
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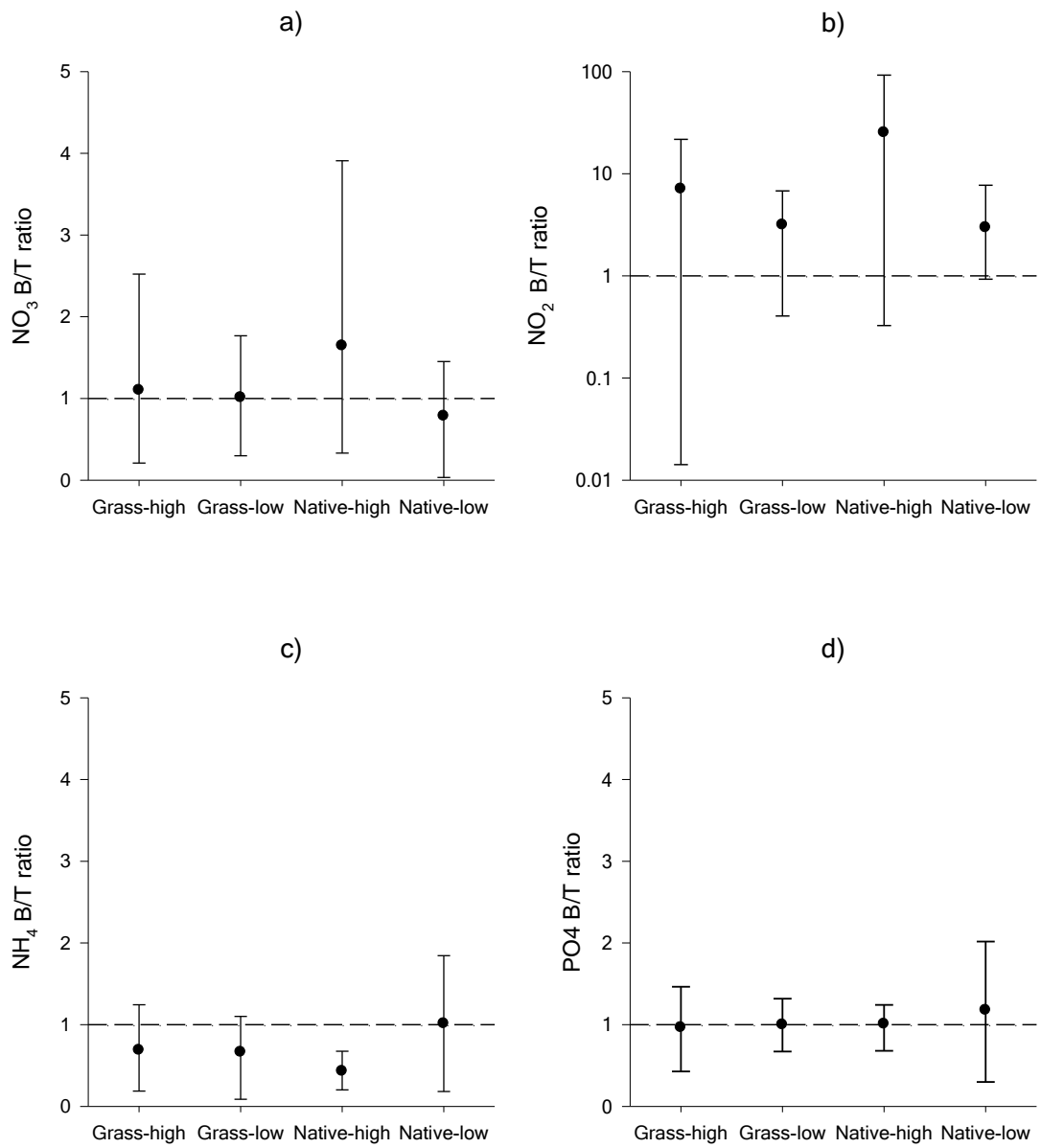
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Tables

Table1: Mean soil nutrient availability at field-edge/riparian boundaries adjacent to high (*n=10*) and low (*n=10*) intensity agricultural land-use activities on the Canterbury Plains. Mean nutrient values are µg nutrient g^{-1} dry resin (d.f. 1,18).

Nutrient	Low intensity mean	SE	High intensity mean	SE	F-stat	P-value
NO_3	8.68	0.29	5.88	1.02	0.834	0.373
NO_2	1.31	0.25	2.21	0.15	1.281	0.272
NH_4	206.79	16.20	263.29	10.39	0.790	0.385
PO_4	34.49	20.19	165.24	3.86	2.203	0.155

Table 2: Seasonal effects on nutrient availability at field-edge/riparian boundaries of 1st - 2nd order agricultural waterways on the Canterbury Plains. Values are median concentrations of resin-adsorbed, KCL-extractable nutrients (µg nutrient g^{-1} dry resin), for 20 sites.

Nutrient	Winter	Spring	Summer	Chi Sq	d.f.	P-value
NO_3	4.82	4.86	3.09	3.84	2	0.14
NO_2	2.68	0.03	0.07	32.86	2	<0.001
NH_4	100	120	320	18.91	2	<0.001
PO_4	74.5	1.2	1.8	3.77	2	<0.001

Table 3. Seasonal effects on soil nutrient availability at field-edge/riparian boundaries of waterways adjacent to high and low intensity agricultural land-use activities (*n=10* each) on the Canterbury Plains. Range and median values are KCL-extractable nutrient concentrations (μg nutrient g^{-1} dry resin).

Season	Land-use Int.		NO_3	NO_2	NH_4	PO_4
Winter	High	Range	3.5 - 30	0.35 - 25	22 - 600	38 -1535
		Median	7.14	4.05	93	194
	Low	Range	2.3 - 12	0.16 - 15.3	29 - 294	1.5 - 328
		Median	4.03	2.6	78	62
		P value	0.6	0.6	0.36	0.13
Spring	High	Range	2.8 - 13.4	0.003 - 0.87	59 - 1095	0.56 - 2.09
		Median	7	0.03	109	1.3
	Low	Range	2.2 - 25	0.008 - 0.7	55 - 427	0.056 - 1.9
		Median	3.5	0.03	143	0.99
		P value	0.54	0.36	0.97	0.07
Summer	High	Range	1.7 - 10.5	0.028 - 7.5	119 - 647	1.33 - 4.34
		Median	4.6	0.07	320	1.8
	Low	Range	1.4 - 41	0.03 - 0.09	127 - 588	1.3 - 4.35
		Median	2.4	0.07	305	1.8
		P value	0.7	0.88	0.97	0.94
Overall	High	Median	5.2	0.07	154	1.86
	Low	Median	3.3	0.07	147	1.87
		P value	0.399	0.819	0.579	0.442

The effect of agricultural nutrients on the growth of three native riparian tree species

Abstract

Planting riparian buffer zones in agricultural landscapes is a widely accepted 'best management' practice for reducing nutrient and sediment inputs to farm waterways. Use of native species in such plantings is widely supported in New Zealand, yet little is known of their ability to act in a buffering capacity within riparian margins, by uptake and storage of nitrogen and phosphorus within biomass. To assess whether uptake of agriculturally-derived nutrients was reflected in the growth rate of adjacent native plants in riparian buffers, I first measured nitrogen (N) and phosphorus (P) concentrations and soil moisture status at riparian/agricultural boundaries adjacent to 15 waterways on the Canterbury Plains (seven with native vegetation and eight with grass). Accumulated growth of three native trees commonly found in such established (over 7 years old) plantings *Cordyline australis* (cabbage tree), *Plagianthus regius* (ribbonwood) and *Pittosporum tenuifolium* (kohuhu), was then compared with predicted growth by age, for each species. Accumulated grass-sward biomass was measured within eight grass-only buffers. Soil nutrient concentrations (g-1 dry resin) ranged from 9–512 µg for phosphate–P, 2.5–26.7 µg for nitrate–N and 87–623 mg for total nitrogen, of which 95% was ammonium–N. Of the three tree species only *C. australis* showed a significant growth response to nutrient resources and this was for nitrate–N. All three native trees and grass showed a significant response to soil moisture status. The near-surface root structure of *C. australis* may confer an advantage over other species in resource acquisition, resulting in more effective nitrate–N removal within shallow riparian soils.

Introduction

The loss of native riparian vegetation and its replacement with non-native species is a worldwide phenomenon (Hladyz et al. 2011). In New Zealand, European settlement and establishment of farming on the Canterbury Plains was accompanied by removal of indigenous vegetation along stream margins and introduction of exotic trees such as willow (*Salix spp.*) and poplar (*Populus spp.*) for functional roles such as bank stabilisation, erosion control and flood protection (Phillips and Daly 2008a). Native riparian vegetation was considered 'difficult to establish' and 'slow-growing' (Marden et al. 2007), leaving its potential to fulfil these roles unexplored, and the species under-valued.

Currently, the majority of riparian buffers along small farm waterways in lowland Canterbury consist of fenced-off pasture grass or, single rows of exotic trees or shrubs (see Chap. 2). However, studies have shown that these provide only limited ability to restrict a range of agricultural pollutants from entering waterways. For example, grass strips are known to be effective at capturing particulate-bound phosphorous in surface run-off due to high stalk density, however removal of nutrients in solution (e.g., nitrate) is not as effective (Dillaha et al. 1988, Cooper et al. 1995). Conversely, single rows of trees or shrubs may retain some of the soluble nutrient load within the soil profile following infiltration, but fail to capture surface run-off (McKergow et al. 2006a, McKergow et al. 2006b). Multi-tiered and multi-species buffers that include grasses, shrubs and trees can reportedly achieve greater nutrient retention as well as many other objectives of riparian environments by ensuring structural and habitat complexity both above- and below-ground (Lee et al. 2000). Variation in canopy structure, ground-cover roughness and rooting depth, density and complexity (Schultz et al. 1995, Marden et al. 2007) enables higher infiltration, and greater nutrient-attenuation from overland flow and shallow ground-waters, whilst also providing bank stabilisation and erosion control, shade, detrital inputs and other stream-habitat requirements (Quinn et al. 2001, Parkyn 2004, Czernin and Phillips 2005, Dosskey et al. 2010).

A resurgence of interest in planting indigenous vegetation, for both aesthetic and bio-diversity value, by local community groups and land management authorities throughout New Zealand has seen widespread planting of native grasses, flaxes and early coloniser, small trees and shrubs along-side waterways (Davis and Meurk 2001a, Ledgard and Henley 2009). This has prompted investigation into their potential to fulfil functional roles necessary for waterway protection, such as providing river-bank stability against erosion and flooding

(Phillips and Marden 2006b, Phillips et al. 2009, Phillips et al. 2011). Trials assessing the performance of several native riparian species have recently been conducted to determine species survival (Bergin 2008, Phillips et al. 2009), canopy coverage (Bergin 2008), above- and below-ground growth and biomass production (Marden et al. 2007, Bergin and Kimbeley 2011), and root tensile strength and site occupancy (Czernin and Phillips 2005, Phillips et al. 2011).

In these trials plant survival was primarily determined by species-site suitability, with non-woody species more resilient in hostile environments (e.g., flood-zones), which significantly reduced survival of woody species (Phillips et al. 2009). Appropriate weed management and pest control significantly enhanced survival, growth and establishment of young native plants in riparian areas (Ledgard and Henley 2009, Smaill et al. 2011). Once established, plant growth-rate and structure can differ with site situation and be related to light availability, planting density (Wang et al. 2012) soil type, disturbance frequency, weed competition (Smaill et al. 2011) and animal browsing or, latitudinal and climatic restraints such as temperature, light and rainfall (Harris et al. 1998, Harris et al. 2003). The endemic monocot cabbage tree, and two native angiosperms, ribbonwood and kohuhu, have consistently out-performed most other species in survival and biomass growth (Phillips et al. 2009, Phillips et al. 2011, Smaill et al. 2011).

A plants above- and below-ground biomass and structure represents its ability to compete for, capture and use available resources such as light, water and nutrients and, incremental growth should reflect this over time. Nutrient uptake and sequestration is correlated strongly with biomass production (Dosskey et al. 2010) and studies have shown increased growth-rates of riparian vegetation resulting from assimilation of agriculturally-derived nutrients (Tufekcioglu et al. 2003). In the United States native and exotic riparian seedling trees showed significantly faster growth (height, leaf, stem and root biomass) in response to effluent treatments containing elevated concentrations of ammonium, nitrate, phosphate (Marler et al. 2001). Similarly, in Western Australia, growth of native *Banksia prionotes* was 3–12 times faster within 20 m of crop and pastoral land, than *B. prionotes* 100–300 m from crops (Grigg et al. 2000). Differences in growth rates were associated with nitrate-enriched ground water and aerial drift of fertiliser from adjacent land management and were most evident in young trees (< 15 yrs), declining with age to 25 years.

In order to determine growth response to nutrient enrichment, establishment of a 'normal' growth by age baseline is required. Plant growth (by age) under 'normal' and consistent conditions should fall along a sigmoid curve (representing growth rates of

seedling, juvenile and mature trees), and departures from it may correspond with higher or lower resource availability. Unfortunately, with few exceptions quantitative data for New Zealand native riparian species does not extend beyond trees of 5–6 yrs old (Harris et al. 1998, Harris et al. 2003, Czernin and Phillips 2005, Marden et al. 2007, Bergin and Kimbeley 2011, Phillips et al. 2011, Smaill et al. 2011). Furthermore, while there are extensive and informative guidelines describing riparian management techniques for New Zealand stream environments and, expected outcomes of good riparian management (Davis and Meurk 2001a, ECAN 2005(a), Ledgard and Henley 2009), there has been little evaluation of the functional performance of different native species and, in particular, their capacity to remove agriculturally-derived nutrients.

In this study I assessed whether three native riparian trees showed differential growth response to resources derived from the adjacent agricultural land. I hypothesize that departures from 'normal' growth-rates for individual species, would be correlated with low and high riparian soil nutrient concentrations and/or soil moisture status. Additionally, in grass-only riparian margins, standing crop of pasture grass was measured to determine its response to agriculturally-derived resources.

Methods

Study area and site location

This research was conducted on the Canterbury Plains on the east coast of New Zealand's South Island (latitude 42–43°, longitude 171–173°). The climate is temperate, with mean annual rainfall ranging from 600-800 mm and mean annual temperatures from 10°–12.5° C (Sturman 2008). Soils range from deep silt loams in eastern lowland areas to stony sandy loams further westward. The agricultural landscape is dominated by cropping, dairying and sheep farming.

Riparian buffer zones of fifteen, 1st – 2nd order farm waterways were selected. Seven were planted riparian buffers containing between 6–10 native species. Five of these were adjacent to dairy grazing and two, adjacent to dry-stock grazing. Planting age (5–12 years) was provided by landowners and a standard two years added to reflect age of plants pre-planting-out. The remaining eight buffer zones contained exotic pasture grasses, mainly perennial ryegrasses (*Lolium multiflorum* & *L. perenne*), tall fescue (*Festuca arundinaceae*) and cocksfoot (*Dactylis glomerata*). Four buffers were adjacent to dairy-grazed pastures, three to dry-stock grazing and one to barley cropping.

Field methods

Ion-exchange resin bags, containing 5 grams (dry weight) of resin beads (Dowex^{TM} Marathon^{TM} MR-3 Mixed Ion Exchange Resin) were used to measure the soil nutrient status at each riparian buffer zone. At each location five resin bags were buried 1 metre apart, at 10cm depth within the riparian soil along a 4m transect parallel and adjacent to the agricultural/riparian boundary (field-edge) (Hart and Binkley 1985, Saggar et al. 1990). Fresh bags were installed for each of three sampling periods (April to July, 2011, July to October, 2011 and October, 2011 to January, 2012) approximating austral winter, spring and summer, and were left in the soil for an average of 92, 97 and 91 days respectively. On retrieval, all five resin bags at each site were bagged together as a composite 'site' sample, and returned to the laboratory for analysis (for further details see Chapter 3). Soil moisture (% saturation) was recorded at each of the five bag positions at each site (ThetaProbe ML2 and Moisture Meter type HH2, AT Delta-T Devices Ltd).

In January 2012, at the native-planted buffers, above-ground measurements were made of native tree and shrub species between the fence and waterway within each 4m lengthwise transect. These included basal stem diameter, crown diameter, canopy height and DBH (diameter at breast height), and depended on the plant growth form. Crown diameter was recorded as the average of the NS and WE dimensions and canopy height was measured using an extendable measure and trigonometry. Growth metrics for a total of 72 plants were collected. At the grass buffer sites a 25 cm^2 quadrat of grass was cut at approximately 10cm from the ground, bagged and returned to the laboratory for drying (60° C) and weighing.

Nutrient analysis

In the laboratory, each composite resin-bag sample was washed, extracted by shaking in 200ml 2M KCL and filtered. Then, resin-adsorbed nitrogen and phosphorus concentrations were analysed by colorimetry. An EasyChem® continuous flow auto-analyser (Systea, Italy) was used to analyse nitrate–N (NO_3), nitrite–N (NO_2) and phosphate–P (PO_4) concentrations and ammonium–N (NH_4) concentration was determined by the indophenol blue method (Solorzan.L 1969) followed by spectrophotometric analysis (Shimadzu, Kyoto, Japan). Chemically analysed nutrient values were converted to resin-available, KCL-extracted concentrations (µg nutrient g^{-1} dry resin*)* for descriptive and statistical analysis (for methods see Chapter 3).

Data Analysis

For each site, average soil PO_4, NO_2, NO_3 and NH_4 concentration was calculated as the mean of values obtained for the three sampling occasions. Total nitrogen (TN), was derived from the sum of NO_2, NO_3 & NH_4 concentrations. Datasets were log10 transformed where necessary to meet normality requirements for statistical analysis. Average soil moisture for each site was calculated as the mean, of the mean of the five soil percent-saturation readings, taken on each of 3 spot-recordings (July, October and January).

Published information on growth rates of native riparian vegetation is sparse, due to this and the low occurrence of each species between the seven native-planted buffers, analysis was limited to the three most commonly occurring and better-documented species. These were cabbage tree, ribbonwood and kohuhu, of which there were a total of 31 individual trees.

In order to calculate a predicted growth curve for each species previously published data by Czernin and Phillips, (2005) and Marden et al, (2007) was used to determine which,

of crown height, crown diameter and basal stem diameter, best-correlated growth with age for each species. For cabbage trees, data from a study located in the Canterbury region comprising 13 destructively sampled trees, aged 1–25 years was used (Czernin and Phillips 2005). Regression was used to convert DBH measurements recorded in my study to root collar dimensions for comparison with the Czernin & Phillips (2005) by the following equation:

$$Y = 1.6217(x) - 2.5391 \ (R2 = 0.9734)$$

An exponential relationship between age and root collar diameter was derived ($y = 0.8322$ $^{0.2688x}$, $R^2 = 0.9376$) from the Czernin and Phillips, (2005). To calculate growth by age equations for ribbonwood and kohuhu, my data was supplemented with data for young trees aged 1–5 years, ten of each species being destructively sampled each year (Marden et al. 2007). For ribbonwood, a 1st order, natural logarithm (ln) relationship between age and canopy height was derived for ribbonwood growth, from 5yr old plants (Marden et al. 2007) and data from this study (y = 1.4014 * ln(x) + 0.9402, $R^2 = 0.5365$). A 1st order, natural logarithm (ln) relationship between age and canopy height was derived for kohuhu (y = 1.3006 * ln(x) + 0.1771, R2 = 0.659) from data for 5yr old plants (Marden et al. 2007), and data for 13 trees, at 7 sites in this study. Deviations of individual trees from derived predicted-growth curves were established. ANOVA and linear models were used to correlate soil nutrient and moisture status at each site, with native tree growth (R, version 14.2.2) (Crawley 2005). Regressions were used to analyse grass biomass response to soil nutrient and soil moisture status.

Results

Analysis of resin-adsorbed nutrient concentrations showed that 93–98.5% of available soil nitrogen (TN) at the field-edge at the 15 streams was in the form of NH_4. TN ranged from 186–623 µg (median 349 µg) at planted tree sites and 87–313 µg (med. 185 µg) at grass sites. NO_3 concentrations were low (range 2.56–26.69 µg), exceeding 10 µg at only one planted tree site, and one grass site (Table 1). PO_4 levels were generally high, but variable ranging from 16–512 µg at native sites (med. 101) and 9–354 µg (med. 22) at grass sites. My earlier study indicated that 95% of PO_4 was transported to field-edges during winter (see Chap. 3 for details).

Mean soil moisture for each site over the 10-month period was relatively low, ranging from 7–37% saturation over all 15 sites, and from 17–23% saturation for the seven tree-planted sites. However, within each season site means ranged from 7–60% saturation. Soil moisture was only slightly correlated with nutrient levels, and only at grass sites for TN ($R^2 = 0.6872$) and NO_3 ($R^2 = 0.3182$).

Root collar diameter for 7–13 year old cabbage trees (*n=7*) ranged from 12.5–34.5 cm. Some plants varied widely from predicted growth, with plants at two sites showing markedly greater growth than expected for their age (Fig. 1a). Canopy heights of 7–12 year old ribbonwood trees (*n=11*) ranged from 2.6–5.7 m (Fig. 1b). A number of trees from several sites were taller than expected for their age. For kohuhu, canopy heights ranged from 2.0–4.3m (Fig. 1c).

Residuals analysis showed that growth of individual native trees (*n=31*) was significantly influenced by soil NO_3 ($F_{1,29} = 4.86$, p = 0.03) and soil moisture status ($F_{1,29} = 5.732$, p = 0.02) but not by TN or PO_4 (Table 2). NO_3 and soil moisture were both site-dependent (ANOVA, p < 0.0001), and NO_3 availability was dependent on soil moisture status ($F_{1,29} = 12.59$, p = 0.001). Analysis of mean residual values by species showed that only growth of cabbage trees (*n=7*) were significantly influenced by NO_3 availability ($F_{3,13} = 4.637$, p = 0.008). There were no statistically significant relationships between nutrients and ribbonwood (*n=11*) or kohuhu (*n=13*). However, all three species were significantly influenced by soil moisture levels ($F_{3,13} = 5.703$, p = 0.01) (Table 3).

Seasonal grass growth in grass filter strips varied considerably between sites, ranging from 0.5–1.8 metres in height. Biomass accumulation was significantly and positively correlated with soil moisture status ($F_{1,6} = 6.49$, p = 0.04), but not with PO_4 ($F_{1,6} = 0.007$, p = 0.938), NO_3 ($F_{1,6} = 2.18$, p = 0.19) or TN ($F_{1,6} = 2.371$, p = 0.17) (Fig. 2).

Discussion

Pre-human settlement, the Canterbury Plains was home to a wide range of native riparian species. These included rushes, sedges and flaxes, shrubs such as coprosmas, hebe and olearia, the monocot cabbage tree, a number of small hardwood trees such as ribbonwood, pittosporums, kowhai, manuka and broadleaf, as well as the longer-lived floodplain species hinau, matai and kahikatea (Meurk 2008). During 150 years of intensifying farming practices the riparian areas of Canterbury streams have been extensively modified, with native species removed and replaced with exotic trees, shrubs and pasture grasses or left unprotected from livestock access, resulting in highly degraded waterways (Phillips and Daly 2008a, ECAN 2010). Over the past three decades many of these native species are being re-planted as part of riparian restoration schemes aimed at improving stream health and riparian function (Collins et al. 2013). Regional land and water managers also recommend planting native vegetation along riparian margins to farmers, to reduce and prevent nutrient inputs to adjacent waterways, especially in at-risk lowland regions (MfE 2000, ECAN 2005(a), 2005(b)). In my survey of riparian areas in the Canterbury agricultural region, I found that although there was evidence of a wide range of newly-planted native species (less than 2 years old), there was a relatively limited number of these in established riparian buffers of 5–13 years old. Among the most common established riparian trees were cabbage trees (*Cordyline australis*), ribbonwood (*Plagianthus regius*) and kohuhu (*Pittosporum tenuifolium*).

The monocot cabbage tree is a found naturally along swampy lake and river margin environments from sea level to 1000 metres above sea level (Czernin and Phillips 2005). It can reach up to 17m in height and has a thick fibrous trunk, spear-shaped leaves and a unique rhizome and root structure (Poole and Adams 1994b). Rapid growth of the rhizome greatly increases the opportunity for new roots to form, and helps anchor the plant and store food and water. A young adult tree has a multitude of spaghetti-like roots, each several mm in diameter, radiating like spokes from the root collar at surface level (Czernin and Phillips 2005). By contrast, ribbonwood and kohuhu grow a typical tap-root structure. Ribbonwood has a divaricate juvenile stage while developing a single sturdy main stem which then supports a partially deciduous, leafy canopy which may reach 17m in height (Allan 1982, Poole and Adams 1994b). Young kohuhu trees often develop several main stems from 5–10cm above the ground becoming bush-like with a considerable proportion of biomass comprising leafy shoots. Kohuhu may reach 10m in height (Allan 1982, Poole and Adams

1994b). Both hardwoods are adapted to a range of environments but can be intolerant of hard frosts and waterlogging (Marden et al. 2007).

Plant uptake is the primary process by which excess agricultural nutrients may be removed by riparian vegetation (Tufekcioglu et al. 2003, Sovik and Syversen 2008, Balestrini et al. 2011). This could imply that faster plant growth will result in higher N and P removal rates (Wang et al. 2012) and therefore, the capacity of individual species to assimilate and store nutrients in biomass, is critical to buffer effectiveness. In this study I assessed the potential of three native trees (cabbage trees, ribbonwood and kohuhu) commonly found established within riparian buffers, to capture and retain agriculturally-derived N and P within biomass. As different species grow at different rates it was necessary to compare growth of study plants, under a gradient of 'known' soil nutrient levels, and predicted growth-for-age estimates based on an average for each species. I used resin-bag methodology to estimate soil nutrient availability at each of seven riparian buffers containing native species and published data was used to construct growth by age estimates for each species, and the growth of 31 individuals.

Soil nitrate (NO_3) concentrations in this study were found to be low in comparison to other N species. For example, NH_4 levels were over 20 times higher than those of NO_3 and comprised the main part of TN. This is consistent with free-draining soils and dry conditions and NH_4 can accumulate in surface soils adsorbed to clay particles (see Chapter 3 for details). Importantly, while some plants, and some conditions favour NH_4 uptake (Wang et al. 2012), higher mobility within the soil profile and a high diffusion gradient favours NO3 uptake and assimilation within plant tissue (Miller and Cramer 2005).

Interestingly, I found that of the three species assessed only cabbage tree growth was correlated with soil NO_3 status and, accumulated growth of each of the three species appeared to be unrelated to either soil TN or soil P status at 10cm depth. This most likely indicates that neither nitrogen nor phosphorus were likely to be a limiting resource at these sites, as might be expected adjacent to agricultural activity. Furthermore, the majority of the individual riparian trees assessed were well above the expected growth curve. In a similar study of riparian buffers across five European countries, no relationship was found between above-ground biomass production and soil extractable NO_3 and NH_4 availability, the authors suggesting that plant growth was not N-limited at these sites (Hefting et al. 2005). My results indicated that soil moisture was the main influence on riparian tree growth. Soil water is considered one of the primary factors limiting plant growth and productivity (Chapin et al. 1987, Mooney et al. 1987) and in fact, the magnitude of effect for optimum growth of stem

height, leaf area, photosynthetic rate and biomass yield of plants has been shown to be water, followed by nitrogen, then phosphorus (Dong et al. 2011). Soil water also increases the availability of dissolved nutrients available for plant-uptake in inorganic ionic forms (NO_3, NH_4 & PO_4) which are more readily-assimilated within plant biomass. This result was surprising as the average field-edge soil moisture (taken from spot-samplings) for each site over the duration of the study was relatively low, with the overall difference between all, only 6% saturation. It would seem unlikely that this narrow margin would explain differences in tree growth, although at individual resin-bag positions recordings of between 6.5–71% saturation occurred over all sampling occasions. However, if soil moisture was a growth-limiting factor then it may also have influenced the biological conditions determining the predominant form of nitrogen available at each site (discussed in Chapter 3).

Root architectural structure determines a species ability to access to nutrients within the soil profile and may confer advantages in resource acquisition, over other species. The rhizome and root structure of the cabbage tree allows it to exploit and dominate a large area of soil around its base, which can be up to 5–10 metres for adult trees (Czernin and Phillips 2005). Marden et al (2007) found that mean below-ground biomass for five year old cabbage trees (2.7 kg) was considerably greater than that of ribbonwood (1.8 kg) and kohuhu (1.2 kg). It may also be possible that the bulk of root biomass of the 7–13 year old ribbonwood and kohuhu trees in this study would most likely be deeper within the soil profile than 10cm, where nutrient concentrations were measured. Thus, root biomass may contribute to cabbage trees potentially being able to more rapidly take advantage of temporary spatial variation in soil water and NO_3. However, in the Marden et al, (2007) study of five year old native species, mean root spread and rooting depths for the three species were similar (i.e., from 2.2–2.6 m spread and 0.3–0.4 m depth) and generally ribbonwood and kohuhu were found to have higher root site-occupancy than cabbage trees (Phillips et al. 2011).

A similar comparison of nutrient up-take capacity of a shallow-rooting fibrous-structured species, *Casaurina cunninghamiana* and *Eucalyptus camaldulensis*, a deeper tap-rooted species, was conducted in an Australian study. It was suggested that nutrient attenuation from surface run-off might be superior by the shallow-rooted, fibrous *C. cunninghamiana* than *E. camaldulensis,* whereas *E. camaldulensis* might more effectively remove nutrients from shallow ground-waters, due to its deeper, tap-root structure (Wang et al. 2012). However, it was concluded that these morphological differences did not significantly affect NO_3 –N removal from either shallow surfaces or deep groundwater flows at the young (3–6 yr old) developmental stage of these trees (Wang et al. 2012)

The cabbage trees in this study showed the highest variation between actual-growth and predicted-growth among the three species. At two of the native riparian buffers, cabbage tree root collar diameters were wider than predicted and, exceeded the average root collar diameter (15.6cm) of 10 year old cabbage trees in a survey conducted at 39 locations nationwide (Bergin and Kimbeley 2011). My predicted growth curve for this species was expected to provide a reasonably accurate estimation of growth by age, due to the wide age-range reported in previously published data (Czernin and Phillips 2005). While their study comprised only 13 trees in total, it was conducted near Christchurch, thus climatic conditions were similar to those of my study (Czernin and Phillips 2005). This was considered an important factor in another study of 29 populations of cabbage trees throughout New Zealand (Harris et al. 1998), as it was found that southern trees had stouter trunks and lower tree-height to trunk-diameter ratios with increasing latitude or altitude and, correspondingly lower temperatures. This is believed to be a general adaptation among plants involving seasonal synchronisation of growth patterns in response to photoperiod and temperature stimuli, to avoid exposure of soft, actively growing tissue to extreme climatic conditions (Harris et al. 1998, Harris et al. 2003).

The fact that ribbonwood and kohuhu growth appeared not to be influenced by nutrient availability may possibly be due to uncertainty around the reliability of the predicted-growth curve for these species. While several recent studies have assessed aspects of the performance of these two species (Phillips et al. 2011, Smaill et al. 2011), quantitative data for above-ground growth by age for ribbonwood and kohuhu was only available for juvenile trees aged 1–5 years old (Marden et al. 2007). However, tree growth-rate during years 1–5 does not reliably predict expected growth of trees older than five years, as the trajectory of growth and length of time at each growth-stage varies with species, growth conditions and, during plant lifespan. Growth of plants in pots (yrs 1–2) typically represents a lag stage, while roots are constricted. Once planted out growth accelerates exponentially before becoming linear. For New Zealand native shrub/small trees the growth-rate decrease occurs around 10–20 years, and growth then begins to plateau towards the end of their relatively short life-span (~40yrs).

Furthermore, the former study was conducted in Gisborne, North Island (38° S) under 'ideal' conditions in (Marden et al. 2007). Seedlings were locally-sourced, 2 year old trees planted in evenly-spaced blocks in a protected riparian environment and, irrigated for the first 3 months. Thus, differences other than nutrient resource availability, such as latitudinal

differences in climate (e.g., temperature and rainfall) and environmental differences (e.g., soil type, shelter) would contribute to differences in growth conditions between these and the Canterbury sites.

Although root structural differences may have contributed to the significant influence of near-surface NO_3 on cabbage tree, but not hardwood growth in this study, neither soil NO_3 or moisture, at 10cm depth, were reliable predictors of growth-rates overall. I found some consistency among riparian buffer sites for either above- or below-predicted values of most species present, indicating that growth-rate may have been site-dependent. For example, at two sites growth of all species was higher than predicted and although these two shared the highest soil moisture (23% sat.) one site had only average NO_3 levels (7.1μg). At two other sites growth was consistently lower than expected, although one site had the second highest NO_3 (9.1μg) and soil moisture (22% saturation). Therefore it is probable that other site-related factors could have confounded nutrient/growth relationships between all three species. These may have included physical aspects of the buffer itself such as bank slope, preferential run-off flow-paths, hot-spots of soil microbial activity as well as plant competition for resources such as light (due to plant growth form or planting-out spacing), which can limit plant responses to below-ground resource heterogeneity (Jackson and Caldwell 1992).

Alternatively, superior adaptations to the generally harsh, open riparian environment of the Canterbury Plains may ensue a growth advantage to cabbage trees, which are a light demanding species that are adapted to a range of moisture conditions from dry to boggy environments (Czernin and Phillips 2005).

In the eight grass filter strips assessed in this study, I found considerable variation in grass height and biomass between sites by the conclusion of the study. Soil moisture was significantly correlated with grass biomass, and appeared the most dominant influence, followed by total nitrogen, on grass growth. Interestingly there was no relationship between grass growth and NO_3 status, indicating that NO_3 at 10cm depth did not influence grass growth within these stream-side margins. Although not part of the present study, the most apparent reason for this was observed to be the difference in adjacent land-use activity and its associated management (Chapter 3). Under winter/spring soil saturation conditions or, intensively managed, dairy-farming conditions (i.e. high stocking rate and effluent- irrigated), well-drained soils and lowland areas such as Canterbury, are prone to substantial nitrate leaching within the paddock, and sub-surface flow is often the dominant pathway for nitrate movement (Ledgard et al. 1999, Sabater et al. 2003, McKergow et al. 2006b). Thus field-edges and riparian buffers may not receive similar inflows of nitrate, as shallow ground-

waters. Although, under low soil moisture conditions and warmer temperatures, the top 10cm of the soil profile is more likely to provide a reasonable indication of overall nutrient availability, due to high rates of plant and microbial activity (Sovik and Syversen 2008).

There are several important caveats to these findings that limit interpretation of nutrient-related, tree-growth data. In particular, the scarcity of established riparian buffers in Canterbury that contain similar species and multiple individuals of each and, of known and, in graduated age groups, gives this study only low statistical power. Additionally, the very different growth forms of the three species meant that inconsistency in growth metric measured was unavoidable, as a completely accurate measurement of biomass growth over tree life was not possible without destructive sampling. Furthermore, estimation of N and P availability was assessed during only a short (10 month) period of the life of the trees, at 10cm depth, and along the field-edge only, which may not accurately indicate resource availability throughout the riparian soil profile or in sub-soils and shallow ground-waters, where many riparian tree species are likely to obtain additional resources. Similarly soil moisture status was assessed on three occasions by spot-sampling.

The results of this study indicate that it is probable that growth of riparian species adjacent to agricultural fields is not nutrient-limited. However, the root structure of cabbage trees may allow it to take advantage of nutrients in surface soils that might be unavailable to other species. While my study failed to find a significant relationship between hardwood growth and nutrient levels in shallow riparian soils, it does not suggest that ribbonwood and kohuhu are ineffective in riparian buffers. In fact, with age and increased rooting depths, and in combination with native grasses and shrubs, all three species should play a role in filtering and removing agricultural nutrients at a range of depths within the soil profile and, from shallow ground-waters (MfE 2000). The hardwoods also provide benefits in addition to their role as nutrient filters. For example *Plagianthus regius* is partially deciduous, therefore contributing detrital inputs to stream food-webs. *Pittosporum* species consistently out-perform other species in lateral and vertical biomass growth, and rapidly develop canopy-cover within riparian margins, providing shade for in-stream temperature regulation. It might also be suggested that cabbage trees, ribbonwood and kohuhu are more frequently observed in established, native riparian buffers in Canterbury due to greater tolerance of environmental conditions that can include strong winds, drought and frosts. Information on New Zealand's smaller trees and shrubs that include those typically used in riparian plantings is sparse and there is a need for baseline data on growth by age for these species. There remains a wide gap in our knowledge regarding the ability of native riparian plants to remove agricultural

nutrients such as nitrogen and phosphorus from soils. Further research should include investigation of the nutrient uptake capacity of all native riparian species, and their ability to take advantage of nutrient spikes in space and time. This will add to the relatively sparse information on growth of native riparian species, as well as determining how they might function in providing a range of functional and ecosystem services necessary to enhancing waterway and biodiversity protection.

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Figure Legends

Figure 1: Growth by age of three New Zealand native riparian species in planted riparian buffers adjacent to pastoral land-use in lowland Canterbury, New Zealand **a)** *C. australis* (*n=7*), **b)** *P. regius* (*n=11*) and, **c)** *P. tenuifolium* (*n=13*) where in a, (x) represents data derived from Czernin & Phillips, (2005), in b & c (x) represents data derived from Marden (2007) and (●) represents data from this study.

Figure 2: Grass biomass accumulation in grass riparian buffers (*n=8*) over 10 months, as influenced by **a)** NO_3 –N, **b)** TN, **c)** PO_4 –P and **d)** soil moisture (% saturation).
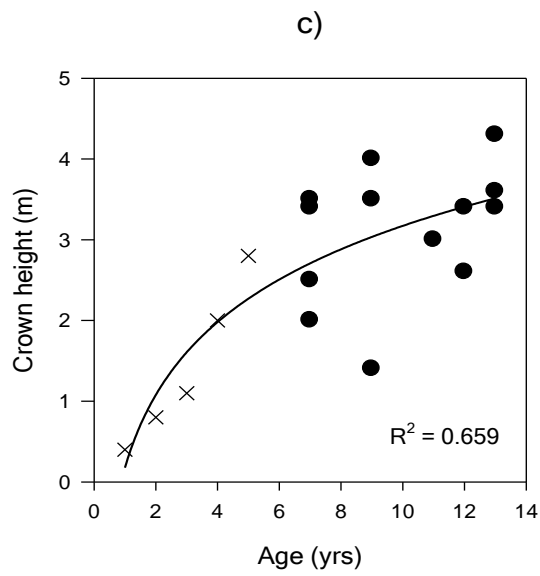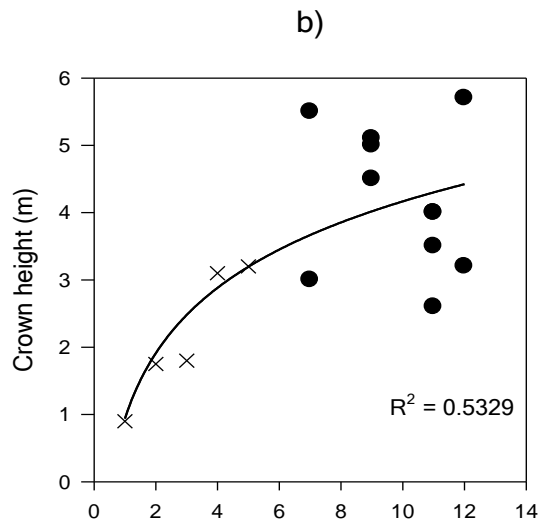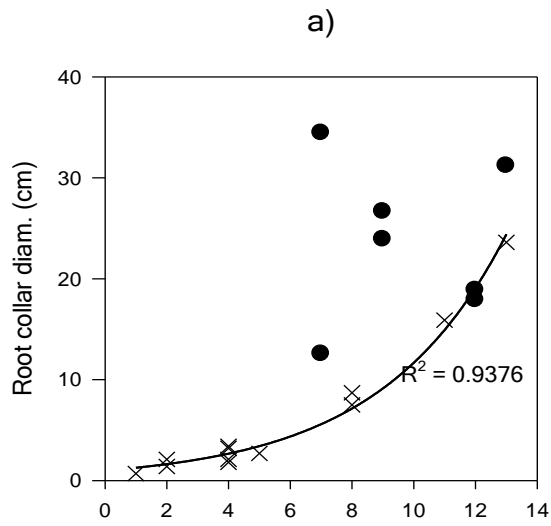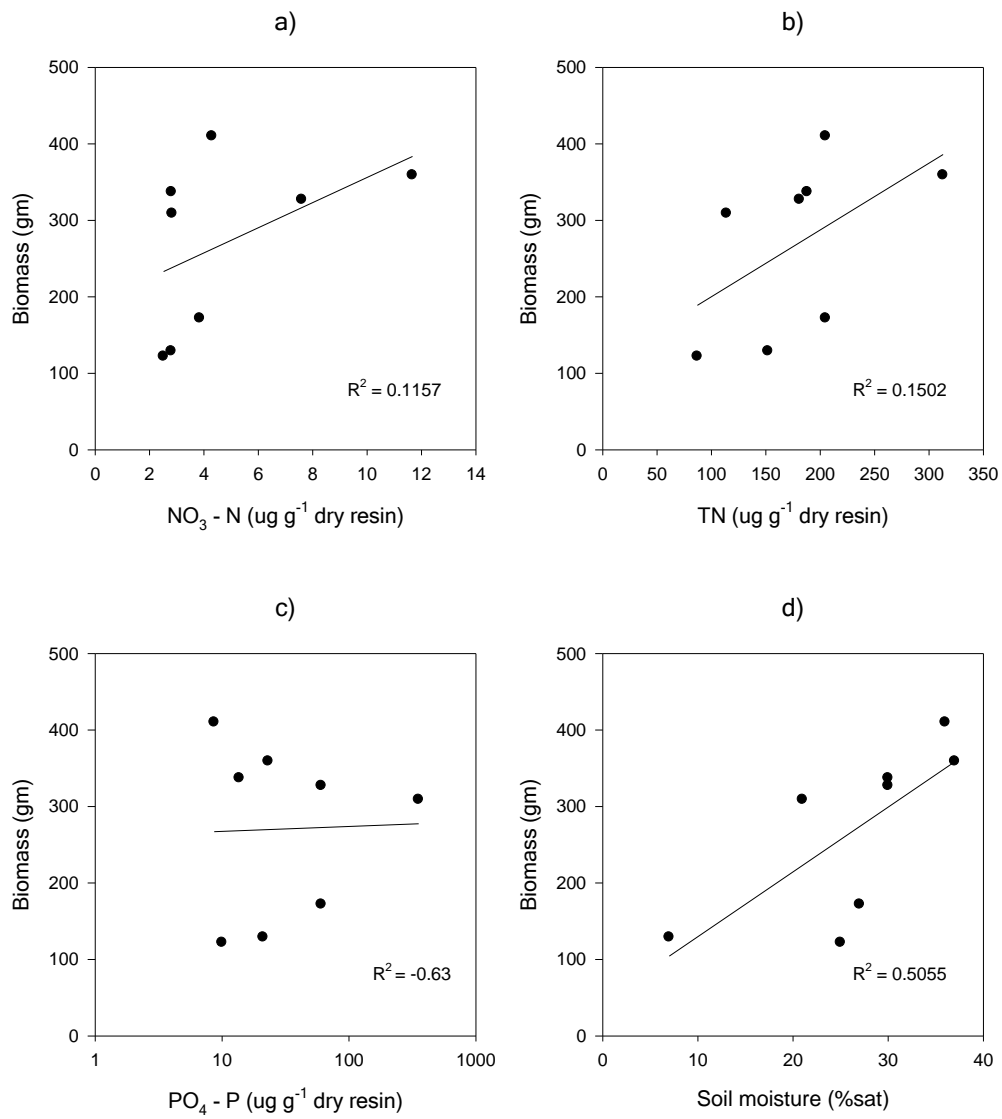
Figure 1

Figure 2

Tables

Table 1: Soil nutrient concentrations (µg g-1 dry resin) and soil moisture status (% saturation) for native-planted riparian buffers (*n=7*) and grass buffers (*n=8*) adjacent to agricultural land in Canterbury, South Island. Nutrient values represent the mean of 3, 90-day in-situ sampling periods. Soil percent-saturation values represent the mean of 3, spot-samples.

	Low	High	Median	Mean	SE
Native					
NO$_3$	2.56	26.69	7.12	8.96	3.06
TN	186	623	349	353	59.05
PO$_4$	16	512	101	137	64.26
SoilSat	17	23	27	20	0.89
Grass					
NO$_3$	2.52	11.67	3.00	4.80	1.14
TN	87	313	185	180	24.25
PO$_4$	9	354	22	69	41.43
SoilSat	7	37	29	27	3.33

Table 2: The influence of nutrients and soil moisture on individual growth (represented by residual values) of 31 native trees (*C. australis, P. regius* & *P. tenuifolium*) within seven planted riparian buffers of 5-13 yrs old in Canterbury, New Zealand. (*) indicates a significant relationship between resource and growth.

Individual tree growth (residuals)	F-Stat	d.f.	P value
NO$_3$	4.86	1,29	0.03 *
TN	1.9	1,29	0.18
PO$_4$	0.745	1,29	0.39
Soil Moisture (% Sat.)	5.732	1,29	0.02 *

Table 3: Influence of nutrient availability and soil moisture on the growth of each of three native tree species in planted riparian buffers (*n=7,* d.f. = 3,13), adjacent to agricultural land in Canterbury, South Island. Nutrient concentration values are resin-extracted µg nutrient g^{-1} dry resin. (*) indicates a significant difference in growth.

Species Nutrient	*C. australis* p value	*P. regius* p value	*P. tenuifolium* p value	All trees	
				F-Stat	P value
NO$_3$	0.008 **	0.3	0.35	4.637	0.02 *
TN	0.71	0.43	0.39	1.027	0.41
PO$_4$	0.49	0.72	0.66	1.75	0.21
Soil Moisture (% Sat.)	0.01 *	0.05 *	0.05 *	5.703	0.01 *

CHAPTER 5

General Discussion

Freshwater is a resource of global importance and yet its continued quality and availability is under serious threat by anthropogenic land-use activity. In New Zealand, agriculture constitutes a significant proportion economy, therefore sustainable farming practices must be realised to prevent further degradation of waterways. Lowland waterways are at the highest risk of severe and continued degradation due to both the intensity of agricultural activity carried out in lowland regions and downstream effects (Larned et al. 2004, Greenwood et al. 2012). Of particular concern are nitrate–N (NO_3) and phosphate–P (PO_4)**.**

Riparian management is a tool that can be used to mitigate water quality issues, and restoration, or rehabilitation, of riparian margins is a practical way to begin to achieve this goal. The effectiveness of using riparian vegetation as 'buffer zones' is supported by an extensive body of research and thus, is promoted as a farming 'best management practice' by local government agencies and land and water resource managers. This thesis set out to investigate the extent to which riparian management is being implemented as a water protection measure and, its effectiveness in retaining agriculturally-derived nutrients within the intensely-farmed, lowland region of the Canterbury Plains.

Extent of riparian management in Canterbury

The survey assessing the extent of riparian management alongside farm waterways found that riparian management appeared not to be a priority on the Plains, and varied in both its occurrence, and degree of implementation (Chapter 2). The majority of farm waterways were fenced off from stock access, but 65% of buffers were less than five metres wide, and most were dominated by grass and exotic weeds and shrubs. Buffers that included trees were generally associated with stock shelter or stream-bank stability, rather than nutrient removal and stream health. Active riparian management such as planting native trees, shrubs and grasses was rare. This may partly be due to farmers remaining cautious about implementing a potentially time-consuming and expensive riparian management strategy, with little or no quantitative evidence to support potentially better water-quality outcomes of native-vegetated buffers, compared with the known benefits of existing grass filter strips.

Exotic weeds and adventive trees and shrubs dominate riparian margins as opposed to native vegetation as a consequence of the extensive landscape modification that both horticultural and pastoral farming has bought to this region. However, the Canterbury region possesses a rich flora of native riparian species (Meurk 2008), and support is fast-growing for their re-establishment to achieve functional outcomes (Davis and Meurk 2001b, Phillips and Marden 2006b, Bergin 2008). This was evident from the number of 'new' native riparian plantings that were noted during the riparian survey, although there are many unmanaged riparian margins along which self-sustaining native vegetation could be established to improve riparian habitat, function and, stream health.

Nutrient dynamics in riparian buffers

In Chapter 3, I first set out to quantify nutrient (nitrogen and phosphorus) concentrations at riparian field-edges of ten high-intensity (dairy) and ten low-intensity (sheep or other dry-stock grazing) land-use activities to provide a measure of typical N and P concentrations that may be entering riparian zones from these activities. My key finding, for these sites, was that season is highly influential in determining soil N and P delivery to riparian margins. I also found that nutrient levels adjacent to dairy paddocks were not significantly higher or different to those adjacent to sheep farming activities, and both were equally variable.

Of particular interest, I found that soil nitrate levels were relatively low adjacent to high intensity dairying operations. While this result was surprising, put within the context of landscape topography, land-use activity and its effects on seasonally mediated soil conditions, it was not unexpected. In dairy-grazed pastures of lowland plains areas, the dominant pathway for NO_3 loss is via leaching within the paddock from beneath high density urine patches. Furthermore, sub-surface flow has been identified as the dominant pathway for NO_3 movement, and can be 5 to 10 times greater than surface run-off (Cooke and Dons 1988, Ledgard et al. 1998, Cameron and Di 2004). In accordance with other reports, my results suggest that in lowland Canterbury, under high intensity management, NO_3 most likely enters waterways via sub-surface flow, passing beneath shallow riparian soils, during the majority of the year.

Nutrient retention of grass and native buffers

My primary focus in Chapter 3 was to compare nutrient retention efficacies of rank pasture-grass buffers and buffers of mixed native vegetation. My key finding was that riparian buffers of mixed native vegetation was slightly more effective than grass at retaining a range of N

and P pollutants within relatively narrow riparian buffers, on the Canterbury Plains. This study is first time that New Zealand native vegetation has been evaluated in this role and nutrient retention quantified and as such, represents a positive outcome that supports planting native buffers for stream health protection.

Nutrient retention by each buffer type was found to be dependent on the chemical form of N or P measured, supporting overseas findings of better retention of particulate-associated nutrients by grass, and dissolved nutrients by herbaceous vegetation. Like field-edge availability, retention was driven by a significant seasonal component, which influenced physical and chemical aspects of the stream and soil environments and, (although not measured in this study), would also have influenced removal rates by mediating nutrient assimilation by plant and soil communities.

The literature suggests that infiltration *rate* and infiltration *depth* are critical to effective removal of nutrient pollutants therefore to optimise pollutant removal in narrow buffers of small farm waterways on the Canterbury Plains, the most effective riparian management may be to combine beneficial aspects of both grass strips and mixed native vegetation in two specific zones. For example, a 1-2 metre grass filter strip adjacent to the fence-line should provide adequate trapping of suspended particulate matter and increase infiltration rate of run-off. A further 3-5 metres (or greater) of mixed herbaceous vegetation, stream-side, would provide greater infiltration depth and spatial distribution of nutrients and thus, higher retention capacity, especially of nutrients in sub-surface flow. This type of zoned design has been previously promoted by Correll (2005) and Mander (2005), with Polyakov et al, (2005) stressing the importance of also taking into account site-scale topography.

Growth of native trees in riparian buffers

For the final component of this work (Chapter 4), I assessed accumulated growth of individuals of three native tree species common to riparian plantings, to determine whether growth was correlated with soil nutrient availability arising from adjacent agricultural activity. Evidence of a relationship between nutrient availability and tree growth could enable correlative comment on species' nutrient-uptake and storage capacity within riparian buffers. Accumulated growth for cabbage trees, ribbonwood and kohuhu varied widely around their 'expected' growth for age although this was generally not related to nutrient levels within shallow soils at riparian boundaries. Cabbage tree growth exhibited a relationship with NO_3-N availability, while the ribbonwood and kohuhu appeared unaffected by nutrient status, although soil moisture levels significantly influenced growth of all three species.

These results were not entirely unexpected as there are multiple influences on plant growth in any environment and at any single location that could cause variation in observed growth. For example, competition for space, light and other resources may confer a growth advantage or, slow growth of individual plants. In addition, each of the species used for this study exhibit vastly different growth forms, therefore correlation between soil nutrient concentration and any single growth metric for each species, may not have been sufficient to detect actual growth of individuals in response to a specific resource, among several limiting resources. Thus, ribbonwood and kohuhu should not be considered ineffective in riparian buffers based on growth in height alone. These species afford other valuable ecosystem services within riparian buffers such as stream-bank stability, provision of detrital inputs to stream food-webs, and shading, which translates to lowered in-stream temperatures and improvement of habitat. Furthermore, these trees are also self-seeding and self-sustaining and therefore should require minimum maintenance following planting-out and, their common occurrence in riparian margins is indicative of their tolerance for environmental conditions on the Canterbury Plains.

Limitations of current study

First and foremost, all components of this study were conducted within existing land-use and riparian environments, for which there is no substitute under controlled conditions. However, the inherent difficulty of conducting research in natural environments is that it often poses dilemmas when endeavouring to maintain replicability to fit a model. A lack of established riparian plantings within the Canterbury region meant that compromises in site characteristics were unavoidable when attempting to standardise aspects such as management regime, riparian slope, width and vegetative composition of native riparian buffers between sites.

Secondly, use of the resin-bag method has been widely accepted for similar investigations of plant-available nutrients. Although treatment methods in this study were consistent across all sites and thus, measured nutrient concentrations comparable, it is worth noting that,

> *"The results of the field experiments with resin bags must be interpreted within the context of the interacting factors regulating ion supply to the bags. First, competition with plants and soil microbes may strongly reduce ion supply to the resin bags. Second, increases in water flow to the resin bags may increase ammonium capture more than that of nitrate. Finally, the greater mobility of*

nitrate will accentuate the role of nitrification in determining resin bag estimates of N availability." Binkley (1984).

While in-situ, there was some invasion of resin bags by plant roots. In the laboratory, during their removal, minor rupturing of the bags was unavoidable and did cause some leakage of beads when cleaned and shaken, however this was kept to a minimum and adjusted for where necessary.

In this study, resin bags were placed at 10cm depth within the soil profile, therefore only accumulated soil nutrients at this depth were measured, it is acknowledged that this may not fully represent soil plant-available N and P status, especially at rooting depths more appropriate of tree species. Deploying additional resin-bags or using some other measure such as soil column samples deeper within the soil profile (i.e., 20cm & 30cm) could have been valuable. In my investigation of native tree growth, this would have provided an additional indication of the nutrient levels that these riparian trees established under. As my values for resin-sorbed nutrient concentrations, and spot-measurements of soil moisture, only represent a brief snap-shot of the conditions under which these trees established in their respective riparian margins.

Directions for future research

This thesis highlights several areas in which further research is required. First, it is apparent that ongoing research into a number of indicators of the growth performance of New Zealand's smaller native trees, especially those being used in riparian plantings is required. This should provide greater depth of baseline data to derive expected growth curves for trees beyond five years old. I suggest this might include glasshouse experiments, or in-situ isotopic studies, to determine nutrient uptake capacities and growth response of these plants when exposed to low-high nitrogen and phosphorus levels.

Secondly, further investigation of the various mechanisms involved in nutrient-riparian zone interactions, and at what depth these occur, needs to be carried out to understand how to reduce nutrient losses to waterways in flat landscapes. These should incorporate seasonal influences on nutrient form and transportation mode in the context of the land-use and associated management regimes.

Finally, how much of the success of riparian planting is dependent on site-specific factors? Polyakov et al, (2005), suggested that where riparian buffers fail to perform their protective functions it is often due to inadequate understanding of the conditions under which riparian buffers perform at their best, associated with low adaptability of their design to local field settings. This thesis supports the need for precision riparian buffering where placement, width and complexity may vary dependent upon the nutrient load and flow-path. Furthermore, the above-ground stature and below-ground dimensions of planted species at maturity must be considered when planning buffer zones to enable the most constructive use of buffer width. Future studies that take such factors into account when aiming to optimise nutrient retention in buffers of 5–10 metres wide are likely to be of particular interest to land managers farming on the Canterbury Plains and, within New Zealand.

Concluding remarks

The majority of riparian buffer zones on the Canterbury Plains are less than optimal in width and complexity to cope with a range of agricultural contaminants. Key findings of this study are that,

- there is considerable scope for improving riparian buffering of most farm waterways with attention to buffer width and composition
- riparian buffers of mixed native vegetation are a viable alternative to exotic weeds and shrubs and retention of a range of agricultural pollutants should be optimised when combined with an upslope grass filter strip
- factors that control nutrient transport such as topography and season, need to be considered in combination with land-use activity when planning riparian buffer zones

In order to create sustainable farming practices that do not harm the environment, continued effort and commitment must be put into providing solutions to the problem of improving and maintaining fresh water quality. Use of riparian buffer zones undoubtedly supports this effort.

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APPENDICES

Appendix 1: Additional Tables – Buffer Width (Chapter 2)

A) Riparian buffer-width distribution for 85 sites across 5 sub-regions and **B)**, riparian buffer width distribution across 4 agricultural land-use types of Canterbury, S.I., New Zealand (3 buffers over 25 m were omitted for greater data clarity and as land-use (other) not typical of the agricultural landscape).
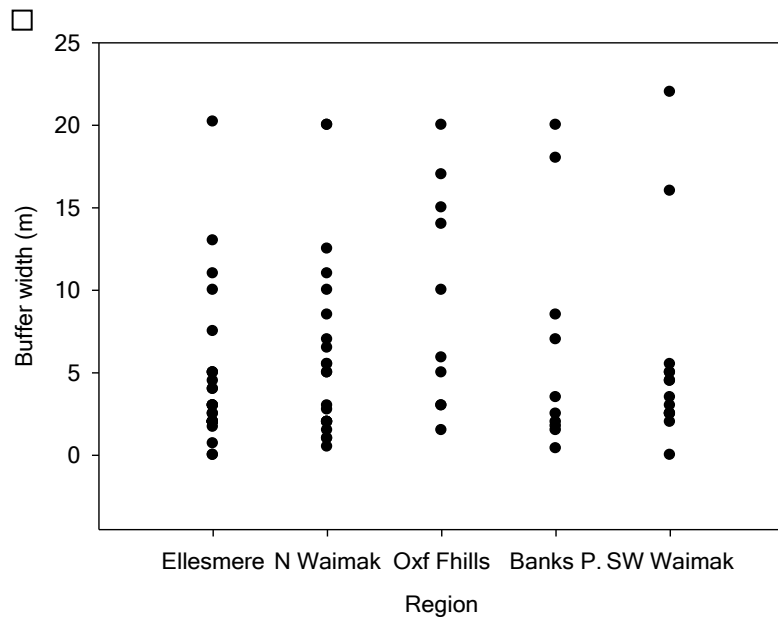
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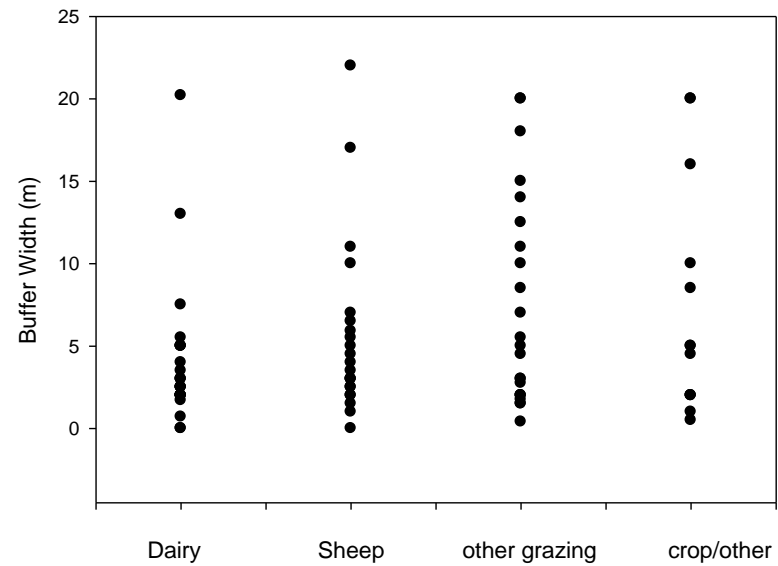
B

Appendix 2. List of vegetation within riparian margins

List of 61 vegetative taxa recorded within riparian margins of 88 small agricultural waterways of the Canterbury region of the South Island, New Zealand. Exotic taxa = X, Native taxa = N, Woody taxa = WO, Weedy taxa = WE.

Scientific name	Family	Name	# of spp.	Origin	Form	CSI
Salix spp.	Salicaceae	willow	2	X	WO	11.7
Populus sp.	Salicaceae	poplar	1	X	WO	2.5
Eucalyptus sp.	Myrtaceae	eucalypt	1	X	WO	3.6
Pinus radiata	Pinaceae	pine	1	X	WO	1.5
Cupressus macrocarpa	Cupressaceae	macrocarpa	1	X	WO	1.4
Sambucus nigra	Adoxaceae	elderberry	1	X	WO	3.0
Acer pseudoplatanus	Sapindaceae	sycamore	1	X	WO	1.7
Crataegus monogyna	Rosaceae	hawthorn	1	X	WO	1.1
Prunus sp.	Rosaceae	cherry	1	X	WO	0.3
Olea europaea	Oleaceae	olive	1	X	WO	0.1
Bambusa vulgaris	Poaceae	bamboo	1	X	WO	0.1
Ulex europaeus	Fabaceae	gorse	1	X	WO	5.3
Cytisus scoparius	Fabaceae	broom	1	X	WO	3.9
Lotus spp.	Fabaceae	lupin/lotus	3	X	WE	0.3
Rubus fruticosus	Rosaceae	blackberry	1	X	WE	2.4
Convolvulus arvensis	Convolvulaceae	convolvulous	1	X	WE	1.0
Solanum spp.	Solanaceae	nightshade	2	X	WE	0.2
Galium aparine	Rubiaceae	sticky weed	1	X	WE	0.6
Digitalis purpurea	Plantaginaceae	foxglove	1	X	WE	0.5
Oenthora stricta	Onagraceae	sand primrose	1	X	WE	0.2
Apiaceae family	Apiaceae	crt/par/celery/hemlk	4	X	WE	0.4
Carduus spp.	Asteraceae	thistle	5	X	WE	2.5
Achillea millefolium	Asteraceae	yarrow	1	X	WE	0.6

Scientific name	Family	Name	spp.	Origin	Form	CSI
Rumex spp.	Polygonaceae	dock	3	X	WE	1.1
Asteraceae family	Asteraceae	dandelion	6	X	WE	0.6
Plantago spp.	Plantaginaceae	plantain	2	X	WE	0.2
Linaria vulgaris	Plantaginaceae	toadflax	1	X	WE	0.1
Polygonum aviculare	Polygonaceae	wireweed	1	X	WE	0.1
Ranunculus spp.	Ranunculaceae	buttercup	5	X	WE	1.5
Trifolium spp.	Fabaceae	clover	4	X	WE	0.6
Cerastium spp.	Caryophyllaceae	chickweed	2	X	WE	0.1
Anagellis arvensis	Primulaceae	scarlet pimpernel	1	X	WE	0.1
Fumaria sp.	Papaveraceae	scrambling fumitory	1	X	WE	0.1
		Exotic pasture grasses	7	X	WE	38.1
Juncus & Gahnia spp.	Juncaceae	rushes & sedges	3	X	WE	1.7
Cortaderia richardii	Poaceae	toi toi	1	N	WE	1.7
Cordyline australis	Agavaceae	cabbage tree	1	N	WE	0.8
Phormium tenax	Agavaceae	flax	1	N	WE	3.5
Carex spp.	Cyperaceae	native grasses	3	N	WE	2.3
Paesia scaberula	Dennstaedtiaceae	bracken	1	N	WE	0.6
Polypodiales order	Pteridaceae	NZ ferns	2	N	WE	0.4
Podocarpus totara	Podocarpaceae	totara	1	N	WO	0.2
Dacrycarpus dacrydioides	Podocarpaceae	kahikatea	1	N	WO	0.1
Prumnopitys taxifolia	Podocarpaceae	matai	1	N	WO	0.2
Kunzea ericoides	Myrtaceae	kanuka	1	N	WO	1.8
Sophora microphylla	Fabaceae	kowhai	1	N	WO	0.3
Nothofagus spp.	Nothofagaceae	beech	1	N	WO	0.8
Dodonaea viscosa	Sapindaceae	akeake	1	N	WO	0.2
Plagianthus regius	Malcaceae	ribbonwood	1	N	WO	1.0
Pittosporum eugenioides	Pittosporaceae	lemonwood	1	N	WO	0.6

Scientific name	Family	Name	spp.	Origin	Form	CSI
Pittosporum tenuifolium	Pittosporaceae	kohuhu	1	N	WO	0.4
Pseudopanax crassifolius	Apiaceae	lancewood	1	N	WO	0.2
Pseudopanax arboreus	Apiaceae	5-finger	1	N	WO	0.1
Griselinia littoralis	Cornaceae	broadleaf	1	N	WO	0.4
Olearia paniculata	Asteraceae	akiraho	1	N	WO	0.9
Coprosma propinqua	Rubiaceae	small-leaved coprosma	1	N	WO	0.5
Hebe spp.	Scrophulariaceae	hebe spp.	3	N	WO	0.5
Discaria toumatou	Rhamnaceae	matagouri	1	N	WO	0.1
Muehlenbeckia australis	Polygonaceae	muehlenbekia	1	N	WE	1.2
Persicaria decipiens	Polygonaceae	swamp weed willow	1	N	WE	0.1
Urtica ferox	Urticaceae	native nettle	1	N	WE	0.1

Appendix 3: Graph – Vegetative Taxa by Cover Score Index

Percentage-cover derived NMRC scores depicting the spatial scale of dominance of individual taxa in riparian buffers of 88 waterways across the agricultural landscape of Canterbury, South Island, New Zealand.
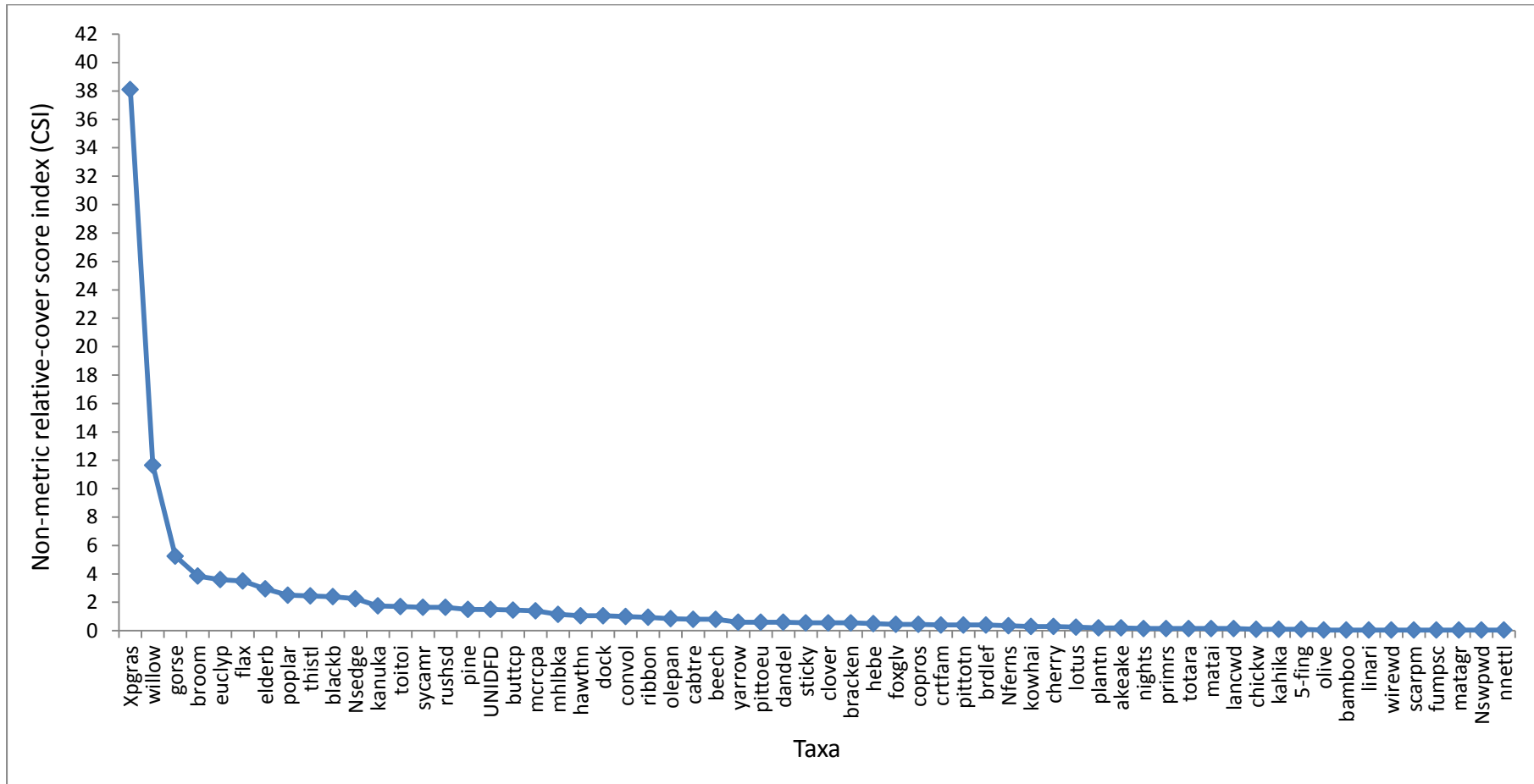
Appendix 4: Map of site locations on the Canterbury Plains

Google map showing the location of the 20 riparian buffers used to investigate nutrient dynamics (Chapter 3).

Appendix 5: Photographs of Riparian Buffers of New Zealand Native Vegetation

A selection of 4 of the native-planted buffers of varying ages used for nutrient budget study (Chapter 3).

Appendix 6: Photographs of Grass Buffers

A selection of 4 grass buffer zones used in nutrient budget study (Chapter 3). Note variability of condition of adjacent land between high-intensity (left), and low-intensity (right), land-use

Appendix 7: Riparian Buffer Retention.

Nutrient retention (%Ret = (T − B) / T) for each buffer by season, within riparian type (native/grass) and land-use intensity (high/low)

			Winter				Spring				Summer			
			NO3	NO2	NH4	PO4	NO3	NO2	NH4	PO4	NO3	NO2	NH4	PO4
KR01	Native	High	-231%	-3253%	-82%	-836%	-28%	8%	-81%	9%	-43%	22%	68%	-12%
KR04	Native	High	54%	65%	92%	100%	-2%	-18%	-4%	19%	13%	-12099%	80%	-10%
KR06	Native	High	56%	94%	70%	82%	46%	90%	46%	23%	67%	67%	64%	32%
KR08	Native	High	26%	-8%	-40%	-43%	17%	-107%	71%	-22%	25%	-157%	34%	-1%
KR13	Native	High	61%	-140%	49%	98%	74%	-375%	87%	13%	67%	-13%	39%	39%
KR10	Native	Low	50%	-5496%	-38%	-46%	82%	48%	24%	-1003%	97%	-872%	51%	14%
KR11	Native	Low	22%	78%	17%	-680%	-18%	34%	34%	0%	-49%	-133%	-76%	70%
KR14	Native	Low	58%	84%	23%	69%	54%	-2868%	37%	37%	6%	4%	-10%	-98%
KR16	Native	Low	41%	83%	88%	32%	3%	-194%	69%	23%	8%	7%	-53%	-2%
KR17	Native	Low	-1100%	-54%	-153%	-74%	-32%	1%	73%	93%	46%	7%	82%	-72%
KR03	Grass	High	35%	53%	-100%	94%	52%	-777%	23%	-51%	46%	-50%	27%	37%
KR05	Grass	High	-51%	67%	63%	-49%	-679%	11%	-211%	14%	-1%	99%	24%	-25%
KR07	Grass	High	75%	-120%	-997%	99%	-781%	-8549%	-781%	-136%	-209%	-13%	-37%	-23%
KR12	Grass	High	12%	-4%	71%	59%	19%	-2017%	77%	-38%	34%	-388%	81%	-29%
KR18	Grass	High	65%	82%	68%	97%	9%	-1175%	-20%	1%	79%	-2694%	60%	57%
KR02	Grass	Low	47%	-1523%	10%	8%	53%	-1639%	74%	-53%	27%	-548%	91%	8%
KR09	Grass	Low	86%	-356%	17%	-800%	72%	98%	50%	60%	70%	-124%	14%	6%
KR15	Grass	Low	25%	-2255%	-222%	-178%	-7%	37%	-88%	-46%	-4%	43%	25%	33%
KR19	Grass	Low	1%	-35%	-120%	-112%	32%	0%	-12%	1%	-1%	60%	-1%	-37%
KR20	Grass	Low	-3%	81%	-62%	43%	-194%	38%	33%	59%	-96%	-504%	40%	-9%
median	Native	High	54%	-8%	49%	82%	17%	-18%	46%	13%	25%	-13%	64%	-1%
values	Native	Low	41%	78%	17%	-46%	3%	1%	37%	23%	8%	4%	-10%	-2%
(n = 5)	Grass	High	35%	53%	63%	94%	9%	-1175%	-20%	-38%	34%	-50%	27%	-23%
	Grass	Low	25%	-356%	-62%	-112%	32%	37%	33%	1%	-1%	-124%	25%	6%
(n = 10)	Native		45%	29%	20%	-5%	10%	-9%	41%	16%	19%	-5%	45%	-2%
	Grass		30%	-19%	-26%	26%	14%	-389%	5%	-18%	13%	-87%	26%	-2%

Appendix 8: Riparian Buffer Retention Efficacy

Retention efficacy of each buffer by season (Eff = (ln(T) – ln(B)) lm^{-1}) within riparian type (native/grass) and land-use intensity (high/low)

			Bwidth (m)	Winter				Spring				Summer			
				NO3	NO2	NH4	PO4	NO3	NO2	NH4	PO4	NO3	NO2	NH4	PO4
KR01	Native	High	3.0	-40%	-117%	-20%	-75%	-8%	3%	-20%	3%	-12%	8%	38%	-4%
KR04	Native	High	5.5	14%	19%	46%	99%	0%	-3%	-1%	4%	2%	-87%	29%	-2%
KR06	Native	High	4.1	20%	69%	29%	41%	15%	56%	15%	6%	27%	27%	25%	9%
KR08	Native	High	5.0	6%	-2%	-7%	-7%	4%	-15%	25%	-4%	6%	-19%	8%	0%
KR13	Native	High	4.0	24%	-22%	17%	98%	34%	-39%	51%	3%	28%	-3%	13%	12%
KR10	Native	Low	2.5	28%	-161%	-13%	-15%	69%	26%	11%	-96%	135%	-91%	29%	6%
KR11	Native	Low	4.0	6%	38%	5%	-51%	-4%	10%	11%	0%	-10%	-21%	-14%	30%
KR14	Native	Low	4.5	19%	41%	6%	26%	17%	-75%	10%	10%	1%	1%	-2%	-15%
KR16	Native	Low	2.5	21%	71%	86%	16%	1%	-43%	47%	10%	3%	3%	-17%	-1%
KR17	Native	Low	4.5	-55%	-10%	-21%	-12%	-6%	0%	29%	60%	14%	2%	38%	-12%
KR03	Grass	High	3.0	14%	25%	-23%	95%	25%	-72%	9%	-14%	21%	-14%	11%	15%
KR05	Grass	High	2.5	-17%	44%	40%	-16%	-82%	5%	-45%	6%	0%	170%	11%	-9%
KR07	Grass	High	4.5	31%	-18%	-53%	103%	-48%	-99%	-48%	-19%	-25%	-3%	-7%	-5%
KR12	Grass	High	4.0	3%	-1%	31%	22%	5%	-76%	37%	-8%	11%	-40%	42%	-6%
KR18	Grass	High	5.0	21%	34%	23%	72%	2%	-51%	-4%	0%	31%	-67%	18%	17%
KR02	Grass	Low	3.5	18%	-80%	3%	3%	22%	-82%	39%	-12%	9%	-53%	69%	2%
KR09	Grass	Low	4.0	50%	-38%	5%	-55%	32%	101%	17%	23%	30%	-20%	4%	2%
KR15	Grass	Low	2.5	12%	-126%	-47%	-41%	-3%	18%	-25%	-15%	-2%	22%	11%	16%
KR19	Grass	Low	2.5	0%	-12%	-32%	-30%	15%	0%	-5%	1%	0%	36%	0%	-13%
KR20	Grass	Low	3.5	-1%	48%	-14%	16%	-31%	13%	11%	26%	-19%	-51%	14%	-3%