Assessment of the effects of upstream land uses and riparian vegetation composition on surface water quality of lowland streams

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Abstract of a thesis submitted in partial fulfilment of the requirements for the Degree of Master of Water Resource Management.

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Non-point sources pollution caused by land-based developments (such as increased in agricultural land, residential and industrial areas) become the major threats to the freshwater quality around the world. In New Zealand, the surface water quality has also been declining, and a significant increase in sediment and nutrients are considered as major water quality problems.

Riparian plantings (vegetation along the riverbank) are recommended as a cost-effective measure because they could reduce sediment and nutrients (nitrogen and phosphorus) inputs from non-point sources through their main functions: infiltration, filtration and absorption. Also, restoring and managing riparian plantings along the waterways is being introduced as one of the best management practice in New Zealand to reduce the impacts of catchment land use on water quality. However, the effectiveness of riparian plantings may vary in accordance with vegetation compositions (such as shaded buffer and grassland buffer) and the width of riparian planting area.

This study aims to assess the relationship between upstream and sub-catchment land-use and water quality and the effectiveness of different riparian vegetation compositions (shaded, un-shaded/grassland and unplanted areas) in reducing nutrient and sediment inputs from upstream and sub-catchment contributing land uses in lowland streams. The Styx River catchment in Christchurch, which has a wide range of riparian vegetation compositions, was chosen as study area. A total nine sampling sites were selected based on three different riparian vegetation compositions (shaded, grassland and unplanted) at the Styx and its main tributaries: Smacks and Kaputone Creeks. A total of 72 water samples were collected over eight dates (fortnightly over five dates and three dates after rain events) from nine sampling sites. Sub-catchment land uses were determined by using Arc GIS software, and defined into four main types: cropland, pastoral land, forested land and built-up area.

Riparian plantings (both shaded and grassland) showed a positive effect on reducing the concentrations of conductivity, turbidity, sediment, phosphorus and nitrogen. The riparian plantings with trees showed more effectiveness in reducing pollutants than grassland areas because mostly, the lowest levels of pollutants (conductivity and phosphorus, turbidity, sediment and nitrogen) were found at shaded sites. However, the proportion of sediment and nutrient fluxes depends on discharge rate. Dissolved oxygen levels showed correlation with water temperature levels.

The upstream and sub-catchment land-use (especially built-up area and pastoral land) was found to have a positive relationship with conductivity and nitrogen (specifically nitrate, total nitrogen and total dissolved nitrogen). The > 5 m wide riparian areas showed more effectiveness in reducing contaminants from built-up and pastoral influence area than solely pastoral land influence area.

In order to determine the effectiveness of riparian plantings, a number of factors (such as length and width of riparian plantings' area and stream shaded area) need to be considered, and a balance between the main functions of the riparian plantings in relation to the sensitivity of a proposed site will also need to be considered. Furthermore, the research suggests that the cooperation of private land owner is critical in establishing and managing riparian plantings along waterways.

Keywords: riparian plantings, land-use, water quality, effects, sediment and nutrient.

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Chapter 1

Introduction

1.1 Background information

Surface water quality often depends on land use activities occurring within a catchment. Numerous studies have investigated land use/land cover changes and identified them as major drivers of water quality deterioration and degradation of freshwater ecosystems at global, regional and local scale (Ahearn et al., 2005; Ganaie et al., 2018; Tafangenyasha & Dube 2008; Zhang et al., 2017) due to the pollutants contributed to the receiving water bodies by these actions. Here, “land-use” is considered as use of the terrestrial landscape for human needs such as agricultural, residential housing, and industrial purposes. Overall, this concept is interrelated with human actions and development, and changes in land-use patterns have driven global and local environmental problems.

Generally, pollution sources can be classified as either point or non-point sources. Sewage treatment plants and industrial discharges (discharge through a pipe or single place) are examples of point sources, as they can be easily monitored and managed (Shrestha et al., 2008). In contrast, non-point sources are associated with distributed discharge sources such as agricultural nutrients and runoff from roads and roofs in urban landscapes. In other words, non-point inputs transport contaminants from different land uses through overland flow during rainfall events and through groundwater flow. Non-point sources are difficult to measure and control (Scholz, 2011) and become persistent and dominant contaminant inputs to most surface water.

In general, agricultural land uses (including pastoral land) and urban landscapes are considered to be non-point pollution sources that contribute nutrients and chemical contaminants into surface channels and waterways through overland flow and into receiving water bodies. Those agricultural and urban land uses are increasing because of increased population, and consequently non-point source pollution of water bodies has become a long-standing and significant topic of interest in water resource management in many parts of the world. For instance, in the Manyame River catchment in Zimbabwe, both residential areas and agricultural activities were reported as the causes of a high degree of pollution compared to forested land (Kibena et al., 2014). In Lake Erhai catchment in China, surface runoff from agricultural land and residential areas was also reported as the major source of contaminants into waterways, based on a study using the Soil and Water Assessment Tool (SWAT; Yuan et al., 2019). Elevated levels of nutrients in surface water create a number of problems, including increased algal production and low dissolved oxygen concentration in fresh water, which, in turn, affects aquatic communities and recreational use (Ribaudo et al., 2003).

Similarly, in New Zealand, the surface water quality has been declining over recent decades as a result of non-point source pollution (Ministry of Environment, 2017). Fine sediment and nutrients (particularly nitrogen and phosphorus) are major contributors to surface water quality degradation (Howard-Williams et al., 2011). Many studies point to current and past land uses, specifically agricultural activities (approximately 40% of the country's land area (Verburg et al., 2010)) and urban developments for heightened nutrient (nitrogen and phosphorus) and sediment concentrations in waterways (Environment Canterbury, 2014; Ford & Taylor, 2006; Morgenstern et al., 2015; MfE, 2017; Wells et al., 2016).

However, nutrient and sediment inputs into waterways depend on catchment land use practices. For example, arable agricultural activities such as dairy farms transport higher amounts of nutrients and sediments into waterways than dry land agriculture (e.g., sheep farms; Monaghan, 2014; Smith et al., 2016). Increasing sediment loads affect water clarity (Davies-Colley et al., 2003) and aquatic ecosystems (Kemp et al., 2011). Excess nutrients cause eutrophication with algal blooms in streams and lakes and consequently degrade ecological values and recreational and aesthetic values. This has drawn attention to the need for reduction of nutrient and sediment inputs into New Zealand's waterways.

The use of riparian plantings has been recommended for reducing nutrients and sediment loads, as they serve as filters to minimise the impacts of adjacent land uses on waterways, especially non-point source pollution (Zhang et al., 2017; Mugni et al., 2013), and they provide recreational value as well as wildlife habitat through the contribution of leaf litter and woody debris. Furthermore, restoring riparian areas, which typically means re-establishing riparian reserves or buffer strips (Kauffman et al., 1997), can often be the most cost-effective measure to reduce non-point source pollution impacts on water quality in streams (McKergow et al., 2016).

However, the effectiveness of riparian plantings depends on various characteristics such as the width of the vegetated riparian area and the vegetation composition of riparian areas (e.g., a riparian vegetation area covered by grasses and/or shrubs, or a riparian area vegetated with mature trees; Connolly et al., 2015). For example, most of the pollutant reduction process takes place within the first 10 m to 15 m (metre) of the riparian planting area, but even a 9 m wide riparian area can effectively remove sediment from surface run-off (Coyne et al., 1995). Forested riparian areas can provide stream shading and prevent bank erosion and woody debris and surface roots create a network of pools where sediment can be trapped (Broadmeadow & Nisbet, 2004). In contrast, riparian grassland areas also have potential to control non-point pollution (Barden et al., 2003). They can effectively trap nutrients and sediment as well as pesticides and fertiliser from non-point pollution source (Mankin et al., 2007).

Although research has addressed the effectiveness of riparian vegetation on water quality, there is limited research in comparative studies of the effects of forested and grassland riparian area on water quality. Thus, this research proposes to investigate the effectiveness of different riparian vegetation compositions for reducing the nutrient and sediment loads from different contributing land uses into surface water in a small river catchment, the Styx River catchment, in Christchurch, New Zealand.

1.2 Styx River catchment

The Styx River, which is situated on the northern edge of Christchurch city, originates in the Harewood area before passing through several land uses (both urban and rural, including pasture) to meet the Waimakariri River at the Brooklands Lagoon. It is approximately 22 km in length with a 7,000 ha catchment area (Christchurch City Council, 2012). Although it has several small drains, there are only two main tributaries: Smacks and Kaputone Creeks (Figure 1.1) (Sourced from doschec_LincolnGIS). Smacks Creek is 2.5 km long and joins the Styx River directly north of the Styx Mill Road/Highsted Road intersection (CCC, 2017). Kaputone Creek is 11 km long, originating from the Belfast area (CCC, 2012).

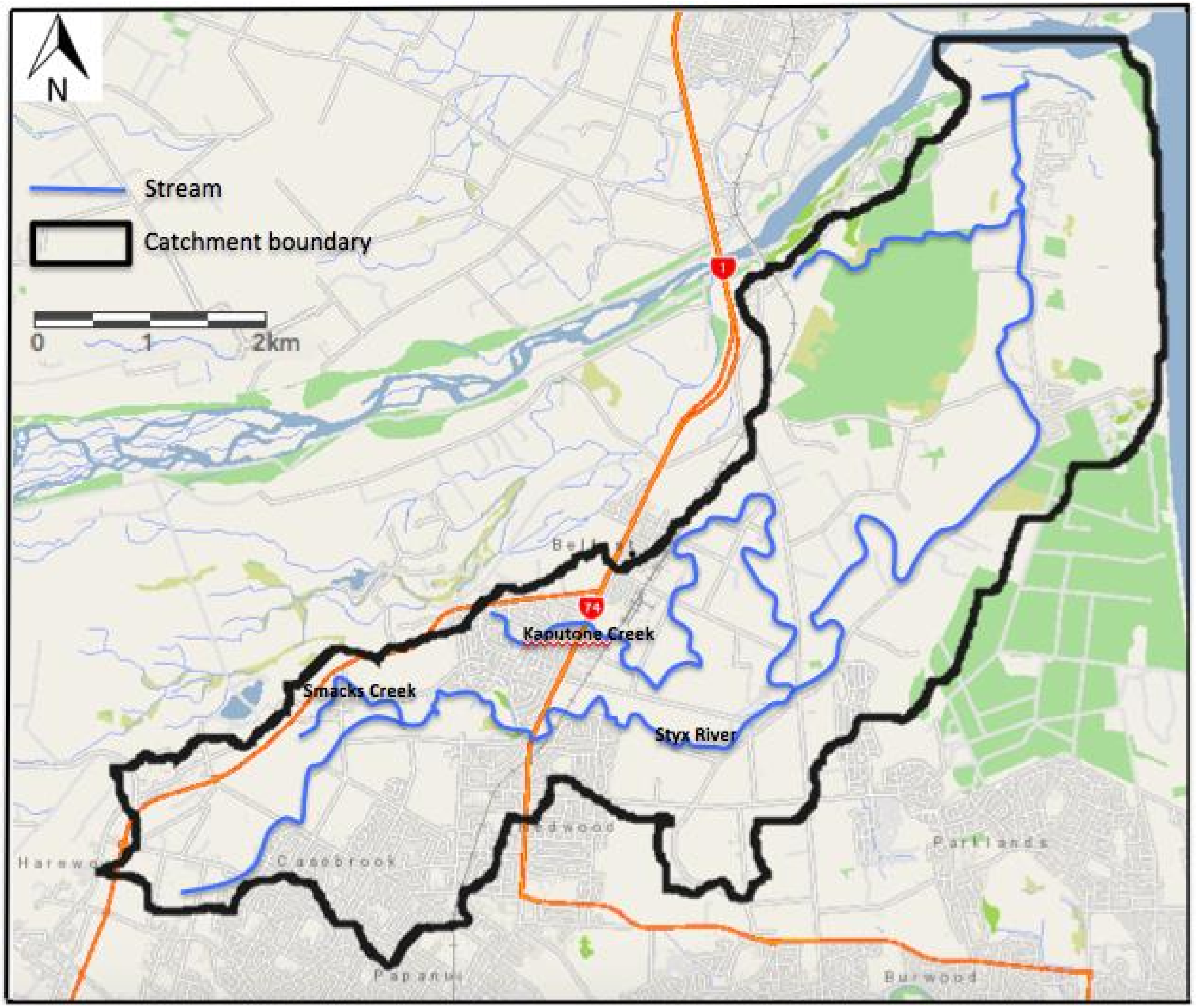

Figure 1.1 Styx River Catchment

In the past, several wetland areas were seen around the Styx River and its outlets. However, its ecological condition has been altered by land-use changes such as agricultural practices (farming and cropland), industries, and human settlements. A report from the Christchurch City council (2017) pointed out that Kaputone Creek has become "one of the most polluted waterways in Christchurch" through increased sediment and siltation as a consequence of land-use changes in the Styx River catchment. Although the water quality status in the Styx River Catchment was assessed as "good" in the Christchurch City council's 2019 report, water quality status in 2018 was reported as "fair" because the challenges of increased sediment and nutrient loads from land use remained, especially after rainfall events (CCC, 2019).

The city council has been developing an ecological restoration programme and a long-term environmental monitoring programme for the Styx River catchment (CCC, 2012). In addition, communities along the Styx River are eager to maintain and improve the water quality by getting involved in a riparian restoration programme and by inviting scholars to research the catchment. The Styx Living Laboratory Trust and Christchurch City Council are major sources of previous data related to Styx River catchment.

The upstream area of the river near Gardiners Road is flanked by riparian vegetation, predominantly with exotic species (as shown in Figure 3.2). Semi mature native tree and shrub plantings along the river could be seen through Harewood Park (see Figure 3.3; CCC, 2012). Then, the river passes through the Willowbank Wildlife Reserve and shrubs, semi mature native trees and open wetland areas in the Styx Mill Conservation Reserve before reaching the Radcliff Road (CCC, 2012).

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Figure 1.2 Downstream from Gardiners Road

(Sourced from (CCC, 2012, p. 15)

Material removed due to copyright compliance

Figure 1.3 Semi mature native planting

(Sourced from (CCC, 2012, p. 15)

Smacks Creek originates as springs in willow woodland near Harewood Park, and flows through reserves, as shown in Figure 3.4 (CCC, 2012), while Kaputone Creek flows through residential areas, open space, and reserve. Some areas of its bank are flanked with riparian vegetation as shown in Figure 3.5.

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Figure 1.4 Smacks Creek (CCC, 2012, p. 19)

Figure 1.5 Kaputone Creek near Northwood Park (Photo: 2020)

To summarise, the Styx River Catchment was selected as the study area because of its declining water quality trend and different riparian vegetation compositions as well as accessibility to the sampling sites. Furthermore, public access is available to a range of different riparian areas with different vegetation compositions.

1.3 Research objectives

This study aims to assess whether riparian vegetation can effectively reduce nutrient and sediment inputs from different land uses to small streams. The study will test the hypothesis that the riparian vegetation composition and the width of riparian vegetation determines the effectiveness of mitigating the input of nutrients, resulting from diffuse overland flow into a spring-fed stream, irrespective of surrounding land-use.

The research aim was achieved by completing the following objectives.

1) Collect water quality data from streams with different riparian vegetation conditions (shaded, grassland and unplanted buffer) and riparian width.

2) Characterise catchment land uses in multiple tributaries.

3) Evaluate the possible influence of riparian vegetation compositions and riparian widths on elevated nutrient and sediment concentrations with respect to immediate upstream land use combinations.

1.4 Thesis outline

This thesis is structured into six thematic chapters.

Chapter 1 is the introductory section of the thesis providing an overview of the thesis and its aim and objectives.

The literature review in Chapter 2 begins by presenting the land use links with water quality by defining two main sources of pollution produced by land uses. Then, a detailed discussion about non-point sources (especially agricultural activities and urbanization) and their effects on water quality in both global and local scales follow. After that, the effectiveness of riparian plantings is highlighted. Finally, land use and water quality in New Zealand and riparian-related sciences in New Zealand are presented.

The methodology in Chapter 3 provides the background information of the study area, with a description of the water quality status, and the geographical and riparian vegetation conditions of the Styx River catchment. The criteria for site selection, methods for assessing riparian vegetation condition, land-use analysis, field and laboratory analysis for water quality, and statistical analysis follow.

Chapter 4 presents the final results, which include the catchment and sub-catchment land-use characteristics, riparian vegetation condition and site characteristics (such as channel width and length, and flow rate) and the statistical analysis of the relationship of riparian vegetation and land use with water quality.

Then, Chapter 5 summarises the key findings and evaluates the effects of upstream and sub-catchment land uses and different riparian vegetation composition on water quality. Lastly, conclusions are presented in Chapter 6.

Chapter 2

Literature Review

2.1 Introduction

The purpose of this chapter is to discuss the global and local water quality problems related to the impacts of contributing land uses and it includes three main sections. First, a review section will focus on the relationship between land uses and water quality at a global scale and at the local scale in New Zealand. The second section will follow with the effectiveness of riparian plantings as a measure to reduce the impacts of land use on surface water quality. The last section will provide the linkage between land use and water quality and the effectiveness of riparian plantings being applied to reduce impacts on water quality at the national and regional scale in New Zealand.

2.2 The relationship between land uses and water quality

Although three-quarters of the earth's surface is covered by water, only 2.5% is available as fresh water and nearly three quarters of this 2.5% is locked up in the glaciers and permanent snow cover of the polar regions (Shiklomanov, 1997; Oki & Kanae, 2006) leaving only a small proportion available for human uses. The amount of fresh water available to support living organisms is very limited and 70% comes from surface water such as rivers, streams and lakes, with the rest sourced from underground aquifers.

Managing limited freshwater resources to allow for continued access to high quality water is a challenging task. Previous research findings have reported that land-use developments ultimately affect the freshwater quality in both surface water (rivers, lakes and streams) and groundwater (Duncan 2014; Liu et al., 2014; Ahearn et al., 2005). Agricultural intensification and urbanization are the common land-use developments leading to water pollution. Agricultural run-off can contaminate surface waterways and groundwater with nutrients, agrichemicals and animal waste (Duncan, 2014). Urban areas can contribute pollutants discharged from storm water and chemical contaminants from business and industries (Lee et al., 2009). The potential pollution sources could be categorised as either point or non-point sources according to their discharge characteristics.

2.2.1 Point source pollution

Point source pollution is characterised as discharges through a single point into a waterway, such as a pipe or discharges originating from a fixed outlet (Gyawali et al., 2013). Sewage treatment plant outlets and industrial discharges are examples of point sources. Possible contaminants from point sources can be easily monitored and regulated at a single point (Shrestha et al., 2008). For example, industrial wastewater effluent could be properly treated before discharge to reduce chemical contamination. Furthermore, point source discharges can be controlled under management schemes such as by setting up national or regional level standards and measures for the quality of industrial effluent before discharge.

2.2.2 Non-point sources

The origin of non-point sources tends to be diffuse and covers large areas. A definition of pollution from non-point sources normally includes the following factors.

- Linkages with land management, which can be controlled by society.
- Transported as part of the hydrological cycle. For instance, nutrients originating on pastoral lands can be washed into nearby waterways via overland flow and/or could infiltrate into the groundwater through the soil profile. Furthermore, chemical contaminants from fertilisers and pesticides, nutrients, sediments and organic and inorganic matters from croplands are potential inputs of agricultural activities.

Contaminants from impervious areas, such as roads, sidewalks, driveways and parking lots as well as industrial areas, can be transported into waterways. Runoff from roads and roofs in urban landscapes are examples of diffuse discharges. Urban runoff potentially carries nutrients, suspended solids, trace metals and bacteria. Sources of these pollutants can be vehicles, fertiliser and pesticide from home gardens, animal manure, construction activities, and run-off from roads and roofs. The agricultural and urban areas are the major cause of non-point source pollution (Ngoye & Machiwa, 2004), but groundwater might also transport pollutants from septic tanks and landfills.

2.2.2.1 Agricultural land use and water quality

Numerous studies on diffuse nutrient pollution have focused on the effects of runoff flowing over agricultural land (Scholz et al., 2010). The United States Environmental Protection Agency (2012) reported that rivers and lakes in the USA have deteriorated because of agricultural pollution, and a study in Finland described diffuse discharges from agricultural lands as being the leading cause of surface water quality degradation (Vuorenmaa et al., 2002).

Previous studies reported that agricultural land uses strongly influence the migration of nutrients (e.g., nitrogen (Duncan, 2014) and phosphorus (Liu et al., 2014)) and sediment loads in waterways (Ma et al., 2011). A study in the River Don system by Ferrier et al. (1995) showed that the level of nitrate increased with agricultural land-use conversions (e.g., from forested land or grassland to cropland). Also, in a comparative study of the effects of forest and agricultural areas on water quality in the USA, Lenat et al. (1994) determined that the highest nitrogen concentrations were produced from agricultural land. Lankoski and Ollikainen (2013) found that in Europe 50%–80% of the total nitrogen and phosphorus loads to waterways resulted from agricultural activities.

The literature on the impacts of agricultural land considers the impacts of all croplands and pastoral land, including irrigated land such as dairy farms, as well as dry land systems such as sheep farms. Donohue et al. (2006) identified that arable and pasturelands were the principal land uses affecting water quality in Irish rivers. Fisher et al. (2000) reported that all agricultural activities (including both crop land and pastoral lands) have strong impacts on water quality degradation based on its practices. For instance, sheep/beef farms can transport nutrients from untreated animal wastes into waterways, while croplands flush the soil particles combined with residues of fertilisers into the waterways.

Areas of concern over the effect of cultivated areas on water quality include the transporting of soil particles combined with chemical contaminants (Donohue et al., 2006). Because of an increasing demand for foods, chemical fertilisers, animal manure and pesticides are being applied more frequently to croplands. This causes nutrient enrichment in surface soils, and overland flows during and after the rain events carry soil particles with their load of nutrients (especially nitrogen and phosphorus) and chemical residues into waterways. Jordan et al. (1997) stated that concentrations of total dissolved nitrate in waterways correlated with the percentage cropland in Rhode River watershed area in the coastal plain of the Chesapeake Bay catchment. Myers et al. (2000) and Frey (2001) found that herbicide concentrations were highest in land uses with predominant row crop systems such as soybean and corn. Ferrier et al. (2001) demonstrated that while arable land cover influences nitrate concentrations, phosphorus concentrations correlate with improved grassland land cover.

Lin et al. (2015) examined agricultural land-use changes in the Red River of the North Basin shared by the US and Canada and reported that when agricultural areas were increased by the planting of bioenergy crops such as corn and soybeans, the increase in total phosphorus load (14.1%) was higher than total nitrogen loads (9.1%). Gakstatter et al. (1978) highlighted that non-point source phosphorus discharges from agricultural land uses and pastoral land caused 72%–82% of the eutrophication of downstream water bodies. The non-point sources load in agricultural areas is usually seasonal, with higher load associated with planting and harvesting activities.

Bouwman et al. (2009) stated that pastoral lands are considered to be one of the major accelerators of nutrient inputs into surface water bodies, since livestock wastes contain high levels of nutrients**.** Similarly, Daniel et al. (2002) reported that excessive loss of nutrients (principally nitrogen and phosphorus) and farm effluent from pastoral land into surface runoff and through leaching is the principal cause of degradation in surface and ground water quality. Hively et al. (2005) concluded that runoff generated from farm areas such as cow paths and farmyards may carry more significant phosphorus loads. International assessments showed that between 2000 and 2050, global livestock production will increase and consequently lead to an increase in nutrient inputs into surface water bodies (Bouwman et al., 2013).

Houston and Brooker (1981) noted that a catchment with predominantly arable land uses showed higher nitrate inputs into surface water bodies than a catchment mainly covered by dry land agriculture. According to the review by Monaghan (2009) on research related to the flows of nitrogen from dairy pasture in New Zealand, Northern Ireland and the USA, animal urine from dairy farms has become the most significant contributor of nitrate inputs into drainage and surface runoff over the last four decades. Haygarth et al. (1998) observed that dairy pastures transported from 59 kg to 194 kg of nitrogen per hectare per year in the United Kingdom. Similarly, a study by Watson et al. (2000) in Ireland showed that 18 kg to 65 kg of nitrate had flushed from one hectare of a dairy farm into adjacent waterways. It is assumed that phosphorus discharge from cattle-grazed land into surface runoff is also higher than from sheep farms (Haygarth et al., 1998; Haynes & Williams, 1993).

2.2.2.2 Urban landscapes and water quality

Increasingly large built-up areas and urban landscapes are a part of land-use change and they create both point and non-point source discharges into surface water bodies (Basnyat et al., 1999). Li et al. (2008) pointed out that the lack of or reduced vegetation cover and increase in impermeable landscapes may lead to higher degrees of overland flow and nutrient wash-out into waterways. Walsh et al. (2005) highlighted that the development of urban areas increases surface water nutrients, as well as changes to surface water hydrology, and also reported that nutrient and pollutant concentrations could increase as a result of urbanization even in low intensity catchments (catchments with low rainfall intensity). Lee et al. (2009) found that urbanization, rather than agricultural land use, was a major factor in water quality degradation according to a study of 144 reservoirs in South Korea.

According to a study in China, Li et al. (2013) argued that urbanization, including increases in residential, industrial, and transportation areas, were significant contributors of dissolved reactive phosphorus to surface water bodies. Similarly, in a study in Australia, nitrogen and phosphorus could be seen in dissolved form in both urban and rural areas (Petrone, 2010). Goonetilleke et al. (2005) also noted that dissolved phosphorus and urban land use had a linear relationship with the area of the urban landscape. They found that urban nutrient inputs doubled between 1973 and 1991.

Mouri et al. (2011) stated that dense populations, economic development, transportation infrastructure, and increases in impermeable surfaces led to an increase in surface runoff and nutrient (both nitrogen and phosphorus) concentration in surface water bodies. Duncan (2005) and Dunk et al. (2007) stated that sources of nutrients in urban runoff include leaf fall, fertiliser and pesticides used in home gardens, waste from birds, dogs and other domestic animals, and industrial debris. Furthermore, Atasoy et al. (2006) outlined that construction sites in urban areas or rural–urban conversion areas can also add total suspended solids (TSS) into surface water bodies through surface runoff. Increasing populations will require the development of urban/residential areas and, as a result, the nutrient and sediment inputs from urban diffuse sources are likely to continue to generate water quality problems in the future.

2.2.2.3 Land use and water quality in New Zealand

In New Zealand, surface water quality has been declining over the last several decades because of non-point sources that primarily come from land-use change to agricultural farming and residential area, coupled with fertiliser use (MoE & Stats, 2017; Ford & Taylor, 2006). Water quality in different catchments across the country is heavily influenced by catchment land uses. However, it also depends on many other factors, such as rainfall, soil type, fertiliser use, number of stock and stock type.

The rate of nutrients and sediment from urban areas can be high because of surface run-off from home gardens, construction sites and industrial and sewage discharge (Chakravarthy et al., 2019). However, it might be less than the loss rates from the agricultural activities.

Houlbrooke et al. (2004) reported that dairy farm expansion significantly increased between 1993 and 2003, and pastoral agriculture (including sheep/beef farms and dairy farms) became the major land use in the middle and downstream catchment areas of New Zealand's waterways (Allan, 2004). Fertiliser and pesticides use in the agricultural industry has also increased in the past few decades (MacLeod & Moller, 2006).

As a result, New Zealand's lowland freshwater ecosystems are degraded or may face degradation due to nutrient and sediment loads from agricultural diffuse discharges. Much research has highlighted that agricultural land use delivers the highest nutrient and sediment loads to waterways (Ford & Taylor, 2006; ECan, 2014; Morgenstern et al., 2015; Wells et al., 2016). Ballentine and Davies-Colley (2009) and Verburg et al. (2010) reported that pastoral agriculture is the biggest source of water pollution in New Zealand.

Ballantine and Davies-Colley (2009) reported that a significant increase in sediment and nutrients (nitrogen and phosphorus) in rivers is increasing over time at the national scale and they are considered as major water quality problems in New Zealand's waterways. Excess sediment and nutrients in rivers and streams lead to eutrophication and murky water creating challenges for aquatic ecosystems.

A large amount of nitrate-nitrogen loss is derived from cropland and pasture grazed areas (Monaghan et al. 2013; Shepherd et al. 2012; Smith et al. 2012). Animal urine is the major nitrogen sources (Monaghan et al., 2013). Melland (2003) and Parfitt et al. (2007) found that nitrogen inputs from sheep farms in New Zealand ranged from 1 kg to 19 kg of nitrogen /ha/year, while Monaghan and Smith (2010) argued that nitrogen losses from dairy farms are higher than from sheep farms.

Furthermore, Houlbrooke et al. (2009) outlined that 56% of dissolved phosphorus and total phosphorus were annually lost from dairy farms through surface runoff. They concluded that arable agriculture has become a significant contributor of phosphorus and sediment. Orchiston et al. (2013) also reported that cattle-grazed land transported high amounts of sediment in surface runoff.

However, McDowell and Wilcock (2008) reported that the major phosphorus losses in New Zealand could be found in particulate form bound in sediment losses from hill landscapes dominated by sheep/beef farms. They documented annual phosphorus loads ranging from 0.1 kg to 2.1 kg per ha and annual sediment losses ranging from 22 kg to 27 kg per ha respectively. They concluded that although much of the phosphorus in river and lake sediments has originated in the erosion of hill landscape sheep pasture, it is less well documented.

2.3 Reduction of nutrient and sediment loads to surface water bodies

In order to reduce the impacts of non-point source pollution, implementing conservation measures is of local and global interest. Better farming practices or best management practices (BMPs) have been introduced in many parts of the world to reduce the non-point nutrient and sediment impacts on surface water bodies (Monoghan et al., 2007). Such practices include reducing or limiting the use of fertiliser, establishing riparian plantings along waterways, fencing waterways to exclude stock access and managing irrigation systems (Müller et al., 2010; McDowell & Campbell, 2011; McDowell & Nash, 2012). Among those practices, many researchers recommended riparian plantings as a cost-effective conservation measures to reduce non-point source pollution to surface waters (Schoonover et al., 2005; Sahu & Gu, 2009; Collins et al., 2012; Zhang et al., 2017).

2.3.1 Riparian plantings

Riparian plantings are all the vegetation (both native and exotic) including trees, shrubs and grasses that grow on the banks of rivers, streams and lakes. They form the boundary between terrestrial land uses and water bodies and serve to filter sediments and nutrients to reduce the direct discharge of pollutants (Connolly et al., 2015; Zhang et al., 2017).

2.3.2 Effects of riparian vegetation on water quality

The streams with extensive riparian buffers (defined as a vegetated buffer strip on the banks of water bodies in this paper) had lower concentrations of nitrate than streams without riparian vegetation (Zhang et al., 2017). Also, Sahu and Gu (2009) reported that riparian vegetation could reduce nitrate loads to waterways, while Basnyat et al. (1999) observed that waterways surrounded by forest and grassland strips could reduce nitrate inputs from urban land use. Sparovek et al. (2002) estimated that riparian buffers removed approximately 89% of the nitrogen from field runoff, while Dal Ferro et al. (2019) showed that riparian plantings can also absorb phosphorus loadings.

Riparian plantings can also control sediment loads to waterways (Ghermandi et al., 2009; Dal Ferro et al., 2019). Other studies stated that riparian plantings can retain sediment, preventing it entering the waterways from across the surrounding areas (Loades et al., 2010; Vigiak et al., 2011). Furthermore, Wilkinson et al. (2009) described riparian plantings also reducing sedimentation created by stream bank erosion (which might be one of the major contributors of sediment) during heavy rain.

The major functions of riparian plantings associated with reducing nutrients and sediment include infiltration, deposition, filtration and adsorption. Infiltration, in which dissolved and particulate chemicals enter into the subsurface, is facilitated by leaf litter in riparian areas that offer high resistance to overland flow and decrease its velocity. For instance, Correll (1997) and Lee et al. (2000) stated that riparian vegetation and leaf litter fall from riparian trees onto the soil surface and so reduce the velocity of runoff and absorb nutrients and sediment. A study by Gilliam (1994) also highlighted that about 50% of phosphorus in runoff could be removed through the infiltration function of riparian vegetated areas.

Filtration, during which solid particles are separated from overland flow, is facilitated by vegetation and litter. For instance, Zhou and Shangguan (2008) stated that riparian plantings retain sediment from surface runoff by slowing the movement of overland flow. Dal Ferro et al. (2019) also found that riparian buffers remove phosphorus by trapping soil particles (including particulate phosphorus and nitrogen).

Riparian plantings can absorb dissolved chemicals into the soil and plants during overland flow, though this function is not very significant because of the short contact time. Clay particles in riparian areas absorb dissolved forms of phosphorus from runoff (Lee et al., 2000). Bowden et al. (2007) also highlighted that riparian plantings could take up large amount of nitrogen to support the production of roots, leaves, and stems. Vance (2001) also confirmed that vegetative uptake is a very important mechanism for removing nitrate from riparian systems, since vegetation (especially trees) can remove nitrates and convert that nitrate to organic nitrogen in plant tissue.

In addition to reducing nutrients and sediment, riparian plantings have other effects on water quality. They can help maintain optimal water temperature by providing shading and control algal growth by reducing light penetration. However, many studies stated that the success of riparian vegetation depends not only on the pollutant loads but also the characteristics of the riparian buffer such as vegetation composition (forested/shaded or unshaded/grassland vegetation) and the size of the strip (e.g., buffer width; Meyer et al., 2005; Connolly et al., 2015).

2.3.3 Forested/shaded riparian areas versus grassland riparian areas

Schoonover et al. (2005) demonstrated that riparian plantings with a shaded buffer could reduce concentrations of nitrate-nitrogen by approximately 97% and phosphate mass by about 78% in runoff from croplands. Other studies also confirmed that forested/shaded buffers play a significant role in removing nitrogen and nitrate. For instance, Henri and Johnson (2005) confirmed that forested riparian areas possess a high capacity to reduce nutrients and sediment loads. Lee et al. (2003) reported that an increase in forested buffer areas could raise the removal efficiency of soluble nutrients by 20% and sediment by 2%.

Riparian grass buffers can also reduce nutrients and sediments in the same way as forested buffer areas (Syversen, 2005). Riparian grassland buffers could reduce total nitrogen and total phosphorus by 50% (Daniels & Gilliam, 1996). Similarly, Mankin et al.

(2007) demonstrated that riparian areas with grass–shrub plantings could reduce total suspended solids by over 90%, and total phosphorus and total nitrogen by 40% to 90%. Aby-Zreig et al. (2003) stated that nutrient loads from runoff could be reduced by increasing grassland cover in riparian areas, while Hook (2003) and Deletric (2005) reported that the riparian grassland could retain sediment effectively.

Although both forested and grassland riparian buffers have their own advantages in reducing sediment and nutrients, there is still disagreement over their potential negative effects on water quality. Meyer et al. (2005) argued that riparian shading trees could wash out leaf litter into the runoff, creating another source of organic matter in waterways. Furthermore, Dillaha et al. (1989) reported that riparian grasslands might trap particulate phosphorus, but that particulate phosphorus can be later released during storm events. However, in a comparative study of the capacity of mixed forested and grass buffer strip, Osborne and Kovacic (1993) stated that on an annual basis, grass buffers were a more efficient sink for phosphorus than forest buffers. This suggests that more research is still needed to determine the capacity of forested and grassland riparian area.

2.3.4 Effectiveness of riparian area buffer widths

Much research related to riparian areas has suggested that riparian buffer width could determine the effectiveness of riparian vegetation (Dillaha et al., 1989; Schoonover et al., 2005; Syversen, 2005; Mankin et al., 2007). A study by Dillaha et al. (1989) reported that 10 m wide grass buffer strips removed phosphorus from runoff by 89%, while 5 m of riparian grassland removed only 61%. Similarly, Schoonover et al. (2005) stated that nutrient reduction within a 10 m wide forested riparian zone was relatively high (between approximately 70% and 90%). Syversen (2005) also found that 10 m wide riparian planting areas showed significantly higher removal efficiencies than 5 m wide areas.

Mankin et al. (2007) described how 8 m wide grassland riparian areas could improve water quality effectively. Their results showed that nutrient and sediment reduction rates were over 90% for total suspended solids and over 45% for total nitrogen and phosphorus. Lee et al. (2003) stated that when the width of grassland riparian areas was increased from 7 m to 16 m, the removal efficiency of soluble nutrients increased by over 20%, though sediment removal increased by only 2%. Duchemin and Hogue (2009) recommended that the width of a riparian area should be at least 5 m to effectively reduce runoff volume, suspended solids and nutrients. However, a study in New Zealand on the effects of riparian buffer width (National Institute of Water and Atmospheric Research Ltd, 2000) suggested that riparian zones should be least 10 m wide to sustain their functions; in very small streams or where there is not enough space for riparian plantings, 5 m wide buffers could be better than nothing and for large waterways, riparian vegetation at least 20 m wide would be most effective.

2.3.5 Reducing nutrient effects on fresh water in New Zealand

Farming practices have changed, and continue to change, to mitigate nutrient loss and the environmental impacts of land uses. Legislatively water quality degradation has been addressed under the Resource Management Act 1991 (RMA). Through the National Policy Statement for Freshwater Management (NPS-FM, 2014), the regional governments have instituted a limits to nutrient losses from agricultural practices to reduce environmental impacts (Greenhalgh & Murphy, 2017).

Furthermore, best management practices for farming are being introduced to reduce the impacts of agricultural land use in New Zealand and maintain the country's "clean and green" reputation (Monoghan et al., 2010; Collins et al., 2012). Riparian restoration, and fencing along the waterways to exclude direct access to stock are parts of this practice. Therefore, regional and district councils are cooperating with local communities around the country in implementing riparian restoration projects to maintain their waterways and ecosystem values.

2.3.6 Riparian planting in New Zealand

In New Zealand, riparian vegetation has been found to improve soil infiltration capacity and reduce sediment and nutrients entering the waterways (Cooper et al., 1995), and Franklin et al. (2015) highlighted that nutrient absorption by plants is high in riparian zones with native plant species. Rutherford and Nguyen (2004) also reported that riparian areas could remove approximately 25% of nitrate from overland flow.

Other catchment studies highlighted that riparian vegetation could reduce suspended solids and phosphorus loads and improve water clarity (Williamson et al., 1996;

Hughes & Quinn, 2014). Wilcock et al. (2009) stated that over eight years of experiments in the Waiokura Stream catchment, Taranaki riparian planting reduced the sediment by 10%–25% and also significantly reduced phosphorus loads from irrigated land.

A previous study of the effectiveness of the riparian zones on water quality in four separate streams/creeks in the Te Waihora catchment, Canterbury, was undertaken by Collins et al. (2012). The results showed significant increases in dissolved oxygen and conductivity and decreases in turbidity in planted sites. However, no other differences were found in other variables between planted and non-planted sites, and Collins et al. (2012) discussed whether the findings might be affected by the insufficient width of all four planted buffers and by gaps in the buffers.

The influence that riparian vegetation has on the amount of nutrients and sediment loads into waterways therefore remains unclear and, if they are effective, to what extent they can reduce these loads. Moreover, as stated in the preceding literature review, if the capacity of riparian buffer areas depends on their width and vegetation composition (e.g., forested versus grassland buffers), the effectiveness of riparian zones for reducing land-use impacts on surface water quality needs to be investigated more fully to determine the most effective combination of type and width.

2.4 Approaches to assess the effects of catchment land uses on water quality

The effects of catchment land uses on water quality are complicated to address and require a combination of several approaches rather than a single approach. The following section provide a review of several approaches that have been applied in previous studies in order to understand the underlying effects of catchment land-use composition on water quality.

In order to identify the effects of diffuse pollutants from catchment land-use composition, computerised water quality modelling tools have been used to evaluate land use and water quality relationship. For example, Mungi et al. (2013) used the "Soil and Water Assessment Tool (SWAT)" developed by Arnold et al. (2012), along with a comparative study of water quality near different agricultural areas or near different land uses to assess the effects of catchment-scale land use on water quality. Lin et al. (2015) also used the SWAT model to compare the effects of agricultural land planted with different crop types (especially corn and soybean) in the Red River basin in the USA. The model was calibrated using observed stream flow, suspended sediments, and nutrients during the 2006 to 2013 period.

Some researchers have used a sub-catchment spatial scale (Kibena et al., 2014), while others used a whole catchment approach (Chu et al., 2013). In both cases, they used geographical information systems (GIS), remote sensing (RS) and water quality monitoring data. For instance, a study in the Manyame River upstream catchment in Zimbabwe (Kibena et al., 2014) used GIS and delineated sub-catchment land uses; they combined these with water quality data of each sub-catchment to compare land use/land cover maps within a specific period (between 1995 and 2012). In contrast, a study verified by Chu et al. (2013) in Tseng-Weng reservoir watershed in Taiwan used a whole catchment approach. They used high-resolution satellite imagery to derive the normalised difference vegetation index (NDVI) in order to identify land use changes between 2001 and 2010 and combined them with water quality monitoring data.

2.5 Approaches to assess the effectiveness of different riparian plantings

Some research studied the effectiveness of shaded/forested riparian areas on water quality by comparing the percentage of riparian shading and water quality data, while the others studied the effects of riparian grassland. For instance, a study by Burrell et al. (2014) in 21 streams of the Canterbury Region of New Zealand assessed riparian cover (from closed canopied to open canopied) to assess the effectiveness of riparian shading in mitigating stream eutrophication in agricultural catchments. In the case of the Mogi-Guacu River catchment in Brazil, Fernandes et al. (2014) assessed the effectiveness of riparian planting by measuring the diameter at breast height (DBH) of trees in riparian forests and compared those results with water quality parameters.

The first method (vegetated and unplanted areas) is mainly used, while the others were applied according to the available riparian data of the study area and research interests. For each riparian study, although riparian width is considered an essential criterion, the information related to riparian length is less well documented.

2.6 Summary

Land use changes significantly contribute to surface water quality degradation, especially with increasing nutrient and sediment loads through non-point pollution. Agricultural and urban activities are major non-point pollution sources to surface water bodies. Impermeable surface such as roads and constructed areas are increased during the development of urban land uses, which causes increases in volumes of nutrients and sediment. There is considerable evidence for correlations between agricultural and urban land uses and elevated nutrients and sediments in surface water bodies.

As agriculture and urban land uses are increasing in both developed and developing countries, water quality problems related to non-point pollution have become a major environmental problem. Similarly, New Zealand's waterways have been facing water quality problems resulted from increasing agricultural and urban land use conversion.

In order to mitigate these water quality issues, much research has suggested that riparian plantings are cost-effective conservation measures, since they can filter nutrients and sediment, but the effectiveness of riparian plantings can vary depending on the vegetation composition (such as forested and grassland riparian areas) and their width.

Therefore the next chapter will discuss how this research investigated the effectiveness of different riparian plantings (forested versus grassland) in reducing nutrient and sediment input from catchment land uses.

Chapter 3
Methodology

3.1 Introduction

The aim of this chapter is to outline how the research was conducted and how this study was designed to meet the study objectives. Firstly, background information on the study area (including land use, water quality trends and riparian vegetation composition) will be presented. Secondly, the study design, data collection methodology and descriptions of each sampling site will be given. Finally, the statistical analysis methods will be presented.

3.2 Study design

In order to understand the effects of upstream contributing land uses and riparian vegetation composition on surface water quality, a comparative study of water quality data near different riparian vegetation (shaded buffer, grassland buffer, and unplanted buffer) and upstream and sub-catchment land use data was conducted.

3.3 Site selection

Sampling sites were selected according to two main criteria: variety of riparian vegetation composition and accessibility. Three different riparian vegetation compositions (shaded, grassland and unplanted riparian areas) within 50 m upstream of the sampling points were prioritised for site selection. Shaded riparian areas were defined as those where at least 40% of the stream was shaded with riparian vegetation, and mainly covered with trees. The grassland areas included at least 3 m of riparian plantings (including short and long grasses, and shrubs) while the unplanted areas as those where the riparian plantings area did not extend beyond 2 m from the edge to the river. Accessibility to the streams and wadeable areas were also taken into consideration for both time management and health and safety reasons.

Site selection was firstly conducted through online databases especially satellite imaginary on Google Earth and Environment Canterbury's maps viewer before beginning field data collection in order to check riparian vegetation condition. After selecting the sites through the online database, characterisation of the sites was carried out to determine the different riparian vegetation composition, to measure riparian widths (50 m upstream from the sampling points) and to check the channel depth on 21st June 2020.

A total of nine water quality sampling sites were selected along the main stem (the Styx River and major tributaries of the Styx River (Smacks and Kaputone Creeks). The three sampling sites were selected at each stream based on three different riparian vegetation compositions, and the sites were named in short form of the name of the streams (SM for Smacks Creek, K for Kaputone Creek and ST for the Styx River) and riparian condition (S=shaded, G=grassland and U=unplanted).

3.4 Land use analysis at sub-catchment scale

The land use characteristics of the Styx River catchment and sub-catchment for each sampling point were determined by using ArcGIS software. All GIS datasets used in this study were obtained from Land Information New Zealand (LINZ) and the Ministry for the Environment (MfE) data services. These GIS dataset layers include land use classifications for New Zealand for the year 2016, a 1 m Digital Elevation Model (DEM) data for Christchurch City, and catchment layers and river lines for Styx River catchment.

First, the 1 m DEM was used to determine flow-path direction and to delineate sub-catchment areas for each sampling site in ArcGIS version 10.6 (ESRI Company, Redlands, CA, USA). The 1 m DEM data were added to the table of contents layers window of ArcGIS, and Hydrology from Spatial Analysis Tools in the ArcGIS tool box was applied. The procedures involved calculating fill from the Hydrology toolset and then calculating flow direction, flow accumulation, stream link, stream order and stream feature. The shape of river lines and a New Zealand map were added to the table of contents, and snap pour point was applied to create points for each sampling site by setting New Zealand Transverse Mercator as the coordinate system. Then, upstream sub-catchment areas for each sampling site were delineated using sampling points as the outlet points through the watershed tool in ArcGIS, and the Raster to Polygon conversion tool was used to produce sub-catchment polygons. Each polygon was exported as a shape file and their areas were calculated in the attribute table.

In order to determine land uses at catchment scale and sub-catchment scale, land use data were added to the table of contents of window in ArcGIS and filter land use data for 2016. The land use layers were first clipped using the Clip tool in order to extract the area of interest defined by the boundary GIS layer of the catchment and sub-catchments.

The 2016 existing landcover GIS layer database for dominant land-use types of Styx River catchment was used to create a land use map for the catchment with 12 dominant land use classes. These land use classes were then reclassified into four main land-use types as the major land use focus of the study: cropland, pasture, forest and built-up. Table 3.1 summarises the original land-use types assigned to each of these major four land-use types.

Table 3.1 Reclassification of dominant land-use types into four main land uses

No.	Four main land-use types	Dominant land uses
1	Crop Land	Crop land annual, Crop land Orchard and Vineyards
2	Pasture	Grassland – high production Grassland – low production
3	Forest	Grassland with woody biomass, Natural Forest, Planted Forest pre-1990, Post-1989 forest, Wetland-open water, Wetland-vegetated non forest
4	Built-up	Settlement, Other

3.5 Assessing riparian planting

The width of riparian plantings area was measured inland perpendicular to the riverbank. The true right and true left sides of the stream bank were determined by facing upstream from the sampling point. The riparian width was measured at 12.5 m intervals from the sampling point to 50 m upstream. The average width was calculated by dividing the sum of widths within 50 m by the number of intervals.

Vegetation composition (generally the percentage of grass, shrubs and matured trees) was also recorded. In this case, plantings over 5 m in height were identified as trees. Shrubs involved vegetation with branches that were between 0.3 m to 5 m in height. The percentage of grass, shrubs and mature trees was calculated according to the area they occupied within the measured riparian area.

3.6 Location of sampling sites

At Smacks Creek, the grassland site (SMG in Figure 3.1) was located in the headwaters of Smacks Creek, and the unplanted site (SMU) was about 600 m to 700 m downstream of the grassland site, while the shaded site (SMS) was located about 100 m downstream of the unplanted site.

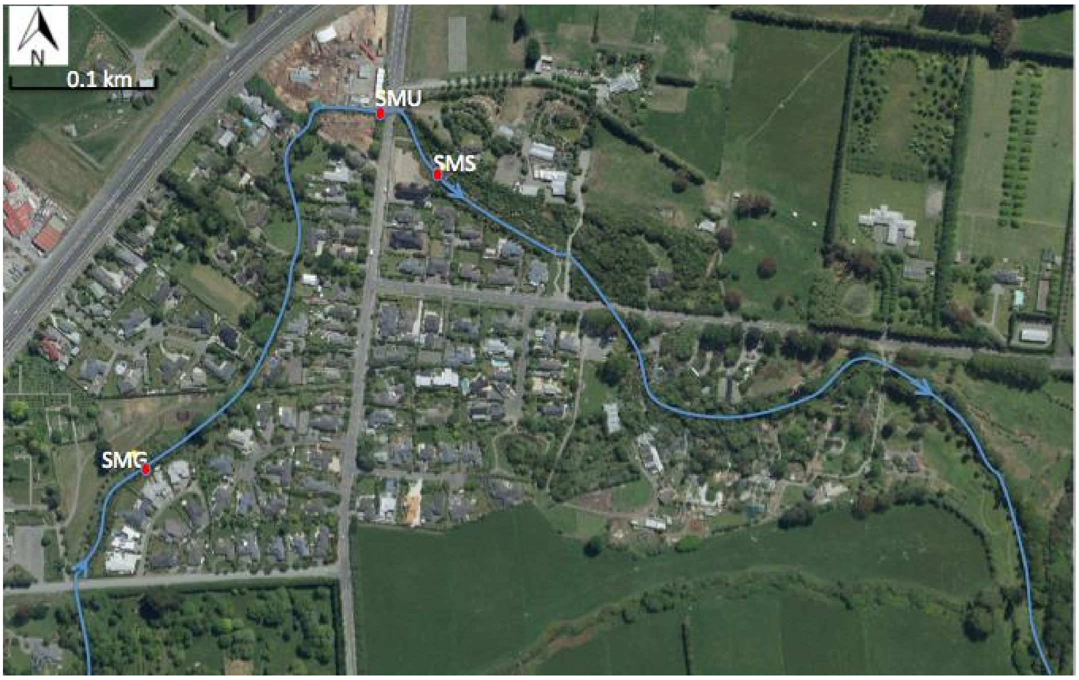

Figure 3.1 Sampling sites at Smacks Creek (stream=blue line, arrow=flow direction and red circle= sampling site)

Shaded site was (KS on Figure 3.2) located at the upstream area of Kaputone Creek within Northwood Park, and downstream of shaded site near a new sub-division was the unplanted site (KU). The grassland site (KG) was located around 150 m to 200 m downstream of the unplanted site.

Figure 3.2 Sampling sites at Kaputone Creek (stream=blue line, arrow=flow direction and red circle= sampling site)

Shaded site (STS in Figure 3.3) was located upstream in the Styx River near Harewood Park, and downstream from the shaded site near the Styx Mill Dog Park was the grassland site (STG). Then, the unplanted site (STU) was located the farthest downstream, near Radcliff Road.

Figure 3.3 Sampling sites at Styx River (stream=blue line, arrow=flow direction and red circle= sampling site)

3.7 Data collection and analysis for water quality data

Water quality sampling was conducted from 28th June 2020 to 1st September 2020. Water samples were collected over eight dates: fortnightly over five dates and on three dates after rain events. The event-based sampling was carried out within 24 h once after rainfall exceeds 3 mm in 24 h periods. The event-based sampling was carried out recognising that nutrients and sediments could be transported into waterways during and after rainfall events (Jones et al., 2011). Sampling was started at the same time of day and sites were visited in the same order to reduce diurnal variation between dates.

3.7.1 Physiochemical water quality data

At every sampling collection, water temperature (TEMP), pH, conductivity (COND) and dissolved oxygen (DO; mg/L) were measured in situ using a Hach HQ40d kit). All parameters were measured with two replicates and an average value was calculated. Turbidity (TURB) was measured using turbidity meter (Thermo Fisher AQ4500 Turbidity meter) in NTUs.

3.7.2 Nutrient analyses: phosphorus and nitrogen

Duplicate samples were collected for nutrient analysis at each sampling time and location. Samples for dissolved nutrients were filtered (0.45 µm) on site. All samples were stored at <5° C until analysis, which took place within 24 h of sample collection.

To analyse phosphorus (dissolved reactive phosphorus (DRP), total dissolved phosphorus, (TDP), total phosphorus (TP) and particulate phosphorus (PP)), samples were collected into new or acid-washed phlypropylene containers.

Dissolved reactive phosphorus (DRP) concentration was measured using the ascorbic acid method (APHA, 2005) at 880 nm using a DR3900 Laboratory Spectrophotometer (Hach Pacific, Auckland, New Zealand). Samples for total phosphorus (TP) and total dissolved phosphorus (TDP) analysis underwent digestion using potassium peroxodisulfate and sodium hydroxide at 120° C. The digested samples were then analysed using the ascorbic acid method (APHA, 2005) as described above. Particulate phosphorus (PP) was estimated by subtracting TDP from PP.

Nitrogen (as nitrate; NO_3-N) concentrations were measured using the cadmium reduction method reported by Caspers et al. (1979). The absorbance was measured at 543 nm using the DR3900 spectrophotometer. Samples for total nitrogen (TN) and total dissolved nitrogen (TDN) analysed underwent persulfate digestion, as described above, followed by Flow Injection Analysis (FIA) (APHA, 2005). The concentrations of particulate Nitrogen (PN) were calculated by subtracting TDN from TN.

A solution containing glutamic acid and nitrophenyl phosphate disodium salt was used as organic standards to confirm recovery of N and P using the persulphate digestion and to estimate the uncertainty of the method.

3.7.3 River discharge

The river discharge was measured at each site using the midsection method (Hipolito & Loureiro, 1988) to determine an estimation of nutrient and sediment fluxes. Briefly, the width of the river at the sampling point was divided into 0.45 m subsections and then the average depth and velocity of each sub-section were measured using a flow meter (Son Tek-SL/son Tek IQ series).

3.7.4 Total suspended solid

Two litres of water was collected at each sample collection for total suspended solids (TSS) analysis. The exception to this was the first round of sample collection on 28th June 2020 when only one litre was collected from each location. The low TSS concentration at this time informed the need to collect a greater volume collection subsequently. The samples were analysed using the standard APHA method (APHA, 2005). However, GE glass microfiber filters with 0.45 µm pore size were used instead of the recommended glass fibre with 2 µm pores used in the standard methods.

3.7.5 Nutrient and sediment flux analysis

Nutrients (both phosphorus and nitrate species) and suspended solid concentrations at each site were multiplied by flow data, and nutrient and sediment flux were calculated. These values were used to make a comparison between land uses and riparian conditions and nutrient and sediment fluxes for each site. Then, the ratio of discharge and nutrient and sediment concentrations was calculated in order to determine at which site discharge carried the high concentrations of nutrient and sediment.

3.8 Statistical analysis

The collected data were tested for normal distribution using SPSS software (SPSS,1988). As the data were not normally distributed, the non-parametric statistic, Kruskal–Wallis test, was used to analyse the significant differences among water quality parameters within treatments ($P < 0.05$). This method was used because its assumptions did not require normality within the data sets (Lowrance, R & Sheridan, J. M., 2005). The significant differences between water quality data were analysed within two main treatments.

1) Treatment-1 was undertaken to determine significant effects of different riparian vegetation composition on water quality parameters. The three groups in treatment-1 was categorised based on different riparian vegetation compositions with different average width of riparian planting area on both true left and true right banks: 1) grassland with >20-40 m (referred to as “category-1” in the Results chapter), 2) shaded with >5-20m (“category-2”) and 3) unplanted with 0-5 m (“category-3”).

2) Treatment-2 was processed to evaluate the effectiveness of riparian vegetation and riparian width on reducing nutrient and sediment loads from land uses. The three groups in treatment-2 were grouped based on three different riparian width with different land uses: 0-5 m width predominantly built-up (referred to as “group-1” in the Results chapter), >5 m riparian width predominantly built up/pastoral land use (“group-2”) and >5 m riparian width predominantly solely pastoral land use (“group-3”).

Kruskal-Wallis test was also used to analyse significant effect of rain event and sites on water quality parameters. Relationships between water quality parameters and land use (crop land, pasture, forest and built-up areas) were also examined using Pearson’s correlation analysis.

Chapter 4

Results

4.1 Introduction

The purpose of this chapter is to present the results of the field and laboratory work and the statistical analysis. First, the catchment land uses for the whole Styx River catchment will be presented. The second, third and fourth sections will follow with site characteristics including riparian vegetation composition and sub-catchment land use for sampling sites at each stream (Smacks Creek, Kaputone Creek and Styx River). Then, the results of the statistical analysis for each water quality parameter will be presented, considering the effects of different riparian vegetation composition and riparian width, rain events and differing sub-catchment land uses.

In terms of the results presented, treatment-1 refer to three different groups of riparian vegetation compositions: shaded (S), grassland (G) and unplanted (U) and average riparian width on both true left and true right banks, while treatment-2 represent three different groups: 0-5 m width predominantly built-up, >5 m riparian width predominantly built up/pastoral land use and >5 m riparian width predominantly solely pastoral land use. The effect of treatment-1 means that results are significantly different between the three different riparian vegetation compositions, while the effect of treatment-2 means that results are significantly different between different riparian width with different contributing land uses. A site effect means that results are significantly different between sites at the three streams based on site situation such as the aquatic plants condition, groundwater leaching or stream bank stability at each site.

4.2 Catchment land use contributions

The state of land use in the 5,832 ha Styx River catchment is graphically presented in Figure 4.1 (Map created based on 2016 land-cover database sourced from LINZ). At the time of this study, the land use in the entire catchment was dominated by pastoral land, while forested land was the second most dominant land use type (Table 4.1). While the majority of land use in the upper Styx catchment of STU (Radcliffe Road near Hawkins Road) was built-up and pastoral land, the lower catchment area was dominated by forested land mixed with pastoral land. Cropland was spread throughout the whole catchment area.

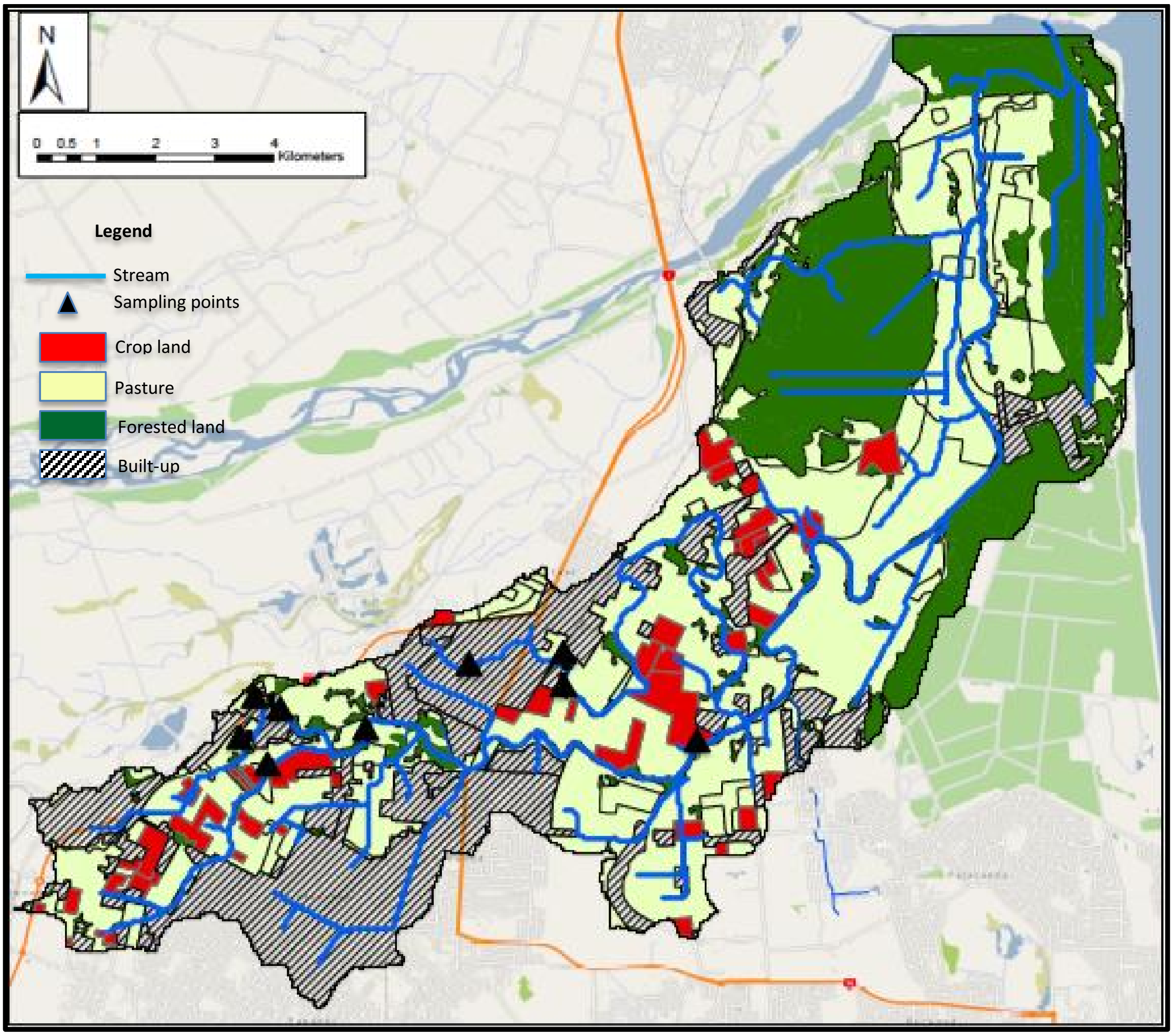

Figure 4.1 Land use contributions of Styx River Catchment

Table 4.1 Descriptive statistics for contributing catchment land use in Styx River Catchment and nine-sampling sites

	Crop (%)	Pastoral Land (%)	Forest (%)	Built up (%)
Whole Catchment	6	45	25	24
Sub-catchment scale for nine sampling sites				
Minimum	0.00	0.00	0.00	0.60
Maximum	21.03	68.90	25.00	100.00
Mean	3.55	20.92	11.54	64.00
Median	0.00	16.20	10.00	74.30
SD[@]	7.08	24.20	7.39	32.30

[@] Standard Deviation.

The result of sub-catchment delineation analysis for each sampling site is shown in Figure 4.2. The nine water quality sampling sites were mainly located at the upper catchment area across the four main land-use types (cropland, pastoral land, forested land and built-up area). Among the nine sampling sites, the percentage of cropland ranged from 0.00% to 21.03%, pastoral land from 0.00% to 68.9%, forested land from 0.00% to 25%, and built-up area from 0.6% to 100% respectively (Table 4.1).

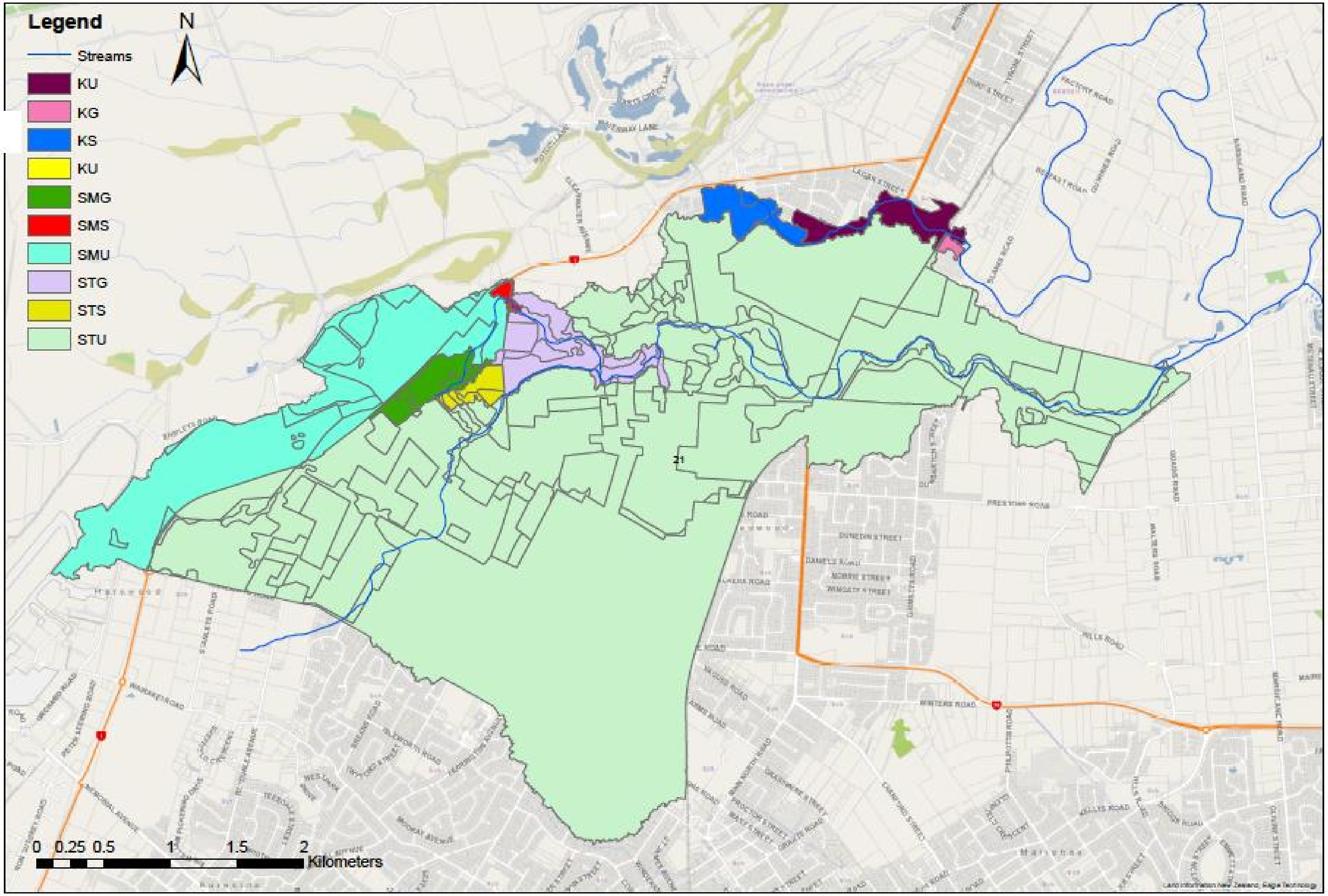

Figure 4.2 Sub-catchments for the nine sampling sites

4.3 Site characterisbtics at Smacks Creek sites

Among the three sampling sites at Smacks Creek, SMG had the widest riparian planting areas on both side of the banks compared to SMU and SMS (Table 4.2). The riparian vegetation at SMG was short grasses, shrubs and trees (Figure 4.3). At SMU, the riparian vegetation consisted of short grasses (Figure 4.4), and the riparian plantings at SMS were covered mostly with shrubs, small numbers of trees and mature trees (Figure 4.5).

Table 4.2 Site characteristics and riparian vegetation condition

	SMG	SMU	SMS
Width of riparian plantings area on the true right bank (m)	5	0	3.7
Width of riparian plantings area on the true left bank (m)	36	0.7	13.2
Channel width (m)	1.8	2	2.3
Channel depth at the deepest point (m)	0.25	0.18	0.13
Sub-catchment area (ha)	19	189	2
Crop Land (%)	0	-	-
Pastoral land (%)	5	16	18
Forest (%)	12	10	17
Built-up (%)	83	74	65

Note: the sites are shown from upstream to downstream, a) SMG = Smacks Creek grassland, b) SMU = Smacks Creek unplanted, c) SMS = Smacks Creek shaded.

The channel at SMS was the widest among the three Smacks Creek sampling sites. The channel was wider and shallower at downstream sites (Table 4.2). The river bed at SMG was covered with gravel and stones, while the river bed at SMU and SMS was mainly covered with small stones. Overall, land use at in the sub-catchments of the sites at Smacks Creek was mainly built-up areas (65%–83%) followed by forestry (10%–17%) and pastoral land (5%–18%) (Table 4.2).

Figure 4.3 Riparian vegetation condition at SMG (Smacks Creek grassland site)

Figure 4.4 Riparian vegetation condition at SMU (Smacks Creek unplanted site)

Figure 4.5 Riparian vegetation condition at SMS (Smacks Creek shaded site)

4.4 Site characteristics at Kaputone Creek sites

Among the three sampling sites at Kaputone Creek, KG had the widest riparian planting areas on both side of the banks compared to KS and KU (Table 4.3). The riparian area at KS was covered with small trees, shrubs and grasses (Figure 4.6), and the riparian vegetation at KU composed of tussock grass, also known as snow grass (Figure 4.7). The riparian area at KG mainly consisted of tussock grass and short grass (Figure 4.8).

Table 4.3 Site characteristics and the width of riparian plantings area

	KS	KU	KG
Width of riparian plantings area on the true right bank (m)	3.9	2.1	11.2
Width of riparian plantings area on the true left bank (m)	28.2	0.9	44
Channel width (m)	2.5	1.5	2.2
Channel depth at the deepest point (m)	0.14	0.06	0.1
Sub-catchment area (ha)	17	1	2
Crop Land (%)	-	-	-
Pastoral land (%)	-	-	1
Forest (%)	-	17	11
Built-up (%)	100	83	88

Note: the sites are shown from upstream to downstream, a) KS = Kaputone Creek shaded, b) KU = Kaputone Creek unplanted, c) KG = Kaputone Creek grassland.

The channel at KS was the widest and deepest among the three Kaputone Creek sampling sites, while the channel at KU was the narrowest (Table 4.3). The river bed at KS was covered with sediment (approximately 0.20 m–0.25 m deep). The river bed at KU was covered with small stones, while the river bed at KG was covered with silt and sand. The sub-catchment land use at the Kaputone Creek sites was mainly built-up areas (83%–100%) followed by forestry (11%–17%) and pastoral land (1%) (Table 4.3).

Figure 4.6 Riparian vegetation condition at KS (Kaputone Creek shaded site)

Figure 4.7 Riparian vegetation condition at KU (Kaputone Creek unplanted site)

Figure 4.8 Riparian vegetation condition at KG (Kaputone Creek grassland site)

4.5 Site characteristics at Styx River sites

Among the three sampling sites at the Styx River, STG had the widest riparian planting area on both side of the banks compared to STS and STU (Table 4.4). At STS, the riparian vegetation condition was covered by mature trees and fully shaded (Figure 4.9). The riparian area at STG was covered by shrubs and some trees (Figure 4.10), and the riparian area at STU consisted of trees and tussock grass (Figure 4.11).

Table 4.4 Site characteristics and the width of riparian plantings area

	STS	STG	STU
Width of riparian plantings area on the true right bank (m)	4.92	45.70	7.6
Width of riparian plantings area on the true left bank (m)	10.08	37.72	0.4
Channel width (m)	4.50	5.30	8.0
Channel depth at the deepest point (m)	0.42	0.25	0.68
Sub-catchment area (ha)	9	44	1336
Crop Land (%)	21	2	8
Pastoral land (%)	68	48	32
Forest (%)	10	25	4
Built-up (%)	1	25	56

Note: the sites are shown from upstream to downstream, a) STS= Styx River shaded, b) STG= Styx River grassland, c) STU= Styx River unplanted.

The channel at STU was the widest and deepest among the three Styx River sampling sites, while the channel at STS was the narrowest (Table 4.4). The deepest area at STU was approximately 1 m. The river bed at STS and STU was mainly covered with silt and sand, while the river bed at STG was covered with small stones, gravel and silt in some area. In general, sub-catchment land-use at the Styx River sites was mainly covered with pastoral land (32 – 68%) followed by built-up areas (1%–56%), forested land (4%– 25%) and crop land (2%–21%) (Table 4.4).

Figure 4.9 Riparian vegetation condition at STS (Styx River shaded site)

Figure 4.10 Riparian vegetation condition at STG (Styx River grassland site)

Figure 4.11 Riparian vegetation condition at STU (Styx River unplanted site)

4.6 Discharge (L/s)

Across all sampling dates at all sites, the lowest discharge was 15.35 L/s at KS, and the highest was 1359 L/s at STU (Table 4.5). The mean discharge value over all sampling sites on all dates ranged from 20.69 L/s to 983.5 L/s.

Table 4.5 Minimum, mean and maximum values for discharge at each sampling sites over the entire sampling period

	Smacks Creek			Kaputone Creek			Styx River		
	SMG	SMU	SMS	KS	KU	KG	STS	STG	STU
Minimum	31.41	50.48	59.67	15.35	16.54	23.26	290.95	311.8	847.5
Mean	37.68	61.33	81.17	20.69	22.80	26.06	304.01	437.0	983.5
Maximum	63.97	72.52	111.47	27.36	29.76	32.49	315.42	530.9	1359

Note: discharge for each sampling site is shown from upstream to downstream. The mean discharge for the Styx River shaded sites (STU) was calculated based on only seven sampling date as the first date of sampling was not available.

Overall, the lowest mean discharge values were found at the Kaputone Creek sites, while the highest value was measured at the Styx River sites (Figure 4.12).

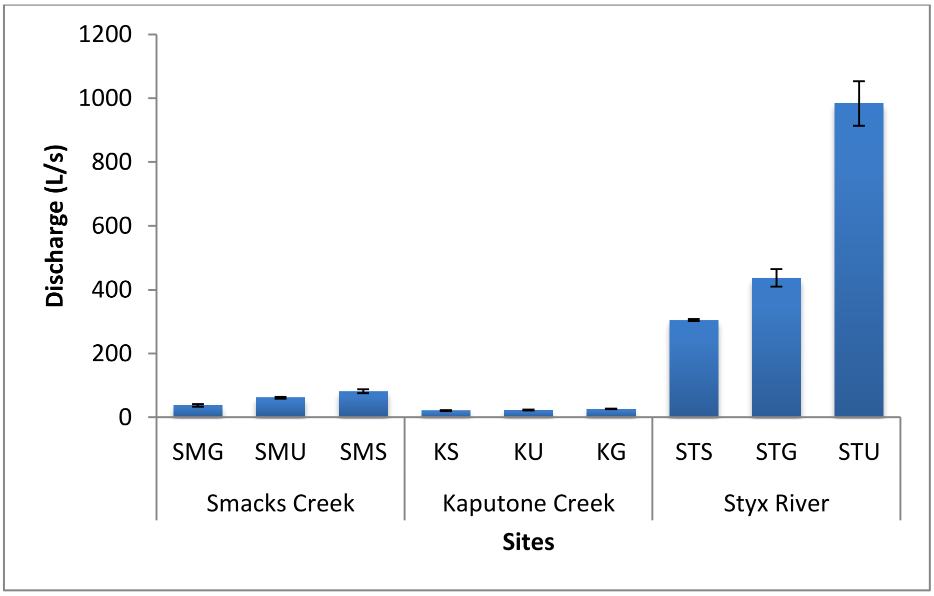

Figure 4.12 Average discharge between sampling sites

4.7 Water quality versus different riparian vegetation composition and upstream and sub-catchment contributing land uses

4.7.1 pH

Across all sampling dates at all sites, the lowest pH was 6.30 at SMG, and the highest pH level was 7.61 at STU (See in Table 1, Appendix A). The mean pH over all sampling sites on all dates ranged between 6.35 and 7.33. The mean pH was found to increase in the downstream direction at both Smacks Creek and the Styx River (Figure 4.13).

There was found to be a significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and $P = 0.008$) or sites ($P = <0.001$) on pH. The pairwise comparison of Kruskal-Wallis test for trearment-1 showed that there was a significant difference between category-1 (grassland with >20-40 m width) and category-3 (unplanted riparian with 0-5 m width) ($P = 0.010$).

Also, there was found to be a significant effect of treatment-2 (different riparian width with different land uses influence) (DF = 2 and $P = 0.006$) on pH. The pairwise comparison of Kruskal-Wallis test for treartmen-2 showed that there was a significant difference ($P = 0.005$) between group-2 (>5 m riparian width predominantly built-up/pastoral land uses) and group-1 (0-5 m width predominantly built-up area).

However, there was no significant effect from rain events ($P = 0.074$) on pH. Also, there was no significant relationship with catchment contributing land use percentages (P = 0.7 for crop land, P = 0.468 for pasture, P = 0.383 for forested land, P = 0.405 for built-up area) and pH.

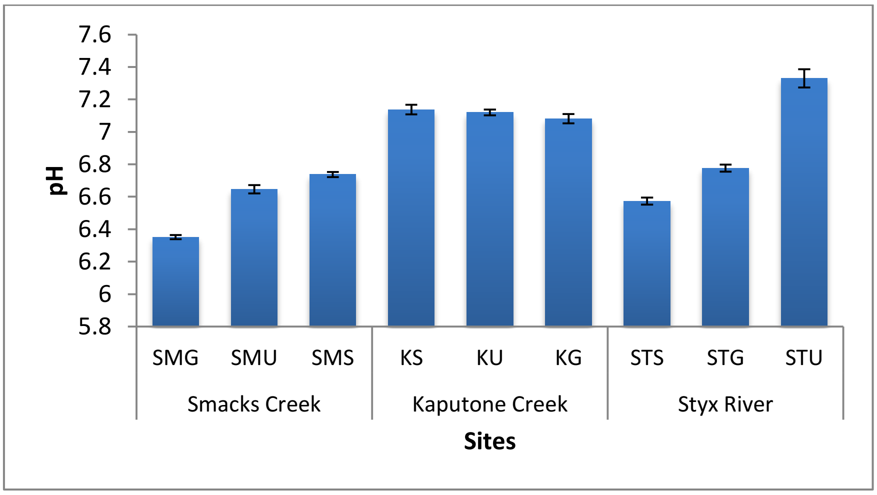

Figure 4.13 Average pH at different riparian vegetation composition

4.7.2 Water temperature

Across all sampling dates at all sites, the lowest water temperature was 9.3 °C at KU, and the highest water temperature was 14.4 °C at KS (See in Table 1, Appendix A). The mean water temperature over all sampling sites on all dates ranged between 11.4 °C and 12.9 °C. The mean water temperature decreased downstream at Kaputone Creek and the Styx River sites (Figure 4.14). Overall, the mean water temperature at the Kaputone Creek sites was slightly lower than the other sites.

There was no significant effect of treatment-1 (different riparian vegetation compositions) (DF=2 and $P = 0.345$) or treatment-2 (different riparian width with different land uses influence) (DF = 2 and $P = 0.357$) or sites ($P = 0.06$) on water temperature. However, there was found to be a significant effect rain events ($P = 0.013$) on water temperature.

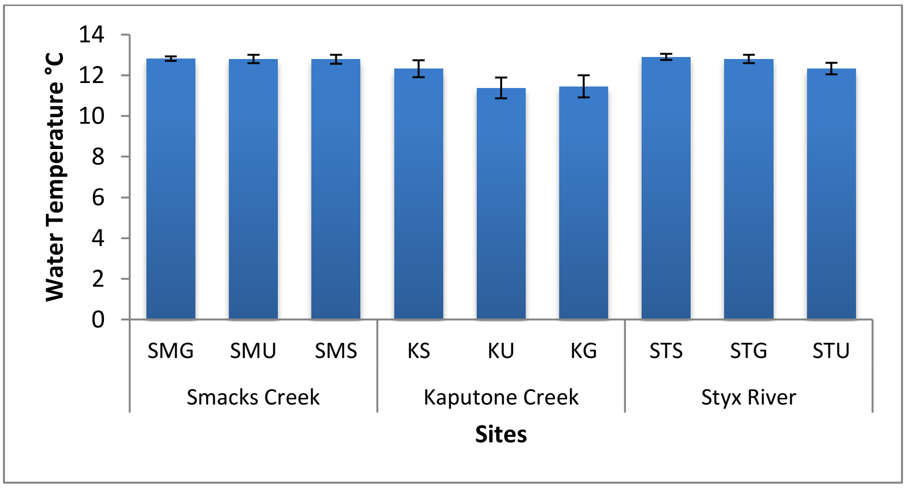

Figure 4.14 Average water temperature at different riparian vegetation composition

There was no significant relationship between catchment contributing land use percentage ($P = 0.388$ for crop land, $P = 0.204$ for pasture, $P = 0.824$ for forest and $P = 0.149$ for built-up area) and water temperature.

4.7.3 Dissolved oxygen

Across all sampling dates at all sites, the lowest dissolved oxygen (DO) was 3.14 mg/L at SMG, and the highest level was 10.49 mg/L at KG (See in Table 1, Appendix A). The mean dissolved oxygen across all the sampling sites on all dates ranged between 3.41 mg/L and 9.74 mg/L. The mean dissolved oxygen showed an increasing trend downstream at all sites

(Figure 4.15). Overall, the highest dissolved oxygen values were found at the Kaputone Creek sites.

There was found to be a significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and P = 0.025) or sites (P = <0.001) on the concentration of dissolved oxygen. The pairwise comparison of Kruskal-Wallis test for treatment-1 showed that there was a significant difference between category-2 (shaded riparian with >5-20 m width) and category-3 (unplanted riparian with 0-5 m width) (P = 0.046).

Also, there was found to be a significant effect of treatment-2 (different riparian width with different land uses influence) (DF = 2 and P = 0.014) on dissolved oxygen. The pairwise comparison of Kruskal-Wallis test for treatment-2 showed that there was a significant difference between group-2 (>5 m riparian width with built-up/pastoral land uses) and group-1 (0-5 m width on both side of the bank with built-up influence) (P = 0.011).

However, there was no significant effect from rain events (P = 0.753) on dissolved oxygen. There was also no significant relationship between sub-catchment land use (P = 0.793 for crop land, P = 0.701 for pasture, P = 0.631 for forest and P = 0.649 for built-up area) and dissolved oxygen.

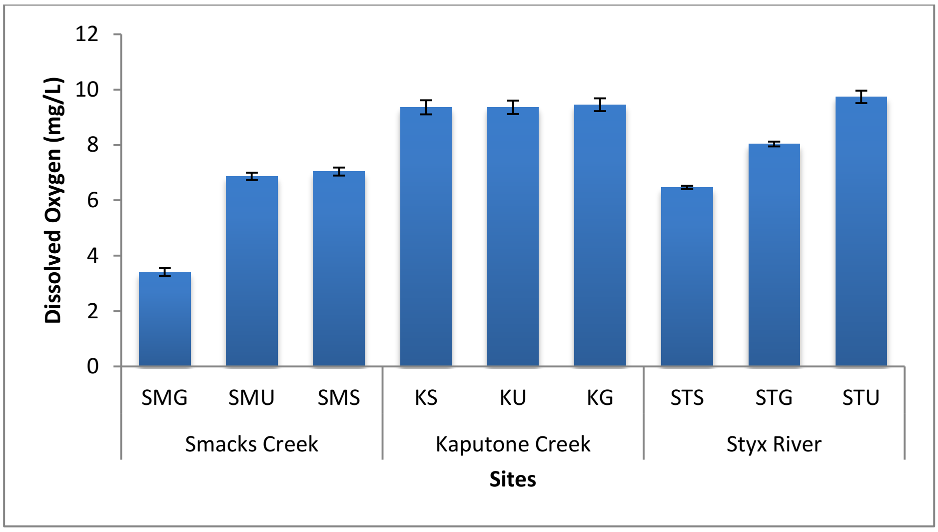

Figure 4.15 Average dissolved oxygen at different riparian vegetation composition

4.7.4 Conductivity

Across all sampling dates at all sites, the lowest conductivity was 108.2 μS/cm at STS, and the highest conductivity was 152.8 μS/cm at KG (See in Table 1, Appendix A)). The mean conductivity over all sampling sites on all dates ranged between 108.5 μS/cm and 137.9 μS/cm. The mean conductivity showed an increasing trend downstream at Kaputone Creek and in the Styx River sites (Figure 4.16). Overall, the highest conductivity was found at the Kaputone Creek sites.

There was found to be a significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and $P = 0.002$) or sites ($P = <0.001$) on conductivity. The pairwise comparison of Kruskal-Wallis test for treatment-1 showed that there was a significant difference between category-2 (shaded riparian with >5-20 m width) and category-3 (unplanted riparian with 0-5 m width) ($P = 0.001$).

Also, there was found to be a significant effect of treatment-2 (different riparian width with different land uses influence) (DF = 2 and $P = <0.001$). The pairwise comparison of Kruskal-Wallis test for trearment-2 showed that there was a significant difference between group-3 (>5 m riparian width with solely pastoral land uses influence) and group-1 (0-5 m width on both side of the bank with built-up influence) ($P = 0.003$). However, there was no significant effect of rain events ($P = 0.921$) on conductivity.

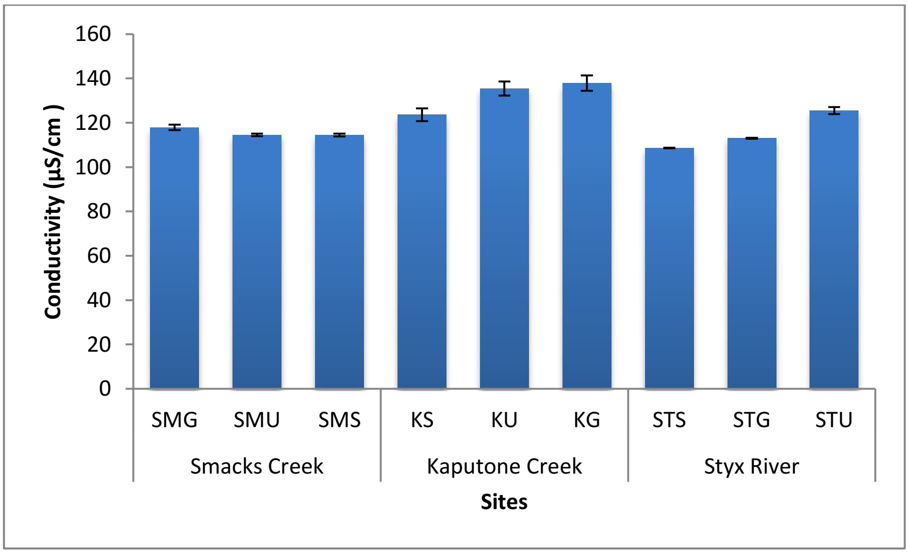

Figure 4.16 Average conductivity at different riparian vegetation composition

Although there was no significant relationship between the percentage of some sub-catchment land uses ($P = 0.243$ for crop land, $P = 0.628$ for forest and $P = 0.057$ for built- up areas) and conductivity, the percentage of pastoral land ($P = 0.042$) showed a significant relationship with conductivity level. The percentage of built-up area had a positive relationship with conductivity at Smacks Creek and Styx River sites (Figure 4.17).

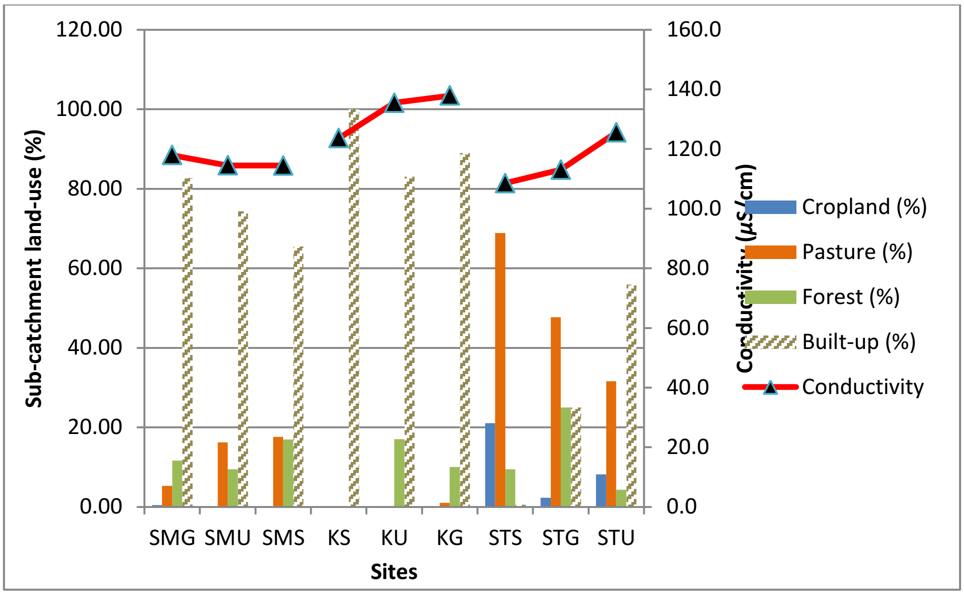

Figure 4.17 The relationship between average conductivity and different sub-catchment land use percentages

4.7.5 Turbidity

Across all sampling dates at all sites, the lowest turbidity was 0 NTU at SMG, and the highest was 3.34 NTUs at KU (See in Table 1, Appendix A). The mean turbidity over all sampling sites on all dates ranged between 0.02 NTUs and 1.74 NTUs. The mean turbidity values increased downstream at the Styx River sites. The highest turbidity was found at the unplanted sites (SMU, KU and STU) (Figure 4.18). Overall, the highest turbidity among sites was found at Kaputone Creek sites.

There was found to be a significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and $P = 0.022$) or sites ($P = <0.001$), or rain events ($P = 0.017$) on turbidity. The pairwise comparison of Kruskal-Wallis test for treatment-1 showed that there was a significant difference between category-2 (shaded riparian with >5-20 m width) and category-3 (unplanted riparian with 0-5 m width) ($P = 0.020$).

Also, there was found to be a significant effect of treatment-2 (different riparian width with different land uses influence) (DF = 2 and $P = 0.004$). The pairwise comparison of Kruskal-Wallis test for treatment-2 showed that there was a significant difference between group-2 (>5 m riparian width with built-up/pastoral land uses) and group-1 (0-5 m width on both side of the bank with built-up influence) ($P = 0.003$).

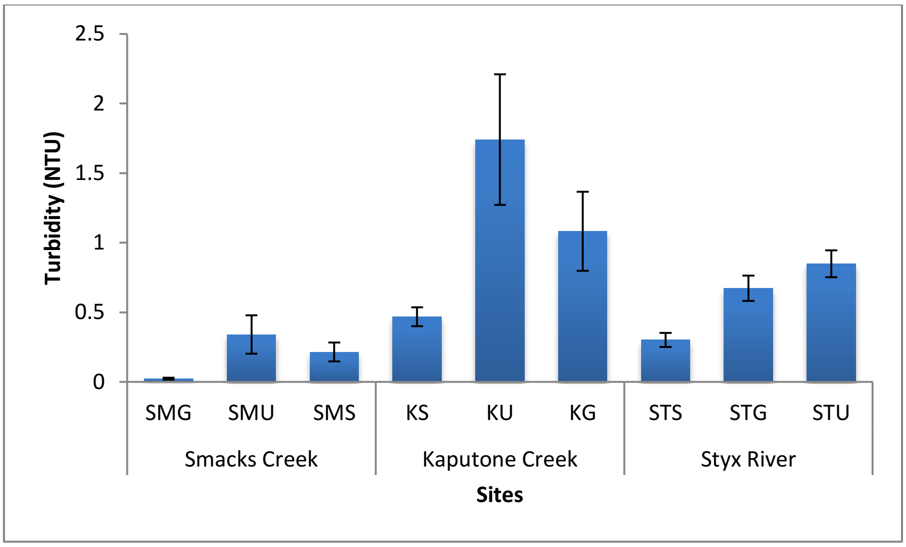

Figure 4.18 Average turbidity at different riparian vegetation composition

There was no significant relationship between the percentage of sub-catchment contributing land uses (*P* = 0.645 for crop land, *P* = 0.463 for pasture, *P* = 0.67 for forest and *P* = 0.584 for built-up area) and turbidity.

4.8 Contaminants versus different riparian vegetation composition and upstream and sub-catchment contributing land uses

4.8.1 Total suspended solid

Across all sampling dates at all sites, the lowest total suspended solid concentration was 0 mg/L at SMG, and the highest was 13 mg/L at KU (See in Table 2, Appendix A). The mean total suspended solids over all the sampling sites on all dates ranged between 0.250 mg/L and 4.690 mg/L. The mean total suspended solid concentration increased downstream at Smacks Creek sites (Figure 4.19). Overall, Smacks Creek sites showed the lowest total suspended solids values.

There was no significant effect of treatment-1 (different riparian vegetation compositions and average riparian width) (DF = 2 and *P* = 0.874) or rain events (*P* = 0.264) on total suspended solids. However, there was a significant effects of sites (p = < 0.001) on total suspended solids. Also, there was found to be a significant effect of treatment-2 (different riparian width with different land uses influence) (DF = 2 and *P* = 0.014). The pairwise comparison of Kruskal-Wallis test for treatment-2 showed that there was a significant difference between group-2 (>5 m riparian width with built-up/pastoral land uses) and group-3 (>5 m riparian width with solely pastoral influence) (*P* = 0.011).

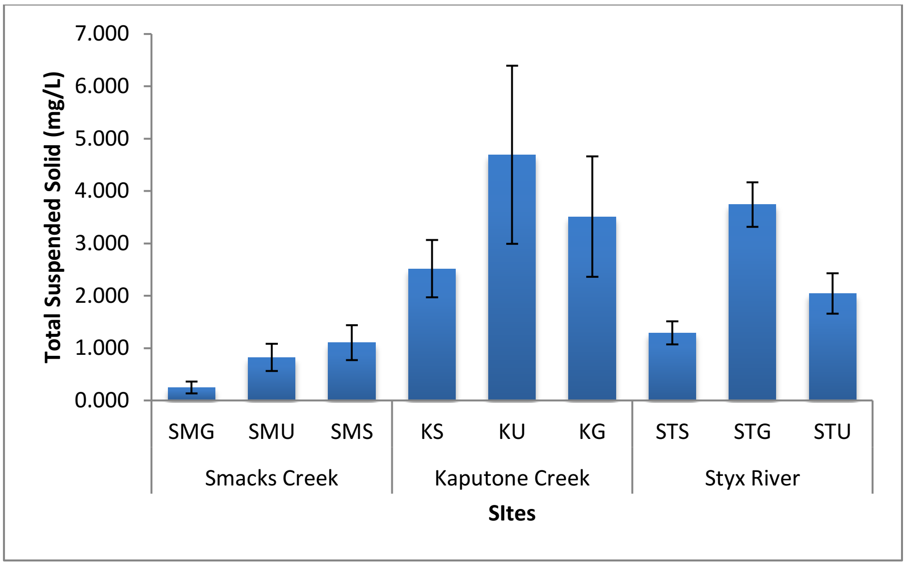

Figure 4.19 Average total suspended solids at different riparian vegetation composition

There was no significant relationship between the percentage of catchment contributing land use (*P* = 0.696 for crop land, *P* = 0.933 for pasture, *P* = 0.182 for forest and *P* = 0.819 for built up areas) and total suspended solids.

4.8.2 Dissolved reactive phosphorus

Across all sampling dates at all sites, the lowest dissolved reactive phosphorus was 0.015 mg/L at SMS, and the highest DRP was 0.086 mg/L at KU (See in Table 2, Appendix A). The mean dissolved reactive phosphorus over all sampling sites on all dates ranged between 0.023 mg/L and 0.057 mg/L. The mean value of dissolved reactive phosphorus concentration increased downstream in Kaputone Creek and the Styx River sites (Figure 4.20). Overall, Kaputone Creek sites were found to have the highest mean DRP.

There was found to be a significant effect of treatment-1 (different riparian vegetation compositions and average riparian width) (DF = 2 and *P* = 0.038) or sites (*P* = <0.001) on dissolved reactive phosphorus. The pairwise comparison of Kruskal-Wallis test for treatment-1 showed that there was a significant difference between category-2 (shaded riparian with >5-20 m width) and category-3 (unplanted riparian with 0-5 m width) (*P* = 0.032).

However, there no significant effect of rain event (*P*= 0.519) or treatment-2 (different riparian width with different land uses influence) (DF = 2 and *P* = 0.069) on dissolved reactive phosphorus.

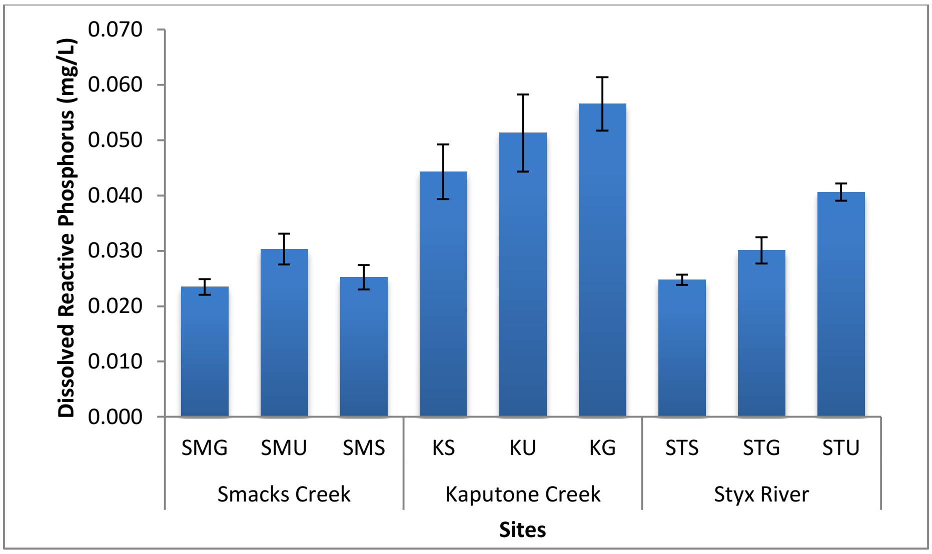

Figure 4.20 Average dissolved reactive phosphorus at different riparian vegetation composition

There was also no significant relationship between the percentage of sub-catchment contributing land use (*P* = 0.407 for crop land, *P* = 0.137 for pasture, *P* = 0.528 for forest and *P* = 0.146 for built-up areas) and dissolved reactive phosphorus.

4.8.3 Total phosphorus

Across all sampling dates at all sites, the lowest total phosphorus (TP) was 0.020 mg/L at SMG, and the highest was 0.099 mg/L at KG (See in Table 2, Appendix A). The mean total phosphorus at all sampling sites on all dates ranged between 0.026 mg/L and 0.062 mg/L. The mean total phosphorus increased downstream at Kaputone Creek and the Styx River sites (Figure 4.21). Overall, Kaputone Creek sites showed the highest mean total phosphorus value.

There was a significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and *P* = 0.025) or sites (*P* = <0.001) on total phosphorus. The pairwise comparison of Kruskal-Wallis test for treatment-1 showed that there was a significant difference between category-2 (shaded riparian with >5-20 m width) and category-3 (unplanted riparian with 0-5 m width) (*P* = 0.025).

Also, there was found to be a significant effect of treatment-2 (different riparian width with different land uses influence) (DF = 2 and *P* = 0.030) on total phosphorus. The pairwise comparison of Kruskal-Wallis test for treatment-2 showed that there was a significant difference between group-2 (>5 m riparian width with built-up/pastoral land uses) and group-1 (0-5 m width on both side of the bank with built-up influence) (P = 0.038).

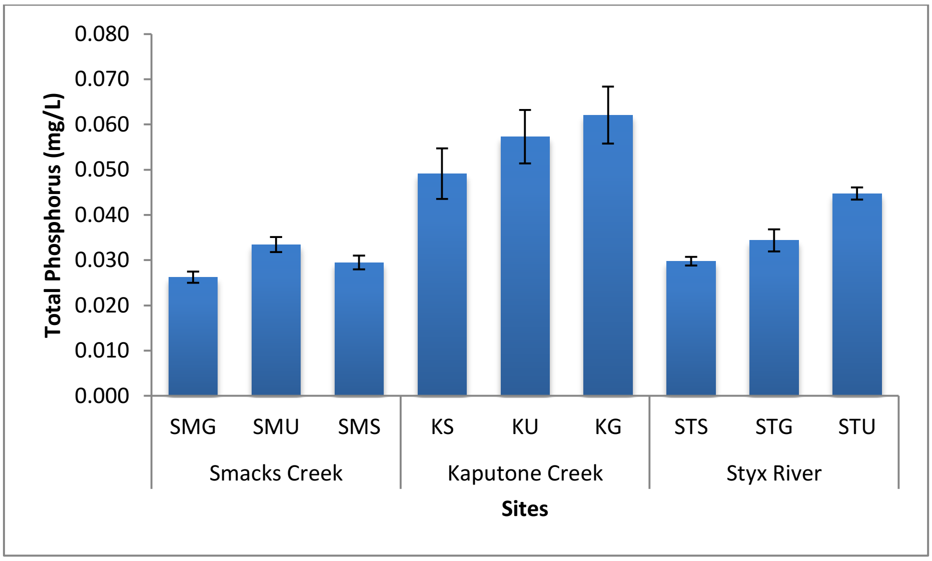

Figure 4.21 Average total phosphorus at different riparian vegetation composition

However, there was no significant effect from rain events (P = 0.249) on total phosphorus. Also, there was no significant relationship between the percentage of sub-catchment contributing land uses (P = 0.369 for crop land, P = 0.137 for pasture, P = 0.668 for forest and P = 0.157 for built-up areas) and total phosphorus.

4.8.4 Total dissolved phosphorus

Across all sampling dates at all sites, the lowest total dissolved phosphorus (TDP) was 0.020 mg/L at SMG, and the highest was 0.088 mg/L at KU (See in Table 2, Appendix A). The mean total dissolved phosphorus over all sampling sites on all dates ranged between 0.026 mg/L and 0.059 mg/L. The mean total dissolved phosphorus increased downstream in Kaputone Creek and the Styx River sites (Figure 4.22). Overall, Kaputone Creek sites were shown to have the highest mean total dissolved phosphorus values.

There was found to be a significant effect of treatment-1 (different riparian vegetation compositions and riparian width) (DF = 2 and P = 0.043) or sites (P = <0.001) on total dissolved phosphorus. The pairwise comparison of Kruskal-Wallis test for treatment-1 showed that there was a significant difference between category-2 (shaded riparian with >5-20 m width) and category-3 (unplanted riparian with 0-5 m width) (P = 0.044).

Also, there was found to be a significant effect of treatment-2 (different riparian width with different land uses influence) (DF = 2 and P = 0.047) on total dissolved phosphorus. The pairwise comparison of Kruskal-Wallis test for treatment-2 showed that there was a significant difference between group-2 (>5 m riparian width with built-up/pastoral land uses) and group-1 (0-5 m width on both side of the bank with built-up influence) (P = 0.052).

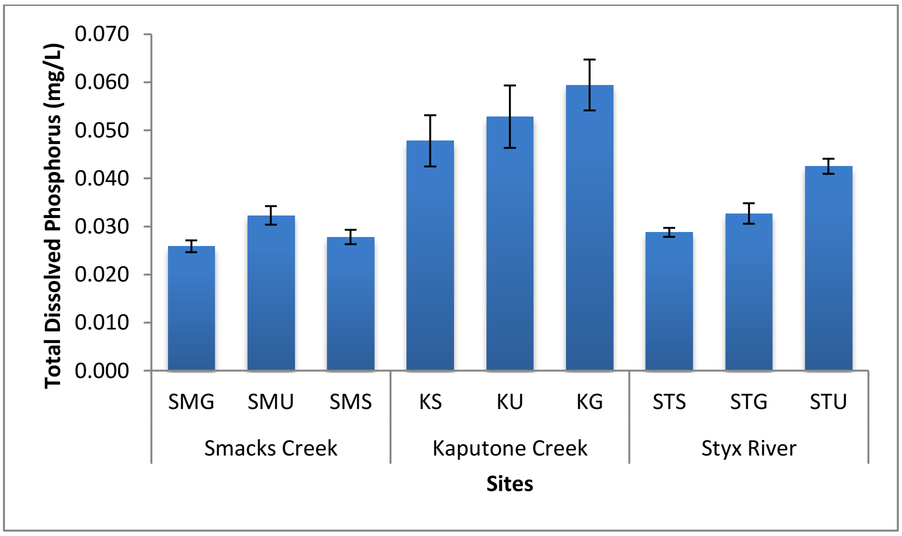

Figure 4.22 Average total dissolved phosphorus at different riparian vegetation composition

However, there was no significant effect of rain events (P = 0.489) on total dissolved phosphorus. There was also no significant relationship between the percentage of sub-catchment contributing land uses (P = 0.32 for crop land, P = 0.111 for pasture, P = 0.621 for forest and P = 0.124 for built-up area) and total dissolved phosphorus.

4.8.5 Particulate phosphorus

Across all sampling dates at all sites, the lowest particulate phosphorus (PP) was 0.000 mg/L at SMG, SMU, STS and STG, and the highest was 0.0241 mg/L at KU (See in Table 2, Appendix A). The mean particulate phosphorus over all the sampling sites on all dates ranged between 0.0004 and 0.0044 mg/L. The mean particulate phosphorus increased downstream at Smacks Creek and the Styx River sites (Figure 4.23). Overall, the shaded sites of Styx River and Kaputone Creek showed the lowest mean particulate phosphorus values, while the highest showed at the unplanted sites of the Styx River and Kaputone Creek (Figure 4.23).

There was no significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and P = 0.239) or treatment-2 (different riparian width with different land uses influence) (DF = 2 and P = 0.258) on particulate phosphorus concentrations.

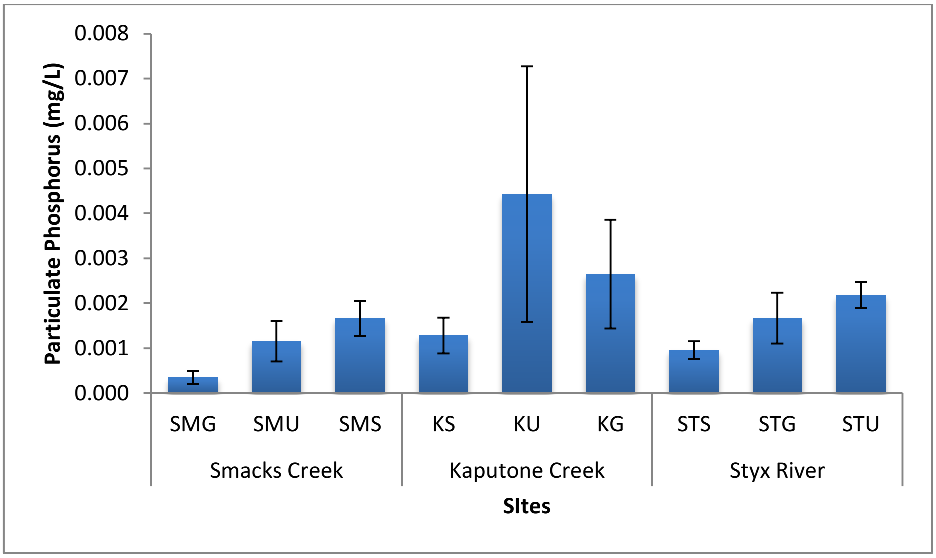

Figure 4.23 Average particulate phosphorus at different riparian vegetation composition

There was also no significant relationship between the percentage of sub-catchment contributing land uses (*P* = 0.738 for crop land, *P* = 0.919 for pasture, *P* = 0.685 for forest and *P* = 0.924 for built-up areas) and particulate phosphorus. However, there was a significant effect of sites (*P* = 0.025) and rain events (*P* = 0.073) on particulate phosphorus concentrations.

4.8.6 Nitrate

Across all sampling dates at all sites, the lowest nitrate concentration was 0.087 mg/L at STS, and the highest was 1.083 mg/L at KS (See in Table 2, Appendix A). The mean nitrate over all sampling sites on all dates ranged from 0.222 to 0.921 mg/L. The mean nitrate concentration slightly increased downstream at Smacks Creek and the Styx River sites, and decreased downstream at Kaputone Creek sites (Figure 4.24). Overall, Kaputone Creek sites were found to have the highest nitrate level, while the Styx River sites showed the lowest nitrate levels.

There was no significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and *P* = 0.997) or treatment-2 (different riparian width with different land uses influence) (DF = 2 and *P* = 0.651) on nitrate concentration. However, there was a significant effect of sites (*P* = <0.001) or rain events (*P* = 0.022) on nitrate concentrations.

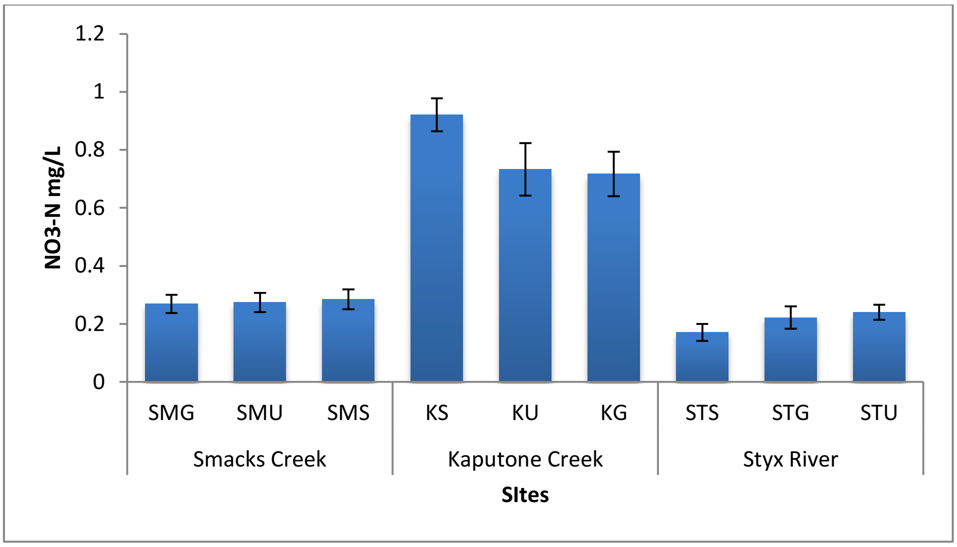

Figure 4.24 Average nitrate at different riparian vegetation composition

There was no significant relationship between the percentage of cropland and forested land in sub-catchment ($P = 0.207$ for crop land and $P = 0.355$ for forest) and nitrate concentrations, but the percentage of pasture and built-up areas ($P = 0.032$ for pasture and $P = 0.031$ for built-up area) showed a significant relationship with nitrate concentrations (Figure 4.25).

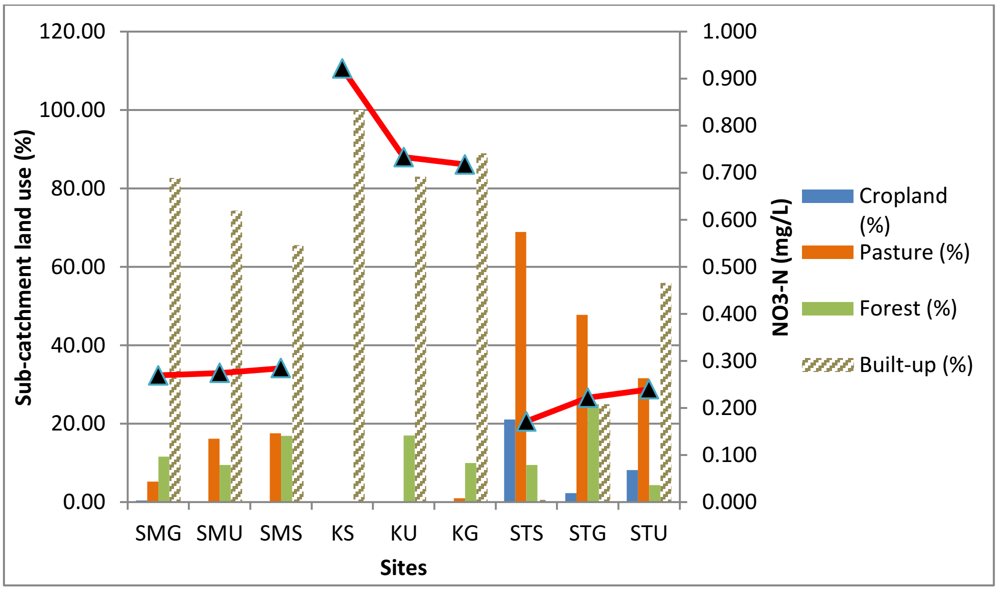

Figure 4.25 The relationship between average nitrate and different sub-catchment land uses percentages

4.8.7 Total nitrogen

Across all sampling dates at all sites, the lowest concentration of total nitrogen (TN) was 0.252 mg/L at STS, and the highest was 2.132 mg/L at KS (See in Table 2, Appendix A). The mean total nitrogen over all sampling sites on all dates ranged from 0.321 mg/L to 1.841 mg/L. The mean total nitrogen values decreased downstream at Smacks Creek and Kaputone Creek sites (Figure 4.26), and increased downstream at the Styx River sites. Overall, Kaputone Creek sites showed the highest mean total nitrogen values.

There was no significant effect from treatment-1 (different riparian vegetation compositions) (DF = 2 and P = 0.799) or treatment-2 (different riparian width with different land uses influence) (DF = 2 and P = 0.226) or rain events (P = 0.776) on total nitrogen concentrations. However, there was a significant effect of sites (P = <0.001) on total nitrogen concentration.

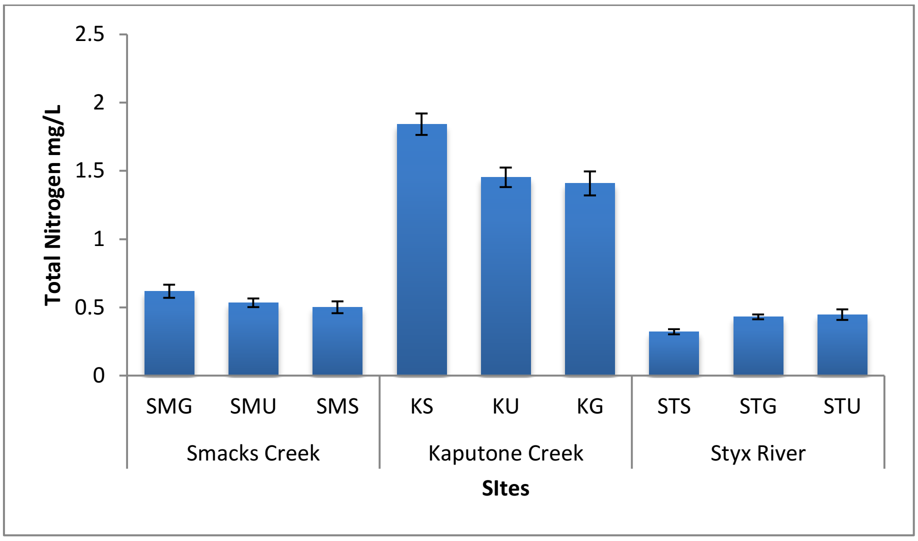

Figure 4.26 Average total nitrogen at different riparian vegetation composition

There was no significant relationship between the percentage of cropland and forested land in sub-catchments (P = 0.199 for crop land and P = 0.345 for forest) and total nitrogen concentrations, but the percentage of pasture and built-up areas (P = 0.028 for pasture and P = 0.026 for built-up areas) showed a significant relationship with total nitrogen concentrations (Figure 4.27).

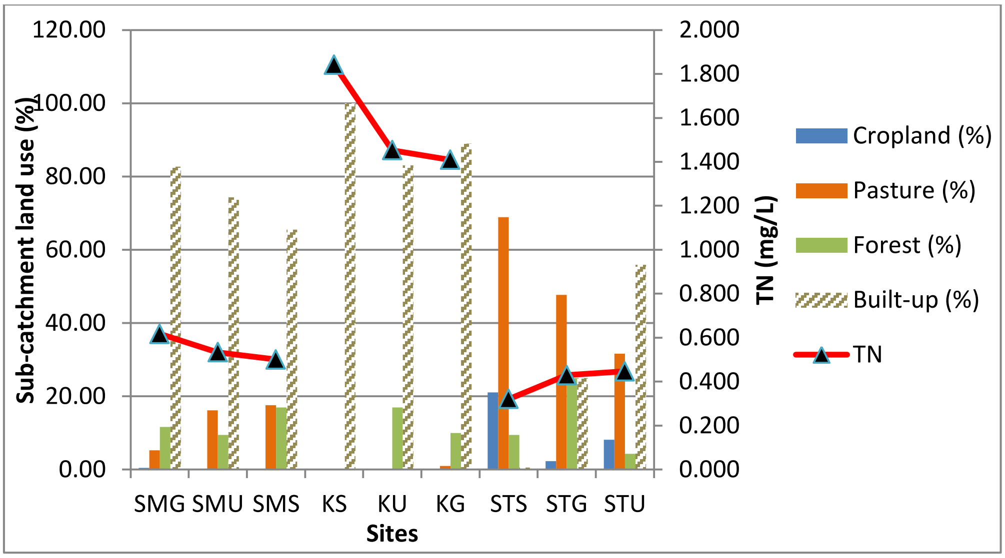

Figure 4.27 The relationship between average total nitrogen and different sub-catchment land uses percentage

4.8.8 Total dissolved nitrogen

Across all sampling dates at all sites, the lowest total dissolved nitrogen level was 0.230 mg/L at SMS, and the highest was 1.905 mg/L at KS (See in Table 2, Appendix A). The mean total dissolved nitrogen over all the sampling sites on all dates ranged from 0.311 to 1.761 mg/L. The mean total dissolved nitrogen level showed a decreasing trend downstream at Smacks Creek and Kaputone Creek sites, but this was reversed at the Styx River sites (Figure 4.28). Overall, Kaputone Creek sites showed the highest mean total dissolved nitrogen values.

There was no significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and $P = 0.736$) or treatment-2 (different riparian width with different land uses influence) (DF = 2 and $P = 0.324$) or rain events ($P = 0.504$) on total dissolved nitrogen concentrations. However, there was a significant effect of sites (P = <0.001)) on total dissolved nitrogen concentrations.

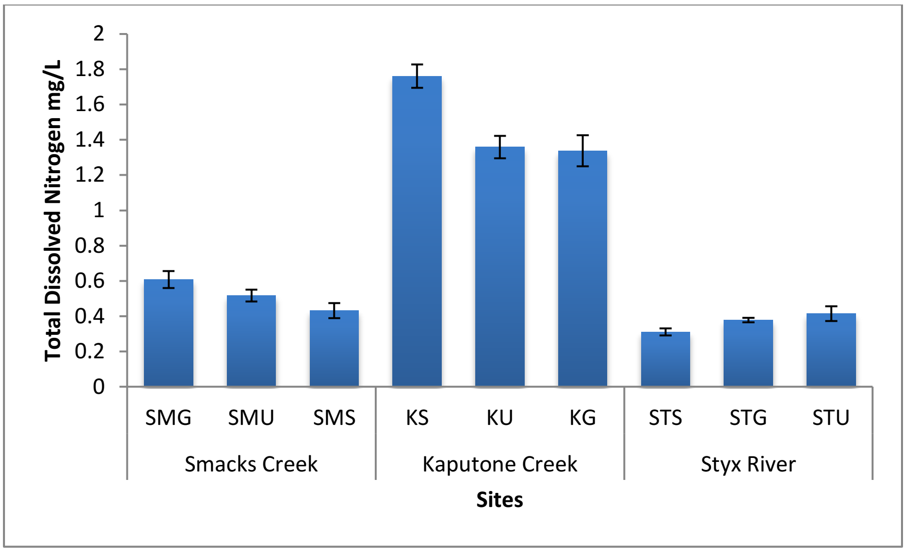

Figure 4.28 Average total dissolved nitrogen at different riparian vegetation composition

There was no significant relationship between the percentage of cropland and forested land in the sub-catchment (*P* = 0.21 for crop land and *P* = 0.312 for forest) and total dissolved nitrogen concentrations, but the percentage of pasture and built-up area (*P* = 0.028 for pasture and *P* = 0.025 for built-up area) showed a significant relationship with total dissolved nitrogen concentrations (Figure 4.29).

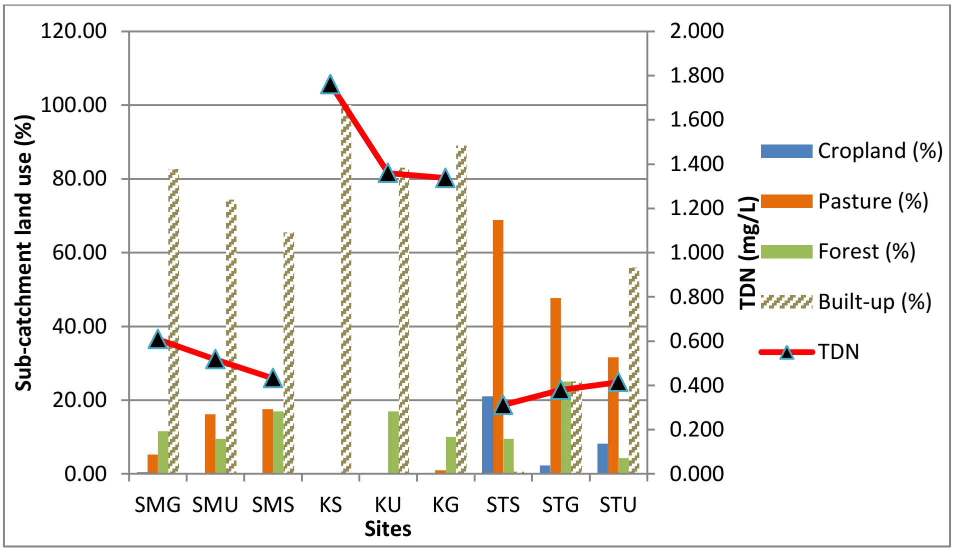

Figure 4.29 The relationship between average total dissolved nitrogen and different sub-catchment land uses percentage

4.8.9 Particulate Nitrogen

Across all sampling dates at all sites, the lowest particulate nitrogen level was 0 mg/L at SMG, STS and STG, and the highest was 0.302 mg/L at STU (See in Table 2, Appendix A). The mean particulate nitrogen over all sampling sites on all dates ranged from 0.008 to 0.094

mg/L. The mean particulate nitrogen level increased downstream at Smacks Creek sites (Figure 4.30). Overall, the highest mean particulate nitrogen values were found at Kaputone Creek sites.

There was no significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and *P* = 0.955) or treatment-2 (different riparian width with different land uses influence) (DF = 2 and *P* = 0.5728) or rain events (*P* = 0.527) on particulate nitrogen concentrations. However, there was a significant effect of sites (*P* = <0.001) on particulate nitrogen concentration.

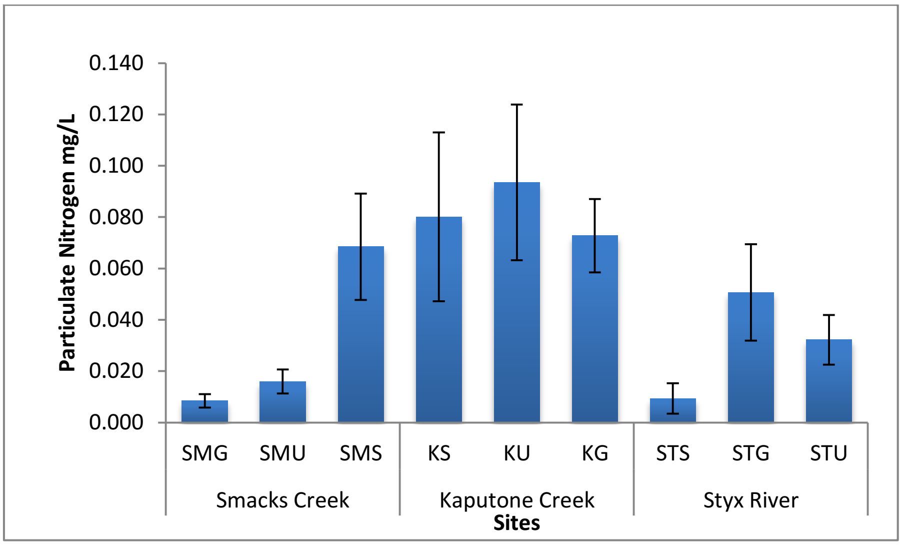

Figure 4.30 Average particulate nitrogen at different riparian vegetation composition

There was no significant relationship between the percentage of sub-catchment contributing land uses (*P* = 0.155 for crop land, *P* = 0.139 for pasture, *P* = 0.747 for forest and *P* = 0.186 for built-up areas) and particulate nitrogen.

4.9 Sediment and nutrients fluxes versus different riparian vegetation composition, and upstream and sub-catchment contributing land uses

4.9.1 Total suspended solid flux

Across all sampling dates at all sites, the lowest total suspended solid flux was 0 kg/day at SMG, and the highest was 346.8 kg/day at STU (See in Table 3, Appendix A). The mean total suspended solid flux over all sampling sites on all dates ranged between 0.76 kg/day and 171.5 kg/day.

There was no significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and *P* = 0.697) or rain events (*P* = 0.236) on total suspended solid fluxes. Also, there was no significant relationship between the percentage of sub-catchment land uses (*P* = 0.078 for crop land, *P* = 0.066 for pasture, *P* = 0.062 for forest and *P* = 0.056 for built-up area) and total suspended solid fluxes.

However, there was a significant effect of the sites (*P* = <0.001) on total suspended solid fluxes. Also, there was found to be a significant effect of treatment-2 (different riparian width with different land uses influence) (DF = 2 and *P* = <0.001) on total suspended solid fluxes. The pairwise comparison of Kruskal-Wallis test for treatment-2 showed that there was a significant difference between group-2 (>5 m riparian width with built-up/pastoral land uses) and group-1 (0-5 m width on both side of the bank with built-up influence) (*P* = 0.023).

Also the discharge showed a positive relationship with total suspended solid fluxes: when discharge increased, total suspended solid flux also increased. The ratio of TSS and discharge showed highest at Kaputone Creek sites. It indicated that discharge at Kaputone Creek sites carried the highest concentrations of TSS (Figure 4.31).

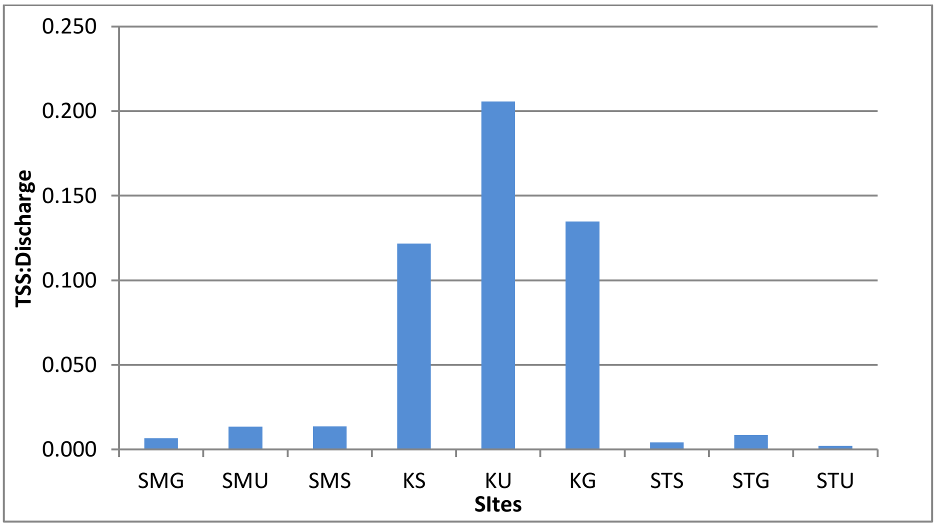

Figure 4.31 The ratio of TSS and discharge at different riparian vegetation composition

4.9.2 Total phosphorus flux

Across all sampling dates at all sites, the lowest total phosphorus flux (TP) was 0.06 kg/day at SMG, KS and KU, and the highest was 4.91 kg/day at STU (See in Table 3, Appendix A). The mean total phosphorus flux at all sampling sites on all dates ranged between 0.09 and 3.75 kg/day. Total phosphorus fluxes consisted of over 90% of TDP. The mean total phosphorus flux increased downstream at all sites.

There was no significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and $P = 0.488$) or rain events ($P = 0.142$) on total phosphorus fluxes. Also, there was no significant relationship between the percentage of sub-catchment land use *(P* = 0.059 for crop land, $P = 0.089$ for pasture, $P = 0.103$ for forest and $P = 0.121$ for built-up area) and total phosphorus fluxes.

However, there was a significant effect of the sites ($P = <0.001$) on total phosphorus fluxes. However, there was a significant effect of the sites ($P = <0.001$) on total suspended solid fluxes. Also, there was found to be a significant effect of treatment-2 (different riparian width with different land uses influence) (DF = 2 and $P = 0.017$) on total phosphorus fluxes. The pairwise comparison of Kruskal-Wallis test for treatment-2 showed that there was a significant difference between group-2 (>5 m riparian width predominantly built-up/pastoral land uses) and group-3 (>5 m width riparian area predominantly pastoral land use) ($P = 0.027$).

Also, and there was also a positive relationship between discharge and total phosphorus flux. The highest ratio of TP and discharge was found at Kaputone Creek sites and the lowest at the Styx River sites (Figure 4.32).

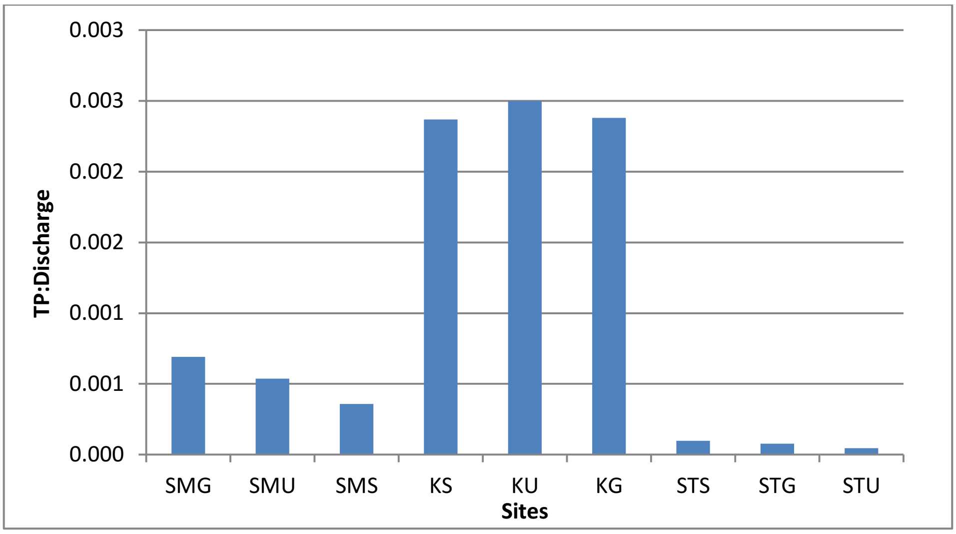

Figure 4.32 The ratio of TP and discharge at different riparian vegetation composition

4.9.3 Total dissolved phosphorus flux

Across all sampling dates at all sites, the lowest dissolved reactive phosphorus (DRP) and total dissolved phosphorus flux (TDP) was 0.05 kg/day at KS, and the highest was 4.67 kg/day at STU (See in Table 3, Appendix A). The mean total dissolved phosphorus flux at all sampling sites on all dates ranged between 0.08 and 3.55 kg/day. Total dissolved phosphorus flux consisted of approximately 90% DRP.

There was no significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and *P* = 0.578) or rain events (*P* = 0.168) on total dissolved phosphorus fluxes. Also, there was no significant relationship between the percentage of sub-catchment land uses (*P* = 0.154 for crop land, *P* = 0.089 for pasture, *P* = 0.103 for forest and *P* = 0.121 for built-up areas) and DRP and TDP fluxes.

However, there was a significant effect of sites (P =< 0.001) on DRP and TDP fluxes. Also, there was found to be a significant effect of treatment-2 (different riparian width with different land uses influence) (DF = 2 and *P* = 0.021) on total dissolved phosphorus fluxes. The pairwise comparison of Kruskal-Wallis test for treatment-2 showed that there was a significant difference between group-2 (>5 m riparian width predominantly built-up/pastoral land uses) and group-3 (>5 m riparian width predominantly pastoral land use) (*P* = 0.025).

Also, there was a positive relationship between discharge and total dissolved phosphorus fluxes. The highest ratio of dissolved reactive phosphorus and discharge, total dissolved phosphorus and discharge was found at Kaputone Creek sites and the lowest at the Styx River sites (Figure 4.33).

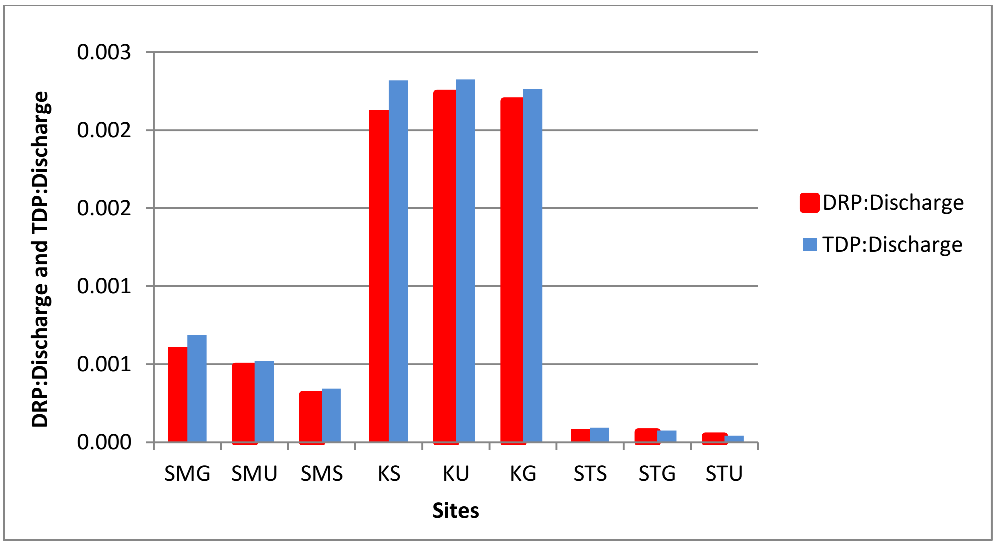

Figure 4.33 The ratio of DRP and discharge, TDP and discharge at different riparian vegetation composition

4.9.4 Particulate phosphorus flux

Across all sampling dates at all sites, the lowest particulate phosphorus flux (PP) was 0.000 kg/day at all sites except STU, and the highest was 0.34 kg/day at STU (See in Table 3, Appendix A). The mean particulate phosphorus fluxes over all sampling sites on all dates ranged between 0.000 and 0.19 kg/day. The mean particulate phosphorus flux increased downstream at all sites.

There was no significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and P = 0.264) or treatment-2 (different riparian width with different land uses influence) (DF = 2 and P = 0.113) or rain events (P = 0.128) on particulate phosphorus fluxes. Also, there was no significant relationship between sub-catchment land use (P = 0.073 for crop land, P = 0.107 for pasture, P = 0.118 for forest and P = 0.132 for built-up area) and particulate phosphorus fluxes.

However, there was a significant effect of sites (P = <0.001) on particulate phosphorus fluxes. Also, there was found to be positive relationship between discharge and particulate phosphorus fluxes. Kaputone Creek sites showed the highest ratio of PP concentration and discharge, and lowest ratio was found at the Styx River sites (Figure 4.34).

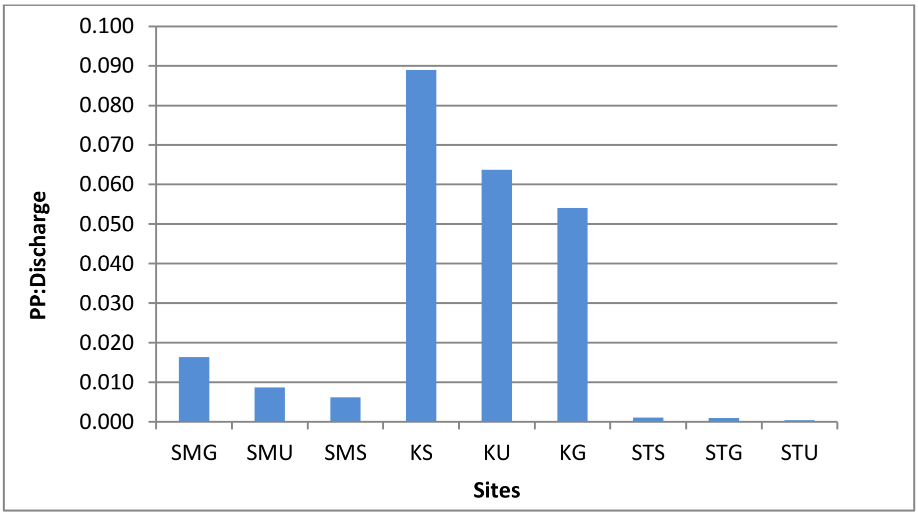

Figure 4.34 The ratio of PP and discharge at different riparian vegetation composition

4.9.5 Total nitrogen flux

Across all sampling dates at all sites, the lowest total nitrogen flux was 1.19 kg/day at SMG, and the highest was 45.96 kg/day at STU (See in Table 3, Appendix A). The mean total nitrogen flux at all sampling sites on all dates ranged from 2.00 kg/day to 36.28 kg/day. Total nitrogen flux consisted of approximately 90% of TDN.

There was no significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and $P = 0.660$) or rain events ($P = 0.530$) on total nitrogen fluxes. There was also no significant relationship between the percentage of sub-catchment land uses ($P = 0.366$ for crop land, $P = 0.227$ for pasture, $P = 0.769$ for forest and $P = 0.301$ for built-up area) and total nitrogen fluxes.

However, there was a significant effect of sites ($P = <0.001$) on total nitrogen fluxes. There was found to be a significant effect of treatment-2 (different riparian width with different land uses influence) (DF = 2 and $P = 0.021$) on total dissolved phosphorus fluxes. The pairwise comparison of Kruskal-Wallis test for treatment-2 showed that there was a significant difference between group-2 (>5 m riparian width predominantly built-up/pastoral land uses) and group-3 (>5 m riparian width predominantly pastoral land use) ($P = 0$).

Also, there was found to be a significant relationship between discharge and total nitrogen fluxes. The highest ratio of TN and discharge was found at Kaputone Creek sites and the lowest at the Styx River sites (Figure 4.35).

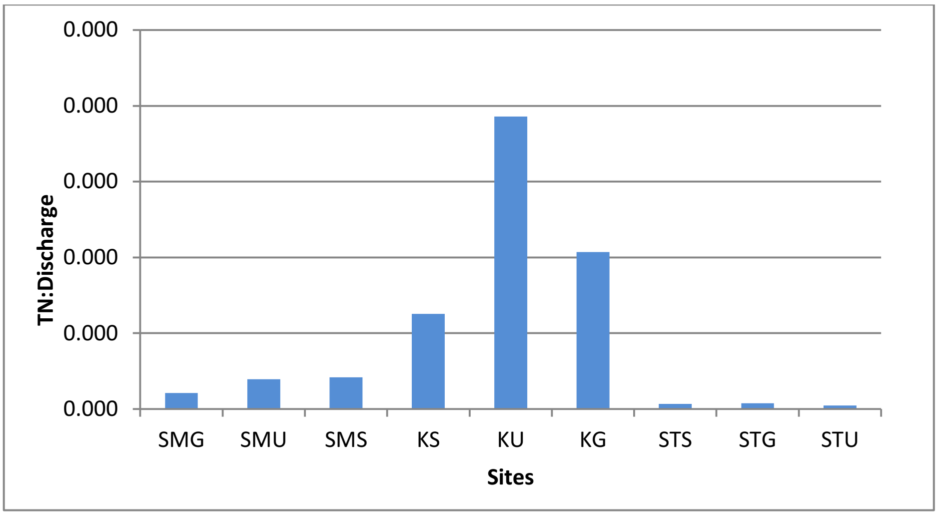

Figure 4.35 The ratio of TN and discharge at different riparian vegetation composition

4.9.6 Total dissolved nitrogen flux

Across all sampling dates at all sites, the lowest total dissolved nitrogen flux was 1.16 kg/day at SMG, and the highest was 44.02 kg/day at STU (See in Table 3, Appendix A). The mean total dissolved nitrogen fluxes at all sampling sites on all dates ranged from 1.97 to 33.19 kg/day. Total dissolved nitrogen flux consisted of 42.3% to 63.8% of NO_3–N.

There was no significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and $P = 0.756$) or rain events ($P = 0.670$) on total dissolved nitrogen fluxes. Also, there was no significant relationship between the percentage of sub-catchment land uses ($P = 0.314$ for crop land, $P = 0.225$ for pasture, $P = 0.731$ for forest and $P = 0.300$ for built-up area) and total dissolved nitrogen fluxes.

However, there was a significant effect of sites ($P = <0.001$) on total dissolved nitrogen fluxes. There was found to be a significant effect of treatment-2 (different riparian width with different land uses influence) (DF = 2 and $P = 0.021$) on total dissolved phosphorus fluxes. The pairwise comparison of Kruskal-Wallis test for treatment-2 showed that there was a significant difference between group-2 (>5 m riparian width predominantly built-up/pastoral land uses) and group-3 (>5 m riparian width predominantly pastoral land use) ($P = 0$).

Also, discharge showed a positive relationship with total dissolved nitrogen fluxes. Figure 4.36 showed that the highest ratio of nitrate (NO_3-N) and discharge, and total dissolved nitrogen and discharge were found at Kaputone Creek sites and the lowest at the Styx River sites.

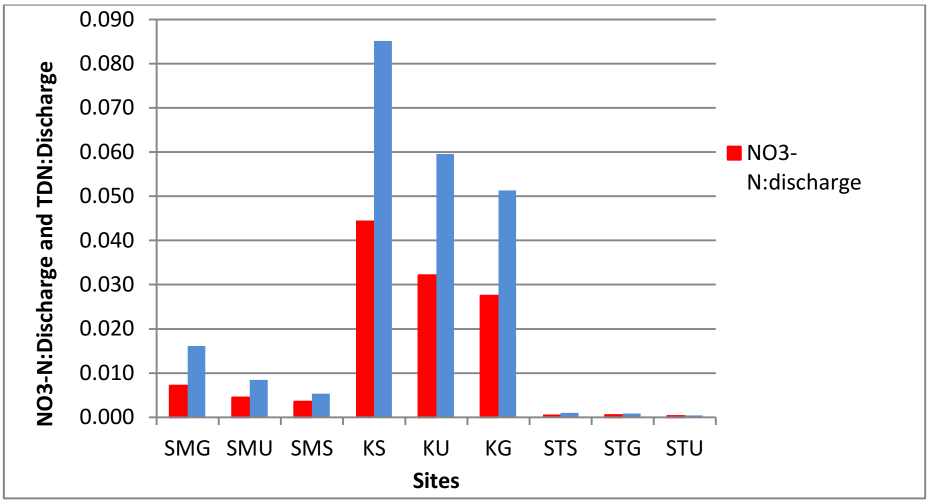

Figure 4.36 The ratio of NO_3-N and discharge, TDN and discharge at different riparian vegetation composition

4.9.7 Particulate nitrogen flux

Across all sampling dates at all sites, the lowest particulate nitrogen flux was 0 kg/day at SMG, STS and STG, and the highest was 0.627 kg/day at STU (See in Table 3, Appendix A).

The mean particulate nitrogen flux at all sampling sites on all dates ranged from 0.03 to 3.09 kg/day. The mean particulate nitrogen flux increased downstream at all sites. Overall, the highest mean particulate nitrogen values were found at Styx River sites.

There was no significant effect of treatment-1 (different riparian vegetation compositions) (DF = 2 and $P = 0.326$) or treatment-2 (different riparian width with different land uses influence) (DF = 2 and $P = 0.021$) or rain events ($P = 0.207$) on particulate nitrogen fluxes. Also, there was no significant relationship between the percentage of sub-catchment land uses ($P = 0.601$ for crop land, $P = 0.2735$ for pasture, $P = 0.846$ for forest and $P = 0.392$ for built-up areas) and particulate nitrogen fluxes.

However, there was a significant effect of sites ($P = <0.001$) on particulate nitrogen fluxes. Also, there was a positive relationship between discharge and particulate nitrogen fluxes. The highest ratio of particulate nitrogen and discharge was found at Kaputone Creek sites and the lowest at the Styx River sites (Figure 4.37).

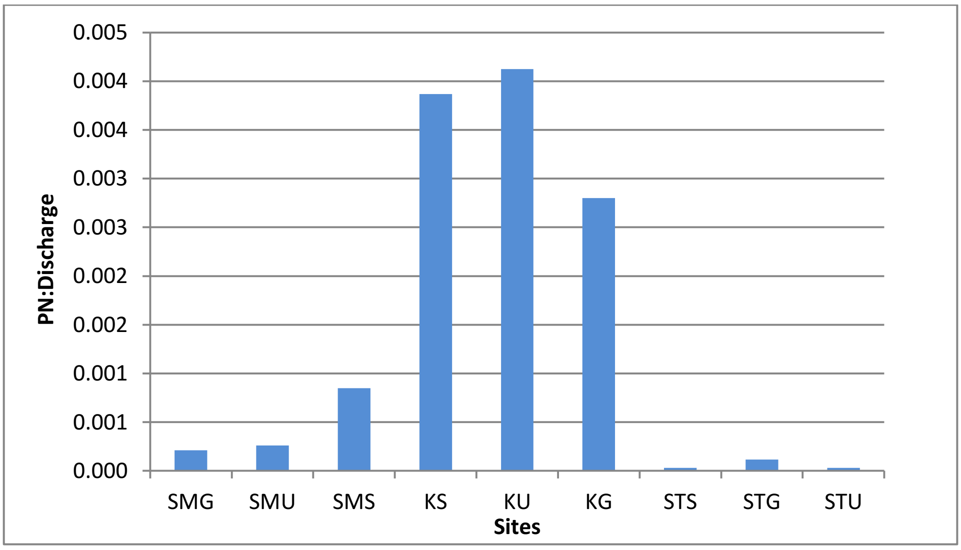

Figure 4.37 The ratio of PN and discharge at different riparian vegetation composition

4.10 Summary

Based on data collected in the field, pastoral land use was the major land use of the Styx River catchment as a whole, while at the sub-catchment scale, built-up area was the primary land use at most of the nine sampling sites.

Kaputone Creek sites showed the highest concentration of nutrients (nitrogen and phosphorus species), conductivity, turbidity and dissolved oxygen, while the lowest nitrate, total nitrogen and total dissolved nitrogen levels were found at Styx River sites. The lowest water temperature level was found at the Kaputone Creek sites. The highest turbidity levels were found at all unplanted sites.

There was a significant effect of treatment-1 (different riparian vegetation compositions and different average riparian width) on pH, dissolved oxygen, conductivity, turbidity, total phosphorus and total dissolved phosphorus. However, there was no significant effect of treatment-1 on water temperature, total suspended solid, nitrogen species, particulate matters and nutrient and sediment fluxes.

In case of treatment-2, it showed significant effect on pH, dissolved oxygen, conductivity, turbidity, total suspended solid, total and total dissolved phosphorus, and most of the nutrient and sediment fluxes except particulate matter.

Sediment and nutrient fluxes (kg/day) were influenced by discharge rate. The highest flux of sediment and nutrients was shown at STU, which had the highest discharge rate, however, the highest ratio of sediment and nutrient concentrations and discharge was found at Kaputone Creek sites.

The next chapter will discuss in detail how upstream contributing land uses and riparian vegetation composition and the width of riparian plantings affect water quality in the study area.

Chapter 5

Discussion

5.1 Introduction

The aim of this chapter is to evaluate the possible influence of riparian vegetation compositions and riparian widths on elevated nutrient and sediment concentrations with respect to immediate upstream land use combinations. Firstly, the condition of riparian vegetation composition and substrate types will be summarised, followed by the effectiveness of different riparian vegetation compositions on water quality parameters. Secondly, the influence of riparian widths with different contributing land uses on water quality will be evaluated. The relationship between contributing land uses and water quality will then be discussed.

In addition, some possible effects of site characteristics (such as stream bank erosion, the existence of in-stream aquatic plants and groundwater leaching at or upstream of the sampling site) and effects of rain events will also be highlighted. Finally, the water quality variables measured in this study will be compared with the acceptable ranges set by different authorities including the Australian and New Zealand Environment and Conservation Council (ANZECC) Guidelines.

5.2 Riparian vegetation conditions and substrate types

The nine sampling sites in this study had varying degrees of riparian vegetation compositions and riparian width. Of the three different riparian vegetation compositions, grassland areas along all streams had the widest riparian planting areas. These included purposefully planted riparian zones such as parks or reserve areas. Apart from the Styx River shaded area, none of the sites had complete shade cover with a closed canopy for 50 m upstream from the sampling points.

The width of riparian planting areas in this study was approximately 4 m to 45 m on the true right bank and 10 m to 37 m on the true left bank. The average riparian area width ranged between 0 and 42 m. Parkyn et al. (2000) found that areas of planting less than 5 m wide are not likely to support self-sustaining vegetation, and that weed control can be a problem in these situations. NIWA (2000) also recommended that the width of riparian plantings should be at least 10 m on both sides of the bank to effectively reduce nutrients and sediment from runoff. According to this advice, the width of riparian planting areas on the true right bank at all sites failed to meet this recommended width.

Substrate types in the three streams included silt/sand/gravels and small stones. Smacks Creek sites KU and STG were mainly covered with gravel and small stones, while the other sites (KS, KG, STS and STU) were found to be covered with silt and sand. The sediment depth at KS was the deepest (0.25 m).

Flow data shows that there is variable discharge across the nine sampling sites. The wider and deeper sites had larger flows than the smaller and shallower stream. For instance, STU (Styx River unplanted site) was approximately 8 m wide and had a mean flow of 900 L/s, whereas KS was about 2.5 m wide and the mean flow was 20 L/s.

5.3 Effects of different riparian plantings composition

5.3.1 Effects of different riparian planting compositions on water quality

Statistically, there were significant main effects of different riparian vegetation compositions on pH, dissolved oxygen, conductivity levels and turbidity but no significant effect was shown on water temperature. The pairwise comparison indicated a significant difference in pH levels at grassland and unplanted sites, and the significant difference in dissolved oxygen, conductivity and turbidity levels at shaded and unplanted sites.

At all the streams, the pH values at unplanted sites were found significantly higher than grassland sites. It can be suggested that grassland riparian areas have an influence on pH levels. In the case of water temperature, Smacks Creek sites showed more consistency, and the lowest levels showed at Kaputone Creek sites. The effect of riparian plantings on water temperature levels is unclear in these results. Although a study by Rutherford et al. (1997) suggested that the riparian shaded area could reduce water temperature levels, the findings in this study showed that water temperature levels at shaded areas were slightly higher than grassland and unplanted sites. This might be because the riparian plantings at sampling sites did not have enough shaded area to affect the water temperature.

The analysis showed that significant differences in dissolved oxygen levels were found between shaded and unplanted sites. However, it is difficult to determine whether dissolved oxygen levels are affected by riparian planting composition. At the Smacks Creek sites, the lowest dissolved oxygen level can be seen at the grassland site and the highest at a shaded one, whilst at the Styx River sites, the lowest is at a shaded site and the highest at the unplanted site. Furthermore, the highest dissolved oxygen values were found at Kaputone Creek sites and the Styx River unplanted site. This may be because of the lack of or few aquatic organisms that could consume dissolved oxygen or because water temperature at those sites influenced dissolved oxygen concentration. Katarial et al. (2011) reported that water temperature affects the solubility of oxygen in water, reducing water's ability to absorb dissolved oxygen at higher temperatures.

The highest conductivity levels were found at the grassland site at Kaputone Creek and Smacks Creek and the unplanted site at the Styx River, while the lowest levels were found at the shaded sites at all streams. In these streams, the shaded riparian plantings were effective in reducing conductivity levels because shaded riparian plantings could effectively reduce other contaminants such as nutrient and sediments.

The highest readings of turbidity levels were recorded from unplanted sites at all streams, while the lowest levels were found both at the grassland site at Smacks Creek and the shaded areas at Kaputone Creek and the Styx River. This suggests that in these streams, the riparian plantings were able to reduce turbidity in the water.

5.3.2 Effects of different riparian vegetation composition on sediment and nutrients

The results of statistical analysis indicated no significant effect of different riparian vegetation compositions on both total suspended solid and nitrogen concentrations. However, there was a significant effect of different riparian vegetation on dissolved reactive phosphorus, total phosphorus and total dissolved phosphorus. The pairwise comparisons showed a significant difference in dissolved reactive phosphorus, total phosphorus and total dissolved phosphorus levels between shaded and unplanted sites.

The lowest total suspended solid concentrations were found at shaded sites at Kaputone Creek and the Styx River, and at the grassland site at Smacks Creek, while the highest levels were found at three streams with different riparian conditions. Therefore, it is unclear which type of riparian vegetation composition is more effective, although previous studies have suggested that riparian vegetation could effectively reduce sediment transport from surface run-off (Ghermandi et al., 2009; Wilkinson et al., 2009).

The lowest concentrations of DRP, TP and TDP were found at the shaded sites of all three streams, and the highest at unplanted sites at Smacks Creek and the Styx River, and at the grassland site at Kaputone Creek. This result highlighted that the shaded riparian plantings effectively reduce phosphorus concentration from surface runoff, and the effects of grassland riparian plantings in reducing phosphorus levels were better than for unplanted areas (Dal Ferro et al., 2019). Furthermore, the findings showed that dissolved reactive phosphorus made up a large percentage of total phosphorus in the study area. In the case of particulate phosphorus, the lowest concentrations were found at shaded sites at Kaputone Creek and the Styx River, and the grassland site at Smacks Creek.

The pattern of nitrogen concentration at Smacks Creek is different for each species, especially nitrate, total nitrogen, and total dissolved nitrogen, and particulate nitrogen had the same pattern as total suspended solid. While nitrate levels increased downstream, total nitrogen and total dissolved nitrogen levels decreased downstream at Smacks Creek. The lowest nitrogen levels were found both at grassland sites and shaded sites. The lowest nitrate levels were found at grassland sites at Smacks and Kaputone Creeks, and at the shaded site at the Styx River, while the lowest total nitrogen and total dissolved nitrogen levels showed at shaded sites at Smacks Creek and Styx River, and at the grassland site at Kaputone Creek. The results imply that riparian plantings with grassland had a positive effect in reducing nitrate, whereas riparian shaded planting was more effective in reducing total nitrogen and total dissolved nitrogen. At grassland sites, this might be due to aquatic plants consuming nitrate for photosynthesis, but at shaded sites, riparian trees could take up more nitrogen for the production of biomass. Bowden et al. (2007) stated that forested riparian areas reduce nitrogen levels which support production systems of trees.

5.3.3 Effects of riparian vegetation compositions on sediment and nutrients fluxes

Statistically, there was no significant effect of riparian vegetation composition on sediment and nutrient fluxes, but the discharge rate was linked with sediment and nutrient fluxes.

Variation in stream flow has a significant influence on water quality (Hayward & Ward, 2008). High rainfall in the catchment results in high overland flow, which in turn increases the loading of nutrients and sediments within a waterway (Waterwatch Victoria, 1996).

The findings showed that the highest ratio of discharge and sediment and nutrient fluxes were found at the Kaputone Creek unplanted site (TSS, PN, DRP, TP and TDP) and shaded site (PP, NO_3–N, TN and TDN), while the lowest ratio was found at the Styx River sites. This suggests that at the Kaputone Creek sites, the flow carried a relatively high input of sediment and nutrient from upstream and/or nearby land uses. Furthermore, the nitrogen concentration was found to be critically higher than phosphorus levels at the study sites.

5.3.4 Effectiveness of forested versus grassland riparian plantings

Overall, the riparian plantings (both shaded and grassland) had a positive effect in reducing contaminants from overland flow compared to unplanted sites. However, it remains unclear whether forested riparian areas are more effective in reducing sediment and nutrient levels than grassland areas, because the results showed that although conductivity levels were found to be lowest at the shaded sites (riparian plantings mainly covered with trees) of all three streams, the lowest values of turbidity, total suspended solids and nutrient levels were found in both forested and grassland areas. For instance, the turbidity, total suspended solid and phosphorus levels at the Kaputone Creek and the Styx River shaded sites were found to be significantly lower than the grassland site at Smacks Creek.

It is likely that gaps in the riparian planting areas and insufficient width of riparian planting areas are still contributing to poor water quality. The results indicated that conductivity, turbidity, total suspended solids and nutrients levels were lower at the shaded site at the Styx River than in other riparian conditions. So, it can be noted that the width of riparian planting area at the Styx River shaded site can be a reference in reducing contaminants effectively, and it is the only site fully shaded with riparian vegetation nearly 5 m on both sides of the stream.

5.4 Upstream and sub-catchment contributing land uses and their relationship with water quality parameters

Built-up areas were found to be the predominant upstream contributing land uses at the Smacks and Kaputone Creek sites, while pastoral land mainly constituted the upstream land uses at the Styx River sites. Small areas of cropland were found only at the Styx River sites and SMG. The largest area of cropland in upstream land uses was found at STS. Forested land was found at all sites with the largest area at STG. Overall, the Styx River sites and SMG only showed a range of four different land uses in the upstream area.

No significant relationship was found between pH, water temperature, dissolved oxygen and turbidity, and the upstream and sub-catchment land uses. Only conductivity showed a significant relationship with the percentage of sub-catchment land uses. (especially the percentage of built-up areas). In this study, the highest conductivity level was measured at the Kaputone Creek sites with predominantly built-up areas in the upstream and sub-catchment areas. This could be due to surface run-off from built-up areas carrying nutrients from home gardens, domestic animals waste and contaminants from construction sites as well as sewage overflows into the nearby stream. A study by Zeb et al. (2011) in the Siran River in Pakistan also suggested that higher conductivity near settlement areas was due to sewage discharge into the river.

The upstream and sub-catchment land uses did not show a significant relationship with total suspended solid and phosphorus concentration. However, total suspended solid and phosphorus concentrations of the most upstream sites at the three streams were generally lower than the downstream sites. It indicated that the downstream sites were likely to be affected by the sediment and nutrient inputs from upstream and sub-catchment land uses.

There was a significant relationship between the percentage of sub-catchment land uses (especially pastoral land and built-up areas) and nitrate, total nitrogen and total dissolved nitrogen, but not correlated with particulate nitrogen. The highest nitrogen levels were found at Kaputone Creek sites, where the highest percentage of built-up areas occurred.

The results indicated that nitrate levels had a positive relationship with both pastoral land and built-up areas. Nitrate concentration increased downstream at Smacks Creek sites where the percentage of pastoral land increased. This could be due to the use of nitrogen fertilizers on nearby pasture crops or animal waste from pastoral land. Monaghan et al. (2013) stated that animal urine can be a major source of nitrate. At the Styx River sites, nitrate levels increased when built-up areas also increased. As well, total nitrogen and total dissolved nitrogen showed a positive relationship with built-up areas. The concentration of total nitrogen and total dissolved nitrogen decreased downstream at the Smacks Creek sites where built-up areas decreased, and then increased downstream at the Styx River sites as built-up areas became more prevalent. It indicated that surface run-off at those sites possibly carried nitrogen from home gardens, domestic animal wastes and sewage overflows from built-up area to the streams (Chakravarthy et al., 2019). Suthar et al. (2010) stated that sewage water from urban area may also affect river water quality near urban area.

5.5 The influence of riparian plantings with different width predominantly different land uses

5.5.1 The influence on water quality

The statistical analysis showed that there was a significant effect of different riparian width with different contributing land uses (group-1: 0-5 m riparian width predominantly built-up areas, group-2: > 5 m riparian width predominantly built-up/pastoral areas and group-3: >5 m riparian width predominantly solely pastoral areas) on pH, dissolved oxygen, conductivity and turbidity levels. The pairwise comparison showed a significant difference in pH, dissolved oxygen and turbidity levels between group-2 and group-1, and a significant difference in conductivity levels between group-3 and group-1. Group-1 includes unplanted sites at all three streams (SMU, KU and STU), SMG, SMS and KG in group-2 and KS, STS and STG in group-3. The pH, dissolved oxygen and turbidity levels at group-1 sites are higher than group-2 sites, while conductivity levels at group-1 sites are higher than group-3 sites. It is difficult to evaluate whether riparian plantings have a positive influence on pH and dissolved oxygen levels. However, in case of turbidity and conductivity levels, it can clearly be seen that riparian plantings with > 5 m width have a positive influence on reducing contaminants from contributing catchment land uses.

5.5.2 The influence on sediment and nutrients

The statistical analysis showed that there was a significant effect of different riparian width with different contributing land uses on total suspended solid, total and total dissolved phosphorus levels only. The pairwise comparison showed a significant difference in total suspended solid between group-2 and group-3, and a significant difference in total and total dissolved phosphorus levels between group-2 and group-1. The total suspended solid levels at group-3 is higher than group-2. This result can be reflected that riparian planting with > 5 m width could effectively be reduced total suspended solids from built-up/ pastoral land, and also, it can be seen that runoff from pastoral land carries the higher amount of total suspended solids. However, the total and total dissolved phosphorus levels at group-2 sites are higher than group-1 sites. It indicates that > 5 m wide riparian area did not have an influence on reducing nutrients from contributing land uses, though previous research by Duchemin and Hogue (2009) recommended that the width of a riparian area should be at least 5 m to effectively reduce runoff volume, suspended solids and nutrients.

5.5.3 The influence on sediment and nutrient fluxes

The statistical analysis showed that there was a significant effect of different riparian width with different contributing land uses on total suspended solid, total and total dissolved phosphorus, and total and total dissolved nitrogen fluxes. The pairwise comparison showed a significant difference in total suspended solid fluxes between group-2 and group-1, and a significant difference in total and total dissolved phosphorus, and, total and total dissolved nitrogen fluxes between group-2 and group-3. The total suspended solid fluxes at group-1 sites are higher than group-2 sites, and total and total dissolved nutrient fluxes at group-3 sites are higher than group-2 sites. The results agree with Duchemin and Hogue (2009)'s recommendation because it can be evaluated that riparian plantings with > 5 m width have a positive influence on reducing sediment and nutrients fluxes from built-up/pastoral areas.

5.6 Effect of site characteristics

5.6.1 Effect of site characteristics on water quality

The statistical analysis showed that there were significant effects of sites on water quality parameters (pH, dissolved oxygen, conductivity and turbidity) but no effects on water temperature. In some cases, although there were effects both from riparian plantings and the sites, the effects of site can be seen more clearly than the effects of riparian planting. For instance, the lowest pH levels were found at the grassland site at Smacks and Kaputone Creek and the shaded site at the Styx River, while the highest pH levels were at the shaded sites at Smacks and Kaputone Creek and the unplanted site at the Styx River. It is difficult to determine which riparian vegetation compositions have a positive effect on pH level. In this case, it can clearly be seen that the pH values of each sampling site are affected by the site characteristics. Aquatic plants were found at the grassland site at Smacks Creek and the high respiration and decomposition rate of aquatic plants at that site might result in decreased pH levels (Schneider et al., 2000). Furthermore, dissolved oxygen at the Smacks Creek grassland site were shown to be critically low, perhaps due to the effect of site characteristics such as high consumption by aquatic organisms or because oxygen entering to a stream from the atmosphere and groundwater discharge is very low in oxygen. NIWA (2021) stated that dissolved oxygen in stream could be low when groundwater inflows contribute low oxygen concentration.

5.6.2 Effects of site characteristics on sediment and nutrients

The statistical analysis showed that there was a significant effect of sites on sediment and nutrients levels. The lowest total suspended solids concentrations were found at the most upstream sites of each stream. So it can be concluded that total suspended solid concentrations were affected by the site characteristics. At Smacks Creek, the shaded site showed the highest values, and this might be because it is the most downstream site of the three sampling sites at Smacks Creek. However, at Kaputone Creek and the Styx River, the second sampling sites (before the most downstream site) showed the highest levels. In this case, although the highest level at Kaputone Creek can be explained because the site is unplanted, the site with the highest level at the Styx River was the grassland site. So, it can be concluded that the effectiveness of grassland is not likely to be enough to reduce the effects of site. Furthermore, these streams seem to allow suspended solids to settle before water drains downstream because total suspended solids concentrations decreased for the most downstream sites at Kaputone Creek and the Styx River. Ryan (1991) also stated that suspend solids could settle in stream beds.

The phosphorus concentrations were found to increase downstream at all sampling sites. This might possibly be due to groundwater inflows carrying phosphorus and surface run-off. The effects of site on nitrogen concentrations can also be clearly seen at Smacks Creek and Kaputone Creek, where the highest nitrogen levels (total nitrogen and total dissolved nitrogen) were found even at the most upstream sites. Also, nitrogen levels at these two streams decreased downstream, and it might be because in-stream vegetation uptake is higher at that site or in the upstream area of that site (Parkyn et al., 2003), or nitrogen is leaching into groundwater upstream or at that site.

5.6.3 Effects of rain events on water quality, nutrients and sediment

Contrary to other water quality parameters, water temperature, turbidity, total suspended solid and particulate phosphorus were correlated with rain events. It can be seen at Kaputone Creek sites where water temperature was significantly higher after rain events. This could be due to thermal pollution caused by run-off transporting pollutants from impervious surface (e.g. roads and parking lots) because those roads and parking absorbs heat and create warm surface run-off. Herb et al. (2008) similarly reported that after rain events, runoff carrying pollutants from impervious surfaces caused an increase in water temperature whilst studying thermal pollution of streams in Minnesota, USA.

Furthermore, total suspended solid values at all sites increased after rain events, and total suspended solid values at the Smacks Creek unplanted site, Kaputone Creek shaded and grassland sites were found to be doubled. The concentrations of particulate phosphorus at SMS, KU, KG and STU were also significantly higher after rain events. High intensity rainfall events with long durations may have carried particulate matter from adjacent land uses. Gong et al. (2016) stated that rainfall intensity and duration can influence total suspended concentrations in surface-runoff.

5.7 Acceptability of water quality compared to the recommended limits

The pH levels at all sampling sites fell within the ideal range of 6.0 to 8.5, as recommended by Chergui et al. (2013), James (1999) and WHO (2008). But the pH values at some sampling sites (except KS, KG and STU) were below 7.2 and out of the range of 7.2 to 7.8 as recommended by ANZECC (2000). However, as the ANZECC ranges are often considered to be overly stringent (e.g., Milne & Perrie 2006), these values are considered to be acceptable as they fall within the range adopted by Chergui et al. (2013), James (1999) and WHO (2008).

Water temperature ranged from 9.3°C to 14.4°C, well below the recommended upper limit of 20°C (Quinn & Hickey, 1990), and Ausseil (2013) suggested that there are low thermal stresses on aquatic ecosystems when the water temperature is less than or equal to 18°C.

Dissolved oxygen levels ranged from 3.14 to 10.49 mg/L. The range of dissolved oxygen level at SMG were less than or equal to 4 mg/L, which is the national bottom line for daily minimum dissolved oxygen concentration recommended by Ausseil (2013). The mean dissolved oxygen concentrations at SMU, SMS and STS were within an acceptable range of 4 to >7.5 mg/L but occasional minor stress on sensitive organisms because of the short periods of lower dissolved oxygen level can still be expected to occur for a few hours each day. The other sites (Kaputone Creek sites, STG and STU) recorded above 7.5 mg/L dissolved oxygen concentration, within acceptable levels.

In New Zealand, there are no published guidelines for conductivity levels, and so conductivity was assessed using only the Australian standard though New Zealand streams do not have salinisation issues like Australia (CCC, 2019). Conductivity ranged from 108 to 152.8 μS/cm, well below the Australian upper limit of 1,500 μS/cm (Waterwatch Victoria, 1996).

Turbidity levels at all sites ranged from 0 to 3.34 NTUs, and are well below the upper limit of 5.6 NTUs as set out by ANZECC (2000). However, the maximum turbidity levels at KU and KG are well above the upper limits of 2 NTUs recommended by Davies-Colley and Wilcock (2004), while other sites are well below 2 NTUs. Total suspended solids at nine sampling sites ranged from 0–13 mg/L, and are below the acceptable level of 25 mg/L recommended by Ryan (1991).

Dissolved reactive phosphorus levels ranged from 0.015 to 0.086 mg/L. These values are well above the upper limits of 0.014 as set out by Ausseil (2013). Total phosphorus concentration ranged from 0.020 to 0.099 mg/L. The mean total phosphorus values at SMG and SMS are just below the upper limit of 0.03 mg/L recommended by ANZECC (2000), while the mean values at other sites are well above the recommended limits.

Nitrate levels ranged from 0.087 to 1.083 mg/L. Apart from STS, the mean values at all other sampling sites are well above the levels of 0.1 mg/L as set out by Davies-Colley and Wilcock (2004). But, except for Kaputone Creek sites, other sites are well below the limits of 0.444 mg/L recommended by ANZECC (2000). Total nitrogen levels ranged from 0.252 to 2.132 mg/L. The mean total nitrogen values at the Styx River sites SMU and SMS are well below the trigger value of 0.614 mg/L recommended by ANZECC (2000) for New Zealand's lowland rivers, whereas the mean values at other sites are above the trigger value.

5.8 Summary

This discussion has highlighted and discussed the key findings of the study in the context of the literature. Riparian plantings (mainly involving mature tree species) were found to have a positive effect on water quality in terms of reducing turbidity, conductivity, total suspended solid and nutrients; however, in some areas, sediment and nutrient inputs from upstream and/or surrounding land uses were still higher than the ability of riparian plantings to remove them. The riparian plantings with > 5 m width do not have enough influence on reducing nutrients from pastoral land. Furthermore, the water quality parameters are likely to be affected by the site characteristics such as stream bank erosion and the existence of aquatic plants.

The effectiveness of riparian plantings was evaluated in terms of the impact of width, stream shade, and different vegetation composition and the effects of site characteristics were also considered. The background and findings of this study are summarized in the upcoming conclusions chapter.

Chapter 6

Conclusion and Recommendations

Global freshwater quality is being threatened by human actions such as land-use changes and development activities. This can also be seen in New Zealand, where water pollution especially in lowland streams has been found. One of the critical threats to water quality in New Zealand is nutrients and sediment resulting from land-use conversions to agricultural and urban land use in catchment areas. Riparian plantings along the river banks have been recommended as a cost-effective measure to manage water quality problems. However, it should be noted that the effectiveness of riparian plantings might differ according to vegetation composition (e.g., mature trees and grassland), riparian width and the amount of contaminants transported from contributing land uses.

This study assessed the water quality status of streams in the Styx River catchment in relation to upstream contributing land uses and the effectiveness of different riparian plantings (mature trees and grassland) in reducing sediment and nutrient loads in lowland streams. Water quality data were collected from nine sampling sites (three different riparian vegetation compositions (shaded, grassland and unplanted) at three streams: Smacks Creek, Kaputone Creek and the Styx River). A total of 72 water quality samples were collected and analysed for physio-chemical water quality, sediment and nutrients (both phosphorus and nitrogen) within different riparian composition and different width predominantly different land uses.

The highest concentrations of dissolved oxygen were also found at Kaputone Creek sites, which also had the lowest water temperature. Water temperature showed no difference between the sites with different riparian vegetation compositions, although some literature has highlighted that water temperature is affected by shaded riparian vegetation. This might be because the riparian plantings at the sampling sites did not have sufficient shading to reduce water temperatures.

However, the results still support previous findings (e.g., Collin et al., 2012 and Parkyn et al., 2003) which have suggested that riparian plantings (both shaded and grassland) have a positive effect on water quality. This can be seen in the decrease in the levels of conductivity, turbidity, sediment, phosphorus and nitrogen at sites with established riparian plantings.

Also, the results highlighted that riparian plantings with trees showed higher effectiveness in reducing pollutants than grassland areas, though the riparian planting areas at the sampling sites did not meet the recommended minimum width of 10 m. It can be seen that the lowest values of conductivity and phosphorus were found at shaded sites and the lowest values of turbidity, sediment and nitrogen were found mostly at shaded sites (at the shaded sites of Kaputone Creek and Styx River, and the grassland site at Smacks Creek).

Only conductivity and nitrogen (nitrate, total nitrogen and total dissolved nitrogen) concentration showed positive relationships with upstream land uses. The increasing trends of these parameters are mostly associated with an increase in land-use development especially for built-up areas and pastoral land. The increase in conductivity, total nitrogen and total dissolved nitrogen was found in the sites mainly covered by built-up areas, whereas only nitrate increased at sites predominantly composed of both pastoral land and built-up areas.

Although the other parameters did not show a statistically significant relationship with land use, the highest amount of pollutants (dissolved reactive phosphorus, total phosphorus, total dissolved phosphorus, nitrate, total and total dissolved nitrogen) were found at the Kaputone Creek sites, where built-up areas were found to be the major upstream and sub-catchment contributing land uses.

In addition, > 5 m wide riparian areas can effectively reduce turbidity and conductivity from built-up/pastoral land uses. The > 5 m wide riparian areas are more effective in reducing sediment and nutrients from combined built-up/ pastoral area than solely pastoral land.

The effectiveness of riparian plantings and different riparian vegetation compositions in reducing sediment and nutrients is not as simple as whether the riparian area is planted with trees or grasses. A number of factors (such as length and width of riparian planting area and stream shaded area) need to be considered. Furthermore, it should be noted that the riparian plantings may not entirely solve the water quality problems. A balance between the main functions of the riparian plantings in relation to the sensitivity of a proposed site (such as the upstream and adjacent land uses and possible contaminant transport) will need to be considered.

In order to restore and/or maintain in-stream water quality, it would be more effective if the riparian plantings (ideally with mature trees and some grass and shrubs just next to the stream bank) began at the upstream area and continued through to the river mouth. This is often impractical because of the private ownership of riparian areas.

If the landowners could be convinced to cooperate in catchment management in order to restore and replant riparian plantings, the situation might change. In order to motivate landowners to be more involved with catchment management, further study to confirm the optimal widths and lengths is needed to most effectively reduce contaminant loads from nearby land uses. Also, a study related to which trees, shrubs and grass species are most effective in mitigating contaminants from surface runoff would be beneficial in establishing more effective riparian planting areas.

Until better information is available on these factors, it is recommended that a minimum 10 m wide riparian planting area on both side of the streams be created or preserved before development begins.

6.1 Scope for further research

This research highlighted that more research needs to be done related to the effectiveness of riparian vegetation on water quality.

For example, the relationship between the percentage of shaded areas and water temperature could be explored as water temperature is correlated with several water quality parameters and also aquatic ecosystem health.

Furthermore, this study raises questions around whether rainfall intensity and duration influence sediment and nutrients concentrations and the optimal dimensions for riparian vegetation to reduce sediment and nutrients from high rainfall intensity and long duration events.

Socio-economic studies of the interests of private land owners in establishing and managing riparian plantings along waterways could also be conducted. These might help us to better understand the local aspects of riparian restoration and why more people aren't doing this and what the barriers are.

6.2 Closing comments

This research has assessed the impacts of non-point pollution resulting from upstream contributing land use and evaluated the effectiveness of different riparian plantings on water quality using a case study in the Styx River catchment. Built-up areas were found to have effects on conductivity levels, total nitrogen and total dissolved nitrogen, and pastoral land showed an effect on nitrate levels. However, no land use effects showed for other water quality variables. The proportions of sediment and nutrient fluxes rely on discharge rate, and the flow at the Kaputone Creek sites carried the high amount of sediment and nutrients.

Riparian plantings (both trees and grasses) showed a positive effect on water quality in terms of decreasing conductivity, turbidity and phosphorus. Varied responses of different riparian plantings were seen. At Kaputone Creek and the Styx River, riparian plantings with trees showed positive effects in reducing conductivity, turbidity and phosphorus, but at Smacks Creek, grassland showed a positive response in reducing contaminants. In the case of nitrogen levels, it showed the lowest level at the Styx River shaded site, which has approximately 5 m-wide riparian plantings with trees on both sides of the bank. It is likely that riparian plantings with trees at least 5 m wide had a considerable effect in reducing nutrients (especially soluble nutrients) from upstream and sub-catchment land uses.

This research has shown the need for realistic goals to be set for maintaining existing riparian plantings and establishing riparian planting areas on both sides of the stream as much as possible. Furthermore, on-going monitoring on catchment land use change, management and establishment of riparian plantings and water quality changes along the streams is identified as being important in providing scientific evidence for the further establishment of riparian plantings and encouraging the cooperation of private land owners in the management of riparian plantings.

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Appendix A

Results of statistical analysis

A.1 Physiochemical water quality

Table 1. Minimum, mean, maximum and *P* of physiochemical water quality parameters at all sampling sites over the entire sampling period

		Smacks Creek			Kaputone Creek			Styx River						
		SMG	**SMU**	**SMS**	**KS**	**KU**	**KG**	**STS**	**STG**	**STU**	***P(tr-1)***	***P(tr-2)***	***P(s)***	***P(r)***
pH	Minimum	6.30	6.49	6.68	7.01	7.04	7.00	6.40	6.71	7.14				
	Mean	6.35	6.65	6.74	7.14	7.12	7.08	6.57	6.78	7.33	*	*	*	-
	Maximum	6.41	6.75	6.8	7.24	7.19	7.26	6.64	6.89	7.61				
Temp	Minimum	12.3	12	12	10.9	9.3	9.5	12.2	12.2	11.4				
	Mean	12.8	12.8	12.8	12.3	11.4	11.5	12.9	12.8	12.3	-	-	*	*
	Maximum	13.3	13.5	13.5	14.4	13.5	13.9	13.6	13.8	13.7				
DO	Minimum	3.14	6.18	6.39	8.43	8.49	8.67	6.15	7.58	8.32				
	Mean	3.41	6.87	7.04	9.36	9.36	9.45	6.47	8.03	9.74	*	*	*	-
	Maximum	4.07	7.43	7.64	10.14	10.34	10.49	6.61	8.30	10.34				
Cond	Minimum	110.4	110.5	110.2	112.7	122.3	124.8	108.0	112.3	121.4				
	Mean	117.9	114.5	114.5	123.6	135.5	137.9	108.5	113.1	125.5	*	*	*	-
	Maximum	120.6	115.8	115.9	134.0	148.4	152.8	109.2	113.8	132.6				

		Smacks Creek			Kaputone Creek			Styx River						
		SMG	**SMU**	**SMS**	**KS**	**KU**	**KG**	**STS**	**STG**	**STU**	***P(tr-1)***	***P(tr-2)***	***P(s)***	***P(r)***
Turbidity	Minimum	0.00	0.01	0.03	0.14	0.22	0.24	0.17	0.38	0.49				
	Mean	0.02	0.34	0.21	0.47	1.74	1.08	0.30	0.67	0.85	-	*	*	*
	Maximum	0.07	1.05	0.64	0.79	3.34	2.23	0.53	0.99	1.20				

Note: the sampling sites for each stream are shown from upstream to downstream.

a) P(tr-1) = P for treatment, b) P(tr-2) = P for treatment, c) P(s) = P for site, d) P(r) = P for rain event

A.2 Sediment and nutrient

Table 2. Minimum, mean, maximum and *P* for contaminants (sediment and nutrients) at each sampling site over the entire sampling period

		Smacks Creek			Kaputone Creek			Styx River						
		SMG	**SMU**	**SMS**	**KS**	**KU**	**KG**	**STS**	**STG**	**STU**	***P(tr-1)***	***P(tr-2)***	***P(s)***	***P(r)***
TSS	Minimum	0.000	0.200	0.200	0.200	0.660	0.400	0.400	1.667	0.333				
	Mean	0.250	0.825	1.105	2.517	4.690	3.510	1.292	3.742	2.043	-	*	*	-
	Maximum	1.000	2.467	2.733	5.067	13.000	10.200	2.200	5.200	4.000				
DRP	Minimum	0.019	0.017	0.015	0.032	0.023	0.044	0.022	0.023	0.036				
	Mean	0.023	0.030	0.025	0.044	0.051	0.057	0.025	0.030	0.041	*	-	*	*
	Maximum	0.029	0.041	0.034	0.076	0.086	0.082	0.029	0.042	0.048				
TP	Minimum	0.020	0.028	0.022	0.032	0.041	0.044	0.027	0.026	0.042				
	Mean	0.026	0.033	0.029	0.049	0.057	0.062	0.030	0.034	0.045	*	*	*	-
	Maximum	0.030	0.041	0.034	0.082	0.091	0.099	0.034	0.048	0.052				
TDP	Minimum	0.020	0.024	0.022	0.032	0.030	0.044	0.026	0.026	0.039				
	Mean	0.026	0.032	0.028	0.048	0.053	0.059	0.029	0.033	0.043	*	*	*	-
	Maximum	0.030	0.041	0.034	0.078	0.087	0.088	0.032	0.044	0.051				
PP	Minimum	0.0000	0.0000	0.0004	0.0002	0.0003	0.0002	0.0000	0.0000	0.0008				
	Mean	0.0004	0.0012	0.0017	0.0013	0.0044	0.0027	0.0010	0.0017	0.0022	-	-	*	*
	Maximum	0.0013	0.0040	0.0034	0.0037	0.0241	0.0109	0.0017	0.0041	0.0031				

		Smacks Creek			Kaputone Creek			Styx River						
		SMG	**SMU**	**SMS**	**KS**	**KU**	**KG**	**STS**	**STG**	**STU**	***P(tr-1)***	***P(tr-2)***	***P(s)***	***P(r)***
NO3–N	Minimum	0.142	0.140	0.144	0.597	0.310	0.380	0.087	0.101	0.119				
	Mean	0.269	0.274	0.285	0.921	0.733	0.717	0.171	0.222	0.240	-	-	*	-
	Maximum	0.386	0.435	0.436	1.083	1.073	0.981	0.334	0.358	0.348				
TN	Minimum	0.409	0.456	0.263	1.376	1.148	0.990	0.252	0.348	0.267				
	Mean	0.617	0.533	0.501	1.841	1.453	1.409	0.321	0.430	0.447	-	-	*	-
	Maximum	0.782	0.707	0.614	2.132	1.785	1.663	0.389	0.486	0.627				
TDN	Minimum	0.398	0.438	0.230	1.329	1.141	0.953	0.244	0.343	0.238				
	Mean	0.608	0.517	0.432	1.761	1.359	1.337	0.311	0.379	0.415	-	-	*	-
	Maximum	0.782	0.704	0.605	1.905	1.609	1.609	0.389	0.451	0.601				
PN	Minimum	0.000	0.003	0.002	0.013	0.007	0.036	0.000	0.000	0.002				
	Mean	0.008	0.016	0.069	0.080	0.094	0.073	0.009	0.051	0.032	-	-	*	-
	Maximum	0.019	0.046	0.177	0.302	0.246	0.151	0.050	0.122	0.096				

Note: the sampling sites for each stream are shown from upstream to downstream.

a) *P(tr)*= *P* for treatment, *P(s)*=*P* for site, c) *P(r)*=*P* for rain event

A.3 Sediment and nutrient fluxes

Table 3. Minimum, mean, maximum and *P* for sediment and nutrient fluxes (kg/day) at each sampling site over the entire sampling period

		Smacks Creek			Kaputone Creek			Styx River						
		SMG	**SMU**	**SMS**	**KS**	**KU**	**KG**	**STS**	**STG**	**STU**	***P(tr-1)***	***P(tr-2)***	***P(s)***	***P(r)***
TSS	Minimum	0.00	1.13	1.13	0.27	1.28	0.84	10.06	63.40	25.10				
	Mean	0.76	4.42	8.86	4.80	8.01	8.09	33.94	139.70	171.50	-	*	*	
	Maximum	3.14	13.56	26.33	11.98	18.58	24.56	58.08	234.40	346.80				
TP	Minimum	0.06	0.12	0.11	0.06	0.06	0.09	0.69	0.81	3.13				
	Mean	0.09	0.18	0.21	0.09	0.12	0.14	0.78	1.32	3.75	-	*	*	-
	Maximum	0.14	0.23	0.29	0.16	0.20	0.21	0.92	2.20	4.91				
	TDP (%)	98.1	96.7	94.2	97.2	92.4	95.8	96.8	94.8	94.8				
	PP (%)	1.8	3.2	5.8	2.5	7.5	4.3	3.3	5.2	5.2				
TDP	Minimum	0.06	0.11	0.11	0.05	0.06	0.09	0.67	0.81	2.90				
	Mean	0.08	0.17	0.20	0.09	0.11	0.13	0.76	1.25	3.55	-	*	*	-
	Maximum	0.13	0.23	0.26	0.16	0.19	0.18	0.88	2.03	4.67				
	DRP (%)	91.2	94.1	91.9	91.0	97.1	94.9	85.9	91.7	94.8				
PP	Minimum	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07				
	Mean	0.00	0.01	0.01	0.00	0.01	0.01	0.03	0.07	0.19	-	-	*	-
	Maximum	0.01	0.02	0.03	0.01	0.05	0.02	0.05	0.17	0.34				

		Smacks Creek			Kaputone Creek			Styx River						
		SMG	**SMU**	**SMS**	**KS**	**KU**	**KG**	**STS**	**STG**	**STU**	***P(tr-1)***	***P(tr-2)***	***P(s)***	***P(r)***
TN	Minimum	1.19	2.14	1.54	2.03	2.36	2.08	6.66	10.94	20.11				
	Mean	2.00	2.84	3.56	3.33	2.84	3.20	8.41	16.17	36.28	-	*	*	-
	Maximum	3.07	4.12	5.91	5.04	4.16	4.30	9.87	22.28	45.96				
	TDN (%)	98.5	96.9	86.5	95.2	94.3	94.9	97.0	87.8	91.5				
	PN (%)	1.5	3.1	13.5	4.8	5.7	5.1	3.0	12.2	8.5				
TDN	Minimum	1.16	2.10	1.34	1.96	1.97	2.00	6.44	10.94	17.92				
	Mean	1.97	2.75	3.08	3.17	2.68	3.04	8.16	14.20	33.19	-	*	*	-
	Maximum	2.99	4.10	4.77	4.33	4.11	4.15	9.80	17.58	44.02				
	NO3–N (%)	42.3	51.6	63.8	50.5	52.6	53.7	55.0	61.8	62.8				
PN	Minimum	0.00	0.02	0.01	0.02	0.02	0.08	0.00	0.00	1.73				
	Mean	0.03	0.09	0.48	0.16	0.16	0.16	0.25	1.97	3.09	-	-	*	-
	Maximum	0.782	0.707	0.614	2.132	1.785	1.663	0.389	0.486	0.627				

Note: the sampling sites for each stream are shown from upstream to downstream.

a) *P(tr)*= *P* for treatment, *P(s)*=*P* for site, c) *P(r)*=*P* for rain event

Appendix B

Water quality and flow data

This section presented the water quality data and calculated discharge rate at each sampling sites.

B.1 Smacks Creek grassland site

	1st Field	2nd Field	3rd Field	4th Field	5th Field	6th Field	7th Field	8th Field
pH	6.38	6.38	6.34	6.32	6.34	6.41	6.30	6.35
Temp (°C)	13.30	12.90	12.30	12.60	12.50	12.90	13.00	13.00
DO (mg/L)	3.14	3.22	3.22	4.07	4.06	3.17	3.16	3.25
Cond (μS/cm)	110.35	116.05	116.35	118.60	120.50	120.40	120.55	120.40
Turb (NTUs)	0.00	0.01	0.02	0.02	0.02	0.04	0.07	0.02
TSS (mg/L)	0.0000	0.1333	0.1333	0.2000	1.0000	0.3333	0.1333	0.0667
DRP (mg/L)	0.0224	0.0190	0.0293	0.0238	0.0210	0.0292	0.0242	0.0190
TP (mg/L)	0.0247	0.0200	0.0302	0.0259	0.0260	0.0296	0.0296	0.0240
TDP (mg/L)	0.0234	0.0200	0.0301	0.0256	0.0256	0.0294	0.0294	0.0236
PP (mg/L)	0.0013	0.0000	0.0001	0.0002	0.0004	0.0002	0.0002	0.0004
NO_3-N (mg/L)	0.142	0.386	0.174	0.327	0.352	0.281	0.208	0.287
TN (mg/L)	0.555	0.712	0.782	0.572	0.760	0.667	0.409	0.478
TDN (mg/L)	0.541	0.710	0.782	0.557	0.741	0.666	0.398	0.474
PN (mg/L)	0.014	0.002	0.000	0.016	0.019	0.001	0.011	0.004
Discharge (L/s)	63.97	31.82	35.45	33.29	36.32	35.43	33.72	31.41
TSS (kg/day)	0.000	0.367	0.408	0.575	3.138	1.020	0.389	0.181
DRP (kg/day)	0.124	0.052	0.090	0.068	0.066	0.090	0.070	0.051
TP (kg/day)	0.137	0.055	0.093	0.074	0.082	0.091	0.086	0.065
TDP (kg/day)	0.129	0.055	0.092	0.074	0.080	0.090	0.086	0.064
PP (kg/day)	0.007	0.000	0.000	0.001	0.001	0.001	0.001	0.001
NO_3-N(kg/day)	0.785	1.061	0.532	0.940	1.105	0.859	0.605	0.777
TN (kg/day)	3.067	1.958	2.395	1.646	2.384	2.042	1.191	1.298
TDN (kg/day)	2.990	1.951	2.395	1.601	2.324	2.038	1.159	1.286
PN (kg/day)	0.077	0.006	0.000	0.045	0.060	0.004	0.032	0.012

B.2 Smacks Creek unplanted site

	1st Field	2nd Field	3rd Field	4th Field	5th Field	6th Field	7th Field	8th Field
pH	6.69	6.75	6.63	6.67	6.63	6.50	6.64	6.66
Temp (°C)	13.50	12.80	12.10	12.00	12.30	13.10	13.40	13.20
DO (mg/L)	6.18	6.68	6.68	7.43	7.24	6.88	6.94	6.92
Cond (μS/cm)	110.45	115.35	114.25	115.75	114.85	115.45	115.00	115.00
Turb (NTUs)	0.26	0.01	0.85	0.05	0.10	0.16	0.25	1.05
TSS (mg/L)	0.2000	0.3333	1.2000	0.9333	0.6000	0.4000	0.4667	2.4667
DRP (mg/L)	0.0410	0.0304	0.0377	0.0238	0.0168	0.0329	0.0335	0.0266
TP (mg/L)	0.0414	0.0320	0.0386	0.0317	0.0280	0.0343	0.0338	0.0278
TDP (mg/L)	0.0411	0.0320	0.0377	0.0303	0.0240	0.0329	0.0336	0.0268
PP (mg/L)	0.0002	0.0000	0.0009	0.0015	0.0040	0.0015	0.0002	0.0010
NO3-N (mg/L)	0.140	0.435	0.192	0.305	0.347	0.299	0.221	0.257
TN (mg/L)	0.460	0.521	0.455	0.707	0.489	0.629	0.497	0.508
TDN (mg/L)	0.445	0.513	0.438	0.704	0.481	0.615	0.450	0.493
PN (mg/L)	0.014	0.008	0.018	0.003	0.008	0.014	0.046	0.015
Discharge (L/s)	65.40	50.48	58.42	67.44	50.50	62.22	72.52	63.63
TSS (kg/day)	1.130	1.454	6.057	5.439	2.618	2.150	2.924	13.561
DRP (kg/day)	0.232	0.132	0.190	0.138	0.073	0.177	0.210	0.146
TP (kg/day)	0.234	0.140	0.195	0.185	0.122	0.185	0.212	0.153
TDP (kg/day)	0.232	0.140	0.190	0.176	0.105	0.177	0.211	0.147
PP (kg/day)	0.001	0.000	0.005	0.009	0.017	0.008	0.001	0.005
NO_3-N(kg/day)	0.793	1.898	0.967	1.774	1.515	1.605	1.385	1.413
TN (kg/day)	2.599	2.272	2.299	4.119	2.135	3.384	3.111	2.795
TDN (kg/day)	2.517	2.236	2.208	4.102	2.100	3.307	2.821	2.710
PN (kg/day)	0.082	0.036	0.090	0.016	0.035	0.077	0.290	0.085

B.3 Smacks Creek shaded site

	1st Field	2nd Field	3rd Field	4th Field	5th Field	6th Field	7th Field	8th Field
pH	6.72	6.80	6.78	6.72	6.70	6.68	6.71	6.79
Temp (°C)	13.50	12.90	12.10	12.00	12.10	13.10	13.40	13.20
DO (mg/L)	6.39	6.81	6.81	7.51	7.64	7.09	7.18	6.88
Cond (μS/cm)	110.20	115.30	115.10	115.85	114.60	115.00	114.85	114.70
Turb (NTUs)	0.22	0.04	0.27	0.03	0.26	0.12	0.15	0.64
TSS (mg/L)	0.2000	0.4400	0.6000	0.4000	1.7333	0.6667	2.7333	2.0667
DRP (mg/L)	0.0224	0.0209	0.0251	0.0146	0.0335	0.0329	0.0260	0.0266
TP (mg/L)	0.0260	0.0220	0.0326	0.0275	0.0344	0.0336	0.0305	0.0292
TDP (mg/L)	0.0240	0.0217	0.0298	0.0258	0.0335	0.0332	0.0271	0.0275
PP (mg/L)	0.0020	0.0004	0.0028	0.0017	0.0009	0.0004	0.0034	0.0017
NO3-N (mg/L)	0.144	0.436	0.220	0.325	0.393	0.282	0.222	0.256
TN (mg/L)	0.466	0.595	0.559	0.262	0.607	0.396	0.614	0.504
TDN (mg/L)	0.366	0.541	0.382	0.229	0.605	0.361	0.495	0.476
PN (mg/L)	0.100	0.054	0.177	0.033	0.001	0.036	0.119	0.028
Discharge (L/s)	65.36	59.67	75.17	67.76	88.44	88.96	111.47	92.50
TSS (kg/day)	1.129	2.268	3.897	2.342	13.245	5.124	26.326	16.517
DRP (kg/day)	0.126	0.108	0.163	0.086	0.256	0.253	0.251	0.212
TP (kg/day)	0.147	0.114	0.212	0.161	0.263	0.258	0.294	0.233
TDP (kg/day)	0.136	0.112	0.193	0.151	0.256	0.255	0.261	0.220
PP (kg/day)	0.011	0.002	0.018	0.010	0.007	0.003	0.033	0.014
NO_3-N(kg/day)	0.812	2.247	1.428	1.902	3.005	2.167	2.133	2.044
TN (kg/day)	2.633	3.069	3.629	1.536	4.637	3.046	5.914	4.028
TDN (kg/day)	2.068	2.791	2.482	1.343	4.626	2.771	4.765	3.803
PN (kg/day)	0.565	0.278	1.147	0.193	0.011	0.275	1.149	0.226

B.4 Kaputone Creek shaded site

	1st Field	2nd Field	3rd Field	4th Field	5th Field	6th Field	7th Field	8th Field
pH	7.08	7.09	7.24	7.02	7.06	7.22	7.20	7.18
Temp (°C)	11.90	12.20	11.50	10.90	11.30	13.30	14.40	13.00
DO (mg/L)	8.66	8.43	8.43	9.92	10.14	9.91	9.65	9.73
Cond (μS/cm)	125.20	134.00	134.00	124.85	112.65	123.65	112.75	121.75
Turb (NTUs)	0.30	0.14	0.52	0.60	0.49	0.43	0.49	0.79
TSS (mg/L)	1.4000	0.2000	5.0667	3.6000	2.6000	1.4000	2.2667	3.6000
DRP (mg/L)	0.0317	0.0493	0.0377	0.0439	0.0440	0.0347	0.0372	0.0759
TP (mg/L)	0.0320	0.0520	0.0591	0.0456	0.0455	0.0380	0.0385	0.0821
TDP (mg/L)	0.0318	0.0518	0.0579	0.0449	0.0444	0.0360	0.0373	0.0784
PP (mg/L)	0.0002	0.0002	0.0011	0.0007	0.0011	0.0020	0.0012	0.0037
NO3-N (mg/L)	0.597	1.082	0.844	1.058	1.083	0.921	0.888	0.894
TN (mg/L)	1.906	1.748	2.132	1.909	1.953	1.376	1.786	1.918
TDN (mg/L)	1.866	1.731	1.830	1.832	1.868	1.329	1.726	1.905
PN (mg/L)	0.040	0.017	0.301	0.077	0.085	0.048	0.059	0.013
Discharge (L/s)	25.34	15.35	27.36	19.30	16.02	17.08	22.11	22.94
TSS (kg/day)	3.065	0.265	11.975	6.002	3.599	2.066	4.329	7.135
DRP (kg/day)	0.069	0.065	0.089	0.073	0.061	0.051	0.071	0.150
TP (kg/day)	0.070	0.069	0.140	0.076	0.063	0.056	0.074	0.163
TDP (kg/day)	0.070	0.069	0.137	0.075	0.061	0.053	0.071	0.155
PP (kg/day)	0.000	0.000	0.003	0.001	0.002	0.003	0.002	0.007
NO_3-N(kg/day)	1.307	1.436	1.994	1.764	1.498	1.359	1.697	1.772
TN (kg/day)	4.174	2.318	5.038	3.182	2.703	2.031	3.411	3.802
TDN (kg/day)	4.086	2.295	4.326	3.054	2.586	1.961	3.297	3.776
PN (kg/day)	0.088	0.023	0.713	0.128	0.117	0.070	0.113	0.026

B.5 Kaputone Creek unplanted site

	1st Field	2nd Field	3rd Field	4th Field	5th Field	6th Field	7th Field	8th Field
pH	7.17	7.06	7.12	7.04	7.11	7.15	7.11	7.19
Temp (°C)	11.60	11.00	10.40	9.30	10.10	12.60	13.50	12.50
DO (mg/L)	9.05	8.49	8.49	10.23	10.34	9.46	9.42	9.44
Cond (μS/cm)	122.30	148.40	148.40	136.00	130.25	134.25	131.65	132.50
Turb (NTUs)	0.43	0.22	0.92	0.52	3.34	2.72	2.82	2.95
TSS (mg/L)	0.6600	0.8000	2.1333	1.9333	13.0000	10.4667	1.6000	6.9333
DRP (mg/L)	0.0410	0.0683	0.0859	0.0402	0.0398	0.0548	0.0576	0.0228
TP (mg/L)	0.0440	0.0689	0.0912	0.0433	0.0408	0.0566	0.0593	0.0543
TDP (mg/L)	0.0428	0.0686	0.0869	0.0407	0.0401	0.0551	0.0584	0.0302
PP (mg/L)	0.0012	0.0003	0.0043	0.0026	0.0006	0.0014	0.0009	0.0241
NO3-N (mg/L)	0.679	0.887	0.310	0.948	1.073	0.815	0.694	0.459
TN (mg/L)	1.506	1.378	1.148	1.425	1.785	1.516	1.619	1.241
TDN (mg/L)	1.466	1.329	1.141	1.278	1.609	1.271	1.597	1.181
PN (mg/L)	0.040	0.049	0.007	0.147	0.176	0.246	0.022	0.061
Discharge (L/s)	25.34	15.35	27.36	19.30	16.02	17.08	22.11	22.94
TSS (kg/day)	1.279	1.676	4.687	3.917	18.577	16.264	4.114	13.527
DRP (kg/day)	0.080	0.143	0.189	0.081	0.057	0.085	0.148	0.044
TP (kg/day)	0.085	0.144	0.200	0.088	0.058	0.088	0.153	0.106
TDP (kg/day)	0.083	0.144	0.191	0.082	0.057	0.086	0.150	0.059
PP (kg/day)	0.002	0.001	0.009	0.005	0.001	0.002	0.002	0.047
NO_3-N(kg/day)	1.316	1.859	0.682	1.920	1.533	1.267	1.785	0.895
TN (kg/day)	2.920	2.888	2.523	2.887	2.550	2.356	4.164	2.422
TDN (kg/day)	2.842	2.785	2.507	2.590	2.299	1.974	4.107	2.303
PN (kg/day)	0.078	0.104	0.016	0.297	0.252	0.382	0.057	0.119

B.6 Kaputone Creek grassland site

	1st Field	2nd Field	3rd Field	4th Field	5th Field	6th Field	7th Field	8th Field
pH	7.00	7.03	7.06	7.03	7.07	7.26	7.12	7.09
Temp (°C)	11.50	11.00	10.90	9.50	9.60	12.60	13.90	12.60
DO (mg/L)	9.04	8.67	8.67	10.13	10.49	9.62	9.41	9.57
Cond (μS/cm)	124.80	152.75	152.75	138.50	132.85	135.95	132.70	132.80
Turb (NTUs)	0.34	0.24	0.82	0.45	0.76	1.93	1.89	2.23
TSS (mg/L)	1.2000	0.4000	4.3333	0.8667	4.3333	10.2000	1.7333	5.0000
DRP (mg/L)	0.0522	0.0512	0.0817	0.0439	0.0482	0.0475	0.0539	0.0740
TP (mg/L)	0.0539	0.0528	0.0985	0.0442	0.0555	0.0526	0.0607	0.0784
TDP (mg/L)	0.0530	0.0512	0.0877	0.0440	0.0540	0.0504	0.0580	0.0772
PP (mg/L)	0.0008	0.0016	0.0109	0.0002	0.0015	0.0022	0.0028	0.0012
NO3-N (mg/L)	0.380	0.844	0.535	0.980	0.981	0.742	0.730	0.549
TN (mg/L)	1.296	1.325	1.188	1.663	1.626	1.655	1.533	0.990
TDN (mg/L)	1.258	1.270	1.036	1.609	1.548	1.541	1.478	0.953
PN (mg/L)	0.038	0.056	0.151	0.054	0.078	0.114	0.055	0.036
Discharge (L/s)	23.26	24.32	24.34	23.45	28.45	27.87	32.49	24.32
TSS (kg/day)	2.412	0.840	9.112	1.756	10.653	24.561	4.866	10.506
DRP (kg/day)	0.105	0.108	0.172	0.089	0.118	0.114	0.151	0.156
TP (kg/day)	0.108	0.111	0.207	0.090	0.136	0.127	0.170	0.165
TDP (kg/day)	0.107	0.108	0.184	0.089	0.133	0.121	0.163	0.162
PP (kg/day)	0.002	0.003	0.023	0.000	0.004	0.005	0.008	0.003
NO_3-N(kg/day)	0.763	1.772	1.124	1.985	2.412	1.787	2.050	1.153
TN (kg/day)	2.605	2.784	2.497	3.368	3.998	3.985	4.304	2.079
TDN (kg/day)	2.528	2.667	2.179	3.258	3.806	3.711	4.149	2.003
PN (kg/day)	0.076	0.117	0.318	0.110	0.192	0.274	0.155	0.076

B.7 Styx River shaded site

	1st Field	2nd Field	3rd Field	4th Field	5th Field	6th Field	7th Field	8th Field
pH	6.64	6.58	6.63	6.60	6.57	6.44	6.59	6.54
Temp (°C)	13.00	12.80	12.80	12.20	12.50	13.30	13.60	13.00
DO (mg/L)	6.15	6.46	6.53	6.61	6.60	6.44	6.37	6.59
Cond (μS/cm)	108.70	109.20	108.35	108.25	108.40	108.45	107.95	109.00
Turb (NTUs)	0.17	0.22	0.25	0.51	0.53	0.17	0.31	0.26
TSS (mg/L)	0.6000	1.8000	1.0000	2.2000	1.5333	1.0667	0.4000	1.7333
DRP (mg/L)	0.0261	0.0228	0.0272	0.0219	0.0293	0.0219	0.0242	0.0247
TP (mg/L)	0.0316	0.0300	0.0338	0.0270	0.0328	0.0279	0.0280	0.0268
TDP (mg/L)	0.0304	0.0296	0.0324	0.0259	0.0318	0.0262	0.0280	0.0260
PP (mg/L)	0.0012	0.0004	0.0014	0.0011	0.0010	0.0017	0.0000	0.0008
NO3-N (mg/L)	0.087	0.334	0.114	0.232	0.200	0.102	0.130	0.170
TN (mg/L)	0.326	0.341	0.326	0.252	0.285	0.265	0.389	0.381
TDN (mg/L)	0.326	0.341	0.276	0.244	0.275	0.261	0.389	0.379
PN (mg/L)	0.000	0.001	0.050	0.008	0.010	0.004	0.000	0.002
Discharge (L/s)	23.26	24.32	24.34	23.45	28.45	27.87	32.49	24.32
TSS (kg/day)	16.144	47.610	27.252	58.078	39.112	28.352	10.055	44.883
DRP (kg/day)	0.702	0.602	0.742	0.579	0.748	0.583	0.608	0.639
TP (kg/day)	0.851	0.794	0.922	0.714	0.837	0.742	0.704	0.694
TDP (kg/day)	0.818	0.783	0.883	0.685	0.812	0.697	0.704	0.673
PP (kg/day)	0.033	0.011	0.039	0.029	0.026	0.045	0.000	0.022
NO_3-N(kg/day)	2.338	8.845	3.118	6.135	5.096	2.698	3.265	4.410
TN (kg/day)	8.780	9.032	8.876	6.661	7.266	7.034	9.781	9.866
TDN (kg/day)	8.784	9.016	7.523	6.440	7.011	6.929	9.780	9.802
PN (kg/day)	-0.005	0.017	1.352	0.221	0.254	0.105	0.001	0.064

B.8 Styx River grassland site

	1st Field	2nd Field	3rd Field	4th Field	5th Field	6th Field	7th Field	8th Field
pH	6.89	6.81	6.82	6.77	6.75	6.74	6.71	6.72
Temp (°C)	13.00	12.40	12.60	12.20	12.20	13.40	13.80	12.80
DO (mg/L)	7.58	7.88	7.88	8.30	8.28	8.06	8.07	8.22
Cond (μS/cm)	112.65	113.75	113.75	112.35	112.30	113.30	113.00	113.40
Turb (NTUs)	0.39	0.59	0.72	0.88	0.99	0.38	0.45	0.98
TSS (mg/L)	4.2000	5.1333	3.7333	5.2000	3.4000	2.6000	1.6667	4.0000
DRP (mg/L)	0.0280	0.0228	0.0335	0.0256	0.0230	0.0420	0.0297	0.0361
TP (mg/L)	0.0300	0.0260	0.0366	0.0348	0.0273	0.0479	0.0349	0.0375
TDP (mg/L)	0.0300	0.0260	0.0354	0.0324	0.0262	0.0442	0.0307	0.0367
PP (mg/L)	0.0000	0.0000	0.0013	0.0025	0.0011	0.0037	0.0041	0.0007
NO3-N (mg/L)	0.114	0.339	0.101	0.341	0.239	0.358	0.157	0.125
TN (mg/L)	0.406	0.451	0.460	0.348	0.380	0.486	0.435	0.471
TDN (mg/L)	0.406	0.451	0.369	0.343	0.376	0.383	0.353	0.350
PN (mg/L)	0.000	0.000	0.091	0.004	0.004	0.102	0.082	0.122
Discharge (L/s)	311.75	379.50	386.75	521.83	495.50	530.93	440.08	429.49
TSS (kg/day)	113.128	168.314	124.749	234.447	145.557	119.269	63.372	148.432
DRP (kg/day)	0.753	0.747	1.120	1.153	0.987	1.928	1.131	1.338
TP (kg/day)	0.809	0.853	1.224	1.570	1.167	2.196	1.326	1.390
TDP (kg/day)	0.809	0.853	1.182	1.459	1.121	2.026	1.169	1.363
PP (kg/day)	0.000	0.000	0.042	0.111	0.046	0.170	0.157	0.027
NO_3-N(kg/day)	3.063	11.102	3.375	15.365	10.236	16.436	5.962	4.620
TN (kg/day)	10.943	14.792	15.375	15.670	16.252	22.277	16.553	17.489
TDN (kg/day)	10.943	14.800	12.318	15.476	16.090	17.578	13.439	12.973
PN (kg/day)	0.000	-0.008	3.056	0.194	0.162	4.698	3.114	4.516

B.9 Styx River unplanted site

	1st Field	2nd Field	3rd Field	4th Field	5th Field	6th Field	7th Field	8th Field
pH	7.28	7.25	7.25	7.41	7.14	7.61	7.48	7.22
Temp (°C)	12.00	11.90	11.80	11.40	12.00	13.30	13.70	12.50
DO (mg/L)	8.32	9.50	9.50	10.01	10.14	10.34	10.02	10.05
Cond (μS/cm)	122.80	132.55	132.55	125.60	121.35	123.90	122.65	122.80
Turb (NTUs)	0.61	1.02	1.17	0.85	0.88	0.49	0.57	1.20
TSS (mg/L)	2.4000	2.4067	4.0000	2.2000	1.4667	1.1333	0.3333	2.4000
DRP (mg/L)	0.0466	0.0418	0.0356	0.0365	0.0398	0.0475	0.0390	0.0380
TP (mg/L)	0.0493	0.0440	0.0427	0.0418	0.0442	0.0517	0.0420	0.0419
TDP (mg/L)	0.0477	0.0419	0.0396	0.0389	0.0423	0.0510	0.0400	0.0388
PP (mg/L)	0.0016	0.0021	0.0031	0.0029	0.0019	0.0008	0.0020	0.0031
NO3-N (mg/L)	0.207	0.348	0.119	0.313	0.290	0.202	0.222	0.218
TN (mg/L)	0.553	0.627	0.399	0.391	0.466	0.413	0.267	0.460
TDN (mg/L)	0.551	0.601	0.303	0.369	0.430	0.388	0.238	0.437
PN (mg/L)	0.002	0.026	0.096	0.022	0.036	0.024	0.029	0.023
Discharge (L/s)	N/A	847.54	1003.41	1359.03	875.82	1060.82	873.07	864.66
TSS (kg/day)	N/A	176.235	346.779	258.325	110.983	103.876	25.144	179.297
DRP (kg/day)	N/A	3.057	3.088	4.291	3.012	4.354	2.945	2.836
TP (kg/day)	N/A	3.223	3.703	4.909	3.346	4.742	3.169	3.131
TDP (kg/day)	N/A	3.071	3.432	4.572	3.201	4.671	3.019	2.899
PP (kg/day)	N/A	0.152	0.271	0.337	0.145	0.070	0.151	0.232
NO_3-N(kg/day)	N/A	25.469	10.299	36.706	21.937	18.478	16.769	16.271
TN (kg/day)	N/A	45.883	34.568	45.960	35.243	37.834	20.114	34.375
TDN (kg/day)	N/A	44.015	26.259	43.379	32.536	35.600	17.924	32.643
PN (kg/day)	N/A	1.868	8.308	2.581	2.707	2.234	2.190	1.732