Investigating innovative drainage management tools to reduce nutrient runoff in surface drains in Rangitikei sand country

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Abstract

Approximately 2.5 million hectares of land in NZ is currently artificially drained. However, while being beneficial to the system, drains have been identified as providing a direct conduit for nutrient loss from agricultural lands to receiving waters, potentially leading to detrimental effects instream. The aim of this thesis was to investigate targeted drainage management tools that can reduce nutrient runoff in surface drains in an intensive sand country dairy farm in the Rangitikei district. Drainage patterns and water quality was characterised, and then macrophyte management, real time swapping; and the potential for harvesting and recycling drainage water was investigated. Surface water samples were collected and analysed weekly for a period of nine months from June 2018 to February 2019 to characterise the drainage patterns and drainage water quality. Monitoring for real time swapping and macrophyte management trials took place between December 2017 and February 2018. Spatial and temporal trends were then analysed, and the potential of harvesting and reuse of drainage water for irrigation was assessed.

Nitrate-N was identified as a problem pollutant in this study, with concentrations varying from 0.06 g/m^3 to 5.96 g/m^3 over the course of a year. OVERSEER estimated an average nitrogen loss of 25.9 g/m^3 from the root zone to drainage waters. However, an overall average of only 3.3 g/m^3 was observed. During the swapping trial period (January – March), average Nitrate-N levels in both the groundwater and drain waters were consistently low (<0.20 g/m^3), meaning that swapping had no effect. Under the macrophyte trial, there was suggestion towards nutrient uptake with increasing macrophyte cover, but ongoing research is needed to find a definitive relationship. Under the Harvesting and Recycling exercise, it was found that nitrate-N attenuation costs are influenced by the concentration of N in the drainage water, and if there is an existing irrigation system. It is more cost effective ($0.34 per Kg N attenuated/yr) if the drainage nitrate-N concentration is higher and is recycled over previously un-irrigated land.

These research findings will help to develop appropriate in-field or edge-of field management practices, and inform nutrient management plans for intensified land use to maintain or enhance water quality in the region. Potential progression could be to

further study options to control and treat surface drainage water by controlled drainage, combined with wetlands, in order to reduce nutrient loads from intensive farms in the region.

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Chapter 1

General Introduction

Chapter 1: Introduction

1.1 Background

Productive farms and clean waterways throughout New Zealand are central to our national identity and international brand as 'Clean and Green' producers. Freshwater is a precious natural resource that underpins not only our productive farms but also our social and cultural values, recreational lifestyle (swimming and fishing) and international eco-tourism. However, the increase in intensive primary production over the last 20 years in New Zealand has been associated with a parallel decrease in water quality. From 2013 to 2017, rivers in pastoral land median nitrate-N levels were modelled to be 9.7 times higher, DRP levels 3.4 times higher, and E. coli levels 14.6 times higher compared with rivers running through native land (Ministry for the Environment & Stats NZ, 2019). In the Manawatu-Wanganui Region, between 50-60% of stream sites failed to meet the regional targets for SIN and DRP (Horizons Regional Council, 2019a). While nitrogen and phosphorous are found naturally in the environment, diffuse sources, such as fertilizer applications, urine patches and farm effluent spreading; all have the potential to leach these nutrients to groundwater and surface waters. Elevated levels of these nutrients in surface waters can lead to major environmental concerns, including but not limited to; nuisance algal blooms, oxygen deficiency, loss of habitat, and a loss of biodiversity.

Traditionally, nutrient management in agriculture has been concerned with optimising the economic return from nutrients used for primary production (Beegle, Carton, & Bailey, 2000). Today, the agronomic and economic requirements of nutrient management remain central, but in addition, the process must consider the potential impact of these nutrients on environmental quality. Therefore, the major challenge facing us today is how to balance the economic and societal welfare benefits that can be gained by growing agriculture with the environmental and eco-tourism benefits of preserving the quality of freshwater ecosystems?

While nutrient-management plans and best management practices have resulted in benefits to farmers and society, implementation of these plans has not been optimal. Several factors have been identified as critical to the successful implementation of nutrient-management planning; among them the use of targeted approaches to managing critical nutrient flow pathways. Today, artificial drains make up the majority of lowland waterways; it is estimated that 2.5 million hectares of land in NZ are currently artificially drained (Manderson, 2018). These drains act as direct conduits for nutrients, sediment and bacteria lost on-farm, and therefore, drainage management should be considered an intrinsic component of successful and sustainable agriculture throughout the country.

The coastal sand belt located in the Rangitikei area of the Horizons region in the North Island is an example of land that was previously sand dunes; but is now in agriculture with help of irrigation and drainage practices. There is a year-round high-water table on largely undeveloped soil. With the aid of modified drainage and irrigation systems, many of the Rangitikei's coastal sandy soils are farmed intensively for dairying, vegetable production, cropping and other intensive land uses. The favourable climate, coarse textured well drained sandy soils, intensive land uses and use of irrigation result in highly productive farm systems which often have relatively large nitrogen leaching losses. A study in the lower Rangitikei area found that the groundwater appears to have a strong reducing environment, conducive for potential denitrification of nitrogen leached from the soil surface (SB Collins et al., 2016). However, a recent study in this area found that the open surface drains that are necessary to lower the water table accumulate nitrate-N and could be a significant contributor to the local streams (Smith, Singh, & Matthews, 2017). It raises the question of whether having deep open drains is beneficial in terms of farm nutrient management and water quality outcomes. Smith et al., 2017 also found that dissolved reactive P was more common in shallow groundwater environments.

For economically sustainable agricultural production to have reduced water quality impacts, it is of increasing importance to develop more efficient and targeted approaches to manage critical nutrient flow pathways including drainage management. Most nutrient management methods focus on how to make a difference above the soil, such as most planting riparian strips and fencing off waterways. My thesis investigates potential of innovative drainage management practices; and looks at the development of targeted management tools that can reduce nutrient runoff in surface drains in an intensive sand country dairy farm.

1.2 Study Objectives

My thesis first characterises the drainage patterns and drainage water quality, and then investigate potential targeted novel in-field and edge-of-field practices to reduce nutrient runoff in drainage waters, and their effects on downstream water quality and ecology on an intensive dairy farm in the Rangitikei district, Manawatu-Wanganui Region. The specific objectives are as follows:

- Review various drainage management practices
- Characterise the drainage flows and water quality patterns
- Assess the potential of harvesting and reuse of drainage water for irrigation
- Assess the management of macrophytes as a tool for nutrient uptake in drain waters

This research aims to help develop appropriate in-field or edge-of-field drainage management practices and inform nutrient management plans for intensified land use to maintain or enhance water quality in the region.

1.3 Thesis Outline

There are five chapters to this thesis. Chapter 1 outlines the study background, aims and objectives. Chapter 2 reviews literature and previous studies on targeted nutrient management methods. Chapter 3 describes the case study farm, including climate, soils and land use, and outlines the methods used to model or monitor drainage flow patterns and drainage water quality parameters on the case study farm under the three different trials (swapping, macrophyte management, and harvesting and recycling). Chapter 4 presents the results from the monitoring and assessments of the studied drainage management practices; and discusses the implications of these results regarding drainage management practices in the study region and across New Zealand. Chapter 5 summarises the research findings and concludes by suggesting further potential avenues of investigation to reduce nutrient loads from intensive farms in the region.

Chapter 2

Literature Review

Chapter Two: Literature Review

2.1 Agriculture and Water Quality

There is increasing recognition globally that non-point source pollution of waterways from agricultural production is major cause of water quality degradation. Intensive agriculture is known to emit significant amounts of nutrients, particularly nitrogen (N) and phosphorus (P), as well as faecal bacteria and sediment (Monaghan et al., 2007). While these emissions are not large by agronomic standards (at least for P) – or relative to the amounts residing within the soil system, the transfer of these pollutants from land to water can result in significant water quality impairment (Monaghan et al., 2007).

Throughout New Zealand, many lowland waterways have been impacted by intensive agricultural land use. Monitoring shows that nitrate trends worsened in 61% of pastoral streams between 1994 and 2013; and that nitrogen leaching from agricultural soils was estimated to have increased 29% from 1990 to 2012 (Gadd, 2016; Larned, Snelder, & Unwin, 2017). Trends in dissolved reactive phosphorus (DRP) were also shown to have worsened at 21% of pastoral sites for the period 1994-2013 (Gadd, 2016; Larned et al., 2017). Between 2008 and 2017, up to 40% of monitored sites throughout the country show degrading trends in terms of Total Nitrogen and DRP (LAWA, 2019). In the Manawatu-Wanganui region specifically, between 50-60% of monitored stream sites failed to meet the regional targets for soluble inorganic nitrogen (SIN) and DRP (Horizons Regional Council, 2019b).

In recent years, New Zealand has seen a pronounced change in its agricultural landscape, with dairy cow numbers increasing by 69%, from 3.84 million in 1994 to 6.49 million in 2015 (Statistics NZ, 2017). By 2016, dairy was New Zealand's largest export sector; comprising of 18% of the country's total goods and service exports (DCANZ, n.d.). Much of the dramatic land use change has occurred in regions such as the Waikato, Southland, and Canterbury (where 70% of the country's irrigated agriculture is currently located) (Statistics NZ, 2017). Although the economic and social benefits of this land use change are widely acknowledged, there has been increasing public concern about the environmental impacts resulting from this. The change from sheep to dairy farming is typically accompanied by increased use of fertiliser, pesticides, and the production of large quantities of animal excreta, deposited both in the field and at the milking shed (Monaghan et al., 2010). Many studies have shown that this excreta is an important source of nutrients and faecal bacteria in waterways, transferred via either overland flow or subsurface drainage pathways, including ones by Monaghan, Paton, Smith, Drewry, and Littlejohn (2005) and Thorrold, Rodda, and Monaghan (1998).

Whilst dairy cows are not the sole contributor to water quality impairment, lactating cows do excrete about 70% of the nitrogen they consume in urine, and inappropriate management of the dairy farm system has the potential to cause significant stream and river pollution (Monaghan et al., 2007). While N and P do occur naturally and are necessary for plant growth, around 30 percent of North Island dairy farms have twice the level of soil nutrients required (Waikato Regional Council, n.d.). Up to one third of the nitrogen entering soils on intensive farms may end up leaching down through the soil into ground water (Waikato Regional Council, n.d.; Mercer, Ledgard & Power, 2011). This nitrogen-rich ground water will eventually flow into and pollute waterways. High levels of nutrients can stimulate algal blooms and nuisance algae growths in the waterways. Excessive periphyton in water can decrease oxygen levels, prevent light from penetrating water, and change the composition of freshwater plant and animal species that live there (Ministry for the Environment & Stats NZ, 2019). High concentrations of nitrogen can be toxic to aquatic species and make water unsafe to drink. The Ministry for the Environment defines targets for periphyton biomass (120 mg/m^2) and cover (30% filamentous algae; 60% cyanobacteria) for gravel/cobble bed streams in New Zealand (Biggs, 2000).

Increasing water quality concerns has triggered proposals and discussions about the development of policy and plans to reduce nutrient losses from agricultural lands to freshwaters. Water quality limits are being set for each catchment in New Zealand under the recently amended National Policy Statement for Freshwater Management (NPS-FM) of 2014. Through this, the agricultural sector is required to take action to reduce their contribution to the degradation of water quality, including N and P pollution (A. J. Daigneault, F. V. Eppink, & W. G. Lee, 2017). Therefore, there is a strong need to find ways to decrease nutrient losses from agricultural land to surface waters. Freshwater is a precious natural resource that underpins not only our productive farms but also our social and cultural values, recreational lifestyle (swimming and fishing) and international eco-tourism. Best on-farm Management Practices (effective and practical methods for preventing or reducing non-point source pollution to help achieve water quality goals) alone are not likely to reduce losses sufficiently, so additional methods of nutrient removal from drainage water are needed (Monaghan et al., 2010). There is a need to provide farmers with cost-effective, practical alternatives to reduce nutrients, especially N and P, losses from farms to the wider environment.

2.2 Nutrient Flow Pathways

The phrase 'nutrient flow pathways' is a blanket term for the transmission pathways from nutrient source to the receiving waterways. The main pathways to water bodies in this context are overland flow (erosion and runoff), subsurface drainage (tile or mole drains), open channel drainage, and percolation to groundwater. Different pathways apply more or less to different nutrients and land types; and can be influenced by changes to water flow via subsurface and surface drains (Hatch, Goulding, & Murphy, 2002). Drainage systems (both surface and subsurface) provide direct conduits that can transport nutrients (N and P) from agricultural fields to surrounding natural waterways.

2.2.1 Nitrogen

Nitrogen (N) is integral to the natural processes of the environment and plays a multitude of roles. As described by (Follett, 2008), it is 'ubiquitous' with all things and is one of the most yield-limiting nutrients. For example, it is essential to agricultural production as vegetative production is reliant on its presence. Atmospheric dinitrogen (N_2) makes up 78% of the atmosphere but is inert. To be accessible in the soil, it must cycle from this form to organic soil forms, then to plant-available mineral forms (Follett, 2008). It is converted to different atmospheric, terrestrial and aquatic forms via the biological and chemical processes in the nitrogen cycle, as shown in Figure 2.1.

Figure 2. 1 Nitrogen cycle overview (Follett, 2008)

Elevated levels of nitrate in freshwaters pose risks to both ecological and human health. There are standards for nitrate concentrations in water bodies, assigned by the World Health Organisation (WHO). For water to be potable, the concentration must be at or below 11.3 milligrams of nitrate-N per litre of water (or 50 milligrams of nitrate per litre of water) (Follett, 2008). In New Zealand, to meet lowland freshwater stream standards for ecological health, the concentrations in water need to be less than 0.44 g/m^3 nitrate-N and 0.021 g/m^3 ammoniacal-N (Davies-Colley, 2000). Due to nitrogen's integral role farm management in New Zealand, it can be difficult to abide by the recommended levels in agricultural catchments and their downstream waterways.

Both livestock urine and defecation deposits are major sources of concentrated leached nitrate (Clark, Caradus, Monaghan, Sharp, & Thorrold, 2007; Houlbrooke, Horne, Hedley, Hanly, & Snow, 2004). The introduction of nitrogen fixing clovers has also increased the amount of nitrate leaching from pastoral catchments, as has the practice of using nitrogen fertilisers and effluent irrigation (Clark et al., 2007). Effluent leachate however, are generally only a problem when application exceeds field capacity, or when inadvertently applied directly into the waterways. A large amount of research in New Zealand and overseas over the past 3 decades has clearly shown that the amount of N excreted by animals, and in particular urine N, is the most important determinant of N losses (including leaching, runoff and gaseous losses) from pastoral farms (Di & Cameron, 2002; Ledgard, 2001). Consequently, it can be argued that on these farms, the amount of N excreted by animals is the primary driving factor of on farm N losses, rather than inefficiencies related to N fertiliser usage (Monaghan et al., 2010).

In pastoral grazing system, a significant amount of nitrogen taken in by an animal is returned to the soil by excreta, but in concentrated hotspots throughout the pastures and yards. Around 70% of the excreta-N is returned in urine patches, and quickly converts to ammonium ions then nitrate (DairyNZ, 2013). Leaching occurs easily as nitrate is negatively charged, so is repelled by similarly charged soil surfaces and moves freely through most soils (Rivett, Buss, Morgan, Smith, & Bemment, 2008).

Nitrate leaching via drainage through the soil profile is the dominant form of N loss on intensively-grazed farms. Movement is dictated by mass flow through soil profiles, and the extent of retention depends on soil and climatic conditions. On poorly drained soil, excess nitrate which does not infiltrate the soils can end up in the waterways via surface run off (Neilen, Chen, Parker, Faggotter, & Burford, 2017). These soil types become high risk as there is minimal soil-water contact, thus providing little opportunity for the filtration or adsorption. A second pathway is vertical flow via matrix flow, or bypass flow, through macropores to the subsurface. If there is mole and tile drainage, there will be cracks that encourage bypass flow, reducing soil water contact. In the North Island of New Zealand, approximately 50% of flat to rolling land is underlain with soils with a high potential for bypass flow (McLeod, Close, & Collins, 2005).

2.2.2 Phosphorus

Excessive fertiliser application, defecation deposits, and effluent irrigation are all sources of P (Aye, Nguyen, Bolan, & Hedley, 2006; Clark et al., 2007). If best practice is not followed, then P losses from fertilisers can account for most P losses from the farm (Hart, Quin, & Nguyen, 2004). In saying that, McDowell et al. (2019) recently suggested that median concentrations of phosphorus in New Zealand streams and rivers are improving (attributed to a greater use of phosphorus loss mitigation strategies and policy instruments).

It is important to regulate P applications as imbalances can easily occur, leading to losses from the agricultural system (Fig. 2.2). This is a problem because only a small amount of P is needed to cause water quality problems in receiving waterways. Lowland DRP concentrations need to be below 0.010 g/m^3 to avoid possible adverse ecosystem effects (Davies-Colley, 2000). It is often a limiting nutrient for nuisance weed and algal growth in surface waters (Leinweber & Turner 2002). Therefore, when it is no longer limiting, provided other requirements are met, increased P concentrations can lead to growths that smother ecosystems and change water chemistry (Leinweber et al. 2002).

Figure 2. 2 Phosphorus transport processes from soil (Leinweber et al. 2020)

Compared to nitrogen leaching however, phosphorus leaching from agricultural systems is generally much less (Owens et al., 2007). The main way that excess phosphorus enters the waterways is in particulate form via surface runoff, or through land erosion events (such as landslips and stock trampling) to which the phosphorus is bound to the sediment (Owens et al., 2007). If there is a higher water table, soil will have a lower infiltration capacity, and therefore surface runoff can be generated more easily (Zimmer & Madramootoo, 1997). This can subsequently carry dissolved reactive phosphorus (DRP) from agricultural lands to receiving waterways, becoming bioavailable in streams, rivers and lakes, and thus supporting algae growth.

A study on the loss of dissolved and particulate P by subsurface drainage by Grant et al. (1996) discussed how the level of losses via the subsurface depended on a variety of factors like soil type and the height of groundwater beneath the surface. They linked high P losses in drainage to the large amounts of manure on the site from grazing cattle and the permanently high water table beneath it. As both of these factors are relevant on dairy farms in the coastal sand belt, it would be beneficial to find whether they increase the rate of P losses to the subsurface. A study by Smith et al. (2017) on a farm on the coastal sand belt however, found that DRP levels in the surface water drains were generally insignificant, except for directly after a rainfall event. It was thought that rather than surface runoff, the phosphorus could entering the waterways via leaching particularly in areas with low anion sorption capacities.

2.3 Agricultural drainage across New Zealand

Agricultural drainage is simply the process of removing excess water from poorly drained agricultural lands. It is necessary to create a well-aerated root environment for plant growth and increased production. Drain installation can lead to major environmental concerns however. Multiple studies highlight how artificial drainage is a key pathway for nutrient loss from agricultural lands to receiving waters. Modified surface and subsurface (mole and tile) drainage systems provide benefits of removing excess soil water for improved plant growth during wetter periods, but also create critical flow pathways of increased nutrient losses from farm production systems (Monaghan, Paton, Smith, & Binet, 2000; Monaghan, Smith, & Muirhead, 2016; Skaggs, Breve, & Gilliam, 1994).

Because drains are also usually part of a network that eventually feed into larger rivers and streams, inputs into them are usually cumulative and can directly impact the receiving waterways. The importance of an individual drain therefore, may be negligible until it is considered for its role in the functioning of the wider waterway network (Hudson & Harding, 2004).

2.3.1 Scope

Today, agricultural drainage makes up the majority of lowland waterways; it is estimated that 2.5 million hectares of land in NZ is currently artificially drained (Fig. 2.3) (Manderson, 2018). As mentioned above, these drains act as direct conduits for nutrients, sediment and bacteria lost on-farm, and therefore, drainage management should be considered an intrinsic component of successful and sustainable agriculture throughout New Zealand.

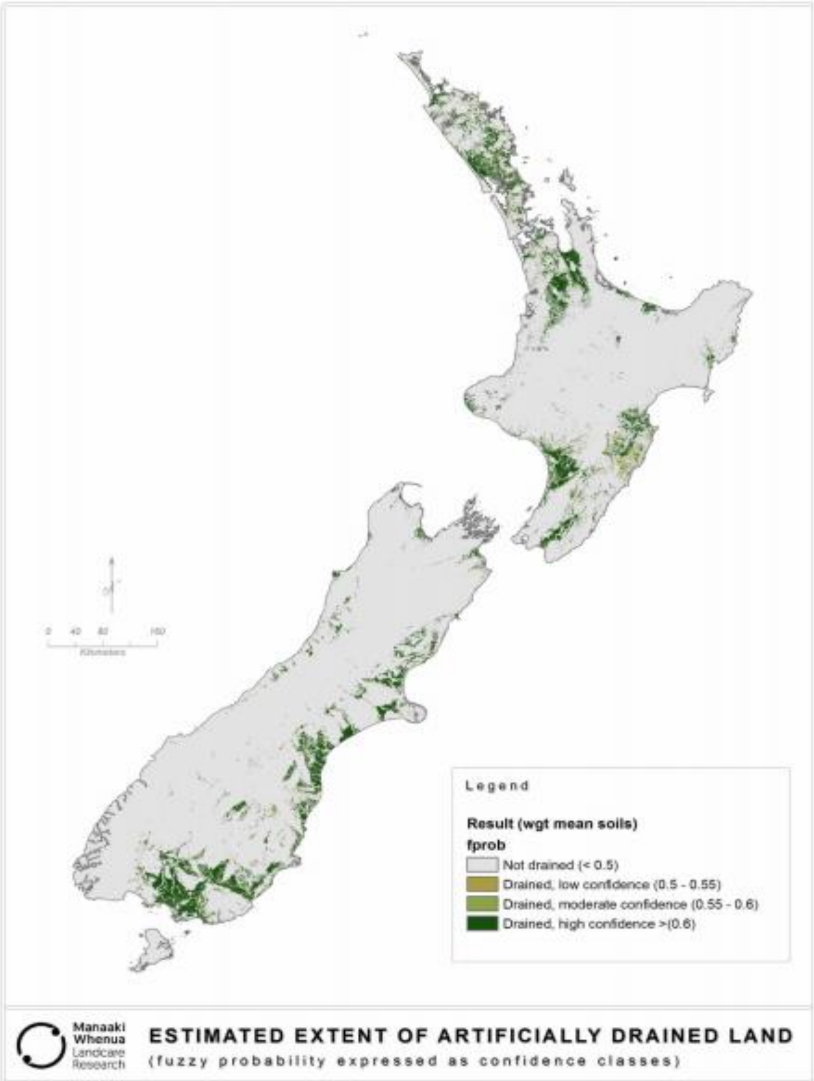

Figure 2. 3 The estimated extent of artificially drained land in NZ (Manderson, 2018)

The presence of artificial drainage is influenced by land use type and intensity (Manderson, 2018). In the Southland region, for example, soils are predominantly fine textured and natural drainage is slow. Because of this, mole-pipe drainage networks are used (Monaghan, 2014). Pearson (2015) estimated that artificial drainage covers potentially three-quarters of agricultural lands in this area. In coastal sand country that is supported by shallow groundwater flows on the other hand, open surface drains are used, such as on the Rangitikei Coastal Belt (Smith et al., 2017).

2.3.2 Open surface drainage management and maintenance

As mentioned above, drains' primary function is to facilitate the removal of excess water efficiently and quickly from the soil profile. With the exception of peatland, the goal is to lower the water table 30 cm below the soil surface within 24 hours, or 50 cm within 48 hours of rain stopping (anything more or less can reduce pasture health) (Dairy NZ, 2015). Good drainage management therefore simply advocates fencing, grass or plantings along the bank, no pugging, little build-up of sediment, and a gently sloping V-shaped channel (Dairy NZ, 2015).

Nuisance macrophyte growth is a particular problem in New Zealand agricultural drains (Hudson & Harding, 2004). Therefore, when excessive macrophyte growth occurs, weed management is commonly undertaken. The main practices currently used by Regional and District Councils to maintain drains fall into three broad categories: mechanical maintenance, chemical control, and biological control. These options all have a level of impact on the waterway, and costs and benefits must be weighed up and the most appropriate methods selected for the specific site and target species (Hudson & Harding, 2004). Additionally, under the RMA, Regional Councils have the responsibilities to manage the effects and use of watercourses or bodies, including that of drainage runoff. This means that surface drains over one metre wide should be treated as waterways and their management adhere to the strict conditions of consent from the council for clearance, fencing and planting (Dairy NZ, 2015). Furthermore, under the Sustainable Dairying: Water Accord all drains should be included in farm riparian plans - whether natural, straightened-natural or artificial- and require fencing. The only exceptions are -as mentioned above- drains less than a metre wide and a foot deep (Dairy NZ, 2015).

2.4 Drainage Management Technologies and Practices

The Dairy Best Practice Catchment programme was undertaken in New Zealand from 2001 to 2011; and provided an opportunity to investigate long-term environmental responses to catchment-scale stream rehabilitation. It involved applying best on-farm management practices throughout five degraded lowland wadeable streams across New Zealand. Water quality was intensively monitored over 10 years. The key findings of the programme were that while stock exclusion fencing and improved farm effluent management reduced suspended sediment and total phosphorus levels, nitrate levels either remained constant or increased (Holmes et al., 2016).

It is evident therefore that Best Management Practices alone are not likely to reduce losses sufficiently, so additional methods of nutrient removal from drainage water are needed (Monaghan et al., 2010). There is a need to provide farmers with economic alternatives to current practices that are leading to the leaching of nutrients, especially N and P, from farms to the wider environment. Low-cost and simple technologies are needed to reduce agricultural export of excess nutrients to waterways. NZ has approximately 2.5 m ha of land with potential for targeted managed of drainage waters (Manderson, 2018). Among the most promising edge-of-field technologies for reducing nutrient loss from drainage waters, are constructed wetlands, controlled drainage, saturated buffers, and bioreactors.

2.4.1 Constructed Wetlands

Constructed wetlands are often cited as being effective at reducing nutrient loads (Fisher & Acreman, 2004). They do this by encouraging sedimentation, sorbing nutrients to sediments, taking up nutrients to plant biomass, and enhancing denitrification (Fisher & Acreman, 2004). In an attempt to mimic natural wetlands and reduce nutrient losses from farm drainage systems, constructed wetlands have been trialled as a way of removing nutrients and other pollutants from drainage water. Constructed wetlands are not a novel idea, as they have been successfully used for treatment for various types of municipal wastewaters for more than four decades. The use of constructed wetlands to remove nitrogen from agricultural drainage waters was first proposed during the late 1980s (Mitsch, 1992; Van der Valk & Jolly, 1992).

Different types of pollutants can be removed by installing constructed wetlands, through a complex inter-connected system of rooted plants, gravel or soil media, bulk water and biomass population (Fountoulakis, Terzakis, Kalogerakis, & Manios, 2009). There are two basic types of constructed wetlands: surface flow, and subsurface flow systems (Kadlec & Wallace, 2008). Surface flow wetlands are similar to natural wetlands, with shallow flow of water (usually less than 60 cm deep) over saturated soil substrate.

Subsurface flow wetlands mostly employ gravel as the main media to support the growth of plants; water flows vertically or horizontally through the substrate where it comes into contact with microorganisms, living on the surfaces of plant roots and substrate (Kadlec & Wallace, 2008), allowing for pollutant removal.

When comparing different studies, the nutrient removal efficiency varies considerably, which makes it difficult to assess the extent to which wetland creation is an efficient measure to reduce eutrophication. Most studies however, suggest that constructed wetlands treating agricultural drainage waters are effective for both nitrogen and phosphorus removal. For example, for non-point source (mostly agricultural and urban runoff) nutrient removal in eastern USA, (Mitsch, 1992) suggested that sustainable removal rates would be 10 to 40 g/ m^2/year for nitrogen and 0.5-5 g/m^2/year for phosphorus, and based this on wetland studies in the last quarter of the 20th century. In New Zealand, a recent review of case studies in New Zealand, undertaken by NIWA for DairyNZ, found that seepage wetlands can reduce the amount of nitrate by up to 75 to 98% (K. Rutherford, Hughes, & McKergow, 2017). A study from a 6 year old Waikato constructed wetland found that nitrate-N concentrations declined from 10.4 to 3.4 mg N, with less than one quarter of the nitrate-N processed sequestered into the wetland plant, and the balance permanently removed by denitrification (Matheson & Sukias, 2010). Another study on constructed wetlands treating subsurface drainage from dairy pastures in Waikato (rain-fed) and Northland (irrigated), New Zealand, found that median nitrate-N concentrations of about 10 g/m^3 in the drainage inflows were reduced by 15 to 67% during passage through the wetlands and annual nitrate-N loads reduced by 16 to 61% (C. Tanner, Long Nguyen, & Sukias, 2003).

Although current literature agrees that constructed wetlands are effective at removing N, wetlands' capacity to remove P tends to be much lower and more variable, and, depending on specific site factors, they can be either sources or sinks of P (Pant, Reddy, & Lemon, 2001). Wetland sediments will generally have a finite capacity to adsorb P and, once saturated, they will stop adsorbing P and can actually become a P source if physicochemical conditions change (Pant et al., 2001). In studies of constructed wetlands treating tile drainage from grazed pastures carried out in New Zealand (Sukias, Tanner, & Stott, 2006; C. Tanner, Nguyen, & Sukias, 2005), the P loads measured at the outlets of farm drainage wetlands had remained higher than at the inflow over periods of 3-5 years, meaning that these wetlands have actually been net sources of P. Similarly, another Waikato study found wetlands can actually be net sources (putting out more than goes in) of some forms of nitrogen at times of higher flows (J. Rutherford & Nguyen, 2004).

A couple of disadvantages face the implementation of constructed wetlands. For example, a study in the Waikato region of New Zealand's North Island has highlighted the potential for flows to be transported across the wetland surface during high rainfall events with minimal nitrate removal (Burns & Nguyen, 2002). Similarly, channels reduce contact time between water and soil, reducing the effectiveness of denitrification. Furthermore, they can be expensive to install, although some NZ regional councils do offer funding for the development of constructed wetlands. Location and area are particularly important; NIWA scientists found that 1-5% of land in a catchment is needed as a wetland to achieve 20-50% reduction of nitrogen levels (C. Tanner, Howard, C., 2011).

2.4.2 Woodchip Bioreactors

Although constructed wetlands are commonly used to remove nutrients, denitrification in these systems can be limited by carbon availability. An emerging solution are denitrifying woodchip bioreactors, which intercept runoff, tile drainage, and/or shallow groundwater from agricultural fields for treatment before leaving the drain to enter a surface water body (Schipper, Robertson, Gold, Jaynes, & Cameron, 2010). They are designed to provide ideal conditions for denitrification and address the carbon limitation issue in a relatively inexpensive approach. They come in various forms, but essentially, bioreactors are subsurface trenches filled with a carbon source, mainly wood chips, through which drainage water is directed (Fig. 2.4) (L. E. Christianson & Helmers, 2011; Pluer, Geohring, Steenhuis, & Walter, 2016; Schipper et al. 2010). The carbon source in the trench serves as a substrate for denitrifying bacteria which break down the nitrate (Christianson & Helmers, 2011; Pluer et al., 2016). Woodchips are by far the most widely used materials in denitrification bioreactors and have shown the ability to deliver long-term (anywhere from 10 to 20 years) nitrate removal while requiring minimum maintenance (Christianson & Helmers, 2011). An anaerobic environment must also be provided for the denitrifying bacteria; and so an outlet control structure is needed to retain water for enough periods in the trench (Christianson & Helmers, 2011; Pluer et al., 2016).

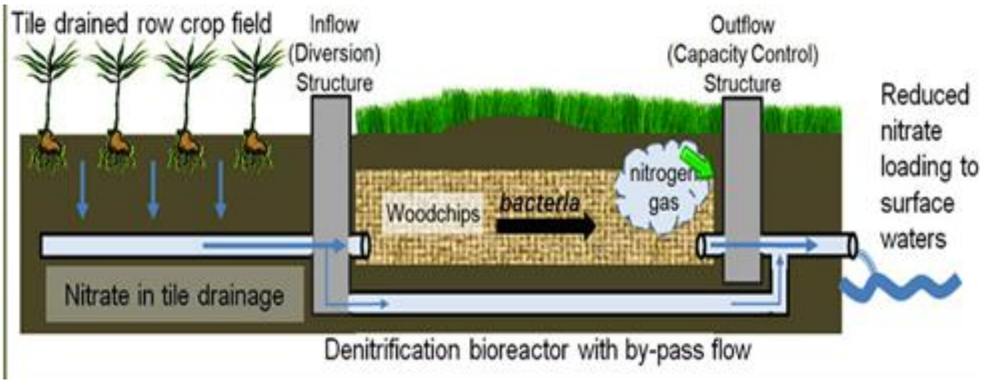

Figure 2. 4 Descriptive illustration of a woodchip bioreactor (Christianson & Helmers, 2011)

The use of organic matter to remove agricultural N was first reported in 1971 (Williford, McKeag, & Johnston, 1971). There has been significant development and improvement of this edge-of-field management practice in the past 20 years (Addy et al., 2016; Christianson & Helmers, 2011; Schipper et al., 2010). To date, the main denitrifying bioreactors types are denitrification walls (intercepting shallow groundwater), and denitrifying beds (intercepting concentrated discharges) (Christianson & Helmers, 2011; Schipper et al., 2010). Denitrification walls consist of trenches penetrating 1 to 2 m into the groundwater, dug perpendicular to groundwater flow paths between the edge-of-field and stream/drain, and filled with either 100% wood chips or sawdust mixed with soil. “Denitrification beds” are containers (sometimes lined) that are filled with wood chips and receive nitrate in concentrated discharges wastewaters or drainage discharge. Denitrification beds have also been installed into existing stream beds or drainage ditches and are specifically referred to as “stream bed bioreactors” (Schipper et al., 2010).

Based on previous studies, most bioreactors show high nitrate removal rates. Results from an in-situ field bioreactor study in Illinois USA showed up to 98% nitrate-N removal efficiency, with an average of 63% nitrate-N removal (Bell, Cooke, Olsen, David, & Hudson, 2015). They found that temperature explained 66% of the variance in N removal rate. A study by Pluer et al. (2016) found that both denitrification walls and beds are successful in nitrate-N removal, with rates generally ranging from 0.01 to 3.6 g N/m^3/day for walls and 2 to 22 g N/m^3/day for beds, with the lower rates often associated with nitrate limitations. Average removal of nitrate-N was 3.23 (41%) and 4 g N/m^3/day (54%) for woodchip and woodchip/ biochar reactors, respectively (Pluer et al., 2016). In New Zealand, Schipper et al. (2010) found that under field operating conditions, woodchip bioreactors demonstrated nitrate removal efficiencies ranging from 33 to 100%, and removal rates of 2–22 g N/m^3/day. A study by Rivas, Barkle, Moorhead, Clague, and Stenger (2019) set up a woodchip bioreactor on a dairy farm in the Waikato and found that nitrate-N removal efficiency was 99% in the first year, but only 48% in the second year. This difference can be attributed to a greater organic carbon availability in the first season. In the Manawatu specifically, a study on wood-based denitrification bioreactors by Christianson, Hanly and Hedley (2011), found N removal rates of 14 % vs. 37 % (depending on whether the drainage water was contained prior to treatment, to facilitate longer, more constant retention times).

Since N denitrification is the primary process in these reactors (Christianson & Helmers, 2011), the singular focus on N treatment is to be expected. However, both N and P are common in agricultural pollution; if denitrifying bioreactors are to be an ideal solution, they need to be designed to treat both nutrients. The study by Rivas et al. (2019) in the Waikato found that in the first season of operation, the woodchip bioreactor released higher concentrations of DRP than what entered. This is thought to be because effective treatment of P uses different processes from those associated with N removal because of its contrasting properties; removal of P typically focuses on sorption rather than biological processes, which are effective for N removal (Sharpley et al., 1994). A few studies on reactors have attempted to use a mixed media solution to facilitate treatment of both N and P (Bock et al., 2015; Coleman, Easton, & Bock, 2019; Pluer et al., 2016; Schipper et al., 2010). Biochar is a soil additive that has a high affinity for sorbing organic matter, nutrients, and metals, including phosphorus (Bock et al., 2015). A laboratory-scale study by (Bock et al., 2015) found that average P removal of 65% was observed in the biochar treatments, compared with an 8% increase in the control. The biochar addition also resulted in average nitrate removal of 86%, compared with only 13% in the control. A study by (Coleman et al., 2019) however, found that biochar only modestly enhanced nitrate removal, and it actually exacerbated phosphorus removal; while a study by (Pluer et al., 2016) showed no significant removal of phosphorus with biochar amendments. An innovative modification was investigated by Hua, Salo, Schmit, and Hay (2016), where an additional section for phosphorus removal was added (Fig. 2.5). The phosphorus removal material consisted of 15% activated alumina, bulked with coarse gravel. The total phosphorus removal rate increased by about 19 times compared to using only woodchips in the bioreactor (Anderson et al., 2016).

Figure 2. 5 Descriptive illustration of a two stage drainage treatment system (Hua et al., 2016)

A major advantage of these systems was the relatively low construction cost and no maintenance for many years (Christianson & Helmers, 2011). An initial cost/benefit analysis demonstrates that denitrifying bioreactors are cost effective and complementary to other agricultural management practices aimed at decreasing nitrogen loads to surface waters (Christianson & Helmers, 2011). There are however, some challenges facing the use of bioreactors. For example, the study by Aldrin et al (2019) observed potential pollution swapping in the bioreactors with the production of methane and hydrogen sulphide when residence times were long.

2.4.3 Riparian Buffer Strips

Riparian buffer strips (RBS) have been one of the most widely used management options worldwide for dealing with protection of surface waters from agricultural diffuse pollution (Parkyn, 2004; S. M. Parkyn, R. J. Davies-Colley, N. J. Halliday, K. J. Costley, & G. F. Croker, 2003; Weller, Baker, & Jordan, 2011). As the interface between terrestrial and aquatic environments, they have an extremely large influence on stream water quality relative to stream size. If managed appropriately, RBS offer multiple functions related to improving water quality, biodiversity, and climate adaptation (Lucy A McKergow, Fleur E Matheson, & John M Quinn, 2016). Importantly, they sustain water quality by sequestering nutrients and faecal contaminants, regulate instream and forest temperature, limit soil erosion and maintain instream biodiversity (Parkyn, 2004; S. M. Parkyn et al., 2003)

Riparian buffer strips (RBS) encompass the vegetated strips of land that extends along streams and rivers (J. Quinn, Cooper, & Williamson, 1993). This can range from a single strip of vegetation (grass filter strips) from which livestock or other agricultural activities are excluded, to a completely vegetated native forest riparian strip (Parkyn, 2004). Wetlands are also considered a form of riparian buffers but are included as their own form of land management practice for the purpose of this review. Depending on desired outcomes, a range of practices and designs can be applied to riparian areas, including stock exclusion, targeted vegetation management, and soil management to promote or limit certain pollution processes. Fencing stream banks and planting riparian buffers have been proposed in New Zealand as a key option to mitigate freshwater contaminants (DairyNZ, 2013).

2.4.3.1 Grass Buffer Strips

Conventionally, vegetated buffer strips (VBS) are established without additional management, using natural grass and herb vegetation with the aim to filter out sediment, sediment associated pollutants (particulate P and N), and faecal bacteria from surface runoff (Gharabaghi, Rudra, Whiteley, & Dickinson, 2002). A study done by Franklin (2014) in New Zealand compared nitrogen extraction efficiency in the introduced Perennial ryegrass (*L. perenne*) against native plants such as toetoe (*Cortaderia fulvida*). She found that the exotic *L. perenne* extracted more soil N than native species. Depending on the management objectives, perennial ryegrass could be incorporated into the grass filter strip with the native species. In saying that, toetoe was also found to extract relatively high soil N (Franklin, 2014). A study in the Bay of Plenty reported grass filters of three metres can reduce N and P loads by 35 to 87% (McKergow, Costley, & Timpany, 2009). Recommendations differ across regions depending on climatic conditions, but in general, regional councils recommend plants such as toetoe, as well as wineberry (*Aristotelia serrata*), pukio (*Carex secta*), red tussock grass (*Chionochloa rubra*) and swamp flax (*Phormium tenax*) for the grass strip. The DairyNZ riparian planner includes a 1 m grass strip as the minimum recommended width (K. Rutherford et al., 2017).

2.4.3.2 Traditional Buffer Strips

Structurally diverse riparian buffers, i.e. those that contain a mix of trees, shrubs and grasses, are much more effective at capturing a wide range of pollutants than a riparian buffer that is solely trees or grass strips (Hawes & Smith, 2005). There was a lot of variability in Total Nitrogen (TN) and Total Phosphorus (TP) retention efficiencies across the literature, with some international studies demonstrating TN removal efficiencies anywhere from 61 to 100%, and TP removal efficiencies of 17 to 99.9% (Mankin, Ngandu, Barden, Hutchinson, & Geyer, 2007).

Much of the variability in surface runoff nitrate retention efficiency could be explained by the effect of rainfall and vegetation type. For example, a study by Neilen et al. (2017) found that during high rainfall, the type of vegetation in the riparian buffer had a major effect on N retention, with lower N exported from grassed versus wooded riparian zones. Aguiar Jr, Rasera, Parron, Brito, and Ferreira (2015) also found that effectiveness is largely controlled by vegetation type; with buffer zones composed of woody vegetation being more effective in nutrient removal when compared to shrub or grass vegetation areas. Woody vegetation has deep rooting systems and woody soils have a higher content of organic matter, allowing for better nutrient adsorption. In a meta-analysis across a great range in buffer widths, Valkama, Usva, Saarinen, and Uusi-Kämppä (2018) highlighted that buffer zones were more effective at reducing N in groundwater (70% reduction) than in surface runoff (33% reduction). However, the meta-analysis found that buffering effectiveness was reduced with buffer zone age and was unrelated to width.

In contrast, other studies in the current compilation support nitrate-N reduction effectiveness increasing with width (Stutter, Kronvang, Ó hUallacháin, & Rozemeijer, 2019). Across New Zealand, Regional Councils generally recommend buffer widths between 5 and 10 m (Appendix 1). However, a review by Lucy A McKergow et al. (2016) found that most riparian buffer widths in NZ are in fact less than 5 m wide, often being a compromise between maintaining productive land and providing ecosystem services. Where possible, it is recommended to establish a multi-tiered riparian system (Fig 2.6) (DairyNZ, 2013).

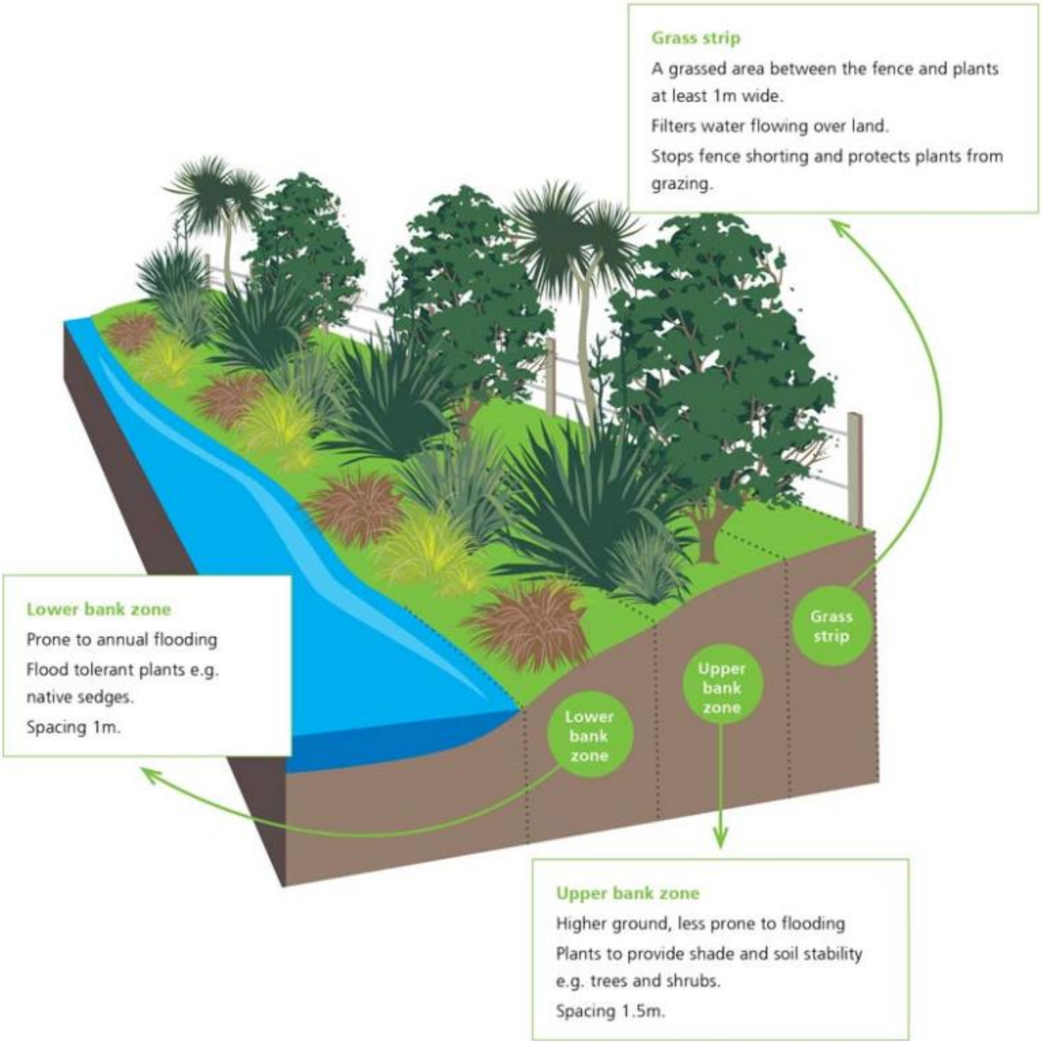

Figure 2. 6 Descriptive illustration of a combined riparian buffer strip (DairyNZ, 2013)

Although globally recognised, not many studies have quantified the effectiveness of RBS in New Zealand. An example by Cooper, Smith, and Smith (1995) showed improved soil infiltration capacity, while a study by S. Parkyn, R. Davies-Colley, N. Halliday, K. Costley, and G. Croker (2003) found varying levels of decreases in N and P in nine streams with RBS when compared to a control stream without an RBS. A study in southern Taranaki, New Zealand, showed that significant water quality improvements occurring between 2001 and 2008 could be attributed to the inclusion of riparian management, in conjunction with other on farm improvements (Ballantine & Davies-Colley, 2014). Both DRP and TP declined by 25 to 40% as a result of increased riparian protection, as well as a reduction in dairy shed effluent pond discharges with conversion to land irrigation, and a 25% reduction in the average application rate of P fertiliser (Ballantine & Davies-Colley, 2014).

There are some key problems which challenge the effectiveness of RBS in NZ. For example, many waterways such as drains, water races, small streams and intermittent/ephemeral channels are excluded from riparian policies (DairyNZ, 2013; L. A. McKergow, F. E. Matheson, & J. M. Quinn, 2016), despite often being direct conduits for on farm contaminants. Secondly, in rainfall-dominated systems, most nitrate export occurs in winter; when plant growth is low & temperature may control denitrification. Low summer soil moisture may also limit denitrification (Luo, Tillman, & Ball, 2000). The implementation of tile drains means that surface flow bypasses plant root zone (Stewart, Mehlhorn, & Elliott, 2007). Equally important, contaminant movement across the landscape is by preferential flow paths, meaning that effective RBS may require wide adjoining strips. While such variable buffer widths would maximise contaminant attenuation, they would also be minimising economic outlay.

2.4.3.3 Saturated Buffers

Sometimes, enhanced RBS management is required to tackle subsurface transfer pathways for nutrients. Saturated buffers are an option for utilizing existing riparian buffers, or zones of vegetation along stream banks or ditches, to treat tile drainage water in addition to surface runoff. Traditionally, tile drains transfer water directly from the field edge to a stream or drainage ditch, thus bypassing the riparian buffer (United States Department of Agriculture, 2016). Saturated buffers utilise the riparian buffer to treat some, or all, of the drainage water that would otherwise flow untreated through the buffer by artificially raising the water table and diverting much of the water from a subsurface drainage system along the buffer (Fig. 2.7) (Jaynes & Isenhart, 2019). As the drain water is introduced to the buffer, the soil becomes saturated. As saturation of the buffer occurs, along with lateral water movement through the buffer, nitrate is removed via denitrification (Jaynes & Isenhart, 2014).

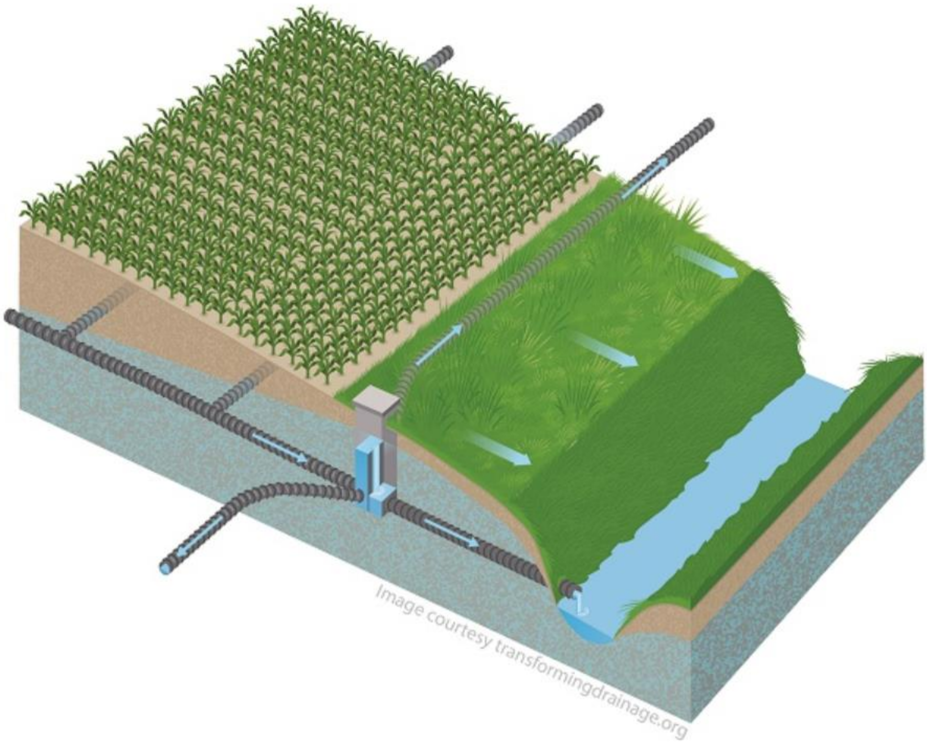

Figure 2. 7 Descriptive illustration of a saturated buffer strip (National Institute of Food and Agriculture, 2019)

Although no research has been conducted on saturated buffers in NZ, some early results for studies overseas indicate that they can be very effective for removing nitrate from tile drain water. A study by D. Jaynes and Isenhart (2018) showed all six sites of their designs of saturated riparian buffers (SRBs) to be effective in removing nitrate-N from the tile drainage water entering the SRB. The annual N removal effectiveness from 8 to 84%. They found that removal effectiveness for nitrate-N was found to depend on the buffer age; the SRBs constructed on “old” VBSs performed at nearly twice the nitrate-N removal efficiency of SRBs newly constructed into new buffer space. In a 2014 study also by D. B. Jaynes and Isenhart (2014), they concluded that all the nitrate-N that entered the buffer was removed and that no measurable nitrate-N reached the stream. Davis, Groh, Jaynes, Parkin, and Isenhart (2018) found that denitrification rates measured in the SRBs established on 20 year old riparian buffers could explain 48 and 77% of the total nitrate-N removal in the SRB. Conversely, this was reduced to 8 and 36% for the SRBs established on the 3 year old riparian buffer and only 4% in SRBs on newly established buffer space. The authors strongly advocated promoting high groundwater levels to use the entire soil column for removal of nitrate-N in SRBs, especially the topsoil (up to 20 cm in depth) where denitrification rates were greatest. Including the topsoil would increase the cumulative denitrification rates up to 100% for the SRB, if established within a 20 year old riparian buffer.

As a consequence of the diversion, saturated buffers could also help reduce the peak flow in streams, although little research has occurred on their potential ability to temporarily store water (United States Department of Agriculture, 2016). Furthermore, there would be no decrease in drainage effectiveness, and on farm economic outlay would not be affected as no additional land would need to be taken out of production (providing there is a pre-existing RBS) (United States Department of Agriculture, 2016). In saying that, it is recommended that the buffer be over 10 m wide to be fully effective (United States Department of Agriculture, 2016), and as discussed above, a pre-established buffer is more ideal as their nitrate removal potential are increased significantly. This presents a difference in opinion between biodiversity and water quality science on how to use and manage riparian buffers. Riparian planting influences instream biodiversity by providing woody debris as tress fall into streams, providing habitat diversity and cover for aquatic invertebrates, fish and freshwater crayfish (Hawes & Smith, 2005; Martin, Kaushik, Trevors, & Whiteley, 1999; Parkyn, 2004; S. Parkyn et al., 2003; Sweeney & Newbold, 2014). A. Daigneault, F. Eppink, and W. Lee (2017) suggest that biodiversity gains will only start to show at 20 -50 m buffer widths, while Sweeney and Newbold (2014) meta-analysis concluded that riparian areas should be at least 30 m wide to protect key aspects of forested small stream ecosystems.

2.4.3.4 Macrophyte Management

Nuisance macrophyte growth is a particular problem in New Zealand agricultural drains; and can reduce water flow, increase sediment deposition, impede drainage causing flooding, and create large daily fluctuations in dissolved oxygen concentrations resulting in overnight anoxia (Collins K. E. et al., 2018). When excessive macrophyte growth occurs, weed management is commonly undertaken. The three main macrophyte management strategies employed in small flowing waterways in New Zealand are mechanical clearance, chemical sprays and hand weeding. Macrophyte control is an expensive task – in the United States, annual costs were estimated to be $100 million in 2005. No cost estimate of macrophyte control is available in New Zealand, although the costs are considerable and expected to be in the tens of millions annually (Collins K. E. et al., 2018).

However, a study by Levi et al. (2015) on nutrient uptake in Denmark, found that macrophytes actually regulate stream function via direct uptake of ammonium (NH_4^+) from the water. Studies using tracer injections of ammonium found that the rates of N uptake in stretches with macrophytes were higher than the rates in stretches without macrophytes (Riis, Dodds, Kristensen, & Baisner, 2012; Simon, Niyogi, Frew, & Townsend, 2007). If drains are considered a place to treat drainage waters before they enter the stream, then there is potential for macrophyte management as a tool to remove nutrients.

As mentioned previously, RBS provide many benefits however, particularly in regards to their shading effects on the in-stream environment. Establishing them can be time consuming, resource intensive, and require a lot of maintenance. Installing artificial shade has been suggested as a useful tool for mimicking the shading effects of RBS. In an 8-month trial in a small drain in the Waikato, (Scarsbrook, Wilcock, Costley, & Nagels, 2000) reduced light levels by 90% with artificial shading. In this study there was no effect on the overall amount of plant cover, however, there was a significant change in both the type and density of plants growing under the shade. This means that there is potential to pursue nutrient management through macrophyte uptake, while still providing the ecological benefits of shading.

2.4.4 Drainage Management

2.4.4.1 Controlled Drainage

Controlled drainage (CD), sometimes called drainage water management, entails using a water control structure to raise the depth of the drainage outlet, holding water in the field during periods when drainage is not needed (Fig. 2.8). Unlike conventional free-draining systems that remove excess soil water to the drain depth, controlled drainage increases water retention and storage within the soil profile. Controlled drainage systems have been installed in Ontario and North Carolina in the USA, and the reports available suggest they greatly reduce the volume of tile flow and associated nutrient loads in drainage waters.

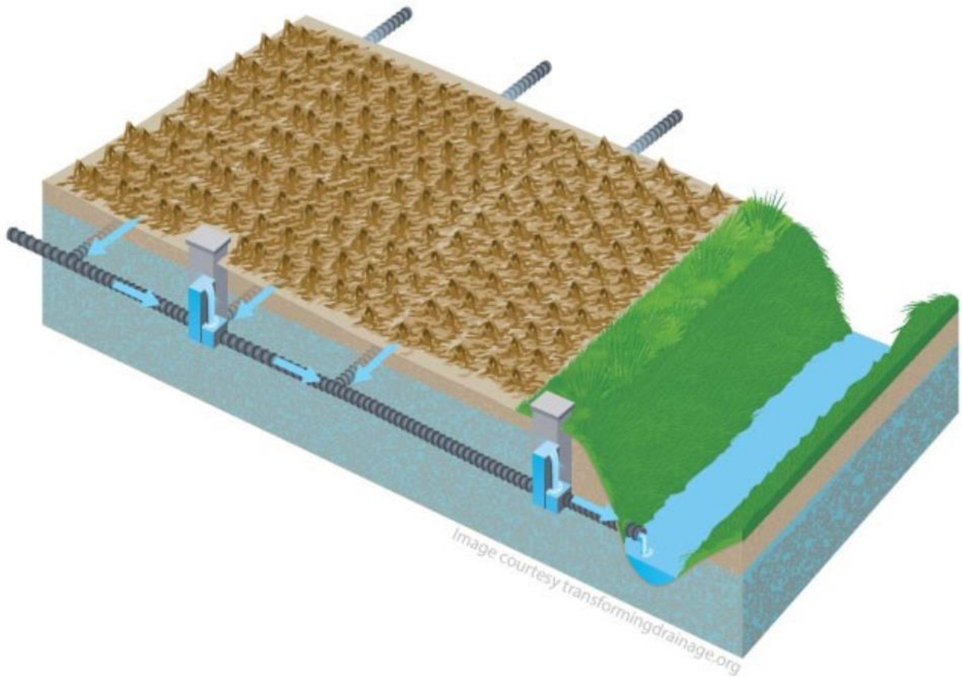

Figure 2. 8 Descriptive illustration of controlled drainage (National Institute of Food and Agriculture, 2019)

For example, in a study in Illinois by (Cooke & Verma, 2012), nitrate loss from tile drains with CD systems was shown to be between 52 and 79%. The Iowa Nutrient Reduction Strategy science assessment summarised multiple field studies in which controlled drainage has been used, and found nitrate loads reducing by 33% (Thompson, Helmers, Isenhart, & Lawrence, 2016). This reduction in nitrate loss is primarily attributed to a reduction in drainage volume however, rather than a reduction in nitrate concentration (Skaggs et al., 1994; Skaggs, Fausey, & Evans, 2012)

Nutrient reduction in CD is feasible via these three processes: firstly, denitrification is enhanced through as an increased anaerobic zone in the soil profile; secondly, as mentioned above, there is a reduced volume of drainage water leaving the field via drains; and thirdly, drainage water is potentially reduced in exposure to soil nitrate through having a reduced depth of the soil profile to infiltrate through (Dinnes et al., 2002). However, for moderately well drained soils, CD may actually have limited effectiveness in reducing nitrate losses as seepage water containing nitrate may eventually make its way to surface waters through different paths (Skaggs et al., 2012).

Although one control structure can effectively control 15-20 acres (Frankenberger et al., 2006), many factors such as topography, soil types, and climate can all influence how an individual field performs under this system. For example, CD will be most effective on relatively flat lands; slopes greater than 0.5% will only allow for drainage control on a small portion of the land surface, and may result in tile blowouts (Jane Frankenberger et al., 2006). Wetter soils can also suffer with CD in place, being more likely to have more runoff and erosion (Jane Frankenberger et al., 2006). Furthermore, while this practice can be used in reverse as a system during water shortage periods; under prolonged dry conditions, there may not be enough water to maintain an elevated water table. In this case, the system would not necessarily offer an advantage over conventional drainage systems. There is unfortunately a lack of studies evaluating the potential of controlled drainage in the New Zealand landscape.

2.4.4.2 Drainage Harvesting

Increasing irrigation water demands and water quality impacts are driving novel edge-of-field practices such as drainage harvesting and use in agricultural landscape. The timing and amount of precipitation does not always coincide with soil water deficit needs. Agricultural drainage occurs mostly in the winter and spring due to excess precipitation, while soil water deficit occurs in mid to late summer (Jane Frankenberger et al., 2017). Drainage harvesting is the practice of capturing water drained from fields during high-flow periods and diverting it into on-farm ponds or reservoirs, where it is stored until it can be used to irrigate with during periods when crop water needs exceed available soil water (Fig. 10) (Jane Frankenberger et al., 2017). This practice can be a closed loop system where the drained water from a field is recirculated onto the same field, or water drained from one field can be used to irrigate a different field. Irrigation may be through subirrigation that raises the soil water table by flooding the subsurface drain tiles; sprinkler systems such as a centre pivot; or other technologies (J. Frankenberger et al., 2017).

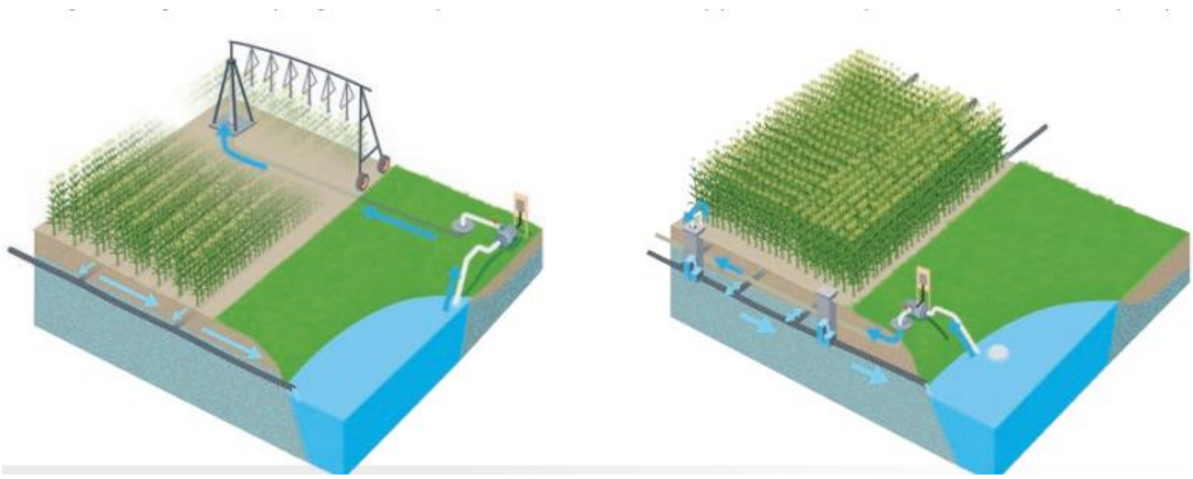

Figure 2. 9 A drainage water recycling system consists of a drainage water storage pond, which is then used for irrigation. Irrigation methods may vary, and may include overhead irrigation (left) or subirrigation (right) (National Institute of Food and Agriculture, 2019)

Relative to conventional drainage, drainage harvesting has several additional major benefits. As much as 70% of water consented for abstraction in New Zealand is consented for irrigation purposes (Srinivasan & Elley, 2017). Irrigation management in New Zealand is rapidly evolving as end-user needs and regulatory policies evolve. Since 2002, the size of irrigated area in New Zealand has almost doubled, and currently stands at 720,000 ha (IrrigationNZ, 2019). Due to the high volume of water used annually for irrigation proposes and the projected increase in the agricultural water requirements, reuse of drainage water for irrigation is an attractive option in sustainable water management. On-farm water harvesting has in fact already been implemented in many areas (albeit without any specific focus on nutrient management and recycling) in New Zealand such as the Kerikeri Irrigation Company's storage pond near the top of the North Island (Kerikeri Irrigation, 2019).

While such a practice can provide an opportunity for irrigation where certain limitations exist, such as inadequate water supplies or poor water quality, it can also optimise the use of nutrients and reduce the impact on the environment. Firstly, downstream water quality would benefit in that contaminants such as N and P (which are generally high in drainage water) are diverted into the water storage pond instead (Nnadi, Newman, Coupe, & Mbanaso, 2015). This can be particularly beneficial if drainage harvesting is made a closed loop system, where the drained water from a field is recirculated onto the same or neighbouring fields. In fact, the application of closed loop systems has been shown to reduce the consumption of water and fertilizers and the environmental pollution that is generally associated with over-irrigation (Massa et al., 2010). Secondly, natural removal processes present in the pond itself, such as settling and denitrification, would reduce nutrient levels in the water. Furthermore, the high concentrations of nutrients such as nitrogen and phosphorus present in drainage water, as well as its relative continuous availability makes it a more attractive option with regards to increased crop yield (Frankenberger et al., 2017; Nnadi et al., 2015).

Limited research has been published on drainage water harvesting systems, particularly under New Zealand farming conditions. In locations where ground and surface water sources are limited, water harvesting dams have been employed on farms to capture and store drainage for use as irrigation water (Mackay, Budding, & Morton, 2008). A hypothetical example provided by (Frankenberger et al., 2017) estimates that a system that captures and stores 3 inches of drain flow, with a concentration of 15 g/m^3 nitrate-N and 0.5 g/m^3 phosphorus, can prevent 9 Kgs of nitrate-N and 0.3 Kgs of phosphorus per acre from reaching downstream waterways. If this system drains 65 ha, it could reduce downstream loads by more than 362 Kgs of nitrate and 12 Kgs of phosphorus/year. In Missouri, a reservoir originally constructed for livestock watering was converted to a closed-loop system to capture drainage water. No results were published on the reduction in nutrient loads, although significant crop yield benefits were observed (Nelson, Meinhardt, & Smoot, 2012). An example from Ohio featured three drainage water recycling systems installed in the 1990s including water storage and irrigation but used them in conjunction with a wetland for water treatment. The wetland field tests showed that natural processes could remove 28% of the nitrate-N present in the drainage inflow (Allred et al., 2014).

Despite the promising benefits of water harvesting and recycling, there are concerns that intensification of such systems may have negative consequences on the downstream social–ecological systems by reducing stream flows (Dile et al., 2016). Additionally, issues of cost, complexity, compatibility with the current farm system, and a perceived uncertainty of actual environmental benefits are identified as key barriers to adoption of some of these technologies.Such a system is likely to be costly, but the large costs can be offset by the expected long-term yield increases due to irrigation (Frankenberger et al., 2017).

2.5 Conclusions

There are many emerging – and already established- in-field and edge-of-field drainage management methods, demonstrating a wide range of nutrient removal efficiencies (Table 2.1).

Table 2. 1 Summary of different drainage management practices and their nitrate-N removal efficiencies

Drainage Management Method	N Removal Efficiencies	References
Constructed Wetlands	15-98%	(Matheson & Sukias, 2010; K. Rutherford et al., 2017; C. Tanner et al., 2003)
Bioreactors	14-100%	(Bell et al., 2015; Rivas et al., 2019; Louis A. Schipper et al., 2010, Christianson, Hanly and Hedley (2011))
RBS	33-100%	(S. Parkyn et al., 2003; Valkama et al., 2018)
Saturated BS	8-84%	(D. Jaynes & Isenhart, 2018)
Controlled Drainage	33-79%	(Cooke & Verma, 2012; Thompson et al., 2016)
Harvesting and recycling	No estimate yet	N/A

However, the methods described above are not necessarily a cure-all. While these practices may reduce instream contamination, they do not address or mitigate land management problems. Their effectiveness is highly dependent on the source, volume, and type of contaminant, as well as the specific local conditions. Many of the methods explored in this review have been researched overseas, with little or no research been done in New Zealand (except for riparian buffers and wetlands). This is particularly true for harvesting and recycling of drainage water, posing a significant knowledge gap. It is possible, that drainage water harvesting and reuse could offer a great opportunity to reduce nutrient losses and irrigation water demands in New Zealand. This is the focus of my thesis research in following chapters.

Chapter 3

Methods and Materials

Chapter 3: Methods and Materials

This thesis aims to characterise the drainage patterns and drainage water quality, and then investigate potential targeted novel in-field and edge-of-field practices to reduce nutrient runoff in drainage waters. These practices include a Swap trial, a Macrophyte Management trial, and a Drainage Harvesting and Recycling trial. These trials were assessed on two intensive dairy, case study farms in the Rangitikei district, Manawatu-Wanganui Region.

3.1 Site description: Hyde Park & Regent Park

The case study took place across two farms belonging to the OB Group; Hyde Park and Regent Park. Both farms are intensive dairy units located in the Rangitikei coastal sand belt in the Santoft area, in the Manawatu-Rangitikei, New Zealand (Fig. 3.1). These farms are an example of land that was previously low productive Sand dunes, but now has the potential to support dairy farm operations through the increasing popularity of intensive irrigation systems.

The area is composed of sand dune complexes, and regionally-significant streams, lakes and wetlands (OB Group, 2019). The Santoft coastal area is a groundwater discharge area, meaning that it is supported by shallow groundwater flows. Furthermore, there is a formation of hard iron sand pans facilitating this perched shallow groundwater situation. Hence, there is a strong need for surface drainage in this area to lower the water table, particularly during the winter season. The farms thus have a series of drains that run parallel to and eventually join with the Kotiata stream that flows through the property. This means that any contaminants from the farm and its drains have the potential to be carried to these larger surface water bodies.

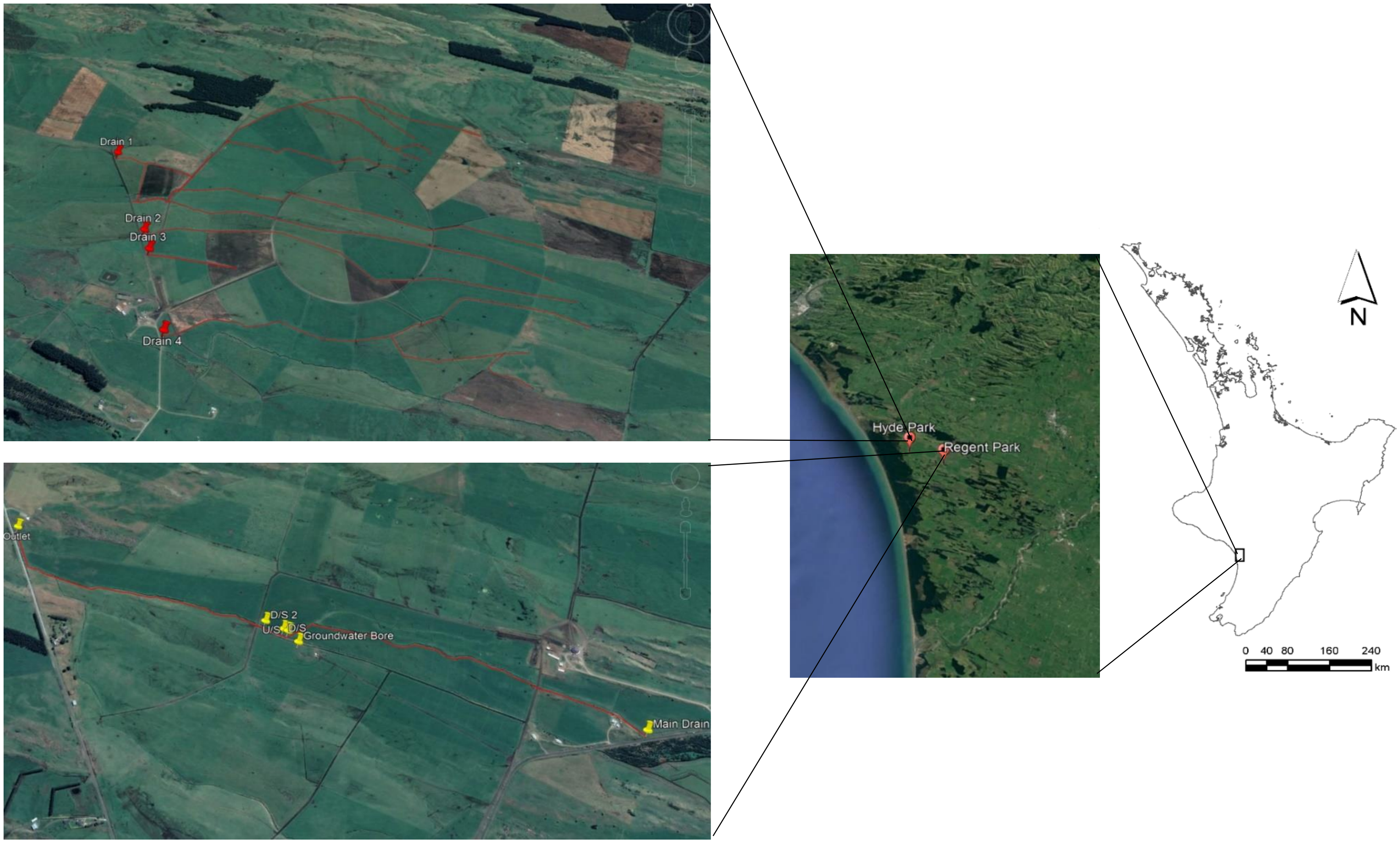

Figure 3. 1 Locations of the trial sites in relation to New Zealand

3.1.1 Climate

The proximity to the sea leads to a warmer than normal climate. Due to this, and the lower altitude, ground temperature tends not to fall below 8 degrees Celsius, allowing for reasonable production through the colder months and into spring.

The Rangitikei coastal sand belt experiences low summer rainfall and receives about 800-900 mm of annual average rainfall (Horizons Regional Council, 2019c). In 2018, rainfall was highest in April, averaging at about 4.3 mm/day; and lowest in January and February (0.88 and 1.5 mm respectively). April, May, July and August were the wettest months of the year, with rainfall averaging over 3.5 mm/day. June had uncharacteristically low rainfall compared to its surrounding months (2.3 mm/day) (Appendix 2).

3.1.2 Land Use

Hyde Park is run as a seasonal supply dairy unit, usually calving through the later part of July and August/September and producing through to May/June. Cow numbers fluctuate between 1,100 and 1,150. Regent Park on the other hand is run as an all year-round supply unit. Cow numbers have varied each year but on average the property has 1,150 (OB Group, 2019).

To supplement pasture growth, the main blocks are fertilised mostly monthly with urea at 60 kg/ha with one superphosphate application per year at 200 kg/ha.

3.1.3 Soil Types

The study sites are peculiar due to their utilisation of land that has very sandy soils and dunes. Parts of the land have been re-contoured, meaning that dunes have been levelled and covered with peat. The site therefore has a combination of soils with weakly developed structure and poor topsoil development. Surveys done by Landvision in 2017 indicated that there are seven different soil types at Hyde Park (LandVision Ltd, 2017a), and eight at Regent Park (LandVision Ltd, 2017b), including sandy loams, loamy sands, and peaty loams. The sandy loams are characterised by mostly medium N leaching vulnerability, very high structural vulnerability, medium to high bypass flow, and low relative runoff potential. Specifically, the Pukepuke black sand has an anion sorption capacity or P retention of 40%, and the Himitangi sand has a value of 21%. This indicates the risk of P loss in the area.

A Land use capability (LUC) map has also been completed for the dairy unit, ranking areas from LUC I-VIII depending on its ability to sustain productive agriculture. Class I is the best and is appropriate for intensive land use, while Class VIII is unsuitable for agricultural uses. According to the map, there are eleven separate LUC groups present at Hyde Park, with a major proportion being in the Class III ranking at 68%. At Regent Park, nine different LUC units and 5 LUC classes were recorded. Approximately 18% of the property is class II land, 44% is class III land, 20% is class IV land, 15% is class VI land and 3% is class VII land.

3.1.4 Irrigation and Drainage

Pasture growth is generally limited by the lack of water during the dry summer months from November to March. Crop water demands are generally higher than rainfall received during the summer season, hence the need for irrigation. Irrigation water for both farms is currently sourced from groundwater aquifers in the Santoft area. However, due to the dry summers combined with low soil water holding capacity (sandy soils), the groundwater in this area is in high demand, with irrigation using over 67% of the total consented water use (Horizons Regional Council, 2019a).

At Hyde Park there are two large centre pivots that cover most of the paddocks in the dairy unit to irrigate 297.5 effective ha of land, and 108.4 ha of the paddocks are also irrigated with effluent. At Regent Park, approximately 275.7 ha (266.4 ha effective) is irrigated with two centre pivot irrigators, and 145.5 ha of the paddocks are also irrigated with effluent.

There is the potential that the shallow groundwater in the Santoft area is so high because it is perched by an impermeable iron pan or a discontinuity in the lithologies underlying the sand in the area. This uncharacteristically high water table has led for the need for surface drainage. This combination of irrigation and drainage practices offers unique opportunities to assess drainage water harvesting for irrigation purpose.

3.1.5 OVERSEER Estimates

First developed in the 1980s, OVERSEER® is a nutrient budgeting software that enables farmers to improve nutrient use on farms by estimating all nutrient inputs and outputs from a farm or block within a farm, including nutrient losses, such N leaching and greenhouse gas emissions (Selbie, Watkins, Wheeler, & Shepherd, 2013). It has now become an essential agricultural management tool in New Zealand. Using OVERSEER, an estimate of the nutrient losses from a farm can be calculated based on farm data. As part of this study, the data from the farm management practices with fertiliser, irrigation, effluent and livestock was entered into OVERSEER to find an estimate of average annual nitrogen and phosphorus losses from the root zone of the farm.

3.2 Field Observations and Experiments

Field observations and experiments were conducted to assess performance of three separate trials; drainage water swap trial, macrophyte management, and drainage harvesting. The performance of drainage water swap and macrophyte management trials were observed over 3 months period during summer season, from December 2017 – February 2018. Assessment of drainage harvesting potential requires a characterisation of drainage flow patterns and water quality parameters. The characterisation of drainage water quality was achieved by sampling and analysis of drain water samples from June 2018 – May 2019.

3.2.1 Characterising Drainage Water Quality

Water Quality

Water samples from drains across the three trials were sampled weekly for a year to capture water quality changes temporally and spatially in the Santoft area. A ‘grab’ sampling was used, using plastic bottles (1L). The bottles were then sealed, placed in a cooler and filtered later in the laboratory using 0.45 μm filter paper and a vacuum flask. They were analysed for the parameters specified in Table 3.1.

All drain water samples were analysed for Total Nitrogen, Nitrate-N Ammonical-N, Total Phosphorus, and DRP concentrations, using continuous flow analysis (Technicon® Autoanalyser II). Analysis of DRP required using the Murphey and Riley method, while Ammonical-N uses a series of reagents (Rivas et al., 2014; Baldwin, 1998; Haygarth et al., 1998). The total N and P samples were analysed using a standard digestion in alkaline persulfate while being autoclaved. These samples were then analysed using continuous flow analysis like with their inorganic forms (Baldwin, 1998).

The drain water samples were also analysed for Nitrite-N, Bromide, Chloride, Flouride, and Sulphide concentrations via ion chromatography.

Environmental Parameters

At each sampling event, an YSI probe (YSI® Professional Plus) or SmartTroll was submerged to monitor field water quality indicator parameters (temperature, dissolved oxygen (DO), conductivity and pH) (Table 3.1).

Percent macrophyte and periphyton cover was visually assessed across five 1m^2 transects that were 10 m apart.

Drain discharge was measured using the Velocity-Area gauging method with stationary current meters as per the protocols outlined in (National Environmental Monitoring Standards 2013). Valeport Model 801 Electromagnetic Open Channel Flow Meters were used.

Table 3. 1 Table showing water quality and ecological parameters, frequency, and methods, for drain water samples collected from the Hyde Park and Regent Park Dairy units monitoring sites

Measure	Occurrence	Parameters		Method
Water Chemistry	Weekly	Totals	(TP, TN)	AutoAnalyser
		Cations	(NH_4^{+})	AutoAnalyser
		Anions	(NO_3-N, NO_2- Br^-, Cl^-, F^-, SO_4^{2-})	Ion Chromatography
Drain flow	Weekly	Discharge		Gauging
In-field Water Quality Parameters	Weekly	Temperature, pH, dissolved oxygen, and conductivity		In field Smart-Troll/YSI
Ecological measures	Weekly	% macrophyte and periphyton cover		Visual Assessment

3.2.2 Swap Trial

The first drainage management trial investigated the potential effects of swapping degraded drain water with relatively clean groundwater in real time for the purpose of recycling nutrients via irrigation over the dry summer period (December 2017 - February 2018). Water quality sampling and analysis was used to determine the nutrient concentrations temporally and spatially along one of Regent Park's drains. Sites were established via GPS to monitor nutrients flowing in and out of the drain. A total of five drain water monitoring sites- two upstream (MD and U/S), and three downstream (D/S, D/S 2, and OL) of the designated swap site, and one groundwater bore site were chosen (Fig. 3.2). They were selected to assess any changes in nutrient concentrations along the drain's length, and any impacts of swapping on the drain water downstream. Although five sites were monitored, only three (U/S, GW, and D/S) were analysed and compared closely for negative effects.

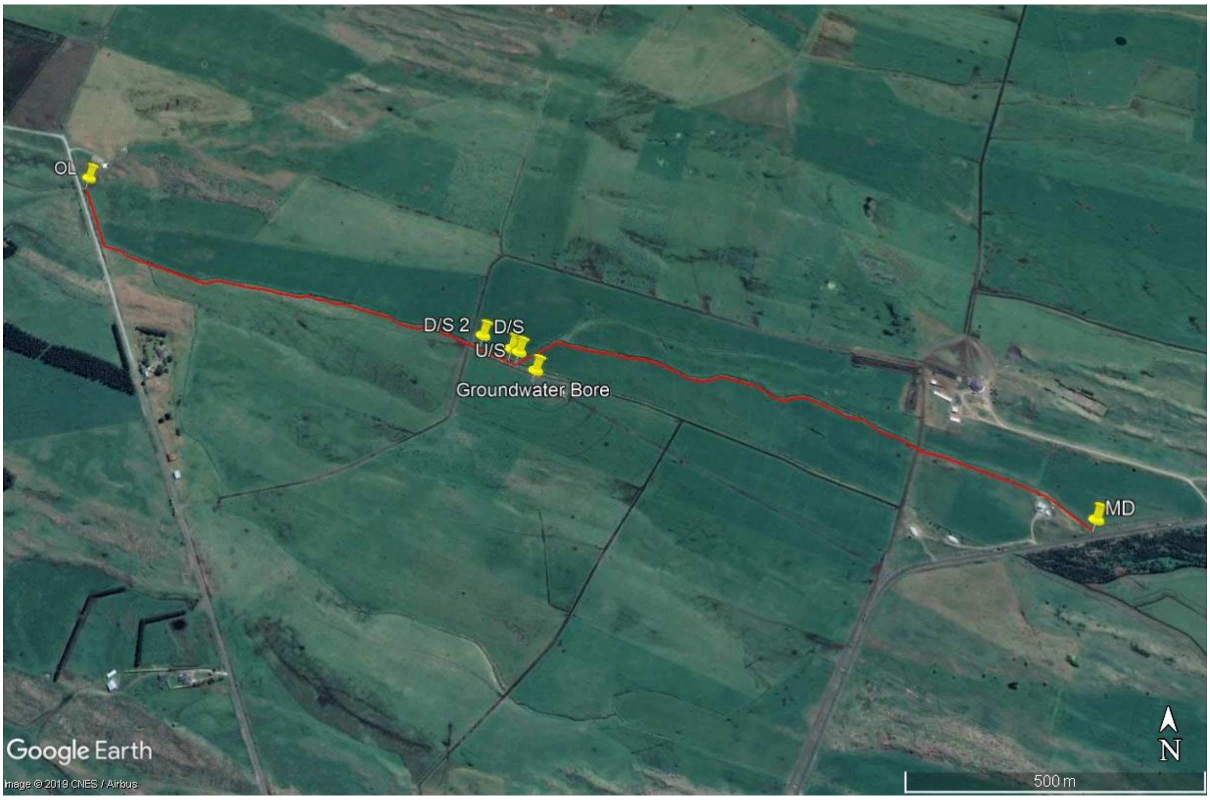

Figure 3. 2 Map of sites for the water swap trial. For reference, the red line represents the digitized drain. The damn was constructed in between U/S and D/S. U/S was added at a later point in time. D/S 2 was also added later and was 50 m downstream to D/S. A regularly used raceway crossed the drain just metres downstream to D/S 2.

The water swap was achieved by pumping drain water to a central pivot irrigator while, at the same time, diverting equal amounts of pumped groundwater to replace the drain flow (Fig. 3.3). A dam was constructed at the swap site and drain water upstream of this dam was diverted to the irrigator via a pump. The volume diverted was adjusted manually each day over the irrigation season.

Figure 3. 3 The swapping site. The water body to the left is the dam. The pump system can be seen to the right, and the irrigator that it feeds into can be seen in the distance

Eight weeks of baseline monitoring was carried out prior to the swapping trial (15/12/2017- 8/02/2018). Monitoring was conducted for 5 weeks during the swapping trial from 7/02/2018 to 9/03/2018. Water samples were collected weekly from each of the five drain sites and analysed for the water quality parameters summarised in Table 3.1. Groundwater samples were taken monthly.

At each site, five 1m^2 permanent assessment transects were set up 10 m apart. At each sampling event, these were used to assess the percent cover of macrophytes and periphyton, as well as in-field water quality parameters (Table 3.1). The results from each transect were averaged for each site.

3.2.3 Macrophyte Management

The second drainage management trial investigated the potential for nutrient uptake in vegetated drains, and if shading would have any effect. Riparian shading was simulated across the entire drain channel by using medium strength (~60%) shade cloth. These were set up and monitored closely over seven weeks in summer (26/01/2018-9/03/2018) and have been left indefinitely to monitor the effects of the shades on drain quality. Drain water quality and ecological measures were taken to observe any relationship between macrophyte cover, and nutrient uptake. If effective, these would be a simple and naturally occurring nutrient management method.

To do this, two 50 m artificially shaded reaches were set up in January 2018 in a stretch of drain at Regent Park farm, in Rangitikei. A site near the outlet of the farm (OL) with a fully established riparian strip on one side was used as a control (approximately 1 km downstream) (Fig. 3.4).

Figure 3. 4 Map of the macrophyte trial sites. For reference, the red line represents the digitized drain. SCNM (shade cloth no macrophytes) is shown 50m downstream to SCM (shade cloth macrophytes). The control site at the outlet (OL) is shown adjacent to the road.

For each of the two treatment sites, one 50 m 60% shade cloth, 3.66 m wide (source Redpath NZ), was placed over the drains. The shade cloth was clipped taught to a wire running along the two banks, supported by waratahs (source Farmlands Co-op) every 5 m, and wooden posts in the corners (Fig. 3.5).

Figure 3. 5 One of the shade cloths at installation

Prior to installing the second shade cloth, any existing macrophyte growth was mechanically removed from the drain. This site became SCNM for shade cloth no macrophytes. The macrophyte growth under the first shade was left untouched (SCM for shade cloth macrophytes). Thus, the two shades compared macrophyte growth in two different situations (existing growth and no pre-existing growth) with the reduction of sunlight via the shade cloth.

Within each drain and control reach, 5 permanent macrophyte assessment transects were set up 10 m apart. When the experiment started, macrophytes were already well established and had 100% cover, and extended above the water. Water samples were collected weekly for each of the seven weeks of monitoring from each of the three sites (SNM, SCNM, and OL) and analysed for the water quality parameters summarised in Table 3.1. % macrophyte and periphyton cover was also assessed weekly as per Table 3.1.

3.2.4 Drainage Water Harvesting

The third drainage management trial assessed the potential of drainage harvesting and reuse for summer irrigation and recycling of nutrients. Assessment of drainage harvesting potential requires a characterisation of drainage flow patterns and water quality parameters. Characterisation of drainage water quality commenced at the beginning of the drainage season (June 2018) and has been ongoing weekly to eventually achieve a complete year profile. Data up until 20/02/19 was used for this thesis. Since only a small volume of degraded drain water was able to be used in the water swap trial, this idea expands on that by determining how much drain and drain water (and its nutrient content) could be harvested over winter and spring seasons, and subsequently used to supplement summer irrigation for nutrient recycling.

Weekly field monitoring was used to determine drainage water quality and discharge (Table 3.1) temporally and spatially across Hyde Park's drains. A total of four drain sites (HPD1, HPD2, HPD3 and HPD4) were chosen and established to monitor drain water nutrients concentrations and discharge across the farm (Fig. 3.6). They were selected to assess differences in drain water quality and discharges across the different drains, their potential for harvesting, and if there were any parameters that would compromise the drainage water's direct placement on field through irrigation. In-stream gauging was used to calculate drainage discharge (Table 3.1).

Figure 3. 6 Map of the harvesting and recycling trial sites. The red lines represent the digitized drains. The pins show the sampling points for each drain

A desk-top exercise based on the gathered drain water quality and flow information, as well as OVERSEER values, was undertaken to assess and compare the feasibility of harvesting drainage water for irrigation. OVERSEER numbers however, are predicted. Scenarios were therefore created to compare the measured drainage water quality values versus the OVERSEER estimates in terms of recycling.

The potential savings in terms of groundwater irrigation and fertiliser application, as well as basic cost-benefit analysis of required farm infrastructure for potential drainage water storage and reuse as irrigation to provide adequate drainage irrigation capabilities were assessed. Capital costs were calculated as including the costs of the storage pond per m^3, the pump, and resource consent. Annual costs were calculated as having power, costs of capital at 8%, repair and maintenance, lost profit from land due to the storage pond, and potential savings in urea per ha. Cost per Kg of nitrate-N attenuated was calculated as the total potential nitrate-N harvested over the annual costs. It should be noted that the numbers used are rough estimates used more so to illustrate the differences in potential harvesting and recycling of drainage water based on the OVERSEER predictions versus the actual nitrate-N concentrations found in sampling.

3.3 Statistical Analysis

For all trials, the data collected from on-site and laboratory analysis were recorded in an Excel model to identify spatial and temporal trends. Further analysis was completed in R Studio version 3.5.2. For each trial, the Shapiro Normality Test was carried out on all the variables. The majority of the data was non-normally distributed, and as such, non-parametric statistical methods were used.

Kruskal Walis tests were carried out to determine if there were any significant differences across and in between the sampling times and sites, and Mann Whitney Pairwise Tests were used to identify where the differences were. Where the data was normally distributed, One-Way ANOVAs and Post Hoc Tukey Tests were used to analyse drain nutrient concentrations across and in between the sites.

Spearmans's Rank Correlation Test was used for the Macrophyte Management Trial to explore the strengths and directions of association between variables such as nitrate-N and % macrophyte cover.

Chapter 4

Results and Discussion

Chapter 4: Results and Discussion

4.1 Drainage Flows and Water Quality Patterns

Weekly water samples from a total of five drains across both Hyde Park and Regent Park were analysed for a full suite of parameters, including nitrate-N and DRP. The Hyde Park samples were collected from June 2018 to February 2019, while the samples for March were used from the trial at Regent Park in 2018. Where data was absent (April and May), values from Smith et al. 2017 were used. These covered two surface water drains, also on Hyde Park.

The collected drain water quality data from both farms was merged into one data set to observe the general drain water quality patterns in the Santoft area. It should be noted that the minimum standards for nitrate-N was 0.25 g/m^3 used by the IC analysis. Therefore, the minimum detection limit would be 0.125 g/m^3. Any nitrate-N values below this limit is not considered reliable as it is lower than the concentration that can be accurately detected by the lab equipment. Similarly, the minimum standards for DRP was 0.05 g/m^3 and so the minimum detection limit would be 0.025 g/m^3.

Table 4.1 summaries the drain water quality parameters analysed. Nitrate-N concentrations showed the most variation, varying from 0.06 g/m^3 to 5.96 g/m^3 (Table 4.1). While the recommended standard for Nitrate-N in lowland drains is 0.44 g/m^3, the average nitrate-N in drainage waters in this area was relatively higher at 1.31 g/m^3. Lowland DRP concentrations need to be below 0.01 g/m^3 to avoid possible adverse ecosystem effects (Davies-Colley, 2000), while the standard ammoniacal-N value for freshwater ecological health is 0.021 g/m^3 (Davies-Colley, 2000). While the drainage waters in the study area in general showed relatively low levels of ammonia-nitrogen (0.06- 4.75 g/m^3), the average (0.19 g/m^3) exceeded the recommended guideline. Average DRP levels on the other hand (0.01 g/m^3), did not exceed either the detection limit, or the recommended guideline (Table 4.1).

Table 4. 1 Drainage Water Quality parameters (April 2016-February 2019), measured at surface water drains at Hyde Park and Regent Park

	Co-unt	Average (g/m^3)	Minimum (g/m^3)	Maximum (g/m^3)	Standard Deviation	Coefficient of Variance
Bromide	185	0.10	0.05	0.13	0.02	0.24
Chloride	187	36.71	1.00	54	9.71	0.26
Fluoride	150	0.17	0.05	0.59	0.11	0.65
Ammonia-N	147	0.19	0.06	4.75	0.58	3.01
Total Nitrogen	156	2.64	0.06	10.12	1.98	0.75
Nitrite-N	47	0.01	0.00	0.05	0.01	0.84
Nitrate-N	182	1.31	0.06	5.96	1.46	1.11
DRP	133	0.01	0.01	0.01	0.00	0.00
Total Phosphorus	159	0.10	0.01	1.06	0.15	1.60
Sulphate	183	10.09	0.20	21.22	4.48	0.44

Total N values ranged from 0.06-10.12 g/m^3, with the average TN concentration at 2.64 g/m^3. This greatly exceeds the recommended guideline of 0.614 g/m^3 of Total N to avoid possible adverse ecosystem effects (Davies-Colley, 2000). Total P concentrations varied from 0.01 to 1.06 g/m^3. The average Total P was measured at 0.10 g/m^3, which also exceeds the recommended guideline value for lowland drains, 0.033 g/m^3 (Davies-Colley, 2000). Arguably, the ANZECC guidelines (Davies-Colley, 2000) could be considered environmentally conservative for highly modified dairy catchments, as nitrogen and phosphorus are so ingrained in farm management practices. The variability in nutrient concentrations is probably due to the size and location of the drains as well as the catchment area and land use at the study farms.

4.1.1 Nitrate Yearly Profile

Among others, Nitrate-N can be identified as a problem pollutant in this study. Its mobility and water-solubility give it more reason for concern as it means that it can easily be transported via a water vector such as surface drains present on the farms. All Nitrate-N values from each week sampled June 2018 to February 2019 were compiled into a boxplot to examine the spatial and temporal variability (Fig. 4.1). It should be noted that no current data was collected in the months March, April and May 2018, so Nitrate-N concentrations measured in drains waters during April 2016 and May 2016 from a previous study on the same farm are used for the purpose of this exercise (Smith et al., 2017). Nitrate-N concentrations for March were used from the trial at Regent Park in 2018. It is evident that there is high spatial variability in Nitrate-N, particularly coming into August (Week 31). This variability is likely due to the size and location of the drains in addition to the catchment area and land use at the study farms.

Figure 4.1 reproduces the yearly variability of Nitrate-N concentrations measured in drain waters at the study farms. All monthly average levels of Nitrate-N exceeded the limit of 0.44 g/m^3, except for February and March (which have the lowest average Nitrate-N levels (0.2 and 0.25 g/m^3 respectively). It is evident that there is also a high temporal variability in the concentration of nitrate-N across the year (Fig. 4.1) (Table 4.2). When a Mann Whitney Pairwise T Test was run, it was evident that most months are statistically different to each other for nitrate-N concentrations (P value <0.05).

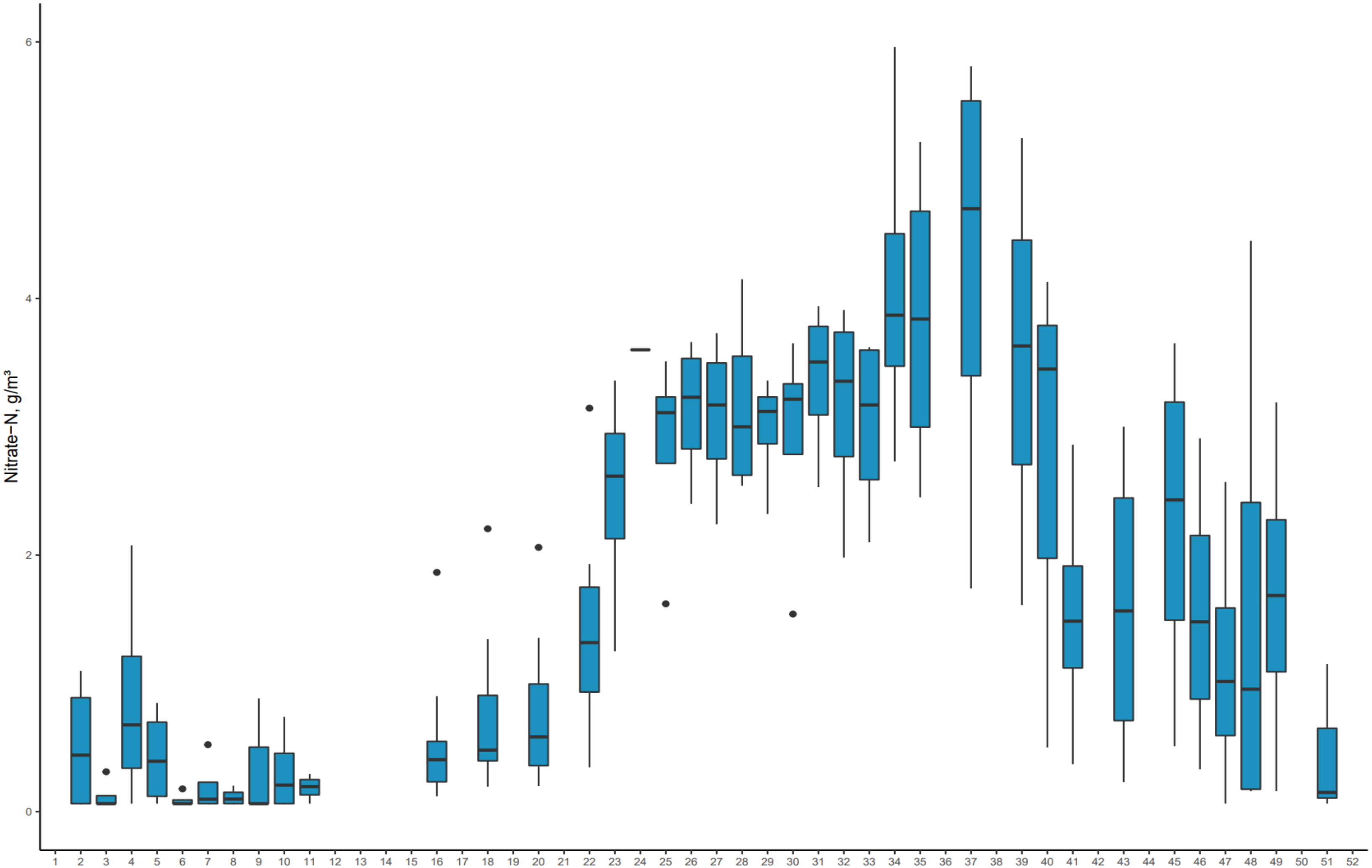

Figure 4. 1 Boxplot showing spatial and temporal variation of Nitrate-N concentrations in the Santoft area over a generalised year. Values are amalgamated from samples taken from both Hyde Park and Regent Park across multiple studies between 2016 and 2019. Week 1 starts in the first week of January, and Week 52 is the last week of December. For reference, Week 23 is the first week of June.

Table 4. 2 Nitrate-N concentrations over a generalised year in the Santoft area, measured at surface water drains at Hyde Park and Regent Park between 2016 and 2019

	Count	Average (g/m^3)	Minimum (g/m^3)	Maximum (g/m^3)	Standard Deviation	Coefficient of Variance
January	16	0.48	0.06	2.08	0.55	113.00%
February	19	0.2	0.06	0.88	0.25	127.20%
March	11	0.25	0.06	0.74	0.23	90.80%
April	16	0.67	0.12	2.21	0.6	89.70%
May	16	1.11	0.2	3.15	0.78	69.90%
June	13	2.87	1.25	3.66	0.73	25.40%
July	16	3.03	1.54	4.15	0.63	20.70%
August	20	3.5	1.98	5.96	0.97	27.60%
September	8	3.88	1.61	5.81	1.52	39.20%
October	11	1.88	0.23	4.13	1.28	68.50%
November	16	1.65	0.06	4.45	1.31	79.50%
December	7	1.15	0.06	3.19	1.07	92.90%
Total	169	1.68	0.06	5.96	1.52	90.40%

Nitrate-N concentrations begin to rise in early autumn, i.e. with the onset of the wetter months in May and June. A study on the same farm found that a similar increasing trend was followed in Nitrate-N concentrations in surface drains between April and September (Smith et al., 2017). The Nitrate-N drain values ranged from <0.25 to 6.59 g/m^3 over these months (Smith et al., 2017) (Appendix 3). Similarly, in the current study, Nitrate-N concentrations varied from 0.12 g/m^3 to 5.96 g/m^3 between June and September on Hyde Park (Table 4.2).

The higher average Nitrate-N concentrations are found in the winter season (June to September), corresponding with higher rainfall and therefore higher drainage through soil profile (Fig. 4.2).

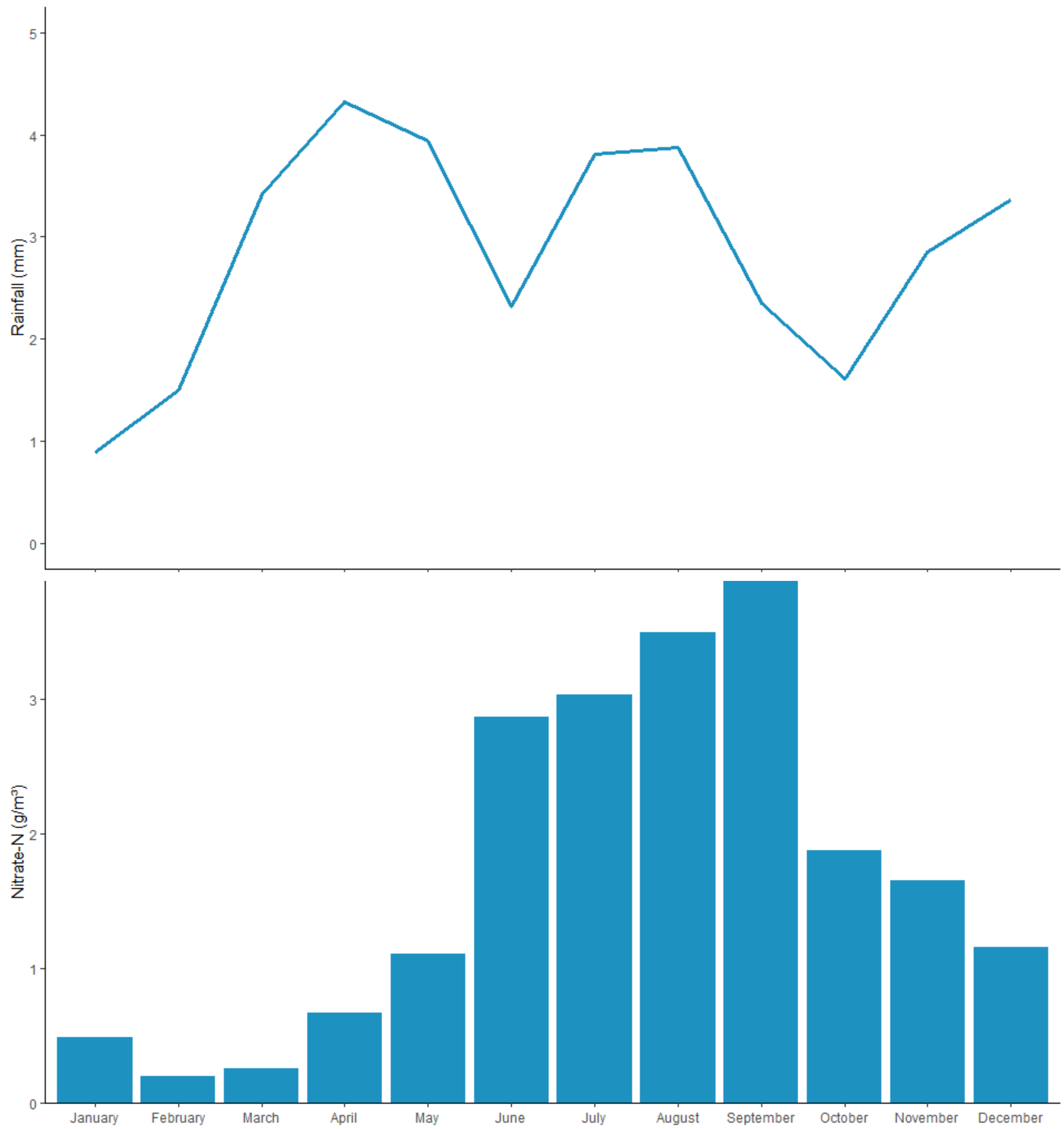

Figure 4. 2 Graph showing monthly average nitrate-N concentrations in the Santoft area from data collected between 2016 and 2019. Average daily rainfall values from between June 2018 and May 2019 are also shown, obtained from (Horizons Regional Council, 2019c)

There was a sudden increase in Nitrate-N concentrations in June (2.87 g/m^3), inconsistent with the sudden drop in rainfall for that month. This is possibly due to first flushes of drainage water washing off nitrate-N that had accumulated in the soil profile during the summer months (with a lack of drainage). The average nitrate-N concentration over these wetter months (June-September) is 3.3 g/m^3 (Appendix 2).

Rainfall was minimal over summer (Fig. 4.2). It is likely that most of the water flowing through the drain, particularly in the later months (January-March), was due to baseflow recession more so than drainage. Hyde Park dairy unit has deeply-dug drains throughout the property that are at the same level as the average height of the water table. The water table tended to remain around 1-1.3 m below ground level and the drains were dug to around 1.3 m deep. This is a technique often employed to control the height of the groundwater table to increase the infiltration capacity and reduce the potential for surface runoff and flooding (Zimmer & Madramootoo, 1997). If surface water is lower than the level of adjacent groundwater water tables, then it is an 'effluent drain' and can be recharged by the groundwater.

Observations over a year from a previous study on the same farm showed that the flow of the drain tends to disappear two-thirds of the way across the farm in one of the main drains by October (Smith et al., 2017). At this point, water flow disappears underground during the dry summer weather. However, there is still some flow at the end points of the main drains before they join the main drain. This is probably due to baseflow recession (Hantush, Kalin, & Govindaraju, 2011). This means that, as the water table decreases during the drier summer period, less and less groundwater is discharged to the surface drains. However, when the water table is high enough in the wet season, the groundwater is quite capable of moving into the surface drains.

Furthermore, groundwater samples collected from December 2017- March 2018 showed Nitrate-N concentrations measuring below the detection limit (<0.125 g/m^3). The average was 0.12 g/m^3. Another study, also conducted on Hyde Park from April-September 2016, gave similar results of groundwater Nitrate-N concentrations measured below the detection limit, except for on two instances- in July and August (0.26-0.83 g/m^3) (Smith et al., 2017). It is evident therefore that there is not much nitrate-N contribution from shallow groundwater into the drains.

There is a sharp drop in average nitrate-N concentrations in drain waters from September to October (Fig. 4.2). This does correspond with the reduction in rainfall (Fig. 4.2), but also, from mid-September, drains were beginning to disappear, and vegetation had grown densely within them. The macrophytes remain dense until around mid-March/April when they start to die back, and late march when they are mechanically cleared. The plant growth and low flow could mean nitrate-N was leaving the system by periphyton or macrophyte uptake or possible in-drain denitrification. A study by Peterson et al. (2001) found that headwater streams and riparian wetlands generally exhibit relatively high nitrate attenuation rates because of their shallowness and high contact area with biologically active sediments and plants. The combination of increasing macrophyte growth and decreasing flow (increasing baseflow recession) could explain the decreasing nitrate-N concentrations in drains during the summer season (January – March) (Fig. 4.2).

4.1.2 Drainage Profile

A drainage profile was constructed using a combination of flow measurements gathered from Hyde Park. Open channel gaugings were taken in the drains from the beginning of June 2018 through to the end of February 2019 (Table 4.3). Over the monitoring period, it was found that HPD4 had the highest average discharge (29.3 l/s) and HPD1 had the lowest average (4.05 l/s). For HPD4, discharge ranged from 7.96 l/s to 61.95 l/s. For HPD1, discharge ranged from no flow to 14.8 l/s. As with the nutrient levels, this variability is likely due to the size and location of the drains in addition to the catchment area and land use at the study farms.

Table 4. 3 Drainage flow (l/s) for each of the four drain sites at Hyde Park, taken between June 2018 to February 2019

Drain	Count	Average (l/s)	Minimum (l/s)	Maximum (l/s)	Standard Deviation	Coefficient of Variation
HPD1	30	4.05	0.00	14.85	3.86	95.4%
HPD2	30	11.19	4.11	22.70	6.28	56.1%
HPD3	30	4.66	1.48	35.46	6.75	144.9%
HPD4	34	29.30	7.96	61.95	14.32	134.7%
Total	124	12.85	0.00	61.95	13.76	107.1%

From the data gathered, the higher average drainage flows are found in the winter season (June to September) (Fig. 4.3) (Appendix 4), corresponding with both the higher rainfall and nitrate-N concentrations (Fig. 4.2). The average combined drain flow over these four months is 16.9 l/s.

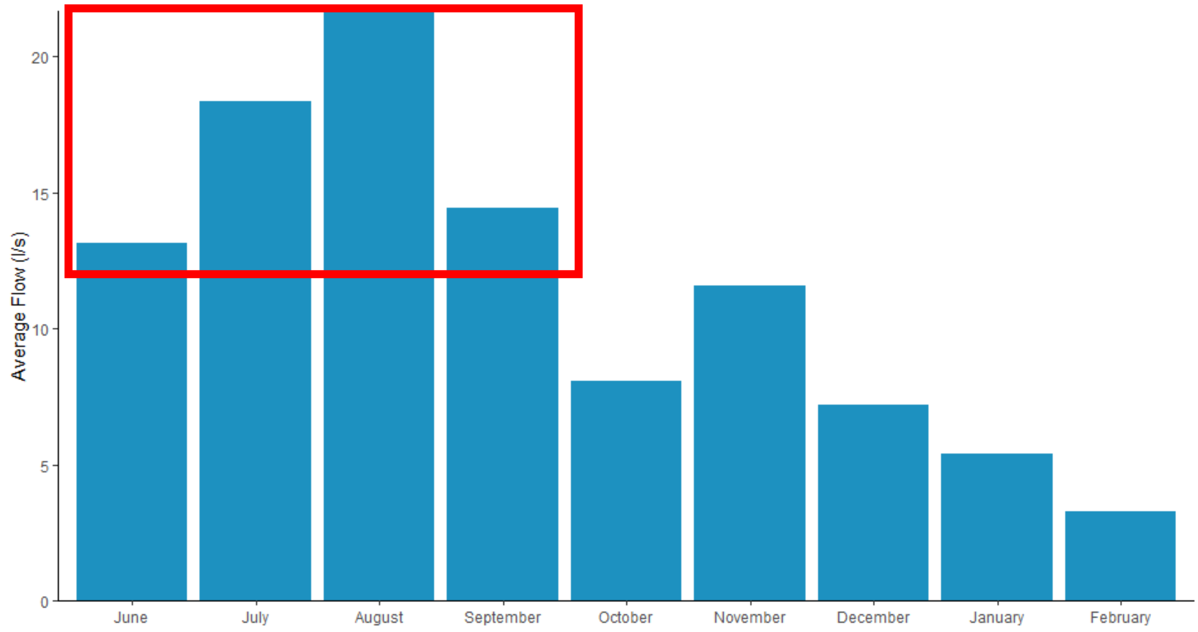

Figure 4. 3 Total average monthly discharge (l/s) of all 4 drains combined on Hyde Park. The red box depicts the highest average months for discharge

4.1.3 Nutrient Losses from the Root Zone

The following outputs for Hyde Park were produced from OVERSEER (Fig. 4.4).

Block name	Total N lost kg N/yr	N lost to water kg N/ha/yr	N in drainage * ppm	N surplus kg N/ha/yr	Added N ** kg N/ha/yr
Irrigated Pukepuke	6,132	74	**25.9**	277	371
Irrigated Himatangi	7,482	74	**25.7**	278	372
Dry Pukepuke	2,521	42	**17.3**	120	161
Effluent Irrigated Puke	7,026	88	**30.5**	340	444
Effluent Irrigated Hima	2,353	88	**30.6**	340	444
Effluent Dry Puke	370	53	**21.5**	217	271
Summer brassica	1,091	55	**19.4**	123	173
Winter brassica	2,233	112	**35.3**	105	141
Dry Pukepuke (to Lake)	5	2	N/A		
Other sources	632				
Whole farm	29,845	68			
Less N removed in wetland	0				
Farm output	29,845	68			

Figure 4. 4 Screenshot of the nitrate-N output from the Hyde Park profile on OVERSEER

Based on the estimates from OVERSEER, the modelled N loss from Hyde Park would be 29,845 kg N per year over the whole farm, with an average of 68 kg N/ha/year lost to drainage water. The greatest N loss to water comes from the Effluent Irrigated Pukepuke and Himatangi blocks (88 Kgs N/ha/yr). The concentrations of N in drainage waters is estimated to be from 17.3-35.3 g/m3 from different blocks on the farm. Interestingly, these estimates are substantially higher than the N concentrations found during sampling across the farms, where the range was from 0.06- 5.96 g/m^3 (Table 4.2).

Under irrigated dairy (Irrigated Himatangi and PukePuke soils), OVERSEER estimated an average nitrogen loss of 25.9 g/m^3 from the root zone to drainage waters. During the monitoring period, the maximum nitrogen concentration in drainage waters on the case study farm at any given point was 5.96 g/m^3, while the overall average was 3.3 g/m^3 (Table 4.2). This disparity between the OVERSEER estimates and the actual drainage results suggests nitrate-N attenuation below the root zone and mixing up of drainage waters with reduced shallow groundwaters in the drains.

A rough representative nitrate-N load for the case study farm was estimated from the average nitrate-N and drainage volumes. The average nitrate-N value for the months monitored on the farm was 1.31 g/m^3 (Table 4.1), while the total average drain flow was

12.85 l/s (Table 4.3). Based on this information, total drainage volume would be 405,212.56 m^3/yr (810 mm), and total nitrate-N flux would be 532.74 Kg/yr. Google Earth was used to calculate an approximate drainage area for Drains 2 and 3 (Fig. 4.5). The result was 50.4 ha. On an approximate per ha basis, I calculated this farm to leach 10.57 Kg N/ha/yr into the waterways.

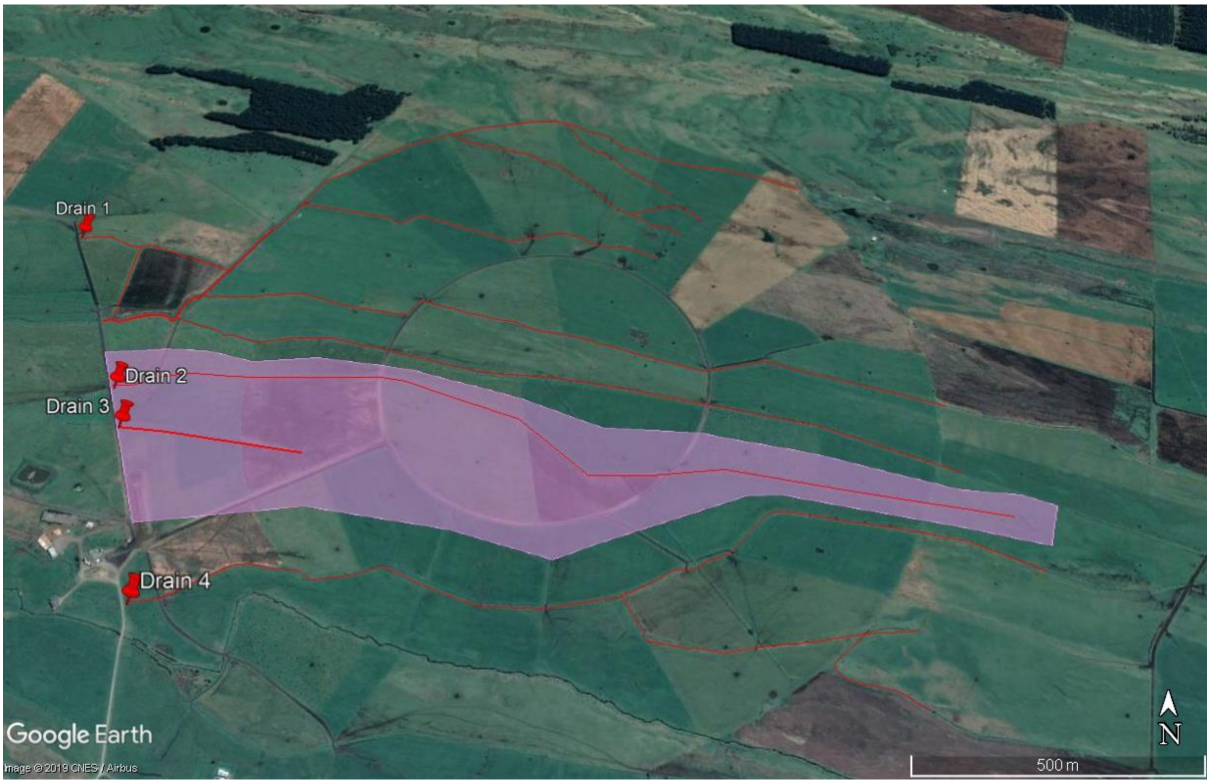

Figure 4. 5 Approximate drainage area for Drains 2 and 3, taken from Google Earth.

This value is drastically lower than the OVERSEER estimate (68 Kg N/ha/yr). Previous research in the Tararua and Rangitikei catchment also shows nitrogen loads measured in the river to be significantly smaller (67 to 94%) than the OVERSEER estimates of N leached from the root zone (Elwan, Singh, Horne, Roygard, & Clothier, 2015; Singh et al., 2017). Groundwater surveys and in-field experiments suggested that potential in-drain nitrogen uptake by periphyton were estimated to account for only between <1 to 5.7% of the estimated total nitrogen loss, whereas the occurrence of subsurface denitrification was the key attenuation process in these catchments (S Collins et al., 2017; Jha, Singh, & McMillan; Rivas et al., 2017). This clearly suggests that the effects of catchment characteristics such as soil type, underlying geology, and subsurface geochemistry should be considered in prediction and accounting of nitrogen flows and its potential attenuation from farms to receiving surface water bodies in agricultural catchments. This is consistent with Smith's study from 2017 on the same case study farm, which found that the redox conditions of groundwater in this area appeared to be conducive to processes that would reduce nitrate-N via denitrification, and that DNRA (dissimilatory nitrate reduction to ammonium; another means for nitrate-N to be transformed under low-oxygen conditions) could also be a reducing process in the subsurface environment.

From the monitored season, it is therefore possible to propose that since the nitrogen being leached on Hyde Park is not necessarily making its way to the surface water, that leaching at Hyde Park (although observed at concentrations over 0.44 g/m^3) is not necessarily a problem due to subsurface reducing conditions.

4.2 Swapping Trial

4.2.1 Summary of Water Quality

Drainage water swapping trial occurred between 8/02/18 and 9/03/18. A total of 6 sites were sampled, but this section will primarily focus on those immediately surrounding the swapping site; U/S (upstream), D/S (downstream), and GW (groundwater bore) (Fig. 4.6, 4.7). A summary of water quality for each of the sites can be found in Appendix 5. During this time, there was no significant difference across drain water quality variables between the three sampling sites, except for Chloride and Bromide (Kruskal Walis P value < 0.05). The Mann Whitney Pairwise/ Tukey HSD tests suggested that the levels of Chloride in the upstream site were significantly different from those in the groundwater (p value = 0.017) and downstream sites (p value= 0.018), while levels of Bromide in the upstream site were also significantly different from the groundwater (p value = 0.007) and downstream sites (p value = 0.024).

Most of the Nitrate-N concentrations across the three sites post swap were measured below the detection limit (<0.125 g/m^3). They also remained under the recommended standard for Nitrate-N in lowland drains (0.44 g/m^3) (Table 4.4). Nitrate-N levels at the U/S site were average at 0.13 g/m^3 and ranged from 0.06 to 0.35 g/m^3 (Fig. 4.7). Nitrate-N levels at the post swap D/S site were averaged at 0.08 g/m^3 and ranged from 0.06 to 0.15 g/m^3 (Fig. 4.7). Groundwater average Nitrate-N levels were measured as 0.12 g/m^3. Despite the mixing of upstream and groundwater nitrate levels, average Nitrate-N levels dropped considerably (albeit not significantly) at the downstream site (0.08 g/m^3). It is possible that this is due to macrophyte growth and uptake between the U/S and D/S sites.

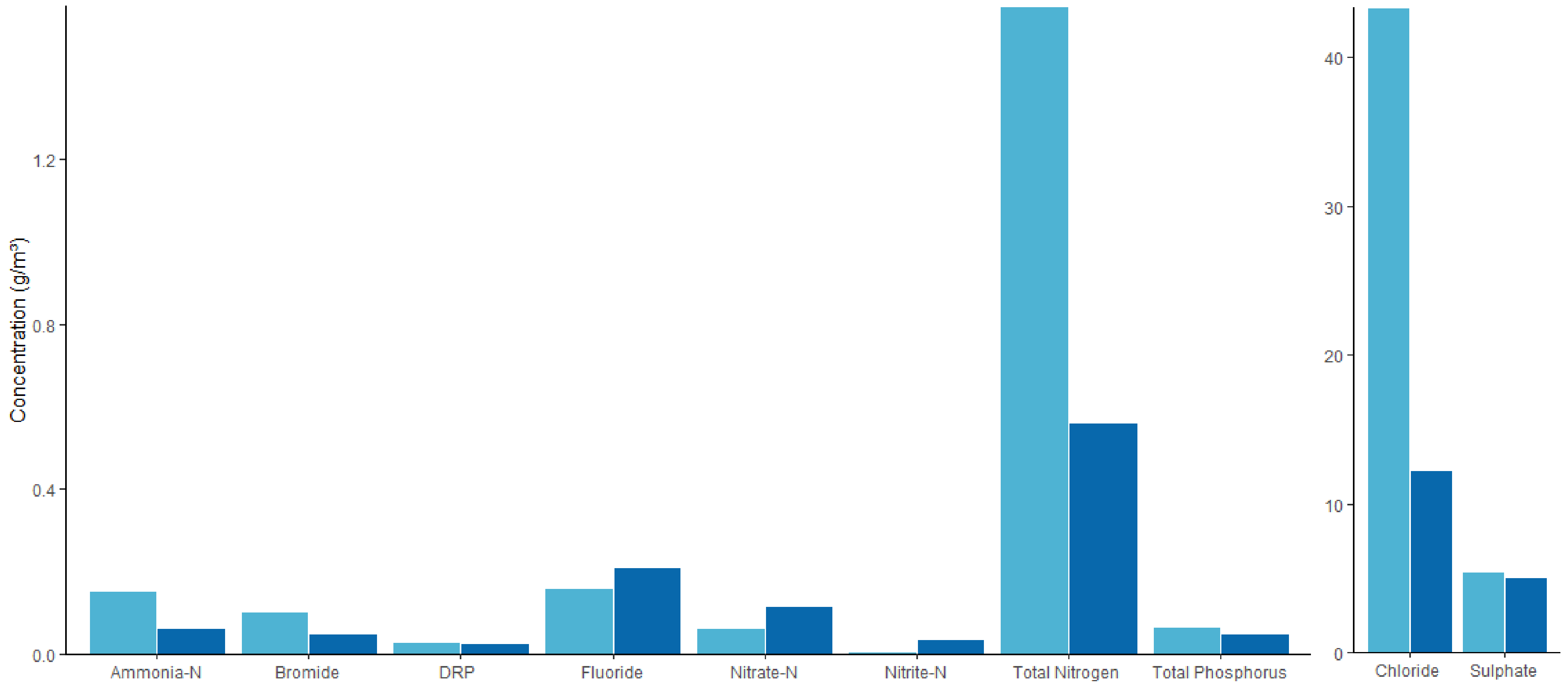

Figure 4. 6 Bargraph showing average concentrations of water quality parameters taken during the swap trial. Light blue annotates baseline data collected weekly at the site D/S between 15/12/2017- 8/02/2018 (pre swap), while dark blue shows average groundwater levels, taken once monthly

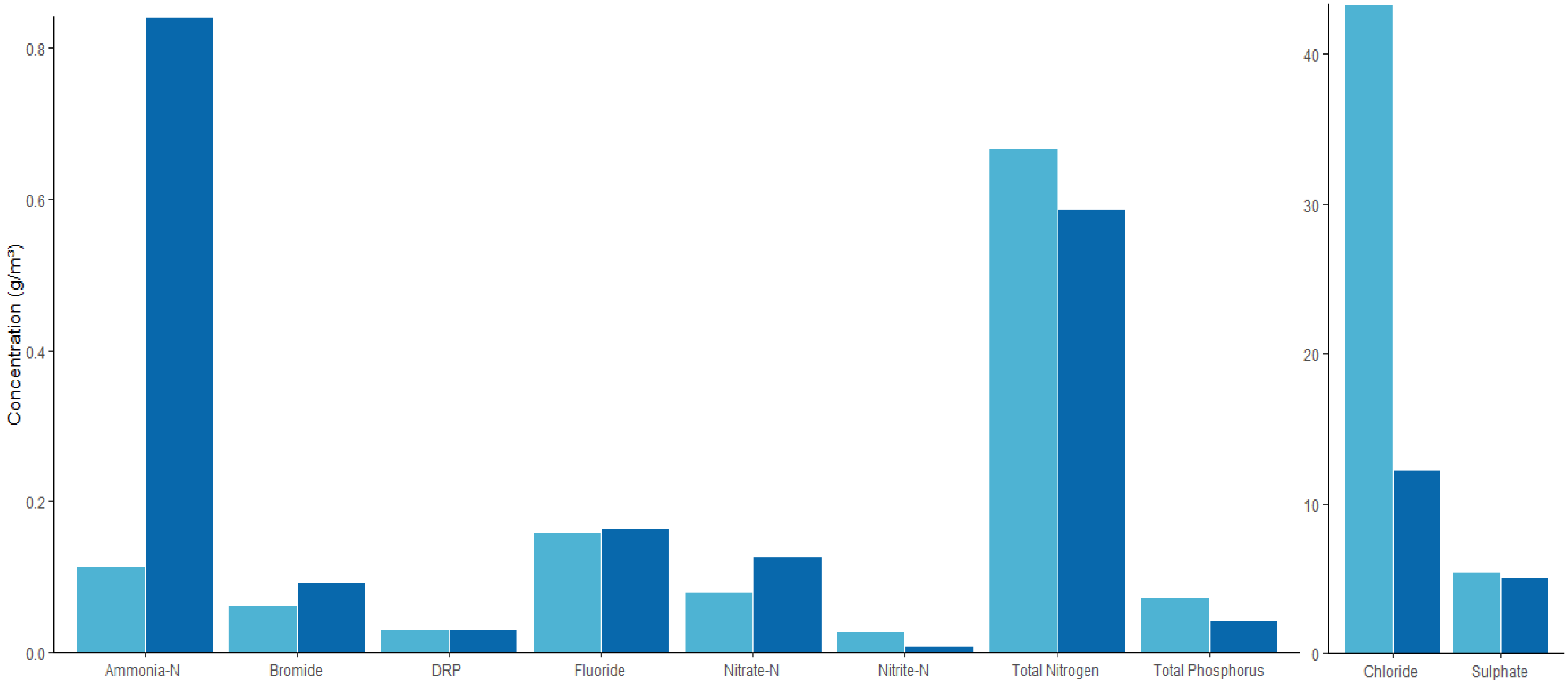

Figure 4. 7 Bargraph showing average concentrations of water quality parameters taken during the swap trial. Light blue annotates baseline data collected weekly at the site D/S between 8/02/2018-9/03/2018 (post swap), while dark blue shows average U/S levels, taken weekly for the whole duration of monitoring

All the DRP concentrations across the three sites averaged just above the detection limit (>0.025 g/m^3). DRP levels at all three sites post swap averaged around 0.03 g/m^3 (Table 4.4). Albeit low, these levels do however slightly exceed the recommended lowland DRP concentrations (0.01 g/m^3) (Davies-Colley, 2000).

Table 4. 4 Average concentrations (g/m^3) of water quality parameters taken during the swap trial. Data from the U/S site was collected weekly for the whole duration of the trial (8/02/18 and 9/03/18); data from the GW site was collected once monthly.

	U/S Average (g/m^3)	GW Average (g/m^3)	Post Swap D/S Average (g/m^3)
Bromide	0.09	0.05	0.06
Chloride	39.22	12.28	19.81
Fluoride	0.17	0.21	0.16
Ammonia	0.84*	0.06	0.11
Total Nitrogen	0.59	0.56	0.67
Nitrite	0.01	0.04	0.03
Nitrate	0.13	0.12	0.08
DRP	0.03	0.03	0.03
Total Phosphorus	0.04	0.05	0.07
Sulphate	6.46	5.07	5.46

* Uncharacteristically high ammonia value likely a result of machine error

4.2.2 Pre and Post Swapping Trial

Eight weeks of baseline monitoring was carried out prior to the swapping trial (15/12/2017- 8/02/2018). Monitoring was conducted for 5 weeks during the swapping trial from 8/02/2018 to 9/03/2018. There was no significant difference pre and post swap for Nitrate-N levels (P values >0.05) at the site D/S, with average Nitrate-N levels both pre and post-swap being 0.06 g/m^3 (Table 4.5).

D/S 2 (100 m downstream from the swapping site) was significantly different (P value <0.05) from all other sites (average Nitrate-N levels were 0.2 g/m3 pre-swap, and 0.35 g/m3 post swap), however it is possible that these uncharacteristically high Nitrate-N values are because the sampling site D/S 2 was located just upstream to a frequently used raceway (Fig. 3.2).

Table 4. 5 Nitrate-N (g/m³) concentrations both pre and post swap for each of the five drain sites. Only U/S and D/S will be assessed closely for this study

Nitrate-N g/m³		MD	U/S	D/S	D/S 2	OL
Pre-Swap	Count	12.00	7.00	8.00	2.00	8.00
	Average	0.06	0.13	0.06	0.20	0.18
	Minimum	0.06	0.06	0.06	0.40	0.06
	Maximum	0.06	0.35	0.06	0.40	0.54
	Standard Deviation	0.00	0.12	0.00	0.11	0.17
	Coefficient of Variance	0.00	0.91	0.00	0.55	0.95
Post Swap	Count			8.00	5.00	5.00
	Average			0.06	0.35	0.10
	Minimum			0.06	0.17	0.06
	Maximum			0.06	0.70	0.23
	Standard Deviation			0.00	0.21	0.08
	Coefficient of Variance			0.00	0.60	0.79

There was also no significant difference pre and post swap for DRP levels (P values >0.05) across all sites, except for at D/S 2 (P Value= 0.08) (Table 4.6). Average DRP concentrations upstream from the swap site (U/S) was 0.03 g/m³ (Table 4.6), while average DRP levels both pre and post swap downstream from the swap site (D/S) were 0.03 g/m³. Average pre and post swap DRP concentrations were significantly different at D/S 2 (100 m downstream from the swapping site). Average DRP levels were 0.03

g/m³ pre-swap, and 0.02 g/m³ post swap. However, as mentioned above, this is likely due to its proximity to the raceway.

Table 4. 6 DRP (g/m³) concentrations both pre and post swap for each of the five drain sites. Only U/S and D/S will be assessed closely for this study

DRP g/m³		MD	U/S	D/S	D/S 2	OL
Pre-Swap	Count	12.00	7.00	8	2	8
	Average	0.03	0.03	0.03	0.03	0.03
	Minimum	0.01	0.02	0.01	0.03	0.01
	Maximum	0.04	0.03	0.04	0.03	0.03
	Standard Deviation	0.01	0.00	0.01	0.00	0.01
	Coefficient of Variance	0.24	0.11	0.32	0.11	0.24
Post Swap	Count			5.00	5.00	5.00
	Average			0.03	0.02	0.02
	Minimum			0.02	0.01	0.01
	Maximum			0.03	0.03	0.03
	Standard Deviation			0.00	0.01	0.01
	Coefficient of Variance			0.14	0.38	0.27

4.2.3 Conclusions from the Swapping Trial

The average rate of water abstracted daily over the swapping trial was 4 L/s (Appendix 6). Given that for the majority of nutrients, concentrations did not differ significantly between the abstraction site (upstream), the receiving site (downstream), and groundwater in the first place; and given that 4 L/s was likely not a substantial enough amount of water to make any noticeable difference, it is not surprising that the swapping trial yielded no discernible improvement in water quality downstream. Also, the nutrient concentrations in drain waters during the summer swap trial months were measured very low, almost similar to the groundwater levels. Average Nitrate-N levels in the groundwater and drain waters were consistently low (<0.20 g/m³) during the summer swap trial period (Table 4.5). Furthermore, the standard for Nitrate-N in lowland drains is 0.44 g/m^3. Over the 13 weeks of sampling, on average, none of the sampling sites reached concentrations higher than 0.44 g/m^3, although OL (the most downstream site) did spike higher than the set standard (0.54 g/m^3) pre-swap (Table 4.5).

This analysis, therefore, rules out real time drain swapping over summer months (January – March), as the levels of Nitrate-N in the drains were consistently low (under 0.44 g/m^3) and so there was no apparent need for drain nutrient management during this period. There is however, still potential for harvesting drain waters during winter and spring months (when nitrate-N concentrations are higher) and then recycling that at a later point in time for summer irrigations.

4.3 Macrophyte Management Trial

4.3.1 Water Quality Parameters

Riparian shading was simulated across a drain channel by using a medium strength (~60%) shade cloth to evaluate the correlation between shade, plant cover, and nutrient concentrations. Three sites were monitored - the control site (OL) comprised a fully established riparian strip on one side, while the two treatment sites comprised the shade cloth over an excavated drain stretch (SCNM) and a shade cloth over a non-excavated stretch (SCM) (Fig. 3.4). These were monitored for 7 weeks between 26/01/2018 and 9/03/2018 for water quality and plant cover. A complete summary of water quality parameters for this trial can be found in Appendix 7.

The Nitrate-N level at the control site (OL) with the riparian strip was averaged at 0.13 g/m^3 (Table 4.7) and ranged from 0.02-0.54 g/m^3. This is significantly lower than both the two treatment sites (p value <0.05), where the average Nitrate-N values were 0.53 and 0.63 g/m3 for this site with existing pre-macrophytes (SCM) and the site where the macrophytes had been excavated (SCNM) respectively (Table 4.7). The two shade cloth sites did not differ significantly from each other for Nitrate-N. The standard for Nitrate-N in lowland drains is 0.44 g/m^3. The average Nitrate-N values for both shade cloth sites slightly exceeded this standard, while the average for the control site did not (0.13 g/m^3).

Although none of the sites differed significantly for DRP (with average values ranging from 0.2 to 0.3 g/m^3) (Table 4.7), they all slightly exceeded the recommended concentration for lowland waters (0.01 g/m^3) to avoid possible adverse ecosystem effects (Davies-Colley, 2000). The standard ammonia-N value for freshwater ecological health is 0.021 g/m3 (Davies-Colley, 2000). All three sites showed relatively high average levels of ammonia-N (ranging from <0.0-8.54 g/m^3) (Table 4.7). Although none of the sites were significantly different from each other, the averages for all sites exceeded the recommended standard. Interestingly, OL had significantly higher average Phosphate-P values (0.28 g/m^3) than both the shade cloths (<0.05 g/m^3). Again, both shade cloths did not differ significantly from each other (P value 0.936). The averages for the shade cloth sites were below 0.05 g/m^3 (Table 4.7).

Table 4. 7 Average concentrations (g/m^3) of water quality parameters and plant cover taken during the macrophyte trial. Data was collected from the two treatment sites (SCM and SCNM) and the control site (OL) weekly for the whole duration of the trial (26/01/2018-9/02/2019).

Average (g/m^3)	SCM	SCNM	OL
Bromide	0.08	0.07	0.08
Chloride	31.99	28.07	35.92
Fluoride	0.14	0.19	0.17
Ammonia-N	0.46	0.80	0.79
Total Nitrogen	0.99	1.00	0.63
Nitrite-N	0.04	0.00	0.01
Nitrate-N	0.53	0.63	0.13
DRP	0.02	0.02	0.03
Total Phosphorus	0.01	0.02	0.03
Sulphate	7.46	7.75	8.60
Macrophyte cover (%)	100	39.24	71.22
Periphyton cover (%)	0	24.93	5.88

All three sites differed significantly for macrophyte cover (P value <0.05) (Table 4.7). SCM ranged from 7.6% to 100% coverage, and the mean was 100%, while SCNM ranged from 7.6% to 73% coverage, and had the lowest mean (39.24%). There was no association between macrophyte and nitrate-N concentration at SCM, though there is a significant strong positive correlation between the two variables at SCNM (rho= 0.79, p value = 0.05).This suggests that at this site, macrophyte cover increases with nitrate-N concentration. The OL site had an average mean of 71.22% for macrophyte cover. Cover ranged from 27.8% to 100%. The correlation coefficient between macrophyte cover and nitrate-N at this site was -0.53 and the p-value was 0.06, meaning that there was a relatively strong negative correlation between the two, albeit not significant. This suggests that, unlike the other two sites, at OL increased macrophyte growth reduces nitrate-N concentrations.

For periphyton cover, SCNM differed significantly from both the other sites (p value <0.05). It had a higher average cover (24.93%), as opposed to 0% and 5.88% for SCM and OL (Table 4.7). It is likely that the increased periphyton cover at SCNM was due to the channel having been mechanically cleared, allowing for unhindered colonisation. The correlation coefficient between periphyton cover and nitrate-N at OL was 0.5357 and the p-value was 0.2357, meaning that there was a relatively strong positive correlation between the two, albeit not significant. There was no association between periphyton and nitrate-N concentration at SCM, and only a weak positive correlation between the two variables at SCNM, although not significant (rho= 0.1109, p value = 0.72).

Shade effects on periphyton biomass are particularly important, especially regarding nitrate-N uptake. In a New Zealand study by J. M. Quinn, Cooper, Stroud, and Burrell (1997) on the effects of shade on periphyton in a pasture environment was investigated by comparisons in 12 replicate channels where shade cloth reduced the photosynthetically available radiation by 0, 60, 90, and 98%. They found that nitrate-N uptake rates by periphyton decreased progressively as shade increased from 60% through 90% to 98%. Overall, nitrate-N uptake rates were significantly higher under 0 and 60% shade than under 90%, where the rates were significantly higher than at 98% shade. The shade cloths in the current study gave ~60% shade. Although shade level was not able to be measured at the OL site, it is likely that it was less than this value, particularly given that the channel was only shaded on one side. This could explain the reduced/non-existent association between periphyton and nitrate-N at the shade cloth sites.

Many local studies have examined the potential for in-drain processes, (particularly macrophyte and periphyton uptake) to reduce drain nutrient loads and have found similar results. For example, an ongoing trial from Whangamata found that in-drain plant uptake reduced drain nutrient loads, but with riparian plant growth and subsequent drain shading, both in-drain plant biomass and associated nutrient uptake declined (Howard-Williams & Pickmere, 2010). An in *situ* short-term nutrient addition experiments showed that periphytic biofilms can reduce summer drain nitrate loads up to 59%, particularly in shallower, faster-flowing drains with higher water column P availability (F. Matheson, Tank, & Costley, 2011).

Conclusions

Vegetated drains have some capacity for nutrient retention. This can be tentatively be implied from the current trial, although more research is needed, as well clearly demonstrated in multiple other studies, including the one by Nguyen and Sukias (2002). It was clear that the shading and mechanical clearance reduced the in-drain plant cover (SCNM), and that nitrate-N levels were higher at these sites. Although there are many factors which could be influencing higher nitrate levels at the shaded sites, previous studies suggest that the higher levels of nitrate-N were likely due to the shading. This raises the question about whether macrophyte management is necessary for waterways such as surface drains, and if riparian shading is even necessary on drains. It appears that macrophyte growth in drains play a major role in reducing nutrient loads. One could consider their presence a naturally occurring filter, similar in function to shallow wetlands, and can manage their growth and harvesting to increase update of nutrients in drains during summer season.

4.4 Drainage Harvest and Reuse

Estimates of nutrient losses by the farm nutrient budgeting model, OVERSEER were used to investigate the potential to harvest and recycle drainage water on the case study farm (about 297.5 ha) (Fig. 3.6). This was then repeated using the actual water quality values attained during sampling and the results were compared to assess effect of different drainage water quality and patterns on potential of drainage harvesting and reuse.

Four potential scenarios were investigated for the potential to harvest and recycle drainage water at Hyde Park, using both in-stream concentrations estimated by OVERSEER, and concentrations obtained from sampling on the case study farms. According to OVERSEER, the average drainage at the farm is 300 mm/yr, while the average irrigation needs are 400 mm/yr. It is assumed that a pond 69,000 m^3 in size is installed on the farm. At the farm's drainage rate, 23 ha of land must be drained to fill the pond.

Scenario 1

Harvest from 30 ha of Effluent Irrigated Pukepuke and irrigate back to the same block using OVERSEER values

Given that one of the highest levels of N (88 Kgs N/ha/yr) lost to drain water is from the Effluent Irrigated Pukepuke Block (Fig. 4.8), that will be the reference block for this desktop exercise. Additionally, the PukePuke sand is characterised as imperfectly drained and is more likely to have artificial drainage installed, therefore making it a more practical soil block to use over the Himatangi sands which are characteristically well drained. The nitrate-N concentration in the drain water is 29.3 g/m^3, therefore the total potential N harvested in the storage pond (69,000 m^3) would be 2,024 Kg N. The original irrigated area (Effluent Irrigated Pukepuke Block) is 90.9 ha. In an effort not to increase capital costs more than they already are ($622,000) by constructing a larger storage pond, a new smaller hypothetical block (30 ha) was carved out of the Effluent Irrigated Pukepuke block (originally 90.9 ha) on OVERSEER (SW Effluent irrigated Puke on Fig 4.8).

Block name	Total N lost	N lost to water	N in drainage *	N surplus	Added N **
	kg N/yr	kg N/ha/yr	ppm	kg N/ha/yr	kg N/ha/yr
Irrigated Pukepuke	6,118	74	**25.8**	277	371
Irrigated Himatangi	7,465	74	**25.7**	278	372
Dry Pukepuke	2,516	42	**17.2**	120	161
Effluent Irrigated Puke	4,692	88	**30.5**	340	444
Effluent Irrigated Hima	2,347	88	**30.5**	340	444
Effluent Dry Puke	369	53	**21.4**	217	271
Summer brassica	1,064	53	**18.9**	124	173
Winter brassica	2,219	111	**35.1**	107	141
Dry Pukepuke (to Lake)	5	2	N/A		
SW Effluent irrigated Puke	2,365	90	**31.1**	326	370
Other sources	631				
Whole farm	29,790	68			
Less N removed in wetland	0				
Farm output	29,790	68			

Figure 4. 8 Screenshot from OVERSEER showing nitrate-N levels lost to the farm broken down by soil blocks. Effluent Irrigated Pukepuke leaches 88 Kg N/ha/yr. The newly created block for Scenario 1 is SW Effluent irrigated Puke, and leaches 90 Kg N/ha/yr.

Given that average irrigation is 400 mm, it would take 120,000 m^3 to irrigate this land. There would be 69,000 m^3 available from the storage pond, and there is 51,000 m^3 available from the current system. The groundwater supply that is used currently for irrigation has an average nitrate-N concentration of 0.12 g/m^3 (Appendix 5). Altogether, 2030.12 Kg N would be available from the combined irrigation water supply. The resulting concentration in the combined water supply for irrigation would be 16.9 g/m^3. The N value on the nutrient concentrations in irrigation water on OVERSEER was updated to reflect this.

Table 4. 8 Table showing the steps taken to calculate the amount of nitrate-N that can be harvested from the Effluent Irrigated Pukepuke Block on Hyde Park and recycled back to the same block over the dry summer months, using a combination of a storage pond and the current irrigation system.

Irrigated area (Effluent Irrigated Pukepuke Block)	30 ha
Total volume needed to irrigate 30 ha	120,000 m^3
N lost /ha from Effluent Irrigated Pukepuke Block (Overseer)	88 Kg N /ha/yr
Concentration of N in drainage (modelled)	29.3 g/m^3
Suggested Storage Pond Volume (~75% of annual drainage)	69,000 m^3
Total N harvested in storage pond	2024 Kg N
Irrigation water needed from current system	51,000 m^3
Natural nitrate-N level in groundwater (irrigation source)	0.12 g/m^3
Total N input from current irrigation system	6.12 Kg N
Total N in irrigation water (storage pond + current system)	2030.12 Kg N
N available to be irrigated over summer months (4) over 30 ha	16.9 Kg N/ha/month
Average levels of N put on land in form of urea	28 Kgs N/ha/month
N lost /ha from Effluent Irrigated Pukepuke Block (Overseer) previously	88 Kg N /ha/yr
N lost /ha from Effluent Irrigated Pukepuke Block (Overseer) post urea update	88 Kg N /ha/yr

On average, about 28 Kgs N/ha/month are fertilised onto the land in the form of urea (60 Kgs urea/ha/month total). Assuming the combined water supply is used over the irrigation season, there would be an additional 16.9 kg N/ha/month spread onto the land. To keep the amount of N entering the system in balance, urea loads over the 4 summer months could be decreased to 20 Kgs urea (9 Kgs N/ha/month) and substituted with the N in the irrigation water.

These values were entered into OVERSEER to ensure that the reuse of N from irrigation would not drive N leaching levels up. N in water is extremely soluble, much more than N in the form of urea. The resulting N leaching rate during drain water reuse, however, was at 88 Kgs N/ha/yr, consistent with the leaching rate before the modelling exercise (88 Kgs N/ha/yr).

Capital costs for setting up such a project, including cost per m^3 of dam, resource consent, and power, would come to approximately $602,000 (Table 4.9). Annual running costs, including cost of capital, and savings in urea, would come to approximately $52,080. The capital cost alone of purpose-built dams may be cost prohibitive, but when expressed as an annual cost per kg N attenuated ($25.7 per kg N attenuated/yr), drainage water recycling may seem more achievable.

Table 4. 9 Table showing the potential capital and annual costs for harvesting from the Effluent Irrigated Pukepuke Block on Hyde Park and recycling back to the same block using OVERSEER estimates (Scenario 1), as well as estimated cost per Kg of N attenuated

Capital Costs ($)		Annual Costs ($)	
Cost/m^3 of dam	8	Repair and Maintenance	510
Total cost of dam	552,000	Cost of capital 8%	48,160
Pump	30,000	Saving in urea	-2640
Resource consent	20,000	Lost profit	4050
		Power	2,000
Total	$602,000	Total	$ 52,080
Total nitrate attenuated in storage pond			2,024 Kg N
Total Annual Costs			$52,080
Cost per kg N attenuated			$25.7 per Kg N

Scenario 2

Harvest from Effluent Irrigated Pukepuke and irrigate back to same block using Water Quality values obtained from drains in sampling

In most regards, this scenario is very similar to the previous one. The crucial difference is that rather than using the modelled nitrate-N concentrations from OVERSEER, an average value was taken the sampling undertaken at Hyde Park. The months with the highest average nitrate-N concentrations were June-September (Table 4.2), where the average N concentration was 3.3 g/m^3. Furthermore, it was clear from sampling that there was enough drainage water captured over those same winter months (178,200 m^{3,} Table 4.3) to meet the irrigation needs for the 30 ha Effluent Irrigated Pukepuke block unaided (120,000 m^3).

If it is assumed that the storage pond was increased in size to meet the demands of irrigation single handed (120,000 m^3 as opposed to 69,000 m^3) and knowing that the concentration of N is 3.3 g/m^3, then the total potential N harvested in the pond would be 398.5 Kg N (Table 4.10).

As with the previous scenario, about 28 Kgs N/ha/month are fertilised onto this block in the form of urea (60 Kgs urea/ha/month total). There would be an additional 3.3 Kg N/ha/month spread onto the land from the drainage water irrigation system. As before, urea loads over the 4 summer months could be decreased to 20 Kgs urea (9 Kgs N/ha/month) and substituted with the N in the irrigation water (Table 4.10).

Table 4. 10 Table showing the steps taken to calculate the amount of nitrate-N that can be harvested from the Effluent Irrigated Pukepuke Block on Hyde Park and recycled back to the same block over the dry summer months, using only drain water and nitrate-N concent

Irrigated area (Effluent Irrigated Pukepuke Block)	30 ha
Total volume needed to irrigate 30 ha	120,000 m^3
Concentration of N in drainage water	3.3 g/m^3
Suggested Storage Pond Volume	120,000 m^3
Total N harvested in storage pond	398.5 Kg N
N available to be irrigated over summer months (4) over 30 ha	3.3 Kg N/ha/month
Average levels of N put on land in form of urea	28 Kgs N/ha/month
N lost /ha from Effluent Irrigated Pukepuke Block (Overseer) previously	88 Kg N /ha/yr
N lost /ha from Effluent Irrigated Pukepuke Block (Overseer) post urea update	90 Kg N /ha/yr

These values were updated on OVERSEER, and the resulting total loss of N was at 90 Kgs N/ha/yr, marginally higher than the leaching rate before we started the modelling exercise (88 Kgs N/ha/yr). The differing rates in N lost from the two Dry Pukepuke blocks would come predominantly from increased leaching rates as less N is lost to the air as volatilisation from the fertiliser (having replaced the fertiliser with irrigation), therefore leaving more to leach into the water.

Capital costs would in this scenario would increase extortionately ($1,010,000) due to the larger pond size, while annual running costs would increase to $89,990, meaning that to recycle drainage water on the Effluent Irrigated Pukepuke block using nitrate-N concentrations from actual sampling, the annual cost per kg N attenuated would be $225.8 per kg N attenuated/yr (Table 4.11).

Table 4. 11 Table showing the potential capital and annual costs for harvesting from the Effluent Irrigated Pukepuke Block on Hyde Park and recycling back to the same block using actual nitrate-N concentrations from sampling (Scenario 2), as well as estimated cost

Capital Costs ($)		Annual Costs ($)	
Cost/m^3 of dam	8	Repair and Maintenance	510
Total cost of dam	960,000	Cost of capital 8%	80,800
Pump	30,000	Saving in urea	-519.79
Resource consent	20,000	Lost profit	7,200
		Power	2,000
Total	$1,010,000	Total	$ 89,990

Total nitrate-N attenuated in storage pond	398.5 Kg N
Total Annual Costs	$89,990
Cost per kg N attenuated	$225.8 per Kg N

Scenario 3

Harvest from Effluent Irrigated Pukepuke but irrigate back to Dry PukePuke using OVERSEER values

To explore the option additional benefits of drain water irrigation on previously dry land, a new hypothetical block was carved out of the Dry Pukepuke block (originally 60 ha) on OVERSEER. Currently, 42 Kgs N/ha/yr are lost to drain water on the Dry Pukepuke soils (Fig. 4.9). Unlike the previous scenario however, the receiving irrigated area would be Dry Pukepuke and so would have no existing irrigation system. At an irrigation rate of 400 mm, the volume in the drain storage pond (69,000 m^3) alone could irrigate 17.25 ha (SW (dry) Pukepuke, Table 4.12). The nitrate-N concentration in the drain water is 29.3 g/m^3, therefore the total potential N harvested in the storage pond would be 2,024 Kg N.

Block name	Total N lost kg N/yr	N lost to water kg N/ha/yr	N in drainage * ppm	N surplus kg N/ha/yr	Added N ** kg N/ha/yr
Irrigated Pukepuke	6,088	73	**25.5**	275	369
Irrigated Himatangi	7,429	73	**25.4**	275	369
Dry Pukepuke	1,790	42	**17.1**	120	161
Effluent Irrigated Puke	7,021	87	**30.3**	339	444
Effluent Irrigated Hima	2,352	87	**30.4**	339	444
Effluent Dry Puke	366	52	**21.3**	216	271
Summer brassica	1,060	53	**18.9**	124	173
Winter brassica	2,197	110	**34.7**	107	141
Dry Pukepuke (to Lake)	5	2	N/A		
SW (dry) Pukepuke	752	50	**17.5**	49	35
Other sources	637				
Whole farm	29,697	68			
Less N removed in wetland	0				
Farm output	29,697	68			

Figure 4. 9 Screenshot from OVERSEER showing nitrate-N levels lost to the farm broken down by soil blocks. Effluent Irrigated Pukepuke leaches 88 Kg N/ha/yr. The newly created block for Scenario 3 is SW (dry) Pukepuke, and leaches 50 Kg N/ha/yr.

Approximately 160 Kgs N/ha/yr is needed to be fertilised onto the Dry Pukepuke block. Assuming the drain storage pond water supply is used over the irrigation season, there would be 29.3 kg N/ha/month spread onto the land from November through to March, resulting in a total of 118 Kgs N/ha/yr (Table 4.12). This means that 42.6 Kgs N/ha/yr will still be needed to be applied in the form of Urea fertiliser.

Table 4. 12 Table showing the steps taken to calculate the amount of nitrate-N that can be harvested from the Effluent Irrigated Pukepuke Block on Hyde Park and recycled back to a newly created Dry Pukepuke Block over the dry summer months (Scenario 3), using only

N lost /ha from Effluent Irrigated Pukepuke Block (Overseer)	42 Kg N /ha/yr
Concentration of N in drainage (modelled)	29.3 g/m³
Suggested Storage Pond Volume	69,000 m³
N harvested in storage pond	2024 Kg N
Area that storage pond alone can irrigate	17.25 ha
N available to be irrigated over summer months (4) over 17.25 ha	29.3 Kg N/ha/month
Average levels of N put on land in form of fertiliser	160 Kg N/ha/yr
Levels of N still needed	42.6 Kg N/ha/yr
N lost /ha from original Dry Pukepuke Block	42 Kg N /ha/yr
N lost /ha from new Dry Pukepuke Block	50 Kg N /ha/yr

When updated on OVERSEER, the resulting N lost to drainage waters would be 50 Kg/ha/yr, compared to a rate of 42 Kg/ha/yr from the original Dry Pukepuke Block. The differing rates in N lost from the two Dry Pukepuke blocks come predominantly from increased leaching rates as less N is lost to the air as volatilisation from the fertiliser (having replaced the fertiliser with irrigation), therefore leaving more to leach. The difference in total N leaching between the two blocks (original Dry Pukepuke Block vs new partly irrigated 17.25 ha Pukepuke Block) would be 138 Kg N/ yr. The total N attenuated would therefore be 2024 – 138 ~ 1886 Kg/yr (Table 4.14).

Before looking at the annual costs per Kg N attenuated for this system, one needs to first consider the cost of setting up new irrigation system and the extra income from milk solids as a result of increased growth due to supplemental irrigation. By irrigating the harvested drain water onto this new block at a rate of 29.3 Kg N/ha/month, a growth response in pasture would likely occur. The subsequent increase in milk solids production would result in an estimated $62,100 extra income (Table 4.13).

Table 4. 13 Table showing potential extra income generated by Scenario 3

N added over summer months (4) per ha	29.3 Kg N/ha/month
N added as urea fertiliser	42.6 Kg N/ha/yr
Response of growth to irrigation	15 Kg DM/ha/mm of irrigation
Total increase in pasture	101,250 Kg DM
Increase in pasture per ha	6 t/ha
Kg Dry Matter/Kg Milk solids	10
Extra milk solids	10,125
Increase in MS/cow	8.6
$/Kg Milk Solids	6
Extra income	$62,100

Capital costs would remain approximately the same (Table 4.14), while annual running costs, including cost of capital for both the dam and the irrigation system, would come to approximately $62,740. There would be an additional cost of $86,250 to set up an irrigation system. The capital cost plus irrigation system set up may seem cost prohibitive, but when expressed as an annual cost per kg N attenuated ($0.34/kg N attenuated/yr), it appears to be almost negligible. This is probably because of the economic benefits come from the additional irrigation water that is created by harvesting the winter drain flows.

Table 4. 14 Table showing the potential capital and annual costs for harvesting from the Effluent Effluent Irrigated Pukepuke Block on Hyde Park and recycled back to a newly created Dry Pukepuke Block over the dry summer months (Scenario 3) using only storage pond water, and OVERSEER values

Irrigation set up ($)		Annual costs ($)	
Irrigation equipment/ha	5000	Cost of capital 8% -irrigation system	6,900
		Cost of capital 8% -dam	48,160
		Lost profit from land	4,050
		Power	2,000
		Urea ($700/t)	1,120
		Repair and maintenance	510
Total cost of irrigation	$ 86,250	Total Annual Costs	$62,740
Net N attenuation		1,886 Kg N	
Extra Income		$62,100	
Total Annual Costs		$62,740	
Cost per kg N attenuated		$0.34	

Scenario 4

Harvest from Effluent Irrigated Pukepuke but irrigate back to Dry PukePuke using Water Quality values obtained from drains in sampling

Again, this scenario is very similar to the previous one, with only difference being that rather than using the modelled nitrate-N concentrations from OVERSEER, the average value was taken the sampling undertaken at Hyde Park (3.3 g/m^3, Table 4.2). The sampled nitrate-N concentration in the drain water was 3.3 g/m^3 in the highest months (June-September), therefore the total potential N harvested in the storage pond (69,000 m^3) would be 229 Kg N.

Assuming the drain storage pond water supply is used over the irrigation season, there would be an additional 3.3 kg N/ha/month spread onto the land from November through to March, resulting in a total of 13.3 Kgs N/ha/yr (Table 4.15). Because this block of land needs 160 Kg N/ha/yr, an additional 146.7 Kg N/ha/yr will need to be added in the form of urea.

When updated on OVERSEER, the resulting N lost to drainage waters would be 50 Kg/ha/yr, compared to a rate of 42 Kg/ha/yr from the original Dry Pukepuke Block. As previously found, the differing rates in N lost from the two Dry Pukepuke blocks come predominantly from increased leaching rates as less N is lost to the air as volatilisation from the fertiliser (having replaced the fertiliser with irrigation), therefore leaving more to leach. The difference in total N leaching between the two blocks (original Dry

Pukepuke Block vs new partly irrigated 17.25 ha Pukepuke Block) would be 138 Kg N/ha/yr. The total N attenuated would therefore be 229 – 138 ~ 91 Kg/yr (Table 4.16).

Table 4. 15 Table showing the steps taken to calculate the amount of nitrate-N that can be harvested from the Effluent Irrigated Pukepuke Block on Hyde Park and recycled back to a newly created Dry Pukepuke Block over the dry summer months (Scenario 4), using only

N lost /ha from Effluent Irrigated Pukepuke Block (Overseer)	88 Kg N /ha/yr
Concentration of N in drainage	3.3 g/m³
Suggested Storage Pond Volume	69,000 m³
N harvested in storage pond	229.14 Kg N
Area that storage pond alone can irrigate	17.25 ha
N available to be irrigated over summer months (4) over 17.25 ha	3.3 Kg N/ha/month
Average levels of N put on land in form of fertiliser	160 Kgs N/ha/yr
Levels of N still needed	146.7 Kg N/ha/yr
N lost /ha from original Dry Pukepuke Block	42 Kg N /ha/yr
N lost /ha from new Dry Pukepuke Block	50 Kg N /ha/yr

Capital costs would remain approximately the same, as would extra income (Table 4.16). Annual running costs would come to $65,820 (Table 4.16), with the majority of the difference coming from increased costs of Urea. When expressed as an annual cost per kg N attenuated, it comes to $40.8.

Table 4. 16 Table showing the potential capital and annual costs for harvesting from the Effluent Effluent Irrigated Pukepuke Block on Hyde Park and recycled back to a newly created Dry Pukepuke Block over the dry summer months (Scenario 4) using only storage pond

Irrigation set up ($)		Annual costs ($)	
Irrigation equipment/ha	5000	Cost of capital 8% -irrigation system	6,900
		Cost of capital 8% -dam	48,160
		Lost profit from land	4050
		Cost of Urea ($700/t)	4,200
		Power	2000
		Repair and maintenance	510
Total cost of irrigation	$ 86,250	Total Annual Costs	$65,820
Net N attenuation		91.1 Kg N	
Extra Income		$62,100	
Total Annual Costs		$65,820	
Cost per kg N attenuated		$40.8	

Conclusions from drainage harvesting and reuse

Although there have been a significant number of on-farm storages constructed in recent years across New Zealand, the majority are within irrigation schemes. The cost of on-farm storage (per m^3 of water stored) is higher than bulk storage (Riley Consultants Ltd, 2010). Capital costs from the scenarios in this study reflect this, with a range of $602,000 to $1,010,000. Disconcerting as that is, modelled attenuation costs per Kg of nitrate-N offset that to some degree. Given that the net attenuation ranges from 91-2024 Kg N observed in this study, the cost of harvesting and recycling drainage waters equates to $0.34-225 per Kg N depending on different scenarios. Ultimately, it was found that costs of harvesting and recycling drainage waters are influenced by two main factors; whether there is a pre-existing irrigation system in place or not, and the concentration of nitrate-N in the drainage water.

Because of the high capital costs, such high attenuation costs ($225 per Kg N), and such little return in terms of savings in urea, Scenario 2 (where water is harvested from Effluent Irrigated Pukepuke and irrigated back to the same block at a concentration of 3.3 g/m^3) is not cost effective. Scenario 1 however (where there is a pre-existing irrigation system), with attenuation costs of $25.7 per Kg N, is significantly more reasonable. If the concentration in drain waters is changed to 3.3 g/m^3 to reflect the actual concentration of nitrate-N found in the drains in the case study (that is, leaching 10 Kgs N/ha/yr as opposed to 88 Kgs); attenuation costs would be $236.6 per Kg N attenuated/yr and only a total of 230 Kgs N would be attenuated from the farm. At the higher concentration of 29.3 g/m^3, a total of 2,024 Kgs would be attenuated from the farm. A simple linear relationship can be formed between the two scenarios on already irrigated land (Fig. 4.10), and it can be seen that it is clearly more cost efficient per Kg N attenuated if there are higher concentrations of nitrate-N in the drains for which to be harvested and recycled.

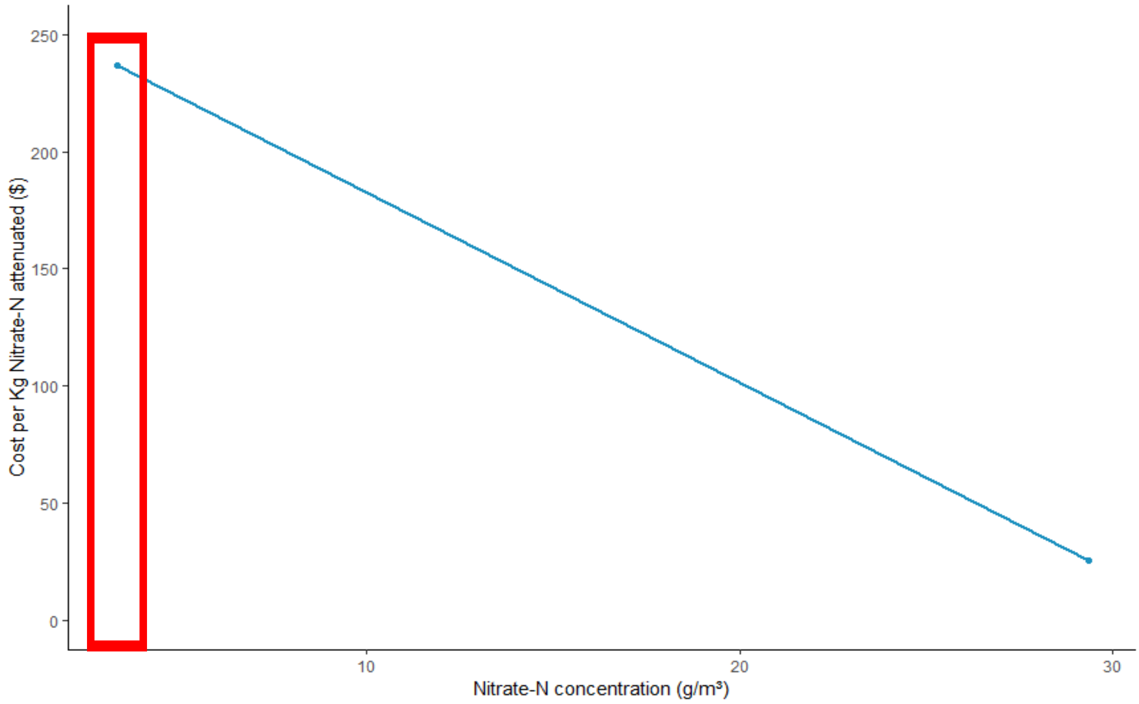

Figure 4. 11 Simple sensitivity analysis showing the different costs per Kg N attenuated for Scenario 1 using the OVERSEER nitrate-N estimates and the actual nitrate-N values from sampling. The red box indicates the actual values for Hyde Park and the associated costs.

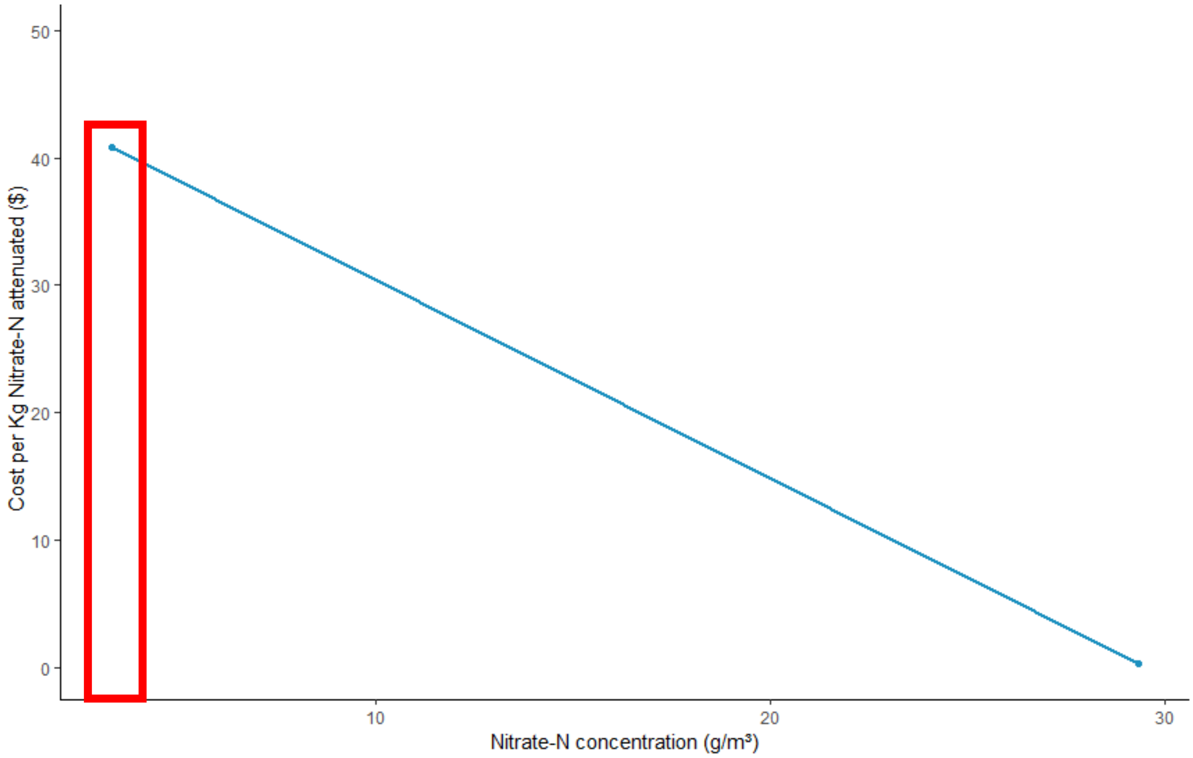

Figure 4. 10 Simple sensitivity analysis showing the different costs per Kg N attenuated for Scenario 3 and 4, using the OVERSEER nitrate-N estimates and the actual nitrate-N values from sampling. The red box indicates the actual values for Hyde Park and the associated costs.

The same thinking can be applied to Scenario 3 and 4 where a new irrigation system is applied (Fig. 4.11). Under Scenario 3, attenuation costs are almost insignificant ($0.34 per Kg N attenuated/yr) at a concentration rate of 29.3 g/m^3, and a total of 2,024 Kgs N are attenuated from the farm. However, at the actual nitrate-N concentration (3.3 g/m^3) (Scenario 4), attenuation costs are relatively higher ($40.8 per Kg N attenuated) and only 229 Kgs N are harvested from the farm.

Both these scenarios show that it is not only more cost efficient per Kg N attenuated, but also that more nitrate-N is harvested from the farm and thus prevented from entering the streams, if there are higher concentrations of nitrate-N in the drains that can be harvested and recycled. As it is, there is a stark difference between the modelled OVERSEER nitrate-N concentrations in water and what was found in the drains. As discussed in 4.1.4, this disparity between the OVERSEER estimates and the actual drainage results suggests nitrate-N attenuation below the root zone and mixing up of drainage waters with reduced shallow groundwaters in the drains. Realistically, harvesting and recycling would not be economical on Hyde Park, but should still be considered as a mitigation tool in areas with higher surface water nitrogen-N concentrations. However, site-specific conditions should first be pre-assessed for cost-effectiveness. For example, capital costs could be reduced on areas with undulating topography as the land would be naturally suited for the construction of storage ponds.

These removal rates can be compared with the mean costs calculated for other field-edge practices by L. Christianson, Tyndall, and Helmers (2013) using similar methods. They found (in USD) removal costs of $2.02 Kg N for controlled drainage, $2.12 Kg N for denitrification bioreactors, and $2.92 Kg N for denitrification wetlands. Albeit, these measures and associated costs are designed to treat drainage waters in Midwestern USA agricultural landscapes. Depending on the concentration of nitrate-N in the drains, and if it is recycled back to already irrigated land, or previously dry land; the cost effectiveness for harvesting and recycling is not only comparable to other field-edge practices, but also comparatively cheaper (Table 4.17).

Table 4. 17 Table comparing estimated costs per N attenuated for different nutrient management techniques

Drainage Management Method	$ per Kg N attenuated	Source
Wetlands	USD 2.92-3.26	Christianson et al. (2013) (Louis A Schipper, Will D Robertson, Arthur J Gold, Dan B Jaynes, & Stewart C Cameron, 2010)
Bioreactors	USD 2.12	Christianson et al. (2013)
Saturated Buffer Strips	USD 2.17	(Addy et al., 2016)
Controlled Drainage	USD 2.02	Christianson et al. (2013)
Harvesting and Recycling	USD 0.22-154 (NZD 0.34-236)	Case Study at Hyde Park

However, Christianson et al. (2013) also emphasised that ‘no individual technology or management approach will be capable of addressing drainage water quality concerns in entirety’, meaning that it is likely that a variety of practices applied across the landscape will be necessary to achieve a higher degree of reduction in nitrate losses in drainage waters. Furthermore, areas where there are naturally high nitrogen reduction processes (such as Hyde Park) would not necessarily benefit environmentally or economically from large changes in farm practices for nitrogen mitigation.

Chapter 5

Summary and Conclusions

Chapter 5: Summary and Conclusions

Artificial drainage is an intrinsic and necessary component of successful agricultural systems, particularly on poorly drained soils and/or shallow groundwater areas. It is estimated that 2.5 million hectares of land in NZ is currently artificially drained. However, while being beneficial to the agriculture production system, these drains act as direct conduits for nutrient loss from agricultural lands to receiving waters. By modifying the soil hydrology, artificial drainage shorten the flow pathway length and time that water-soluble nutrients such as nitrate take to flow from the soil profile to waterways. High levels of nutrients can stimulate algal blooms and nuisance algae growths in the waterways, leading to repercussive in-stream ecological effects. Between 2008 and 2017, up to 40% of monitored sites throughout the country showed degrading trends in terms of Total Nitrogen and DRP.

As such, this thesis aimed to help develop appropriate in-field and/or edge-of-field drainage management practices, focusing on the surface drains, and inform nutrient management plans for intensified land use to maintain or enhance water quality in the Manawatu-Wanganui Region. In particular, the thesis focused on nutrient losses in coastal sandy soils in the region, where surface drainage is generally required to keep naturally shallow groundwater levels below the root zone during the wet season, and irrigation is generally practiced to maintain soil water supply for plant growth in the summer season. The thesis focused development of targeted and effective drainage management tools that can reduce nutrient runoff in surface drains in intensive sand country farms in the coastal Rangitikei catchment. Specifically, the thesis objectives were to characterise drainage flow and water quality patterns; to assess the potential of harvesting and reuse of drainage water for irrigation; and to assess the management of macrophytes as a tool for nutrient uptake in drain waters. To achieve this, the thesis developed and implemented a combination of detailed field measurements and experiments (during December 2017 – February 2019) to characterise the drainage patterns and water quality and assess The potential of harvesting and reuse of drainage water for irrigation, and the potential management of macrophytes as a tool for nutrient uptake in drain waters.

Drainage Characterisation

Weekly drain water quality samples and in stream flow gaugings were taken in 4 selected surface drains at Hyde Park (located in the Santoft are) between June 2018 and February 2019. The collected drain water samples were analysed for a range of nutrients, including nitrate-N and DRP. The nitrate-N and discharge values from each week sampled were compiled to examine the spatial and temporal variability across a year. No current data for nitrate-N was collected in the months March, April and May 2018, so nitrate-N concentrations measured in drains waters during April 2016 and May 2016 from a previous study on the same farm were used to characterise drain nitrate-N concentrations through the year.

The main findings of this case study were that there is high temporal variability in drainage flow and nitrate-N. The observed nitrate-N, for example, varied from 0.06 to 5.96 g/m3 throughout the year, and drain flow varied from 0 - 62 l/s. In winter season (June-September), the drains are characterised by high flows (average of 17 l/s) and high nitrate-N concentrations (average of 3.3 g/m^3), while in summer season (November-February), the drains are characterised by low flows (average of 6.8 l/s) and low nitrate-N (average of 0.52 g/m^3). This is probably because reduced shallow groundwater becomes the primary source of drainage water in the summer months, while nitrate-N rich water percolation from the soil profile is the primary source of drainage in the winter. It should be noted that based on the drain sampling, it was estimated that approximately 11 Kg N/ha/yr was leaving the farm, whereas OVERSEER estimated 68 Kg N/ha/yr as leaching from the root zone. This suggests nitrate-N attenuation below the root zone and mixing up of drainage waters with reduced shallow groundwaters in the drains. This is supported by a previous study (Smith et al. 2017) reporting reduced groundwater conditions conducive for subsurface denitrification at the study site.

Groundwater – drainage water swapping for summer irrigation

The first drainage management trial investigated the potential effects of swapping degraded drain water with relatively clean groundwater in real time for the purpose of recycling nutrients via irrigation over the dry summer period (December 2017- February

2018). In one of the study drain, eight weeks of baseline monitoring was carried out prior to the swapping trial (December 2017- February 2018). The drain flow was swapped with groundwater at a rate of 4 l/s. Further monitoring was conducted for 5 weeks during the swapping trial from February 2018 to March 2018. However, due to the afore mentioned low nitrate-N levels in the drain during the summer months, the opportunity for real time drain – groundwater swapping was limited. Average nitrate-N levels in both the groundwater and drain waters were consistently low (< 0.20 g/m^3) over the summer. The average nitrate-N levels both the pre and post-swap were 0.06 g/m^3 in the study drain. The nitrate-N concentrations did not differ significantly ($P > 0.05$) between the abstraction site (upstream), the receiving site (downstream), and groundwater in the first place, suggesting that the potential of real time drain water swapping with groundwater for summer irrigation offers negligible benefits in terms of reducing nutrient flows in drains. This is particularly for this case study where the low nitrate-N levels (< 0.2 g/m^3) were observed in the drain during the summer months. This also indicates that there was no apparent need for drain nutrient management during the summer periods, where the drain flow appears to be dominated by flows from reduced (low nutrient levels) shallow groundwaters.

Macrophyte management to reduce nutrient levels

Macrophyte can be managed to help uptake nutrients from drain water, particularly during the spring and summer season. The second drainage management trial was set up to explore the potential of nutrient uptake by macrophytes, and if shading influenced its effectiveness. This was observed at three consecutive sites along the same drain monitored weekly between January 2018 and March 2018. One site had an established riparian strip on one side, while the other two had artificial shadings. Pre-existing macrophytes were removed from under one of the shade cloths. The site under the natural riparian strip had the lowest average nitrate-N concentration (0.13 g/m^3), and it was significantly lower than the two shade cloth sites ($P > 0.5$). Average macrophyte cover at this site was 71%, as compared to 39% and 100% at the shade sites (SCNM and SCM respectively). Both positive and negative associations were found between the nutrient uptake and macrophyte cover at two of the sites during this trial (site with the riparian strip, and shade cloth site with no pre-existing macrophytes), although the relationship was not significant ($P>0.05$). Other studies have found that macrophyte growth in waterways plays a major role in reducing nutrient loads, and as such one should consider their presence a naturally occurring filter before entering the streams. The relationship between shading and its effect on nutrient uptake efficiency is unclear in this study, however, and more research is needed to assess the potential of macrophytes management to reduce nutrient flows in drains.

Drainage harvesting and reuse for irrigation

Drainage harvesting and reuse for irrigation offers an innovative way of capturing and reuse of nutrients from drain flows, reducing water quality impacts and benefiting plant growth. Four potential scenarios were investigated for the potential to harvest and recycle drainage water at Hyde Park, using both average drainage nitrate concentrations estimated by OVERSEER (29.9 g/m^3), and average drainage nitrate concentrations obtained from sampling on the case study farms (3.3 g/m^3). Drain nitrate concentrations are expected to vary spatially and effect on the performance and cost-effectiveness of drainage water harvesting and reuse.

The first scenario looked at the potential to irrigate onto an effluent irrigated block, where an irrigation system was already in place, using average nitrate-N concentrations (29.9 g/m^3) estimated by OVERSEER. A storage pond 69,000 m^3 in size was assumed, and the harvested drain water served to supplement the already existing irrigation. Under this scenario, capital costs would come to $602,000 and annual running costs would be $52,080. Additionally, 2,024 Kgs N would be harvested from the farm at a rate of $25.7 per Kg N attenuated/yr.

The second scenario explored the potential to irrigate onto the same block of land, but using average drainage nitrate-N concentrations measured in the field sampling. Furthermore, it was assumed that the storage pond was increased in size to meet the demands of irrigation without aid from the existing system. A storage pond of 120,000 m^3 was assumed, with the capital costs over 1 million dollars and annual running costs of about $89,990. It was found, however, that under such a scenario, only 398.5 Kg N would be harvested from the farm, and it would come at a rate of $225.8 per Kg N attenuated/yr. This scenario was ruled out as impractical.

The third scenario investigated the potential to irrigate onto a new smaller block of previously un-irrigated land. It was assumed that an irrigation system would be implemented, and only drain water harvested in the storage pond (69,000 m^3) would supply new irrigation systems installed. For this scenario, average drainage nitrate-N concentrations were estimated by OVERSEER. Under this scenario, capital costs would come to $602,000 and annual running costs would be $62,740. There would be an additional cost of $86,250 to set up an irrigation system, but $62,100 of that would be returned as extra income as a result of increased productivity by irrigating and fertilising the land. The capital cost plus irrigation system set up may seem cost prohibitive, but considering additional productivity benefits and when expressed as an annual cost per kg N attenuated ($0.34 per Kg N attenuated/yr), it appears to be almost negligible. A total of 1,886 Kgs N would be harvested and reuse on the farm.

The fourth scenario was almost identical to the third as describe above, except that the average nitrate-N concentrations were attained the field sampling. In this case, capital costs would remain approximately the same, as would the extra income. Annual running costs would come to $65,820, with most of the difference coming from increased costs of Urea. When expressed as an annual cost per Kg N attenuated, it comes to $40.8. However, only 91.1 Kg N would be harvested from the farm.

Ultimately, there were two main factors influencing costs-effectiveness of drainage harvesting and recycling; the concentration of nitrate-N in the drainage water, and whether the land is already irrigated or not. It is more cost effective per Kg N attenuated if there are higher concentrations of nitrate-N in the drains for which to be harvested and recycled, and if there is the possibility to irrigate it onto previously dry land. Given the low nitrate-N concentrations and existing irrigation, drainage harvesting and recycling would not realistically be economical on the case study farm. However, it should still be considered as a mitigation tool in areas with higher drain nitrogen-N concentrations, but site-specific conditions need to be pre-assessed for cost-effectiveness. For example, capital costs could be reduced on areas with undulating topography as the land would be naturally suited for the construction of storage ponds.

This study has clearly identified scope for potential drainage management practices. There are however, some areas that could be explored further to expand the research:

- A high-resolution profile of drainage flow and water quality could be carried out, particularly on different soils and landscapes, in order to better assess potential drainage management methods;

- Since there is potential for a naturally occurring filter, more in depth observations drain macrophytes and nutrient uptake, and their relationship with shade could be explored;
- A more detailed cost benefit analysis for harvesting and recycling of drainage water could be investigated in order to realistically and accurately quantify the costs of undertaking such a nutrient management method;
- Create a cost benefit analysis for other drainage management methods, such as wetlands or bioreactors, in order to make them more realistic and achievable to the landowner;
- Look into other potential scenarios for harvesting and recycling of drainage water, on different soil blocks on the case study farm, but also on different farms altogether, and;
- Investigate the potential for using this information to inform and alter nutrient and land management planning, and the validity of including information like this in a nutrient budgeting programme like OVERSEER®.

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Appendices

Appendices

Appendix 1

Table of buffer recommended riparian buffer widths by Regional Councils across New Zealand

Regional Council	Minimum Recommended Width	Source
Northland	5-10 m	(Northland Regional Council, 2005)
Auckland	10 m	(Auckland Regional Council, 2001)
Waikato	Under the regulatory approach; 3 m from the top of the bank. Under the incentive approach; beyond 3 m from the top of the bank i.e. 4-5 m	(Campbell, 2002)
Bay of Plenty	Depends on soil type, slope, and adjacent land use	(Bay of Plenty Regional Council, 2008)
Gisborne	5 m if the stream is under 2 m wide, 10 m if the stream is over 2 m, 20 if bordering a lake or marine system	(Gisborne Regional Council, 2015)
Hawke’s Bay	No specific recommendation	
Taranaki	5 m	(Tarankai Regional Council, 2009)
Manawatu-Whanganui	10 m	(Manawatu-Wanganui Regional Council, 2014)
Wellington	5 m	(Wellington regional Council, 2003)
Nelson	5-10 m in rural zones	(Nelson Regional Council, 2015)
Marlborough	For forestry areas, 5 m if stream is under 3 m wide, 10 m if the stream is equal to or over 3 m wide	(Marlborough District Council, 2015)
Canterbury	2-3 m	(Environment Canterbury, 2005)
West Coast	Irrespective of stream width, 3 m if bordering pasture and slope is under 12, 10m if slope is over 12. If bordering indigenous vegetation, 5 m if stream is under 3 m wide and slope is under 12. 10 m otherwise. If bordering lakes, 20 m.	(West Coast Regional Council, n.d.)
Otago	5 m riparian vegetative, or/including 1 m grass buffer	(Otago Regional Council, 2005)
Southland	The wider the better. However, Dairy NZ recommends 3-5 m	(MacGibbon, 2001)

Appendix 2

Average daily values (mm) for each month between June 2018 and May 2019 taken at the Raumai Climate Station (Horizons Regional Council, 2019c)

Year	Month	Rainfall (mm)
2018	June	2.31
2018	July	3.81
2018	August	3.87
2018	September	2.35
2018	October	1.61
2018	November	2.85
2018	December	3.37
2019	January	0.88
2019	February	1.50
2019	March	3.41
2019	April	4.32
2019	May	3.94

Appendix 3

Table showing data from Smith et al. (2017), collected from 8 drain sites on Hyde Park in 2016

	Count	Average (g/m^3)	Minimum (g/m^3)	Maximum (g/m^3)	Standard Deviation	Coefficient of Variance
Ammonia-N	32	0.20	0.06	0.62	0.14	69.6%
Total Nitrogen	32	1.33	0.69	2.48	0.46	34.2%
Nitrate-N	32	0.89	0.12	3.15	0.74	83.1%
DRP	32	0.01	0.01	0.01	0.00	0.0%

Appendix 4

Table showing monthly summary of discharges (l/s) from the four drains on Hyde Park between June 2018 and February 2019

Month	Count	Average (l/s)	Minimum (l/s)	Maximum (l/s)	Standard Deviation	Coefficient of Variance
June	6	13.13	0.59	35.49	17.02	129.7%
July	7	18.38	2.02	36.97	15.07	82.0%
August	29	21.69	4.11	61.95	16.82	77.5%
September	21	14.43	2.60	48.79	14.81	102.7%
October	15	8.07	1.48	18.13	6.39	79.2%
November	17	11.56	1.46	35.11	10.96	94.9%
December	7	7.18	0.00	17.06	6.78	94.4%
January	12	5.39	0.00	28.48	7.95	147.5%
February	11	3.26	0.13	13.54	4.52	138.5%

Appendix 5

Table summarising different water quality values for each site during the Swap Trial (December 2017-March 2018)

	Bromide	Chloride	Fluoride	Ammonia-N	Total Nitrogen	Nitrite-N	Nitrate-N	DRP	Total Phosphorus	Sulphate
<u>MD</u>										
Count	12	12	12	11	12	2	12	12	12	12
Average	0.12	36.87	0.21	1.1	2.97	0.01	0.06	0.03	0.11	1.48
Minimum	0.1	33.14	0.14	0.06	0.54	0.01	0.06	0.01	0.01	0.06
Maximum	0.13	39.75	0.47	11.24	10.12	0.01	0.06	0.04	0.79	2.58
Standard Deviation	0.01	2.18	0.09	3.36	3.84	0	0	0.01	0.22	0.75
Coefficient of Variance	9.00%	5.90%	44.50%	307.20%	129.20%	9.00%	0.00%	24.40%	191.30%	50.70%
<u>U/S</u>										
Count	7	7	7	7	6	2	7	7	7	7
Average	0.09	39.22	0.17	0.84	0.59	0.01	0.13	0.03	0.04	6.46
Minimum	0.05	16.71	0.1	0.06	0.06	0.01	0.06	0.02	0.01	3.96
Maximum	0.11	48.54	0.31	4.41	0.75	0.01	0.35	0.03	0.09	7.41
Standard Deviation	0.02	10.85	0.07	1.63	0.26	0	0.12	0	0.03	1.2
Coefficient of Variance	21.60%	27.70%	42.00%	193.10%	44.80%	23.80%	91.40%	11.10%	69.60%	18.60%
<u>GW</u>										
Count	3	3	3	3	3	1	3	3	3	3
Average	0.05	12.28	0.21	0.06	0.56	0.04	0.12	0.03	0.05	5.07
Minimum	0.04	10.91	0.13	0.06	0.48	0.04	0.06	0.01	0.02	4.79
Maximum	0.06	14.72	0.26	0.06	0.65	0.04	0.23	0.04	0.08	5.3
Standard Deviation	0.01	2.11	0.07	0	0.08	0	0.1	0.01	0.03	0.26
Coefficient of Variance	21.90%	17.20%	32.90%	0.00%	14.80%	0	81.50%	44.20%	55.20%	5.10%
<u>Pre Swap D/S</u>										
Count	8	8	8	8	8	5	8	8	8	8
Average	0.1	43.37	0.16	0.16	1.57	0.01	0.06	0.03	0.07	5.42
Minimum	0.08	36.35	0.1	0.06	0.57	0	0.06	0.01	0.01	0.41

Maximum	0.13	48.05	0.31	0.68	5.25	0.02	0.06	0.04	0.19	8.17
Standard Deviation	0.01	4.14	0.07	0.22	1.62	0	0	0.01	0.06	2.64
Coefficient of Variance	13.20%	9.50%	44.90%	140.20%	102.80%	61.80%	0.00%	32.30%	92.00%	48.80%
Post Swap D/S										
Count	5	5	5	4	5	2	5	5	5	5
Average	0.06	19.81	0.16	0.11	0.67	0.03	0.08	0.03	0.07	5.46
Minimum	0.05	12.83	0.12	0.06	0.48	0.01	0.06	0.02	0.01	4.86
Maximum	0.08	32.15	0.18	0.27	0.83	0.05	0.15	0.03	0.12	6.42
Standard Deviation	0.01	8.2	0.02	0.1	0.14	0.03	0.04	0	0.04	0.57
Coefficient of Variance	22.10%	41.40%	13.70%	90.90%	20.90%	109.40%	50.50%	13.60%	57.10%	10.50%
Pre Swap D/S 2										
Count	2	2	2	2	2	1	2	2	2	2
Average	0.11	43.15	0.1	0.06	1.04	0.03	0.2	0.03	0.08	6.51
Minimum	0.12	49.04	0.09	0.06	1.05	0.03	0.4	0.03	0.08	7.45
Maximum	0.12	49.04	0.11	0.06	1.05	0.03	0.4	0.03	0.08	7.45
Standard Deviation	0.02	8.34	0.01	0	0.01	0	0.11	0	0	1.33
Coefficient of Variance	22.10%	19.30%	12.90%	0.00%	1.40%	0	54.60%	10.90%	4.80%	20.50%
Post Swap D/S 2										
Count	5	5	5	4	4	4	5	5	5	5
Average	0.06	19.84	0.17	0.2	0.81	0.01	0.35	0.02	0.05	6.34
Minimum	0.05	14.34	0.15	0.06	0.56	0	0.17	0.01	0.02	5.63
Maximum	0.07	28.63	0.19	0.59	1.12	0.02	0.7	0.03	0.08	7.57
Standard Deviation	0.01	5.31	0.02	0.27	0.26	0.01	0.21	0.01	0.02	0.74
Coefficient of Variance	12.80%	26.80%	9.10%	136.00%	31.70%	52.90%	60.40%	37.70%	43.20%	11.70%
Pre Swap OL										
Count	8	8	8	8	8	3	8	8	8	8
Average	0.09	41.76	0.18	0.08	0.75	0.01	0.18	0.03	0.03	8.49
Minimum	0.08	32.5	0.09	0.06	0.52	0	0.06	0.01	0.01	5.82
Maximum	0.11	47.42	0.59	0.16	0.99	0.03	0.54	0.03	0.06	10.46

Standard Deviation	0.01	4.51	0.17	0.04	0.15	0.01	0.17	0.01	0.01	1.46
Coefficient of Variance	10.40%	10.80%	90.20%	48.10%	20.70%	110.70%	94.80%	24.30%	50.20%	17.20%
Post Swap OL										
Count	5	5	5	4	5	0	5	5	5	5
Average	0.07	26.58	0.16	2.18	0.45	0	0.1	0.02	0.03	8.78
Minimum	0.06	24.52	0.14	0.06	0.29	0	0.06	0.01	0.02	7.98
Maximum	0.07	32.96	0.19	8.54	0.75	0	0.23	0.03	0.04	9.2
Standard Deviation	0.01	3.6	0.02	4.24	0.17	0	0.08	0.01	0.01	0.48
Coefficient of Variance	7.70%	13.50%	11.10%	194.30%	38.50%	0	79.40%	27.30%	23.20%	5.40%

Appendix 6

Table showing the volume of drain water abstracted daily from the upstream dam at the Swap Site, and put back into the downstream site as groundwater during the swap trial (February-March 2018).

Date	Flow (l/s)
2/02/2018	5
3/02/2018	4
4/02/2018	3
5/02/2018	3
6/02/2018	3
7/02/2018	3.2
8/02/2018	3.2
9/02/2018	4
10/02/2018	4
11/02/2018	5
12/02/2018	5
13/02/2018	2.5
14/02/2018	5.5
15/02/2018	4.5
16/02/2018	3.5
17/02/2018	4
18/02/2018	3.5
19/02/2018	3.5
20/02/2018	4
21/02/2018	
22/02/2018	5
23/02/2018	4.5
24/02/2018	3
25/02/2018	2.5
26/02/2018	3
27/02/2018	5
28/02/2018	5
1/03/2018	4.8
2/03/2018	3.5
3/03/2018	3.2
4/03/2018	3.5
5/03/2018	3.3
6/03/2018	3.3
7/03/2018	5.5
8/03/2018	5.5
Average	3.9

Appendix 7

Table summarising different water quality values for each site during the Macrophyte Management Trial

	Nutrients (g/m^3)										% Cover	
	Bromide	Chloride	Fluoride	Ammonia-N	Total Nitrogen	Nitrite-N	Nitrate-N	DRP	Total Phosphorus	Sulphate	Macrophyte	Periphyton
OL												
Count	13	13	13	13	13	3	13	13	13	13	13	13
Average	0.08	35.92	0.17	0.79	0.63	0.01	0.13	0.03	0.03	8.6	71.22	5.88
Minimum	0.06	24.52	0.09	0.06	0.29	0	0.02	0.01	0.01	5.82	27.8	0
Maximum	0.11	47.42	0.59	8.54	0.99	0.03	0.54	0.03	0.06	10.46	100	20
Standard Deviation	0.02	8.68	0.13	2.34	0.21	0.01	0.15	0.01	0.01	1.16	25.02	9.19
Coefficient of Variance	19.00%	24.20%	73.10%	295.90%	33.80%	110.70%	111.60%	25.00%	48.00%	13.50%	35.10%	156.40%
SCM												
Count	7	7	7	7	7	1	7	7	7	7	7	7
Average	0.08	31.99	0.14	0.46	0.99	0.04	0.53	0.02	0.01	7.46	100	0
Minimum	0.06	21.97	0.09	0.06	0.84	0.04	0.04	0.01	0.01	5.76	100	0
Maximum	0.11	44.72	0.17	2.34	1.21	0.04	0.84	0.03	0.03	8.68	100	0
Standard Deviation	0.02	9.79	0.03	0.85	0.12	0	0.29	0.01	0.01	1.02	0	0
Coefficient of Variance	26.70%	30.60%	19.40%	184.70%	12.60%	0	54.60%	25.80%	36.60%	13.70%	0.00%	#DIV/0!
SCNM												
Count	7	7	7	7	6	1	7	7	7	7	7	7
Average	0.07	28.07	0.19	0.8	1	0	0.63	0.02	0.02	7.75	39.24	24.93

Minimum	0.06	20.32	0.12	0.06	0.87	0	0.32	0.01	0.01	5.71	7.6	9.2
Maximum	0.1	44.09	0.42	4.75	1.13	0	0.88	0.03	0.06	8.78	73	37.8
Standard Deviation	0.02	9.22	0.1	1.75	0.11	0	0.21	0.01	0.02	0.96	22.03	10.61
Coefficient of Variance	22.90%	32.90%	55.30%	218.80%	10.60%	0	33.80%	38.30 %	81.00%	12.40%	56.10%	42.60%