The Ability of Detainment Bunds to Mitigate the Impact of Pastoral Agriculture on Surface Water Quality in the Lake Rotorua Catchment

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ABSTRACT

Identifying and implementing cost-effective mitigation strategies are necessary to achieve reductions in the anthropogenic phosphorus (P) and nitrogen (N) loads that contribute to eutrophication and toxic algal blooms in Lake Rotorua, in the Bay of Plenty Region on the North Island of New Zealand. Storm generated surface runoff from grazed pastures, that cover ~48% of Lake Rotorua's catchment, contribute 67% of the total N (TN) and 43% of the total P (TP) loads delivered from the catchment to the lake. Detainment bunds (DBs) are a novel mitigation strategy targeted at decreasing nutrient and sediment losses by impeding and temporarily ponding stormflows for up to 3 days. A DB is an earthen, stormwater retention structure, approximately 1.5-2 m high and 20-80 m long, constructed on pastures across the flow path of targeted low-order ephemeral streams.

Two DBs on pastures in the Lake Rotorua catchment, with 20 and 55 ha catchments, were monitored over 12 months. Nearly 20 storm events resulted in ponding at each site. Detailed hydrological analyses were conducted for each storm in order to establish water balances, as well as to analyse contaminate loads delivered to, and discharged from the DBs. Surface runoff flows were measured, and samples were collected, to determine the DB mitigation performance and to identify the processes affecting the outcomes. The DBs prevented an estimated 51-59% of the annual suspended sediment loads, 47-68% of the annual TP loads, and 57-72% of the annual TN loads delivered to the DBs in runoff, from reaching the lake. An estimated 43-63% of the annual surface runoff delivered to the DBs infiltrated the soil, as a result of increased residence times of surface runoff on well-drained pasture soils. Soil infiltration was mainly responsible for decreased contaminant loads delivered to surface waters downstream of the bunds, while sorption and sedimentation also contributed to some load reductions. The inability to impound only portions of the runoff generated during rare, high magnitude storm events limited the performance of DBs. Furthermore, declining soil infiltration rates and increasing soil P concentrations in the ponding areas could affect the longer-term performance of DBs. A cost: benefit analysis of the DB strategy was conducted in order to compare the cost-effectiveness of DBs to other nutrient migration strategies, with results demonstrating that the DB strategy is a highly cost-effective edge of field mitigation option available to pastoral farmers in the Lake Rotorua catchment.

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RELEVANT PUBLICATIONS AND PRESENTATIONS

Publications to date

Peer-reviewed journal articles

Levine, B., Burkitt, L., Horne, D., Condron, L., Tanner, C., & Paterson, J. (2019). Preliminary assessment of the ability of detainment bunds to attenuate sediment and phosphorus transported by surface runoff in the Lake Rotorua catchment. *Animal Production Science, 60*, 154-158. doi:10.1071/AN18544

Chapter 3

Levine, B., Horne, D., Burkitt, L., Tanner, C., Sukias, J., Condron, L., & Paterson, J. The ability of detainment bunds to decrease surface runoff leaving pastoral catchments: Investigating a novel approach to agricultural stormwater management. Accepted by the journal Agricultural Water Management.

Thesis chapters composed of submitted manuscripts currently under review for publication in peer-reviewed journals

Chapter 4

Levine, B., Burkitt, L., Horne, D., Tanner, C., Sukias, J., Condron, L., & Paterson, J. The ability of detainment bunds to decrease sediment loss from pastoral catchments in surface runoff: Investigating a novel stormwater mitigation strategy. Submitted to: Hydrological Processes.

Chapter 5

Levine, B., Burkitt, L., Horne, D., Tanner, C., Sukias, J., Condron, L., & Paterson, J. The ability of detainment bunds to reduce phosphorus losses in surface runoff from pastoral catchments: A novel strategy for the nutrient mitigation toolbox. Submitted to: Ecological Engineering.

Chapter 6

Levine, B., Horne, D., Burkitt, L., Tanner, C., Sukias, J., Condron, L., & Paterson, J. Effect of impeding stormflows with detainment bunds on nitrogen transport in surface runoff from pastoral catchments. Submitted to: Soil Research.

Conference/workshop proceedings

Levine, B.M., Burkitt, L., Horne, D., Tanner, C., Condron, L., and Paterson, J. (2019). Phosphorus Mitigation Project: Quantifying the ability of detainment bunds to attenuate phosphorus and sediment loads in surface runoff from grazed pasture in the Lake Rotorua catchment. In: Nutrient Loss Mitigations for Compliance in Agriculture. Occasional Report No. 32, 7p. (Eds L. D. Currie and C. L. Christensen). Palmerston North, New Zealand: Fertiliser and Lime Research Centre, Massey University.

Levine, B.M., Burkitt, L., Horne, D., Tanner, C., Condron, L., and Paterson, J. (2018). Phosphorus Mitigation Project: Preliminary data investigating nutrient attenuation using detainment bunds in the Lake Rotorua catchment. Rotorua, New Zealand: NZ Land Treatment Collective Conference.

Levine, B.M., Burkitt, L., Horne, D., Tanner, C., Condron, L., and Paterson, J. (2017). Phosphorus Mitigation Project: Mitigation of surface P runoff using detainment bunds. In: Science and policy: nutrient management challenges for the next generation. Occasional Report No. 30, 7p. (Eds L. D. Currie and M. J. Hedley). Palmerston North, New Zealand: Fertiliser and Lime Research Centre, Massey University.

Presentations

Levine, B.M., Burkitt, L., Horne, D., Tanner, C., Condron, L., and Paterson, J. (2020) "Phosphorus Mitigation Project: Quantifying the ability of Detainment Bunds to mitigate nutrient losses from pastoral agriculture in the Lake Rotorua watershed." Presented at the Farmed Landscapes Research Centre Workshop. February 2020, Palmerston North, New Zealand

Levine, B.M., Burkitt, L., Horne, D., Tanner, C., Condron, L., and Paterson, J. (2019). "Quantifying the Ability of Detainment Bunds to Attenuate Sediments and Nutrients by Temporarily Ponding Surface Runoff in the Lake Rotorua Watershed, New Zealand." Presented at the Agronomy Science of America International Annual Meeting. November, 2019, San Antonio, Texas.

Levine, B.M., Burkitt, L., Horne, D., Tanner, C., Condron, L., and Paterson, J. (2019). "Quantifying the ability of detainment bunds to attenuate phosphorus and sediment loads in surface runoff from grazed pasture in the Lake Rotorua catchment, New Zealand." Presented at the Land Use and Water Quality Conference. June 2019, Aarhus, Denmark.

Levine, B.M., Burkitt, L., Horne, D., Tanner, C., Condron, L., and Paterson, J. (2019). "Phosphorus Mitigation Project: Quantifying the ability of detainment bunds to attenuate phosphorus and sediment loads in surface runoff from grazed pasture in the Lake Rotorua catchment." Presented at the Fertiliser and Lime Research Centre Workshop. February 2019, Palmerston North, New Zealand.

Levine, B.M., Burkitt, L., Horne, D., Tanner, C., Condron, L., and Paterson, J. (2018). Phosphorus Mitigation Project: Preliminary data investigating nutrient attenuation using Detainment Bunds in the Lake Rotorua catchment." Presented at the NZ Land Treatment Collective Conference, March 2018, Rotorua, New Zealand.

Levine, B.M., Burkitt, L., Horne, D., Tanner, C., Condron, L., and Paterson, J. (2017). "Phosphorus Mitigation Project: Detainment Bunds as a strategy to reduce P transfer in surface runoff from farms." Presented at the Waikato-Bay of Plenty Soils Conference, December 2017, Hamilton, New Zealand

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LIST OF ABBREVIATIONS

A	Awahou
ASC	Anion storage capacity
CEC	Cation exchange capacity
CSA	Critical source area
DB	Detainment bund
DP	Dissolved phosphorus
DIN	Dissolved inorganic nitrogen
DRP	Dissolved reactive phosphorus
EDI	Effective depth of interaction
H	Hauraki
M	Bay of Plenty Regional Council climate monitoring site at Oturoa Road
n	Number of observations
N	Nitrogen
NH4+	Ammonium
NO3-	Nitrate
NZD	New Zealand dollar
P	Phosphorus
PP	Particulate phosphorus
RMA	Resource Management Act
SDA	Stormwater detention areas
SS	Suspended sediment
TLI	Trophic level index

CHAPTER 1: Introduction

1.1 Background and rationale of the study

Lake Rotorua, in the Bay of Plenty Region on the North Island of New Zealand, is recognised as a 'taonga', or treasured natural resource, and provides valuable ecosystem services (Land and Water Forum, 2010). Anthropogenic nutrient loading has caused ecological degradation, eutrophication and toxic algal blooms in the lake (Environment Bay of Plenty, 2009). Targets have been set to reduce nutrient loading from the catchment in order to improve lake water quality (Bay of Plenty Regional Council, 2012). Achieving load reduction targets may become more difficult with time as climate change contributes to warmer conditions increasing internal nutrient loading in the lake (Burns, 2001; Burns et al., 2005), and causes more dramatic hydrologic conditions that exacerbate erosion and nutrient losses in runoff from the catchment (Ministry for the Envrironment, 2019; Ockenden et al., 2016).

Pastoral agriculture is commonly associated with eutrophication and the deterioration of freshwater ecosystems in New Zealand, due to nutrient losses in runoff resulting from interactions between land management, landscape features and precipitation patterns (Verburg et al., 2010). Pastoral dairy and drystock farms cover ~48% of Lake Rotorua's 42,000 ha surface area catchment (Fig. 1.1), and contribute 67% of the total nitrogen (TN), and 43% of the total phosphorus (TP) loading from the catchment (Bay of Plenty Regional Council, 2012). Storm generated surface runoff leaving grazed pastures in the catchment is responsible for a significant portion of the annual nutrient loads delivered to Lake Rotorua (Environment Bay of Plenty, 2009).

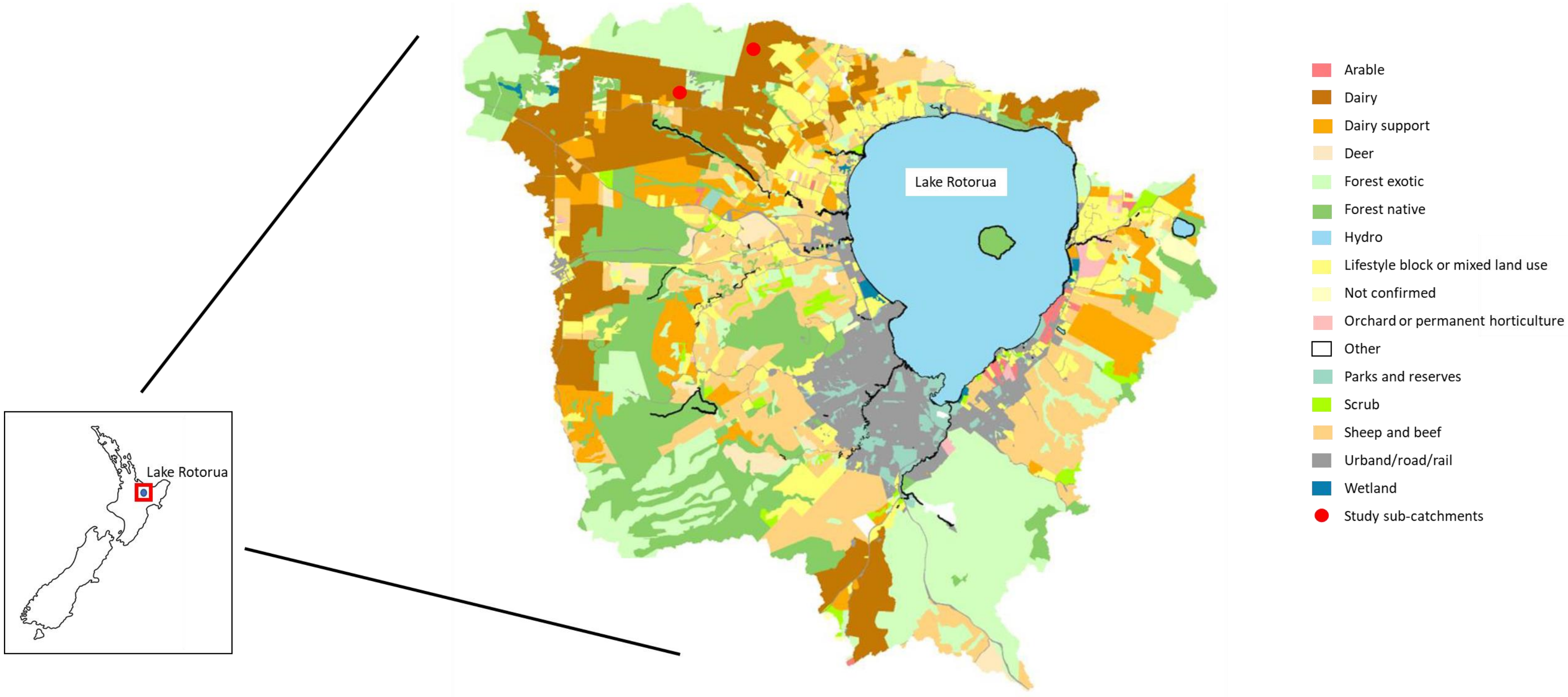

Figure 1.1: Map of land use within the Lake Rotorua catchment with study sub-catchments marked and an inset of map of New Zealand, adapted from McBride et al. (2018).

Detainment bunds (DBs) are a mitigation strategy targeted at reducing nutrient losses from pastures in the Lake Rotorua catchment by increasing surface runoff residence times by impeding stormflow and temporarily ponding water. A DB is an earthen, stormwater retention structure, approximately 1.5-2 m high and 20-80 m long, constructed on pastures across the flow path of targeted low-order ephemeral streams. A preliminary study, which served as a proof-of-concept for the strategy, found that DBs facilitated sedimentation and retained P enriched sediments (Clarke, 2013). Prior to the research reported in this thesis, there was no definitive quantification of the impact of the DB strategy on annual sediment and nutrient losses from pastures in the Lake Rotorua catchment.

Identifying and implementing cost-effective mitigation strategies that address nutrient losses is becoming more salient as pastoral land use in the Lake Rotorua catchment intensifies and climate change contributes to conditions that are conducive to greater nutrient loading (Edgar, 2008; Ministry for Primary Industries, 2016). Gathering scientific evidence to determine the efficacy of a mitigation strategy, and generating accurate cost: benefit analyses, help overcome challenges related to implementing appropriate mitigation strategies (Bieroza et al., 2019). This process is particularly important for novel strategies such as DBs, which require demonstration at the field scale.

1.2 Research hypotheses and objectives

1.2.1 Research hypotheses

Since contaminant loads in surface runoff are determined by the volume of runoff and the concentrations of contaminants, it was hypothesised that the DBs' ability to effectively mitigate nutrient losses from the DB catchments would be affected by multiple processes. First, increasing the residence time of surface runoff, by impeding stormflow on well-drained soils prevalent in pastures in the Lake Rotorua catchment, would facilitate significant soil infiltration, and therefore decrease runoff volumes and dissolved nutrient loads discharged from the DB catchments. Secondly, impeding stormflow would reduce the kinetic energy of flowing water, causing sediment deposition in the ponding area and a decrease in the concentration of sediment-bound nutrients. Lastly, increasing the residence times of runoff could allow greater time for chemical processes such as sorption to occur, that would decrease the concentration of dissolved nutrients such as dissolved reactive P, and ammonium.

1.2.2 Research objectives

The research reported in this thesis aimed to provide insight into the DB strategy's function and viability as an option available to pastoral farmers attempting to mitigate nutrient losses that contribute to eutrophication in Lake Rotorua. Achieving the research objective required field monitoring and sample collection at 2 DBs located on pastures in the Lake Rotorua catchment for one calendar year. Specific objectives were:

a) To quantify the volume of runoff delivered to DBs and characterise the size of runoff events.
b) To identify the fate of water delivered to DBs, and the factors impacting the ability of DBs to reduce the volume of surface runoff.
c) To quantify the ability of DBs to reduce sediment, P and N loads transported in surface runoff from pastures in the Lake Rotorua catchment.
d) To identify the factors affecting the ability of DBs to decrease sediment, P and N loads from being delivered to Lake Rotorua.
e) To perform a cost: benefit analysis on the DB strategy to compare the cost-effectiveness of the various nutrient mitigation strategies that target surface runoff available to pastoral farmers in the Lake Rotorua catchment.
f) To identify approaches to maintain and improve DB mitigation performance.
g) To assess the applicability of the DB strategy beyond the Lake Rotorua catchment.

A detailed hydrological analysis was carried out for every storm-generated runoff event that took place during the year-long study period at the 2 DB sites. Water balances were calculated and used along with runoff sample analyses, to determine contaminate loads delivered to, and discharged from the DBs for each storm event. The data from each event was used to quantify the strategy's cumulative effect on annual loads of suspended sediments, phosphorus, and nitrogen, generated and discharged from the DB catchments, and analysed to identify the processes affecting treatment efficiency. By quantifying the effectiveness of 2 DBs with varying characteristics, and analysing the data to identify the mechanisms contributing to DB performance, this research built on the understanding of DB function, and developed recommendations for maintaining and enhancing DB performance, as well as provided insight into the utility of DBs beyond the Lake Rotorua catchment.

1.3 Thesis outline

The review of the literature presented in Chapter 2 supports the motivation behind the investigations carried out and presented in this thesis and provides a foundation for the discussion of the study results presented in Chapters 3 to 7. Topics in the literature review include New Zealand's lake water quality issues (Section 2.1), P and N mobilisation and transport in pastoral agricultural systems (Section 2.2), and nutrient mitigation strategies in the Lake Rotorua catchment (Section 2.3)

Chapters 3 to 6 were written as stand-alone chapters that addressed the specific objectives of this research, and each chapter was submitted for publication in peer-reviewed journals. The studies presented in Chapters 3 to 6 collected data and samples during the same ~20 runoff events that occurred during the 12-month study. Chapter 3 describes a study investigating the effect of DBs on hydrology by measuring surface runoff volumes that were delivered to the DB, infiltrated the soil as a result of impeding stormflow, and discharged from the DBs. A detailed analysis of the processes affecting the proportion of runoff delivered to the DB being discharged were also explored and discussed in Chapter 3.

The detailed hydrological analysis presented in Chapter 3 was heavily relied upon to calculate contaminant inflow and discharge loads that were reported in Chapters 4, 5 and 6, as well as during the discussion of factors affecting DB treatment efficiencies. Chapter 4 describes an investigation into the DB strategy's ability to facilitate sedimentation and quantified the effect of DBs on sediment loads discharged from the DB catchments. Chapters 5 and 6 describe studies that investigated the ability of DBs to decrease P and N loads discharged from the study sub-catchments, respectively, and quantified the loads of P and N prevented from reaching Lake Rotorua as a result of the DB treatment. In addition to quantifying the contaminant load treatment efficiencies of the DBs, Chapters 4 to 6 discuss the process driving DB performance based on the similarities and differences observed in the data between the 2 sites. Each sample collected during the runoff events was analysed for each of the contaminants collected. Therefore the 'Materials and methods' section in Chapters 4, 5 and 6 report the same sample collection procedure, but describe the different sample analysis procedures for the respective contaminants investigated in each chapter. Chapter 7 presents key findings and conclusions from this research and includes a cost: benefit analysis and recommendations for areas of further research on DBs.

CHAPTER 2: Literature Review

2.1 Freshwater resources in New Zealand

Freshwater has long been a valued resource in New Zealand. The Maori ('first nation') people of Aotearoa/New Zealand recognise freshwater as a taonga (treasure), and have traditional obligations to protect freshwater so as to "leave a worthy inheritance for future generations" (Land and Water Forum, 2010). Coupled with Maori tradition, there is a high level of concern for water quality in the general public, partly due to the population's reliance on rivers and lakes for economic, social and cultural well-being (Larned et al., 2016). A recent nation-wide public opinion poll found that water pollution is New Zealand's number one concern (Cosgrove, 2019) .

Government policies aim to protect and improve the health of rivers and lakes that provide valuable ecological services. The Resource Management Act of 1991 (RMA) is the primary piece of legislation for managing air, soil, freshwater and coastal marine areas based on the principle of sustainable management, while recognising the Treaty of Waitangi in decision making (Ministry for the Envrionment, 2017). Under the RMA, the National Policy Statement for Freshwater Management of 2011 requires regional councils to set objectives to "maintain or improve the overall quality of freshwater within a region." In efforts to achieve these objectives, regional councils are required to sustainably manage land use and development in order to safeguard freshwater ecosystems (Ministry for the Envrionment, 2011).

Although policies aimed at protecting New Zealand's water resources have been established, recent studies have found excess nutrient loads causing eutrophication in many freshwater systems. A 2015 study of 77 river sites showed that 49% of monitored sites had enough nitrogen (N), and 32% had enough dissolved phosphorus (P), to trigger nuisance periphyton growth (Larned et al., 2015). Another study analysing data from 2005-2009 concluded that 44% of the 112 lakes assessed were eutrophic or worse, and estimated that 32% of all 3820 New Zealand lakes >1 ha in area were eutrophic or worse, and that water quality was deteriorating in twice as many lakes as there were improving.

2.1.1 Lake Rotorua

Lake Rotorua, is located in the Bay of Plenty Region, on the North Island of New Zealand. The Lake Rotorua basin was formed 220-230 thousand years ago as a result of a rhyolitic eruption (Wood, 1992). The polymictic lake has a surface area of 81 km^2 and average depth of 11 m (Burger et al., 2007). Lake Rotorua has a topographical surface water catchment of 502.1 km^2, and total groundwater catchment of 537.9 km^2 (White et al., 2014). Roughly two-thirds of the inflow to the lake comes from 9 major streams, with the remainder coming from smaller streams, springs, groundwater upwelling to the lake bed, and direct rainfall to the lake (Dare, 2018). Lake Rotorua provides valuable ecosystem services, including cultural significance to the Maori people, and contributes substantially to the New Zealand tourism industry, with a world-renowned trout fishery and other recreational opportunities (Burns et al., 2005).

2.1.1.1 Lake Rotorua water quality

Water quality impairment due to excess nutrients entering Lake Rotorua has been recognised since the 1960s (Edgar, 2008). Land use intensification and population increases in the catchment over the past 60 years have contributed to increased P and N inputs causing ecological degradation, eutrophication and toxic algal blooms in the lake (Environment Bay of Plenty, 2009). The lake experienced severe cyanobacteria Dolichospermum (formerly known as Anabaena) sp. and Microcystis sp. blooms in the early 2000's (McBride et al., 2018).

Studies have found temporal and spatial variability of P and N limiting phytoplankton growth in the lake (McBride et al., 2018). Concentrations above 0.013 g total P (TP) m^{-3} and 0.11 g total N (TN) m^{-3} are considered to stress lakes such as Lake Rotorua in New Zealand (ANZG, 2018). Median concentrations in Lake Rotorua were 0.02 g TP m^{-3} and 0.30 g TN m^{-3} from 2013-2017 (Stats NZ, 2019). Therefore, Lake Rotorua is considered eutrophic despite nutrient concentration decreases contributing to improved water quality since 2002, with fewer algae blooms and increased water clarity (Hamill, 2018). Various interventions have contributed to improved water quality trends including ceasing to discharge municipal wastewater into the lake, regional rules to cap land-based inputs, reticulation of sewage from smaller communities, N removal from water delivered from Tikitere geothermal field, and alum dosing in the Utuhina and Puarenga streams to remove biologically available P (Stephens et al., 2018).

Many challenges exist to limit nutrient loading from internal and external sources and continue improving trends in order to achieve Lake Rotorua water quality objectives (Stephens et al., 2018). Climate change is contributing to conditions that make improving lake water quality more difficult (Donald et al., 2019) including wetter winters, hotter and drier summers, and greater storm intensities in this region (Ministry for the

Envrironment, 2019). These more dramatic hydrologic conditions have the potential to overwhelm land management strategies and exacerbate nutrient and sediment losses, particularly from agricultural headwater catchments (Kleinman et al., 2006; McDowell & Sharpley, 2002; Ockenden et al., 2017). Increases in runoff magnitudes and flashiness of flows caused by climate change are also likely to decrease the natural processing of N in low order streams, and lead to greater N loading to Lake Rotorua (Alexander et al., 2007). Also, internal P and N loading from lake sediments has been shown to increase during warmer months (Burns et al., 2005).

Another longer-term concern for nutrients entering Lake Rotorua is the very slow and lagged response of streams and the lake to anthropogenic contamination from groundwater inputs (Morgenstern et al., 2015). The mean residence time for groundwater feeding into streams range from 30-145 years in the catchment, implying that the majority of the nitrate load discharged into the lake is from land use activities taking place at least 30 years ago (Morgenstern et al., 2015). Anthropogenic nitrate loads into the lake from groundwater are expected to increase well into the future due to recent dairy conversions and intensified N fertiliser applications (Morgenstern et al., 2015). While Morgenstern et al. (2015) discusses that dissolved P is efficiently retained by the ashfall soils in the catchment with thick unsaturated zones that readily sorb P, the study makes no estimate as to when soils could become saturated and anthropogenic P starts reaching the groundwater.

2.1.1.2 Lake Rotorua water quality objectives

At the time of writing, the Bay of Plenty Regional Council is in the process of approving Plan Change 10 to the Regional Water and Land Plan, which sets rules for Lake Rotorua nutrient management (Hamill, 2018). Currently, the target trophic level index (TLI) stated for acceptable lake health in the Lake Rotorua Action Plan is 4.2 (Environment Bay of Plenty, 2009). The TLI is an annual average value determined by water quality variables and equivalents to trophic levels, with lower TLI values corresponding to less eutrophic conditions (Burns et al., 2000). Table 2.1, and the equations below, give an explanation of how TLI's are calculated for eutrophic conditions.

Table 2.1: Measurements used to determine the trophic level index values of bodies of water considered eutrophic

Lake type	Trophic level	Chlorophyll-a	Secchi Depth	TP	TN
		(mg m^{-3})	(m)	(mg P m^{-3})	(mg N m^{-3})
Eutrophic	4.0 to 5.0	5.0 – 12	2.8 – 1.1	20 – 43	213 - 458

Trophic level values are calculated for each of the key variables (TLx) using the equations below for each year and the annual value for each variable:

TLc = 2.22 + 2.54 log (Chlorophyll-a)

TLs = 5.56 + 2.60 log((1/Secchi Depth) – (1/40))

TLp = 0.218 + 2.92 log(TP)

TLn = -3.61 + 3.01 log(TN)

The TLI and its standard error is then calculated for each year using the equation below:

TLI = 0.25(TLc + TLs + TLp + TLn)

(Burns et al., 2000)

When establishing plans to reach TLI targets, internal nutrient loads contributed 360 t N y^{-1} and 36 t P y^{-1} to the lake, while 556 t N y^{-1} and 39 t P y^{-1} were delivered from the lake catchment (Environment Bay of Plenty, 2009). One recent study using streamflow data calculated 23 t P y^{-1} of the 49 t P y^{-1} delivered to the lake was anthropogenically sourced (Tempero et al., 2015). Another study, using data from Tempero et al. (2015) and other data sources, estimated the annual P load to be 46 t P y^{-1}, with 18 to 27 t P y^{-1} coming from anthropogenic sources (Hamill, 2018). Hamill (2018) also calculated the average TP load from 2007-2014 to be 42 t P y^{-1}, with anthropogenic sources contributing 17 to 19 t P y^{-1}.

Trends for the key variables determining the TLI have shown improvements since 2001, and the 4.2 TLI target level has been frequently reached since 2012 (Stephens et al., 2018). To maintain TLI targets, total lake inputs would need to be 37 t P y^{-1}, and 435 t N y^{-1} (Environment Bay of Plenty, 2009). External input targets for 2029 were set at 29 t P y^{-1}, and 386 t N y^{-1} (Environment Bay of Plenty, 2009). To achieve these goals, models have estimated anthropogenic TP loading would need to be reduced to 8–13 t P y^{-1} (Hamilton et al., 2015).

2.2 Anthropogenic nutrient loading to freshwaters

2.2.1 Internal loading

Anthropogenic nutrient loads from several decades of inputs from historical town sewage and agriculture are stored in Lake Rotorua sediments (Environment Bay of Plenty, 2009). Physical, chemical and biological processes taking place within the lake sediments throughout the year can result in the release of a significant portion of annual P and N loads to the lake (Burger et al., 2007; Burns et al., 2005; Environment Bay of Plenty, 2009). For instance, P released from sediments in Lake Rotorua account for roughly half of the total annual P loads (Bay of Plenty Regional Council, 2012), while warm periods causing more anoxic conditions can lead to significantly greater contributions to annual P loads by lake sediments (Burns et al., 2005). Also, N in the form of ammonia may be released from sediments under less severe anoxic conditions than P (Burger et al., 2007; Burns, 2001). Also, particulate organic matter decomposition releases biologically available P and dissolved inorganic N, mostly as ammonium (NH_4^+), which can oxidise to nitrate (NO_3^-) (Burger et al., 2007; McDowell et al., 2013).

2.2.2 Land use

Numerous studies have acknowledged that aspects of land use activities and agricultural management strategies, including nutrient, crop, livestock and soil management, interact with biophysical factors such as soil, rainfall and topography, to contribute and control nutrient losses from soils to surface runoff (Buda et al., 2009; McDowell et al., 2002; Sharpley et al., 2001; Withers & Jarvie, 2008). In intensive agriculture, plant available nutrients are lost from the system through nutrient cycling, retention in soils, product export, animal transfer and via runoff (Ward et al., 1985). Nutrient losses generally exceed the rate of natural replenishment in New Zealand, so regular inputs through fertilisation and supplemental feeding are essential in pastoral agricultural systems to increase plant production to provide food for livestock (Abrahamson & Darkey, 1988).

Pastoral agriculture is commonly associated with eutrophication and the deterioration of freshwater ecosystems in New Zealand (Verburg et al., 2010). Pastoral agriculture covers ~48% of the Lake Rotorua surface catchment and contributes 67% of the annual total N (TN), and 43% of the annual total P (TP) delivered to the lake (Bay of Plenty Regional Council, 2012). Nutrient inputs through feed and fertiliser, high nutrient return rates in animal excreta, and erosion, are significant drivers of nutrient loss from pastures in New Zealand (Monaghan et al., 2007). Treading by grazing animals may decrease infiltration rates and porosity, and impair plant growth, increasing the likelihood of surface runoff and erosion (McDowell et al., 2003; Ward et al., 1985). Year-round grazing and high stocking rates used to graze crops are common practices in New Zealand, and are associated with increased erosion during typically wet winters (Monaghan et al., 2007). The crushing of plant roots and shoots by livestock may also release P from plant cells which is then available for P loss (McDowell et al., 2003). Converting land from forest to intensively managed pastures affects hydrological conditions due to soil alterations, and changes in the percentage of ground cover and biodiversity which have been linked to increased runoff volumes and deteriorating water quality in surface waters (Bilotta et al., 2007).

2.2.3 Runoff generation

Surface runoff generation is controlled by soil moisture at the start of a rainfall event, inherent soil infiltration properties, and rainfall intensity (Kleinman et al., 2006). Pastoral catchments in New Zealand have been found to have more extreme hydrological responses to rainfall, with higher peak flows, a greater proportion of water yield as stormflow, and a greater temporal variability in water yields, which contributes to greater N and P losses compared to native bush and pine forest catchments (Cooper & Thomsen, 1988).

Runoff generally occurs as a combination of infiltration excess and saturation excess in intensive pasture systems in temperate regions such as those found in New Zealand and the Lake Rotorua catchment (Dougherty et al., 2004; Muller et al., 2010). The contribution of infiltration excess and saturation excess to runoff generation is controlled by antecedent soil moisture, soil properties affecting infiltration rates, and rainfall magnitude and intensity, and may occur simultaneously during a single storm (Kleinman et al., 2006) and therefore may be highly variable spatially and temporally (Dougherty et al., 2004). In intensive pasture systems, saturation excess typically dominates runoff generation because of relatively high infiltration rates due to ground cover and soil structure resulting from high levels of organic matter (Dougherty et al., 2004). However, infiltration excess can be the main contributor of runoff in high traffic areas such as laneways, stock camps, water troughs and gateways where soil disruption or compaction has occurred (Dougherty et al., 2004; Lucci et al., 2012).

In temperate areas, processes in headwaters dominate the response of surface water catchments to rainfall (Ockenden et al., 2016). Headwaters, including ephemeral streams, are estimated to contribute 70% of the mean annual water volume to second-order streams and 55% of higher order rivers (Alexander et al., 2007). Headwater sub-catchments have been found to be responsible for the majority of the streamflow and hydrochemical responses to storms in a stream network (Bieroza et al., 2018).

2.2.4 Sediment and nutrient mobilisation and transport

Agriculture relies on nutrients and soil for production, however hydrologic processes can overwhelm nutrient and soil management strategies, causing losses of valuable productive resources, viz. nutrients and soils, and contribute to water quality degradation (Kleinman et al., 2011; Kleinman et al., 2006; McDowell et al., 2004; Monaghan et al., 2000). The interaction between runoff and soil determines whether potential nutrient and sediment losses are translated into actual losses (McDowell et al., 2008). Places in the landscape where nutrients that may be mobilised overlap with hydrologic flow pathways are considered critical source areas (CSAs), since they are at increased risk of contributing to significant nutrient losses (Sharpley et al., 1994). Critical source areas in pastoral agriculture such as gateways, troughs lanes, and near barns and trees, may represent a small proportion of a catchment, but may be responsible for the majority of nutrients and sediments exported from these areas (McDowell et al., 2004; Pionke et al., 2000). Pastures in low-order catchments in New Zealand have been found to account for an average of 73% of the annual loads of TN and dissolved reactive P (DRP) delivered to small streams, and 84% of the suspended sediments (SS) (McDowell et al., 2017).

The quantity and form of contaminants available for transport depend on factors such as land use and management, soil type, topography, climate, and antecedent soil conditions (Letcher et al., 1999). Hydrologic and chemical factors controlling sediment and nutrient mobilisation affect concentration and load responses, and are highly variable, temporally and spatially (Pionke et al., 1996). The variability reflects the frequency and intensity of storm events affecting runoff generation, viz. the dominant pathway of nutrient transport, and aspects of land management such as soil, nutrient, crop and livestock management (Withers & Jarvie, 2008).

Variable precipitation patterns with very wet winters and dry summers interspersed with large storms, create challenges for nutrient management and loss mitigation on New Zealand's pastoral farms, particularly those on sloping landscapes (McDowell et al., 2013). Wetter soils often have a greater potential for surface flow, and thus potential for more sediment and nutrient mobilisation than dry soils (McDowell et al., 2004). However, hydrophobicity in very dry soils may produce surface runoff, and slaking and dispersion effects, that result in the loss of P-rich clay-sized material (McDowell & Sharpley, 2002).

Seasonality and stormflow characteristics affect the concentration of contaminants in runoff. Soil P tends to increase during warmer, drier months, due to mineralisation of organic P, and decrease in winter due to more frequent and increased runoff and/or leaching (Abell et al., 2013; Lucci et al., 2012). High concentrations of SS, P and N have been observed during the first storm events after prolonged drought conditions, due to a gradual accumulation of solutes and particulates (Bieroza et al., 2019). Nutrient concentrations may increase or decrease with increased stormflow as a result of processes such as flushing from CSAs, or dilution by rainwater (Abell et al., 2013). Suspended sediment concentrations typically increase with higher stormflows due to erosive processes (Abell et al., 2013). A study of nutrients entering streams in the Lake Rotorua catchment found that dissolved N and P concentrations are less correlated with storm generated runoff than particulate N and P concentrations (Rutherford & Timpany, 2008).

Similar to the findings of Lucci et al. (2012), P loads in runoff can be affected by seasonal patterns affecting soil P concentration, with greater P loads mobilised during drier periods when soil P concentration builds up (Abell et al., 2013). More intense storms have been found to generate greater magnitudes of runoff, which tend to mobilise and transport greater quantities of sediments and nutrients from pastures in New Zealand (Cooke, 1988; Smith & Monaghan, 2003) and the Lake Rotorua catchment, specifically (Abell et al., 2013; Dare, 2018). Rare, large storms have been found to be responsible for the majority of the annual sediment and nutrient loading in streams in the Lake Rotorua catchment (Abell et al., 2013; Dare, 2018).

2.2.5 Sediment and phosphorus

Due to the potential for P to be transported with sediments, P and SS movement is inextricably linked (Kronvang, 2007). Soil hydrology strongly influences transport of sediment, and soil texture, organic matter content, soil structure and permeability are factors in soils erodibility (Harrod & Theurer, 2002). The main processes driving erosion and associated PP mobilisation includes the impact of raindrops and soil 'wetting-up', causing slaking and soil particle dispersion, and the detachment and transport of soil particles by the force of flowing water (McDowell et al., 2003). Rainfall intensity and droplet size determine the erosive power of the rain (Kleinman et al., 2006). Rainfall seasonal distribution also has an effect on erosion since it is closely related to ground cover and grass length in pastures (Smith, 1987). Slope pitch and slope length are the most important topographical factors controlling erosion, with the steeper and longer the slopes generally having a greater the risk of erosion due to higher flow velocity and associated higher erosive energy of flowing water (Owens, 2005). The magnitude of the erosion is therefore a function of climate, vegetation, soil type and topography.

Phosphorus in soil may be bound to particulates (PP) or dissolved in solution (DP) (Haygarth et al., 1998) and can exist in either organic and inorganic forms (Condron et al., 2005). Total P is the sum of DP in solution, and PP, which is associated with soil minerals and organic material (Haygarth et al., 1998). Factors controlling the dynamics between the PP and DP fractions, and organic and inorganic forms, influence the quantities, and potential environmental impacts, of mobilised P (Haygarth et al., 2005).

The form of organic P, consisting of undecomposed organic residues, microbes, and organic matter, plays a critical role in determining the dynamics, biological availability, and mobility of soil P (Condron et al., 2005). Organic P forms in the soil include relatively labile phospholipids, nucleic acids, inositols, fulvic acids and humic acids. Immobilisation is the biological conversion of inorganic P to organic P performed by plants and microbes, which subsequently release organic P upon cell death and decay (Condron et al., 2005).

Orthophosphate, PO_4^{3-}, is the inorganic form of P utilised by plants and is typically the most abundant form of P found in nature (Holtan et al., 1988). Mineralisation occurs through the hydrolysis of organic P, by chemical and/or biological reactions (Condron et al., 2005). The form, distribution and retention of inorganic P in the soil is regulated by temperature, pH, redox potential, P concentration in soil solution, and concentrations of Fe, Al, and Ca minerals in the soil (Reddy & DeLaune, 2008).

Dissolved reactive P describes inorganic P which is immediately available to plants and algae, and is of particular relevance to water quality in lakes receiving surface runoff (McDowell et al., 2004). Particulate P, attached to soils and sediment, may become bioavailable over time through desorption and mineralisation processes (Ekholm, 1998). The portion of particulate bound P that can be potentially transformed into DRP under natural conditions is considered bioavailable P (Boström et al., 1988).

Total P is advocated for assessing the nutrient status of lakes due to the potential immediate biological uptake of DRP, and PP's potential source of biologically available P in aquatic systems in the long-term. However, TP measurements in surface runoff could be a poor predictor of P bioavailability in receiving waters since TP might be predominately composed of PP, which may not become available for biological uptake (McDowell et al., 2004). The bioavailability of PP can vary from 10 to 90%, depending on the physical and chemical properties of the PP (Daniel et al., 1998), and the pH, redox potential (Eh) and temperature influencing PP mineralisation processes in the receiving waters (Boström et al., 1988).

The form of P in soil affects its solubility and susceptibility to mobilisation by runoff (Ward et al., 1985). Particulate P is released into surface runoff during erosion events, while DP is transferred by interactions between soil and sediments, and water (Haygarth et al., 2005). Phosphorus mobilisation occurs within the top few mm of soil that interacts with rainfall and surface runoff, referred to as the effective depth of interaction (EDI) (Ahuja et al., 1981). The mixing of water and soil in the EDI, caused by the impact of raindrops and/or the flow of runoff, drives detachment of PP, and mass transfer or desorption of DP into runoff water (Sharpley et al., 1981a). The rainfall period exerts the greatest influence determining the EDI, while other factors include soil characteristics such as soil surface conditions, soil P sorption capacity, and soil type (Ahuja et al., 1981).

Dissolved P and PP may be mobilised and transferred independently or together and may change during a single storm event and/or seasonally, depending on a number of variables (Jordan-Meille & Dorioz, 2004). The temporal variability in P loss throughout the year may be influenced by changes controlling the EDI such as climate conditions and land management activities (Heathwaite & Dils, 2000). The vulnerability of soil to physical damage, and the relative magnitude of sediment and PP transferred in surface runoff depends on soil type, soil P concentration, soil P sorption capacity, rainfall intensity, the rate of flow, pasture-plant cover, stocking rate and slope (McDowell & Wilcock, 2007).

Particulate P can be responsible for the majority of P loss from pastures where effluent or manure has been recently spread, since most P in effluent or manure is in small, easily mobilised particulate form (McDowell et al., 2008). Smaller particles that are more prone to mobilisation and transport by rainfall and surface runoff contain more P than coarser particles, due to greater surface areas providing more P sorption sites (Sharpley, 1985). Steeper slope gradients have been found to contribute to higher erosion rates and likely greater contributions of PP to TP in runoff (Kleinman et al., 2006).

Soil and hydraulic characteristics within the EDI control DP mobilised in runoff (Sharpley et al., 1981a). The availability of DP transfer from soil to runoff depends on the P sorption capacity of the soil controlling desorption-dissolution reactions, fertiliser reaction products, and decaying plant residues (Sharpley et al., 1992). Depending on the concentration of soil P and dissolved P in runoff, soils may be a source or sink of dissolved P transported from pastures in surface runoff (McDowell et al., 2001). Greater soil P at the surface has shown to contribute to greater amounts of DRP in surface runoff, with multiple studies showing a linear relationships between the two (Pote et al., 1996).

Desorption reactions release PP from soil particles into solution, while sorption is the removal of DP from solution and abiotic retention in the particulate phase (Reddy & DeLaune, 2008). The rate and amount of P sorbed and desorbed varies with temperature, the time of the reaction, the DP concentration and other chemical factors (Berkheiser et al., 1980). Readily desorbable P with a high solubility constant may be rapidly released from the surface of minerals (Harrod & Theurer, 2002). Desorption of rapidly exchangeable soil P is transferred to runoff in solution when concentrations of P in runoff are lower than that of soils, which is common with surface runoff conditions (McDowell et al., 2008).

The sorption process is controlled by the concentration of P in soil solution and the ability of P in solution to be replenished by that in the solid phase (Berkheiser et al., 1980). Soils adsorb P when the concentration of P added to the system is higher than the concentration previously in soil solution and P sorption sites are available on soil particles (Reddy & DeLaune, 2008). Sorption occurs in two stages, adsorption and absorption (Barrow, 2015). In the first stage, inorganic P readily adsorbs to Fe and Al hydrous oxide coatings on the surface of soil particles via ligand exchange. In the second stage of sorption, absorption occurs when some of adsorbed P penetrates or diffuses into the solid phase and forms discrete orthophosphate minerals (Barrow, 2015).

Chemical interactions and biological activity controlling sorption and desorption affect the ratio of PP:DP in surface runoff (Condron et al., 2005). The rate of flow and equilibrium P concentration of sediments being transported may influence the potential transition between DP and PP during transport. Increased amounts of SS in runoff may decrease the concentration of DP via P sorption, since finer sediments with high sorption capacities are preferentially mobilised by surface runoff (Sharpley et al., 1981b).

2.2.5.1 Phosphorus loss and erosion in the Lake Rotorua catchment

Agricultural runoff can transfer P in surface runoff and subsurface flow (McDowell & Sharpley, 2003). Generally, P loss in subsurface flow is less than that in surface flow due to P sorption in subsoils (Ward et al., 1985). Surface runoff contributes most of the P entering Lake Rotorua from agricultural sources since subsurface P transport is low due to the prevalence of soils with high P sorption capacities in the catchment (Morgenstern et al., 2015).

Phosphorus transported by surface runoff in the Lake Rotorua catchment is present as both the biologically available DRP and PP (Rutherford & Timpany, 2008). Dissolved P and PP transported from pastures by surface runoff have been identified as significant drivers of Lake Rotorua eutrophication (Abell & Hamilton, 2013; Burger et al., 2007; Tempero et al., 2015). An estimated 71-79% of P delivered to the lake from anthropogenic sources in the catchment is sediment bound (Hamill, 2018). Major sources of PP in the Lake Rotorua catchment are erosion and cattle excreta (Tempero et al., 2015). Studies have found that PP entering Lake Rotorua is able to be released under anoxic conditions and contribute dissolved P in the water column over time, contributing to eutrophication (Abell & Hamilton, 2013; Burger et al., 2007).

Phosphorus inputs utilised in intensive pastoral agriculture, such as fertiliser and manure application, and supplemental feed sourced from outside of the catchment, increase the source potential for P loss by promoting P accumulation in soil (Sharpley et al., 1994). Important influences on P loss from pastoral agriculture in the Lake Rotorua catchment, include soil P concentration management, stock access to waterways, effluent management, and land application of effluent and fertilisers (Hill, 2018).

The concentration of P considered critical for pasture growth, is ~0.2-0.3 mg L^{-1} (Daniel et al., 1993) while the economic optimal Olsen P concentrations in the Lake Rotorua catchment range from 15-30 mg L^{-1} for drystock farms, and 35-45 mg L^{-1} for dairy farms in the catchment (McDowell, 2010). Soil P concentrations that exceed those required for maximum plant growth have occurred in many farms in New Zealand due to excessive fertiliser applications, and pose increased risks for P loss in surface runoff (Monaghan et al., 2007). Since soil P concentrations are proportional to the magnitude of P losses from soils in runoff, high Olsen P levels in soils are likely to contribute to P in surface runoff leaving pastures in the Lake Rotorua catchment (McDowell, 2010).

2.2.6 Nitrogen

Pasture productivity relies on a sustained N supply (Hall, 2008). Nitrogen inputs into pastoral agricultural systems include fertilisers, fixation, effluent spreading and feed supplements that are deposited in animal excreta, with external sources increasing as a result of efforts to increase production (Thorrold & Doyle, 2007). Due to rapid plant uptake and leaching, most N in soils is present in water-insoluble organic complexes, with a small proportion in ionic forms in the soil solution, in mineral forms, or ionic forms adsorbed on to soil colloid surfaces (Cameron et al., 2002).

Nitrogen mobilised and transported by surface runoff may be in dissolved or particulate forms. Ammonium and nitrate are forms of dissolved inorganic nitrate, which are able to stimulate primary productivity and cause eutrophication in N-limited aquatic systems (McKergow et al., 2007). Decomposition of particulate organic matter from dung and soil releases organic N that may become biologically available dissolved inorganic N, mostly as ammonium, which can subsequently undergo nitrification to nitrate (Burger et al., 2007; McDowell et al., 2013).

Nitrogen losses in surface runoff are influenced by factors such as drainage, soil characteristics, slope of the landscape, land use, the presence of grazing animals, and the application of fertiliser (Greenhill et al., 1983). The relationship between soil N concentrations in the surface soil, and the concentration of N forms in surface runoff is difficult to define compared to P, since soil N concentrations are dynamic and readily influenced by changes in soil chemical, physical and biological properties (Burkitt, 2014). Nitrogen transported by surface runoff is highly reactive and may undergo chemical transformations, assimilation and uptake in biological material, and permanent removal via denitrification, depending on environmental conditions and the form of N (Alexander et al., 2007).

Surface runoff from pastoral agriculture typically has elevated concentrations of N (Ledgard et al., 1999). The concentrations of the different N forms in surface runoff can be strongly influence by flow paths and residence times of surface runoff in the landscape (Alexander et al., 2007). Factors including lower surface runoff volumes, higher temperatures stimulating nutrient cycling, fertiliser applications, dung deposited and urine pooled on the soil surface, and the accumulation of dead plant material, may contribute to increased N concentrations in runoff from late summer to early autumn compared to other seasons (Cooke & Cooper, 1988). Particulate organic N has been found to be the dominant form of N in surface runoff from pastures during the winter, due to high rates of erosion (Cooke & Cooper, 1988). Climate change is likely to increase nitrate loading in receiving waters due to greater storm intensities and increase nitrate concentrations during low flows in summer caused by drought (Ockenden et al., 2016).

2.2.6.1 Nitrogen loss in the Lake Rotorua catchment

Rainfall and subsequent runoff mobilises and transports N from pastures to Lake Rotorua (Dare, 2018). Most of the ammonium is transported from pastures by surface runoff in the Lake Rotorua catchment opposed to subsurface drainage, since ammonium is readily adsorbed onto silicate clay and organic matter with high cation exchange capacities common in the catchment's soils (McDowell et al., 2008; Reddy & DeLaune, 2008). Ammonium concentrations in soils are generally low because ammonium is readily nitrified to nitrate by soil microorganisms (Burkitt, 2014). Nitrate, which is negatively charged, is not adsorbed by positively charged soil surfaces, so large losses of nitrate occur when water drains through the soil profile and results in nitrate leaching into the groundwater (Burkitt, 2014). Nitrate leaching into the groundwater is unlikely to undergo denitrification in the Lake Rotorua catchment due to relatively oxic groundwaters, and is expected to reach the lake (Morgenstern et al., 2015).

Inputs increasing N concentrations in the soil also increase the potential for N loss in runoff (Hatch et al., 2002). Fertiliser inputs, high stocking rates which generate dung and urine spots highly concentrated in N, and year round grazing driving erosion, all contribute to N losses being greater from intensive pastoral agriculture, compared to other rural land uses in New Zealand (Elliott, 2005). Due to its high solubility, N transport from the land to receiving waters are controlled by the hydrological conditions that expand both laterally and vertically during periods of wetting in temperate regions such as the Lake Rotorua catchment (Alexander et al., 2007). On average, pastoral agriculture in the Lake Rotorua catchment loses 29 kg N ha^{-1} y^{-1} and is responsible for 578 t N y^{-1} (77%) of the annual N loads delivered to the lake from the catchment (Donald et al., 2019).

2.3 Mitigation strategies

2.3.1 Need and implementation

The New Zealand Ministry for Primary Industry hopes to double the value of New Zealand exports from $32 billion to $64 billion between 2012 and 2025 (Ministry for Primary Industries, 2016). Achieving these goals will require greater intensification of pastoral agricultural production (Howard-Williams et al., 2010). Balancing the economic drive for increased pastoral production with environmental policies set out by the National Policy Statement will be a major challenge for farmers and regulators in New Zealand (Edgar, 2008; Howard-Williams et al., 2010). Therefore, it is becoming increasingly important to identify and utilise cost-effective mitigation strategies that prevent nutrient losses generated by surface runoff from pastures, particularly since climate change is likely to exacerbate nutrient and sediment losses due to more dramatic hydrologic conditions in New Zealand (Ministry for the Envrironment, 2019; Ockenden et al., 2016).

Although a range of nutrient management options exist, identifying, implementing and maintaining appropriate strategies may be difficult (Osmond et al., 2019). Identifying optimal mitigation strategies is a major challenge since the efficacy of the strategy depends on unique climate, landscape and management characteristics (Hill, 2018). The performance of a mitigation strategy may vary spatially and temporally due to the interaction between unique landscape and climate factors that affect hydrochemical responses to rainfall, and the various mechanisms involved that affect the ability of a strategy to mitigate contaminants (McKergow et al., 2007).

Measures to overcome challenges related to identifying and adopting mitigation strategies rely on gathering scientific evidence to determine strategy efficacy, and utilising the local knowledge of landowners (Bieroza et al., 2019). Financial incentives and a strong understanding of the local agricultural systems have the greatest impact on the adoption of mitigation strategies, while education and technical assistance are also important (Osmond et al., 2019). Mitigation strategies are more likely to be adopted if they are minor adjustments to farm management practices, with minimal cost and impact on the farm system or production levels, compared to the installation of more complex and/or expensive edge of field approaches (Hill, 2018). Also, communicating potential challenges, trade-offs and time lags involved with certain mitigation strategies to stakeholders, can improve their participation in future mitigation programs (Bieroza et al., 2019).

Cost: benefit analyses are important tools for assisting decision makers attempting to implement appropriate mitigation strategies (Bieroza et al., 2019; McDowell, 2010). In order to develop useful cost benefit analyses, the efficacy of mitigation strategies must be proven by quantifying their performance in the field. One way to improve the usefulness and accuracy of cost: benefit analyses is to include mitigation strategy performance assessments in implementation budgets, and by making data accessible to researchers and decision makers (Bieroza et al., 2019).

'Scaling up' mitigation efforts is often required to have a significant impact at a catchment scale. Various challenges exist to scaling up due to greater demands of time and resources, determining parties responsible for financial burdens, and the need for co-ordination and inclusion of a greater diversity of stakeholder groups with potentially varied interests (Osmond et al., 2019). It has been reported that utilising a combination of strategies is the most effective approach to manage nutrients lost from pastures in surface runoff (Quinn et al., 2009). However, suites of strategies do not always lead to water quality improvements, potentially due to the inability of techniques to remove nutrients from runoff, the lack of redundancy in the system, and the possibility of strategies becoming a source of nutrients (Osmond et al., 2019). Still, due to the complicated relationship between land use and hydrology, multiple strategies should be utilised to target various sources of pollution, including land (in-field) and in-stream (off-field) networks, for mitigation programs to be effective at the catchment scale (Bieroza et al., 2019; McDowell, 2010).

2.3.2 Off-field mitigation

Various 'off-field' interventions have contributed to improved trends in Lake Rotorua water quality (Hamill, 2018), including ceasing to discharge Rotorua municipal wastewater into the lake, reticulating sewage from smaller communities, N removal from water in the Tikitere geothermal field, and alum dosing to lock up P in the Utuhina and Puarenga streams (Stephens et al., 2018). While effective, alum dosing is not considered a sustainable mitigation strategy due to potential toxicological effects (Tempero et al., 2015). In-lake remediation techniques, such as hydraulic flushing for direct algal control, P locking (geoengineering), floating wetlands, bio-manipulation and macrophyte harvesting, have also been considered (Donald et al., 2019).

2.3.3 In-field mitigation

Due to the influence of hydrology on nutrient mobilisation and transport, storm periods have been recognised as important opportunities for mitigating nutrient loses from farms (Gburek & Sharpley, 1998). Since recognising the critical role hydrology plays in the impact of pastoral agriculture on water quality, a range of mitigation strategies have been developed to target nutrient losses driven by different hydrological pathways. Nutrient loss mitigation methods include those that reduce the amount of nutrients imported on to the farm in feeds and fertilisers, nutrient mobilisation control methods that affect solubilised nutrients, detached particles and incidental transfer of manure and fertilisers, and transport control methods that target mobilised nutrients (Haygarth et al., 2009).

In-field management options for controlling nutrient losses by surface runoff may focus on controlling nutrient inputs and/or controlling nutrient outputs, and may be classified as 'on-farm management', 'amendments' or 'edge of field' strategies (McDowell, 2010). The cost and effectiveness of implementing a mitigation strategy may vary drastically depending on the technique and the location the technique is to be implemented (Bieroza et al., 2019). Generally, the closer the mitigation strategy is to the source of pollution, the more efficient and lower the cost to implement (Bieroza et al., 2019). Therefore, the benefits derived from implementing mitigation strategies, in terms of cost, follow this sequence: farm management > amendment > edge of field > in-stream (Bieroza et al., 2019).

Important farm management techniques for limiting nutrient losses from pastures include fertiliser best-management practices such as testing soil P concentrations to determine appropriate fertilisation rates, and soil and stock management approaches that attempt to prevent soil erosion (Howard-Williams et al., 2010; Sharpley et al., 1994). While some land management practices are able to successfully control erosion and associated PP losses, they may have negligible impacts on DP losses due to 'legacy P' in soils, which can be common in New Zealand pastures due to historically high fertiliser inputs (Daniel et al., 1993). Also, intense hydrological conditions may overwhelm land management strategies attempting to minimise erosion and nutrient mobilisation (Kleinman et al., 2011).

Storm periods have been recognised as important opportunities for mitigating nutrient losses from farms since they may overwhelm farm management strategies aimed to control nutrient losses (Gburek & Sharpley, 1998). Amendments, such as applying sorbents and flocculants as DP sorption and erosion mitigation measures, are some approaches to address storm generated runoff overwhelming farm management strategies.

Identifying and mitigating losses from CSAs should be a top priority, since the approach can be highly cost-effective, given the areas requiring mitigation are usually relatively small, while potential nutrient losses are high (Pionke et al., 1996). Ideally, all CSAs within a catchment should be identified, and mitigation strategies implemented in these areas, to maximise mitigation efforts and minimise cost (Osmond et al., 2019). However, identifying potentially small areas across large landscapes is challenging. A farm scale spatial tool that is able to spatially identify CSAs, and compare the cost and environmental effectiveness of different mitigation scenarios, MitAgator (https://ballance.co.nz/mitagator), has recently been developed in New Zealand, although the tool currently has limited accessibility to farmers in the Lake Rotorua catchment (Hill, 2018). Track and lane management, either by engineering methods (runoff diversion berms), or using sorbents, can be effective approaches to mitigate the impact of common CSAs in pastoral agricultural systems (Hill, 2018; McDowell & Nash, 2012). Minimising fertiliser applications, or implementing livestock exclusion or reductions to CSAs are also approaches to reduce losses from these areas (Howard-Williams et al., 2010).

2.3.3.1 Edge of field mitigation

Common edge of field mitigation strategies that address surface runoff include sedimentation ponds, constructed wetlands, and riparian buffer strips, and stormwater detention areas (SDAs). Edge of field mitigation strategies that increase stormflow residence time have been found to decrease surface runoff flows and facilitate sediment deposition by decreasing the kinetic energy of flowing water (McKergow et al., 2007). A wide range of performances have been reported for edge of field mitigation strategies (McDowell, 2010). Also, it is acknowledged that while increased residence times can facilitate soil infiltration which contributes to surface runoff contaminant mitigation (Skaggs et al., 1994), the effect on runoff volumes have been underreported (McKergow et al., 2007).

Edge of field methods that induce sedimentation include SDAs (Shukla et al., 2017), wetlands, and grass buffer strips and stream-bank vegetation (Hart et al., 2004). The longevity of sediment attenuation is influenced by the type of mitigation strategy.

Strategies such as grass filter strips are likely to retain sediments for much briefer periods of time (days to months), compared to other strategies where sediments are blanketed over a wide area (up to dozens of years) such as with SDAs (McKergow et al., 2007).

Grass buffer strips specifically target PP in surface runoff (Hill, 2018; McDowell, 2010). Studies investigating buffer strips report that soil types with higher infiltration capacity can reduce runoff to a greater degree than soils having lower infiltration rates, and that greater infiltration due to flow impediment decreases erosion and the transport of sediments and nutrients (Dosskey, 2001). A study of fenced grass buffer strips within paddocks in the Lake Rotorua catchment decreased P losses from the buffer strip area by 40%, compared to a grazed control during two runoff events (McKergow et al., 2007). However, buffers may be less effective if surface runoff becomes chanellised or the strips become clogged with sediment (McDowell, 2010). Many factors, including buffer width, vegetation type, soil type and soil P sorption capacity status, interact with varying hydrologic factors which affects the complex cycling of P in buffer zones, and controls the ability of buffer strips to be sediment and nutrient sinks (Dosskey, 2001; Osmond et al., 2019). Mobilisation of organic matter and sediments, and desorption contributing to DP in runoff, can decrease the effectiveness of buffer strips over time (Osmond et al., 2019).

Treatment wetlands may also be an effective strategy to target sediment and nutrients transported by surface runoff (Osmond et al., 2019). However, multiple studies have reported both positive and negative nutrient retention by wetlands due to the complex variables affecting nutrient cycling and the potential for previously deposited sediments and senescent organic matter to be flushed downstream, especially during high flow events (Tanner & Sukias, 2011). The ability of wetlands to retain nutrients also changes over time, since dissolved P and N may eventually be released from enriched sediments or decaying organic matter in the wetland (Hill, 2018; Tanner & Sukias, 2011). Besides not consistently decreasing nutrients, and being expensive and complex to construct and maintain, wetlands are likely to be located lower in the Lake Rotorua catchment, while strategies that reduce P losses further upstream are likely to be more effective and have a greater downstream impact (Bieroza et al., 2018; Hill, 2018; Ockenden et al., 2017).

Another mitigation method that increases stormflow residence times by impounding surface runoff is SDAs, which are commonly used in flood protection, but are increasingly being used for water quality improvement, in agricultural and urban settings, particularly in the USA (Shukla et al., 2017; Stanley, 1996). Agricultural SDAs is a collective term used for natural or manmade depressions, ponds, and reservoirs (Shukla et al., 2017). Due to the nonuniformity in location and design, and the varying approaches used to investigate their performance, studies on SDAs are often not directly applicable to one another. However, the processes for N and P retention for the varying SDA applications are similar, such as sedimentation, plant uptake, soil adsorption, and microbial conversions (e.g., nitrification–denitrification in the case of N) (Shukla et al., 2017).

Sedimentation ponds are one type of SDA that utilises the ponding to decrease the kinetic energy of flowing water and affects particle size transported in surface flow, due to the sinking of coarse sand-sized particles (McDowell et al., 2003). An investigation of sedimentation ponds in Idaho, USA found sediment retention efficiencies of 65 to 76% resulted in P retention efficiencies of 25 to 33% (Brown et al., 1981). McDowell et al. (2006) found that sedimentation and sorption decreased TP and DRP loads discharged from a sedimentation pond fed by a perennial stream. Phosphorus discharged from sedimentation ponds are likely to be in dissolved forms or attached to smaller, less dense particles that do not settle readily (Brown et al., 1981). Also, sedimentation ponds have been found increase nitrate discharges, and become sources of P and SS due to meteorological and antecedent hydrological conditions (Bieroza et al., 2019).

Dry detention ponds are another type of SDA that are dry except for periods after storms, in which surface runoff is temporarily impounded (Stanley, 1996). During impoundment, runoff is passively drained by soil infiltration and a constant discharge from an outlet pipe. Sedimentation and the reduction of surface runoff discharged from the dry detention pond are responsible for sediment and nutrient load reductions (Harper et al., 1999). A 6 month study of a detention pond receiving runoff from a 10 ha catchment in Florida, USA reported that the soil infiltration of the 70% influent was primarily responsible for effective load reductions of total suspended solids (99%), TP (84%) and TN (86%) (Harper et al., 1999).

2.3.3.2 Comparing studies of mitigation strategy effectiveness

The discussion of complex spatial and temporal factors influencing runoff generation, and contaminant mobilisation, transport and fate, elucidates the variable nature of hydrochemical characteristics of surface runoff. Because the efficacy of mitigation strategies are affected by antecedent conditions, the magnitude of runoff occurring during a storm, the form and quantities of contaminants delivered in runoff, and the potential fate of various attenuated contaminants, the performance of mitigation strategies will also vary spatially and temporally (Haygarth et al., 2009). Therefore, comparing the performance of different mitigation strategies in similar locations, similar strategies in different locations, or even a strategy in a location over various time periods has limited practicality and usefulness because of the multiplicity variables.

Spatial and temporal factors affecting characteristics within a targeted catchment and a mitigation strategy instalment include, but are not limited to: climate, precipitation patterns, topography, land use and management, ground cover and vegetation type, sediment and detritus build-up in the mitigation area, and soil characteristics including soil type, moisture, porosity, chemistry and nutrient concentrations (McDowell et al., 2013; Shukla et al., 2017). These factors, which may vary temporally within and between seasons and years, affect the hydrochemical characteristics of runoff generated in a catchment (magnitudes of runoff, and contaminant forms and concentrations), as well as characteristics within the mitigation area itself, which interact to affect the potential performance of mitigation strategies (McDowell et al., 2013; Shukla et al., 2017; Tanner & Sukias, 2011).

Spatial differences between mitigation strategy geographic locations would affect those factors mentioned in regard to temporal variability(McDowell et al., 2013). Additionally, other spatial variables affecting runoff and contaminant delivery, and mitigation efficacy, include the location of the strategy within the catchment and the size of the catchment contributing runoff to a mitigation strategy location (Shukla et al., 2017).

The performance of a mitigation strategy, often driven by hydraulic retention times, are also affected by the way the spatial and temporal variables previously mentioned interact with mitigation strategy design factors, which vary by location (Dosskey, 2001; Shukla et al., 2017). For instance, there is nonuniformity between buffer widths and vegetation, treatment wetland size, hydraulic capacity and vegetation, and sedimentation pond and stormwater detention area ponding volumes per hectare of contributing catchment at various locations (Dosskey, 2001; Hamill et al., 2010; Hill, 2018; McKergow et al., 2007; Shukla et al., 2017; Stanley, 1996; Tanner & Sukias, 2011). Also, whether maintenance of a mitigation strategy area affects the efficacy and the schedule in which the maintenance is carried out, if at all, can also affect mitigation efficacy.

To compound the complexity of interactions between the factors discussed so far, there is variability in the approach mitigation strategy performance is assessed and reported on in the literature. Differences in the methodologies and reporting between studies include: the temporal and spatial scale of investigations, simulated versus natural environment experiments, modelling versus in-field results, varying sample and data collection regimes, the age of an edge of field mitigation strategy instalment, and reported results such as whether dissolved and/or particulate and/or total nutrient forms were investigated, and whether concentrations and/or loads and/or yields are reported. The lack of consistency between studies amplifies the difficulty in attempting to organize and compare the results of studies investigating different and similar mitigation strategies in the literature.

A study by Haygarth et al. (2009) acknowledged the unconsolidated nature of studies reporting on P mitigation strategies in agricultural settings. This study proposes an approach of assessing and reporting the potential cost-effectiveness of a range of P mitigation strategies that used a process of collating an inventory of potential P mitigation methods, identifying the varying ranges of P transfer from key model farm typologies, and the potential application and cost-effectiveness of mitigation methods to model farm systems. While Haygarth et al. (2009) recognises uncertainties with estimated loss and treatment coefficients in their study, they point out that adopting a uniform method of assessing and reporting the cost-effectiveness of mitigation strategies described in this study would be useful for comparing potential mitigation options.

While direct comparisons between studies investigating the performance of various mitigation strategies under various settings may not be useful, identifying the functions responsible for mitigation performance, and roughly what results one could expect under certain scenarios can inform investigators and decision makers about the potential for what strategy might be appropriate in a specific area. Because identifying and comparing the costs and benefits of strategies helps overcome challenges related to implementing appropriate mitigation approaches (Bieroza et al., 2019), efforts have been made to organize and present findings from various studies in ways that attempt to be useful to scientists and policy makers. For instance, Cuttle et al. (2016) developed a ‘method-centric user manual’ inventorying 44 agricultural diffuse water pollution mitigation strategies, describing how each approach work in controlling N, P and fecal indicator organism, their cost and effectiveness and their potential application within different farming systems and soil types. This study used model farms representative of the main UK farming sectors closely defined in terms of farmed area, field size, cropping, livestock numbers and ages, housing period, fertiliser and manure/slurry management, using typical values obtained from published data, as well as expert judgements to fill the gaps where scientific data were lacking, to estimate pollutant losses at the whole-farm scale. Although the authors of this study considered themselves ‘successful in providing provisional estimates of cost and effectiveness in an accessible form,’ they acknowledge ‘a number of limitations to its content and application.’ These limitations include the estimated values in the ‘User Manual’ being strictly valid for farms matching the defined model farm types which could not be representative of the full range of farms found within a particular farming sector or of different soils and climate zones, the lack of consideration for potential existing mitigation methods on actual farms and the varying costs of implementing applicable mitigation methods, the sensitivity of methods calculating baseline pollutant losses based on the proportion of a farm contributing to losses, and the uncertainty arising from the difficulties of extending results from what was often a limited number of research studies to a whole-farm scale and to different soils. Consequently, Cuttle et al. (2016) make clear that ‘estimates of cost and effectiveness only apply to the model farms and cannot be simply extrapolated to the whole of a farming sector across farms of different sizes and in different regions.’

An example of an effective way of summarising, organising and reporting the information applicable to the application of farm-scale strategies to mitigate the loss of water quality contaminants to water in New Zealand was presented by McDowell et al. (2013). An adaptation of Table 4.1 from the McDowell et al. (2013) report is presented below as an example of how information about various edge of field mitigation strategies can be effectively presented (Table 2.2). This table design allows users to clearly see under what farming system the strategy can be used, how the strategy functions, the effectiveness and cost-effectiveness of the strategies in regards to various contaminants, reasons for performance variability and limited strategy implementation of a strategy, co-benefits of the strategy, and listed references that allow users to quickly locate how the studies used in compiling the table potentially varied.

Table 2.2: Information applicable to comparing edge of field farm-scale strategies to mitigate the loss of water quality contaminants to water adapted from McDowell et al. (2013) Table 4.1.

Strategy	Range of applications	Description of function	Effectiveness	Relative cost	Reasons for variability	Factors limiting uptake	Co-benefits	References
Constructed wetlands	All farming enterprises	Modification of landscape features such as depressions and gullies to form wetlands. Slow water movement encourages deposition of suspended sediment and entrained contaminants (e.g. P). Compared to many natural wetlands, constructed wetlands can be designed to remove contaminants from waterways by: 1) decreasing flow rates and increasing contact with vegetation – thereby encouraging sedimentation; 2) improving contact between inflowing water, sediment and biofilms to encourage contaminant uptake and sorption; and 3) creating anoxic and aerobic zones to encourage bacterial nitrogen processing, particularly denitrification loss to the atmosphere. Performance varies depending on wetland size and configuration, hydrological regime, and contaminant type and form. An adaptation has seen the inclusion of floating wetlands (emergent wetland plants grown hydroponically on floating mats) to remove significant quantities of dissolved P from artificial urban stormwater compared to unplanted mats. However, it is also noted that while the regular harvesting and removal of plants growing on wetland sediments may increase P removal from the wetland, unless the biomass has an economic value, harvesting is not a cost-effective strategy. Although relatively easy to construct and maintain, constructed wetlands also remove land from production, which impairs their cost-effectiveness.	Very high [N]; Medium [P]; High [SS]	High [N]; Very high [P]; Medium [SS]	Wetland performance depends on intercepting the maximum amount of run-off from the catchment at the right flow rate.	No suitable areas on farm (i.e. catchment lies outside of farm area).	Flood attenuation, wildlife habitat and biodiversity	Headley and Tanner (2007); McKergow et al. (2007a); Tanner et al. (2005), .

Sediment traps	All farming enterprises	Stock pond or earth reservoir constructed at natural outlet of zero-order catchment. In-stream sediment traps are useful for the retention of coarse sized sediment and sediment-associated N and P, but do little to retain N and P bound to fine sediment. As the P sorptive capacity of fine particles is much greater than coarse particles (w/w basis), sediment traps can be ineffective at decreasing P loss if the soil in finely textured and/or surface runoff is dominated by fines.	Low [P]; Very high [SS]; Low [E. coli]	Very high [P]; Very high [SS]; Very high [E. coli]	Although design can be modified to maximise removal via settling, traps are ineffective at high flows when most sediment is transported	May require resource consent	Potential to buffer storm events and therefore potential downstream flooding.	Hicks DL (1995); Hudson (2002); McDowell et al. (2006)
Vegetated buffer strips	All farming enterprises	Vegetated buffer strips work to decrease contaminant loss in surface runoff by a combination of filtration, deposition, and improving infiltration. The upslope edge of the strip is where most large particles and particulates (sediment and entrained N, P and E. coli) are filtered-out, and the speed of surface runoff slows enough that deposition occurs. If the hydrology allows, a more important mechanism that decreases contaminant loss is infiltration (i.e. there is no water for transport overland into streams). This deposits of particulate material onto the soil surface or vegetation and increases the interaction and sorption of dissolved P with the soil.	High [P]; High [SS]; Low [E. coli]	High [P]; High [SS]; Very high [E. coli]	Buffer strips do have major flaws: 1) the strip can quickly become clogged with sediment; 2) they function poorly in areas that are often saturated due to limited infiltration; 3) they function best under sheet flow, whereas most surface runoff tends to converge into small channels that can bypass or inundate strips; and, 4) grassed buffer strips function best when the number of tillers is greatest, which generally occurs where biomass is harvested (i.e. under grazing).	Land adjacent to stream may not be available or suitable for a buffer strip.	Potential to stabilise stream banks.	Longhurst (2009); McKergow et al. (2007a,b;); Redding et al. (2008); Smith (1989)

2.3.4 Efforts to mitigate the impact of pastoral agriculture on surface water quality in the Lake Rotorua catchment

Sustainable in-field mitigation approaches to improve Lake Rotorua water quality involve identifying anthropogenic sources of nutrients in the catchment, and implementing strategies to reduce losses from land, or attenuating mobilised nutrients before they reach the lake (Tempero et al., 2015). Identifying and implementing appropriate, cost-effective strategies to improve Lake Rotorua water quality is a priority of the Bay of Plenty Regional Council and local stakeholders, due to the lake's contribution to regional tourism revenue and significant cultural values (Bay of Plenty Regional Council, 2012). In order to reach Lake Rotorua water quality objectives, effective mitigation strategies that target the right contaminants in the right place, need to be identified at the farm scale and implemented at the catchment scale (McDowell, 2012; Osmond et al., 2019). Mitigation strategies that are effective during intense storm events are important to identify in the Lake Rotorua catchment since studies have shown that rare, high magnitude events may cause the majority of the annual nutrient loading into the lake (Abell et al., 2013; Dare, 2018). Identifying mitigation strategies that are effective during more extreme hydrological conditions will become more important over time due to the effects of climate change (Ockenden et al., 2017). Deploying mitigation strategies, and managing nutrient losses from headwater subcatchments, are also critical for improving water quality, and decreasing the potential for eutrophication in receiving waters (Bieroza et al., 2018; Ockenden et al., 2017). Studies of varying catchment sizes have found that hydrochemical conditions in downstream waters are strongly connected to distant landscape characteristics and respond relatively quickly to changes in in upstream sources, such as the implementation of nutrient mitigation strategies (Alexander et al., 2007). In New Zealand pastoral catchments, where an average of 77% of contaminants are derived from low-order streams, focussing mitigation efforts on preventing contaminant loading to headwaters may be more cost-effective than trying to mitigate their impact further downstream (McDowell et al., 2017).

The prevalence of PP, and the proportion of annual nutrient loads delivered to Lake Rotorua during rare, high magnitude storm events, highlights the importance of utilising best land use practices to control erosion and implementing mitigation strategies that facilitate sedimentation (Abell et al., 2013; Tempero et al., 2015). McDowell (2010) has suggested that multiple land management mitigation strategies will need to be implemented, including potentially novel approaches, to reach Lake Rotorua nutrient load reduction goals, although identifying effective mitigation strategies that are easily adopted, is likely to be challenging.

A novel strategy to address sediment and nutrient losses from pastoral agriculture has been developed and implemented in the Lake Rotorua catchment for the first time is referred to as detainment bunds (DBs). Detainment bunds are a type of SDA that impedes stormflow and increases surface runoff residence times with earthen, stormwater retention structures that form temporary ponds (Fig 2.1). A DB may be approximately 1.5-2 m high and 20-80 m long, and constructed on pastures across the flow path of targeted low-order ephemeral streams. Currently, the DB site selection and design protocol for the Bay of Plenty region promotes a minimum ratio of 120 m^3 pond volume capacity: 1 ha of contributing catchment area, and local government regulations stipulate a 10,000 m^3 maximum impoundment volume (Paterson & Clarke, 2013).

Figure 2.1: Photo of a pond formed on pasture by a detainment bund impeding the flow of surface runoff generated during a storm event. The fencing protects sampling equipment from livestock.

A DB can be purpose-built, or constructed by modifying an existing structure, such as a raised raceway that divides a paddock. Due to design and regulatory limitations, some landscapes, such as steep mountainous country with incised valley floors and flat flood plains, are not appropriate for DB locations due to their topography (Paterson, 2019). The ‘Detainment Bund Handbook’ used to advise parties interested in implementing the strategy, describes features specific to DBs to include an upstand riser connected to an outlet pipe that passes through the bund and discharges ponded runoff on the downstream side of the DB (Paterson & Clarke, 2013) (Fig. 2.2 and 2.3). The upstand riser is a ~1 m diameter vertical pipe reaching to ~20 cm below the lowest point of the DB, installed near the bund at the low point of the ponding area. Ponded water may be discharged from the outlet pipe if the pond height exceeds that of the riser (Fig. 2.4). During large runoff events that overwhelm the pond storage capacity and discharge rates from the upstand riser, water may be discharged via an 'emergency spillway' at the lowest point of the DB. The surface of the spillway is protected by either a mat material, compacted substrate, or stable grass cover.

Figure 2.2: Photo of an upstand riser installed in the ponding area of a detainment bund. The riser is connected to an outlet pipe that passes through the bund, which discharges ponded runoff on the downstream side.

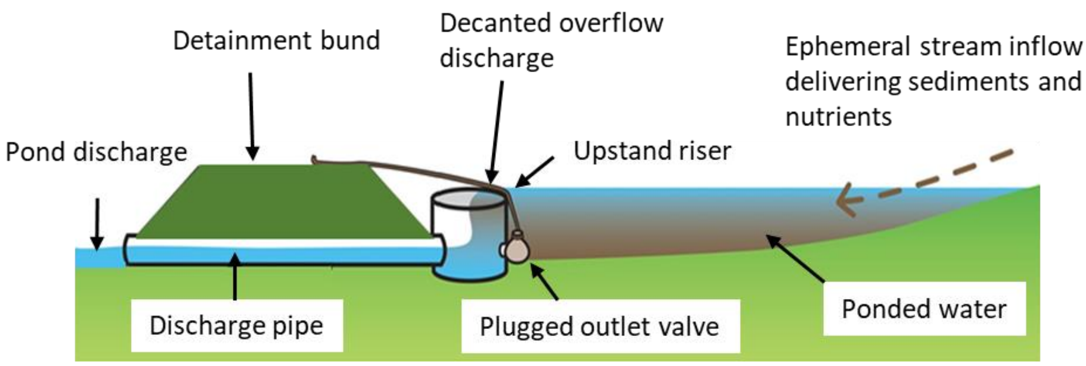

Figure 2.3: Cross-section of ponding area showing the ephemeral stream inflow delivering sediments and nutrients, and ponding behind a detainment bund. If the pond height exceeds the height of the upstand riser then 'decanted overflow' is discharged via a pipe passing through the bund wall.

a)  b) 

Figure 2.4: Photos of detainment bund pond below the height of the upstand riser (a), and breaching the upstand riser (b). Ponded water may be discharged from the outlet pipe on the downstream side of the bund if the pond height exceeds that of the riser.

The ability of DBs to increase surface runoff residence time suggest that similar mechanisms driving the efficacy of proven mitigation strategies described earlier, are similar to those of DBs. Of the different types of SDAs and other edge of field mitigation strategies previously mentioned, DBs perform most similarly to dry detention ponds. While both dry detention ponds and DBs are dry between storm events, and induce soil infiltration during the temporary retention of surface runoff, they differ in that dry detention ponds are passively drained by a constant discharge from an outlet pipe, while DB ponds can be rapidly drained by unplugging an outlet valve. Although ryegrass-based pastures in New Zealand have some tolerance to saturated soil conditions, early adopters of the DB strategy consider that restricting the inundation period to a maximum of 3 days reduces the risk of pasture damage in the ponding area (Paterson & Clarke, 2013).

Over 20 DBs have been constructed in the Lake Rotorua catchment since 2010. A preliminary study of DBs in the Lake Rotorua catchment found that P enriched sediments were deposited in ponding areas (Clarke, 2013). This finding served as a proof-of-concept for the strategy, however, the ability of DBs to effectively mitigate annual sediment and nutrient losses from pastures in the Lake Rotorua catchment have not been thoroughly investigated and quantified. Due to the necessity of identifying effective, and potentially novel mitigation strategies in order to achieve Lake Rotorua water quality objectives, it is important to scientifically investigate the functionality of DBs and assess whether they are a potential mitigation option available for pastoral famers in the area.

2.3.5 Methods for determining the efficacy of detention bunds

2.3.5.1 Sample collection

Surface runoff flows were measured and water samples were collected during storm events at the 2 DBs investigated during a 12-month investigation during this thesis. Flowmeters (UNIDATA® 6527 Starflow® QSD), and Isco® (California, USA) 6712 portable auto-samplers, capable of filling 24 x 1 L bottles collected inflow and measured and sampled runoff inflows and discharges to determine the effect of the mitigation strategy on surface runoff volumes, suspended sediments (SS), P and N.

Inflow auto-samplers collected a 1 L sample every 20 min for the first 10 samples, then one 1 L sample/h thereafter (Harmel et al., 2003; Stanley, 1996). The mouth of a rain guarded 750-mL self-sealing bottle using a ping-pong ball inside the bottle, was installed at ground level near the pond outlet valve to capture a sample of the initial flush of surface runoff generated before the inflow auto-sampler was triggered.

Discharge auto-samplers were programmed to collect a 1-L sample each hour (Harmel et al., 2003; Stanley, 1996). Sampled discharge flows were generated if the pond height exceeded the upstand riser height during a storm event (i.e. 'overflow discharge'), and when the outlet valve at the base of the riser was opened to release the pond at the end of the event treatment (i.e. 'release discharge'), typically on the third day of ponding (Fig. 2.3).

2.3.5.2 Sample analysis

Because nutrient transformations during handling and storage may occur quickly (Haygarth & Edwards, 2009), efforts were made to collect samples from the field within 24 h of the end of the ponding event and refrigerate collected samples at 4 °C prior to subsampling (within ~24 hr of collection). Separate subsamples (~30 mL) were taken from the 1-L field samples for dissolved P and N, and total P and N analysis. The samples analysed for dissolved P and N were filtered through a 0.45μm cellulose acetate membrane filter. Both the filtered and unfiltered subsamples were subsequently frozen until analysis. Although it has been noted in the literature that freezing samples is not advisable due to physical transformations that may occur as a result of cell lysis (Haygarth & Edwards, 2009), freezing is common practice in studies similar to those in this thesis (e.g.,) and offers a level of practicality when field sites are located 370 km away and an overnight trip is required for each sampling event. It is also important to note that all samples collected in this study were frozen, therefore any impact of freezing is assumed to be uniform across all samples. After subsamples were obtained, the remaining field sample was kept refrigerated until being analysed for SS concentration.

The standard gravimetric filter analysis procedure was used to determine SS concentrations in this thesis (American Public Health Association, 2005). Filter papers (Whatman GF/C 70 mm) were rinsed with deionised water then pre-dried in the oven at 105°C for 1 day before being weighed. After drying, the filters were cooled in a desiccator, and then re-weighed prior to filtering the water samples. After the remaining field samples (~900mL) were filtered, the filters were again oven dried at 105°C for 1 day and cooled in a desiccator before being weighed.

Total P and N concentrations were determined using the unfiltered subsamples that were digested using the alkaline persulphate digestion method of Hosomi and Sudo (1986). Both the digested unfiltered and the filtered DRP subsamples were analysed for P concentrations following the standard molybdenum blue method (Murphy & Riley, 1962) using automated flow injection analysis (QuikChem 8000 FIA+; Lachat Instruments, Loveland, CO). The molybdenum blue method may overestimate P in DRP samples in comparison with chromatographic determinations (Haygarth & Edwards, 2009), although this procedure is still commonly used throughout the literature. Unfiltered TN subsamples that were digested were analysed for nitrate-N concentrations using the FIA with Lachat QuickChem methods [10-107-04-1-A (NO^-_3-N). Filtered subsamples were analysed for concentrations of nitrate-N and ammonium-N using the FIA with Lachat QuickChem methods [10-107-04-1-A (NO^-_3-N), 10-107-06-2-B (NH_4^+)].

Prior to sample analysis, calibration curves were established on the analysis equipment using 6 standard solutions for P (0.05-0.8 ppm) and 7 standard solutions for N (0.25-12 ppm). After each 10 sample batch was analysed, a batch of 3 solutions (blank-standard-blank) was run to determine any concentration drift during analysis. Samples below the lower detection limit were included in the data used to calculate mean flow proportional concentrations, and samples exceeding the upper limit of the standards were diluted and reanalysed.

2.3.5.3 Data analysis

All surface runoff occurring from 1 December 2017 to 30 November 2018 was measured at the DB sites investigated in this thesis. Storm periods causing surface runoff to occur are referred to as ‘events’. Depending on the amount of runoff delivered to the DB, events were differentiated into 2 types according to the mode(s) in which ponded water was discharged from the DB. ‘Overflow Events’ occurred during larger runoff events when inflow continued to be delivered to the pond after the pond height exceeded the height of the upstand riser, generating ‘overflow discharge’ (Fig. 3.1). After 3 days of ponding, any residual ponded water was evacuated when the outlet valve was opened, creating ‘release discharge’. Therefore, ‘Overflow Events’ had both ‘overflow discharge’ and ‘release discharge’ components (Fig. 3.3). In contrast, ‘Non-overflow Events’ were smaller storms that did not contribute enough runoff to overtop the upstand riser. Non-overflow Events included events which, at the end of the 3-day treatment period, either had a portion of ponded runoff to discharge by opening the outlet valve, or had no runoff to discharge due to leakage and infiltration into the soil.

Data from each site were analysed to calculate annual results and to compare results based on event types. The volume of water delivered to and discharged from the bunds were compared to determine the volume of water infiltrating the soils during ponding. Changes to concentrations of contaminants (SS and the different forms of P and N) were calculated as the percent difference between inflow and outflow concentrations (percent change in concentration=(outflow-inflow)/inflow)*100). The inflow and discharge loads of contaminants were also compared. The results of these data were analysed to determine factors influencing DB performance.

Statistical analyses of the data were conducted in R ver. 3.6 (R Development Core Team, 2019) to determine relationships between factors affecting what was delivered to the pond in terms of runoff and contaminants and the effect of the DB treatment on runoff volumes, and contaminant concentrations and loads. Because the data was highly non-normal/heavily skewed, as is typical with many data from environmental science, linear models were inappropriate to analyse the data. Instead regression modelling proceeded as follows:

1. A suitable linear or generalised linear mixed model was chosen based on the form of the response data. Site was left in every model as a random effect which allowed different y-axis intercept for each site in each regression model. For modelling the mean of the response variable:

i) The Beta model with logit link was chosen for response variables which were proportions (percentages). This ensures that all responses stay within 0 and 1 (or 0 and 100%).

ii) The lognormal or gamma models with canonical link (log and inverse respectively) were used according to the shape of the response variable and the Akaike Information Criterion (AICc) produced. A canonical link is one which puts the mean of the response variable on the scale of the linear predictor, or in other words, allows a simple interpretation.

iii) In rare instances, a linear mixed model was able to be estimated because the response variable resembled a normal distribution.

2. If there were more than about 8 or more predictor variables, variable selection was performed with a Random Forest Technique (Genuer et al., 2015), and all the variables associated with explanatory or predictive power along with the next few variables in decreasing order of performance by this technique, were included in the first model. Automatic model testing based on the corrected AICc was used to narrow the model choice to a few models. A limited amount of forward and backward stepwise (variable) elimination was undertaken in between these steps. To make the most parsimonious model (model with the fewest, most important terms), variables that had an estimate of zero or nearly zero for their coefficient were removed from the model. Some would criticise this, as it does tell us that a variable has no apparent effect, which is of course informative. Nevertheless, in order to make the smallest, most meaningful model, these variables have been removed along with the variables that are “not significant” in terms of their contribution to information used in the model.

3. The initial attempt at parameter estimation involved functions performing automatic differentiation using the Laplace approximation. Whether or not this function’s algorithm converged allowing parameter estimation, an attempt was made with the Monte Carlo Markov Chain (MCMC) technique to estimate the same parameters.

CHAPTER 3: Hydrology

Research highlights

- Impeding stormflow with detainment bunds increased residence time and facilitated soil infiltration.
- Surface runoff reaching downstream waterways decreased by 43% and 63%.
- Soil infiltration rates in ponding areas decreased due to repetitive ponding.
- The novel mitigation strategy is likely to decrease sediment and nutrient loading downstream.

3.1 Introduction

Storm generated surface runoff drives contaminant losses from pastoral catchments that cause water quality degradation in receiving waters. Areas where precipitation patterns are variable, with very wet winters, and dry summers interspersed with large storms, contribute to hydrological conditions that create challenges for nutrient management and loss mitigation (McDowell et al., 2013). Wetter soils often have a greater potential for surface runoff than drier soils (McDowell et al., 2004), although hydrophobicity in very dry soils may contribute to high rates of surface runoff (McDowell & Sharpley, 2002). Furthermore, climate change is projected to cause more dramatic hydrological conditions that result in greater runoff and associated nutrient losses from agricultural areas (Ministry for the Envrironment, 2019; Ockenden et al., 2016).

In temperate areas, processes in headwaters dominate the response of surface water catchments to rainfall (Ockenden et al., 2016). Headwaters, including ephemeral streams, are estimated to contribute 70% of the mean annual water volume to second-order streams and 55% of higher order rivers (Alexander et al., 2007). Headwater sub-catchments have been found to be responsible for the majority of the streamflow and hydrochemical responses to storms in a stream network (Bieroza et al., 2018).

A range of mitigation strategies have been developed to target different hydrological pathways to address nutrient losses from pastures (McDowell et al., 2013). Edge of field mitigation strategies that increase stormflow residence time have been found to decrease surface runoff flows and facilitate sediment deposition by decreasing the kinetic energy of flowing water (McKergow et al., 2007). Since nutrient loads transported by runoff are the product of the volume of runoff and the concentration of contaminants, mitigation strategies that decrease runoff volumes are likely to mitigate associated nutrient losses (Gburek & Sharpley, 1998). While soil infiltration in the flow pathway of surface runoff has been found to reduce contaminant loads, the impacts of specific mitigation strategies on surface runoff volumes have rarely been reported (McKergow et al., 2007).

Common edge of field mitigation strategies that address surface runoff include riparian buffer strips and stormwater detention areas (SDAs) such as sedimentation ponds and constructed wetlands. Stormwater detention areas are commonly used for flood protection, but are increasingly being used for water quality improvement in agricultural and urban settings (Shukla et al., 2017; Stanley, 1996). Agricultural SDA is a collective term used for natural or manmade depressions, ponds, and reservoirs (Shukla et al., 2017). Due to the nonuniformity in location and design, and the varying approaches used to investigate their mitigation performance, studies on SDAs are often not directly applicable to one another. However, the processes for contaminant retention and treatment for the varying SDA applications are similar, such as soil infiltration, sedimentation, plant uptake, soil adsorption, and microbial uptake (Shukla et al., 2017).

Detainment bunds (DBs) impede stormflow and increase surface runoff residence times with earthen, stormwater retention structures that form temporary ponds (Fig 3.1). Detainment bunds are a type of SDA utilised in the Lake Rotorua catchment, in New Zealand, that were developed to address phosphorus (P) loss starting in 2010 (Clarke, 2013). A design feature that sets DBs apart from other stormwater detention structures is the incorporation of an 'upstand riser' that is connected to an outlet pipe that passes through the bund and discharges ponded runoff on the downstream side of the DB (Fig. 3.1**)**. The upstand riser is a ~1 m diameter vertical pipe reaching to ~20 cm below the lowest point of the DB, installed near the bund at the low point of the ponding area. Ponded water is discharged from the outlet pipe if the pond height exceeds that of the riser. During large runoff events that overwhelm the pond storage capacity and discharge rates from the upstand riser, water may also be discharged via an 'emergency spillway' at the lowest point of the bund. The surface of the emergency spillway is protected by either a mat material, compacted substrate, or stable grass cover. An outlet valve connected to the upstand riser at ground-level is able to be unplugged to rapidly drain the pond. Although ryegrass-based pastures in New Zealand have some tolerance to saturated soil conditions, early adopters of the DB strategy restrict the inundation period to a maximum of 3 days to reduce the risk of pasture damage in the ponding area (Paterson & Clarke, 2013).

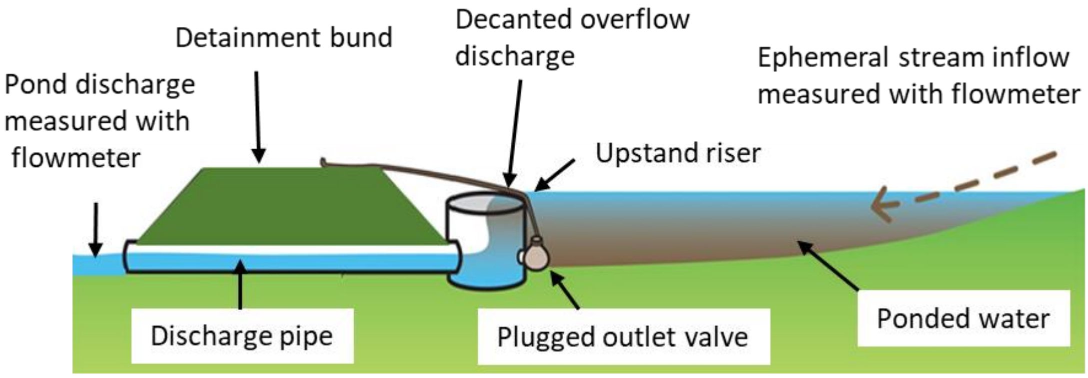

Figure 3.1: Cross-section of ponding area showing the ephemeral stream inflow ponding behind a detainment bund. If the pond height exceeds the height of the upstand riser then ‘decanted overflow’ is discharged via a pipe passing through the bund wall. Inflow and discharges are measured with flowmeters.

In the Lake Rotorua catchment, DBs may be approximately 1.5-2 m high and 20-80 m long, and are constructed on pastures across the flow path of targeted low-order ephemeral streams. A DB can be purpose-built, or constructed by modifying an existing structure, such as a raised raceway that divides a paddock. The local DB site selection and design protocol promotes a minimum ratio of 120 m^3 pond volume capacity per 1 ha of contributing catchment area, and a 10,000 m^3 maximum pond volume due to local regulatory requirements (Paterson & Clarke, 2013). Because of design and regulatory limitations, some landscapes such as steep hill country with incised valley floors, and flat flood plains, are not appropriate for DB locations due to their topography (Paterson, 2019).

Because hydrological factors play a critical role in determining the impacts of agriculture on water quality, it is essential to develop a thorough understanding of the effect the DB strategy has on surface runoff hydrology. A preliminary study of 3 non-consecutive ponding events at a single DB site in the Lake Rotorua catchment reported that discharge volumes were 30-67% lower than event inflow volumes (Levine et al., 2019). While the previous study serves as a proof of concept, there is currently no definitive research quantifying the impact of the DB strategy on surface runoff volumes discharged from pastoral landscapes.

The main objective of this present study was to measure the effect of DBs on annual surface runoff volumes at 2 sites in the Lake Rotorua catchment. We also aimed to identify factors that influence the hydraulic performance of DBs. Due to the prevalence of well-drained soils in pastures in the catchment which could allow for significant infiltration, we hypothesised that increasing surface runoff residence time by impeding stormflow will facilitate soil infiltration during the ponding period and decrease runoff volumes discharged from the catchment. If DBs are capable of decreasing the volume of surface runoff leaving pastures by facilitating soil infiltration, particularly during large storm events, we expect that the strategy would also decrease nutrient and sediment loads delivered to surface waters downstream of the DB catchment. Investigations into effect of the DB strategy on nutrient and sediment loads are recorted on in subsequent studies. Because to the importance of hydrology on contaminant losses from pastures contributing to water quality degradation, the results of this present study could offer insight into whether DBs could be an effective nutrient mitigation strategy in the Lake Rotorua catchment and other areas where DBs could be located.

3.2 Materials and methods

3.2.1 Site description

Lake Rotorua in the North Island of New Zealand is culturally recognised as a 'taonga', or treasured natural resource, that provides valuable ecosystem services (Land and Water Forum, 2010). Anthropogenic nitrogen (N) and P loading has caused ecological degradation, eutrophication, and toxic algal blooms in the lake since the 1960's (Burns, 2001). Large storms are responsible for significant portions of the annual runoff and nutrient loads delivered to streams (Abell et al., 2013; Dare, 2018).

Dairy stock are typically grazed on pastures throughout the year in the temperate maritime climate of New Zealand. Pastoral dairy and drystock farms cover ~48% of Lake Rotorua's 42,000 ha surface catchment, and contribute 67% of the total N and 43% of the total P loads to the lake (Bay of Plenty Regional Council, 2012). The 2012 Lake Rotorua Nutrient Management Plan has set water quality targets to reduce anthropogenic loads by 320 t N y^{-1} and 10 t P y^{-1}, respectively (Bay of Plenty Regional Council, 2012). Various in-field and edge of field nutrient mitigation options, including novel approaches, are required to achieve nutrient load reduction targets (McDowell, 2010).

Two DB sites located on pastoral dairy farms in the north-western portion of the Lake Rotorua catchment were monitored during this 12-month study (Fig. 3.2). A digital elevation model derived from LiDAR data (2 m resolution) was used to identify appropriate locations to construct the DBs, measure the contributing catchment area, the area downstream of the inflow monitoring site, and the pond areas, and determine pond slopes. Site selection criteria for this study stipulated that a single main ephemeral stream delivered runoff to the DB ponding area in a manner that allowed for accurate measurements of inflow volumes. The 2 DBs varied in catchment size but had similar pond storage volume to catchment size ratios (Table 3.1).

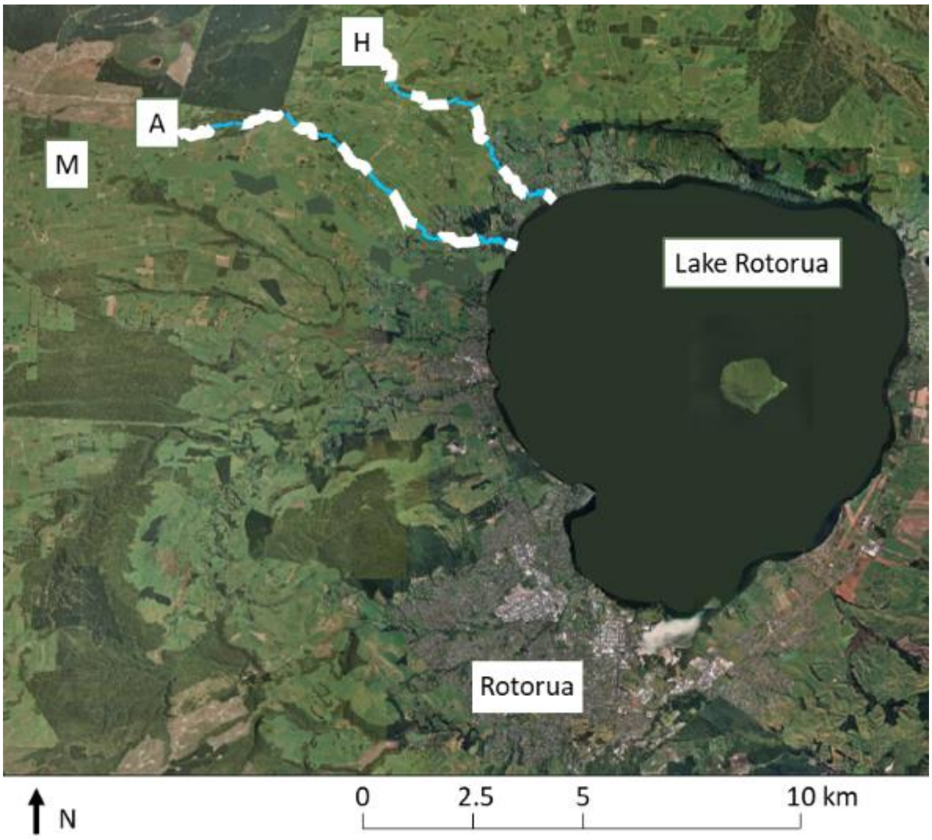

Figure 3.2: Map of Lake Rotorua with study sites labelled with initials Awahou (A) and Hauraki (H) and the Bay of Plenty Regional Council climate monitoring site at Oturoa Road (M). White and blue dashed lines show path of surface runoff from detainment bund sites to Lake Rotorua.

Table 3.1: Characteristics of detainment bund (DB) sites.

Characteristic		Hauraki	Awahou
Grid Reference		38°00'21"S 176°11'03"E	38°01'43"S 176°07'54"E
Year DB constructed		October 2011	June 2012
Topography of catchment		Flat, rolling and hill	Mainly rolling
Percentage of catchment with slope (%)	0°-7.9°	69	69
	8.0°-15.9°	16	19
	16°-25.9°	9	9
	>26°	5	3
Size of DB entire DB catchment (ha)		55.0	19.7
Area of DB catchment downstream of inflow monitoring (ha)		8.3	1.8
Height of bund at spillway (m)		1.56	1.80
Height of upstand riser (m)		1.36	1.60
DB pond volume ($m^{3)}$		4,894 m^3 at upstand riser 7,110 at m^3 at spillway	1,652 m^3 at upstand riser 2,244 m^3 at spillway
Ratio of pond volume: catchment area (m^3: ha)		89:1 at upstand riser 129:1 at spillway	84:1 at upstand riser 114:1 at spillway
Pond area at pond filled to upstand riser and spillway (m^2)		9,564 m^2 at upstand riser 12,221 m^2 at spillway	2,610 m^2 at upstand riser 2,940 m^2 at spillway
Average slope of ponding area (degree)		0.76°	1.64°

3.2.2 Soil characteristics

The 2 DB study catchments were in the Mamaku region of the Lake Rotorua catchment, which has relatively coarse textured, volcanic tephra soils (Landcare Research, 2017). The Mamaku Ignimbrite soils found in this region were deposited 220-230 thousand years ago during the formation of the Rotorua Caldera (Milner et al., 2003).

At the Hauraki site, soils in the ponding area are in the Oropi series, which is classified as Vitric Hapludand in the USA soil classification system, and Buried-allophanic Orthic Pumice in the New Zealand soil classification system. These soils have a dark brown, loamy sand topsoil which overlays a dark yellow-brownish sandy loam, on a layer of yellow-brown silt loam (Rijkse & Guinto, 2010).

At the Awahou site, the soils in the ponding area are in the Waiteti series, which is classified as Andic Haplohumod in the USA soil classification system, and Typic Orthic Podzols in the New Zealand soil classification system. These soils have a dark reddish-brown friable loamy sand, over dark brown loamy sand, which in turn is over a yellowish-brown loamy sand on sand (Rijkse & Guinto, 2010).

The Oropi and Waiteti soils are both classified as Hydrologic Soil Group A, which are free draining and very permeable soils even in the slowest horizon with infiltration rates measuring greater than 72 mm/h (Rijkse & Guinto, 2010).

3.2.3 Equipment and monitoring

Field monitoring was conducted from 1 December 2017 to 30 November 2018. Rainfall was measured at each site using UNIDATA® (Willetton, Western Australia) 6506B tipping bucket (0.5 mm) rain gauges. Pond heights were continuously measured with ENVCO® (Auckland, New Zealand) PT12 pressure transducers installed near the base of the upstand riser. Flowmeters (UNIDATA® 6527 Starflow® QSD), which were fitted to pipes at both DBs, were used to measure inflows and discharges, with the exception of inflow measurements at the Hauraki site, where a 160° V-notch weir and float/counterweight for height measurement was deployed. Inflow (i.e. upstream) monitoring occurred at elevations high enough to avoid inundation by the pond. Rainfall, pond height and flow rates were collected at 5-minute intervals, and stored using serial digital interface communications linked to telemetered UNIDATA® Neon® 2013 F 3G External Memory Metering Module data loggers. Standard quality controls were followed for calibration and maintenance of the monitoring equipment (NIWA, 2004).

Additional historical rainfall data was collected from the Bay of Plenty Regional Council's climate monitoring site at Oturoa Road (550 m above sea level, 3 km away from the Awahou site and 7 km from the Hauraki site) (Fig. 3.2) (Bay of Plenty Regional Council, 2018). These data were used to calculate the average seasonal rainfall pattern over the 10 years prior to the study.

Soil moisture for the Hauraki site was measured at 5-minute intervals on a paddock ~650 m from the bund, using a soil moisture probe (Model AOS220A-20, AovicTech, Beijing, China) inserted 15 cm into the soil at 45°. Soil moisture data for the Awahou site was obtained from the Oturoa Road Bay of Plenty Regional Council monitoring site (Fig. 3.2), where soil moisture percentage was measured at 15-min intervals in the top 25 cm of soil using an Aquaflex (Christchurch, New Zealand) SI.99

soil moisture and temperature sensor (Bay of Plenty Regional Council, 2018). The soil moisture data was used to calculate the average soil moisture percentage in the 24 h period prior to each runoff event at each site (Deasy et al., 2009).

Evaporation data was calculated using the Penman method and climate data, measured ~17 km from the study sites (-38.146 S, 176.258 E) (NIWA, 2019). The maximum hourly evaporation rate over any 72-h period was 0.3 mm/h during this study period, suggesting that evaporation from the ponds was negligible, and was therefore not considered in calculating water lost from the pond.

3.2.4 Event types

Event types were differentiated according to the mode(s) in which ponded water was discharged from the DB. ‘Overflow Events’ occurred during larger runoff events when inflow continued to be delivered to the pond after the pond height exceeded the height of the upstand riser, generating ‘overflow discharge’ (Fig. 3.1). After 3 days of ponding, any residual ponded water was evacuated when the outlet valve was opened, creating ‘release discharge’. Therefore, ‘Overflow Events’ had both ‘overflow discharge’ and ‘release discharge’ components (Fig. 3.3). In contrast, ‘Non-overflow Events’ were smaller storms that did not contribute enough runoff to overtop the upstand riser. Non-overflow Events included events which, at the end of the 3-day treatment period, either had a portion of ponded runoff to discharge by opening the outlet valve, or had no runoff to discharge due to leakage and infiltration into the soil. Throughout all ponding events, ‘leak discharge’ from an intractable leak at the connection point of the outlet pipe and the base of the upstand riser generated a constant measured flow of ~2-4 m^3/h at both sites. Attempts at sealing this leak during the study period were unsuccessful.

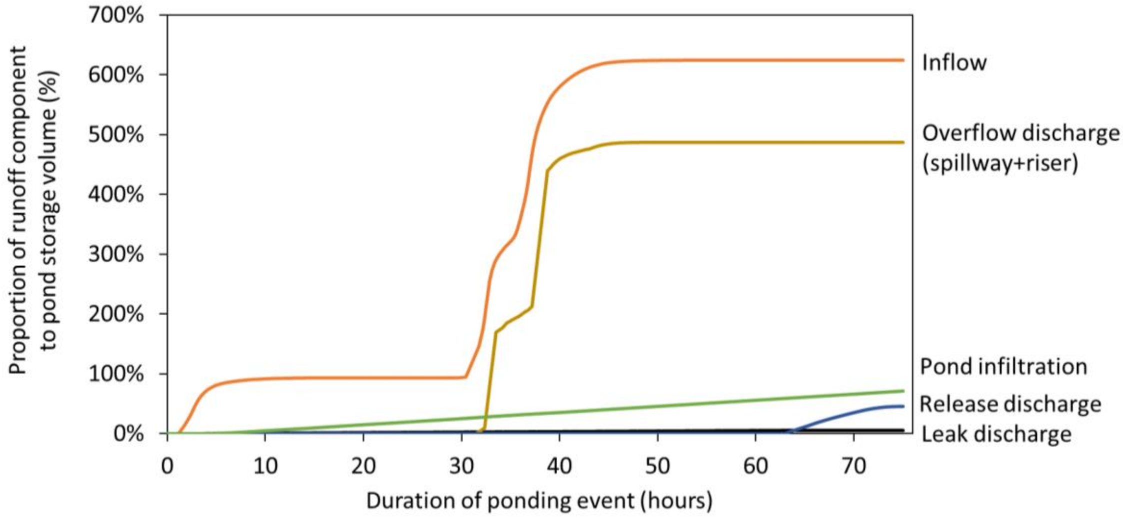

Figure 3.3: Example of the proportion of the different runoff components including inflow, overflow discharge (combining spillway and riser), release discharge, soil infiltration occurring in ponding area, and leak discharge, as a percentage of the detainment bund pond storage volume, throughout the duration of a typical Overflow Event.

3.2.5 Runoff data analysis

Inflow and discharge volumes were calculated by multiplying the measured flow rate by the time elapsed between flow measurements (Harmel et al., 2003). Event inflow volumes were corrected on a pro rata basis (increased by 15% at the Hauraki site, 9% at Awahou) to account for the small catchment area not measured by the inflow flowmeter between the inflow monitoring point (i.e. upstream) and the DB (Table 3.1).

Volume measurements, including inflow, infiltration and discharges are expressed as yields (mm) i.e. volume per unit of contributing catchment area. The percentage of rainfall occurring as inflow was calculated by dividing the total rainfall, measured onsite with a rain gauge, by the inflow yield. Overflow discharge from the upstand riser, release discharge (which occurred during Overflow Events and some Non-overflow Events) and leak discharge (throughout ponding during all events) were measured at the outlet pipe (Fig. 3.1).

For all events apart from those with emergency spillway discharge, the infiltration volume was determined by subtracting the measured leak volume, and upstand riser overflow and release volumes if applicable, from the event inflow. Event infiltration rates (mm/h) were calculated by dividing the infiltration yield (mm) by the duration of event ponding.

The emergency spillway was breached during the 2 high runoff magnitude Overflow Events that occurred at both sites. Since both the spillway discharge and soil infiltration were unmeasured, the event average soil infiltration rates were calculated for each of the Overflow Events to determine the volume discharged over the spillway for each respective event. During these events, whenever the water level was below the spillway height, it was possible to calculate the infiltration rate using; inflow and discharge volumes measured by the flowmeters, and the change in the volume of water being held in the pond. These calculations were performed on an hourly time-step. The change in the volume of water in the pond was calculated using pond height data which were used to determine the inundated ponding area. The average pond inundation area during each time-step was the mean value of the area inundated by ponded water, calculated at five-minute intervals, using measurements of pond heights and the slope of the ponding area. The slope of the ponding area was determined using LiDAR data and GIS software that calculated the ponding area at various elevations. The equation used to find the slope of the pond was integrated to determine the pond volume at various pond heights measured at the lowest point of the pond. The calculated infiltration rate was applied to the period during the breach of the spillway, thus enabling a calculation of the entire event infiltration volume. The spillway volume was then determined by subtracting the measured leak, upstand riser overflow, and release discharge volumes, and the calculated infiltration volumes, from the event inflow.

During the Overflow Event at the Awahou site on 28th April 2018, surface runoff bypassed the inflow monitoring station due to flows overwhelming and flanking the inlet pipe. The ‘MissForest’ package (Stekhoven & Bühlmann, 2012) was used to run a ‘random forest’ model 1000 times using R version 3.6 (R Development Core Team, 2019), to estimate the inflow volume during this event. The model utilised the following data from both sites for each ponding event to determine the inflow volume of this event: average soil moisture in the 24 hours prior to inflow delivered to the DB, duration of rainfall, total rainfall, average rainfall intensity, duration of rainfall intensity greater than 6 and 12 mm/h, maximum rainfall intensity, and the runoff volume delivered to the DB. The minimum and maximum 95% confidence interval values of the predicted values were within 92% of the mean value used as the volume for the event.

3.2.6 In-field soil infiltration measurement

Soil infiltration rates were collected in the field at the 2 DB sites using 40-cm diameter stainless-steel rings charged with water approximately 50 mm high, on 16th September 2019. Seven replicate rings were used to measure the infiltration rates near the upstand riser in the lowest portion of the ponding area, as well as outside of the ponding area upstream of the DB.

3.3 Results

Annual rainfall in the monitor year was 2,162 mm at the Hauraki site, and 2,098 mm at the Awahou site (Fig. 3.2). Annual rainfall at the sites were very similar to the 10-year average of 2,197 mm y^{-1} measured at a nearby climate monitoring station (Bay of Plenty Regional Council, 2018), although the seasonal rainfall distribution during this study varied slightly from the 10-year seasonal averages (Fig. 3.4). The summer of the study year had the greatest seasonal rainfall total at both sites, which differed from the 10-year average (Fig. 3.4). Local rainfall can vary substantially between Lake Rotorua subcatchments, although both sites showed similar seasonal rainfall patterns during the study year (Abell et al., 2013).

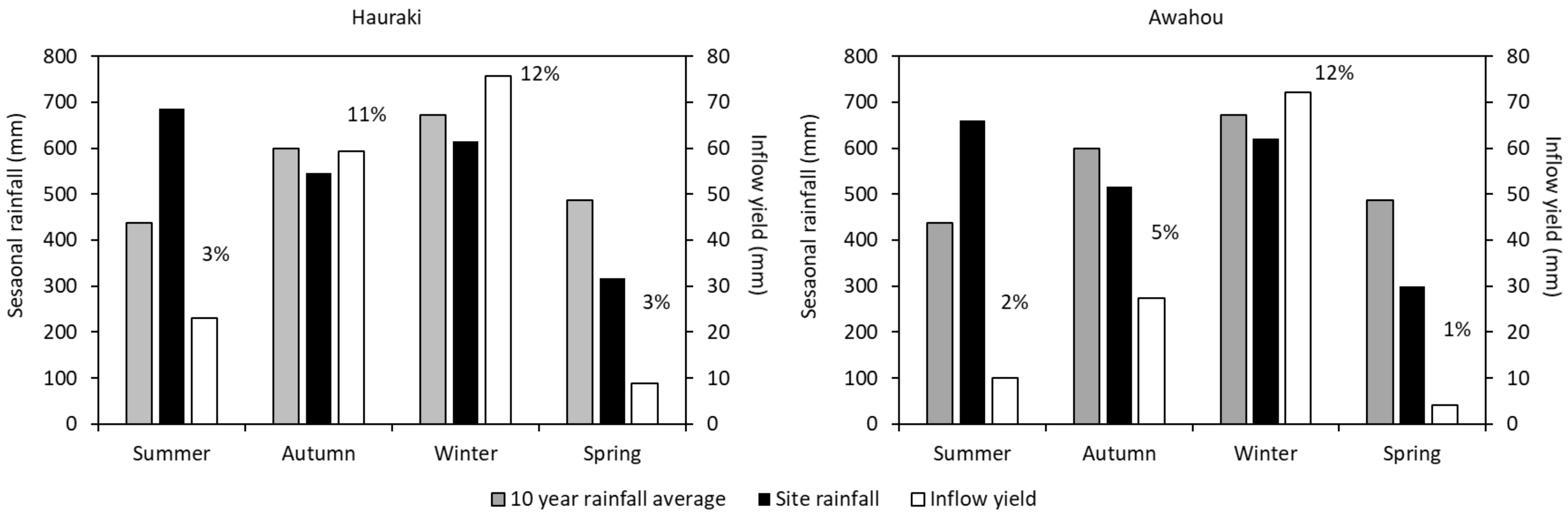

Figure 3.4: A comparison of the 10 year (2007-2017) average seasonal rainfall measured at the Bay of Plenty Regional Council's climate monitoring site at Oturoa Road, to the cumulative seasonal rainfall measured at the study sites from 1 December 2017 to 30 November 2018. Also, cumulative seasonal inflows of surface runoff measured as seasonal inflow, and the percentage of seasonal rainfall occurring as runoff (%) are shown.

The total annual inflow yield at the Hauraki site was 167 mm (91,801 m^3) compared to 114 mm (22,404 m^3) at the Awahou site (Fig. 3.5). These measurements show that only a small fraction of the annual rainfall generated significant surface runoff (8% at the Hauraki site, and 5% at the Awahou site). Seasonal inflow yields, and the percentage of rainfall occurring as runoff, were greatest during the winter period at both sites (Fig. 3.4). The major difference in the seasonal runoff patterns between the two sites during the study period was the greater proportion of autumn rainfall delivered as runoff to the Hauraki site (11%) compared to the Awahou site (5%).

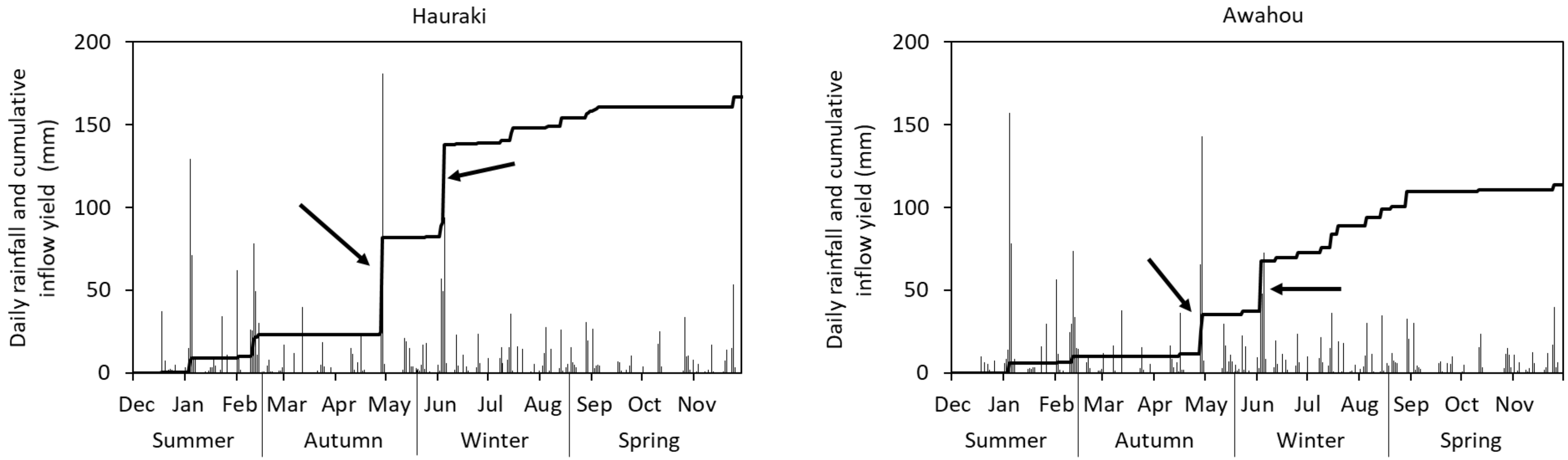

Figure 3.5: Daily rainfall totals (mm) (bars) and the cumulative inflow yield (mm) (line) over the duration of the 12-month study at both sites. Arrows point to high runoff magnitude Overflow Events occurring on the same dates at both sites. Note: austral seasons are labelled with corresponding months.

Storm generated surface runoff resulted in 18 ponding events at the Hauraki site, and 19 ponding events at the Awahou site. However, not all of the measured inflow contributed to measurable ponding events. On occasion, inflow yields were relatively small and presumably the soil was sufficiently dry to accommodate this inflow without generating measurable ponding, and the entire inflow was considered to infiltrate the soil. Ponding events occurred most prevalently during the winter months (~50% of the events at both sites) compared to the other seasons (Table 3.2). Two high runoff magnitude Overflow Events occurred at both sites during the same storm events (Fig. 3.5).

Table 3.2: Number of each type of ponding event occurring during each austral season at the two study sites (H= Hauraki site, A= Awahou site).

		Site	Number of Overflow Events	Number of non-Overflow Events	Total number of events
Summer	Dec-Feb	H	0	5	5
		A	0	3	3
Autumn	Mar-May	H	1	1	2
		A	1	2	3
Winter	June-Aug	H	1	8	9
		A	1	9	10
Spring	Sept-Nov	H	0	2	2
		A	0	2	2
Total number of events		H	2	16	18
		A	2	17	19

The total annual discharge yield from the Hauraki DB was 116 mm (62,969 m^3), and 65 mm (12,807 m^3) at the Awahou DB. The ~50 mm difference occurring between annual inflows and discharges at both sites was attributed to soil infiltration occurring in the ponding area. Leak discharges accounted for 5% (6 mm = 3,221 m^3) of the total annual discharge from the Hauraki site and 26% (17 mm = 3,267 m^3) at the Awahou site.

The calculated event infiltration rates ranged from 5 to 24 mm/h at the Hauraki site, and from 3 to 16 mm/h, at the Awahou site (Fig. 3.6). The calculated mean event infiltration rates were 13 mm/hr and 9 mm/hr at the Hauraki and Awahou sites, respectively.

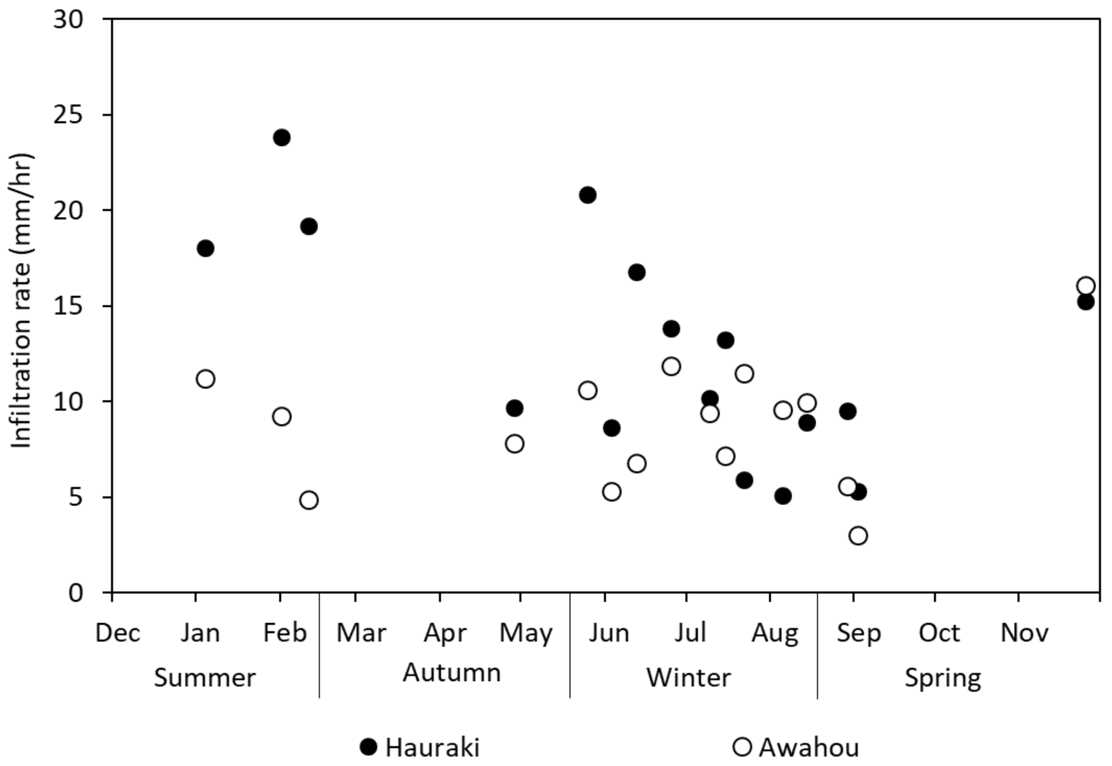

Figure 3.6: Calculated infiltration rates for each ponding event during the 12-month study at both sites, determined by the event yield infiltrating the soil in the ponding area and the duration of the ponding event. Note: austral seasons are labelled with corresponding months.

Field measured infiltration rates collected adjacent to the upstand riser in the lowest portion of the ponding area fell within the range of calculated event infiltration rates (Table 3.3). Interestingly, the mean measured infiltration rates outside the ponding area exceeded both the measured inside the ponding area, and the calculated event maximum infiltration rates for both sites (Fig. 3.6, Table 3.3). The mean calculated event infiltration rate, and mean field measured rates inside the ponding area, were greater at the Hauraki site than the Awahou site, while measured infiltration rates outside the ponding area were very similar between the 2 sites.

Table 3.3: Mean infiltration rates measured adjacent to the upstand riser in the lower portion of the ponding area, and at an area outside the ponded area at both sites. The standard deviation surrounding the means are reported in brackets.

Site	Infiltration in the ponding area (mm/h)	Infiltration outside the ponding area (mm/h)
Hauraki	19 (14)	36 (15)
Awahou	12 (7)	37 (11)

Two high runoff magnitude Overflow Events occurred during the study period at each site in which ‘overflow discharge’ was generated when the pond height exceeded the height of the upstand riser and then the emergency spillway (Fig. 3.1, Table 3.2). The combined inflow yield of these two Overflow Events was 114 mm (62,938 m^3) at the Hauraki site, and 54 mm (10,571 m^3) at the Awahou site, which accounted for 69% and 47% of the total annual inflow at the sites, respectively. The combined overflow discharge yields (riser overflow and spillway discharges combined) at the Hauraki site was 92 mm (50,579 m^3) at the Hauraki site, and 37 mm (7,367 m^3) at the Awahou site, which accounted for 80% and 58% of the total annual discharge from the DBs. Infiltration yields during Overflow Events were 16 mm (8,650 m^3) at the Hauraki site, and 12 mm (2,335 m^3) at the Awahou site, accounting 14% and 22% of the Overflow Event inflow yields at each site, respectively.

The fate of discharged runoff must be considered when evaluating the effect of DBs on the proportion of surface runoff inflows reaching downstream surface waters. It is unlikely that any of the overflow discharge (riser overflow and spillway discharges combined) infiltrated the soil downstream of the DBs, since runoff was still being generated in the catchments during this discharge period. Furthermore, any leak discharge generated while runoff was being generated in the catchment was also likely to have reached downstream surface waters. In contrast, discharges, including leakage after runoff generated in the catchment had ceased, and the release discharge generated when the pond outlet valve was opened after the prescribed 3 day period, were expected to infiltrate the well-drained soils downstream of the DB before connecting with a downstream surface waterway.

At the Hauraki site, 21 mm (11,622 m^3), or 18% of the annual discharge yield was expected to infiltrate the soil downstream of the bund, and 23 mm (4,603 m^3), or 36% of the annual discharge yield was expected to infiltrate the soil downstream of the Awahou site. The combined infiltration occurring during the ponding period and downstream of the bund is likely to have prevented 73 mm (40,324 m^3) of surface runoff from reaching downstream surface waters at the Hauraki site, and 72 mm (14,118 m^3) at the Awahou site. The total surface runoff likely to be prevented from reaching downstream surface waters equated to 43% of the inflow at the Hauraki site, and 63% at the Awahou site (Fig. 3.7).

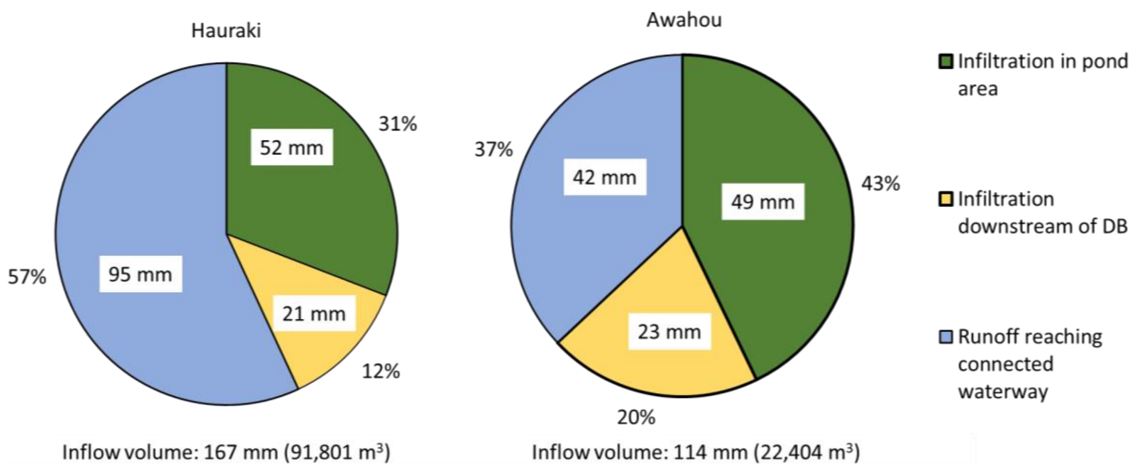

Figure 3.7: Yield (mm) and proportion of annual inflow infiltrating the soil in the ponding area, infiltrating the soil downstream of the detainment bund (DB), and likely to reach surface waters downstream of the bund.

3.4 Discussion

The major finding of this study was that the DBs prevented 43% and 63% of the inflow runoff from reaching surface waters downstream of the Hauraki and Awahou sites, respectively. The results of this study show, for the first time, that in a typical year in the Lake Rotorua catchment, DBs can substantially decrease surface runoff volumes by impeding stormflow and facilitating soil infiltration by increasing residence time on sufficiently permeable soils. Soil infiltration occurs during the ponding period (typically 3 days) and when water that is discharged from the DB infiltrates dried soil downstream of the bund. The results of this study positively support the hypothesis set out in section 3.1 and demonstrate, for the first time, that in a typical year in the Lake Rotorua catchment, DBs can substantially decrease surface runoff volumes by impeding stormflow, which facilitated soil infiltration by increasing surface runoff residence time on sufficiently permeable soils.

Related studies have found that mitigation strategies that are capable of decreasing surface runoff volumes also reduce nutrient loads transported by surface runoff (Harper et al., 1999; McKergow et al., 2007). The results of this study suggest that by decreasing surface runoff discharges, DBs in the Lake Rotorua catchment are likely capable of decreasing sediment and nutrient loads transported by surface runoff from pastures, and should be considered a viable stormwater mitigation strategy in the catchment.

Given differences between the study site geomorphologies, and likely varying management factors and localised differences in storm rainfall intensity and duration, the results reported in this study for the 2 sites are surprisingly consistent. The difference in annual inflow yields and the percentage of rainfall as runoff between sites [167 mm delivered (8% of rainfall) to the Hauraki DB, and 114 mm delivered (5% of rainfall) to the Awahou DB], is at least partially explained by the greater total rainfall that occurred during the high runoff magnitude Overflow Events at the Hauraki site. During the two Overflow Events, which occurred on the same dates at each site, 423 mm of rainfall occurred resulting in 114 mm of inflow at the Hauraki site, compared to 369 mm of rainfall resulting in 54 mm of inflow at the Awahou site. The 56 mm difference in rainfall contributed to the 60 mm difference in inflow yields during the Overflow Events, and the 53 mm difference in annual inflow yields between the sites. The proportion of annual inflows delivered during the rare high magnitude Overflow Events was 69% at the Hauraki site, and 47% at the Awahou site, which is consistent with other studies that found rare high magnitude storm events contribute the majority of annual runoff, and associated sediment and nutrient loads delivered to streams in the Lake Rotorua catchment (Abell et al., 2013; Dare, 2018).

The Overflow Events at the 2 sites were also responsible for the differences in the proportion of the annual inflow yields undergoing soil infiltration before reaching downstream surface waters. Due to the greater rainfall contributing to greater inflows at the Hauraki site, 67% of the cumulative Overflow Event inflow yield went over the upstand riser and spillway as overflow discharge at the Hauraki site, compared to 47% at the Awahou site. These overflow discharges accounted for 80% of total annual discharge at the Hauraki site and 58% at the Awahou site. Since we assumed infiltration did not occur downstream of the DB while runoff was still being delivered to the pond, all of the overflow discharge is likely to have reached downstream waterways. This highlights the importance of locating and constructing DBs to maximise the pond volume: catchment ratio in order to avoid excessive overflow discharges which were generated when enough runoff was delivered to the DB and the pond height exceeded that of the upstand riser and emergency spillway.

Still, the DBs prevented 19% and 29% of the Overflow Event inflow from reaching downstream waterways from the Hauraki and Awahou site, respectively. The results showing that a smaller portion of the inflow yield escaped as overflow discharge from the Awahou DB compared to the Hauraki DB, despite the similar ratios for pond volume to catchment area, suggest that the potential magnitude and intensity of large storm events, and subsequent runoff, are important to consider when sizing and locating a bund. These results also suggest that the size of very large storm events have implications for the effectiveness of DBs as a mitigation tool used to address sediments and nutrients transported in surface runoff, since greater contaminant loads have been found to be transported from pastures during greater magnitude runoff events in New Zealand (Cooke & Dons, 1988; Smith & Monaghan, 2003), and the Lake Rotorua catchment specifically (Abell et al., 2013; Dare, 2018).

The relative importance of the size of pond storage versus the infiltration rate in the ponded area is clearly illustrated in Figure 3.3 where the overflow discharge is seen to be much greater than soil infiltration. These results suggest that the currently suggested minimum 120 m^3 pond capacity: 1 ha contributing catchment ratio could be increased in order to avoid excess overflow discharge and increase the potential for ponded and released water to infiltrate the soil. However, it is important to consider the costs and benefits of the DB strategy before revising the protocol, since such measures could significantly limit the relatively small number of viable DB locations in the Lake Rotorua catchment (~300 DBs with 120 m^3: 1 ha ratios (Paterson, 2019)), and restrict the potential benefits of their implementation.

Multiple factors could be responsible for the seasonal trend observed between soil infiltration rates being lower during colder months, including stocking rates and timing, as well as the potential for macropores to develop during the warmer, drier seasons. The influence water temperature on viscosity could also contribute to lower soil infiltration rates in the winter months. The viscosity of water decreases by 41% when decreasing temperature from 25 to 5 C° (Korson et al., 1969), which is roughly the range of temperatures observed in the ponded water at the DB sites during this study. The influence of seasonal and diurnal temperature changes on infiltration rates have been observed in studies investigating stormwater management (Emerson & Traver, 2008; Jaynes, 1990). Because infiltration rates affect DB discharge volumes, the influence of temperature on infiltration rates should be considered when developing DB design protocols, and when developing models used to estimate DB performance. This is particularly important in climates where the majority of annual runoff occurs in winter, when soil infiltration rates are likely to be the slowest due to greater water viscosity.

There was good agreement between the calculated event infiltration rates and field measured infiltration rates (Table 3.3 and Fig. 3.6). Both of these rates were considerably smaller than the >72 mm/h value reported by Rijkse and Guinto (2010) when investigating soil types in the region, and likely reflects the effect of treading damage under intensive dairying on soil infiltration rates (McDowell et al., 2003). The decline in soil infiltration rates in the lower portion of the ponding area compared to rates outside of the ponding area might also be due to increased treading damage in the lower-lying and wetter ponding areas (Curran Cournane et al., 2010), or perhaps the large volumes of water that infiltrated the soil and caused deterioration in soil structure in the ponding area. Furthermore, given that the DBs in this current study have been operating for 6 years, and that during the course of this 12-month study approximately 789 kg and 1,280 kg of sediments were deposited at the Hauraki and Awahou ponding areas, respectively (Chapter 4), deposited sediments may have formed a less permeable layer on top of existing soils, and/or be clogging soil pores, and reducing infiltration rates. Soil pore clogging by sediments has been observed in studies investigating the effects of sedimentation on soil infiltration rates (Rice, 1974) and soil filter permeability (Reddi et al., 2000). This theory is further supported by the greater decrease in the infiltration rates in the Awahou ponding area, where greater quantities of sediments were deposited in a smaller ponding area. The results of this current study suggest DBs may facilitate greater infiltration during the first few years after construction, however infiltration rates may decline over time. Therefore, the effect repetitive ponding on the long-term ability of DBs to decrease surface runoff volumes should be the subject of future research to further inform the development of DB design protocols and performance models. Strategies to remediate soil infiltration rates in the ponding area should also be investigated.

This study identified factors that will vary spatially and temporarily which affected the proportion of runoff infiltrating the soil before reaching surface waters downstream of DBs including: precipitation patterns, particularly the magnitude of Overflow Events; soil permeability and changes to infiltration rates due to sedimentation and repetitive ponding; and the catchment size:pond volume ratio. This study should be expanded to collect longer-term data from more DB locations to investigate ways to maximise the proportion of surface runoff infiltrating soils since decreasing runoff discharges contribute to lower contaminant discharges. Lastly, results from this current study and future studies should be used to develop models that estimate runoff and contaminant yields delivered to, and treated by DBs in specific locations, based on hydrologic and landscape conditions. The development of these models will help decision makers determine the applicability of DBs as a stormwater mitigation strategy in their catchments.

3.5 Conclusion

This 12-month study found 2 detainment bunds prevented 43% (at the Hauraki site) and 63% (at the Awahou site) of the annual surface runoff from reaching surface waters downstream of targeted pastoral areas in the Lake Rotorua catchment. The detainment bunds effectively decreased surface runoff volumes by impeding stormflow, which increased runoff residence time and facilitated soil infiltration during the ponding phase, and when the ponded runoff was released onto what was dried soils downstream of the detainment bunds. By decreasing surface runoff discharges and associated contaminant loads, the detainment bund strategy should be considered a viable nutrient mitigation strategy in places where stormflows contribute to water quality degradation and soils are sufficiently permeable.

The study found that the magnitude of surface runoff generated during rare high magnitude storm events, soil infiltration rates, and the DBs' pond volume: catchment size ratios are important factors determining the ability of DBs to decrease surface runoff volumes reaching downstream surface waters. Therefore, in order to optimise DB performance, it is essential to maximise the pond volume: catchment size ratio, which limited the ability of DBs to impound the entire volume of runoff delivered to the bund during rare, large storm events during this study, and have been found to be responsible for significant runoff and nutrient loading to surface waters in the Lake Rotorua catchment. This study found infiltration rates in the ponding area declined due to repetitive ponding over the lifespan of a DB, which could affect the longer-term hydrochemical treatment efficiencies of the DB mitigation strategy. Methods to remediate declining infiltration rates in ponding areas should be investigated.

The study presented in this chapter tested and positively support the hypothesis that DBs would effectively impede stormflow and thereby facilitate soil infiltration by temporarily ponding surface runoff. By decreasing surface runoff discharges, we also hypothesized that the DB strategy is able to decrease nutrient loads transported from pastoral catchments by facilitating sedimentation and soil infiltration, thus decreasing the transport of sediment bound and dissolved nutrients. This hypotheses will be tested in the following three chapters of this thesis.

CHAPTER 4: Sediments

Research highlights

- Detainment bunds facilitated sedimentation on pastures by impeding stormflow.
- Annual sediment loads discharged from the catchments decreased by 1280 kg (59%) and 789 kg (51%).
- Soil infiltration and sedimentation processes contributed to load decreases.

4.1 Introduction

Land use developments and the clearing of native forests have accelerated the already naturally high erosion rates across New Zealand and caused significant sedimentation in lakes and streams (Ministry for the Envrionment, 2019). Pastoral agriculture in New Zealand is strongly associated with eutrophication and degraded freshwater ecosystems (Verburg et al., 2010). Treading by grazing animals increases the likelihood of surface runoff and erosion by physically disturbing the soil, decreasing infiltration rates and porosity, and impairing plant growth (Bilotta et al., 2007; McDowell et al., 2003; Ward et al., 1985). Year-round grazing and high stocking rates used to graze crops are common practices in New Zealand and contribute to increased erosion rates (Monaghan et al., 2007). Pastures in low-order stream catchments have been found to account for an average of 84% of the annual sediment loads delivered to small streams in New Zealand (McDowell et al., 2017).

Suspended sediments (SS) are organic and inorganic particles transported in suspension by water (Bilotta et al., 2009). Rainfall and surface runoff cause erosion and transport SS which may lead to sedimentation in downstream surface waters which, in turn, degrade aquatic ecosystems by disrupting habitats and food webs (Howard-Williams et al., 2010), and delivering sediment-bound nutrients that contribute to eutrophication (Dare, 2018).

Since the 1960’s, water quality in Lake Rotorua has declined due to nitrogen (N) and phosphorus (P) inputs from residential, commercial, industrial and agricultural developments in the catchment in the Bay of Plenty Region of New Zealand’s North Island (Environment Bay of Plenty, 2009). An estimated 42% of the annual P delivered to the lake comes from pastoral dairy and drystock farms which cover ~48% of the 42,000 ha Lake Rotorua surface catchment (Bay of Plenty Regional Council, 2012). Between 71-79% of the anthropogenic P delivered to the lake is sediment bound

(Hamill, 2018), and a portion of that may become biologically available under anoxic conditions which occur in Lake Rotorua and contribute to lake eutrophication (Abell & Hamilton, 2013).

Addressing erosion is a challenge for pastoral farmers in New Zealand, particularly those on sloping landscapes and under the variable precipitation patterns associated with very wet winters, and dry summers interspersed with highly erosive storm events (McDowell et al., 2013). Erosion is likely to be intensified by the more dramatic hydrological conditions caused by climate change (Ministry for the Envrironment, 2019; Ockenden et al., 2016).

The 2012 Lake Rotorua Management Plan has set a target to reduce annual P loads delivered from the catchment in order to restore lake water quality (Bay of Plenty Regional Council, 2012). Achieving Lake Rotorua water quality targets by addressing P loading from pastoral agriculture will require multiple nutrient mitigation strategies and may benefit from the development of new technologies (McDowell, 2010). Due to the prevalence of the contribution of sediment bound P to annual P loads delivered to the lake, mitigation strategies that prevent erosion and the transport of SS should decrease P loading from the catchment.

Stormwater detention areas (SDAs) are natural or manmade depressions, ponds, and reservoirs, commonly used for flood protection, but are increasingly being used for water quality mitigation strategies in agricultural and urban settings (Shukla et al., 2017; Stanley, 1996). Mitigation strategies that increase stormflow residence time, such as SDAs, have been found to decrease surface runoff flows, leading to increased sediment deposition by lowering the kinetic energy of flowing water (Dosskey, 2001; McKergow et al., 2007; Stanley, 1996). However, the type of mitigation strategy affects the duration over which sediments are attenuated. Studies have found that sediment retention times are brief (days to months) in concentrated areas such as narrow grass filter strips and constructed treatment wetlands, while strategies where sediments are blanketed over a wide area may have retention times of up to hundreds of years (McKergow et al., 2007).

Previous research has found that ponding surface runoff can decrease discharge concentrations and loads of sediments and particulate bound P by decreasing the kinetic energy of flowing water (Brown et al., 1981; Harper et al., 1999; Levine et al., 2019; McDowell et al., 2006; Stanley, 1996). Detainment bunds (DBs), a form of SDAs, are earthen storm water retention structures constructed on pastures across the flow path of low-order ephemeral streams, and temporarily pond up to 10,000 m^3 of surface runoff. DBs were implemented in Lake Rotorua headwater catchments in 2010 as a mitigation strategy to target P losses from pastures (Clarke, 2013). Studies of various catchment sizes have found that locating mitigation strategies in catchment headwaters could be especially important because hydrochemical conditions in downstream waters are strongly connected to distant landscape characteristics, and may respond relatively quickly to changes in in upstream sources such as the implementation of nutrient mitigation strategies (Alexander et al., 2007).

Preliminary studies of DBs in the Lake Rotorua catchment found that P enriched sediments were deposited in DB ponding areas (Clarke, 2013), and that a DB effectively decreased the runoff volumes, and sediment and P loads, discharged during 3 non-consecutive ponding events (Levine et al., 2019). A concurrent study currently in review focused on the hydrology of the same ponding events at the same DB sites as this present study reported 31 and 43% of the annual runoff delivered to the DBs infiltrated the soil in the ponding area, and noted that deposited sediments could be developing a less permeable surface soil layer and/or clogging soil pore spaces and causing infiltration rates to decline in the ponding areas (unpublished data).

Although erosion is recognised for its potential impact on aquatic ecosystems, there is a need to progress the understanding of the transport and fate of sediments lost in runoff from intensively managed pastures (Haygarth et al., 2006). To determine if DBs provide a viable strategy for pastoral farmers to improve Lake Rotorua water quality, it is important to quantify their ability to decrease SS loads delivered downstream from pastures. The main objective of this study was to measure the effect of the DB strategy on SS concentrations and yields delivered to two DBs and identify the factors influencing the results. The DBs were installed on pastures downstream of 55 ha and 20 ha catchments, mainly used for pastoral agriculture and draining to Lake Rotorua. Previous studies on DBs and related mitigation strategies suggest ponding surface runoff facilitates sedimentation, although there is currently no definitive research quantifying the impact of the DBs on annual sediment loads transported from pastures in the Lake Rotorua catchment. We hypothesised that ponding surface runoff will facilitate sedimentation and result in lower discharge concentrations. The ability of DBs to decrease SS concentrations along with findings from the concurrent study that showed decreased runoff outflows facilitated by soil infiltration, will result in decreased annual SS loads discharged from the DB catchments.

4.2 Materials and Methods

4.2.1 Site descriptions

The 2 DBs investigated during this current study, along the studies described in Chapters 3, 5 and 6 which investigated the effect of DBs on hydrology and nutrients during the same storm events from 1 December 2017 to 30 November 2018, were located on pastoral dairy farms in the north-western portion of the Lake Rotorua catchment. Table 4.1 presents relevant site characteristics reported in Chapter 3. The Oropi series soils at the Hauraki site, and Waiteti series soils at the Awahou site are both free draining, with >72 mm/h permeability in slowest horizon (Rijkse & Guinto, 2010). Measured infiltration rates in the contributing catchment outside of the DB ponding area were considerably lower permeability reported by Rijkse and Guinto (2010) (Table 4.1), which likely reflects the effect of treading damage under intensive dairying on soil infiltration rates (McDowell et al., 2003).

Table 4.1: Characteristics of detainment bund (DB) sites.

Characteristic		**Hauraki**	**Awahou**
Grid Reference		38°00'21"S 176°11'03"E	38°01'43"S 176°07'54"E
Year DB constructed		October 2011	June 2012
Topography of catchment		Flat, rolling and hill	Mainly rolling
Size of DB entire DB catchment (ha)		55.0	19.7
Area of DB catchment downstream of inflow monitoring (ha)		8.3	1.8
Percentage of catchment with slope (%)	0°-7.9° 8.0°-15.9° 16°-25.9° >26°	69 16 9 5	69 19 9 3
Height of bund at spillway (m)		1.56	1.80
Height of upstand riser (m)		1.36	1.60
DB pond volume ($m^{3)}$		4,894 m^3 at upstand riser 7,110 at m^3 at spillway	1,652 m^3 at upstand riser 2,244 m^3 at spillway
Ratio of pond volume: catchment area (m^3: ha)		89:1 at upstand riser 129:1 at spillway	84:1 at upstand riser 114:1 at spillway
Pond area at pond filled to upstand riser and spillway (m^2)		9,564 m^2 at upstand riser 12,221 m^2 at spillway	2,610 m^2 at upstand riser 2,940 m^2 at spillway
Measured infiltration rates inside and outside ponding area[a] (mm/h)		Inside: 19 Outside: 36	Inside: 12 Outside: 37
Soil classifications		New Zealand: Buried-allophanic Orthic Pumice USA: Vitric Hapludand	New Zealand: Typic Orthic Podzols USA: Andic Haplohumod

[a] See Chapter 3

4.2.2 Event types

Event types in this study were differentiated according to the mode(s) in which ponded water was discharged from the DB, as described in Chapter 3. ‘Overflow Events’ occurred during larger runoff events when inflow continued to be delivered to the pond after the pond height exceeded the height of the upstand riser (Fig. 4.1). After 3 days of ponding, any residual ponded water was evacuated when the outlet valve was opened, creating 'release discharge'. Therefore, 'Overflow Events' had both overflow and release discharge components. In contrast, 'Non-overflow Events' were smaller storms that did not contribute enough runoff to overtop the riser. Non-overflow Events included events when at the end of the 3-day treatment period, either had a portion of ponded runoff to discharge by opening the release valve, or all ponded runoff leaked and infiltrated the soil so there was no water left to discharge.

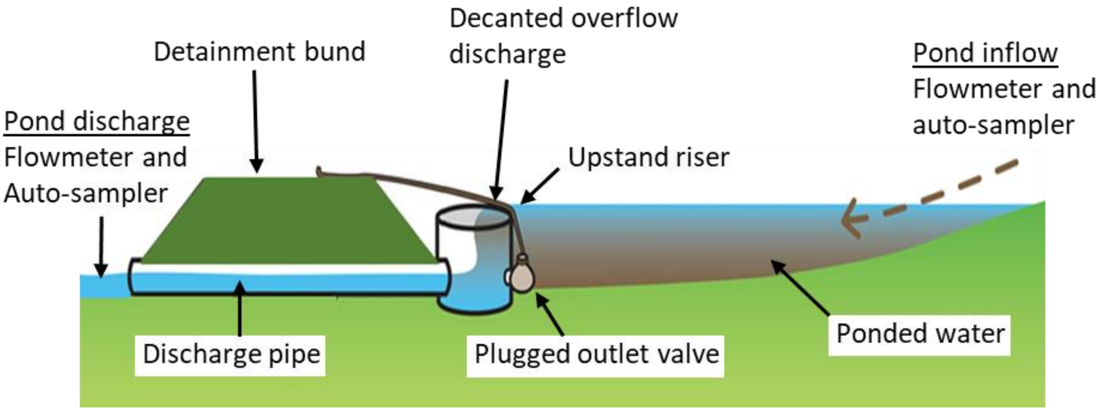

Figure 4.1: Cross-section of the ponding area showing the ephemeral stream inflow ponding behind a detainment bund. If the pond height exceeds the height of the upstand riser then 'decanted overflow' is discharged via a pipe passing through the bund. Inflows and discharges are measured with flowmeters which triggers auto-sampler collections.

4.2.3 Equipment and sampling

The equipment and procedure for collecting surface flow data delivered to, and discharged from the DBs, was described in Chapter 3. Isco® (California, USA) 6712 portable auto-samplers, capable of filling 24 x 1 L bottles collected inflow and discharge samples at each site when triggered by a telemetered UNIDATA® Neon® 2013 F 3G External Memory Metering Module data loggers linked to UNIDATA® 6527 Starflow® QSD flowmeters. The auto-samplers were triggered to collect 1 L samples when flows exceeded 7 L/s (Harmel et al., 2002). Calibration and maintenance of the monitoring equipment followed standard quality controls (NIWA, 2004).

Inflow auto-samplers collected a 1 L sample every 20 min for the first 10 samples, then one 1 L sample/h thereafter (Harmel et al., 2003; Stanley, 1996). The mouth of a rain guarded 750-mL self-sealing bottle using a ping-pong ball inside the bottle, was installed at ground level near the pond outlet valve to capture a sample of the initial flush of surface runoff generated before the inflow auto-sampler was triggered. The ping-pong ball bottle sample was used as the concentration of the initial runoff and used in calculating event inflow loads

Discharge auto-samplers were programmed to collect a 1-L sample each hour (Harmel et al., 2003; Stanley, 1996). Sampled discharge flows were generated if the pond height exceeded the upstand riser height during a storm event (i.e. ‘overflow discharge’) (Fig. 4.1), and when the outlet valve at the base of the riser was opened to release the pond at the end of the event treatment (i.e. ‘release discharge’), typically on the third day of ponding.

Throughout all ponding at both sites an intractable leak at the connection point of the outlet valve pipe and the base of the upstand riser generated a continual flow of ~2-4 m^3/h. Attempts at sealing this leak during the study period were unsuccessful. Under normal sampling conditions, the leak flow was too low to trigger the auto-samplers. Leak samples were collected during 4 events at the Hauraki site, and 1 event at the Awahou site in order to characterise the SS concentrations of the leak discharge.

4.2.4 Sample analysis

Water samples were collected from the field within 24 h of the end of the ponding event and kept refrigerated at 4 °C prior to subsampling that occurred within ~24 h of sample collection. Two separate subsamples (~30 mL) were taken from the field sample after vigorously shaking the bottle, to analyse total and dissolved N and P. The remaining field sample was kept refrigerated until being analysed for SS concentrations used in this current study, following the standard procedure from the American Public Health Association (2005). Filter papers (Whatman GF/C 70 mm) were rinsed with deionised water then pre-dried in the oven at 105°C for 1 day before being weighed. After drying, the filters were cooled in a desiccator, and then re-weighed prior to filtering the water samples. After the remaining field samples (~900mL) were filtered, the filters were again oven dried at 105°C for 1 day and cooled in a desiccator before being weighed.

4.2.5 Mean flow proportional concentration calculations

Event and annual mean flow proportional (MFP) SS concentrations were calculated by dividing the inflow and discharge loads by their respective volume (Tanner & Sukias, 2011). The average difference between the event MFP inflow and leak samples collected during 5 events was +3%, with no consistent increase or decrease. Due to the negligible difference between the MFP inflow and leak concentrations, the MFP inflow concentration was applied to the entire leak volume for each respective event in which the leak discharge was not sampled. The applied leak concentration was used to calculate the event MFP discharge concentrations and event discharge loads. All inflow and discharge MFP concentrations will be referred to only as inflow and outflow concentrations.

4.2.6 Load and yield calculations

Loads (kg) of SS in inflows, and each discharge type, were determined for each ponding event. Inflow loads of SS were calculated by multiplying the measured concentration of the runoff samples collected by the ping-pong ball sample bottle and auto-samplers, and using interpolated concentrations based on the linear rate of change between measured concentrations, by the interval flow volume measured every 5 minutes. Inflow loads were corrected on a pro rata basis (15% increase at the Hauraki site and 9% increase at the Awahou site) to account for the small catchment area between the inflow monitoring site and the DB (Table 4.1).

Discharge loads were calculated for overflow discharge (combining upstand riser and spillway breaching), release discharge (which occurred during Overflow events and Non-Overflow events), and leak discharge (all events). The load of each discharge type was calculated from flow measurements and sample concentrations taken from the DB outlet pipe, except for emergency spillway breaching. Emergency spillway loads were calculated by applying the MFP concentration of the overflow discharge generated by ponded water discharged by going over the upstand riser to the volume breaching the spillway calculated in Chapter 3. Yields refer to the load per unit of contributing catchment area and expressed as mm for runoff volumes, and kg ha^{-1} for SS loads.

4.2.7 Data analysis

Events at each site were analysed to calculate annual results and to compare event types. Changes to concentrations were calculated as the percent difference between inflow and outflow concentrations (percent change in concentration=(outflow-inflow)/inflow)*100). Differences between inflow and outflow concentrations are referred to as a the ‘trapping efficiency’. Differences between inflow and outflow yields are referred to as a ‘yield treatment efficiency’. Inflow yield data for each site was organised by austral seasons (i.e. summer from December to February) to compare differences between the sites and identify seasonal patterns for SS inflow yields.

4.3 Results and discussion

The same 18 ponding events at the Hauraki site, and 19 ponding events at the Awahou site that were reported on in Chapters 3, 5 and 6 were assessed during this present 12-month study. Ponding events occurred most often during the winter months compared to the other seasons (Table 3.2). Water samples were analysed for 13 of these ponding events at the Hauraki site, and 14 events at the Awahou site, since not all ponding events generated flow rates high enough to trigger inflow auto-samplers. Discharge samples were collected during 10 events at the Hauraki site, and 13 events at the Awahou site, since not all events generated discharge flows to be sampled due to leakage and soil infiltration.

4.3.1 Concentrations

The annual SS inflow concentration was 17 g m^{-3} at the Hauraki site, and 96 g m^{-3} at the Awahou site. Inflow concentrations peaked in the winter months at both sites during this study, although there was no clear temporal trend for inflow concentrations (Fig. 4.2). These results are similar to the findings of Smith (1987) who found that SS concentrations were higher in winter runoff when pasture lengths were low, and concentrations were lower in the spring and summer when pasture lengths were longer. During this present study, event inflow concentrations did not tend to correspond to event runoff magnitudes, and varied widely between events (Fig. 4.2). Various factors could have contributed to the lack of a relationship between runoff magnitudes and concentrations including land management factors (Kleinman et al., 2002), storm frequencies affecting source exhaustion (Edwards & Withers, 2008), antecedent moisture conditions affecting susceptibility to erosion (McDowell & Sharpley, 2002) and pasture length affecting the transport potential of SS (Smith, 1987).

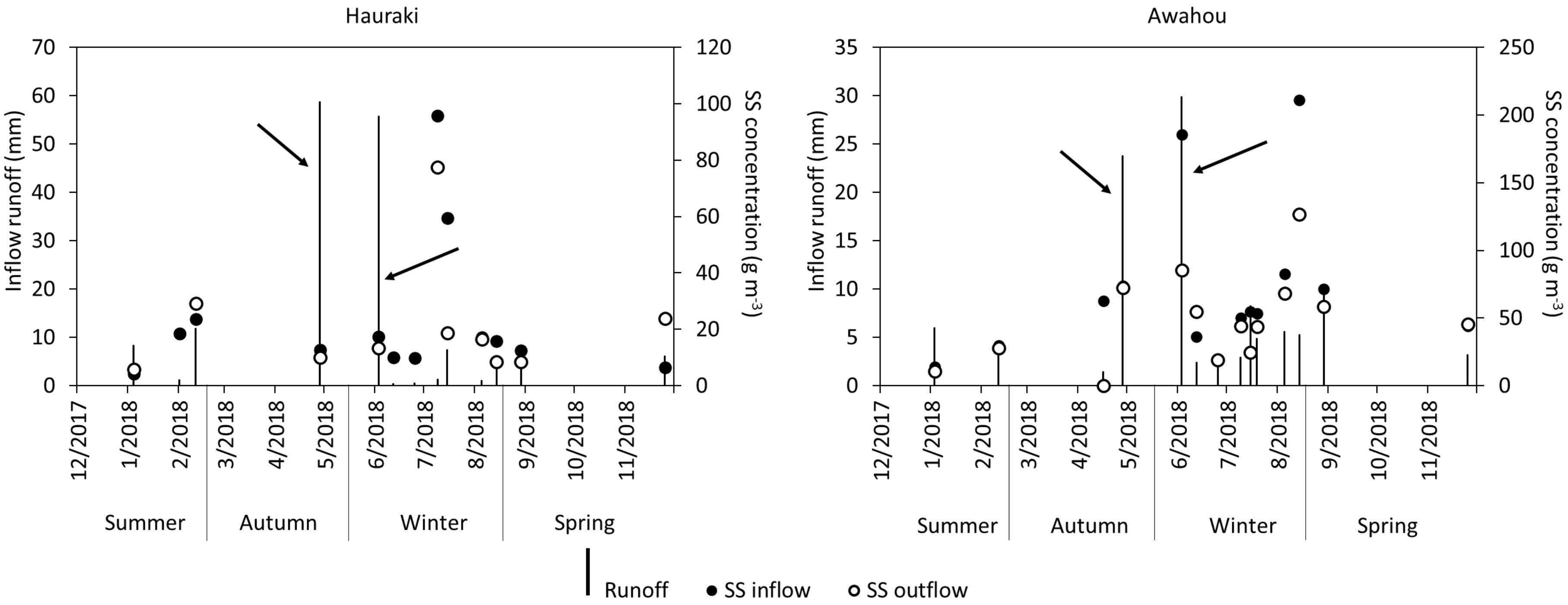

Figure 4.2: Inflow runoff yields (mm) and mean flow proportional suspended sediment (SS) concentrations (g m^{-3}) of inflow and discharge for each event at each site, with arrows pointing to high runoff magnitude Overflow Events. Dates are presented as month and year with austral seasons labelled. Note: Both y-axes are different between the sites.

The annual MFP SS discharge concentration was 28% lower than inflows at the Hauraki site, and 29% lower at the Awahou site. These results suggest that DBs effectively facilitated sedimentation during ponding supporting the hypothesis set out described in section 4.1 and demonstrate that deposited sediments are attenuated in the ponding area. Discharge concentrations were lower than inflows during 7 of the 10 events analysed at the Hauraki site, and 10 of the 13 events analysed at the Awahou site (Fig. 4.2).

Inconsistencies in trapping efficiencies were observed between and within event types at both sites with no apparent temporal trends (Table 4.2) (Fig. 4.2). Outflow concentrations were lower than inflows during 7 of the 10 events analysed at the Hauraki site, and 10 of the 13 events analysed at the Awahou site (Fig. 4.2). On average, the concentration decreased 31% at the Hauraki site and 25% at the Awahou site during events in which concentrations decreased. During events in which concentrations increased, the concentration increased 109% and 18% on average at the at the Hauraki and Awahou site, respectively. The large increase observed at the Hauraki site was the result of 1 of the 3 events in which the concentration increased, when the outflow concentration was 270% higher than inflow (Table 4.2). This extreme increase could be the result of the inflow concentration during this event being very low (6 g m^{-3}) compared to other events, contributing to the second lowest event inflow concentration measured in the study and measuring one-third of the annual MFP inflow concentration. Due to the low inflow concentration during this event, a slight increase in outflow concentration would result in a high proportional increase.

Table 4.2: Mean flow proportional (MFP) concentrations of suspended sediments (SS) for inflow and discharges across all events, MFP concentration ranges for each event type, and changes to MFP concentrations by percentage (%), comparing inflows to discharges.

Site	Event type	MFP SS concentration (g m^{-3})		Percentage change
		Inflow	Discharge	(%)
Hauraki	All events	17	12	-28
	Overflow Event range	13 – 17	10 –13	-22 – -21
	Non-Overflow Event range	4 – 96	6 – 77	-69 – +270
Awahou	All events	96	68	-29
	Overflow Event range	74 – 186	73 – 85	-54 – -1
	Non-Overflow Event range	14 – 211	11 – 127	-55 – +50

The wide range of trapping efficiencies observed between events in this study were likely influenced by multiple factors. Treading damage, deposited animal excreta (McDowell et al., 2003) and previously deposited sediments in the ponding area (Barber & Quinn, 2012) could have contributed to SS discharged from the DB that was not accurately accounted for by the pro rata correction of the unmeasured contributing catchment area, and so would have affected the trapping efficiency results.

Variations in particle sizes delivered to the DBs, which were not measured in this study, could have also contributed to the varying trapping efficiencies observed between events and the sites. Heavier particles (i.e. sand) settle more readily than smaller particles (i.e. silt and clay) which more likely to be transported and/or remobilised and discharged from the DBs (McDowell et al., 2003). A previous study of DBs found that sediments deposited at higher elevations in the ponding area typically had greater proportions of coarse size sand particles than lower elevations, suggesting that finer sediments take longer to settle than coarser particles in DBs (Clarke, 2013).

During this present study, a greater proportion of large particles may have been delivered to the DBs during higher magnitude runoff events due to their greater erosive power, particularly Overflow Events. Therefore, differential transport of grain sizes could be partially responsible for SS concentrations decreasing during all Overflow events in this study, while this was not the case for all Non-Overflow Events (Table 4.2). While Non-Overflow Events had greater variation in concentration changes than Overflow Events (standard deviation= 21.8 during Overflow Events, 71.3 during Non-Overflow Events), median trapping efficiencies were similar between event types (Fig. 4.3)

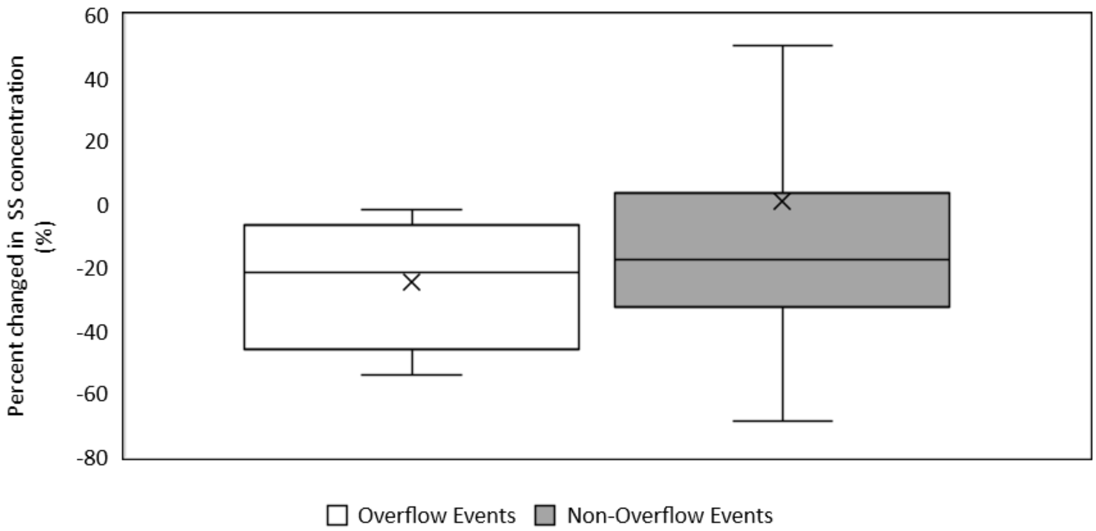

Figure 4.3. Side-by-side box and whisker plot comparing the percent change in suspended sediment (SS) concentration during Overflow and Non-Overflow events occurring at both study sites during the 12-month study. Centre lines represent the medians, box limits indicate the 25th and 75th percentiles and x's indicate the mean event percent concentration change.

During Overflow Events at both sites, the SS concentration difference between the portions of inflow contributing to overflow discharge (i.e. ponded surface water discharged via the upstand riser and emergency spillway), termed Flow A, and the subsequent overflow discharge, termed Flow B, did not decrease to the same extent as the concentration decreased between the overflow discharge (Flow B) and the following release discharge (Flow C) (Table 4.3). These results are somewhat surprising since we would expect the decanting of the uppermost layer of water performed by the upstand riser (Fig. 4.1) and emergency spillway would be highly effective at preventing SS discharge. The data suggests however, that longer pond residence times experienced by the release discharge compared to the overflow discharge (an average of 14 hours between overflow discharge and the following release discharge at both sites), allowed for greater sedimentation to occur. Longer retention times have been found to increase sediment removal efficiencies in a study of sedimentation ponds (Brown et al., 1981).

Table 4.3: Mean change in suspended sediment (SS) concentrations between the portion of inflow contributing to overflow discharge (Flow A), and the runoff discharged over the upstand riser (Flow B), and the mean concentration change between the overflow discharge (Flow B) and the release discharge generated when the outlet valve was opened to drain the pond (Flow C), during Overflow Events at both sites.

Mean change in SS concentration between:	**Hauraki (%)**	**Awahou (%)**
Portion of inflow contributing to overflow discharge (Flow A) and the runoff discharged over the upstand riser (Flow B)	-37	-20
Overflow discharge (Flow B) and release discharge generated upon opening the outlet valve to drain the pond (Flow C)	-41	-84

The data suggests ponding runoff for longer than 3 days (Clarke, 2013) could result in greater trapping efficiencies, however, this could risk damaging pasture productivity. Removing the upstand riser/outlet valve/discharge pipe installation, and allowing all ponded water to infiltrate the soil, would prevent the discharge of the bottommost portion of ponded water where SS are likely to concentrate and/or be stirred up by turbulence when unplugging the outlet valve to drain the pond. Also, placing the outlet valve 10-cm above ground level would enable a small portion of the ponded water left after draining the pond to infiltrate the soil. This change would also prevent the discharge of a lower portion of ponded runoff, and would decrease the area potentially affected by prolonged inundation compared to avoiding the release procedure entirely. Lastly, approaches to achieve greater trapping efficiencies could include the use of flocculants that would aggregate SS and facilitate greater sedimentation.

4.3.2 Yields and loads

The key finding of this 12-month study was that impeding stormflow with DBs resulted in 789 kg and 1280 kg of SS being attenuated in the Hauraki and Awahou DB ponding areas, respectively. The SS load reductions were equivalent to 51% and 60% of the annual SS inflow loads at the Hauraki and Awahou sites, respectively. The proportion of the annual SS load reduced by the DBs exceeded the proportion of the annual runoff inflow infiltrated the soil in the ponding areas, 31% at Hauraki and 43% at Awahou, reported in Chapter 3.These results positively support the hypothesis that DBs would facilitate sedimentation and attenuate SS transported in stormflows.

The results also suggest DBs should also be effective at reducing P losses from pastures in the Lake Rotorua catchment, due to the high proportion of sediment bound P delivered to the lake (Hamill, 2018). The benefit of reducing SS loads discharged from the DB catchments, and potentially mobilised downstream of the DB catchments, would also decrease the particulate P loads delivered to receiving surface waters. Clarke (2013) found that the mean P concentration of sediments deposited in the same DB ponding areas as this present study ranged from ~1.5-3 g P kg^{-1} of sediment dry weight. Taking the findings of Clarke (2013) into account, the results of this present study suggest that DBs could decrease particulate P losses delivered to Lake Rotorua by 1.2-2.4 kg y^{-1} (Hauraki DB) and 1.9-3.8 kg y^{-1} (Awahou DB).

The sediment loads deposited in the ponding area in the current study are likely to be lower than the loads prevented from reaching surface waters downstream of the DBs as a result of the mitigation strategy. This is because some portion of sediments discharged from the DBs could be permanently entrained in the soil, which typically occurs in pastures (Smith, 1987), but would also enhanced by the reduced surface runoff magnitudes occurring downstream of the DBs as a result the impediment of stormflows described in the unpublished concurrent study. Additionally, erosion and SS mobilisation occurring downstream of the DBs is likely to decrease as a result of the mitigation strategy effectively decreasing surface runoff magnitudes. The extent of these downstream benefits was beyond the scope of this study and should be investigated in the future.

Annual SS inflow yields were 28 kg ha^{-1} at the Hauraki site, and 109 kg ha^{-1} at the Awahou site, although runoff inflow yields were greater at the Hauraki site than the Awahou site (Fig 4.4). The annual SS inflow yields at both sites in this study were much lower than the estimated annual SS yields entering streams in the same area of the Lake Rotorua catchment from May 2010 to May 2012 (479-741 kg ha^{-1} y^{-1}) (Abell et al., 2013). Factors affecting the catchments' hydrological responses to precipitation, including antecedent soil conditions and localised differences in storm rainfall intensity and duration, and differences between the catchment sizes, geomorphologies, and land use and management factors, are likely to have affected runoff generation and erosion (Dougherty et al., 2004), and likely accounted for the SS inflow yield differences between study sites in this present study and the results reported by Abell et al. (2013).

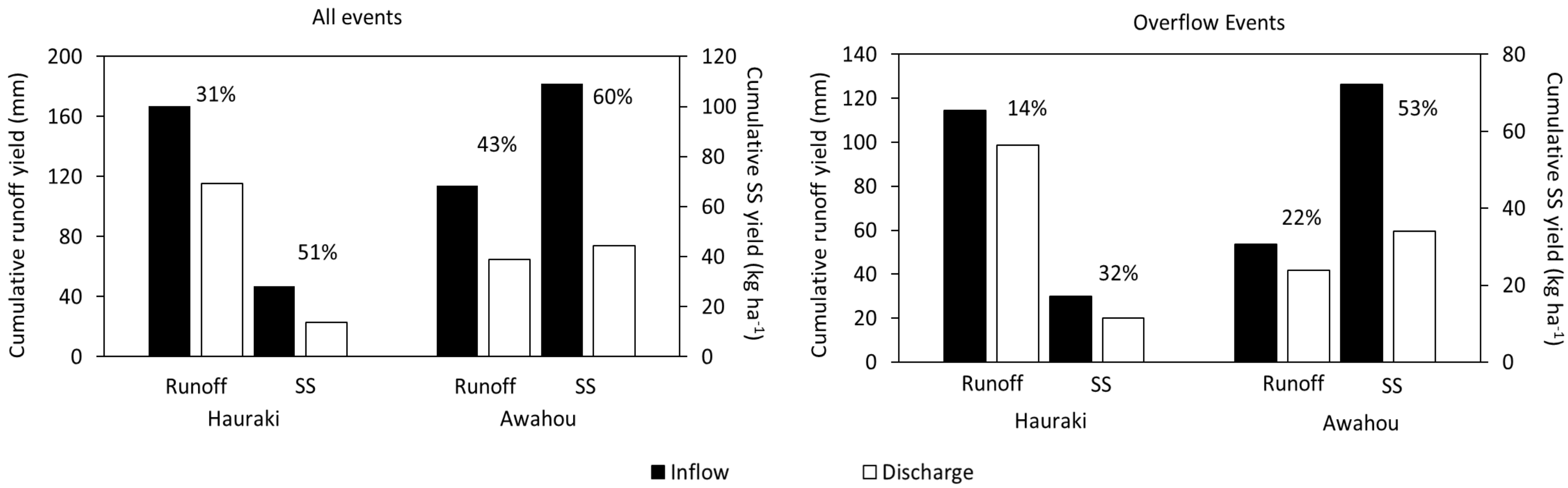

Figure 4.4: Cumulative annual, and Overflow Event only, inflow and discharge runoff yields (mm) and suspended sediment (SS) yields (kg ha^{-1}). Note: Difference in both y-axes between sites.

At both sites during this present study, runoff and SS inflow yields were lowest in the spring and increased during each subsequent season, peaking during the winter period (Fig. 4.5). These results were not surprising, as the contributing catchment is grazed by dairy cattle, and soil treading damage and erosion is likely to increase when soils are wet (McDowell et al., 2003). Additionally, greater SS yields tended to correspond with greater runoff yields during individual events at each site, particularly during the high runoff magnitude Overflow Events (Fig. 4.6). The positive relationship between event runoff and SS yields contrast with the lack of relationship between event runoff yield and SS concentration, likely due to the effects of source exhaustion and dilution (Abell et al., 2013). The results of this present study are also consistent with other studies that found greater runoff magnitudes tend to mobilise and transport greater quantities of sediments and nutrients from pastures in New Zealand (Cooke, 1988; Smith & Monaghan, 2003) and the Lake Rotorua catchment, specifically (Abell et al., 2013; Dare, 2018). The higher SS yields measured at the Awahou site while higher runoff yields occurred at the Hauraki site suggests differences in factors affecting erosion between the catchments at the two sites, such as precipitation patterns, geomorphologies, soil types and land use and management (Dougherty et al., 2004).

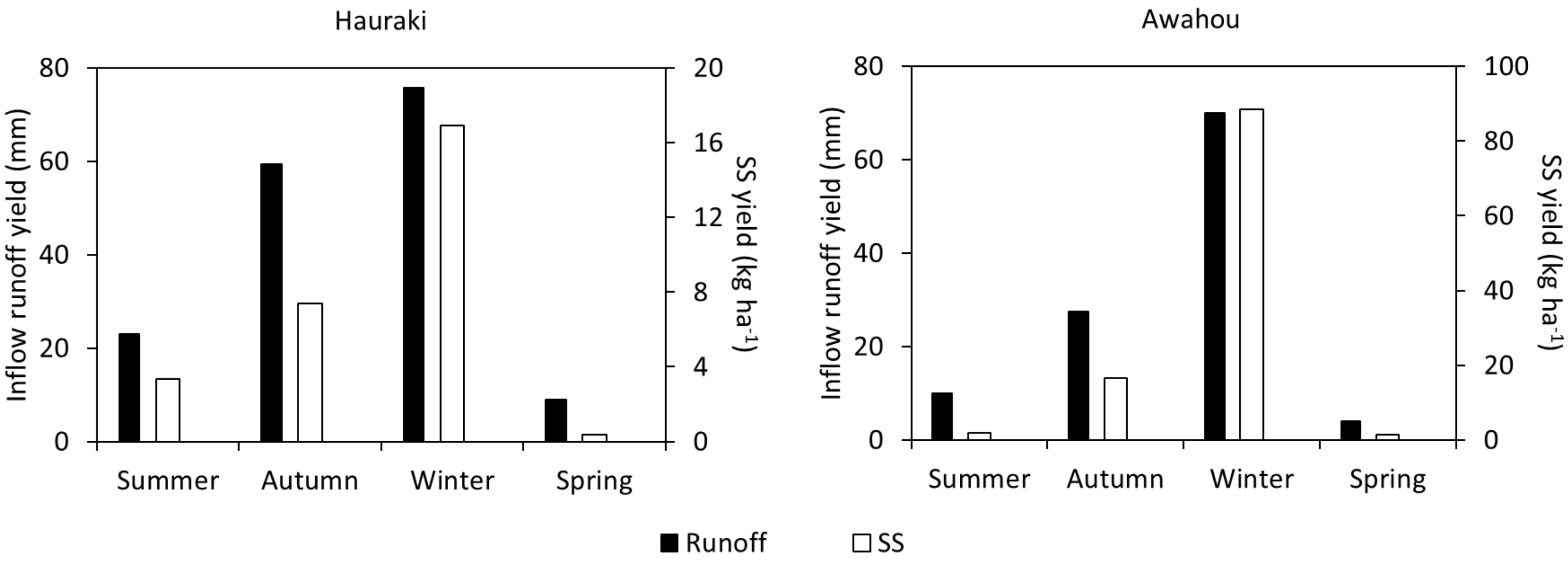

Figure 4.5: Cumulative seasonal runoff inflow (mm) and suspended sediments (SS) inflow yields (kg ha^{-1}) for each season at each site. Note: Difference between the 'SS yield' y-axis between sites.

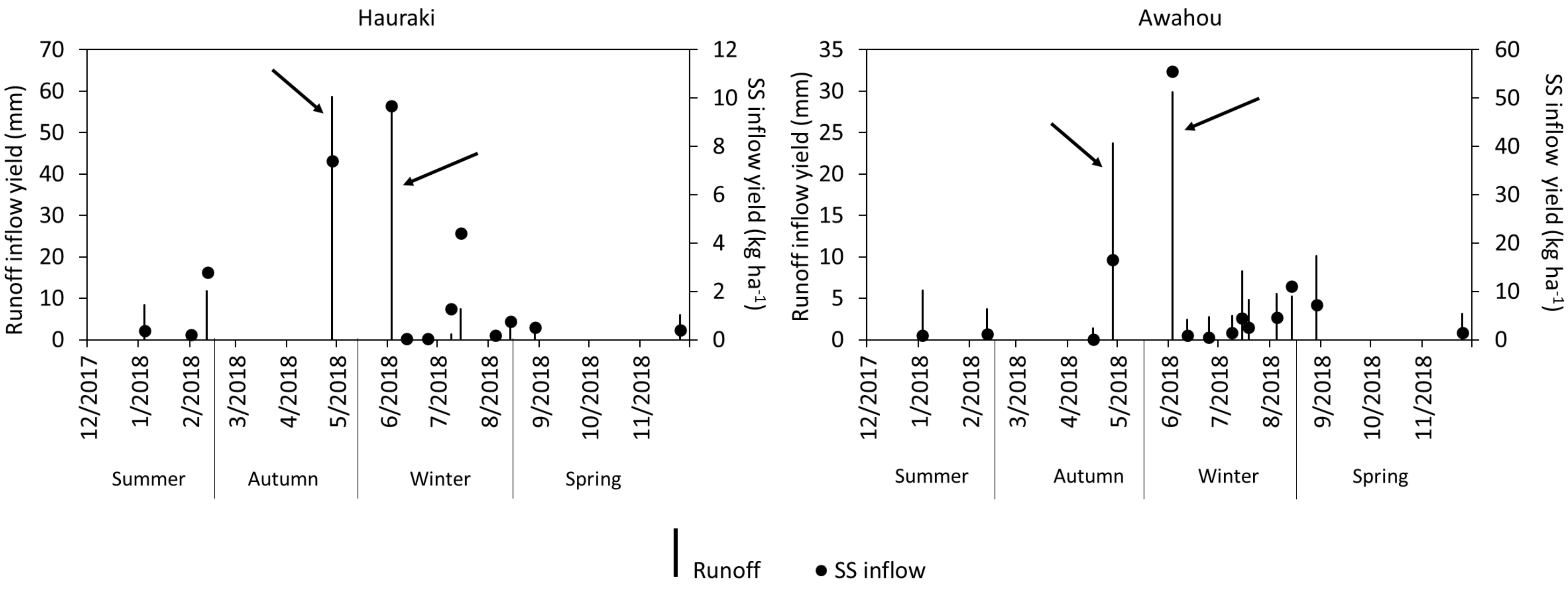

Figure 4.6: Event inflow runoff yield (mm) and suspended sediment (SS) yields (kg ha^{-1}) at both sites, with arrows pointing to high runoff magnitude Overflow Events. Dates are presented as month and year with austral seasons labelled. Note: both y-axes are different between the sites.

The results of this study demonstrate the DBs at both sites were able to consistently decrease SS loads discharged from the DB catchments, even during rare, large events, despite the outflow concentrations not being consistently lower than inflow concentrations. These results emphasize the important role soil infiltration plays in DBs effectively decreasing SS outflow loads. The greater inflow magnitudes during Overflow Events at the Hauraki site contributed to a greater portion of runoff undergoing overflow discharge compared to the Awahou site, and consequently, the difference in the portion of inflow undergoing soil infiltration and SS yield treatment efficiencies between the sites during the high magnitude events (Fig. 4.4).

The results from the rare Overflow Events emphasise the importance of DBs being effective during these high magnitude events, since Overflow Events were responsible for 61% and 66% of the annual SS inflow loads at the Hauraki and Awahou sites, respectively, and 39% and 59% of annual SS yields attenuated. This finding is important to note, since large storm events have been found to be responsible for the majority of SS loading to streams in the Lake Rotorua catchment (Abell et al., 2013).

The ability of DBs to consistently decrease SS loads, particularly during large runoff events, is noteworthy because some land management strategies are overwhelmed by extreme hydrologic conditions (Kleinman et al., 2006; McDowell & Sharpley, 2002; McKergow et al., 2007). Sediment deposition across the relatively wide DB ponding area was observed during this study, and likely contributed to the consistency in DB performance (McKergow et al., 2007). Importantly, it is likely sediments deposited in the DB ponding area will be attenuated for longer periods of time compared to other mitigation strategies, such as buffer strips and treatment wetlands, that have more concentrated sediment deposition areas and are susceptible to flushing during high magnitude events (McKergow et al., 2007). The ability of the DB to impede the stormflow of each runoff event reduced the kinetic energy of water, which enables the transfer and/or remobilisation sediments, and particularly the ‘first-flush’ of the initial runoff, could have had a major influence on the DBs’ ability to decrease SS loads transported in surface runoff during each event in this study (Bieroza et al., 2019).Although DBs effectively attenuated SS loads during Overflow Events, these large magnitude events still generated 84% of the annual SS outflow yields at the Hauraki site, and 77% at the Awahou site. These results are likely related to the majority of the annual runoff outflow also occurring during Overflow Events at both sites (Table 4.4**)**. Results from a concurrent study suggest that soil infiltration played a key role in reducing runoff outflow from DBs and this was critical to the reduction of SS loads in the current study. This highlights the importance of optimising DB design to maximise soil infiltration of ponded runoff and avoiding excess overflow discharge during high magnitude events.

Table 4.4: Percentage (%) of annual runoff and suspended sediments (SS) inflow and discharge yields which occurred during Overflow Events at each site.

Site	Flow type	Runoff (%)	SS (%)
Hauraki	Inflow	69	61
	Discharge	85	84
Awahou	Inflow	47	66
	Discharge	64	77

The contribution of soil infiltration to annual SS yield treatment efficiencies is also important to note because infiltration rates in the ponding area were found to be lower than those outside the ponding area, likely due to some influence of repetitive ponding (Table 4.1**).** The results of this present study support previous findings, that deposited sediments clog soil pores and/or form a less permeable surface soil layer (Hendrickson, 1934; Reddi et al., 2000; Rice, 1974). Therefore, infiltration rates, and consequently SS yield treatment efficiencies, will be highest in newly constructed DBs, and are likely to decrease over time. Additionally, infiltration rates and SS yield treatment efficiencies would be likely to decline faster in locations with higher erosion rates and greater SS loads being deposited in DB ponding areas.

During this study, outflow concentrations were lower than inflow concentrations in only 70% and 77% of the events at the Hauraki and Awahou sites respectively, and SS yield treatment efficiencies were greater than runoff yield treatment efficiencies. These results indicate that sedimentation facilitated by impeding stormflows with DBs caused lower SS outflow concentrations. Therefore, DBs would still decrease SS outflow yields in areas where soil infiltration rates and pond storage to catchment area ratios are lower than those in this present study, although yields are not likely to decrease to the same extent. Other factors influencing the proportion of runoff infiltrating the soil and sediment sizes delivered to the DBs would affect yield treatment efficiencies.

Revising the DB design to remove the upstand riser/outlet valve/discharge pipe installation would prevent SS leak and release discharges. Removing the leak and release discharge loads from the annual SS outflow loads would have prevented an additional 147 kg of SS from being discharged from the Hauraki site, and an additional 216 kg at the Awahou site, increases of 16% and 14% of the annual SS load attenuated at each site, respectively. The costs and benefits of revising the DB design should be assessed because the increased inundation period could damage pasture productivity.

Despite hundreds of kgs of sediments being deposited in the DB ponding area during the 12-month study period, and presumably during each of the 6 years since the DBs were constructed, there was no observable build-up of sediments in the ponding area. Although previously deposited sediments may be remobilised in subsequent ponding events, and soil infiltration rates have been show in a concurrent study to be decreasing in the ponding areas, the finding that SS outflow loads were effectively reduced by the DBs suggests the monitored DBs will be able to continue to effectively attenuate SS well into the future. However, sediment deposition and innovations that increase trapping efficiencies could, in turn, decrease yield treatment efficiencies in the long-term, due to greater quantities of deposited sediments contributing to further decreases in soil infiltration rates and increased sediment remobilisation. Methods of mitigating declines in the soil infiltration rates observed in the concurrent study and affecting SS attenuation by DBs, such as aerating the pond area soils or employing subsoil amendments, should be investigated. Future investigations should also characterise sediment sizes (distribution of sand, silt, and clay) in the DB catchments, mobilised during runoff events, attenuated in the DB ponding area, and discharged from the DB, in order to provide further insight into the ability of DBs to attenuate SS and associated P in the short- and long-term.

4.4 Conclusion

The results of this current study found that DBs located on pastures in the Lake Rotorua catchment attenuated 789 kg SS at the Hauraki site, and 1280 kg SS at the Awahou site, accounting for 51% and 59% of the annual inflow SS loads, respectively. Large portions of the annual SS yields attenuated by the DBs occurred during high runoff magnitude events, which delivered the majority of annual surface runoff and SS yields to the bunds.

The annual SS yield treatment efficiencies observed in this study were related to changes in SS concentrations through deposition, and the portion of runoff infiltrating the soil in the ponding area. Greater SS outflow yields occurred with greater runoff outflows, which emphasises the importance of optimising DB design to maximise the amount of runoff infiltrating the soil. The impoundment of runoff generally decreased event SS concentrations, suggesting that DBs may effectively decrease SS loads where soil infiltration rates, and pond storage to catchment area ratios, are not as high as those this present study.

While this study found DBs consistently decreased SS outflow yields from the DBs, identifying methods to improve trapping efficiencies, such as integrating the use of flocculants, or allowing the bottommost layer of the pond to infiltrate the soil rather than be released, would improve yield treatment efficiencies. Also, cost: benefit analyses should be conducted to determine whether removing pond discharge mechanisms (i.e. riser/outlet valve/discharge pipe unit) would be beneficial, keeping in mind this might affect pasture productivity and performance longevity. Longer-term studies in a higher number of DB locations should also be conducted in the Lake Rotorua catchment to further understand the strategy's potential to effectively mitigate pastoral farming's impact on surface water quality. Future investigations should also characterise sediment sizes in the DB catchments, mobilised during runoff events, attenuated in the DB ponding area, and discharged from the DB, in order to provide further insight into the ability of DBs to attenuate SS in the short- and long-term. Studies should also investigate the cause of declining soil infiltration rates in the ponding area and methods for maintaining or rehabilitating infiltration rates in order to maintain SS yield treatment efficiencies over the life of the DB.

Because of the potential for sediment-bound P transport to contribute to eutrophication, the evidence of the ability of DBs to facilitate sedimentation and attenuate sediments in the ponding area presented in this present chapter will be useful in understanding if DBs are a useful nutrient mitigation strategy by decreasing P losses from pastoral catchments. The following chapter will investigate how sedimentation affects P attenuation by DBs, as well as how soil infiltration facilitated by temporarily ponding surface runoff are likely to affect P delivery to Lake Rotorua.

CHAPTER 5: Phosphorus

Research highlights

- Annual concentrations of total P and dissolved reactive P decreased.
- Concentration decreases and soil infiltration combined to decrease P load discharges.
- The strategy prevented an estimated 12 and 44 kg P from reaching downstream surface waters, decreases of 39 and 60%.
- Detainment bunds are an effective P mitigation option in the Lake Rotorua catchment

5.1 Introduction

Anthropogenic phosphorus (P) loading contributes to eutrophication in the culturally significant, and economically valuable Lake Rotorua, located on the North Island of New Zealand (Tempero et al., 2015). Concentrations above 0.013 g total P (TP) m^{-3} are considered to stress lakes such as Lake Rotorua in New Zealand (ANZG, 2018). Median concentrations in Lake Rotorua were 0.02 g TP m^{-3} from 2013-2017, and 0.0002 g dissolved reactive P (DRP) m^{-3} from 2009-2013 (Stats NZ, 2019).

Surface runoff contributes the majority of P to Lake Rotorua from anthropogenic sources since P transport in subsurface runoff is low due to the prevalence of soils with high P sorption capacities in the catchment (Morgenstern et al., 2015). Phosphorus transported by surface runoff in the Lake Rotorua catchment is present as biologically available DRP, and particulate forms bound to sediments (Rutherford & Timpany, 2008). An estimated 71-79% of P delivered to the lake from the catchment is sediment-bound (Hamill, 2018). Sediment-bound P entering Lake Rotorua is able to be released under anoxic conditions and contribute to DRP in the water column (Abell & Hamilton, 2013; Burger et al., 2007).

The average P load delivered to Lake Rotorua from 2007-2014 was estimated to be 42 t P y^{-1} with 17-19 t P y^{-1} derived from anthropogenic sources (Hamill, 2018). Dairy and drystock farms account for ~43% of the 42,000 ha Lake Rotorua surface area catchment, and contribute ~43% of the annual P load delivered to the lake, with average P losses from pastures estimated to be 0.84 kg ha^{-1} y^{-1} (Bay of Plenty Regional Council, 2012). Managing P losses in surface runoff from pastures that contribute to water quality degradation are difficult to manage in the Lake Rotorua catchment, since soil P concentrations are proportional to the magnitude of P losses from soils in runoff (McDowell et al., 2001), and the economically optimal Olsen P levels in pastures (15-45 mg L^{-1}) are orders of magnitude higher than those that contribute to lake eutrophication (McDowell, 2010).

To achieve acceptable water quality levels, the 2012 Lake Rotorua Management Plan has set goals to reduce anthropogenic P loading to the lake by 3.5 t y^{-1} by 2019, and 10 t y^{-1} by 2029, from the 2012 annual loading estimates of 39 t P y^{-1} (Bay of Plenty Regional Council, 2012). A more recent study estimated that loads from the catchment need to decrease by 8-13 t Py^{-1} to achieve lake water quality targets (Tempero et al., 2015). Land-based mitigation strategies are capable of reducing P loads delivered to Lake Rotorua by up to 35 t y^{-1}, which will help achieve water quality targets (Donald et al., 2019).

Climate change contributes to conditions that make improving water quality in Lake Rotorua more challenging (Donald et al., 2019). Nutrient losses in runoff from pastures are likely to increase due to greater storm intensities, wetter winters, and hotter, drier summers, projected for New Zealand as a result of climate change (Ministry for the Envrironment, 2019; Ockenden et al., 2016). Also, internal P loading from lake sediment has been shown to increase during warmer months, and has the potential to be the source of the majority of annual P loading in Lake Rotorua when severe deoxygenation of the hypolimnion occurs (Burns et al., 2005; Donald et al., 2019).

Achieving Lake Rotorua water quality targets by addressing P loading from pastoral agriculture will require combining multiple nutrient mitigation strategies and potentially developing new technologies (McDowell, 2010). Sedimentation has been identified as one of the primary mechanisms involved in nutrient mitigation strategies that target surface runoff (Brown et al., 1981; Carter et al., 1974; Stanley, 1996). Studies have found that ponding surface runoff decreased TP loads by facilitating sorption of dissolved P and sedimentation of P-enriched particles (Brown et al., 1981; Harper et al., 1999; McDowell et al., 2006).

Since 2010, detainment bunds (DBs) have been used as a potential strategy to mitigate P transported from pastures in the Lake Rotorua catchment by impeding stormflow and temporarily ponding surface runoff (Clarke, 2013). Detainment bunds are ~1.5-2 m high stormwater retention structures constructed on pastures across the flow path of low-order ephemeral streams, with the ability to pond up to 10,000 m^3 of surface runoff. Further details about DB design and operation were presented in Chapters 3 and 4.

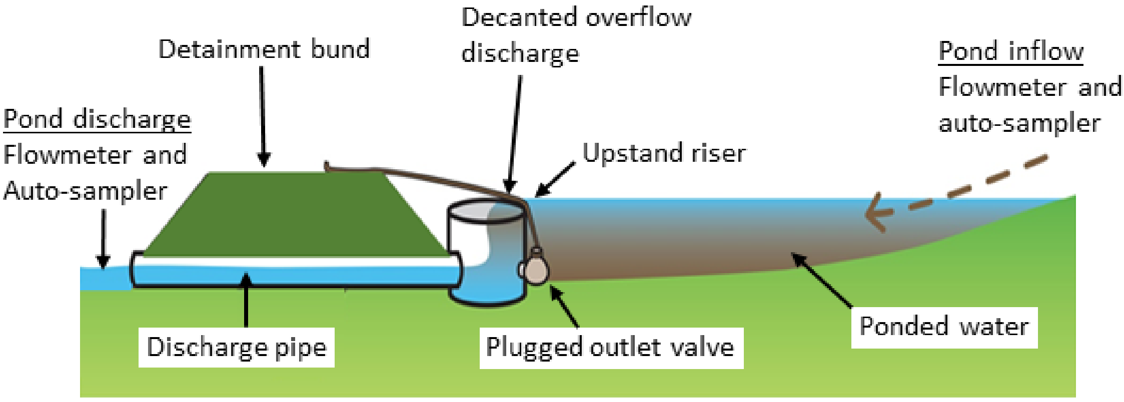

Figure 5.1: Cross-section of ponding area showing the ephemeral stream inflow ponding behind a detainment bund. If the pond height exceeds the height of the upstand riser then 'decanted overflow' is discharged via a pipe passing through the bund wall. After approximately three days of ponding, the plug is removed from the outlet valve at the bottom of the upstand riser to empty the pond. Inflows and discharges are measured with flowmeters which triggers auto-sampler collections.

Preliminary studies have found that DBs retained P enriched sediments during ponding events (Clarke, 2013), and sediment, TP and DRP loads discharged from the DB catchment were lower than inflows during a study of 3 non-consecutive ponding events at a DB due to lower discharge volumes, sedimentation and sorption (Levine et al., 2019). Previous chapters in this thesis reported that annual surface runoff discharged from the catchments at the 2 DBs in this current study decreased by 31 and 43% due to soil infiltration (Chapter 3), and annual sediment loads decreased by 1,280 kg y^{-1} (59%) and 789 kg y^{-1} (51%) respectively (Chapter 4). However, no previous studies have definitively quantified the ability of DBs to reduce annual P losses from pastures in the Lake Rotorua catchment. The main objective of this study was to quantify the effect of DBs on the TP and DRP concentrations and loads at 2 sites in the Lake Rotorua catchment, and identify the mechanisms affecting mitigation performance. We hypothesised that the DB strategy will decrease TP and DRP discharges as a result of lower runoff volumes discharged from the DBs, sorption lowering DRP concentrations, and sedimentation during ponding lowering TP concentrations. This study also investigated the effect of repetitive ponding on soil P concentrations in the ponding area, and whether ponding areas might become a potential source of P in DB discharges. By comparing the results from the 2 DBs that had similar and differing site characteristics, this study also identified factors affecting the ability of the DBs to prevent P from reaching downstream surface waters.

5.2 Materials and methods

5.2.1 Site descriptions and event descriptions

This current study reporting on the effect of DBs on TP and DRP concentrations and loads, along the studies described in Chapters 3, 4 and 6 which investigated the effect of DBs on hydrology, sediments and nitrogen, respectively, took place at the same 2 DBs, located on pastoral dairy farms in the north-western portion of the Lake Rotorua catchment, during the same storm events from 1 December 2017 to 30 November 2018. Detailed site characteristics are listed in Table 5.1.

Table 5.1: Characteristics of detainment bund (DB) sites.

Characteristic		**Hauraki**	**Awahou**
Grid Reference		38°00'21"S 176°11'03"E	38°01'43"S 176°07'54"E
Year DB constructed		October 2011	June 2012
Topography of catchment		Flat, rolling and hill	Mainly rolling
Size of DB entire DB catchment (ha)		55.0	19.7
Percentage of catchment with slope (%)	0°-7.9°	69	69
	8.0°-15.9°	16	19
	16°-25.9°	9	9
	>26°	5	3
Area of DB catchment downstream of inflow monitoring (ha)		8.3	1.8
Height of bund at spillway (m)		1.56	1.80
Height of upstand riser (m)		1.36	1.60
DB pond volume ($m^{3)}$		4,894 m^3 at upstand riser 7,110 at m^3 at spillway	1,652 m^3 at upstand riser 2,244 m^3 at spillway
Ratio of pond volume: catchment area (m^3: ha)		89:1 at upstand riser 129:1 at spillway	84:1 at upstand riser 114:1 at spillway
Pond area at pond filled to upstand riser and spillway (m^2)		9,564 m^2 at upstand riser 12,221 m^2 at spillway	2,610 m^2 at upstand riser 2,940 m^2 at spillway
Measured infiltration rates inside and outside ponding area[a] (mm/h)		Inside: 19 Outside: 36	Inside: 12 Outside: 37
Soil description		Oropi series- Free draining with >72 mm/h permeability in slowest horizon	Waiteti series- Free draining with >72 mm/h permeability in slowest horizon
Measured infiltration rates inside and outside ponding area[a] (mm/h)		Inside: 19 Outside: 36	Inside: 12 Outside: 37
Sediment load deposited in ponding area during current study period[b] (kg)		789	1,280
Anion storage capacity (%)		46	85
Olsen P in catchment contributing to ponding area (mg L^{-1})		35	38

a See Chapter 3 **b** See Chapter 4

5.2.2 Event types

Event types reported on in this study and the studies presented in Chapters 3, 4 and 6 were differentiated according to the mode(s) ponded water was discharged from the DB. 'Overflow Events' occurred during larger runoff events when inflow continued to be delivered to the pond after the pond height exceeded the height of the upstand riser (Fig. 5.1). After 3 days of ponding, any residual ponded water was evacuated when the outlet valve was opened, creating 'release discharge'. Therefore, 'Overflow Events' had both overflow and release discharge components. In contrast, 'Non-overflow Events' were smaller storms that did not contribute enough runoff to overtop the riser. Non-overflow Events included events when at the end of the 3-day treatment period, either had a portion of ponded runoff to discharge by opening the release valve, or all ponded runoff leaked and infiltrated the soil so there was no water left to discharge.

5.2.3 Equipment and sampling

The equipment and procedure for collecting surface flow data delivered to, and discharged from the DBs, was described in Chapter 3. The same samples collected and analysed in Chapter 4 were analysed in this current study. Surface runoff sample collection equipment and procedures were presented in Chapter 4. Isco® (California, USA) 6712 portable auto-samplers, capable of filling 24 x 1 L bottles collected inflow and discharge samples at each site when triggered by a telemetered UNIDATA® Neon® 2013 F 3G External Memory Metering Module data loggers linked to UNIDATA® 6527 Starflow® QSD flowmeters. The auto-samplers were triggered to collect 1 L samples when flows exceeded 7 L/s (Harmel et al., 2002). Calibration and maintenance of the monitoring equipment followed standard quality controls (NIWA, 2004).

Inflow auto-samplers collected a 1 L sample every 20 min for the first 10 samples, then one 1 L sample/h thereafter (Harmel et al., 2003; Stanley, 1996). The mouth of a rain guarded 750-mL self-sealing bottle using a ping-pong ball inside the bottle, was installed at ground level near the pond outlet valve to capture a sample of the initial flush of surface runoff generated before the inflow auto-sampler was triggered. The ping-pong ball bottle sample was used as the concentration of the initial runoff and used in calculating event inflow loads

Discharge auto-samplers were programmed to collect a 1-L sample/h (Harmel et al., 2003; Stanley, 1996). Sampled discharge flows were generated if the pond height exceeded the upstand riser height during a storm event (i.e. 'overflow discharge') (Fig.

4.1), and when the valve at the base of the riser was opened to release the pond at the end of the event treatment, typically on the third day of ponding (i.e. 'release discharge').

Throughout all ponding at both sites, an intractable leak at the connection point of the outlet valve pipe and the base of the upstand riser generated a continual flow of ~2 -4 m^3 h^{-1}. Attempts at sealing this leak during the study period were unsuccessful. During 4 events at the Hauraki site, and 3 events at the Awahou site, auto-samplers were programmed to collect samples of the leak in order to characterise the TP and DRP concentrations of this discharge.

5.2.4 Sample analysis

Water samples were collected from the field within 24 h of the end of the ponding event and kept refrigerated at 4 °C prior to subsampling (within ~24 hr of collection). Separate subsamples (~30 mL) were taken from the field sample for TP and the DRP analysis. The DRP subsamples were filtered (<0.45 µm) and both the filtered and unfiltered subsamples were subsequently frozen until analysis. Unfiltered TP subsamples were digested using the alkaline persulphate digestion method of Hosomi and Sudo (1986). Both the digested and the filtered DRP subsamples were analysed for P concentrations following the standard molybdenum blue method (Murphy & Riley, 1962) using automated flow injection analysis (QuikChem 8000 FIA+; Lachat Instruments, Loveland, CO).

5.2.5 Mean flow proportional concentrations

The same calculations used to determine the mean flow proportional (MFP) concentrations of sediments in Chapter 4 were used to calculate the event and annual MFP TP and DRP concentrations in this present study, by dividing the inflow and discharge loads by their respective volumes (Tanner & Sukias, 2011). The average difference between MFP inflow and leak samples collected during 7 events were -2% for TP, and +7% for DRP, and there was no consistent increase or decrease for either contaminant. Therefore, the MFP inflow concentration was applied to the entire leak volume for each respective event in which the leak discharge was not sampled. The applied leak concentration was used in calculating each event's MFP discharge concentrations. All event inflow and discharge MFP concentrations will be referred to as inflow and discharge concentrations. Changes to concentrations were calculated as the percent difference between inflow and discharge concentrations.

5.2.6 Loads and yields calculation

The same calculations used to determine load and yields for sediments in Chapter 4 were used to determine TP and DRP loads for the inflow, and each discharge type for each ponding event in this present study. Loads of TP and DRP were calculated by multiplying the measured concentration of the runoff samples and interpolated concentrations based on the linear rate of change between measured concentrations, by the interval flow volume measured every 5 minutes. Inflow loads were corrected on a pro rata basis (15% increase at the Hauraki site and 9% increase at the Awahou site) to account for the small catchment area between the inflow monitoring site and the DB (Table 5.1).

Discharge loads were calculated for overflow discharge (combining upstand riser and spillway breaching), release discharge (which occurred during Overflow events and Release events), and leak discharge (which occurred throughout each ponding event). The upstand riser overflow discharge loads and release discharge loads were calculated from flow measurements and sample concentrations taken from the DB outlet pipe. Leak loads were calculated by multiplying the leak volume by the respective event's MFP inflow concentration. Emergency spillway loads were calculated by multiplying the mean overflow discharge concentration measured in this current study by the volume breaching the spillway reported in Chapter 3. Yields refer to the load per unit of contributing catchment area, therefore runoff volumes are expressed as mm, and loads are expressed as kg ha^{-1}. The percent difference between inflow and discharge yields were reported as 'yield treatment efficiencies'.

Chapter 3 suggested that discharges occurring while runoff was not being generated in the catchment was likely to infiltrate the soil downstream of the DB before reaching downstream surface waters. In this current study, DRP discharge loads that were assumed to have infiltrated the soil downstream of the DBs, and were subtracted from TP discharge loads to estimate the TP loads prevented from reaching Lake Rotorua.

5.2.7 Contributing catchment soil analysis

The range of soil types, topographies and management areas within the DB catchments at both study sites were soil sampled to a depth of 7.5cm in September 2018, to characterise the soil Olsen P (Olsen et al., 1954), and anion storage capacity (ASC), which is the standard single point P sorption measure used in New Zealand (Saunders, 1965). These data, in addition to farm management data, were used to parametrise the

OverseerFM® model in order to estimate the soil P maintenance requirements to maintain a stable Olsen P concentration in the ponding area at each site. The amount of P required to increase the Olsen P concentration by 1 mg L^{-1} was determined from Roberts and Morton (2012).

5.2.8 Ponding area soil analysis

Soil cores were collected from the 0-7.5 cm depth using a standard soil sampler in January 2018 and January 2019, at both DB sites. Triplicate bulk samples composed of 15 soil cores were collected near the upstand riser, and outside the ponding area upstream of the bund. Samples from near the upstand riser, at the lowest point of the pond, were collected from a ~ 3 x 8m area. Elevations outside of the ponding area were confirmed with a theodolite and separated into 3 roughly equivalent sized sample zones. Samples from outside of the ponding area were composed of 5 soil samples from each of the 3 zones to avoid an over- or under- representation of the area near the ephemeral stream pathway. A sample for each of the triplicates was collected <50 cm away from each other at each of the 5 sample positions within the 3 zones (Fig. 5.2). Upon returning to the laboratory, soils were air dried for one week prior to grinding and sieving (<2 mm). Samples were then analysed for Olsen P (Olsen et al., 1954), and the mean Olsen P concentration at each elevation for each sampling year was calculated from the triplicate samples collected from both sites.

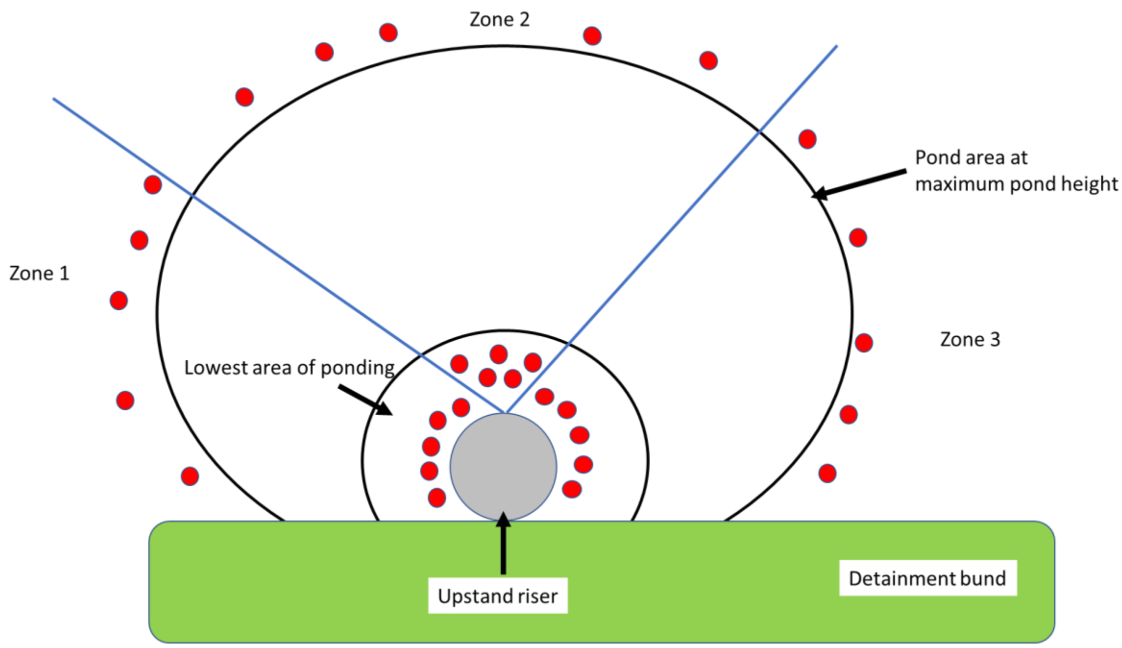

Figure 5.2: Overhead schematic displaying soil core sampling locations inside the lowest area of ponding and in the 3 zones outside of potential ponding. Red dots demarcate point where triplicate samples were collected.

5.3 Results and Discussion

5.3.1 Events

The same data and samples collected during the 18 ponding events at the Hauraki site, and 19 ponding events at the Awahou site, and reported on in Chapters 3, 4 and 6, were assessed during this present 12-month study. Inflow samples were collected during 13 of these ponding events at the Hauraki site, and 14 events at the Awahou site, since not all ponding events generated flow rates high enough to trigger auto-samplers. Discharge samples were collected during 10 events at the Hauraki site and 13 events at the Awahou site since not all events generated discharge flows to be sampled due to leakage and soil infiltration.

5.3.2 Concentration

Event TP and DRP inflow concentrations varied throughout the year, with no relationship to runoff magnitudes at either site (Fig. 5.3). Event inflow concentrations of TP and DRP tended be lowest during the winter at both sites, although this was more consistent at the Awahou site than the Hauraki site.

Various factors affected the concentrations of TP and DRP delivered to and discharged from the DBs in this study in runoff. The temporal variability in P concentrations mobilised in runoff throughout the year were influenced by changes controlling the effective depth of interaction (EDI) between the soil and runoff, such as climate conditions and land management activities (Heathwaite & Dils, 2000). The rainfall period has been found to exert the greatest influence determining the EDI, while other factors include soil characteristics such as soil surface conditions, soil P sorption capacity, and soil type (Ahuja et al., 1981).

Soil P tends to increase during warmer, drier months, due to mineralisation of organic P, and decrease in winter due to more frequent and increased runoff and/or leaching (Abell et al., 2013; Lucci et al., 2012). High concentrations of sediments and P have been observed during the first storm events after prolonged drought conditions, due to a gradual accumulation of solutes and particulates (Bieroza et al., 2019). Phosphorus concentrations in runoff have been found to both increase or decrease with increased stormflow as a result of processes such as flushing from CSAs, or dilution by rainwater (Abell et al., 2013).

The form of P, whether particulate-bound P (PP), or dissolved in solution, would affect the susceptibility of P to mobilised by runoff (Ward et al., 1985). The vulnerability of soil to physical damage, and the relative magnitude of PP transferred in surface runoff depends on soil type, soil P concentration, soil P sorption capacity, rainfall intensity, the rate of flow, pasture-plant cover, stocking rate and slope (McDowell & Wilcock, 2007).

During this present study, DRP made up a higher proportion of TP at the Hauraki site compared to the Awahou site (Table 5.2). Greater quantities of suspended sediments in runoff, which were found to occur at the Awahou site and reported on in Chapter 4, could have decreased the concentration of dissolved P during transport via P sorption, since finer sediments with high sorption capacities are preferentially mobilised by surface runoff (Sharpley et al., 1981b). Particulate P has been found to be responsible for the majority of P loss from pastures where effluent or manure has been recently spread, since most P in effluent or manure is in small, easily mobilised particulate form (McDowell et al., 2008).

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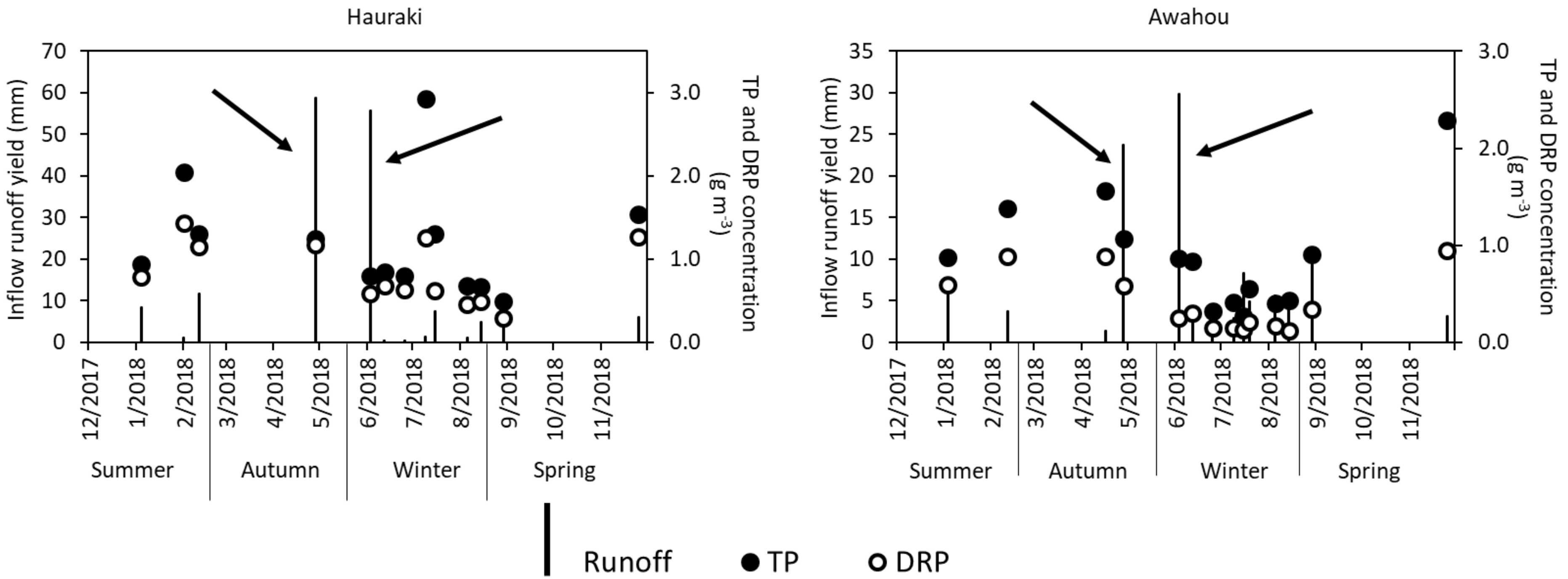

Figure 5.3: The inflow runoff yields (mm), and mean flow proportional total P (TP) and dissolved reactive P (DRP) concentrations (g m^{-3}) of inflows and discharges, for each event at each site, with arrows pointing to high runoff magnitude Overflow Events. Dates are presented as month and year with austral seasons labelled. Note: The 'Inflow runoff yield' y-axis difference between the sites.

Table 5.2: Annual mean, and event type range, of the mean flow proportional (MFP) inflow and discharge concentrations (g m^{-3}) of total P (TP) and dissolved reactive P (DRP), and the percent change in MFP concentrations comparing inflows and discharges (%).

		MFP TP concentration (g m^{-3})			MFP DRP concentration (g m^{-3})		
Site	**Event type**	**Inflow**	**Discharge**	**Change**	**Inflow**	**Discharge**	**Change**
Hauraki	Annual	1.03	0.93	-10%	0.85	0.73	-14%
	Overflow range	0.80 –1.25	0.70 – 1.10	-12% – -11%	0.58 –1.18	0.51 – 0.94	-20% – -13%
	Non-Overflow range	0.49 – 2.93	0.55 – 3.27	-12% – +113%	0.29 – 1.27	0.31 – 2.79	-15% – +119%
Awahou	Annual	0.81	0.57	-30%	0.35	0.29	-18%
	Overflow range	0.86 – 1.07	0.42 – 0.61	-61% – -29%	0.25 – 0.58	0.26 – 0.28	-55% – +14%
	Non-Overflow range	0.27 – 2.29	0.12 – 1.58	-55% – +22%	0.11 – 0.94	0.06 – 0.92	-54% – +42%

The TP inflow and discharge concentrations of each event exceeded levels that stress Lake Rotorua (0.013 g TP m^{-3}) (ANZG, 2018) (Table 5.2). Event inflow and discharge concentrations were also well above the most recently reported median concentrations of TP from 2013-2017 (0.02 g m^{-3}) and DRP from 2009-2013 (0.0002 g m^{-3}) in Lake Rotorua (Stats NZ, 2019) (Table 5.2). High inflow concentrations were not surprising since surface runoff would have interacted with high P status soils (Olsen P concentrations >34 mg L^{-1} at both sites) (Table 5.1), as well as fertilisers and deposited faecal matter, which could have been mobilised, and delivered PP and dissolved P to the DBs (McDowell et al., 2001).

Annual TP and DRP concentrations decreased to a greater degree, and event concentrations decreased more consistently, at the Awahou site compared to the Hauraki site (Tables 5.2 and 5.3). Complex physical and chemical interactions between variables determining the fate of mobilised sediments and P likely contributed to the wide range of concentration treatment efficiencies of TP and DRP between event types and sites in this study (Table 5.2) (Letcher et al., 1999; McDowell et al., 2004). Additionally, the pro rata correction inaccurately estimating the contribution of unmeasured P from the DB catchment downstream of the inflow sampling location likely influenced concentration treatment efficiency results. Although fertiliser was not applied to the ponding areas during the current study, mobilisation of sediments, high soil P levels, deposited animal excreta, and soil treading, in the ponding area, would likely have had some impact on the concentration of contaminants discharged from the DBs.

Table 5.3: The number of events (n=) in which the mean flow proportional (MFP) concentration of suspended sediments (SS), total P (TP) and dissolved reactive P (DRP) discharge was lower than the inflow, the mean percentage increase during events when the contaminant MFP concentration increased, and the mean percentage decrease during events when the contaminant concentration decreased.

Site	Contaminant	Number of events MFP concentrations decreased (n=)	Mean increase during events concentration increased (%)	Mean decrease during events concentration decreased (%)
Hauraki[A]	SS	7	75	31
	TP	4	22	11
	DRP	3	16	16
Awahou[B]	SS	9	15	26
	TP	9	13	26
	DRP	7	24	19

[A]10 events analysed, [B]13 events analysed

Sedimentation has been identified as the primary mechanism involved in mitigation strategies affecting surface runoff contaminant concentrations (Stanley, 1996). As such, the decreases in TP concentrations observed in this study would occur when delivered sediment-bound P underwent sedimentation, and/or when dissolved P concentrations decreased due to adsorption onto sediment particles that were deposited in the ponding area, as found in previous studies (Brown et al., 1981; McDowell et al., 2006; Sharpley et al., 1981b). Any P discharged from the DB was either in dissolved form, and/or bound to sediments too small to settle out in the pond, as previously observed by Brown et al. (1981).

The investigation in Chapter 4 found that annual SS concentrations decreased 28% at the Hauraki site, and 29% at the Awahou site. Variations in particle sizes delivered to the DBs, which were not measured in this study or in Chapter 4, could have affected TP and DRP concentration treatment efficiencies, and contributed to the differences in concentration changes observed between events and the sites in this present study. This is because large particles that have greater densities settle more readily, but have less P sorption sites available for P enrichment, compared to smaller sized particles with lower densities that are less likely to settle, and more likely to be transported and discharged

(McDowell et al., 2003; Sharpley, 1985). These factors could also explain the greater SS concentration treatment efficiencies reported in Chapter 4, compared to TP concentration treatment efficiencies observed in this present study, and SS concentration treatment efficiencies exceeding TP concentration treatment efficiencies in studies of sedimentation ponds (Brown et al., 1981) and a dry detention pond (Harper et al., 1999).

The number of events in which SS concentrations decreased compared to TP and DRP concentrations differed at the Hauraki site, but were similar at the Awahou site (Table 5.3). The greater consistency of TP and SS concentrations decreasing at the Awahou could be due to SS inflow concentrations being much higher at the Awahou site (96 g m^{-3}) than the Hauraki site (17 g m^{-3}) (Chapter 4). These results suggest sedimentation processes facilitated by the DBs treat TP concentrations more effectively than ponding affects DRP concentrations. Therefore, factors affecting the sediment and DRP concentrations in runoff, such as seasonal hydrologic conditions and land management factors, are likely to cause performance variations temporally at the same site, and spatially between sites (Pionke et al., 1996). For instance, the greatest proportion of annual runoff inflow and the lowest proportion of TP as DRP in inflow occurred during the winter at both sites, which is likely due more frequent and increased runoff and/or leaching (Abell et al., 2013; Lucci et al., 2012), and coincided with the lowest TP and DRP concentration treatment efficiencies (data not shown).

We might expect the size of the ponding area could affect the build-up of soil P in the ponding area and therefore the concentration treatment efficiencies, since the P load attenuated in the ponding area, which is described in the next section, would be more concentrated in a smaller ponding area, compared to a larger one. However, the annual TP and DRP concentration decreased to a greater extent at the Awahou site (Table 5.2), which had a smaller ponding area, and higher ASC and a lower mean Olsen P concentration in the ponding area, compared to the Hauraki site (Tables 5.1 and 5.4). These results suggest soils with lower ASCs and greater soil P concentrations building up in the ponding area, likely contributed to the less consistent event concentration decreases, and lower annual concentration treatment efficiencies observed at the Hauraki site during this study.

Table 5.4: Mean Olsen P concentrations (mg L^{-1}) in the ponding area and outside of the ponding area measured in January 2018 and January 2019. Standard deviations are reported in parenthesis.

	Hauraki		**Awahou**	
	2018	**2019**	**2018**	**2019**
	Olsen P (mg L^{-1})			
Ponding area	135 (14)	142 (18)	42 (10)	45 (1)
Outside ponding area	51 (8)	50 (3)	40 (5)	41 (5)

Besides the ponding area potentially contributing dissolved P to ponded surface runoff, events in which DRP concentrations were higher in discharges than inflows could be a result of rapidly exchangeable sediment-bound P undergoing desorption in the ponding area (McDowell & Sharpley, 2001). During 4 events (3 at Awahou and 1 at Hauraki) the DRP concentrations increased while TP concentrations decreased, suggesting desorption could have occurred. These events occurred in June and July suggesting seasonal variations might have influenced these results. During this present study, the winter months had more consistently wetted soils likely contributing to greater erosion and associated sediment-bound P in runoff, along with lower DRP inflow concentrations during the winter due to due to more frequent and increased runoff and/or leaching which has been observed in other studies investigating P in runoff in New Zealand (Abell et al., 2013; Lucci et al., 2012).

5.3.3 Loads and yields

Impeding stormflow with DBs effectively decreased annual TP and DRP loads discharged from Hauraki site by 39% and 41%, respectively, and 60% and 53% at the Awahou site. Annual TP loads were reduced from 94.9 kg to 58.3 kg at the Hauraki site, and from 18.2 kg to 7.3 kg at the Awahou site, and annual DRP loads were reduced from 78.0 kg to 45.8 kg, and from 7.9 kg to 3.7 kg at the sites, respectively.

The percentage decrease in the TP and DRP loads observed in this present study exceeded the decrease in annual runoff volumes discharged from the DBs measured in Chapter 3, which were 31% and 43% lower than inflows at the Hauraki and Awahou DBs, respectively (Fig. 5.4). The results suggesting that TP decreased to a greater degree than runoff, and event DRP concentrations decreased occasionally, positively supporting the hypothesis that soil infiltration, sedimentation and sorption would contribute to decreased P loads discharged from the DB catchments. The data suggests that soil infiltration is the primary mechanism responsible for yield treatment efficiencies at both sites, which emphasises the importance of optimising DB site selection and design for soil infiltration, in order for the strategy to most effectively mitigate P leaving pastures in surface runoff.

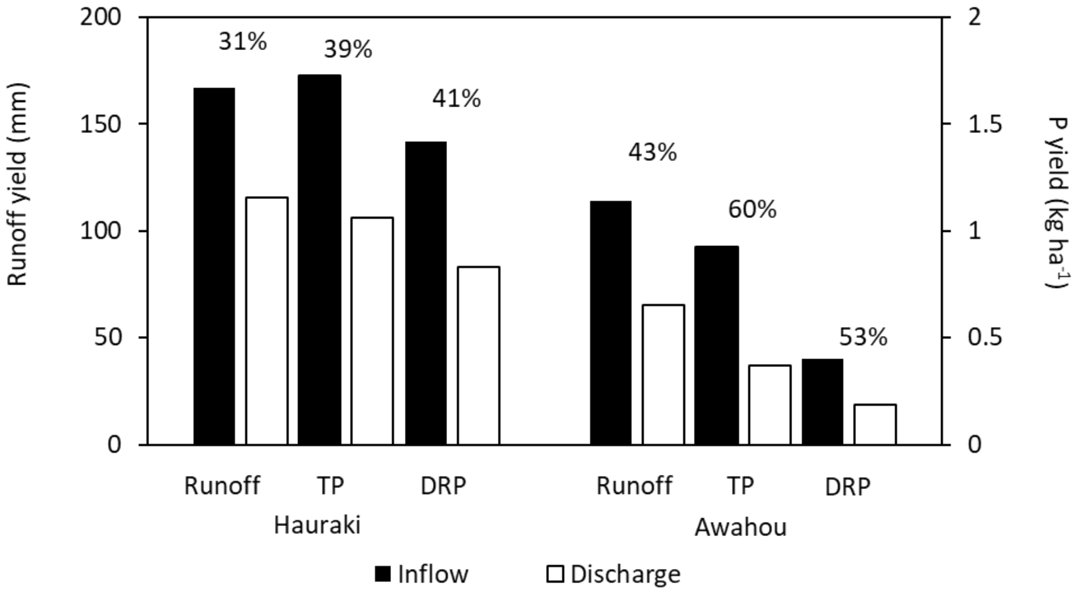

Figure 5.4: Annual total inflow and discharge runoff yields (mm), and TP and DRP yields (kg ha^{-1}) occurring at both sites. Percentage decrease in yields are also shown (%).

The annual proportion of TP as DRP for inflows was 82% at the Hauraki site, and 43% at the Awahou site, which is consistent with the higher SS inflow concentrations at the Awahou site reported in Chapter 4. During this present study, decreases in DRP loads made up 88% of the TP load attenuated in the ponding area at the Hauraki site, and 38% at the Awahou site. These results suggest that DBs are able to effectively prevent DRP from being discharged from the DB catchments, which is important since DRP is highly bioavailable, and can immediately contribute to algal blooms when conditions are conducive to primary productivity, particularly during the summer (Correll, 1998). Additionally, the DRP discharged from the DB while runoff was not being generated in the catchment likely infiltrated the soil prior to reaching downstream surface waters. When combining the TP and DRP loads attenuated in the DB ponding area with the discharged DRP loads infiltrating downstream of the bunds, the annual TP load prevented from reaching downstream waterways in surface runoff was 44.4 kg P at the Hauraki site, and 12.4 kg P at the Awahou site, which accounted for 47% and 68% of the annual inflow loads at each site, respectively. These TP loads estimated to be prevented from reaching downstream surface waters is likely conservative since we assumed any sediment-bound P discharged from the DB was likely to reach downstream surface waters due to the potential for sediment remobilisation during subsequent runoff events. However, some of the sediments could be permanently entrained, and/or desorption could occur, releasing dissolved P that could be taken up by plants and/or is mobilised by subsequent runoff and is sorbed deeper in the soil profile if that runoff infiltrates the soil. These results highlight the impact of increasing surface runoff residence times by impeding stormflows on pastures with DBs, which facilitated soil infiltration during ponding and downstream of the bund, and along with sedimentation, prevented large portions of annual P loads from reaching downstream surface waters.

Annual inflow yields at the Hauraki site were 1.7 kg TP ha^{-1} and 1.4 kg DRP ha^{-1}, and 0.9 kg TP ha^{-1} and 0.4 kg DRP ha^{-1} at the Awahou site. The annual TP inflow yields at both sites exceeded the average P yield delivered to Lake Rotorua from pastures in the catchment (0.84 kg P ha^{-1} y^{-1}) (Bay of Plenty Regional Council, 2012). Event inflow yields of TP and DRP varied throughout the year, and greater TP and DRP yields tended to correspond with greater runoff yields at both sites (Fig. 5.5). Loads of sediment and nutrients transported from pastures to receiving water bodies has been found to be largely dictated by the volume of runoff during a runoff event (Braskerud et al., 2000; McDowell et al., 2008), with the total event rainfall during runoff events found to be a good predictor of TP loading in receiving waters (Ockenden et al., 2016).

The DB decreased the TP yield discharged from the Awahou site to below the catchment average (0.37 kg P ha^{-1}), however this was not the case for the Hauraki site (1.1 kg P ha^{-1}). The difference in these results could be due to inflow yields being greater at the Hauraki site than the Awahou site, as well as the factors affecting the yield treatment efficiencies described in the following paragraphs.

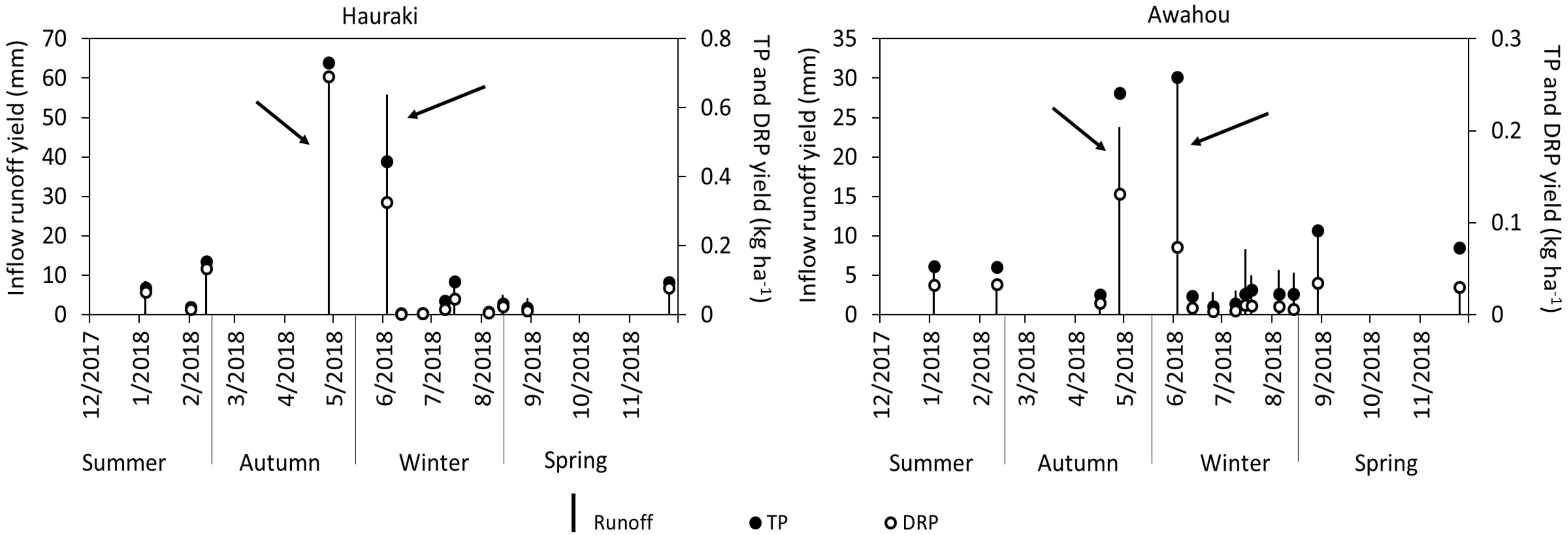

Figure 5.5: Event runoff (mm), total P (TP), and dissolved reactive P (DRP) (kg/ha) inflow yields at each site during each event in this study, with arrows pointing to high magnitude Overflow Events. Dates are presented as month and year with austral seasons labelled. Note: Both y-axes are different between the sites.

Although the high runoff magnitude Overflow Events accounted for only 2 of the 18 events at the Hauraki site and 2 of the 19 events at the Awahou site, they were responsible for the majority of the annual TP and DRP inflow and discharge yields at both sites (Fig. 5.6). Similarly, Chapter 3 reported that Overflow Events were responsible for 69% and 47% of the annual runoff inflow yields at the Hauraki and Awahou site, respectively. Results of this current study, and concurrent studies, are consistent with studies that found the majority of P loading in streams in the Lake Rotorua catchment were the result of large, rare storm events (Abell et al., 2013; Dare, 2018). Studies have also found that larger storms generating more intense runoff may mobilise greater quantities of P from New Zealand pastures than smaller runoff events (Cooke, 1988; Smith & Monaghan, 2003) and a few large storm events can be responsible for the majority of P losses from New Zealand agricultural catchments (Cooke, 1988; Rutherford & Timpany, 2008; Smith, 1987).

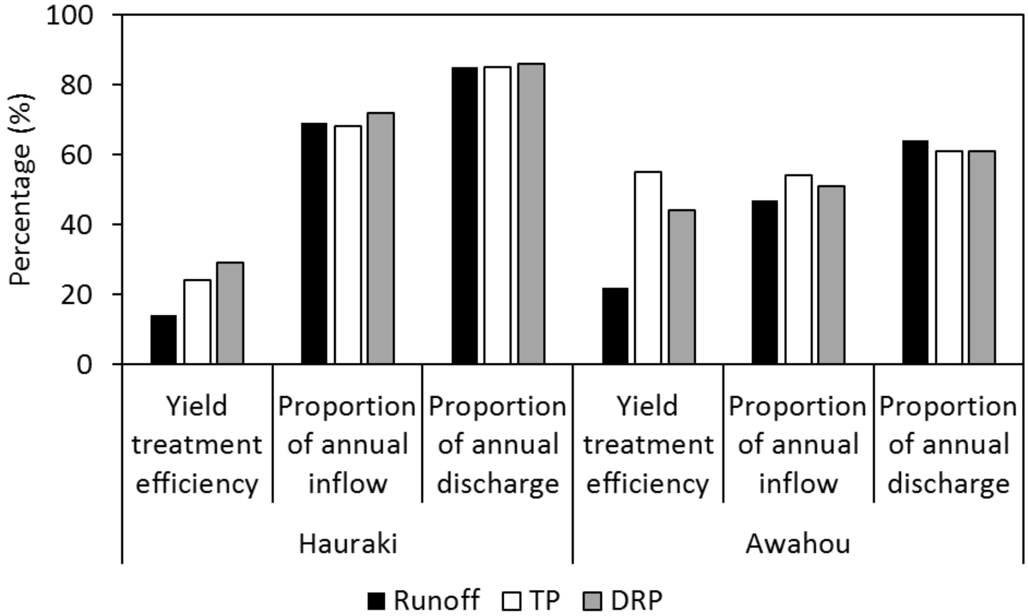

Figure 5.6: Yield treatment efficiencies (%) during Overflow Events, and the proportions (%) of annual inflow and discharge yields of runoff, total P (TP) dissolved reactive P (DRP) occurring during high runoff magnitude Overflow Events at each site.

Cumulative runoff, TP and DRP inflow yields during Overflow Events were much greater at the Hauraki site than the Awahou site (Fig. 5.6). The greater cumulative inflow runoff delivered to the Hauraki site during Overflow Events caused a greater proportion of the inflow to be discharged from the DB as overflow discharge (i.e. over the top of the upstand riser and emergency spillway). The difference in the proportion of inflow being discharged as overflow discharge likely contributed, at least partially, to the Awahou DB decreasing runoff, TP and DRP yields more effectively than the Hauraki DB during the Overflow Events in this study (Fig. 5.6).

The yield treatment efficiencies of TP and DRP were greater than the proportion of runoff infiltrating the soil at both sites during the Overflow Events (Fig. 5.6). Also, attenuated yields of TP and DRP during Overflow Events accounted for 42% and 51% of the annual yield decreases at the Hauraki site, respectively, and 50% and 42% at the Awahou site. These results demonstrate that in addition to being effective during smaller, more frequent runoff events, DBs are also capable of decreasing P yields during rare, high magnitude runoff events, which contributed to a large portion of the annual load decreases, despite runoff yields not infiltrating the soil to as great a degree (Fig. 5.6).

Results comparing mean Olsen P concentrations in the lowest ponding area and outside the ponding area, as well as changes in concentrations of soil P over the course of the study suggest P attenuated in the ponding area contributed to the increased soil P at both sites (Table 5.4). The differences in soil P concentration increases was likely affected by the soil P maintenance requirement and ASC for the soil type at each site (46% at Hauraki, and 85% at Awahou), and the load of P attenuated in the ponding areas. Data suggests that the P loads estimated to have been deposited in the lower portion of the ponding area were similar to the calculated loads necessary to achieve the observed soil P concentration changes (Table 5.5). Potentially, some of the DRP attenuated in the ponding area infiltrated deeper into the soil than the layer sampled via macropores, which could explain why the values presented in the study are slightly higher than the P loads necessary to obtain the observed increases in soil P concentrations.

Table 5.5: Phosphorus yields required to maintain and increase soil Olsen P concentrations *(Roberts & Morton, 2012)*, and the P yield estimated to have been deposited in the average ponding area at each site.

Site	**Maintenance P yield**	**Yield required to raise soil P concentration by 1 mg L^{-1}**	**Yield required to reach observed change**	**Yield deposited in average ponding area during study**
	kg P ha^{-1} y^{-1}	kg P ha^{-1} y^{-1}	kg P ha^{-1} y^{-1}	kg P ha^{-1} y^{-1}
Hauraki	53	7	102	119
Awahou	36	18	90	135

Olsen P concentrations at the lowest pond elevation were ~85 mg L^{-1} area in June 2012 at the Hauraki site (historical data did not exist for the Awahou site) (Clarke, 2013). Based on these results, 115 kg P ha^{-1} y^{-1} would need to have been deposited in the lower ponding area between the historic and current studies at the Hauraki site, to increase Olsen P concentrations by the observed ~8.8 mg L^{-1} y^{-1}. This value is very close to the 119 kg P estimated to have been deposited across the lower ponding area during this present study, suggesting that the load attenuated during this present study could be typical compared to previous years, which corresponds with the similarity between rainfall during this current study period and the 10-year average rainfall occurring at the sites reported in Chapter 3.

5.4 Implications of detainment bund treatment results

This study reports, for the first time, that DBs were able to effectively decrease annual P loads transported from pastures in surface runoff as a result of the combination of soil infiltration, sedimentation and sorption of dissolved P, positively supporting the hypothesis set out in section 5.1. The results of this current study suggest DBs were able to consistently decrease TP and DRP loads discharged from the DB catchments during every storm event in this study period, even during rare, high magnitude runoff events. This is a significant finding since some land management and edge of field strategies may be overwhelmed by extreme hydrologic conditions (Kleinman et al., 2006; McDowell & Sharpley, 2002). Also, since large storm events have been found to be responsible for the majority of P loading to streams in the Lake Rotorua catchment (Abell et al., 2013; Dare, 2018), DBs have the potential to be a particularly effective and important mitigation option for pastoral farmers in the area.

The performance of DBs reported in this current study are made more significant when considering that processes in headwaters of temperate catchment areas are likely to dominate the hydrochemical responses of downstream surface waters to rainfall (Bieroza et al., 2018; Ockenden et al., 2016). Due to the limited viable locations determined by landscape and regulatory parameters, DBs are most likely to be located on pastures in the headwaters of the Lake Rotorua catchment (Paterson, 2019) which is important to note since pastoral catchments have been found to account for an average of 73% of the annual loads of DRP delivered to low-order streams in New Zealand (McDowell et al., 2017). Also, since nutrient losses in runoff from pastures are likely to increase due to more dramatic hydrological conditions driven by climate change, the ability of DBs to attenuate

P transported in surface runoff will become more important over time (Ministry for the Envrironment, 2019; Ockenden et al., 2016).

The annual TP and DRP concentrations discharged during each ponding event remained well above the median concentrations in Lake Rotorua at both sites, and the TP yield discharged from the Hauraki sites remained above the average yield lost from pastures in the catchment. These results suggest that while DBs are effective at reducing annual TP concentrations and yields, improvements to DB performance along with the implementation of other mitigation strategies needs to take place in order for pastoral farmers to make greater progress towards reducing their impact on Lake Rotorua water quality.

While this study found that locating DB ponds on soil types with sufficient infiltration capabilities gives the potential to achieve the greatest yield treatment efficiencies, the data suggests utilising DBs on soils with lower infiltration rates than those in this present study could still achieve some P load reductions, due to strategy's ability to facilitate sedimentation. Besides optimising DBs for soil infiltration, adopting approaches to improve concentration treatment efficiencies would also increase yield treatment efficiencies. The results of this current study reporting an increase in soil P concentrations in the ponding areas suggest that ponding areas, particularly those in the lower portions where ponding occurs most often, and on pastures with low ASC soils, should be strategically managed in order to avoid ponding areas becoming a P source to discharged runoff. Strategic management of ponding areas could include cut-and-carry management approaches, and fencing off lower ponding areas to avoid excess treading to decrease erosion, and potentially decrease impacts to soil infiltration rates. Also, improving treatment efficiencies by integrating methods to sorb dissolved P with the DB strategy, such as P socks or alum dosing (Tempero et al., 2015), or using flocculants to aggregate P enriched soil particles to reduce mobilisation of P-enriched sediments (Braskerud, 2002b), should also be investigated.

Lastly, revising the DB design to avoid releasing sediment-bound P that is likely to be remobilised and delivered to downstream surface waters in subsequent runoff events should be considered. This could be achieved by preventing leak and release discharges by removing the upstand riser/outlet valve/discharge pipe installation (Fig. 5.1). However future studies should investigate the costs and benefits of this approach, since longer inundation periods could reduce pasture productivity. During this present study, removing the upstand riser structure would have increased the TP load prevented from reaching Lake Rotorua by approximately 3.3 kg P at the Hauraki site, and 1.6 kg P at the Awahou site, increases of 7% and 12% of the annual load prevented from reaching Lake Rotorua, respectively. Some increase in the TP load prevented from reaching Lake Rotorua could also be achieved raising the outlet valve to 10 cm above ground-level, with less potential impacts to pasture productivity, although the difference in loads would not be as profound as avoiding the release and leak discharges entirely.

5.5 Conclusion

This current study found that 2 DBs located on pastures in headwaters of the Lake Rotorua catchment attenuated 39% to 60% of the annual TP load and 41% to 53% of the annual DRP loads discharged from the DB catchments. When including the portion of DRP discharged from the DB likely to infiltrate the soil before reaching downstream surface waters, this study estimated that 12.3 kg TP y^{-1} and 44.3 kg TP y^{-1} was prevented from reaching the lake, decreases of 68% and 47%, respectively. While soil infiltration was primarily responsible for the yield treatment efficiencies overserved in this study, the data suggests sedimentation processes facilitated by impounding surface runoff may effectively decrease P loads where sediment-bound P makes up a large proportion of TP. Therefore, DBs could be effective where soil infiltration rates are not as high as those in this current study.

Identifying the ability of DBs to effectively decrease P loads during large storm events is an important finding of this study, since it is common for large, but less frequent runoff events to be responsible for the majority of annual TP loading into Lake Rotorua from the catchment. Also, the ability of DBs to consistently decrease TP loads in storm generated surface runoff, and the significant role headwater subcatchments play in determining downstream hydrochemical responses to rainfall events, highlight the significance of identifying the effectiveness of DBs as a strategy to mitigate annual anthropogenic P loading from pastures.

Results of soil P testing at the study locations suggest ponding areas could increase P concentrations in runoff and decrease treatment efficiencies in the long-term. Future studies should investigate strategies to improve treatment efficiencies such as implementing cut-and-carry management in the ponding area, and integrating the use of flocculants and sorbents. This study should be expanded upon to collect longer-term data from more DB locations, and results should be incorporated into nutrient budgeting and management models that allow policy makers, consultants and farmers to account for the ability of DBs to reduce P loading in downstream surface waters.

While the main objective of utilising the DB strategy in the Lake Rotorua catchment is to address P losses from pastures in surface runoff which contributes to eutrophication, there is also the potential for ‘nitrogen (N) by-catch’ which will decrease the nitrogen load reaching the lake which also contributes to eutrophication. In the following chapter we test the hypothesis that the factors contributing DBs effectively decreasing sediment-bound and dissolved P losses described in this present chapter, may also decrease N loads being delivered to Lake Rotorua.

CHAPTER 6: Nitrogen

Research highlights

- An estimated 10% of annual total N losses from pastures occurred in surface runoff.
- Annual concentrations of organic N and dissolved inorganic N in surface runoff decreased as a result of detainment bund treatment.
- Concentration decreases and soil infiltration combined to decrease N load discharges
- Detainment bunds prevented 86 and 51 kg N from reaching Lake Rotorua
- Nitrogen 'by-catch' is an added benefit of utilising detainment bunds

6.1 Introduction

Surface runoff from intensively managed pastoral agriculture typically has elevated concentrations of nitrogen (N) (Ledgard et al., 1999). Nitrogen transported in surface runoff is highly reactive, and may undergo chemical transformations, assimilation and plant uptake, and permanent removal via denitrification (Alexander et al., 2007). Ammonium and nitrate are forms of dissolved inorganic nitrogen (DIN) which are capable of stimulating primary productivity, and cause eutrophication in N-limited aquatic systems (Wurtsbaugh et al., 2019). Decomposition of organic matter releases biologically available DIN, mostly as ammonium, which can subsequently be nitrified (Burger et al., 2007; McDowell et al., 2013).

Rainfall, and subsequent runoff, mobilise and transport organic N and DIN from pastures to Lake Rotorua and contribute to lake eutrophication (Burger et al., 2007) . Fertiliser inputs, year round grazing and associated excrement deposits contribute to N losses being greater from intensive pastoral agriculture compared to other rural land uses in New Zealand (Elliott, 2005). Most of the organic N and ammonium is exported from such pastures in the Lake Rotorua catchment by surface runoff, since ammonium is readily adsorbed onto the silicate clay and organic matter with high cation exchange capacities (CECs) common in the catchment's soils, during subsurface transport (McDowell et al., 2008; Reddy & DeLaune, 2008). Nitrate leaching out of the root zone and deep into groundwater is prevalent in the Lake Rotorua catchment due to fertiliser applications and the presence of livestock depositing concentrated patches of urine onto well-drained pasture soils (Morgenstern et al., 2015). Since the groundwater in the catchment is relatively oxic, nitrate leaching out of the root zone is unlikely to undergo significant denitrification and is a major source of N loading to Lake Rotorua (Morgenstern et al., 2015).

Internal and external sources of N contribute to Lake Rotorua eutrophication (Burger et al., 2007; Donald et al., 2019). Pastoral agriculture in this area, which covers ~48% of the 42,000 ha Lake Rotorua surface catchment, loses an estimated 29 kg N ha^{-1} y^{-1} and is responsible for 578 t N y^{-1} (77%) of the annual N loads delivered to the lake (Donald et al., 2019). The 2012 Lake Rotorua Management Plan has set targets to reduce N loads to the lake from 2012 losses of 755 t N y^{-1} to 435 t N y^{-1}, in order to achieve lake water quality objectives (Bay of Plenty Regional Council, 2012).

Multiple land management mitigation strategies will need to be implemented, including potentially novel approaches, to reach the N load reduction goals required to achieve Lake Rotorua water quality objectives (McDowell, 2010). Climate change contributes to more dramatic hydrological conditions with increased storm intensities, wetter winters, and hotter, drier summers, making nutrient loading reductions from the catchment more challenging (Ministry for the Envrironment, 2019; Ockenden et al., 2016). The increased runoff magnitudes, and flashiness of flows caused by climate change will likely increase N loads lost in runoff from pastures, and decrease the natural processing of N in low order streams (Alexander et al., 2007).

Some mitigation strategies that target N will have phosphorus (P) ‘by-catch’, and vice versa (Donald et al., 2019). Detainment bunds started being utilised as a novel mitigation strategy to address P losses in surface runoff from pastoral agriculture in the Lake Rotorua catchment in 2010 (Clarke, 2013). A DB is an earthen stormwater retention structure, ~20-80 m long by ~1.5-2 m high, constructed on productive pasture across the flow path of targeted low-order ephemeral streams. By impeding stormflow, DBs are capable of temporarily ponding up to 10,000 m^3 of surface runoff. The current DB design protocol recommends a minimum pond volume of 120 m^3 per 1 ha of contributing catchment, and suggests rapidly draining the pond by opening a plugged outlet valve after a 3 day holding period to avoid impairing pasture productivity (Paterson & Clarke, 2013) (Fig. 6.1).

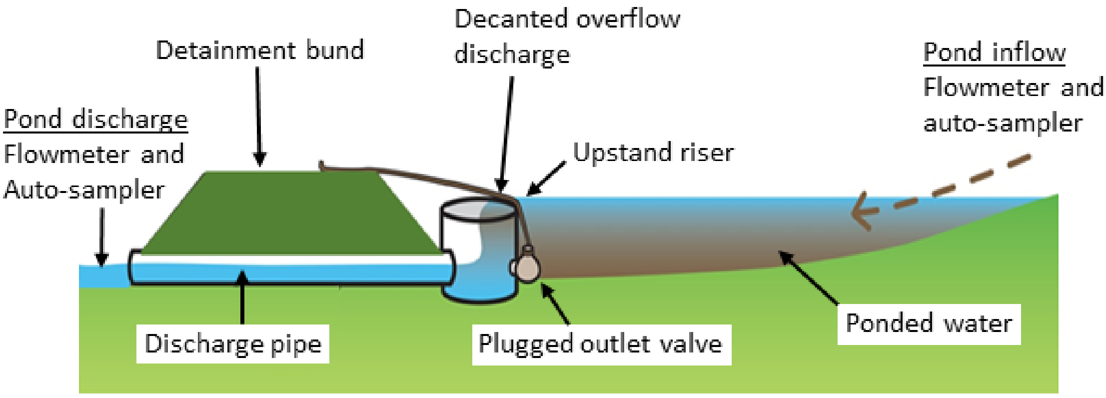

Figure 6.1: Cross-section of ponding area showing the ephemeral stream inflow ponding behind a detainment bund. If the pond height exceeds the height of the upstand riser then 'decanted overflow' is discharged via a pipe passing through the bund wall. Inflows and discharges are measured with flowmeters which triggers auto-sampler collections.

No previous studies have investigated the ability of DBs to treat N in stormflows from pastures in the Lake Rotorua catchment. The main objective of this current study was to quantify the ability of DBs to decrease N loads transported by surface runoff from pastures in the Lake Rotorua catchment and identify mechanism affecting attenuation performance. Based on the results reported in Chapters 3, 4 and 5 we hypothesised that DBs will decrease the loads of N discharged from the DB catchment in surface runoff due to sedimentation decreasing particulate bound organic N loads, and soil infiltration decreasing the volumes of runoff, and consequently dissolved N loads. Also, ammonium sorbing on to suspended sediments and chemical changes causing nitrate to be removed from surface runoff could contribute to decreased N loads discharged from the catchment. This study also calculated the TN prevented from reaching Lake Rotorua from the DB catchments, based on the potential fate of the various forms of N delivered to the DBs.

6.2 Materials and methods

6.2.1 Site descriptions and event descriptions

This current study reporting on the effect of DBs on organic N and DIN concentrations and loads coincided with concurrent studies investigating the impact of DBs on surface runoff volumes in Chapter 3, and sediment and P concentrations and loads in Chapters 4 and 5, respectively. The previous chapters presented detailed site descriptions and relevant data listed in Table 6.1.

Table 6.1: Characteristics of detainment bund (DB) sites.

Characteristic	**Hauraki**	**Awahou**
Grid Reference	38°00'21"S 176°11'03"E	38°01'43"S 176°07'54"E
Year DB constructed	October 2011	June 2012
Topography of catchment	Flat, rolling and hill	Mainly rolling
Size of DB entire DB catchment (ha)	55.0	19.7
Area of DB catchment downstream of inflow monitoring (ha)	8.3	1.8
Height of bund at spillway (m)	1.56	1.80
Height of upstand riser (m)	1.36	1.60
DB pond volume ($m^{3)}$	4,894 m^3 at upstand riser 7,110 at m^3 at spillway	1,652 m^3 at upstand riser 2,244 m^3 at spillway
Ratio of pond volume: catchment area (m^3: ha)	89:1 at upstand riser 129:1 at spillway	84:1 at upstand riser 114:1 at spillway
Pond area at pond filled to upstand riser and spillway (m^2)	9,564 m^2 at upstand riser 12,221 m^2 at spillway	2,610 m^2 at upstand riser 2,940 m^2 at spillway
Measured infiltration rates inside and outside ponding area[a] (mm/h)	Inside: 19 Outside: 36	Inside: 12 Outside: 37
Soil description	Oropi series- Free draining with >72 mm/h permeability in slowest horizon	Waiteti series- Free draining with >72 mm/h permeability in slowest horizon
Measured infiltration rates inside and outside ponding area[a] (mm/h)	Inside: 19 Outside: 36	Inside: 12 Outside: 37

[a] Chapter 3

Event types reported on in this study and the studies presented in Chapters 3, 4 and 5, were differentiated according to the mode(s) ponded water was discharged from the DB. 'Overflow Events' occurred during larger runoff events when inflow continued to be delivered to the pond after the pond height exceeded the height of the upstand riser (Fig. 5.1). After 3 days of ponding, any residual ponded water was evacuated when the outlet valve was opened, creating 'release discharge'. Therefore, 'Overflow Events' had both overflow and release discharge components. In contrast, 'Non-overflow Events' were smaller storms that did not contribute enough runoff to overtop the riser. Non-overflow Events included events when at the end of the 3-day treatment period, either had a portion of ponded runoff to discharge by opening the release valve, or all ponded runoff leaked and infiltrated the soil so there was no water left to discharge.

6.2.2 Equipment and sampling

The equipment and procedure for collecting surface flow data delivered to, and discharged from the DBs, was described in Chapter 3, and surface runoff sample collection equipment and procedures were presented in Chapter 4. Isco® (California, USA) 6712 portable auto-samplers, capable of filling 24 x 1 L bottles, collected inflow and discharge samples at each site when triggered by UNIDATA® 6527 Starflow® QSD flowmeters linked to a telemetered UNIDATA® Neon® 2013 F 3G External Memory Metering Module dataloggers. The auto-samplers were triggered to collect 1 L samples when flows exceeded 7 L s^{-1} (Harmel et al., 2002). Calibration and maintenance of the monitoring equipment followed standard quality controls (NIWA, 2004).

Inflow (i.e. upstream) auto-samplers collected a 1 L sample every 20 min for the first 10 samples, then one 1 L sample h^{-1} thereafter (Harmel et al., 2003; Stanley, 1996). A rain guarded 750-mL self-sealing bottle using a ping-pong ball inside the bottle, was installed level with the bottom of the pond outlet valve to capture a sample of the initial flush of surface runoff generated before the inflow auto-sampler was triggered. The ping-pong ball bottle sample was used as the concentration of the initial runoff and used in calculating the inflow load for each event.

Discharge auto-samplers were programmed to collect a 1-L sample (Harmel et al., 2003; Stanley, 1996). Sampled discharge flows were generated if the pond height exceeded the upstand riser height during pond filling, and when the valve at the base of the riser was opened to release the pond at the end of the event treatment.

Throughout all ponding at both sites an intractable leak at the connection point of the outlet valve pipe and the base of the upstand riser generated a continual flow of ~2 - 4 m^3/h. Attempts at sealing this leak during the study period were unsuccessful. During 4 events at the Hauraki site, and 3 events at the Awahou site, auto-samplers were programmed to collect samples of the leak in order to characterise the TN, nitrate-N and ammonium-N concentrations of this discharge.

6.2.3 Sample analysis

The same samples collected and analysed in this current study were also analysed for suspended sediments in Chapter 4, and P in Chapter 5. Water samples were collected from the field within 24 h of the end of the ponding event and kept refrigerated at 4 °C prior to subsampling (within ~24 hr of collection). Separate subsamples (~30 mL) were taken from the field sample for TN, and nitrate-N and ammonium-N analysis. The subsample used to analyse nitrate-N and ammonium-N analysis was filtered (<0.45 µm), while the TN sample was not filtered. Both the filtered and unfiltered subsamples were subsequently frozen until analysis. Unfiltered TN subsamples were digested using the alkaline persulphate digestion method of Hosomi and Sudo (1986). Filtered subsamples were analysed for concentrations of nitrate-N and ammonium-N using the FIA with Lachat QuickChem methods [10-107-04-1-A (NO^{-}_{3}-N), 10-107-06-2-B (NH_4^{+})]. References to total inorganic nitrogen (TIN) include nitrate-N and ammonium-N. Organic N was calculated by subtracting TIN from TN (Tanner & Sukias, 2011).

6.2.4 Mean flow proportional concentrations

The same calculations used to determine the mean flow proportional (MFP) concentrations of sediments in Chapter 4, and TP and DRP concentrations in Chapter 5 were used to determine MFP concentrations of TN, nitrate-N and ammonium-N concentrations in this present study, and were calculated by dividing inflow and discharge loads by their respective volume (Tanner & Sukias, 2011). There was no consistent difference between MFP inflow and leak samples collected during 7 events that were sampled for this analysis. Therefore, the MFP inflow concentration was applied to the entire leak volume for each respective event in which the leak discharge was not sampled. The applied leak concentration was used in calculating the event MFP discharge concentrations. All inflow and discharge MFP concentrations will be referred to only as inflow and discharge concentrations. Changes to concentrations were calculated as the percent difference between inflow and discharge concentrations.

6.2.5 Loads and yields calculation

The same calculations used to determine load and yields for contaminants in surface runoff in Chapter 4 and 5 were used to determine TN, nitrate-N and ammonium-N loads for the inflow and each discharge type of each ponding event in this present study. Contaminant loads were determined for the inflow, and each discharge type for each ponding event. Loads of TN, nitrate-N and ammonium-N were calculated by multiplying the measured concentration of the runoff samples and interpolated concentrations based on the linear rate of change between measured concentrations, by the interval flow volume measured every 5 minutes. Inflow loads were corrected on a pro rata basis (15% increase at the Hauraki site and 9% increase at the Awahou site) to account for the small catchment area between the inflow monitoring site and the DB (Table 6.1). Organic N loads were calculated by subtracting TIN loads from TN loads (Tanner & Sukias, 2011).

Discharge loads were calculated for overflow discharge (combining upstand riser and spillway breaching) which occurred during rare, high magnitude runoff events, release discharge (which occurred during Overflow Events and Release Events), and leak discharge (which occurred throughout each ponding event). The upstand riser overflow discharge loads and release discharge loads were calculated from flow measurements and sample concentrations taken from the DB outlet pipe. Leak loads were calculated by multiplying the leak volume by the respective event's MFP inflow concentration. Emergency spillway loads were calculated by multiplying the mean overflow discharge concentration measured in this current study by the volume breaching the spillway calculated in Chapter 3. Yields refer to the load per unit of contributing catchment area, therefore runoff volumes are expressed as mm, and loads are expressed as kg ha^{-1}. The percent difference between inflow and discharge yields were reported as 'yield treatment efficiencies'. Inflow yield data for each site was also organised by austral seasons (example: summer from December to February) to compare differences between the sites and identify seasonal patterns for N inflow yields.

Chapter 3 suggested that discharges occurring while runoff was not being generated in the catchment was likely to infiltrate the soil downstream of the DB before reaching downstream surface waters. In order to ensure conservative estimates of DB performance during this current study, it was assumed any nitrate-N infiltrating the soil would reach the groundwater and be delivered to Lake Rotorua. Therefore, to calculate the nitrate-N prevented from reaching Lake Rotorua, we considered the changes to nitrate-N loads as a result of the changes in concentration between inflow and discharges, and excluded the portion that infiltrated the soil from loads prevented from reaching the lake. Due to the prevalence of soils with high CECs, we assumed ammonium-N infiltrating the soil would be sorbed and eventually taken up by plants. We also assumed minimal organic N leaching, and that any organic N prevented from reaching downstream surface waters would decompose, and eventually be taken up by plants. Therefore, the decreases in nitrate-N loads due to the decreases in nitrate-N concentrations, the organic N and ammonium-N prevented from being discharged from the DB, and the loads of organic N and ammonium-N released when runoff was not being generated in the catchment (i.e.. discharges not including overflow discharge) were combined to determine the TN load prevented from reaching Lake Rotorua as a result of the DB treatment.

6.3 Results and Discussion

The same data and samples collected during the 18 ponding events at the Hauraki site, and 19 ponding events at the Awahou site, and reported on in Chapters 3, 4 and 5 were assessed during this present 12-month study. Inflow samples were collected during 13 of the ponding events at the Hauraki site, and 14 events at the Awahou site, since not all ponding events generated flow rates high enough to trigger auto-samplers. Discharge samples were collected during 10 events at the Hauraki site, and 13 events at the Awahou site, since not all events generated discharge flows to be sampled due to leakage and soil infiltration.

6.3.1 Concentration

Annual organic N inflow concentrations were higher at the Hauraki site, while nitrate-N and ammonium-N inflow concentrations were higher at the Awahou site (Table 6.2). No distinct temporal trends were observed for nitrate-N and ammonium-N inflow concentrations, while organic N concentrations were typically higher in the winter months at the Hauraki site, but not at the Awahou site. (Fig. 6.2). The event inflow concentrations and concentration treatment efficiencies varied greatly within and between event types at both sites (Table 6.2). Still, annual MFP concentrations decreased as a result of the DB treatment for all N species measured.

Table 6.2: Annual mean flow proportional (MFP) inflow and discharge concentrations (g m^{-3}), and event type ranges, of organic N, nitrate-N and ammonium-N, and the percent concentration changes (%) based on comparing event MFP inflow and discharge concentrations.

		Hauraki		
Concentration		**Annual**	**Overflow range**	**Non-overflow range**
Organic N	Inflow (g m^{-3})	1.12	0.59-1.14	0.52-4.5
	Discharge (g m^{-3})	0.92	0.53-0.97	0.53-4.5
	Change	-17%	-15% - -11%	-54% - +87%
Nitrate-N	Inflow (g m^{-3})	0.25	0.11-0.33	0.05-1.01
	Discharge (g m^{-3})	0.23	0.08-0.35	0.08-1.19
	Change	-10%	-30% - +8%	-32% - +78%
Ammonium-N	Inflow (g m^{-3})	0.28	0.16-0.16	0.04-6.58
	Discharge (g m^{-3})	0.15	0.07-0.16	0.05-5.38
	Change	-45%	-59% - +2%	-68% - +189%

		Awahou		
Concentration		**Annual**	**Overflow range**	**Non-overflow range**
Organic N	Inflow (g m^{-3})	2.18	2.00-2.57	0.72-4.6
	Discharge (g m^{-3})	1.61	0.94-1.85	0.53-4.5
	Change	-26%	-53% - -28%	-31% - +245%
Nitrate-N	Inflow (g m^{-3})	0.35	0.22-0.65	0.08- 1.0
	Discharge (g m^{-3})	0.27	0.24-0.27	0.03-1.0
	Change	-22%	-58% - +13%	-59% - +61%
Ammonium-N	Inflow (g m^{-3})	0.51	0.20-0.51	0.04-1.97
	Discharge (g m^{-3})	0.38	0.08-0.41	0.02-1.7
	Change	-26%	-62% - -19%	-68% - +42%

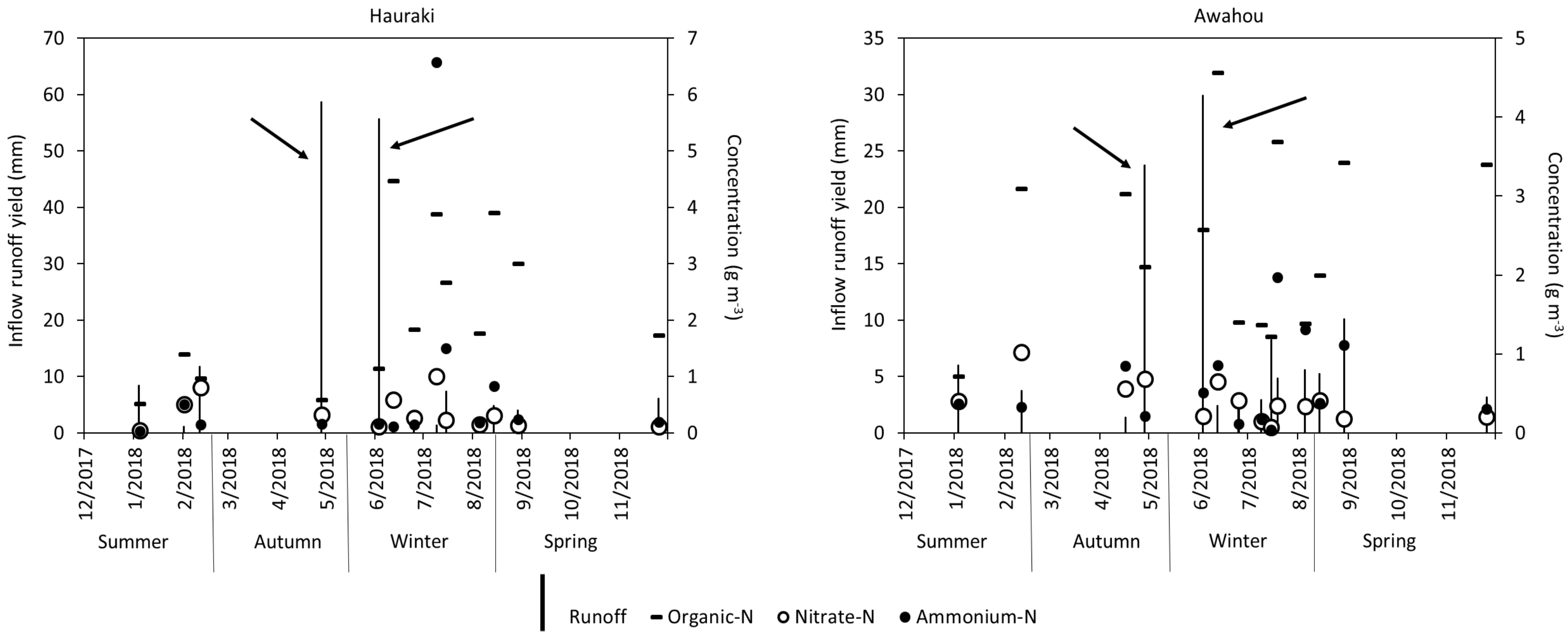

Figure 6.2: Inflow runoff yields (mm) and inflow mean flow proportional concentrations (g m-^{3}) of organic N, nitrate-N and ammonium-N for each event at each site during this study. Arrows point to Overflow Events occurring on the same date at both sites. Note different y-axis scales for runoff yields and N species concentrations between the sites.

The MFP inflow and discharge TN concentrations of each event were well above those 0.11 g TN m^{-3} concentration considered to stress lakes such as Lake Rotorua (ANZG, 2018) (Table 6.2). Event inflow and discharge concentrations in this study also exceeded the most recently reported median concentrations of N species in Lake Rotorua (0.30 g TN m^{-3} from 2013-2017, and 0.007 g NO_3-N m^{-3} and 0.008 g NH_4 m^{-3} from 2009-2013) (Stats NZ, 2019) (Table 6.2). High inflow concentrations were not surprising since surface runoff from pastoral agriculture typically has elevated concentrations of N (Ledgard et al., 1999), with studies finding ammonium-N and nitrate-N concentrations in excess of 1 g m^{-3} in surface runoff from New Zealand pastures (Smith, 1987; Smith & Monaghan, 2003).

Complex interactions between physical and chemical variables can potentially affect the fate of N delivered to, and impounded by the DBs (Ponnamperuma, 1972; Sánchez-Rodríguez et al., 2019). During ponding, increased nitrate-N concentrations may have occurred as a result of microbial nitrification of ammonium, anaerobic ammonium oxidation (anamox), or leaching from plants or soil (Reddy & DeLaune, 2008), while decreases in nitrate-N concentrations may occur through microbial denitrification, assimilatory nitrate reduction to ammonia, or uptake by microbes (immobilisation) or plants (Friedl et al., 2018; Matheson et al., 2002; Nie et al., 2019). Denitrification (and presumably also anammox) would only have been likely to occur if soils in the ponding area became anoxic, since under most conditions, very little or no denitrification is likely to occur in the water column (Reddy & DeLaune, 2008). Ammonium-N concentrations may have increased due to mineralisation of organic N, or assimilatory and/or dissimilatory nitrate reduction to ammonia, and may have decreased through sorption to soil, plant litter and suspended sediments, ammonia volatilisation from the water surface, microbial nitrification, or (in association with nitrate) anammox. Again, microbial transformations and volatilisation were unlikely to have occurred to a significant extent in the water column during the 3-day ponding period (Jayaweera & Mikkelsen, 1991).

In addition to commonly acknowledged complexities including the previously mentioned chemical and physical interactions, the DB ponding areas in this study were on productive pasture, downstream of the inflow monitoring stations. The ponding area could have therefore contributed N that was not accurately accounted for in the pro rata correction of the contributing catchment area (Table 6.1). Ammonium-N and nitrate-N could move out of the soils downstream of the inflow monitoring station and into ponded water due to diffusion, advection, bioturbation and mixing at, or near the soil-floodwater interface (Reddy & DeLaune, 2008). Elevation gradients within in the ponding area affecting inundation frequencies could have affected soil properties and influenced the soil N available for mobilisation, similar to processes in river floodplains (Woodward et al., 2015). Large quantities of ammonium-N and nitrate-N may be mobilised in runoff from the wetting of dried soils (Qui & McComb, 1996; Valett et al., 2005), and the wetting-drying cycles in the ponding area may have contributed to the wide range of concentration changes observed in this study (Woodward et al., 2015).

6.3.2 Yields

Annual inflow yields delivered in surface runoff to the Hauraki site were 2.8 kg TN ha^{-1}, and 3.5 kg TN ha^{-1} at the Awahou site. As expected, these were much lower than the 29 kg N ha^{-1} y^{-1} average total N losses from pastures estimated for the Lake Rotorua catchment (Donald et al., 2019), due to nitrate losses leaching into the groundwater, which is the prevalent source of N loss from pastures in this catchment (Morgenstern et al., 2015).

Organic N made up 68% and 72 % of the TN inflows at the Hauraki and Awahou site, respectively. Ammonium-N composed 53% of the TIN at the Hauraki site, and 59% at the Awahou site. The proportion of TN as organic N was the highest in the spring and generally decreased with each following seasonal period at both sites, while the proportion of TN as nitrate-N was lowest in the spring and increased with each subsequent season at both sites (Fig. 6.3).

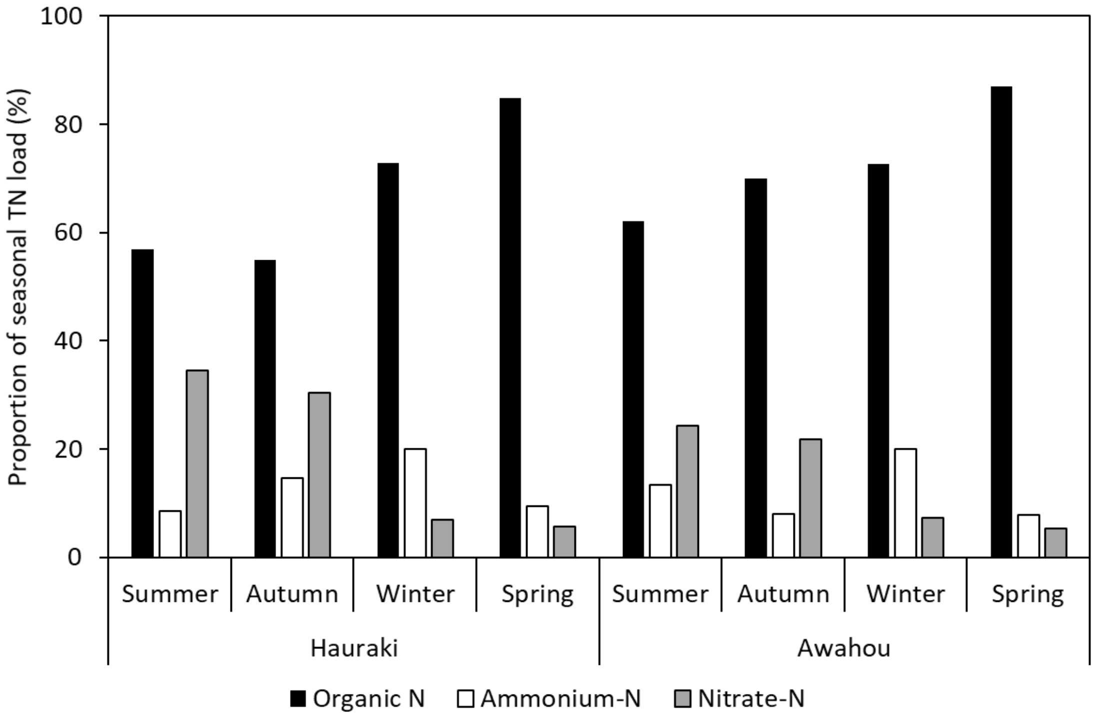

Figure 6.3: Proportions (%) of total nitrogen (TN) as organic nitrogen (N), ammonium-N and nitrate-N during each austral season, at each site.

When examining the cumulative seasonal runoff yields, the winter period was responsible for the majority of the annual runoff, organic and ammonium-N inflow yields at both sites (Fig. 6.4). The greatest nitrate-N inflow yield occurred during the winter at the Hauraki site, and during the autumn period at the Awahou site. At both sites, spring had the lowest contaminant inflow yields, and winter was the only season ammonium-N inflow yields exceeded nitrate-N inflow yields. The findings of the current study are consistent with other research that found that most N surface runoff losses from pastures in New Zealand occurred in the winter, although other studies attributed more significant losses in the spring, while the current study found more substantial losses occurring in the autumn (Cooke & Cooper, 1988; Smith, 1987; Smith & Monaghan, 2003). The larger autumn yield observed in the current study is likely due to a high magnitude storm event resulting in an Overflow Event in late-April (autumn) that delivered 24%, 19%, 45% and 20% of the annual runoff, organic N, nitrate-N and ammonium-N inflow yields respectively, at the Hauraki site, and 36%, 20%, 40% and 10% at the Awahou site (Fig 6.4).

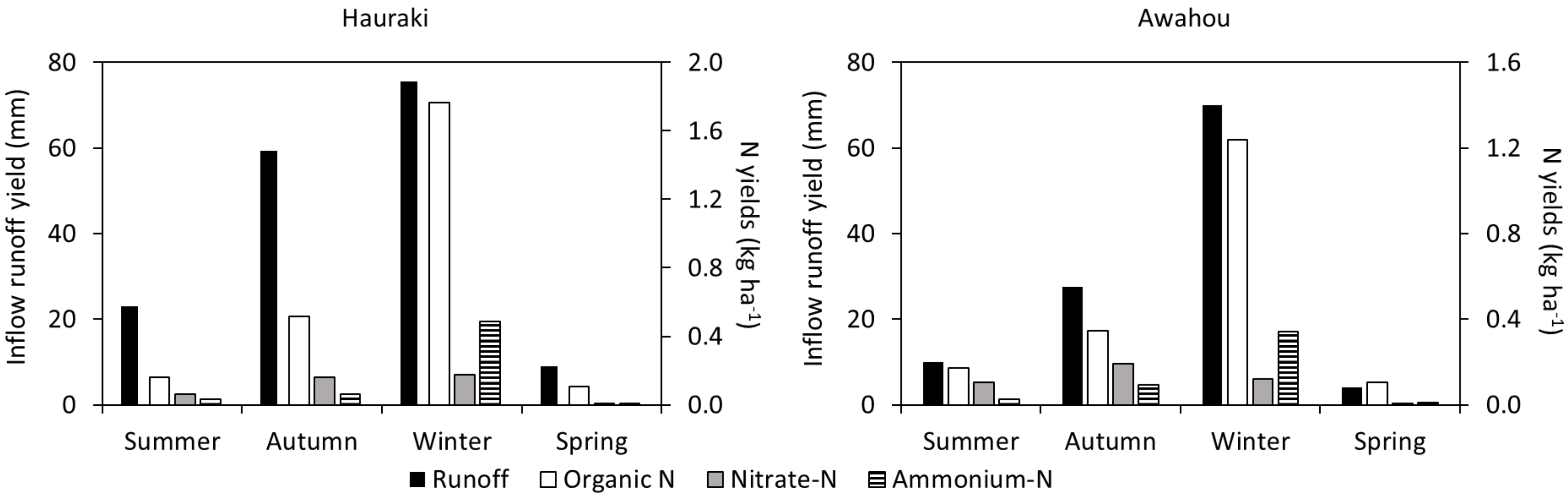

Figure 6.4: Cumulative seasonal runoff inflow yields (mm), and organic N, nitrate-N and ammonium-N inflow yields (kg ha^{-1}) at the two sites in this study. Note: Different 'N yield' y-axis between the sites.

Events with the greater runoff inflow yields tended to have greater TN inflow yields, mostly as a result of greater organic N yields being delivered during these events. These results were likely due to the greater erosive force of runoff transporting particulate organic N during higher magnitude runoff events (Fig. 6.5). These findings are supported by the results reported in Chapter 4 which found that 61% and 66% of the annual SS loads were delivered to the Hauraki and Awahou sites during the rare Overflow Events, respectively (Chapter 4), and another study of New Zealand pastures which found that increased particulate organic N losses were associated with higher rates of erosion (Cooke & Cooper, 1988).

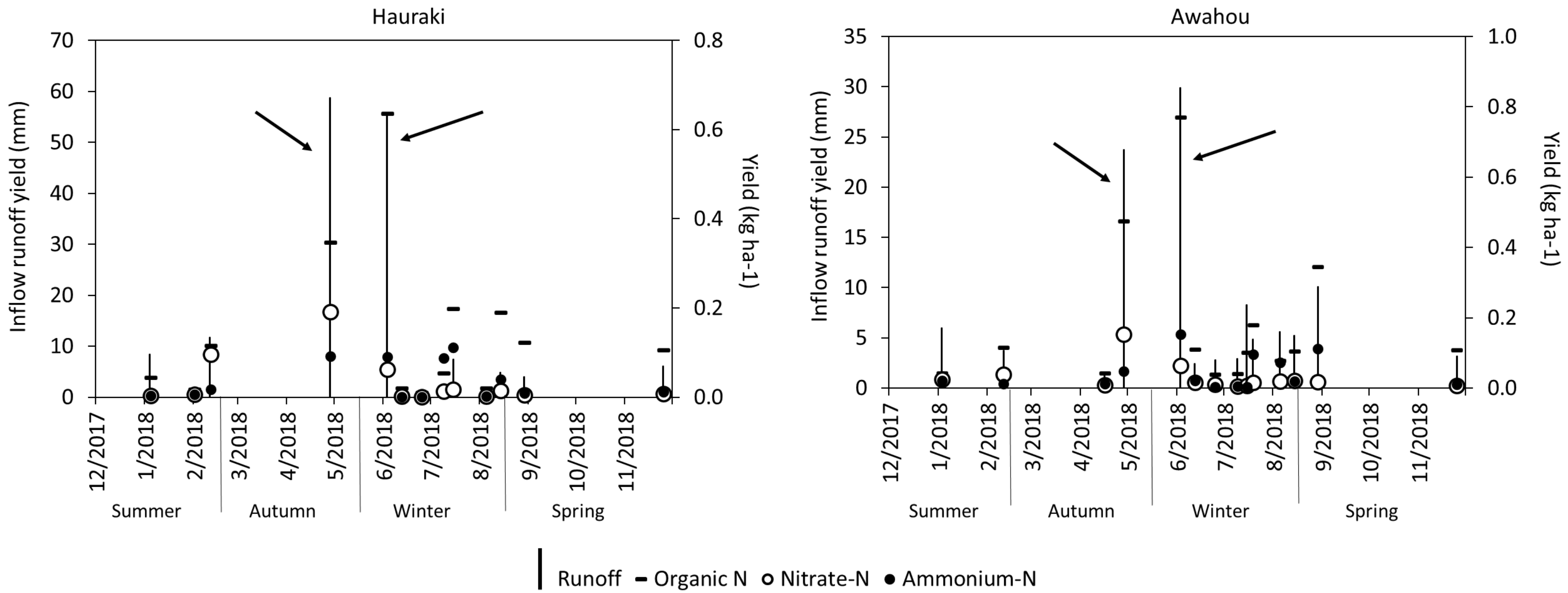

Figure 6.5: Inflow runoff yields (mm), and organic N, nitrate-N and ammonium-N yields (kg ha^{-1}) for each event at each site during this study, with arrows pointing to high runoff magnitude Overflow Events. Dates are presented as month and year with austral seasons labelled. Note: Difference in both y-axis between the sites.

Overflow events accounted for 2 of the 18 events at the Hauraki site, and 2 of the 19 events at the Awahou site, but were responsible for significant portions of the annual organic, nitrate-N and ammonium-N inflow loads at both sites (Table 6.3). The results of this current study are consistent with a study that found large storm events were responsible for significant portions of annual organic N and ammonium-N exports from pastures (Cooke & Cooper, 1988) as well as the findings in Chapters 3, 4 and 5 which reported that significant portions of runoff, sediments and P delivered to the DBs during Overflow Events.

Table 6.3: Proportions of annual inflow of runoff, organic N, nitrate-N and ammonium-N occurring during Overflow Events at each site.

	Hauraki		**Awahou**	
	Inflow (%)	Discharge (%)	Inflow (%)	Discharge (%)
Runoff	69	85	47	64
Organic N	53	69	49	57
Nitrate-N	60	83	53	59
Ammonium-N	39	65	34	46

Decreased TN yields discharged from the DBs were the result of changes in the concentrations of organic N, nitrate-N and ammonium-N, and the volume of runoff infiltrating the soil, positively supporting the hypothesis set out in section 6.1. Annual yield treatment efficiencies of runoff, organic N and ammonium, but not nitrate, were greater at the Awahou site than the Hauraki site (Fig. 6.6). The greater portion of runoff infiltrating the soil at the Awahou site is at least partially responsible for the greater yield decreases compared to the Hauraki site. The Overflow Events, discussed in more detail below, contributed greater runoff inflow yields, and consequently generated a greater amount of overflow discharge (ponded water going over the upstand riser and spillway) at the Hauraki site, and is likely the primary reason for the difference in yield treatment efficiencies between the sites (Fig. 6.6).

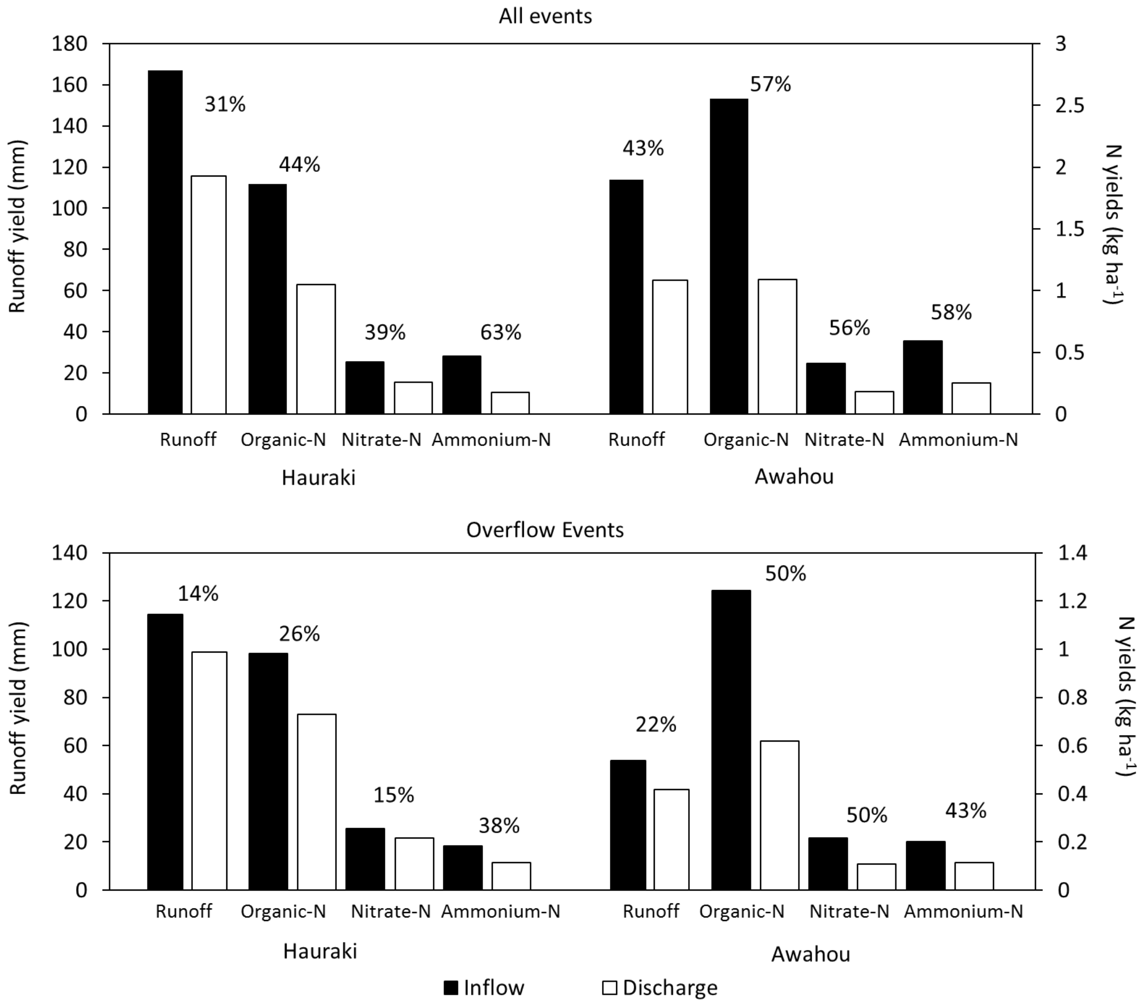

Figure 6.6: Cumulative annual, and Overflow Event only, inflow and discharge yields for runoff, organic N, nitrate-N and ammonium-N at the two sites in this study. The percentage difference between inflow and discharge yields also shown (%).

Organic N, ammonium-N and nitrate-N yields attenuated in the ponding area during high runoff magnitude Overflow Events accounted for 29%, 30% and 21% of the annual yield decreases at the Hauraki site, respectively, and 44%, 41% and 22% at the Awahou site. The results from the Overflow Events suggest that impeding stormflow during high magnitude runoff events played a key role in effectively decreasing N yields transported in surface runoff, and highlights the importance of locating DB ponds on soil types with sufficient infiltration capabilities, and maximising the pond volume capacity to catchment size ratio in order to avoid excess overflow discharge and to therefore optimise DB yield treatment efficiencies.

The form of N delivered to the DB affected whether the contaminant would potentially reach Lake Rotorua. To conservatively calculate the ability of DBs to prevent N loads from reaching Lake Rotorua, we assumed the entire nitrate load infiltrating the soil leached through the root and zone and would eventually reach the lake due to the relatively oxic groundwater in the catchment (Morgenstern et al., 2015). Therefore, the only potential effect the DB treatment would have on nitrate-N loads would be through decreasing the nitrate-N concentration, although some could be assimilated, taken up by plants, or denitrified in the soil before reaching groundwater. During 5 events at the Hauraki site, and 9 events at the Awahou site, nitrate-N concentrations in discharges were lower than inflows, and an estimated 0.01 kg nitrate-N y^{-1} was prevented from reaching Lake Rotorua from the Hauraki site, and 1.2 kg nitrate-N y^{-1} from the Awahou site. These results suggest that nitrate-N removal from surface runoff did not play a major role in the DBs ability to decrease TN yields from reaching Lake Rotorua.

Due to the prevalence of soils with high CECs in the catchment, ammonium-N infiltrating the soil in the ponding area, or downstream of the DB, was likely readily sorbed during subsurface transport (McDowell et al., 2008; Reddy & DeLaune, 2008). Therefore, it was assumed that the ammonium-N in solution would eventually be taken up by plants. It was also assumed that minimal dissolved organic N leached into the groundwater, and that any organic N attenuated in the ponding area, or discharged from the DB when runoff was not being generated in the catchment (i.e. excluding overflow discharges), would eventually decompose to bioavailable inorganic forms and be taken up by plants. Therefore, the TN load reaching Lake Rotorua included nitrate-N infiltrating the soil or discharged from the DB, and organic N and ammonium-N load discharged from the DB during overflow discharge.

At the Hauraki site, 44.8 kg of organic N, and 16.3 kg of ammonium-N, was attenuated by the DB in the ponding area. Additionally, 21.3 kg of organic N and 3.9 kg ammonium-N were discharged from the DB outside of overflow discharge, and was therefore likely prevented from reaching downstream surface waters. Including the 0.01 kg nitrate-N removed from surface runoff, 86.3 kg TN was prevented from reaching Lake Rotorua from the Hauraki site, a 57 % decrease in N loading.

At the Awahou site, 29.0 kg of organic N, and 6.7 kg of ammonium-N, was attenuated by the DB in the ponding area, and an additional 10.7 kg of organic N, and 2.9 kg ammonium-N was discharged from the DB outside of overflow discharge and likely did not reach downstream surface waters. Including the 1.2 kg nitrate-N removed from surface runoff, 50.5 kg TN was prevented from reaching Lake Rotorua in surface runoff from the Awahou site, a 72% decrease in N loading.

The results of this study suggest that DBs effectively mitigated N losses transported in surface runoff from the DB catchments by facilitating soil infiltration and sedimentation. Methods of improving the proportion of runoff infiltrating the soil discussed in Chapter 3, and the use of flocculants to increase sedimentation of particulate organic N would improve DB performance (Braskerud, 2002a). However, due to the prevalence of nitrate leaching into oxic groundwater, the strategy is not likely to effectively prevent significant portions of the annual N loads lost from pastures from reaching Lake Rotorua, since the strategy targets losses in surface runoff and not nitrate leaching into the soil.

Annual N losses from pastoral agriculture in this catchment averaged ~29 kg N ha^{-1} y^{-1} (Donald et al., 2019), while this present study found that ~10% of that average was delivered to the DBs in surface runoff. Of the estimated 10% of annual N losses transported to the DBs in surface runoff during this present study, the DB strategy prevented ~60-70% of the TN transported in surface runoff from potentially reaching Lake Rotorua. Therefore, the DB strategy only decreased annual N loading to the lake from the DB catchments by ~6-7% as the strategy is targeting surface runoff processes and not leaching. Meanwhile, the DBs investigated in this present study were estimated to decrease total P loads from reaching Lake Rotorua by 47-68% (Chapter 5). Still, when applying the DB mitigation strategy to nutrient budgeting and management models, the results of this study suggest that 'N by-catch' should be considered a benefit of utilising DBs.

6.4 Conclusion

This current study found annual yields of organic N, nitrate-N and ammonium-N discharged from 2 DB catchments in surface runoff decreased by 39% to 63%. Considering the potential fates of the various forms of N, an estimated 86.3 kg N and 50.5 kg N were prevented from reaching Lake Rotorua from the 2 sites, equivalent to 57% and 72% of the annual inflow loads, respectively. Nitrate-N removal from surface runoff accounted for very little of the N loads prevented from reaching Lake Rotorua, and sedimentation and sorption of dissolved ammonium-N in the portion of runoff that infiltrated the soil contributed to the majority of the DBs' efficacy. Although the DBs' main objective is to mitigate P losses in surface runoff from pastures in the Lake Rotorua catchment, results of this current study suggest that 'N by-catch' is an added benefit of the strategy, and should be considered in nutrient budgeting and management models when implementing DBs in the landscape. The results of this study positively support the hypothesis that hypothesised that DBs will decrease the loads of N discharged from the DB catchment in surface runoff due to sedimentation decreasing particulate bound organic N loads, and soil infiltration decreasing the volumes of runoff, and consequently dissolved N loads. Also, the study found that ammonium sorbing on to suspended sediments prevented an appreciable load of N from reaching downstream waterways and surface runoff, although chemical changes did not occur to a great extent that and little nitrate was prevented from reaching Lake Rotorua.

The results of the study demonstrate that DBs can effectively decrease N loads discharged from the DB catchments in surface runoff, and prevent significant portions of N transported in surface runoff from reaching Lake Rotorua. However, the concentrations of nitrate-N and ammonium-N discharged from the DBs were not consistently lower than inflows, and discharge concentrations remained well above median lake concentrations. Also, DBs do not have a major impact on the TN losses from pastures from reaching the lake due to the prevalence of nitrate leaching and delivery by groundwater to Lake Rotorua.

Future work on DBs should investigate ways to optimise DBs for soil infiltration and improve treatment efficiencies with the use of flocculants. Also, the results from the current study should be incorporated into nutrient budgeting and management models to allow policy makers, regulators and farmers to account for the mitigation capacity of DB’s to reduce N losses to surrounding water bodies.

The results of this present study, and the three preceding it, positively supporting the hypotheses that by impeding stormflow and temporarily ponding surface runoff, DBs will facilitate soil infiltration of ponded water, thus decreasing dissolved nutrient losses from pastures, as well as facilitating sedimentation which could contribute to further nutrient attenuation. In the following chapter we will summarize the findings of this thesis in order to determine whether DBs are an effective nutrient mitigation strategy in the Lake Rotorua catchment, the implications of the findings in this thesis, recommendations for maintaining and improving DB performance, consider the applicability of DBs beyond the Lake Rotorua catchment, and set priorities for further developments and research into the DB strategy.

CHAPTER 7: Summary and recommendations

This chapter is a summary of the key findings and conclusions of this thesis. Section 7.1 is a synthesis of the insights gained in the experiments, and places them in the context of using detainment bunds (DBs) as a strategy to help achieve Lake Rotorua water quality objectives. Section 7.2 considers factors affecting the long-term efficacy of the DB strategy and makes recommendations for maintaining and improving DB performance. Section 7.3 investigates the applicability of DBs in a broader context by considering their utility beyond the Lake Rotorua catchment. Section 7.4 describes some priorities for future research.

7.1 Detainment bunds as an effective mitigation strategy in the Lake Rotorua catchment

Numerous challenges face the various stakeholders involved in improving Lake Rotorua's water quality. Reaching water quality objectives will require a decrease in phosphorus (P) and nitrogen (N) loading from anthropogenic sources, particularly pastoral agriculture. Pastoral agriculture covers 48% of the Lake Rotorua catchment, and is estimated to be responsible for almost half of the ~40 t P yr^{-1}, and 77% of the 578 t N yr^{-1} delivered to the lake from the catchment (Bay of Plenty Regional Council, 2012). Surface runoff from pastoral agriculture is strongly associated with eutrophication and degraded freshwater ecosystems in New Zealand (Verburg et al., 2010) due to the use of fertilisers, the prevalence of year-round grazing at high stocking rates, and the grazing of crops, which contribute to increased erosion and mobilised sediment bound nutrients during typically wet winters (Monaghan et al., 2007). Also, managing P losses from pastures in surface runoff is difficult to manage since the economically optimal Olsen P concentrations in pasture soils in the Lake Rotorua catchment (15-45 mg L^{-1}) are orders of magnitude higher than those that contribute to lake eutrophication (McDowell, 2010) and soil P concentrations are proportional to the magnitude of P losses from soils in runoff (McDowell et al., 2001). Therefore implementing mitigation strategies to reduce nutrient losses from pastures in the catchment will be necessary to achieve the 3.5 t P y^{-1} and 10 t P y^{-1} P load reduction targets by 2019 and 2029, respectively, and in order for N loads to decrease by 320 t N y^{-1} by 2022 (Bay of Plenty Regional Council, 2012).

Due to the influence of hydrology on sediment and nutrient mobilisation and transport, storm periods have been recognised as important opportunities for mitigating contaminant losses from pastures (Gburek & Sharpley, 1998). Interest in DBs as a potential mitigation strategy for P loss in surface runoff from pastures began in 2010. This current thesis builds upon a Masters thesis which provided a 'proof-of-concept' by identifying the ability of DBs to attenuate P-enriched sediments in the ponding area as a result of impeding stormflows (Clarke, 2013). The results of the studies in this current thesis quantified the ability of 2 DBs to attenuate significant portions of the annual loads of SS, P, and N transported in surface runoff from pastures in the Lake Rotorua catchment. The studies also noted that it was important to account for the form of P and N attenuated in the ponding area or discharged from the DBs, as the contaminant form would likely influence whether they would be prevented from reaching Lake Rotorua due to the DB treatment. Accordingly, the studies in this current thesis calculated the contaminant loads prevented from reaching Lake Rotorua as a result of DBs increasing surface runoff residence times by impeding stormflows. An analysis of the data gathered in these studies also identified the mechanisms by which DBs effectively reduced nutrient loads from reaching Lake Rotorua, and concluded that DBs are an important option to add to the nutrient mitigation toolbox available to farmers in the catchment.

The investigations in this thesis found that DBs decreased nutrient losses from pastures during every storm event that occurred during the 12-month study period. One potential reason why DBs were able to consistently decrease contaminant loads discharged in runoff is that they impede the flow of even the highest magnitude stormflows, as well as the 'first flush' of sediments and nutrients that may have accumulated during inter-storm periods which can be rapidly mobilised when rainfall initiates surface runoff (Bieroza et al., 2019). The proportion of sediment and nutrient loads attenuated in the ponding area was affected by the decrease in surface runoff volume, mainly attributed to soil infiltration, and changes to contaminant concentrations (Tables 7.1, 7.2 and 7.3). Therefore, weather patterns, and land use and hydrological conditions influencing the magnitude of runoff and the concentration and form of nutrients delivered to the DB, impacted the quantity of nutrients prevented from reaching Lake Rotorua as a result of the DB strategy. However, the spatial and temporal variability of factors affecting runoff and contaminant transport, and the fate of contaminants transported in runoff is highlighted by the results of the extensive statistical analysis described in section 2.3.5.3 of this thesis which found no statistically significant effects, based on the regression models.

Table 7.1: Summary of the cumulative annual runoff yields (mm) and contaminant loads (kg) delivered to the detainment bunds in inflow, and discharged as overflow, and combined release and leak discharges. The cumulative annual runoff yield and contaminant loads attenuated in the ponding area, and the estimated load prevented from reaching Lake Rotorua are presented, as well as the proportion of annual inflow runoff and contaminants estimated to have been prevented from reaching the lake (%).

	Hauraki							
	Runoff (mm)	**Suspended sediment (kg)**	**Total P (kg)**	**Dissolved reactive P (kg)**	**Total N (kg)**	**Organic N (kg)**	**Nitrate-N (kg)**	**Ammonium-N (kg)**
Inflow	167	1543	95	78	152	103	23	26
Overflow discharge (riser and spillway)	92	607	47	37	53	36	11	6
Release and leak discharge (all events)	22	147	12	9	37	21	12	4
Load attenuated in the ponding area	51	789	37	32	70	45	9	16
Load prevented from reaching Lake Rotorua	72	789	44	40	86	66	0.01	20
Percentage of annual inflow prevented from reaching Lake Rotorua	43	51	47	51	57	64	0	78

Table 7.1 continued:

	Awahou							
	Runoff (mm)	**Suspended sediment** (kg)	**Total P** (kg)	**Dissolved reactive P** (kg)	**Total N** (kg)	**Organic N** (kg)	**Nitrate-N** (kg)	**Ammonium-N** (kg)
Inflow	114	2151	18	8	70	50	8	12
Overflow discharge (riser and spillway)	37	634	4	2	15	11	2	2
Release and leak discharge (all events)	28	216	3	2	20	11	6	3
Load attenuated in the ponding area	49	1280	11	4	40	29	4	7
Load prevented from reaching Lake Rotorua	72	1280	12	6	51	40	1	10
Percentage of annual inflow prevented from reaching Lake Rotorua	63	59	68	71	72	79	15	75

Table 7.2: Summary of the proportion of annual inflows delivered to detainment bunds during Overflow Events (%), the percentage of loads attenuated during these events (%), the proportion of the annual load attenuated during Overflow Events (%), and the percent change in mean flow proportional contaminant concentrations between inflow and discharges during Overflow Event s(%).

	Runoff	**Suspended sediment**	**Total P**	**Dissolved reactive P**	**Total N**	**Nitrate-N**	**Ammonium-N**
				Hauraki			
Proportion of annual inflow during Overflow events (%)	69	61	68	72	51	60	39
Overflow Event attenuation (%)	14	32	24	29	25	15	38
Proportion of annual load attenuated during Overflow Events (%)	31	39	42	51	28	23	23
Percent change in contaminant concentration (%)		-22	-12	-18	-13	-1	-28
				Awahou			
Proportion of annual inflow during Overflow events (%)	47	66	54	51	47	53	34
Overflow Event attenuation (%)	22	54	56	44	50	51	44
Proportion of annual load attenuated in pond during Overflow Events (%)	24	59	50	43	40	48	26
Percent change in contaminant concentration (%)		-41	-43	-29	-36	-37	-28

Table 7.3: Percent change (%) in annual mean flow proportional contaminant concentrations as a result of the detainment bund treatment.

Site	Suspended sediment	Total P	Dissolved reactive P	Total N	Organic N	Nitrate-N	Ammonium-N
Hauraki	-28%	-10%	-14%	-21%	-17%	-10%	-45%
Awahou	-29%	-30%	-18%	-25%	-26%	-22%	-26%

Multiple factors, including soil infiltration, sedimentation, and chemical processes, contributed to the DB's ability to decrease the contaminant loads transported in surface runoff from reaching surface waters downstream of the bunds, positively supporting the hypothesis described in section 1.2.1. The ability of DBs to decrease surface runoff volumes by increasing the residence time of runoff on the relatively well-drained soils in the ponding area is an important finding since contaminant loads delivered to surface water in runoff are the product of the concentration of contaminants and the volume of runoff. Soil infiltration occurring in the ponding area decreased annual runoff yields discharged from the DB catchments by 31% at the Hauraki site, and 43% at the Awahou site (Chapter 3). Additionally, the ponded runoff released from the DBs after approximately 3 days of detention was likely to have infiltrated the soil downstream of the bund. During the 3 days of ponding between the storm front and releasing the ponded water, the soil downstream of the DB was likely to have dried and have restored reasonably rapid soil infiltration capabilities. Overall, when both in-pond and downstream infiltration was accounted for, 43% and 63% of the surface runoff delivered to the Hauraki and Awahou sites were prevented from reaching downstream surface waters, respectively (Chapter 3).

The ability of DBs to facilitate soil infiltration has an obvious impact on the loads of dissolved nutrients discharged from the DB catchments. However, the form of the dissolved nutrient affected whether the DBs ability to decrease the load transported in surface runoff actually resulted in decreased loading in receiving waters downstream. The soils in the Lake Rotorua catchment generally have high anion storage capacities (ASCs) and high cation exchange capacities (CECs), and have the ability to sorb dissolved P and ammonium that infiltrate the soil (Morgenstern et al., 2015; Reddy & DeLaune, 2008). Nitrate leaching through the root zone, however, is unlikely to undergo removal from solution as a result of denitrification during subsurface transport due to the relatively oxic groundwater in the Lake Rotorua catchment, and was expected to reach the lake in this current study (Morgenstern et al., 2015). Therefore, in order to conservatively estimate the DBs' ability to treat N loads, we assumed that any nitrate infiltrating the soil would leach into the groundwater and reach Lake Rotorua, although some may have been denitrified in the soil, and/or taken up by plants.

Sediments reaching Lake Rotorua may cause aquatic ecosystem degradation by disrupting aquatic habitats and food webs (Howard-Williams et al., 2010), and by delivering sediment-bound nutrients that contribute to eutrophication (Dare, 2018). Sedimentation has been identified as the primary mechanism involved in mitigation strategies affecting surface runoff sediment and nutrient concentrations (Stanley, 1996). During this current study, the DB strategy was found to facilitate sedimentation, decreasing SS concentrations in the majority of ponding events at both sites, and decreasing the loads of sediments discharged from the DB catchments to a greater degree than runoff discharges. Also, sediments attenuated by the DB were blanketed across the relatively wide ponding area, and are therefore more likely to be effectively held behind the DB as opposed to other strategies such as buffer strips and treatment wetlands that may have sediments flushed out during high magnitude runoff events (McKergow et al., 2007).

In Chapter 4, in order to conservatively calculate the sediments prevented from reaching surface waters downstream of the DBs, we only considered the loads attenuated in the pond, since some portion of the sediments that were discharged on release could eventually be remobilised by future runoff events, especially the high magnitude overflow events that breach the emergency spillway. However, it is also possible that some of these discharged sediments could be permanently entrained in the soil. Since we assumed any sediment-bound P discharged from the DB was likely to reach downstream surface waters, the conservative nature of the sediment load attenuation estimates also pertain to total P (TP) loads reported to be prevented from reaching Lake Rotorua in Chapter 5. However, some of these sediments could be permanently entrained, and/or desorption could occur releasing dissolved P that is taken up by plants and/or is mobilised into water that is sorbed and retained deeper in the soil due to infiltration.

Rare, large storm events have been found to be responsible for the majority of runoff, and sediment and nutrient loads delivered to surface waters in the Lake Rotorua catchment (Abell et al., 2013; Dare, 2018). Results from the 2 largest storm events during this current study, which were responsible for large portions of the annual inflow runoff and contaminant yields, point to the importance of sedimentation and chemical processes such as sorption in the ponded area, in mitigating contaminant losses during these very large storm events. Although only relatively small portion of the inflow runoff volumes infiltrated the soil in the ponding area during these large Overflow Events (14 to 22%), the quantities of sediments and nutrients attenuated in the ponding area were, by contrast, approximately twice as large (24 to 56%) (Table 7.2). Additionally, significant portions of the annual sediment and nutrient loads attenuated at both sites occurred during these large events (Table 7.2). The ability of DBs to perform effectively during high magnitude storm events will become more important over time since climate change is likely to increase the number and/or the intensity of large storms, and increase runoff and associated erosion and nutrient losses (Ministry for the Envrironment, 2019; Ockenden et al., 2016). Also, the results from Overflow Events highlight the importance of placing DBs in locations that maximise the pond storage: catchment size ratio in order to minimise overflow discharges during large runoff events that are responsible for the majority of contaminants that are discharged from the DBs (Table 7.2).

During Overflow Events at both sites, the SS concentration difference between the portions of inflow contributing to overflow discharge (i.e. ponded surface water going over the top of the upstand riser and emergency spillway) (Flow A) and the resulting overflow discharge (Flow B) did not decrease to the same extent as the concentration decreased between the overflow discharge (Flow B) and the following release discharge (Flow C) (Table 4.3). These results are somewhat surprising since we would expect the decanting of the uppermost layer of water performed by the upstand riser and emergency spillway would be highly effective at preventing SS discharge. The data suggests however, that longer pond residence times experienced by the release discharge compared to the overflow discharge (an average of 14 hours between overflow discharge and the following release discharge at both sites), allowed for greater sedimentation to occur. Longer retention times have been found to increase sediment removal efficiencies in a study of sedimentation ponds (Brown et al., 1981).

Although the DB's main objective is to mitigate P losses from pastures in the Lake Rotorua watershed, results of this study suggest that 'N by-catch' could be an added benefit of the strategy, and should be considered in nutrient budgeting and management models that calculate the effects of implementing DBs in the landscape. This present study found that only ~10% of the estimated average annual N yield lost from pastoral agriculture in the Lake Rotorua catchment was delivered to the DBs in surface runoff, likely due to the prevalence of nitrate leaching common in the catchment (Chapter 6). The ability of DBs to mitigate nutrients transported in surface runoff effectively prevented 47% to 68% of the TP loads from reaching Lake Rotorua during this study, and 57% and 72% of the total N (TN) loads were prevented from reaching downstream surface waters in surface runoff. However, the annual TN load prevented from being transported to Lake Rotorua calculated in this study equates to approximately only 7% of the combined TN losses in surface and subsurface runoff from the DB catchments.

7.1.1 Implications of results

The ability of DBs to decrease contaminant loads during every storm event delivering surface runoff to the DBs during this 12-month study is very important to recognise. At the onset of the project, it was predicted that up to 6 ponding events would occur during the 12-month sample period, while 18 and 19 ponding events at the 2 sites actually occurred during this study. Only 2 Overflow Events occurred at both sites throughout the study, illustrating the patchiness of high intensity rainfall events occurring in this region. These rare, high magnitude events delivered 114 mm and 54 mm of surface runoff to the Hauraki and Awahou sites, respectively, and accounted for 69% and 47% of the annual inflow runoff (Chapter 3). On average, these Overflow Events delivered 4.4 times the pond volume capacity at the Hauraki site, and 2.4 times at the Awahou site. Due to the potential for large portions of annual contaminant loading into surface waters in the Lake Rotorua catchment during rare, high magnitude storm events (Abell et al., 2013; Dare, 2018), the ability of DBs to provide a degree of treatment of sediment and nutrient loads during Overflow Events taking place in this present study is an important result to highlight.

The consistency of the DBs' ability to prevent sediment and nutrient loads from reaching Lake Rotorua during all storm events is also an important factor to consider when comparing DBs to land management and other edge of field mitigation strategies. Some land management strategies may be overwhelmed by hydrologic conditions, especially during large storms or very wet periods (Kleinman et al., 2006). Edge of field mitigation strategies, such as grass buffer strips and treatment wetlands, may not effectively mitigate dissolved nutrients, particularly when surface runoff flow rates are high (McKergow et al., 2007). Additionally, these edge of field mitigation strategies may become a source of nutrient enriched particles that are flushed out of buffer strips or wetlands in surface runoff during large runoff events, or release dissolved nutrients as a result of organic matter decomposition, or soil P enrichment and desorption of P into discharged runoff (McKergow et al., 2007; Tanner & Sukias, 2011). By comparison, the results of this present study found that DBs effectively reduced sediment and nutrient loads year-round after 7 years of being in use, and during the most extreme runoff events.

The average TP yield prevented from reaching Lake Rotorua by the 2 DBs in this study was 0.72 kg TP ha^{-1} yr^{-1} (Chapter 5). Given this value, 70% of the ~20,000 ha of pastoral agriculture in the catchment would need to be treated by DBs if the 2029 load reduction target (10 t P yr^{-1}) was to be achieved with DBs alone. This is not feasible, however, since the number of viable DB locations are limited by landscape characteristics and regulations described by Paterson (2019). A scoping study using a computer modelling program (Detainment Bund Applicability Model) to find appropriate locations for DBs in the Lake Rotorua catchment identified 300 ‘confirmed mock-up sites’ that met the 120 m^3 pond volume: 1 ha contributing catchment area ratio criteria. The sites identified in the model had an average catchment size of ~15 ha and therefore ~4,500 ha of the Lake Rotorua catchment could be treated by DBs (Paterson, 2019). By applying average results of the current study to the potential area treated by DBs in the catchment, an estimated 3.2 t TP yr^{-1} could be prevented from reaching Lake Rotorua with DBs, which is nearly equal to the 2019 P load reduction target (3.5 t P yr^{-1}) and is equivalent to 32% of the 2029 goals. Therefore, in the longer-term, it is clear that DBs are one of multiple mitigation strategies that will need to be implemented in order to reach Lake Rotorua nutrient load reduction targets.

Various mitigation options are available to farmers in the Lake Rotorua catchment attempting to decrease nutrient losses from their pastoral areas (McDowell, 2010). Cost: benefit analyses are important tools for assisting decision makers attempting to identify and implement appropriate mitigation strategies (Bieroza et al., 2019; McDowell, 2010). In order to develop useful cost benefit analyses, the efficacy of mitigation strategies must be measured in field studies. This is particularly important for novel strategies such as DBs.

Factors influencing the cost to build a DB, and the efficacy of the DB to attenuate contaminants are unique to each potential DB location, and will therefore result in a range of cost: benefit ratios that vary spatially and temporally. Although a rigorous cost: benefit analysis for DBs will be important, it is beyond the scope of this thesis, particularly because it is difficult to estimate the cost of constructing a specific DB. The 2 DBs investigated in this thesis are examples that demonstrate the potentially wide variations in construction costs. The estimated construction cost at the Awahou site was $1000, versus $9500 at the Hauraki site, due to the existing infrastructure that was able to be utilised at the Awahou site. Labour and material costs for the more than 20 DBs constructed since 2010 also varied widely (from ~$1,000-$20,000) due to differences in existing infrastructure at locations, and the access farmers had to equipment and in-kind contributions. A further complication to calculating costs is that the Bay of Plenty Regional Council currently offers a subsidy to construct new DBs, but it is uncertain how long this incentive will be in place, and whether other regional councils will offer similar incentives. The estimated average cost of the ~20 previously constructed DBs in the Lake Rotorua catchment was $10,000, which did not include the costs associated with newly enforced regulatory protocols expected to add an additional $10,000 in costs to each bund (J. Paterson, PMP Project Manager, *pers. comm.*, 2019). Due to the numerous unique factors contributing to varying costs of DB construction, more information is required on the likely costs to build DBs in a range of situations before comprehensive financial analyses can be performed.

Additional costs including those associated with DB maintenance, and disruption to productivity, must also be considered in the cost: benefit analyses of mitigation strategies. One of the goals, and potential advantages of the DB strategy is that DBs can fit into pastoral farm systems with minimal disruption to pasture productivity since ponds form on pastures that do not need to be taken out of production, and the manually operated outlet valve enables limited inundation periods. However, approaches to maintain or improve DB performance discussed later in Section 7.2, such as revisions to DB design that increase ponding duration, and/or soil P management strategies in the ponding area, could add to the cost of operating a DB in the long-term.

To perform a rough cost: benefit analysis, we considered the average construction cost to be approximately $20,000 per DB, which includes the new regulatory enforcement costs, for each of the 300 potential DBs in the Lake Rotorua catchment, which together, could potentially attenuate 3.2 t P yr^{-1}, based on the results in Chapter 5. If all the costs associated with a DB are accounted for – including, the cost of borrowing money to construct a DB, the potential labour, repair and maintenance costs, and the potential loss of production from the ponded area – then the likely cost benefit ratio of DBs is in the range of $120 to $140 kg^{-1} P attenuated. The cost-effectiveness of DBs based on analysis compares favourably with other edge of field techniques as reported by McDowell (2010) in a study of mitigation strategies able to be utilised in the Lake Rotorua catchment (Table 7.4). Additionally, the benefits of reduced surface runoff magnitudes occurring downstream of the DBs that would presumably decrease erosion and nutrient mobilisation and transport further down in the catchment was not considered in the cost: benefit analysis described above.

Table 7.4: Summary of efficacy and cost in New Zealand dollars (NZD) of P mitigation strategies in the Lake Rotorua catchment adapted from McDowell (2010). Detainment bund values (bold) are based on the results from the current study.

Strategy	**Effectiveness (%)**	**Cost** (NZD kg^{-1} P conserved)
	Management	
Optimum soil test P	5-20	Highly cost-effective[1]
Low solubility P fertiliser	0-20	0-30
Stream fencing	10-30	5-65
Greater effluent pond storage	10-30	30
Low rate effluent application to land	10-30	45
	Amendment	
Tile drain amendments	50	25-100
Restricted grazing of cropland	30-50	150-250
Alum to pasture	5-30	150->500
Alum to grazed cropland	30	160-260
	Edge of field	
Detainment bunds	**47-68**	**120-140**
Grass buffer strips	0-20	>250
Sorbents in and near streams	20	350
Retention dams/water recycling[2]	10-80	>500
Constructed wetlands[3]	-426-77	>500
Natural seepage wetlands[3]	<10	>500

[1] depends on existing soil test P concentration, but no cost if already in excess of optimum

[2] upper bound only applicable to retention dams combined with water recycling

[3] potential for wetlands to act as a source of P renders upper estimates for cost infinite

The location of the DB within the landscape is another factor to recognise when considering the significant benefits of the strategy. Detainment bunds tend to be located in the headwaters of the Lake Rotorua catchment, and target pastures which are recognised as significant contributors to nutrient loading to the lake. Headwaters play a significant role in hydrochemical responses to storms, and receiving waters have been shown to respond rapidly to changes in contaminant sources in headwaters (Alexander et al., 2007). Pastures in low-order catchments in New Zealand have been found to account for an average of 73% of the annual loads of TN and DRP delivered to small streams, and 84% of the SS (McDowell et al., 2017). Therefore, due to their important placement in the landscape, and high contaminant attenuation performance and cost-effectiveness, DBs are an important option in the nutrient mitigation toolbox available to pastoral farmers in the Lake Rotorua catchment.

7.2 Recommendations for optimising and maintaining DB performance

Certain factors should be considered for optimising and maintaining the long-term performance of DBs. Since DB performance depends on soil infiltration, sedimentation and chemical processes, it is important to maintain or enhance these processes to optimise DB efficacy.

The studies in this thesis identified that the DB’s ability to decrease annual contaminant loading to downstream surface waters was limited by their ability to impound the entire runoff volumes during the Overflow Events, since overflow discharge reduced the proportion of annual runoff infiltrating the soil before reaching downstream surface waters. Therefore, design protocols that maximise the pond volume: catchment are ratio are essential to optimise DB performance. While this study emphasises the importance of maximising the pond volume: catchment area ratio, it is also important to recognise the benefits of installing DBs where the opportunity for implementation exists. Viable locations for DBs are limited by particular landscape characteristics and potential regulatory requirements (Paterson, 2019). Therefore, results of this study should be used to develop models that are able to establish the optimal minimum pond volume: catchment area ratio in specific locations based on the unique hydrological conditions at each potential DB site, and maximise the cost-effectiveness of implementing DBs in a catchment by forecasting the potential contaminant loads attenuated at specific sites.

Another way to maximise yield treatment efficiencies would be to decrease release discharges that are generated when unplugging the outlet valve. While discharged dissolved nutrients are able to infiltrate the soils downstream of the DB following the ponding period, sediments and sediment-bound nutrients discharged from the DB may potentially be remobilised and transported in subsequent runoff events. Raising the outlet valve 5-10 cm above ground level, or removing the upstand riser/outlet valve/discharge pipe unit all together would likely to increase sediment attenuation, however both approaches could affect pasture productivity in lower ponding areas since inundation may last longer than three days. The potential effects of lengthening the duration of inundation to pasture productivity should be investigated and considered when calculating the overall costs of utilising the DB strategy. However, the benefits of preventing the sediments and sediment bound nutrients from being discharged from the bund could be significant in locations where erosion rates are high and nutrient enriched particles are prevalent. Had there been no discharge pipe at either site, an additional 16% and 7% of SS and TP, respectively, is estimated to have been prevented from reaching Lake Rotorua from the Hauraki site, and 14% and 12% of SS and TP from the Awahou site (Chapters 4 and 5).

Results of this study found evidence of increased soil P concentrations building up in the ponding area (Chapter 5). The average event TP and DRP concentration decreased to a greater extent at the Awahou site, which had much lower soil P concentrations in the ponding area than the Hauraki site (Chapter 5). These results suggest soil P concentrations building up in the ponding area may have contributed to the lower TP concentration treatment efficiencies at the Hauraki site, although the varying proportions of TP as DRP, and sedimentation being a more effective concentration treatment mechanism, also likely affected these results. Additionally, much of the difference in yield treatment efficiencies between the sites was due to the greater portion of inflow runoff undergoing overflow discharge at the Hauraki site.

That the Hauraki site had higher soil P concentrations than the Awahou site in the ponding, but similar P yields were deposited in the ponding area during this study, and the two sites had similar soil P concentrations outside the ponding area (Chapter 5, Table 5.1), suggest that ponding areas on pastures with low ASC soils, particularly those in the lower portions where ponding occurs most often, should be strategically managed in order to maximise treatment efficiencies. Approaches to avoid P enriched sediment mobilisation and discharges from ponding areas would be to exclude livestock from the lower ponding area, at least during wetter winter periods, and/or and implementing cut and carry activities to avoid excess treading and erosion. Integrating P socks or alum dosing with the DB strategy to sorb dissolved P, and using flocculants to aggregate sediments, thereby decreasing their transport potential, should also be investigated. Also, farmers should be aware that soil P concentration build-up in the ponding area is likely and to exclude this area from P fertiliser applications.

Efforts to optimise and maintain DB performance could increase the cost of utilising DBs. Aerating soils, excluding livestock and applying resources to cut and carry management in the lower ponding area, and/or sacrifices to some pasture productivity by extending inundation periods beyond the recommended 3 days, would increase the costs of utilising DBs when implementing strategies to maintain or improve DB performance. However, as some of these measures may also enhance the performance of DBs, the cost: benefit ratio may not change or may improve. While results of this study should help decision makers determine the suitability of the DB strategy under specific conditions, the finding that DB treatment performance is likely to change over time could make estimating long-term DB efficacy, and therefore the cost-effectiveness, more complicated.

7.3 Applicability of DBs beyond the Lake Rotorua catchment

Due to limitations on construction design and potential regulatory limitations, some landscapes are not appropriate for DBs, due to their topography, such as steep mountainous country with incised valley floors and flat flood plains (Paterson, 2019). However, developing models that are able to predict DB performance based on conditions unique to each site using the information gathered in this thesis, would help determine whether DBs are appropriate in certain locations, and assist in achieving maximum cost-effectiveness of DBs implemented in catchments.

By identifying factors that affect the ability of DBs to prevent sediment and nutrient loading to downstream surface waters, the findings presented in this thesis should help decision makers determine the utility of DBs in areas outside of the Lake Rotorua catchment. The ability of DBs to decrease dissolved nutrient loading in downstream surface waters will likely depend on the ability of ponded runoff and discharges to infiltrate the soil in locations where dissolved nutrients make up a significant proportion of total nutrients. However, the likelihood of dissolved nutrients reaching surface waters downstream of DBs is affected by the capacity of DBs to facilitate soil infiltration, as well as whether the soils in the catchment are capable of sorbing dissolved nutrients in the case of DRP or ammonium, or nitrate undergoing denitrification during subsurface transport. During this study, large portions of dissolved contaminants infiltrated the soil due to the permeable soils present at the study sites. In the case of the Lake Rotorua catchment, DRP and ammonium infiltrating the soil would have likely been sorbed due to the prevalence of soils with high ASCs and CECs. In areas where well drained soils do not have as high a capacity to sorb DRP and ammonium, soil infiltration may not prevent these dissolved nutrients from reaching downstream waters as effectively. Where soils with low ASCs and low CECs are present, it may be important to integrate the use sorbents with the DB in order for the strategy to effectively mitigate dissolved nutrients.

As previously discussed, nitrate leaching through the root zone is not expected to undergo denitrification in the Lake Rotorua catchment due to the relatively oxic groundwater (Morgenstern et al., 2015). In areas where denitrification may occur in the soil or groundwater, DBs that facilitate soil infiltration could be an effective approach to mitigating nitrate loads transported in surface runoff. In areas where nitrate concentrations in surface runoff are high and infiltration rates are low, installing treatment wetlands downstream of the DBs could be an effective strategy to mitigate N loading in downstream surface waters. Also, DBs installed upstream of treatment wetlands could enhance the performance and lifespan of treatment wetlands due to the DBs ability to buffer high magnitude stormflows and retain sediments which have been found to compromise treatment wetland mitigation performance (McKergow et al., 2007; Tanner & Sukias, 2011).

Greater runoff volumes discharged from DBs during storm events corresponded to greater contaminant load discharges, and the annual DB performances in this present study were limited by their ability to impound runoff volumes during rare, high magnitude Overflow Events. Therefore, soil infiltration, and the ability of DBs to impede stormflow during high magnitude events, were key drivers in decreasing contaminant loads reaching downstream surface waters. These results suggest that in order for DBs in other areas to have similar contaminant yield treatment efficiencies as those in this study, it is likely that a similar proportion of surface runoff delivered to the DB would need to infiltrate the soil. However, the proportion of dissolved nutrients and sediment bound nutrients delivered to the DB would affect whether soil infiltration in the ponded area, and/or downstream of the DB, prevented loads from reaching downstream surface waters due to the potential remobilisation of sediment-bound nutrients in subsequent runoff events. Results from Chapter 4 demonstrated that the DBs effectively facilitated sedimentation, even during Overflow Events, suggesting that DBs should be able to effectively decrease sediment and sediment bound nutrients discharged from the DB catchments where soil infiltration rates and/or pond volume: catchment area ratios are not as high as those in the current study. Many factors, including particles sizes in runoff, nutrient enrichment of soil particles, dissolved nutrient concentrations in runoff would affect the ability of DBs to prevent nutrients from reaching downstream surface waters during Overflow and Non-Overflow Events.

An important factor identified during this study likely to affect the long-term efficacy of DBs is the lower soil infiltration rates measured in the ponding area compared to the soils outside the ponding area (Chapter 3). Although this study could not definitively conclude that attenuated sediments in the ponding area were responsible for the decreased soil infiltration rates, the data, along with related studies (Reddi et al., 2000; Rice, 1974), suggest this is at least a partial explanation. Concerns about declining soil infiltration rates may not be an issue when DBs are utilised in areas where erosion is not as prevalent as those in this study. However, in areas where erosion is more intense than those measured in this present study, soil infiltration rates in the ponded area could decrease more rapidly, and cause sharper declines in yield treatment efficiencies. In this case, most treatment would then be provided by sedimentation in the ponding area and dissolved nutrients discharged from the DBs infiltrating the soil downstream of the bund.

7.4 Priorities for DB development and research

The results of this current study emphasised the important role decreased runoff volumes discharged from the DBs played in preventing contaminants from reaching Lake Rotorua. While the current study offers a degree of support to the minimum pond volume: catchment size ratio established in the current DB site selection and design protocol, results also suggest that building DBs with greater ratios would contribute to greater treatment efficiencies. Developing models that are able to predict hydrologic responses to rainfall and DBs ability to retain portions of runoff during large storm events in specific locations, would help identify the most cost-effective locations to utilise DBs.

Additional benefits of utilising DBs could be derived from reducing stormflow magnitudes downstream of DBs. A project by the New Zealand Transportation Agency using DBs, with the main objective of protecting road infrastructure from flooding, began in 2018. Besides protecting infrastructure from flood pulses, the ability of DBs to decrease stormflow magnitudes would likely reduce sediment and nutrient mobilisation and transport occurring downstream of the bunds. These added benefits that were not within the scope of this thesis should be considered in cost: benefit analyses of the DB strategy.

Evidence of soil infiltration rates declining in the ponding area was recognised in this study (Chapter 3). Since soil infiltration was found to play a major role in yield treatment efficiencies, future research should investigate methods of maintaining or improving soil infiltration rates in DB ponding areas, and perhaps downstream of the bund. Future research should investigate the cause of declining infiltration rates since the cause will likely affect the appropriate measures to maintain and/or rehabilitate soil infiltration rates. If the large volumes of water that pond on and move through the soil are causing deterioration in soil structure, it might be necessary to apply subsurface amendments to the soil. If deposited sediments formed a lower permeability layer on top of existing soils and/or are clogging soil pores, then aerating the ponding area might restore infiltration rates.

Methods of improving concentration treatment efficiencies should also be investigated to improve contaminant yield treatment efficiencies. Identifying the most effective types of sorbents and flocculants, and the most effective location to incorporate them along with the DB strategy, whether in the ephemeral stream path upstream or downstream of the DB, or potentially in the ponding area, should be investigated. Also, farmers adopting DBs could place outlet valves 5-10 cm above ground level in order to let the bottom most layer of the pond, where sediments concentrate, to infiltrate the soil. The costs and benefits associated with decreasing release volumes, whether by raising the outlet valve or doing away with the discharge pipe system entirely should be investigated.

Efforts should be made to raise public awareness of the DB strategy and its ability to mitigate nutrient losses from pastures. Developing a simple decision tree to help decision makers determine whether DBs might be appropriate in a location would assist with the implementation of the strategy. Steps for farmers interested in implementing DBs into their pastoral agricultural system include:

- Accessing the ‘Detainment Bund Handbook’ (Paterson & Clarke, 2013).
- Assessing site topography to determine its fit within the Detainment Bund Applicability Model, which uses a GIS program for assessing a catchment’s suitability for the installation of DBs based on the pond storage volume: catchment size ratio (Paterson, 2019).

- Determining soil infiltration rates in potential ponding areas using basic methods such as rings or cylinders pushed into soil and charged with water.
- Communicating with farmers who have already implemented DBs in their agricultural system.

Key findings of this current study, which have advanced the understanding of the function and effectiveness of DBs as a nutrient mitigation strategy, include:

a) Identifying the key role soil infiltration played in decreasing the contaminant loads from potentially reaching Lake Rotorua.
b) Identifying the ability of DBs to decrease contaminant loads during rare, high magnitude runoff events.
c) Identifying decreased soil infiltration rates and the build-up of soil P concentrations in the ponding area, which may compromise the long-term performance of DBs.
d) Identifying that the DB strategy is a highly cost-effective P mitigation strategy.

This current study should be expanded to collect longer-term data from more DB locations. Results from this current study, and future studies, should be used in algorithms that estimate runoff and contaminant yields delivered to, and treated by DBs in specific locations, based on hydrologic and landscape conditions. These models should be integrated into nutrient management software such as OverseerFM®, MitAgator® and others, to increase the adoption of DBs and allow policy makers and farmers to account for the capacity of DB’s to reduce nutrient losses to surrounding water bodies.

References

Abell, J. M., & Hamilton, D. P. (2013). Bioavailability of phosphorus transported during storm flow to a eutrophic, polymictic lake. *New Zealand Journal of Marine and Freshwater Research, 47*(4), 481-489. doi:10.1080/00288330.2013.792851

Abell, J. M., Hamilton, D. P., & Rutherford, J. C. (2013). Quantifying temporal and spatial variations in sediment, nitrogen and phosphorus transport in stream inflows to a large eutrophic lake. *Environmental Science Processes & Impacts, 15*, 1137-1152. doi:10.1039/c3em00083d

Abrahamson, M., & Darkey, C. (1988). *Phosphours requirements and use for pastoral systems in New Zealand* (Vol. 31). Lincoln College: Tussock Grasslands and Mountain Lands Institute.

Ahuja, L., Sharpley, A., Yamamoto, M., & Menzel, R. (1981). The depth of rainfall-runoff-soil interaction as determined by ^{32}P. *Water Resources Research, 17*(4), 969-974. doi:10.1029/WR017i004p00969

Alexander, R. B., Boyer, E. W., Smith, R. A., Schwarz, G. E., & Moore, R. B. (2007). The role of headwater streams in downstream water quality. *Journal of the American Water Resources Association, 43*(1), 41-59. doi:10.1111/j.1752-1688.2007.00005.x

American Public Health Association. (2005). Standard methods for the examination of water and wastewater. In. Washington, DC, USA: Federation, Water, Environmental, and American Public Health Association.

ANZG. (2018). *Australian and New Zealand guidlines for fresh and marine water quality*. Canberra ACT, Australia Australian and New Zealand Governments and Australian state and territory governments Retrieved from www.waterquality.gov.au/anz-guidelines.

Barber, N. J., & Quinn, P. F. (2012). Mitigating diffuse water pollution from agriculture using soft-engineered runoff attenuation features. *Area, 44*(4), 454-462. doi:10.1111/j.1475-4762.2012.01118.x

Barrow, N. J. (2015). A mechanistic model for describing the sorption and desorption of phosphate by soil. *European Journal of Soil Science, 66*(1), 9-18. doi:10.1111/ejss.12198_2

Bay of Plenty Regional Council. (2012). *Improving water quality in Lake Rotorua: Information on the way land is used*. Whakatane, New Zealand: Bay of Plenty Regional Council.

Bay of Plenty Regional Council. (2018). Rotorua at Upper Oturoa Rd: Soil Moisture. Retrieved from http://monitoring.boprc.govt.nz/MonitoredSites/cgi-bin/hydwebserver.cgi/points/details?point=1314

Berkheiser, V. E., Street, J. J., Rao, P. S. C., Yuan, T. L., & Ellis, B. G. (1980). Partitioning of inorganic orthophosphate in soil-water systems. *C R C Critical Reviews in Environmental Control, 10*(3), 179-224. doi:10.1080/10643388009381680

Bieroza, M., Bergström, L., Ulén, B., Djodjic, F., Tonderski, K., Heeb, A., Svensson, J., & Malgeryd, J. (2019). Hydrologic extremes and legacy sources can override efforts to mitigate nutrient and sediment losses at the catchment scale. *Journal of Environmental Quality, 48*, 1314-1324. doi:10.2134/jeq2019.02.0063

Bieroza, M. Z., Heathwaite, A. L., Bechmann, M., Kyllmar, K., & Jordan, P. (2018). The concentration-discharge slope as a tool for water quality management. *Science of the Total Environment, 630*, 738-749. doi:10.1016/j.scitotenv.2018.02.256

Bilotta, G. S., Brazier, R. E., & Haygarth, P. M. (2007). The impacts of grazing animals on the quality of soils, vegetation, and surface waters in intensively managed grasslands. *Advances in Agronomy, 94*, 237-280. doi:10.1016/S0065-2113(06)94006-1

Bilotta, G. S., Krueger, T., Brazier, R. E., Freer, P. B., Hawkins, J. M. B., Haygarth, P. M., Macleode, C. J. A., & Quinton, J. N. (2009). Assessing catchment-scale erosion and yields of suspended solids fromimproved temperate grassland. *Journal of Environmental Monitoring, 12*, 731-739. doi:10.1039/B921584K

Boström, B., Andersen, J. M., Fleischer, S., & Jansson, M. (1988). Exchange of phosphorus across the sediment-water interface. *Hydrobiologia*, 229-244. doi:10.1007/978-94-009-3109-1_14

Braskerud, B. C. (2002a). Design considerations for increased sedimentation in small wetlands treating agricultural runoff. *Water Science and Technology, 45*, 77-85. doi:10.2166/wst.2002.0209

Braskerud, B. C. (2002b). Factors affecting phosphorus retention in small constructed wetlands treating agricultural non-point source pollution. *Ecological Engineering, 19*, 41-61. doi:10.1016/S0925-8574(02)00014-9

Braskerud, B. C., Lundekvam, H., & Krogstad, T. (2000). The impact of hydraulic load and aggregation on sedimentation of soil particles in small constructed wetlands. *Journal of Environment Quality, 29*, 2013-2020. doi:10.2134/jeq2000.00472425002900060039x

Brown, M. J., Bondurant, J. A., & Brockway, C. E. (1981). Ponding surface drainage water for sediment and phosphorus removal. *Tansactions of ASAE, 24*(6), 1478-1481. doi:10.13031/2013.34477

Buda, A. R., Kleinman, P. J. A., Srinivasan, M. S., Bryant, R. B., & Feyereisen, G. W. (2009). Effects of hydrology and field management on phosphorus transport in surface runoff. *Journal of Environment Quality, 38*(6), 2273-2284. doi:10.2134/jeq2008.0501

Burger, D. F., Hamilton, D. P., Pilditch, C. A., & Gibbs, M. M. (2007). Benthic nutrient fluxes in a eutrophic, polymictic lake. *Hydrobiologia*, *584*(1), 13-25. doi:10.1007/s10750-007-0582-0

Burkitt, L. (2014). A review of nitrogen losses due to leaching and surface runoff under intensive pasture management in Australia. *Soil Research, 52*, 621-636. doi:10.1071/SR13351

Burns, N. (2001). *Trophic level trends in 12 Rotorua district lakes, 1990 to 2000.* Paper presented at the A symposium on Research Needs of the Rotorua Lakes, Rotorua, New Zealand.

Burns, N., Bryers, G., & Bowman, E. (2000). *Protocol for monitoring trophic levels of New Zealand lakes and reservoirs*. (99/2). Pauanui, New Zealand: Ministry for the Environment

Burns, N., McIntosh, J., & Scholes, P. (2005). Strategies for Managing the Lakes of the Rotorua District, New Zealand. *Lake and Reservoir Management, 21*(1), 61-72. doi:10.1080/07438140509354413

Cameron, K., Di, H., & Condron, L. (2002). Nutrient and pesticide transfer from agricultural soils to water in New Zealand. In PM Haygarth & S. Jarvis (Eds.), *Agriculture, hydrology and water quality* (pp. 373–393). Wallingford, UK: CABI Publishing.

Carter, D., Brown, M. J., Robbins, C., & Bondurant, J. A. (1974). Phosphorus associated with sediments in irrigation and drainage waters for two large tracts in southern Idaho. *Journal of Environment Quality, 3*(3), 287-291. doi:10.2134/jeq1974.00472425000300030022x

Clarke, D. (2013). *The performance of Detainment Bunds (DBs) for attenuating phosphorus and sediment loss from pastoral farmland.* (Master of Science Masters thesis), University of Waikato, Hamilton, New Zealand. Retrieved from https://hdl.handle.net/10289/7993

Condron, L. M., Turner, B. L., & Cade-Menun, B. J. (2005). Chemistry and dynamics of soil organic phosphorus. In *Phosphorus: agriculture and the environment* (pp. 87-121).

Cooke, J. G. (1988). Sources and sinks of nutrients in a New Zealand hill pasture catchment II. Phosphorus. *Hydrologlcal Processes, 2*, 123-133. doi:10.1002/hyp.3360020203

Cooke, J. G., & Cooper, A. B. (1988). Sources and sinks of nutrients in a New Zealand hill pasture catchment III. Nitrogen. *Hydrological Processes, 2*, 135-149. doi:10.1002/hyp.3360020204

Cooke, J. G., & Dons, T. (1988). Sources and sinks of nutrients in a New Zealand hill pasture catchment I. Stormflow generation. *Hydrological Processes, 2*, 109-122. doi:doi.org/10.1002/hyp.3360020202

Cooper, A. B., & Thomsen, C. E. (1988). Nitrogen and phosphorus in streamwaters from adjacent pasture, pine, and native forest catchments. *New Zealand Journal of Marine and Freshwater Research, 22*(2), 279-291. doi:10.1080/00288330.1988.9516300

Correll, D. (1998). The role of phosphorus in the eutrophication of receiving waters: A review. *Journal of Environment Quality, 27*, 261-266. doi:10.2134/jeq1998.00472425002700020004x

Cosgrove, R. (2019). Water pollution is now New Zealanders' Number One Concern [Press release]. Retrieved from https://fishandgame.org.nz/news/water-pollution-is-now-new-zealanders-number-one-concern/

Curran Cournane, F., McDowell, R. W., & Condron, L. M. (2010). Effects of cattle treading and soil moisture on phosphorus and sediment losses in surface runoff from pasture. *New Zealand Journal of Agricultural Research, 53*(4), 365-376. doi:10.1080/00288233.2010.509903

Cuttle, S. P., Newll-Price, J. P., Harris, D., Chadwick, D. R., Shepherd, M. A., Anthony, S. G. A., MacLeod, C. J. A., Haygarth, P. M., & Chambers, B. J. (2016). A method-centric 'User Manual' for the mitigation of diffuse water pollution from agriculture. *Soil Use and Management, 32*, 162-171. doi:10.1111/sum.12242

Daniel, T. C., Edwards, D. R., & Sharpley, A. N. (1993). Effect of extractable soil surface phosphorus on runoff water quality. *Transactions of the ASAE, 36*(4), 1079-1085. doi:10.13031/2013.28437

Daniel, T. C., Sharpley, A. N., & Lemunyon, J. L. (1998). Agricultural phosphorus and eutrophication: A symposium overview. *Journal of Environment Quality, 27*, 251-257. doi:10.2134/jeq1998.00472425002700020002x

Dare, J. (2018). *Trends and State of Nutrients in Lake Rotorua Streams 2002-2016*. Whakatane, New Zealand: Bay of Plenty Regional Council.

Deasy, C., Brazier, R. E., Heathwaite, A. L., & Hodgkinson, R. (2009). Pathways of runoff and sediment transfer in small agricultural catchments. *Hydrological Processes, 23*, 1349-1358. doi:doi.org/10.1002/hyp.7257

Donald, R., Bruere, A., & Park, S. (2019). *Lake Rotorua Science Review - Summary Report*. Whakatane, New Zealand: Bay of Plenty Regional Council.

Dosskey, M. (2001). Toward quantifying water pollution abatement in response to installing buffers on crop land. *Environmental Management, 28*(5), 577–598. doi:10.1007/s002670010245

Dougherty, W. J., Fleming, N. K., Cox, J. W., & Chittleborough, D. J. (2004). Phosphorus Transfer in Surface Runoff from Intensive Pasture Systems at Various Scales. *Journal of Environment Quality, 33*(6), 1973. doi:10.2134/jeq2004.1973

Edgar, N. B. (2008). Icon Lakes in New Zealand: Managing the Tension Between Land Development and Water Resource Protection. *Society & Natural Resources, 22*(1), 1-11. doi:10.1080/08941920802223325

Edwards, A. C., & Withers, P. J. A. (2008). Transport and delivery of suspended solids, nitrogen and phosphorus from various sources to freshwaters in the UK. *Journal of Hydrology, 350*(3-4), 144-153. doi:10.1016/j.jhydrol.2007.10.053

Ekholm, P. (1998). Algal-available phosphorus originating from agriculture and municipalities. *Monographs of the Boreal Environment Research, 11,* 1-64.

Elliott, A. H., Alexander, R.B., Schwarz, G.E., Shankar, U., Sukias, J.P.S. and McBride, G.B. (2005). Estimation of nutrient sources and transport for New Zealand using the hybrid mechanistic-statistical model SPARROW. *Journal of Hydrology (New Zealand), 44*(1), 1-27.

Emerson, C., & Traver, R. (2008). Multiyear and seasonal variation of infiltration from storm-water best management practices. *Journal of Irrigation and Drainage Engineering, 134*(5), 598-605.

Environment Bay of Plenty. (2009). *Lakes Rotorua and Rotoiti Action Plan*. Environment Bay of Plenty.

Friedl, J., De Rosa, D., Rowlings, D. W., Grace, P. R., Müller, C., & Scheer, C. (2018). Dissimilatory nitrate reduction to ammonium (DNRA), not denitrification dominates nitrate reduction in subtropical pasture soils upon rewetting. *Soil biology and biochemistry, 125,* 340-349. doi:10.1016/j.soilbio.2018.07.024

Gburek, W. J., & Sharpley, A. N. (1998). Hydrologic controls on phosphorus loss from upland agricultural watersheds. *Journal of Environmental Quality, 27*(2), 267-277. doi:10.2134/jeq1998.00472425002700020005x

Genuer, R., Poggi, J.-M., & Tuleu-Malot, C. (2015). VSURF: An R Package for Variable Selection Using Random Forests.

Greenhill, N., Peverill, K., & Douglas, L. (1983). Nutrient concentrations in runoff from pasture in Westernport, Victoria. *Australian Journal of Soil Research, 21,* 139–145. doi:10.1071/SR9830139

Hall, R. (2008). Nitrogen in soil. In *Soil essentials* (pp. 25-27). Melbourne: Landlinks Press, CSIRO Publishing.

Hamill, K., MacGibbon, R., & Turner, J. (2010). *Wetland feasibility for nutrient reduction to Lake Rotorua*. Whakatane, New Zealand: Bay of Plenty Regional Council

Hamill, K. D. (2018). *Anthropogenic phosphorus load to Rotorua review and revision*. Whakatane, New Zealand: Bay of Plenty Regional Council.

Hamilton, D. P., McBride, C. G., & Jones, H. F. E. (2015). *Assessing the effects of alum dosing of two inflows to Lake Rotorua against external nutrient load reductions:*

Model simulations for 2001-2012. University of Waikato, Hamilton, New Zealand: ERI Report.

Harmel, R., King, K., & Slade, R. (2003). Automated storm water sampling on small watersheds. *Applied Engineering in Agriculture, 19*(6), 667-674. doi:10.13031/2013.15662

Harmel, R., King, K., Wolfe, J., & Torbert, H. (2002). Minimum flow considerations for automated storm sampling on small watersheds. *Texas Journal of Science, 54*(2), 177-188.

Harper, H. H., Herr, J. L., Baker, D., & Livingston, E. H. (1999). *Performance evaluation of dry detention stormwater management systems*. Paper presented at the Stormwater Research & Watershed Management Conference, Tampa, Florida.

Harrod, T. R., & Theurer, F. D. (2002). Sediment. In P. M. Haygarth & S. C. Jarvis (Eds.), *Agriculture, hydrology and water quality* (pp. 155-171). Wallingford, UK: CABI Publishing.

Hart, M. R., Quin, B. F., & Nguyen, M. L. (2004). Phosphorus Runoff from Agricultural Land and Direct Fertilizer Effects. *Journal of Environmental Quality, 33*(6), 1954. doi:10.2134/jeq2004.1954

Hatch, D., Goulding, K., & Murphy, D. (2002). Nitrogen. In S. J. PM Haygarth (Ed.), *Agriculture, hydrology and water quality*. Wallingford, UK: CABI Publishing.

Haygarth, P. M., ApSimon, H., M. Betson, D. Harris, R. Hodgkinson, & Withers, P. J. A. (2009). Mitigating diffuse phosphorus transfer from agriculture according to cost and efficiency. *Journal of Environment Quality, 38*, 2012-2022. doi:10.2134/jeq2008.0102

Haygarth, P. M., Bilotta, G. S., Bol, R., Brazier, R. E., Butler, P. J., Freer, J., Gimbert, L. J., Granger, S. J., Krueger, T., Macleod, C. J. A., Naden, P., Old, G., Quinton, J. N., Smith, B., & Worsfold, P. (2006). Processes affecting transfer of sediment and colloids, with associated phosphorus, from intensively farmed grasslands: an overview of key issues. *Hydrological Processes, 20*, 4407-4413. doi:10.1002/hyp.6598

Haygarth, P. M., Condron, L. M., Heathwaite, A. L., Turner, B. L., & Harris, G. P. (2005). The phosphorus transfer continuum: linking source to impact with an interdisciplinary and multi-scaled approach. *Sci Total Environ, 344*(1-3), 5-14. doi:10.1016/j.scitotenv.2005.02.001

Haygarth, P. M., & Edwards, A. C. (2009). *Water sample collection, handling, preparation and storage* (J. L. Kovar & G. M. Pierzynski Eds.): Virginia Tech University.

Haygarth, P. M., Hepworth, L., & Jarvis, S. C. (1998). Forms of phosphorus transfer in hydrological pathways from soil under grazed grassland. *European Journal of Soil Science, 49*(1), 65-72. doi:10.1046/j.1365-2389.1998.00131.x

Heathwaite, A. L., & Dils, R. M. (2000). Characterising phosphorus loss in surface and subsurface hydrological pathways. *Science of the Total Environment*, *251*, 523-538. doi:10.1016/S0048-9697(00)00393-4

Hendrickson, B. H. (1934). The choking of pore-space in the soil and its relation to runoff and erosion. *Transactions American Geophysical Union, 15*, 500-505. doi:10.1029/TR015i002p00500

Hill, R. (2018). *A review of land-based phosphorus loss and mitigation strategies for the Lake Rotorua catchment. Technical report produced for Lake Rotorua Technical Advisory Group.*

Holtan, H., Kamp-Nielsen, L., & Stuanes, A. (1988). Phosphorus in soil, water and sediment: An overview. *Hydrobiologia*, *170*, 19-34. doi:10.1007/978-94-009-3109-1_3

Hosomi, M., & Sudo, R. (1986). Simultaneous determination of total nitrogen and total phosphorus in freshwater samples using persulphate digestion. *International Journal of Environmental Studies, 27*, 267-275. doi:10.1080/00207238608710296

Howard-Williams, C., Davies-Colley, R., Rutherford, K., & Wilcock, R. (2010). *Diffuse pollution and freshwater degradation: New Zealand perspectives.* Paper presented at the Issues and Solutions to Diffuse Pollution: Selected Papers from the 14th International Conference of the IWA Diffuse Pollution Specialist Group, DIPCON 2010, Québec, Canada.

Jayaweera, G. R., & Mikkelsen, D. S. (1991). Assessment of ammonia volatilization from flooded soil systems. In N. Brady (Ed.), *Advances in Agronomy* (pp. 303-356). New York: Academic Press.

Jaynes, D. (1990). Temperature variations effect on field-measured infiltration. *Soil Science Society of America Journal, 54*, 305-312.

Jordan-Meille, L., & Dorioz, J. M. (2004). Soluble phosphorus dynamics in an agricultural watershed. *Agronomie, 24*(5), 237-248. doi:10.1051/agro:2004021

Kleinman, P. J. A., Sharpley, A., Buda, A., McDowell, R. W., & Allen, A. (2011). Soil controls of phosphorus in runoff: Management barriers and opportunities. *Canadian Journal of Soil Science, 91*(3), 329-338. doi:10.4141/cjss09106

Kleinman, P. J. A., Sharpley, A. N., Moyer, B. G., & Elwinger, G. F. (2002). Effect of mineral and manure phosphorus sources on runoff phosphorus. *Journal of Environmental Quality, 31*(6), 2026-2033. doi:10.2134/jeq2002.2026

Kleinman, P. J. A., Srinivasan, M. S., Dell, C. J., Schmidt, J. P., Sharpley, A. N., & Bryant, R. B. (2006). Role of rainfall intensity and hydrology in nutrient transport via surface runoff. *Journal of Environmental Quality, 35*(4), 1248-1259. doi:10.2134/jeq2006.0015

Korson, L., Drost-Hansen, W., & Millero, F. (1969). Viscosity of water at various temperatures. *Journal of Physical Chemistry*, *73*(1), 34-39.

Kronvang, B., Vagstad, N., Behrendt, H., Bøgestrand, J., Larsen, S. E. (2007). Phosphorus losses at the catchment scale within europe: An overview. *Soil Use and Management, 23*, 104-116.

Land and Water Forum. (2010). *Report of the Land and Water Forum: A Fresh Start for Fresh Water*. Wellington, New Zealand: Land and Water Forum.

Landcare Research. (2017). Smap-Online Software. Retrieved from http://smap.landcareresearch.co.nz/home

Larned, S. T., Snelder, T., Unwin, M., McBride, G. B., Verburg, P., & McMillan, H. (2015). *Analysis of water quality in New Zealand lakes and rivers*. Christchurch, New Zealand: NIWA.

Larned, S. T., Snelder, T., Unwin, M. J., & McBride, G. B. (2016). Water quality in New Zealand rivers: current state and trends. *New Zealand Journal of Marine and Freshwater Research, 50*(3), 389-417. doi:10.1080/00288330.2016.1150309

Ledgard, S., Penno, J., & Sprosen, M. (1999). Nitrogen inputs and losses from clover/grass pastures grazed by dairy cows, as affected by nitrogen fertilizer application. *Journal of Agricultural Science*, *132*, 215-225. doi:10.1017/S002185969800625X

Letcher, R. A., Jakeman, A. J., Merritt, W. S., McKee, L. J., Eyre, B. D., & Baginska, B. (1999). *Review of techniques to estimate catchment exports*. Sydeny, Australia: Environment Protection Authority.

Levine, B., Burkitt, L., Horne, D., Condron, L., Tanner, C., & Paterson, J. (2019). Preliminary assessment of the ability of detainment bunds to attenuate sediment and phosphorus transported by surface runoff in the Lake Rotorua catchment. *Animal Production Science*, *60*, 154-158. doi:10.1071/AN18544

Lucci, G. M., McDowell, R. W., & Condron, L. M. (2012). Phosphorus source areas in a dairy catchment in Otago, New Zealand. *Soil Research, 50*(2), 145-157. doi:10.1071/SR12030

Matheson, F. E., Nguyen, M. L., Cooper, A. B., Burt, T. P., & Bull, D. C. (2002). Fate of 15N-nitrate in unplanted, planted and harvested riparian wetland soil microcosms. *Ecological Engineering*, *19*, 249-264. doi:10.1016/S0925-8574(02)00093-9

McBride, C. G., Abell, J. M., & Hamilton, D. P. (2018). Long-term nutrient loads and water quality for Lake Rotorua: 1965 to 2017. ERI report 123. Environmental Research Institute, The University of Waikato, Hamilton, New Zealand. 70 pp.

McDowell, R. W. (2010). *The efficacy of strategies to mitigate the loss of phosphorus from pastoral land use in the catchment of Lake Rotorua*. Environment Bay of Plenty.

McDowell, R. W. (2012). Minimising phosphorus losses from the soil matrix. *Current Opinions in Biotechnology, 23*(6), 860-865. doi:10.1016/j.copbio.2012.03.006

McDowell, R. W., Biggs, B. J. F., Sharpley, A. N., & Nguyen, L. (2004). Connecting phosphorus loss from agricultural landscapes to surface water quality. *Chemistry and Ecology, 20*(1), 1-40. doi:10.1080/02757540310001626092

McDowell, R. W., Cox, N., & Snelder, T. H. (2017). Assessing the yield and load of contaminants with stream order: Would policy requiring livestock to be fenced out of high-order streams decrease catchment contaminant loads? *Journal of Environmental Quality, 46*, 1038-1047. doi:10.2134/jeq2017.05.0212

McDowell, R. W., Drewry, J. J., Paton, R. J., Carey, P. L., Monaghan, R. M., & Condron, L. M. (2003). Influence of soil treading on sediment and phosphorus losses in overland flow. *Australian Journal of Soil Research, 41*(5), 949. doi:10.1071/sr02118

McDowell, R. W., Houlbrooke, D. J., Muirhead, R. W., Muller, K., Shepherd, M., & Cuttle, S. P. (2008). *Grazed Pastures and Surface Water Quality*: Nova Publishers.

McDowell, R. W., McGrouther, N., Morgan, G., Srinivasan, M. S., Stevens, D., Johson, M., & Copland, R. (2006). Monitoring of the impact of farm practices on water quality in the Otago and Southland deer focus farms. *Proceeding of the New Zealand Grassland Association, 68*, 183-188.

McDowell, R. W., & Nash, D. M. (2012). A review of the cost-effectiveness and suitability of mitigation strategies to prevent phosphorus loss from dairy farms in New Zealand and Australia. *Journal of Environment Quality, 41*, 680-693. doi:10.2134/jeq2011.0041

McDowell, R. W., Sharpley, A., Condron, L., Haygarth, P., & Brookes, P. (2001). Processes controlling soil phosphorus release to runoff and implications for agricultural management. *Nutrient Cycling in Agroecosystems, 59*(3), 269-284. doi:10.1023/A:1014419206761

McDowell, R. W., & Sharpley, A. N. (2001). Approximating phosphorus release from soils to surface runoff and subsurface drainage. *Journal of Environment Quality, 30*, 508-520. doi:10.2134/jeq2001.302508x

McDowell, R. W., & Sharpley, A. N. (2002). The effect of antecedent moisture conditions on sediment and phosphorus loss during overland flow: Mahantango Creek catchment, Pennsylvania, USA. *Hydrological Processes, 16*(15), 3037-3050. doi:10.1002/hyp.1087

McDowell, R. W., & Sharpley, A. N. (2003). Uptake and Release of Phosphorus from Overland Flow in a Stream Environment. *Journal of Environment Quality, 32*(3), 937. doi:10.2134/jeq2003.0937

McDowell, R. W., Sharpley, A. N., Kleinman, P. J. A., & Gburek, W. J. (2002). Hydrological Source Management of Pollutants at the Soil Profile Scale. In P. M. Haygarth & S. C. Jarvis (Eds.), *Agriculture, Hydrology and Water Quality* (pp. 197-223): CAB International.

McDowell, R. W., Wilcock, B., & Hamilton, D. P. (2013). *Assessment of Strategies to Mitigate the Impact or Loss of Contaminants from Agricultural Land to Fresh Waters*. Wellington, New Zealand: Ministry for the Environment.

McDowell, R. W., & Wilcock, R. J. (2007). Sources of sediment and phosphorus in stream flow of a highly productive dairy farmed catchment. *J Environ Qual, 36*(2), 540-548. doi:10.2134/jeq2006.0352

McKergow, L., Tanner, C., Monaghan, R., & Anderson, G. (2007). *Stocktake of diffuse pollution attenuation tools for New Zealand pastoral farming systems*. Hamilton, New Zealand: NIWA.

Milner, D., Cole, J., & Wood, C. (2003). Mamaku Ignimbrite: a caldera-forming ignimbrite erupted from a compositionally zoned magma chamber in Taupo Volcanic Zone, New Zealand. *Journal of Volcanology and Geothermal Research, 122,* 243-264. doi:10.1016/S0377-0273(02)00504-8

Ministry for Primary Industries. (2016). *Situation and outlook for primary industries 2016*. Wellington, New Zealand: Minisitry for Primary Industries.

Ministry for the Envrionment. (2011). *New start for freshwater*. Office of the Minister for the Envrionment and Minister of Agriculture.

Ministry for the Envrionment. (2017). Introduction to the RMA. Retrieved from http://www.mfe.govt.nz/rma/introduction-rma

Ministry for the Envrionment. (2019). Environment Aotearoa 2019 Summary. *Environment Aotearoa 2019.* Retrieved from https://www.mfe.govt.nz/environment-aotearoa-2019-summary

Ministry for the Envrironment. (2019, 2019/03/22). Likely climate change impacts in New Zealand. Retrieved from https://www.mfe.govt.nz/climate-change/likely-impacts-of-climate-change/likely-climate-change-impacts-nz

Monaghan, R., Hedley, M., Di, H., McDowell, R. W., Cameron, K., & Ledgard, S. (2007). Nutrient management in New Zealand pastures—recent developments and future issues. *New Zealand Journal of Agricultural Research, 50*(2), 181-201. doi:10.1080/00288230709510290

Monaghan, R. M., Paton, R. J., Smith, L. C., & Binet, C. (2000). Nutrient losses in drainage and surface runoff from a cattle-grazed pasture in Southland. *Proceedings of the New Zealand Grassland Association, 62*, 99-104.

Morgenstern, U., Daughney, C. J., Leonard, G., Gordon, D., Donath, F. M., & Reeves, R. (2015). Using groundwater age and hydrochemistry to understand sources and dynamics of nutrient contamination through the catchment into Lake Rotorua, New Zealand. *Hydrology and Earth System Sciences, 19*(2), 803-822. doi:10.5194/hess-19-803-2015

Muller, K., Srinivasan, M., Trolove, M., & McDowell, R. W. (2010). Identifying and linking source areas of flow and P transport in dairy-grazed headwater catchments, North Island, New Zealand. *Hydrological Processes, 24*, 3689-3705. doi:10.1002/hyp.7809

Murphy, J., & Riley, J. (1962). A modified single solution method for the determination of phosphate in natural waters. *Analytica chimica acta, 27*, 31-36. doi:10.1016/S0003-2670(00)88444-5

Nie, S., Zhu, G.-B., Singh, B., & Zhu, Y.-G. (2019). Anaerobic ammonium oxidation in agricultural soils-synthesis and prospective. *Environmental pollution, 244*, 127-134. doi:10.1016/j.envpol.2018.10.050

NIWA. (2004). Environmental data operations manual. In *AS/NZS ISO 9001:2003*. Wellington, NZ: NIWA Science and Technology Series, National Institute of Water and Atmospheric Research.

NIWA. (2019). Rotorua Penman evaporation. *National Climate Database.* Retrieved from https://cliflo.niwa.co.nz

Ockenden, M. C., Deasy, C. E., Benskin, C. M. H., Beven, K. J., Burke, S., Collins, A. L., Evans, R., Falloon, P. D., Forber, K. J., Hiscock, K. M., Hollaway, M. J., Kahana, R., Macleod, C. J. A., Reaney, S. M., Snell, M. A., Villamizar, M. L., C.Wearing, Withers, P. J. A., Zhou, J. G., & Haygarth, P. M. (2016). Changing climate and nutrient transfers: Evidence from high temporal resolution concentration-flow dynamics in headwater catchments. *Science of the Total Environment, 548-549*, 325-339. doi:10.1016/j.scitotenv.2015.12.086

Ockenden, M. C., Hollaway, M. J., Beven, K. J., Collins, A. L., Evans, R., Falloon, P. D., Forber, K. J., Hiscock, K. M., Kahana, R., Macleod, C. J. A., & Tych, W. (2017). Major agricultural changes required to mitigate phosphorus losses under climate change. *Nature Communications*, *8*(161), 1-9. doi:10.1038/s41467-017-00232-0

Olsen, S. R., Cole, C. V., Watanbe, F. S., & Dean, L. A. (1954). *Estimation of Available Phosphorus in Soils by Extraction with Sodium Bicarbonate*. Washington DC: U.S. Government Printing Office.

Osmond, D. L., Shober, A. L., Sharpley, A. N., Duncan, E. W., & Hoag, L. K. (2019). Increasing the effectiveness and adoption of agricultural phosphorus management strategies to minimize water quality impairment. *Journal of Environmental Quality*, *48*, 1204-1217. doi:10.2134/jeq2019.03.0114

Owens, P. (2005). Conceptual models and budgets for sediment management at the river basin scale. *Journal of Soils and Sediments, 5*(4), 201-212.

Paterson, J., & Clarke, D. (2013). *The Detainment Bund Handbook*. Whakatane, New Zealand: Bay of Plenty Regional Council.

Paterson, J. H. (2019). *The DB Applicability Model: A GIS model for assessing catchments' suitability for the installation of Detainment Bunds PS*^{120} *to mitigate storm water runoff*. Rotorua, New Zealand: Bay of Plenty Regional Council and NZ Ministry for the Environment.

Pionke, H. B., Gburek, W. J., & Sharpley, A. N. (2000). Critical source area controls on water quality in an agricultural watershed located in the Chesapeake Basin. *Ecological Engineering, 14*, 325-335. doi:10.1016/S0925-8574(99)00059-2

Pionke, H. B., Gburek, W. J., Sharpley, A. N., & Schnabel, R. R. (1996). Flow and nutrient export patterns for an agricultural hill-land watershed. *Water Resources Research, 32*(6), 1795-1804. doi:10.1029/96WR00637

Ponnamperuma, F. (1972). *The Chemistry of Submerged Soils*. New York: Academic Press.

Pote, D. H., Daniel, T. C., Moore, P. A., Nichols, D. J., Sharpley, A. N., & Edwards, D. R. (1996). Relating extractable soil phosphorus to phosphorus losses in runoff. *Soil Science Society of America Journal, 60*(3), 855-859. doi:10.2136/sssaj1996.03615995006000030025x

Qui, S., & McComb, A. (1996). Drying-induced stimulation of ammonium release and nitrification in reflooded lake sediment. *Marine and Freshwater Research, 47*, 531-536. doi:10.1071/MF9960531

Quinn, J. M., Wilcock, R. J., Monaghan, R. M., McDowell, R. W., & Journeaux, P. R. (2009). Grassland farming and water quality in New Zealand. *Tearmann: Irish journal of agri-environmental research, 7*, 69-88.

R Development Core Team. (2019). R: A Language and Environment for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://www.R-project.org

Reddi, L., Ming, X., Hajra, M., & Lee, I. (2000). Permeability reduction of soil filters due to physical clogging. *Journal of Geotechnical and Geoenvironmental Engineering, 126*(3), 236-246. doi:10.1061/(ASCE)1090-0241(2000)126:3(236)

Reddy, K. R., & DeLaune, R. D. (2008). *Biogeochemistry of Wetlands: Science and Applications*: CRC Press.

Rice, R. (1974). Soil clogging during infiltration of secondary effluent. *Journal of Water Pollution Control Federation, 46*(4), 708-716.

Rijkse, W. C., & Guinto, D. F. (2010). *Soils of the Bay of Plenty Volume 2: Central Bay of Plenty*. Whakatane, NZ: Environment Bay of Plenty.

Roberts, A. H. C., & Morton, J. D. (2012). *Fertiliser use on New Zealand dairy farms: the principles and practice of soil fertility and fertiliser use on New Zealand dairy farms*: New Zealand Fertiliser Manufacturers' Research Association.

Rutherford, K., & Timpany, G. (2008). *Storm nutrient loads in Rotorua streams*. Hamilton, New Zealand: NIWA.

Sánchez-Rodríguez, A. R., Hill, P. W., Chadwick, D. R., & Jones, D. L. (2019). Typology of extreme flood event leads to differential impacts on soil functioning. *Soil biology and biochemistry, 129*, 153-168. doi:10.1016/j.soilbio.2018.11.019

Saunders, W. (1965). Phosphate retention in New Zealand soils and it's relation to free sesquioxides, organic matter and other soil properties. *New Zealand Journal of Agricultural Research, 8*(30), 30-57. doi:10.1080/00288233.1965.10420021

Sharpley, A. (1985). The Selective Erosion of Plant Nutrients in Runoff. *Soil Science Society of America Journal, 49*, 1527-1534. doi:10.2136/sssaj1985.03615995004900060039x

Sharpley, A., Ahuja, L., & Menzel, R. (1981a). The release of soil phosphorus to runoff in relation to the kinetics of desorption. *Journal of Environmental Quality, 10*(3), 386-391. doi:10.2134/jeq1981.00472425001000030029x

Sharpley, A. N., Chapra, S. C., Wedepohl, R., Sims, J. T., Daniel, T. C., & Reddy, K. R. (1994). Managing Agricultural Phosphorus for Protection of Surface Waters: Issues and Options. *Journal of Environment Quality, 23*(3), 437. doi:10.2134/jeq1994.00472425002300030006x

Sharpley, A. N., McDowell, R. W., & Kleinman, P. J. A. (2001). Phosphorus loss from land to water: integrating agricultural and environmental management. *Plant and Soil, 237*(2), 287-307. doi:10.1023/A:1013335814593

Sharpley, A. N., Menzel, R. G., Smith, S. J., Rhoades, E. D., & Olness, A. E. (1981b). The sorption of soluble phosphorus by soil material during transport in runoff from cropped and grassed watersheds. *Journal of Environmental Quality, 10*(2), 211-215. doi:10.2134/jeq1981.00472425001000020018x

Sharpley, A. N., Smith, S. J., Jones, O. R., Berg, W. A., & Coleman, G. A. (1992). The transport of bioavailable phosphorus in agricultural runoff. *Journal of Environmental Quality, 21*, 30-35. doi:10.2134/jeq1992.00472425002100010003x

Shukla, S., Shukla, A., Knowles, J., & Harris, W. (2017). Shifting nutrient sink and source functions of stormwater detention areas in sub-tropics. *Ecological Engineering, 102*, 178-187. doi:10.1016/j.ecoleng.2017.01.034

Skaggs, R. W., Brevé, M. A., & Gilliam, J. W. (1994). Hydrologic and water quality impacts of agricultural drainage. *Critical Reviews in Environmental Science and Technology, 24*(1), 1-32. doi:10.1080/10643389409388459

Smith, C. M. (1987). Sediment, phosphorus, and nitrogen in channelised surface run-off from a New Zealand pastoral catchment. *New Zealand Journal of Marine and Freshwater Research, 21*(4), 627-639. doi:10.1080/00288330.1987.9516268

Smith, L. C., & Monaghan, R. M. (2003). Nitrogen and phosphorus losses in overland flow from a cattle-grazed pasture in Southland. *New Zealand Journal of Agricultural Research, 46*(3), 225-237. doi:10.1080/00288233.2003.9513549

Stanley, D. (1996). Pollutant removal by a stormwater dry detention pond. *Water Environment Research, 68*(6), 1076-1083. doi:10.2175/106143096X128072

Stats NZ. (2019). Lake water quality. Retrieved from https://statisticsnz.shinyapps.io/lake_water_quality/

Stekhoven, D. J., & Bühlmann, P. (2012). MissForest—non-parametric missing value imputation for mixed-type data. *Bioinformatics, 28*(1), 112-118. doi:10.1093/bioinformatics/btr597

Stephens, T., Hamill, K., & McBride, C. (2018). *Lake Rotorua: Trends in Water quality (2001-2017)*. Technical report produced for Lake Rotorua Technical Advisory Group.

Tanner, C. C., & Sukias, J. P. S. (2011). Multiyear Nutrient Removal Performance of Three Constructed Wetlands Intercepting Tile Drain Flows from Grazed Pastures. *Journal of Environment Quality, 40*(2), 620. doi:10.2134/jeq2009.0470

Tempero, G., McBride, C., Abell, J., & Hamilton, D. (2015). *Anthropogenic phosphorus loads to Lake Rotorua*. Hamilton, New Zealand: Bay of Plenty Regional Council.

Thorrold, B., & Doyle, P. (2007). Nature or nurture – forces shaping the current and future state of dairy farming in New Zealand and Australia. In D. Chapman, D. Clark, K. Macmillan, & D. Nation (Eds.), *Meeting the challenges for pasture-based dairying* (pp. 450-460). Melbourne, Australia: National Dairy Alliance.

Valett, H. M., Baker, M. A., Morrice, J. A., Crawford, C. S., Jr, M. C. M., Dahm, C. N., Moyer, D. L., Thibault, J. R., & Ellis, L. M. (2005). Biogeochemical and metabolic responses to the flood pulse in a semiarid floodplain. *Ecology, 86*, 220-234. doi:10.1890/03-4091

Verburg, P., Hamill, K., Unwin, M., & Abell, J. (2010). *Lake water quality in New Zealand 2010: Status and trends*. Hamilton, New Zealand: NIWA.

Ward, J. C., Talbot, J. M., Denne, T., & Abrahamson, M. (1985). *Phosphorus losses through transfer, soil erosion and runoff: processes and implications*: Center for Resource Managment: Lincoln Collge and University of Canterbury.

White, P., Tschritter, C., Levett, A., & Cusi, M. (2014). *Lake Rotorua catchment boundry relevant to Bay of Plenty Regional Council's water and land management policies*. GNS Science Consultancy.

Withers, P. J., & Jarvie, H. P. (2008). Delivery and cycling of phosphorus in rivers: a review. *Science of the Total Environment, 400*(1-3), 379-395. doi:10.1016/j.scitotenv.2008.08.002

Wood, C. (1992). Geology of the Rotorua geothermal system. *Geothermics, 21*, 25-41. doi:10.1016/0375-6505(92)90066-I

Woodward, K., Fellows, C., Mitrovic, S., & Sheldon, F. (2015). Patterns and bioavailability of soil nutrients and carbon across a gradient of inundation frequencies in a lowland river channel, Murray–Darling Basin, Australia. *Agriculture, Ecosystems and Environment*, 1-8. doi:10.1016/j.agee.2015.02.019

Wurtsbaugh, W. A., Paerl, H. W., & Dodds, W. K. (2019). Nutrients, eutrophication and harmful algal blooms along the freshwater to marine continuum. *Wiley Interdisciplinary Reviews: Water, 6*, e1373. doi:doi.org/10.1002/wat2.1373

Appendix

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MASSEY UNIVERSITY
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STATEMENT OF CONTRIBUTION
DOCTORATE WITH PUBLICATIONS/MANUSCRIPTS

We, the candidate and the candidate's Primary Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated below in the *Statement of Originality*.

Name of candidate:	Brian Levine
Name/title of Primary Supervisor:	Lucy Burkitt
Name of Research Output and full reference:	
The ability of detainment bunds to decrease surface runoff leaving pastoral catchments: Investigating a novel approach to agricultural stormwater management	
In which Chapter is the Manuscript /Published work:	Chapter 3
Please indicate:	
• The percentage of the manuscript/Published Work that was contributed by the candidate:	80
and	
• Describe the contribution that the candidate has made to the Manuscript/Published Work:	
Brian developed the experimental protocol, assisted in setting up the study site locations and proceeded to collect and analyse samples and data gathered from the field. Brian also prepared the manuscript that was submitted for publication.	
For manuscripts intended for publication please indicate target journal:	
Agricultural Water Management	
Candidate's Signature:	Brian Levine — Digitally signed by Brian Levine Date: 2020.03.30 16:40:19 +13'00'
Date:	30/3/2020
Primary Supervisor's Signature:	Lucy Burkitt — Digitally signed by Lucy Burkitt Date: 2020.03.30 19:58:20 +13'00'
Date:	30/03/2020

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Name of candidate:	Brian Levine
Name/title of Primary Supervisor:	Lucy Burkitt
Name of Research Output and full reference:	
The ability of detainment bunds to decrease sediment loss from pastoral catchments in surface runoff: Investigating a novel stormwater mitigation strategy	
In which Chapter is the Manuscript /Published work:	Chapter 4
Please indicate:	
• The percentage of the manuscript/Published Work that was contributed by the candidate:	80
and	
• Describe the contribution that the candidate has made to the Manuscript/Published Work:	
Brian developed the experimental protocol, assisted in setting up the study site locations and proceeded to collect and analyse samples and data gathered from the field. Brian also prepared the manuscript that was submitted for publication.	
For manuscripts intended for publication please indicate target journal:	
Hydrological Processes	
Candidate's Signature:	Brian Levine Digitally signed by Brian Levine Date: 2020.03.30 16:40:19 +13'00'
Date:	30/3/2020
Primary Supervisor's Signature:	Lucy Burkitt Digitally signed by Lucy Burkitt Date: 2020.03.30 20:00:19 +13'00'
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Name of candidate:	Brian Levine
Name/title of Primary Supervisor:	Lucy Burkitt
Name of Research Output and full reference:	
The ability of detainment bunds to reduce phosphorus losses in surface runoff from pastoral catchments: A novel strategy for the nutrient mitigation toolbox	
In which Chapter is the Manuscript /Published work:	Chapter 5
Please indicate:	
• The percentage of the manuscript/Published Work that was contributed by the candidate:	80
and	
• Describe the contribution that the candidate has made to the Manuscript/Published Work:	
Brian developed the experimental protocol, assisted in setting up the study site locations and proceeded to collect and analyse samples and data gathered from the field. Brian also prepared the manuscript that was submitted for publication.	
For manuscripts intended for publication please indicate target journal:	
Ecological Engineering	
Candidate's Signature:	Brian Levine Digitally signed by Brian Levine Date: 2020.03.30 16:40:19 +13'00'
Date:	30/3/2020
Primary Supervisor's Signature:	Lucy Burkitt Digitally signed by Lucy Burkitt Date: 2020.03.30 19:59:23 +13'00'
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STATEMENT OF CONTRIBUTION
DOCTORATE WITH PUBLICATIONS/MANUSCRIPTS

We, the candidate and the candidate's Primary Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated below in the *Statement of Originality*.

Name of candidate:	Brian Levine
Name/title of Primary Supervisor:	Lucy Burkitt
Name of Research Output and full reference:	
Effect of impeding stormflows with detainment bunds on nitrogen transport in surface runoff from pastoral catchments	
In which Chapter is the Manuscript /Published work:	Chapter 6
Please indicate:	
• The percentage of the manuscript/Published Work that was contributed by the candidate:	80
and	
• Describe the contribution that the candidate has made to the Manuscript/Published Work:	
Brian developed the experimental protocol, assisted in setting up the study site locations and proceeded to collect and analyse samples and data gathered from the field. Brian also prepared the manuscript that was submitted for publication.	
For manuscripts intended for publication please indicate target journal:	
Soil Research	
Candidate's Signature:	Brian Levine Digitally signed by Brian Levine Date: 2020.03.30 16:40:19 +13'00'
Date:	30/3/2020
Primary Supervisor's Signature:	Lucy Burkitt Digitally signed by Lucy Burkitt Date: 2020.03.30 20:09:50 +13'00'
Date:	30/03/2020

(This form should appear at the end of each thesis chapter/section/appendix submitted as a manuscript/ publication or collected as an appendix at the end of the thesis)

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