



THE UNIVERSITY OF  
**WAIKATO**  
*Te Whare Wānanga o Waikato*

Research Commons

<http://researchcommons.waikato.ac.nz/>

## Research Commons at the University of Waikato

### Copyright Statement:

The digital copy of this thesis is protected by the Copyright Act 1994 (New Zealand).

The thesis may be consulted by you, provided you comply with the provisions of the Act and the following conditions of use:

- Any use you make of these documents or images must be for research or private study purposes only, and you may not make them available to any other person.
- Authors control the copyright of their thesis. You will recognise the author's right to be identified as the author of the thesis, and due acknowledgement will be made to the author where appropriate.
- You will obtain the author's permission before publishing any material from the thesis.

**The role of mānuka (*Leptospermum scoparium*)  
riparian plantings in the mitigation of diffuse  
agricultural nitrogen**

A thesis

submitted in fulfilment

of the requirements for the degree

of

**Masters of Science (Research) in Environmental Sciences**

at

**The University of Waikato**

by

**Olivia María Adamson**



THE UNIVERSITY OF  
**WAIKATO**  
*Te Whare Wānanga o Waikato*

2021

---

# Abstract

---

The disruption of nutrient cycles in agricultural settings, particularly pastoral farming, is responsible for up to 70% of the nitrogen (N) loads entering streams in New Zealand (NZ) — resulting in the widespread degradation of freshwater ecosystems. Riparian plantings are one strategy to mitigate the losses of N from land to water, removing N by denitrification and plant uptake. Some types of vegetation are more effective than others at intercepting N, due to their ecology, impact on soil quality and root exudates. Mānuka (*Leptospermum scoparium*) is a plant native to NZ, and the species could be a good candidate to mitigate N losses due its ability to tolerate high N loads and co-benefits such as farm diversification through apiculture or essential oil production. Previous work also suggests that *L. scoparium* could be a biological nitrification inhibitor (BNI), limiting nitrate (NO<sub>3</sub><sup>-</sup>) production in soil.

The aim of this study was to investigate the potential of mānuka to intercept and remove N in an experimental riparian buffer in Lake Waikare. Currently, the lake is eutrophic due to high nutrient and sediment inputs. Significant restoration efforts have been made in recent years, led by many hapuu around the lake. All are very passionate and determined to recover the mauri of this ecosystem and reconnect the people to the lake, as well as provide economic opportunities. The riparian band had been established for four years and was set up as a series of experimental plots with different vegetation types, including plots solely in mānuka, and grassed controls. The plots are on a Perch-Gley Ultic Soil underlain by a slowly permeable clay layer— the Hamilton Ash. Further, the study sought to explore the relationship between soil physical properties and N cycling, and to identify influences expected by the local hydrology— as there was evidence of perching and lateral flow in the site. A series of suction-cup lysimeters were installed and pore water was sampled seven times between May and July 2021. Samples were analysed for total N (TN), total Kjeldahl N (TKN), NO<sub>3</sub><sup>-</sup> and ammonium (NH<sub>4</sub><sup>+</sup>).

The results show that the riparian buffer is effectively removing N from subsurface flows, with TN declining from an average of 9.32 mg/l at 1 m into the buffer to 2.03 mg/l at 7 m. Although N concentrations were higher under mānuka, the total amount of

N extracted from those plots was 21% less due to high rainfall interception by the canopy (63.9%), limiting the transport of the solute. Indicators of soil physical quality, bulk density ( $\rho_b$ ) and macroporosity (MP), were investigated as potential causes for differences in the movement of N. There was a 17% improvement in  $\rho_b$ , and 38% improvement in MP relative to 2017 measurements, before the riparian band was established. No appreciable differences in soil physical properties were found between vegetation types at this early stage in the inception of the riparian buffer, although changes are likely to occur as the mānuka trees mature and the impact of roots is more developed. Shallow wells were deployed to monitor the dynamics of the water table, as there was evidence of winter perching. Dip wells demonstrated the existence of a short-lived perched water table, resulting in lateral flow, and likely impacting the degree of soil gleying in different parts of the riparian band. Water and contaminants are likely to move laterally over the Hamilton Ash towards the drain, supported by the accumulation of  $\text{NO}_3^-$  at the boundary with the clay layer.

Future work should be devoted to better quantifying  $\text{NO}_3^-$  leaching from the riparian plot by incorporating a drainage model, as well as further investigating the relative importance of N removal pathways at this site (i.e. denitrification and plant uptake)—including the potential for the lower part of the riparian band, closest to the drain, to become a hotspot for nitrous oxide ( $\text{N}_2\text{O}$ ) emissions. It would also be of value to revisit this study in 4-10 years to explore how the buffer impacts soil quality and N dynamics in the subsurface as it matures.

# Acknowledgements

---

If there is one thing that the last 18 months have reminded me it is that it really does “take a village”. I am very grateful to the many people who have helped me get to the finish line.

Firstly, I would like to thank my supervisors. Dr. Tanya O’Neill: thank you for your support, many proof-reads of my chapters, helping me get organised and our many coffee meetings by the lake, it’s been wonderful. Dr. María Jesus Gutierrez-Gines: thank you too for your support, encouragement and reassurance throughout this journey— it has been immensely valuable. Dr. Matthew Taylor: thank you for sharing your insight from your many years of experience in the field, and for helping me dig in those lysimeters— it was hard work!

I am also very grateful to ESR for their Masters Scholarship, and the University of Waikato for further financial support, which has lifted a weight off my shoulders and allowed me to focus on my research more fully. The Chemical analysis of soil pore water and the remote monitoring equipment was also funded by ESR's Native Ecosystems SSIF Fund.

An immense thank you to Matahuru whanau for allowing me to carry out this research on their whenua, as well as Ngaa Muka for further supporting this project. Thank you for welcoming me into the learning community, for sharing your knowledge and stories, and thank you for all the awesome mahi you are doing to restore Lake Waikare to health. Thank you specifically to Tawera Nikau, Glen Tupuhi and Liz Duggan for revising the section about Lake Waikare and making sure it is as accurate as possible.

Thank you to the many people who have provided technical help and advice, especially over the last few months: Kristin Bohm for all the help with R and graph-making. Thank you for your patience with my millions of questions— you have truly expedited the writing of the results chapter! Kevin Wang and Ackshat Bisht for help and advice in downloading and dealing with weather station data, and providing great aerial photos of

the plots. Another big thank you to David Campbell for lending me the dip wells, and for helping me calibrate the water table data.

Of course, a big thank you to my lovely friends Alice, Emily, Ally and Tsitsi. Thank you for the coffee breaks, the laughs, the tips and tricks and for answering any and all of my questions. Thank you for walking this road together.

To Adam and Valerie El-Gamel, for welcoming me into their home and into their family for much of my graduate journey. Thank you, thank you, thank you.

Robert Acksyn: thank you for being my biggest cheerleader and co-conspirer. You are a huge inspiration. Thank you for always uplifting me and challenging what is deemed “possible” every single day.

Clementina Benedetto: my long-time, and long-distance friend, who has encouraged and supported me every step of the way. I miss you!

To my family, for always supporting my academic (and many other!) adventures. It has been tough road to walk so far from home, and where I am today would not be possible without you.

And lastly to Ollie, my love: this thesis is as much yours as it is mine. From installing lysimeters, many field trips to collect samples in the rain, always being there to listen to what I was up to (even when it was a little tantrum) and supporting each and every one of my dreams and aspirations— while reminding me to play, and laugh, and have the most outrageous fun along the way. Thank you.

# Table of Contents

---

Abstract .....	i
Acknowledgements.....	iii
Table of Contents .....	v
List of Figures .....	ix
List of Tables.....	xii
<b>Chapter 1 Introduction.....</b>	<b>1</b>
1.1. Background .....	1
1.2. Objectives .....	4
1.3. Thesis outline.....	5
<b>Chapter 2 Literature Review.....</b>	<b>7</b>
2.1. Introduction.....	7
2.2. Planetary boundaries.....	7
2.3. The N cycle .....	9
2.3.1. Inputs.....	10
2.3.2. Internal cycling.....	11
2.3.3. Losses.....	11
2.3.4. Controls on N cycling .....	13
2.4. N in New Zealand .....	15
2.5. Movement of N from land to water .....	17
2.5.1. Contaminant transfer pathways.....	18
2.5.1.1. Surface transport .....	20
2.5.1.2. Shallow subsurface transport .....	20
2.5.1.3. Quantifying contaminant transfer pathways .....	22
2.5.1.3.1. Suction cup lysimeters .....	25
2.5.1.3.2. Use in studies .....	27
2.5.2. Concluding thoughts on contaminant transfer pathways .....	29
2.6. Mitigating N losses: riparian plantings .....	30
2.6.1. Potential economics of riparian planting.....	30
2.6.2. Effects of riparian planting on freshwater quality.....	31
2.6.2.1. Riparian plantings and N cycling.....	32

2.6.2.2.	Effect of buffer width on N cycling .....	32
2.6.2.3.	Effect of vegetation type and age on N cycling .....	33
2.6.2.4.	Native NZ vegetation: mānuka .....	36
2.6.3.	Concluding thoughts on riparian planting .....	38
2.7.	Soil quality .....	40
2.7.1.	Soil quality reporting and indicators .....	41
2.7.1.1.	Total carbon .....	43
2.7.1.2.	Total nitrogen and total mineralisable nitrogen .....	44
2.7.1.3.	Bulk density .....	45
2.7.1.4.	Macroporosity .....	45
2.7.2.	Concluding thoughts on soil quality .....	46
2.8.	Lake Waikare restoration project .....	47
2.8.1.	Location and physical characteristics .....	47
2.8.2.	History and land uses .....	49
2.8.3.	Lake Waikare learning community and vision maatauranga .....	52
2.8.3.1.	Scientific work at Lake Waikare .....	53
<b>Chapter 3</b>	<b>Methodology .....</b>	<b>56</b>
3.1.	Experimental plots .....	56
3.1.1.	Vegetation .....	57
3.1.2.	Soil setting .....	58
3.2.	Overview of the monitoring design .....	60
3.3.	Soil physical properties .....	64
3.4.	Hydrology .....	65
3.4.1.	Rainfall, soil moisture and electrical conductivity .....	65
3.4.2.	Water table monitoring .....	65
3.4.2.1.	Design .....	65
3.4.2.2.	Deployment .....	66
3.4.2.3.	Data collection .....	67
3.4.2.4.	Data analysis .....	69
3.5.	Nitrogen concentrations in pore water .....	69
3.5.1.	Design .....	69
3.5.2.	Deployment .....	71
3.5.3.	Sampling .....	73

3.5.4.Laboratory analysis .....	74
3.5.5.Data analysis .....	74
<b>Chapter 4 Results .....</b>	<b>75</b>
4.1. Soil physics .....	75
4.2. Hydrology: rainfall, water table, soil moisture and electrical conductivity .....	76
4.3. Nitrogen movement.....	82
4.3.1.Volume of pore water samples .....	82
4.3.2.Cumulative N extracted throughout sampling period .....	82
4.3.3.Factors determining N concentrations .....	83
4.3.3.1.Influence of distance from fence on N .....	84
4.3.3.2.Influence of depth on N concentrations .....	85
4.3.3.3.Influence of plot on N concentrations .....	86
4.3.3.4.Influence of vegetation on N concentrations .....	87
<b>Chapter 5 Discussion.....</b>	<b>89</b>
5.1. General setting .....	89
5.2. Soil physics .....	89
5.2.1. Soil physical properties after land use change: from dairy pasture to riparian buffer .....	90
5.2.2. Soil physical properties in the riparian plots: grass versus mānuka buffer ..	92
5.3. Nitrogen in soil solution.....	94
5.3.1. Effectiveness of the riparian buffer .....	94
5.3.2. Impact of depth on N concentrations .....	98
5.3.3. Impact of subsurface flows on N concentrations .....	98
5.3.4. Impact of vegetation type on N concentrations .....	100
5.3.5. Water fluxes under mānuka compared to control .....	102
5.3.6. Water fluxes and overall N losses .....	103
5.3.7. Impact of soil and hydrology on N concentrations .....	104
5.4. Limitations and further research .....	105
5.5. Summary of key points .....	108
<b>Chapter 6 Conclusion.....</b>	<b>109</b>
6.1. Introduction.....	109
6.2. Influence of vegetation type on soil physical properties.....	109
6.3. Subsurface water flows at the southern edge of Lake Waikare .....	109

6.4. Influence of riparian vegetation and soil quality on N fluxes .....	110
Appendices .....	112
Appendix 1: Volume of pore-water samples obtained .....	112
Appendix 2: Raw data of N concentrations .....	114
Appendix 3a: 3-dimensional view of TN concentrations.....	118
Appendix 3b: 3-dimensional view of TKN concentrations .....	119
Appendix 4a: Correlations between TN - TKN, and TN - NO <sub>3</sub> - .....	120
Appendix 4b: TKN and NO <sub>3</sub> - as percentages of TN .....	120
Appendix 4c: Composition of TN in pore water samples.....	121
Appendix 5: Cumulative TN extracted from each plot and vegetation type throughout the sampling period (mg).....	121
<b>Bibliography .....</b>	<b>122</b>

# List of Figures

---

Figure 2.1: The current status of variables for seven of the nine planetary boundaries (sourced from Steffen et al., 2015).....	8
Figure 2.2: The N cycle in terrestrial ecosystems, specifically in an agricultural setting (adapted from McLaren & Cameron, 1996).....	10
Figure 2.3: Relationship between the physiological controls of N dynamics and the large-scale environmental factors that mediate them (sourced from Schimel & Holland, 2005).....	14
Figure 2.4: Urea fertiliser applied by dominant farm type in New Zealand, 2002-2019 (sourced from StatsNZ, 2020).....	16
Figure 2.5: Partitioning of contaminant transfer pathways and associated lag times (adapted from Stenger, 2019).....	19
Figure 2.6: Flow chart for guidance on appropriate soil water sampling methods according to different needs and circumstances (sourced from Weihermuller, 2007).....	24
Figure 2.7: Schematic of a suction cup lysimeter (adapted from SDEC, n.d.) .....	26
Figure 2.8: <i>L. scoparium</i> tree (left), leaves (middle) and seed pods (right).....	37
Figure 2.9: Infographic summarising the seven soil quality indicators (sourced from MfE & StatsNZ, 2021).....	42
Figure 2.10: Map of Lake Waikare in the context of New Zealand's North Island (marked in red).....	48
Figure 2.11: (Above) Lake Waikare in 2018. (Below) Toxic algae blooms at Lake Waikare in December 2019 (sourced from O'Dwyer, 2020). .....	51
Figure 2.12: (top) aerial image of the location of the experimental riparian plots before planting, in 2016 (Waikato Regional Council, 2016) and (bottom) aerial image of the experimental riparian plots after 4 years of establishment, in July 2021. During this time a bridge was being built for easier access to Lake Waikare (University of Auckland, 2021).....	55
Figure 3.1: (A) Aerial photo of the experimental plots (July 2021), (B) aerial photo of Lake Waikare highlighting the location of the plots and (C) outline of the experimental plots and their vegetation (produced in GIS). .....	57
Figure 3.2: Soil profile to 40 cm in the paddock upslope of the experimental plots (under maize in the summer of 2020-2021).....	58
Figure 3.3: Sequence of soils from representative soil layers at the experimental riparian plots. (A) soil core showing typical Mangatawhiri clay loam topsoil from	

control plot C2, 0-16 cm depth. (B) ~30 cm depth in control plot C2. (C) Hamilton Ash clay encountered at 50-60 cm interval in mānuka plot M1. .60

Figure 3.4: Schematic of monitoring design deployed in this study, showing lysimeters, dip wells, soil cores, rain gauges and soil sensors in a mānuka plot (not to scale).....61

Figure 3.5: GIS map of the experimental riparian site picturing the boundaries of each experimental plot, lysimeters at different depths, dip wells, soil sensors and rain gauges.....62

Figure 3.6: Panoramic photograph of monitoring setup on plot C1, showing the location of the experimental plots relative to the lake, drain and paddock. For lysimeter label references see Appendix 1. ....63

Figure 3.7: Soil coring process. (A) digging the core into the soil with the help of a serrated knife. (B) the soil core, just removed from the ground. (C) wrapping the core for safe transport and storage. ....64

Figure 3.8: Dip wells.....66

Figure 3.9: Installation of dip well C. Pictured with dutch auger. ....67

Figure 3.10: Collecting water table measurements at plot C2 using the “bubbler” .....68

Figure 3.11: Extracting data from the LevelSCOUT data logger in dip well “C” .....68

Figure 3.12: Lysimeters used in this study (to sample 10-16.5 cm depth, 30-36.5 cm and 50-56.5 cm) .....70

Figure 3.13: Lysimeters being tested for air leaks and vacuum-holding capacity .....71

Figure 3.14: Lysimeter installation process (A) coating the instrument in a slurry of soil and water, (B) installed lysimeter, with most of the soil removed during excavation replaced around it, and (C) lysimeter covered by bucket for protection and ease of location.....72

Figure 3.15: (A) Sampling from a 10 cm depth lysimeter on plot C2 and (B) Samples back in the lab. ....74

Figure 4.1: Average bulk density ( $\rho_b$ ) and macroporosity (MP) at control and mānuka experimental plots, calculated from three different soil cores. ....76

Figure 4.2: A) Daily precipitation (mm) as measured by weather station next to the experimental riparian plots at Lake Waikare. Pore water sampling dates indicated by arrows. B) Daily rainfall expressed in mm water column. Note that during late June and early July, the weather station was offline due to technical issues, which is why some data is missing, which appears in the rain gauge graphs. ....77

Figure 4.3: Water table depth (cm) as measured manually and by a LevelSCOUT data logger at the experimental riparian plots in Lake Waikare.....	78
Figure 4.4: Measurements for volumetric water content (m <sup>3</sup> /m <sup>3</sup> ) from soil sensors deployed at the experimental plots in Lake Waikare (15 and 30 cm depth). Pore water sampling dates indicated by arrows. ....	80
Figure 4.5: Measurements for bulk electric conductivity (dS/m) from soil sensors deployed at the experimental plots at Lake Waikare (15 and 30 cm depth). Pore water sampling dates indicated by arrows. Note difference in y-axis scale between 15 and 30 cm depth. ....	81
Figure 4.6: Box plots displaying differences in TN, NO <sub>3</sub> <sup>-</sup> and TKN at different distances from the fence, including the mean value. Graphs display the median, 25th and 75th percentiles, and mean, minimum and maximum values. Letters indicate groups that are significantly different (p-value <0.05). Note different scales on y-axis for TKN. Note that statistical analysis were performed on log transformed data, not on the raw data shown in these graphs. ....	85
Figure 4.7: Box plots displaying differences in TN, NO <sub>3</sub> <sup>-</sup> , TKN and NH <sub>4</sub> <sup>+</sup> at different depths within the soil profile, including the mean value. Graphs display the median, 25th and 75th percentiles, and mean, minimum and maximum values. Letters indicate groups that are significantly different (p-value <0.05). Note different scales on y-axis for TKN and NH <sub>4</sub> <sup>+</sup> . Note that the statistical analysis were performed on log transformed data, not in the raw data shown in these graphs. ....	86
Figure 4.8: Box plots displaying TN, NO <sub>3</sub> <sup>-</sup> and TKN concentrations in each plot, including the mean value. Graphs display the median, 25th and 75th percentiles, and mean, minimum and maximum values. Letters indicate groups that are significantly different (p-value <0.05). Note that the scale on y-axis for TKN is one order of magnitude lower than for TN and NO <sub>3</sub> <sup>-</sup> . Note that the statistical analysis were performed on log transformed data, not in the raw data shown in these graphs. ....	87
Figure 4.9: Box plots displaying TN, NO <sub>3</sub> <sup>-</sup> and TKN concentrations in each vegetation type, including the mean value. Graphs display the median, 25th and 75th percentiles, and mean, minimum and maximum values. Letters indicate groups that are significantly different (p-value <0.05). Note that the scale on y-axis for TKN is one order of magnitude lower than for TN and NO <sub>3</sub> <sup>-</sup> . Note that the statistical analysis were performed on log transformed data, not in the raw data shown in these graphs. ....	88
Figure 5.1: Summary of soil physical properties at the experimental riparian plots at Lake Waikare. Conditional highlighting shows whether the values are within (green) or outside (red) target ranges. Targets from Sparling et al. (2008) were used for pb, while those from Mackay et al. (2013) were used for MP (revised values). ....	89

## List of Tables

---

Table 4.1: Bulk density ( $\rho_b$ ) and macroporosity (MP) results for three different soil cores at each experimental plot, compared to samples from 2017 (pasture), before the riparian band was planted. Conditional highlighting shows whether the values are within (green) or outside (red) target ranges. Targets from Sparling et al. (2008) were used for $\rho_b$ , while those from Mackay et al. (2013) were used for MP (revised values). .....	75
Table 4.2: Cumulative TN extracted from each plot and each vegetation type (mg/l N) throughout the sampling period. ....	83
Table 4.3: ANOVA results of significant interactions between variables and N species as indicated by p-values. *** $p < 0.001$ , ** $p < 0.01$ , * $p < 0.05$ , ° $p < 0.1$ .....	84

# Chapter 1

## Introduction

---

### 1.1. Background

The health of freshwater ecosystems worldwide have been compromised over recent decades (Steffen et al., 2015). In New Zealand (NZ), it is estimated that 46% of lakes larger than 1 hectare (1,758 lakes) are in poor or very poor ecological health (MfE, 2020). The intensification of agriculture is partly responsible for this decline, particularly in terms of the disruption of biogeochemical cycles, such as that of nitrogen (N), and associated pollution (MfE, 2020). From a nutrient point of view (e.g. N), sustainable food production can be defined as a system with no nutrient leakage into local groundwater and rivers (Willett et al., 2019). However, according to Stats NZ (2020), approximately 7% of all N fertiliser applied in NZ leaches from the soil, ending up in freshwater reservoirs. Howard-Williams et al. (2010) estimated that diffuse sources of N & phosphorus (P) from modified landscapes, predominantly pastures, accounted for 75% of the total flux delivered to sea.

The first step in addressing and restoring freshwater quality is having the knowledge of when, where, how and in what quantities contaminants are moving— which allows to prioritise action in ‘hotspots and hot-moments’ and to develop assessment methods that recognise lag periods between mitigation actions and improvements in ecosystem health. In the longer term, this understanding can be utilised to reorganise land use in a way that reflects the natural capacity of a system to absorb and attenuate the effects of activity (Srinivasan et al., 2020). Additionally, mitigation strategies can be utilised to reduce the impacts of agricultural systems on freshwater (e.g. stream fencing, nitrification inhibitors, precision agriculture, cover crops) (McDowell et al., 2013).

Riparian planting is one of the potential strategies to mitigate the adverse effects of agriculture on the landscape and freshwater which has garnered interest in NZ and worldwide (Franklin et al., 2015). In the appropriate setting, this mitigation strategy has the potential to filter overland flow and reduce sedimentation, minimise peak flood flows, stabilise river banks (Marden et al., 2005), exclude stock, regulate stream

temperature (Parkyn et al., 2003), provide organic matter as an in-stream food source, provide habitat for fish and aquatic invertebrates (Collier et al., 1995; Parkyn et al., 2003; Collins et al., 2013), accumulate soil carbon (Trotter et al., 2005) and create new sources of income for farmers (Daigneault et al., 2017). Particular attention has been given to the potential of riparian plantings to manage nutrients (Fennessy & Cronk, 1997; Mayer et al., 2007; Franklin et al., 2015; Lyu et al., 2021). For example, a meta-analysis of 89 individual riparian buffers from 45 published studies concluded that the effectiveness of riparian buffers at controlling nutrient export varies widely but they are capable of removing large quantities of nitrate ( $\text{NO}_3^-$ ) from water flowing through riparian zones (Mayer et al., 2007).

Riparian restoration has been occurring in NZ for over 30 years (Collins et al., 2013) and the positive effect on stream quality improvements have been reported by several studies (Collier et al., 1995; Parkyn et al., 2003). The vegetation and soils in the riparian zone are largely responsible for the attenuation of pollutants, and understanding their functioning can help to maximise their potential to mitigate pollution and remediate freshwater ecosystems. However, most studies on riparian plantings have been based on the analysis of stream, river and lake water quality that resulted from collective impacts of various riparian planting strips in a catchment. Less information is available, and less attention has been given to how each riparian strip itself would contribute to the mitigation of contaminant transfers.

Plant traits play a large role in mediating the N cycle (van Groenigen et al., 2015), largely because they modify the physical and chemical characteristics of soil (Franklin et al., 2019) — and due to their role in supporting microbial communities with nutrients and energy (van Groenigen et al., 2015; Lyu et al., 2021). In contrast, some plants such as myrtaceae species, may antagonise pathogens in the soil (Prosser et al., 2016; Halford et al., 2021).

Although the literature addressing optimal buffer widths (Vidon & Hill, 2004; Mayer et al., 2007) and positions (Zhou et al., 2010) is abundant, there is a lack of science-based advice when recommending the best plant species for the management of N (Franklin et al., 2019). When native plants are used, riparian plantings have been proposed as an opportunity to increase on-farm biodiversity (providing and improving habitat for bird life and aquatic species) (Dybala et al., 2019; Kelly, 2019; Lind et al., 2019), which has

been in a steady decline in NZ (MfE & StatsNZ, 2018). Van Groningen et al. (2015) highlighted that there is a lack of understanding in the relationships between plant traits, soil characteristics and specific N cycling variables, particularly under field conditions. Some of the critical factors in regulating microbial communities, and thus N cycling include macroporosity (MP), bulk density ( $\rho_b$ ), total carbon (TC) and total N (TN) (Lyu et al., 2021).

The dominant local hydrological processes are also critical in determining whether a riparian buffer will be effective— including the dominant contaminant transfer pathways (e.g. surface versus subsurface flow, and types and direction of subsurface flow) (Singh & Stenger, 2018; Srinivasan et al., 2020).

Mānuka (*Leptospermum scoparium*) is a wide-ranging species belonging to the myrtaceae family, native to NZ. A study by Esperchuetz et al. (2017b) found that, although mānuka is adapted to live in low-fertility environments, it can thrive in high nutrient environments. In addition, leaf extracts of this species have been found to decrease  $\text{NO}_3^-$  production relative to *Lolium perenne* (perennial ryegrass) by 60% (Downward, 2013). These results indicate that mānuka could potentially be a Biological Nitrification Inhibitor species (BNI) — limiting the levels of  $\text{NO}_3^-$  in soil (N remaining as ammonium,  $\text{NH}_4^+$ ) and thus reducing N losses via leaching and denitrification (given that  $\text{NH}_4^+$  is held more readily in soil than  $\text{NO}_3^-$ ). These results are supported by the findings of Esperchuetz et al. (2017a) who found that, after a series of urea fertiliser applications, mānuka significantly reduced  $\text{NO}_3^-$  leaching compared to pine (2 kg ha<sup>-1</sup> versus 53 kg ha<sup>-1</sup>). Moreover, Esperchuetz et al. (2017a) and Halford et al. (2021) indeed found differences in the speciation of N under mānuka, having less  $\text{NO}_3^-$  and more  $\text{NH}_4^+$  compared to *Pinus radiata* and *L. perenne*.

Some co-benefits of utilising mānuka on-farm include potential economic returns (Daigneault et al., 2017) from essential oil production which has been shown to increase in high nutrient environments (Seyedalikhani et al., 2019), as well as opportunities for apiculture. Moreover, studies have found that mānuka may also diminish the proliferation of pathogens in soil (Prosser et al., 2016; Gutierrez-Gines et al., 2021).

The aforementioned research indicates that mānuka has high potential for mitigating the loss of N in agricultural systems, specifically in riparian plantings. However, this potential remains to be investigated outside of the laboratory, including the quantification of NO<sub>3</sub><sup>-</sup> leaching (Halford et al., 2021). This information will fill a knowledge gap in the understanding of the role that mānuka may play in the mitigation of diffuse agricultural pollution.

Lake Waikare, located in the Lower Waikato-Waipā Region, suffers from a dramatic loss of biodiversity, toxic algal blooms and loss of cultural ecosystem services over the last decades. The mana whenua at Lake Waikare (customary authority exercised by an iwi or hapu in an identified area) are determined to restore the mauri (life-force) of the lake. In this spirit, Matahuru Marae, the landowners on the south margins of the lake, have transformed a plot of farmland for scientists to perform research on the role of mānuka-dominated ecosystems for water quality improvements and ecological restoration. The riparian band was set up as a series of experimental plots (a grass control, mānuka and mixed native species) which was leveraged in this study to investigate the movement of N under different vegetation types.

The soil underlying the experimental plots is the Mangatawhiri clay loam (an Ultic Soil in the New Zealand Soil Classification), comprised of a rich and well-structured A horizon, and underlain by the widespread and less permeable weathered Hamilton Ash Formation. This soil setting suggests that subsurface lateral flow might be an important pathway for contaminants moving towards the lake, which might influence N dynamics. The hypothesis remains to be tested.

## **1.2. Objectives**

In summary, different plants impact soil quality in different ways, affecting, in turn, the soil's ability to attenuate pollutants such as N (Franklin et al., 2015; Franklin et al., 2019). Previous work suggests that mānuka has a large potential to impact N dynamics in soil and thus mitigate the impacts of diffuse agricultural pollution (Downward, 2013; Esperschuetz et al., 2017a; Halford et al., 2021). This thesis was a continuation of the aforementioned research and seeks to contribute to the understanding of riparian

buffers, NO<sub>3</sub><sup>-</sup> leaching, the ecology of *L. scoparium* and local hydrological processes in the Lake Waikare region.

In short, the question this study sought to answer was:

*Do mānuka riparian plantings impact N dynamics in the subsurface, compared with unplanted riparian margins (control)? How? What are the potential reasons?*

The specific objectives of this study were to:

1. Investigate how vegetation type (mānuka and grass) influences the physical properties of soil (using soil quality indicators, bulk density and macroporosity),
2. Establish an understanding of subsurface water flows at the southern margin of Lake Waikare (using dip wells) focusing on the expected sub-surface lateral flow, and
3. Explore how the interaction between vegetation type and soil quality may influence N fluxes.

The overarching aim of this research is to provide stakeholders (including iwi, researchers and government bodies) with a better understanding of whether and how native plants can play a role in mitigating diffuse nutrient pollution. The aspirational goal of this research is that riparian plantings can be designed more effectively to cater for local hydrological conditions and desired outcomes while providing co-benefits such as carbon sequestration, return of biodiversity and farm diversification.

### **1.3. Thesis outline**

Chapter two reviews the literature on contaminant transfer pathways (specifically N), riparian plantings, the role of native NZ species in attenuating diffuse pollution and an overview of soil quality indicators.

Chapter three introduces the research site, Lake Waikare, and provides a detailed methodology used for collecting and processing soil and water samples, as well as water table monitoring.

Chapter four presents the main findings of this research.

Chapter five provides an evaluation and discussion of the results presented in chapter four, including a discussion on the validity of the findings and comparisons with existing literature. The study's limitations and recommendations for further research are also included here.

Chapter six is a synthesis of the main findings of this research and assesses the completion of objectives set out in chapter one.

# Chapter 2

## Literature Review

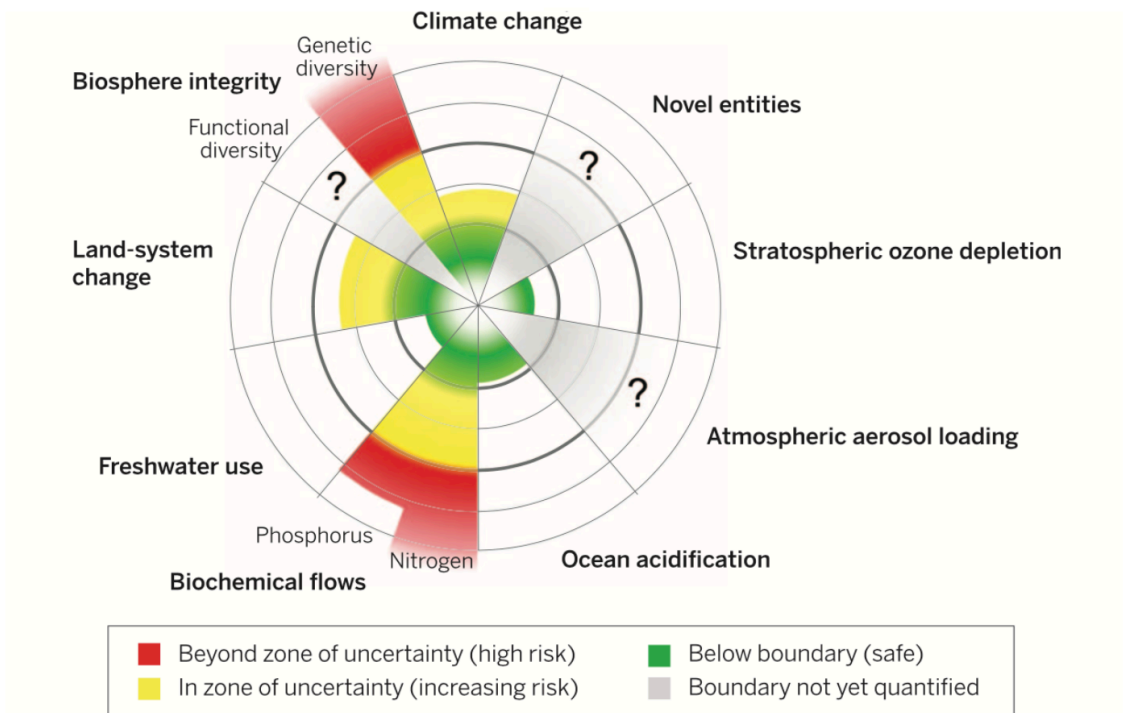
---

### 2.1. Introduction

The following chapter will review the processes and frameworks that underpin the transfer of N from land to water, as well as a review of methods for measuring  $\text{NO}_3^-$  leaching and its horizontal movement through the subsurface environment. The literature relating to the use of riparian plantings to mitigate agricultural nutrients will then be presented, as well as a review of soil quality and its indicators. This is followed by an overview of Lake Waikare: its characteristics, value and previous work that contributes to the present research project.

### 2.2. Planetary boundaries

The Planetary Boundaries Framework (Figure 2.1) attempts to delineate a “safe operating space” for humanity, where human activities are below the threshold of earth system destabilisation. The framework combines the scientific understanding of the earth system with the precautionary principle to determine where the boundaries of safe human activity lie (Steffen et al., 2015). Steffen et al. (2015) have identified the boundary for the biogeochemical flow of N between 62–82 Tg N yr<sup>-1</sup>. This limit has been widely exceeded, with the magnitude of the flux currently estimated at ~150 Tg N yr<sup>-1</sup>. It is estimated that human activities have increased the rate of reactive N fixation from 15 Tg N yr<sup>-1</sup> in 1860 to 165 Tg N yr<sup>-1</sup> in 2000 (Galloway et al., 2003). The transgression of this boundary is due to the burning of fossil fuels and the intentional biological fixation of N for synthesising agricultural fertilisers (the Haber–Bosch process) (Galloway et al., 2003).



**Figure 2.1: The current status of variables for seven of the nine planetary boundaries (sourced from Steffen et al., 2015)**

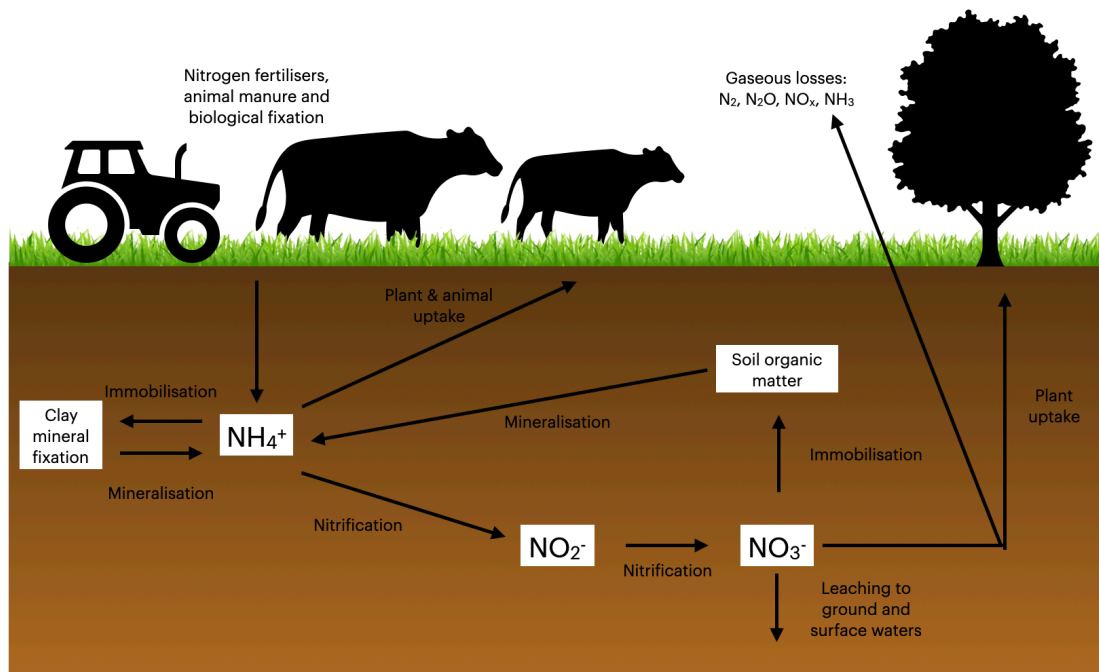
The Haber-Bosch process was created in 1910 to industrially convert dinitrogen gas (N<sub>2</sub>) from the atmosphere into ammonia (NH<sub>4</sub><sup>+</sup>), a reactive form of N, later used in fertilisers. This process removed the main constraint to global food production, which, in combination with advances in breeding and pesticides, allowed crop yields to increase dramatically. This has been a great scientific feat that provided great health and welfare benefits to society. However, agriculture today is almost completely dependent on synthetic fertilisers: the Food and Agriculture Organisation (FAO) estimates that 37.4% of the Earth’s land surface is used for growing crops and grazing animals (FAOSTAT). Although this is a success story in many ways, the disruption of the natural N cycle has also had a variety of adverse effects on the environment, including the widespread pollution of freshwater resources, the degradation of aquatic habitats and increasing greenhouse gas concentrations in the atmosphere. From a nutrient point of view, sustainable food production can be defined as a system with no nutrient leakage into local groundwater and rivers (Willett et al., 2019). Understanding the interphases and biogeochemical fluxes between land and freshwater is thus vital in understanding how to maintain and improve overall ecosystem health— particularly in agricultural landscapes.

### 2.3. The N cycle

N is an essential element of life. N is largely unavailable to organisms, although it is abundant in both the earth's crust ( $18 \times 10^{-18}$  units) and in the atmosphere (72% of the gas in the atmosphere, or  $3.8 \times 10^{15}$  tonnes of  $N_2$ ) (McLaren & Cameron, 1996). Molecular N is unreactive due to its strong triple bond and must be converted into reactive forms in order to be utilised. The inorganic reactive forms of N include: ammonia ( $NH_3$ ) and ammonium ( $NH_4^+$ ), nitrogen oxide ( $NO_x$ ), nitric acid ( $HNO_3$ ), nitrous oxide ( $N_2O$ ), nitrite ( $NO_2^-$ ) and nitrate ( $NO_3^-$ ). N is also a key constituent of organic compounds such as urea, amines, proteins, and nucleic acids. In excess, reactive N can have a number of adverse effects on the atmosphere, biosphere and hydrosphere as well as human health (Galloway et al., 2003). Converting reactive N back into  $N_2$  is the only way to terminate any adverse effects onto the receiving environment (Galloway et al., 2003).

According to Brady & Weil (2008), the A horizon of soils usually contains between 0.02-0.5% N (around 0.15% in cultivated soils) amounting to 3.5 Mg of N, as well as another 1-2 Mg in the litter layer of forested soils. Soils may also contain between 10 to 20 times as much N as does the standing vegetation, for both forested and cultivated areas.

N is fixed by soil microorganisms, by lightning, or by industrial manufacturing processes, to be transformed into reactive forms. In soils, N must be converted from its unavailable soil organic forms to available inorganic forms,  $NH_4^+$  and  $NO_3^-$ , in order to be utilised by plants. The transformation of N between its different forms, as well as its transfer between the plants-soil-atmosphere continuum, is referred to as the N cycle (Figure 2.2).



**Figure 2.2: The N cycle in terrestrial ecosystems, specifically in an agricultural setting (adapted from McLaren & Cameron, 1996)**

The following sections briefly review the transfer pathways of N between systems, its internal cycling, storage sites and losses in the context of an agricultural setting.

### 2.3.1. Inputs

N may enter an agricultural landscape in the form of fertilisers, by biological fixation and by deposition of livestock excrements or dead plant material.

There are several types of N fertilisers including inorganic and organic options. In NZ, the most common inorganic forms are ammonium sulphate, ammonium nitrate and di- or mono-ammonium phosphate. Organic fertilisers are generally in the form of urea, liquid N or ‘blood and bone’— the latter most common for garden plants rather than commercial crops (McLaren & Cameron, 1996). In areas that have not been fertilised, inorganic-N usually accounts for less than 1-2% of total N (TN) (Brady & Weil, 2008).

In the field, atmospheric N is fixed by bacteria (namely rhizobia) which possess specialised enzymes that are able to break the strong triple bond in  $\text{N}_2$ . Legumes, such as clover, chickpeas and gorse are common examples of plants that host N-fixing bacteria. Animal wastes also add N to the soil organic matter pool (SOM), where N gets rapidly mineralised into  $\text{NH}_4^+$ . Worldwide, the size of these fluxes has been

estimated by Galloway et al. (2003) at around 120 Tg N from fertilisers and biological fixation, and an additional 50 Tg N from already existent reactive N (crop residues and manure). To a lesser extent, atmospheric deposition may also input N into a system, and has a number of adverse effects in soil and water ecosystems. The burning of fossil fuels results in the transformation of  $N_2$  gas into  $NO_x$ , which can, through a series of reactions, increase the acidity of rainwater, leading to eutrophication. Although the extent of N inputs through atmospheric deposition is negligible in NZ, European countries have had issues in managing  $NO_x$  and associated acid rain in the past (Menz & Seip, 2004).

### **2.3.2. Internal cycling**

Organic N comprises the bulk of soil N (95-99%) with the remainder being in its inorganic forms (Brady & Weil, 2008). Once in the soil system, organic N has two possible pathways: (1) it may be immobilised into microbial biomass or mineral surfaces or (2) become mineralised to form  $NH_4^+$ . Further,  $NH_4^+$  might become adsorbed into clay minerals due to their overwhelmingly negative charge, or it might undergo nitrification. Nitrification refers to the bacterial oxidation of  $NH_4^+$  to form  $NO_2^-$  and subsequently  $NO_3^-$ . For nitrification, the amount of  $NH_4^+$  that is converted to  $NO_3^-$  is directly related to oxygen availability (rate of nitrification decreases with less oxygen) but the proportion that is lost as N gases is larger (increases with more oxygen) (Schimel & Holland, 2005). Both gases usually represent about 1% of the  $NH_4^+$  nitrified (Schimel & Holland, 2005).

### **2.3.3. Losses**

There are five pathways for the loss of N from a system: denitrification, leaching, volatilisation, erosion and product removal.

Denitrification occurs in poorly drained or waterlogged soils where facultative anaerobic bacteria use  $NO_3^-$  as an electron acceptor in place of  $O_2$  to carry out their metabolic reactions. By this mechanism,  $NO_3^-$  is converted to NO,  $N_2O$  and subsequently  $N_2$ . There are a number of factors that impact the rate at which

denitrification proceeds: the availability of organic carbon (C), pH (very slow under acidic conditions) and temperature (very slow <10 °C) (McLaren & Cameron, 1996). In soils with a low C:N ratio, denitrifying bacteria become “wasteful” and are likely to produce N<sub>2</sub>O as the dominant product of denitrification. However, when N supply is more limited, microbes are more likely to use it to its full potential and reduce it all the way to N<sub>2</sub> gas. A meta-analysis of N<sub>2</sub>O emissions from European agriculture illustrates the magnitude of fluxes: under dry conditions (~500 mm) and with a 450 kg ha<sup>-1</sup> input of N fertiliser, N<sub>2</sub>O emissions remained below 3 kg N<sub>2</sub>O-N ha<sup>-1</sup>. With more rain (~1500 mm) emissions triple to 10 kg N<sub>2</sub>O-N ha<sup>-1</sup> (Rees et al., 2013).

N<sub>2</sub>O is a powerful greenhouse gas, with a global warming potential of 298 CO<sub>2</sub>-equivalents, and around 65% of the N<sub>2</sub>O released into the atmosphere every year comes from soil (Schimel & Holland, 2005). The fluxes of gases from land to atmosphere are large at the global scale, however, they may seem small when looked at the scale of a single ecosystem. Although an ecosystem may mineralise 100kg N ha<sup>-1</sup> yr<sup>-1</sup>, it might only lose 1 kg N<sub>2</sub>O-N ha<sup>-1</sup> yr<sup>-1</sup> (Schimel & Holland, 2005). It can also be difficult to model the fluxes of trace gases since they are highly variable across time and space, occurring in “hot-spots and hot-moments” (Galloway et al., 2003; McClain et al., 2003; Schimel & Holland, 2005; Bernhardt et al., 2017).

Leaching accounts for a larger loss of NO<sub>3</sub><sup>-</sup> relative to denitrification in NZ. Given that SOM has a largely negative charge, there are relatively few sites for NO<sub>3</sub><sup>-</sup> to be stored in the soil. NO<sub>3</sub><sup>-</sup> is also readily dissolved in water and is mainly transported in soil solution in the subsurface environment. Such characteristics make NO<sub>3</sub><sup>-</sup> very mobile and vulnerable to being leached out of the system and into ground and surface waters. Over 80% of excess N in soil may be lost through leaching, depending on the agricultural system (Billen et al., 2013). Soil quality is especially important in mediating leaching under high rainfall conditions, whilst during dry periods, soil C and N stores (potential indicators of soil microbial activity) mediate NO<sub>3</sub><sup>-</sup> leaching (i.e. the main removal mechanism is denitrification) (Neilen et al., 2017). In NZ most leaching losses occur between May and August, when the winter rains bring soils up to, or near field capacity and there is an abundance of NO<sub>3</sub><sup>-</sup> in soil solution due to slower plant growth at this time of the year (McLaren & Cameron, 1996). The estimated losses by NO<sub>3</sub><sup>-</sup> leaching in the Waikato, Manawatū-Wanganui and Canterbury regions totals 75 million kg NO<sub>3</sub><sup>-</sup>-N

$\text{y}^{-1}$  ( $> 40 \text{ kg N ha}^{-1} \text{ y}^{-1}$ ) (Dymond et al., 2013). Globally, Billen et al. (2013) estimate that 43% of the world's continental area is the destination for 84% of anthropogenic N losses. These hotspots are located in Europe, the Americas, India, China and NZ (Billen et al., 2013).

Volatilisation refers to the reversible reaction that transforms  $\text{NH}_4^+$  into  $\text{NH}_3$  gas. This reaction occurs as a result of the degradation of organic materials and fertilisers. As shown by equation 1, the main control of this reaction is pH, with the reaction proceeding in the forward direction at high pH levels. Given that clay and humus both adsorb  $\text{NH}_3$  gas, losses via volatilisation are largest when these soil components are scarce, or when  $\text{NH}_3$  is not in contact with them. Hence, one mitigation strategy to diminish the loss of N by volatilisation is incorporating crop residues, fertilisers and manures into the topsoil instead of leaving them in the soil surface. This mitigation strategy may reduce volatilisation losses by 25-75% (Brady & Weil, 2008).



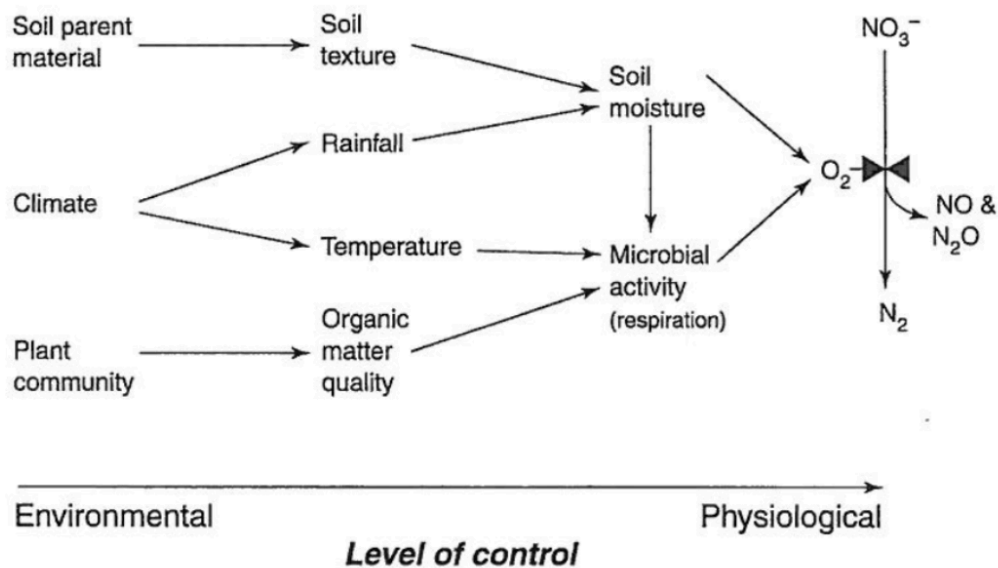
N is also removed from a system through harvesting crops and the export of animal products such as milk and meat. Finally, soil erosion is another process through which N may be lost from a system.

Globally, of the 170 Tg N added to agricultural landscapes, 33 and 16 Tg are consumed by animals and humans respectively while the remainder (121 Tg) is either reintegrated to the system as crop residues or manure, or is lost to air and water pathways. As a general rule, the more N is added to agricultural landscapes, the more N is transferred to the atmosphere and hydrosphere (Galloway et al., 2003).

#### **2.3.4. Controls on N cycling**

N dynamics, both within soil and in the forms released as gases, are controlled by a range of factors that operate at different scales: from soil characteristics, climate and vegetation, to microbial activity. Moreover, since N cycling occurs at microbial, plot, ecosystem and global levels (Figure 2.3), it is important (and challenging) to integrate

the understanding in all these dimensions (Schimel & Holland, 2005). One useful way to incorporate these scales is the framework of proximal and distal controls. For example, proximate controls are those that directly regulate the processes that influence the production of a GHG. These include excess N in soil, soil moisture, temperature, organic carbon (OC), and pH (McLaren & Cameron, 1996). In contrast, distal controls are those that indirectly influence proximate controls. For example, irrigation and rainfall determine moisture levels in soil, while biological fixation and fertilisation type and intensity determine the levels of N. While land-managers cannot control soil moisture or biological fixation directly, the management practices implemented impact these indirectly and allow them to manage their GHG footprint. The mitigation measures applied must thus seek to manipulate the distal controls that will affect the proximate controls of GHG production (see section 2.6.2.1).



**Figure 2.3: Relationship between the physiological controls of N dynamics and the large-scale environmental factors that mediate them (sourced from Schimel & Holland, 2005).**

The phenomenon of  $\text{NO}_3^-$  leaching also illustrates this idea well. Although leaching occurs due to the chemical properties of  $\text{NO}_3^-$ , the physical properties of soil can facilitate or diminish losses. While water (and thus  $\text{NO}_3^-$ ) would readily drain out of a sandy soil, a clayey soil would retain more water, and provide more sites for  $\text{NO}_3^-$  to be adsorbed onto (Brady & Weil, 2008). Soil texture also impacts the levels of  $\text{NO}_3^-$  leaching since it dictates how much C can be held in the soil. Clayey and silty soils release more organic C relative to sandy soils and have higher rates of N immobilisation

(Neilen et al., 2017). On another level, the amount of leaching is also closely associated with C stores in the soil, which serves to fuel microbes: when there is not enough soil organic carbon (SOC) for microbes to process N, N accumulation results, which is subsequently leached (Neilen et al., 2017).

Land use is another major factor in determining leaching losses due to its impact on the level of N inputs and soil physics. N inputs in agricultural systems often overwhelm the assimilative capacity of vegetation. This often results in nitrifying organisms producing polluting forms of N, such as  $\text{NO}_3^-$  (Villegas et al., 2020). Urine patches, for example, are a “hotspot” for N losses due to the concentration of  $\text{NO}_3^-$  in a small area, and in excess to what the system can assimilate or attenuate (McClain et al., 2003; Bernhardt et al., 2017; Stenger et al., 2019). Mineralisation (and thus the amount of  $\text{NO}_3^-$  in soil solution) can also be increased by land management practices such as ploughing (McLaren & Cameron, 1996), which increase  $\text{O}_2$  in the soil, breaks up organic matter and makes N caught in plant tissues and animal manure more bioavailable (Uchida & Akiyama, 2013).

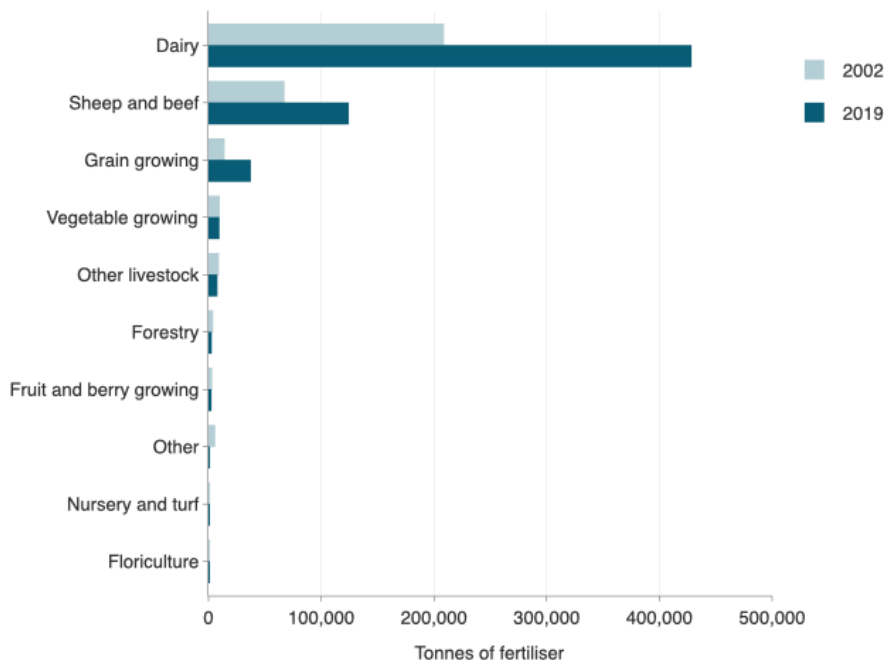
## **2.4. N in New Zealand**

During the past 200 years, since European settlement, NZ landscapes have changed dramatically, shifting from native forest to agricultural land. Today, only 49% of the land area is covered by indigenous forest, tussock grassland, scrub and shrub-land. The remainder is comprised of modified ecosystems, namely exotic forests and pastures, croplands, and urban areas (MfE & StatsNZ, 2021). Between 2002 and 2016 only, there has been a 42% increase in the land area occupied by dairy farming, with great environmental consequences (MfE & StatsNZ, 2018). MfE (2018) estimates that 192 million tonnes of soil are lost every year from erosion, 44% of which is due to pastoral farming. Out of the seven soil quality indicators (section 2.7), two raise reason for concern: low macroporosity (MP) and high levels of phosphorus (P). Not only is the land resource becoming degraded, it is also becoming smaller: there has been urban expansion onto 29% of class 1 and 2 lands (Land Use Capability), which account for just 5% NZ land.

In the past 15 years, there has also been an intensification of current uses, such as higher stocking rates— particularly in dairy farming. These changes in land use have resulted

in the loss of indigenous biodiversity, higher emissions of greenhouse gases and the degradation of freshwater quality. The decline in the area of land under native vegetation has resulted in the decimation of habitats: 82% of native NZ birds, bats, reptiles, and frogs are classified as threatened or at risk of extinction (MfE & StatsNZ, 2021).

NZ’s export economy highly depends on land, with primary-production (agriculture, horticulture and forestry) and tourism comprising 3.7% and 5.7% of the national gross domestic product respectively. In 2016 these figures equated to \$35.4 billion for agriculture (half of the country’s total export earnings), while tourism expenditure in NZ was \$14.7 billion (MfE & StatsNZ, 2018). NZ soils have a naturally low fertility, though sufficient to support native vegetation. Current levels of agricultural production are possible due to improvements in breeding and pesticide use, as well as the use of N fertilisers, which has grown 627% between 1990 and 2015 (StatsNZ, 2020). The overwhelming majority of the fertiliser applied nation-wide is used in dairy and sheep & beef farms (427,325 and 123,292 tonnes respectively) (Figure 2.4) (StatsNZ, 2020).



**Figure 2.4: Urea fertiliser applied by dominant farm type in New Zealand, 2002-2019 (sourced from StatsNZ, 2020).**

According to Stats NZ (2020), approximately 7% of all N applied in NZ leaches from the soil, ending up in freshwater reservoirs. Moreover, Howard-Williams et al. (2010)

estimated that diffuse sources of N and P from modified landscapes, predominantly pastures, accounted for 75% of the total flux delivered to sea (while the remaining 25% is natural). In a modelling exercise, Elliot et al. (2005) estimated that only 3.2% of the total N load entering streams comes from point sources, and that 70% of the N loads entering streams in NZ comes from diffuse pastoral farming (similar to the figure proposed by Howard-Williams, 2010). There are inherent challenges in controlling diffuse sources of pollution, and in measuring the inputs versus outputs while calculating the attenuation capacity of a system. For instance, 45% of the N load arrives at the coast, while the rest (55%) is attenuated throughout its journey (Elliott et al., 2005). However, the extent of attenuation largely differs across and between landscapes. The quality and characteristics of soil play a large role in determining the attenuation capacity of a soil ( section 2.7).

Best management farming practices for the protection of freshwater resources include stock exclusion from streams and rivers, nutrient budgeting, timing of irrigation and riparian planting (Fennessy & Cronk, 1997; Parkyn et al., 2003; Franklin et al., 2015). Despite many of these measures being adopted, water quality in NZ continues to decline (MfE, 2020). According to the Ministry for the Environment (MfE), most rivers in urban, farming and forestry areas are polluted (MfE, 2020). The quality of freshwater in NZ has come to the forefront of public debate in recent years, with 82% of New Zealanders reporting concern about the issue (Cosgrove, 2019).

## **2.5. Movement of N from land to water**

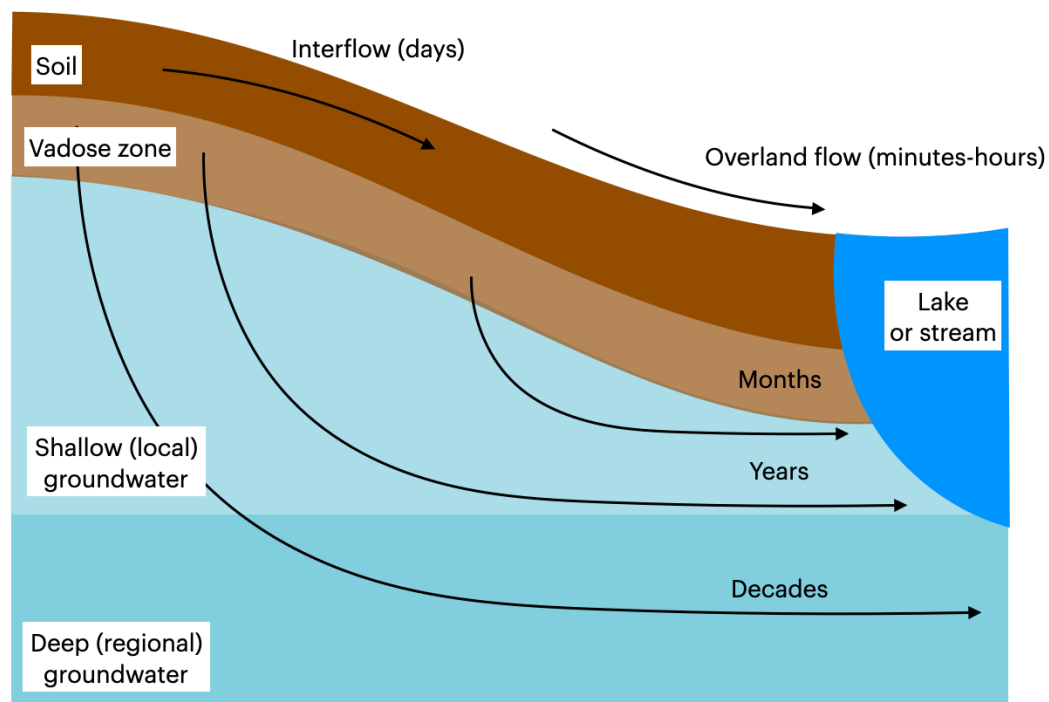
The intensification of agriculture impacts surface waters by increasing nutrient leaching and overland flow carrying sediments and pathogens, which leads to water eutrophication. Eutrophication refers to the process through which water bodies become enriched with sediments and nutrients. This decreases light penetration and utilisation for macrophytes and invertebrates, increases turbidity and temperature, and leads to states of anoxia. The excess of nutrients can also favour toxic algal blooms. Eutrophication naturally occurs in the order of hundreds of years but has been accelerated dramatically by land use changes from natural to managed ecosystems (Brady & Weil, 2008). Knowledge of the dominant hydrological processes in a

catchment is an important first step in determining adequate management practices for best water quality outcomes. Soil health is a vital component to consider. Soils that are in good health have good structure, host a diverse microbiome and are less likely to undergo extreme levels of erosion under normal conditions, or leach excessive nutrients. As contaminants travel from source to receptor, they undergo a variety of transformations, dependent on their environment. Drawing links between the sources of contaminants, soils, vadose zones and groundwater — and their associated attenuation capacities — is thus the first step in managing land-based activities and improving environmental outcomes, including freshwater quality and habitats (Tanner et al., 2017; Srinivasan et al., 2020).

The first step in understanding the transfer of contaminants, such as N, between land and water, is examining the characteristics of each environment. According to Barkle et al. (2014b), “the process of determining the maximum land-use intensity commensurate with maintaining freshwater quality targets suffers from paucity of reliable data on the fate of nutrients along the vadose zone–groundwater pathways between the land and the receiving surface water body”.

### **2.5.1. Contaminant transfer pathways**

Where, when, and to what extent a land use or land management change affects freshwater quality depends on the physical and biogeochemical characteristics of the pathways that link land (source) and freshwater (receptor) (Stenger et al., 2016). These pathways can be divided into surface flow (runoff or overland flow, plus artificial drainage) and subsurface flows, which include water moving through the rooting zone (interflow), shallow groundwater flow (local scale) and deep groundwater flow (regional scale) (Singh & Stenger, 2018). Contaminants may move through one or more of these pathways, which have different characteristics and residence times (Figure 2.5).



**Figure 2.5: Partitioning of contaminant transfer pathways and associated lag times (adapted from Stenger, 2019).**

The factors that influence the partitioning of precipitation and irrigation water throughout the different pathways are: soils, climate, topography, geology, vegetation and land use and management (Singh & Stenger, 2018). Surface and near surface transport pathways are more important during and immediately after a rain event (Barkle et al., 2014b). These pathways differ in their location within a catchment, the nutrient loads they carry, their natural attenuation capacity within a system and potential mitigation options.

Having a grasp of both the contaminants' biogeochemistry (transformations and storage) and transport (by applying an understanding of hydrological processes) is necessary to predict the magnitude, form and timing of contaminant inputs to water (Srinivasan et al., 2020). Knowledge about when and how contaminants are moving, allows to prioritise action in 'hotspots' and during critical times, such as a rain event, and to develop assessment methods that recognise lag periods between mitigation action and any improvements. In the longer term, this understanding can be utilised to

reorganise land use to reflect the natural capacity of a system to absorb and attenuate the effects of activity (Srinivasan et al., 2020).

#### **2.5.1.1. Surface transport**

Overland flow, or runoff, is the sheet-like movement of water over the ground surface. This occurs when the infiltration capacity of soil is exceeded or when the surface soil layer is saturated, meaning that antecedent soil moisture conditions are important in determining whether runoff will occur, as well as soil conditions such as compaction or hydrophobicity (Ward & Robinson, 2000). Overland flow occurs in the order of minutes to hours and particularly during rain events, sometimes reaching velocities up to 250 m hr<sup>-1</sup> (Neilen et al., 2017; Singh & Stenger, 2018). Areas where overland flow is generated are also source-areas for contaminants such as P, faecal microbes and sediment. Although N is mainly transported through interflow and groundwater, particulate N may be mobilised in overland flow as well, especially if rain follows a recent fertilisation or tillage event. When overland flow is the dominant pathway, water flows relatively quickly, which does not allow processes such as denitrification or immobilisation to occur: virtually all N may be lost to adjacent waterbodies (Fennessy & Cronk, 1997).

#### **2.5.1.2. Shallow subsurface transport**

In the subsurface, water may flow laterally as interflow (Figure 2.5), move downwards through the profile as percolation, or eventually move upwards as evaporation loss to the atmosphere (Ward & Robinson, 2000). The direction of movement occurs from areas of relatively high total potential to low total potential (defined as the sum of matric potential or suction, plus gravitational potential) (Ward & Robinson, 2000). The downward movement of water into drier and deeper soil horizons is important, given that it determines the amount of water retained in the root zone, the number of pores left for gas exchanges with the atmosphere and the water-filled pore space for storage until the following rain event.

Interflow occurs when the soil's lateral hydraulic conductivity exceeds the vertical hydraulic conductivity, leading to down-slope movement (McLaren & Cameron, 1996).

Soils are remarkably heterogenous and profiles are often comprised of layers with distinctive characteristics. It is common for the undisturbed soils surface horizons (e.g. those that are not compacted) to have a higher saturated hydraulic conductivity than deeper layers, where fine clays may accumulate, sometimes creating an impermeable boundary (Ward & Robinson, 2000). There, a perched water table may develop, above which water may flow laterally. This is the predominant mechanism hypothesised in the well-known Maimai experimental catchments (Westland, NZ) (McGlynn et al., 2002).

The slope of the water table determines the hydraulic gradient, with interflow moving down slope (Davie, 2004). The ability of a soil to transmit water is determined by the distribution of pore sizes and their connectivity (McLaren & Cameron, 1996; Davie, 2004; Brady & Weil, 2008). Macropores in particular (defined as those larger than 60  $\mu\text{m}$ ) (section 2.7.1.4) play an important role in the transmission of soil water. Davie (2004) reports that the movement of water through a saturated soil is usually slow, with a typical velocity of around 13  $\text{mm hr}^{-1}$  in NZ soils. An exception to this figure is the more rapid infiltration of water through earthworm holes, burrows and cracks into the soil— highlighting that the physical and biological characteristics of soils have a large effect on hydrology, and therefore contaminant transport.

In addition, some rapid interflow may result from the so-called “piston displacement” mechanism. This idea recognises that in certain situations, as rainfall enters the soil, it displaces the already existent water stored in the soil, causing the “older” water to exit the soil first. This means that the streamflow that is generated is not by water from the rainfall event, but by older water that has been stored in the soil. For this mechanism to effectively occur, soil water storage must already be full, thus rendering the pre-existent soil moisture conditions critical. This mechanism is supported by tracer studies (McGlynn et al., 2002) and has important implications for nutrient dynamics as it implies that nutrient-rich water will take a long time to reach the stream (Davie, 2004). Due to its solubility and negative charge,  $\text{NO}_3^-$  readily moves through this subsurface pathway.

Water quality improvement relies heavily on “adequate” residence times, which allow biological and physicochemical processes to occur (Fennessy & Cronk, 1997). Indirect flow pathways make residence times longer, thus maximising the ability of a system to process nutrients and other chemicals (e.g. through denitrification) — a reason why riparian buffers can be effective at removing  $\text{NO}_3^-$  (Fennessy & Cronk, 1997). For

instance, Fennessey and Cronk (1997) estimated that residence times in a 17-meter buffer strip ranged between 5 and 190 days. In a 27-meter buffer strip, residence times increased to between 12 and 1000 days (calculated from Haycock & Pinay, 1992). The tortuosity and variability of soil conductivity, however, make it extremely difficult to measure residence times in the subsurface and thus explain the high uncertainty in these figures. Data is also scarce.

### **2.5.1.3. Quantifying contaminant transfer pathways**

An understanding of the dominant contaminant transfer pathways allows to design and implement mitigation strategies that directly target the most influential pathways for freshwater quality. There are both direct and indirect methods to quantify transfer pathways. Indirect methods include tracer studies, concentration-discharge relationships and models.

Silica dioxide ( $\text{SiO}_2$ ) has long been used as a tracer to quantify the contribution of different pathways to streamflow. The concentration of silica increases proportionally to the waters contact time with Si-bearing minerals (due to dissolution) (Barkle et al., 2014a). Bromide is also a common tracer (e.g. Russow et al., 2013).

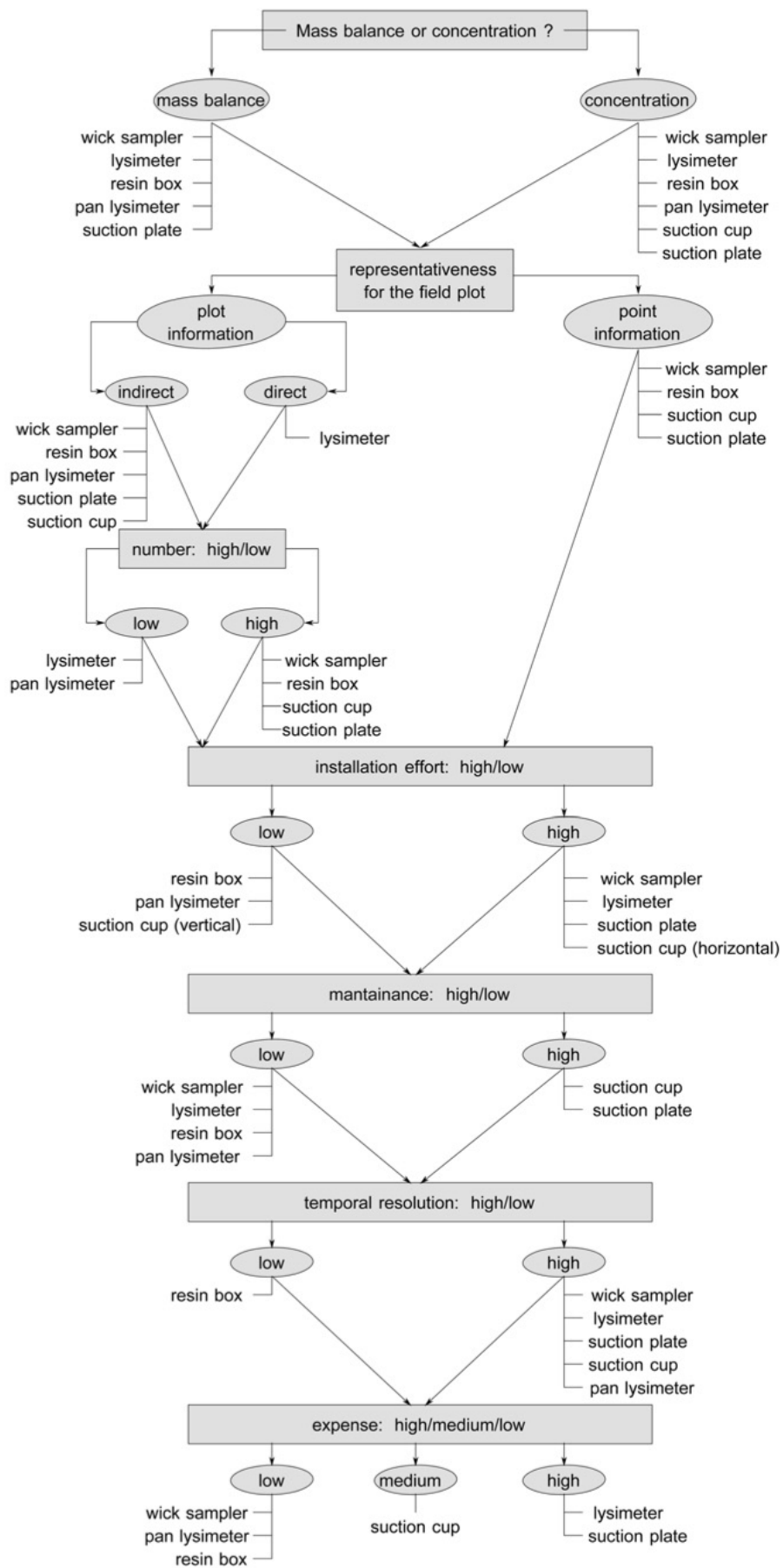
Indirect methods can also be used, including the analysis of Concentration-Discharge Relationships (CDRs) (Singh & Stenger, 2018). CDRs simply plot log concentration against log flow, painting a simple and rapid picture of the dominant processes in a catchment. This tool provides important information including whether the concentration of the solute either (1) increases with increasing flow (positive CDR, transport limits delivery rather than abundance of the solute), (2) decreases with increasing flow (negative CDR, abundance or production of solute limits delivery rather than available transport) or (3) shows little variation in concentration relative to large variations in discharge (chemo-static behaviour). The technique thus provides an integrated understanding of a system, identifying the main sources of contaminants and adequate mitigation strategies (Singh & Stenger, 2018).

There are two ways of setting up a CDR: the first relies on short-term and high resolution data and is most useful to understand the dominant hydrological and biogeochemical processes in a catchment. The second is utilising longer-term but lower resolution data, an approach which is most valuable as a screening tool to determine

sources of contaminants and most impactful mitigation options (Bowes et al., 2014; Singh & Stenger, 2018).

On the other hand, models are also useful and usually more accessible to land managers. The Overseer™ model was developed with the aim of producing a nutrient budgeting tool that is user-friendly to landowners and farmers. The model has recently been critiqued for a number of reasons: being a steady state model attempting to simulate a dynamic, continually varying system, using monthly time-steps that do not capture “hot moments” and not accounting for the heterogeneity of water and nutrient distribution in the soil profile, amongst others (Panel, 2021). Despite its flaws, the principles applied in Overseer remain useful. The model quantifies “source loads” (e.g. leaching losses from land) and “delivery loads” (which appear at the receiving environment such as a lake or river) in an attempt to create harmony between farm production, economics and environmental protection.

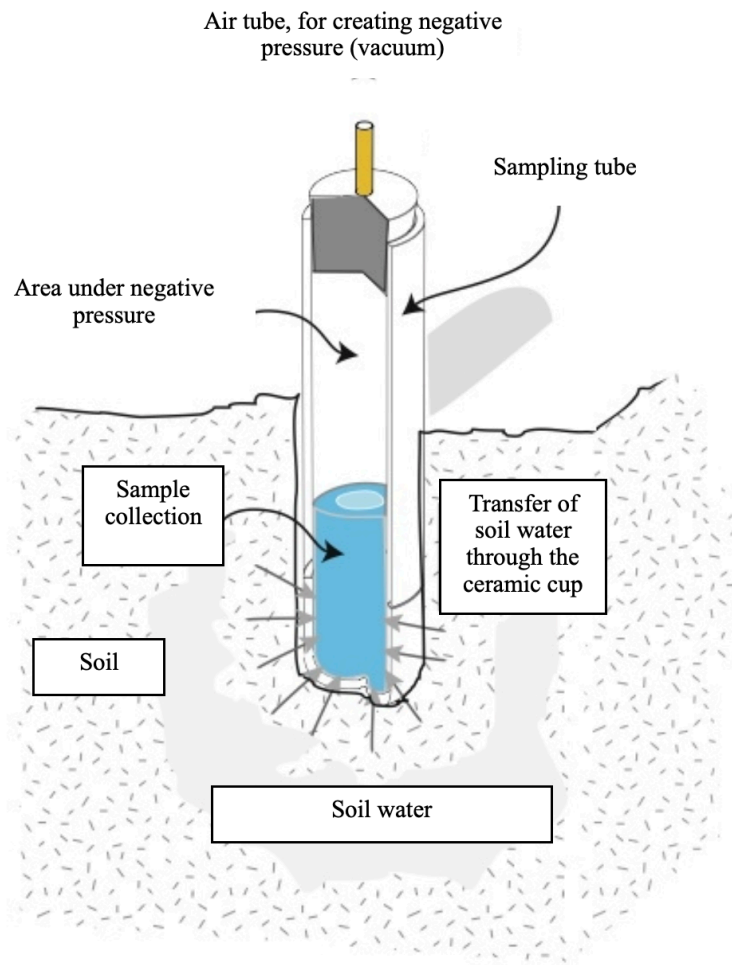
There are several methods to directly measure the subsurface movement of water, including soil physical measurements such as MP and hydraulic conductivity, or instruments like tensiometers. In terms of measuring contaminants in subsurface water, some common methods include: suction plates, pan lysimeters, wick samplers and suction cup lysimeters. Weihermuller et al. (2007) provide a thorough review of the advantages and disadvantages of each method, presenting a simple flow chart of appropriate sampling methods for different needs and circumstances (Figure 2.6).



**Figure 2.6: Flow chart for guidance on appropriate soil water sampling methods according to different needs and circumstances (sourced from Weihermuller, 2007).**

### **2.5.1.3.1. Suction cup lysimeters**

An understanding about the movement of elements and compounds through soil can be informed through the analyses of the chemical composition of soil pore water. Pore water is defined as the liquid matrix mediating between soil, roots and vegetation and is influenced by a complex web of biogeochemical interactions (Nieminen et al., 2013a). Suction cup lysimeters have been used since the early 1900s and are still the most frequently used method for extracting pore water (Weihermüller et al., 2007). They are referred to by a number of different names in the literature, including: suction cup lysimeters, suction cups, vacuum lysimeters and tension cup lysimeters. The technique samples pore water by producing a negative pressure (via a vacuum system) that draws soil water into a porous ceramic cup by capillary connections. Similar to tensiometers, water flows from the soil into the cup until the pressure inside the cup equalises with that of the soil water. The water flow rate through the ceramic cup and the soil dictate the amount of time it will take for equalisation to occur (Nieminen et al., 2013a). The cup is attached to a tube that comes up to the ground surface, where the user can draw samples for collection with a syringe or pump (Figure 2.7).



**Figure 2.7: Schematic of a suction cup lysimeter (adapted from SDEC, n.d.)**

According to Corral et al., (2016) tension and zero-tension lysimeters represent different soil solution processes. Tension lysimeters, such as suction cups, are reported to sample soil water under unsaturated flow conditions and from pores with longer residence times — i.e. water that has relatively more interaction with soil components. Hence, this method delivers a better representation of the bioavailable materials in soil, rather than those that are “just passing through”. This is particularly true for ions such as Na, K, Mg, Cl. However,  $\text{NO}_3^-$  appears to show no differences whether sampled via tension or zero-tension lysimeters. In contrast to tension lysimeters, zero-tension lysimeters, such as funnel lysimeters, sample free-draining water that has a relatively short residence time and does not interact with soil components as much (Cabrera Corral et al., 2016). Zero-tension lysimeters have been proposed to be a more suitable method to measure the flux of solutes between soil horizons, although Nieminen et al. (2013b) argue that it is difficult to draw definite conclusions and that the impact of lysimeter type is highly

site-dependent. They are considered a suitable method for sampling soil pore water in natural ecosystem studies (Nieminen et al., 2013b).

Some of the benefits of employing suction cup lysimeters in comparison to other methods are that they are low cost, low tech, easy to install, able to collect samples at different soil depths, easy to deploy in quantity and for long periods (and thus address different spatial and temporal scales), and can carry out periodic sampling with minimal disturbance of the soil. Suction cup lysimeters are able to collect samples on a continuous and discontinuous basis. The advantage of discontinuous sampling is that the presence (or not) of a solute at a particular point in time can be identified. Moreover, since the negative pressure is not constant, preferential flow pathways are less likely to develop (as would with a continuous sampling design).

Some limitations of the method are that there has to be good soil-lysimeter contact to prevent the creation of preferential flow pathways along the lysimeter walls (and thus inaccurate results), and requires soils with a relatively high water content ( $> -100$  kPa) (Weihermüller et al., 2007). Tension-lysimeters do not necessarily represent free-draining water— which could be a limitation of this method. They are also unable to record rapid changes in solute concentration (as happens during heavy rainfall), and they do not identify preferential flow pathways (Weihermüller et al., 2007). Soils are also inherently heterogeneous, and the placement of cups on different horizons throughout a landscape can create very variable results (Weihermüller et al., 2007; Cabrera Corral et al., 2016).

#### **2.5.1.3.2. Use in studies**

Suction cup lysimeters have specifically been used to study N in the vadose zone. For example, Zhou et al. (2010) used suction cup lysimeters with the goal of quantifying the impact of land use change on  $\text{NO}_3^-$  concentrations in soil and shallow groundwater, and to assess the potential of riparian plantings to mitigate losses. The study was conducted in maize-soybean rotations in central Iowa and focused on native riparian tall grasses. Two lysimeters were installed in each plot (riparian and cropland) to 1 m depth and were sampled monthly during the growing season between 2005 and 2008. The study found that  $\text{NO}_3^-$  concentrations in the vadose zone of upslope cropland to reach a maximum of 8.5 mg/l in April and then dropping to 0.4 mg/l in September. Under the

riparian zones, similar concentrations were found: 0.1-11.1 mg/l. Downslope and under cropland,  $\text{NO}_3^-$  concentrations remained higher throughout the year (between 3.1–10.6 mg/l). In contrast, under the riparian buffer, concentrations reached their maximum in April (2.9 mg/l) and declined to almost zero by July. This study showed that the conversion of grassland to cropland increased the  $\text{NO}_3^-$  concentrations in the vadose zone and shallow groundwater, and that grassed riparian buffers were effective at mitigating this impact.

Another study investigated the transfer pathways and residence times of N from an almond orchard to the Merced River, in California (Domagalski et al., 2008). The authors utilised wells and suction cup lysimeters (deployed at depths between 1 m and 6 m) in combination with models. Nitrate concentrations in soil solution showed that 63% of the N fertiliser applied was transported through an unsaturated soil zone, with the mean concentration of nitrate being 17 mg/l. In relation to residence times, the study revealed transport times of less than one year to over 6 years through a 50-100 m wide riparian zone. Domagalski et al. (2008) also found more strongly reducing conditions under the riparian zone than in the orchard, resulting in higher rates of denitrification.

NZ-based studies have also used suction cup lysimeters (Chibnall, 2013; Fraser et al., 2013; Trolove et al., 2019). Fraser et al. (2013) examined the impact of tillage intensity and winter cover crops on  $\text{NO}_3^-$  leaching over a 7 year period. Soil solution concentrations were calculated from suction cup samples (through flow injection analysis, FIA) in combination with drainage volumes. The suction cups were installed at a 60 cm depth on a transect along the length of the subplot treatments. The study reported that approximately 30% of the  $\text{NO}_3^-$  leaching variability between treatments was due to residual soil mineral N. They also found that cover crops were able to reduce leaching by ~50% during drier winters, and that tillage had little influence on leaching. The largest leaching losses occurred in an unfertilised fallow plot, which had no plant sinks to catch leaching N (derived from SOM mineralisation). Trolove et al. (2019) focused their work on  $\text{NO}_3^-$  leaching during pasture renewal— and different plant sequences that might be effective at reducing losses. Like Fraser et al. (2013), the study used suction cups to obtain  $\text{NO}_3^-$  concentrations (one in each treatment plot, at 1 m depth) and predicted drainage volumes using a model, in this case, the soil water module of APSIM. There was little difference in leaching between treatments although

they did find that the timing of leaching reflected on differences in plant N uptake and fallow period. The authors also found that in tilled plots, treading resulted in compaction and reduced  $\text{NO}_3^-$  leaching by around 40%— which was explained due to increased denitrification.

Lastly, Chibnall (2013) utilised suction cup lysimeters to determine the contribution of dissolved organic carbon (DOC) leaching to the C budget of a dairy farm in NZ. One hundred instruments were deployed and the samples were analysed for both DOC and TN. The study found that the average TN concentration was  $2.06 \pm 0.58$  mg/l at 30 cm depth.

### **2.5.2. Concluding thoughts on contaminant transfer pathways**

In summary, in order to address water quality outcomes — particularly in terms of contaminants like N— it is necessary to understand both (1) the contaminants biogeochemistry and (2) its main transport pathways. Due to its solubility and negative charge,  $\text{NO}_3^-$  readily moves in subsurface water flows. In turn, soil characteristics and vegetation are important factors in determining the partitioning of precipitation and irrigation water throughout the different water flow pathways— and therefore contaminant transport (Davie, 2004). MP and bulk density ( $\rho_b$ ) are two measures that give insight into the transmission of water through the soil. Barkle et al. (2014c) highlight the lack of reliable data on the fate of nutrients in the subsurface, due to the difficulty of obtaining measurements and large associated uncertainties. This is especially true when considering that soils are inherently heterogeneous and often have layers of distinctive characteristics— such as impermeable boundaries that may cause perching and lateral flow (Ward & Robinson, 2000). However, an understanding about the movement of elements and compounds through soil can be informed through the analyses of the chemical composition of soil water— which can be collected by suction cup lysimeters. This knowledge can then be used to design and implement mitigation strategies that directly target the most influential pathways for freshwater quality.

## **2.6. Mitigating N losses: riparian plantings**

Riparian zones, buffer zones and buffer strips are all terms that have been used to describe the transition between terrestrial and aquatic areas, where streams, rivers, lakes, wetlands and groundwater converge and interact (Zhao et al., 2016). Planting riparian zones is one of the potential strategies to mitigate the adverse effects of agriculture on the landscape which has garnered interest in NZ and worldwide (Howard-Williams, 2010; Franklin et al., 2015). There has been extensive research into the restoration of riparian zones as a means of reducing nutrient and sediment pollution in freshwater, as well as providing other ecosystem services— including filtering overland flow and reducing sedimentation, minimising peak flood flows, accumulating soil carbon (Trotter et al., 2005), bank stabilisation (Marden et al., 2005), stock exclusion, regulating stream temperature (Parkyn et al., 2003), provisioning of organic matter in-stream as a food source, provision of habitat for fish and aquatic invertebrates (Collier et al., 1995; Parkyn et al., 2003; Collins et al., 2013), and creating new streams of income for farmers (Daigneault et al., 2017).

The following sections give a broad overview of the research relating to riparian buffers, including their general benefits and their potential to mitigate diffuse nutrient pollution in agricultural settings.

### **2.6.1. Potential economics of riparian planting**

The initial establishment of riparian plantings can be costly, depending on the size of the buffer and the plants utilised. Although the literature recommendation for buffer width varies, it is generally accepted that wider is better (section 2.6.2.2). According to Environment Southland (2020), 100 metres of planting and 3 years maintenance would cost \$1195 for double rows of rows of native plants, and \$668 for a single row of shelter trees. Budgets should factor in ongoing maintenance costs and loss of profits from the retired land (MPI, 2016). Some estimates purport that riparian planting could yield net benefits of between NZ\$ 1.7 billion and NZ\$5.2 billion/yr when applied at a national level on all streams flowing through agricultural land (Daigneault et al., 2017). These figures consider benefits in the form of GHG emissions mitigation, reductions in N leaching and P loss, sedimentation and biodiversity gain, and relevant costs such as fencing, alternative stock water supplies, restoration planting and opportunity costs

(Daigneault et al., 2017). The potentially high initial costs of establishment could be offset by economic benefits down the line. In a similar fashion to silvopasture, farmers obtain economic benefits from plant products such as honey and timber, as well as their usual crops and animal products (Esperschuetz et al., 2017b). Native NZ plants, mānuka (*Leptospermum scoparium*) and kānuka (*Kunzea spp.*) can be used to produce essential oils. When planted in riparian areas, the species might even yield larger amounts of high-quality essential oils due to N fertilisation and luxury uptake. It is unlikely that the higher input of N or other nutrients will affect the quality of the essential oils (Seyedalikhani et al., 2019). It is estimated that the potential for the production of mānuka essential oils is NZ\$ 36,900 ha<sup>-1</sup> (Seyedalikhani et al., 2019). Mānuka honey is also a growing industry, with a 35% increase in total honey exports from 2015 to 2016, retailing between NZ\$12–148 kg<sup>-1</sup> (MPI, 2017).

### **2.6.2. Effects of riparian planting on freshwater quality**

Riparian plantings have received much attention in recent decades as an effective way to protect and improve freshwater from the adverse impacts of adjacent agricultural land. Amongst their benefits, buffer strips are purported to significantly reduce the import of sediment, pathogens and nutrients into surface and groundwater by filtering runoff and taking up nutrients, as well as “catching” leaching nutrients (Fennessy & Cronk, 1997; Parkyn et al., 2003). Being that excess nutrients are currently a major threat to freshwater environments in NZ (MfE, 2020), the research surrounding riparian plantings has been focused on this aspect. A meta-analysis of 89 individual riparian buffers from 45 published studies concluded that the effectiveness of riparian buffers at controlling nutrient export varies widely but that, indeed, they are capable of removing large quantities of NO<sub>3</sub><sup>-</sup> from water flowing through riparian zones (Mayer et al., 2007). Moreover, in a comparative study of nine riparian planting projects in the North Island of NZ, Parkyn et al. (2003) found that fenced and planted streams regained clarity and channel stability rapidly, but there were varied responses in terms of nutrients and faecal contaminants.

### **2.6.2.1. Riparian plantings and N cycling**

In relation to managing N inputs into surface waters, riparian plantings act by providing a zone for denitrification and N uptake by plants to occur, before N-loaded surface and subsurface flows reach a waterbody (Fennessy & Cronk, 1997). Throughout this study, nitrate removal will refer to the combined effect of denitrification and plant uptake. Neilen et al. (2017), propose that the relative importance of N uptake by vegetation and microbial processes is mediated by the hydrological conditions of the area, largely dependent on soil quality.

As discussed in section 2.3, some of the proximal factors that impact N cycling include moisture, aeration, the length and tortuosity of transport pathways, OC and microbial communities. These elements are, in turn, determined by distal factors, such as soil type and hydraulic properties (e.g. MP), landscape position, buffer width and vegetation. One of the factors mediating the efficiency of N cycling is the impact of vegetation type on soil quality (van Groenigen et al., 2015). Vegetation types (distal) have varying impacts on the degree of soil structure as well as OC and N requirements (proximal) (Schultz et al., 1995; Bharati et al., 2002; Franklin et al., 2019) (section 2.6.2.3). On the other hand, different plant species will have different N requirements.

### **2.6.2.2. Effect of buffer width on N cycling**

Multiple studies have emphasised that the efficiency of nitrate removal relies largely on the width of the riparian zone, which ensures that the residence time of flowing water is long enough to (1) allow plants to intercept and uptake N and (2) allow denitrification to occur (Fennessy & Cronk, 1997), as long as the hydrogeomorphology allows for water to infiltrate into the subsurface (Vidon & Hill, 2004) (i.e. the soil has a relatively low  $\rho_b$ , is not compacted at the surface and there is sufficient MP). One study found that up to 81% of contaminant removal capacity of a buffer strip is determined by width (Phillips, 1989). However, ideal buffer widths are contested, with the general agreement that wider buffers are likely to be more effective than narrower buffers— up to a certain threshold. Trends in buffer effectiveness are only apparent across a larger range of buffer size categories (Mayer et al., 2007).

According to a meta-analysis by Fennessy & Cronk (1997), a buffer zone of 20 to 30 m width can remove 100% of  $\text{NO}_3^-$  inputs. The study reports that this width is ideal for

improving surface water quality and that no additional benefits in terms of N cycling have been found in the literature with larger widths.  $\text{NO}_3^-$  removal efficiencies of 50, 75, and 90% were estimated among buffers approximately 5, 50, and 150 m wide (Mayer et al., 2007), suggesting that 30 m may be wide enough in other scenarios. Some studies propose that narrow buffers (defined as <15 m) are still capable of removing significant amounts of N, while others found that they might act as sources of N to riparian zones (Mayer et al., 2007). This could be attributed to high rainfall events in combination with pulses of rapid nitrification (e.g. after the tillage of adjacent agricultural land) (Mayer et al., 2007).

Some studies have found that the rate of removal remained constant through the entire length of the buffers, emphasising the importance of width (Mayer et al., 2007). Others propose that  $\text{NO}_3^-$  removal rates vary throughout the length of a buffer strip and are spatially focused (Fennessy & Cronk, 1997), being that there is relatively more OC at the edge of a buffer strip (Gilliam and Skaggs, 1988), where  $\text{NO}_3^-$  enters the subsurface and provides the best conditions for denitrification to occur.

Differences reported in the literature may be related to the flow pathways of each study site: subsurface removal appears not to be linked to width, while surface removal is. Subsurface removal might not be so tightly linked with buffer width since other factors influence denitrifying microbes on this pathway including: soil type, hydrology (soil saturation, groundwater flow paths), temperature and OC supply (Mayer et al., 2007). In contrast, the removal of N in surface flow relies on erosion control and the filtering of particulate N by vegetation. In this case, grassy buffers may be more efficient at filtering particulate nitrogen (compared to woody buffers) in sediments as it reduces channelised flow (Mayer et al., 2007).

Another consideration when deciding on the width of a riparian buffer, is any additional benefits sought to be gained, such as the provision of habitat, carbon sequestration, the diversification of farm income and cost of establishment.

### **2.6.2.3. Effect of vegetation type and age on N cycling**

Vegetation age and type determines how much N can be uptaken and also affects soil properties (Franklin et al., 2019), impacting the potential for denitrification. The

removal or production of  $\text{NO}_3^-$  has been attributed to variations in the conditions in the soils' rhizosphere and soil pore water. This might reflect differences in root biomass and morphology as well as plant canopies which influence rain percolation, differing evapotranspiration, and some influence of root exudation or leaf litter on soils (Franklin et al., 2015).

The largest proportion of N uptake occurs at the time of maximum biomass production (i.e during the growing season) (Fennessy & Cronk, 1997). Young vegetation may thus uptake N and P more rapidly than mature plants. Hence, some have suggested harvesting riparian vegetation to maintain the bulk of the buffer as an early succession, maximising the uptake of nutrients. However, this approach might prove too disruptive to other flora and fauna and thus any biodiversity gains made by the operation . Moreover, even without harvesting, after the growing season plants may die and reincorporate those nutrients into the system (Fennessy & Cronk, 1997). In the case of mature plants, luxury uptake into leaves may result in higher quality plant products such as essential oils (in the case of mānuka) (Esperschuetz et al., 2017b; Seyedalikhani et al., 2019).

The type of riparian vegetation is also of consideration. Hardwood basal area and grass coverage have been found to be predictors of the extent of N mineralization (Evans et al., 2011). There is evidence to argue for both using grasses over woody plants, and vice versa.

On one hand, numerous studies support that woody vegetation is more effective at reducing N exports to surface waters than herbaceous species (Haycock & Pinay, 1992; Fennessy & Cronk, 1997; Neilen et al., 2017). A few reasons have been identified to explain this. Firstly, woody vegetation usually has deeper tapping roots which are able to “catch” leaching  $\text{NO}_3^-$ , as compared to pasture's shallower rooting system (Esperschuetz et al., 2017b). By utilising the excess N from pasture, trees also improve the fertiliser use efficiency in farms. Secondly, trees supply a source of OC for microorganisms in the deeper profiles, where it is usually less concentrated as compared to topsoil (Fennessy & Cronk, 1997). Haycock & Pinay (1992) compared the  $\text{NO}_3^-$  removal efficiency of a riparian poplar forest versus a grass dominated buffer. They found that the forest removed 100% of  $\text{NO}_3^-$  under all flow conditions, versus 84%

removal by the herbaceous cover, precisely because of their higher biomass and deeper-tapping roots (as well as exudates). Additionally, woody vegetation has litterfall, serving as another source of OC to “fuel” denitrifying bacteria (Fennessy & Cronk, 1997). This factor is particularly important in those environments where denitrification is the main pathway of N removal instead of plant assimilation (Haycock & Pinay, 1992; Fennessy & Cronk, 1997; Neilen et al., 2017). Moreover, the difference in quality of plant litter between woody and grassy vegetation is significant (nutrient rich, lignin poor versus nutrient poor and lignin rich) (Schimel & Holland, 2005). The C:N ratio of litterfall-covered soils is in the range of 70:1, whereas grassy vegetation it is 17:1 (Neilen et al., 2017). Thirdly, woody vegetation is also able to assimilate more N due to its larger above-ground biomass (Neilen et al., 2017). The higher root biomass, and deeper root system also mean that soil structure is improved (plus root exudates, which increase soil flocculation) and so water is able to infiltrate into the soil profile and reach the anaerobic zones where denitrification is to take place. Because of this, some researchers have suggested that dairy shed effluent could be irrigated onto riparian plants as an alternative to grazed pastures (deeper tapping roots and OC at depth) (Franklin et al., 2015).

On the other hand, herbaceous vegetation is also useful as riparian buffers, trapping sediment and sediment-bound pollutants, such as P (Fennessy & Cronk, 1997). For example, in a greenhouse and field trial researchers found that native NZ monocot species (herbaceous) were more efficient at sequestering N than native woody dicots (Franklin et al., 2015).

In some cases, differences in the effectiveness of woody versus grassy riparian buffers might be related to rainfall. Grassy vegetation has shown lower  $\text{NO}_3^-$  leaching losses under high rainfall conditions as compared to woody riparian buffers. In contrast, under lower rainfall conditions, soil type and OC became more important factors in mediating N losses (indicators of microbial activity), which were lower in wooded areas (Neilen et al., 2017). A combination of woody and grassy vegetation may thus be the most effective way of minimising the flow of surface and subsurface  $\text{NO}_3^-$  from farmland into adjacent waterbodies (Fennessy & Cronk, 1997). As highlighted by Franklin et al. (2015), mixed plantings maximise the benefits of employing different types of

vegetation by simultaneously attenuating pollutants through various pathways, including root interception, biomass acquisition and soil rhizosphere denitrification.

#### **2.6.2.4. Native NZ vegetation: mānuka**

Given that plant species impact nutrient cycling in soils in different ways (Franklin et al., 2019), land managers can use that information to design agricultural landscapes that improve environmental outcomes (Dollery et al., 2019). If planting riparian bands, native NZ species could offer more benefits than exotics: many are adapted to low fertility soils and can improve native ecosystems and biodiversity (Franklin et al., 2015; Lind et al., 2019).

Mānuka (*Leptospermum scoparium*) (Figure 2.8) and kānuka (*Kunzea spp.*) are otherwise known as “tea trees” and are members of the Myrtaceae family, which includes 3100 species of cosmopolitan distribution (Stephens, 2005). Both species are similar in appearance, with the exception that *L. scoparium* is smaller, typically between 4-8 m in height and reaching up to 12 m (Stephens, 2005). Leaves are dark-green (7-20 mm x 2-6 mm) and sharp in comparison to kānuka’s— the easiest way to tell these very similar species apart. Mānuka flowers are white and occasionally pink or red with flowering occurring between October and February, triggered by temperature and day-length cues (Stephens, 2005). During the first five years of establishment, mānuka allocates a large percentage of its growth to root biomass (~40% according to Marden et al., 2005), reaching a depth of 30 cm in that time, and slowing thereafter. Mature trees of 13-50 years of age might reach between 50 cm and 80 cm depth depending on soil type (Marden et al., 2005).



**Figure 2.8:** *L. scoparium* tree (left), leaves (middle ) and seed pods (right)

Although the species such are known for establishing on low-fertility and acidic soils, and coping with harsh conditions such as severe exposure to high winds and waterlogged conditions (McLaren & Cameron, 1996), there is evidence that they can also survive and thrive in very high nutrient environments (Esperschuetz et al., 2017a). Franklin et al. (2015) argue that some native species could surpass the ability of pasture grasses to establish quickly in high-N agricultural soils. For instance, one laboratory study found a 117% increase in the total dry biomass of mānuka after the application of biosolids (equivalent to 1250 kg N ha<sup>-1</sup>). This increase is significantly larger than that experienced by the *P. radiata* control samples, which increased 86% (Esperschuetz et al., 2017a). The uptake of N, P and trace elements were also increased in the biosolids treatment, but not to concerning levels. Despite the apparent stimulation of aboveground biomass production, Esperchuetz et al. (2017a) found no significant effects in root biomass. Despite high N inputs, NO<sub>3</sub><sup>-</sup> leaching levels were low in all treatments (<2 kg ha<sup>-1</sup>). It is likely that in a high N scenario, where N and P are readily available, plants would accumulate more N than is needed for growth (i.e. luxury uptake), rendering them effective nutrient sinks (Fennessy & Cronk, 1997; Seyedalikhani et al., 2019). In contrast to the aforementioned findings, Franklin et al. (2015) found that other native NZ plants had a limited growth response to N. The study also found that the higher root biomass of native species may mean they are more tolerant to N than those of ryegrass (which died at high N loads), and that this might indeed translate into luxury uptake to foliage. The authors also found that there was less N accumulation in the soil under native species, therefore reducing the potential for N leaching (Franklin et al., 2015).

Mānuka may also be a Biological Nitrification Inhibitor (BNI)— i.e. may produce phytochemicals that prevent the bacterial oxidation of ammonium, producing nitrate. This may be through the inhibition of the enzyme ammonia monooxygenase or through toxicity to nitrifying bacteria (*Nitrosomonas* and *Nitrospira sp*) and some archaea (Downward, 2013). Mānuka, unlike many pioneer species, does not fix N. Given that the species is adapted to growing in low fertility environments, limiting nitrification might be an evolutionary adaptation. A study by Villegas et al. (2020) found that Guinea grass (*Megathyrsus maximus*), a BNI plant that is important for livestock farming in the tropics, reduced N<sub>2</sub>O emissions by between 30-70% due to lower abundance of ammonia oxidising bacteria and archaea, and lower rates of nitrification. Having less NO<sub>3</sub><sup>-</sup> in the soil reduces N losses via leaching and denitrification.

In a laboratory study, Downward (2013) found that *L. scoparium* leaves reduced nitrification by 67% relative to ryegrass (*P. radiata* decreased nitrification by 60% and *C. macrocarpa* by 87%). A lysimeter study had similar results, suggesting that this might also be the case in roots (Esperschuetz et al., 2017a). Esperschuetz et al. (2017a) found that, after a series of urea applications, mānuka significantly reduced NO<sub>3</sub><sup>-</sup> leaching compared to pine (2kg NO<sub>3</sub><sup>-</sup> ha<sup>-1</sup> versus 53 kg NO<sub>3</sub><sup>-</sup> ha<sup>-1</sup>). Moreover, both Esperschuetz et al. (2017a) and Halford et al. (2021) indeed found differences in the speciation of N under mānuka, having less NO<sub>3</sub><sup>-</sup> and more NH<sub>3</sub> compared to *P. radiata* and *L. perenne*. In contrast, Bowman (2020) did not find such difference compared to *L. perenne*, although their study included relatively young mānuka stands (~18 months old).

The ability of mānuka to adapt to a range of conditions and survive in marginal lands makes the species suitable for silvopastoral systems, shelter belts and riparian zones (Esperschuetz et al., 2017b). For these reasons, there has been research into using mānuka and kānuka in carbon farming endeavours (Trotter et al., 2005), for bank stabilisation (Marden et al., 2005), and as nurse crop for other vegetation and mature native forest (Stephens, 2005).

### **2.6.3. Concluding thoughts on riparian planting**

As discussed in the above sections, there are numerous benefits to the establishment of riparian buffers, and their effectiveness in reducing N losses to freshwater is well

researched. Their effectiveness is constructed by a variety of interplaying factors, where adjusting one factor may emphasise or undermine the importance of others (Mayer et al., 2007).

Evans et al., (2011), warned that, although riparian plantings indeed have merit in improving nutrient cycling they are not a silver bullet and their suitability needs to be assessed on a case-by-case basis— they may not provide the same level of N pollution mitigation or ecosystem services in place A as they do in place B. Their efficiency needs to be assessed in reference to the dominant biogeochemical processes occurring in a landscape as well as the position along a river network. An example of an ineffective riparian planting can be found in the Willamette River (Oregon, USA). After heavy winter rainfalls, runoff is the predominant water flow pathway due to very poorly drained soils. The result is that dissolved N is delivered directly into surface waters, without enough retention time in the riparian zones (or the adequate conditions) for denitrification to occur. During the late fall and early spring, low temperatures in the area mean that most vegetation is dormant and microbial activity is minimal (i.e there is little potential for denitrification) (Evans et al., 2011). This means that in this case, the right vegetation needs to be chosen to maximise the filtering of particulate N in surface flow— likely by herbaceous vegetation, and that the riparian buffer must be wide to maximise its efficiency.

Moreover, riparian plantings can be costly and usually requires on-going management. Treating this strategy as a silver bullet may lead to programmes and guidelines that do not achieve their goals (Evans et al., 2011). This approach needs to be combined with other management practices such as timing fertiliser applications to reduce nutrient losses, adequate irrigation practices and reducing other point and non-point sources of nitrogen. Water quality improvements start at the headwaters, where most of a stream's water originates, and continues down through the catchment, forming a continuous buffer though the length of the stream (Parkyn et al., 2003). Resolving issues in water quality requires taking an ecosystem perspective and integrated watershed management to improve environmental outcomes on land and in freshwater. This does include the restoration of degraded riparian areas, although the strategy should be treated as a 'bonus' to maximise nitrate removal. In summary, the consensus is that, with adequate knowledge and consideration of the locally dominant hydrological and biogeochemical

processes, riparian plantings can be extremely effective at mitigating N pollution, and that they should be used in combination with other best-management farming practices.

To date, the research surrounding the potential of riparian plantings and native NZ plants (specifically mānuka) to attenuate the flow of N from land to water seems promising. Nevertheless, the bulk of this research has been conducted in a laboratory, or in plant-scale studies (Downward, 2013; Esperschuetz et al., 2017b; Seyedalikhani et al., 2019; Gutierrez-Gines et al., 2021; Halford et al., 2021) — which do not represent field processes and conditions, and are still to be studied.

## **2.7. Soil quality**

Section 2.3 has reviewed the biogeochemical cycle of N in soils, particularly in agricultural landscapes, with emphasis on NZ (section 2.4). Section 2.5 has delved into the transfer of N from land to water, while section 2.6 has explored riparian plantings as a mitigation option— particularly including native plants such as mānuka. The medium underpinning the cycling and transfer of N is soil— whose characteristics are largely influenced by land use (largely vegetation, distal control), affecting the proximate controls of N cycling such as soil moisture, availability of OC and the microbial community. The following section is dedicated to understanding the different components of soil quality.

Soil quality is defined as “the capacity of a specific soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation” (Soil Science Society of America, 1995, as cited in Lilburne et al. (2004). More succinctly, soil quality is “fitness for use” (Larsen and Pierce, 1992). The targets for different soil quality indicators are thus suited to fit particular land uses, whether it be cropping, pastoral farming or indigenous forest.

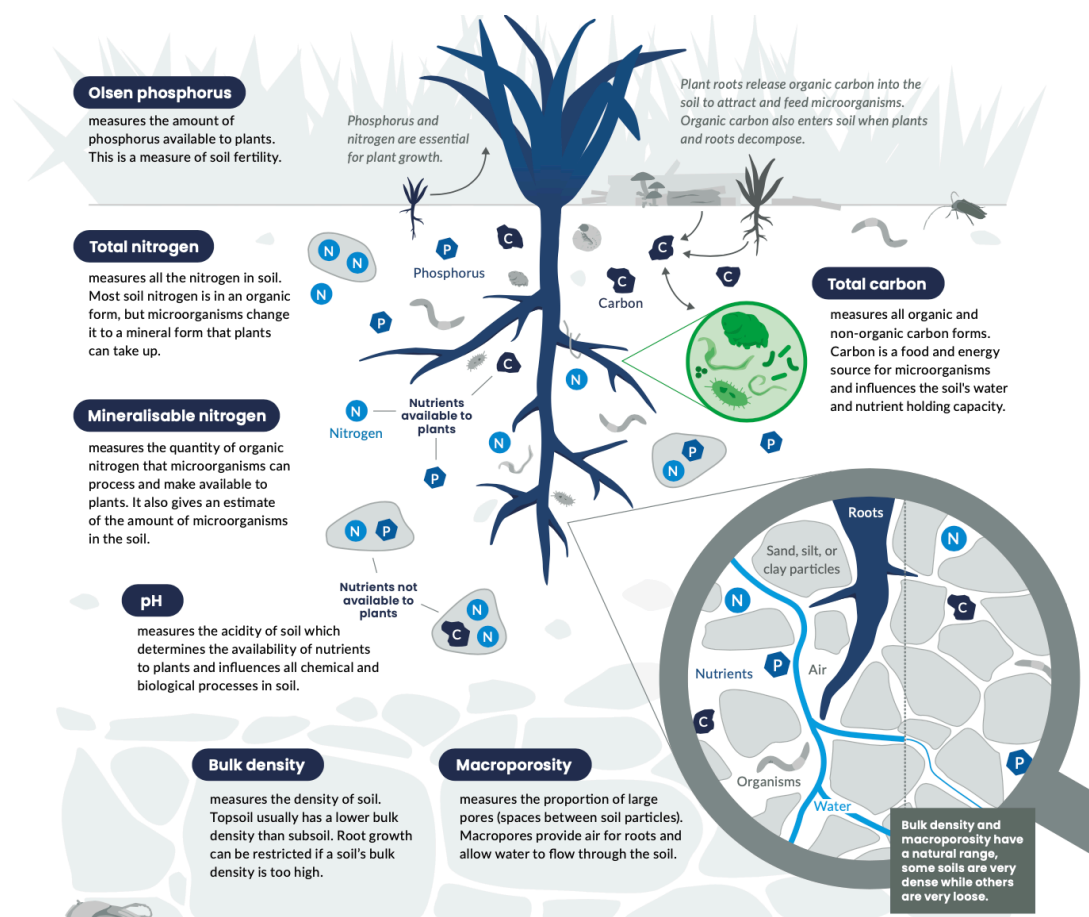
The degradation of soil resources, therefore, involves a reduction in productivity or ecosystem services provided. This could involve the soils becoming less fertile, eroded or contaminated, which might decrease production or destroy a valued ecosystem as a result of land use change and/or management practices. The deterioration of soil might

include compaction, loss of MP and erosion (physical); contamination, acidification, and nutrient excess/deficit (chemical); or loss of biodiversity and habitat (biological).

### **2.7.1. Soil quality reporting and indicators**

Soil monitoring in NZ is mandated by national legislation including the Resource Management Act (1991) and Environmental Reporting Act (ERA) (2015), which hold local authorities accountable to monitor and report on the state of the environment in five distinct domains. The ERA requires state of the environment reports at 3-yearly cycles, including land and freshwater.

The “500 Soils” project was born in 1995 out of the necessity to have consistent and thorough data on land use management practices and soil quality, for both remaining accountable to international agreements on environmental performance and reporting at the regional scale (Sparling & Schipper, 2002; Lilburne et al., 2004). The methodology measures seven representative soil quality indicators that impact key characteristics and functions of soil, specifically in the topsoil (0-10 cm). The indicators were selected to best represent the dynamic changes that might happen in the most intensive land uses: dairy pasture and cropland. They include TC, TN, total mineralisable N, Olsen P, pH, pb and MP (Figure 2.9). The most relevant indicators for this study will be discussed in further depth in following sections. Sparling & Schipper (2002) found that soil order and land use explained 55-76% of the variability in soil properties while the seven soil quality indicators explained 87% of the total variability.



**Figure 2.9: Infographic summarising the seven soil quality indicators (sourced from MfE & StatsNZ, 2021)**

The Waikato Regional Council currently has 150 active soil quality monitoring sites across nine soil orders and six land-uses, representative of the region. It takes 5 years to get through all of the sites, with some having been sampled four times over the last 20 years. In 2018, 96% of dairy pasture sites had concerning soil quality (especially MP and Olsen P), as well as 94% of other pastures, 89% of horticultural sites and 63% of the forestry sites (Waikato Regional Council, 2018). A review on soil quality in the Waikato Region found that in 2015, 10% of the surveyed sites met the targets for all seven indicators, down from 17% in 2006 (Taylor et al., 2017). The report also identified that key issues in the region's soil quality were soil compaction, excess nutrients, loss of soil organic matter (SOM) and a decrease in the levels of biological activity (Waikato Regional Council, Taylor et al., 2017; 2018). These issues persist despite having been identified as concerns in a similar reports released in 2010 and 2017 (Taylor et al., 2010; Taylor et al., 2017).

### **2.7.1.1. Total carbon**

SOM is critical for the adequate functioning of soils: it provides an energy source for biota, absorbs solar energy that warms the soil and binds toxic elements (e.g. Al). SOM serves as a source and storage of nutrients, buffers pH and improves soil structure, increasing the soil's water storage capacity and resistance to erosion (Balesdent & Mariotti, 1996). Given that SOM is comprised of ~50% C, examining the dynamics of soil organic carbon (SOC) is vital in understanding how to maintain and improve soil health. C reserves in soil are important in regulating biogeochemical cycles, particularly that of N, to which it is tightly linked. A decline in C levels in the soil are associated with a decrease in biological activity, particularly microbes, due to a lack of sources for their metabolic activities. Given that the N cycle is microbially mediated, low levels of C have a large impact on N cycling and, consequently, overall system productivity.

The measurement of TC includes both the organic and inorganic fractions (carbonates). However, since NZ soils have very low to negligible amounts of carbonates, TC is usually a good measure of SOC. When it comes to C contents in soil, no upper limits in terms of amount are defined (with the exception of organic soils, which are excluded from analysis by definition, already consisting of >50% C) (Sparling, 2008).

Soil type is a major influence on the size and stability of C stocks (~50%) (Sparling & Schipper, 2002). Coarser sandy soils generally hold less C than clayey soils due to the amount of binding sites. Finer soils also hold more water. Land use change and management practices also have a large impact— 27% according to Sparling & Schipper (2002). In general C contents under indigenous forest and pasture are larger than under plantation forestry and cropland (Sparling & Schipper, 2002). These figures emphasise the importance of matching land use to soil order as well as adopting adequate management practices that aim at conserving SOC.

Recognising the potential of soils to mitigate anthropogenic greenhouse gas emissions, the “4 per 1000” initiative was launched at United Nations Framework Convention on Climate Change (UNFCCC) twenty-first conference of the parties (COP21). The program's ambition is to increase SOM content, and thus carbon sequestration in soils, by 0.4% each year, worldwide. Given that there are areas of the world where land is severely depleted in C (Stockmann & S.Y., 2015), restoring such stocks could be a

powerful tool in addressing the UN Sustainable Development goals, tackling climate change, soil health and food security.

In NZ, the average SOC stocks (to a depth of 30 cm) are 98.7 t C/ha (Minasny, 2017). Meeting the 4‰ initiative would require a SOC accumulation rate of 0.4 t C/ha/year (Minasny et. al., 2017). In working towards this goal, we must explore the land uses and practices that are most effective in sequestering and stabilising soil carbon— whilst regarding food security and social and economic outcomes.

#### **2.7.1.2. Total nitrogen and total mineralisable nitrogen**

Most N in soil is contained within SOM and, in order to become bioavailable it must be mineralised by microbes via ammonification and nitrification (section 2.3). In the topsoil, up to 90% of the TN occurs in organic forms (Sparling, 2008). It is estimated that only 1.5 to 3.5% of the organic N in soil mineralises every year (Brady & Weil, 2008). The fraction that can be readily decomposed by organisms to become available to plants is measured as total mineralisable N (LMF, 2009). Measuring TN gives an indication of the available reserves for plant growth, the accumulation of N in soil and the potential for N leaching at times when plant demand is low (during winter for example) (Brady & Weil, 2008). Given that N is highly associated with SOM (which, as discussed is vital in ensuring adequate soil structure and biochemistry), the ratio of C:N is important.

N levels in soil are equally determined by land use and soil order (43% each) while the rest is attributable to management practices (Sparling & Schipper, 2002). Pastoral systems have a higher N content than any other land use, partly due to the dominance of white clover, a N-fixing plant, and N-fertiliser inputs in intensive systems. This is evidenced in the much lower C:N ratio for pastures (between 10:1 and 15:1) compared to that of plantation forests, tussock grassland, and indigenous forest (between 15:1 and 20:1) (Sparling & Schipper, 2002). Optimum TN and mineralisable N levels in soil depend on land use whether it is for productivity (“more is better”) or environmental protection (“less is better”) (Sparling, 2008). In terms of the N movement from land to water, the amount of N that arrives at a waterbody depends highly on the excessive

accumulation of N in soil that may later be leached under wet conditions before it has a chance to be taken up by plants or become denitrified to N<sub>2</sub> gas.

### **2.7.1.3. Bulk density**

A thorough understanding of the physical characteristics of soil is vital for appropriate land use management, since they dictate the soil's suitability for plant growth just as much as chemistry (McLaren & Cameron, 1996).  $\rho_b$  refers to the mass of solids contained in a sample relative to its volume (i.e. how closely packed the soil particles are). For instance, a soil with a high  $\rho_b$  ( $> 1.3 \text{ t/m}^3$ ) is more compact in comparison to one with a low  $\rho_b$  ( $< 0.8 \text{ t/m}^3$ ) (McLaren & Cameron, 1996). A compacted soil will not allow water or air to move freely through it, restricting root penetration, plant growth and the health of the microbiome. On the other hand, soils with a very low  $\rho_b$  might be too loose and prone to erosion, as well as having little water-holding capacity so plants might find it difficult to establish or acquire nutrients and water (Sparling, 2008).  $\rho_b$  also impacts the infiltration of water, where a compacted soil increases the likelihood of overland flow and thus nutrient loss and soil erosion. A compacted subsoil layer could also limit the downward movement of water, resulting in lateral flow above the impermeable layer. Compaction is also significant since it may alter the dynamics of microbial species that mediate the N cycle by limiting the diffusion of air (and thus O<sub>2</sub>) into the soil. This might increase the proportion of NO and N<sub>2</sub>O as products of nitrification and denitrification (Schimel & Holland, 2005). Compaction also impacts the soil's organic reserves (TC, TN, TMN) (Sparling & Schipper, 2002).

Soil order is the major factor determining  $\rho_b$  (Sparling & Schipper, 2002) due to different characteristics such as SOM content, texture, constituent minerals and porosity (Sparling, 2008). Management practices such as the use of heavy equipment or vehicles, high stocking rates, cultivation and loss of SOM are all factors that can contribute to compaction (Sparling & Schipper, 2002; LMF, 2009).

### **2.7.1.4. Macroporosity**

When a soil becomes compacted, macropores ( $> 60 \mu\text{m}$ ) are the first ones to collapse (thus MP has an inverse relationship with  $\rho_b$ ) (McLaren & Cameron, 1996; Sparling,

2008). Macropores are also the first ones to drain after a rainfall event. MP is used as a soil quality indicator, rather than total porosity since macropores are responsible for most of the aeration and drainage in soil. This is also a very sensitive indicator of changes in soil quality, often due to management practices. Low MP in soil is associated with the same issues as a high pb, as well as the restriction of air movement (Sparling, 2008). MP is calculated as a function of moisture release curves, particle density and pb (Sparling, 2008) and is expressed as a percent pore volume over the volume of soil. In a 2002 review of soil quality in NZ, Sparling & Schipper found that the highest pb occurred in cropland soils, which also had the lowest MP. On the other end of the spectrum, soils under indigenous vegetation generally have higher MP (10-24% v/v) (Sparling & Schipper, 2002).

MP impacts the N cycle given that nitrifying organisms require aerobic conditions to produce  $\text{NO}_2^-$  and  $\text{NO}_3^-$  ions, and are thus favoured in well-drained and well aerated soils (Brady & Weil, 2008). Less aeration results in less N fixation and nitrification, and thus, a decrease in plant productivity. MP also determines the amount of water that can be held in large pores and form micro-sites where denitrification can occur in unsaturated conditions. When soils are very dry, microbial activity and thus gas production, are limited. As the soil starts getting wetter and the volumetric water content (VMC) approaches 30%, nitrification is the dominant process, producing NO. As water filled pore space goes over 60% VMC, the rate of nitrification slows sharply and denitrification becomes dominant, with  $\text{N}_2\text{O}$  as the main product. As the soil become saturated, denitrification continues until all the available  $\text{NO}_3^-$  is used, and  $\text{N}_2$  is mainly produced.

A rapid flush of nitrification might occur after the first rains of a dry period, tilling (introduces  $\text{O}_2$ ) or the sudden warming of frozen soil as it thaws during spring (Brady & Weil, 2008), when MP suddenly increases. Additionally, increased soil aeration is associated in an increased capacity of the soil to uptake methane ( $\text{CH}_4$ ), another powerful greenhouse gas (Saggar et al., 2008).

### **2.7.2. Concluding thoughts on soil quality**

Both the chemical and physical properties of soil impact its functioning— affecting microbial communities as well as the flow of air and water— ultimately dictating the

rate and efficiency of nutrient cycling. TOC, TN,  $\rho_b$  and MP are key indicators of soil quality, which are used throughout NZ for environmental monitoring. Having data about these key indicators, and an understanding of the synergies between them can help to create a more comprehensive picture of the processes mediating the transfer of N from land to freshwater. From this perspective, TOC reflects on the available food sources for denitrifying bacteria (converting  $\text{NO}_3^-$  into  $\text{N}_2$  and  $\text{N}_2\text{O}$ ). TN reflects on the amount of N in soil which has the potential to be transported in subsurface water, or leach into groundwater, ultimately reaching rivers and lakes.  $\rho_b$  impacts the ability of plant roots to penetrate deeper into the soil profile, potentially “catching” leaching  $\text{NO}_3^-$  and uptaking it.  $\rho_b$  also influences the ability of water to infiltrate into the soil— either allowing water to move freely or impeding its downward movement (for example above an impermeable layer). MP may alter the dynamics of microbial species that mediate the N cycle by limiting the diffusion of air (and thus  $\text{O}_2$ ) into the soil.

## **2.8. Lake Waikare restoration project**

The intention of the following section is to frame this study in the current efforts to restore Lake Waikare. In this area of Aotearoa (NZ), the environmental state, impacts and potential solutions are complex— with multiple interrelationships of causes, effects, interactions and actors involved. The following is an attempt to identify some of these factors in order to explain where this project fits, and where it can be useful for the region. Information was gathered from published sources as well as informal interviews and participation in various hui (meetings). This section has been written and reviewed by participants in this project to increase its accuracy<sup>1</sup>. Much of the information in this section has been kindly shared by local whaanau (families) for the purposes of this project, and all of the information and stories below belong to them. Thank you for providing insight into the local history and importance of the lake.

### **2.8.1. Location and physical characteristics**

Lake Waikare is part of the Whangamarino Wetland catchment (797.1 km<sup>2</sup>), located in the Waikato Region of Aotearoa (Figure 2.10). The catchment includes Lake Waikare,

---

<sup>1</sup> Note that the following sections have been written using Waikato-Tainui conventions, using double vowels instead of macrons in reo words (maori language). English translations have also been provided.

Whangamarino Wetland, Whangamarino River, and the Matahuru and Waerenga streams. The catchment drains via the Whangamarino River into the Waikato River between Meremere and Mercer (Lawrence & Ridley, 2018).



**Figure 2.10: Map of Lake Waikare in the context of New Zealand's North Island (marked in red)**

The Whangamarino Wetland receives water from Lake Waikare ( $37^{\circ}28'26''\text{S}$ ,  $175^{\circ}13'55''\text{E}$ ) and the Waikato River catchment as part of the Lower Waikato-Waipā Flood Control Scheme (LWWFCS). The wetland is recognised as an area of significant ecological value and is protected by the Ramsar Convention for the protection of wetlands (RAMSAR, 1992). Whangamarino wetland is also the second largest in the North Island, while Lake Waikare is the largest in the Lower Waikato Basin (Lawrence & Ridley, 2018).

The lake's surface area is 3,442 ha and is relatively shallow, with an average depth of 1.5 m, and a maximum depth of 1.8 m (Dean-Speirs, 2014). The lake's catchment (~21,055 ha) is delimited by the Waikato River (west), the Hapuakohe Range (east), the Taupiri Range (south) and a low ridge on which Te Kauwhata-Waerenga Road separates Lake Waikare from Whangamarino Wetland to the north (Lawrence & Ridley, 2018). To the west lie four small lakes, three of which are linked to Waikare via drains. These are

Ohinewai, Kopuera, Rotokawau and Penewaka Lagoon (not linked) (Reeves, 2012). The main inflow is Matahuru stream.

### **2.8.2. History and land uses**

Before the colonisation of Aotearoa by Britain, it is most likely that the Lake Waikare catchment was covered with secondary forests on the slopes and a rich freshwater wetland ecosystem on the flats, with scattered stands of kahikatea (*Dacrycarpus dacrydioides*). The landscape would have likely evolved, with taller species such as rewarewa (*Knightia excelsa*), kamahi (*Weinmannia racemosa*), mangaeo (*Litsea calicaris*), rimu (*Dacrydium cupressinum*), and tanekaha (*Phyllocladus triomanoides*) becoming dominant (Lawrence & Ridley, 2018).

The lake was a ‘food-bowl’ for all iwi (tribes) within whose rohe (territory, area of land) it falls (Glen Tupuhi, 2021, personal communication). According to the narratives gathered during waananga (learning occasion) in Waikare Marae and Matahuru Marae the marae used to be a mahinga kai, a food production centre for the whaanau. For instance, there used to be a large migration of tuna (eel) around the lake which was an important fishery— purported to be the biggest in the Waikato Region, with a yield of up to 85 tons each year (Dean-Speirs, 2014). Grass carp used to make a restorative medicinal broth for health. Taro plantings were also abundant in the area, and many were discovered during excavations for the Waikato expressway (Warren Gumbly, 2019, presentation at waananga). The lake used to host a variety of important plants including rongoaa species (medicinal), raupoo (*Typha orientalis*) for doing floral arrangements, ferns, and kawakawa. Harakeke (flax) used to be collected from around the lake and hapuu (sub-tribes) would take it to the Auckland markets to trade with. Waka ama (canoeing) was often practiced on the lake’s healing waters.

Significant changes have occurred in the area since European settlement. Deforestation of the erosion-prone hillsides became widespread. Today, only 7.58% of the catchment remains under native vegetation (Dean-Speirs, 2014), which causes sediment and nutrients to drain straight into Lake Waikare. The lake bed, which used to be pumice sand is now soft sediment. The tuna has become smaller and the abundance of kooura (freshwater crayfish) and kaaeo (freshwater mussels) has declined drastically. Pest

species such as grass carp first, and the more aggressive koi carp recently, have replaced native species.

Before European settlement, the lower Waikato Region had poorly drained soil with land that was periodically flooded. The LWWFCS was launched in the 1960s. Due to its integration into the flood control scheme, the lake level now fluctuates seasonally at about 0.3 meters (Lawrence & Ridley, 2018) and approximately 840 ha of seasonally inundated land were lost. According to Reeves et al. (2012) the scheme has resulted in a considerable degradation of ecologically valuable ecosystems due to the modification of the hydrological regime, which prevents the flushing of the lake that occurred during large floods. Moreover, the flood control scheme “liberated” land for agriculture, which has become intensive in the area and further aggravates water quality issues and the loss of habitat: it is estimated that 67% of the wetland has been lost since 1963 (Reeves, 2012).

Ownership of the lakebed was recently returned from Land Information NZ (LINZ) and Department of Conservation to Waikato-Tainui (2008). Today there is limited access to the lake and its cultural values and mauri (life force, vital essence) are significantly impacted by its currently poor environmental state.

According to scientist Moritz Lehmann, the lake is the second most polluted in the country (after Lake Ellesmere in the Canterbury Region) (O'Dwyer, 2020b). Lake Waikare is in a hypertrophic state due to extremely high levels of suspended sediment (Dean-Speirs, 2014) and blue-green algal blooms are frequent, historically during the summer (Figure 2.11). In 2021 the lake had health warnings throughout winter for the first time (Glen Tupuhi, 2021, personal communication) and in June 2021, the lake water was a bright red colour (personal observation).



**Figure 2.11: (Above) Lake Waikare in 2018. (Below) Toxic algae blooms at Lake Waikare in December 2019 (sourced from O'Dwyer, 2020).**

Between 2010 and 2012, 9 out of the 10 samples taken annually from Lake Waikare exceeded the recreational guideline levels for blue green algae (Dean-Speirs, 2014). One study revealed that most nutrients and sediment inputs come from the Matahuru stream inflow. The catchment (which is mostly steep pastoral lands) contributes 60% of TN, 53% of TP, and circa 75% of the annual sediment load of the lake (Dean-Speirs, 2014). A freshwater scientist from the Waikato Regional Council told news outlets that there was still an increasing trend of nitrogen in Lake Waikare in 2020 (O'Dwyer, 2020a). The intensification of land use in recent decades is partly responsible for aggravating the water quality issues. Another great threat to biodiversity (and water

quality) in the lake is the dominance of exotic pest fish including koi carp, rudd and catfish. There have also been non-compliant discharges of wastewater from the Te Kauwhata's wastewater treatment plant. According to O'Dwyer (2020b), discharge has not been compliant in levels of P, N and *E. coli* for three years. Despite these negative impacts, Lake Waikare still hosts high biodiversity and a range of habitat for indigenous species— including six threatened species (Reeves, 2012).

### **2.8.3. Lake Waikare learning community and vision maatauranga**

Significant efforts have been made in recent years to start restoring Lake Waikare, with efforts being made by many hapuu around the lake— including Matahuru and Ngaa Muka as well as the Te Kauwhata community as a whole (the township on the northern end on the lake). All are very passionate and determined to recover the mauri of this ecosystem and reconnect the people to the lake as well as provide economic opportunities to revitalise the area. To this end the Lake Waikare Steering Group was formed in 2007 to assess management options. In 2017 the Lake Waikare Learning Community was established (Wallace, 2017). With the aim of establishing the purpose and vision of the Learning Community's outcomes, a document was compiled to summarise the values and expectations of hapuu and whaanau in relation to the lake. This was done through historical accounts of kaumautā and kuia (male and female elders respectively, a person of status within the whaanau) from a hui held at Waikare Marae and short interviews with whaanau participating in a Community Planting Day. Although the participating adults did not believe that the youth would gather kai or flax, they still considered it important for them to understand their whakapapa (lineage, descent) and practice manaakitanga (hospitality, kindness, generosity, support) — an important principle related with food gathering. Below is a summary of the environmental indicators aspired to, as established at in the Vision Mātauranga project, designed to understand the desired outcomes of the mana whenua:

- Improved water quality, specifically clarity and safety for recreational use (to swim and sail in),
- Bushes and trees grow, plants take hold and flourish, supporting birds and other species coming back to the area (including native ducks and teal), rongoa species planted,

- Wildlife flourishes, birdsongs increased (more sightings of tui, keruru, bellbirds etc),
- Ridding of koi carp and other pests, increased abundance of eels and fish: mullet, inanga, grass carp, and freshwater mussels,
- Building of pest proof fence to allow for the release of kiwi,
- Establishment of miro miro to attract birds and nature walkways, and
- Establishment of mānuka for bees.

### **2.8.3.1. Scientific work at Lake Waikare**

The restoration of the lake is expected to be very complex due to the number of point and non-point sources of pollution, physical characteristics of the lake (shallow, very fine sediment that is constantly resuspended by pest fish and wind mixing) and ecosystem modification (LWWFCS, draining of wetland, conversion to pasture, invasion by pests) (Barnes, 2002). Currently, there are a variety of different projects underway involving different groups from across the rohe, all aiming at the common vision of recovering the environmental and cultural values of Lake Waikare. One such project involves Matahuru iwi (tribe).

A report exploring mitigation options to reduce sediment inputs to Lake Waikare and the Whangamarino wetland concluded that all options had prohibitive costs, uncertain outcomes and unacceptable results for stakeholders (Reeves, 2012). In terms of riparian management, livestock exclusion was deemed to be highly applicable to the Matahuru Catchment, with the potential to reduce N, P, sediments and faecal microbial loads. Riparian buffers were classified as moderately applicable to the Matahuru catchment. Benefits identified were the reduction of N and P loads, improved aquatic habitat, biodiversity values and landscape aesthetics. The one drawback that was identified was the need to actively manage the vegetation. Grass filter strips were also deemed to be highly applicable. Importantly, the report identified knowledge gaps that need to be addressed in order to effectively enact mitigation measures, including better quantifying sediment and nutrient input into the waterbodies (Reeves, 2012).

Nikau Estate Trust and Matahuru Marae whanaau, landowners on the south margins of the lake, transformed a plot of farmland in 2017 in a commitment to reassert their kaitiakitanga (guardianship, stewardship) on their lake and surrounding whenua (land) (Figure 2.12). This mainly involved the planting of native vegetation in the riparian areas, set up as a series of experimental plots with different combinations of vegetation, to carry out scientific research. Mostly funded by Waikato River Authority, the research is being done with the support of scientists from ESR, the Centre for Integrated Biowaste Research (CIBR), Waikato Regional Council, Ecoquest Education Foundation, Waikato University, Auckland University, Canterbury University and Manaaki Whenua Landcare Research. There are many research projects being carried out at the experimental plots, investigating different aspects of the restoration effort including microbiology, hydrology, soil and water quality. The vision of mana whenua is to totally fence the lake and streams and plant them in native vegetation to protect them from the adverse effects of the adjacent land uses, especially sediments and nutrients. The scientists are exploring which native plants or combination of native plants would optimise riparian outcomes.



**Figure 2.12: (top) aerial image of the location of the experimental riparian plots before planting, in 2016 (Waikato Regional Council, 2016) and (bottom) aerial image of the experimental riparian plots after 4 years of establishment, in July 2021. During this time a bridge was being built for easier access to Lake Waikare (University of Auckland, 2021).**

# Chapter 3

## Methodology

---

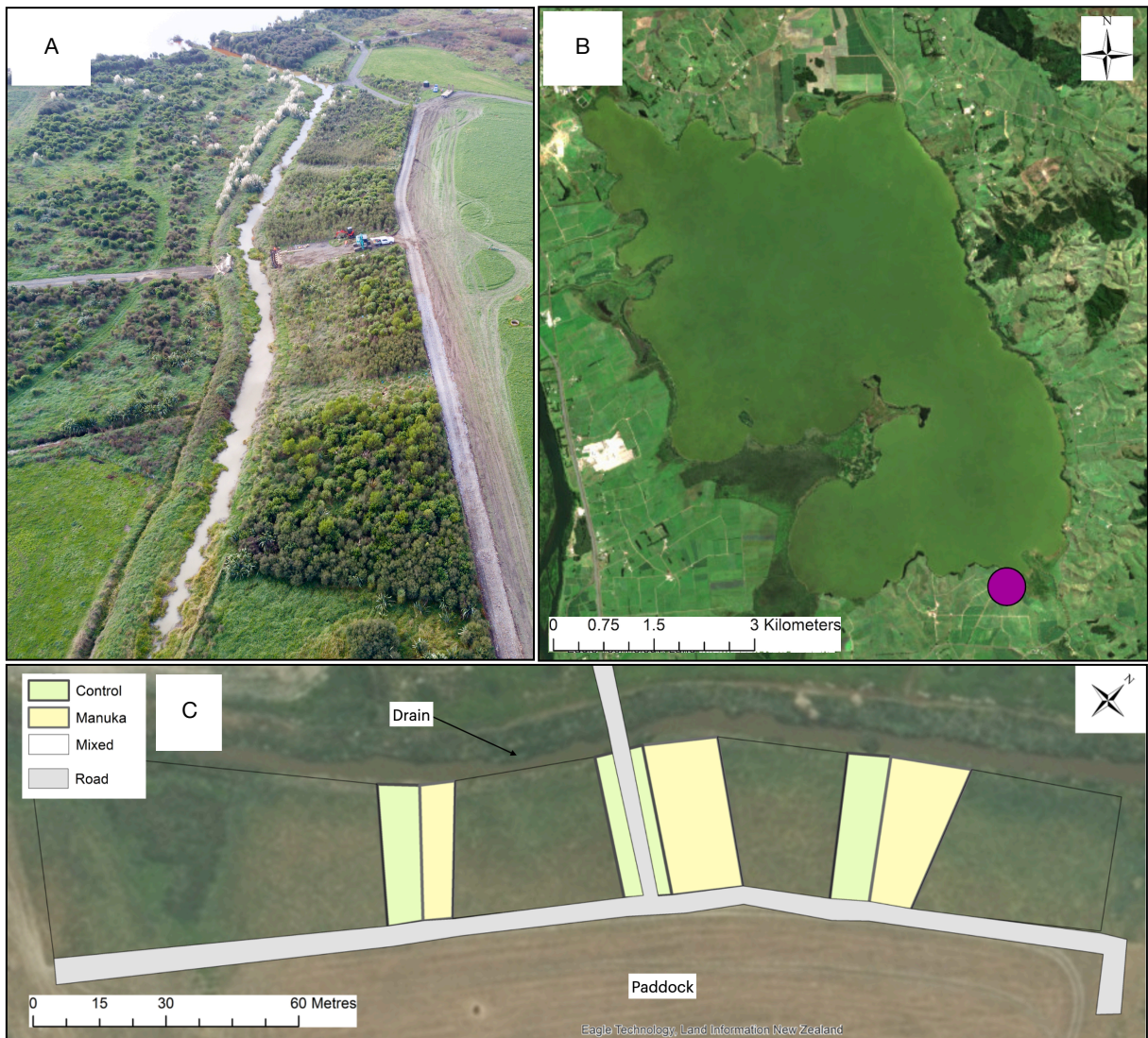
### 3.1. Experimental plots

An opportunity that arises from the poor ecological state of the Lake Waikare catchment is to investigate alternative land uses for this area (e.g. mānuka planting). This can assist in decision making to match land use with land suitability in the catchment, while improving water quality and providing continued economic benefits from this land. First, however, the impacts of land use change (LUC) must be assessed, alongside whether they will achieve the desired outcome.

In this spirit, 4 ha of productive land were transformed by Nikau Estate Trust to set up experimental plots (-37.474036, 175.231869) (Figure 3.1). One of the plots comprises of a 272 m x 30 m riparian band planted at a density of about 1 plant/m<sup>2</sup> during the winter of 2017. The riparian strip runs along a drain that flows into Lake Waikare, and downslope of a paddock used for dairy cow grazing until October 2019, when it was converted to maize cropping for 2020. After the maize was cultivated in February 2021, the field was used for silage production. A single fertilisation event occurred between 2020 and 2021, after the harvest of maize (April 2021), where koi carp fertiliser was utilised. The paddock and riparian band are in a gentle slope towards the drain (10°). This riparian band is divided into 10 experimental plots:

- three mānuka plots,
- three unplanted but fenced plots (grassed), and
- four plots with a mixture of mānuka and 20 other native species appropriate for lower Waikato region in between the mānuka and unplanted plots.

The mānuka and grassed plots are each 30 m long and approximately 10 m wide, which is large enough to reap benefits for the mitigation of nutrient pollution (Fennessy & Cronk, 1997) (section 2.6.2.2). At the time of planting, there was a surplus of plants, which resulted in some of the control plots being slightly narrower by 1-2 m. The distance from the plots to the lake is approximately 300 m and 100 m from the farthest and closest plots respectively (along the length of the drain).



**Figure 3.1: (A) Aerial photo of the experimental plots (July 2021), (B) aerial photo of Lake Waikare highlighting the location of the plots and (C) outline of the experimental plots and their vegetation (produced in GIS).**

### 3.1.1. Vegetation

The objectives of this study focused exclusively on mānuka riparian plantings. Thus, a subset of the experimental plots was selected: Control 1 (C1), Control 2 (C2), Mānuka 1 (M1) and Mānuka 2 (M2) (Figure 3.5). Plots M1 and M2 had slightly different vegetation densities— approximately 1.6 plant/m<sup>2</sup> and 1 plant/m<sup>2</sup> respectively. The control plots were fenced but unplanted, with a plant cover currently dominated by *Dactylis glomerata* (also known as cocksfoot).

Although initially the plan was to use two sets of adjacent mānuka and control/grassed buffers, a road was constructed on one of the control plots for farm operations (February 2021). Moreover, half-way through the course of the project (late May 2021) a gravel

road was built upslope of the experimental plots (after the installation of the monitoring equipment). Lastly, in July 2021, a bridge was built over the drain downslope from the plots, which involved the movement of lots of soil and a significant amount of heavy vehicle traffic. This might have had implications for the movement of runoff and subsurface flow from the paddock and through the riparian plantings.

### 3.1.2. Soil setting

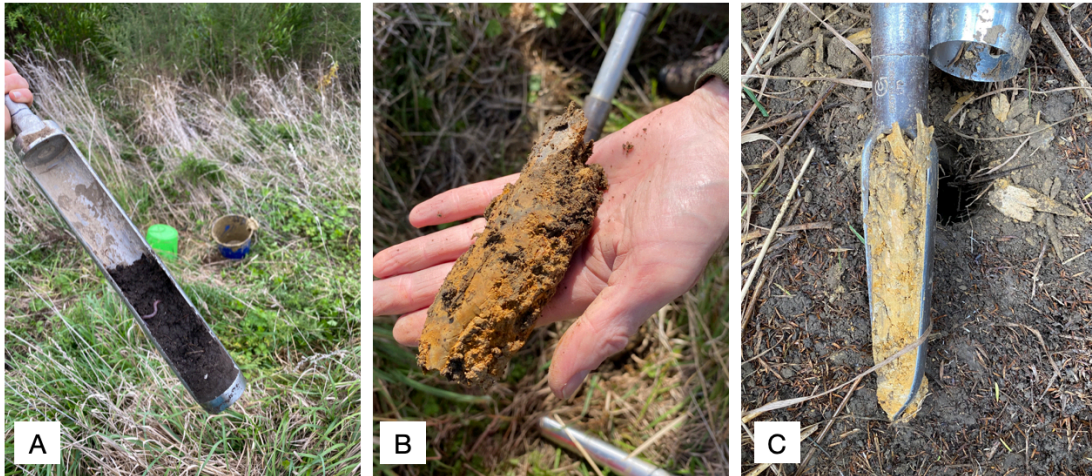
The soil in the experimental plots belongs to the Mangatawhiri clay loam series (to approximately 30 cm, depth to which it is ploughed), part of the Kainui soil series, classified as an Perch-Gley Ultic soil. This is underlain by an orange-coloured, clay-rich layer below: the Hamilton Ash (Figure 3.2).



**Figure 3.2: Soil profile to 40 cm in the paddock upslope of the experimental plots (under maize in the summer of 2020-2021).**

The Kainui soil is widespread around the north of Hamilton and the Waikato lowlands, and is comprised of two distinct parts: the upper layer varies between 0.4 and 0.7 m in thickness and is made up of thin tephra layers deposited in the late Quaternary (last 50,000 years). Underneath are the Hamilton Ash beds comprised of strongly weathered clays (Lowe, 2019b). The Hamilton Ash “layer” remains largely undated and could be as young as 74,000 years and as old as 125,000 years (Lowe, 2019b). The shallower

portion of the soil is comprised of thin layers of rhyolitic and andesitic tephra originating from the Taupo Volcanic Zone (TVZ). These tephras were deposited one by one, meaning that each of them was an “A” horizon at one point in time (Lowe, 2019b). Over time, there had been significant intermixing between the tephras due to the ongoing top-down soil forming processes at the land surface (including biological activity and hydrological processes) while tephras continued to be deposited— i.e. upbuilding pedogenesis. This process results in the deepening of the soil profile as a downward moving front, and reflects the influences of the land surface through time, as well as the interaction between layers. This top portion of the soil is characterised by a well-moderately drained topsoil (Lowe, 2019b) and a firm, poorly drained subsoil at about 30-50 cm— reflecting the properties of the Hamilton Ash. The boundary between the TVZ tephras and the Hamilton Ash is marked by the Rotoehu Ash, followed by an irregular lithological discontinuity (Lowe, 2019a). The Hamilton Ash beds are comprised of an approximately 3 m deep series of strongly weathered yellow, brown and red-ish clayey tephras, predominantly halloysite. The upper parts of the Hamilton Ash beds show evidence of prolonged periods at saturation or near-saturation as a result of perching on the 2bBt(f) paleosol horizon (Figure 3.3)— a slowly permeable, clay-rich paleosol (Lowe, 2019b). These soils are characterised by relatively high bulk density ( $\rho_b$ ) (around 1.2 kg/m<sup>3</sup>; Kuman, 2019), slow permeability and are vulnerable to treading damage when wet (Manaaki Whenua-Landcare Research New Zealand, 2021). The soil setting suggests that there is a sub-surface lateral flow along the top of the Hamilton Ash layer (where N would accumulate), through the riparian plots and towards the drain, as water would not be able to infiltrate past the impermeable Hamilton Ash layer, due to its high  $\rho_b$ . The area may remain saturated for several days a year, or near-saturation for months at a time during the winter time.



**Figure 3.3: Sequence of soils from representative soil layers at the experimental riparian plots. (A) soil core showing typical Mangatawhiri clay loam topsoil from control plot C2, 0-16 cm depth. (B) ~30 cm depth in control plot C2. (C) Hamilton Ash clay encountered at 50-60 cm interval in mānuka plot M1.**

As discussed in section 2.7, the physical characteristics of soil have a large impact on drainage. Observation of the soil profile reveals that each layer has very different properties, which will impact the movement of water and contaminants. The topsoil (Figure 3.3A) looks very permeable, has a good soil structure, and likely allows the easy movement of water. At around 30 cm depth, (Figure 3.3B), the soil becomes more clayey and sticky, with a dark orange colour. This is evidence of prolonged periods at saturation or near-saturation. At a depth of 50-60 cm, the clay becomes even thicker and paler and exhibits mottling, evidence of winter perching over this layer.

### **3.2. Overview of the monitoring design**

The monitoring design thus addresses each of the objectives outlined in section 1.2 (Figures 3.5, 3.5 & 3.6). N species in soil solution were sampled using suction cup lysimeters. The local hydrology was monitored using dip wells (to determine the depth of the water table and its direction of movement) and above- and below-canopy rain gauges. Soil moisture and electrical conductivity (EC) sensors were also deployed. Lastly, physical indicators of soil quality were quantified (macroporosity and bulk density). Each of these components are described in further detail in the following sections.

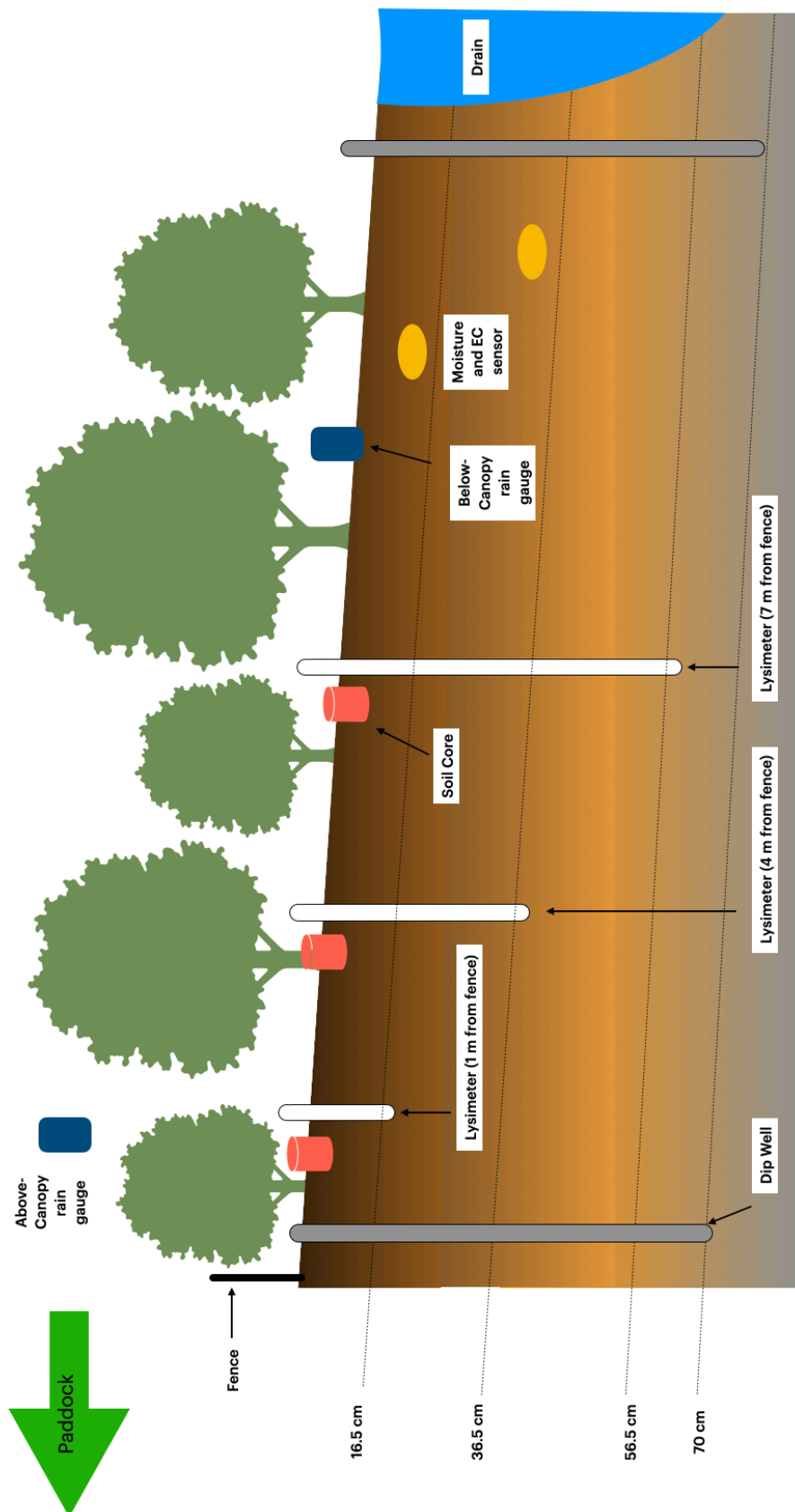


Figure 3.4: Schematic of monitoring design deployed in this study, showing lysimeters, dip wells, soil cores, rain gauges and soil sensors in a mānuka plot (not to scale).

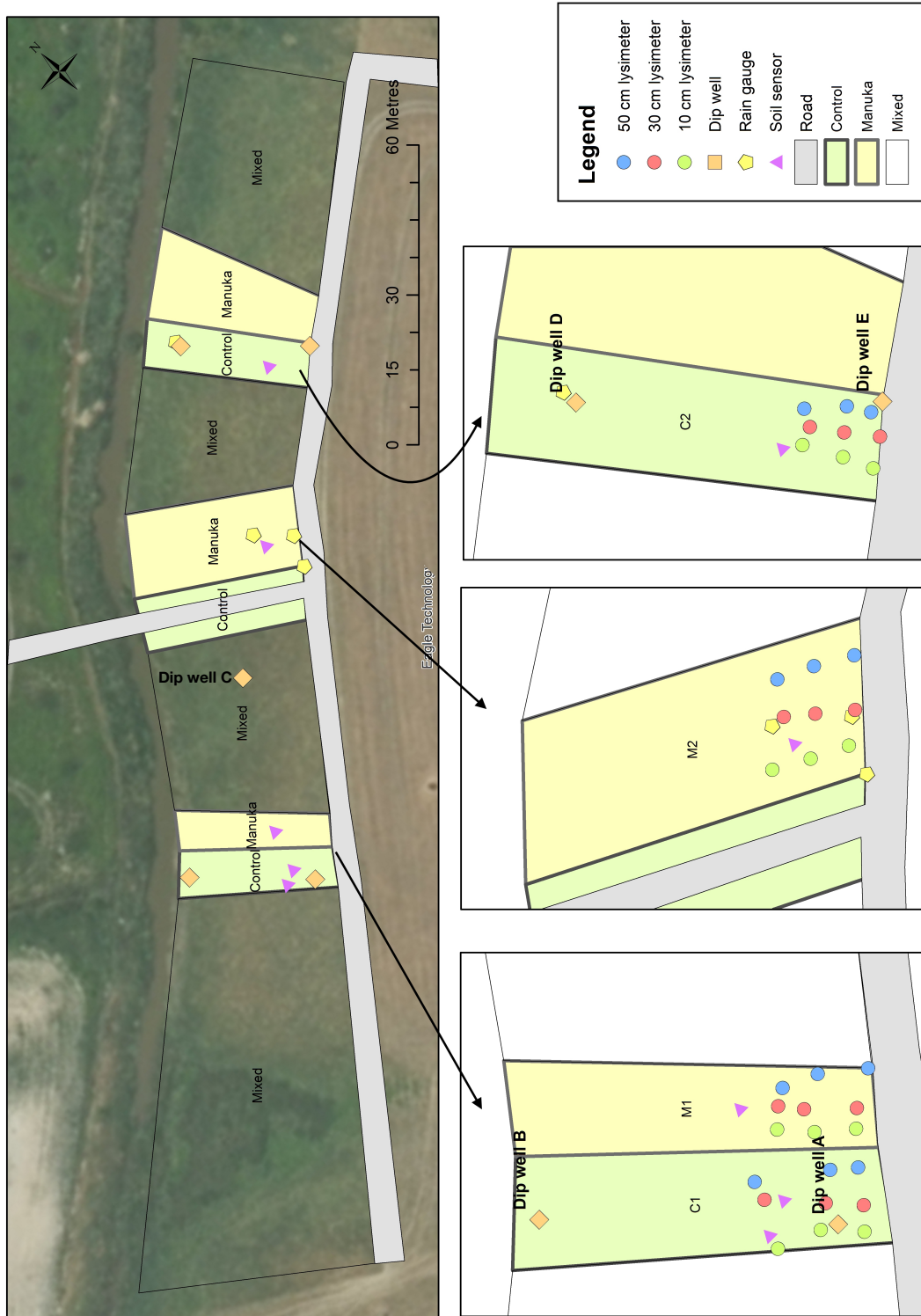
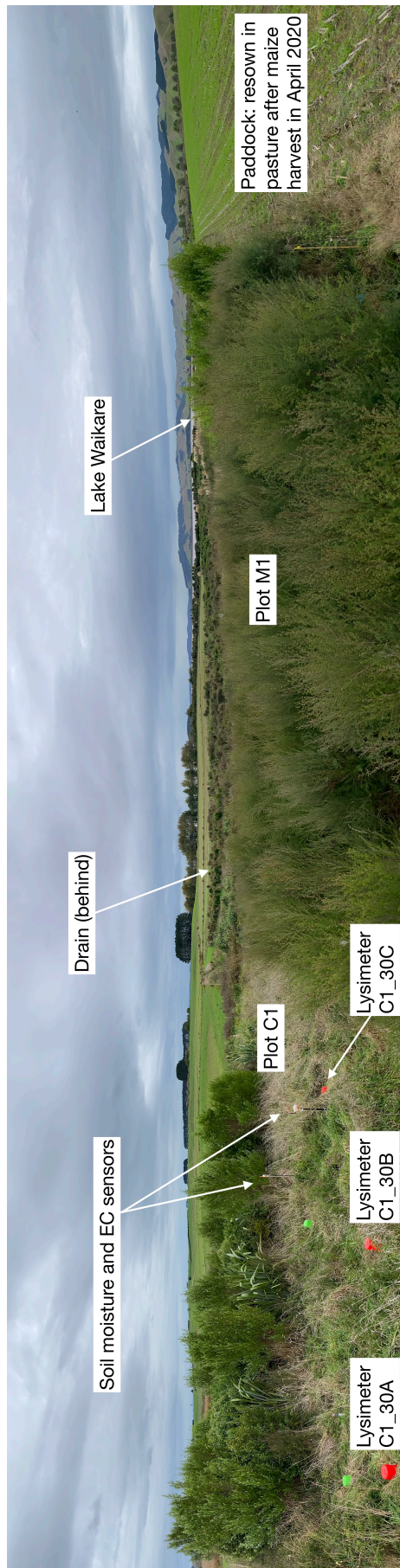


Figure 3.5: GIS map of the experimental riparian site picturing the boundaries of each experimental plot, lysimeters at different depths, dip wells, soil sensors and rain gauges.



**Figure 3.6: Panoramic photograph of monitoring setup on plot C1, showing the location of the experimental plots relative to the lake, drain and paddock. For lysimeter label references see Appendix 1.**

### 3.3. Soil physical properties

To test the hypothesis that mānuka would change the soil structure compared with the control, and potentially alter drainage and water/N movement, soil quality indicators macroporosity and bulk density were determined. Three 15 cm deep topsoil cores were taken from each of the experimental plots on March 25th 2021, in the same distribution as the suction cup lysimeters: 1 m, 3 m and 7 m from the fence (Figure 3.4).

First, the vegetation was pushed aside to reveal the bare soil underneath, and a standard sized soil core (15 cm diameter) was then placed on the soil surface, avoiding any large cracks. A serrated knife was used to cut into the ground surface and make room for the core to penetrate into the soil. The coring ring was manually pushed into the soil, making sure to exert even pressure on all sides and not disturb the inside of the core. A spade was then used to lift the whole area upwards and remove the sample with care. The bottom end of the core was trimmed to make sure it was flush with the core (flat ends) and then wrapped in plastic for transport and storage until analysis at Landcare Research by standard methods (Figure 3.7).

The results were then analysed in Excel with descriptive statistics and compared to NZ-specific soil quality targets (Sparling, 2008; Mackay, 2013), as well as to historical measurements from the same site taken in 2017— before the riparian buffer was established (collected by Matthew Taylor, Waikato Regional Council).



**Figure 3.7: Soil coring process. (A) digging the core into the soil with the help of a serrated knife. (B) the soil core, just removed from the ground. (C) wrapping the core for safe transport and storage.**

## **3.4. Hydrology**

### **3.4.1. Rainfall, soil moisture and electrical conductivity**

As emphasised by Parkyn et al. (2003), riparian vegetation type will only have an effect on N loads leached during wet conditions, but not in dry conditions. Thus, precipitation data from the weather station, automatic rain gauges and the soil sensors, was used to test this hypothesis. A weather station and automatic above- and below-canopy rain gauges were installed by Auckland University in June 2021. Soil moisture and EC sensors were installed in each experimental plot in December 2020 by Auckland University and ESR. These were installed at 15 cm depth and 30 cm depth. Both datasets were processed and plotted using R statistical software.

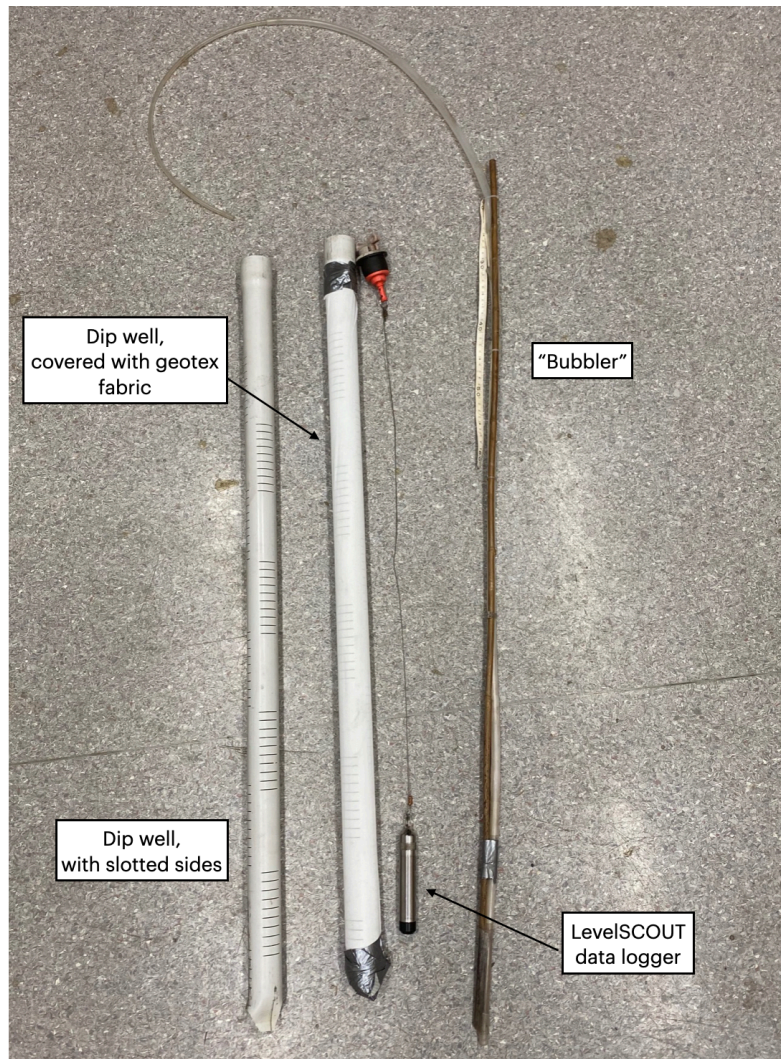
### **3.4.2. Water table monitoring**

Dip wells were deployed in the experimental plots at Nikau Farm with the purposes of:

- Ascertaining the depth of the water table and see whether it intersects with the lysimeters (turning the lysimeters into wells themselves),
- Verifying whether there is a perched water table at the site, or a transient water table above the Hamilton Ash layer,
- Determining the direction of water movement (i.e. diagonally from top left corner of the plot to bottom right, or straight from pasture to drain), and
- Providing useful information for other projects related to the Lake Waikare Learning Community (see section 2.8.3)

#### **3.4.2.1. Design**

The dip wells used in this study were made out of 40 mm wide PVC piping, with slotted sides, as to allow water to flow into them. The bottom ends of the instruments were heated with a heat gun and sealed to prevent soil from entering the tube during installation. The wells were also wrapped in geotex fabric to stop soil and insects from entering the well. The top end, protruding from the soil surface, was sealed with a rubber bung. One of the dip wells was equipped with an automatic data logger, which continuously collected data from one site (Figure 3.8).



**Figure 3.8: Dip wells**

### **3.4.2.2. Deployment**

Five dip wells were installed on March 25th 2021. On the day of installation, the soil was extremely dry and difficult to dig. Big cracks were noticed at some of the sites. They were installed on the four corners of the experimental plots used in this study, plus one in the middle (dip well C), containing the data logger (Figure 3.5).

To install them, the vegetation in the selected site was cleared by simply pushing it away, to reveal the bare soil underneath. A dutch auger of a similar diameter to the wells was subsequently used to dig a hole to about 65-70 cm depth (Figure 3.9), the depths which a perched water table was expected to occur over the Hamilton Ash. The well was then inserted in the hole and some of the removed soil was replaced around the well to make sure it stayed secure. GPS coordinates were then taken, as well as estimated altitude above sea level.



**Figure 3.9: Installation of dip well C. Pictured with dutch auger.**

### **3.4.2.3. Data collection**

The depth of the water table was measured during every site visit using a “bubbler”: a simple piece of bamboo with a tape measure attached to it, as well as a flexible piece of tubing. The tube is used to blow into and, when it touches the water table, the operator can hear bubbles, making note of the depth with the tape measure (Figure 3.10). The data collected by the automatic sensor was extracted twice (one month after deployment and at the end of the sampling period) (Figure 3.11).



**Figure 3.10: Collecting water table measurements at plot C2 using the “bubbler”**



**Figure 3.11: Extracting data from the LevelSCOUT data logger in dip well “C”**

#### **3.4.2.4. Data analysis**

Measurements from the LevelSCOUT data logger were calibrated to barometric pressure using information from nearby Kopuatai Bog monitoring station (provided by David Campbell, from the University of Waikato). The manual dip well measurements were then incorporated into the dataset and plotted using Excel.

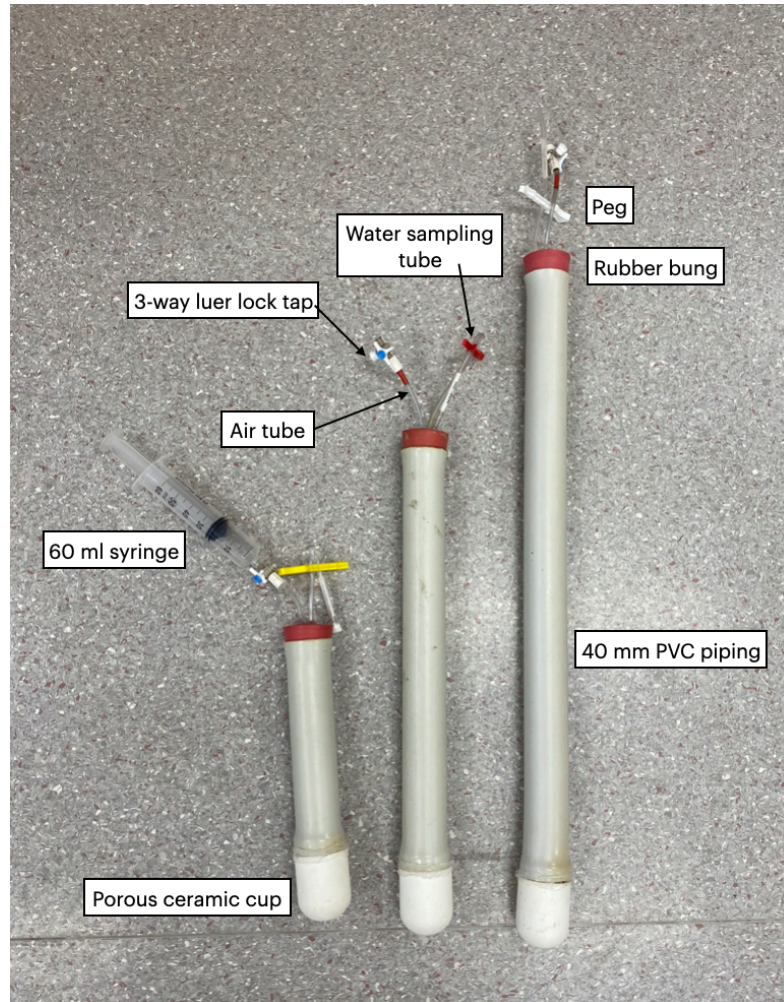
### **3.5. Nitrogen concentrations in pore water**

In this study, suction-cup lysimeters were used to collect pore water samples for measuring the concentration of TN, TKN and  $\text{NO}_3^-$  and  $\text{NH}_4^+$ . This method samples soil water under unsaturated conditions and from pores with longer residence times, meaning that the water sampled will have had more interaction with soil components than free-draining water would (under saturated flow) (Cabrera Corral et al., 2016) (section 2.5.1.3.1). The lysimeters were deployed to three depths to reach each of the key soil horizons in the profile; assumed to have different hydraulic characteristics (section 3.1.2). This allowed to obtain a better spatial resolution of N and water dynamics in this soil.

#### **3.5.1. Design**

The instruments utilised in this study were purpose-built and made of 6.5 cm long ceramic cups, attached to 4 cm wide PVC drainage pipes. The PVC pipes were cut to three different lengths (20, 40 and 60 cm) to reach the desired soil horizons. The piping was heated with a heat gun to expand and fit around the ceramic cups, forming a tight seal. To ensure that no air leaks existed, a silicone sealing agent was applied on the intersection. Rubber bungs were fitted with two pieces of 0.3 cm PVC tubing. The first reached from the bottom of the ceramic cup to 10 cm over the bung to collect sample with, and was closed with plastic pegs. The second piece of tubing was about 15-20 cm and was fitted with 3-way luer lock taps that allowed to vacuum air out of the instruments without letting air back inside. Silicone was also used around the bungs and tubes to ensure a tight seal and, in some cases, extra sealing was required between the 3-way taps and tube (Figure 3.12). The instruments were tested by submerging them in water and pressurising them so as to reveal any air leaks, evidenced by bubbles.

Subsequently, the lysimeters were vacuum sealed and left for 24 hours to ascertain they were functioning properly. They were left in water since they would otherwise become dry, and air could escape through the empty pores in the ceramic cup (Figure 3.13).



**Figure 3.12: Lysimeters used in this study (to sample 10-16.5 cm depth, 30-36.5 cm and 50-56.5 cm)**



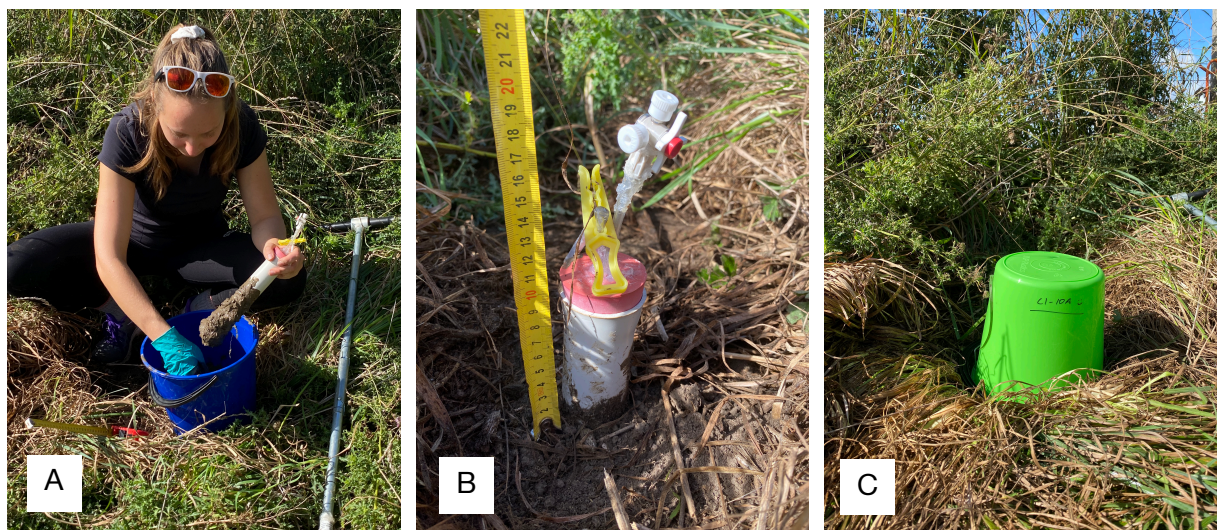
**Figure 3.13: Lysimeters being tested for air leaks and vacuum-holding capacity**

### **3.5.2. Deployment**

The suction cup lysimeters were installed in the experimental plots on April 19th and 20th, 2021. A total of 36 lysimeters were deployed in two paired sites, with nine in each plot (Figure 3.5). There were two different treatments (control and mānuka buffers), and pore water was collected at three different depths (approximately 10, 30 and 50 cm) and at three different distances from the fence (1, 4 and 7 m). This distance was chosen to get a good evaluation of changes in nitrate throughout the plot— as it is suggested that most of the nitrate is assimilated or denitrified in the first few meters of a riparian band (Fennessy & Cronk, 1997). The transects were separated from each other by approximately 2 m to get samples that are representative of the whole plot, and to avoid creating unnatural zones of dryness around the lysimeters, impacting on the results. In some cases, the 2 m distance was not exact as other instruments deployed in these plots had to be avoided (belonging to other studies). Each transect was set up to reach a different soil horizon of different characteristics: 10-16.5 cm (accounting for the length of the ceramic cup), 30-36.5 cm and 50-56.5 cm below the ground surface.

During installation it was important to minimise the disturbance of the site as much as possible (Nieminen et al., 2013b). Thus, holes of a diameter similar to that of the lysimeters were created with a dutch auger. It is critical that the ceramic cup makes

good contact with the surrounding soil, so the instruments were coated in a slurry of soil and water (Figure 3.14A). Most of the soil excavated during this procedure was also used to fill in any gaps that remained around the instrument (Figure 3.14B). Each lysimeter was then covered with a plastic bucket and secured with weed mat pegs to ensure they were safe from being damaged by the sun, or accumulated soil or insects (Figure 3.14C). This also made the instruments easy to locate in the tall grasses and very dense mānuka plots when sampling was to occur in likely cold and wet conditions. The buckets in each transect were also colour coded for ease of sampling: (green = 10 cm, red= 30 cm and blue = 50 cm).



**Figure 3.14: Lysimeter installation process (A) coating the instrument in a slurry of soil and water, (B) installed lysimeter, with most of the soil removed during excavation replaced around it, and (C) lysimeter covered by bucket for protection and ease of location**

Given that soils are remarkably heterogeneous, the decision on the number of samplers deployed was made around having a large enough number to capture a representative picture of the processes occurring in the plots while being mindful of how much disturbance was made to the site given the relatively small area of the plots (yielding unrealistic results). In this case, what took most importance was sampling frequency over number of sampling points. The instruments were left to settle for four weeks before the first sampling event as, during the augering of holes, a portion of soil was removed and replaced by the lysimeter. This period theoretically allowed the soil to naturally re-arrange itself around the instruments, and minimise the creation of preferential flow pathways along the length of the lysimeters. Moreover, given that there is not a constant flow of water through the cup, resulting in sorption, it is

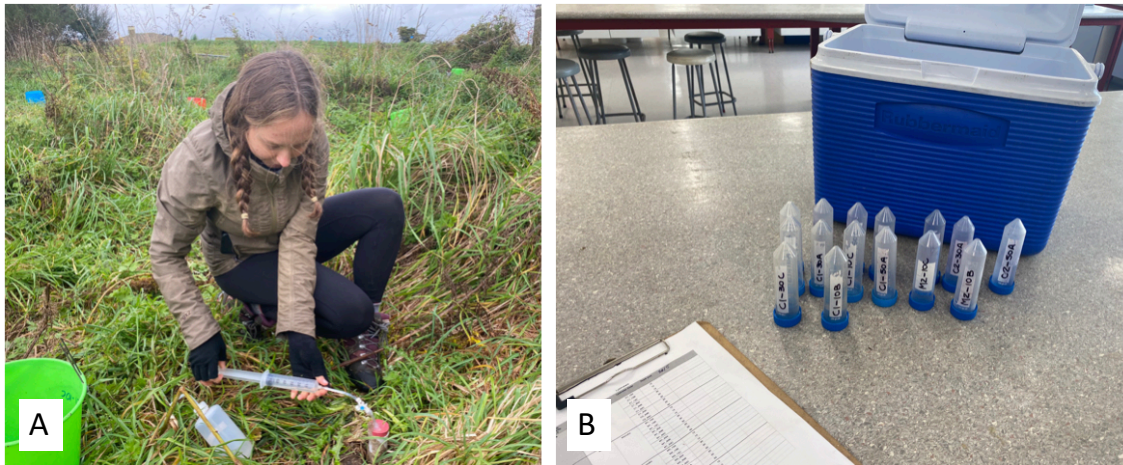
recommended that the first water sampled is discarded (Weihermüller et al., 2007; Chibnall, 2013).

### **3.5.3. Sampling**

Suction cups were sampled between May and July 2021, given that most leaching losses occur when the winter rain bring soils up to, or near field capacity and there is an abundance of NO<sub>3</sub><sup>-</sup> in soil solution due to slower plant growth (McLaren & Cameron, 1996). Sampling occurred every 1-2 weeks (Weihermüller et al., 2007; Zhou et al., 2010; Clague et al., 2013; Nieminen et al., 2013a; Clague et al., 2015; Sigua et al., 2017).

As most N moves through the soil in soil solution, particularly as through-flow after a rain event, sampling occurred after precipitation events. The site was visited the day before a rain event was forecast and any water in the lysimeters was emptied. A suction of -50 kPa was then applied with a 60 ml syringe (Clague et al., 2015; Cabrera Corral et al., 2016). This was achieved by removing 324, 717 and 1109 ml of air (equivalent to 3, 6 and 9 syringe-fulls of air for each lysimeter size respectively), and the 3-way tap. The site was revisited approximately 24 hours after the event and samples were collected (Figure 3.15). Syringes were rinsed with distilled water in between samples to avoid cross-contamination. The sample volume obtained was recorded on the field using graded falcon tubes (Nieminen et al., 2013a). Suction cup lysimeters best suit experimental designs that have short sampling intervals due to the need of having the instruments hold a vacuum for a length of time, and the tendency of NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup> to be altered by biological processes such as nitrification and assimilation (Weihermüller et al., 2007).

Transport to the laboratory was carried out in a closed cooler box with ice packs in an effort to maintain the samples as intact as possible and prevent chemical and biological alteration from happening (Weihermüller et al., 2007; Nieminen et al., 2013a). Samples were then frozen at -20 °C until the time of analysis (Figure 11). Overall, seven sampling events were carried out.



**Figure 3.15: (A) Sampling from a 10 cm depth lysimeter on plot C2 and (B) Samples back in the lab.**

### **3.5.4. Laboratory analysis**

The pore water samples were sent to Analytica Laboratories for analysis using a flow-injection analyser, with a detection limit of 0.002 mg/l. They were analysed for:  $\text{NO}_3^-$ ,  $\text{NO}_2^-$ ,  $\text{NH}_4^+$  and Total Kjeldahl N (TKN). TN was calculated as the sum of all N species. As recommended by Nieminen et al. (2013b), the samples from each depth and sampling occasion were analysed separately to conserve resolution about spatial and temporal variations.

### **3.5.5. Data analysis**

Data was analysed through analysis of variance (ANOVA) using R statistical software to test the statistical significance of the observed differences in N concentrations between distances from the fence, depth, plots and vegetation type. P-values of less than 0.05 were reported as statistically significant. Leaching losses on each sampling date were calculated using the sum of TN concentrations in each plot for each date and multiplied by the volume of sample collected.

# Chapter 4

## Results

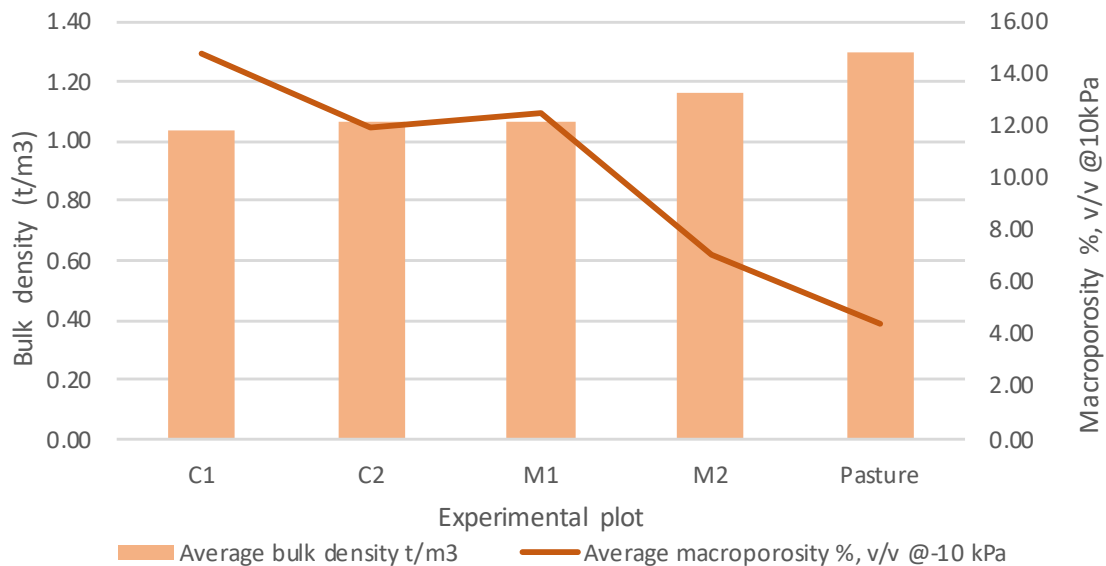
### 4.1. Soil physics

In this study, bulk density ( $\rho_b$ ) and macroporosity (MP) results were compared to the NZ soil quality targets for pasture, being that the previous land use at this location (Table 4.1). This allowed for a fairer comparison between vegetation types than comparing the soil physics results to targets for forested systems.

**Table 4.1: Bulk density ( $\rho_b$ ) and macroporosity (MP) results for three different soil cores at each experimental plot, compared to samples from 2017 (pasture), before the riparian band was planted. Conditional highlighting shows whether the values are within (green) or outside (red) target ranges. Targets from Sparling et al. (2008) were used for  $\rho_b$ , while those from Mackay et al. (2013) were used for MP (revised values).**

Core #	<u><math>\rho_b</math> (t/m<sup>3</sup>)</u>				<u>Macroporosity (v/v%, @-10 kPa)</u>			
	1	2	3	Average	1	2	3	Average
C1	0.97	1.16	0.97	1.03	17.10	11.70	15.60	14.80
C2	1.09	1.11	1.00	1.07	12.20	13.50	10.30	12.00
M1	1.06	1.10	1.04	1.07	12.90	10.40	14.30	12.50
M2	1.14	1.22	1.13	1.16	9.20	4.40	7.70	7.10
Pasture	1.35	1.38	1.18	1.30	3.20	2.60	7.50	4.43
Average control				1.05				13.40
Average mānuka				1.11				9.81
Average riparian				1.08				11.60

Figure 4.1 presents the average results from Table 4.1 for comparison between plots. The figure clearly shows the inverse relationship between  $\rho_b$  and MP (e.g. C1 has the lowest average  $\rho_b$  and highest average MP, while pasture had the highest average  $\rho_b$  and lowest average MP). The average  $\rho_b$  between all riparian plots (control and mānuka) was 1.08 t/m<sup>3</sup> compared to an average 1.30 t/m<sup>3</sup> in the paddock, representing a 17% change. The average MP between all riparian plots was 11.6% compared to 4.43% in the paddock, showing a change of 38%.

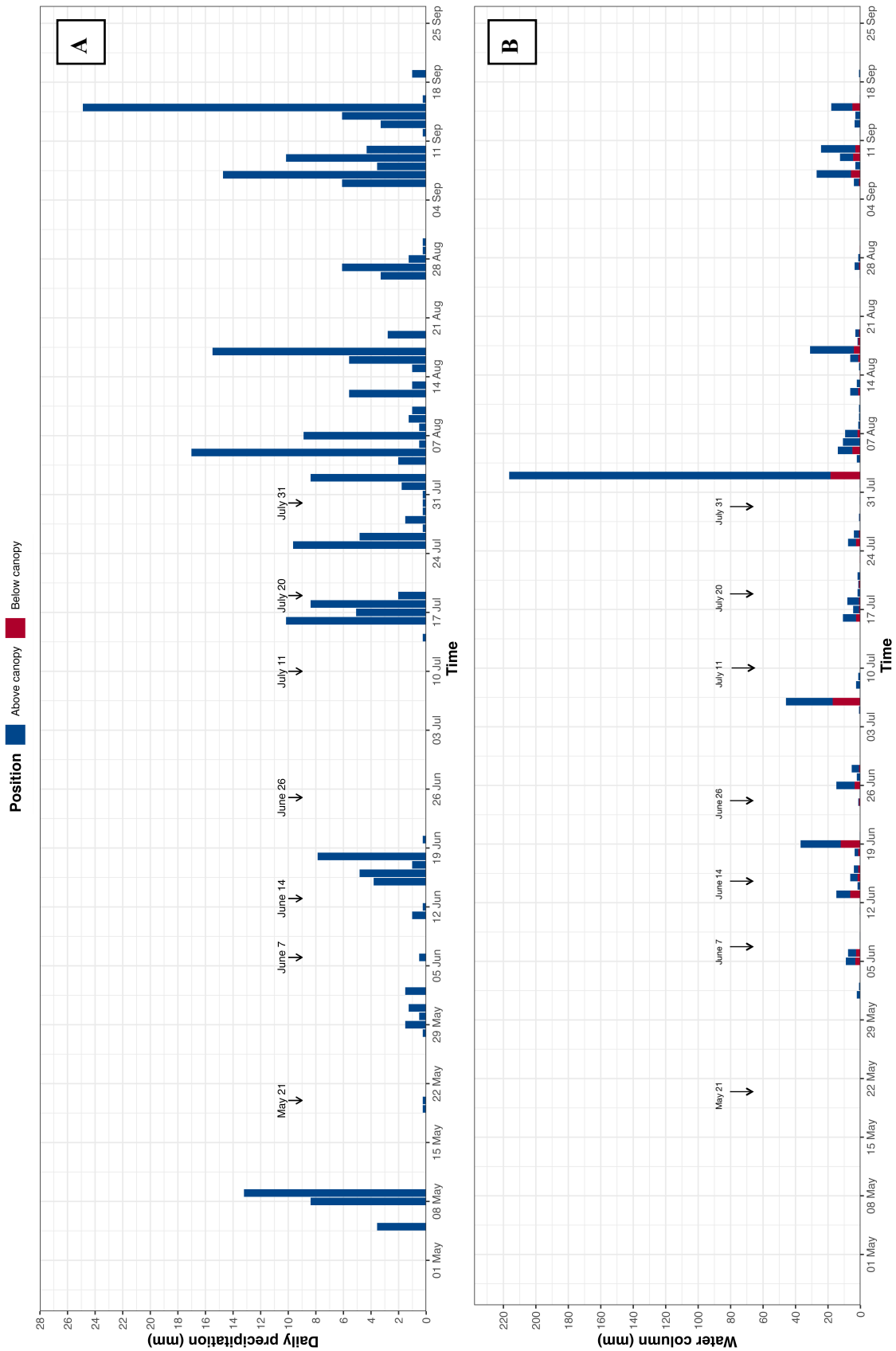


**Figure 4.1: Average bulk density ( $\rho_b$ ) and macroporosity (MP) at control and mānuka experimental plots, calculated from three different soil cores.**

Plots C1, C2 and M1 showed relatively similar physical characteristics and mostly fall within target ranges. Plot M2, in contrast was the most compacted— with higher  $\rho_b$  and lower MP compared to the other riparian plots (1.16 t/m<sup>3</sup> and 7.1% respectively). Plot C1 was the least compacted, with the lowest  $\rho_b$  and highest MP (1.03 t/m<sup>3</sup> and 14.8% respectively). The pasture samples were the most compacted overall, with MP being well below the lower target of 12% (4.43% average). The two plots under control and the two under mānuka also differed from each other. C1 appears to be less compact than C2, while M1 appears to be less compact than M2.

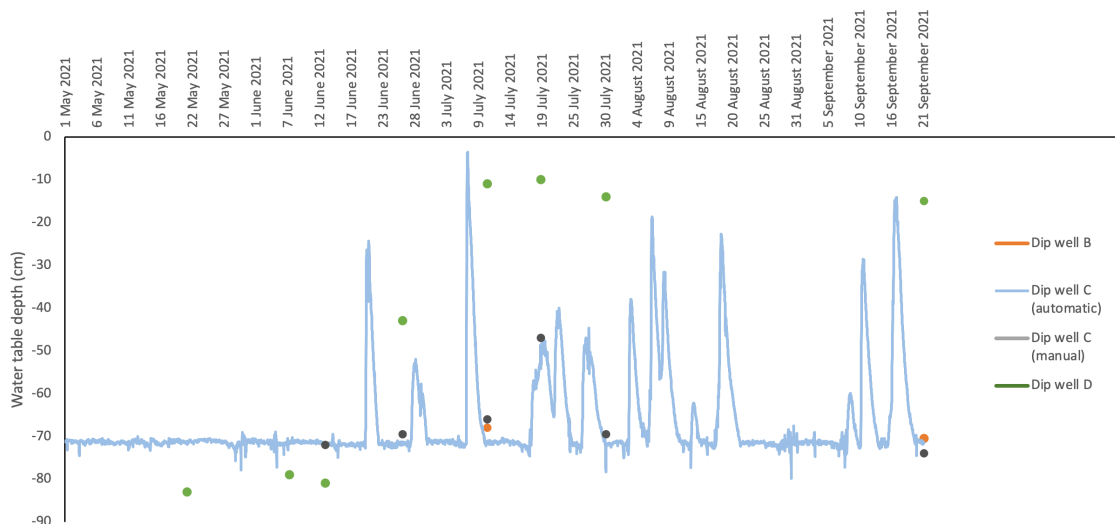
## **4.2. Hydrology: rainfall, water table, soil moisture and electrical conductivity**

Figure 4.2 presents the rainfall data as collected by the weather station next to the riparian plots, and by automatic rain gauges above- and below- mānuka canopy. It is important to note that the rain gauge measurements are given in mm water column, which is not the daily precipitation as measured by the weather station (in mm). Therefore, the data sets are not compatible. The red bars in Figure 4.2B, indicate below-canopy rainfall and are a subset of the blue bars. The average difference in above-below canopy rainfall was 63.9%, while the cumulative rainfall difference (over 3.5 months approximately) below the mānuka canopy was 76.3% lower than above-canopy.



**Figure 4.2: A) Daily precipitation (mm) as measured by weather station next to the experimental riparian plots at Lake Waikare. Pore water sampling dates indicated by arrows. B) Daily rainfall expressed in mm water column. Note that during late June and early July, the weather station was offline due to technical issues, which is why some data is missing, which appears in the rain gauge graphs.**

Figure 4.3 shows the depth of the water table through time, with both manual and automatic measurements, which coincide well. The peaks in water table height at dip well “C” aligned with peaks of rainfall events. Dip well “C” aligned with peaks of rainfall events. Dip well “D”, which was lower down the slope and next to the drain, was the only other one which consistently had water within it. The water level at this location remained closer to the surface (up to 10 cm depth), and seemingly for longer (although the frequency of the data is significantly smaller). For example, on July 20th, the water table depth at “C” was 52 cm while at “D” it was 10 cm. The water table at “D” remained close to the surface throughout September (at 15 cm depth), when measurements finalised. Closer to the fence, and to the locations of pore-water sampling, the water table remained relatively deep and was only measured on dip well “E” (in plot C2) once on July 20th, at 64.5 cm depth— which is almost within the reach of the 50cm lysimeters (sampling to 56.5 cm depth approx.).



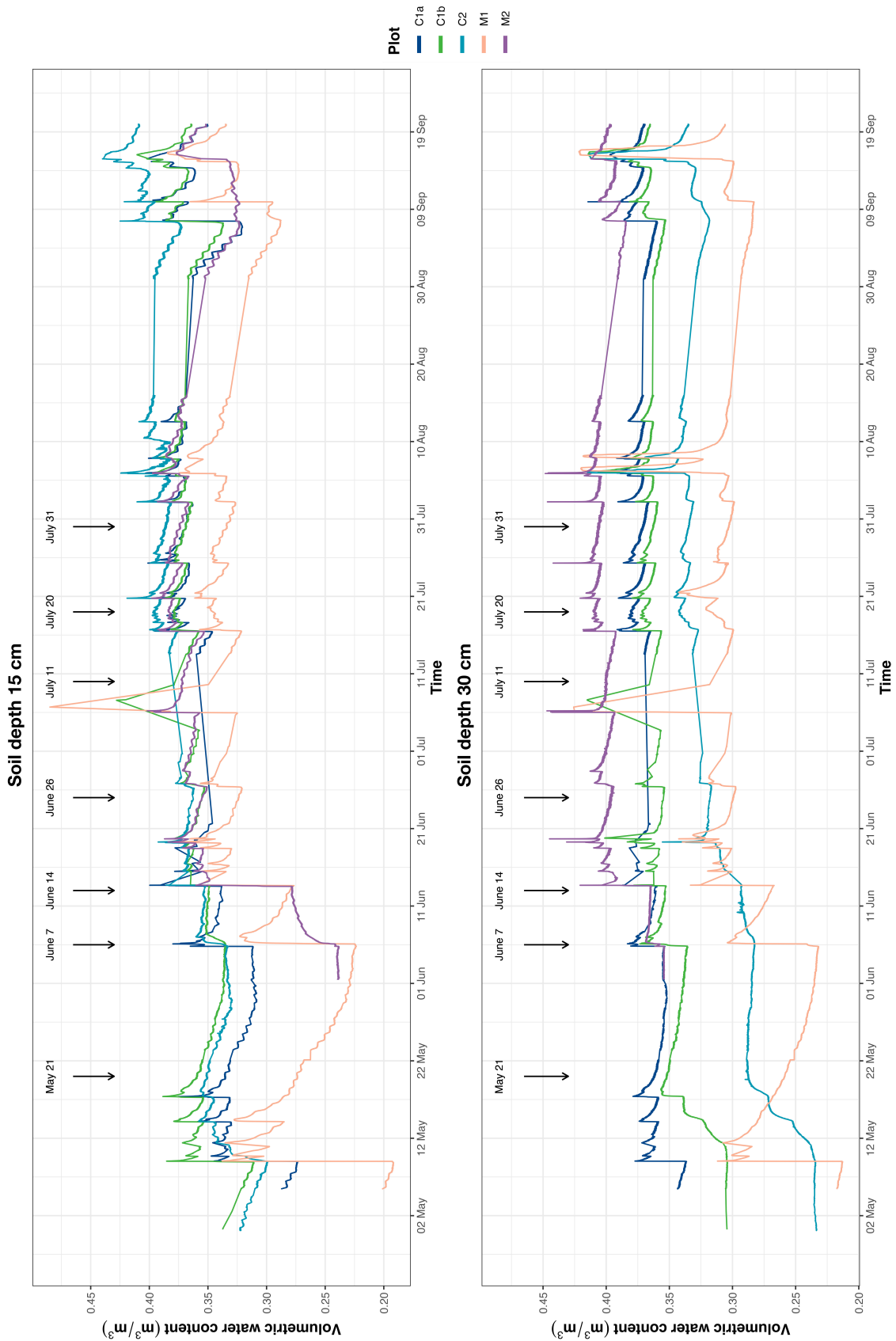
**Figure 4.3: Water table depth (cm) as measured manually and by a LevelSCOUT data logger at the experimental riparian plots in Lake Waikare.**

Throughout the sampling period, soil moisture (VMC) and electricity conductivity (EC) were monitored at 15 and 30 cm depths (Figures 4.5 and 4.6). Early in the sampling season (between May and June) there were a series of small and sporadic rain events, with daily rainfall mostly remaining below 2 mm (with the exception of two larger events on May 8-9 and June 15-18). These events align well with spikes in VMC. EC and VMC are also well correlated. The flat lines during periods such as between 26th June and 11th July indicates that no data was recorded due to connection and technical problems.

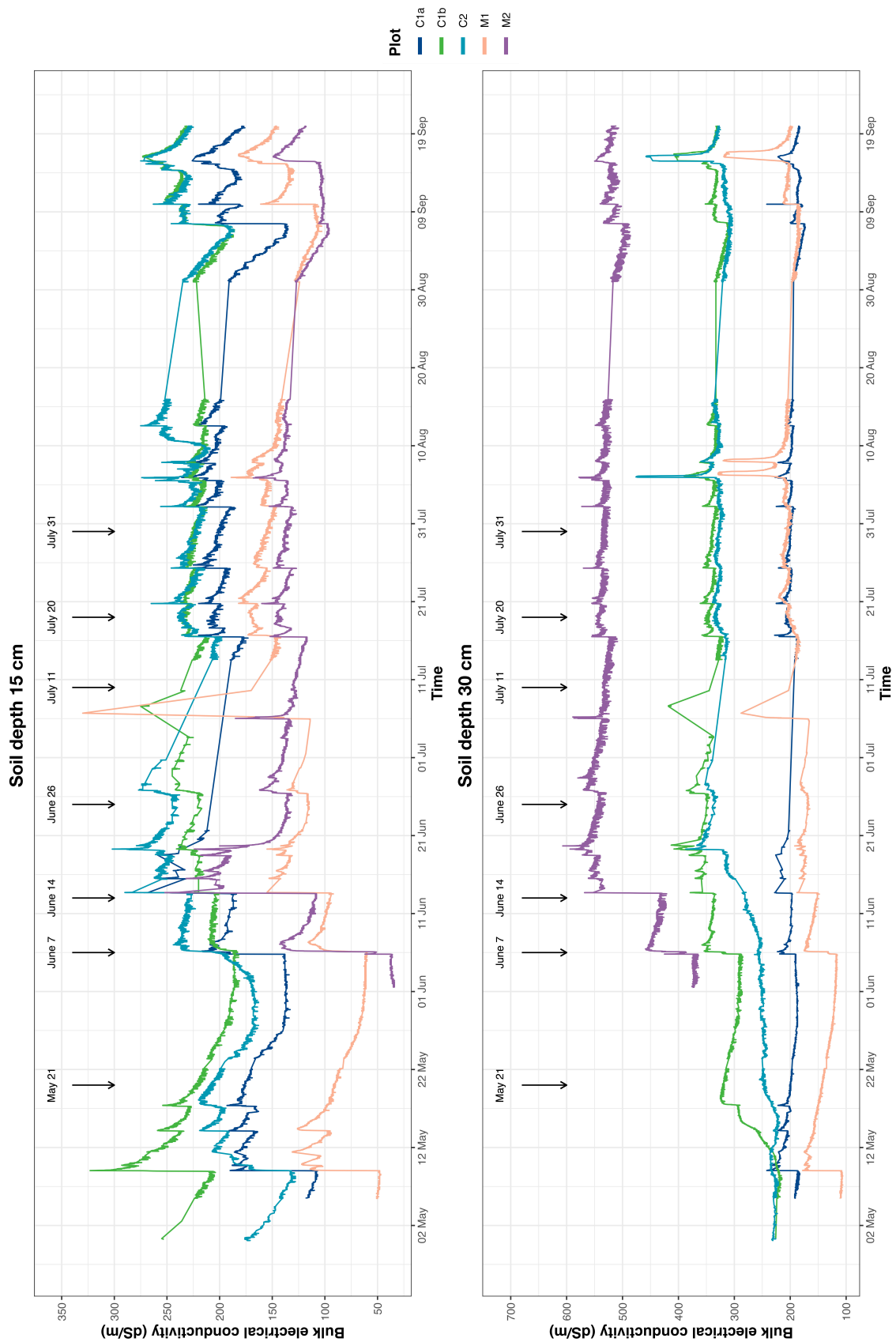
At 15 cm depth the control plots had similar VWC during May, while plot M1 remained markedly drier (e.g. 0.2 m<sup>3</sup>/m<sup>3</sup> on May 4th in M1 compared to 0.32 m<sup>3</sup>/m<sup>3</sup> in plot C1). The soil sensor at plot M2 was offline during May, but presumably would have behaved similarly to M1. In mid-June the VWC for all plots became similar, although plot M1 remained notably drier and drained to a larger extent by comparison. Plot M2 remained similar to control plots, possibly because it was lower on the slope. C2 was the wettest by mid-August.

There was more variation in VWC between the plots at 30 cm depth, pointing to a difference in water table height and/or soil physical properties (including both texture and structure). At 30 cm depth plot M2 had the highest VWC throughout the whole period, even more so than C2 which was lowest down the slope. This coincides with the higher EC at this plot as well. M1 was the driest plot at this depth, and also had the lowest EC.

EC at 30 cm depth was remarkably higher than at 15 cm depth (note axis scales on Figure 64.). For example, on June 14th, M2 measured more than double the EC at 30 cm compared to at 15 cm (550 and 250 dS/m respectively). At 15 cm depth, EC was consistently higher under the control plots. In contrast, at 30 cm depth, M2 had the highest EC out of all of the plots, while M1 and C1a had the lowest. It is interesting to note the differences in EC between the two sensors in plot C1, which show that there is high spatial variability within plots as well. Sensors C1a and C1b measured similar VMC at 30 cm but EC was more variable— pointing to the heterogeneity of soil and contaminant transfer pathways.



**Figure 4.4: Measurements for volumetric water content (m<sup>3</sup>/m<sup>3</sup>) from soil sensors deployed at the experimental plots in Lake Waikare (15 and 30 cm depth). Pore water sampling dates indicated by arrows.**



**Figure 4.5: Measurements for bulk electric conductivity (dS/m) from soil sensors deployed at the experimental plots at Lake Waikare (15 and 30 cm depth). Pore water sampling dates indicated by arrows. Note difference in y-axis scale between 15 and 30 cm depth.**

### **4.3. Nitrogen movement**

#### **4.3.1. Volume of pore water samples**

The volumes obtained from the soil pore water samplers (suction cups) are shown in Appendix 1. These volumes are only an indication, instead of a real quantification of potential leaching volume. The nature of suction cup technology is that it is a qualitative measure, not a quantitative measure. The objective was to obtain enough sample for the analysis of N, and not necessarily to quantify drainage. However, given the inability to extract samples from some plots during the first month, these results reveal interesting trends that are relevant to the present discussion on N movement.

During the first three sampling events (late May to mid-June), more samples were obtained from the control plots (particularly at 10 cm depth), and more so from C2 than from C1— even though the plots had similar VWC at 15 cm and there was lower VWC at 30 cm in C2 (Figure 4.4). Only five samples were obtained from both of the mānuka plots during this time, and the volumes were insufficient for laboratory analysis (<25 ml). The soil sensors showed that plot M2 had relatively high VWC at 30 cm depth (compared to control plots) but still did not yield samples. During the last four sampling events, samples were obtained more consistently from all plots. Relatively small amounts of sample were obtained from 30 and 50 cm depth in plot M2, although more sample was extracted from the 10 cm depth. On the 20th of July, the lysimeters at 50 cm depth in plot C1 and M1 had >300 ml of sample in them. It may be that the water table was being sampled at this point, although the water table was not observable from dip well “A” (Figure 4.3). In summary, more samples were obtained from the control plots, and more from M1 than from M2. In general, the suction cups in the topsoil produced more sample than those at deeper profiles (except when the saturated zone was being sampled). The full dataset is presented in Appendix 1.

#### **4.3.2. Cumulative N extracted throughout sampling period**

Table 4.2 shows the cumulative TN extracted from each plot and the average per vegetation type throughout the sampling period (i.e. concentration of TN \* volume of sample obtained). As discussed in following section, N concentrations under the mānuka plots tended to be higher than under the control plots (full N dataset in

Appendix 2). However, the amount of sample extracted was significantly smaller— in other words, there was potentially less drainage under the manuka plots (although drainage was not modelled during the experiment). Overall, 20.93% more N was extracted from the control plots (Appendix 5).

**Table 4.2: Cumulative TN extracted from each plot and each vegetation type (mg/l N) throughout the sampling period.**

Plot	Sampling date						
	21 May	7 June	14 June	26 June	11 July	20 July	31 July
C1	0.07	0.53	0.91	2.43	4.35	6.58	7.02
C2	1.16	2.60	4.47	6.33	8.72	11.63	13.19
<b>Average Control</b>	<b>0.62</b>	<b>1.57</b>	<b>2.69</b>	<b>4.38</b>	<b>6.54</b>	<b>9.11</b>	<b>10.11</b>
M1	-	-	-	1.65	3.18	7.14	9.90
M2	-	-	-	1.42	3.03	3.98	6.08
<b>Average Mānuka</b>	-	-	-	<b>1.54</b>	<b>3.11</b>	<b>5.56</b>	<b>7.99</b>
Control-Mānuka				2.85	3.43	3.55	2.12
% difference				64.95	52.49	38.93	20.93

### 4.3.3. Factors determining N concentrations

Table 4.3 shows the variables considered in this study (plot, vegetation, depth, distance from fence and sampling day) and their statistical effect on the different N species (TN, TKN, NO<sub>3</sub><sup>-</sup> and NH<sub>4</sub><sup>+</sup>) measured in pore water samples. Here, TKN represents dissolved organic N (<45 μm).

TN is affected by all variables analysed, in significance order *distance from fence* > *depth* > *plot* > *vegetation*. It is important to note the strong significance of the interactions between variables, i.e Vegetation \* distance from fence, Vegetation\* depth, Distance from fence\* depth, and Vegetation\* distance from fence \* depth. These interactions are better represented in the graphs in Appendix 3, with a 3-dimensional

view of TN and TKN concentrations through depth and distance from fence for each plot.

NO<sub>3</sub><sup>-</sup> and TN were highly correlated (R<sup>2</sup>= 0.99), while TKN and TN showed no correlation (R<sup>2</sup>= 4x10<sup>-6</sup>) (Appendix 4a). This is because in the samples with the highest TN concentration, NO<sub>3</sub><sup>-</sup> was the main species, making up to 99% of the TN. In contrast, TKN was the main N species (over 90%) in those samples with the lowest TN concentration (see Appendix 4b). The average TN concentration found, regardless of plot, vegetation, distance or depth was 5.57 mg/l. For NO<sub>3</sub><sup>-</sup> it was 4.26 mg/l.

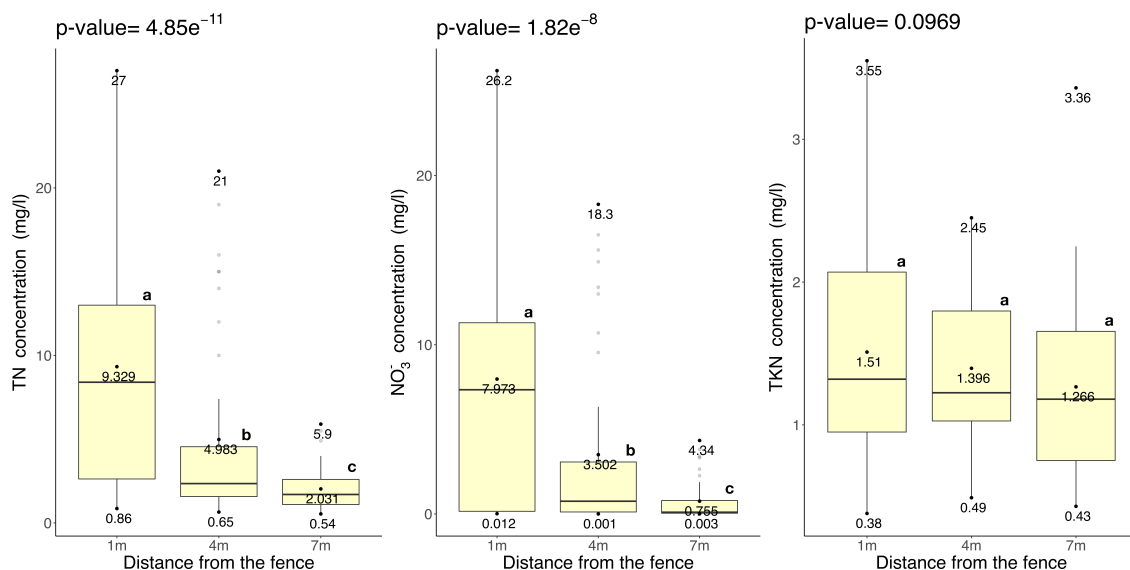
**Table 4.3: ANOVA results of significant interactions between variables and N species as indicated by p-values. \*\*\* p < 0.001, \*\* p < 0.01, \* p < 0.05, ° p < 0.1**

N Species	Distance from fence	Depth	Plot	Vegetation	Distance from fence * Depth	Distance from fence * Vegetation	Vegetation * Depth	Distance from fence * Depth * Vegetation
TN	***	***	**	°	***	***	**	**
NO <sub>3</sub> <sup>-</sup>	***	***	**	*	**	***		
TKN	°	***	*	*	**			
NH <sub>4</sub>		*						

#### 4.3.3.1. Influence of distance from fence on N

ANOVA analysis identified that distance from the fence exerted a significant impact on N concentrations. Both TN and NO<sub>3</sub><sup>-</sup> were highest at 1 m from the fence and declined progressively at 4 m and 7 m (e.g. from a mean of 9.32 mg TN/l to 2.03 mg TN/l; Figure 10). Statistical analysis found that distance had some influence on TKN concentrations as well, although groups were not significantly different from each other. Figure 10, however, shows that TKN concentrations may decrease farther into the riparian buffer, with the median mean and interquartile ranges both diminishing (e.g. mean was 1.51 mg/l at 1 m compared to 1.26 mg/l at 7 m).

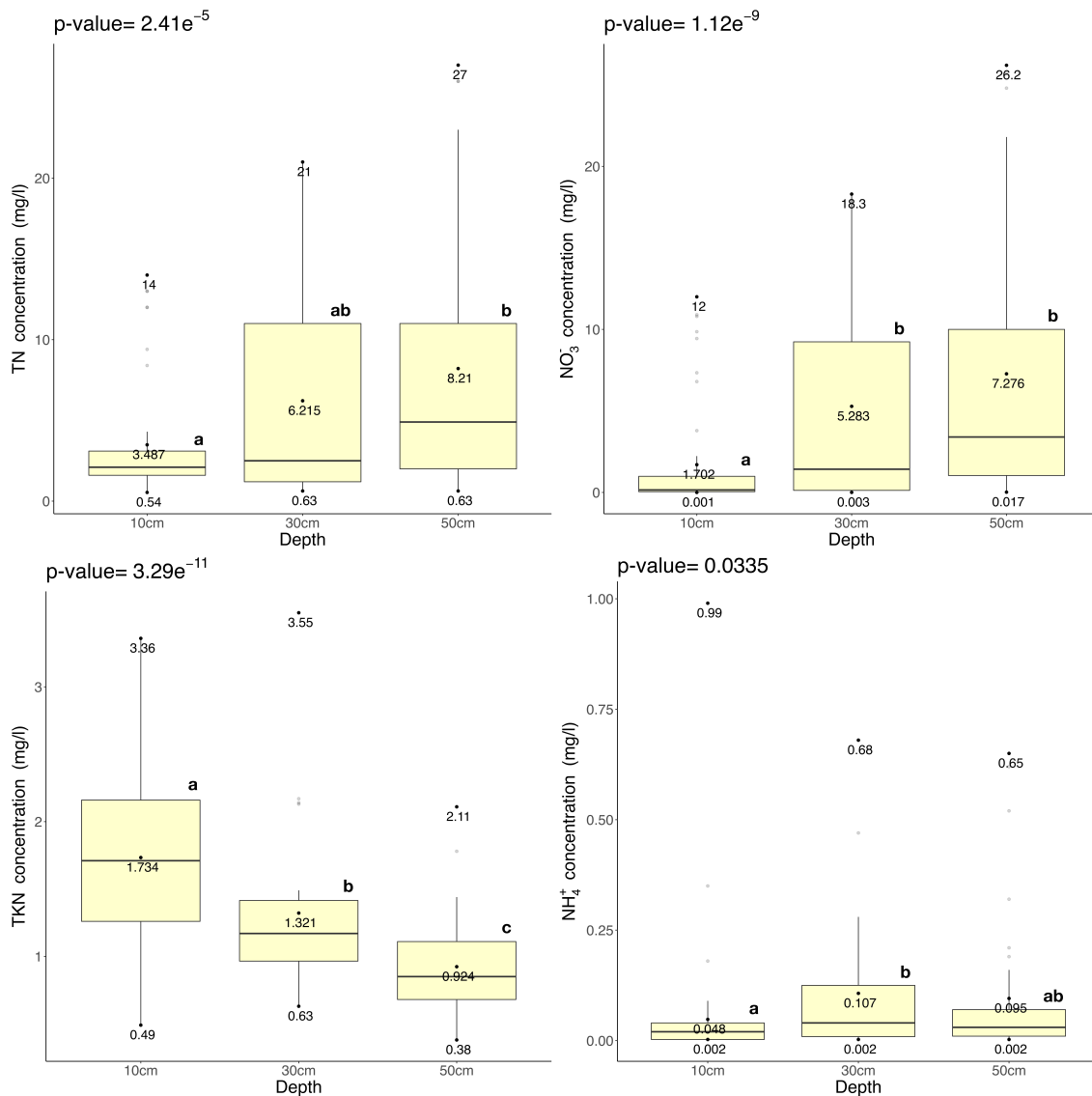
Distance from the fence also interacted with depth and vegetation, impacting TN and NO<sub>3</sub><sup>-</sup> concentrations (see Appendix 3 for a 3D view of interactions): there was more TN at 1 m from the fence and 50 cm depth, and more TN at 1 m from the mānuka plots, compared to the controls.



**Figure 4.6: Box plots displaying differences in TN, NO<sub>3</sub><sup>-</sup> and TKN at different distances from the fence, including the mean value. Graphs display the median, 25th and 75th percentiles, and mean, minimum and maximum values. Letters indicate groups that are significantly different (p-value < 0.05). Note different scales on y-axis for TKN. Note that statistical analysis were performed on log transformed data, not on the raw data shown in these graphs.**

#### 4.3.3.2. Influence of depth on N concentrations

Figure 4.7 displays the distribution of N species throughout the soil profile. TKN was the species most impacted by depth: concentrations were highest closer to the surface — at 10 cm, the mean concentration was 1.73 mg/l, while at 50 cm it was 0.92 mg/l. In contrast, NO<sub>3</sub><sup>-</sup> concentrations increased with depth, from a mean of 1.70 mg/l at 10 cm to 7.27 mg/l at 50 cm. Statistical tests showed that each depth had a significantly distinct concentration of TKN and NO<sub>3</sub><sup>-</sup>. For TN, however, depths 10 cm and 50 cm were significantly different from each other, while 30 cm was a “middle point” or “transitional” layer. ANOVA showed that depth had a significant impact on NH<sub>4</sub><sup>+</sup> (p-value= 0.034), with the 10 and 30 cm layers being different from each other. However, NH<sub>4</sub><sup>+</sup> comprised a small proportion of N in this system compared to other species (Appendix 2). Depth also had a strong interaction with vegetation type (Appendix 3): In general TN was higher in deeper horizons, but at 50 cm it was higher in the control plots compared to mānuka.

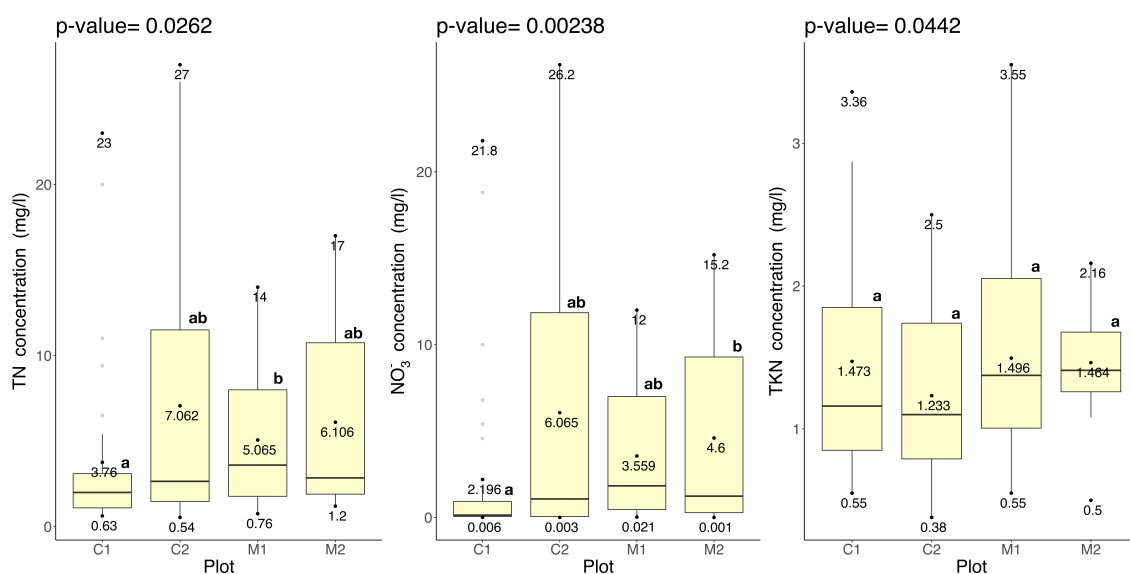


**Figure 4.7: Box plots displaying differences in TN, NO<sub>3</sub><sup>-</sup>, TKN and NH<sub>4</sub><sup>+</sup> at different depths within the soil profile, including the mean value. Graphs display the median, 25<sup>th</sup> and 75<sup>th</sup> percentiles, and mean, minimum and maximum values. Letters indicate groups that are significantly different (p-value < 0.05). Note different scales on y-axis for TKN and NH<sub>4</sub><sup>+</sup>. Note that the statistical analysis were performed on log transformed data, not in the raw data shown in these graphs.**

#### 4.3.3.3. Influence of plot on N concentrations

Plot impacted NO<sub>3</sub><sup>-</sup> more than TN by an order of magnitude (p-values = 0.0023 and 0.026 respectively) (Figure 4.8). NO<sub>3</sub><sup>-</sup> concentrations were significantly different between plots C1 and M2, with the latter having more NO<sub>3</sub><sup>-</sup>. C1 is highest on the slope, having a relatively lower water table, although VWC was similar to other plots (Figure 4.4). Plot M2 is lower on the slope and maintained relatively high VWC throughout the season (particularly at 30 cm depth, which coincides with this plot having the highest EC at this depth). However, the average NO<sub>3</sub><sup>-</sup> concentration at M2 is lower than that of

plot C2 (4.6 compared to 6.06 mg/l). In terms of TN, plots C1 and M1 were the most different, with the manuka plot having more TN (average 3.76 and 5.06 mg/l respectively)— which is noteworthy since they are adjacent. Plots M2 and C2 were similar, where M2 had a lower mean concentration of  $\text{NO}_3^-$  (4.6 mg/l) than C2 (6.07 mg/l). For TKN, ANOVA results returned a significant impact, although further analysis did not recognise differences between the plots.



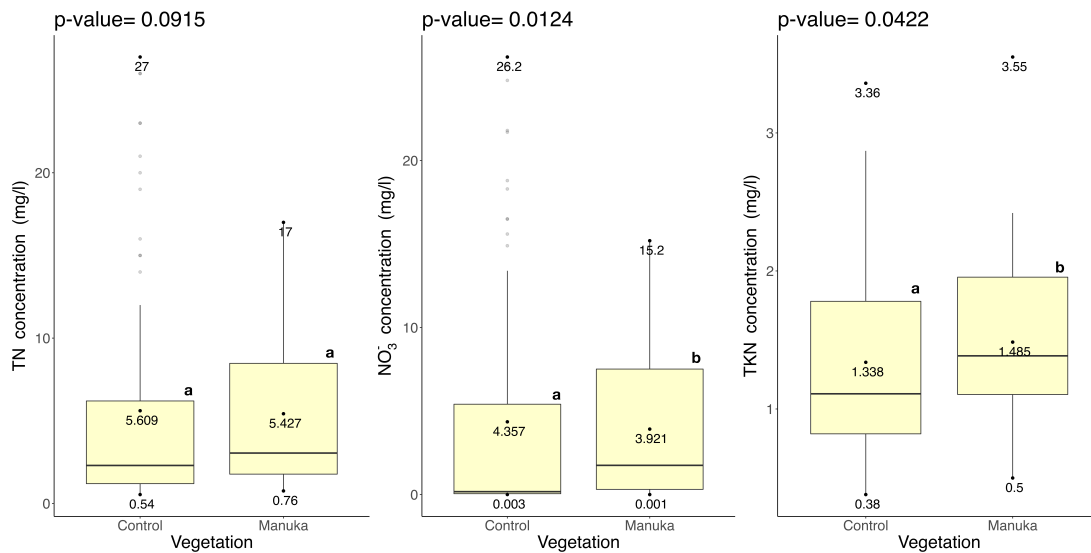
**Figure 4.8: Box plots displaying TN,  $\text{NO}_3^-$  and TKN concentrations in each plot, including the mean value. Graphs display the median, 25<sup>th</sup> and 75<sup>th</sup> percentiles, and mean, minimum and maximum values. Letters indicate groups that are significantly different (p-value <0.05). Note that the scale on y-axis for TKN is one order of magnitude lower than for TN and  $\text{NO}_3^-$ . Note that the statistical analysis were performed on log transformed data, not in the raw data shown in these graphs.**

#### 4.3.3.4. Influence of vegetation on N concentrations

There were statistically significant differences in N species concentrations between vegetation types for TN,  $\text{NO}_3^-$  and TKN (Figure 4.9). The most significant difference was for  $\text{NO}_3^-$  concentration, where the median was higher under the mānuka plots. Vegetation also had a significant, albeit smaller, impact on TKN concentrations— where again, the mānuka plots had higher concentrations. Statistical analysis found no difference in TN concentrations between the mānuka and control plots, although ANOVA showed a small impact. Note too, that the range of TN and  $\text{NO}_3^-$  concentrations

found was large for both vegetation types, but especially for the controls. For example TN ranged from 0.54 to 27 mg/l in the control plots, with many outliers with high concentrations.

Lastly, vegetation type interacted with distance from the fence and depth. There was a strong interaction between these three variables (i.e distance from the fence \* depth \* vegetation; p-value = 0.00286).



**Figure 4.9: Box plots displaying TN, NO<sub>3</sub><sup>-</sup> and TKN concentrations in each vegetation type, including the mean value. Graphs display the median, 25<sup>th</sup> and 75<sup>th</sup> percentiles, and mean, minimum and maximum values. Letters indicate groups that are significantly different (p-value < 0.05). Note that the scale on y-axis for TKN is one order of magnitude lower than for TN and NO<sub>3</sub><sup>-</sup>. Note that the statistical analysis were performed on log transformed data, not in the raw data shown in these graphs.**

# Chapter 5

## Discussion

### 5.1. General setting

The soil setting at this site suggests that drainage is impeded due to the relatively high  $\rho_b$  of the Hamilton Ash layer (around  $1.2 \text{ t/m}^3$ ) (Kuman, 2019), resulting in subsurface lateral flow through the riparian plots and towards the drain (Figure 3.5). The area studied was originally under dairy pasture and was converted into a series of experimental riparian plots in 2017, with the hopes of mitigating the transfer of contaminants, particularly N, into Lake Waikare. The primary mechanisms of N removal in riparian buffers are denitrification and plant uptake.

### 5.2. Soil physics

In relation to this study, soil quality targets are useful in providing context for the  $\rho_b$  and MP results shown in Table 4.1, and summarised in Figure 5.1 below. The  $\rho_b$  target for the soil type in question is between  $0.8$  and  $1.2 \text{ t/m}^3$  (Sparling, 2008), while the MP target for pastures is between  $12$  and  $30\%$  (Mackay, 2013). The  $\rho_b$  of the soils sampled is relatively high given that these are Ultic Soils, which are inherently “heavier” due to their high clay content. Figure 4.1 clearly shows the inverse relationship between  $\rho_b$  and MP (e.g. C1 has the lowest average  $\rho_b$  and highest average MP, while pasture had the highest average  $\rho_b$  and lowest average MP (McLaren & Cameron, 1996).

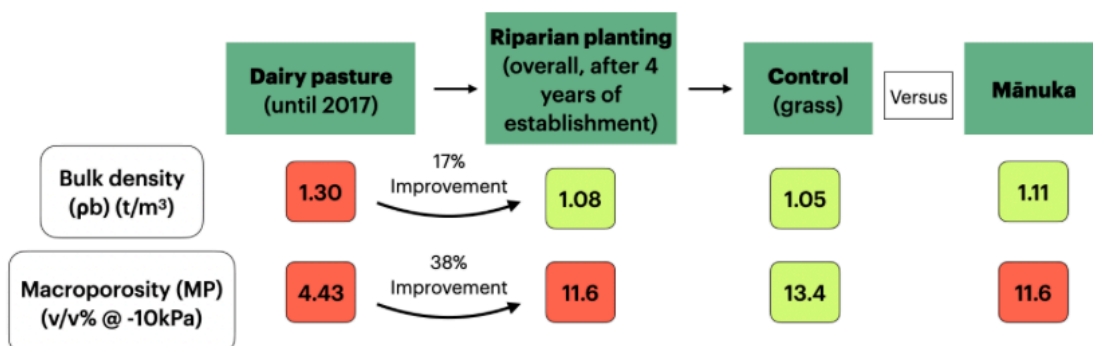


Figure 5.1: Summary of soil physical properties at the experimental riparian plots at Lake Waikare. Conditional highlighting shows whether the values are within (green) or outside (red) target ranges. Targets from Sparling et al. (2008) were used for  $\rho_b$ , while those from Mackay et al. (2013) were used for MP (revised values).

### **5.2.1. Soil physical properties after land use change: from dairy pasture to riparian buffer**

This study has found a positive change in the soil physical properties,  $\rho_b$  and MP, within four years of riparian planting and land use change (i.e. decreased  $\rho_b$  and increased MP compared to the dairy pasture in 2017) (Table 4.1, Figure 5.1). The average  $\rho_b$  between all riparian plots (control and mānuka) was 1.08 t/m<sup>3</sup> compared to an average of 1.30 t/m<sup>3</sup> in the paddock, representing a 17% improvement in the soil quality indicator in the four years since fencing off and planting. The average MP between all riparian plots was 11.6% compared to 4.43% in the paddock, showing again, a 38% improvement. The effect of plant roots, macro invertebrates, earthworm activity and lack of traffic by machinery, livestock or cultivation are likely causes for this improvement (McLaren & Cameron, 1996).

It is important to note that the soil quality targets are not strict limits that should be attained— targets are meant to be general guidelines about what is “better” or “worse” from both a production and an environmental protection standpoint. This is especially the case for MP, a target which is hard to define, as there are many contributing factors (Sparling & Schipper, 2002). Another consideration is that targets have been set based on economic land uses such as cropping, forestry and pastoral systems, and not for native forest (or restoration and mitigation projects such as the riparian band at Lake Waikare). Relative to the MP target for forestry (most similar land use; minimum of 22%) there is still improvement to be made, and with the mānuka still being very young (4 years old), ongoing changes are likely— soils under indigenous vegetation generally have higher MP (10-24%) (Sparling & Schipper, 2002). Taylor et al. (2009) found that MP in mature forests across the Waikato Region was 31.5% (under Allophanic and Podzolic soils). Based on these results, it can be expected that  $\rho_b$  will continue to decrease while MP will increase over the coming years. A soil’s MP is strongly correlated with its infiltration capacity (Taylor et al., 2009). A soil with high MP will require a very intense rainfall event to overwhelm the macropore and drainage system and produce overland flow (McLaren & Cameron, 1996). Hence, the soil quality improvements in the riparian band will provide a better flood defence during intense rainfall events— which are predicted to increase in frequency and intensity as climate

change advances (MfE, 2018). Less overland flow also reduces erosion, which is severe in the Lake Waikare catchment (Dean-Speirs, 2014). Macroporosity impacts the N cycle given that nitrifying organisms require aerobic conditions to produce  $\text{NO}_2^-$  and then  $\text{NO}_3^-$  ions, and are thus favoured in well-drained and well aerated soils. Less aeration results in less air for roots and less N fixation and nitrification (given that many beneficial microorganisms are aerobic), and thus, a decrease in plant productivity. Macroporosity is also important because it determines the amount of water that can be held in large pores (i.e. providing drainage) and form micro-sites where denitrification can occur in unsaturated soil conditions. Hence, this is a desirable characteristic for the management of N in all arable systems, as well as in riparian plantings, which remove N through denitrification and uptake.

These results are in agreement with other studies which have found higher MP in riparian buffers compared to adjacent grazed pastures (Bharati et al., 2002; Kumar et al., 2008; Kumar et al., 2012). For example, Kumar et al. (2008) found that MP under cottonwood trees (*Populus deltoides*) and a grassed riparian buffer to be 5.7 and 4.5 times higher after four years than in a continually grazed pasture, while  $\rho_b$  was 11.2% lower. The authors attributed this to deeper root systems in the buffer, which helped to move water under saturated conditions down the soil profile. Another study by Bharati et al. (2002) found improvements in soil quality (namely  $\rho_b$ ) after six years of riparian buffer establishment (compared to grazed pasture and cropped fields), and suggest that this is an effective step in reducing N, P, and sediment pollution. The same study also found the riparian buffer had infiltration rates five times higher than both grazed pasture and cropping, while  $\rho_b$  was significantly lower (e.g.  $1.10 \text{ t/m}^3$  in the grass filter compared to  $\sim 1.45 \text{ t/m}^3$  in the pasture). Similarly, Taylor et al. (2009) found that MP was 5.5 times larger under mature forested land than under grazed pastures (on average 31.6% and 5.8% respectively).

The physical characteristics of the soil under dairy pasture (pre-riparian establishment, in 2017) are in accordance with reports issued in the past decade, which have consistently reported low MP in pastoral farming sites as a critical issue since 2010 (Taylor et al., 2010; Taylor et al., 2017; MfE, 2021). This translates to increased chances of runoff, particularly on a site with a shallow perched water table—potentially leading

to a rapid saturation of the soil. Compaction also restricts the flow of water and air movement, providing little chance for water to infiltrate under heavy rainfall and little diffusion of oxygen for microorganisms (which play a key role in mediating the N cycle).

### **5.2.2. Soil physical properties in the riparian plots: grass versus mānuka buffer**

The soil physical properties measured at the control and mānuka plots appear to be similar (with the exception of plot M2, Figure 3.5) (Table 4.1, Figure 4.1). Upon land use change, pb improved across all plots, although there was a more marked improvement in the control plots relative to mānuka: from an original 1.30 t/m<sup>3</sup> under dairy pasture to 1.05 and 1.11 t/m<sup>3</sup> respectively— bringing the indicator within the target range (Figure 5.1). MP also increased more in the grass control plots than in the mānuka plots (from 4.4% under original pasture to 13.4% and 11.6%). In a similar fashion, the aforementioned study by Bharati et al. (2002) found that soils under silver maple had a higher pb compared to that of grasses (~ 1.2 and ~1.1 t/m<sup>3</sup> respectively), but also had a higher infiltration rate. Meek et al. (1992) sustains that there is only a very small, or no relationship between pb and infiltration rate. The authors attributed the higher infiltration under the silver maples to higher MP due to more earthworm activity and greater root biomass (after six years of establishment). Such could be the case at Lake Waikare too. Although infiltration was not measured directly, the volume of pore water samples extracted from the mānuka plots was smaller (Appendix 1), potentially indicating that water was readily draining through the soil instead of staying at shallower depths, or draining through cracks in the soil (personal observation). At 15 cm depth, the mānuka plots were somewhat drier than the controls, especially at the start of winter, while at 30 cm plot M1 was the driest (Figure 4.5). Similarly in the study by Bharati et al. (2002) the grass filter was wetter than the silver maples. This is likely the combined effect of high rainfall interception by the canopy (63.9% in this study, Figure 4.2), and the easier movement of water through the drier areas or through cracks in the soil (personal observation). Indeed, Ghestem et al. (2011) have found that roots (which are expected to be larger and more abundant in the mānuka plots) may produce

preferential flow and induce subsurface lateral flow, which Halford et al. (2021) also suggested might have happened in their trial with mānuka.

Although the control plots showed a greater improvement in physical soil quality indicators, the differences observed between the control and mānuka plots were not very large (although lack of repetition prevents a robust statistical analysis). The similarity between vegetation types could be attributed to the originally poor condition of the soil (meaning that fencing alone would already create an improvement) and the young age of the riparian plants— so differences due to vegetation type might not yet be detectable. Moreover, changes in many soil properties are generally slow and can take several years before a statistically measurable difference can be found.

$\rho_b$  and MP were similar between the control plots (C1 and C2), but differed between the mānuka plots, where M2 was more compacted (1.07 t/m<sup>3</sup> and 12.5% for M1, and 1.16 t/m<sup>3</sup> and 7.1% for M2). One potential cause could relate to the higher planting density of plot M1 (1.6 versus 1.0 plants/m<sup>2</sup> in M2), which would mean a larger root biomass in M1. Plot M2 also has a lower MP, which could indeed mean less infiltration and more runoff. This is reflected in the pore water samples collected from the site: more pore water sample was collected from plot M1 than from M2, indicating that there might be more infiltration into M1. Although plot M2 was more compacted than M1, it had higher VMC. It might be that, being more compacted, it would be necessary to exert more negative pressure to obtain samples in M2— water moving more slowly, which is why it was wetter. The area of compactness, which appears to restrict drainage at 30 cm depth, could also reflect differences in subsurface micro-topography, namely, the distribution of the Hamilton Ash layer. Lysimeters may have tapped into the thick clay layer in plot M2 (but not in M1), where water was being tightly held by clay particles. Historical photos of the riparian plots (Figure 2.11, dated to 2016) do not show any obvious reasons for compaction on plot M2 (such as machinery or livestock paths). Another reason for the higher  $\rho_b$  in plot M2 may be due to its positioning lower on the slope (Figure 3.5) where the water table was found to be closer to the surface for longer periods of time (Figure 4.4), making it more likely to have a higher degree of gleying and a higher proportion of clay (which tends to increase  $\rho_b$ ). Indeed, the soil type in question is a Perched-Gley Ultic Soil. In support of this hypothesis is the fact that the

plot is relatively wet, particularly at 30 cm depth (Figure 4.5), even more so than plot C2 which was lower down the slope (Figure 3.5). When VWC is high and the plastic limit is exceeded, disconnection of pores and compaction could occur. In turn, this reduces the soil's capacity to regulate water, which leads to wetter soil with less strength, creating a feedback cycle. The relationship between MP and VWC might also extend to plots C1 and C2: MP in C2 might be lower than in C1 (if ever so slightly), due to the shallower depth of the water table at this point, as well as marginally higher VWC (Figure 4.4 and 4.5).

### **5.3. Nitrogen in soil solution**

When it comes to N in agricultural systems, much of the discussion revolves around  $\text{NO}_3^-$  being the species most at risk of being leached from the soil. However, all forms of N can eventually transform into  $\text{NO}_3^-$  as they travel downstream, and may cause adverse effects. For this reason, all species are included in the analysis below. However, different N species do represent different components and processes (Figure 2.2). Given its positive charge (and the overwhelming negative charge of soil)  $\text{NH}_4^+$  is well adsorbed onto soils. It is the first breakdown product of N inputs such as biological fixation, fertiliser or urine, and is usually rapidly nitrified to  $\text{NO}_2^-$  and into  $\text{NO}_3^-$ . Due to its short lifespan in soils and negligible concentrations found in this site (0.28% of TN, Appendix 4c),  $\text{NO}_2^-$  was excluded from the present analysis. TKN is a measure of  $\text{NH}_4^+$  and dissolved organic N (DON) which is a source of energy for microorganisms—converted into  $\text{NH}_4^+$  and then  $\text{NO}_3^-$ . TN is simply the sum of the all N species. In this system, the majority of TN in high concentration samples was  $\text{NO}_3^-$  while, in low TN samples, TKN was the main species (see Appendix 4b). TN concentrations were found to be impacted by all the variables considered in this study: distance from the fence, depth, plot and vegetation type. Thus, the following discussion will be based firstly on TN results, and then broken down into relevant species.

#### **5.3.1. Effectiveness of the riparian buffer**

TN concentrations decreased significantly throughout the length of the riparian band, from an average of 9.32 mg/l at 1 m from the fence to 2.03 mg/l at 7 m.  $\text{NO}_3^-$  was the

main form of N being removed, decreasing 10-fold within the sampled area (from an average of 7.97 to 0.75 mg/l) (Figure 4.6). This shows that the buffer is fulfilling its purpose and successfully removing N from subsurface flow. Considering that only the first 7 of the 30 m of the buffer were sampled, it can be expected that concentrations will continue to decrease farther into the riparian band, and be substantially lower at the bottom of the plots than what was found in this study (i.e. close to the drain, Figure 3.5).

A “back of the envelope” calculation suggests that the rate of N removal decreases along the length of the buffer together with N concentrations: on average, more N was removed between 1 - 4 m compared to 4 - 7m (4.34 mg/l vs 2.95 mg/l TN removed). Fennessy & Cronk (1997a) propose that  $\text{NO}_3^-$  removal rates indeed vary throughout the length of a buffer strip. The authors suggest that there is relatively more OC at the edge of a buffer, where  $\text{NO}_3^-$  enters the subsurface and provide the best conditions for denitrification to occur (better soil structure compared to the pasture upslope allows for infiltration, although there is also a gravel track at the edge of the buffer in this site, which might change water flow pathways). In contrast, Mayer et al. (2007) found that N removal rates were constant along the length of a buffer, emphasising the importance of buffer width.

Regardless of the consistency of removal rate, findings are in agreement with the body of literature, emphasising that  $\text{NO}_3^-$  removal relies largely on buffer width. This ensures that the residence time of water is long enough to allow plants to intercept and uptake N, and allow denitrification to occur, as long as the hydrogeomorphology allows for water to infiltrate into the subsurface — i.e. the soil has a relatively low  $\rho_b$ , is not compacted at the surface and there is sufficient MP (Fennessy & Cronk, 1997; Vidon & Hill, 2004; Mayer et al., 2007). This appears to be the case at the experimental plots, where further increases in MP and thus infiltration capacity can be expected as the mānuka trees reach maturity and the grassed plots continue to develop.

In addition to its nitrification inhibition abilities, it was hypothesised that denitrification would be more efficient in the mānuka plots as its roots would be deeper tapping and deliver C for denitrifying bacteria to use as a substrate. The deeper tapping roots would also improve infiltration, especially in the deeper layers (further enhancing denitrification) and would be able to uptake  $\text{NO}_3^-$  as it leached downward (section

2.6.2.3). However, few samples were obtained from the lysimeters deployed to 50 cm, while soil properties in the topsoil were found to be similar between the vegetation types—which has not provided enough data to test the hypothesis.

Although some high  $\text{NO}_3^-$  concentrations were found throughout the course of sampling (up to 26.2 mg/l), the average concentrations found, regardless of plot, vegetation, distance or depth, were 4.26 mg/l which is below the World Health Organisation drinking water standard (i.e. 11.3 mg/l). However, these concentrations are well above the national bottom line for ecosystem health in lakes, which sets  $\text{NO}_3^-$  toxicity at 3.5 mg/l (New Zealand Government, 2020). For comparison, in a riparian zone in central Iowa, Zhou et al. (2010) found 0.1-11.1 mg/l  $\text{NO}_3^-$  to a depth of 100 cm. They also found that seasonal variations were substantial, from a maximum of 8.5 mg/l in spring and then dropping to 0.4 mg/l in autumn— meaning that year round monitoring of the buffer at Lake Waikare could provide further insight on the dominant processes occurring onsite. In NZ, Fraser et al. (2013) found  $\text{NO}_3^-$  concentrations between 1-10 mg/l under winter cover crops (to a depth of 60 cm), while Chibnall (2013) found an overall average of  $2.06 \pm 0.58$  mg/l (to a depth of 30 cm) in a dairy farm in the Waikato Region. The highest concentrations reported in the literature were up to 77 mg/l (Fraser et al., 2013). Further research is also necessary to incorporate N concentrations found in lysimeters with a drainage model to be able to quantify actual N leaching, and report it as kg N/ha.

Although this study demonstrated that the riparian band is successfully removing N from subsurface flow, the identification and assessment of the fate of that N was outside the scope of this study. N transformations and removal pathways underpin the results obtained and provide an insight into the processes occurring. For instance, vegetation age and type determine the extent of canopy closure, as well as root development and depth, impacting soil properties (Franklin et al., 2019). In turn, this affects the relative importance of N removal processes: plant uptake and denitrification (Mayer et al., 2007).

Although the largest proportion of N uptake occurs at the time of maximum biomass production, (during the growing season, spring/summer; Fennessy & Cronk, 1997), the

experimental sites are located in an area which experiences mild winters, where vegetation continues to grow (as measured by other researchers on-site), so uptake may have been possible. In addition, the decline in N concentrations between 1 and 7 m from the fence is large— and it is unlikely that vegetative uptake alone is responsible for all of the N removal (research is currently being carried out onsite with the goal of quantifying uptake). As discussed above, MP in the riparian plots is significantly higher than that of the pasture upslope, allowing for better infiltration. The lack of treading and roots of riparian vegetation are also very likely to improve soil structure at depth (although not measured in this study). Moreover, the lateral flow over the Hamilton Ash layer is likely to lead to high levels of saturation (section 5.3.3). The combination of better infiltration and better soil structure is thus conducive for denitrification to proceed efficiently— with the existence of both aerobic and anaerobic micro-sites (where water-filled pore spaces > 60%, Schimel & Holland, 2005). For example, it is possible that plot C1 had lower concentration of  $\text{NO}_3^-$  because it was more efficient at denitrification due to its consistently higher VMC in comparison to plot M1 (Figure 4.4 and Figure 4.8). The fact that these plots were adjacent (and thus had similar environmental conditions), gives more weight to this hypothesis. Likewise, closer to the drain and towards the lake (area not sampled), where the water table appears to remain closer to the surface for longer (Figure 4.3), saturation might be more constant, and so would denitrification potential, becoming a hotspot for N removal when N concentrations are already low. On the other hand, it is also possible that riparian zones act as hotspots for  $\text{N}_2\text{O}$  production when denitrification proceeds inefficiently (e.g. when the C:N ratio is low) (McLaren & Cameron, 1996). Tradeoffs between water quality and greenhouse gas emissions thus need to be assessed (Liu et al., 2016).

Temperature is another factor that influences the rate of denitrification: below 5 °C denitrification is very slow, while it reaches a maximum between 25-30 °C (Skiba, 2008). In NZ, most  $\text{NO}_3^-$  leaching losses occur during the winter (McLaren & Cameron, 1996). The weather station installed on site shows that although nighttime air temperatures sometimes dropped below 5 °C during June and July, daytime temperatures usually remained between 5-15 °C (data not shown; it is important to note that soil has a thermal buffering capacity so it remains more stable). Hence, denitrification is indeed proceeding during winter. However, during summer, when

denitrification potential would be highest temperature-wise, the case might be that the soil is too dry (so denitrifying bacteria are estivating)— supported by the fact that large cracks in the soil were observed in the mānuka plots in December 2020 and March 2021 (personal observation).

### **5.3.2. Impact of depth on N concentrations**

Soil depth was the single factor that most impacted TKN concentrations, which showed a clear decline with depth (Figure 4.7). This was to be expected as, in the topsoil, up to 90% of the TN occurs in organic forms (Sparling, 2008). It is estimated that only 1.5 to 3.5% of the organic N in soil mineralises every year (Brady & Weil, 2008). TKN is associated with organic matter is easily accessible to aerobic microorganisms in the topsoil (which at this site has good structure and likely aeration) and was probably being consumed there— rapidly converting into  $\text{NH}_4^+$ ,  $\text{NO}_2^-$  and  $\text{NO}_3^-$  which were then moved down the soil profile ( $\text{NO}_3^-$  concentrations indeed increased with depth; Figure 4.7). The increase in  $\text{NO}_3^-$  concentrations with depth can also be attributed to the soil's physical characteristics: with increasing depth, there is an increasing proportion of clay in this soil (section 3.1.2), which easily holds water (and solutes within pore water, such as  $\text{NO}_3^-$ ) (Brady & Weil, 2008). Soil texture also impacts the levels of  $\text{NO}_3^-$  leaching since it dictates how much C can be held in the soil. Usually clayey and silty soils release more organic C relative to sandy soils and thus have higher rates of N immobilisation (although some clays, like allophane protect SOM) (Neilen et al., 2017). The increase of  $\text{NO}_3^-$  with depth is further supported by the fact that EC and  $\text{NO}_3^-$  are usually correlated (Miyamoto, 2015) and EC was higher at 30 cm depth than at 15 cm depth (Figure 4.5).

### **5.3.3. Impact of subsurface flows on N concentrations**

The well-structured surface layers allow for water to infiltrate into the soil, until it reaches the more compacted and impermeable Hamilton Ash layer, forcing water to move laterally towards the lake instead. During rain events, a short-lived perched water table develops at the boundary with the Hamilton Ash (peaks in water table height at dip well “C” aligned with peaks of rainfall events, Figure 4.3). This is also evidenced by

soil characteristics, such as mottling and gleying at 30 cm depth. Moreover, a large volume of pore water samples was extracted on July 20th (Appendix 1), which suggested that the lysimeters were sampling the perched water table.

It is expected that as  $\text{NO}_3^-$  is travelling through the buffer, it is also moving downward until it reaches the Hamilton Ash layer, where it accumulates (net movement being diagonally towards the lake, Figure 3.5 and Table 4.2). This raises the question of whether N is actually being removed, or if it is simply slowly moving through/over the Hamilton Ash layer (due to its high clay content and slow permeability): Appendix 3a shows that even if at 1 m from the fence TN concentrations at 10 cm and 50 cm depth are similarly high, at 7 m the concentration decreases for all plots (particularly clear for plot C2 between July 11th and 31st). This shows that N is predominantly moving laterally over the ash layer rather than vertically downwards, and simultaneously accumulating in the deeper horizons (where pore water, and the  $\text{NO}_3^-$  within it, is held). The implication is that N will have a longer residence time in the Hamilton Ash layer than in shallower profiles, meaning that the full mitigation impact of the riparian buffer will not be apparent in the drain/lake's water quality for years to come: The N-loaded water that is absorbed by the Hamilton Ash layer may have long residence times. For example, Davie (2004) reports that the movement of water through a saturated soil is usually slow, with a typical velocity of around  $13 \text{ mm hr}^{-1}$  in New Zealand soils—which would translate to approximately 96 days for water to move throughout the 30 m band at this site. An exception to this figure is the more rapid infiltration of water through wormholes, burrows and cracks into the soil (i.e preferential flow), which were observed in the mānuka plots between May and June.

It can also be expected that the perched water table and lateral flow also exist at the top of the paddock upslope, so that most of the water moving through that paddock will indeed go through the riparian plots (instead of continuing to leach downwards straight into the shallow groundwater or very slowly permeate through the Hamilton Ash)—which is why the concentration of  $\text{NO}_3^-$  might also be so high in the Hamilton Ash: it is an accumulation of N from the whole paddock upslope.

Although N does not predominantly move through overland flow, a proportion of it may be carried as particulate N, especially plant debris and animal excreta. When the area close to the drain is saturated (Figure 4.3, e.g on July 21st at dip well D), there is little

capacity for infiltration of any runoff generated within the plots (which has been collected for other studies at the site). The potential for N attenuation within runoff is low, unless it becomes trapped in the vegetation itself— particularly in the control plots (Fennessy & Cronk, 1997). One solution could be extending the riparian zone upslope, so that the zone of N removal is expanded, so that by the time subsurface water reaches the saturated zone, N loading is very low. The compromise is a loss in productivity for farmer, unless the area could be made profitable through the production of mānuka honey or essential oils. On the other hand, the higher level of saturation close to the drain may increase the efficacy of denitrification, when N concentrations in the subsurface are already expected to be substantially smaller than those found in this study (as N would have been removed through the length of the buffer) (Venterink et al., 2002; Schimel & Holland, 2005).

#### **5.3.4. Impact of vegetation type on N concentrations**

In general, the results of this study show higher N concentrations in soil pore water under mānuka compared to the grassed control (Figure 4.8, Appendix 2), not fulfilling the hypothesis of mānuka inhibiting nitrification. Based on previous work, it was expected that N losses would be smaller under mānuka compared to the control. In a laboratory study, Downward (2013) demonstrated that mānuka extracts inhibited the activity of nitrifying bacteria, while Esperchuetz et al. (2017) found that leaching was substantially smaller under mānuka and kānuka compared to *Pinus radiata* (2 kg ha<sup>-1</sup> compared to 53 kg ha<sup>-1</sup> NO<sub>3</sub><sup>-</sup>). Further Halford et al. (2021), found that NO<sub>3</sub><sup>-</sup> and NH<sub>4</sub><sup>+</sup> concentrations to 20 cm depth were significantly lower under mānuka compared to pasture. Our results are more in accordance with those of Neilen et al. (2017) who found NO<sub>3</sub><sup>-</sup> concentrations in barrel lysimeter leachates to be higher under woody vegetation than under grasses (0.23 ± 0.31 mg/l versus 0.14 ± 0.16 mg/l respectively). Although the concentrations found at the experimental plots might seem high in comparison, their study was conducted on soil cores, which do not necessarily reflect field processes and conditions. Generally, NO<sub>3</sub><sup>-</sup> concentrations were higher in the mānuka plots, with a median value of 1.75 mg/l compared to 0.17 mg/l in the controls. However, it is important to note that the highest NO<sub>3</sub><sup>-</sup> concentrations were found under the control plots (up to 26.2 mg/l) (Figure 4.9).

On average,  $\text{NH}_4^+$  comprises a small proportion of the N in this system (3.1%, Appendix 2b). The fact that a significant difference in  $\text{NH}_4^+$  concentrations with depth was detected, albeit small, means that further investigation is necessary. The impact of depth on  $\text{NH}_4^+$  is relevant in this study, as some literature suggests that mānuka could be a BNI (biological nitrification inhibitor) species, eliminating nitrifying bacteria and thus reducing  $\text{NO}_3^-$  concentrations in the soil (N would remain as  $\text{NH}_4^+$ ) (Downward, 2013; Esperschuetz et al., 2017; Halford et al., 2021). If that were the case,  $\text{NH}_4^+$  would remain available in soil for a longer time, for when the plant requires it (i.e. more  $\text{NH}_4^+$  in soil under mānuka compared to the control plots). Figure 4.7 shows that the range of  $\text{NH}_4^+$  concentrations found at 30 cm depth was larger than at 10 or 50 cm depth. According to Marden et al. (2005) mānuka allocates a large percentage of its growth to root biomass during its first five years of establishment, reaching a depth of 30 cm in that time— which coincides with the depth at which differences in  $\text{NH}_4^+$  concentration were found. Clays, which become more abundant at this depth, are negatively charged, and can thus strongly bind  $\text{NH}_4^+$  (Brady & Weil, 2008), so it might not be the effect of vegetation driving these differences. There was no significant interaction between vegetation and depth, and like Bowman (2020), no significant differences in  $\text{NH}_4^+$  concentrations were found between vegetation types.

Despite the differences with previous studies, the results herein present high variability, while the lower N contents under the controls might only be true in very general terms: there also was a significant interaction between vegetation type and other variables (i.e. soil depth and distance from fence; Table 4.3), which indicates that under certain conditions, this trend may be reversed (Appendix 3a). However, these interactions remain unclear as a small number of samples was extracted from the 50 cm layer under the mānuka plots. This was caused by drier soil conditions due to rainfall interception and presumably higher transpiration rates, as well as the Hamilton Ash layer tightly binding water (meaning that more negative pressure would have been needed to extract pore water). Such factors add to the complexity of untangling the processes that were taking place in this site.

### 5.3.5. Water fluxes under mānuka compared to control

Although not hypothesised during the experimental design, results show that differential water flows under mānuka and control may be an important factor explaining N results. Some factors mediating water fluxes include: rainfall interception, root demand and preferential flow.

A seven year field trial in NZ found that  $\text{NO}_3^-$  concentrations under permanent pasture were between 1-10 mg/l (Fraser et al., 2013) similar to what was found in the experimental plots (Figure 4.8). Fraser et al. (2013) also found that  $\text{NO}_3^-$  leaching increased with increasing rainfall, which agrees with more N losses found under the control sites, which received relatively more rainfall than the mānuka sites (due to higher interception and evapotranspiration in the latter, by 63.9%; Figure 4.2). The interception of rainfall by the mānuka canopy translated to more evapotranspiration and less water reaching the ground— meaning that there was a lack of water to transport the solute and that N was simply less diluted. Although the present study did not measure infiltration directly, Figure 4.4 shows that, particularly during the start of winter, plot M1 was drier and drained to a larger extent between rain events, when nitrification would continue to accumulate  $\text{NO}_3^-$  that would be lost only once rainfall intensity allowed for water to reach the ground and increase VMC above field capacity. As the buffer matures, it can be expected that MP was also increase allowing more infiltration (section 5.2.1).

The findings also suggest that the intermittent wetting and drying cycles of the soil under the mānuka canopy might contribute to  $\text{NO}_3^-$  and  $\text{NH}_4^+$  accumulation (Venterink et al., 2002), which may then be “flushed” during the wetter winter months, or when the short-lived perched water table develops. Wetting and drying cycles such a large rainfall event being followed by dry periods where macropores drain to a large extent might stimulate mineralisation. Upon re-wetting denitrification might increase again, but without a concurrent decline in mineralisation, resulting in the accumulation of  $\text{NO}_3^-$  and  $\text{NH}_4^+$  (Venterink et al., 2002). For this reason Neilen et al (2017) found that, under saturated conditions, riparian zones may become a source of N. This could be a potential reason for the higher concentrations of N under the mānuka plots: the canopy intercepts a large proportion of rainfall (63.9%, Figure 4.2), meaning that the soil remains drier (particularly upslope, like M1, Figure 4.4), until a large rain event that

overwhelms that canopy and is able to wet the soil to a larger extent— only to become dried out again in-between large rain events. On July 7th for example, it appears that there were spells of heavy rains, as a similar amount of water was recorded in rain gauges above and below canopy at certain times of the day (Figure 4.2; more data not shown). This is also visible in VMC for all plots, but particularly M1 (Figure 4.4).

Moreover, there might also be a higher demand of water from the relatively larger mānuka roots compared to the control. For example, the removal of a forest canopy has led to a rise in the water table due to a reduction in evapotranspiration (Trousedell and Hoover, 1995). However, the role of roots in extracting soil water by evapotranspiration is limited during cold, wet seasons (Ghestem et al., 2011).

Preferential flow pathways might also be a reason for differences between vegetation types. Preferential flow might occur along 1) cracks in the soil produced due to dryness over the summer months (personal observation); and 2) mānuka roots (as also observed by Halford et al., 2021). These factors contribute towards rapid water flows that do not allow time for N attenuation processes to occur. Ghestem et al. (2011) have found that roots may also induce subsurface lateral flow. This highlights one of the disadvantages of suction-cup lysimeters, as they lack the ability to identify preferential flow pathways (Weihermüller et al., 2007).

### **5.3.6. Water fluxes and overall N losses**

Although N concentrations under the mānuka plots were generally higher than those found under the grassy controls, the amount of water extracted (proxy for drainage) was considerably smaller, resulting in 21% less TN being extracted from the mānuka plots. A reduction in drainage is important because water management might be one of the main factors mitigating N losses in these riparian bands, rather than uptake and/or transformations. This might also explain why the concentration was higher under mānuka than under grass— N is simply less diluted and there was a lack of water to transport the solute. According to climate change projections for NZ, heavy rains are expected to become more frequent (MfE, 2018), making water management increasingly important, which appears to be more beneficial for N mitigation in the mānuka plots. However, the infiltration capacity of a soil is more important than canopy effects in the distribution of heavy rainfall (Keim & Skaugset, 2003). More heavy

rainfall in coming years and the limited depth for infiltration (due to perching over Hamilton Ash layer) may lead to large amounts of overland flow— and N being carried through that pathway (as particulate N, plant matter and animal excreta). The literature suggests that herbaceous buffers are more efficient at intercepting contaminants that move overland— also including P and sediment (which is also an issue in the Lake Waikare catchment) (Franklin et al., 2015; Neilen et al., 2017). Thus, a combination of both herbaceous and woody vegetation (like mānuka) might be most efficient at mitigating the flow of contaminants into Lake Waikare in coming decades. This might mean extending the riparian zone upwards, and adding a band of herbaceous planting above the mānuka plots.

### **5.3.7. Impact of soil and hydrology on N concentrations**

Heterogeneity and consequent spatial differences in soil texture and physical properties might be a major factor driving differences in N concentrations between plots due to their influence on hydrology. For instance, other researchers carrying out experiments at the site have also observed that the drainage in the upslope plots (C1 and M1) is good and becomes progressively poorer closer to the lake. Nevertheless, and despite large variability, other studies have shown no difference in the volume of runoff between mānuka and pasture (personal communication, David Clarke, EcoQuest Education Foundation). On the other hand, water fluxes also impact soil characteristics, such as the degree of gleying— which in turn define the existence and longevity of the perched water table. With the current information, it is unclear whether subsurface flows run directly parallel to the plots, or if it moves through all vegetation types (diagonally towards the drain, Figure 3.5)— which is highly likely— confounding the impact of vegetation type on N dynamics. In order to be able to better compare N losses at the plots, a higher resolution understanding of hydrological fluxes would be necessary, as well as longer-term monitoring. It would also be helpful to further model leaching losses.

The differences between plots are supported by the fact that plot had a larger influence on N concentrations than vegetation (Table 4.3), suggesting that hydrology, or the general heterogeneity of soil, was more important at mediating N cycling in this site

than vegetation type, at this stage (which might change as the buffer matures). Figure 4.7 exemplifies this well: Plot C1, which was higher on the slope (and had a lower water table) had a lower average TN concentration than Plot C2, which was lower on the slope and a higher water table (shallower). Plot location impacted  $\text{NO}_3^-$  10x more than TN (i.e. samples with high N concentrations, Appendix 4b) — so it can be inferred that hydrology and soil heterogeneity is the main driver of nitrification and denitrification.

#### **5.4. Limitations and further research**

This study sought to investigate the impact of vegetation type (grassed control versus mānuka) on the movement of N through the riparian bands, and focused on obtaining N concentration measurements from different plots. Although N concentrations were higher under the control plots, there is evidence to suggest that there is less drainage from the mānuka plots, largely due to the canopy's interception of rainfall. However, it cannot be discarded that less pore water samples were extracted due to preferential flow along root channels and cracks in the soil. The main limitation of this study is that TN extracted (i.e. TN concentration x volume of sample) was used as a proxy for drainage rather than directly quantifying leaching. Hence, future research at this site might be dedicated to creating a soil water balance to estimate leaching, which would provide a more complete picture of the processes occurring at the experimental site. This might be done utilising a simple bucket model (i.e. drainage = precipitation - evaporation - soil storage) where above/below canopy rainfall are measured for each vegetation type in further detail, evaporation is estimated by the FAO5<sub>6</sub> equation (if not measured directly), and storage calculated by soil water deficit (as developed by Woodward et al., 2001) (Chibnall, 2013). Alternatively, models such as APSIM or EVACROP could be used (Vogeler et al., 2020). A quantification of root biomass and morphology, which might limit denitrification potential due to short residence times, would also be useful.

It would also be of value to revisit this study in 4-10 years to explore how the buffer impacts soil quality and N dynamics in the subsurface as it matures. Moreover, and as highlighted in section 2.5.1.3.1, there are inherent disadvantages to using suction cup lysimeters as a method for collecting pore water samples. These include 1) the fact that

they sample soil water under unsaturated flow conditions and from pores with longer residence times (not free-draining; i.e.: water that has relatively more interaction with soil components), 2) they are not able to record rapid changes in solute concentration (as happens during heavy rainfall), and 3) they do not identify preferential flow pathways (Weihermüller et al., 2007).

The number of variables considered in this study was relatively large (distance from the fence, depth, vegetation type, plot, sampling date), providing a lot of data on one single variable. This has painted a broad picture of the factors mediating N in the experimental plots and revealed trends. In the future it would be useful to take more replicates of the same measurements for the purposes of significance testing (which was limited in the present study). Given that the effects of distance and depth are relatively well understood, the analysis could be constrained to one or two depths, and with three replicates per distance from fence per plot, and all the way to the drain to provide a higher resolution understanding of potential N transfers to Lake Waikare. In addition, lysimeters could be installed in the paddock upslope to monitor how much N is entering the plots, and if the rate of N removal is different within them. Moreover, rains persist throughout the end of winter and early spring in the area (August-October) where N measurements could continue— as seasonal variations in N transfer may be large (Zhou et al., 2010). Further research might also be devoted to understanding gaseous N losses in the riparian buffer, as they have the potential to become hotspots for N<sub>2</sub>O emissions, as a by-product of denitrification. Hence trade-offs between water quality improvements, biodiversity and greenhouse gas emissions would need to be assessed (Liu et al., 2016; Lyu et al., 2021).

This research was novel because the potential of mānuka to mitigate the loss of N had not yet been tested in the field. However, this also meant that the study had to contend with the disadvantages of working in a fully operating farm: mid-way through the study, a gravel track was built over the top of one of the control plots which was intended to be used for N measurements (Figure 3.5). In this way there would have been two adjacent paired sites (M2 and C2), instead of them being separated by approximately 30 m. The soils at the site are very heterogeneous (personal observation), creating further variability between plots that was hard to untangle. Further research might also include

a third paired site at this location, or other mānuka-dominated riparian sites around the country. For the latter, it is important to note that there may exist genetic and chemical differences between and amongst populations of *L. scoparium* (Downward, 2013). In fact the mānuka at Lake Waikare is locally known as swamp mānuka, and has distinctive characteristics compared with mānuka from other regions, such as softer and longer leaves, more similar to those of kānuka, and the capacity to thrive in swamp conditions.

Quantifying denitrification and plant uptake as pathways for N removal was outside the scope of this study, although other research provides insight into the utility of native species in sequestering N into their biomass. In a greenhouse and field trial researchers found that native NZ monocot species (i.e herbaceous; *Cordyline australis*, *Phormium tenax*, *Carex virgata* and *Austroderia richardii*) were more efficient at uptaking N than native woody dicots, including mānuka (Franklin et al., 2015). This might suggest that a mixture of native species might be best at intercepting N fluxes and removing them. As mentioned in section 3.1, the experimental site at Lake Waikare indeed include plots of mixed vegetation— to which the methodology employed in this study might be extended to evaluate the merits of mixed plantings. The strategy might reap the benefits of both herbaceous and woody vegetation types. Grasses, for instance, are useful as riparian buffers as they can trap particulate N (Fennessy & Cronk, 1997), while woody buffers may aid in increasing infiltration and providing OC at depth to fuel denitrifying bacteria (Haycock & Pinay, 1992; Neilen et al., 2017).

Another aspect that might be delved into further is soil physical properties. A limited number of soil cores were collected (three per plot, 12 in total), meaning that statistical significance testing could not been carried out robustly. Nevertheless, the results are in accordance with the body of literature as well as reports published in the last decade. One factor to consider when analysing the soil physics results presented here is that soil cores were taken from the topsoil only (0-15 cm depth), hence the results do not represent the soils' physical characteristics at depth, where some of the pore water N samples have been collected. In order to fully assess changes in soil quality, this investigation might be deferred for some years, until the full effects of vegetation type are evident (perhaps another 4 years), and at that point, soil profiles dug and soil

physical properties assessed, horizon by horizon. In the meantime, the present study provides a good overall evaluation of present day differences in soil physical properties between plots.

Throughout the broader literature and this study, hydrology was found to have a large impact on N cycling. Soil moisture, precipitation and water table were monitored, but detailed measurements were outside the scope of this study. Water table measurements in particular could be improved: the dip wells close to the fence recorded little data and it was evident from dip well “C” that peaks in water table height were sharp and short-lived, showing how useful continuous data loggers are — being it impossible to achieve such resolution with manual measurements. Dip wells should also be installed to a greater depth (well into Hamilton Ash layer, ~1 m), and all contain continuous data loggers.

## **5.5. Summary of key points**

In summary, the experimental riparian plots at Lake Waikare are part of a complex 3-dimensional system, where there is an interplay of many factors. The hydrology of the area is impacted by both the soil setting and vegetation, affecting in turn, N transfers through the riparian buffer and likely into the lake. Soil quality improved under both vegetation types relative to measurements taken before the buffer’s establishment. To some degree, pb and MP improved more in the control plots, although it is likely that ongoing changes will occur in the mānuka plots as the trees continue to mature. The Mangatawhiri clay loam is classified as a Perch-Gley Ultic Soil. The existence of a subsurface lateral flow over the Hamilton Ash layer was evidenced by spikes in water table height during rain events, soil gleying and mottling and the higher bulk density of the Hamilton Ash layer (Kuman, 2019). Pore water results show that the buffer is effectively removing N from the subsurface, with a decline in N concentrations between 1 and 7 m from the fence. Concentrations were highest in the mānuka plots, although overall N losses were largest in the control plots due to higher water flows. These results suggest that in terms of losses, at this stage of the buffers development, water management is more impactful than management of N itself.

# Chapter 6

## Conclusion

---

### 6.1. Introduction

The main objective of this thesis was to determine the efficacy of mānuka riparian buffers at intercepting and removing N in subsurface flows compared to a grassed control. Further aims were to explore soil quality as a factor influencing differences in N removal between vegetation types, and to confirm the existence of a short-lived perched water table and shallow subsurface lateral flow towards Lake Waikare. The following sections summarise the key findings of this study as presented in chapters four and five, and circle back to the objectives set in chapter one.

### 6.2. Influence of vegetation type on soil physical properties

A positive change in soil physical properties was observed after riparian establishment at Lake Waikare. Overall, pb improved by 17% while MP improved by 38% in four years. The effect of plant roots, macro invertebrates, earthworm activity and lack of traffic (machinery, livestock or cultivation) are likely causes for this improvement. Vegetation type did not have a substantial influence on soil physical properties, which may be due to the originally poor condition of the soil (meaning that fencing alone would already create an improvement) and the young age of the riparian plants— so that differences due to vegetation type might not yet be detectable. Moreover, changes in many soil properties are generally slow and can take several years before a statistically measurable difference can be found. In coming years, a greater improvement in soil physical attributes is likely. This would provide benefits such as decreasing overland flow, providing a flood defence and potentially increasing the capacity of the soil to process and remove contaminants such as N.

### 6.3. Subsurface water flows at the southern edge of Lake Waikare

This study has shown evidence of the existence of a short-lived perched water table at the Hamilton Ash boundary, over which water and contaminants are likely to move laterally towards the drain. A number of observations support this theory. Good soil

structure in the surface horizons allows for infiltration through the soil profile, until water reaches the more impermeable and compact Hamilton Ash layer, at which point lateral movement is favoured. Gleying can be seen from 30 cm depth, as well as mottling— and the Mangatawhiri clay loam is indeed classified as a Perch-Gley Ultic Soil. Personal observations suggest that the micro-topography of the area is very heterogeneous, while soil sensors show restricted drainage at 30 cm depth in plot M2. Higher N concentrations at depth also suggest that N-loaded water is perched upon the clay-rich Hamilton Ash layer. Perching is further evidenced by the large volume of pore water samples extracted on July 20th, which suggested that the lysimeters could be sampling the perched water table.

#### **6.4. Influence of riparian vegetation and soil quality on N fluxes**

Independent of vegetation type, this study showed that the riparian buffer is effectively removing N from the subsurface throughout its length. It can be expected that at the bottom of the buffer N concentrations will be substantially lower than what was found in this study (given that only the first 7 of the 30 m of the buffer were sampled), helping to attenuate N inputs into the already eutrophied Lake Waikare. TN concentrations were not significantly different between vegetation types, but  $\text{NO}_3^-$  concentrations were. Higher  $\text{NO}_3^-$  concentrations were found in the mānuka plots, although, throughout the sampling period, 21% less TN was extracted from those plots when correcting for the volume of samples obtained. The main reason for this is the interception of rainfall by the mānuka canopy (63.9%), meaning that less water reached the soil below (also reflected in VMC)— in other words, it is a lack of transport for N, rather than smaller amounts of N driving differences amongst vegetation types. It is also likely that preferential flow through root channels, earthworm holes and cracks in the soil played a role. Wetting and drying cycles of the soil might also contribute to the accumulation of  $\text{NO}_3^-$  and  $\text{NH}_4^+$  in the soil by increasing mineralisation and diminishing denitrification (Venterink et al., 2002; Neilen et al., 2017). The lower N contents under the grassed control, might also only be true in very general terms, as there is also an interaction between vegetation, depth and distance from fence. It is likely that the buffer is still too young for mānuka roots to be fully developed and have a significant impact on N cycling, although, compared to the dairy paddock that was in its place until 2017, large

improvements have been observed in soil quality. As the buffer continues to mature, continued improvements are expected, as well as observable co-benefits such as improvements in habitat and biodiversity.

Overall, this study has provided insight into the main factors mediating the movement of N at this location, and it is hoped that this study will contribute to the scientific understanding of riparian buffer functioning-- and more specifically the role of a native NZ species, mānuka. Most importantly, the aspiration is that this study has provided further insight into the mitigation of N to support mana whenua in their efforts to restore Lake Waikare to health, reinstate its role as a "food bowl" for the area and provide co-benefits such as habitat restoration and farm diversification.

# Appendices

## Appendix 1: Volume of pore-water samples obtained

The labelling system utilised for the lysimeters was as follows:

- C1, C2 = control plots
- M1, M2 = mānuka plots
- 10 = 10 cm depth; 30 = 30 cm depth; 50 = 50 cm depth
- A= 1 m from the fence ; B= 4 m from the fence; C= 7 m from the fence

**Table: volume of pore water samples obtained from suction-cup lysimeters at the riparian experimental plots in Lake Waikare.**

Lysimeter Code	Sample volume (ml)						
	May 21	Jun 7	Jun 14	Jun 26	Jul 11	Jul 20	Jul 31
C1_10A	10*	12.5*	21*	35	82	92	45
C1_10B	36	35	27	32.5	40	42	30
C1_10C	3*	22*	27.5	47.5	82	104	60
C1_30A	19*	40	25	27.5	15*	55	35
C1_30B				17*	25	18*	10*
C1_30C	16*	11*	12.5*	11*	20*		15*
C1_50A	4*	40	30	41	62	132	15*
C1_50B				17.5*	20*	88	30
C1_50C		87.5	33	34	129	175	45
C2_10A	44	60	53.5	68	107	118	72
C2_10B	36	100	91	88	115	110	
C2_10C	45	95	77.5	65	87	102	22*
C2_30A	14*	12.5*	12.5*			20	15*
C2_30B	45	50	76	35	47	30	102
C2_30C		23*	16*	23*	13*	50	35
C2_50A	37	9*	15*	27	27	49	25

Lysimeter Code	Sample volume (ml)						
	May 21	Jun 7	Jun 14	Jun 26	Jul 11	Jul 20	Jul 31
C2_50B	23*	7*	24	27	42	45	30
C2_50C			26	24	90	62	37
M1_10A			9*	91	35	130	119
M1_10B				37	59	57	32
M1_10C				8*	30	40	27
M1_30A			4*		27	90	35
M1_30B				26	30	11*	22*
M1_30C				37	47	47	30
M1_50A				17*	12*	80	47
M1_50B				54	67	82	52
M1_50C				34	65	54	35
M2_10A				64	119	132	117
M2_10B	6*			35	65	77	65
M2_10C	14*	12.5*		37.5	45	75	40
M2_30A				26	30	16*	30
M2_30B				9*	17*	16*	17*
M2_30C					3*	20*	17*
M2_50A					22*	12*	20*
M2_50B							10*
M2_50C					17*	40	15*

\* samples not analysed due to insufficient volume (minimum 25 ml)

## Appendix 2: Raw data of N concentrations

**Table: Concentration of different N species found in pore water samples at the riparian experimental plots in Lake Waikare (see appendix 1 for labelling system)**

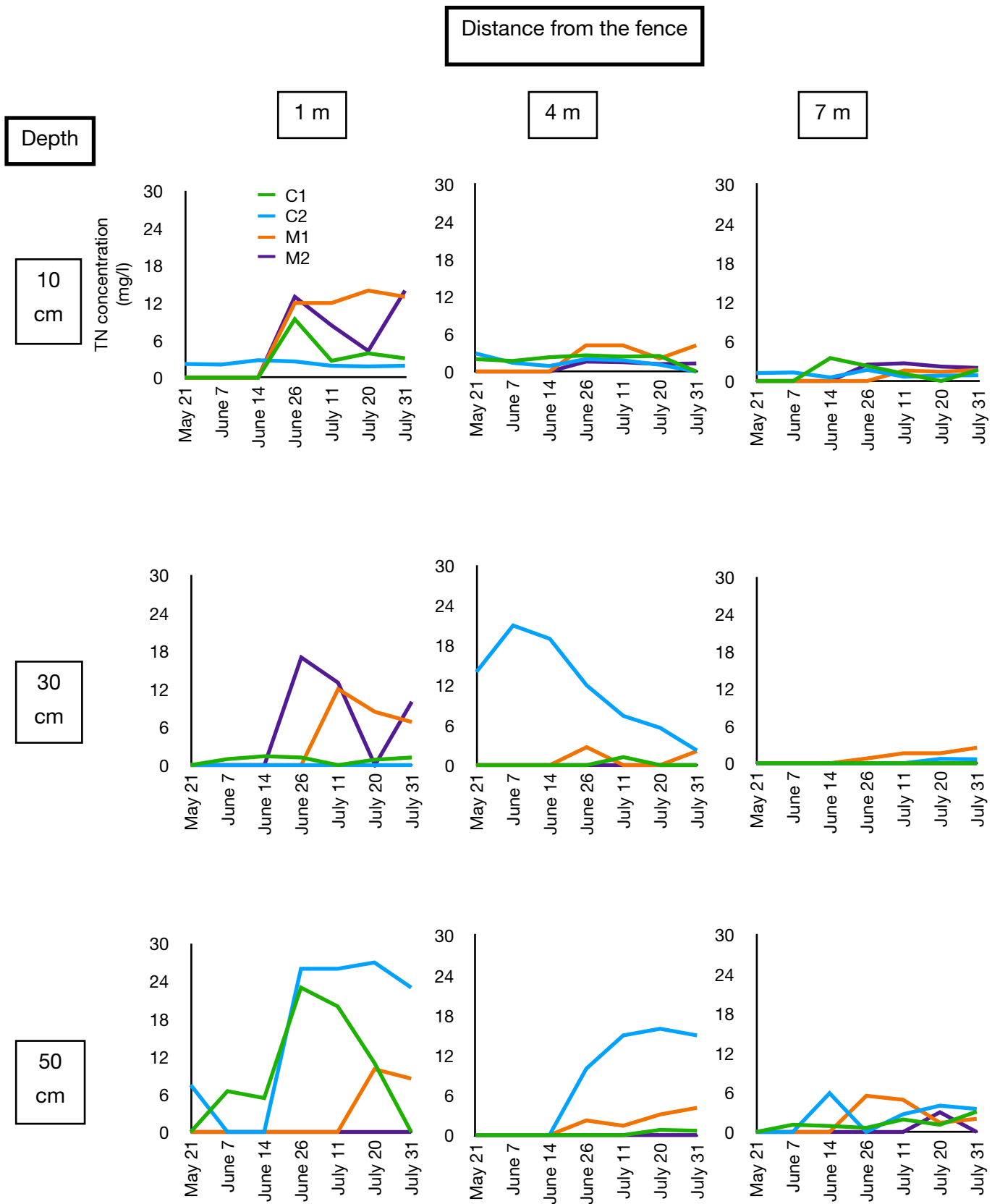
Lysimeter	Date	Concentration (mg/l)			
		NO <sub>3</sub> <sup>-</sup>	NH <sub>4</sub>	TKN	TN
C1_10A	21-May				
C1_10B	21-May	0.300	0.007	1.690	2.000
C2_10A	21-May	0.092	0.003	2.070	2.200
C2_10B	21-May	1.070	0.180	1.780	2.900
C2_10C	21-May	0.072	0.003	1.090	1.200
C2_30B	21-May	13.000	0.090	1.330	14.000
C2_50A	21-May	6.740	0.009	0.720	7.500
C1_10B	7-Jun	0.052	0.003	1.680	1.700
C1_30A	7-Jun	0.133	0.003	0.810	0.950
C1_50A	7-Jun	5.410	0.020	1.110	6.500
C1_50C	7-Jun	0.445	0.003	0.660	1.100
C2_10A	7-Jun	0.046	0.020	2.060	2.100
C2_10B	7-Jun	0.263	0.010	1.140	1.400
C2_10C	7-Jun	0.053	0.060	1.190	1.300
C2_30B	7-Jun	18.300	0.160	2.170	21.000
C1_10B	14-Jun	0.459	0.030	1.850	2.300
C1_10C	14-Jun	0.084	0.080	3.360	3.500
C1_30A	14-Jun	0.045	0.100	1.360	1.400
C1_50A	14-Jun	4.570	0.003	0.850	5.400
C1_50C	14-Jun	0.153	0.003	0.760	0.910
C2_10A	14-Jun	0.334	0.040	2.470	2.800
C2_10B	14-Jun	0.066	0.003	0.820	0.890
C2_10C	14-Jun	0.049	0.003	0.490	0.540
C2_30B	14-Jun	16.500	0.270	2.140	19.000
C2_50C	14-Jun	3.810	0.010	2.110	5.900
C1_10A	26-Jun	6.800	0.060	2.510	9.400
C1_10B	26-Jun	0.166	0.040	2.450	2.600
C1_10C	26-Jun	0.028	0.008	2.250	2.300
C1_30A	26-Jun	0.136	0.040	1.030	1.200
C1_50A	26-Jun	21.800	0.190	1.110	23.000

Lysimeter	Date	Concentration (mg/l)			
		NO <sub>3</sub> <sup>-</sup>	NH <sub>4</sub>	TKN	TN
C1_50C	26-Jun	0.079	0.030	0.550	0.630
C2_10A	26-Jun	0.084	0.090	2.500	2.600
C2_10B	26-Jun	0.385	0.003	1.620	2.000
C2_10C	26-Jun	0.100	0.990	1.550	1.700
C2_30B	26-Jun	10.700	0.050	1.460	12.000
C2_50A	26-Jun	24.800	0.060	0.940	26.000
C2_50B	26-Jun	9.540	0.003	0.850	10.000
M1_10A	26-Jun	9.860	0.040	2.210	12.000
M1_10B	26-Jun	1.980	0.070	2.190	4.200
M1_30B	26-Jun	1.540	0.090	1.170	2.700
M1_30C	26-Jun	0.024	0.003	0.740	0.760
M1_50B	26-Jun	1.380	0.050	0.810	2.200
M1_50C	26-Jun	4.340	0.070	1.150	5.500
M2_10A	26-Jun	10.900	0.350	2.070	13.000
M2_10B	26-Jun	0.115	0.020	1.430	1.600
M2_10C	26-Jun	0.313	0.050	2.160	2.500
M2_30A	26-Jun	15.200	0.040	1.490	17.000
C1_10A	11-Jul	1.090	0.030	1.640	2.700
C1_10B	11-Jul	0.172	0.020	2.180	2.400
C1_10C	11-Jul	0.012	0.003	1.110	1.100
C1_30B	11-Jul	0.081	0.150	1.160	1.200
C1_50A	11-Jul	18.800	0.030	0.980	20.000
C1_50C	11-Jul	0.074	0.210	1.780	1.900
C2_10A	11-Jul	0.025	0.003	1.900	1.900
C2_10B	11-Jul	0.527	0.003	1.190	1.800
C2_10C	11-Jul	0.031	0.030	0.620	0.650
C2_30B	11-Jul	6.330	0.010	1.070	7.400
C2_50A	11-Jul	25.700	0.010	0.380	26.000
C2_50B	11-Jul	13.400	0.020	1.110	15.000
C2_50C	11-Jul	2.260	0.040	0.430	2.700
M1_10A	11-Jul	9.440	0.003	2.280	12.000
M1_10B	11-Jul	2.230	0.010	1.970	4.200
M1_10C	11-Jul	0.031	0.080	1.540	1.600

Lysimeter	Date	Concentration (mg/l)			
		NO <sub>3</sub> <sup>-</sup>	NH <sub>4</sub>	TKN	TN
M1_30A	11-Jul	8.250	0.470	3.550	12.000
M1_30C	11-Jul	0.199	0.280	1.440	1.600
M1_50B	11-Jul	0.751	0.003	0.650	1.400
M1_50C	11-Jul	3.400	0.320	1.440	4.900
M2_10A	11-Jul	7.340	0.003	1.080	8.400
M2_10B	11-Jul	0.172	0.070	1.350	1.500
M2_10C	11-Jul	0.749	0.020	1.990	2.700
M2_30A	11-Jul	11.700	0.003	1.280	13.000
C1_10A	20-Jul	1.280	0.040	2.640	3.900
C1_10B	20-Jul	0.110	0.005	2.400	2.500
C1_30A	20-Jul	0.012	0.003	0.850	0.860
C1_50A	20-Jul	10.000	0.007	0.600	11.000
C1_50B	20-Jul	0.019	0.020	0.760	0.780
C1_50C	20-Jul	0.055	0.620	0.990	1.100
C2_10A	20-Jul	0.022	0.003	1.780	1.800
C2_10B	20-Jul	0.014	0.003	1.040	1.100
C2_10C	20-Jul	0.037	0.003	0.780	0.820
C2_30B	20-Jul	4.180	0.070	1.370	5.600
C2_30C	20-Jul	0.047	0.008	0.660	0.700
C2_50A	20-Jul	26.200	0.030	0.680	27.000
C2_50B	20-Jul	15.600	0.100	0.900	16.000
C2_50C	20-Jul	3.320	0.003	0.680	4.000
M1_10A	20-Jul	12.000	0.010	2.080	14.000
M1_10B	20-Jul	0.141	0.020	1.910	2.100
M1_10C	20-Jul	0.057	0.030	1.370	1.400
M1_30A	20-Jul	7.550	0.003	0.880	8.400
M1_30C	20-Jul	0.231	0.240	1.380	1.600
M1_50A	20-Jul	9.460	0.020	0.840	10.000
M1_50B	20-Jul	1.880	0.160	1.180	3.100
M1_50C	20-Jul	0.830	0.020	0.550	1.400
M2_10A	20-Jul	3.790	0.020	0.500	4.300
M2_10B	20-Jul	0.004	0.003	1.190	1.200
M2_10C	20-Jul	0.535	0.008	1.670	2.200

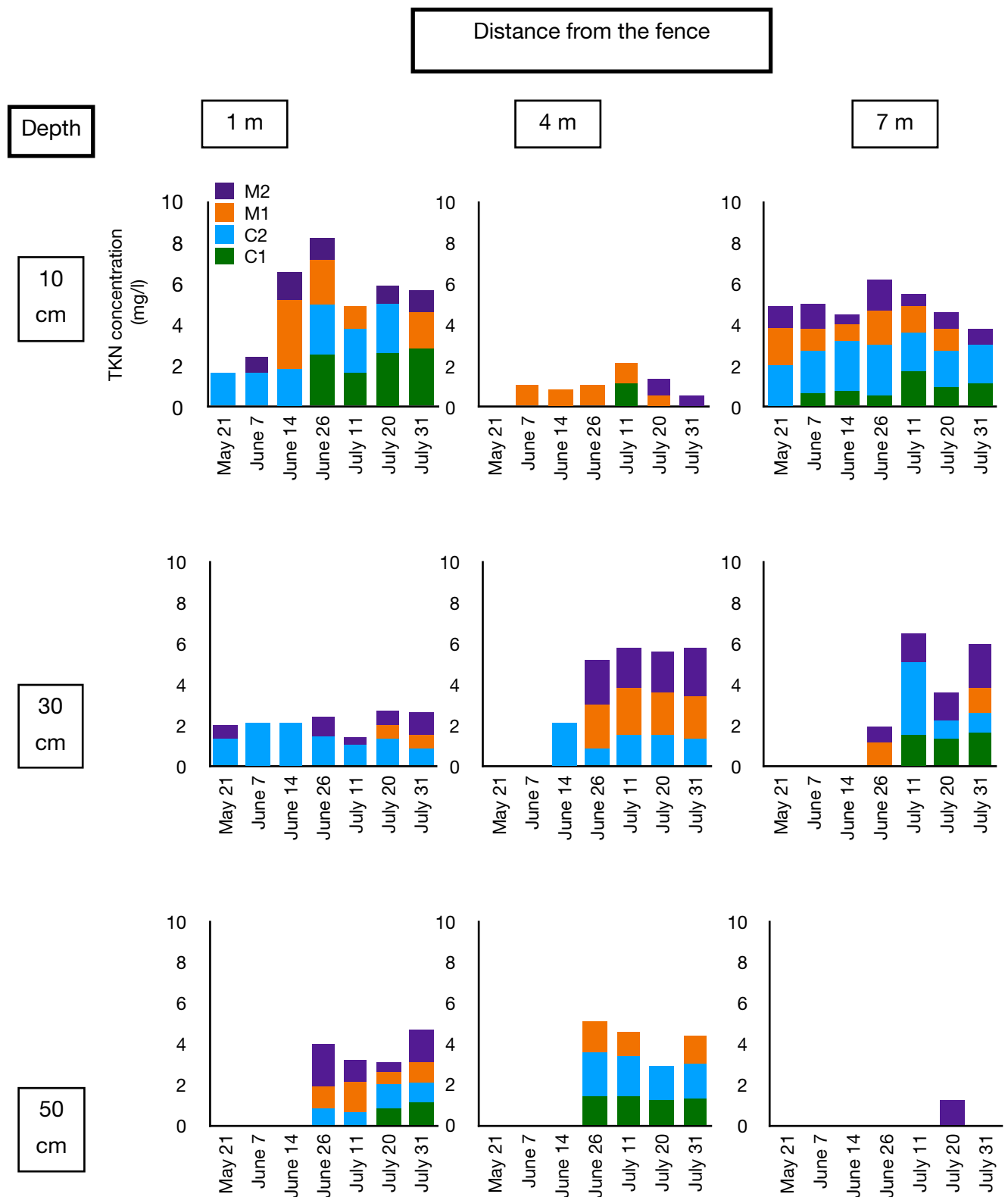
Lysimeter	Date	Concentration (mg/l)			
		NO <sub>3</sub> <sup>-</sup>	NH <sub>4</sub>	TKN	TN
M2_50C	20-Jul	1.720	0.520	1.260	3.000
C1_10A	31-Jul	0.174	0.020	2.870	3.100
C1_10B	31-Jul	0.108	0.010		
C1_10C	31-Jul	0.006	0.010	1.710	1.700
C1_30A	31-Jul	0.110	0.060	1.090	1.200
C1_50B	31-Jul	0.017	0.040	0.630	0.650
C1_50C	31-Jul	1.880	0.650	1.180	3.100
C2_10A	31-Jul	0.020	0.010	1.850	1.900
C2_10C	31-Jul	0.030	0.020	0.840	0.880
C2_30B	31-Jul	1.310	0.020	0.900	2.200
C2_30C	31-Jul	0.003	0.008	0.630	0.630
C2_50A	31-Jul	21.700	0.020	1.150	23.000
C2_50B	31-Jul	14.900	0.010	0.490	15.000
C2_50C	31-Jul	2.650	0.010	0.820	3.500
M1_10A	31-Jul	10.800	0.020	2.080	13.000
M1_10B	31-Jul	1.780	0.030	2.420	4.200
M1_10C	31-Jul	0.021	0.010	1.640	1.700
M1_30A	31-Jul	5.750	0.010	1.050	6.800
M1_30B	31-Jul	0.974	0.020	1.150	2.100
M1_30C	31-Jul	0.359	0.680	2.130	2.500
M1_50A	31-Jul	7.410	0.030	1.090	8.500
M1_50B	31-Jul	3.070	0.060	0.990	4.100
M1_50C	31-Jul	1.040	0.130	0.990	2.000
M2_10A	31-Jul	12.000	0.060	1.600	14.000
M2_10B	31-Jul	0.001	0.003	1.260	1.300
M2_10C	31-Jul	0.312	0.020	1.700	2.000
M2_30A	31-Jul	8.750	0.010	1.390	10.000

### Appendix 3a: 3-dimensional view of TN concentrations



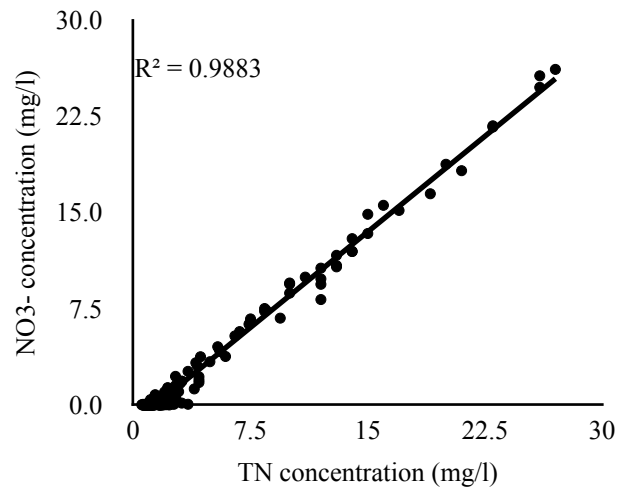
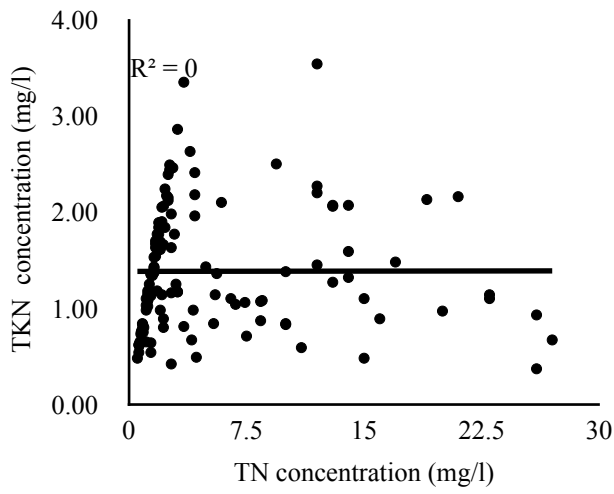
**Figure: 3-dimensional representation of TN concentrations through depth and distance from the fence, in the manuka and control riparian plots at Lake Waikare. Plots share axis scales and legend for readability.**

### Appendix 3b: 3-dimensional view of TKN concentrations

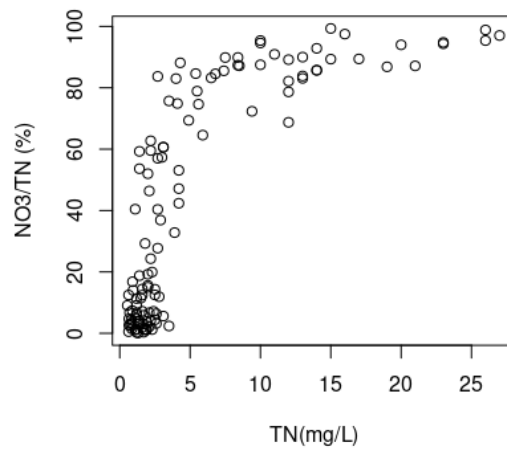
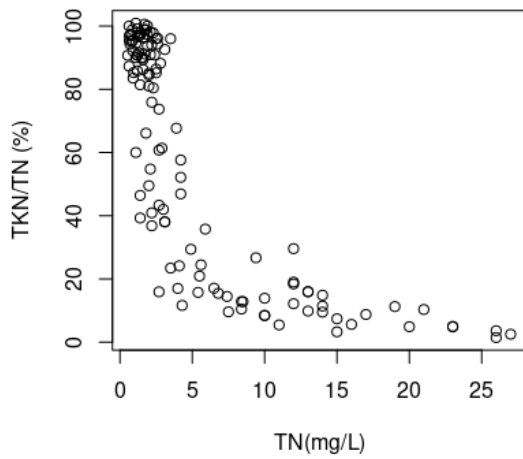


**Figure: 3-dimensional representation of TKN concentrations through depth and distance from the fence, in the manuka and control riparian plots at Lake Waikare. Plots share axis scales and legend for readability.**

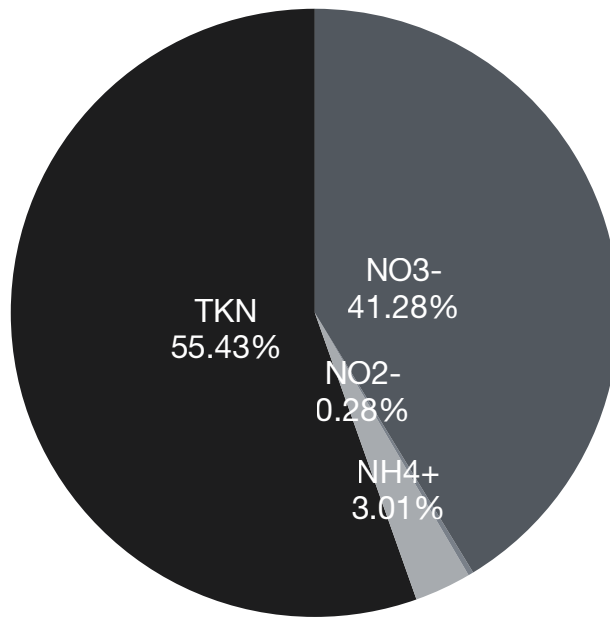
### Appendix 4a: Correlations between TN - TKN, and TN - NO<sub>3</sub><sup>-</sup>



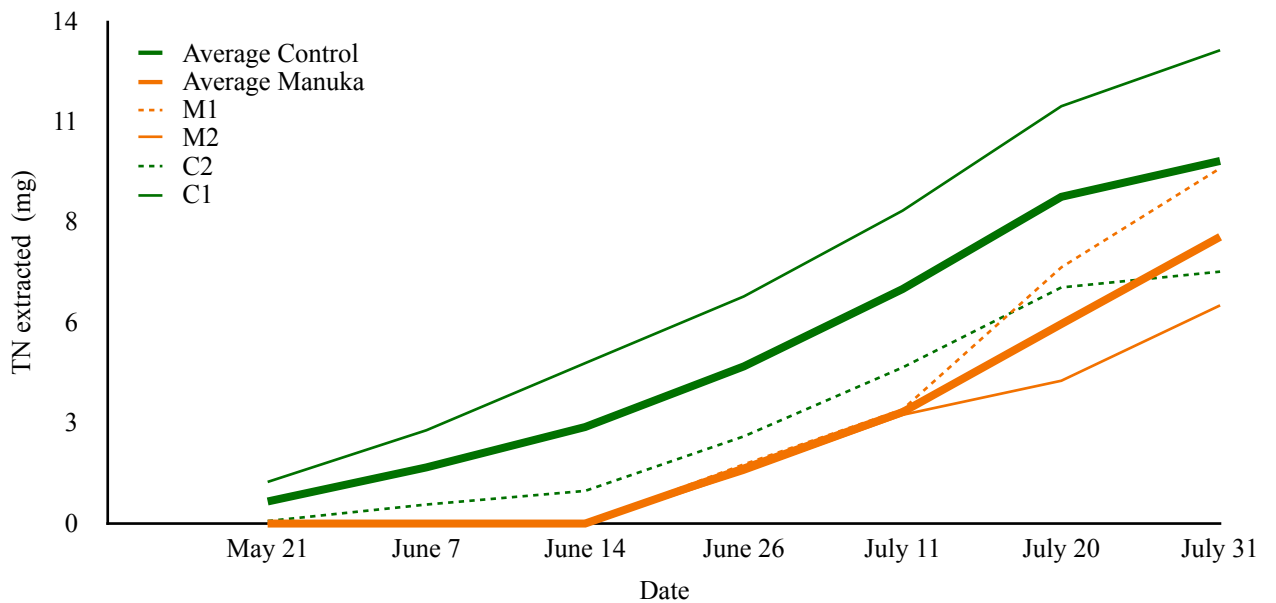
### Appendix 4b: TKN and NO<sub>3</sub><sup>-</sup> as percentages of TN



### Appendix 4c: Composition of TN in pore water samples



### Appendix 5: Cumulative TN extracted from each plot and vegetation type throughout the sampling period (mg).



## Bibliography

---

- Balesdent, J., & Mariotti, A. (1996). Measurement of Soil Organic Matter Turnover Using <sup>13</sup>C Natural Abundance. In T. W. Boutton & S. Yamasaki (Eds.), *Mass Spectrometry of Soils*: Marcel Dekker Inc.
- Barkle, G., Stenger, R., Moorhead, B., & Clague, J. (2021). The importance of the hydrological pathways in exporting nitrogen from grazed artificially drained land. *Journal of hydrology (Amsterdam)*, 597, 126218. doi:10.1016/j.jhydrol.2021.126218
- Barkle, G. F., Stenger, R., Burgess, C., & Wall, A. (2014). Quantifying groundwater contribution to stream flow generation in a steep headwater catchment. *Journal of Hydrology*, 1(53), 23-40.
- Barkle, G. F., Stenger, R., & Wöhling, T. h. (2014). Fate of urine nitrogen through a volcanic vadose zone. *Soil Research*(52), 658-670.
- Barnes, G. E. (2002). Water quality trends in selected shallow lakes in the Waikato region, 1995 – 2001. Retrieved from Hamilton, Waikato Regional Council (Environment Waikato):
- Bernhardt, E. S., Blaszcak, J. R., Ficken, C. D., Fork, M. L., Kaiser, K. E., & Seybold, E. C. (2017). Control Points in Ecosystems: Moving Beyond the Hot Spot Hot Moment Concept. *Ecosystems*, 20(4), 665-682. doi:10.1007/s10021-016-0103-y
- Bharati, L., Lee, K. H., Isenhardt, T. M., & Schultz, R. C. (2002). Soil-water infiltration under crops, pasture, and established riparian buffer in Midwestern USA. *Agroforestry systems*, 56(3), 249-257. doi:10.1023/A:1021344807285
- Billen, G., Garnier, J., & Lassaletta, L. (2013). The nitrogen cascade from agricultural soils to the sea: modelling nitrogen transfers at regional watershed and global scales. *Philosophical transactions. Biological sciences*, 368(1621), 20130123-20130123. doi:10.1098/rstb.2013.0123
- Bowes, M. J., Jarvie, H. P., Naden, P. S., Old, G. H., Scarlett, P. M., Roberts, C., . . . Collins, A. L. (2014). Identifying priorities for nutrient mitigation using

- river concentration–flow relationships: The Thames basin, UK. *Journal of Hydrology*, 517, 1-12. doi:<https://doi.org/10.1016/j.jhydrol.2014.03.063>
- Bowman, H. J. (2020). The influence of New Zealand native vegetation on rhizosphere chemical concentration and speciation in riparian zones. (Master of Science in Environmental Science). University of Canterbury, Christchurch, New Zealand.
- Brady, N. C., & Weil, R. R. (2008). *The Nature and Properties of Soil* (V. R. Anthony Ed. 14th Edition ed.). New Jersey, United States: Pearson Education International.
- Cabrera Corral, F. J., Bonachela Castaño, S., Fernández Fernández, M. D., Granados García, M. R., & López Hernández, J. C. (2016). Lysimetry methods for monitoring soil solution electrical conductivity and nutrient concentration in greenhouse tomato crops. *Agricultural Water Management*, 178, 171-179. doi:<https://doi.org/10.1016/j.agwat.2016.09.024>
- Chibnall, E. J. (2013). Contribution of Dissolved Organic Carbon Leaching to the Annual Carbon Budget of a Dairy Farm (Thesis, Master of Science (MSc)). University of Waikato, Hamilton, New Zealand.,
- Clague, J. C., Stenger, R., & Clough, T. J. (2013). The Impact of Relict Organic Materials on the Denitrification Capacity in the Unsaturated–Saturated Zone Continuum of Three Volcanic Profiles. *Journal of Environmental Quality*, 42(1), 145-154. doi:<https://doi.org/10.2134/jeq2012.0239>
- Clague, J. C., Stenger, R., & Clough, T. J. (2015). Evaluation of the stable isotope signatures of nitrate to detect denitrification in a shallow groundwater system in New Zealand. *Agriculture Ecosystems & Environment*, 202, 188-197. doi:10.1016/j.agee.2015.01.011
- Collier, K. J., Cooper, A. B., Davies-Colley, R. J., Rutherford, J. C., Smith, C. M., & Williamson, R. B. (1995). *Managing riparian zones: A contribution to protecting New Zealand's rivers and streams*. Retrieved from Wellington, New Zealand.:
- Collins, K. E., Doscher, C., Rennie, H. G., & Ross, J. G. (2013). The Effectiveness of Riparian 'Restoration' on Water Quality-A Case Study of Lowland Streams

- in Canterbury, New Zealand. *Restoration ecology*, 21(1), 40-48. doi: 10.1111/j.1526-100X.2011.00859.x
- Connor, S., Nelson, P. N., Armour, J. D., & Hénault, C. (2013). Hydrology of a forested riparian zone in an agricultural landscape of the humid tropics. *Agriculture, ecosystems & environment*, 180, 111-122. doi:10.1016/j.agee.2011.12.006
- Cosgrove, R. (2019). Water Pollution is now New Zealander's Number One Concern. Retrieved from <https://fishandgame.org.nz/news/water-pollution-is-now-new-zealanders-number-one-concern/>
- Council, W. R. (2018). Soil Quality. Retrieved from <https://www.waikatoregion.govt.nz/environment/land-and-soil/land-and-soil-monitoring/soil-quality-report-card/>
- Cuttle, S. P., & Bourne, P. C. (1993). Uptake and leaching of nitrogen from artificial urine applied to grassland on different dates during the growing season. *Plant and Soil*, 150(1), 77-86. doi:doi:10.1007/BF00779178
- Daigneault, A. J., Eppink, F. V., & Lee, W. G. (2017). A national riparian restoration programme in New Zealand: Is it value for money? *Journal of Environmental Management*, 187, 166-177. doi:10.1016/j.jenvman.2016.11.013
- Davie, T. (2004). Soil water, runoff and streamflow generation. In J. Harding, Mosley, P, Pearson, C and Sorrel, B. (Ed.), *Freshwaters of New Zealand*. Christchurch: Caxton Press.
- Dean-Speirs, T. a. N., Keri. (2014). Waikato region shallow lakes management plan. Retrieved from Hamilton: <https://www.waikatoregion.govt.nz/services/publications/tr201459/>
- Dollery, R., Li, S., & Dickinson, N. M. (2019). Nutrient-enriched soils and native N-fixing plants in New Zealand. *Journal of Plant Nutrition and Soil Science*, 182(1), 104-110. doi:<https://doi.org/10.1002/jpln.201800482>
- Domagalski, J. L., Phillips, S. P., Bayless, E. R., Zamora, C., Kendall, C., Wildman Jr, R. A., & Hering, J. G. (2008). Influences of the unsaturated, saturated, and riparian zones on the transport of nitrate near the Merced River, California,

USA. *Hydrogeology journal*, 16(4), 675-690. doi:10.1007/s10040-007-0266-x

Downward, R. L. (2013). Nitrification inhibition by common plants in New Zealand's agricultural landscapes. (BSc (Hons)). Lincoln University, New Zealand.

Dybala, K. E., Steger, K., Walsh, R. G., Smart, D. R., Gardali, T., Seavy, N. E., & Macinnis-Ng, C. (2019). Optimizing carbon storage and biodiversity co-benefits in reforested riparian zones. *The Journal of applied ecology*, 56(2), 343-353. doi:10.1111/1365-2664.13272

Dymond, J. R., Ausseil, A. G., Parfitt, R. L., Herzig, A., & McDowell, R. W. (2013). Nitrate and phosphorus leaching in New Zealand: a national perspective. *New Zealand Journal of Agricultural Research*, 56(1), 49-59. doi:10.1080/00288233.2012.747185

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

Environment-Southland. (2020). A guide to riparian planting options and costs. In E. Southland (Ed.).

Esperschuetz, J., Anderson, C., Bulman, S., Katamian, O., Horswell, J., Dickinson, N. M., & Robinson, B. H. (2017). Response of *Leptospermum scoparium*, *Kunzea robusta* and *Pinus radiata* to contrasting biowastes. *Science of The Total Environment*, 587-588, 258-265. doi:<https://doi.org/10.1016/j.scitotenv.2017.02.134>

Esperschuetz, J., Balaine, N., Clough, T., Bulman, S., Dickinson, N. M., Horswell, J., & Robinson, B. H. (2017). The potential of *L. scoparium*, *K. robusta* and *P. radiata* to mitigate N-losses in silvopastoral systems. *Environmental Pollution*, 225, 12-19. doi:<https://doi.org/10.1016/j.envpol.2017.03.042>

Evans, D. M., Schoenholtz, S. H., Wigington Jr., P. J., & Griffith, S. M. (2011). Nitrogen Mineralization in Riparian Soils along a River Continuum within a Multi-Land-Use Basin. *Soil Science Society of America Journal*, 75(2), 719-728. doi:<https://doi.org/10.2136/sssaj2010.0162>

- FAOSTAT. (2021). Food and Agriculture Database. Retrieved from <http://www.fao.org/faostat/en/#home>
- Fennessy, S., & Cronk, J. (1997). The Effectiveness and Restoration Potential of Riparian Ecotones for the Management of Nonpoint Source Pollution, Particularly Nitrate. *Critical Reviews in Environmental Science and Technology*, 27, 285-317. doi:10.1080/10643389709388502
- Franklin, H. M., Dickinson, N. M., Esnault, C. J. D., & Robinson, B. H. (2015). Native plants and nitrogen in agricultural landscapes of New Zealand. *Plant and Soil*, 394(1), 407-420. doi:10.1007/s11104-015-2622-2
- Franklin, H. M., Robinson, B. H., & Dickinson, N. M. (2019). Plants for nitrogen management in riparian zones: A proposed trait-based framework to select effective species. *Ecological management & restoration*, 20(3), 202-213. doi:10.1111/emr.12380
- Franklin, H. M., Woods, R. R., Robinson, B., & Dickinson, N. (2017). Nitrous oxide emissions following dairy shed effluent application beneath *Kunzea robusta* (Myrtaceae) trees. *Ecological Engineering*, 99, 473-478. doi:<https://doi.org/10.1016/j.ecoleng.2016.11.064>
- Fraser, P. M., Curtin, D., Harrison-kirk, T., Meenken, E. D., Beare, M. H., Tabley, F., . . . Francis, G. S. (2013). Winter nitrate leaching under different tillage and winter cover crop management practices. *Soil Science Society of America Journal*, 77(4), 1391-1401.
- Galloway, J. M., Aber, J. D., Erisman, J. M., Seitzinger, S. P., Howarth, R. W., Cowling, E. B., & Cosby, J. B. (2003). The Nitrogen Cascade. *Bioscience*, 53(4), 341-356. doi:10.1641/0006-3568(2003)053[0341:TNC]2.0.CO
- 2
- Ghestem, M., Sidle, R. C., & Stokes, A. (2011). The Influence of Plant Root Systems on Subsurface Flow: Implications for Slope Stability. *Bioscience*, 61(11), 869-879. doi:10.1525/bio.2011.61.11.6
- Gutierrez-Gines, M. J., Alizadeh, H., Alderton, E., Ambrose, V., Meister, A., Robinson, B. H., . . . Horswell, J. (2021). Phytoremediation of microbial contamination

in soil by New Zealand native plants. *Applied Soil Ecology*, 167, 104040.  
doi:<https://doi.org/10.1016/j.apsoil.2021.104040>

Halford, S., Gutiérrez-Ginés, M. J., Rees, A., Alderton, I., Schaller, K., Smith, R., & Horswell, J. (2021). Effect of mānuka (*Leptospermum scoparium*) on nitrogen and *Escherichia coli* reductions in soils: a field experiment. *Plant and Soil*. doi:10.1007/s11104-021-05035-3

Haycock, N., & Pinay, G. (1992). Nitrate retention in grass and poplar vegetated buffer strips during winter. *Journal of Environmental Quality*.

Howard-Williams, C., Davies-Colley, R., Rutherford, K. and Wilcock, R. (2010). Diffuse pollution and freshwater degradation: New Zealand perspectives— Issues and Solutions to Diffuse Pollution. Paper presented at the IWA Diffuse Pollution Specialist Group, Paris. [https://www.landcareresearch.co.nz/uploads/public/researchpubs/Howard\\_williams\\_2013\\_Diffuse\\_pollution\\_and\\_freshwater\\_degradation.pdf](https://www.landcareresearch.co.nz/uploads/public/researchpubs/Howard_williams_2013_Diffuse_pollution_and_freshwater_degradation.pdf)

Keim, R. F., & Skaugset, A. E. (2003). Modelling effects of forest canopies on slope stability. *Hydrological processes*, 17(7), 1457-1467. doi:10.1002/hyp.5121

Kuman, D. (2019). Determination of Optimum Moisture Content and degradation of shear strength over time for Hamilton Ash materials. (Master of Science in Earth Sciences). University of Waikato, Hamilton, New Zealand. Retrieved from <https://researchcommons.waikato.ac.nz/bitstream/handle/10289/13318/thesis.pdf?sequence=4>

Kumar, S., Anderson, S. H., Bricknell, L. G., Udawatta, R. P., & Gantzer, C. J. (2008). Soil hydraulic properties influenced by agroforestry and grass buffers for grazed pasture systems. *Journal of soil and water conservation*, 63(4), 224-231. doi:10.2489/jswc.63.4.224

Kumar, S., Anderson, S. H., Udawatta, R. P., & Kallenbach, R. L. (2012). Water infiltration influenced by agroforestry and grass buffers for a grazed pasture system. *Agroforestry systems*, 84(3), 325-335. doi:10.1007/s10457-011-9474-4

- Lal, R. (2004). Soil carbon sequestration to mitigate climate change. *Geoderma*, 123(1), 1-22. doi:10.1016/j.geoderma.2004.01.032
- Lawrence, L., & Ridley, G. (2018). Lake Waikare and Whangamarino Wetland Catchment Management Plan. Retrieved from
- Lilburne, L., Sparling, G., & Schipper, L. (2004). Soil quality monitoring in New Zealand: development of an interpretative framework. *Agriculture, ecosystems & environment*, 104(3), 535-544. doi:10.1016/j.agee.2004.01.020
- Lind, L., Hasselquist, E. M., & Laudon, H. (2019). Towards ecologically functional riparian zones: A meta-analysis to develop guidelines for protecting ecosystem functions and biodiversity in agricultural landscapes. *Journal of Environmental Management*, 249, 109391. doi:<https://doi.org/10.1016/j.jenvman.2019.109391>
- Liu, W., Xiong, Z., Liu, H., Zhang, Q., & Liu, G. (2016). Catchment agriculture and local environment affecting the soil denitrification potential and nitrous oxide production of riparian zones in the Han River Basin, China. *Agriculture, Ecosystems & Environment*, 216, 147-154. doi:<https://doi.org/10.1016/j.agee.2015.10.002>
- LMF. (2009). Land and Soil Monitoring: A guide for SoE and regional council reporting. Retrieved from <https://www.mfe.govt.nz/publications/land/land-and-soil-monitoring-guide-soe-and-regional-council%20reporting>.
- Lowe, D. J. (2019a). Origin and classification of Kainui silt loam: update on the leopard that changed its spots. *New Zealand Soil News*, 67(1), 75-85.
- Lowe, D. J. (2019b). Using soil stratigraphy and tephrochronology to understand the origin, age, and classification of a unique Late Quaternary tephra-derived Ultisol in Aotearoa New Zealand.
- Lyu, C., Li, X., Yuan, P., Song, Y., Gao, H., Liu, X., . . . Yu, H. (2021). Nitrogen retention effect of riparian zones in agricultural areas: A meta-analysis. *Journal of cleaner production*, 315, 128143. doi:10.1016/j.jclepro.2021.128143
- Mackay. (2013). Soil Quality Indicators: The Next Generation. Retrieved from

- Marden, M., Rowan, D., & Phillips, C. (2005). Stabilising characteristics of New Zealand indigenous riparian colonising plants. *Plant and soil*, 278(1/2), 95-105. doi:10.1007/s11104-004-7598-2
- Manaaki Whenua-Landcare Research New Zealand. (2021). Soils Portal: Describing Soils. Retrieved from <https://soils.landcareresearch.co.nz/describing-soils/nzsc/>
- Mayer, P. M., Reynolds Jr., S. K., McCutchen, M. D., & Canfield, T. J. (2007). Meta-Analysis of Nitrogen Removal in Riparian Buffers. *Journal of Environmental Quality*, 36(4), 1172-1180. doi:<https://doi.org/10.2134/jeq2006.0462>
- McClain, M. E., Boyer, E. W., Dent, C. L., Gergel, S. E., Grimm, N. B., Groffman, P. M., . . . Pinay, G. (2003). Biogeochemical Hot Spots and Hot Moments at the Interface of Terrestrial and Aquatic Ecosystems. *Ecosystems*, 6(4), 301-312. doi:10.1007/s10021-003-0161-9
- 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, prepared by AgResearch, NIWA and the University of Waikato for the Ministry for the Environment. Retrieved from
- McGlynn, B. L., McDonnel, J. J., & Brammer, D. D. (2002). A review of the evolving perceptual model of hillslope flowpaths at the Maimai catchments, New Zealand. *Journal of hydrology (Amsterdam)*, 257(1-4), 1-26. doi:10.1016/S0022-1694(01)00559-5
- McLaren, R. G., & Cameron, K. C. (1996). *Soil Science — Sustainable Production and Environmental Protection (Second Edition ed.)*. Australia: Oxford University Press.
- Meek B.D., R. E. R., Carter L.M., DeTar W.R. and Urie A.L. (1992). Infiltration rate of a sandy loam soil: effects of traffic, tillage, and plant roots. *Soil Sci Soc Am J*, 56, 908-913.
- Menz, F. C., & Seip, H. M. (2004). Acid rain in Europe and the United States: an update. *Environmental Science & Policy*, 7(4), 253-265. doi:<https://doi.org/10.1016/j.envsci.2004.05.005>

- MfE. (2018). Climate Change Projections for New Zealand: Atmosphere Projections Based on Simulations from the IPCC Fifth Assessment, 2nd Edition. Retrieved from Wellington:
- MfE. (2020). Our Freshwater. Retrieved from <https://www.mfe.govt.nz/sites/default/files/media/Environmental%20reporting/our-freshwater-report-2020.pdf>
- MfE. (2021a). New Zealand's Greenhouse Gas Inventory 1990-2019. Retrieved from New Zealand:
- MfE. (2021b). Our Land. Retrieved from
- Minasny, M., Mcbratney, Angers, Arrouays, Chambers , Winowiecki. (2017). Soil carbon 4 per mille. *Geoderma*, 292 (C), 59-86.
- Miyamoto, T., Kameyama, K. and Iwata, Y. . (2015). Monitoring Electrical Conductivity and Nitrate Concentrations in an Andisol Field Using Time Domain Reflectometry. *JARQ*, 49(3), 261-267.
- MPI. (2016). Stock Exclusion Costs Report. Retrieved from <https://www.mpi.govt.nz/dmsdocument/16537-Ministry-for-Primary-Industries-Stock-Exclusion-Costs-Report>
- MPI. (2017). Ministry for Primary Industries 2016 apiculture monitoring programme. Retrieved from Wellington, New Zealand.:
- Neilen, A. D., Chen, C. R., Faggotter, S. J., Ellison, T. L., & Burford, M. A. (2016). Biotic and abiotic controls on nitrogen leaching losses into waterways during successive bovine urine application to soil. *Journal of Environmental Management*, 176, 11-20. doi:<https://doi.org/10.1016/j.jenvman.2016.03.024>
- Neilen, A. D., Chen, C. R., Parker, B. M., Faggotter, S. J., & Burford, M. A. (2017). Differences in nitrate and phosphorus export between wooded and grassed riparian zones from farmland to receiving waterways under varying rainfall conditions. *Science of The Total Environment*, 598, 188-197. doi:<https://doi.org/10.1016/j.scitotenv.2017.04.075>
- New Zealand Government. (2020). National Policy Statement for Freshwater Management 2020,

- Nieminen, T. M., Derome, K., Meeseburg, H., & De Vos, B. (2013). In M. Ferretti & R. Fischer (Eds.), *Forest Monitoring : Methods for Terrestrial Investigations in Europe with an Overview of North America and Asia*. London, UNITED KINGDOM: Elsevier.
- Nikau, T. (2021, August 19th 2021). [Health warnings on Lake Waikare].
- O'Dwyer, E. (2020a). Scientists track the changing colour in one of New Zealand's most polluted lakes. Retrieved from <https://www.stuff.co.nz/environment/123144492/scientists-track-the-changing-colour-in-one-of-new-zealands-most-polluted-lakes>
- O'Dwyer, E. (2020b). The Last Lake: Waikato lakes among most polluted in the world - scientist. Stuff. Retrieved from <https://www.stuff.co.nz/national/123075186/the-last-lake-waikato-lakes-among-most-polluted-in-the-world--scientist>
- O'Dwyer, E. (2020). Lake Waikare turns green as toxic algae flourish in Summer. Stuff. Retrieved from <https://www.stuff.co.nz/environment/118356458/lake-waikare-turns-green-as-toxic-algae-flourish-in-summer>
- O'Dwyer, E. (2021). Council's three years of discharge into one of nation's most polluted lakes. Stuff. Retrieved from <https://www.stuff.co.nz/national/125527768/councils-three-years-of-discharge-into-one-of-nations-most-polluted-lakes>
- Parkyn, S. M., Davies-Colley, R. J., Halliday, N. J., Costley, K. J., & Croker, G. F. (2003). Planted Riparian Buffer Zones in New Zealand: Do They Live Up to Expectations? *Restoration Ecology*, 11(4), 436-447. doi:<https://doi.org/10.1046/j.1526-100X.2003.rec0260.x>
- Parlance, J. Y., Steenhuis, S. T., Haverkamp, R., Barry, D. A., Culligan, P. J., Hogarth, W. L., . . . Stagnitti, F. (1999). Soil Properties and Water Movement. In M. B. P. a. J. W. Hopmans (Ed.), *Vadose Zone Hydrology*. Oxford University: Oxford University Press.
- Phillips, J. D. (1989). An evaluation of the factors determining the effectiveness of water quality buffer zones. *Journal of hydrology (Amsterdam)*, 107(1), 133-145. doi:10.1016/0022-1694(89)90054-1

- Prosser, J. A., Anderson, C. W. N., Horswell, J., & Speir, T. W. (2014). Can manuka (*Leptospermum scoparium*) antimicrobial properties be utilised in the remediation of pathogen contaminated land? *Soil Biology and Biochemistry*, 75, 167-174. doi:<https://doi.org/10.1016/j.soilbio.2014.04.003>
- Prosser, J. A., Woods, R. R., Horswell, J., & Robinson, B. H. (2016). The potential in-situ antimicrobial ability of Myrtaceae plant species on pathogens in soil. *Soil Biology and Biochemistry*, 96, 1-3. doi:<https://doi.org/10.1016/j.soilbio.2015.12.007>
- RAMSAR. (1992). Ramsar sites information services: Whangamarino. Retrieved from <https://rsis.ramsar.org/ris/443>
- Rees, R. M., Augustin, J., Alberti, G., Ball, B. C., Boeckx, P., Cantarel, A., . . . Wuta, M. (2013). Nitrous oxide emissions from European agriculture - An analysis of variability and drivers of emissions from field experiments. *Biogeosciences*, 10(4), 2671-2682. doi:10.5194/bg-10-2671-2013
- Reeves, P., & Hancock, N. (2012). Ecological Impacts of the Flood Control Scheme on Lake Waikare and the Whangamarino Wetland, and Potential Mitigation Options. Retrieved from
- Russow, R., Tauchnitz, N., Spott, O., Mothes, S., Bernsdorf, S., & Meissner, R. (2013). Nitrate turnover in a peat soil under drained and rewetted conditions: results from a [<sup>15</sup>N]nitrate-bromide double-tracer study: Dedicated to Professor Dr habil. Hans Faust on the occasion of his 85th birthday. *Isotopes in environmental and health studies*, 49(4), 438-453. doi:10.1080/10256016.2013.831089
- Saggar, S., Tate, K. R., Giltrap, D. L., & Singh, J. (2008). Soil-atmosphere exchange of nitrous oxide and methane in New Zealand terrestrial ecosystems and their mitigation options: a review. *Plant and Soil*, 309(1), 25-42. doi:10.1007/s11104-007-9421-3
- Schimel, J., & Holland, E. A. (2005). Global Gases. In J. J. Fuhrman, D. M. Sylvia, P. G. Hartel, & D. A. Zuberer (Eds.), *Principles and Applications of Soil Microbiology* (pp. 491-509). Upper Saddle River: Pearson-Prentice Hall.

- Schultz, R. C., Colletti, J. P., Isenhardt, T. M., Simpkins, W. W., Mize, C. W., & Thompson, M. L. (1995). Design and placement of a multi-species riparian buffer strip system. *Agroforestry systems*, 29(3), 201-226. doi:10.1007/BF00704869
- ScienceAdvisoryPanel. (2021). Overseer whole-model review Assessment of the model approach. Retrieved from <https://www.mpi.govt.nz/dmsdocument/46360-Overseer-whole-model-review-Assessment-of-the-model-approach>
- SDEC. (n.d.). Ceramic suction lysimeters Ø 31mm & Ø 63mm - SPS 200. Crop and soil science. Retrieved from <https://agronomie.sdec-france.com/ceramic-succion-lysimeters-31-63-mm-for-crop-science.html>
- Seyedalikhani, S., Esperschuetz, J., Dickinson, N. M., Hofmann, R., Breitmeyer, J., Horswell, J., & Robinson, B. H. (2019). Biowastes to augment the essential oil production of *Leptospermum scoparium* and *Kunzea robusta* in low-fertility soil. *Plant Physiology and Biochemistry*, 137, 213-221. doi:<https://doi.org/10.1016/j.plaphy.2019.02.008>
- Sigua, G. C., Stone, K. C., Bauer, P. J., Szogi, A. A., & Shumaker, P. D. (2017). Impacts of irrigation scheduling on pore water nitrate and phosphate in coastal plain region of the United States. *Agricultural water management*, 186, 75-85. doi:10.1016/j.agwat.2017.02.016
- Singh, S. K., & Stenger, R. (2018). Indirect Methods to Elucidate Water Flows and Contaminant Transfer Pathways through Meso-scale Catchments – a Review. *Environmental Processes*, 5(4), 683-706. doi:10.1007/s40710-018-0331-6
- Skiba, U. (2008). Denitrification. In S. E. Jørgensen & B. D. Fath (Eds.), *Encyclopedia of Ecology* (pp. 866-871). Oxford: Academic Press.
- Sparling, G. P. (2008). Provisional targets for soil quality indicators in New Zealand. Lincoln, N.Z.: Lincoln, N.Z. : Manaaki Whenua - Landcare Research New Zealand.
- Sparling, G. P., & Schipper, L. A. (2002). Soil Quality at a National Scale in New Zealand. *Journal of Environmental Quality*, 31(6), 1848-1857. doi:<https://doi.org/10.2134/jeq2002.1848>

- Srinivasan, M. S., Muirhead, R. W., Singh, S. K., Monaghan, R. M., Stenger, R., Close, M. E., . . . Hodson, R. (2020). Development of a national-scale framework to characterise transfers of N, P and Escherichia coli from land to water. *New Zealand Journal of Agricultural Research*, 1-28. doi: 10.1080/00288233.2020.1713822
- StatsNZ. (2020). Agriculture. Retrieved from <https://www.stats.govt.nz/topics/agriculture>
- Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., . . . Sörlin, S. (2015). Planetary boundaries: Guiding human development on a changing planet. *Science*, 347(6223), 1259855. doi:10.1126/science.1259855
- Stenger, R., Rivas, A., Wilson, S., Friedel, M., Barkle, G., Clague, J., . . . Kusabs, I. (2019). Critical Pathways Programme: Unravelling Sub-Catchment Scale Nitrogen Delivery to Waterways.
- Stenger, R., Wilson, S., Barkle, G., Close, M., Woodward, S., Burberry, L., . . . Lilburne, L. (2016). Transfer pathways programme (TPP)— New research to determine pathway-specific contaminant transfer from the land to water bodies.
- Stephens, J. M. C., Molan, P.C. & Clarkson, B.D. (2005). A review of *Leptospermum scoparium* (Myrtaceae) in New Zealand. *New Zealand Journal of Botany*, 43, 431-449.
- Stockmann, U., Padarian, J., McBratney, A., Minasny, B., de Brogniez, D., Montanarella, L., Hong, & S.Y., R., B.G., Field, D.J. (2015). Global soil organic carbon assessment. *Glob. Food Sec.*, 6, 9-16.
- Tanner, C., Muirhead, R. W., Burger, D., Carrick, S., Close, M., Monaghan, R. M., . . . Tipa, G. (2017). A framework for understanding the linkages between land and water quality impacts. Retrieved from <http://flrc.massey.ac.nz/publications.html>
- Taylor, M. D., Cox, N., Littler, R., & Drewry, J. J. (2017). Trends in soil quality monitoring data in the Waikato region 1995- 2015. Retrieved from

- Taylor, M. D., Kim, N. D., Hill, R. B., & Chapman, R. (2010). A review of soil quality indicators and five key issues after 12 yr soil quality monitoring in the Waikato region. *Soil use and management*, 26(3), 212-224. doi:10.1111/j.1475-2743.2010.00276.x
- Taylor, M. D., Mulholland, M., & Thornburrow, D. (2009). Infiltration characteristics of soils under forestry and agriculture in the Upper Waikato catchment. Hamilton East [N.Z: Environment Waikato.
- TheAgribusinessGroup. (2016). Ministry for Primary Industries Stock Exclusion Costs Report. Retrieved from
- Trolove, S., Thomas, S., van der Klei, G., Beare, M., Cichota, R., & Meenken, E. (2019). Nitrate leaching losses during pasture renewal – Effects of treading, urine, forages and tillage. *Science of The Total Environment*, 651, 1819-1829. doi:<https://doi.org/10.1016/j.scitotenv.2018.09.333>
- Trotter, C., Tate, K. R., Scott, N., Townsend, J., Wilde, H., Lambie, S., . . . Pinkney, T. (2005). Afforestation/reforestation of New Zealand marginal pasture lands by indigenous shrublands: The potential for Kyoto forest sinks. <http://dx.doi.org/10.1051/forest:2005077>, 62. doi:10.1051/forest:2005077
- Tupuhi, G. (2021). [Lake waikare as a food bowl].
- Tupuhi, G., & Baker, V. (2017). Measuring the benefits of riparian vegetation restoration on the health and well-being of Lake Waikare and surrounding hapū and whānau. Retrieved from
- Uchida, Y., & Akiyama, H. (2013). Mitigation of postharvest nitrous oxide emissions from soybean ecosystems: a review. *Soil science and plant nutrition* (Tokyo), 59(4), 477-487. doi:10.1080/00380768.2013.805433
- van Groenigen, J. W., Huygens, D., Boeckx, P., Kuyper, T. W., Lubbers, I. M., Rütting, T., & Groffman, P. M. (2015). The soil N cycle: new insights and key challenges. *SOIL*, 1(1), 235-256. doi:10.5194/soil-1-235-2015
- Venterink, H. O., Davidsson, T. E., Kiehl, K., & Leonardson, L. (2002). Impact of drying and re-wetting on N, P and K dynamics in a wetland soil. *Plant and soil*, 243(1), 119-130. doi:10.1023/A:1019993510737

- Vidon, P. G. F., & Hill, A. R. (2004). Landscape controls on nitrate removal in stream riparian zones: landscape controls on nitrate removal. *Water resources research*, 40(3). doi:10.1029/2003WR002473
- Villegas, D., Arevalo, A., Nuñez, J., Mazabel, J., Subbarao, G., Rao, I., . . . Arango, J. (2020). Biological Nitrification Inhibition (BNI): Phenotyping of a Core Germplasm Collection of the Tropical Forage Grass *Megathyrsus maximus* Under Greenhouse Conditions. *Frontiers in Plant Science*, 11(820). doi: 10.3389/fpls.2020.00820
- Vogeler, I., Hansen, E. M., Nielsen, S., Labouriau, R., Cichota, R., Olesen, J. E., & Thomsen, I. K. (2020). Nitrate leaching from suction cup data: Influence of method of drainage calculation and concentration interpolation. *Journal of environmental quality*, 49(2), 440-449. doi:10.1002/jeq2.20020
- Wallace, C. (2017). Ground-breaking science set to help Te Kauwhata lake restoration. Stuff. Retrieved from <https://www.stuff.co.nz/waikato-times/news/93676514/groundbreaking-science-set-to-help-te-kauwhata-lake-restoration>
- Ward, R. C., & Robinson, M. (2000). *Principles of Hydrology* (4th Edition ed.). Berkshire, England: McGraw-Hill International Limited.
- Weihermüller, L., Siemens, J., Deurer, M., Knoblauch, S., Rupp, H., Göttlein, A., & Pütz, T. (2007). In Situ Soil Water Extraction: A Review. *Journal of Environmental Quality*, 36(6), 1735-1748. doi:<https://doi.org/10.2134/jeq2007.0218>
- Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., . . . Murray, C. J. L. (2019). Food in the Anthropocene: the EAT–Lancet Commission on healthy diets from sustainable food systems. *The Lancet*, 393(10170), 447-492. doi:[https://doi.org/10.1016/S0140-6736\(18\)31788-4](https://doi.org/10.1016/S0140-6736(18)31788-4)
- Woodward, S., Barker, D. J., & Zyskowski, R. F. (2001). A practical model for predicting soil water deficit in New Zealand pastures. *New Zealand Journal of Agricultural Research*, 44, 91-109. doi:10.1080/00288233.2001.9513464
- Woodward, S. J. R., Stenger, R., & Bidwell, V. J. (2013). Dynamic analysis of stream flow and water chemistry to infer subsurface water and nitrate fluxes in a

lowland dairying catchment. *Journal of Hydrology*, 505, 299-311.  
doi:<https://doi.org/10.1016/j.jhydrol.2013.07.044>

Zhao, S., Zhou, N., & Liu, X. (2016). Occurrence and controls on transport and transformation of nitrogen in riparian zones of Dongting Lake, China. *Environmental Science and Pollution Research*, 23(7), 6483-6496. doi: 10.1007/s11356-015-5865-9

Zhou, X., Helmers, M. J., Asbjornsen, H., Kolka, R., & Tomer, M. D. (2010). Perennial Filter Strips Reduce Nitrate Levels in Soil and Shallow Groundwater after Grassland-to-Cropland Conversion. *Journal of environmental quality*, 39(6), 2006-2015. doi:10.2134/jeq2010.0151