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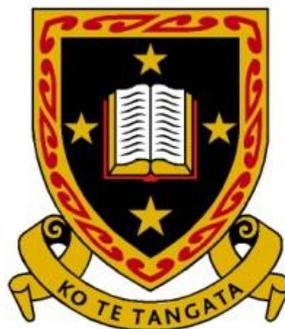
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**Constructed treatment wetlands:
Tools to attenuate diffuse agricultural pollution and
enhance the biodiversity of eutrophic peat lake ecosystems**

A thesis
submitted in fulfilment
of the requirements for the degree
of
Doctor of Philosophy in Biological Sciences
at
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by
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Abstract

Agricultural land use is a major source of nitrogen (N), phosphorus (P) and suspended sediment (SS) to aquatic ecosystems. Elevated inputs of N, P and SS associated with agricultural expansion and intensification have led to widespread degradation of surface waters throughout the world, causing eutrophication and excessive sedimentation of many rivers, lakes, wetlands, and estuaries. In excess, these contaminants cause diverse negative impacts on lake ecosystems such as prolific growth of phytoplankton and aquatic plants, toxic algal blooms and collapse of macrophyte beds, high turbidity and in-filling, as well as reduced biodiversity, health and resilience. Shallow peat lakes in the Waikato region of New Zealand have catchment soils that are highly erodible and readily leach nutrients, with low capacity to assimilate elevated loads of N, P and SS. These lakes have some of the poorest water quality amongst lakes in New Zealand.

Constructed treatment wetlands (CTWs) are used internationally as technologies to improve the quality of stormwater and wastewater from municipal, industrial and agricultural point sources. Further, CTWs are becoming increasingly prevalent as mitigation tools to manage diffuse pollution from agricultural runoff. However, discrepancies among studies of the performance of CTWs treating diffuse sources of N, P and SS have created uncertainty for end-users. As such, more field-based research is required to establish the suitability of CTWs as management tools for diffuse agricultural runoff where flow rates are unregulated and pollutant concentrations can be highly variable. Improving our knowledge of the efficacy of CTWs within intensive agricultural landscapes will increase confidence and encourage more widespread implementation of these mitigation tools by those involved with water quality management.

Actions are underway to restore several highly eutrophic Waikato peat lakes that have catchments used for intensive dairy production. Local land-care groups, landowners and lake managers are working collaboratively to improve water quality and enhance biodiversity. CTWs have been implemented as mitigation tools to manage diffuse sources of N, P and SS and reduce loads to the receiving

lakes. Restoration group members additionally anticipate that CTWs will improve biodiversity through provision of supplementary habitat within the highly modified, homogeneous agricultural landscape. The primary objective of my research was to evaluate the dual benefits of CTWs as tools for nutrient and sediment attenuation, and restoration of biodiversity within peat lake ecosystems. The efficacy of CTWs treating diffuse agricultural pollution is influenced by three key elements: CTW morphology, internal contaminant cycling, and the composition of influent constituents. I quantified the magnitude, composition and variability of constituent inputs in surface waters draining small subcatchments of five shallow peat lakes. Up to 26 channelised streams were sampled seasonally over 18 months from 2010 to 2011. Subcatchments had predominantly peat, peaty loam or clay loam soil types and ranged in area from c. 1 to 195 ha (mean 23 ha; median 6 ha). Extensive spatial and temporal variation in nutrient and sediment loads was evident, driven by seasonality and differences between subcatchment soil types and farm-scale management practices. Total N concentrations were highly variable (0.24 - 13.55 mg L⁻¹; median 2.13 mg L⁻¹), attributable to varying concentrations of nitrate-N and particulate organic-N, which ranged from 0.01 to 10.28 mg L⁻¹ and 0.01 to 4.63 mg L⁻¹, respectively. Total P concentrations ranged widely (0.01 - 3.02 mg L⁻¹; median 0.13), with exceptionally high concentrations of dissolved P in watercourses draining subcatchments with deep (≥7 m) peat soils and very low surface-water pH (< 4) (mean 1.29 mg PO₄-P L⁻¹; n=15). These results stress the importance of considering soil type when deriving appropriate environmental targets and controls for diffuse pollution from intensive agricultural peat lake catchments. Lake-catchment nutrient loads calculated from subcatchment daily loads were greater than many of those reported elsewhere in New Zealand, indicating the significance of the nutrient problem faced by water quality managers of Waikato's shallow peat lakes.

The efficacy of CTWs as management tools to attenuate diffuse sources of N, P, and SS from agricultural subcatchments of the five peat lakes was investigated concurrently. Different potential predictors of CTW performance were evaluated

for up to 26 CTWs and the effect of morphological and environmental variables on internal nutrient cycling and treatment performance were elucidated. All CTWs were comprised of a sedimentation pond 'module', with around half including shallow wetland modules planted with native plant species, and three with additional sedimentation pond modules. Inflows were surface-flow watercourses diverted from modified or artificial drainage networks, and outflows were either surface-flow (through drainage channels or culverts) or filtration (through vegetated riparian margins). Morphological predictors of CTW performance included area (range 7 – 1950 m²), depth (0.2 – 2.1 m), volume (12 – 2030 m³), wetland to catchment area ratio (0.01 – 1.18), hydraulic retention time (0.2 – 37.2 h), and hydraulic loading rate (0.4 – 130 m d⁻¹). The presence/absence of macrophytes, the outlet type and the number of CTW modules were included in analyses as categorical variables. Reductions in N, P and SS differed considerably among CTWs, driven by varying influent concentrations and dominant forms of N, P, and SS, as well as CTW morphologies and internal nutrient cycling. Generally, CTWs with larger areas and volumes had higher removal rates of nitrate, total N and coarse sediments, while deeper CTWs more effectively reduced particulate N and volatile SS. Macrophytes improved removal of nitrate and P, whereas filtration outlets frequently increased ammonium. Greater accumulated sediment depths were associated with reduced P removal efficiency, signifying the importance of CTW maintenance such as periodic removal of accumulated sediment. Increasing the number of CTW modules generally improved pollutant removal performance; thus, implementing individualised CTW treatment-train concepts is recommended.

Zooplankton communities were studied to investigate the value of CTWs as tools to improve the biodiversity of peat lake ecosystems through provision of supplementary habitat. Zooplankton are an essential component of healthy functioning lake and wetland ecosystems. Despite this, zooplankton communities within CTWs in agricultural landscapes remain unstudied. Zooplankton taxa richness, total abundance and community composition were compared among three habitat types (lakes, CTWs and drainage ditches) within the five study catchments. CTWs supported zooplankton species otherwise absent from lake

and drain habitats, increasing the biodiversity of the highly-modified peat lake catchments. Taxa richness of CTWs was higher than that of drains, and a few CTWs had greater diversity than some of the lakes. Zooplankton communities were significantly influenced by habitat area, depth and pH, as well as ammonium and phosphate concentrations, water temperature, dissolved oxygen, and macrophyte cover. This thesis explored opportunities for refining CTW designs, to enhance habitat diversity and support zooplankton species that improve ecosystem function and may increase CTW performance in agricultural landscapes.

The results of my research present key design considerations for surface flow CTWs treating diffuse agricultural pollution from peat lake catchments used for intensive dairy production. Collectively, the findings of this thesis provide a scientific basis for more comprehensive, holistic CTW designs to mitigate inputs of N, P and SS, and quantitative evidence of the value of agricultural CTWs as water quality and restoration management tools for peat lake ecosystems. This thesis also provides recommendations for future research to improve our understanding of the complex hydrological and biogeochemical processes driving nutrient losses from peat lake catchments used for intensive dairy production, to inform restoration and rehabilitation of shallow peat lake ecosystems in New Zealand.

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Table of contents

ABSTRACT	I
ACKNOWLEDGEMENTS	V
TABLE OF CONTENTS.....	VII
LIST OF FIGURES.....	XI
LIST OF TABLES.....	XV
PREFACE.....	XIX
CHAPTER 1	1
1 General introduction.....	1
1.1 Agriculture and water quality	1
1.2 Constructed treatment wetlands.....	3
1.3 The Waikato region.....	7
1.4 Aim and objectives.....	9
1.5 Thesis overview.....	10
1.6 References	12
CHAPTER 2	19
2 Spatial and temporal complexity of nutrient and sediment loads to peat lakes from intensive agricultural catchments	19
2.1 Abstract.....	19
2.2 Introduction	20
2.3 Methods.....	24
2.3.1 Study sites	24
2.3.2 Sampling	29
2.3.3 Sample analyses	30
2.3.4 Daily Areal Yields (DAY) and Daily Instantaneous Loads (DIL)	30
2.3.5 Statistical analyses.....	31
2.4 Results.....	33
2.4.1 Grouping factors.....	37
2.4.2 Relative importance of grouping factors.....	39
2.4.3 Mechanisms underlying grouping factors	41
2.4.4 Daily Areal Yields (DAY) and Daily Instantaneous Loads (DIL)	46
2.5 Discussion	51

2.5.1	Daily Areal Yields (DAY) and Daily Instantaneous Loads (DIL)	51
2.5.2	Variability.....	52
2.5.3	Seasonal variability	53
2.5.4	Soil variability	54
2.5.5	Managing diffuse pollution.....	58
2.6	Conclusions	59
2.7	References.....	61
CHAPTER 3.....		69
3	Constructed treatment wetland design considerations to mitigate diffuse pollution from intensive agricultural land use in New Zealand peat lake catchments	69
3.1	Abstract	69
3.2	Introduction	70
3.3	Methods.....	74
3.3.1	Study site	74
3.3.2	Sampling	79
3.3.3	Sample analyses.....	80
3.3.4	Statistical analyses.....	81
3.3.4.1	Examining attenuation patterns	82
3.3.4.2	Assessing morphological predictors of CTW performance	82
3.4	Results.....	84
3.4.1	Attenuation patterns.....	84
3.4.2	Morphological predictors of CTW performance.....	88
3.4.3	Physico-chemical variables	92
3.5	Discussion.....	95
3.5.1	Morphological predictors of CTW performance.....	95
3.5.2	Internal nutrient cycling	96
3.5.3	Macrophytes.....	97
3.5.4	Influent constituent composition	99
3.6	Conclusions and recommendations	102
3.7	Acknowledgements.....	105
3.8	References.....	105
3.9	Supplementary Material	112
CHAPTER 4.....		115
4	Constructed treatment wetlands provide habitat for zooplankton communities in agricultural peat lake catchments	115
4.1	Abstract	115

4.2	Introduction	116
4.3	Methods.....	119
4.3.1	Study sites	119
4.3.2	Field Sampling	122
4.3.3	Sample analyses	123
4.3.4	Statistical analyses.....	124
4.4	Results.....	126
4.4.1	Zooplankton community composition	126
4.4.2	Environmental variables.....	131
4.4.3	Habitat variables influencing zooplankton communities	134
4.5	Discussion	136
4.5.1	Zooplankton diversity.....	136
4.5.2	Zooplankton assemblages & functional attributes	137
4.5.3	Habitat characteristics.....	139
4.6	Conclusions	141
4.7	Acknowledgements.....	142
4.8	References	142
4.9	Supplementary Material	148
CHAPTER 5		149
5	Concluding Discussion.....	149
5.1	Research summary.....	149
5.2	Synthesis	149
5.2.1	Composition of influent constituents.....	149
5.2.2	Agricultural CTW treatment performance	153
5.2.3	Zooplankton biodiversity.....	155
5.3	Recommendations for management and future research	157
5.4	References	160
5.5	Supplementary Material	165
5.5.1	Relative water level data	165
5.5.2	Comparison of subcatchment DAY.....	168
6	Appendix	169
6.1	Guidance and considerations for design and installation of constructed wetlands in shallow lake catchments: Waikato Regional Council Internal Technical Series 2016/10.....	169

List of figures

Figure 1.1 Simplified schematic cross-sections of the principal types of CTW including (A) free water surface, (B) horizontal subsurface flow, and (B) vertical flow systems	5
Figure 1.2. Location of study area showing peat lakes in the Waikato region. Shading indicates urban area. Inset: North Island, New Zealand.	9
Figure 2.1 Location of study area showing peat lakes in the Waikato region. Shading indicates urban area. Inset: North Island, New Zealand.	25
Figure 2.2 Modern landscape features of the Hamilton Basin showing antecedent hills partly buried by volcanogenic alluvium (Hinuera Formation) and post-Hinuera lakes and peat bogs (reproduced with permission from D.J. Lowe, 2010). The location of the study lakes is identified by: 1 (the Horsham Downs lakes), 2 (Lake Koromatua), and 3 (Serpentine Lakes).	26
Figure 2.3 Estimates of components of variation for each grouping factor obtained by the multivariate PERMANOVA using all constituents, and physico-chemical and morphological variables from all sites across five seasons. The variability explained by each term is expressed as a percentage of the total variation.	37
Figure 2.4 Univariate DistLM results relating environmental variables to variation of individual nutrient species and suspended solid concentrations. NH ₄ -N: ammonium; NO ₃ -N: nitrate; ORG-N: organic nitrogen; PO ₄ -P: phosphate; TN: total nitrogen; TP: total phosphorus; VSS: volatile suspended solids; Non-VSS: non-volatile suspended solids.	42
Figure 2.5 PCA ordination plots based on environmental variables, nutrient and suspended sediment concentrations. Sites are plotted according to; A) soil type, B) season, and C) uniformly with vector overlays of environmental variables and all inflow constituents. NH ₄ -N: ammonium; NO ₃ -N: nitrate; ORG-N: organic nitrogen; PO ₄ -P: phosphate; TN: total nitrogen; TP: total phosphorus; VSS: volatile suspended solids; Non-VSS: non-volatile suspended solids; Q: discharge; Cond.: specific conductivity; DO: dissolved oxygen; Temp.: water temperature	46
Figure 3.1 Simplified schematic cross-sections of the principal types of CTW including (A) free water surface, (B) horizontal subsurface flow, and (B) vertical flow systems	71

- Figure 3.2 Modern landscape features of the Hamilton Basin showing antecedent hills partly buried by volcanogenic alluvium (Hinuera Formation) and post-Hinuera lakes and peat bogs (reproduced with permission from D.J. Lowe, 2010). The location of the study lakes is identified by: 1 (the Horsham Downs lakes), 2 (Lake Koromatua), and 3 (Serpentine Lakes). 75
- Figure 3.3 Location of studied peat lakes in the central Waikato (inset), North Island, New Zealand. Lake areas of open water and riparian margins were digitised from 2016 aerial photographs. Catchment boundaries were delineated from 2008 LiDAR (Light Detection and Ranging) data (source; Waikato Regional Council). 77
- Figure 3.4. Mean concentrations of nitrogen ($\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, Org-N) plotted by outlet type (A) surface flow and (B) filtration; and phosphorus (DRP, PP) plotted by outlet type (C) filtration and (D) surface flow; and (E) total suspended solids (VSS and non-VSS). Sampling locations include upstream (US) of the CTW, within the sedimentation pond module (Trap), within the wetland module (W/L), and downstream of the CTW for surface flow outlets (DS), and filtration outlets (DS – Piez). CTWs are ordered from highest to lowest influent concentrations..... 85
- Figure 3.5. Upstream and downstream concentrations of nitrogen ($\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, Org-N, TN), phosphorus (DRP, PP, TP) and suspended solids (VSS, non-VSS) for all samples collected over five seasons, plotted on 1:1 axes. The diagonal line indicates a 1:1 relationship, that is equal upstream and downstream values. Points falling below the line represent reductions in constituent concentrations whilst points above indicate increases. Samples for N and P are plotted according to CTW outlet type (surface flow and filtration), and samples for VSS and non-VSS are plotted according to the presence/absence of macrophytes. 87
- Figure 3.6 A schematic summary of the primary processes and patterns of attenuation and cycling of nitrogen ($\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, Org-N), phosphorus (DRP, PP) and suspended solids (VSS, non-VSS) occurring each module (sedimentation pond, shallow macrophyte, and open-water pond) throughout a CTW treatment-train. DNRA = dissimilatory nitrate reduction to ammonium..... 104
- Figure 4.1 Location of studied peat lakes in the central Waikato (inset), North Island, New Zealand. Lake areas of open water and riparian margins were digitised from 2016 aerial photographs. Catchment boundaries were delineated from 2008 LiDAR (Light Detection and Ranging) data (source; Waikato Regional Council)... 121

Figure 4.2 Mean values for taxa richness and total abundance of zooplankton (a, e) and total abundances of zooplankton species most dissimilar between habitat types including; Bdelloids (b), Brachionus spp. (c), Copepod nauplii (d), Cyclopoid copepodites (f), Keratella spp. (g) and Lecane spp. (h) plotted on a log₁₀-scaled axis for each habitat type (lake, CTW and drain)..... 127

Figure SM 5.1 Water levels of the Mangawara Stream (A) and Puniu River (B) immediately north and south of the study area, respectively... 167

List of tables

Table 2.1 Characteristics of study lakes including morphology, number of streams studied (n), trophic status, Trophic Level Index (TLI), water clarity, and average concentrations of chlorophyll a (Chl a), total phosphorus (TP) and total nitrogen (TN). Source: Hamilton et al. (2010)	27
Table 2.2 Stream subcatchment characteristics and morphologies. Effluent pond = storage facility for dairy farm effluent from milking shed and/or feed-pads for deferred irrigation to land. A = fenced both sides, planted with native vegetation; B = fenced both sides with \geq of 3 wires; C = fenced both side with < 3 wires; D = fenced one side; E = unfenced	28
Table 2.3 Mean values (\pm SD) for physico-chemical attributes of the 26 stream sites summarised by: a) lake-catchment, b) season, c) dominant soil type, and d) all samples across all sites and seasons.	34
Table 2.4 Mean concentrations (\pm SD) of nutrient species and suspended solids across all stream sites for the five seasons sampled summarised by: a) stream site; b) season; c) dominant soil type, and; d) all samples.....	36
Table 2.5 Results of pairwise comparisons of constituents for: a) the interaction between lake-catchment and soil types; b) seasons; c) soil types; d) lake-catchments.....	38
Table 2.6 PERMANOVA results for a) all constituents, and for individual nutrient species and suspended solids: b) $\text{NH}_4\text{-N}$, c) $\text{NO}_3\text{-N}$, d) ORG-N , e) $\text{PO}_4\text{-P}$, f) TN , g) TP , h) VSS , i) Non-VSS . Bold font in Table 2.6 denotes significant relationships.....	40
Table 2.7 Multivariate DistLM results relating environmental variables to variations in stream nutrient and suspended solids concentrations. Cum. R^2 : cumulative percentage of variation explained after entering this variable; SS: Sum of squares; F pseudo: test statistic; P-perm: p-value; Prop.: proportion of explained variance; df: degrees of freedom.	41
Table 2.8 Daily areal yields ($\text{kg ha}^{-1} \text{d}^{-1}$) for total nitrogen (TN) and phosphorus (TP) for the streams sampled, summarised by season, dominant soil type, and lake-catchment.	48
Table 2.9 Daily instantaneous loads (DIL) of total nitrogen (TN) and phosphorus (TP) totalled for each lake-catchment, summarised by season.....	49

Table 3.1 Characteristics of the study lakes including morphology, trophic status, Trophic Level Index (TLI), water clarity, and mean concentrations of chlorophyll a (Chl a), total phosphorus (TP) and total nitrogen (TN), number of major in-flows, number of CTWs and CTW codes. Source (excluding CTWs): (Hamilton et al. 2010).	76
Table 3.2. Description of Constructed Treatment Wetlands (CTWs) and summary of morphological variables. Trap = sedimentation pond-module, WL = wetland-module, WCAR = Wetland to Catchment Area Ratio, HRT = hydraulic retention time, HLR = hydraulic loading rate.	78
Table 3.3. Summary of relationships between CTW morphological predictors of performance and percentage removal (%Δ), upstream (US) and downstream (DS) concentrations of nitrogen (NH ₄ -N, NO ₃ -N, Org-N, TN), phosphorus (DRP, PP, TP) and suspended solids (VSS, non-VSS). Positive percentage values are indicative of a reduction in nutrient concentrations from upstream to downstream of the CTW, whilst negative values indicate an increase. The rho coefficient is given for Spearman's rank correlations (%Δ) and r for Pearson's correlations (US and DS). Significant correlations are in bold, and statistical significance is indicated by *, ** and *** for P-values <0.05, <0.01, and <0.001, respectively.	89
Table 3.4. The effects of the number of CTW modules (one, two or three), outlet type (surface or filtration), and the presence/absence of macrophytes on percentage removal (%Δ) of nitrogen (NH ₄ -N, NO ₃ -N, Org-N, TN), phosphorus (DRP, PP, TP) and suspended solids (VSS, non-VSS) are given. Positive percentage values are indicative of a reduction in nutrient concentration from upstream to downstream of the CTW, whilst negative values indicate an increase. Chi-squared values (χ ²) from Kruskal-Wallis tests for the effects of the number of modules, outlet type, and presence/absence of macrophytes on removal percentages are given with medians and standard deviations (SD). Significant correlations are in bold, and statistical significance is indicated by *, ** and *** for P-values <0.05, <0.01, and <0.001, respectively.	90
Table 3.5. Summary of relationships between physico-chemical variables (water temperature, specific conductivity, DO, pH, influent flow rate Q, and sediment depth) and percentage removal (%Δ), upstream (US) and downstream (DS) concentrations of nitrogen (NH ₄ -N, NO ₃ -N, Org-N, TN), phosphorus (DRP, PP, TP) and suspended solids (VSS, non-VSS). The rho coefficient is given for Spearman's rank correlations (%Δ) and r for Pearson's	

correlations (US and DS). Significant correlations are in bold, and statistical significance is indicated by *, ** and *** for P-values <0.05, <0.01, and <0.001, respectively. 93

Table 4.1 Characteristics of the study lakes including morphology, trophic status, Trophic Level Index (TLI), water clarity, and average concentrations of chlorophyll a (Chl a), total phosphorus (TP) and total nitrogen (TN), number of major in-flows, number of CTWs and CTW codes. Source (excluding CTWs): Hamilton et al. (2010) 120

Table 4.2 SIMPER analysis (square-root transformed data) showing the main taxa contributing to the variation in community composition between lake-CTW and drain-lake habitats, and at least 80% between drain-CTW habitats. Contribution % is the proportion of dissimilarity between habitat pairs contributed by each taxa 129

Table 4.3 Relative abundance (%) of zooplankton groups Cladocerans, Copepods (including Calanoids, Cyclopoids and nauplii), Ostracods, Rotifers and Others for lake, CTW and drain habitats. These measurements represent the mean abundance in which the relative abundance of each zooplankton taxon was estimated for each site (n = 40) then averaged across habitat type (lake n = 5; CTW n = 27, drain n = 8). Letters in superscript denote significant differences (P < 0.05) between habitat types.... 130

Table 4.4 Mean (\pm standard deviation) of environmental variables including physicochemical, biotic and water quality for each habitat type. Letters in superscript denote significant differences (Kruskal Wallis test, P < 0.01) between habitat types. Connectivity with downstream lake being High, Medium, or Low (3, 2, 1, respectively). * lakes were considered to have high connectivity as there is unimpeded flow for exchange of water and aquatic organisms throughout the habitat..... 131

Table 4.5 SIMPER analysis showing the environmental variables contributing up to 70 % of the variation among lake, CTW and drain habitats. Mean values for each variable have been back-transformed to original values to clarify dissimilarities. Contribution % is the percentage of dissimilarity between habitat groups contributed by each variable..... 133

Table 4.6 Results from DistLM analysis of environmental variables driving variation in zooplankton community composition among lake, CTWs and drain habitats. Cum (%) = Cumulative percentage of variance; Prop (%) = Proportion of explained variance; ** P < 0.01; * P < 0.05; n.s = not significant 135

Table SM 3.1 Results from linear regressions between morphological predictors of CTW performance and percentage removal of nitrogen (NH ₄ -N, NO ₃ -N, Org-N, TN), phosphorus (DRP, PP, TP) and suspended solids (VSS, non-VSS) analysed separately for CTWs with macrophytes present and absent. R ² value, F-statistic, and direction of association (positive +, negative -) are given for significant relationships only. Degree of significance is indicated by *, ** and *** for P-values <0.05, <0.01, and <0.001, respectively.....	112
Table SM 3.2 Results from Spearman’s rank correlations between percentage removal (%Δ) and analogous upstream (US) concentrations of nitrogen (NH ₄ -N, NO ₃ -N, Org-N, TN), phosphorus (DRP, PP, TP) and suspended solids (VSS, non-VSS). The rho coefficient and P values are given; significant correlations are in bold, and statistical significance is indicated by *, ** and *** for P-values <0.05, <0.01, and <0.001, respectively.....	113
Table SM 4.1 Zooplankton taxa recorded from lake (L), constructed treatment wetland (CTW) and drain (D) habitats within peat lake catchments. Samples were collected from the Central Waikato region, New Zealand, between 2-8th February 2011	148
Table SM 5.1 Comparison of mean daily instantaneous yields (DAY, Kg ha ⁻¹ day ⁻¹) of ammonium (NH ₄ -N), nitrate (NO ₃ -N), particulate organic nitrogen (Org-N) and phosphate (PO ₄ -P) for subcatchments with clay loam, peaty loam, and deep peat dominated soil types within peat lake catchments of Kaituna, Kainui, Komakorau, and Koromatua. The landowners of each subcatchment are referred to as ‘A’, ‘B’, ‘C’ and ‘D’ to maintain their anonymity. The potential subcatchment nutrient sources stated were identified during the period of sampling (June 2010 – November 2011) therefore are not representative of any management or infrastructure changes which may have occurred subsequently on farms.....	168

Preface

This thesis is comprised of five chapters; Chapters 2-4 were prepared as individual manuscripts for submission to peer-reviewed scientific journals. There is therefore some repetition of methodological details, tables and figures.

Collectively, these chapters form a coherent body of work that makes an original contribution to my chosen thesis topic. Except where stated otherwise, the work in this thesis, including study design, field and laboratory work, data analyses and writing, was undertaken solely by myself while under the supervision of Professor David Hamilton (University of Waikato and Australian Rivers Institute), Dr Ian Duggan (University of Waikato), and Dr John Quinn (National Institute of Water & Atmospheric Research).

Co-authors for each chapter are listed below. All co-authors provided technical advice where necessary and reviewed the relevant chapters. Dr Ian Duggan completed the zooplankton identification for samples collected and analysed in Chapter 4.

Chapter 2 has been submitted to the New Zealand Journal of Agricultural Research as “Spatial and temporal complexity of nutrient and sediment loads to peat lakes from intensive agricultural catchments”. Authors: Rebecca S. Eivers, David P. Hamilton, and John M. Quinn.

Chapter 3 has been submitted to the New Zealand Journal of Marine and Freshwater Research as “Constructed treatment wetland design considerations to mitigate diffuse pollution from intensive agricultural peat lake catchments” and is currently *under review*. Authors: Rebecca S. Eivers, David P. Hamilton, John M. Quinn, and Ian C. Duggan.

Chapter 4 has been published as “Constructed treatment wetlands provide habitat for zooplankton communities in agricultural peat lake catchments” in the international journal *Wetlands*. Authors: Rebecca S. Eivers, Ian C. Duggan, David P. Hamilton, and John M. Quinn.

Chapter 1

1 GENERAL INTRODUCTION

1.1 AGRICULTURE AND WATER QUALITY

Diffuse pollution from agricultural catchments has led to widespread degradation of lakes, wetlands, rivers and estuaries in New Zealand (Howard-Williams et al. 2010; Wilcock 1986; Dymond et al. 2013; Hooda et al. 2000) and worldwide (Novotny 1999; Gulati & van Donk 2002; Smith 2003). More recently, the expansion and intensification of dairy farming to meet growing global demand for milk solids has accelerated water quality decline and eutrophication of inland waters due to increased nutrient and sediment loads (UNEP 2013; Wilcock et al. 2013; PCE 2013; MfE/StatsNZ 2015a). Agricultural land use conversions from sheep and beef to dairy farms in New Zealand equated to almost 158,000 ha in the four years to 2012 (PCE 2015), and more than 6.6 million dairy cattle comprised the national herd in 2016, a 175% increase over two decades (StatsNZ 2017). Concurrently, New Zealand's dairy exports have increased 4-fold and presently account for approximately one-third of total export earnings (Wheeler 2014).

Proliferations of cyanobacterial algal blooms in iconic lakes and toxic benthic-algal mats in many rivers (Wood et al. 2017; Wood et al. 2006), as well as microbial pathogen contamination of bathing waters (MfE 2018; Larned et al. 2016) have heightened public concerns regarding the declining water quality of many New Zealand streams, rivers, lakes, wetlands, estuaries and aquifers (PCE 2015; Hughey et al. 2013). Mounting public pressure to protect waterbodies from further degradation prompted the dairy industry to release The Dairying and Clean Streams Accord (van der Hayden et al. 2003), superseded by the Sustainable Dairying: Water Accord (DairyNZ 2013), to solidify the commitment of the New Zealand dairy sector to improving environmental performance and reducing dairy farming impacts on waterbodies.

Concurrently, the Land and Water Forum was established to develop a common direction for freshwater management in New Zealand and to provide advice to the Government for reform of freshwater management. Their first report, *A Fresh Start for Freshwater* (LAWF 2010), identified shared outcomes and goals for freshwater management nationally and made a comprehensive list of recommendations. Of relevance to the dairy industry were: setting limits for water quantity and quality; mechanisms for achieving targets, such as good management practices, robust industry standards, and effective policy frameworks; a review of the drainage legislation to better align with protection of wetlands and biodiversity; and establishment of a National Policy Statement for Freshwater Management (NPS-FM) to convey the impetus of the Land and Water Forum's advice. The NPS-FM was introduced in 2011 and has been amended twice (MfE 2014, 2017), following submissions from the Land and Water Forum and interested stakeholders, including iwi and hapū (representing New Zealand's indigenous people), the primary sector, local government, environmental and recreational NGOs, scientists, and communities.

Both the NPS-FM and Sustainable Dairying: Water Accord aim to improve or maintain the water quality of receiving environments within intensive agricultural catchments, ultimately through encouraging widespread implementation of environmental best practice on farms and enforcing limits on contaminant discharges to meet water quality objectives for nitrogen (N), phosphorus (P), suspended sediments (SS), and faecal microbial pathogens (managed by *E. coli* objectives). Numerous and varied mitigation measures have been developed in attempts to alleviate the negative impacts of intensive agriculture on aquatic environments (McKergow et al. 2007). These include methods for improved application of effluent, fertilisers, irrigation, and nitrification inhibitors, better management of soil health, stocking rates, watercourses and riparian areas, and more sustainable grazing regimes (Monaghan et al. 2009; Collins et al. 2007; Aarons & Gourley 2012; Di et al. 2009; Cooper et al. 1995; Hansen et al. 2012; McDowell & Catto 2005; Houlbrooke et al. 2004). However, as the dairy industry continues to further intensify farms to increase production (milk solids per hectare), there is a risk that current

strategies to mitigate pollutant losses may be inadequate for reducing contaminant discharges to meet limits from more intensive land use, particularly for N (PCE 2015).

A collaborative research project trialling best management practices aligned with the Dairying and Clean Streams Accord was initiated in 2001 in five regionally representative dairy catchments across New Zealand (Wilcock et al. 2007). Broadly, the aim of the study was to integrate environmentally sustainable practices into dairy farming alongside the industry's policy to increase productivity. While management practices were successful in reducing SS in all catchments and *E. coli* in three of the streams, trends in N and P losses showed no consistent pattern, and in fact N exports increased in two catchments following further intensification over a >5 year period (Wilcock et al. 2013; Monaghan & De Klein 2014; PCE 2015). The authors report that although changes in farm management lead to some improvements in water quality, concentrations of total N (TN), total P (TP) and *E. coli* still exceeded guidelines for ecosystem protection and contact recreation (MfE/MoH 2003; ANZECC 2000), and implementing a greater range of effective BMPs is required to achieve desired reductions in nutrient and faecal contaminants. The findings from this pivotal study suggest further research is needed to refine and improve more targeted, robust and effective mitigation measures to minimise diffuse pollution impacts on waterways from catchments used for intensive dairy production in New Zealand.

1.2 CONSTRUCTED TREATMENT WETLANDS

Constructed treatment wetlands (CTWs) are used internationally as technologies to improve water quality (Kadlec & Knight 1996) and are effective in reducing levels of SS, N and P, as well as organic matter and pathogens (Vymazal & Kröpfelová 2008; Dunne et al. 2012; Kadlec 2010; Braskerud 2003, 2002a). CTWs are designed and created to emulate and enhance the natural processes and functions of wetland ecosystems involving wetland vegetation, soils, and microbial and aquatic communities (Mitsch & Gosselink 2007; Kadlec & Wallace 2008g). The primary processes responsible for nutrient and sediment

attenuation by CTWs include plant uptake, sedimentation, accretion of new soils, sorption, volatilisation, and microbially mediated chemical processes such as denitrification (Kadlec & Wallace 2008d). The principal types of CTWs are defined by their hydrological designs and include free water surface (FWS), horizontal subsurface flow (HSSF), and vertical flow (VF) systems (Figure 1.1). Each is designed to enhance specific wetland characteristics to maximise desired processes and improve treatment performance of specific pollutants (Kadlec & Wallace 2008g).

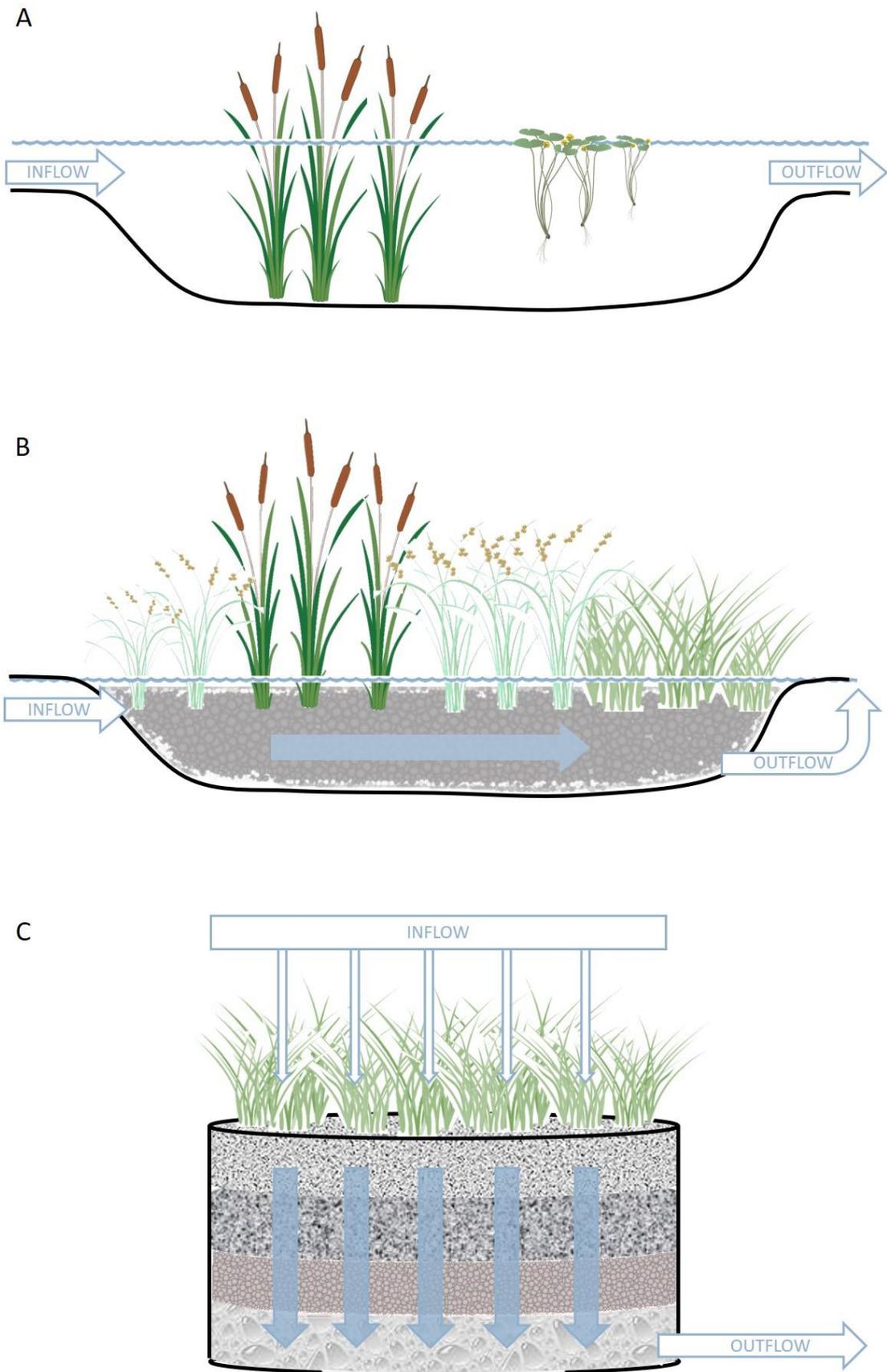


Figure 1.1 Simplified schematic cross-sections of the principal types of CTW including (A) free water surface, (B) horizontal subsurface flow, and (B) vertical flow systems

Both HSSF and VF CTWs are commonly used for primary and secondary treatment of municipal and, more recently, agricultural wastewater (Knight et al. 2000; Morari & Giardini 2009; Nivala et al. 2013). Created in series, HSSF and VF CTWs exploit nitrification-denitrification processes, often generating highly efficient removal of TN (Vymazal 2011a). The FWS CTWs, which more closely resemble natural wetlands with areas of open water, floating macrophytes, and emergent plants, typically follow HSSF and VF systems as a 'polishing' stage in the treatment train before effluent is discharged to surface waters (Kadlec & Wallace 2008g).

Constructed treatment wetlands have become more prevalent as a mitigation tool to manage diffuse pollution from agricultural runoff (Wang et al. 2018; Vymazal 2017; Vymazal & Brezinova 2015; Ballantine & Tanner 2010). A number of studies have shown significant reductions in nitrate-N, P, sediment, and *E. coli* from various agricultural wetlands including FWS (Kadlec 2006; Wilcock et al. 2012; Braskerud 2002b), HSSF (Kadlec et al. 2005; Tanner et al. 2005), riparian (Burns & Nguyen 2002; Rutherford et al. 2009; Hoffmann et al. 2011), and restored wetlands (Fennessy et al. 1994; Hoffmann & Baattrup-Pedersen 2007; Hoffmann et al. 2012). However, others have reported less encouraging outcomes, such as highly variable treatment performances, attributable to seasonal and hydrological drivers (Dunne et al. 2005; Blankenberg et al. 2008; Arheimer & Pers 2017), as well as internal cycling and release of N and P from stored sediments (Hoffmann et al. 2012; Novak et al. 2004; Tanner et al. 2005; Braskerud 2002a; Tanner & Sukias 2011). The discrepancies among studies of performance of CTWs treating diffuse sources of N, P and SS from intensive agricultural catchments creates uncertainty for end-users and may hinder implementation of these management tools. To ensure CTWs can be confidently included in the suite of current agricultural BMPs in New Zealand, greater clarity is required around appropriate designs, placements and maintenance regimes to maximise treatment performance and improve reliability.

1.3 THE WAIKATO REGION

Dairy farming has been widespread in the Waikato for more than fifty years and the region currently supports 38% of New Zealand's dairy herd (StatsNZ 2017). Once a vast mosaic of wetlands, including swamp, marsh, fen and peat bog ecosystems, as well as numerous peat lakes (Shearer 1997; Lowe & Green 1987), the landscape of the central and lower Waikato today is typified by highly productive dairy farms benefiting from the rich, organic soils of these historic environs (MfE/StatsNZ 2015b). The abundance, size and ecological integrity of peat bogs and lakes in the Waikato have declined drastically since the late 1800s ensuing extensive drainage, cultivation and conversion to pasture, threatening the indigenous and endemic flora and fauna of these ecosystems (Hunt 2007; Beard 2010; Johnson & Gerbeaux 2004). Many of the region's remaining 31 peat lakes are eutrophic to hypertrophic and experience frequent cyanobacterial algal blooms as a result of high nutrient and sediment loads associated with this change in land use (Hamilton et al. 2010). Waikato peat lakes are typically shallow (< 6 m mean depth) and historically had low levels of productivity owing to the ombrotrophic bogs from which they were formed (Clymo 1983; Shearer 1997). As such, the lakes are particularly susceptible to eutrophication, having less capacity to absorb increased inputs of N, P, and sediment (Thompson & Greenwood 1997; Scheffer 1998). As intensive dairy farming continues to expand throughout the Waikato, with 28,400 ha of land use conversions to dairy from 2008-2014 (PCE 2015), there is a growing need for practical, cost effective and proven mitigation measures to remediate water quality impacts on these sensitive ecosystems.

Restoration measures are currently being implemented in several Waikato peat lake catchments in collaborations between landowners, local government agencies and the Department of Conservation (the government agency charged with the conservation of New Zealand's natural and historical heritage), to improve lake water quality and restore biodiversity values (Peters et al. 2008; Waikato Regional Council 2006; Thompson & Greenwood 1997). Methods include retiring areas of marginal pasture to create esplanade reserves, fencing

and planting of riparian and wetland areas around the lake margins and along the banks of inlet waterways, and creation of FWS CTWs to intercept inflows and improve water quality. To date, however, there has been little or no measurable improvement in lake water quality (Edwards et al. 2010), suggesting a lack of understanding about the sources, quantities and characteristics of N, P and sediment inputs to the lakes, and the scale of action required to address these loads.

Extensive new drainage networks were excavated during the 1990s in many peat lake catchments, associated with conversions of dry stock farms to support high intensity dairy farming, causing substantial increases in sediment loads to the downstream lakes. Lakes Kaituna and Komakorau, 5 km north of Hamilton city (Figure 1.2), were severely impacted by in-filling, causing average depths to decline from 4 to 1 m over c. 10 years (Hamilton et al. 2010). Consequently, FWS CTWs were initially comprised of a simple in-stream silt-trap or sedimentation pond, designed to capture coarse particulates and prevent further in-filling of the lakes. Subsequent designs included shallow areas planted with native macrophytes to encourage nutrient uptake, and open-water areas designed to provide feeding habitat for native wetland birds (A. Hayes, Lakes Kaituna and Komakarau Lake Care Group chairperson, pers. comm.). The FWS CTWs were predominantly created in situ based on the observations and experience of the landowner, excavator operator, and local council staff. The location and area available for CTWs were frequently restricted to the boundaries of the Department of Conservation management reserve around the lake, except where landowners were willing to forego areas of 'unproductive' land adjacent to the reserve. No previous monitoring of the target watercourses was conducted, nor modelling to estimate catchment or subcatchment loads of N, P or sediment, to determine the size or design of the CTWs. Accordingly, there is considerable uncertainty surrounding the efficacy of these treatment systems, generating demand for empirical evidence and critical assessment of the applicability and benefits of CTWs in support of peat lake restoration.

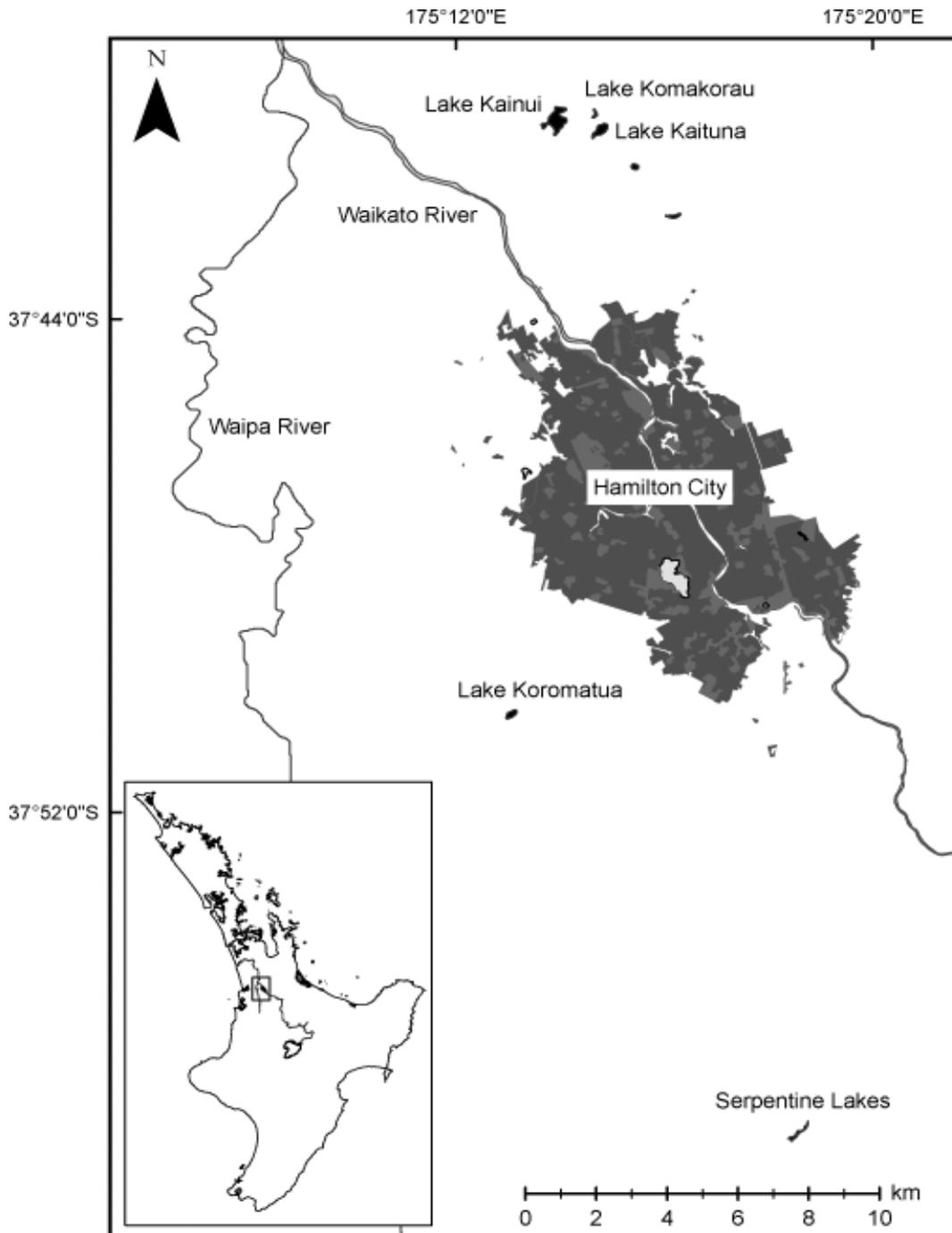


Figure 1.2. Location of study area showing peat lakes in the Waikato region. Shading indicates urban area. Inset: North Island, New Zealand.

1.4 AIM AND OBJECTIVES

Quantitative assessments of freshwater environments frequently focus on the flow and volume of water, concentrations of its constituents including nutrients and sediments, and the diversity and abundance of aquatic organisms (Woods and Howard-Williams, 2004). The overall aim of this thesis was to draw together these fundamental elements of freshwater systems, and to better understand

the value of CTWs in intensive agricultural catchments for remediation and restoration of downstream peat lakes.

The first objective was to identify the variability and magnitude of nutrient and sediment inputs to shallow peat lakes through (i) seasonal synoptic surveys of physico-chemical and water quality related variables from the major inflows to the lakes, and (ii) estimation of nutrient and sediment loads based upon subcatchment-scale water balances and measured concentrations of N, P and SS.

The second objective was to investigate the efficacy of CTWs as tools to support peat lake restoration through (i) evaluating the pollutant removal performance of existing CTWs across selected peat lake catchments, and (ii) comparing the biodiversity of CTWs with lake and stream habitats within peat lake catchments.

1.5 THESIS OVERVIEW

This thesis comprises three main chapters (Chapters 2-4) which have been prepared for, or published in, peer-reviewed scientific journals to address the previously stated objectives.

Chapter 2 aims to quantify concentrations of nutrients, including soluble and particulate forms of N and P, and suspended sediments, in surface waters derived from small peat lake subcatchments with intensive agricultural land use. This chapter investigates the key factors driving variations in runoff composition (expected to vary seasonally, with soil type, and across farm management units), to improve knowledge of how to address eutrophication in the lakes at the catchment scale.

Chapter 3 aims to determine the efficacy of CTWs implemented as mitigation tools to improve the water quality of inflows to shallow peat lakes. Different predictors of CTW performance are evaluated, as well as morphological and environmental variables that may enhance or impede treatment efficiency, to inform improvements in design of CTWs treating diffuse pollution from intensive agricultural catchments. The findings are intended to support stakeholders interested in, and concerned with water quality management, including local

landowners and community care groups, as well as government agencies and industry advisors.

Zooplankton communities are essential components of healthy functioning lake and wetland ecosystems and can be representative of aquatic biodiversity and ecosystem health (Lougheed & Chow-Fraser 2002; Boix et al. 2005; Duggan et al. 2001). Chapter 4 compares zooplankton community composition between the lake, streams and CTW habitats of peat lake catchments to assess the biodiversity value of CTWs within intensive agricultural landscapes.

Opportunities are explored for refining CTW designs to enhance zooplankton biodiversity and potentially improve treatment efficiency.

Chapter 5 synthesises the principal conclusions of the preceding chapters, consolidating recommendations for management and restoration of eutrophic shallow peat lakes using CTWs. Research areas are presented to improve knowledge of nutrient losses from peat-dominated catchments in New Zealand, internal nutrient cycling in CTWs processing diffuse pollution from intensive agricultural land use, and niche habitat preferences for zooplankton beneficial to CTW pollutant removal performance.

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Chapter 2

2 SPATIAL AND TEMPORAL COMPLEXITY OF NUTRIENT AND SEDIMENT LOADS TO PEAT LAKES FROM INTENSIVE AGRICULTURAL CATCHMENTS

2.1 ABSTRACT

The expansion and intensification of agriculture has led to widespread degradation of water quality due to increased sediment and nutrient loads. Shallow peat lakes are particularly susceptible to these effects, particularly where catchment soils have high rates of nutrient loss. This study sought to quantify concentrations of nutrients, including soluble and particulate forms of nitrogen and phosphorus, and suspended solids, in surface waters derived from small peat lake subcatchments with intensive agricultural land use. The primary objective was to identify the key factors driving variations in runoff composition, to better understand how to address eutrophication in these lakes. Up to twenty-six waterways constituting the dominant inflows to five shallow peat lakes were sampled over five seasons. Subcatchments ranged in area from 1.3 to 195 ha, with soil types consisting of predominantly peat, peaty loam and clay loam soils. Nutrient and sediment concentrations varied significantly with catchment soil type and season. Total nitrogen concentrations were highly variable, ranging from 0.24 to 13.6 mg L⁻¹ (median 2.13), attributable to variations in constituent species of nitrogen. Total phosphorus concentrations ranged from 0.01 to 3.0 mg L⁻¹ (median 0.13). Nutrient species were significantly correlated with surface water pH (ammonium-nitrogen $r = -0.47$; nitrate-nitrogen $r = 0.45$; organic-nitrogen $r = -0.42$; and dissolved reactive phosphorus $r = -0.50$), suggesting that soil type in peat environments strongly influences nutrient cycling and runoff. These results indicate that water managers must consider soil types when deriving appropriate environmental targets and controls for diffuse pollution from intensive agricultural catchments.

2.2 INTRODUCTION

Diffuse pollution from intensive agricultural catchments is a significant source of degradation of waterways (Carpenter et al. 1998). Recent expansion and intensification of dairy production has led to elevated loads of nitrogen (N), phosphorus (P) and suspended solids (SS) which increase primary production and cause eutrophication in receiving waters (Wilcock & Nagels 2001; Burns 1991; Bouwman et al. 2013). Excessive P loads have been closely linked to lake eutrophication and toxic cyanobacteria blooms (Schindler 2006), which can be harmful to public health (Mur et al. 1999). Suspended solids adversely impact lake ecosystems both directly, through increased turbidity, infilling and smothering of littoral habitats (Scheffer 1998), and indirectly through increased P availability associated with settled and re-suspended sediments (Correll 1998; Tammeorg et al. 2016).

Reliable estimates of catchment nutrient and sediment loads are important for effective management and restoration of lakes and waterways (Fox & Argent 2009; McColl 1978). Empirical and modelled load estimates are frequently used to evaluate and predict land use pressures on water bodies, however these estimates have inherent uncertainties and variability due to the complex processes driving nutrient and sediment fate and transport, causing management quandaries for custodians of water resources (Elliott et al. 2005). This variability is related partly to catchment characteristics, including land use and land management practices, topography and soil type, as well as to climatic conditions and seasonality (Elliott & Sorrell 2002; Søndergaard et al. 2012). Furthermore, small headwater catchments, particularly those of ephemeral watercourses, and catchments with artificial drainage networks, have highly variable rates of discharge and nutrient and sediment concentrations (Richards & Holloway 1987; Tiemeyer & Kahle 2014). To improve confidence in, and enable the formulation of robust, effective and achievable load-based targets for regulation and management, these processes require quantification at local scales.

Agricultural land use intensity varies with specific local (landowner) and regional management methods used to optimise production, driving spatial variations in nutrient and sediment losses. In New Zealand, where dairy production has recently intensified, N leaching has increased rapidly with higher application rates of inorganic nitrogenous fertiliser to improve pasture yields, and increased quantities of imported supplementary feeds (Clark et al. 2007; Wright 2013). This has enabled higher stocking rates and more intensive grazing regimes, leading to, for example, increased nitrate leaching from urine patches and losses of P caused by soil “pugging” (Haynes & Williams 1993; McDowell et al. 2003). Expanded effluent applications, irrigation, and artificial drainage have also contributed to higher diffuse exports of N and P (Houlbrooke et al. 2004; de Klein et al. 2006; McDowell et al. 2011). The mobilisation of mineral and organic SS is similarly influenced by the aforementioned practices and is strongly affected by the management of riparian areas (McDowell 2006; Wilcock et al. 2009). Groundwater nitrate concentrations, and inputs to streams, are expected to increase substantially through the cumulative effects of leaching which reflects intensification of agriculture in New Zealand from c. 1955 (Morgenstern & Daughney 2012). These lag times and the spatial variability of groundwater systems impart additional variability to N cycling at local and regional scales. Soil porosity, permeability and hydraulic conductivity, as well as organic matter content and microbial communities, strongly influence N cycling and losses to waterways (Patrick & Tusneem 1972; Vagstad et al. 1997). Surface and subsurface runoff from organic soils generally has lower nitrate concentrations than that from mineral soils due to higher denitrification rates supported by large stores of soil carbon (Di & Cameron 2002). However, mineralised organic soils cultivated from drained peatlands have been shown to release high levels of nitrate under certain hydrological conditions, suggesting drained peatlands may be an important source of diffuse N pollution (Tiemeyer & Kahle 2014). Soil type and structure also strongly influence P sorption and buffering capacity, and susceptibility to erosion, and are highly important in defining the P retention capacity of catchments (McLaren & Cameron 1996; de Vente & Poesen 2005). Quantifying P losses at catchment scale therefore requires careful consideration

of both natural and anthropogenic sources as well as the transport processes mediating the movement of P to waterways. Phosphorus is predominantly mobilised during rainfall events, although dissolved reactive P (DRP) can be leached from some soils via subsurface pathways as a result of low anion storage capacity and artificial drainage (McDowell et al. 2004). Hart & Cornish (2012) found that soil buffering capacity is one of the key drivers of P loss from pastoral land in New South Wales, Australia and emphasised that the form in which P is mobilised should govern the management method used to reduce P losses from agricultural catchments.

Nutrient and SS losses from catchments dominated by pastoral agriculture vary seasonally due to changes in rainfall, temperature and day length. Winter and spring in New Zealand are characterised by higher rates of nutrient leaching and increased generation of particulates in surface runoff (Quinn & Stroud 2002; Smith & Monaghan 2003). Concentrations and forms of N species ($\text{NO}_3\text{-N}$, $\text{NH}_4\text{-N}$ and organic nitrogen; ORG-N) are mediated by temperature and day length, which influence evapotranspiration rates and soil moisture content, microbiological activity, mineralisation, nitrification and denitrification rates, as well as in-stream processes (Burns et al. 2009; Vagstad et al. 1997). Summer and autumn are characterised by increased uptake of nutrients by plants, generally reducing losses to waterways (Davies-Colley & Nagels 2002), except during heavy rainstorm events (Abell et al. 2013; Granger et al. 2010). Regionally, applications of urea on silt-loam soils in Canterbury, New Zealand, show increased N leaching associated with soil saturation as well as lower temperatures (Black et al. 1985).

Many lowland peat lakes in the Waikato region of New Zealand have catchments dominated by dairy production and have poor water quality, with frequent cyanobacterial blooms (Hamilton et al. 2010; Wood et al. 2014). Prior to drainage and cultivation for agriculture, the lakes had low levels of productivity and supported a diverse native and endemic flora and fauna (Johnson & Gerbeaux 2004; McLay et al. 1992). Nowadays most of these lakes are eutrophic to hypertrophic as a result of high nutrient and sediment loads, causing loss of much of their natural character (Beard 2010; Shearer 1997). Restoration actions, including fencing and planting of riparian areas around lake margins and along

the banks of inlet waterways, as well as construction of sedimentation ponds and wetlands, are being implemented to reduce nutrient and sediment runoff to selected lakes. However, there has been little or no measurable improvement in lake water quality (Edwards et al. 2010), suggesting a lack of understanding about the sources, quantities and characteristics of N, P and SS inputs to the lakes, and the scale of action required to address these loads.

The aim of this study was to quantify the concentrations of N, P and SS over five seasons in waterways constituting the dominant inflows to five shallow peat lakes in the Waikato region of New Zealand. The inflows are small (mean catchment area 23 ha; median 6 ha), with modified or artificially drained subcatchments predominantly used for intensive dairy production. It was hypothesised that seasonality, soil properties, individual lake catchments (as a proxy for local variation in land use intensity), and stream subcatchment size would be important drivers of variability in nutrient and SS concentrations. Our key objectives were to (i) evaluate the relative importance of season, soil type, stream subcatchment, and lake catchment in influencing nutrient and SS concentrations in surface waters, (ii) examine underlying environmental mechanisms driving relationships with water constituents, and (iii) calculate instantaneous areal nutrient yields and loads for the subcatchments of each stream to examine relative loads at the lake-catchment scale.

2.3 METHODS

2.3.1 Study sites

The research was carried out in the Waikato region of New Zealand (37.8°S, 175.2°E) where there are a number of shallow lakes with elevated nutrient levels and trophic status (Hamilton et al. 2010) which are located in catchments dominated by dairy production. Stocking rates ranged from 3.13 to 3.52 cows ha⁻¹ at the time of study (cf. an average of 2.95 and 2.87 cows ha⁻¹ for the Waikato Region and New Zealand, respectively) (LIC 2015). Five shallow peat lakes with active restoration programmes were selected as study sites: Kainui, Kaituna, Komakorau, Koromatua and Serpentine North (Figure 2.1). Lakes Kainui, Kaituna and Komakorau are within the Kainui restiad peat bog located in the Waikato District (Horsham Downs), north of Hamilton city (population c. 215,000; StatsNZ 2018). Lake Koromatua is on the edge of the Rukuhia restiad peat bog whilst Serpentine North is on the fringe of the Moanatuatua restiad peat bog, both in the Waipa District south of Hamilton city (Clarkson et al. 2004) (Figure 2.2). The topography of the lake catchments is flat to rolling lowland hills <70 m above sea level (mean ~ 34 m asl).

The central Waikato has a temperate maritime climate and rainfall is generally plentiful year-round (Chappell 2014). Annual rainfall across the lake catchments (approximately 35 km from north to south) ranges from 1100 to 1300 mm (Dravitzki & McGregor 2011). The region is characterised by relatively warm temperatures in the summer (December – March) with mean daily maximums between 20 and 25 °C, and relatively cold temperatures during winter (July – August) with mean daily maximums ranging from 0 – 8°C (Chappell 2014).

A summary of the lake morphologies, number of streams draining to each lake, stocking rate of the dairy farm in the lake catchment, trophic status, Trophic Level Index (TLI; Burns et al. 2000), water clarity, and average concentrations of chlorophyll *a* (Chl *a*), TP and TN is given in Table 2.1. Twenty-six streams constituting the major inflows to the five lakes were sampled.

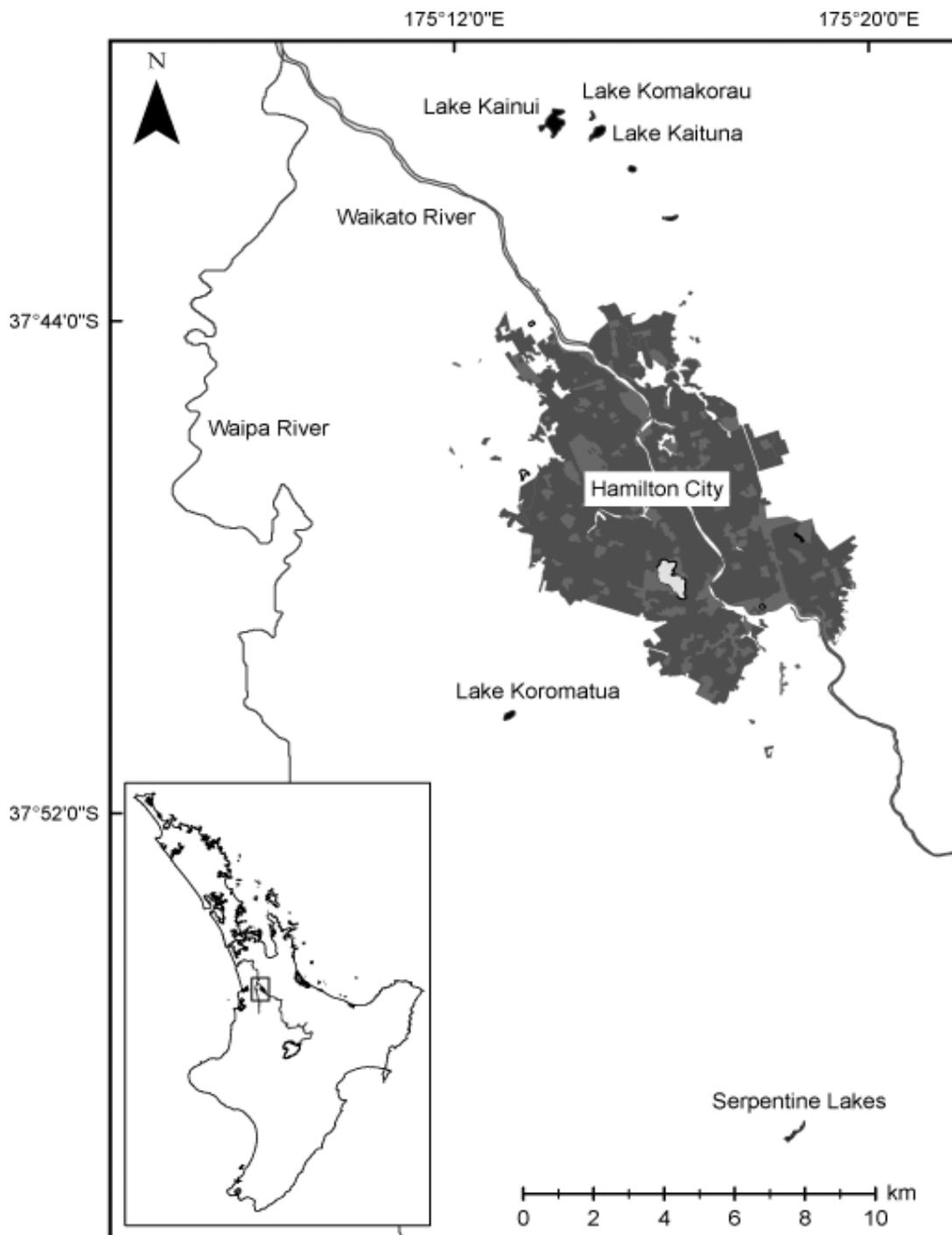


Figure 2.1 Location of study area showing peat lakes in the Waikato region. Shading indicates urban area. Inset: North Island, New Zealand.

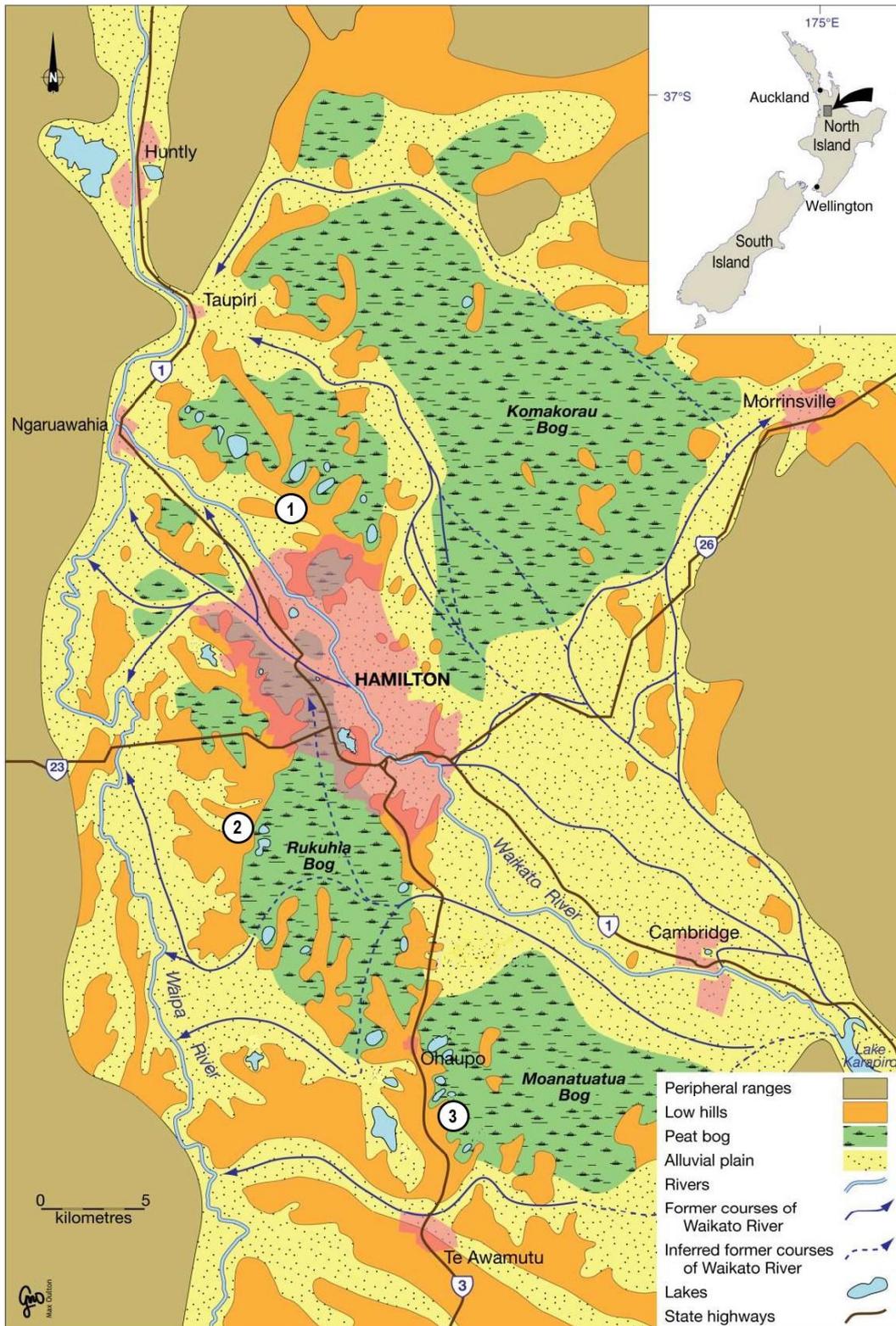


Figure 2.2 Modern landscape features of the Hamilton Basin showing antecedent hills partly buried by volcanogenic alluvium (Hinuera Formation) and post-Hinuera lakes and peat bogs (reproduced with permission from D.J. Lowe, 2010). The location of the study lakes is identified by: 1 (the Horsham Downs lakes), 2 (Lake Korumatua), and 3 (Serpentine Lakes).

Table 2.1 Characteristics of study lakes including morphology, number of streams studied (*n*), trophic status, Trophic Level Index (TLI), water clarity, and average concentrations of chlorophyll *a* (Chl *a*), total phosphorus (TP) and total nitrogen (TN). Source: Hamilton et al. (2010)

Lake and code	Lake area (ha)	Max. depth (m)	Catchment area (ha)	Major inflows (<i>n</i>)	Stocking rate (cows ha ⁻¹)	Trophic state	TLI	Secchi depth (m)	Chl <i>a</i> (mg m ⁻³)	TN (mg m ⁻³)	TP (mg m ⁻³)
Kainui (KN)	25	6.7	260	9	3.20	Hypertrophic	6.18	0.50	45	3041	72
Kaituna (KT)	15	1.3	589	10	3.52	Hypertrophic	6.00	0.32	6	2509	208
Komakorau (KO)	3	1.1	619	3	3.52	Hypertrophic	6.22	0.2	9	2488	200
Koromatua (KR)	7	1.0	67	3	3.13	Hypertrophic	6.99	0.14	32	1492	938
Serpentine North (SN)	5	3.8	163	1	3.19	Eutrophic	4.48	2.02	4	570	48

The streams are described in Table 2.2 including: lake catchment, stream code (unique identifier for each site), subcatchment morphological characteristics (dominant soil type, drainage area, channel width and macrophyte cover), riparian management (fenced and/or planted condition of the stream banks), whether the subcatchment is irrigated with dairy farm effluent (from storage ponds for dairy shed effluent), and, if there is a storage pond for dairy shed and/or feed-pad effluent within the subcatchment. In the Waikato region, applying effluent to land is a permitted activity if certain standards are met including no direct discharges to natural waters.

Table 2.2 Stream subcatchment characteristics and morphologies. Effluent pond = storage facility for dairy farm effluent from milking shed and/or feed-pads for deferred irrigation to land. A = fenced both sides, planted with native vegetation; B = fenced both sides with \geq of 3 wires; C = fenced both side with < 3 wires; D = fenced one side; E = unfenced

Lake	Stream Code	Dominant soil type	Drainage area (ha)	Channel width (m)	Macro-phytes ^α	Rip. mgmt ^β	Irrigated effluent	Effluent pond
Kainui	KN1	Clay loam	17.9	1.40	Yes	B	Yes	No
Kainui	KN2	Peat	122.2	2.04	No	B	No	No
Kainui	KN3	Peat	1.3	2.38	Yes	C	No	No
Kainui	KN4	Peat	1.6	1.40	Yes	C	No	No
Kainui	KN5	Peat	1.5	1.98	Yes	C	No	No
Kainui	KN6	Peat	5.8	3.00	Yes	C	No	No
Kainui	KN7	Peat	0.8	1.50	Yes	C	Yes	No
Kainui	KN8	Peaty loam	15.9	2.00	No	C	Yes	No
Kainui	KN9	Clay loam	15.1	3.33	No	C	Yes	Yes
Kaituna	KT1	Clay loam	77.9	2.65	Yes	B	No	No
Kaituna	KT2	Peat	195.9	5.00	No	B,C	Yes	No
Kaituna	KT3	Peaty loam	3.2	2.80	Yes	C	Yes	No
Kaituna	KT4	Peaty loam	1.0	1.80	Yes	B	Yes	No
Kaituna	KT5	Clay loam	2.9	1.70	Yes	B	Yes	No
Kaituna	KT6	Clay loam	1.7	1.30	Yes	B	Yes	No
Kaituna	KT7	Clay loam	1.3	1.10	Yes	B	Yes	No
Kaituna	KT8	Clay loam	5.4	1.70	Yes	B	Yes	No
Kaituna	KT9	Peaty loam	2.6	1.60	Yes	B	Yes	No
Kaituna	KT10	Clay loam	59.1	4.60	No	B,C	Yes	No
Komakorau	KO1	Peaty loam	5.4	2.30	Yes	B,C	Yes	No
Komakorau	KO2	Clay loam	18.7	2.65	Yes	B	Yes	No
Komakorau	KO3	Peaty loam	5.3	2.50	Yes	B,C	Yes	Yes
Koromatua	KR1	Peaty loam	40.3	3.95	Yes	B	No	Yes
Koromatua	KR2	Clay loam	17.0	2.26	Yes	C,E	No	No
Koromatua	KR3	Clay loam	5.6	3.00	Yes	C,E	No	No
Serpentine North	SN1	Clay loam	9.3	1.30	Yes	A	No	No

^α Yes = macrophyte cover > 50 %

^β Rip. mgmt = Riparian management

2.3.2 Sampling

All twenty-six streams were sampled once during winter (June-August 2010) and once during summer (December 2010-January 2011), between 0800 and 1800 h on each occasion. Fifteen of the streams with permanent flows (KN1, KN2, KN3, KN5, KN6, KN8, KN9, KO3, KR1, KR2, KR3, KT1, KT2, KT10, and SN1) were also sampled during autumn (March-May), winter (July-August) and spring (September-November) 2011.

Winter and autumn sampling of lakes Kainui, Kaituna and Komakorau occurred at water levels between the 55th and 90th percentiles of daily water levels (calculated for the two-year study period from 2010 to 2012). Spring samples were collected at water levels between the 50th to 55th percentiles, and summer samples between the 25th to 50th. Winter and autumn sampling of lakes Koromatua and Serpentine occurred at water levels between the 75th and 95th percentiles of daily water levels over the two-year study period. Spring samples were collected at water levels between the 60th and 65th percentiles, and summer samples between the 55th and 60th.

Water temperature, dissolved oxygen (DO), specific conductivity and pH were measured concurrently with water sample collection, using a YSI 6000 UPG Multi Parameter Sonde (Yellow Springs Instruments, Ohio, USA).

Measurements were made of inflow channel width, wetted width, water depth, sediment depth and water velocity, following standard protocols (Harding et al. 2009)

Subcatchment areas for each stream were calculated using ArcMap 10.0 software (ArcGIS, Environmental Systems Research Institute Inc., CA, USA). Areas were delineated from a digital elevation model (DEM) created from Light Detection and Ranging (LiDAR) data provided by the Waikato Regional Council. Soil types and land use were determined for each subcatchment from the Land Cover Database version 3 (LCDB v3.0) and the New Zealand Soil Classification (NZSC) database (Landcare Research Ltd, 2012).

Water samples were collected using a 1 l measuring jug attached to a pole immediately upstream of the boundary fence between the lake margin and

adjacent dairy farms. Samples were collected in 50 ml centrifuge tubes (Greiner Bio-1, Germany) and 1 l bottles and placed on ice in the field prior to analysis for nutrient and TSS concentrations.

Water samples for analysis of filterable nutrients were syringe-filtered (Whatman GF/C 0.45 μm) in the field and were frozen ($-20\text{ }^{\circ}\text{C}$) upon return to the laboratory, before analysis. Water samples for analysis of total phosphorus (TP) and total nitrogen (TN) were collected unfiltered and frozen prior to analysis (details below).

2.3.3 Sample analyses

Water samples for suspended solids were filtered in the laboratory through pre-combusted ($550\text{ }^{\circ}\text{C}$ for 2 h) and pre-weighed glass fibre filters (Whatman GF/C 0.45 μm). Total suspended solids (TSS) concentrations were determined gravimetrically following drying ($105\text{ }^{\circ}\text{C}$ for a minimum of 8 h) and volatile suspended solid (VSS) concentrations determined following subsequent ashing ($550\text{ }^{\circ}\text{C}$ for 4 h).

Dissolved nutrient concentrations of ammonium ($\text{NH}_4\text{-N}$), nitrite ($\text{NO}_2\text{-N}$), nitrate ($\text{NO}_3\text{-N}$), and phosphate ($\text{PO}_4\text{-P}$) were determined with an Aquakem 200 discrete analyser (Thermo Fisher) using standard colorimetric methods (APHA 2005). Limits of detection were 0.001 mg L^{-1} for $\text{NO}_2\text{-N}$ and $\text{NO}_3\text{-N}$, 0.002 mg L^{-1} for $\text{NH}_4\text{-N}$ and 0.001 mg L^{-1} for $\text{PO}_4\text{-P}$. Total P and TN were determined following alkaline persulphate digestion (APHA 2005) and analysis as for $\text{PO}_4\text{-P}$ and $\text{NO}_3\text{-N} + \text{NO}_2\text{-N}$, respectively, using a Lachat QuickChem[®] Flow Injection Analyser (FIA + 8000 Series, Zellweger Analytics, Inc.). A range of check standards were analysed concurrently with samples to confirm analytical detection limits. Total organic nitrogen (ORG-N) concentrations were calculated by subtracting the sum of $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$ and $\text{NO}_2\text{-N}$ from TN.

2.3.4 Daily Areal Yields (DAY) and Daily Instantaneous Loads (DIL)

Areal water yields for each stream site were calculated using measured instantaneous discharge for each sampling occasion divided by the subcatchment area (ha) and extrapolated to give daily water yields (expressed as

L ha⁻¹ d⁻¹). Water yields were then multiplied by measured nutrient concentrations to give daily areal yields (DAY; kg ha⁻¹ d⁻¹) for NH₄-N, NO₃-N, ORG-N, PO₄-P, TN and TP. Daily areal nutrient yields for each subcatchment were used to calculate area-weighted lake catchment yields and were additionally subtotaled by season and soil type. Daily instantaneous loads (DIL) were calculated by multiplying the daily discharge (L d⁻¹) by measured nutrient concentrations (kg L⁻¹) to give nutrient loads (kg d⁻¹) for NH₄-N, NO₃-N, ORG-N, PO₄-P, TN and TP from each stream subcatchment. DIL were totalled for each lake catchment and summarised seasonally.

2.3.5 Statistical analyses

Multivariate data consisted of physicochemical variables (water temperature, dissolved oxygen saturation, specific conductivity and pH), subcatchment drainage area (drainage area, ha), flow rate (Q), and water quality constituents including nutrients (NH₄-N, NO₃-N, Org-N, PO₄-P, TN, TP) and non-volatile and volatile suspended solids (Non-VSS and VSS). Nitrite was excluded from the analysis as values were <0.001 mg N L⁻¹ for all samples. Data were evaluated using the software Primer 6 (version 6.1.15, Primer-E Ltd., Plymouth Marine Laboratory, Plymouth, UK) with the PERMANOVA + add-in (version 1.0.5).

Data were analysed using a four-factor (season, lake-catchment, predominant subcatchment soil type, and sampling site nested within lake-catchment and soil type) permutational multivariate analysis of variance, PERMANOVA (Anderson et al. 2008; Anderson 2001a). Pair-wise comparisons of group means were completed in the case of a significant factor interaction effect, to assess which levels of the factor were significant (PERMANOVA Anderson 2001a, McArdle & Anderson 2001b). A Type III PERMANOVA for unbalanced designs was performed on the basis of a Euclidean distance matrix (Biondini et al. 1991) calculated from log(X + 0.1) transformed and normalised multivariate data (means were subtracted from each value which were then divided by the standard deviation of the value). Significance was determined by running 9,999 permutations of residuals under a reduced model (Anderson 2001b; Anderson & ter Braak 2003). Following subsequent analyses described below, additional

univariate PERMANOVA analyses were run to test the response of individual nutrient species, VSS and non-VSS to grouping factors.

Distance-based linear modelling, DISTLM (Legendre & Anderson 1999) was used to further examine the significance of grouping factors indicated by the PERMANOVA analyses. An assessment was made of the relative contributions of environmental, morphological and spatial predictor variables in structuring the composition of constituents from all sites across all five seasons. The DistLM procedure tests for significant differences in multivariate response variables to explanatory or predictor variables based on a selected distance-based measure in the form of a resemblance matrix (Anderson et al. 2008). The model partitions the variation in the response variables (nutrient species, VSS and non-VSS) to groups of predictor variables (water temperature, specific conductivity, dissolved oxygen, pH, subcatchment drainage area, and flow rate) as in other canonical approaches (Anderson et al. 2008). The step-wise selection procedure based on 9,999 permutations was used to select and test predictor variables with an adjusted R^2 selection criterion to eliminate insignificant variables. Univariate DISTLM models were also run to examine relationships between predictor variables and individual constituents.

Principle Component Analysis (PCA) (Rao 1964) was used to visualise patterns in the multivariate data indicated by the PERMANOVA and DISTLM analyses. An unconstrained ordination of transformed and normalised data was run using the PCA procedure in Primer 6 based upon the Euclidean distance matrix (Anderson et al. 2008). PCA plots included the grouping factors used in the PERMANOVA analyses; soil type, season and lake-catchment. Additionally, Spearman-rank correlation values were calculated using untransformed data to examine relationships between environmental, morphological and spatial variables and water quality constituents.

2.4 RESULTS

Physicochemical attributes of inflows, including wetted width, water depth, velocity, discharge and sediment depth, as well as water temperature, specific conductivity, percentage of DO saturation and pH, are summarised by lake-catchment, season, dominant soil type and across all stream sites and seasons in Table 2.3.

Table 2.3 Mean values (\pm SD) for physico-chemical attributes of the 26 stream sites summarised by: a) lake-catchment, b) season, c) dominant soil type, and d) all samples across all sites and seasons.

a.	Lake catchment	Wetted width (m)	Water Depth (m)	Velocity (m s ⁻¹)	Discharge (m ³ d ⁻¹)	Sediment depth (m)	Temperature (°C)	Conductivity (mS cm ⁻¹)	DO (%)	pH
	Kainui	1.12 (0.37)	0.25 (0.15)	0.05 (0.07)	1247 (1950)	0.22 (0.15)	14.07 (3.12)	0.25 (0.05)	96 (37)	4.76 (0.87)
	Kaituna	1.36 (0.54)	0.44 (0.20)	0.09 (0.11)	7303 (14595)	0.05 (0.06)	14.20 (2.89)	0.26 (0.08)	57 (34)	5.41 (0.43)
	Komakorau	1.05 (0.40)	0.25 (0.11)	0.13 (0.12)	2445 (1748)	0.18 (0.16)	12.65 (1.67)	0.29 (0.06)	48 (27)	5.93 (0.63)
	Koromatua	1.64 (0.55)	0.32 (0.15)	0.11 (0.12)	4122 (4349)	0.52 (0.49)	14.64 (4.32)	0.16 (0.06)	71 (34)	5.60 (0.45)
	Serpentine South	0.77 (0.17)	0.26 (0.10)	0.11 (0.13)	1685 (1772)	0.03 (0.01)	15.31 (2.73)	0.14 (0.04)	69 (28)	5.66 (0.32)
b.	Season									
	Winter 2010	1.06 (0.46)	0.30 (0.16)	0.09 (0.12)	2253 (3155)	0.13 (0.17)	12.37 (1.14)	0.29 (0.07)	58 (35)	5.47 (0.68)
	Summer 2010	1.50 (0.84)	0.24 (0.14)	0.00 (0.00)	21 (12)	0.11 (0.13)	19.69 (0.17)	0.13 (0.08)	26 (22)	5.29 (0.46)
	Autumn 2011	1.46 (0.52)	0.40 (0.19)	0.11 (0.11)	8870 (16723)	0.12 (0.13)	15.05 (1.54)	0.23 (0.06)	104 (35)	5.06 (0.76)
	Winter 2011	1.29 (0.41)	0.32 (0.19)	0.08 (0.09)	3274 (4298)	0.18 (0.19)	11.14 (1.97)	0.20 (0.04)	104 (19)	5.04 (0.83)
	Spring 2011	1.31 (0.51)	0.28 (0.17)	0.05 (0.06)	2205 (4366)	0.28 (0.40)	18.07 (1.93)	0.20 (0.05)	56 (22)	5.33 (0.85)
c.	Dominant soil type									
	Peat	1.20 (0.56)	0.27 (0.17)	0.06 (0.10)	4934 (14195)	0.15 (0.16)	13.83 (3.76)	0.26 (0.05)	98 (42)	4.38 (0.68)
	Peaty loam	1.35 (0.57)	0.34 (0.14)	0.07 (0.09)	2659 (3917)	0.15 (0.15)	13.51 (3.18)	0.27 (0.07)	64 (32)	5.48 (0.57)
	Clay loam	1.24 (0.44)	0.33 (0.20)	0.09 (0.10)	3486 (5325)	0.21 (0.32)	14.77 (2.76)	0.19 (0.07)	66 (34)	5.67 (0.37)
d.	All samples									
	Mean	1.26 (0.51)	0.32 (0.18)	0.08 (0.10)	3659 (8423)	0.18 (0.25)	14.17 (3.18)	0.23 (0.08)	74 (38)	5.26 (0.76)
	Max.	2.8	0.86	0.47	66437	1.33	21.88	0.43	191	6.76
	Min.	0.42	0.05	0.00	5	0.00	6.67	0.06	1	3.59

Stream nutrient and SS concentrations varied widely across the five lakes and five seasons (Table 2.4A). Total nitrogen concentrations ranged from 0.24 to 13.58 mg L⁻¹ (mean 2.72 mg L⁻¹). Most of this variation was attributable to changes in concentrations of NO₃-N and ORG-N, which ranged from 0.01 to 10.31 mg L⁻¹ and 0.01 to 4.63 mg L⁻¹, respectively (Table 2.4D). Total phosphorus concentrations ranged from 0.01 to 3.0 mg L⁻¹ (mean 0.43 mg L⁻¹) mirroring the relative changes in PO₄-P concentrations. Mean NH₄-N and ORG-N concentrations were greatest from sites with peat, followed by peaty loam, and were two to three times higher than those with clay loam soils (Table 2.4C). Mean TN concentrations were less variable between soil types however, attributable mostly to differences in NO₃-N and Org-N concentrations. Collectively, peat sites generally had higher TP concentrations but lower VSS and non-VSS concentrations compared with peaty loam and clay loam sites.

Table 2.4 Mean concentrations (\pm SD) of nutrient species and suspended solids across all stream sites for the five seasons sampled summarised by: a) stream site; b) season; c) dominant soil type, and; d) all samples.

A Lake	NH ₄ -N mg L ⁻¹	NO ₃ -N mg L ⁻¹	ORG-N mg L ⁻¹	PO ₄ -P mg L ⁻¹	TN mg L ⁻¹	TP mg L ⁻¹	VSS mg L ⁻¹	Non-VSS mg L ⁻¹
Kainui	0.23 (0.26)	1.77 (3.07)	1.55 (1.13)	0.55 (0.61)	3.56 (3.44)	0.73 (0.78)	1.30 (0.98)	2.29 (1.98)
Kaituna	0.28 (0.30)	1.05 (1.15)	0.74 (0.85)	0.03 (0.04)	2.08 (1.44)	0.13 (0.15)	2.32 (1.38)	4.43 (3.59)
Komakorau	0.20 (0.31)	2.23 (2.15)	0.94 (0.75)	0.06 (0.06)	3.37 (2.36)	0.20 (0.16)	4.15 (2.18)	7.92 (5.25)
Koromatua	0.10 (0.10)	0.70 (0.84)	0.92 (1.09)	0.41 (0.60)	1.73 (1.19)	0.48 (0.64)	1.36 (0.81)	2.50 (1.22)
Serpentine Nth	0.09 (0.02)	1.62 (0.90)	0.97 (1.18)	0.04 (0.04)	2.68 (1.55)	0.14 (0.09)	0.27 (0.17)	0.94 (0.54)
B Season								
Winter 2010	0.22 (0.26)	1.92 (3.10)	1.09 (0.89)	0.29 (0.60)	3.23 (3.61)	0.49 (0.78)	1.40 (1.20)	2.26 (1.97)
Summer 2010	0.07 (0.07)	0.33 (0.39)	2.09 (1.16)	0.241(0.33)	2.50 (1.32)	0.33 (0.39)	1.66 (0.04)	3.07 (0.04)
Autumn 2011	0.21 (0.21)	1.60 (1.33)	0.45 (0.39)	0.29 (0.46)	2.27 (1.16)	0.38 (0.49)	1.64 (1.99)	3.97 (4.87)
Winter 2011	0.25 (0.29)	1.56 (1.67)	2.24 (1.13)	0.36 (0.54)	4.06 (1.34)	0.46 (0.62)	1.75 (1.19)	2.84 (1.76)
Spring 2011	0.18 (0.28)	0.37 (0.43)	0.44 (0.46)	0.27 (0.41)	0.98 (0.65)	0.36 (0.49)	1.56 (0.62)	2.76 (0.82)
C Dominant soil type								
Peat	0.36 (0.32)	0.33 (0.58)	1.74 (1.18)	0.74 (0.60)	2.46 (1.39)	0.96 (0.79)	1.61 (1.49)	2.67 (3.52)
Peaty loam	0.20 (0.18)	1.86 (2.62)	1.11 (1.04)	0.30 (0.52)	3.18 (3.06)	0.41 (0.54)	2.06 (1.78)	3.30 (3.49)
Clay loam	0.13 (0.20)	1.74 (2.26)	0.73 (0.80)	0.03 (0.04)	2.60 (2.70)	0.11 (0.12)	1.40 (0.84)	3.02 (1.74)
D All samples								
Mean	0.21 (0.25)	1.40 (2.14)	1.11 (1.06)	0.30 (0.51)	2.72 (2.51)	0.43 (0.61)	1.59 (1.29)	2.96 (2.74)
Max.	1.23	10.31	4.63	2.30	13.58	3.00	6.50	16.10
Min.	0.01	0.01	0.01	0.01	0.24	0.01	0.10	0.20

2.4.1 Grouping factors

The multivariate PERMANOVA including all constituents showed each grouping factor explained a significant amount of the variation in the data. Season ($F_{\text{ratio}}=5.49$, $P<0.01$), soil type ($F_{\text{ratio}}=3.34$, $P<0.01$) and site ($F_{\text{ratio}}=2.46$, $P<0.01$) were the more highly significant grouping factors, followed by lake catchment ($F_{\text{ratio}}=2.47$, $P<0.05$).

Individually, season and stream site accounted for the greatest proportions of variation explained by the PERMANOVA analysis (14.2% and 14.1% respectively; Figure 2.3), followed by soil type (12.8%) and lake (12.6%). The interaction effect between lake and soil type explained the greatest proportion of variation overall (20.3%) and pairwise comparisons indicated that this was driven mostly by differences in soil types between lake catchments rather than different soil types within lake-catchments (Table 2.5A).

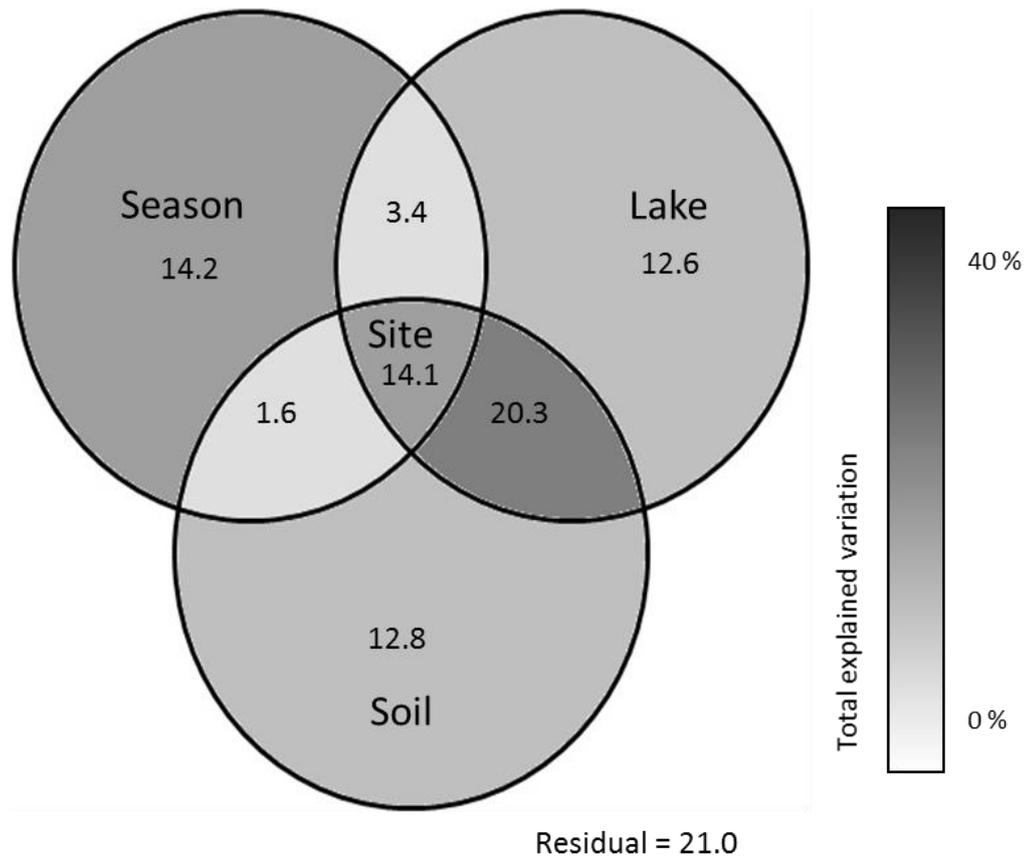


Figure 2.3 Estimates of components of variation for each grouping factor obtained by the multivariate PERMANOVA using all constituents, and physico-chemical and morphological variables from all sites across five seasons. The variability explained by each term is expressed as a percentage of the total variation.

Lake Kainui differed significantly from Lake Kaituna in the relative proportion of peat soil within the catchment ($t = 2.09$, $P = 0.01$), and differed from lakes Koromatua ($t = 1.65$, $P = 0.0001$) and Serpentine North ($t = 3.12$, $P = 0.0001$) in the proportion of clay loam soil in the catchment. The lake-soil interaction effect for Lake Kainui was further investigated using pairwise comparisons, which showed constituent concentrations from the predominantly peat soil subcatchments were significantly different from sites with peaty loam ($t = 2.31$; $P = 0.010$) and clay loam soils ($t = 3.09$; $P = 0.002$).

Pairwise comparisons between seasons indicated samples collected in spring 2011 differed most from other seasons, followed by autumn 2011 ($P < 0.01$; Table 2.5B). Pairwise comparisons between soil types found peat sites differed significantly from clay loam ($P < 0.05$; Table 2.5C) and comparisons between lakes showed constituent inputs to Kainui differed significantly from Kaituna and Koromatua ($P < 0.05$; Table 2.5D).

Table 2.5 Results of pairwise comparisons of constituents for: a) the interaction between lake-catchment and soil types; b) seasons; c) soil types; d) lake-catchments.

A	Kainui	Kaituna	Komakorau	Koromatua
Kaituna	Peat **			
Komakorau	n.s.	n.s.		
Koromatua	Clay loam **	Peaty loam **	n.s.	
Serpentine Nth	Clay loam **	Clay loam **	n.s.	n.s.

B	Winter 2010	Summer 2010	Autumn 2011	Winter 2011
Summer 2010	n.s.			
Autumn 2011	n.s.	n.s.		
Winter 2011	n.s.	n.s.	**	
Spring 2011	**	n.s.	**	**

C	Clay loam	Peat
Peat	*	
Peaty loam	n.s.	n.s.

D	Kainui	Kaituna	Komakorau	Koromatua
Kaituna	*			
Komakorau	n.s.	n.s.		
Koromatua	*	n.s.	n.s.	
Serpentine Nth	n.s.	*	n.s.	n.s.

2.4.2 Relative importance of grouping factors

Univariate PERMANOVAs showed the variation explained by each factor; season, lake-catchment, soil type and site, differed considerably for individual water quality constituents (Table 2.6). Site explained a significant ($P < 0.01$) proportion of the variation in concentrations of $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, $\text{PO}_4\text{-P}$, TP and VSS. The lake/soil interaction effect also accounted for significant ($P < 0.05$) proportions of the variation for $\text{NO}_3\text{-N}$, $\text{PO}_4\text{-P}$ and TP concentrations, and soil type alone had a significant effect ($P < 0.01$) on $\text{PO}_4\text{-P}$ and TP. Soil type was significantly related to ORG-N concentrations, which along with $\text{NO}_3\text{-N}$ and TN, was significantly related ($P < 0.05$) to season and the season/lake interaction.

Table 2.6 PERMANOVA results for a) all constituents, and for individual nutrient species and suspended solids: b) NH₄-N, c) NO₃-N, d) ORG-N, e) PO₄-P, f) TN, g) TP, h) VSS, i) Non-VSS. Bold font in Table 2.6 denotes significant relationships.

Source	df	a. All constituents			b. NH ₄ -N			c. NO ₃ -N			d. ORG-N			e. PO ₄ -P		
		MS	F _{pseudo}	P	MS	F _{pseudo}	P	MS	F _{pseudo}	P	MS	F _{pseudo}	P	MS	F _{pseudo}	P
Season	4	13.52	5.49	0.000	0.34	0.73	0.553	1.75	6.57	0.002	6.12	29.43	0.000	0.05	0.85	0.485
Lake	4	17.08	2.47	0.019	1.18	0.76	0.499	1.79	2.21	0.141	0.98	3.24	0.053	2.44	4.94	0.031
Soil	2	18.47	3.34	0.008	2.94	2.30	0.148	1.61	2.50	0.131	2.00	7.73	0.005	5.16	13.97	0.002
Season x Lake	13	3.39	1.38	0.086	0.16	0.33	0.962	0.65	2.44	0.028	0.48	2.32	0.035	0.05	1.00	0.462
Season x Soil	7	3.14	1.27	0.187	1.29	2.74	0.033	0.32	1.19	0.345	0.14	0.66	0.701	0.06	1.10	0.382
Lake x Soil	4	16.20	2.51	0.012	0.91	0.62	0.622	3.25	4.28	0.024	0.69	2.46	0.121	4.13	9.04	0.002
Site	17	6.06	2.46	0.000	1.42	3.01	0.006	0.71	2.66	0.012	0.27	1.29	0.273	0.42	8.02	0.000
Residual	19	2.46			0.47			0.27			0.21			0.05		
Total	79															

Source	df	f. TN			g. TP			h. VSS			i. Non-VSS		
		MS	F _{pseudo}	P									
Season	4	4.65	13.32	0.000	0.08	0.77	0.551	0.37	0.66	0.575	0.17	0.37	0.810
Lake	4	2.33	3.65	0.036	2.07	2.33	0.135	3.64	2.58	0.111	2.65	3.42	0.043
Soil	2	1.01	1.92	0.194	4.70	7.04	0.012	0.34	0.30	0.710	0.71	1.09	0.358
Season x Lake	13	0.79	2.28	0.038	0.13	1.26	0.298	0.58	1.05	0.438	0.55	1.19	0.336
Season x Soil	7	0.40	1.15	0.363	0.09	0.91	0.516	0.46	0.83	0.552	0.37	0.81	0.578
Lake x Soil	4	1.50	2.53	0.090	3.72	4.52	0.019	0.24	0.21	0.911	1.75	2.43	0.109
Site	17	0.56	1.60	0.164	0.77	7.65	0.000	1.24	2.23	0.030	0.68	1.48	0.198
Residual	19	0.35			0.10			0.55			0.46		
Total	79												

2.4.3 Mechanisms underlying grouping factors

Distance-Based linear modelling (DistLM) was used to investigate significant predictors or drivers of nutrient species, VSS and non-VSS concentrations following the PERMANOVA analyses which had identified the significance of soil type, season and lake-catchment. The DISTLM was constructed using pH, subcatchment drainage area, conductivity, temperature, DO and flow as predictor variables. These variables together explained 37.4% of the variation in stream constituent concentrations across all sites (Cum. R²; Table 2.7).

Individually, pH explained the greatest proportion of the variation (23%) followed by DO and water temperature (6% and 5% respectively; *P*-perm < 0.001). Subcatchment drainage area was also significant (*P*-perm < 0.01) accounting for 3% of the variation in constituents across all sites.

Table 2.7 Multivariate DistLM results relating environmental variables to variations in stream nutrient and suspended solids concentrations. Cum. R²: cumulative percentage of variation explained after entering this variable; SS: Sum of squares; F pseudo: test statistic; P-perm: p-value; Prop.: proportion of explained variance; df: degrees of freedom.

Variable	Cum. R ²	SS	F _{psuedo}	<i>P</i> -perm	Prop.	df
pH	0.222	146.5	23.53	0.000	0.23	78
Dissolved Oxygen	0.272	36.9	6.33	0.000	0.06	77
Temperature	0.313	31.2	5.68	0.000	0.05	76
Conductivity	0.337	19.7	3.72	0.003	0.03	75
Drainage Area	0.357	17.1	3.32	0.008	0.03	74
Flow	0.374	15.0	2.98	0.014	0.02	73
Residual	0.626	365.7			0.58	

Univariate DISTLM models were used to further investigate the relationship between individual constituents and environmental variables. The variation explained by predictor variables for each univariate DistLM has been combined and is presented in Figure 2.4. Similar to the multivariate DistLM, the univariate analyses showed pH was the most significant (*P* < 0.01) predictor of most nutrient concentrations as well as non-VSS. Dissolved oxygen was a significant predictor for the variation in NH₄-N, NO₃-N, VSS, non-VSS (*P* < 0.01), and TN (*P* < 0.05). Temperature significantly explained part of the variation in Org-N and TN (*P* < 0.01), as well as NH₄-N (*P* < 0.05). Subcatchment drainage area explained a significant (*P* < 0.01) proportion of the variation in NH₄-N and ORG-N concentrations, with weak or no relationships with other nutrient species, VSS or non-VSS.

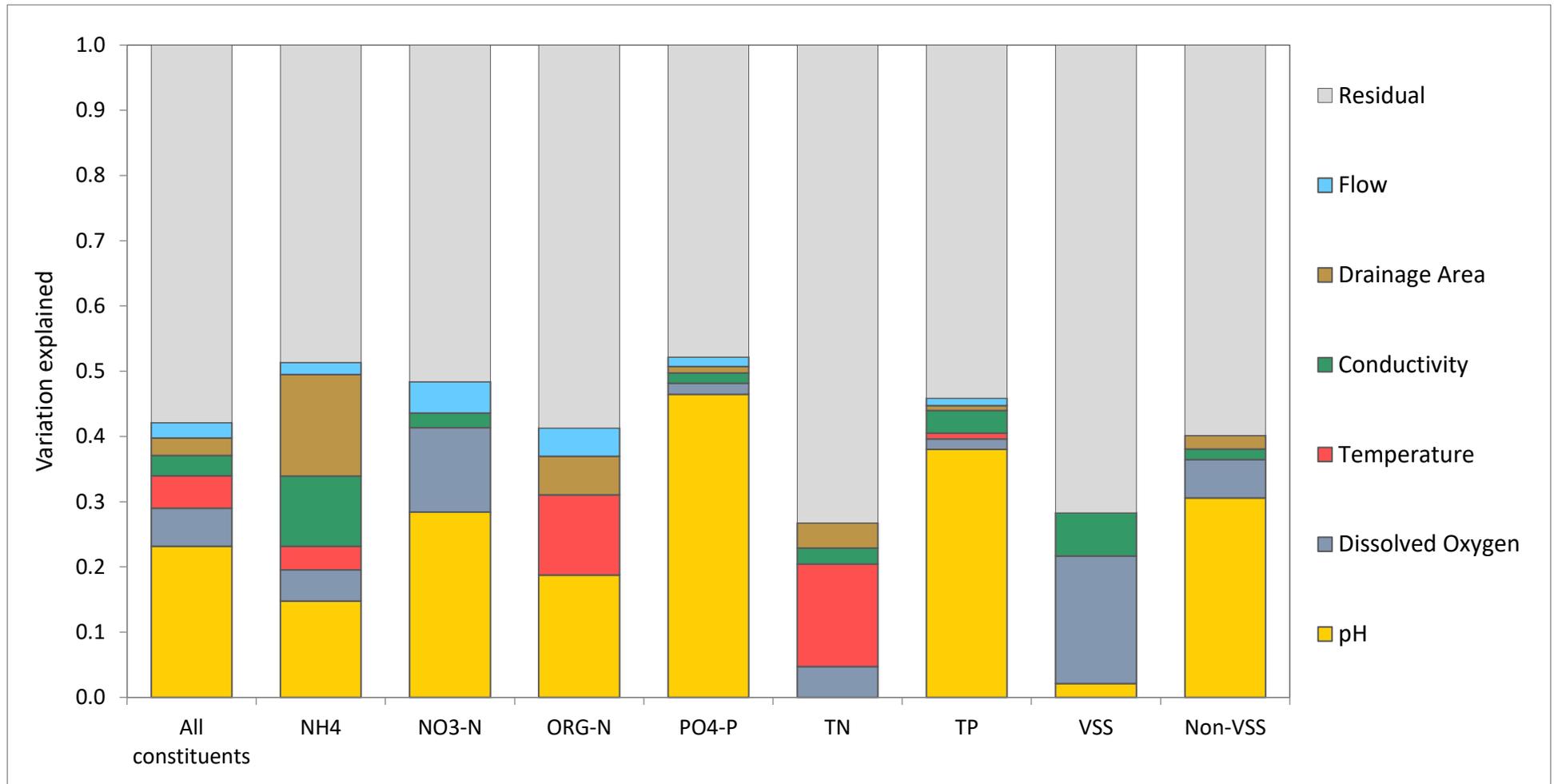


Figure 2.4 Univariate DistLM results relating environmental variables to variation of individual nutrient species and suspended solid concentrations. *NH₄-N*: ammonium; *NO₃-N*: nitrate; *ORG-N*: organic nitrogen; *PO₄-P*: phosphate; *TN*: total nitrogen; *TP*: total phosphorus; *VSS*: volatile suspended solids; *Non-VSS*: non-volatile suspended solids.

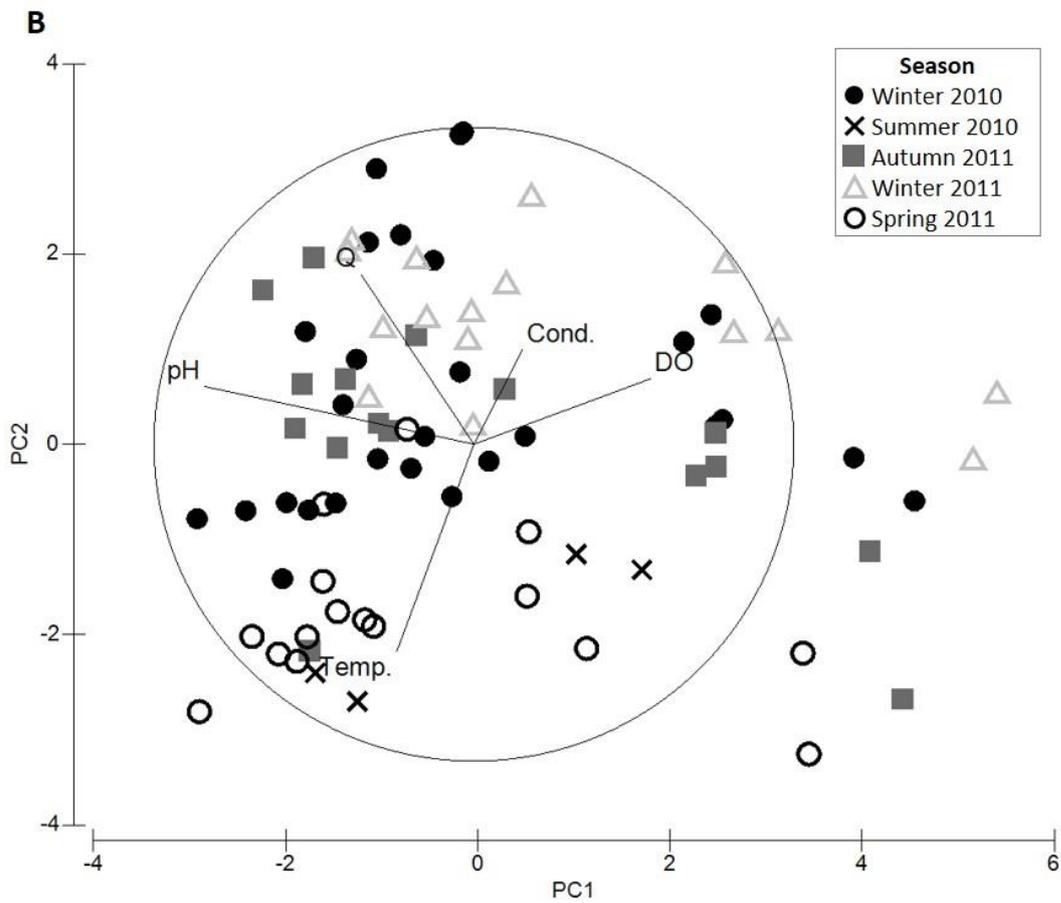
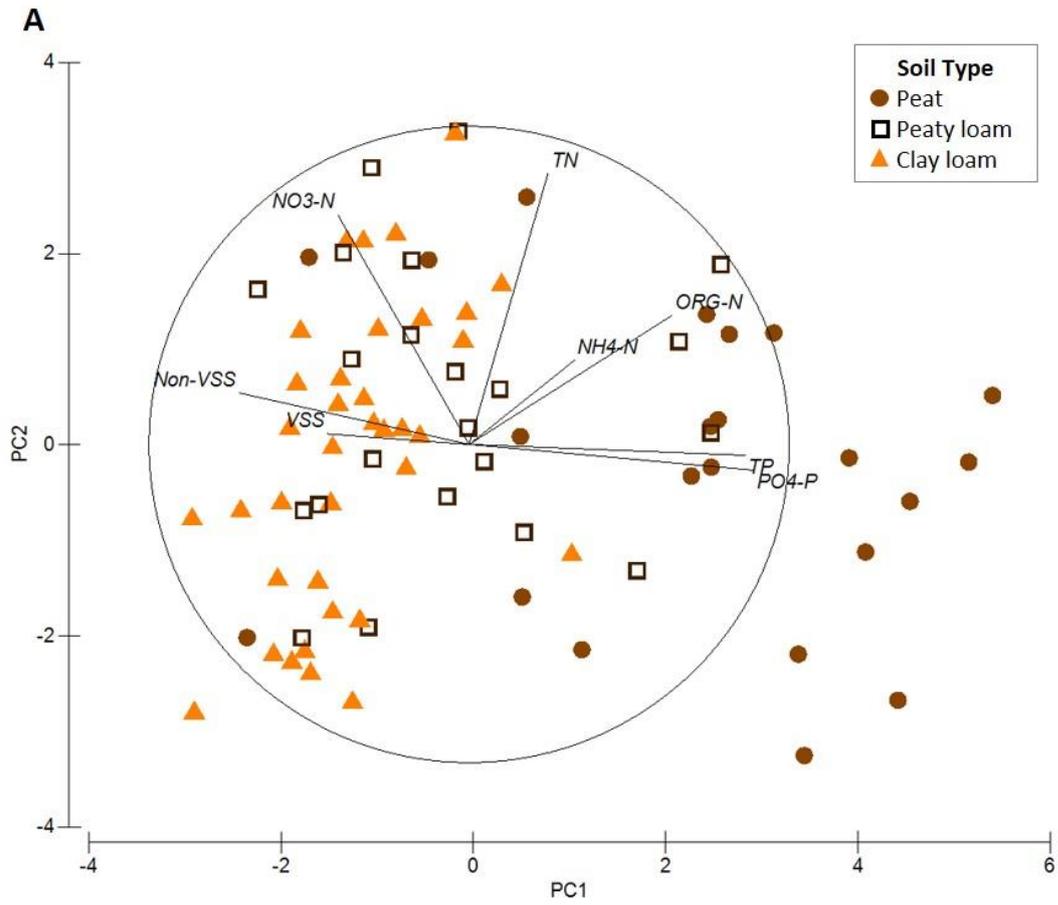
The PCA analysis, based on continuous environmental variables (temperature, DO, specific conductivity, pH, and flow), nutrient species (NH₄-N, NO₃-N, ORG-N, PO₄-P, TN, TP) and suspended solids (VSS and non-VSS), was run to examine relationships amongst soil type, season and site. Sediment depths within the stream channel, subcatchment drainage areas and TSS were excluded from the PCA analysis due to strong correlations with other variables including flow, VSS and non-VSS.

The PCA reinforced the relationships identified by the PERMANOVA and DISTLM analyses. The first PCA axis explained 32.3% (eigenvalue 4.2) of the variation in the data and represented a gradient across soil types (Figure 2.5A). Pairwise comparisons between soil types supported this pattern and showed sites within peat soils differed significantly ($P < 0.05$) from those of clay loam (Table 2.5C). Axis 1 was strongly negatively correlated with pH (-0.410) and non-VSS (-0.351) but was positively correlated with PO₄-P (0.433), TP (0.420) and ORG-N (0.309; Figure 2.5C).

Axis 2 accounted for 18.4% (eigenvalue 2.4) of the variation in the data set and represented a seasonal gradient (Figure 2.5B). This pattern is supported by the findings from the pairwise analysis between seasons, which indicated samples collected in spring 2011 differed most from other seasons, followed by autumn 2011 (Table 2.5B). Flow showed a strong negative correlation with axis two (-0.347) along with TN (-0.550) and NO₃-N (-0.466), whilst temperature gave a positive correlation (0.423; Figure 2.5C).

The third PCA axis explained a further 15.6% (eigenvalue 2.03), contributing to a cumulative variation explained in the dataset of 66.4%. Conductivity showed a strong negative correlation with axis 3, along with VSS (-0.551), non-VSS (-0.341), and NH₄-N (-0.321), while dissolved oxygen was positively correlated with it (0.334; Figure 2.5C). Axis three may be indicative of the variation driven by differences between lake-catchments as indicated by pairwise comparisons which showed Lake Kainui differed significantly ($P < 0.05$) from lakes Kaituna and Koromatua, while Lake Serpentine North differed from Lake Kaituna (Table 2.5D).

Spearman rank-correlation performed on untransformed data indicated several significant relationships between environmental variables and constituents, reflecting those of the PCA. Nitrate concentrations were positively correlated with flow (0.491) and pH (0.449), and pH with non-VSS (0.495), while $\text{NH}_4\text{-N}$, ORG-N, $\text{PO}_4\text{-P}$, and TP were significantly negatively correlated with pH (-0.445, -0.433, -0.479, -0.498 respectively). Temperature was negatively correlated with ORG-N and TN (-0.417 and -0.375 respectively) whilst dissolved oxygen was negatively correlated with VSS (-0.396) and non-VSS (-0.416), and positively correlated with TN (0.386) and $\text{PO}_4\text{-P}$ (0.310).



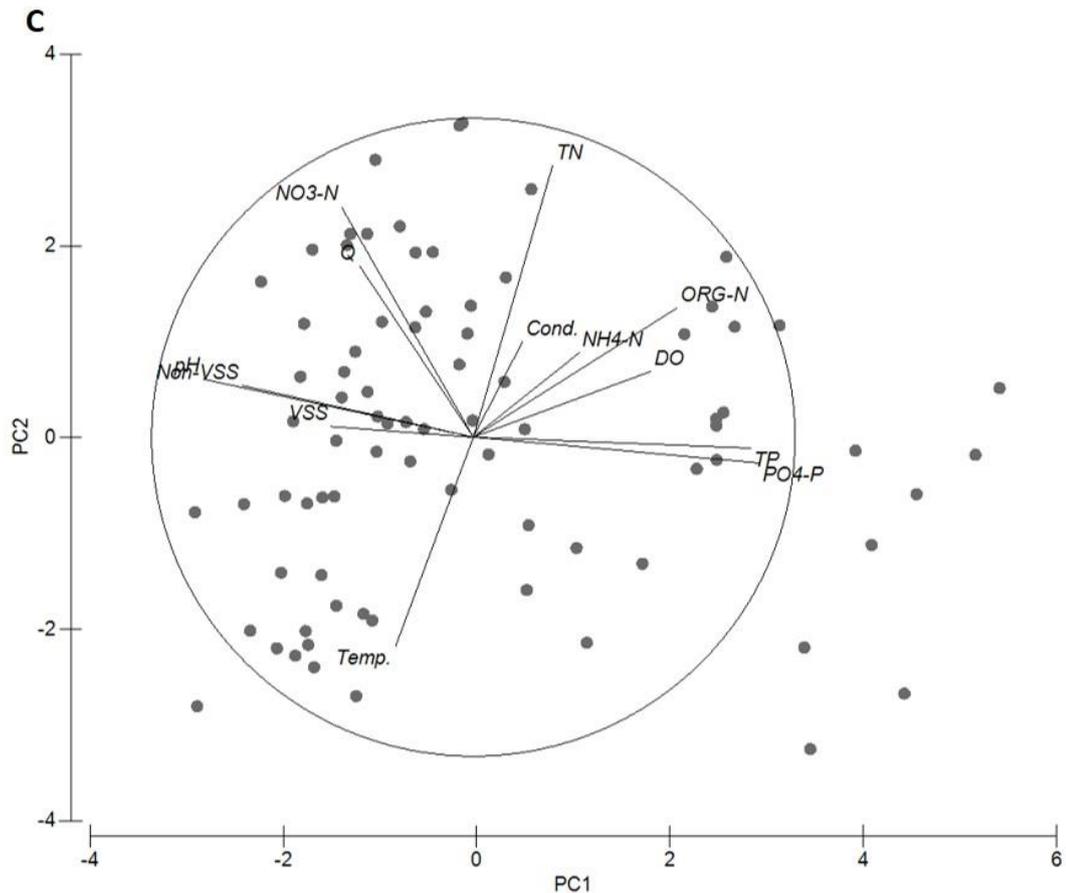


Figure 2.5 PCA ordination plots based on environmental variables, nutrient and suspended sediment concentrations. Sites are plotted according to; A) soil type, B) season, and C) uniformly with vector overlays of environmental variables and all inflow constituents. *NH₄-N*: ammonium; *NO₃-N*: nitrate; *ORG-N*: organic nitrogen; *PO₄-P*: phosphate; *TN*: total nitrogen; *TP*: total phosphorus; *VSS*: volatile suspended solids; *Non-VSS*: non-volatile suspended solids; *Q*: discharge; *Cond.*: specific conductivity; *DO*: dissolved oxygen; *Temp.*: water temperature

2.4.4 Daily Areal Yields (DAY) and Daily Instantaneous Loads (DIL)

Water yields varied significantly seasonally ($F_{\text{pseudo}}=181.3$; $P < 0.001$), being highest in autumn 2011 ($3.30 \text{ L s}^{-1} \text{ ha}^{-1}$) followed by winter 2010 ($2.09 \text{ L s}^{-1} \text{ ha}^{-1}$) and lowest in summer 2010 ($0.02 \text{ L s}^{-1} \text{ ha}^{-1}$). Water yields were intermediate during winter and spring 2011 (1.74 and $1.45 \text{ L s}^{-1} \text{ ha}^{-1}$, respectively).

Subcatchments draining peat soils had significantly lower mean water yields ($1.48 \text{ L s}^{-1} \text{ ha}^{-1}$; $F_{\text{pseudo}} = 5.55$; $P < 0.01$) compared with peaty loam and clay loam sites (2.33 and $2.18 \text{ L s}^{-1} \text{ ha}^{-1}$, respectively). Water yields from peat subcatchments also had high variability ($\text{SD} = 2.50 \text{ L s}^{-1} \text{ ha}^{-1}$), largely attributable to the variability of site KN6. The wetted width and depth of KN6 were relatively stable over the five seasons ($1.34 \pm 0.08 \text{ m}$ and $0.4 \pm 0.06 \text{ m}$, respectively),

however water velocity was more variable ($0.07 \pm 0.09 \text{ m s}^{-1}$), contributing to relatively high, fluctuating discharge and water yields ($4.71 \pm 4.54 \text{ L s}^{-1} \text{ ha}^{-1}$).

Daily areal yields (DAY; $\text{kg ha}^{-1} \text{ d}^{-1}$) of TN and TP for each lake-catchment have been summarised by soil type and season (Table 2.8). DAY of TN for Lake Kainui varied seasonally but were similar between peat and clay loam subcatchments, while DAY of TP from the peat subcatchments of Lake Kainui (mean $0.42 \text{ kg ha}^{-1} \text{ d}^{-1}$) were ~50-fold and ~15-fold greater than peaty loam (mean $0.01 \text{ kg ha}^{-1} \text{ d}^{-1}$) and clay loam (mean $0.03 \text{ kg ha}^{-1} \text{ d}^{-1}$), respectively. By contrast, DAY of TP for Lake Kaituna did not differ between soil types, and DAY of TN were lowest from peat subcatchments (mean $0.36 \text{ kg ha}^{-1} \text{ d}^{-1}$) compared with peaty loam (mean $0.69 \text{ kg ha}^{-1} \text{ d}^{-1}$) and clay loam (mean $0.59 \text{ kg ha}^{-1} \text{ d}^{-1}$). DAY of TN from peaty loam soils were highest for Lake Komakorau (mean $1.69 \text{ kg ha}^{-1} \text{ d}^{-1}$), while highest DAY of TP were exported to Lake Koromatua from peaty loam soils (mean $0.24 \text{ kg ha}^{-1} \text{ d}^{-1}$).

Table 2.8 Daily areal yields ($\text{kg ha}^{-1} \text{d}^{-1}$) for total nitrogen (TN) and phosphorus (TP) for the streams sampled, summarised by season, dominant soil type, and lake-catchment.

Lake	Kainui								Kaituna							
	Peat		Peaty loam		Clay loam		Total		Peat		Peaty loam		Clay loam		Total	
Soil Type	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP
	$\text{kg ha}^{-1} \text{d}^{-1}$		$\text{kg ha}^{-1} \text{d}^{-1}$		$\text{kg ha}^{-1} \text{d}^{-1}$		$\text{kg ha}^{-1} \text{d}^{-1}$		$\text{kg ha}^{-1} \text{d}^{-1}$		$\text{kg ha}^{-1} \text{d}^{-1}$		$\text{kg ha}^{-1} \text{d}^{-1}$		$\text{kg ha}^{-1} \text{d}^{-1}$	
Winter 2010	2.94	0.67	3.49	0.03	2.96	0.08	0.63	0.02	0.22	0.01	0.75	0.07	1.00	0.08	0.24	0.01
Summer 2010-11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Autumn 2011	1.28	0.75	0.17	0.00	0.55	0.01	0.13	0.03	1.26	0.06	2.59	0.03	1.25	0.02	0.99	0.04
Winter 2011	1.59	0.57	0.27	0.00	2.66	0.04	0.29	0.02	0.31	0.01	0.08	0.00	0.61	0.01	0.30	0.01
Spring 2011	0.14	0.09	0.02	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.11	0.00	0.03	0.00

Lake	Komakorau				Koromatua				Serpentine South					
	Peaty loam		Clay loam		Total		Peaty loam		Clay loam		Clay loam			
Soil Type	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP		
	$\text{kg ha}^{-1} \text{d}^{-1}$		$\text{kg ha}^{-1} \text{d}^{-1}$		$\text{kg ha}^{-1} \text{d}^{-1}$		$\text{kg ha}^{-1} \text{d}^{-1}$		$\text{kg ha}^{-1} \text{d}^{-1}$		$\text{kg ha}^{-1} \text{d}^{-1}$			
Winter 2010	3.91	0.16	1.21	0.03	1.22	0.04	0.30	0.31	0.55	0.02	0.30	0.20	0.50	0.02
Summer 2010-11	0.00	0.00	-	-	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.00
Autumn 2011	2.69	0.14	-	-	0.48	0.02	0.21	0.18	0.83	0.03	0.34	0.12	1.05	0.03
Winter 2011	1.86	0.06	-	-	0.33	0.01	0.35	0.12	1.47	0.03	0.55	0.08	0.58	0.02
Spring 2011	0.02	0.01	-	-	0.00	0.00	0.41	0.60	0.35	0.04	0.31	0.39	0.04	0.01

Table 2.9 Daily instantaneous loads (DIL) of total nitrogen (TN) and phosphorus (TP) totalled for each lake-catchment, summarised by season.

Lake	Area ha	Winter 2010		Summer		Autumn 2011		Winter 2011		Spring 2011	
		TN kg d ⁻¹	TP kg d ⁻¹								
Kainui	25	115.0	4.2	0.0	0.0	23.4	6.4	52.6	2.8	1.6	0.6
Kaituna	15	87.5	4.6	0.0	0.0	356.3	14.0	106.5	2.5	10.2	0.4
Komakorau	3	35.8	1.3	0.0	0.0	14.2	0.7	9.8	0.3	0.1	0.1
Koromatua	7	18.9	12.8	0.2	0.0	21.2	7.7	34.8	5.2	19.6	24.8
Serpentine North	5	4.6	0.2	0.1	0.0	9.8	0.2	5.4	0.2	0.4	0.1

Daily instantaneous loads (DIL) for TN and TP for each subcatchment were summed for lake catchments and summarised by season (Table 2.9). Combined TN DIL for lake catchments were greatest during autumn 2011 (mean 85 kg d⁻¹), followed by winter 2010 and 2011 (mean 52 and 41 kg d⁻¹, respectively), then spring 2011 (mean 6 kg d⁻¹). Combined TP DIL were similarly highest in autumn 2011 (mean 5.8 kg d⁻¹), followed by spring 2011 (mean 5.2 kg d⁻¹) then winter 2010 (mean 4.6 kg d⁻¹). DIL ranged between individual subcatchments from 0.01 to 248 kg d⁻¹ for TN (mean 12 kg d⁻¹) and 0.01 to 25 kg d⁻¹ for TP (mean 1.1 kg d⁻¹) over the seasons sampled. Total DIL of TN exported were highest for Lake Kaituna (mean 112 kg d⁻¹), particularly during autumn and winter of 2011 (356 kg d⁻¹ and 107 kg d⁻¹, respectively). DIL of TP to Lake Koromatua were high (mean 19 kg d⁻¹), and moderate for Lake Kaituna (mean 4.3 kg d⁻¹) and Lake Kainui (mean 2.8 kg d⁻¹; Table 2.9).

2.5 DISCUSSION

The shallow peat lakes in this study received substantial nutrient and sediment loads despite their small subcatchment sizes (62% of sites < 10 ha, 81% of sites < 20 ha) and often ephemeral nature of the watercourses. Load estimates are commonly used to inform lake and catchment managers as well as policy makers, to support local, regional and national objectives to minimise nutrient and sediment exports to aquatic ecosystems. In this study, both DAY and DIL were calculated for individual subcatchments so that it was possible to identify the most important sources of N and P to the peat lakes. This approach enabled simplification of complex spatial and temporal variability, thus assisting water managers to implement more targeted nutrient management at subcatchment scales.

2.5.1 Daily Areal Yields (DAY) and Daily Instantaneous Loads (DIL)

Daily areal yields of N were highly variable between individual subcatchments (mean 0.5, SD 0.8 kg N ha⁻¹ d⁻¹) as well as across seasons, comparable to seasonal patterns of N loss from catchments dominated by dairy production in Waikato (Wilcock et al. 1999) and Taranaki regions (Wilcock et al. 2009) of New Zealand. Exports of N increased substantially during autumn and winter, with whole lake catchment DAY (range 0.5 – 9.4 kg N ha⁻¹ d⁻¹) up to nearly one-third of annual nitrate-N losses reported elsewhere in New Zealand (mean 35 kg N ha⁻¹ y⁻¹, equivalent to c. 0.09 kg N ha⁻¹ d⁻¹) (Monaghan et al. 2000; Wilcock et al. 1999). Stream sites KR1 (draining to Lake Koromatua) and KN9 (draining to Lake Kainui) had particularly high mean DAY of N (1.8 and 1.4 kg N ha⁻¹ d⁻¹, respectively), effectively highlighting N-leaching “hotspots”. Compared to the seasonal patterns in DAY of N, there was less variability with soil types, however, losses of nitrogen species (NO₃-N, NH₄-N and ORG-N) were significantly different between soil types.

Daily aerial yields of P varied most between soil types and were greatest from peat subcatchments of Lake Kainui (0.09 – 0.75 kg P ha⁻¹ d⁻¹), and peaty loam subcatchments of lakes Komakorau (0.01 – 0.16 kg P ha⁻¹ d⁻¹) and Koromatua

(0.12 – 0.60 kg P ha⁻¹ d⁻¹). DAY of P were high relative to annual yield estimates from similar studies in New Zealand and Australia (see review by Drewry et al. (2006)). By comparison, Abell et al. (2013) estimated annual yields of 0.90 and 1.34 kg P ha⁻¹ yr⁻¹ (c. 0.003 and 0.004 kg P ha⁻¹ d⁻¹) for two stream subcatchments of Lake Rotorua, a large (8100 ha), eutrophic lake in the Bay of Plenty region of the North Island, and Verburg et al. (2013) report similar yields (mean 0.6 kg P ha⁻¹ yr⁻¹, c. 0.002 kg P ha⁻¹ d⁻¹) from their study of subcatchments of Lake Brunner with intensive dairy farming land use in the South Island. Nexhip & Austin (1998) report loads of 6-17 kg P ha⁻¹ yr⁻¹ (c. 0.016 – 0.047 kg P ha⁻¹ d⁻¹) from border-dyke irrigated dairy farms in Australia, while in a study investigating P losses in surface drainage from land used for dairy grazing in Southland, McDowell & Monaghan (2015) report loads over 18 months of 87 kg P ha⁻¹ (c. 0.159 kg P ha⁻¹ d⁻¹) from organic soils recently developed (for dairy) and 9.0 kg P ha⁻¹ (c. 0.016 kg P ha⁻¹ d⁻¹) from intergrade soils. These load estimates more closely approximate those from our study, particularly for peat subcatchments of Lake Kainui.

Converting DAY to daily instantaneous loads (DIL) enabled some insight into the magnitude of nutrient exports from our subcatchments relative to the area of the shallow peat lakes. For example, Lake Koromatua had an areal load of 99 g m⁻² yr⁻¹ TN and 49 g m⁻² yr⁻¹ TP, far exceeding estimates of Verburg et al. (2013) for Lake Brunner of 12 g m⁻² yr⁻¹ TN and 0.5 g m⁻² yr⁻¹ TP. Analysis of 20 years of data from water quality monitoring of Lake Mendota (Wisconsin, USA) revealed extreme fluctuations of daily P loads (90th percentile values 6 to 80 kg TP d⁻¹) exported from its large (60400 ha) predominantly agricultural catchment (Carpenter et al. 2014). Areal loads, however, were relatively small (90th percentile range 0.06 - 0.75 g m⁻² yr⁻¹ TP) owing to the large surface area of the lake (3960 ha). To improve water quality and meet restoration objectives for Waikato's shallow peat lakes, concerted efforts will be required to adequately reduce such substantial loads of N and P.

2.5.2 Variability

Akin to nutrient loads, concentrations of N and P ranged considerably across stream sites and seasons, with TN varying >55-fold and TP >200-fold. Smith

(1987) describes strong seasonal variation and similarly high concentrations of TN and TP in surface run-off immediately following heavy rainfall events from a New Zealand pastoral catchment, as does Wilcock et al. (1999) from a larger catchment (1510 ha) dominated by intensive dairy farming. Molenat et al. (2008) show similarly large variations in NO₃-N exports amongst seasons and catchments from small agricultural headwaters in France, and high concentrations (> 1 mg L⁻¹ TN and > 0.5 mg L⁻¹ TP) and variability have been reported from streams in catchments dominated by dairy farming in Australia (Adams et al. 2014; Smith et al. 2013) and the Northern Hemisphere (Ulén et al. 2007; Kyllmar et al. 2006; Granger et al. 2010). Concentrations of VSS and non-VSS from our study, however, were low relative to values reported elsewhere for streams from catchments dominated by dairy production (McDowell 2006; Adams et al. 2014) suggesting these constituents may be a lower management priority for the lakes in our study. The relatively small size and low gradient of the streams, as well as extensive fencing, may have contributed to the low levels of SS. Notwithstanding, storm events are a major mechanism for delivering high sediment and nutrient loads from agricultural catchments (Abell et al. 2013; Elliott & Basher 2011).

The variability of nutrient concentrations was related to interactions amongst season and soil type, as well as lake catchment characteristics which, as posited by Adams et al. (2014), are likely to be influenced by local land management practices, critical source areas and groundwater inputs. Interpreting the different responses of individual nutrient species to these drivers enabled us to identify specific environmental and hydro-physicochemical mechanisms influencing the flux of nutrients and SS from the subcatchments of this study.

2.5.3 Seasonal variability

Temporal patterns in rainfall, temperature and evaporation are associated with the seasonality of New Zealand's temperate maritime climate. Seasonality was the most important driver of variations in concentrations of Org-N, NO₃-N and TN. Total N was strongly negatively related to water temperature in this study, indicative of less N leaching during the spring and summer and likely to be

associated with increased productivity of pasture grasses and greater uptake by in-stream processes during these seasons (Brougham 1959; Peterson et al. 2001). Concentrations of TN were considerably higher during autumn and winter due to varying elevated levels of $\text{NO}_3\text{-N}$, Org-N and $\text{NH}_4\text{-N}$, the relative contributions of which were influenced by stream flow, pH and DO saturation. Quinn & Stroud (2002) similarly found stream flow was strongly positively related to $\text{NO}_3\text{-N}$ concentrations, indicating the importance of considering hydrological processes such as surface runoff, leaching and subsurface lateral movement (Abell et al. 2013) when evaluating exports of $\text{NO}_3\text{-N}$ from agricultural land use.

2.5.4 Soil variability

Soil type strongly affects nutrient and SS mobility and transport, thereby influencing rates of nutrient leaching and sediment erosion (McLaren & Cameron 1996). Dissolved oxygen saturation and pH are significantly influenced by soil type and, in this study, peat and peaty loam soils in particular. Streams draining predominantly peat subcatchments had lower pH, reduced DO, lower flows and higher concentrations of $\text{NH}_4\text{-N}$, ORG-N and $\text{PO}_4\text{-P}$. Streams from clay loam subcatchments had $\text{NO}_3\text{-N}$ concentrations more than five-fold higher than those from peat which, as previously discussed, was closely related to stream flow.

Higher flows from clay loam soils are likely to be related to both the lower porosity of these soils compared with peat and the topography of clay loam subcatchments in our study, which are generally steeper than those of the low-lying peat. The clay loam soils in this study are classified as orthic granular and typically have low permeability, becoming sticky and plastic after heavy rainfall, which results in periods of perched water (Hewitt 2010). Subsequent rainfall can transport previously perched water into waterways, along with additional $\text{NO}_3\text{-N}$ from the topsoil porewater, leading to elevated in-stream $\text{NO}_3\text{-N}$ concentrations. Such processes may partially explain the findings of our research where $\text{NO}_3\text{-N}$ concentrations were highest in streams draining clay-dominated subcatchments.

Peat, by comparison, has high porosity (Clymo 1983) and becomes saturated during wetter months as water tables rise, creating anoxic conditions which can

enhance denitrification (Rutherford & Nguyen 2004). The high organic matter content of peat soils also provides an essential carbon source for denitrifying bacteria to metabolise $\text{NO}_3\text{-N}$ (McLaren & Cameron 1996). de Klein et al. (2006) state that soil-stored $\text{NO}_3\text{-N}$ released as nitrous oxide following denitrification is associated with water-filled pore spaces in soil and seasonal rainfall patterns. However, Tiemeyer & Kahle (2014) report substantial $\text{NO}_3\text{-N}$ losses from degraded peatland used as intensive grassland in Germany, which they attribute to mineralisation of organic matter, leading to nitrification, coupled with large fluctuations in groundwater levels and discharge rates.

Peat subcatchments had higher levels of $\text{NH}_4\text{-N}$ than clay loam, with mean values exceeding (> 17-fold) the 80th percentile value ($0.021 \text{ mg NH}_4\text{-N L}^{-1}$) for reference sites outlined in the ANZECC (2000) guidelines for lowland streams. $\text{NH}_4\text{-N}$ was highly negatively correlated with DO, indicative of reducing conditions in peat soils, particularly when soils are saturated during autumn and winter. Reducing conditions will prevent nitrification by ammonium-oxidising bacteria such as *Nitrosomonas* and *Nitrobacter*, and, as the activity of such microbial communities can be altered by temperature (Avrahami et al. 2003; Tourna et al. 2008), ammonium oxidation may be further suppressed during cooler months. High in-stream concentrations of $\text{NH}_4\text{-N}$ were also correlated with low pH. The activity of *Nitrosomonas* in particular is strongly influenced by soil pH (Stephen et al. 1998; Pommerening-Röser & Koops 2005), and the acidic environment of peat can inhibit nitrification, particularly where there are peat soils in agricultural land use (Allison & Prosser 1991). Other microbial processes, however, can support nitrification in low-pH environments, for example, those associated with surface growths of biofilm produced by *Nitrobacter* (Keen & Prosser 1987) and via ureolytic ammonia oxidisers during urea hydrolysis (Burton & Prosser 2001).

Streams draining predominantly peat subcatchments also had higher concentrations of $\text{PO}_4\text{-P}$ compared with clay loam. Mean values for peat subcatchments exceeded (> 74-fold) the 80th percentile values ($0.01 \text{ mg PO}_4\text{-P L}^{-1}$) for reference sites in lowland streams (ANZECC 2000). Peat soils generally have low anion storage capacity (ASC) (O'Connor et al. 2001) and if subjected to

large quantities of P from fertilisers, effluent applications, or manure, will release $\text{PO}_4\text{-P}$ to waterways (Van der Elst & Kinloch 1980). Morton et al. (2003) examined concentrations of $\text{PO}_4\text{-P}$ in overland flow from dairy-farm soils with different ASCs and found high desorption rates associated with low-ASC soils (e.g. Waikoikoi pallic soil, ASC 15%) and high sorption for high-ASC soils (e.g. Egmont allophanic soil, 83% ASC). The $\text{PO}_4\text{-P}$ concentrations leached from soils with low ASC in their study were markedly lower ($\sim 0.03 - 0.4 \text{ mg L}^{-1}$) than those arising from the peat subcatchments in our study ($0.1 - 2.3 \text{ mg L}^{-1}$), suggesting other mechanisms may be responsible for our very high observed $\text{PO}_4\text{-P}$ losses. O'Connor et al. (2001) report the ASC of organic soils increases ($\sim 30\text{-}80\%$) following development of peat from a 'raw' to a 'developed' state via drainage and cultivation, reducing the carbon content and increasing mineralisation. The peat-lands in our study have undergone significant drainage and cultivation since c. 1950 (Hunt 2007) during which time the ASC of the topsoil will have increased, therefore processes other than ASC are likely to be influencing $\text{PO}_4\text{-P}$ leaching from our peat subcatchments.

There was a negative exponential relationship between concentrations of $\text{PO}_4\text{-P}$ and surface water pH in streams draining peat subcatchments. Sites with very low surface-water pH (< 4.5) had $\text{PO}_4\text{-P}$ concentrations four to five-fold greater than sites with pH between 4.5 and 6. The very low-pH (< 4.5) sites drained peat subcatchments with deep ($\geq 7 \text{ m}$), acidic, fibric organic soils in which the peat materials are weakly decomposed compared with shallow (1-2 m) mellow-mesic organic soils which are moderately decomposed (Hewitt 2010; Davoren et al. 1978). The development and cultivation of peat soils to support agriculture results in different physical properties between the upper ploughed layer (c. 0-15 cm) and the underlying, relatively unaffected peat (McLay et al. 1992). The physical differences between the mineralised upper layer and undeveloped peat beneath (for examples see Davoren et al. 1978) are likely to extend to biogeochemical processes and influence nutrient cycling. Furthermore, the aforementioned differences between the upper ploughed layer and the underlying peat are likely to be more pronounced in deep peat soils compared with shallow soils, as peat physicochemical characteristics alter with increasing

depth of organic deposits (Van der Elst & Kinloch 1980). The exponential increase in $\text{PO}_4\text{-P}$ concentrations along a gradient of increasing peat depth and acidity shown in our research may provide evidence for different nutrient cycling and dynamics between deep and shallow peat soils.

Plant-available P declines in organic soils where pH falls below 5.5 due to high concentrations of soluble iron (Fe) and aluminium (Al) which readily fix P (Lucas & Davis 1961; Dawson 1956), a finding supported by O'Connor et al. (2001) from their research on peat soils in the Waikato region. The ASC of the upper mineralised ploughed layer in deep peat sites may be considerably higher than that of the undeveloped underlying peat where pH falls below 4.5. With downward migration of nutrients and water through the soil profile, P will tend to move into deeper layers where changes in pH and oxygen saturation (redox state) may lead to conditions favouring its release, as suggested by the results of this study.

Redox state significantly influences soil biogeochemistry and strongly controls the mobility of P, with the greatest release of sorbed and precipitated P occurring under acid reducing conditions (Richardson & Vaithiyanathan 2009). The transition zone between the developed and undeveloped peat layers is likely to align closely with the water table and the point where the peat is saturated. Redox state changes through the peat profile due to aerobic decomposition in the upper layer, which rapidly exhausts free oxygen within the soil, accelerated by increasingly saturated soil microenvironments rich in organic matter, thereby establishing anaerobic, reducing conditions and fostering P release (Langmuir 1997). This redox interface is likely to be only a few millimetres or centimetres deep in saturated, organic soils and will therefore be acutely sensitive to hydrological changes associated with rainfall, evaporation, barometric pumping, artificial drainage and water table manipulation (Langmuir 1997). The location and thickness of the redox interface will change in relation to water table depths and the availability of undecomposed organic matter. The interface may therefore be of greater importance in deep peat where there is a large reserve of undecomposed organic matter. As redox is reduced through the interface, the

peat between the upper decomposed, mineralised layer and the lower undecomposed layer may become partially decomposed, resulting in release of humic acids (Van der Elst & Kinloch 1980). These acids reduce the pH of the soil environment and, at $\text{pH} < 4.5$, mobilise hydroxides of Fe and Al, thereby encouraging further release of soluble inorganic P (Fox et al. 1990). Once released, P moves readily through undecomposed peat, enriching adjacent waterways (Hogg & Cooper 1964). This process provides a plausible explanation for the very high concentrations of $\text{PO}_4\text{-P}$ in streams draining deep peat subcatchments in our study.

2.5.5 Managing diffuse pollution

Barling & Moore (1994) advocate for controlling agricultural contaminants at their source, however, where appropriate controls are impractical, or cost or time prohibitive, alternative management methods will be required. Our findings suggest management practices aimed at reducing N, P and sediment losses from dairy farms should be specific to soil type. To reduce $\text{NO}_3\text{-N}$ losses from clay soils, remediation measures will need to target reductions in direct flows to waterways by intercepting transport pathways, as well as enhancing the denitrification potential within the catchment. Implementing best management practices (BMPs) for riparian areas, including adequate fencing and planting, have been shown to reduce $\text{NO}_3\text{-N}$ losses as well as nutrient and sediment runoff generally (Smith 1989; Wilcock et al. 2009; Cooper et al. 1995). Reinstating wetlands in riparian areas, as well as in gullies and depressions in the landscape where overland flow occurs in heavy rainstorm events, is also likely to enhance nutrient and sediment removal via filtration, sedimentation and denitrification, as well as by slowing flows (Rutherford & Nguyen 2004; Reddy et al. 1999; Hoffmann et al. 2011). Furthermore, improving soil management practices in order to maintain the naturally high organic content of peat, or employing methods to enhance the organic carbon content in top-soil may facilitate denitrification processes (Schipper et al. 2010). Condrón et al. (2000) and others (Gomiero et al. 2011; Dalgaard et al. 1998) show that organic dairy farms can

have greater carbon content in their top-soils compared to conventional systems, improving N cycling and reducing N-losses.

Targeting $\text{NH}_4\text{-N}$ losses from agricultural peat soils requires careful consideration of soil microbial communities and requisite environmental conditions to promote N cycling and atmospheric release. For example, Mahmood & Prosser (2006) report subspecies of *Nitrosospira*, important in nitrification, have different sensitivities to urea, with consequent effects on N cycling in soils, while soil management practices (Sun et al. 2004), fertiliser applications (Webster et al. 2005), and soil particle surface parameters (Jiang et al. 2011) also affect microbial processes. Moreover, identifying the predominant sources of $\text{NH}_4\text{-N}$ which are generally related to land management practices, including animal excreta (stocking rates and grazing rotations), decomposing organic matter and N fertilisers, will assist with reducing $\text{NH}_4\text{-N}$ exports.

Managing excessive $\text{PO}_4\text{-P}$ losses from peat soils under intensive agriculture in New Zealand requires renewed, collaborative research between agricultural scientists and hydro-geochemists. Clarifying and expanding our knowledge of the hydrogeochemical processes occurring at the redox interface between upper mineralised and deeper, undeveloped peat layers, and its influence on P cycling, would help to inform better water table management for artificial drainage networks within agricultural peat lands and minimise $\text{PO}_4\text{-P}$ losses. Landowners generally attempt to manage water levels to maximise the productivity of upper cultivated peat layers but management goals can be extended to include minimising losses of valuable nutrient resources for environmental benefits.

2.6 CONCLUSIONS

Lake catchment nutrient loads in this study were higher than many of those reported elsewhere in New Zealand and internationally, indicating the magnitude of the nutrient problem faced by water quality managers of Waikato's shallow peat lakes. Calculating DAY and DIL for each subcatchment enabled simplification of the complex variability in nutrient exports within the lake catchments. This information can assist water managers to prioritise and

implement more targeted nutrient management, for example where N or P 'hotspots' were identified. Furthermore, concerted efforts to reduce nutrient and sediment losses from small subcatchments are likely to be more effective than those focussed further downstream, as pollutant sources can be more easily identified and managed.

A wealth of knowledge exists to inform BMPs targeted at minimising losses of $\text{NO}_3\text{-N}$ and SS from catchments dominated by dairy production, however there is little information for the management of $\text{NH}_4\text{-N}$ and $\text{PO}_4\text{-P}$. Our research shows considerable quantities of these nutrient species are lost from cultivated peat soils compared with clay loam. Improving the management of peat soils to sustain agricultural productivity whilst minimising nutrient losses and drainage impacts is crucial. Revision and widespread implementation of existing guidelines detailing BMPs for peat farming is strongly recommended (Peters et al. 2008; WRC 2006). This study has identified some of the critical environmental drivers and soil biogeochemical variables influencing nutrient export to shallow peat lakes. Targeted and appropriate management of peat (deep and shallow), peaty loam and clay loam soils would assist with meeting environmental standards for nutrient loss limits from agricultural catchments, cognisant of soil type and land use intensity. Quantifying the influence of different mechanisms and underlying processes driving nutrient and sediment losses at fine spatial scales (farm or small subcatchment scale) will enable appropriate controls to be developed in relation to local hydrological and biogeochemical conditions. For example, reducing nutrient loads at their source, improving soil conditions to enhance nutrient attenuation and processing, and intercepting pollutants along connectivity pathways. In light of New Zealand's recent adoption of a National Policy Statement for Freshwater Management (MfE 2014) specific local knowledge of pollutant sources and transport pathways, as identified by our study, will be critical to achieving desired water quality outcomes.

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Chapter 3

3 CONSTRUCTED TREATMENT WETLAND DESIGN

CONSIDERATIONS TO MITIGATE DIFFUSE POLLUTION FROM INTENSIVE AGRICULTURAL LAND USE IN NEW ZEALAND PEAT LAKE CATCHMENTS

3.1 ABSTRACT

Constructed treatment wetlands (CTWs) have been implemented as mitigation tools to manage diffuse pollution from intensive agricultural catchments. This study investigates CTW efficacy and evaluates different predictors of performance in shallow Waikato (New Zealand) peat lake catchments, exploring morphological and environmental variables which influence treatment efficiency. Reductions in nitrogen (N), phosphorus (P) and suspended solids (SS) differed considerably across CTWs, driven by varying influent concentrations and dominant forms of N, P, and SS, as well as CTW morphologies. Generally, CTWs with larger areas and volumes improved removal performance of nitrate, total N and coarse sediments, while deeper CTWs more effectively reduced particulate N and volatile SS. Macrophytes improved removal of nitrate and P, whereas filtration outlets frequently increased ammonium. Greater accumulated sediment depths significantly reduced P removal efficiency, signifying the importance of CTW maintenance. Increasing the number of CTW modules generally improved performance, thus implementing individualised treatment-train concepts or sequential wetland modules is recommended.

3.2 INTRODUCTION

Constructed treatment wetlands (CTWs) have been used extensively for treatment of municipal and agricultural wastewater both internationally (Wang et al. 2018; Knight et al. 2000) and in New Zealand (Tanner et al. 2012; Rambags et al. 2016; Nguyen 2000). The eutrophication of surface waters and degradation of water quality due to diffuse agricultural pollution, particularly from intensive dairy production (MfE/StatsNZ 2017; Flávio et al. 2017; Davis et al. 2015; Carpenter et al. 1998), has led to widespread application of agricultural CTWs in Europe (Koskiaho et al. 2003; Blankenberg et al. 2015; Arheimer & Pers 2017), North America (Reddy et al. 2006; Mitsch et al. 2005; Díaz et al. 2012), Asia (Lu et al. 2009; Nakamura 2009; Maniquiz et al. 2011) and increasingly New Zealand (Tanner & Kadlec 2013; Tanner et al. 2005; Wilcock et al. 2012).

CTWs treating diffuse pollution have proved effective for reducing suspended solids (SS) (Dunne et al. 2012; Braskerud 2003), nitrogen (N) (Vymazal 2017; Lu et al. 2009), phosphorus (P) (Kadlec 2016; Johannesson et al. 2011), and faecal bacteria (Wilcock et al. 2012; Tanner et al. 1995a). However, several studies have reported divergent findings such as negative removal rates (Blankenberg et al. 2008; Hoffmann et al. 2012; Tanner et al. 1995b; Johannesson et al. 2017), generating some uncertainty regarding the application of CTWs in agricultural landscapes and confusion concerning design concepts. Discrepancies in agricultural CTW efficacy may be a result of methodological variability between studies, as well as the applicability and accuracy of chosen methods to represent CTW performance in an agricultural context.

Reliable and accurate estimates of treatment performance can be difficult to determine as CTWs are 'open' systems, greatly influenced by both external environmental drivers and internal biogeochemical processes. Different types of CTWs (e.g. surface-flow, horizontal and vertical sub-surface flow; Figure 3.1) with distinct applications and hydraulic regimes (e.g. stormwater, wastewater, diffuse pollution) also function differently. As such, universal parameters that predict treatment performance cannot be expected (Kadlec & Wallace 2008d). Moreover, agricultural surface-flow CTWs have 'unregulated flows' with, at

times, hydrological and pollutant loading rates similar to both continuous-flow wastewater systems and event-driven stormwater wetlands, presenting further difficulties when attempting to predict their pollutant removal performance.

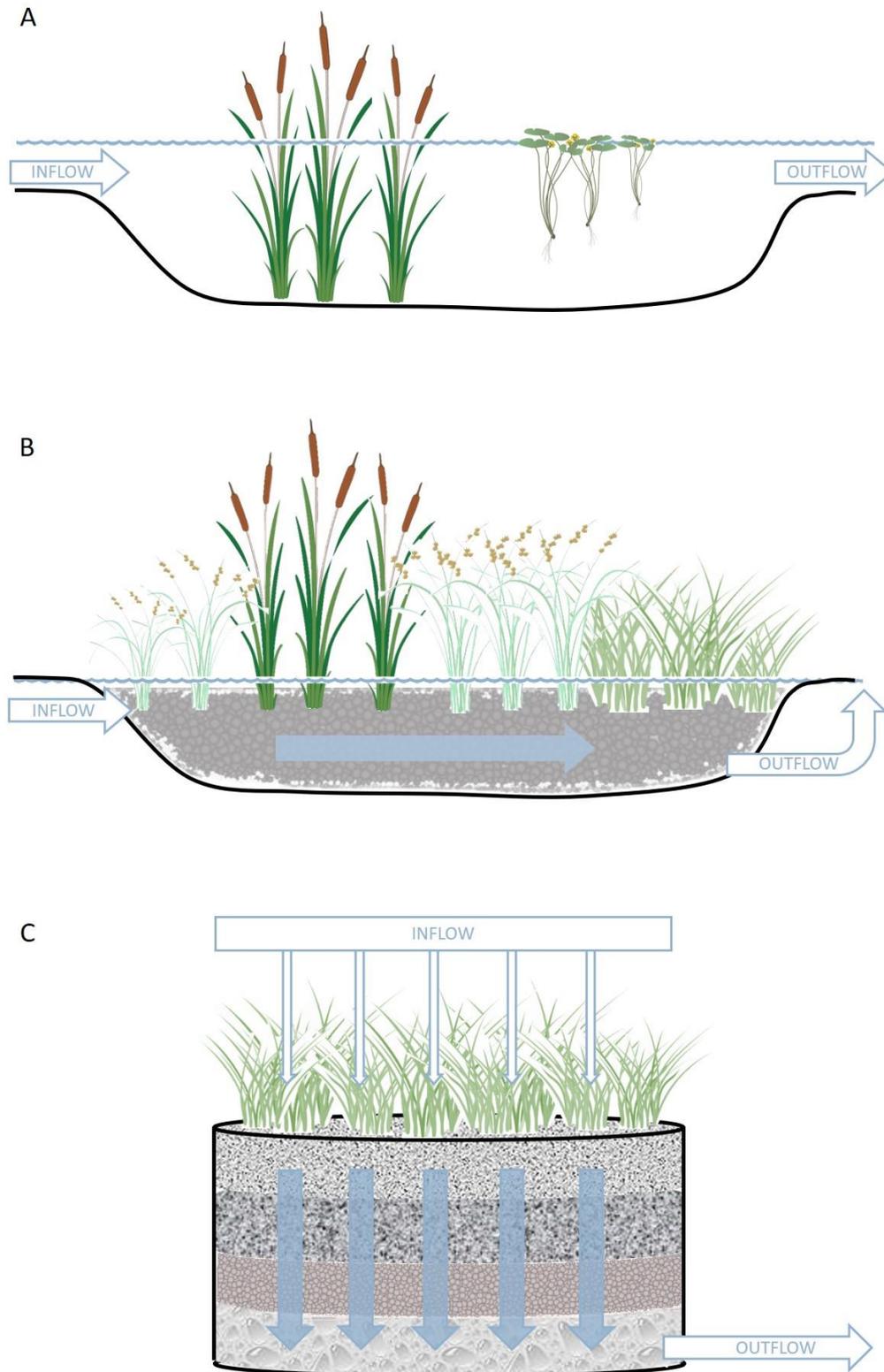


Figure 3.1 Simplified schematic cross-sections of the principal types of CTW including (A) free water surface, (B) horizontal subsurface flow, and (C) vertical flow systems

Kadlec & Wallace (2008d) describe treatment performance as being represented by i) central treatment tendency, and ii) anticipated variability away from that central tendency. Calculating meaningful trends in central treatment tendency requires large datasets collected over many annual cycles or via high-frequency sampling, which is often costly and resource intensive. Nevertheless, while useful for comparisons with similar studies, the transferability of treatment trends from one system to another is questionable given well recognised inter-system variability (Kadlec & Wallace 2008d). Alternatively, aggregated datasets, combining performance data from a number of CTWs, can be useful for exploring trends in treatment performance for a particular application (e.g. diffuse agricultural pollution) and assessing broader relationships with morphological and environmental variables (Carleton et al. 2001). Although aggregated datasets may not accurately predict the performance of an individual CTW, the trends in central treatment tendency for a given application can be useful in creating 'rules of thumb' for design considerations (Kadlec & Wallace 2008d). Design parameters frequently used to summarise CTW performance include wetland-to-catchment area ratio (WCAR), hydraulic retention time (HRT) and hydraulic loading rate (HLR) (Braskerud 2002a, 2002b; Kadlec & Wallace 2008b). Mean depth, area and volume are therefore important characteristics of CTWs that can also be related to removal performances of N, P and SS, as well as the presence of macrophytes (Brisson & Chazarenc 2009). The functional benefits of macrophytes in CTW can be attributed to physical and biological processes including, sedimentation of SS and associated P (through trapping particles and hindering resuspension), plant uptake of N and P, and facilitation of microbially mediated chemical processes such as denitrification and nutrient sorption by associated biofilms (Braskerud 2001; Rodrigo et al. 2018).

Numerous free water surface CTWs have been created in the lower Waikato region of New Zealand to improve the water quality of small watercourses draining to shallow peat lakes downstream. The lakes are eutrophic to hypertrophic as a result of agricultural land use in their catchments, primarily supporting intensive dairy production (Hamilton et al. 2010). Monitoring of the CTWs to determine contaminant removal performance is either lacking or has

given inconclusive results (Sukias et al. 2009). Consequently, their efficacy remains unclear. Landowners, community care groups, and local government authorities require information and guidance on agricultural CTW attenuation patterns and design parameters to instil confidence, inform future management decisions, and provide leverage for justification of future investment.

The main objectives of this research were (i) to determine the morphological predictors of CTW performance that best relate to positive removal rates of water quality constituents, including nutrients, suspended solids and accumulated volumes of sediment, (ii) to evaluate the effects of the number of CTW modules, outlet type, and macrophytes on removal performance, and (iii) to investigate physicochemical variables that may promote or constrain removal performance of CTWs treating diffuse agricultural pollution.

3.3 METHODS

3.3.1 Study site

This study was carried out in the lower Waikato region, New Zealand (37.8°S, 175.2°E), where there are a number of eutrophic shallow peat lakes located within catchments used almost entirely for intensive dairy production. Five peat lakes were selected as study sites where CTWs have been implemented as nutrient and sediment mitigation tools at the inflows to the lakes: Kainui, Kaituna, Komakorau, Koromatua and Serpentine North. These were the only Waikato peat lakes with CTWs at the time the research was conducted and the 26 CTWs were similar in age, constructed between 1999 and 2001. Lakes Kainui, Kaituna and Komakorau are within the Kainui restiad peat bog located in the Waikato District (Horsham Downs), north of Hamilton city (population c. 215,000; StatsNZ 2018). Lake Koromatua is on the edge of the Rukuhia restiad peat bog whilst Serpentine North is on the fringe of the Moanatuatua restiad peat bog, both in the Waipa District south of Hamilton city (Clarkson et al. 2004) (Figure 3.2). The topography of the peat lake catchments is flat to rolling lowland hills <70 m above sea level (mean ~ 34 m asl).

The central Waikato has a temperate maritime climate and rainfall is generally plentiful year-round (Chappell 2014). Annual rainfall across the lake catchments (approximately 35 km from north to south) ranges from 1100 to 1300 mm (Dravitzki & McGregor 2011). The region is characterised by relatively warm temperatures in the summer (December – March) with mean daily maximums between 20 and 25 °C, and relatively cold temperatures during winter (July – August) with mean daily maximums ranging from 0 – 8°C (Chappell 2014).

The location of the study lakes and positioning of CTWs within their catchments are provided in Figure 3.3. A summary of lake morphologies, trophic status, Trophic Level Index (TLI; Burns et al. 2000), water clarity, mean concentrations of chlorophyll *a* (Chl *a*), total phosphorus (TP) and total nitrogen (TN), number of major inflows, and number of CTWs and their site codes is given in Table 3.1.

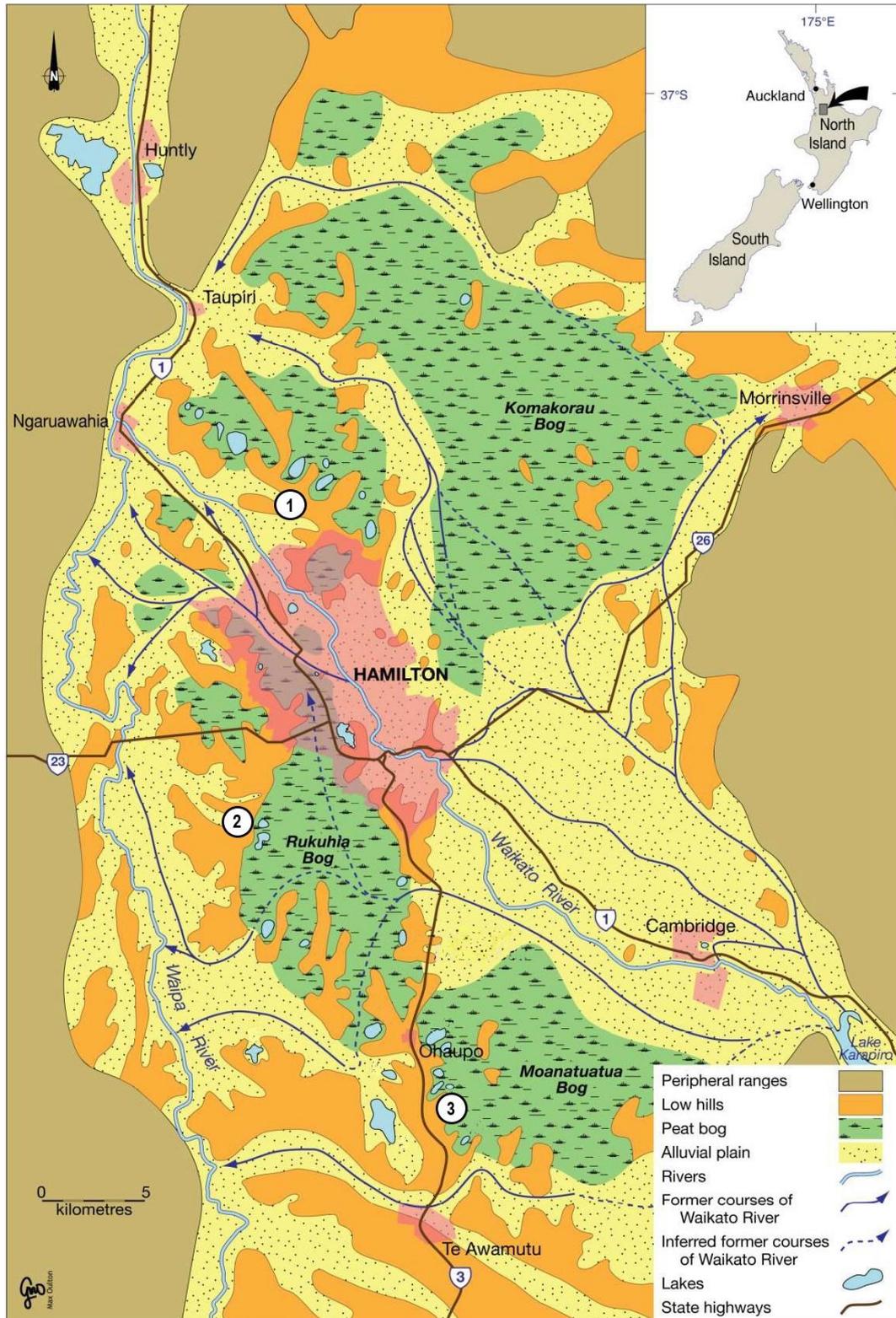


Figure 3.2 Modern landscape features of the Hamilton Basin showing antecedent hills partly buried by volcanogenic alluvium (Hinuera Formation) and post-Hinuera lakes and peat bogs (reproduced with permission from D.J. Lowe, 2010). The location of the study lakes is identified by: 1 (the Horsham Downs lakes), 2 (Lake Koroimata), and 3 (Serpentine Lakes).

Table 3.1 Characteristics of the study lakes including morphology, trophic status, Trophic Level Index (TLI), water clarity, and mean concentrations of chlorophyll a (Chl a), total phosphorus (TP) and total nitrogen (TN), number of major in-flows, number of CTWs and CTW codes. Source (excluding CTWs): (Hamilton et al. 2010).

Lake	Kainui	Kaituna	Komakorau	Koromatua	Serpentine North
Lake code	KN	KT	KO	KR	SN
Lake area (ha)	25	15	3	7	5
Max. depth (m)	6.7	1.3	1.1	0.8	3.8
Catchment area (ha)	260	589	619	67	163
Trophic state	Hypertrophic	Hypertrophic	Hypertrophic	Hypertrophic	Eutrophic
TLI	6.18	6.00	6.22	6.99	4.48
Secchi depth (m)	0.50	0.32	0.20	0.14	2.02
Chl a (mg m ⁻³)	45	6	9	32	4
TP (mg m ⁻³)	72	208	200	938	48
TN (mg m ⁻³)	3041	2509	2488	1492	570
Inflows (n)	10	10	3	3	2
CTWs (n)	9	10	3	3	1
CTW codes	KN1, KN2, KN3, KN4, KN5, KN6, KN7, KN8, KN9	KT1, KT2, KT3, KT4, KT5, KT6, KT7, KT8, KT9, KT10	KO1, KO2, KO3	KR1, KR2, KR3	SN1

All CTWs were comprised of at least one sedimentation pond ‘module’, with around half including a shallow wetland module planted with native species, and three with additional sedimentation pond modules. Each of the different CTW modules occurred in series. The inflows to the CTWs are surface-flow watercourses diverted from modified or artificial drainage networks, and the outflows are either via surface-flow (through drainage channels or culverts), or filtration (through vegetated riparian margins). Characteristics of the 26 CTWs are summarised in Table 3.2, including: outlet type, number of modules, site code, area, subcatchment drainage area, WCAR, mean depth, maximum depth, mean volume, mean HRT, and mean HLR.

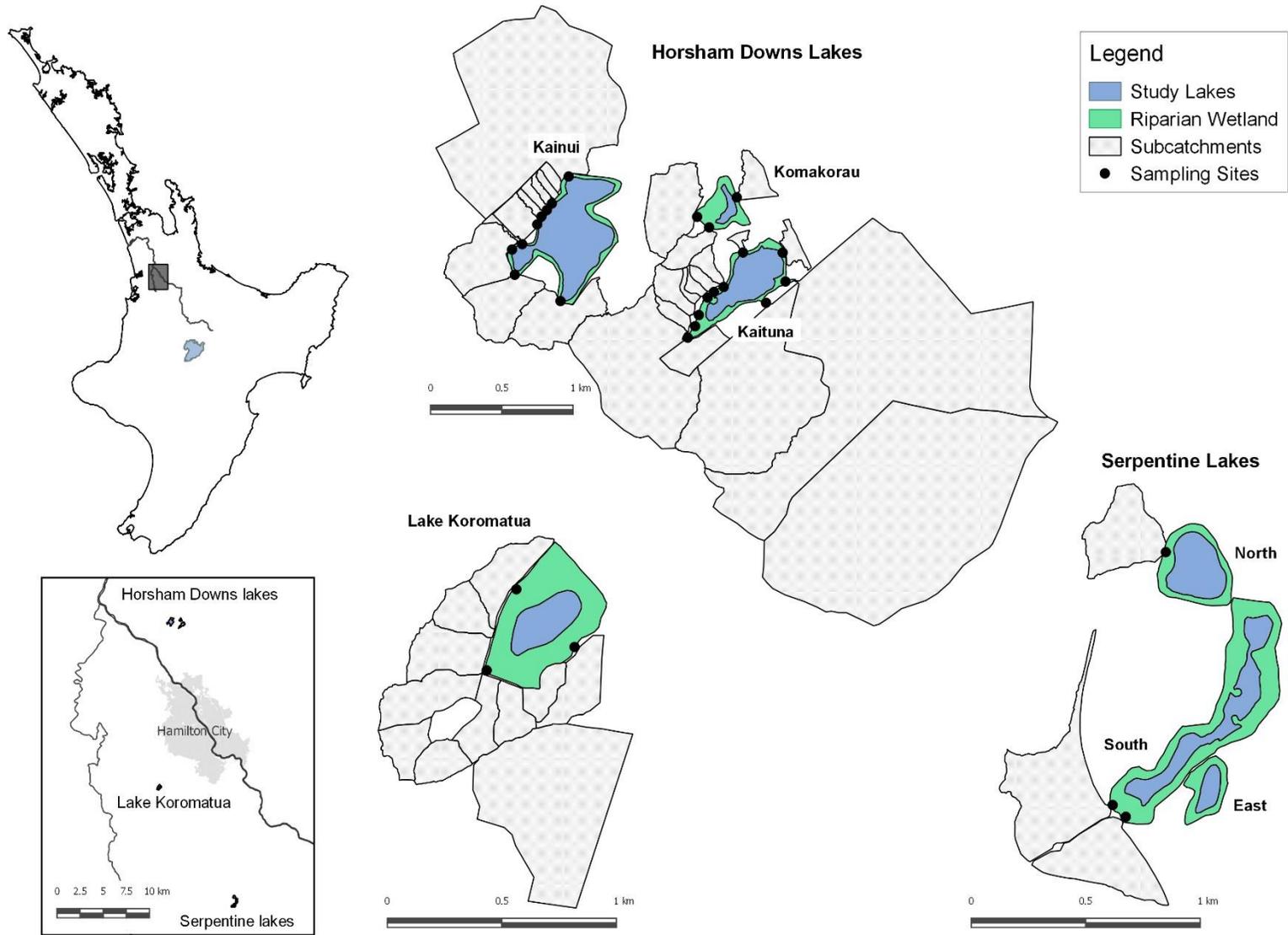


Figure 3.3 Location of studied peat lakes in the central Waikato (inset), North Island, New Zealand. Lake areas of open water and riparian margins were digitised from 2016 aerial photographs. Catchment boundaries were delineated from 2008 LiDAR (Light Detection and Ranging) data (source; Waikato Regional Council).

Table 3.2. Description of Constructed Treatment Wetlands (CTWs) and summary of morphological variables. Trap = sedimentation pond-module, WL = wetland-module, WCAR = Wetland to Catchment Area Ratio, HRT = hydraulic retention time, HLR = hydraulic loading rate.

Lake	Code	Modules	Outlet	CTW area (m ²)	Sub catchment (ha)	WCAR (%)	Mean depth (m)	Max. depth (m)	Mean volume (m ³)	Mean HRT (hr)	Mean HLR (m/d)
Kainui	KN1	Trap + WL + Trap	Surface flow	1950	17.9	1.09	1.1	1.8	2030	35.4	0.7
Kainui	KN2	Trap + WL	Surface flow	425	122.2	0.03	0.6	1.7	131	2.8	2.7
Kainui	KN3	Trap	Filtration	158	1.3	1.18	0.7	1.4	108	37.2	0.4
Kainui	KN4	Trap	Surface flow	54	1.6	0.33	0.2	0.8	10	4.4	1.0
Kainui	KN5	Trap	Surface flow	81	1.5	0.54	0.5	1.1	38	8.3	1.4
Kainui	KN6	Trap + WL	Surface flow	209	5.8	0.36	0.5	0.9	96	1.0	11.3
Kainui	KN7	Trap	Surface flow	75	0.8	0.89	0.4	0.4	30	6.4	1.5
Kainui	KN8	Trap + WL	Surface flow	525	15.9	0.33	0.5	1.5	242	4.1	2.7
Kainui	KN9	Trap + WL	Surface flow	75	15.1	0.05	0.5	1.1	38	0.3	37.3
Kaituna	KT1	Trap + WL + Trap	Surface flow	185	77.9	0.02	0.7	1.6	126	0.2	89.3
Kaituna	KT2	Trap + WL + Trap	Surface flow	290	195.9	0.06	0.2	1.4	205	0.2	21.7
Kaituna	KT3	Trap	Filtration	1080	3.2	0.20	0.8	0.8	55	4.6	4.4
Kaituna	KT4	Trap	Filtration	65	1.0	0.07	0.4	0.8	2	0.2	57.7
Kaituna	KT5	Trap	Filtration	7	2.9	0.03	1.1	1.2	8	0.9	27.1
Kaituna	KT6	Trap + WL	Filtration	8	1.7	0.55	0.8	1.2	77	10.4	1.9
Kaituna	KT7	Trap	Filtration	95	1.3	0.16	1.3	1.7	27	2.3	13.2
Kaituna	KT8	Trap + WL	Filtration	21	5.4	0.10	0.5	0.8	31	1.6	8.9
Kaituna	KT9	Trap	Filtration	54	2.6	0.08	0.7	0.9	14	2.8	5.9
Kaituna	KT10	Trap + WL	Surface flow	20	66.8	0.04	1	1.6	290	1.6	15.2
Komakorau	KO1	Trap	Filtration	65	5.4	0.12	0.9	1.3	61	3.0	7.4
Komakorau	KO2	Trap	Filtration	140	18.7	0.07	0.9	1.5	129	0.7	31.5
Komakorau	KO3	Trap	Filtration	50	5.3	0.09	0.9	1.9	44	0.5	39.1
Koromatua	KR1	Trap	Surface flow	52	40.3	0.01	1.4	2.1	75	0.3	130.0
Koromatua	KR2	Trap + WL	Filtration	241	17.0	0.14	0.6	1.7	152	0.6	27.2
Koromatua	KR3	Trap	Surface flow	162	5.6	0.29	1	2.7	138	1.5	13.3
Serpentine Nth	SN1	Trap + WL	Filtration	320	9.3	0.34	0.2	0.4	73	1.0	5.3

3.3.2 Sampling

Agricultural runoff is typically episodic in nature and loading patterns are somewhat difficult to predict owing to variability of on-farm practices (Eivers et al. submitted). To account for the anticipated variability in nutrient and sediment runoff, all 26 CTWs were sampled across the 5 peat lake catchments during winter 2010 (June-July). Following preliminary investigations, 15 CTWs (KN1, KN2, KN3, KN5, KN6, KN8, KN9, KO3, KR1, KR2, KR3, KT1, KT2, KT10, and SN1) were retained for further sampling during the following summer (February), autumn (May), winter (July) and spring (October-November) in 2011. These CTWs were selected because their inflows were less prone to drying out. CTW KN2 had two outlets downstream of two separate shallow wetland modules; thus, two wetland and downstream samples were collected for this site, referred to as KN2a and KN2b.

Measurements were made of the physical attributes of inflows to each CTW, including channel width, wetted width, water depth, and velocity, which were subsequently used to calculate influent flow rate (Q) following Harding et al. (2009). CTW areas were determined using a hand-held global positioning system (GPS) unit (Garmin Ltd, Kansas, USA; GPS Unit type) and validated using field measurements made with a 5 m measuring staff and 30 m measuring tape. Five to ten measurements of water and sediment depth were made along a minimum of three transects across each CTW, from which mean water depth and accumulated sediment volume were calculated. Macrophyte presence/absence and percentage cover estimates were made concurrently. Depths for small CTWs (< 5 m width) were measured from the bank, whilst larger CTWs (> 5 m width) were measured by wading into the CTW or from a rowboat. Area and water depth measurements were used to calculate CTW volume.

Subcatchment drainage areas for each CTW were calculated using ArcMap 10.0 (ArcGIS, Environmental Systems Research Institute Inc, CA, USA). Subcatchments were delineated from a digital elevation model (DEM) created from Light Detection and Ranging (LiDAR) data provided by the Waikato Regional Council.

The WCAR is defined as the ratio of CTW area to subcatchment drainage area, expressed as a percentage (Strecker et al. 1992).

Hydraulic loading rate was calculated as $HLR = Q/A$, where Q is inflow (m^3d^{-1}) and A is CTW area (m^2). Hydraulic retention time was calculated as $HRT = V/Q$, where V is volume of the CTW containing water (m^3) (Kadlec & Wallace 2008e).

Water samples for analysis of nutrient and SS concentrations were collected using a 1 L measuring jug attached to a pole. Samples were collected immediately upstream (US) and downstream (DS) of each CTW, and within each sedimentation pond module (Trap) and wetland module (WL). Piezometers were installed to collect the 'downstream' samples from the CTWs with filtration outlets (KN3, four; KO3, six; KR2, eight; and SN1, six piezometers). Individual samples were processed separately, and mean values were used for statistical analyses. Water temperature, dissolved oxygen (DO), specific conductivity and pH were measured concurrently at each water sampling location with a YSI 6000 UPG Multi Parameter Sonde (Yellow Springs Instruments, Ohio, USA).

Water samples for total and filterable nutrients were collected in 50 mL centrifuge tubes (Greiner Bio1, Germany) and for SS in opaque 1 L bottles. Filterable nutrient samples were immediately syringe-filtered upon collection (Whatman GF/C 0.45 μm). All nutrient samples were then placed on ice in the field and were frozen upon return to the laboratory until processing. Water samples for suspended solids analyses were stored in the dark at 4 °C and processed in the laboratory within three days of collection. Downstream SS samples were not collected for CTWs with filtration outlets.

3.3.3 Sample analyses

Suspended solids samples were filtered in the laboratory through pre-combusted (550 °C for 2 h) and pre-weighed glass microfibre filters (Whatman GF/C 0.5 μm). Total suspended solid (TSS) concentrations were determined gravimetrically following drying (105 °C for a minimum of 8 h) and volatile suspended solid (VSS) concentrations were determined following subsequent ashing (550 °C for 4 h).

Non-volatile suspended solids (Non-VSS) were calculated from the difference between TSS and VSS.

Filtered nutrient samples were analysed for concentrations of ammonium ($\text{NH}_4\text{-N}$), phosphate ($\text{PO}_4\text{-P}$), nitrate + nitrite ($\text{NO}_3\text{-N} + \text{NO}_2\text{-N}$) and nitrite ($\text{NO}_2\text{-N}$). Concentrations of nitrate were determined by the difference between $\text{NO}_3\text{-N} + \text{NO}_2\text{-N}$ and $\text{NO}_2\text{-N}$. Analyses were carried out using an Aquakem 200 discrete analyser (Thermo Fisher) with standard colorimetric methods (APHA 2005). Limits of detection were $0.001 \text{ mg N L}^{-1}$ for $\text{NO}_2\text{-N}$, $\text{NO}_3\text{-N}$, $0.002 \text{ mg N L}^{-1}$ for $\text{NH}_4\text{-N}$ and $0.001 \text{ mg P L}^{-1}$ for $\text{PO}_4\text{-P}$. Total P and TN were determined following alkaline persulphate digestion (APHA 2005) and analysis for $\text{PO}_4\text{-P}$ and $\text{NO}_3\text{-N} + \text{NO}_2\text{-N}$, respectively, using a Lachat QuickChem[®] Flow Injection Analyser (FIA + 8000 Series, Zellweger Analytics, Inc.). A range of check standards were analysed concurrently with samples to confirm analytical detection limits. Concentrations of total particulate organic nitrogen (ORG-N) were calculated by subtracting the sum of $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$ and $\text{NO}_2\text{-N}$ from TN. Concentrations of total particulate phosphorus (PP) were calculated by subtracting $\text{PO}_4\text{-P}$ from TP. Phosphate is referred to herewith as dissolved reactive phosphate (DRP).

3.3.4 Statistical analyses

Data consisted of physicochemical variables (water temperature, DO, conductivity, pH, influent flow rate Q, sediment depth and macrophyte cover), morphological predictors of CTW performance (CTW area, CTW depth, CTW volume, WCAR, HLR, and HRT), and water quality constituents including nutrients ($\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, Org-N, TN, DRP, PP, TP), suspended solids (VSS and non-VSS) and accumulated volume of sediment. Nitrite was excluded from the analysis as values were $<0.001 \text{ mg N L}^{-1}$ for all samples, below the detection limit of the analyser. Samples for SS were not collected from filtration outlets and could not be collected from surface flow outlets on every sampling occasion due to low water levels; therefore, there are fewer data points ($n=40$) than those of nutrients ($n=87$).

To test the morphological predictors of CTW performance, percentage removal values ($\%\Delta$) were calculated for each nutrient species as well as for VSS and non-

VSS. Percentage removal values were calculated as $\% \Delta = 100 \times [(C_{US} - C_{DS}) \div C_{US}]$ where C_{US} is concentration (mg L^{-1}) upstream of the CTW and C_{DS} is concentration downstream. The volume of accumulated sediment was averaged across all seasons sampled.

3.3.4.1 Examining attenuation patterns

Nitrogen ($\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, Org-N), P (DRP, PP) and SS (VSS, non-VSS) attenuation was examined visually for the CTWs by plotting mean (of seasonal samples) concentrations for each sampling location within the CTW including upstream (US), within the sedimentation trap module (Trap), within the wetland module (WL), and downstream of the CTW (DS).

Upstream and downstream nutrient concentrations for all samples (from five seasons and all CTWs) were plotted on 1:1 scatterplots to examine attenuation patterns for nutrient species, VSS and non-VSS. Points below the diagonal line represent constituent reductions through the CTW, whilst points above the diagonal line represent constituent increases.

3.3.4.2 Assessing morphological predictors of CTW performance

Percentage removal data were strongly skewed by negative values and followed non-normal distributions that could not be normalised through transformations. Thus, non-parametric data analyses were subsequently completed using R statistical computing software (R Core Team 2017).

Spearman's rank correlations were run using the Hmisc package in R (Harrell & Dupont 2017) to investigate relationships between percentage removal of N ($\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, Org-N , TN), P (DRP, PP, TP) and SS (VSS, non-VSS), CTW morphological predictors of performance (CTW area, CTW depth, CTW volume, WCAR, HLR and HRT), and physico-chemical variables (water temperature, specific conductivity, DO, pH, influent flow rate Q, and sediment depth).

Kruskal-Wallis one-way ANOVAs on ranks were run using the R packages 'dplyr' (Wickham et al. 2017), 'FSA' (Ogle 2017), and 'lattice' (Sarkar 2008), to test for effects of the number of CTW modules (One, Two or Three), outlet type (surface-

flow and filtration), and macrophytes (present or absent) on removal percentages of N, P and SS.

Simple linear regression models were run using the R package 'rcompanion' (Mangiafico 2017) to investigate relationships between accumulated sediment volume and morphological predictors of CTW performance following transformations to improve normality: sediment volume (cube root), CTW area (log), CTW depth (not transformed), CTW volume (Freeman-Tukey), WCAR (log), HLR (log), and HRT (Freeman-Tukey).

Following preliminary analyses, relationships between downstream concentrations of N, P and SS, and CTW morphological predictors of performance and physico-chemical variables, were analysed to test for linear relationships otherwise concealed by the highly negative skew of the percentage removal values, given the limited relationships identified in previous analyses. Pearson's correlations were run using the 'psych' package in R (Revelle 2017) following data transformations to meet assumptions of normality where appropriate: CTW area (Freeman-Tukey), CTW depth (log), CTW volume (log), WCAR (log), HLR (Freeman-Tukey), HRT (Freeman-Tukey), NH₄-N (Freeman-Tukey), NO₃-N (log), Org-N (cube root), TN (log), DRP (Freeman-Tukey), PP (Freeman-Tukey), TP (Freeman-Tukey), VSS (cube root), and non-VSS (Freeman-Tukey). Upstream concentrations were included in the analyses to ensure significant correlations were specific to downstream concentrations only and therefore attributable to CTW.

The influence of macrophytes was further analysed to investigate confounding effects, which may have masked relationships between CTW morphological predictors of performance and percentage removal of N, P and SS. Positive percentage removal values of N, P and SS from surface flow CTWs with and without macrophytes were plotted separately with each of the CTW morphological predictors (area, depth, volume, WCAR, HRT and HLR), and linear models were subsequently run to test for significant relationships.

A difference in the relationship between WCAR and VSS percentage removal values for CTWs with and without macrophytes was explored further. A single-

factor ANOVA was run using the 'stats' package in R (R-Core-Team 2017) to test for the effect of macrophyte presence/absence on percentage removal of VSS for CTWs with WCAR values ≤ 0.05 after identification of an apparent threshold (WCAR = 0.05) influencing results.

3.4 RESULTS

3.4.1 Attenuation patterns

Generally, removal rates of P were positive, while N and SS removal rates were more variable. Influent concentrations of N varied by more than 6-fold among CTWs, with $\text{NO}_3\text{-N}$ the dominant form of N for two thirds of the CTWs (10) and particulate organic N (Org-N) dominant in the remaining five (Figure 3.4A). Slight trends in N reduction occurred in some CTWs (KN8, KN1, KT1, KR1, KR2), while N remained largely unaltered in others, or in some instances increased. Large increases in $\text{NH}_4\text{-N}$ occurred in a number of samples collected from the downstream piezometers (DS –piez) of CTWs with filtration outlets (Figure 3.4B). Influent concentrations of P varied by up to 29-fold due to very high concentrations of DRP for five CTWs in particular (KN5, KR1, KN3, KN2, KN6), with concentrations 2 to 5-fold greater than the mean ($0.30 \text{ mg DRP L}^{-1}$) (Figure 3.4D). Trends in P reduction were shown for a number of CTWs (KN3, SN1, KN5, KN2, KN6, KN1), whereas concentrations remained relatively constant in others, particularly CTWs with very low influent levels of P (Figure 3.4C-D). Total suspended solid concentrations were less variable than N and P, except for CTW KT2 (Figure 3.4E). Relatively large reductions in SS occurred in CTWs KR3 and KN7, while only slight reductions or increases occurred in the remaining 14 CTWs from which samples for SS were collected.

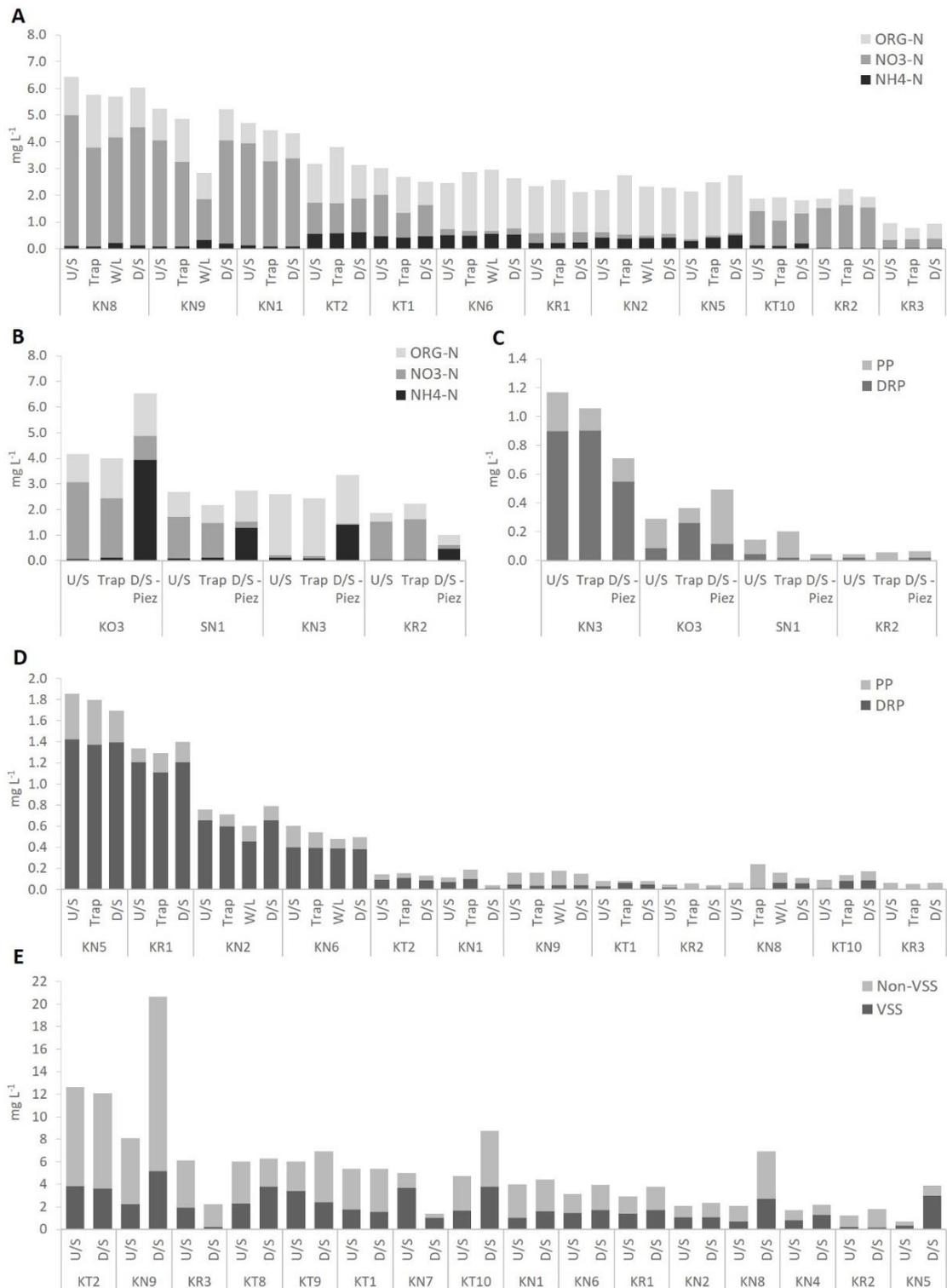


Figure 3.4. Mean concentrations of nitrogen (NH₄-N, NO₃-N, Org-N) plotted by outlet type (A) surface flow and (B) filtration; and phosphorus (DRP, PP) plotted by outlet type (C) filtration and (D) surface flow; and (E) total suspended solids (VSS and non-VSS). Sampling locations include upstream (US) of the CTW, within the sedimentation pond module (Trap), within the wetland module (W/L), and downstream of the CTW for surface flow outlets (DS), and filtration outlets (DS – Piez). CTWs are ordered from highest to lowest influent concentrations.

The scatterplots presented in Figure 3.5 show relatively consistent $\text{NH}_4\text{-N}$ reductions, except those samples collected from CTWs with filtration outlets. Nitrate was reduced for most samples, with the greatest reductions occurring from CTWs with filtration outlets. Attenuation patterns for Org-N, however, were highly variable with more increases (49) through the CTWs than reductions (38). Subsequently, attenuation of TN showed similar numbers of reductions and increases across the CTWs with no clear pattern evident. Both DRP and PP had comparable patterns of reductions, with fewer increases compared to N. Reductions in PP were generally greater than DRP (Figure 3.5). Neither DRP or PP attenuation appeared to be influenced by CTW outlet type. Reductions in TP were greater overall than increases (52 cf. 35). Volatile suspended solids showed slightly greater instances of reductions than increases, both for CTWs with and without macrophytes (Figure 3.5). Non-VSS samples, however, had fewer reductions and showed no obvious effect of macrophyte presence/absence.

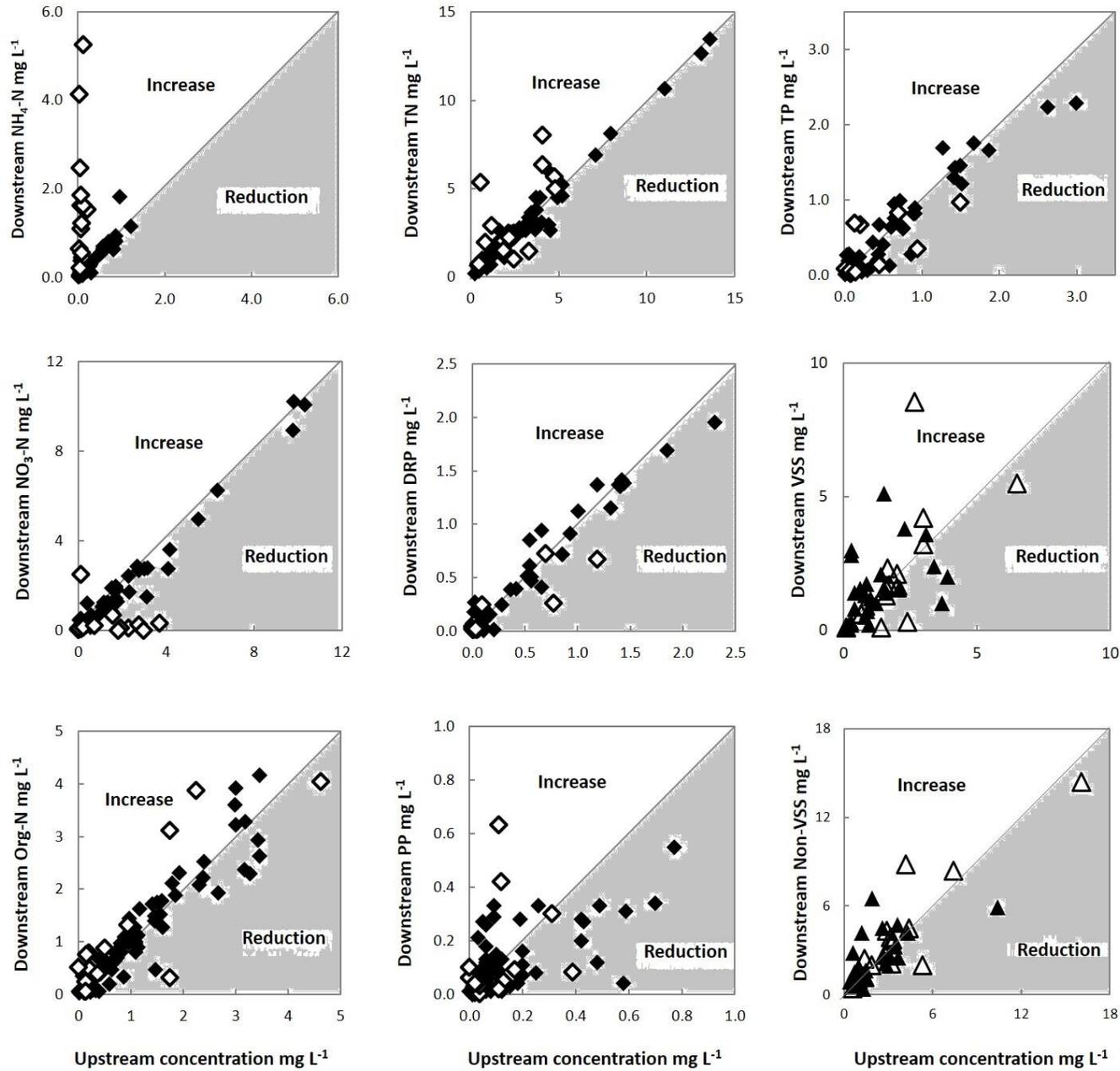


Figure 3.5. Upstream and downstream concentrations of nitrogen ($\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, Org-N, TN), phosphorus (DRP, PP, TP) and suspended solids (VSS, non-VSS) for all samples collected over five seasons, plotted on 1:1 axes. The diagonal line indicates a 1:1 relationship, that is equal upstream and downstream values. Points falling below the line represent reductions in constituent concentrations whilst points above indicate increases. Samples for N and P are plotted according to CTW outlet type (surface flow and filtration), and samples for VSS and non-VSS are plotted according to the presence/absence of macrophytes.

3.4.2 Morphological predictors of CTW performance

Percentage removal values for $\text{NH}_4\text{-N}$ were not correlated with CTW morphological predictors of performance (Table 3.3) while removal rates of $\text{NO}_3\text{-N}$ were very significantly positively correlated with CTW area, and TN with CTW volume. Percentage removal values of Org-N and VSS were not correlated with CTW morphological predictors of performance, however downstream concentrations were both significantly negatively correlated with CTW depth (Table 3.3). Neither percentage removal rates nor downstream concentrations of non-VSS were correlated with CTW morphological predictors of performance (Table 3.3). Percentage removal rates of DRP, PP and TP had significant negative correlations with HLR, while TP was also significantly positively correlated with HRT. Correlations between downstream concentrations of DRP, PP and TP with CTW depth, area and volume were deemed irrelevant given the same relationships occurred with equivalent upstream concentrations (Table 3.3). Accumulated sediment volume had highly significant, positive linear relationships with CTW area ($R^2=0.62$, $P<0.001$) and CTW volume ($R^2=0.49$, $P<0.001$), however no relationships were evident with CTW depth, WCAR, HRT or HLR.

Table 3.3. Summary of relationships between CTW morphological predictors of performance and percentage removal (%Δ), upstream (US) and downstream (DS) concentrations of nitrogen (NH₄-N, NO₃-N, Org-N, TN), phosphorus (DRP, PP, TP) and suspended solids (VSS, non-VSS). Positive percentage values are indicative of a reduction in nutrient concentrations from upstream to downstream of the CTW, whilst negative values indicate an increase. The rho coefficient is given for Spearman's rank correlations (%Δ) and r for Pearson's correlations (US and DS). Significant correlations are in bold, and statistical significance is indicated by *, ** and * for P-values <0.05, <0.01, and <0.001, respectively.**

Nutrient	Variable	% Δ rho	P	U/S r	P	D/S r	P	Nutrient	Variable	% Δ rho	P	U/S r	P	D/S r	P
NH4-N	CTW Area	0.14		0.16		-0.01		DRP	CTW Area	0.02		0.00		-0.02	
	CTW depth	0.04		-0.24		-0.15			CTW depth	0.01		-0.29	**	-0.25	*
	CTW volume	0.14		0.03		-0.08			CTW volume	-0.01		-0.13		-0.13	
	WCAR	-0.05		-0.23		-0.09			WCAR	0.17		-0.07		-0.15	
	HRT	0.05		-0.12		-0.15			HRT	0.20		0.16		0.04	
	HLR	-0.05		0.06		0.11			HLR	-0.21	*	-0.23	*	-0.10	
NO3-N	CTW Area	0.26	**	0.23		0.12		PP	CTW Area	0.08		-0.30	***	-0.32	***
	CTW depth	-0.20		0.13		0.15			CTW depth	-0.19		-0.23	*	-0.05	
	CTW volume	0.07		0.26		0.17			CTW volume	-0.05		-0.38	***	-0.31	***
	WCAR	0.05		-0.13		-0.17			WCAR	0.15		0.01		-0.08	
	HRT	0.08		-0.37		-0.30			HRT	0.20		0.02		-0.07	
	HLR	-0.12		0.39		0.32			HLR	-0.22	*	-0.08		0.05	
Org-N	CTW Area	-0.14		-0.03		0.05		VSS	CTW Area	-0.23		-0.07		0.03	
	CTW depth	0.11		-0.20		-0.30	***		CTW depth	0.20		-0.18		-0.33	*
	CTW volume	-0.03		-0.12		-0.09			CTW volume	-0.07		-0.15		-0.11	
	WCAR	-0.18		-0.01		0.06			WCAR	-0.18		-0.14		-0.01	
	HRT	-0.13		0.16		0.17			HRT	-0.16		-0.11		0.00	
	HLR	0.15		-0.21		-0.24			HLR	0.19		0.02		-0.03	
TN	CTW Area	0.20		0.17		0.08		Non-VSS	CTW Area	-0.15		0.15		0.17	
	CTW depth	0.09		-0.04		-0.08			CTW depth	0.28		0.21		0.02	
	CTW volume	0.25	*	0.13		0.03			CTW volume	0.07		0.22		0.18	
	WCAR	-0.06		-0.02		-0.02			WCAR	0.04		-0.08		-0.07	
	HRT	-0.02		-0.12		-0.11			HRT	0.08		0.10		0.06	
	HLR	0.04		0.09		0.07			HLR	-0.01		0.09		0.14	
TP	CTW Area	0.00		-0.19		-0.20									
	CTW depth	-0.19		-0.29	**	-0.15									
	CTW volume	-0.13		-0.31	***	-0.24	*								
	WCAR	0.15		-0.10		-0.19									
	HRT	0.21	*	0.09		0.04									
	HLR	-0.24	*	-0.17		-0.07									

The number of CTW modules had a significant effect on the percentage removal values of NH₄-N, NO₃-N and TN (Table 3.4). Filtration outlet type had a significant positive effect on percentage removal of TP, while in contrast, it had a significant negative effect on removal rates of NH₄-N (Table 3.4), supporting patterns observed in Figure 3.5. The presence of macrophytes significantly affected percentage removal rates of NO₃-N and TP (Table 3.4).

Table 3.4. The effects of the number of CTW modules (one, two or three), outlet type (surface or filtration), and the presence/absence of macrophytes on percentage removal (%Δ) of nitrogen (NH₄-N, NO₃-N, Org-N, TN), phosphorus (DRP, PP, TP) and suspended solids (VSS, non-VSS) are given. Positive percentage values are indicative of a reduction in nutrient concentration from upstream to downstream of the CTW, whilst negative values indicate an increase. Chi-squared values (χ^2) from Kruskal-Wallis tests for the effects of the number of modules, outlet type, and presence/absence of macrophytes on removal percentages are given with medians and standard deviations (SD). Significant correlations are in bold, and statistical significance is indicated by *, ** and * for P-values <0.05, <0.01, and <0.001, respectively.**

Nutrient	Variable	% Δ		% Δ		% Δ		χ^2	df	P value
		median	SD	median	SD	median	SD			
NH ₄ -N	# modules	One		Two		Three		6.35	2	0.042 *
		-13.6	2215	-16.1	594	2.65	29			
	Outlet	Surface Flow		Filtration						
		-6.20	113	-228	2423			14.0	1	0.000 ***
	Macrophytes	Present		Absent				1.52	1	0.217
		-13	1715	-5	136					
NO ₃ -N	# modules	One		Two		Three		13.25	2	0.001 ***
		-5.52	356	25.1	49	3.78	34			
	Outlet	Surface Flow		Filtration						
		8.89	53	3.53	400			0.01	1	0.934
	Macrophytes	Present		Absent				8.67	1	0.003 **
		13	273	-6	68					
Org-N	# modules	One		Two		Three		1.37	2	0.503
		-0.88	435	-9.66	5912	0.38	132			
	Outlet	Surface Flow		Filtration						
		-7.56	316	-2.21	7602			0.03	1	0.860
	Macrophytes	Present		Absent				0.95	1	0.331
		-9	5090	1	453					
TN	# modules	One		Two		Three		6.44	2	0.040 *
		-12.8	151	-0.79	31.7	5.66	25.1			
	Outlet	Surface Flow		Filtration						
		-1.07	24	-2.70	171			0.22	1	0.636
	Macrophytes	Present		Absent				0.00	1	0.946
		-1	116	-1	28					
DRP	# modules	One		Two		Three		0.26	2	0.880
		2.63	658	5.31	329	7.54	350			

Nutrient	Variable	% Δ		% Δ		% Δ		χ ²	df	P value
		median	SD	median	SD	median	SD			
	Outlet	Surface Flow		Filtration						
		0.23	557	28.51	109			3.42	1	0.064
	Macrophytes	Present		Absent						
		8	174	-5	763			2.43	1	0.119
PP	# modules	One		Two		Three				
		3.64	1232	8.17	389	-0.56	89	0.95	2	0.621
	Outlet	Surface Flow		Filtration						
	2.14	147	15.1	1427			0.32	1	0.572	
	Macrophytes	Present		Absent						
	11	966	-2	129			0.05	1	0.829	
TP	# modules	One		Two		Three				
		5.40	100	-1.29	184	10.95	64	1.81	2	0.404
	Outlet	Surface Flow		Filtration						
	0.00	148	26.9	142			3.90	1	0.048 *	
	Macrophytes	Present		Absent						
	7.95	113	-9.07	193			4.83	1	0.028 *	
VSS	# modules	One		Two		Three				
		0.00	295	-16.13	218	-5.00	52	1.20	2	0.548
	Outlet	Surface Flow		Filtration						
	n/a		n/a				n/a			
	Macrophytes	Present		Absent						
	-20.00	259	0.00	70			0.20	1	0.655	
Non-VSS	# modules	One		Two		Three				
		7.15	65	-23.81	193	0.00	10	2.57	2	0.276
	Outlet	Surface Flow		Filtration						
	n/a		n/a				n/a			
	Macrophytes	Present		Absent						
	0.00	182	-5.26	43			2.91	1	0.088	

3.4.3 Physico-chemical variables

No relationships were evident between water temperature and constituent removal rates. However, highly significant negative correlations occurred with concentrations of Org-N, TN, PP and TP (Table 3.5). Specific conductivity was significantly negatively correlated with removal percentages of non-VSS, and positively correlated with concentrations of $\text{NH}_4\text{-N}$, PP and TP. Dissolved oxygen concentration was negatively correlated with removal rates of Org-N, and positively with downstream concentrations of Org-N and VSS, as well as both upstream and downstream concentrations of TN (Table 3.5). Water pH was negatively correlated with removal percentages of Org-N, and highly negatively correlated with concentrations of $\text{NH}_4\text{-N}$, Org-N, DRP and TP. On the other hand, pH was highly positively correlated with concentrations of $\text{NO}_3\text{-N}$, TN and non-VSS. Influent flow rate was significantly negatively correlated with percentage removal and upstream concentrations of DRP, PP and TP, as well as concentrations of Org-N (Table 3.5). In contrast, flow rate had highly significant positive correlations with concentrations of $\text{NO}_3\text{-N}$ and upstream TN. Sediment depth had a very significant negative correlation with percentage removal of TP, and was positively correlated with downstream concentrations of $\text{NO}_3\text{-N}$.

Table 3.5. Summary of relationships between physico-chemical variables (water temperature, specific conductivity, DO, pH, influent flow rate Q, and sediment depth) and percentage removal (%Δ), upstream (US) and downstream (DS) concentrations of nitrogen (NH₄-N, NO₃-N, Org-N, TN), phosphorus (DRP, PP, TP) and suspended solids (VSS, non-VSS). The rho coefficient is given for Spearman's rank correlations (%Δ) and r for Pearson's correlations (US and DS). Significant correlations are in bold, and statistical significance is indicated by *, ** and * for P-values <0.05, <0.01, and <0.001, respectively.**

Nutrient	Variable	% Δ	P	U/S r	P	D/S r	P
		rho					
NH4	Temperature	-		-		-	
	Conductivity	-		0.23	*	0.29	**
	DO	-		-		-	
	pH	-		-0.46	***	-0.22	*
	Q	-		-		-	
	Macrophyte P/A	-		n/a		-	
	Sediment depth	-		n/a		-	
NO3	Temperature	-		-		-	
	Conductivity	-		-		-	
	DO	-		-		-	
	pH	-		0.46	***	0.31	**
	Q	-		0.51	***	0.40	***
	Macrophyte P/A	0.32	**	n/a		-0.22	*
	Sediment depth	-		n/a		0.25	*
ORG-N	Temperature	-		-0.48	***	-0.47	***
	Conductivity	-		-		-	
	DO	-0.24	*	-		0.21	*
	pH	-0.21	*	-0.37	***	-0.28	**
ORG-N	Q	-		-0.26	**	-0.22	*
	Macrophyte P/A	-		n/a		-	
	Sediment depth	-		n/a		-	
TN	Temperature	-		-0.41	***	-0.39	***
	Conductivity	-		-		-	
	DO	-		0.28	**	0.30	**
	pH	-		-		-	
	Q	-		0.22	*	-	
	Macrophyte P/A	-		n/a		-	
	Sediment depth	-		n/a		-	
DRP	Temperature	-		-		-	
	Conductivity	-		-		-	
	DO	-		-		-	
	pH	-		-0.57	***	-0.62	***
	Q	-0.25	*	-0.24	**	-	
	Macrophyte P/A	-		n/a		-	
	Sediment depth	-		n/a		-	
PP	Temperature	-		-0.24	*	-0.35	***
	Conductivity	-		0.40	***	0.34	***
	DO	-		-		-	
	pH	-		-		-	

Nutrient	Variable	% Δ <i>rho</i>	<i>P</i>	U/S <i>r</i>	<i>P</i>	D/S <i>r</i>	<i>P</i>
	Q	-0.21	*	-0.25	*	-	
	Macrophyte P/A	-		n/a		-	
	Sediment depth	-		n/a		-	
TP	Temperature	-		-0.23	*	-	
	Conductivity	-		0.28	**	0.27	**
	DO	-		-		-	
	pH	-		-0.50	***	-0.49	***
	Q	-0.28	**	-0.28	**	-	
	Macrophyte P/A	0.24	*	n/a		-	
	Sediment depth	-0.29	**	n/a		-	
VSS	Temperature	-		-		-	
	Conductivity	-		-		-	
	DO	-		-		-0.31	*
	pH	-		-		-	
	Q	-		-		-	
	Macrophyte P/A	-		n/a		-	
	Sediment depth	-		n/a		-	
Non-VSS	Temperature	-		-		-	
	Conductivity	-0.32	*	-		-	
	DO	-		-		-	
	pH	-		0.42	**	0.42	**
	Q	-		-		-	
	Macrophyte P/A	-		n/a		-	
	Sediment depth	-		n/a		-	

In CTWs with macrophytes present, significant positive relationships were evident between removal rates of NH₄-N and DRP and CTW depth, and VSS with HRT (Table SM 3.1). For CTWs without macrophytes, removal rates of NH₄-N had significant positive relationships with CTW area, volume, WCAR and HRT. Removal rates of Org-N were also positively correlated with CTW volume, VSS with WCAR and HRT, and non-VSS with CTW depth and HRT (Table SM 3.1).

Collectively, CTWs with WCAR <0.05 and macrophytes present had significantly greater removal rates of VSS than those with macrophytes absent (mean 38% and 7% respectively, F-stat=7.27, *P*<0.05). Macrophyte cover itself had highly significant negative correlations with CTW depth (*rho*=-0.41, *P*<0.001), CTW volume (*rho*=-0.25, *P*<0.01), flow rate (*rho*=-0.38, *P*<0.001), and sediment depth (*rho*=-0.26, *P*<0.01).

3.5 DISCUSSION

The efficacy of CTWs treating diffuse agricultural pollution is influenced by three key elements: CTW morphology, internal cycling, and the composition of influent constituents. The results of our study demonstrate the complexity of interplay between these elements and highlight important relationships that CTW design and maintenance regimes must take into consideration to achieve more effective attenuation of N, P and SS.

3.5.1 Morphological predictors of CTW performance

Numerous and varied relationships were evident between different morphological predictors of CTW performance and removal rates of various forms of N, P and SS. Increasing CTW area was associated with significantly greater volumes of accumulated sediment and NO₃-N reduction, supporting relationships described by others in New Zealand (Tanner et al. 2010) and internationally (Carleton et al. 2001; Jayasooriya et al. 2016). More effective attenuation of TN and settled sediment similarly occurred with increasing CTW volume. Although no obvious relationships were apparent between CTW morphologies and removal rates of Org-N and VSS, lower concentrations occurred downstream of CTWs which had greater depths, possibly due to particulate settling of plankton and organic suspended matter (Kadlec & Wallace 2008f). Removal performance of TP improved with longer HRT, likely assisted by processes of PP sedimentation, as well as biological uptake, chemical precipitation and adsorption of DRP (Kadlec & Wallace 2008c). Moreover, these processes appeared impeded in CTWs with elevated HLR where removal rates of DRP, PP and TP declined.

Superimposed upon the linear relationships between CTW morphology and removal performances were categorical influences pertaining to the number of CTW modules, outlet type, and the presence or absence of macrophytes. Greater reductions in NO₃-N occurred in CTWs with two modules, while NH₄-N removal performance was improved with three modules, as was TN generally. Creating CTWs with two or more modules is therefore likely to improve overall treatment efficiency, a finding consistent with the concept of constructed wetland 'treatment

trains' reported by Greenway (2005) and others (Jayasooriya et al. 2016; Pier et al. 2015; Vymazal 2011b). The absence of relationships between the number of CTW modules and removal performance of DRP, PP and TP may be explained by the confounding effects of internal physico-chemical cycling; for example, the highly significant negative relationship between sediment depth and removal rates of TP, as well as additional morphological and biotic variables such as macrophytes and CTW outlet type.

3.5.2 Internal nutrient cycling

CTWs with filtration outlets had significantly greater removal rates of TP compared to surface flow outlets, while $\text{NH}_4\text{-N}$ concentrations increased with approximately commensurate $\text{NO}_3\text{-N}$ reductions. The increases in $\text{NH}_4\text{-N}$ may be related to dissimilatory $\text{NO}_3\text{-N}$ reduction to ammonium (DNRA), a bacterial-mediated heterotrophic process occurring in anaerobic soils whereby $\text{NO}_3\text{-N}$ is converted to $\text{NH}_4\text{-N}$ (Stouthamer 1988). While denitrification has been previously considered the primary reductive process of $\text{NO}_3\text{-N}$ in freshwater environments (Scott et al. 2008; Sgouridis et al. 2011; Seitzinger 1988), the importance of DNRA for N-cycling has, in recent times, gained more attention (Zhu et al. 2017). Under certain biogeochemical conditions DNRA can dominate N-cycling in both natural and constructed wetland ecosystems. For example, DNRA processes prevail in the presence of specific plant species (Zhang et al. 2017), where the availability of organic carbon is high relative to $\text{NO}_3\text{-N}$ (Burgin & Hamilton 2007), and where sediment oxygen demand and reduced sulphur concentrations are high (Zhu et al. 2017; Scott et al. 2008). Morrissey et al. (2013) suggest DNRA bacterial communities may be poor competitors, proliferating in more oligotrophic conditions, while denitrifiers dominate under more eutrophic environments, particularly under high $\text{NO}_3\text{-N}$ concentrations. Bowden (1987) stated that sediments are the single largest pool of N in wetland ecosystems, and Braskerud (2002a) described decreases in N retention as CTWs age, which he attributed to the conversion of trapped Org-N to inorganic forms and their subsequent release. Correspondingly, the very large increases in $\text{NH}_4\text{-N}$ collected from many of the downstream piezometers, with concentrations often exceeding influent TN, may be attributed to mineralisation of

previously stored Org-N via ammonification from anaerobic sediments, where nitrification is inhibited, within the filtration outlets (Verhoeven 2009).

This existing research and our current findings highlight the need to consider the differential responses of DNRA and denitrifying communities to co-varying resources and geochemical gradients. However, in the absence of a clear consensus regarding successful and consistent methods of manipulating these reductive processes, we suggest CTWs should be designed to accommodate the occurrence of both. Excluding filtration outlets or modules from CTW designs would be imprudent given improved P attenuation, demonstrated by this study, as well as NO₃-N reduction via plant uptake and denitrification, described extensively elsewhere (Reddy et al. 1989; Tanner 1996; Matheson & Sukias 2010; Vymazal 2013). Alternatively, including an additional module downstream of the filtration-module, designed to target NH₄-N removal (described below), would improve the overall treatment efficiency of the CTW should DNRA, or other releasing processes, occur persistently or intermittently within the filtration-module. As previously mentioned, NH₄-N removal was significantly improved in CTWs with three modules (compared to one or two), providing support for this design concept.

3.5.3 Macrophytes

Constructed treatment wetlands with macrophytes had improved NO₃-N removal performance, indicative of biological uptake by plants and biofilms, as well as removal via denitrification (Vymazal 2013). Total P removal was also improved by macrophyte presence, likely through processes of biological uptake, sorption, precipitation and settling processes, conceivably enhanced by extended true HRTs (Tanner et al. 1995b). Greater plant species richness has also been found to increase P (Geng et al. 2017) and inorganic N removal efficiencies in CTW systems (Engelhardt & Ritchie 2001; Zhang et al. 2010).

Investigating the influence of macrophytes on CTW performance from relationships between morphological predictors and the removal rates of N, P and SS, with macrophytes either present or absent, revealed a number of previously unappreciated relationships (Table SM 3.1). Of particular interest were significant positive relationships between NH₄-N reduction and CTW area, WCAR, volume and

HRT in the absence of macrophytes, suggesting that surface flow CTWs with adequate areas of open water have better $\text{NH}_4\text{-N}$ removal performance than those without. $\text{NH}_4\text{-N}$ can be removed from solution to the atmosphere via ammonia (NH_3) volatilisation, a process governed by surface partial pressure of NH_3 (a function of temperature, pH, and concentration of total ammoniacal N) and the ability of gaseous NH_3 to move across the air-liquid interface (influenced by wind speed and the physical character of the air-water boundary) (Macintyre et al. 1995; VanderZaag et al. 2008). In a study comparing NH_3 volatilisation from marsh-pond-marsh CTWs treating swine waste in North Carolina, USA, Poach et al. (2004) found pond sections (11 x 20 x 0.75 m) exhibited significantly greater rates of NH_3 volatilisation than marsh sections of equivalent size (planted with *Typha latifolia* and *Schoenoplectus americanus*), accounting for 54-79% of N removal at higher loads. Harper et al. (2004) and VanderZaag et al. (2008) report similar findings on NH_3 emissions from CTWs processing dairy wastewater. Hence, incorporating open-water pond sections or modules in agricultural CTW treatment-train designs, downstream of macrophyte dominated marsh or filtration modules, may increase N removal rates through sequentially reducing both $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$. The application of this concept to CTWs processing diffuse agricultural runoff requires *in situ* research in New Zealand, however, given NH_3 volatilisation requires specific physiochemical conditions including warm water temperatures ($> 20\text{ }^\circ\text{C}$), surface water pH > 6.5 , air movement above the water surface, and relatively high loads of ammoniacal-N (Poach et al. 2004).

In the absence of macrophytes, significant positive relationships between HRT and removal performance of VSS and non-VSS were also discovered (Table SM 3.1), indicating that, as anticipated, CTWs with longer HRTs had improved VSS and non-VSS reduction. Further scrutiny of the highly significant relationship between WCAR and VSS reductions for CTWs without macrophytes revealed a WCAR threshold of approximately 0.05, below which VSS reductions were substantially improved in the presence of macrophytes as compared to CTWs of similar size without macrophytes (mean removal rates 38 % and 7%, respectively). This suggests that macrophytes greatly increase VSS removal efficiencies in relatively small CTWs (WCAR < 0.05) and signifies their importance in CTW designs, particularly where available land is

restricted. In this study macrophyte cover had highly significant negative relationships with CTW depth, volume, flow rate and sediment depth, as plants thrived in CTWs with shallow areas and low flows, typical of plant communities in CTW environments (Tanner 1996; Vymazal 2011b).

When including macrophytes in agricultural CTW designs, rigorous selection of macrophyte species suitable to the site, precise design morphologies (e.g. area and depth profiles) appropriate to the growth habit of the selected plant species (e.g. submerged, emergent and/or floating), and meticulous weed management are strongly recommended to ensure plants successfully establish and provide the desired physical and biological functions (refer Tanner et al. 2006). Annual and short-lived perennial plants, including native and exotic species, can impair CTW performance, becoming a significant source of nutrients during senescence (Kröger et al. 2007); therefore, careful maintenance is required to avoid their establishment.

3.5.4 Influent constituent composition

It is critically important to make allowance for the inevitable variability in pollutant loads and composition when designing agricultural CTWs and anticipating differences in attenuation and removal performances. In this study both influent concentrations of N, P and SS and the predominant forms of nutrients varied extensively across CTWs, greatly influencing attenuation efficiencies. Kadlec & Wallace (2008a) attribute the variable attenuation efficiencies of CTWs treating diffuse agricultural pollution to their event-driven nature (cf. controlled hydraulic loads of wastewater systems), and design concepts that range from very high loadings, often intended for sediment attenuation, to comparatively low loadings intended for nutrient removal. Such extensive variability was not initially anticipated here given the similarity of soil types, topography, climatic conditions and intensive agricultural land use of the five peat lake catchments studied. However, the high degree of hydrological modification and variability between peat, peaty loam and clay loam dominated soil types at the subcatchment scale profoundly influenced localised diffuse pollution patterns, discernible from strong relationships with water pH and flow rate (Eivers et al. submitted). High

concentrations of $\text{NH}_4\text{-N}$, Org-N and DRP occurred where pH and flow were low, associated with low-gradient, peat-dominated subcatchments, whilst $\text{NO}_3\text{-N}$ and non-VSS concentrations were greatest with higher flow rates and more neutral pH , typical of rolling clay loam subcatchments.

Consideration of the key contaminants of concern for the target watercourse and/or downstream waterbody is crucial for effective CTW design. Different forms of nutrients and SS require different removal processes, thus understanding the specific form(s) of N, P and SS to be treated is imperative to ensure CTWs are designed accordingly. In the case of N for example, $\text{NO}_3\text{-N}$, the dominant form of N for more than half of our CTWs, is frequently reduced via denitrification and plant assimilation, processes which are enhanced in CTWs with shallow areas of dense macrophyte beds (Vymazal 2013). In contrast, $\text{NH}_4\text{-N}$ can be removed via NH_3 volatilisation, an important process occurring in areas of open water which is greatly reduced by the presence of emergent vegetation (Poach et al. 2004). Additionally, larger, deeper areas of open water enhance sedimentation of coarse particles more typical of non-VSS whilst shallow, macrophyte dominated areas can improve VSS attenuation (Jayasooriya et al. 2016).

The removal performance of CTWs follows the 'mass action rule', whereby the removal rate of a pollutant is greater at higher concentrations (Kadlec & Wallace 2008e). In our study, removal percentages of $\text{NH}_4\text{-N}$, Org-N , TN , PP and TP increased significantly with greater upstream concentrations, consistent with this rule (Table SM 3.2), as found by others (Lu et al. 2009; Harper et al. 2004; Vymazal 2007; Koskiaho et al. 2003). Conversely, no such relationships were apparent for VSS or non-VSS, which had comparatively low influent concentrations (median 1.4 and 2.25 mg L^{-1} , respectively). Accordingly, designing CTWs to target specific contaminants that occur at concentrations breaching water quality thresholds or limits, can improve removal performances and assist landowners, land-care groups and managers to achieve more specific water quality goals. For example, to reduce $\text{NO}_3\text{-N}$ concentrations to below 0.8 mg L^{-1} in subcatchment "A", and DRP concentrations to below 0.05 mg L^{-1} in "B", to achieve the water quality standards

stipulated by New Zealand's National Policy Statement for Freshwater Management (NPS-FM) (MfE 2014).

Finally, occurrences of negative removal rates in agricultural CTWs are not uncommon in the literature (Koskiaho et al. 2003; Tanner et al. 1995a; Carleton et al. 2001; Wilcock et al. 2012), whereby generally low influent nutrient concentrations increase through the system as natural biogeochemical processes seek equilibria at sediment-water and water-air interfaces, driving internal fluxes and nutrient cycling. Evidence from our research of both TN and TP increasing at low concentrations is apparent in Figure 3.5. Moreover, biological processes occurring in areas of open-water, such as nutrient assimilation with phytoplankton growth, can increase VSS and particulate organic nutrients, as evidenced in a number of the CTWs with surface-flow outlets in this study (e.g. Org-N in KN1, KN2, KN5; PP in KN2, KN1; and VSS in KN5, KN8, KN9, KT10; Figure 3.4). Higher downstream concentrations of Org-N also occurred concurrently with elevated DO (Table 3.5), indicative of photosynthetic activity.

3.6 CONCLUSIONS AND RECOMMENDATIONS

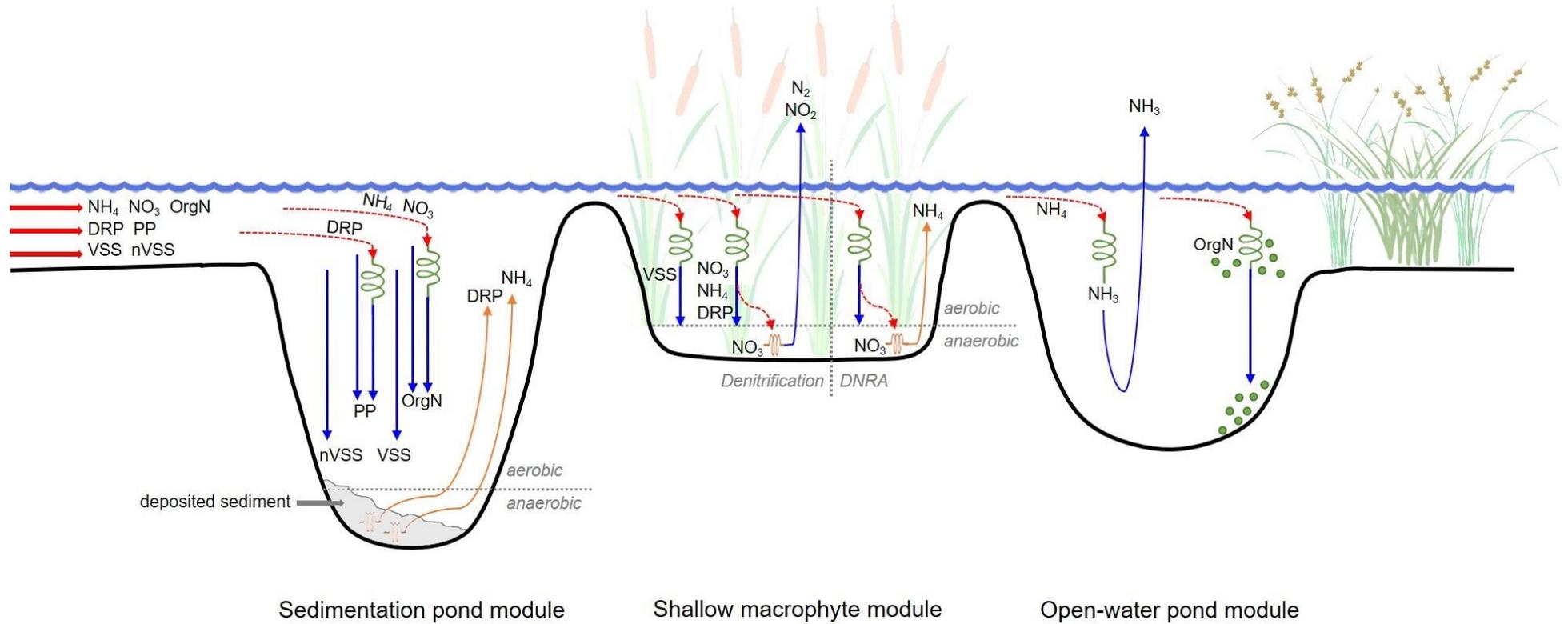
Generally, CTWs with larger areas and volumes improved removal performance of $\text{NO}_3\text{-N}$, TN and sediment capture, while deeper CTWs reduced particulate Org-N and VSS concentrations. In the absence of macrophytes, greater area, WCAR, volume and HRT improved $\text{NH}_4\text{-N}$ reductions, as did CTWs with three modules, plausibly because of NH_3 volatilisation in the third open-water sedimentation-module. Extended HRT increased removal efficiency of TP, and improved VSS and non-VSS removal performance in CTWs without macrophytes, while removal rates of DRP, PP and TP were consistently reduced in CTWs with high influent flows and HLR. CTWs with two modules improved $\text{NO}_3\text{-N}$ reduction, likely due to denitrification occurring within the second wetland-module as macrophytes generally improved $\text{NO}_3\text{-N}$ and TP reduction, and VSS in CTWs with WCARs <0.05 . Negative removal rates of TP occurred in CTWs with large volumes of accumulated sediment, indicative of possible P release from anaerobic sediments. Increases in $\text{NH}_4\text{-N}$ from filtration outlets were likely due to ammonium generation from anaerobic processes such as DNRA and ammonification of previously stored organic N from wetland sediments, combined with low rates of nitrification.

We recommend implementing treatment-train design concepts for CTWs processing diffuse agricultural pollution as encouraged by Wilcock et al. (2012), among others (Jayasooriya et al. 2016; Wang et al. 2018). To provide clarity for key stakeholders and water quality managers, Figure 3.6 summarises the primary processes and patterns of nutrient and sediment attenuation and cycling occurring throughout a CTW treatment-train. The design proposed comprises a deep sedimentation pond-module (≥ 1.5 m deep), shallow macrophyte module (~ 0.3 m deep), and open-water pond-module of moderate depth (~ 1 m) in series. For further guidance and considerations refer to Eivers (2016).

Our findings demonstrate the benefits of agricultural CTWs for reducing N, P and SS inputs to downstream waterbodies, while illuminating how dynamic and complex internal processes can obscure obvious positive effects. Consideration of nutrient cycling and retentive processes must be carefully factored in to CTW design and maintenance to ensure optimum pollutant removal performance. Additionally,

effective monitoring of CTWs treating diffuse agricultural pollution requires sampling plans designed to include consideration of high leaching seasons and random weather events, as well as appropriate sampling frequency and methodologies to measure key retentive processes, detection of critical forms of N, P, and SS, and calculation of appropriate and applicable predictors of CTW performance.

Figure 3.6 A schematic summary of the primary processes and patterns of attenuation and cycling of nitrogen ($\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, Org-N), phosphorus (DRP, PP) and suspended solids (VSS, non-VSS) occurring each module (sedimentation pond, shallow macrophyte, and open-water pond) throughout a CTW treatment-train. DNRA = dissimilatory nitrate reduction to ammonium.



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3.9 SUPPLEMENTARY MATERIAL

Table SM 3.1 Results from linear regressions between morphological predictors of CTW performance and percentage removal of nitrogen (NH₄-N, NO₃-N, Org-N, TN), phosphorus (DRP, PP, TP) and suspended solids (VSS, non-VSS) analysed separately for CTWs with macrophytes present and absent. R² value, F-statistic, and direction of association (positive +, negative -) are given for significant relationships only. Degree of significance is indicated by *, ** and * for P-values <0.05, <0.01, and <0.001, respectively.**

Nutrient	Variable	Macrophytes							
		Present				Absent			
		R ²	F-stat	+ / -	P	R ²	F-stat	+ / -	P
NH₄	CTW Area				n.s	0.82	26.61	+	**
	CTW depth	0.35	5.88	+	*				n.s
	CTW volume				n.s	0.57	9.16	+	*
	WCAR				n.s	0.74	19.84	+	**
	HRT				n.s	0.55	8.42	+	*
	HLR				n.s				n.s
NO₃	CTW Area				n.s				n.s
	CTW depth				n.s				n.s
	CTW volume				n.s				n.s
	WCAR				n.s				n.s
	HRT				n.s				n.s
	HLR				n.s				n.s
ORG-N	CTW Area				n.s				n.s
	CTW depth				n.s				n.s
	CTW volume				n.s	0.30	4.75	+	*
	WCAR				n.s				n.s
	HRT				n.s				n.s
	HLR				n.s				n.s
TN	CTW Area				n.s				n.s
	CTW depth				n.s				n.s
	CTW volume				n.s				n.s
	WCAR				n.s				n.s
	HRT				n.s				n.s
	HLR				n.s				n.s
DRP	CTW Area				n.s				n.s
	CTW depth	0.61	29.95	+	***				n.s
	CTW volume				n.s				n.s
	WCAR				n.s				n.s
	HRT				n.s				n.s
	HLR				n.s				n.s
PP	CTW Area				n.s				n.s
	CTW depth				n.s				n.s
	CTW volume				n.s				n.s
	WCAR				n.s				n.s
	HRT				n.s				n.s
	HLR				n.s				n.s
TP	CTW Area				n.s				n.s
	CTW depth				n.s				n.s
	CTW volume				n.s				n.s
	WCAR				n.s				n.s
	HRT				n.s				n.s

Nutrient	Variable	Macrophytes							
		Present				Absent			
		R ²	F-stat	+ / -	P	R ²	F-stat	+/-	P
	HLR				n.s				n.s
VSS	CTW Area				n.s				n.s
	CTW depth				n.s				n.s
	CTW volume				n.s				n.s
	WCAR				n.s	0.95	147.9	+	***
	HRT	0.37	5.75	+	*	0.62	11.56	+	**
	HLR				n.s	0.45	5.63	-	*
Non-VSS	CTW Area				n.s				n.s
	CTW depth				n.s	0.91	41.85	+	**
	CTW volume				n.s				n.s
	WCAR				n.s				n.s
	HRT				n.s	0.69	9.02	+	*
	HLR				n.s				n.s

Table SM 3.2 Results from Spearman's rank correlations between percentage removal (%Δ) and analogous upstream (US) concentrations of nitrogen (NH₄-N, NO₃-N, Org-N, TN), phosphorus (DRP, PP, TP) and suspended solids (VSS, non-VSS). The rho coefficient and P values are given; significant correlations are in bold, and statistical significance is indicated by *, ** and * for P-values <0.05, <0.01, and <0.001, respectively.**

% Δ	Constituent Concentration	rho	P	
NH ₄ -N	US NH ₄	0.35	0.001	***
NO ₃ -N	US NO ₃ -N	0.20	0.060	
Org-N	US Org-N	0.22	0.043	*
TN	US TN	0.29	0.006	**
DRP	US DRP	0.18	0.097	
PP	US PP	0.35	0.001	***
TP	US TP	0.40	0.000	***
VSS	US VSS	0.24	0.130	
Non-VSS	US Non-VSS	0.19	0.230	

Chapter 4

4 CONSTRUCTED TREATMENT WETLANDS PROVIDE HABITAT FOR ZOOPLANKTON COMMUNITIES IN AGRICULTURAL PEAT LAKE CATCHMENTS

4.1 ABSTRACT

Zooplankton are an essential component of healthy functioning lake and wetland ecosystems. Despite this, zooplankton communities within constructed treatment wetlands (CTWs) in agricultural landscapes remain unstudied. Taxa richness, total abundances and community composition were evaluated for zooplankton assemblages from three habitat types (lakes, CTWs and drainage ditches) within five intensive agricultural peat lake catchments in New Zealand. Relationships to water quality, physicochemical and biotic habitat variables were examined. Zooplankton were dominated by cladocerans, copepods, ostracods and rotifer taxa, representing a range of communities typical of lake and pond habitats. CTWs supported species otherwise absent from lake and drain habitats, increasing the overall biodiversity of the highly-modified peat lake catchments. Taxa richness of CTWs was higher than that of drains, and a few CTWs had greater diversity than several lakes. The morphological variables area and depth contributed to the greatest differences between habitats, followed by pH, inorganic nitrogen, conductivity and temperature. Correspondingly, zooplankton communities were significantly influenced by habitat area, depth and pH, as well as ammonium, phosphate, water temperature, dissolved oxygen, and macrophyte cover. Opportunities were explored for refining CTW designs to enhance zooplankton biodiversity and potentially improve treatment efficiency through increasing the complexity and diversity of CTW habitat niches.

4.2 INTRODUCTION

Constructed treatment wetlands (CTWs) are used globally as technologies to improve water quality (Kadlec & Knight 1996) and are effective in reducing levels of suspended solids, nitrogen and phosphorus, as well as organic matter and pathogens (Vymazal & Kröpfelová 2008; Vymazal 2007; Dunne et al. 2012; Kadlec 2010). CTWs are designed and created to emulate and enhance the natural processes and functions of wetland ecosystems involving wetland vegetation, soils, and microbial and aquatic communities (Mitsch & Gosselink 2007; Kadlec & Wallace 2008g). The efficacy of CTWs depends upon the functionality, resilience and ecological integrity of the wetland ecosystems ability to acclimate to changes in hydrology, pollutant loads and water chemistry. Ecological integrity integrates physical, chemical and biological integrity; the latter refers to an ecosystem's capacity to support and maintain a balanced, integrated, and adaptive biological system with a full range of elements, processes and biotic interactions (Karr 1996).

Biodiversity is an essential component of biological integrity. As part of this biodiversity, zooplankton communities provide a critical link for the flow of energy and nutrients between primary producers and higher trophic levels (Gray et al. 2012; Kattel 2012). Such assemblages are also important in maintaining the ecological integrity of shallow lakes and wetlands (Moss et al. 2003; Van den Broeck et al. 2015), and have been included in a number of biotic indices developed to evaluate the ecological quality and integrity of wetlands (Lougheed & Chow-Fraser 2002; Boix et al. 2005). Zooplankton communities of wastewater treatment ponds, including high rate algal ponds, have been well researched, primarily regarding removal or control of zooplankton that can limit algal production and reduce treatment efficacy (Montemezzani et al. 2015; Schlüter et al. 1987). Methods for manipulating zooplankton community composition and relative abundances within aquaculture ponds have also been extensively researched (Geiger 1983; Milstein et al. 2006; Piasecki et al. 2004). Yet, in contrast to studies of such intensively managed and controlled pond systems, no detailed studies exist of zooplankton within agricultural CTWs. Knowledge of

community composition, feeding guilds and habitat preferences of zooplankton within CTWs could contribute to improved treatment system design and help to optimise the assimilation of nutrients and reduction of pathogens.

Wetlands, including swamp, marsh, fen and peat bog ecosystems, as well as numerous peat lakes, were once key landscape features in the central Waikato region of New Zealand (Shearer 1997), supporting a diverse indigenous and endemic flora and fauna (Lowe & Green 1987). However, wetlands and lakes have declined in abundance, size and ecological integrity following extensive peatland drainage, cultivation and conversion to pasture beginning in the late 1800s (Hunt 2007). Wetland extent in the Waikato has been reduced by approximately 92 % from an estimated pre-human area of >356,000 ha to c. 28,000 ha. Wetland areas have been largely replaced by highly productive, intensive dairy farming (MfE/StatsNZ 2015a). Many of the remaining 31 peat lakes in the Waikato region have poor water quality and frequent cyanobacterial algal blooms, resulting from elevated nutrient and sediment levels associated with this change in land use (Hamilton et al. 2010), causing loss of much of their natural character and native biodiversity (Beard 2010; Shearer 1997).

Restoration actions are currently being implemented in several Waikato peat lake catchments to reduce nutrient and sediment runoff and improve lake water quality (Peters et al. 2008; WRC 2006). Methods include retiring areas of marginal pasture to create esplanade reserves, fencing and planting of riparian and wetland areas around the lake margins and along the banks of inlet waterways, and creation of free surface-flow CTWs to intercept inflows and improve water quality. Within such highly modified, intensive agricultural landscapes, CTWs may feasibly enhance the biodiversity of the lake catchment through provision of habitat isolated from toxic algal blooms and adverse environmental conditions ubiquitous within the lakes themselves.

In this study, we assessed the biodiversity of CTWs by examining zooplankton communities from different aquatic habitats in five shallow peat lake catchments with intensive dairy production as the predominant land use. Habitat types, comprising lakes and their associated CTWs and drainage ditches (heavily

modified and/or channelised artificial streams, referred to as drains herein), were studied as part of a greater body of research examining whether CTWs are effective tools for peat lake restoration. Zooplankton community composition was predicted to differ between habitat types, driven primarily by differences in habitat size (area and depth) and complexity (emergent macrophyte cover), water quality variables, and the presence or absence of fish. Based on these factors, we predicted that the biodiversity of zooplankton in CTWs would be higher than that of drains and lower than those of lake habitats.

Our key objectives were to:

- (i) compare zooplankton communities (taxa richness, abundances and community composition) between CTW, lake, and drain habitat types;
- (ii) examine environmental mechanisms driving differences in the zooplankton communities; and
- (iii) discuss opportunities for refining CTW designs to enhance zooplankton biodiversity.

4.3 METHODS

4.3.1 Study sites

The research was carried out in the central Waikato region, New Zealand (37.8°S, 175.2°E), where there are a number of shallow peat lakes located within agricultural catchments which are almost entirely used for intensive dairy production. Five shallow peat lakes were selected as study sites where CTWs have been implemented as mitigation tools for the inflows of several lakes: Kainui, Kaituna, Komakorau, Koromatua and Serpentine North. These were the only Waikato lakes with CTWs at the time the research was conducted and the CTWs were similar in age, constructed between 1999 and 2001. Lakes Kainui and Kaituna are within the Kainui restiad peat bog in the Waikato District, north of Hamilton city. Lake Koromatua is on the edge of the Rukuhia restiad peat bog, while Lake Serpentine is on the fringe of the Moanatuatua restiad peat bog, both in the Waipa District south of Hamilton city (Clarkson et al. 2004). The central Waikato has a temperate maritime climate with annual rainfall across the lake catchments (approximately 35 km from north to south) ranging from 1100 mm to 1300 mm (Dravitzki & McGregor 2011).

Three habitat types were sampled within each lake catchment and included five lakes, eight drains, and 27 CTWs (40 sampling locations in total). The location of the study lakes and positioning of habitat types within their catchments are provided in Figure 4.1. A summary of the lake morphologies, trophic status, Trophic Level Index (TLI; Burns et al. 2000) , water clarity, average concentrations of chlorophyll *a* (Chl *a*), total phosphorus (TP) and total nitrogen (TN), the number of major inflows to each lake, and the number and CTWs and their site codes is given in Table 4.1.

Table 4.1 Characteristics of the study lakes including morphology, trophic status, Trophic Level Index (TLI), water clarity, and average concentrations of chlorophyll *a* (Chl *a*), total phosphorus (TP) and total nitrogen (TN), number of major in-flows, number of CTWs and CTW codes. Source (excluding CTWs): Hamilton et al. (2010)

Lake	Kainui	Kaituna	Komakorau	Koromatua	Serpentine North
Lake code	KN	KT	KO	KR	SN
Lake area (ha)	25	15	3	7	5
Max. depth (m)	6.7	1.3	1.1	0.8	3.8
Catchment area (ha)	260	589	619	67	163
Trophic state	Hypertrophic	Hypertrophic	Hypertrophic	Hypertrophic	Eutrophic
TLI	6.18	6.00	6.22	6.99	4.48
Secchi depth (m)	0.50	0.32	0.20	0.14	2.02
Chl <i>a</i> (mg m ⁻³)	45	6	9	32	4
TP (mg m ⁻³)	72	208	200	938	48
TN (mg m ⁻³)	3041	2509	2488	1492	570
Inflows (<i>n</i>)	10	10	3	3	2
CTWs (<i>n</i>)	9	10	3	3	1
CTW codes	KN1, KN2, KN3, KN4, KN5, KN6, KN7, KN8, KN9	KT1, KT2, KT3, KT4, KT5, KT6, KT7, KT8, KT9, KT10	KO1, KO2, KO3	KR1, KR2, KR3	SN1

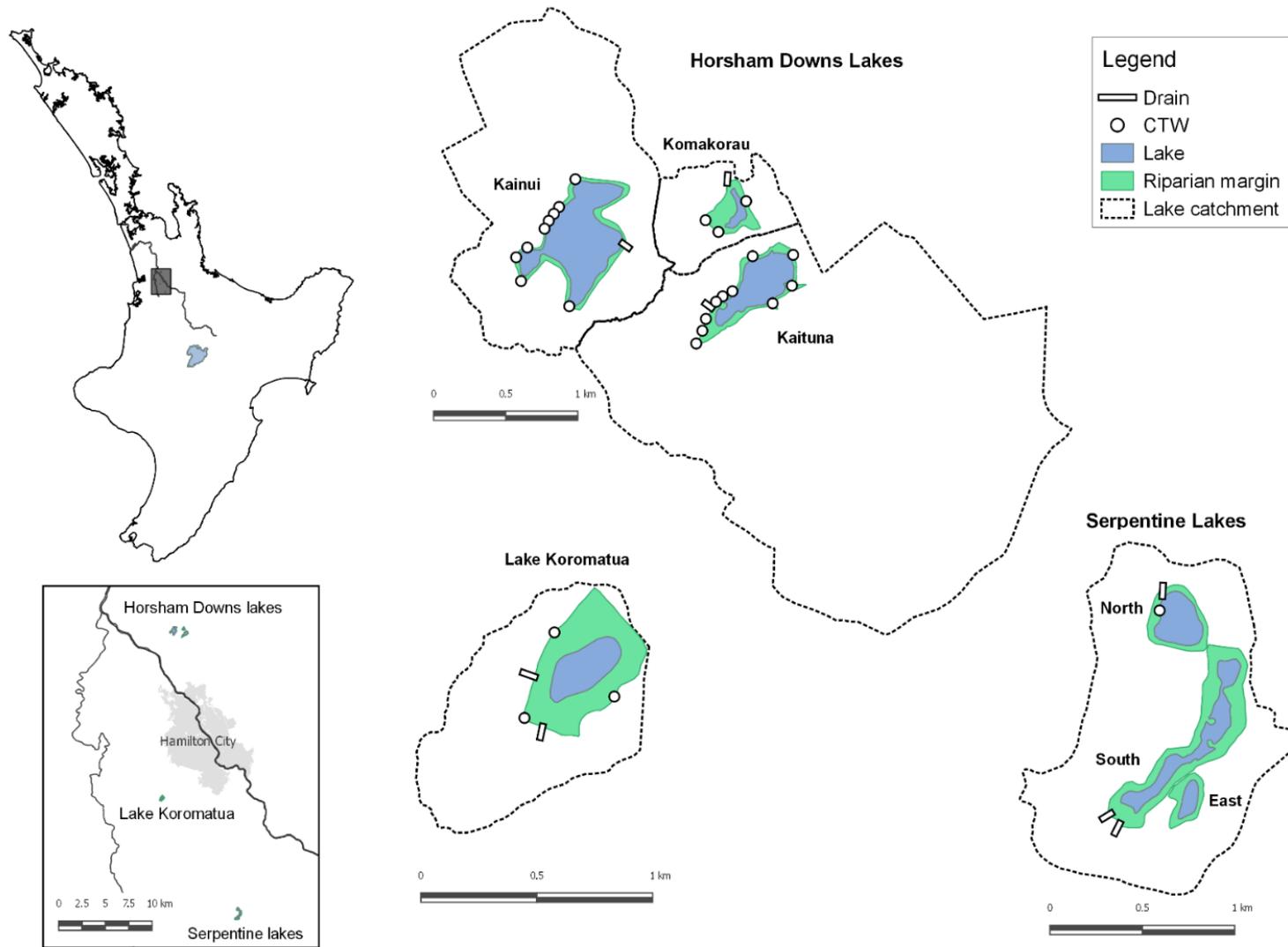


Figure 4.1 Location of studied peat lakes in the central Waikato (inset), North Island, New Zealand. Lake areas of open water and riparian margins were digitised from 2016 aerial photographs. Catchment boundaries were delineated from 2008 LiDAR (Light Detection and Ranging) data (source; Waikato Regional Council)

4.3.2 Field Sampling

Sites were sampled in late summer (2-8 February) 2011, following two weeks of persistent heavy rain (c. 215 - 235 mm across the study area), causing refilling of the CTWs after a dry spring and early summer (c. 125 – 135 mm over the preceding 12 weeks) (CliFlo 2017). We acknowledge that our sampling is a snapshot of the late-austral summer zooplankton community and environmental conditions, and that greater zooplankton diversity and variety of conditions would likely be encountered if seasonal or higher frequency sampling was undertaken. Single measurements of physicochemical variables were made at each site, including wetted area, depth, connectivity with the downstream lake (high = 3, medium = 2, low = 1, no connectivity = 0), as well as water temperature, dissolved oxygen concentration (DO), specific conductivity and pH, using a YSI 6000 UPG Multi Parameter Sonde (Yellow Springs Instruments, Ohio, USA). Area of the lakes was calculated from digitised aerial images (2016) and was measured manually in the field for CTWs and drains (over the 10 m reach surveyed). Sites with 'no connectivity' to the downstream lake occurred where the original drain outlet had been filled to create a 'filtration' outlet for the CTW. Water samples were collected for measurement of nutrients, Chl *a*, suspended solids and zooplankton, concurrently, from the middle of CTWs and drains from a depth of approximately 0.3 m using a 1 L measuring jug on a 2 m pole. Samples were collected from lake habitats at a similar depth, from platforms built by duck-hunters which extend out into the lake approximately 5 – 10 m from the shore.

Water samples for particulate and filterable nutrients were collected in 50 ml centrifuge tubes (Greiner Bio1, Germany) and for suspended solids in opaque 1 L bottles. Filterable nutrient samples were syringe-filtered (Whatman GF/C 0.45 µm) in the field and the filter paper retained and wrapped in aluminium foil for Chl *a* analysis. Samples were placed on ice after collection, with nutrient and Chl *a* samples frozen upon return to the laboratory. Water samples for suspended solids analyses were stored in the dark at 4 °C and processed in the laboratory within three days of collection.

Biotic variables measured included percentage cover of emergent macrophytes, visual presence/absence of iron flocculants, and observed presence/absence of fish. A minimum of two G-minnow traps were set per CTW and drain over four nights of trapping. Fish were identified to species level using descriptions from McDowall (2000). The presence/absence of fish for the lakes was determined from the literature, and percentage cover of emergent macrophytes estimated for the entire lake shore using aerial photographs. Iron flocculants were measured by the presence of rust-coloured particles collected on the filter paper retained for Chl *a* analyses.

Zooplankton samples were collected by pouring 3 to 10 L of water through a zooplankton net (40 µm mesh) and were immediately preserved in ethanol (> 50% final concentration) before subsequent identification and enumeration in the laboratory.

4.3.3 Sample analyses

Chlorophyll *a* samples were analysed using a calibrated fluorometer following acetone extraction (Arar & Collins 1997). Filtered nutrient samples were analysed for concentrations of ammonium (NH₄-N), phosphate (PO₄-P), nitrate + nitrite (NO₃-N + NO₂-N) and nitrite (NO₂-N). Concentrations of nitrate were determined by the difference between NO₃-N + NO₂-N and NO₂-N. Analyses were carried out using an Aquakem 200 discrete analyser (Thermo Fisher) with standard colorimetric methods (APHA 2005). Limits of detection were 0.001 mg N L⁻¹ for NO₂-N, NO₃-N, 0.002 mg N L⁻¹ for NH₄-N and 0.001 mg P L⁻¹ for PO₄-P. Total P and TN were determined following alkaline persulphate digestion (APHA 2005) and analysis for PO₄-P and NO₃-N + NO₂-N, respectively, using a Lachat QuickChem® Flow Injection Analyser (FIA + 8000 Series, Zellweger Analytics, Inc.). A range of check standards were analysed concurrently with samples to confirm analytical detection limits. Concentrations of total organic nitrogen (ORG-N) were calculated by subtracting the sum of NH₄-N, NO₃-N and NO₂-N from TN.

Suspended solids samples were filtered in the laboratory through pre-combusted (550 °C for 2 h) and pre-weighed glass microfibre filters (Whatman GF/C 0.5 µm). Total suspended solids (TSS) concentrations were determined gravimetrically

following drying (105 °C for a minimum of 8 h) and volatile suspended solid (VSS) concentrations were determined following subsequent ashing (550 °C for 4 h). Non-volatile suspended solids (Non-VSS) were calculated from the difference between TSS and VSS.

Zooplankton samples were enumerated in the laboratory using a dissecting microscope at c. 30x magnification until at least 300 individuals, or the whole sample, was counted. Identification was performed using a compound microscope to the lowest level practical, using standard taxonomic guides (Chapman et al. 2011; Shiel 1995).

4.3.4 Statistical analyses

Taxa richness for zooplankton was calculated as the total number of taxa present in each sample. Relative abundances of major zooplankton groups; cladocerans, copepods (including calanoids, cyclopoids and copepod nauplii), ostracods, rotifers and 'others' (including amphipods, dipterans, hemipterans, hydracarina and tardigrades) were calculated for each habitat type and compared using Kruskal-Wallis single factor analysis (Statistica Software version 8.0; Statsoft, Tulsa, OK, USA). The relative abundances of major zooplankton groups were first estimated for each site ($n = 40$) and then averaged across habitat types (lake $n = 5$, CTW $n = 27$, drain $n = 8$) to derive mean relative abundances. This method was chosen as it better incorporated the variability among the sites within each habitat type.

Multivariate analyses were undertaken using Primer 6 (version 6.1.15, Primer-E Ltd., Plymouth Marine Laboratory, Plymouth, U.K) with the PERMANOVA + add-in (version 1.0.5) to determine whether habitat differences existed between lakes, CTWs and drains (based on environmental variables), and to assess patterns of zooplankton community composition. The environmental variables included in analyses were physicochemical (mean depth, area, water temperature, DO, specific conductivity and pH), water quality-related (NH₄-N, NO₃-N, Org-N, PO₄-P, Non-VSS and VSS) and biotic (Chl a , percentage cover of emergent macrophytes, iron flocculant and fish presence/absence). Volume was excluded from the analyses due to high correlations with area and depth

measurements. All analyses of environmental variables were based on Euclidean distance matrices (Biondini et al. 1991) performed on $\log(x+1)$ transformed and standardised data (zero mean and unit variance). All analyses of zooplankton data were based on Bray-Curtis similarity matrices (Bray & Curtis 1957) and performed on square-root transformed data as recommended by Anderson et al. (2008).

Prior to undertaking multivariate analyses, a Pearson correlation was performed to identify any highly correlated physicochemical variables. Connectivity was excluded from this analysis as it was not applicable to lake samples, as well as TN and TP due to high correlations with Org-N and $\text{PO}_4\text{-P}$, respectively. Individual nutrient species were selected for inclusion as they had stronger relationships with zooplankton community composition than TN and TP alone.

The variation in environmental variables among habitats was analysed using a single factor (habitat type) permutational multivariate analysis of variance, PERMANOVA (Anderson et al. 2008; Anderson 2001a). A Type III PERMANOVA for unbalanced designs was performed and significance was determined by 9,999 unrestricted permutations of the raw data (Anderson 2001b; Anderson & ter Braak 2003). Pair-wise comparisons of group (habitat) means were completed in the case of a significant factor effect to assess between which pairs of habitat types significant differences occurred (McArdle & Anderson 2001). Variables contributing most to the variation among habitats were then identified using similarity percentage analysis (SIMPER), which calculates the average dissimilarity between all pairs of samples and assesses the relative dissimilarity contributed by each variable (Clarke 1993).

The variation in zooplankton community composition between habitat types was similarly evaluated using a single-factor Type III PERMANOVA with pair-wise comparisons of group (habitat) means completed to assess levels of significance. The taxa contributing most to differences in zooplankton community composition between habitat types were determined using the SIMPER procedure.

Non-parametric multifactor multiple regression, using the Distance-based Linear Modelling routine in Primer 6 (DistLM), was used to test for the influence of environmental habitat variables (physicochemical, water quality-related and biotic attributes) in structuring the variation in zooplankton community composition (Anderson et al. 2008). The DistLM procedure tests for significant differences in multivariate response variables to explanatory variables based on a selected distance-based measure in the form of a resemblance matrix (Anderson et al. 2008). The step-wise selection procedure based on 9,999 permutations was used to select and test habitat variables with an adjusted R^2 selection criterion to eliminate insignificant variables.

4.4 RESULTS

4.4.1 Zooplankton community composition

Seventy-three taxa were identified from the three habitat types, including 7 cladoceran, 3 copepod, and 55 rotifer species, ostracods and 8 macroinvertebrate taxa (amphipods, dipterans, hemipterans, hydracarinae and tardigrades). Total number of taxa recorded for each habitat type was greatest from CTWs (52), followed by lakes (40), and drains (20) (Table SM 4.1). Lake samples had the highest, although variable, mean taxon richness (15.2, SD=4.9) followed by CTWs (8.7, SD=3.8) and drains (6.8, SD=2.9; Figure 4.2A). Similarly, mean zooplankton total abundance was highest in lake samples (262 animals L^{-1}), followed by CTWs and then drain samples (70 and 46 animals L^{-1} , respectively; Figure 4.2E).

Differences in mean total abundances and richness of zooplankton were not statistically significant amongst habitat types due to high within-group variability. Lakes Komakorau and Koromatua had highest taxa richness (21 and 19, respectively), while the greatest richness from CTW habitats was recorded from KN1 (entering Lake Kainui) and KR2 (entering Lake Koromatua) (both 16). The CTW KT2, adjacent to Lake Kaituna, had the lowest richness with only cyclopoid copepodites and copepod nauplii recorded from the sample despite the CTW having high lake connectivity and a large area (1080 m^2).

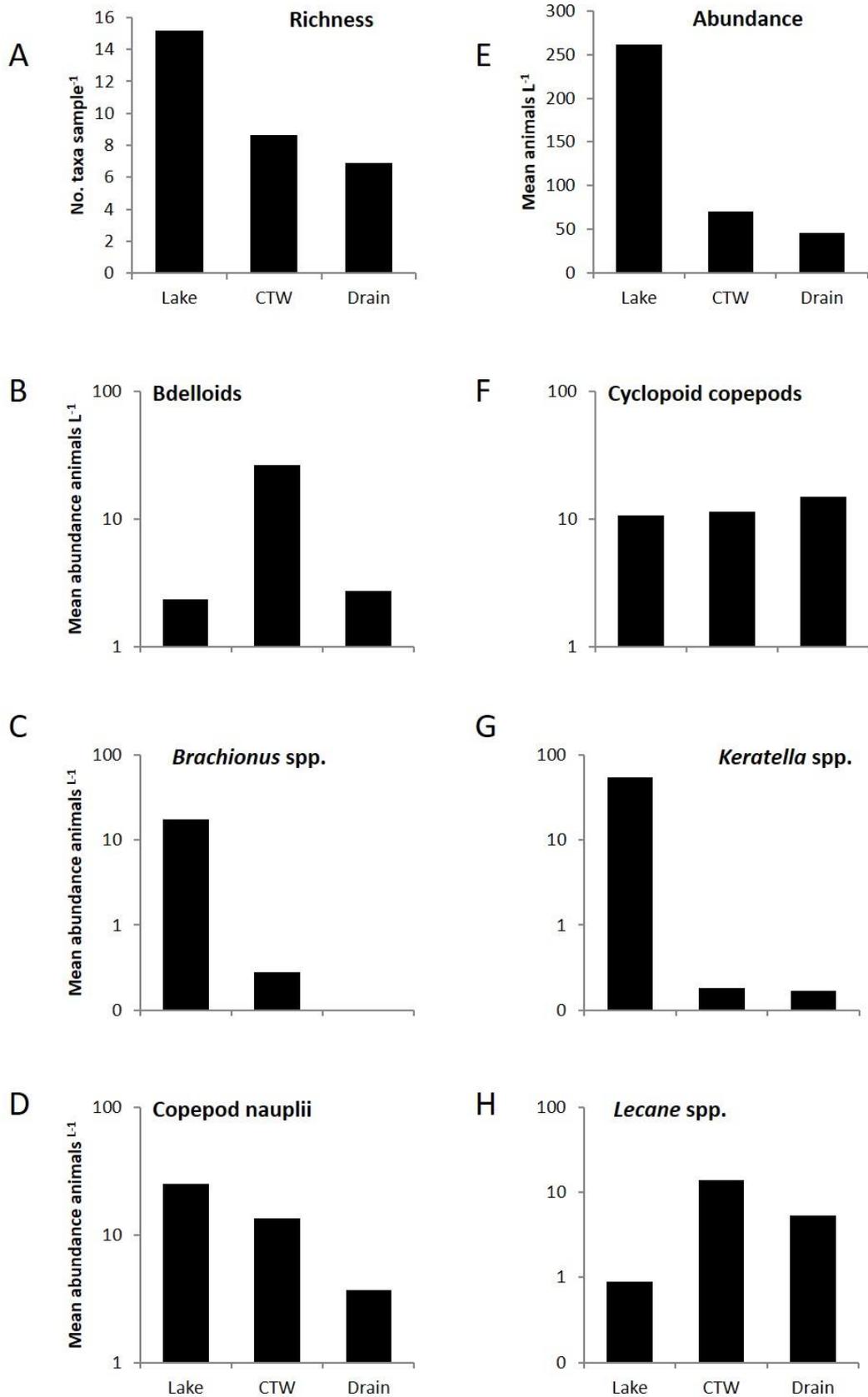


Figure 4.2 Mean values for taxa richness and total abundance of zooplankton (a, e) and total abundances of zooplankton species most dissimilar between habitat types including; *Bdelloids* (b), *Brachionus* spp. (c), *Copepod nauplii* (d), *Cyclopoid copepodites* (f), *Keratella* spp. (g) and *Lecane* spp. (h) plotted on a log₁₀-scaled axis for each habitat type (lake, CTW and drain)

Lake zooplankton assemblages were distinctly different from CTW and drain communities (SIMPER analysis, average Bray-Curtis dissimilarities 83.7 and 86.6, respectively). Differences were driven by the predominance of cladocerans, particularly the rotifers *Brachionus calyciflorus* and *Keratella tropica* (Figure 4.2C, G), as well as *Polyarthra vulgaris*, *Synchaeta longipes*, and the cladoceran *Bosmina meridionalis* (Table 4.2), all of which were at higher abundances in lake habitats but virtually absent from the CTWs and drains. Cumulatively, these taxa, as well as high densities of copepod nauplii, accounted for 45% and 48% of the lake-CTW and lake-drain differences, respectively (Table 4.2).

There was a relatively subtle difference in the dominant taxa typical of drain and CTW habitats, which is reflected by an average dissimilarity of 67.9 derived from the SIMPER analysis (Table 4.2) and represented graphically in Figure 4.2. Cumulatively, cyclopoid copepodites (10.8), copepod nauplii (9.3) and bdelloid rotifers (9.9) contributed 44.1% to the dissimilarity between CTW and drain zooplankton assemblages (Table 4.2). High relative abundances of *Lecane* spp. within CTW habitats (Figure 4.2H) also contributed to 9% of the community compositional differences in comparison to drain habitats (Table 4.2).

Table 4.2 SIMPER analysis (square-root transformed data) showing the main taxa contributing to the variation in community composition between lake-CTW and drain-lake habitats, and at least 80% between drain-CTW habitats. Contribution % is the proportion of dissimilarity between habitat pairs contributed by each taxa

Species	Mean	Mean	Mean	Contribution %
	abundance	abundance	dissimilarity	
	Drain	Wetland		
Cyclopoid copepodites	2.83	2.14	10.76	15.8
Bdelloids	1.21	3.13	9.94	14.6
<i>Copepod nauplii</i>	1.57	2.91	9.33	13.7
<i>Lecane rhytida</i>	0.59	1.27	6.08	9.0
Ostracods	1.55	0.27	4.04	5.9
<i>Cephalodella intuta</i>	0.83	0.42	3.12	4.6
<i>Lecane hamata</i>	0.14	0.68	2.75	4.1
Mosquito larvae	0.55	0.33	2.74	4.0
<i>Lecane lunaris</i>	0.84	0.30	2.67	3.9
<i>Lepadella patella</i>	0.24	0.40	2.02	3.0
<i>Cephalodella tenuiseta</i>	0.07	0.34	1.55	2.3
Mean dissimilarity = 67.9				
	Lake	Wetland		
<i>Keratella tropica</i>	5.31	0.05	8.14	9.7
<i>Bosmina meridionalis</i>	4.60	0.06	7.36	8.8
<i>Polyarthra vulgaris</i>	4.15	0.00	6.22	7.4
<i>Brachionus calyciflorus</i>	2.53	0.02	5.38	6.4
Copepod nauplii	4.47	2.91	5.20	6.2
<i>Synchaeta longipes</i>	2.56	0.00	5.05	6.0
Bdelloids	1.08	3.13	4.38	5.2
Cyclopoid copepodites	2.39	2.14	4.30	5.1
Mean dissimilarity = 83.7				
	Drain	Lake		
<i>Keratella tropica</i>	0.14	5.31	8.66	10.0
<i>Bosmina meridionalis</i>	0.00	4.60	7.98	9.2
<i>Polyarthra vulgaris</i>	0.00	4.15	6.61	7.6
Copepod nauplii	1.57	4.47	6.31	7.3
<i>Brachionus calyciflorus</i>	0.00	2.53	5.92	6.8
<i>Synchaeta longipes</i>	0.00	2.56	5.47	6.3
Cyclopoid copepods	2.83	2.39	4.86	5.6
Mean dissimilarity = 86.6				

Comparison of the relative abundances of the major zooplankton groups (cladocerans, copepods, ostracods and rotifers) for each habitat type supported the finer-scale differences between zooplankton assemblages determined by the SIMPER analysis. Cladocerans and calanoid copepods were significantly more

abundant in lake habitats compared with CTWs and drains (Table 4.3). Copepod nauplii and cyclopoid copepodids were common within each habitat (Figure 4.2D, F), but differed in their relative abundances (Table 4.3). Rotifers were also significantly more abundant in lake and CTW habitats compared with drains (Table 4.3), due to the relatively high abundances of bdelloids and *Lecane* spp. within CTW habitats (Figure 4.2B, H).

Table 4.3 Relative abundance (%) of zooplankton groups Cladocerans, Copepods (including Calanoids, Cyclopoids and nauplii), Ostracods, Rotifers and Others for lake, CTW and drain habitats. These measurements represent the mean abundance in which the relative abundance of each zooplankton taxon was estimated for each site (n = 40) then averaged across habitat type (lake n = 5; CTW n = 27, drain n = 8). Letters in superscript denote significant differences (P < 0.05) between habitat types

Relative zooplankton abundance (%)			
	Lake	Wetland	Drain
Zooplankton			
Cladocerans	25.02 ^a	1.77 ^b	0.30 ^b
Copepods	18.19	44.96	59.24
(Calanoids)	(1.55) ^a	(0.03) ^b	(0) ^b
(Cyclopoids)	(4.37)	(13.96)	(39.51)
(Nauplii)	(12.27)	(30.98)	(19.74)
Ostracods	0.51	0.56	8.71
Rotifers	56.17 ^a	50.92 ^a	28.62 ^b
TOTAL	99.89	98.22	96.88
Others			
Amphipods	0	0	0.37
Diptera	0.06	1.69	2.75
Hemiptera	0	0.09	0
Hydracarina	0.05	0	0
Tardigrades	0	0.01	0
TOTAL	0.11	1.78	3.12

4.4.2 Environmental variables

Lake habitats were significantly larger and more alkaline than CTWs and drains, and both lake shore and CTW habitats had significantly greater depths than drains (Table 4.4). Mean values of physicochemical, biotic and water quality variables were not significantly different between habitat types due to high variability within habitats.

Table 4.4 Mean (\pm standard deviation) of environmental variables including physicochemical, biotic and water quality for each habitat type. Letters in superscript denote significant differences (Kruskal Wallis test, $P < 0.01$) between habitat types. Connectivity with downstream lake being High, Medium, or Low (3, 2, 1, respectively). * lakes were considered to have high connectivity as there is unimpeded flow for exchange of water and aquatic organisms throughout the habitat

Environmental Variables		Habitat Type					
		Lake (n=5)		CTW (n=27)		Drain (n=8)	
Physicochemical							
Area	(m ²)	108000 ^a	\pm 91627	237 ^b	\pm 408	10 ^b	\pm 4
Depth	(m)	0.81 ^a	\pm 0.11	0.90 ^a	\pm 0.36	0.22 ^b	\pm 0.10
Connectivity	(H/M/L = 3/2/1)	3.00*	\pm 0	1.93	\pm 0.87	1.88	\pm 0.35
Temperature	(°C)	24.1	\pm 2.0	21.8	\pm 1.9	20.9	\pm 2.2
Conductivity	(mS cm ⁻¹)	0.18	\pm 0.04	0.25	\pm 0.05	0.20	\pm 0.09
DO	(mg L ⁻¹)	6.72	\pm 3.82	3.92	\pm 4.16	2.59	\pm 2.07
pH		6.84 ^a	\pm 1.24	5.09 ^b	\pm 0.72	5.52 ^b	\pm 0.52
Biotic							
Macrophytes	(% cover)	7	\pm 3	34	\pm 33	65	\pm 33
Chl α	(μ g L ⁻¹)	48.8	\pm 50.8	27.6	\pm 27.2	6.6	\pm 10.1
Fish	(P/A = 1/0)	1	\pm 0.00	0.48	\pm 0.51	0.38	\pm 0.52
Iron floc	(P/A = 1/0)	0.24	\pm 0.26	0.50	\pm 0.42	0.90	\pm 0.47
Water quality							
NH ₄ -N	(mg L ⁻¹)	0.04	\pm 0.04	0.11	\pm 0.15	0.22	\pm 0.34
NO ₃ -N	(mg L ⁻¹)	0.11	\pm 0.10	0.12	\pm 0.19	0.41	\pm 0.57
Org-N	(mg L ⁻¹)	3.92	\pm 1.81	3.18	\pm 1.19	2.37	\pm 1.16
PO ₄ -P	(mg L ⁻¹)	0.06	\pm 0.10	0.24	\pm 0.45	0.01	\pm 0.01
VSS	(mg L ⁻¹)	20.5	\pm 22.2	20.3	\pm 21.4	15.8	\pm 14.9
NVSS	(mg L ⁻¹)	13.5	\pm 17.7	12.7	\pm 13.6	8.9	\pm 9.7

Based on PERMANOVA, habitat type accounted for a significant proportion of the variation in biotic and abiotic environmental variables across all sites ($F_{\text{pseudo}} = 4.78, P = 0.0001$). Pairwise comparisons indicated significant differences between each habitat type (Lake, Drain $t=2.72$; Lake, CTW $t=2.10$; Drain, CTW $t=2.07, P < 0.01$). The morphological variables area and depth contributed to the greatest dissimilarity between habitat types, followed by pH, inorganic-N and conductivity (SIMPER results, Table 4.5.) Water temperature was also important, contributing to 6.8% of the dissimilarity between lake and CTW habitats, and 7.7% between lake and drain habitats.

Table 4.5 SIMPER analysis showing the environmental variables contributing up to 70 % of the variation among lake, CTW and drain habitats. Mean values for each variable have been back-transformed to original values to clarify dissimilarities. Contribution % is the percentage of dissimilarity between habitat groups contributed by each variable

Variable	Mean value	Mean value	Mean value of dissimilarity	Contribution %
	Drain	CTW		
Depth (m)	0.22	0.90	3.64	10.7
NO ₃ -N (mg L ⁻¹)	0.41	0.12	3.12	9.2
Conductivity (mS cm ⁻¹)	0.197	0.254	2.94	8.7
Chlorophyll <i>a</i> (µg L ⁻¹)	6.57	27.55	2.46	7.3
Org-N (mg L ⁻¹)	2.37	3.18	2.19	6.5
NH ₄ -N (mg L ⁻¹)	0.22	0.11	2.18	6.4
PO ₄ -P (mg L ⁻¹)	0.01	0.24	2.14	6.3
Macrophytes (% cover)	65	34	2.10	6.2
Iron floc (P/A = 1/0)	0.9	0.5	2.01	5.9
Mean dissimilarity = 33.90				
	Lake	CTW		
Area (m ²)	108000	237	5.93	15.6
pH	6.84	5.09	5.39	14.2
Temperature (°C)	24.12	21.83	2.58	6.8
Org-N (mg L ⁻¹)	3.92	3.18	2.28	6.0
DO (mg L ⁻¹)	6.72	3.92	2.25	5.9
Conductivity (mS cm ⁻¹)	0.176	0.254	2.20	5.8
Chlorophyll <i>a</i> (µg L ⁻¹)	48.81	27.55	2.14	5.7
NVSS (mg L ⁻¹)	13.46	12.69	2.05	5.4
Mean dissimilarity = 37.91				
	Drain	Lake		
Area (m ²)	10	108000	10.30	20.7
Temperature (°C)	20.89	24.12	3.83	7.7
pH	5.52	6.84	3.63	7.3
Chlorophyll <i>a</i> (µg L ⁻¹)	6.57	48.81	3.62	7.3
Org-N (mg L ⁻¹)	2.37	3.92	3.54	7.1
Macrophytes (% cover)	65	7	3.19	6.4
NH ₄ -N (mg L ⁻¹)	0.22	0.04	2.88	5.8
Iron floc (P/A = 1/0)	0.9	0.2	2.83	5.7
Mean dissimilarity = 50.00				

4.4.3 Habitat variables influencing zooplankton communities

Habitat type was determined by PERMANOVA to account for a significant proportion of the variation in zooplankton community composition ($F_{\text{pseudo}} = 3.59, P = 0.0001$). Pairwise comparisons indicated significant differences between each habitat type based on zooplankton community composition (Lake-Drain $t=2.19, P < 0.01$; Lake-CTW $t=2.21, P < 0.01$; Drain-CTW $t=1.40, P < 0.05$).

The DistLM procedure assigned the greatest proportion of the variation in zooplankton community composition among habitat types to area (22.1 %) followed by pH (16.7 %) and $\text{NH}_4\text{-N}$ (10.4 %; Table 4.6). Area was a key driver of the difference between lakes and CTWs, as well as between lakes and drains based on SIMPER analyses (Table 4.5), while differences in pH were significant between lakes and CTWs. The DistLM marginal tests also determined water temperature and DO were significant drivers of the variation in community composition between lake habitats and CTWs, as well as between lakes and drains (Table 4.6).

Table 4.6 Results from DistLM analysis of environmental variables driving variation in zooplankton community composition among lake, CTWs and drain habitats. Cum (%) = Cumulative percentage of variance; Prop (%) = Proportion of explained variance; ** P < 0.01; * P < 0.05; n.s = not significant

Marginal Tests			Step-wise Sequential Tests			
Variable	Pseudo-F		Variable	Pseudo-F	Cum (%)	Prop (%)
Area	3.50	**	Area	3.50	**	22.1
pH	3.01	**	pH	2.76	**	16.7
Iron Flocculant	2.40	**	NH ₄ -N	1.76	*	10.4
Temperature	2.31	**	Conductivity	1.68	n.s	9.8
Macrophytes	1.93	*	Iron Flocculant	1.55	n.s	8.9
DO	1.89	*	Temperature	1.47	n.s	8.3
NH ₄ -N	1.92	*	Depth	1.46	n.s	8.1
Conductivity	1.87	*	DO	1.36	n.s	7.5
PO ₄ -P	1.81	*	Macrophytes	1.50	n.s	8.1
Depth	1.76	*				
Fish	1.73	*				
NVSS	1.75	*				
Chlorophyll <i>a</i>	1.40	n.s				
VSS	1.38	n.s				
NO ₃ -N	1.36	n.s				
Org-N	1.27	n.s				

The DistLM analysis indicated conductivity and depth were significant explanatory variables, reflecting differences between CTWs and drains, and supporting previous analyses (Table 4.5). PO₄-P was a significant explanatory variable in marginal tests due to elevated concentrations in CTWs (mean=0.24 mg N L⁻¹) compared to lakes and drains (0.06 and 0.01 mg N L⁻¹, respectively). Finally, NH₄-N concentration, iron flocculant and emergent macrophyte cover, each of which were elevated in drains, were also identified as significant by the DistLM marginal tests (Table 4.6), supporting the SIMPER results (Table 4.5). Mean NH₄-N concentrations from drains (0.22 mg N L⁻¹) were twice as high as those from CTWs (0.11 mg N L⁻¹) and more than five-fold greater than lake concentrations (0.04 mg N L⁻¹; Table 4.5).

4.5 DISCUSSION

4.5.1 Zooplankton diversity

Our results suggest agricultural CTWs support greater zooplankton diversity than drain habitats and can increase the overall biodiversity of highly modified peat lake catchments. For example, within the catchment of Lake Kainui, 15 taxa were recorded from the lake itself, five additional taxa were recorded from drains, and a further 17 taxa from CTW habitats. Collectively, CTWs had the highest total diversity (54 taxa) followed by lakes (40) and then drains (20) (Table SM 4.1). Mean zooplankton taxa richness for CTWs (c. 9) was intermediate between lake (15) and drain (6) habitats, although no significant differences were apparent due to high variability within habitat types.

The diversity of zooplankton from the habitats in our study is comparable to that from freshwater ecosystems elsewhere. For example, zooplankton taxa richness for our CTW habitats was similar to floodplain ponds of Truman Lake, a reservoir in Missouri (Medley & Havel 2007), midsummer zooplankton assemblages from wetlands in the mid-west USA (Beaver et al. 1999), and small (area <1 ha) shallow lakes in south-eastern Wisconsin, USA (Dodson et al. 2005). The lower taxa richness of drain habitats in this study was similar to that of agriculturally impacted wetlands in Wisconsin (mean 3.8) studied by Dodson & Lillie (2001), while richness from lakes Komakorau (21) and Koromatua (19) was similar to eutrophic shallow lake ecosystems in the northern hemisphere (Søndergaard et al. 2005), as well as a number of lakes in the North Island of New Zealand (Duggan et al. 2001).

Notably, two of the CTWs (KN1 and KR2) had greater taxa richness (both 16) than lakes Kainui (15), Kaituna (12) and Serpentine South (9), despite having considerably smaller areas (0.2 and 0.02 ha compared to 25, 15, and 5 ha, respectively). Søndergaard et al. (2005) similarly found a weak relationship between zooplankton taxa richness and habitat size following an investigation of almost 800 Danish shallow lakes (median depth 1.5 m), ranging from 0.01 to 4200 ha, going on to suggest that ponds and small lakes are important

biodiversity components in agricultural landscapes. CTWs KN1 and KR2 were relatively complex habitats, with deep (> 1 m) and shallow (< 0.5 m) zones, areas of open water and moderate macrophyte cover; thus, greater zooplankton diversity may be expected owing to a greater range of potential habitats and niches (Lucena-Moya & Duggan 2011).

4.5.2 Zooplankton assemblages & functional attributes

CTW-zooplankton assemblages differed from lake and drain communities owing to relatively few cladocerans and a predominance of rotifers, primarily comprising large numbers of bdelloid rotifers and *Lecane* species. Bdelloid rotifers are primarily benthic and are able to survive extended dry periods via anhydrobiosis (Crowe et al. 1992), making them well adapted to CTW habitats in artificially drained agricultural peat lake catchments where water levels can fluctuate widely. Like most rotifers, bdelloids are suspension feeders and thrive on dead organic matter and bacteria (Ricci 1984). Along with excess sediment and nutrients, pathogenic bacteria have been identified as one of New Zealand's three major water quality problems (MfE/StatsNZ 2017). Runoff from agricultural catchments is known to transport pathogenic bacteria including *Escherichia coli*, *Salmonella*, *Campylobacter* and *Shigella*, as well as the pathogenic protozoans *Cryptosporidium* and *Giardia* (Jamieson et al. 2002; Hooda et al. 2000; Ballantine & Davies-Colley 2014). Bacterivorous zooplankton have been shown to reduce pathogenic bacteria from water (Schallenberg et al. 2005) and are likely to play an important role in the treatment efficiency of CTWs in agricultural landscapes. Therefore, the prevalence of these species in the CTWs of this study is encouraging.

While zooplankton such as bdelloids feed directly on bacteria, metazooplankton including calanoid and cyclopoid copepods as well as large cladocerans such as *Ceriodaphnia dubia* (Galbraith & Burns 2010), consume bacteria indirectly via predation on ciliates (Hansen 2000). Bacterivorous ciliates are important for reducing bacterial densities in effluent from sewage treatment plants (Madoni 2003; Curds et al. 1968) and may also suppress viruses (Pinheiro et al. 2007). The calanoid copepod *Calamoecia lucasi* was abundant in Lake Koromatua and

present in CTW KR2 and lakes Kainui, Komakorau, and Serpentine North, while *C. dubia* was present in CTWs KN5 and KT5 as well as Lake Kainui. Furthermore, early copepodid instars (copepod nauplii and cyclopoid copepodites), which feed preferentially on ciliates (Hansen 2000), were prevalent in CTW habitats, comprising approximately 45 % of relative abundances. The presence of these zooplankton species in the CTWs of this study is promising as they may contribute to reducing pathogenic bacterial communities entering the downstream lakes. Wilcock et al. (2012) likewise suggest that enhancing communities of bacterivorous microorganisms is likely to increase the efficacy of agricultural wetland treatment systems through consumption of harmful bacteria.

Drain-zooplankton assemblages differed from CTWs and were driven by a lack of rotifers and relatively large numbers of ostracods, cyclopoid copepodites and mosquito larvae. Drain habitats generally supported lower zooplankton diversity and abundances, exempting the drain site of Lake Kainui (KND). KND had low macrophyte cover compared to other drain habitats as a result of being cleared in the preceding spring, which may have contributed to greater zooplankton taxa richness (11) and higher abundances (247 animals L⁻¹).

Lake-zooplankton assemblages were distinctly different from CTW and drain communities, due to greater abundances of cladocerans and pelagic rotifer species. Cladoceran zooplankton play a vital role in energy transfer and food web dynamics (Kattel 2012) and were particularly abundant in lake habitats, comprising a quarter of the zooplankton community. The cladoceran *Bosmina meridionalis* was one of the dominant species from our lake samples, second only to the rotifer *Keratella tropica*, each species being commonly abundant in eutrophic New Zealand lakes (Duggan et al. 2002). Interestingly, two species more typical of lakes with low trophic state, *Polyarthra vulgaris* and *Synchaeta longipes*, were abundant in lakes Koromatua and Serpentine North, respectively, despite the lakes being highly eutrophic.

4.5.3 Habitat characteristics

Each habitat type supported significantly different zooplankton communities (refer to results of PERMANOVA and pair-wise comparisons), including between drains and CTWs. Habitat morphologies including surface area, water depth and macrophyte cover were most strongly associated with variations in zooplankton composition, whilst significant water quality parameters included pH, conductivity, iron flocculant and concentrations of PO₄-P and inorganic-N. Drain habitats were smaller and shallower than CTWs, with dense macrophyte cover, high concentrations of NH₄-N and NO₃-N, and high frequency of iron flocculant occurrence. CTW habitats were intermediate in size between lakes and drains, relatively deep and with high Chl *a*, Org-N and VSS concentrations, indicative of elevated phytoplankton biomass. Levels of PO₄-P were also highest in CTWs, particularly in those with low pH. Lakes were naturally the largest habitats and moderately deep with warmer water temperatures, high DO and neutral pH. Similar to CTW habitats, lakes had elevated Chl *a*, Org-N and VSS concentrations due to high phytoplankton densities, which in turn supported greater zooplankton abundances.

As evidenced by the results from this study, dense macrophyte cover in small, shallow watercourses suppresses phytoplankton growth and can cause stagnant, reducing conditions, elevating NH₄-N levels. Nonetheless, while excessive macrophyte growth can have negative impacts on stream communities (Collier et al. 1999), macrophytes improve the structural complexity of freshwater habitats and strongly influence community structure and diversity (Lucena-Moya & Duggan 2011; Warfe & Barmuta 2006).

Macrophytes are an important refuge against predation for pelagic cladocerans, as described by Timms & Moss (1984) following observations of different water clarity between connected shallow wetland ecosystems in the Norfolk Broads. The size of macrophyte beds is also important, particularly for populations of horizontally migrating cladocerans such as *Ceriodaphnia* and *Bosmina* species. Lauridsen et al. (1996) suggest numerous small refuges (c. 2 m diameter) are likely to support higher densities of these species rather than single large refuges

(> 10 m diameter), which are more typically dominated by macrophyte-associated, non-migrating cladocerans such as *Chydorus* and *Simocephalus* species. Each of the aforementioned cladoceran species occurred in a number of the lake and CTW habitats in our study; thus, it is possible similar dynamics may occur in Waikato peat lake ecosystems. Furthermore, the structural complexity of different macrophytes provides habitat niches favoured by different zooplankton taxa (Lucena-Moya & Duggan 2011). Thus, the variety of emergent, submerged and floating macrophytes within a wetland ecosystem will influence zooplankton biodiversity and biomass.

Finally, many of the CTWs in this study were designed to capture coarse sediment to reduce infilling of the shallow peat lakes downstream. While deposited sediment was not measured in this study, the effects of sedimentation, particularly in cultivated catchments, requires consideration as populations of most zooplankton taxa can persist over long periods through egg banks (Hairston 1996). Gleason et al. (2003) investigated the effects of sediment loads from intensive agricultural activities on the emergence of zooplankton from wetland soil egg banks and found burial by sediment as little as 0.5 cm deep reduced invertebrate emergence by 99%. Duggan et al. (2002) also found limited diversity in lakes with high sediment loads. Designing CTWs with distinct areas of open water and variable depths, by incorporating sedimentation forebays as well as open-water habitats isolated from sediment inputs, will help to minimise any adverse sedimentation effects on zooplankton communities and promote greater diversity.

4.6 CONCLUSIONS

CTWs can improve the overall biodiversity of highly-modified peat lake catchments, by supporting zooplankton species otherwise absent from lake and drain habitats. Zooplankton taxa richness and abundances were broadly higher from CTWs than drain habitats, and a few CTWs supported greater diversity than several lakes. The results from our research suggest CTWs afford dual benefits for peat lake restoration within intensive agricultural landscapes through provision of habitat for zooplankton communities as well as water quality improvements.

To further enhance zooplankton communities, we propose creating CTWs with variable depths and areas deeper than existing drains, with larger areas of open water and moderate to low levels of diverse macrophyte cover. Opportunities exist to manipulate the influence of macrophyte beds through careful design, construction, plant selection and maintenance to support targeted zooplankton species which can sustain grazing on phytoplankton and improve water quality treatment (Schriver et al. 1995). Just as recent research has extended our knowledge of zooplankton population dynamics within high rate algal ponds in support of improved wastewater treatment (Montemezzani et al. 2016, 2017), expanding our understanding of the lifecycles and habitat requirements of bacterivorous zooplankton and ciliates in wetlands could additionally inform CTW designs to improve operational efficiency whilst concurrently supporting greater zooplankton biodiversity.

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4.9 SUPPLEMENTARY MATERIAL

Table SM 4.1 Zooplankton taxa recorded from lake (L), constructed treatment wetland (CTW) and drain (D) habitats within peat lake catchments. Samples were collected from the Central Waikato region, New Zealand, between 2-8th February 2011

Phylum / Suborder / Subclass	Genus / Species	Habitat	Phylum / Suborder / Subclass	Taxa	Habitat
Cladocera	<i>Alona camboui</i>	L, CTW	Rotifera	<i>Keratella procurva</i>	L
Amphipoda	Amphipod	D	Rotifera	<i>Keratella tecta</i>	L
Hemiptera	<i>Anisops wakefieldi</i>	CTW	Rotifera	<i>Keratella tropica</i>	L, CTW, D
Rotifera	<i>Anuraeopsis fissa</i>	L, CTW	Rotifera	<i>Lecane bulla</i>	L, CTW
Rotifera	<i>Anuraeopsis navicula</i>	L	Rotifera	<i>Lecane closterocerca</i>	CTW, D
Rotifera	<i>Asplanchna brightwelli</i>	L	Rotifera	<i>Lecane hamata</i>	L, CTW, D
Rotifera	<i>Asplanchna priodonta</i>	L	Rotifera	<i>Lecane hornemanni</i>	CTW
Rotifera	Bdelloids	L, CTW, D	Rotifera	<i>Lecane lunaris</i>	L, CTW, D
Cladocera	<i>Bosmina meridionalis</i>	L, CTW	Rotifera	<i>Lecane rhytida</i>	CTW, D
Rotifera	<i>Brachionus angularis</i>	L, CTW	Rotifera	<i>Lepadella accuminata</i>	L, CTW
Rotifera	<i>Brachionus budapespinensis</i>	L	Rotifera	<i>Lepadella patella</i>	CTW, D
Rotifera	<i>Brachionus calyciflorus</i>	L, CTW	Rotifera	<i>Lophocharis salpina</i>	D
Rotifera	<i>Brachionus quadridentatus</i>	L, CTW	Hydracarina	Mites	L
Copepoda	<i>Calamoecia lucasi</i>	L, CTW	Rotifera	<i>Monomatta</i> sp.	CTW
Rotifera	<i>Cephalodella catellina</i>	CTW	Diptera	Mosquito larvae	CTW, D
Rotifera	<i>Cephalodella forficula</i>	CTW, D	Rotifera	<i>Mytilina bisulcata</i>	CTW
Rotifera	<i>Cephalodella intuta</i>	CTW, D	Rotifera	Notommata sp.	L
Rotifera	<i>Cephalodella tenuiseta</i>	CTW, D	Ostracoda	Ostracods	L, CTW, D
Rotifera	<i>Cephalodella ventripes</i>	CTW	Rotifera	<i>Platyais quadricornis</i>	L, CTW
Cladocera	<i>Ceriodaphnia dubia</i>	L, CTW	Rotifera	<i>Polyarthra dolichoptera</i>	L
Diptera	Chironomid larvae	L, CTW, D	Rotifera	<i>Polyarthra vulgaris</i>	L
Cladocera	<i>Chydorus</i> sp.	L, CTW	Rotifera	<i>Proales</i> sp.	D
Rotifera	<i>Colurella uncinata</i>	CTW	Cladocera	<i>Pseudomoina lemnae</i>	CTW
Rotifera	<i>Conochilus natans</i>	CTW	Hemiptera	Sigara sp.	CTW
Copepoda	Copepod nauplii	L, CTW, D	Cladocera	<i>Simocephalus expinosus</i>	CTW, D
Copepoda	Cyclopoid copepods	L, CTW, D	Rotifera	<i>Synchaeta longipes</i>	L
Rotifera	<i>Dicranophorus forcipatus</i>	L, CTW	Rotifera	<i>Synchaeta oblonga</i>	CTW
Rotifera	<i>Encentrum saunderiae</i>	D	Rotifera	<i>Synchaeta pectinata</i>	CTW
Rotifera	<i>Euchlanis meneta</i>	L	Rotifera	<i>Taphocampa selenura</i>	CTW
Rotifera	<i>Filinia longiseta</i>	L	Tardigrada	Tardigrades	CTW
Rotifera	<i>Filinia novaezealandiae</i>	L	Rotifera	<i>Testudinella patina</i>	CTW
Rotifera	<i>Hexarthra intermedia</i>	L	Rotifera	<i>Trichocerca brachyura</i>	CTW
Rotifera	<i>Hexarthra mira</i>	L	Rotifera	<i>Trichocerca stylata</i>	L, CTW
Cladocera	<i>Ilyocryptus sordidus</i>	CTW	Rotifera	<i>Trichocerca teniour</i>	CTW
Rotifera	<i>Itura myersi</i>	CTW	Rotifera	<i>Trichotria tetractis</i>	L
Rotifera	<i>Keratella cochlearis</i>	L, CTW	Rotifera	Rotifer unknown sp.	CTW
Rotifera	<i>Keratella javana</i>	CTW			

Chapter 5

5 CONCLUDING DISCUSSION

5.1 RESEARCH SUMMARY

My thesis has demonstrated the dual benefits of constructed treatment wetlands (CTWs) for biodiversity and water quality enhancement within intensive agricultural peat lake catchments. I examined the attenuation efficiencies of nutrient and sediment inputs, and compared zooplankton communities within CTWs, lake and drain habitats. Important design characteristics for optimising pollutant removal performance, and the significance of biogeochemical processes influencing water chemistry and constituent composition, have been identified. Quantifying the magnitude and variability of nutrient loads between peat, peaty loam and clay loam subcatchments of the lakes has provided valuable data from which to inform management. My research has determined that zooplankton biodiversity within peat lake catchments is enhanced by the presence of CTWs, and I explored the habitat attributes of CTWs that may enrich zooplankton diversity to improve pollutant removal performance. Collectively, the findings of this thesis provide a scientific basis for more comprehensive, holistic design considerations for CTWs to mitigate diffuse pollution from intensive agricultural peat lake catchments, and quantitative evidence of the value of agricultural CTWs as water quality and restoration management tools.

5.2 SYNTHESIS

5.2.1 Composition of influent constituents

To achieve lake restoration goals and meet targets for water quality and swimmability outlined in legislation such as New Zealand's National Policy Statement for Freshwater Management (NPS-FM) (MfE 2017, 2018), knowledge of

the magnitude, variability and source of catchment pollutant loads is critical (Fox & Argent 2009; Hamilton et al. 2016). Chapter 2 achieved the first objective of my thesis, which was to quantify the variability and magnitude of nutrient and sediment inputs to five shallow peat lakes with agricultural catchments supporting intensive dairy production through (i) seasonal synoptic surveys of physico-chemical and water quality related variables from the major inflows to the lakes, and (ii) estimation of nutrient and sediment loads based upon subcatchment-scale water yields and measured concentrations of N, P and SS. The relative importance of subcatchment soil type, seasonality, and farm-scale management practices influencing the variability in pollutant loads at the catchment scale was elucidated, and underlying environmental and biogeochemical processes were explored.

The magnitude of N and P inputs to the five peat lakes from the small subcatchments studied (62% of sites < 10 ha, 81% of sites < 20 ha) was far greater than expected. Calculating subcatchment Daily Areal Yields (DAY) and Daily Instantaneous Loads (DIL) from measured constituent concentrations and flow volumes enabled comparisons to be made with similar studies, providing an indication of the scale of pollutant loads to the study lakes. Substantial exports of N occurred during autumn and winter when lake-catchment DAY (range 0.5 – 9.4 kg N ha⁻¹ d⁻¹) were up to one-third of annual yields reported by other studies in the Waikato (Wilcock et al. 1999) and Southland (Monaghan et al. 2000) regions of New Zealand. Yields of P from several subcatchments were extremely high compared with other studies in New Zealand (Abell et al. 2013; Verburg et al. 2013; Smith & Monaghan 2003), although more closely aligned to those reported by McDowell & Monaghan (2015) of 87 kg P ha⁻¹ (over 18 months) from a trial conducted on a flat (<1% slope) dairy farm with organic soils in Southland. Daily aerial yields of P were greatest from deep peat subcatchments of Lake Kainui (0.09 – 0.75 kg P ha⁻¹ d⁻¹), and peaty loam subcatchments of lakes Komakorau (0.01 – 0.16 kg P ha⁻¹ d⁻¹) and Koromatua (0.12 – 0.60 kg P ha⁻¹ d⁻¹), and considerably lower from clay loam dominated subcatchments (0.001 – 0.08 kg P ha⁻¹ d⁻¹). These differences revealed

the acute susceptibility of organic soils to leach P from intensive agricultural catchments.

These comparisons should be regarded with caution, however, as areal loads from my study were derived from five seasonal sampling occasions over 18 months, and, in some instances, samples were collected at flows above seasonal averages (derived from two nearby permanent water-level monitoring stations) and may therefore have overestimated loads (Figure SM 5.1). Nonetheless, the lake-catchment yield estimates provide a useful starting point from which to collate data, offer insights to the scale of effort required to address nutrient and sediment exports, and facilitate between-lake comparisons that can assist lake and water quality managers to prioritise restoration actions and allocation of funds. For example, DILs of N to Lake Kaituna were consistently highest across all seasons, followed by lakes Kainui and Koromatua, while the DILs of P were similarly high across all three lakes, although they varied seasonally (Table 2.9). Consequently, managers may prioritise mitigation actions to reduce N losses to Lake Kaituna and chose to gather additional data across lake subcatchments to clarify patterns of P inputs.

Subcatchment loads of N, P and SS varied widely spatially and temporally, influenced by differing soil types and seasonal drivers. Watercourses draining subcatchments with deep peat soils had relatively high concentrations of $\text{NH}_4\text{-N}$ and $\text{PO}_4\text{-P}$, while those from clay loam subcatchments had greater concentrations of oxidised N ($\text{NO}_3\text{-N}$) and particulate-P. Temporal variations in pollutant loads were driven primarily by seasonal changes in rainfall, temperature, and day length, influencing nutrient uptake by plants, and hydrologically mediated processes such as surface runoff, $\text{NO}_3\text{-N}$ leaching, and denitrification. Comparing yields between subcatchments with similar soil types and topography made it possible to identify potential sources of nutrients. These related to farm infrastructure (milking-sheds, feed-pads, silage-stacks, effluent ponds and irrigation equipment), critical source areas or pollutant 'hotspots' where stock assemble frequently at high densities (raceways, troughs,

shade-trees) and sources associated with variations in land use intensity and management at the farm scale (stocking rates, grazing regimes, and conventional vs. best management practices (BMPs)). Four examples, comparing mean DAY for ten subcatchments representing key soil types, demonstrate the application of this concept and are summarised in Table SM 5.1.

Identifying the sources and transport mechanisms of nutrient and sediment inputs to waterbodies is vital for the development and execution of specific and appropriate management plans for effective mitigation of diffuse pollution from intensive agricultural land use. Implementing BMPs should follow a hierarchical approach, initially focused on minimising the source and quantities of pollutants (Barling & Moore 1994), followed by targeted mitigation methods to intercept primary transport pathways (Monaghan et al. 2008). For example, to curtail losses of sediment and particulate forms of N and P, BMPs should firstly focus on good pasture management alongside appropriate stocking rates and grazing regimes (McDowell & Wilcock 2007). Secondary BMPs, such as effective stock exclusion from watercourses by well-constructed fences with adequate set-backs from stream banks, as well as planting of riparian margins, will help to filter surface runoff of N, P and sediment to waterways (Smith 1989; Wilcock et al. 2009; McKergow et al. 2016). However, managing dissolved forms of N and P, such as $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$ and $\text{PO}_4\text{-P}$, can be more problematic as transport vectors are via subsurface pathways and are therefore more difficult to intercept. Additionally, leaching is often exacerbated by subsurface artificial drainage in agricultural catchments (Monaghan et al. 2007a; McDowell et al. 2004), and significantly influenced by soil characteristics and hydrological processes which are challenging to manipulate from a management perspective (Monaghan et al. 2007b; Wilcock et al. 2009).

Supporting landowners to address nutrient and sediment losses requires a quantitative understanding of the environmental drivers and underlying mechanisms influencing the source, quantity and transport of contaminants.

Chapter 2 discussed various hydrological and biogeochemical mechanisms governing

the release and transport of $\text{PO}_4\text{-P}$ and $\text{NH}_4\text{-N}$ from deep and shallow peat soils used to support intensive agriculture. However, a comprehensive and complete understanding of the drivers of such processes and the interactive complexities between them is universally lacking. Improving our knowledge of these mechanisms is pivotal to advancing BMPs to effectively minimise $\text{PO}_4\text{-P}$ and $\text{NH}_4\text{-N}$ losses from organic soils to meet water quality targets and objectives for peat lake restoration in New Zealand.

5.2.2 Agricultural CTW treatment performance

Constructed treatment wetlands (CTWs) have been implemented as management tools to alleviate water quality impacts from intensive agricultural land use on several shallow peats lakes in the Waikato region of New Zealand. However, due to a lack of response in the lake ecosystems and an absence of CTW monitoring, the efficacy of these treatment systems has remained in doubt. The second objective of my thesis was to investigate the value of free water surface agricultural CTWs as tools to support peat lake restoration in catchments used for intensive dairy production. There were two parts to this objective pertaining to the value of CTWs as pollutant removal tools for improving the quality of surface waters, and as restoration tools for improving biodiversity. Chapter 3 attended to the first part of this objective, by evaluating the pollutant removal performance of 26 CTWs across 5 peat lake catchments sampled seasonally over 18 months. Relationships between removal rates of inorganic and particulate forms of N, P and SS, and various morphological predictors of CTW performance, as well as physico-chemical and environmental variables, were identified from the aggregated dataset.

The treatment performance of the CTWs was influenced by design morphologies, internal nutrient cycling, and the composition of influent constituents. Numerous and varied relationships were evident between different morphological predictors of CTW performance and removal rates of various forms of N, P and SS.

Predominantly, CTWs with larger areas and volumes improved removal performance of $\text{NO}_3\text{-N}$, TN and sediment accumulation, while deeper CTWs reduced particulate

organic-N and volatile suspended sediment (VSS) concentrations. Removal efficiency of TP improved with longer HRT, likely due to greater sedimentation of PP, as well as biological uptake, chemical precipitation and adsorption of $\text{PO}_4\text{-P}$ (Kadlec & Wallace 2008a). These processes appeared to be impeded in CTWs with elevated hydraulic loading rates where removal rates of $\text{PO}_4\text{-P}$, PP and TP declined. Removal rates of $\text{NO}_3\text{-N}$ were greater in CTWs with two modules, while three modules improved reductions in $\text{NH}_4\text{-N}$. Filtration outlets improved removal rates of TP, and macrophytes enhanced the removal of $\text{NO}_3\text{-N}$ and TP, as well as VSS and non-VSS, particularly in smaller CTWs (Wetland to Catchment Area Ratio <0.05), in general agreement with the CTW literature (Vymazal 2013; Kadlec & Wallace 2008c).

Several design parameters and environmental variables displayed negative relationships with removal rates of alternate forms of N, P and SS, which were related to internal nutrient cycling. For example, $\text{NH}_4\text{-N}$ concentrations increased in CTWs with filtration outlets, likely due to generation from anaerobic processes such as Dissimilatory Nitrate Reduction to Ammonium (DNRA) (Zhu et al. 2017) and ammonification of previously stored organic N from wetland sediments, combined with low rates of nitrification (Bowden 1987). Furthermore, the presence of macrophytes appeared to impede $\text{NH}_4\text{-N}$ removal, conceivably due to inhibition of NH_3 volatilisation (Poach et al. 2004). Moreover, negative removal rates of TP occurred in CTWs with large volumes of accumulated sediment, indicative of P release from anaerobic sediments (Dunne et al. 2011). VSS and particulate organic nutrients increased in open-water areas, likely due to nutrient assimilation, and growth and reproduction of phytoplankton. The dynamic and complex relationships between pollutant removal performance and CTW design characteristics, as well as interrelated biogeochemical processes and environmental variables, demonstrates the critical importance of scientifically informed, appropriate CTW design and effective, rigorous maintenance routines.

Influent composition significantly influences the removal performance of CTWs (Kadlec & Wallace 2008b; Koskiaho et al. 2003; Vymazal 2007) and correspondingly

should govern CTW design. In this study both influent concentrations of N, P and SS and the predominant forms of nutrients varied extensively across CTWs, greatly influencing attenuation efficiencies. Such broad variability was not initially anticipated given the similarity of soil types, topography, climatic conditions and intensity of agricultural land use among the five peat lake catchments in this study. However, the high degree of hydrological modification and variability between peat, peaty loam and clay loam dominated soil types at the subcatchment scale profoundly influenced localised diffuse pollution patterns, as discussed in Chapter 2 and summarised in section 5.2.1. Accordingly, designing CTWs to target removal of specific water quality contaminants, such as $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$ and $\text{PO}_4\text{-P}$, which are presently more difficult to manage via land-based BMPs, and those occurring at concentrations breaching water quality limits, can improve removal performances and assist landowners, land-care groups and managers to achieve more specific water quality goals.

5.2.3 Zooplankton biodiversity

Chapter 4 addressed part (ii) of objective two, by investigating the efficacy of CTWs as tools to support peat lake restoration by comparing the zooplankton biodiversity of CTWs with lake and drain habitats within five highly modified, agricultural peat lake catchments. Lake restoration frequently centres on improving the functionality, resilience and biological integrity of lake ecosystems (Hamilton et al. 2016).

Zooplankton communities are essential components of lake and wetland ecosystems and can be representative of biodiversity and ecosystem health (Lougheed & Chow-Fraser 2002; Boix et al. 2005; Duggan et al. 2001). Comparison of zooplankton community composition between habitat types revealed CTWs supported species otherwise absent from lake and drain habitats, thereby increasing the biodiversity of highly modified peat lake catchments. Collectively, CTWs had the highest total diversity (54 taxa) followed by lakes (40) and then drains (20). Notably, two of the CTWs (KN1 and KR2) had greater taxa richness (both 16) than lakes Kainui (15), Kaituna (12) and Serpentine South (9), despite having considerably smaller areas.

Søndergaard et al. (2005) similarly found a weak relationship between zooplankton taxa richness and habitat size and asserted that ponds and small lakes are important biodiversity components in agricultural landscapes.

Bacterivorous zooplankton species, as well as metazooplankton which prey on ciliates that consume bacteria and viruses (Galbraith & Burns 2010), occurred in several CTW and lake habitats. Both bacterivorous zooplankton and ciliates have been shown to reduce pathogenic bacteria (Schallenberg et al. 2005; Madoni 2003; Curds et al. 1968) and may therefore play an important role in the performance of CTWs in agricultural landscapes. Wilcock et al. (2012) also suggest communities of bacterivorous microorganisms are likely to increase the efficacy of agricultural treatment wetlands through consumption of harmful bacteria. Although not directly addressed in my study, the presence of these zooplankton species in CTWs is encouraging as they may contribute to reducing pathogenic bacterial communities entering the downstream lakes.

Zooplankton composition was most strongly associated with habitat morphologies, including surface area, water depth and macrophyte cover, as well as water quality parameters including pH, conductivity, iron floc and concentrations of PO₄-P and inorganic-N. Drain habitats were smaller and shallower than CTWs, with dense macrophyte cover, high concentrations of NH₄-N and NO₃-N, and prevalent iron floc. CTW habitats were intermediate in size between lakes and drains, relatively deep and with high Chl *a*, Org-N and VSS concentrations, indicative of elevated phytoplankton biomass. Macrophyte cover in CTWs was moderate (mean 34%) although ranged from 5-100 %. Macrophytes are an important refuge against predation for cladoceran zooplankton (Timms & Moss 1984) and improve the structural complexity of freshwater habitats, providing habitats favoured by different zooplankton taxa and influencing diversity (Lucena-Moya & Duggan 2011). Moreover, the size of macrophyte beds has been shown to be important for horizontally migrating cladocerans such as *Ceriodaphnia* and *Bosmina* species (present in CTW and lake habitats of this study), with numerous small refuges (< 2 m

diameter) supporting higher densities of these species compared to a single, large bed (> 10 m diameter) (Lauridsen et al. 1996). Thus, the size and variety of emergent, submerged and floating macrophyte beds within a CTW ecosystem will influence zooplankton biodiversity and biomass. These relationships require further investigation so that CTW designs can be improved to create diverse habitats to support zooplankton species with a variety of niches that are beneficial to ecosystem function and CTW efficacy within intensive agricultural landscapes. These findings successfully met the second objective of my thesis, demonstrating that CTWs effectively deliver dual benefits for peat lake restoration through provision of habitat to support diverse zooplankton communities as well as achieving water quality improvements.

5.3 RECOMMENDATIONS FOR MANAGEMENT AND FUTURE RESEARCH

There is a wealth of knowledge available to inform BMPs targeted at minimising losses of $\text{NO}_3\text{-N}$ and SS from catchments dominated by dairy production. However, there is little information for the management of dissolved forms of N and P. The findings from Chapter 2 of this thesis show considerable quantities of $\text{PO}_4\text{-P}$ and $\text{NH}_4\text{-N}$ are lost from cultivated peat soils compared with clay loam. There is a paucity of research in New Zealand and internationally pertaining to the vulnerability of dairy farmed peat soils to leach dissolved forms of nutrients, particularly in catchments with varying depths of peat and fluctuating aerobic/anaerobic conditions driven by periods of saturation and drying, typical of many artificially drained and cultivated peat soils. Research is required to more accurately quantify these nutrient losses and to identify underlying biogeochemical processes at fine spatial scales (farm or subcatchment scale). Filling this knowledge gap will enable suitable nutrient management methods to be developed, and appropriate water quality limits to be set, with respect to local hydrological and biogeochemical constraints.

It is critically important to make allowance for the inevitable variability in pollutant loads and composition when designing agricultural CTWs. Manipulating desired biogeochemical wetland processes and habitat diversity throughout a series of CTW treatment-train modules is recommended to enhance biodiversity, ensure comprehensive pollutant removal performance and improve the value of agricultural CTWs as both water quality and restoration management tools (Wilcock et al. 2012; Jayasooriya et al. 2016; Wang et al. 2018).

To enhance zooplankton communities, I recommend creating CTW modules with variable depths, including areas deeper than existing drains, with larger areas of open water and moderate to low levels of diverse macrophyte cover. Incorporating sedimentation forebays or modules isolated from open-water habitats is important for minimising adverse effects of sedimentation on zooplankton egg banks and emergence (Gleason et al. 2003; Duggan et al. 2002). Opportunities exist to manipulate the influence of macrophyte beds through careful design, construction, and plant selection. This could support specific zooplankton species that graze on phytoplankton and pathogenic bacteria, resulting in improved water quality treatment (Rodrigo et al. 2018; Schriver et al. 1995). Expanding our understanding of the lifecycles and habitat requirements of bacterivorous zooplankton and ciliates in wetlands is needed to inform CTW designs and improve operational efficiency, while concurrently supporting greater zooplankton biodiversity.

A conceptual CTW treatment-train designed to optimise agricultural pollutant removal performance is presented in Figure 3.6 (Chapter 3). It summarises the primary processes and patterns of nutrient and sediment attenuation and cycling occurring throughout a series of modules. The proposed design comprises a deep sedimentation pond-module (≥ 1.5 m deep), shallow macrophyte module (~ 0.3 m deep), and open-water pond-module of moderate depth (~ 1 m) in series. This design is a simplification of more comprehensive design concepts and recommendations presented in *'Guidance and considerations for design and installation of constructed wetlands in shallow lake catchments'*, a technical report

produced in conjunction with this PhD thesis for the Waikato Regional Council (Eivers 2016). A copy of this document is provided in section 6.1 (Appendix).

This thesis demonstrates the benefits of agricultural CTWs for reducing N, P and SS inputs to downstream waterbodies, while illuminating how dynamic and complex internal processes can obscure obvious positive effects. Consideration of nutrient cycling and retentive processes must be carefully factored into CTW design and maintenance to ensure optimum pollutant removal performance. Additionally, effective monitoring of CTWs treating diffuse agricultural pollution requires sampling plans designed to include consideration of high leaching seasons and stochastic weather events, as well as appropriate sampling frequency and methodologies to measure key retention processes, detection of critical forms of N, P, and SS, and calculation of appropriate and applicable predictors of CTW performance.

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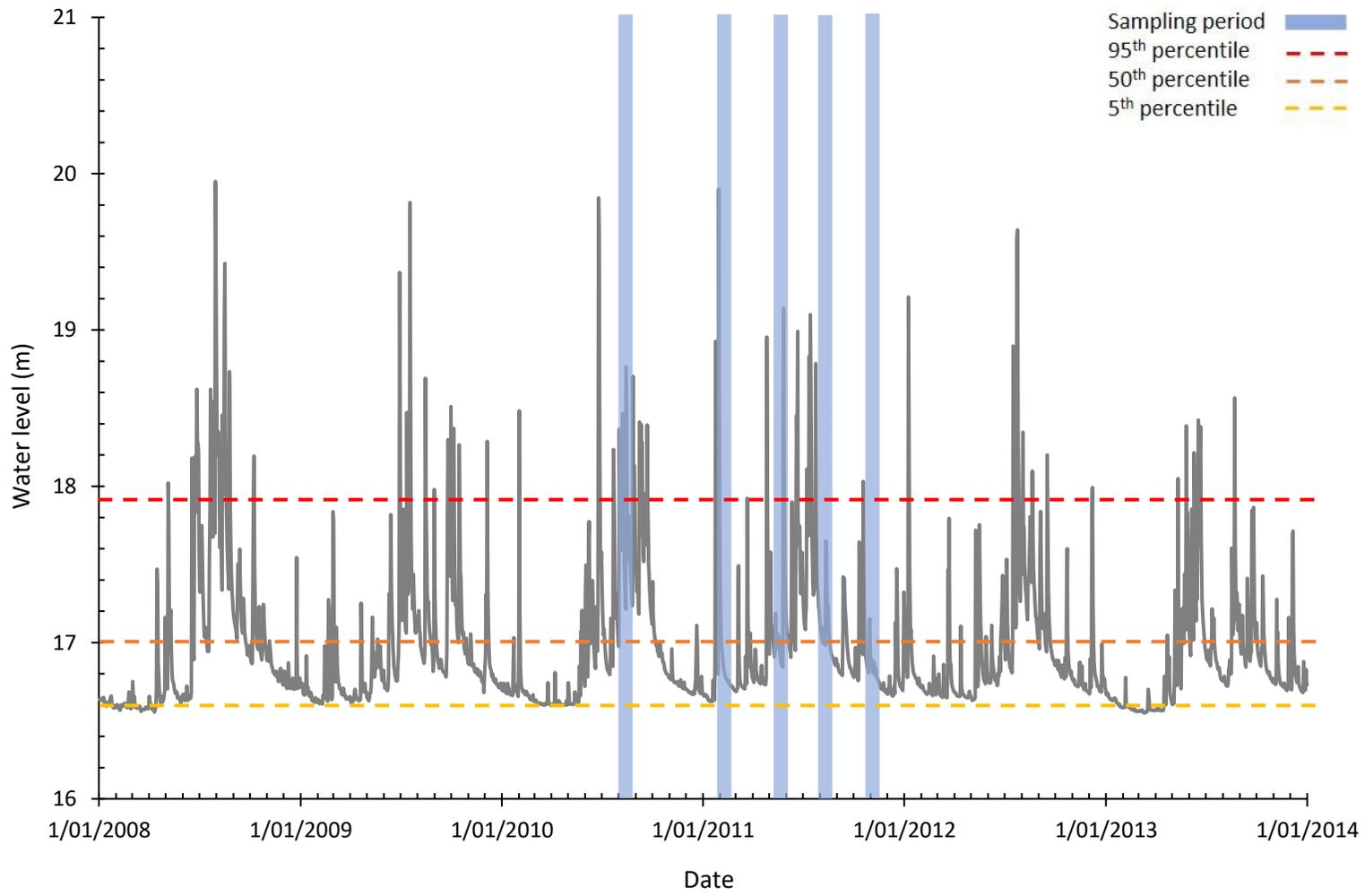
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5.5 SUPPLEMENTARY MATERIAL

5.5.1 Relative water level data

A



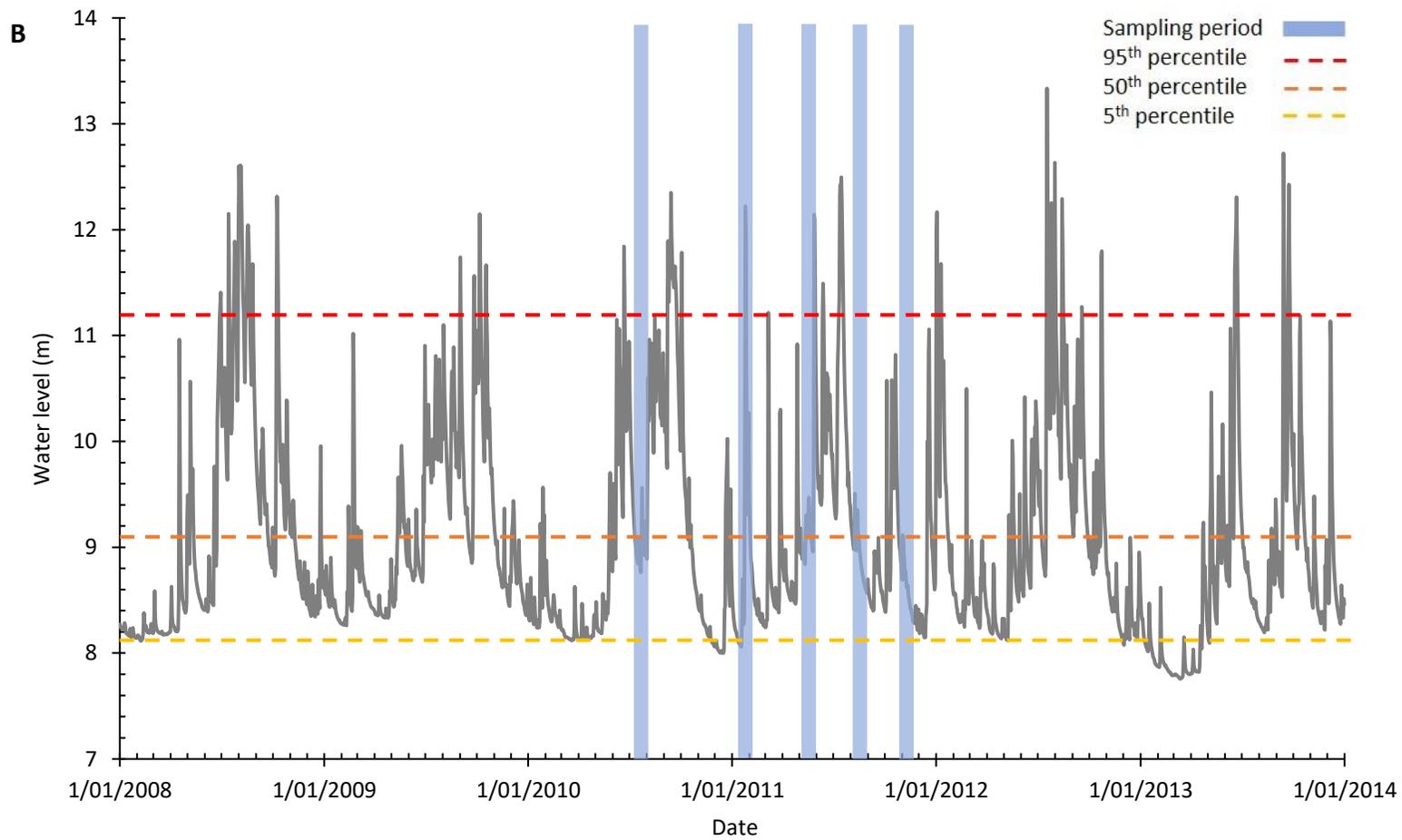


Figure SM 5.1 Water levels of the Mangawara Stream (A) and Puniu River (B) immediately north and south of the study area, respectively.

5.5.2 Comparison of subcatchment DAY

Table SM 5.1 Comparison of mean daily instantaneous yields (DAY, Kg ha⁻¹ day⁻¹) of ammonium (NH₄-N), nitrate (NO₃-N), particulate organic nitrogen (Org-N) and phosphate (PO₄-P) for subcatchments with clay loam, peaty loam, and deep peat dominated soil types within peat lake catchments of Kaituna, Kainui, Komakorau, and Koromatua. The landowners of each subcatchment are referred to as 'A', 'B', 'C' and 'D' to maintain their anonymity. The potential subcatchment nutrient sources stated were identified during the period of sampling (June 2010 – November 2011) therefore are not representative of any management or infrastructure changes which may have occurred subsequently on farms

Lake	Kaituna	Kaituna	Kainui	Kainui	Kainui	Komakorau	Koromatua	Kainui	Kainui	Kainui
Soil type	Clay loam	Clay loam	Clay loam	Clay loam	Peaty loam	Peaty loam	Peaty loam	Deep peat	Deep peat	Deep peat
Site	KT1	KT10-1	KN1	KN9	KN8	KO3	KR1	KN3	KN5	KN6
Subcatchment area (ha)	77.9	59.1	17.9	15.1	15.9	5.3	40.3	1.3	1.5	5.8
Mean DAY (Kg ha⁻¹ day⁻¹)										
NH₄-N	0.048	0.004	0.008	0.012	0.008	0.028	0.038	0.004	0.059	0.219
NO₃-N	0.270	0.126	0.153	1.062	0.741	1.364	0.060	0.009	0.002	0.130
Org-N	0.163	0.017	0.014	0.298	0.238	0.420	0.157	0.134	0.228	0.611
PO₄-P	0.002	0.000	0.002	0.008	0.001	0.033	0.224	0.049	0.107	0.135
Landowner	A	B	C	C	C	B	D	C	C	C
Potential subcatchment nutrient source(s)	Conventional management; cropping	Widespread use of BMPs; effluent irrigation	Silage stack; feed storage pit	Milking shed; effluent pond; sloped raceway flows to stream	Effluent irrigation	Effluent ponds with historic overflow to stream	Milking shed; cropping; effluent irrigation	Single linear channel drains 2 paddocks; ephemeral	Single linear channel drains 2 paddocks; ephemeral	Network of 4 channels drain 5 paddocks; spring fed

6 APPENDIX

6.1 GUIDANCE AND CONSIDERATIONS FOR DESIGN AND INSTALLATION OF CONSTRUCTED WETLANDS IN SHALLOW LAKE CATCHMENTS: WAIKATO REGIONAL COUNCIL INTERNAL TECHNICAL SERIES 2016/10

**Guidance & considerations for
design and installation of
constructed wetlands in shallow
lake catchments**

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Table of contents

EXECUTIVE SUMMARY	VI
1	INTRODUCTION
.....	1
1.1 What is best practice? 2	
2	THE WAIKATO REGIONAL PLAN
.....	3
3	PRE-DESIGN SCOPING CONSIDERATIONS
.....	4
4	TYPES OF CONSTRUCTED TREATMENT SYSTEM DESIGNS
.....	7
4.1 Filtration wetland 7	
4.2 Sedimentation ponds 8	
4.3 Open-water wetland 10	
4.4 Floating treatment wetland 12	
5 PRELIMINARY DESIGN GUIDELINES FOR CONSTRUCTED TREATMENT SYSTEMS	
.....	13
5.1 Filtration wetland 13	
5.1.1 Key design considerations:13	
5.1.2 Maintenance 14	
5.1.3 Schematic diagrams 14	
5.1.4 Tips for construction 15	
5.2 Sedimentation pond 16	
5.2.1 Key design considerations:16	
5.2.2 Maintenance 17	
5.2.3 Schematic diagrams 17	
5.2.4 Tips for construction 19	
5.3 Open-water wetland 22	
5.3.1 Key design considerations:22	

5.3.2	Maintenance	22
5.3.3	Schematic diagrams	23
5.3.4	Tips for construction	27
5.4	Floating treatment wetland	29
5.4.1	Key design considerations:	29
5.4.2	Maintenance	29
5.4.3	Schematic diagrams	29
5.4.4	Tips for construction	31
6DESIGNING TREATMENT SYSTEMS LARGE CATCHMENT (> 50 HA)	33
7SUMMARY AND RECOMMENDATIONS	36
7.1	Attenuation efficiency	36
7.2	Catchment discharge calculations	36
APPENDIX 1	DESKTOP SITE SCOPING SHEET	38
APPENDIX 2	SITE SCOPING FIELD SHEET	40
APPENDIX 3	DESKTOP INFORMATION GUIDANCE.....	43
APPENDIX 4	SITE DATA INPUT SPREADSHEET	46
APPENDIX 5	EXTERNAL PEER REVIEW COMMENTS AND RESPONSES	47
REFERENCES	53

Table of figures

- Figure 1 The sequence of steps recommended for effective use of the pre-design scoping tools and this guideline document for constructed treatment systems for surface water inflows to shallow lakes 6
- Figure 2 A schematic example of a filtration wetland created within an existing farm drain 8
- Figure 3 A schematic example of a silt-trap created adjacent to an existing farm drain 9
- Figure 4 A schematic example of an open-water wetland created at the end of a farm drain on the edge of a lake 11
- Figure 5 A schematic diagram of a floating treatment wetland 12
- Figure 6 Schematic diagrams to provide guidance for designing filtration wetlands. a) Plan view, b) cross sectional view, and c) cross sectional view with planting zones 15
- Figure 7 Schematic diagrams to provide guidance for designing sedimentation ponds. a) Plan view, b) cross sectional view, and c) plan view with planting zones 19
- Figure 8 Schematic diagrams to provide guidance for designing open-water wetlands; e) plan view with outlet channel and planting zones, and f) plan view with filtration outlet and planting zones 27
- Figure 9 Schematic diagrams to provide guidance for designing floating treatment wetlands. a) Plan view, and b) cross sectional views of the diversion channel and the excavated pond for the floating wetland raft 30
- Figure 10 Schematic diagrams illustrating the constructed wetland “treatment train” created in the Lake Ngaroto catchment. a) Plan view, and b) plan view with planting plan 35

Executive summary

This document has been prepared to assist interested parties including management agencies (Regional and District council, Department of Conservation, Fish&Game, NZ Landcare Trust) Iwi, and other stakeholders (landowners and care groups, consultants, agricultural advisors and contractors) with the design of constructed treatment systems to mitigate sediment and nutrient runoff from agricultural land within small (c.5 – 25 ha) lake catchments of the Waikato Region.

The content has been developed on the basis of recent (PhD) research, and practical construction experience at lakes Kainui, Kaituna, Komakorau, Koromatua, Serpentine North and South, and Lake Ngaroto. The effectiveness of these systems for attenuating nutrients and sediment has not been fully quantified, and it is anticipated that guidelines will be updated and modified on the basis of future monitoring and as best practice evolves.

In developing these guidelines, the following considerations have been taken into account:

- Minimising possible adverse effects on the environment;
- Compliance with the requirements of the Waikato Regional Plan;
- Practical and effective construction and maintenance;
- Reduction of nutrient and sediment losses from agricultural land use; and
- Enhancement of biodiversity values

These guidelines have been developed to address:

1. Pre-design considerations;
2. Types of constructed treatment system designs;
3. Design guidelines for effective constructed treatment systems;
4. Pre-design scoping tools:
 - a. Desktop Scoping Guidance (pre site visit);
 - b. Site Scoping Field Sheet (during site visit);
 - c. Desktop Information Guidance (post site visit); and
 - d. Site Data Input Spreadsheet.
5. Recommendations for large (> 50 ha) catchments.

Introduction

Diffuse pollution from intensive agricultural catchments has been associated with the of degradation waterway both within New Zealand (White 1983; Wilcock 1986) and internationally (Carpenter et al. 1998; U.S.EPA 2003). Intensive dairy farming has expanded throughout the world and New Zealand to meet growing global demand for milk solids (PCE 2013; UNEP 2013). Concomitant with the expansion and intensification of this land use in New Zealand the quality of receiving waters has declined due to elevated nutrient and sediment levels (MfE 2007; Baskaran et al. 2009; Quinn et al. 2009; PCE 2013).

Intensive dairy farming has been widespread in the lower Waikato region for more than fifty years and currently supports 30% of New Zealand's dairy herd (Statistics 2013). This region also includes the largest area of peatland in New Zealand, including thirty-one peat lakes (Lowe & Green 1987). Waikato peat lakes are shallow (< 6 m mean depth) and are naturally tea-coloured and acidic owing to the formerly extensive peat bogs from which they were formed (Clymo 1983; Shearer 1997). Prior to drainage and cultivation for agriculture the lakes would have had naturally low levels of productivity due to limited nutrient supply (Johnson & Gerbeaux 2004; Beard 2010) As a result these ecosystems are particularly vulnerable to elevated nutrient and sediment levels which cause eutrophication and may lead to toxic algal blooms (Hamilton et al. 2010).

Reducing N, P and SS inputs to lakes can help prevent algal blooms and improve water clarity and quality. Improving catchment and riparian management to decrease diffuse nutrient and sediment sources is strongly advocated by many regional and district councils, however this occurs less frequently in practice, particularly for small and ephemeral watercourses. Modified streams and farm drains have been shown to deliver high pollutant loads from intensive agricultural catchments to shallow Waikato lakes (Eivers et al. 2014; Tempero & Hamilton 2014).

Different treatment systems can be implemented to attenuate N, P and SS in surface waters draining agricultural catchments to reduce inputs to downstream lakes. These include sedimentation ponds and constructed wetlands which have been widely used to manage and process runoff from agricultural, urban and industrial sources throughout the world, proving to be effective in treating high levels of SS, N and P, as well as reducing metals, organics and pathogens (Kadlec & Wallace 2008). In New Zealand the use of constructed wetlands to reduce direct inflows of N, P and SS to downstream water bodies has become more widespread, although they have focused largely on dairy waste waters (Tanner et al. 1995; Tanner & Kloosterman 1997), subsurface drainage (Tanner et al. 2005; Tanner & Sukias 2011), or relatively large and costly constructed wetlands (Hudson et al. 2009). In the Waikato constructed wetlands created in aid of peat lake restoration have either not been monitored to gauge their effectiveness, or have yielded inconclusive results (Sukias et al. 2009).

These guidelines aim to assist the creation of effective sedimentation ponds and constructed wetlands targeted at small streams, drains and ephemeral watercourses draining to shallow peat lakes from intensive agricultural catchments in the Waikato region. The guidelines have been informed by research investigating the efficacy of constructed wetland and sedimentation ponds on inlet drains of six shallow Waikato peat lakes (Eivers In prep) and an extensive review of current literature from New Zealand and overseas.

What is best practice?

Best practice reflects the most recent, up to date and innovative practice that contributes to the enhancement and improvement of the existing environment. The final developed guidelines are intended to assist those involved with the design, installation and maintenance of constructed treatment systems for the reduction of sediment and nutrients losses to waterways from agricultural land use. The final guidelines shall help to ensure the most appropriate treatment systems are installed and maintained effectively whilst protecting and enhancing the values of the surrounding environment including associated streams, lakes and wetlands.

The final guidelines may also assist those involved with authorising and auditing constructed treatment systems within the Waikato Region, as they provide a set of designs from which to make comparisons with.

The Waikato Regional Plan

The Waikato Regional Plan has been developed by the Waikato Regional Council under the Resource Management Act 1991, and is intended to provide direction regarding the use, development and protection of natural and physical resources in the Waikato region. The proposed Plan contains modules covering Matters of Significance to Maori, Water, River and Lake Beds, Land and Soil, Air, and Geothermal Resources.

The plan contains rules that mediate various activities in the Waikato Region, including the construction of treatment systems such as sedimentation ponds and constructed wetlands as management tools for improving water quality and assisting ecosystem restoration.

Pre-design scoping considerations

Careful consideration should be given to the characteristics of different watercourses relating to catchment, land use intensity, climate and hydrology as these parameters will determine the appropriate type (or combination of) treatment system design.

Fixed and variable inputs will influence the hydrology of a site as well as nutrient and sediment characteristics. Fixed inputs relate to the **catchment** (including area, elevation, slope, soil type, and watercourse length) and **climate** (including precipitation, temperature, wind and evaporation).

Variable inputs relate to land-use intensity and include diffuse pollutant sources (fertiliser use, effluent application, stocking rate, irrigation and riparian management) and point sources (associated with effluent pond and tile drain discharges, as well as runoff from nutrient “hot spots” e.g. milking sheds, feed-pad, feed storage areas, troughs, shade trees, bridges, underpasses and race-ways).

Integral to the choice of an appropriate location along the watercourse, and the type and design of treatment, is the consideration of ongoing maintenance. Sedimentation ponds and constructed wetlands are management tools that require on-going maintenance to ensure continued effectiveness and performance. As N, P and SS accumulate within sedimentation ponds and constructed wetlands, their attenuation capacity will decline. Sediment build-up will occur more rapidly than N and P in sites therefore removing trapped sediment and sludge will be required. Choosing a location in the catchment of the watercourse that can be readily accessed by tractor, trailer and excavator is essential to allow for sediment removal. For example, creating a sedimentation pond near a maintained stock race and leaving specified areas around the pond unplanted with shrubs and trees to allow excavator access.

Pre-design scoping tools are provided in Appendices 1, 2, 3 and 4, and include:

1. A desktop scoping guidance sheet for use *before* the site visit;

2. A scoping field sheet for use *during* the site visit;
3. A desktop information guidance sheet for use *after* the site visit; and
4. A site data input spreadsheet into which data collected is entered and collated. This datasheet will assist the user with selecting the appropriate treatment systems design(s) for the site.

Note: Complete a separate scoping field sheet and data input spreadsheets for *each* potential site identified during the field visit.

The flow diagram in Figure 6.1 illustrates the sequence of steps recommended for effective use of the pre-design scoping tools and these preliminary guidelines for constructed treatment systems for surface water inflows to shallow lakes.

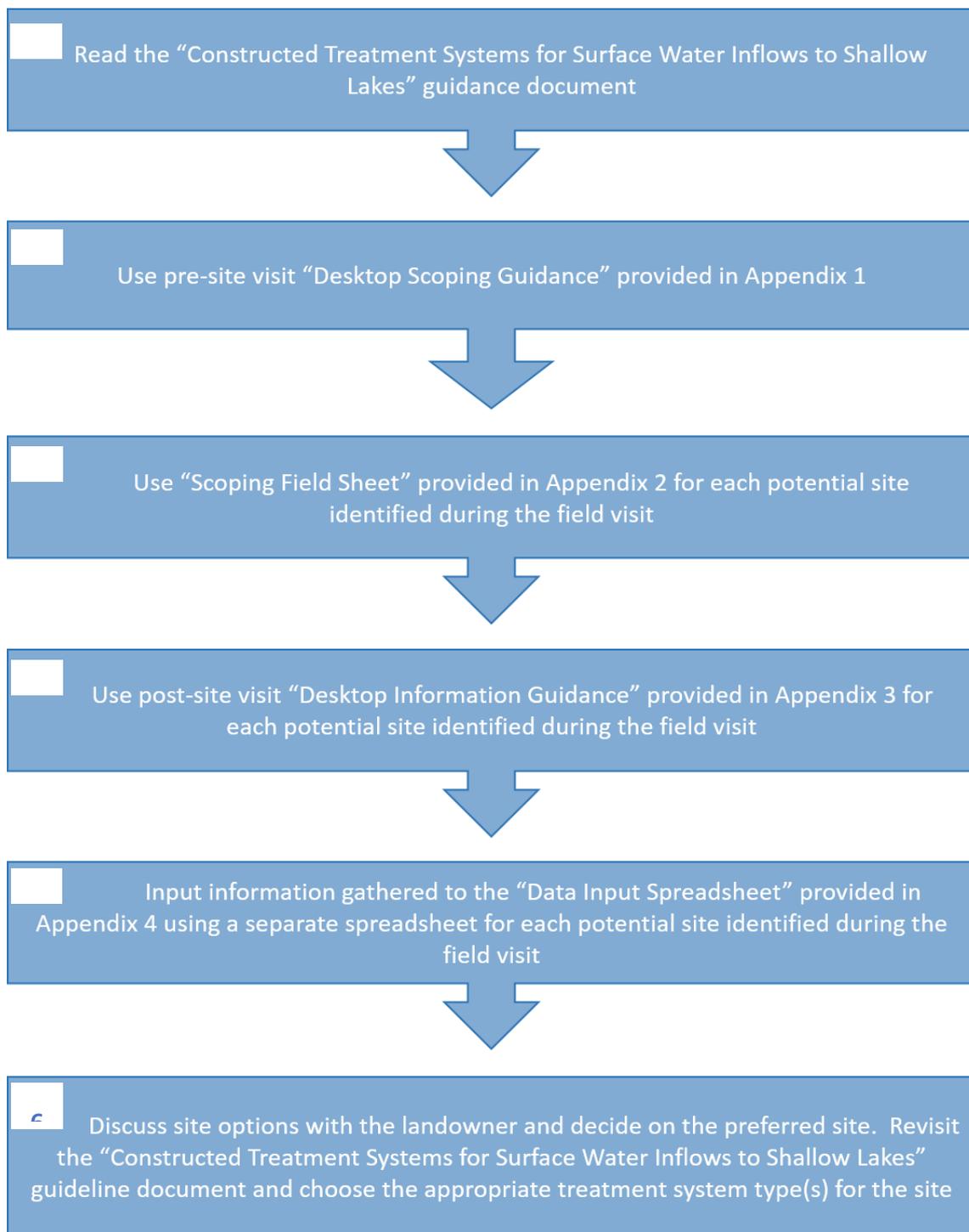


Figure 6.1 The sequence of steps recommended for effective use of the pre-design scoping tools and this guideline document for constructed treatment systems for surface water inflows to shallow lakes

Types of constructed treatment system designs

The constructed treatment system selected will depend on the type(s) of pollutants targeted for attenuation as defined by the objectives of the project as well as the physical and environmental factors of each site. For example, if the project aim is to retain coarse and fine suspended sediment within a limited area then a sedimentation pond and floating wetland treatment system would be most appropriate. In many cases the objectives of the project may be to attenuate coarse and fine sediments as well as particulate and dissolved forms of nitrogen and phosphorus, which would involve a treatment train comprising sedimentation pond(s), floating, open-water, and filtration wetlands.

Filtration wetland

Filtration wetlands are treatment systems suitable for small catchments ($\sim \leq 5$ ha) with intermittent streams or drains, that are best created in low lying areas or hollows with shallow water tables and low gradients (Figure 6.2). The purpose of a filtration wetland is to filter out fine particles of sediment and particulate forms of N and P, enhance N uptake by plants and algae, improve denitrification during low flows and under anoxic conditions, whilst allowing for effective drainage of adjacent productive land. A number of studies have demonstrated the capacity of riparian areas and wetland environments to assimilate N and reduce losses to downstream water bodies (Cooke & Cooper 1988; Cooper 1990; Burns & Nguyen 2002).

A filtration wetland should have shallow water depths (0.1 m at base flow, 0.3 m at high flows) and a wide channel (2 – 4 m) densely planted with native wetland plants including grasses, rushes and sedges. The depth and width chosen must ensure an equivalent cross-sectional area for flow is maintained to allow for adequate drainage. For example, an existing channel 0.5 m deep and 1 m wide has a cross-sectional area of 0.5 m^2 , therefore a filtration wetland with a depth of 0.25 m must be at least 2 m wide to retain a cross-sectional area of 0.5 m^2 .

Care must be taken to ensure the filtration wetland allows for adequate drainage. Once established, the vegetation will have a greater resistance to flow than a clear channel. Increasing the cross-sectional area to facilitate drainage is recommended. Trials of this design concept are recommended.



Figure 6.2 A schematic example of *n* filtration wetland created within an existing farm drain

Sedimentation ponds

Sedimentation ponds are treatment systems designed to attenuate suspended sediment from watercourses transporting agricultural runoff for the purpose of reducing sediment accumulation in downstream water bodies (Figure 6.3).

Sedimentation ponds are best suited to small to medium sized catchments (~5 – 25 ha). For larger catchments, a number of sedimentation ponds in series will be more appropriate with the number relative to the size and hydrology of the catchment.

Sedimentation ponds can benefit both the aquatic environment and land owners, as sediment captured in the treatment system can be used as a source of fertiliser when they are routinely cleaned. If sedimentation ponds are constructed according to the suggested design guidelines they can effectively attenuate sediment whilst maintaining drainage and ensuring adjacent land remains productive.



Figure 6.3 *A schematic example of a silt-trap created adjacent to an existing farm drain*

Open-water wetland

Open-water wetlands, also termed surface-flow or free-water surface wetlands (Kadlec and Wallace, 2008) are designed to enhance nutrient uptake by the aquatic environment by improving habitat for flora and fauna, thereby enhancing biodiversity and ecosystem services (Kadlec 1995; Mitsch & Gosselink 2007). Ecosystem services include denitrification, increased solar exposure of faecal microbes, and nutrient cycling and uptake via food-web processes. Open-water wetlands are best suited to small to medium sized catchments (~5 – 25 ha). For larger catchments, a number of open-water wetlands in series will be more appropriate with the number relative to the size and hydrology of the catchment. Open-water wetlands should be generally placed in low lying areas with shallow water tables, low gradients and low to medium flow rates. The area of an open-water wetland will depend on practical and environmental constraints as well as the size of the catchment, however depths should range from 0.3 – 1.8 m. Different depths support different wetland and aquatic plants whilst helping to reduce draught stress (Raulings et al. 2010). Native shrubs and trees planted along riparian margins will provide shade necessary to keep water temperatures cool. Matheson and Sukias (2010) have shown effective N reduction through denitrification occurring in a 260 m² open-water wetland treating farm drainage waters at Toenepi, New Zealand.

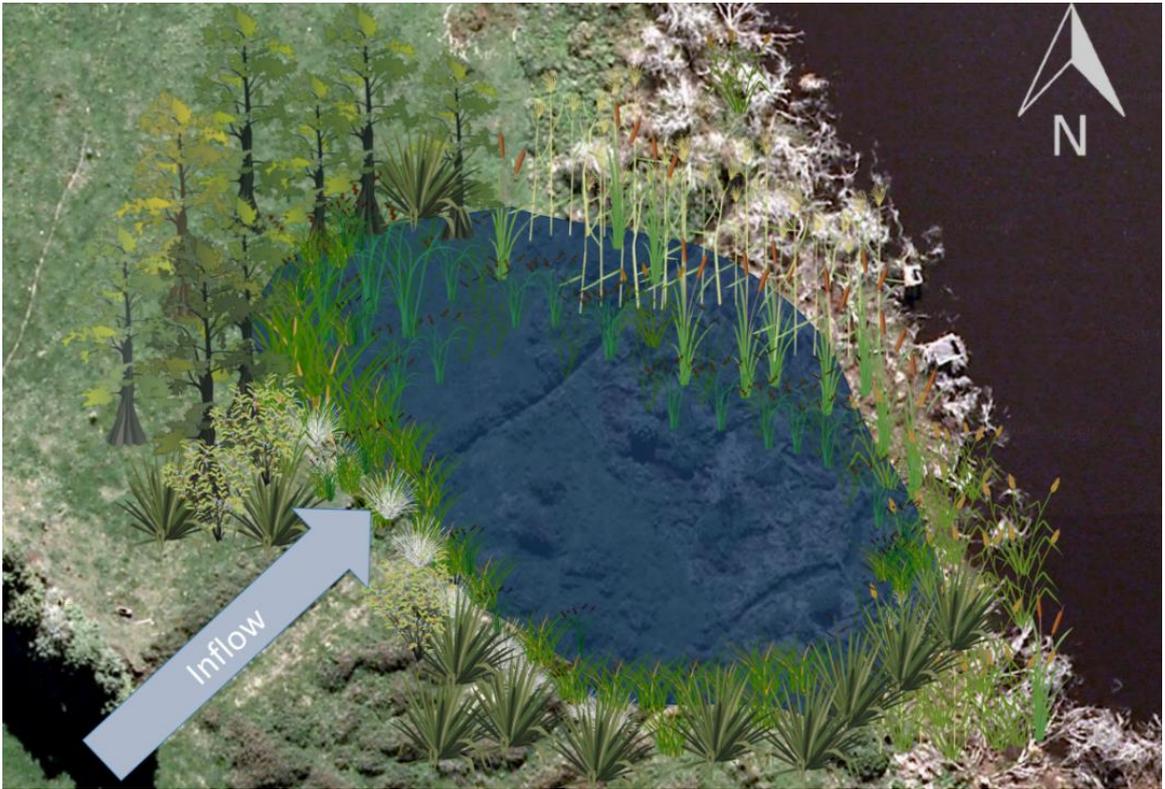


Figure 6.4 *A schematic example of an open-water wetland created at the end of a farm drain on the edge of a lake*

Floating treatment wetland

Floating treatment wetlands (FTW) consist of floating rafts through which aquatic or wetland plants grow, their roots emerging beneath the raft in the water as shown in Figure 6.5 (Tanner & Sukias 2011). FTWs remove nutrients and fine particles from the water by entrapment and uptake via the plants root systems, as well as through bacterial and algal biofilms on the root hairs and the matrix of the floating raft. FTWs provide shade which helps to cool the water and provide shelter for aquatic fauna. Trout and koura have been observed using FTWs as habitat in a number of lakes where they have been trialled in the Bay of Plenty (Rotorua-Lakes 2014).

FTWs are best suited to medium sized catchments ($\sim 25\text{-}50$ ha) with low to medium flows (\sim average velocity $< 0.15\text{ ms}^{-1}$, \sim average discharge $< 0.025\text{ m}^3\text{ s}^{-1}$). Correct flow rates are important to ensure an adequate water residence time within the FTW. If flow rates are too high, the contact time between the water and the FTW will not be long enough to allow for effective nutrient uptake by the plant roots and biofilms. If flows are too low, the water beneath the FTW can become stagnant, leading to anoxic conditions. Anoxic sediments can release nutrients such as phosphorus and ammonium into the water column thereby reducing the efficacy of the FTW.

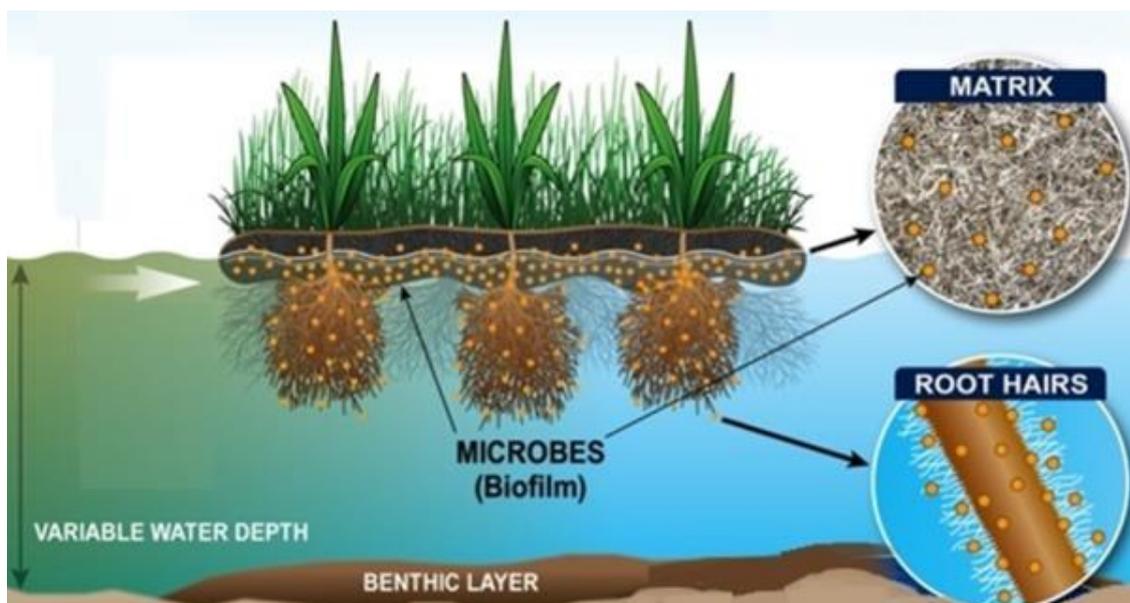


Figure 6.5 A schematic diagram of a floating treatment wetland

Preliminary design guidelines for constructed treatment systems

Filtration wetland

Key design considerations:

- i. Placement

Within ephemeral watercourses or low lying areas with low gradient and shallow water table
- ii. Water depth 0.1 to 0.3 m
- iii. Width 2 to 4 m (adjust ensure cross sectional flow area of the channel is maintained)
- iv. Flow rates Ephemeral to low flows (\sim velocity ≤ 0.05 ms^{-1} , \sim discharge ≤ 0.005 $\text{m}^3 \text{s}^{-1}$)
- v. Plants

Order plants well in advance, preferably from local suppliers who source plants or seeds locally, or from within the same or similar ecological district. Larger plants are preferable (PB3) as they will establish more rapidly, provide cover more quickly and require less weed maintenance. Ideally planting should occur during spring and early summer (September to December) to ensure correct water levels for species and allow for plants to fully utilise the growing season.

- a. **Inner Wetland** – Space 4 plants/ m^2 (0.5 m centres) to adequate depths and stake in with a bamboo stake or similar to avoid uprooting by waterfowl or floating away in higher flows.

Species: *Carex virgata*, *C. secta*, *Machaerina articulata*, *Isolepis prolifera*, *typha orientalis*, *Eleocharis acuta*, *Schoenoplectus tabernaemontani*

- b. **Wetland margin**

Species: *Austroderia toetoe*, *Carex geminata*, *C. lessonia*, *C. subdola*, *C. sinclairii*, *Coprosma propinqua*, *C. tenuicaulis*, *Cyperus ustulatus*, *Isachne globosa*, *Juncus pallidus*, *Macherina rubiginosa*, *Macherina arthrophylla*, *Phormium tenax*

- c. **Riparian zone**

Species: *Aristotelia serrata*, *Coprosma lucida*, *Cordyline australis*, *Hebe stricta*, *Kunzea ericoides*, *Leptospermum scoparium*, *Meliccytus ramiflorus*, *Myrsine australis*, *Plagianthus divaricatus*

Maintenance

- i. **During establishment** – Ensure the site is well fenced to exclude stock. If waterfowl may be a problem during plant establishment, a low hot wire may be necessary. Initial fortnightly visual inspections will be required by the landowner to check for damage from waterfowl or livestock, stressed or dead plants, and weed encroachment. Hand weeding or spot spraying with suitable herbicides may be necessary.
- ii. **Operational** - Following the first year of growth, regular spring maintenance should be established during which infill planting, weed control and checks for embankment erosion are carried out. The effectiveness of the filtration wetland is dependent on well established, wetland plants that provide good cover and filtration across the entire area of the wetland. Operational maintenance is therefore essential.

Schematic diagrams

The following diagrams provide guidance for designing filtration wetlands. The plan view (Figure 6.6 a) shows the required minimum **doubling of the width** of the existing channel, and the area to be densely planted within the wetland. The cross sectional view (Figure 6.6 b) shows the cross sectional flow area is maintained by increasing the channel width and reducing the channel depth. The cross sectional view with planting zones (Figure 6.6 c) is to assist with planting the recommended plants (outlined above in point 6.1.1.1 v) in the correct areas.

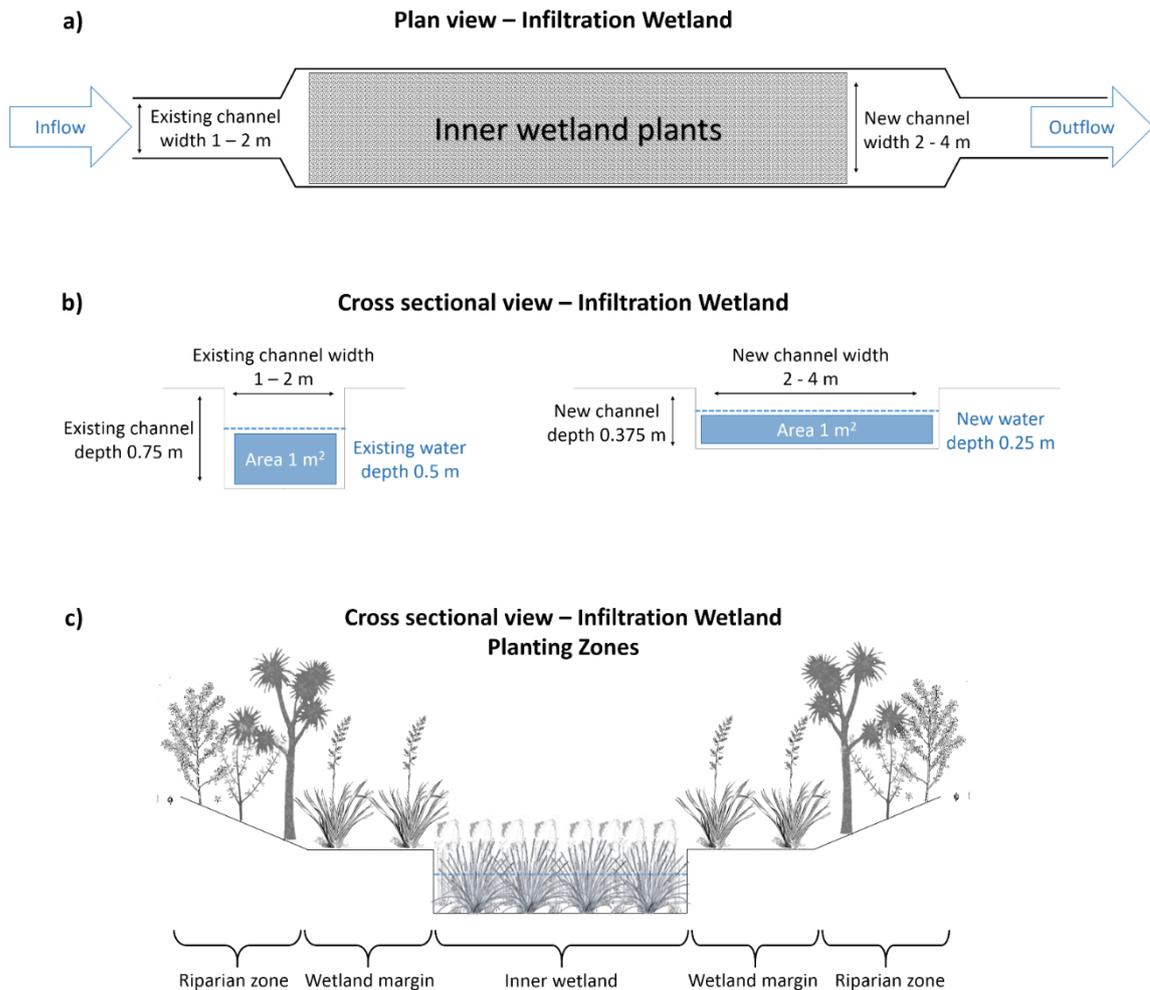


Figure 6.6 Schematic diagrams to provide guidance for designing filtration wetlands. a) Plan view, b) cross sectional view, and c) cross sectional view with planting zones

Tips for construction

- i. Construction should occur during periods of low flows in early summer to allow plants to establish before autumn. Alternatively, filtration wetlands can be excavated during late summer when watercourses are dry and planted in early autumn.
- ii. A fish survey will be required to determine the presence of native fish species in the watercourse prior to construction, preferably during late winter to early spring. If native fish are caught, then trapping must be carried out immediately before construction works begin with fish caught held for the period of disturbance, and subsequently released upstream. If the watercourse is dry at the time of construction, a black mudfish search (if caught previously) will be required to ensure none are aestivating in the damp channel.
- iii. Excavation of the watercourse should commence from the downstream end of the filtration filter, working in an upstream direction.
- iv. Excavated soil can be used on site to fill in holes or dips in fields around the farm. A tractor a trailer unit would be adequate to disperse excavated soil.
- v. Complete planting of the filtration wetland at the suggested times adhering to the recommended planting zones depicted in Figure 6.6 c.

Sedimentation pond

Key design considerations:

i. Placement

Near to sediment runoff “hotspots” in low lying areas adjacent to watercourses with access to well maintained stock races or farm tracks

ii. Depth 1.5 – 1.8 m

iii. Width 8 - 10 m diameter

iv. Flow rates Low to medium (average velocity $\leq 0.15 \text{ ms}^{-1}$, average discharge $\leq 0.025 \text{ m}^3 \text{ s}^{-1}$)

v. Plants

Order plants well in advance, preferably from local suppliers who source plants or seeds locally, or from within the same or similar ecological district. Larger plants are preferable (PB3) as they will establish more rapidly, provide cover more quickly and require less weed maintenance. Ideally planting should occur during spring and early summer (September to December) so plants can fully utilise the growing season, although autumn planting is also suitable. The purpose of the plants is to provide shade to help keep the water cool within the sedimentation pond, and provide filtration of surface water over riparian areas during flood events. It is essential to leave area(s) unplanted **to allow excavator access for maintenance.**

a. **Open-water zone**

Species: *Potamogeton ochreatus*, *Myriophyllum propinquum* (refer Martin & Reeves 2013 for guidance on establishing submerged macrophytes in ponds and wetlands)

b. **Sedimentation pond margin**

Species: *Carex lessonia*, *C. geminata*, *Gahnia xanthocarpa*, *Juncus pallidus*, *J. edgariae*, *Phormium tenax*, *Austroderia toetoe*, *Cyperus ustulatus*

c. **Bund**

Species: *Exotic pasture grass*

d. **Riparian zone**

Species: *Aristotelia serrata*, *Coprosma lucida*, *Cordyline australis*, *Hebe stricta*, *Kunzea ericoides*, *Leptospermum scoparium*, *Melicytus ramiflorus*, *Myrsine australis*, *Plagianthus divaricatus*

Maintenance

- i. **During establishment** – Ensure the site is well fenced to exclude stock. If waterfowl may be a problem during plant establishment, a low hot wire may be necessary. Initial fortnightly visual inspections will be required by the landowner to check for damage from waterfowl or livestock, stressed or dead plants, and weed encroachment. Hand weeding or spot spraying with suitable herbicides may be necessary.
- ii. **Operational** - Following the first year of growth, regular spring maintenance should be established during which infill planting, weed control and checks for embankment erosion are carried out. Accumulated sediment and sludge will require routine cleaning, the regularity of which will be site specific, depending on the SS loads transported by the watercourse.

Cleaning the sedimentation pond will require a medium to large excavator with a boom reach of appropriate length (e.g. 8 – 10 m) to ensure the pond can be cleaned to the original excavated depth and width. Removed sediment and sludge can be transported away and spread on adjacent fields either using a tractor and trailer, or a manure spreader or “honey wagon”. Nutrient analyses of the “sludge” removed is recommended to enable fertiliser estimates of the distributed sludge.

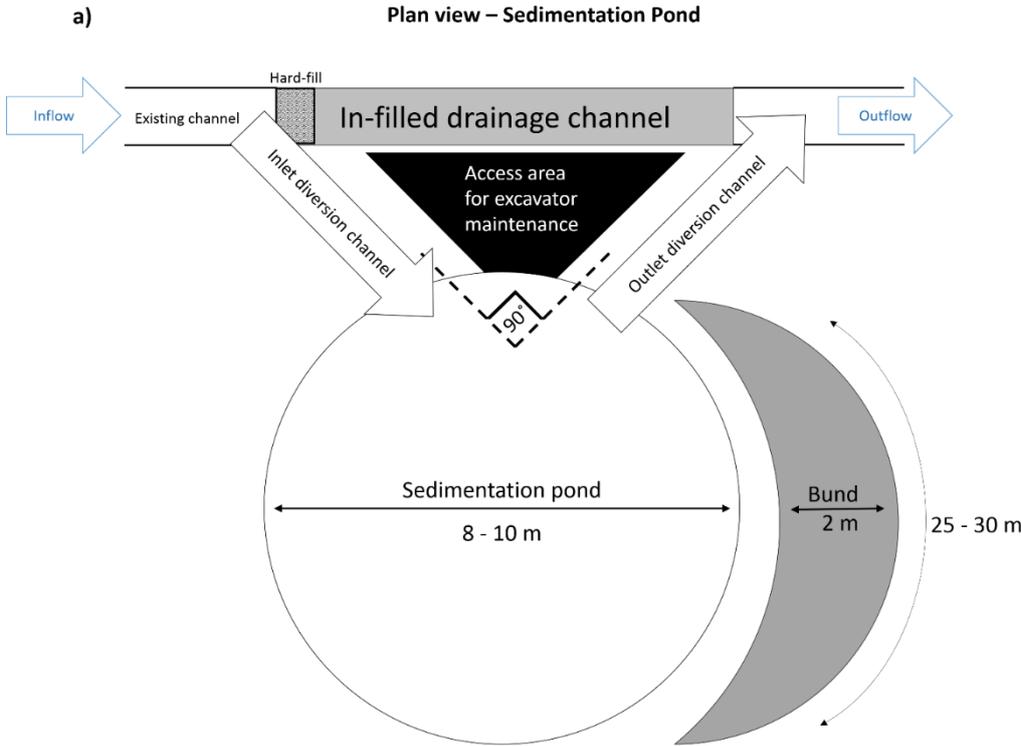
Using a measuring staff to monitor sediment deposition is recommended as the additional thickness of the staff as compared to a long ruler can help the user *feel* the difference between the sludge, accumulated sediment and the bottom of the sedimentation pond. A kayak or small boat (dingy or inflatable) can be used to monitor sediment accumulation, however if deemed unpractical, a float tube with waders may be another option.

Schematic diagrams

The following diagrams provide guidance for effective sedimentation pond designs. The plan view (Figure 6.7 a) shows the layout of the treatment system relative to the channel of the existing watercourse. Included is the sedimentation pond, inlet and outlet diversion channels, flood protection bund, the area required for excavator access, and the area of the existing channel to be infilled. The cross sectional view (Figure 6.7 b) shows the recommended channel depth and width for the diversion

channels, the diameter, depth and batter slope for the edges of the pond, and the width, height and batter slope for the bund. The horizontal:vertical ratio for the batter edges of the sedimentation pond will depend on the stability and condition of the soil. Heavy clay and loosely consolidated moist soils such as peat require a flatter slope (e.g. 1:1) whilst steeper slopes ($\frac{1}{2}$:1) are suitable for well cemented soils. The contractor and designer will need to agree on the most appropriate batter slope depending on the soil conditions of the site.

A flatter slope for the bund batter will help promote vegetation growth. The plan view with planting zones (Figure 6.7 c) is to assist with planting recommended plants (outlined above in point 6.2.1 v) in the correct areas.



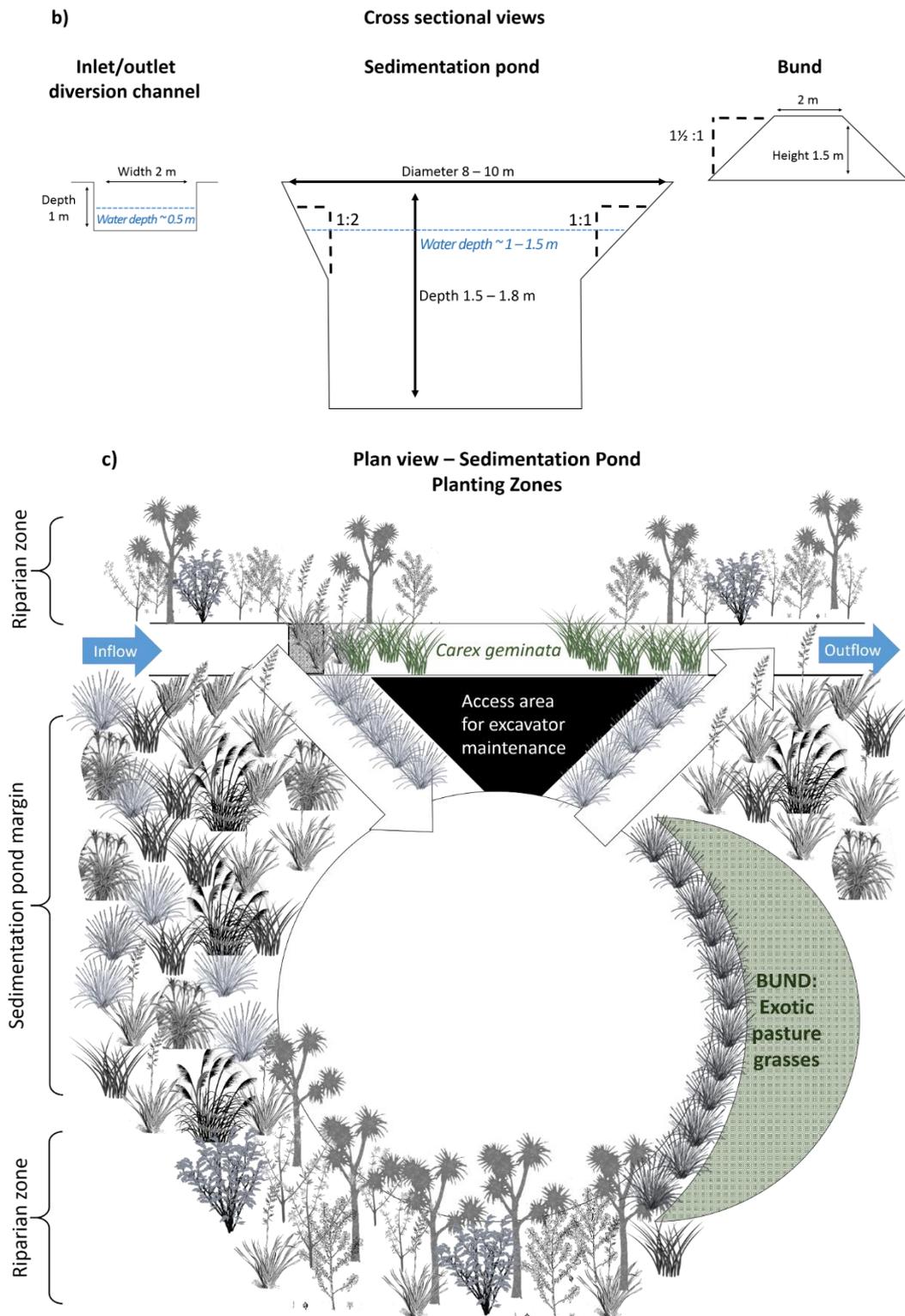


Figure 6.7 Schematic diagrams to provide guidance for designing sedimentation ponds. a) Plan view, b) cross sectional view, and c) plan view with planting zones

Tips for construction

- i. Construction should occur during lowest flows in late summer to early autumn (February – March).
- ii. A fish survey will be required to determine the presence of native fish species in the watercourse prior to construction, preferably during late winter to early spring. If native fish are caught, then trapping must be carried out immediately

- before construction works begin with fish caught held for the period of disturbance, and subsequently released upstream. If the watercourse is dry at the time of construction, a black mudfish search (if caught previously) will be required to ensure none are aestivating in the damp channel.
- iii. Prior to the commencement of works, the treatment system design should be stepped out, measured and marked using paint with the contractor. It is essential to orient the inlet and outlet diversion channels at 90° from each other (refer Figure 6.7 a) as this design promotes back eddies within the pond, increasing the residence time of the water and promoting sediment settling. Additionally, ensure an area large enough for an excavator to access the sedimentation pond for future maintenance is retained (refer Figure 6.7 a, c). Two access areas may be required depending on the size of the sedimentation pond, and the reach of available excavators.
 - iv. Excavation of the treatment system should begin with the sedimentation pond. Regular measurements should be made during the excavation to ensure the correct dimensions of the sedimentation pond and batter slopes are achieved. The spoil removed should first be used to create the bund, then stored alongside the existing channel to be used as fill at the completion of works. Excess excavated soil can be used around the farm to fill in holes or dips in fields using a tractor and trailer unit.
 - v. Secondly, excavate the outlet diversion channel, and thirdly the inlet diversion channel beginning at the downstream end and moving upstream, being careful not to breach the bank between the existing channel and the diversion channel at this point.
 - vi. Before damming the existing channel and allowing water to flow into the treatment system, check the levels of the inlet and outlet diversion channels with the contractor using a dumpy level or equivalent to ensure flows will move downstream effectively.
 - vii. Once satisfied with levels, dam the existing channel immediately below the start of the inlet diversion using hard-fill as shown in Figure 6.7 a. It is essential to use hard-fill material at the upstream end of the existing to create an effective diversion and prevent erosion. Old concrete troughs or equivalent can be used from around the farm property. Use the spoil stored alongside the channel to effectively seal the dam.
 - viii. Once the water in the existing channel begins to pool behind the dam, excavate an opening in the bank to allow the water to flow into the inlet diversion channel. It can take some time for the sedimentation pond to fill and begin flowing out the diversion channel. This will vary from site to site, sometimes taking minutes, hours or days.
 - ix. Finally, fill in the existing drainage channel below the hard-fill dam as shown in Figure 6.7 a. It is recommended to over fill the drain as the level will drop over time as the soil settles and compacts. The final height of the in-filled channel should be slightly below the level of the adjacent field to allow for drainage during floods, and so as to act as a high-flow bypass during heavy rainfall events.

- x. Complete planting of the sedimentation pond treatment system at the suggested times adhering to the recommended planting zones depicted in Figure 6.7 c.

Open-water wetland

Key design considerations:

i. Placement

Low lying areas with low gradient and shallow water table. Open-water wetlands may be created adjacent to existing watercourses, with flows diverted into and out of the wetland. Alternatively, open-water wetlands may be created at the “end” of an existing watercourse, with water diverted into the wetland and allowed to “infiltrate” through the wetland margin and riparian zone to the downstream waterbody (refer Figure 6.4).

ii. Water depth 0.3 m (shallow zone) – 1.8 m (open-water zone)

iii. Width Minimum 5 m diameter including open-water and shallow wetland zones

iv. Flow rates Low to medium (average velocity $\leq 0.15 \text{ ms}^{-1}$, average discharge $\leq 0.025 \text{ m}^3 \text{ s}^{-1}$)

v. Plants

a. **Open-water zone**

Species: *Lemna minor*, *Potamogeton ochreatus*, *Myriophyllum propinquum* (refer Martin & Reeves 2013 for guidance on establishing submerged macrophytes in ponds and wetlands)

b. **Shallow zone**

Species: *Macherina articulata*, *Eleocharis sphacelata*, *Typha orientalis*, *Schoenoplectus tabernaemontani*

c. **Wetland margin**

Species: *Austroderia toetoe*, *Carex geminata*, *C. lessonia*, *C. subdola*, *C. sinclairii*, *Coprosma propinqua*, *C. tenuicaulis*, *Cyperus ustulatus*, *Isachne globosa*, *Juncus pallidus*, *Macherina rubiginosa*, *Macherina arthropphylla*, *Phormium tenax*

d. **Riparian zone**

Species: *Aristolelia serrata*, *Coprosma lucida*, *Cordyline australis*, *Hebe stricta*, *Kunzea ericoides*, *Leptospermum scoparium*, *Meliccytus ramiflorus*, *Myrsine australis*, *Plagianthus divaricatus*

Maintenance

- i. **During establishment** – Ensure the site is well fenced to exclude stock. If waterfowl may be a problem during plant establishment, a low hot wire may be

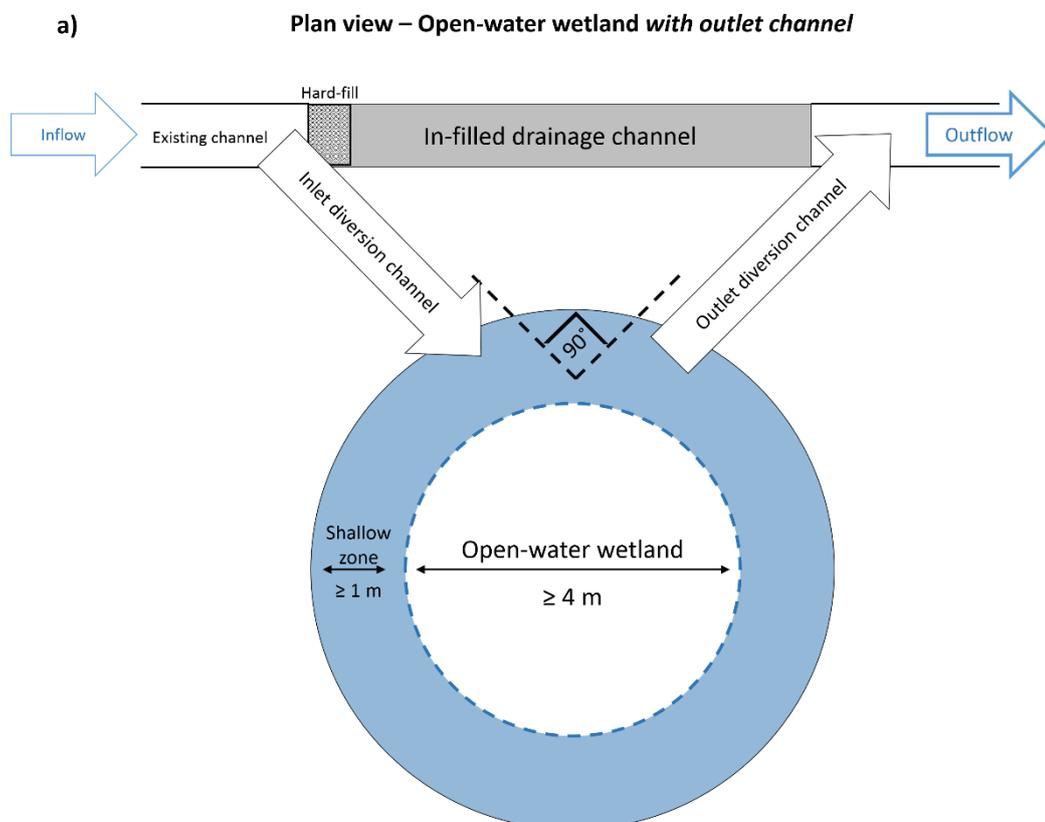
necessary. Initial fortnightly visual inspections will be required by the landowner to check for damage from waterfowl or livestock, stressed or dead plants, and weed encroachment. Hand weeding or spot spraying with suitable herbicides may be necessary.

- ii. **Operational** - Following the first year of growth, regular spring maintenance should be established during which infill planting, weed control and checks for embankment erosion are carried out. The effectiveness of the wetland will be enhanced by well-established wetland plants providing habitat diversity and good cover across all zones of the wetland. Maintenance, including removal of some wetland plants, will be required to ensure the open-water doesn't become colonised by species such as *E. sphacelata*, *T. orientalis*, *S. tabernaemontani*.

Schematic diagrams

The following diagrams provide guidance for effective open-water wetland designs.

The first plan view (Figure 6.10 a) shows the layout of the treatment system relative to the channel of the existing watercourse. Included is the open-water wetland, inlet and outlet diversion channels, and the area of the existing channel to be infilled. The second plan view (Figure 6.10 b) shows an open-water wetland without an outlet channel, instead the water *infiltrates* through the wetland margin and riparian zones to the downstream waterbody.



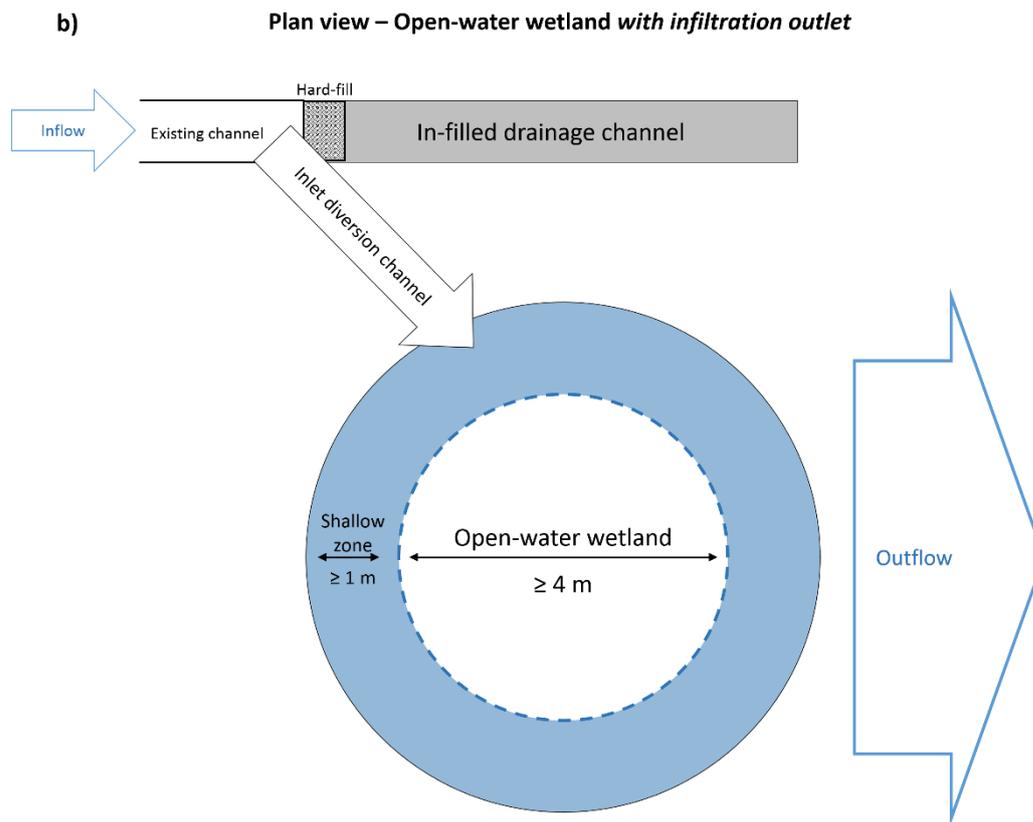


Figure 6.8 Schematic diagrams to provide guidance for designing open-water wetlands; a) Plan view with outlet channel, b) plan view with filtration outlet.

The first cross sectional view (Figure 6.10 c) shows the recommended channel depth and width for the diversion channels, the open-water and shallow zone diameters and depths, and recommended batter slopes for the edges of the wetland. The horizontal:vertical ratio for the batter edges of the wetland will depend on the stability and condition of the soil. Heavy clay and loosely consolidated moist soils such as peat require a flatter slope (e.g. 1:1) whilst steeper slopes ($\frac{1}{2}$:1) are suitable for well cemented soils. The contractor and designer will need to agree on the most appropriate batter slope depending on the soil conditions of the site.

The second cross sectional view (Figure 6.10 d) as well as the additional plan views (Figure 6.10 e, f) show planting zones to assist with planting recommended plants (outlined above in point 6.3.1 v) in the correct areas.

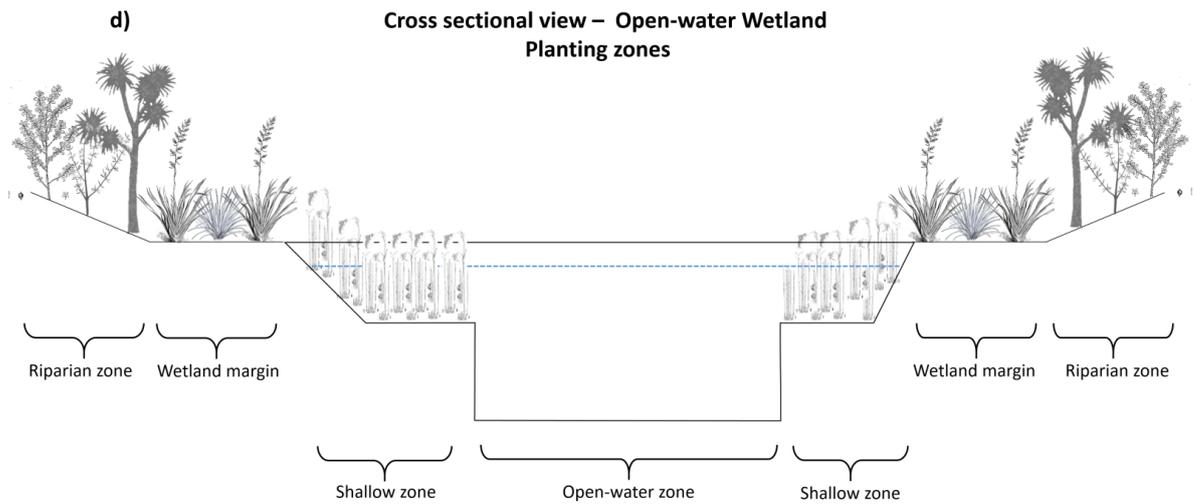
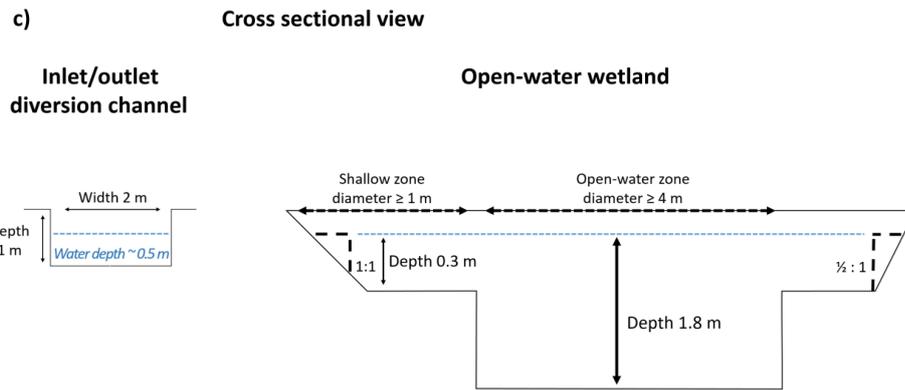
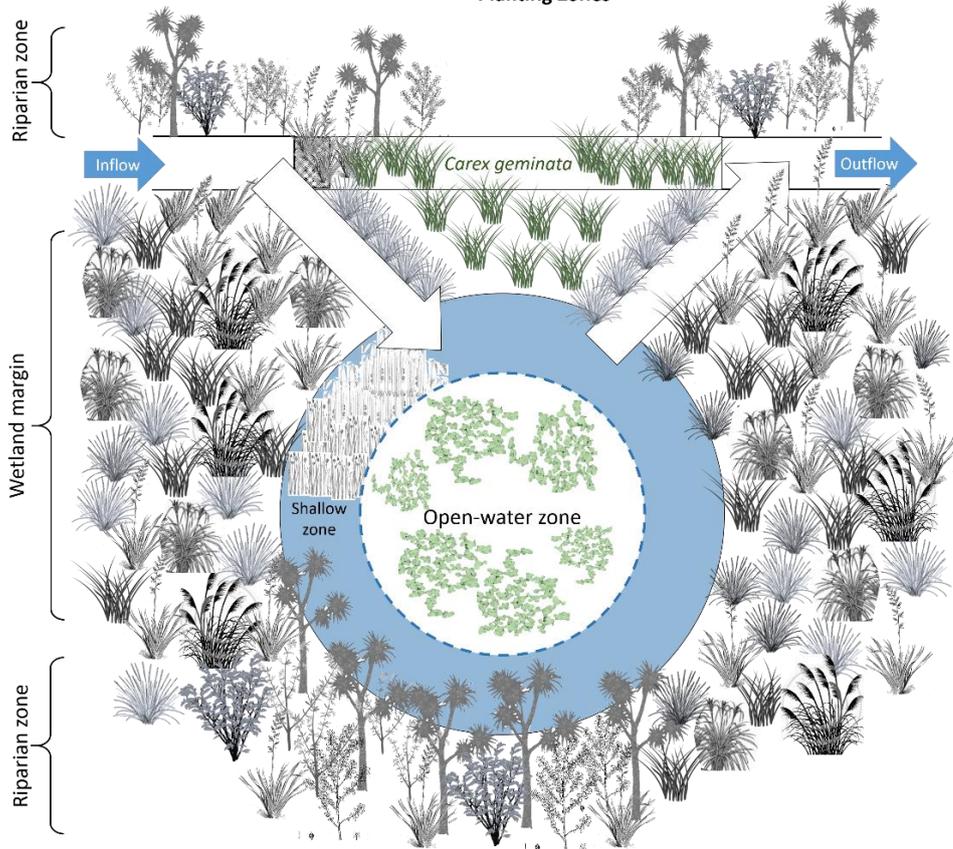


Figure 6.9 Schematic diagrams to provide guidance for designing open-water wetlands; c) cross sectional view, d) cross sectional view and planting zones

e) Plan view – Open-water wetland with outlet channel
Planting Zones



f) Plan view – Open-water wetland with infiltration outlet
Planting Zones

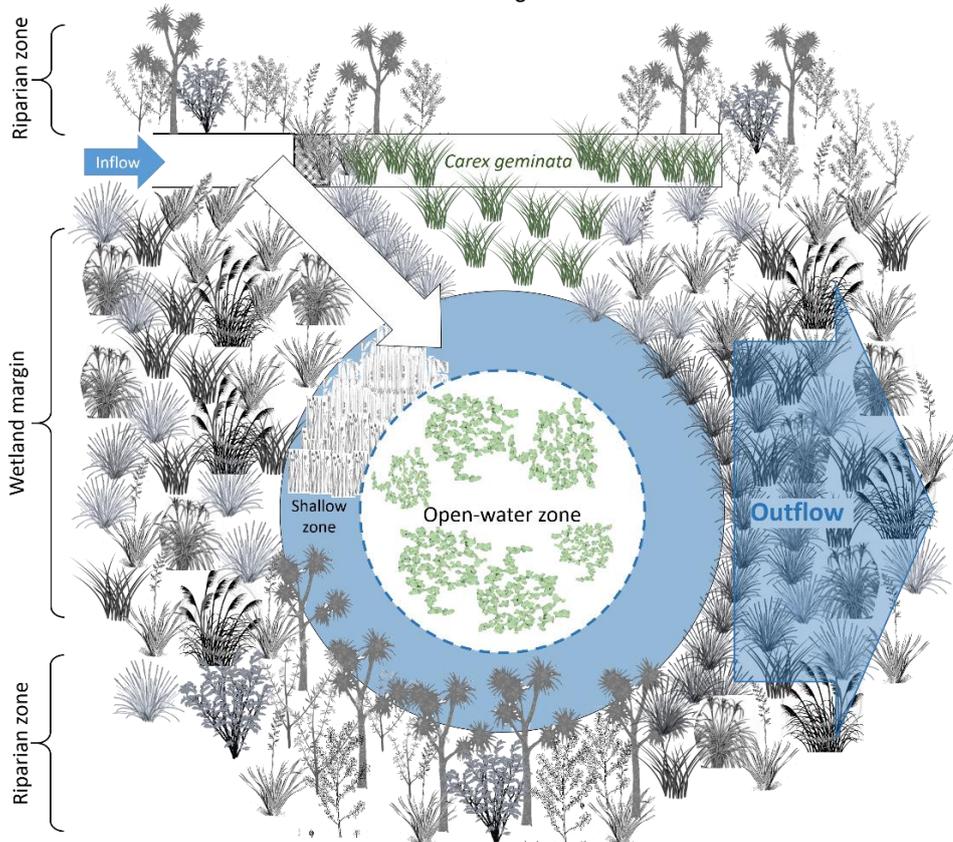


Figure 6.10 Schematic diagrams to provide guidance for designing open-water wetlands; e) plan view with outlet channel and planting zones, and f) plan view with filtration outlet and planting zones

Tips for construction

- i. Construction should occur during lowest flows in late summer to early autumn (February – March).
- ii. A fish survey will be required to determine the presence of native fish species in the watercourse prior to construction, preferably during late winter to early spring. If native fish are caught, then trapping must be carried out immediately before construction works begin with fish caught held for the period of disturbance, and subsequently released upstream. If the watercourse is dry at the time of construction, a black mudfish search (if caught previously) will be required to ensure none are aestivating in the damp channel.
- iii. Prior to the commencement of works, the open-water wetland design should be stepped out, measured and marked using paint with the contractor. It is important to orient the inlet and outlet diversion channels at 90° from each other (refer Figure 6.10 a) as this design promotes back eddies within the pond, increasing the residence time of the water within the wetland.
- iv. Firstly, excavate the open-water wetland. Regular measurements should be made during the excavation to ensure the correct dimensions of the wetland and batter slopes are achieved. The spoil removed should be stored alongside the existing channel to be used as fill at the completion of works. Excess excavated soil can be used around the farm to fill in holes or dips in fields using a tractor and trailer unit.
- v. Secondly, excavate the outlet diversion channel if applicable.
- vi. Thirdly, excavate the inlet diversion channel beginning at the downstream end and moving upstream, being careful not to breach the bank between the existing channel and the diversion channel at this point.
- vii. Before damming the existing channel and allowing water to flow into the treatment system, check the levels of the inlet and outlet diversion channels (if applicable) with the contractor using a dumpy level or equivalent to ensure flows will move downstream effectively.
- viii. Once satisfied with levels, dam the existing channel immediately below the start of the inlet diversion using hard-fill as shown in Figure 6.10 a. It is essential to use hard-fill material at the upstream end of the channel to be in-filled to create an effective diversion and prevent erosion. Old concrete troughs or equivalent can be used from around the farm property. Use the spoil stored alongside the channel to effectively seal the dam.
- ix. Once the water in the existing channel begins to pool behind the dam, excavate an opening in the bank to allow the water to flow into the inlet diversion channel. It can take quite some time for the open-water wetland to fill and begin flowing out the diversion channel (if applicable).
- x. Finally, fill in the existing drainage channel below the hard-fill dam as shown in Figure 6.10 a. It is recommended to over fill the drain as the level will drop over time as the soil settles and compacts. The final height of the in-filled channel

should be slightly below the level of the adjacent field to allow for drainage during floods, and so as to act as a high-flow bypass during heavy rainfall events.

- xi. Complete planting of the open-water wetland at the suggested times adhering to the recommended planting zones depicted in Figure 6.10 d, e, f.

Floating treatment wetland

Careful consideration of water depths and flows will be necessary to avoid creating stagnant areas. Still, warm water can cause anoxic sediment conditions during which phosphorus and ammonium can be released into the water column and lead to algal blooms. Early consultation with the contractor(s) who will produce and install the FTW is crucial to ensure the design parameters incorporate the contractor's FTW specifications (e.g. standard width and depth requirements).

Key design considerations:

- | | | |
|------|-------------|---|
| i. | Placement | Low lying areas with low gradient and shallow water table in close proximity to existing watercourses and water bodies. |
| ii. | Water depth | 1.3 – 1.8 m (TBC by FTW contractor) |
| iii. | Width | ≥ 4 m (TBC by FTW contractor) |
| iv. | Flow rates | Medium (average velocity ≤ 0.15 ms ⁻¹ , average discharge ≤ 0.025 m ³ s ⁻¹) |
| v. | Plants | <i>Carex virgata</i> (TBC by FTW contractor) |

Maintenance

- i. **During establishment** – Ensure the site is well fenced to exclude stock. If waterfowl may be a problem during plant establishment, a low hot wire may be necessary. Initial fortnightly visual inspections will be required by the landowner to check for damage from waterfowl or livestock, stressed or dead plants, and weed encroachment. Hand weeding or spot spraying with suitable herbicides may be necessary.
- ii. **Operational** - Following the first year of growth, regular spring maintenance should be established during which infill planting and weed control are carried out. The effectiveness of the FTW is dependent on well establish, thriving plants for roots that provide good filtration throughout the FTW. Operational maintenance is therefore essential.

Schematic diagrams

The following diagrams provide guidance for FTWs however the contractor will have the greatest input into the final designs. The plan view (Figure 6.11 a) shows the layout of the treatment system relative to the channel of the existing watercourse. Included is the excavated area for the floating wetland raft, inlet and outlet diversion

channels, and the area of the existing channel to be infilled. The cross sectional view (Figure 6.11 b) shows the recommended channel depth and width for the diversion channels, the excavated pond for the floating wetland raft, and recommended batter slopes for the edges of the wetland. The horizontal:vertical ratio for the batter edges of the wetland will depend on the stability and condition of the soil. Heavy clay and loosely consolidated moist soils such as peat require a flatter slope (e.g. 1:1) whilst steeper slopes ($\frac{1}{2}:1$) are suitable for well cemented soils. The contractor and designer will need to agree on the most appropriate batter slope depending on the soil conditions of the site.

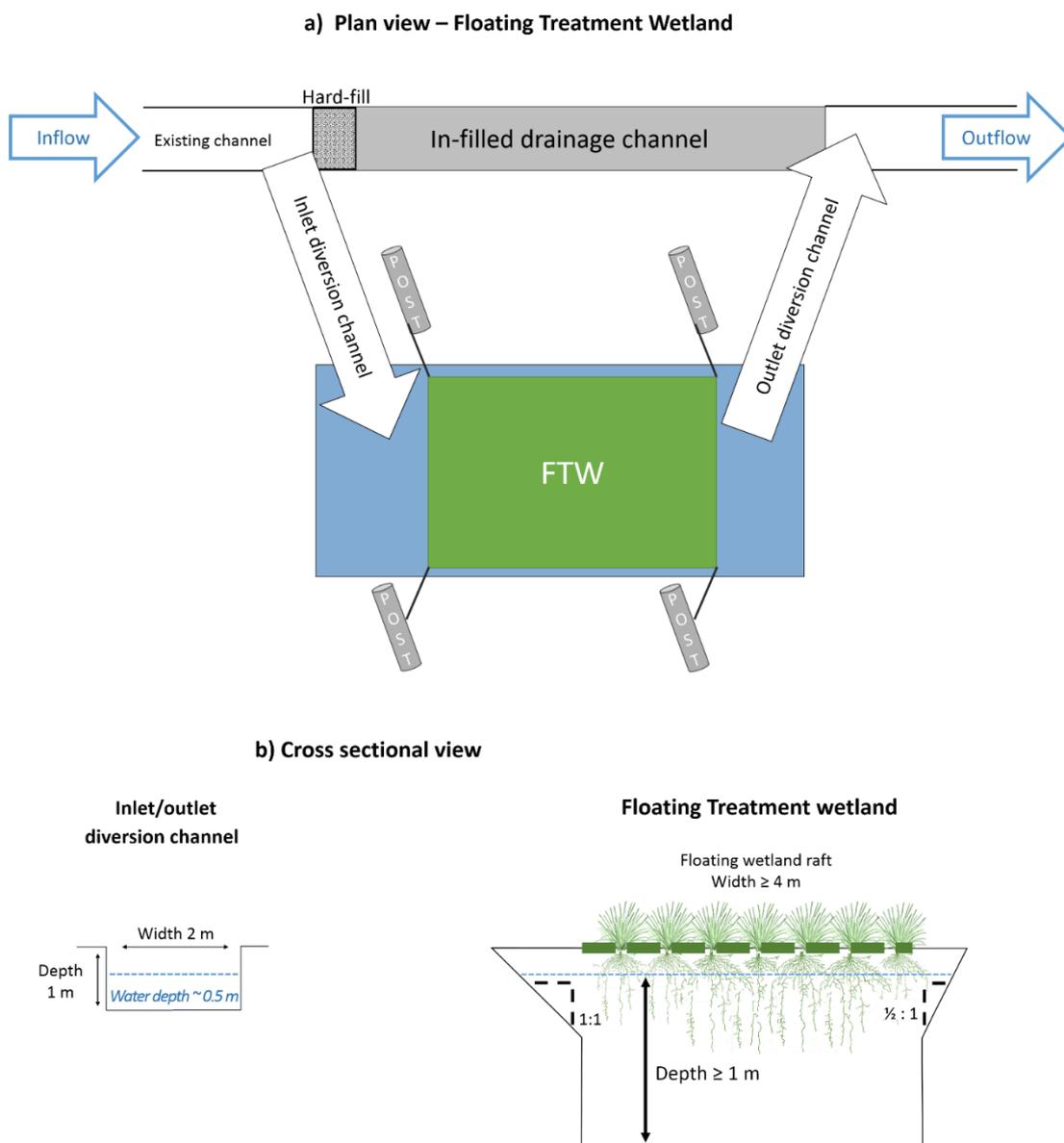


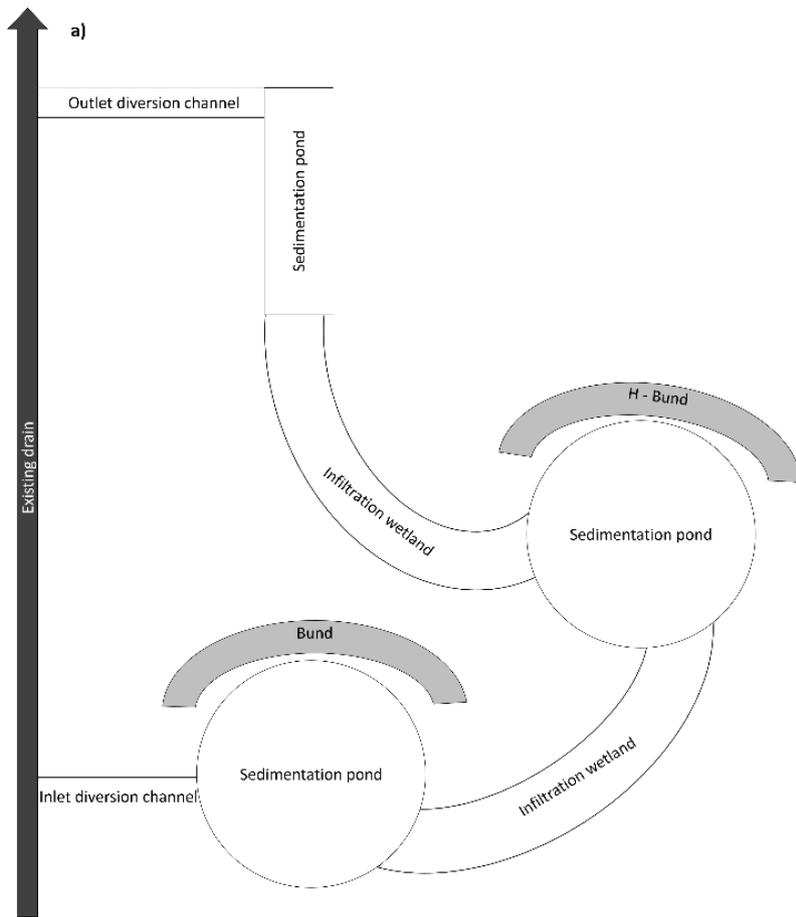
Figure 6.11 Schematic diagrams to provide guidance for designing floating treatment wetlands. a) Plan view, and b) cross sectional views of the diversion channel and the excavated pond for the floating wetland raft

Tips for construction

- i. Construction should occur during lowest flows in late summer to early autumn (February – March).
- ii. A fish survey will be required to determine the presence of native fish species in the watercourse prior to construction, preferably during late winter to early spring. If native fish are caught, then trapping must be carried out immediately before construction works begin with fish caught held for the period of disturbance, and subsequently released upstream. If the watercourse is dry at the time of construction, a black mudfish search (if caught previously) will be required to ensure none are aestivating in the damp channel.
- iii. Prior to the commencement of works, the area to be excavated to allow for the FTW raft should be stepped out, measured and marked using paint with the contractor. The location of the posts to which the FTW will be fastened should also be clearly marked.
- iv. Firstly, excavate the area required to allow for the FTW. Regular measurements should be made during the excavation to ensure the correct dimensions are achieved. The spoil removed should be stored alongside the existing channel to be used as fill at the completion of works. Excess excavated soil can be used around the farm to fill in holes or dips in fields using a tractor and trailer unit.
- v. Secondly, excavate the outlet diversion channel, and thirdly the inlet diversion channel beginning at the downstream end and moving upstream, being careful not to breach the bank between the existing channel and the diversion channel at this point.
- vi. Before damming the existing channel and allowing water to flow into the treatment system, check the levels of the inlet and outlet diversion channels with the contractor using a dumpy level or equivalent to ensure flows will move downstream effectively.
- vii. Once satisfied with levels, dam the existing channel immediately below the start of the inlet diversion using hard-fill as shown in Figure 6.7a. It is essential to use hard-fill material at the upstream end of the channel to be in-filled to create an effective diversion and prevent erosion. Old concrete troughs or equivalent can be used from around the farm property. Use the spoil stored alongside the channel to effectively seal the dam.
- viii. Once the water in the existing channel begins to pool behind the dam, excavate an opening in the bank to allow the water to flow into the inlet diversion channel. It can take quite some time for the open-water wetland to fill and begin flowing out the diversion channel.
- ix. Thirdly, fill in the existing drainage channel below the hard-fill dam as shown in Figure 6.7a. It is recommended to over fill the drain as the level will drop over time as the soil settles and compacts. The final height of the in-filled channel should be slightly below the level of the adjacent field to allow for drainage during floods, and so as to act as a high-flow bypass during heavy rainfall events.
- x. The FTW contractor will install the strainer posts, raft and wetland plants once the area excavated had filled with water following their own installation methodology.

Designing treatment systems large catchment (> 50 ha)

Designing constructed treatment systems for large catchments (> 50 ha) will require input from a suitably qualified and experienced specialist. During heavy rainfall events larger catchments will have greater flow rates and higher volumes of discharge which the constructed wetland must accommodate in order to achieve treatment efficiency. It is important to design appropriately sized constructed wetland treatment systems to ensure the larger volumes of water can be retained within the wetland for adequate lengths of time to allow for coarse sediments to settle out of the water column, for fine sediment and particulates to be filtered from the water column, and for biological process to occur to attenuate nutrients. For such watercourses a constructed wetland “treatment train” is recommended, whereby numerous sedimentation ponds as well as filtration and open-water wetlands can be created in sequence. A recent example of a constructed wetland treatment train has been created in the Lake Ngaroto catchment near Ohaupo, south of Hamilton City (Figure 6.12). The Lake Ngaroto constructed wetland was funded by NZ Landcare Trust as part of a three year project focused on improving the water quality of Lake Ngaroto (funded by MfE's CEF fund). The constructed wetland is a “demonstration” site and can be visited following arrangement with the landowner. Further information about the project can be obtained from NZ Landcare Trust (2015).



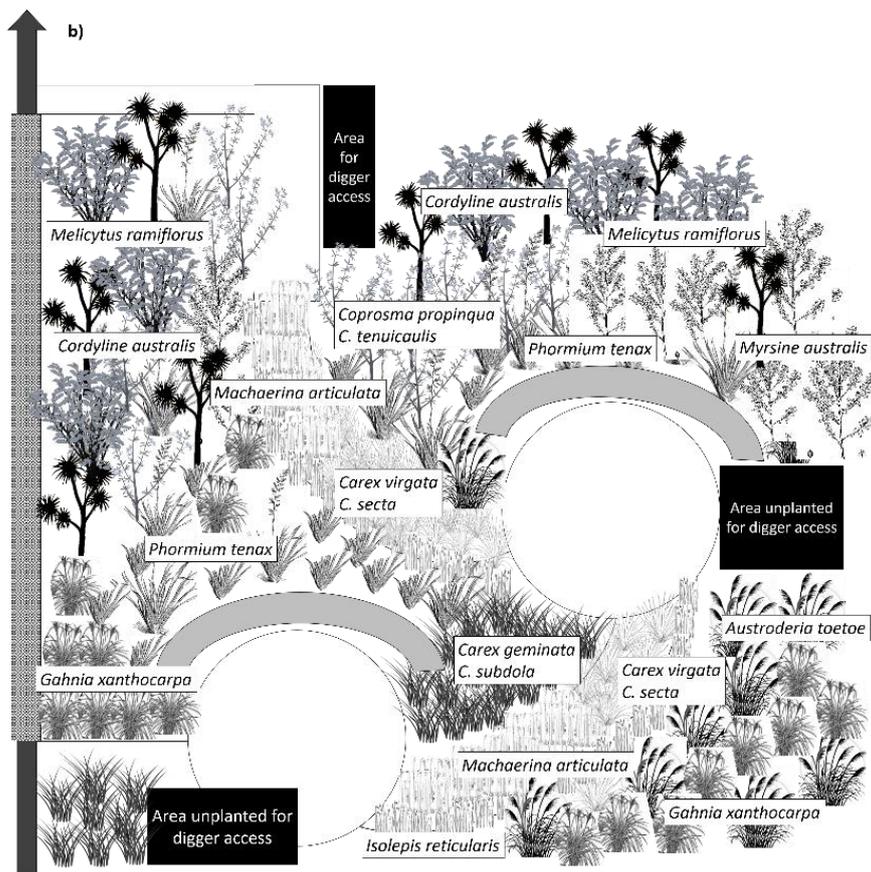


Figure 6.12 Schematic diagrams illustrating the constructed wetland “treatment train” created in the Lake Ngaroto catchment. a) Plan view, and b) plan view with planting plan

Summary and recommendations

Constructed wetland treatment systems can effectively reduce inputs of sediment and nutrients to shallow lakes if carefully designed and maintained. These guidelines describe a number of possible designs including filtration, open-water and floating wetlands as well as sedimentation ponds. General guidance is provided to assist local government agencies, landowners and care groups, consultants, agricultural advisors and contractors when creating constructed treatment systems to improve water quality. Treatment systems as management tools should be considered as secondary measures after best management practices have been implemented on farms and all efforts made to minimise nutrient and sediment runoff to waterways.

Attenuation efficiency

Further monitoring and research is required to determine the attenuation efficiencies of the constructed wetland treatment system designs described. Provision of expected rates of reduction of N, P and SS is outside the scope of these preliminary guidelines. Research carried out by Eivers et al. (in prep) describes the considerable variation in nutrient and sediment loads transported by artificial drains and modified watercourses within intensive agricultural land use to shallow Waikato peat lakes. This research also explores the large variation in the design and construction of treatment wetlands and sedimentation ponds currently being used in the Waikato. Deriving meaningful and accurate rates of pollutant removal from such examples is not scientifically feasible or robust. Pilot projects trialling and monitoring the constructed wetland treatment systems described in these preliminary guidelines is recommended. Data collected from monitoring can be used to determine the attenuation efficiencies of each design, and feedback into the design parameters. Through maintaining consistent designs, the effectiveness of the treatment systems in lieu of seasonal variation, different soil types and land use intensities can be more conclusively established.

Catchment discharge calculations

The size of treatment system designs in these guidelines are described as best suited to small to medium sized catchments (~5 – 25 ha). The scope of this work was to provide guidance towards developing constructed treatment systems for surface water inflows to shallow peat lakes. Tanner et al. (2010) provide useful guidance around appropriate sizing of constructed wetlands for tile drain flows, however the principles

described may be less applicable to watercourses transporting predominantly surface flows. Many of the inflows to shallow lakes in the Waikato are fed by small subcatchments within the greater catchment of the lake. The research behind this work focused on such subcatchments therefore these treatment system designs give effect to the size, topography and discharges typical of many shallow lake subcatchments in the central Waikato. Further research is required to more accurately predict discharges from subcatchments for which no data has been previously collected. This will allow treatment systems to be sized more accurately relative to discharges from the watercourse or size of the catchment to ensure adequate hydraulic residence times for effective pollutant removal. Surface flows from small, often artificial subcatchments are highly dynamic with variable seasonal flow rates and large, flashy peaks in discharge during heavy rainstorm events (Tempero & Hamilton 2014). Furthermore, catchment boundaries can be difficult to define due to the low-lying and undulating nature of the topography of such lake catchments, as well as the artificial drainage networks typical of many watercourses flowing to shallow lakes from catchments which have been heavily modified to support intensive agriculture. Development of a hydrological catchment model which can estimate discharges to artificial drainage networks from agricultural land use would be of significant benefit to the continued development of these guidelines.

Appendix 1 Desktop site scoping sheet

DESKTOP SCOPING STEPS

STEP ONE Request farm paddock map from landowner

- Familiarise self with the layout of the farm property
- Print map for use during the site scoping visit



STEP TWO Examine the farm and surrounding area using [Google Maps](http://www.googlemaps.co.nz)

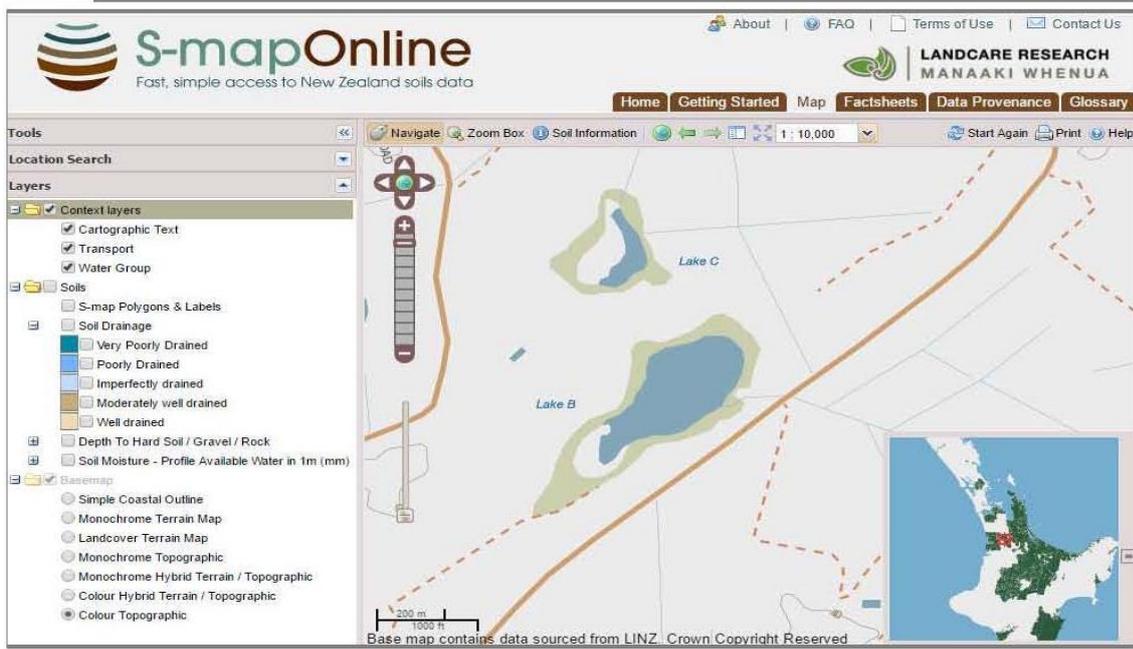
- Identify the following around the farm property:

Watercourses	<input checked="" type="checkbox"/>	Lakes	<input checked="" type="checkbox"/>
Ponds	<input checked="" type="checkbox"/>	Wetlands	<input checked="" type="checkbox"/>
- Save and print at least 3 aerial maps for use during the site scoping visit

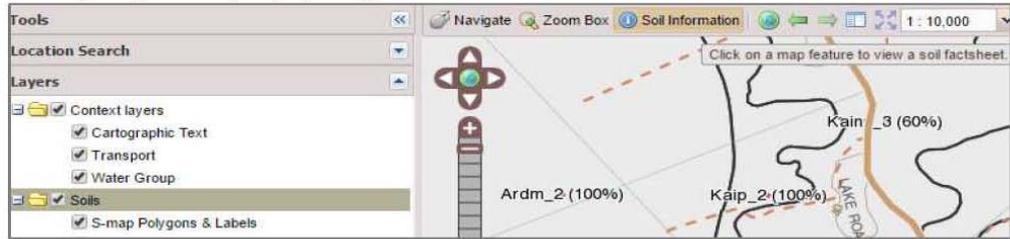
STEP THREE Examine the farm and surrounding area using SMAP smap.landcareresearch.co.nz

- Select "Maps & Factsheets >" [Maps & factsheets >](#)
- Accept Terms & Conditions [I Accept](#)
- Close "Help" box [Close](#)
- Zoom-in to location of the farm property using "Zoom Box" or Location Search [Navigate](#) [Zoom Box](#)
- With default "Context layers" ticked "on", identify the following around the farm property:

Watercourses	<input checked="" type="checkbox"/>	Lakes	<input checked="" type="checkbox"/>
Ponds	<input checked="" type="checkbox"/>	Wetlands	<input checked="" type="checkbox"/>
- Save and print map for use during the site scoping visit (see example map below)



- 7 Tick "on" the "Soils" layers
Tick "on" "S-map Polygons & Labels"
Select "Soil Information" button
Identify soil types and key soil properties, *if available*



- 8 For each map, adjust the "Layer Transparency" by sliding the square up/down the verticle bar to adjust the opacity of the **Soil Order** and **Drainage** maps whilst examining the study area

Save & print map for use during site scoping visit



- 9 Tick "on" the "Soils Drainage" layer to examine the study area, *if available*

Save & print map for use during site scoping visit



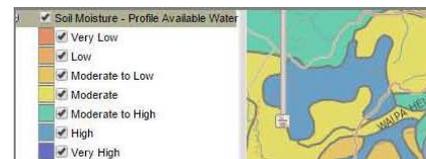
- 10 Tick "on" the "Depth To Hard Soil/Gravel/Rock" layer to examine the study area, *if available*

Save & print map for use during site scoping visit



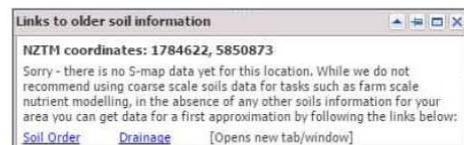
- 11 Tick "on" the "Soil Moisture - Profile Available Water" layer to examine the study area, *if available*

Save & print map for use during site scoping visit



- 12 If there is no detailed S-map data in the location of the farm property, a "Links to older soil info" box will appear. Click on the **Soil Order** and **Drainage** hyperlinks to obtain approximate soil information

Save & print maps for use during site scoping visit



- 13 Save and print soils maps for use during the site scoping visit

DESKTOP SCOPING STEPS COMPLETE

Appendix 2 Site Scoping Field Sheet

Complete a separate scoping field sheet for each potential site identified during the field visit.

Site Scoping Field Sheet

Question	OVERALL FARM		Yes / No / TICK
1	What water quality pollutant do you wish to target?	Sediment	<input type="text" value="Yes"/>
		Nitrogen	<input type="text" value="Yes"/>
		Phosphorus	<input type="text" value="Yes"/>
			<input type="text" value="3"/>
2	Where on the property are potential sites? Examine	Aerial photography map	<input type="text" value="TICK"/>
		Farm paddock map	<input type="text" value="TICK"/>
3	Clarify length of watercourses including artificial drains.	Aerial photography map	<input type="text" value="TICK"/>
		Farm paddock map	<input type="text" value="TICK"/>
4	What is the approximate catchment boundaries of the watercourses?	Aerial photography map	<input type="text" value="TICK"/>
		Farm paddock map	<input type="text" value="TICK"/>
INDIVIDUAL CATCHMENTS			
5	How can the watercourse be classified?	Existing in natural state	<input type="text" value="TICK"/>
		Existing in modified state	<input type="text" value="TICK"/>
		Artificial drain	<input type="text" value="TICK"/>
6	Does the watercourse flow year round?	Yes - Go to Q 7	<input type="text" value="Yes"/>
		No - Go to Q 8	<input type="text" value="No"/>
			<input type="text" value=""/>
7	Are there smaller watercourses which do dry further up the catchment?	Yes - Go to Q 3	<input type="text" value="Yes"/>
		No - Large/complex site	<input type="text" value="No"/>
8	Does the watercourse dry out completely?	Yes - Go to Q 10	<input type="text" value="Yes"/>
		No - Go to Q 9	<input type="text" value="No"/>
9	Identify different sections of the watercourse on maps	Dry areas [DR]	<input type="text" value="TICK"/>
		Damp areas [DM]	<input type="text" value="TICK"/>
		Shallow pools [SP]	<input type="text" value="TICK"/>
		Deep pools [DP]	<input type="text" value="TICK"/>
		Springs [SR]	<input type="text" value="TICK"/>
10	Is the slope of the watercourse relatively uniform?	Yes - Go to Q 12	<input type="text" value="Yes"/>
		No - Go to Q 11	<input type="text" value="No"/>
11	Identify features within the watercourse and riparian zone on maps	Slumps [SM]	<input type="text" value="TICK"/>
		Depressions [DR]	<input type="text" value="TICK"/>
		Dips [DI]	<input type="text" value="TICK"/>
		Bogs [BO]	<input type="text" value="TICK"/>
		Flooding areas [FL]	<input type="text" value="TICK"/>
12	Is the full length of the watercourse fenced?	Yes - Go to Q 14	<input type="text" value="Yes"/>
		No - Go to Q 13	<input type="text" value="No"/>
13	Identify areas of the watercourse which are NOT fenced	Aerial photography map	<input type="text" value="TICK"/>
		Farm paddock map	<input type="text" value="TICK"/>
14	Which paddocks drain to the watercourse?	Farm paddock map	<input type="text" value="X Y Z"/>
15	What is the approximate area of paddocks draining to the watercourse?	Sum areas of paddocks	<input type="text" value="X+Y+Z"/>

16	Can the catchment boundary be defined more accurately?	Aerial photography map Farm paddock map	<input type="checkbox"/> TICK <input type="checkbox"/> TICK
----	--	--	--

HOT SPOT IDENTIFICATION

17	Locate on the aerial and farm maps:	Troughs [TR] Feed Waggon [FW] Shade Trees [ST] Raceways [RW] Bridges Underpass [BU] Milking Shed [MS] Feed Pad [FP] Holding Pen/Yards [HP] Silage Storage [SS] Grain Silo [GS] Fertiliser Storage [FS] Effluent Pond [EP] Tile Drains [TD]	<input type="checkbox"/> TICK <input type="checkbox"/> TICK
----	-------------------------------------	--	--

18	Does raceway runoff drain into watercourses?	Yes - Go to Q 19 No - Go to Q 20	<input type="checkbox"/> Yes <input type="checkbox"/> No
----	--	-------------------------------------	---

19	Locate where raceway runoff enters watercourses	Aerial photography map Farm paddock map	<input type="checkbox"/> TICK <input type="checkbox"/> TICK
----	---	--	--

20	Does runoff from highways or roads drain into watercourses?	Yes - Go to Q 21 No - Go to Q 22	<input type="checkbox"/> Yes <input type="checkbox"/> No
----	---	-------------------------------------	---

21	Locate where road runoff enters watercourses	Aerial photography map Farm paddock map	<input type="checkbox"/> TICK <input type="checkbox"/> TICK
----	--	--	--

DIFFUSE POLLUTION

22	What is the current stocking rate?	Cows / ha	<input type="text"/>
----	------------------------------------	-----------	----------------------

23	What is the current application rate for N fertiliser?	kg N / ha / yr	<input type="text"/>
----	--	----------------	----------------------

24	What is the current application rate for P fertiliser?	Kg P / ha / yr	<input type="text"/>
----	--	----------------	----------------------

25	Are areas of the farm routinely irrigated?	Yes - Go to Q 26 No - Go to Q 27	<input type="checkbox"/> Yes <input type="checkbox"/> No
----	--	-------------------------------------	---

26	Locate which areas are irrigated	Aerial photography map Farm paddock map	<input type="checkbox"/> TICK <input type="checkbox"/> TICK
----	----------------------------------	--	--

27	Is effluent irrigated?	Yes - Go to Q 28 No	<input type="checkbox"/> Yes <input type="checkbox"/> No
----	------------------------	------------------------	---

28	Locate which areas are irrigated with effluent	Aerial photography map Farm paddock map	<input type="checkbox"/> TICK <input type="checkbox"/> TICK
----	--	--	--

Input collected information into the Site Data Sheet upon return to the office, where appropriate

Appendix 3 Desktop Information Guidance

Desktop Information Guidance - CLIMATIC DATA

Collate climate data for the site following the steps outlined below using The National Climate Database Cliflo <http://cliflo.niwa.co.nz/>

A. Click **Subscribe On-line**



B. Read **Cliflo Subscription: Introduction**

C. Follow **Steps 1-6**



D. Click **Continue to Username** and complete the Application

E. Once subscribed & logged in, progress to **Database Query** and complete steps 1 - 4

1. Datatype

Click on **select datatype(s)**

Select **Statistics Calculated from Observations**

Select **Monthly Statistics**

Select **Wind**, tick **Mean wind speed (m/s) (33)**

Click **Add**

Select **Rain**, tick **Total rainfall (mm) (00)**; and

Tick **Maximum 1-day rainfall (mm) (41)**

Click **Add**

Select **Mean Temperature**

Tick **Mean air temperature (°C) (02)**

Click **Add**

Select **Evaporation and PET**

Tick **Total Penman Potential evapo-transpiration (mm) (34)**

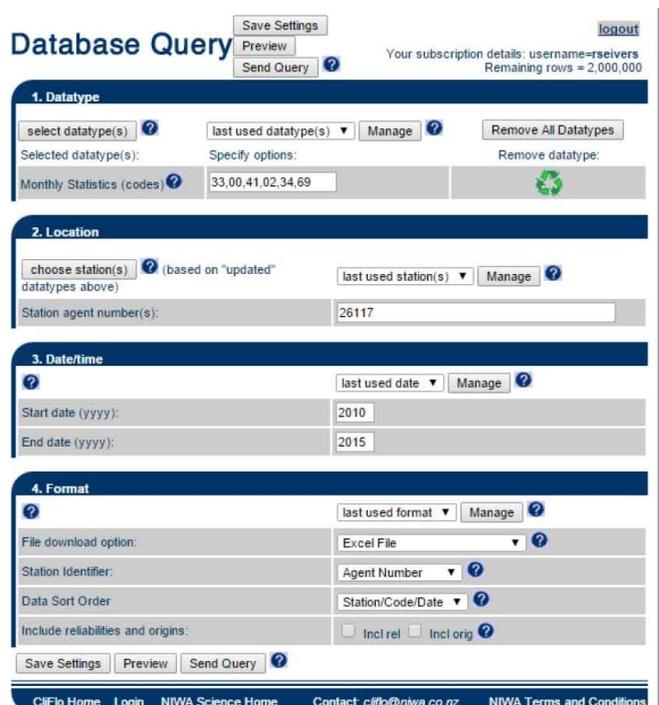
Click **Add**

Select **Measured Soil Moisture**

Tick **Mean soil moisture (%) (69)**

Click **Add**

Select **Close Window** at the top of the form



2. Location

Click on [choose station\(s\)](#)

For **Options** Select • All datatypes must exist at station (Boolean AND)

For **Find station using:** Select • Lat/Long: based on circle radius (km)

Enter Latitude & Longitude in decimal degrees, and radius of the circle to 50 (km)

Select **Station Status:** to Open Stations

Select **File download option:** to HTML Table

Click [Get Station List](#)

Lat/Long: based on circle with radius (km) -37.9519 175.3153
 50
 Station Status: Open Stations
 File download option: HTML Table

Select the preferred station following the instructions for **Station Listing Using Datatypes**

Click [Add Selected Stations](#)

Select	Agent Number	Network Number	Start Date	End Date	Percent Complete	Name	Lat (dec deg)	Long (dec deg)	Dist Km
<input checked="" type="checkbox"/>	26117	C75734	01-Dec-2005	31-Jan-2015	100	Hamilton, Ruakura 2 Ews	-37.77567	175.30506	19.6
<input type="checkbox"/>	23899	C85316	01-Jun-2003	31-Jan-2015	90	Te Kuiti Ews	-38.33356	175.1534	44.7

Select **Database Query Form**

3. Date/time

Set **Start date (yyyy):** and **End date: (yyyy)**

Five years is recommended to allow for biannual variation

3. Date/time
 Start date (yyyy): 2010
 End date (yyyy): 2015

4. Format

Set **File download option:** to Excel File

Set **Station Identifier:** to Agent Number

Set **Date Sort Order** to Station/Code/Date

Click [Preview](#) & review the information given

Select **Back to Database Query Form** when satisfied

Click [Send Query](#)

Save file and open with Microsoft excel



4. Format
 File download option: Excel File
 Station Identifier: Agent Number
 Data Sort Order: Station/Code/Date
 Include reliabilities and origins: Incl rel Incl orig
 Save Settings Preview Send Query

Ruakura Cliffo Monthly Totals 2010-2015.xlsx - Excel

FILE HOME INSERT PAGE LAYOUT FORMULAS DATA REVIEW VIEW PIVOTABLE TOOLS ANALYZE DESIGN

Clipboard Paste Cut Copy Format Painter Font: Calibri, Size: 11, Bold, Italic, Underline, Paragraph: Merge & Center, Alignment: Center, Number: 00%

Styles: Normal, Check cell, Bad, Good, Input, Explanatory...

1 Station information: Agent Number Network N Latitude (dec Longitude (Height, Posn, J Observing Authority

2 Name Hamilton, Ruakura 2 EWS 26117 C75794 -37.77567 175.30506 45 H Niwa/Agresearch

3 Hamilton, Ruakura 2 EWS 26117 C75794 -37.77567 175.30506 45 H Niwa/Agresearch

4 Note: Position precision types are: "W" = based on whole minutes, "M" = estimated to tenth minute,

5 G = derived from gridref, "E" = error cases derived from gridref,

6 H = based on GPS readings (NZGD49), "D" = by definition i.e. grid points.

7

9 Statistics codes in this query are:

Code	Description	Units
0	Total Rainfall	Mm
2	Mean Air Temperat	Celsius
33	Mean Wind Speed	M/Sec
34	Total Penman Potet	Mm
41	Maximum 1-Day Rai	Mm
69	Mean Soil Moisture	Percent

17 Note: Statistics calculations are based on Local Time.

18 DAY OF EXTR gives the Local-Time day for a Monthly Extreme.

19 Stats: Monthly

Station	Mon-YYYY(local)	Stat Code	Stat Value	Day of Extr
26117	Jan-10	0	154.4	-
26117	Feb-10	0	24.4	-
26117	Mar-10	0	15.4	-
26117	Apr-10	0	30.6	-
26117	May-10	0	109.2	-
26117	Jun-10	0	232	-
26117	Jul-10	0	55.6	-
26117	Aug-10	0	157.8	-
26117	Sep-10	0	150	-
26117	Oct-10	0	31.4	-
26117	Nov-10	0	21.6	-
26117	Dec-10	0	104.6	-
26117	Jan-11	0	215.8	-
26117	Feb-11	0	13.4	-
26117	Mar-11	0	159.2	-
26117	Apr-11	0	110.8	-
26117	May-11	0	125	-
26117	Jun-11	0	166	-
26117	Jul-11	0	151.2	-

20 Station

Mon-YYYY(local)	Stat Code	Stat Value	Day of Extr
26117	Jan-10	0	154.4
26117	Feb-10	0	24.4
26117	Mar-10	0	15.4
26117	Apr-10	0	30.6
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26117	Jul-10	0	55.6
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26117	Mar-11	0	159.2
26117	Apr-11	0	110.8
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26117	Jun-11	0	166
26117	Jul-11	0	151.2

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Mon-YYYY(local)	Stat Code	Stat Value	Day of Extr
26117	Jan-10	0	154.4
26117	Feb-10	0	24.4
26117	Mar-10	0	15.4
26117	Apr-10	0	30.6
26117	May-10	0	109.2
26117	Jun-10	0	232
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26117	Apr-11	0	110.8
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26117	Jun-11	0	166
26117	Jul-11	0	151.2

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26117	Apr-11	0	110.8
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Mon-YYYY(local)	Stat Code	Stat Value	Day of Extr
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Appendix 4 Site Data Input Spreadsheet

Complete a separate data spreadsheet for each potential site identified during the site scoping field visit.

Site information		Example		Recommended treatment system			
Name/Code		551		Infiltration	Sedimentation	Openwater	Treatment
Easting		2713847		WL	pond	WL	train
Northing		6359774		4	5	3	1
Altitude (masl)		74		best treatment system type yellow = good treatment system type orange = possible treatment system type purple = not recommended			
Downstream water body		Lake Serpentine South					
Length of the watercourse (m)		455					
Subcatchment size (ha)		15.4					
CLIMATE DATA from CliFlo Database							
Date Range - Monthly from 2010 - 2015		90th Percnt.	Median	Mean	10th Percnt		
Total Rainfall mm		166		93	93	22	
Mean Air Temp °C		19		14	14	9	
Mean Wind Speed m/s		3		2	2	2	
Total PET mm		133		89	82	25	
Max. 1-Day Rain mm		56		25	28	9	
Mean Soil Moisture %		48		36	35	21	
Q. OVERALL FARM				Yes / No / TICK			
1 What water quality pollutant do you wish to target?		Sediment		<input type="checkbox"/> Yes			
		Nitrogen		<input type="checkbox"/> Yes			
		Phosphorus		<input type="checkbox"/> Yes			
2 Where on the property is the site?		Using a GIS mapping software, locate the site and record the co-ordinates above					
3 Clarify length of watercourse, including artificial drains.		Using a GIS mapping software, draw along the length of the watercourse and record the approximate length					
4 What is the approximate catchment boundary of the watercourse?		a) Using a GIS mapping software, draw around the catchment boundary identified with the landowner; OR b) Calculate an accurate catchment area using Lidar and a Digital Elevation Model (DEM) - employ a GIS expert if required					
5 How can the watercourse be classified? (cross out that not applicable)		Existing in natural state		A consent will be required			
		Existing in modified state		A consent will be required			
		Artificial drain		Consent unlikely			
6 Does the watercourse flow year round? [Yes / No]		<input type="checkbox"/> No					
7 Are there smaller watercourses which do dry further up the catchment? [Yes / No]		<input type="checkbox"/> Yes					
8 Does the watercourse dry out completely? [Yes / No]		<input type="checkbox"/> No					
10 Is the slope of the watercourse relatively uniform? [Yes / No]		<input type="checkbox"/> No					
12 Is the full length of the watercourse fenced? [Yes / No]		<input type="checkbox"/> Yes					
HOT SPOT IDENTIFICATION							
18 Does raceway runoff drain into watercourses? [Yes / No]		<input type="checkbox"/> No					
20 Does runoff from highways or roads drain into watercourses? [Yes / No]		<input type="checkbox"/> Yes					
DIFFUSE POLLUTION							
25 Are areas of the farm routinely irrigated? [Yes / No]		<input type="checkbox"/> No					
27 Is effluent irrigated?		<input type="checkbox"/> No					

For Site Data Input Spreadsheets refer to DMH 3264098 - Save As a copy of the spreadsheet for each site being assessed.

Appendix 5 External Peer Review

Comments and Responses

This document summarises the responses made to the comments from Paula Reeves and Chris Tanner following their peer review of the “*Guidelines for Constructed Treatment Systems for Surface Water inflows to Shallow Lakes*” (referred to as the *draft guidelines* herein). Responses are shown in [track changes](#). Where stated, amendments have been made to the *draft guidelines* where appropriate.

Rebecca Eivers



27th November 2015

From:	Paula Reeves [Paula.Reeves@waipadc.govt.nz]	Sent:	Fri 19/06/2015 2:46 p.m.
To:	Tracie Dean-Speirs		
Cc:	'c.tanner@niwa.co.nz'		
Subject:	Feedback on constructed treatment systems for surface water inflows		

Hi Tracie,

Thank you for the opportunity to provide feedback on the Constructed Treatment Systems for Surface Water Inflows to Shallow Lakes. It is a great initiative and one that will be invaluable to those of us working to improve water quality in shallow lakes.

My feedback follows, some of which is based on recent experiences planting infiltration wetlands, sediment ponds and open water wetlands:

- A summary table of the different CTS would be useful near the beginning of the document (or start of section 5) with key information for those people who are trying to decide whether to use a system and which one is likely to be most appropriate.
[The designs are summarised in section 5. It is important the user reads the entire draft guidelines and uses the associated decision support tools provided in appendices 1, 2, 3 and 4 to inform their decision of which treatment system\(s\) will be most appropriate.](#)
 - More information on the effectiveness of CTS for attenuating pollutants is needed. Does Rebecca’s research not give us some conclusive results that can be included in the guidelines? This could be added to the summary table to give people some idea of what type of reductions in pollutants could be expected (could be a range rather than a single figure).
[The designs require trials and monitoring to determine expected efficiencies. This is recommended in Section 8.1.](#)
 - The guidelines frequently recommend planting with trees and shrubs to provide shade to a CTS to keep water temperatures low. However if a waterway/body becomes shaded then the abundance of the wetland and aquatic plant species are likely to be reduced to low covers compromising the effectiveness of the CTS. Taller trees and shrubs could be used to enhance the habitat values of a CTS but would be best limited to the southern area of a system to ensure that the key wetland and aquatic species aren’t compromised by shading. Wetland plant species will also shade the water reducing water temperatures.
[Many of these treatment systems are/will be constructed in open pasture with very high levels of solar irradiance. It is expected that in-stream temperatures and local micro-climates will remain warm enough to sustain aquatic plant growth with riparian plantings including trees. Where a treatment system is to be constructed in a subcatchment/area which that is already well shaded, then riparian plantings including tree species may not be appropriate. These decisions will be site specific and remain at the discretion of the user of the guidelines.](#)
 - The description of an open water wetland needs a little more information on why it’s effective (pg 7). What processes in open water enhance nutrient uptake? [\(Amendment made to draft guidelines\).](#)
 - What is draught stress? The depth ranges provided (0.3-1.0m) for an open water wetland could potentially be completely colonised by some wetland plants. [\(Amendment made to draft guidelines\).](#)
 - *Eleocharis sphacelata*, *raupo*, and *Schoenoplectus tabernaemontanii* could all potentially spread by rhizome to occupy water depths of 1m especially during summer months when water levels are likely to fall well below this. [\(Amendment made to draft guidelines\).](#)
- Plant species selection: I would recommend *Isolepis prolifera* ahead of *Isolepis reticularis* as it is more common in high nutrient environments in the Waikato, is likely to establish and spread quicker and is more densely matted so should be more effective at trapping sediment. Definitely more bang for your buck!
Gahnia xanthocarpa is mostly found in well established wetland / swamp forest – NZPCN says it likes humus

rich soil so I don't think it's likely to thrive in the places most CTS are located. *Austroderia toetoe* is a better option. *Carex subdola* and *Carex sinclairii* would be better on wetland margins than inner wetlands – neither are likely to establish in water depths greater than 0.3m. *Juncus edgariae* is ubiquitous in wetlands and is highly likely to establish on its own accord so I never recommend people plant it (and most people think of it as a weed!). ([Amendment made to draft guidelines](#)).

Experimental but potentially a good wetland margin plant for CTS is *Isachne globosa* (swamp millet grass). Once established likely to spread and suppress many of the exotic annual weeds and grasses. Other good species for wetland margins in peat soils are *Macherina rubiginosa* and *Macherina arthropylla*. *Coprosma propinqua* and *Coprosma tenuicaulis* like high water tables and would fit the wetland margin category better than the riparian zone. The standard coloniser species for riparian margins could be added to the list of species for the riparian zone i.e. karamu, manuka, kanuka, koromiko, wineberry, ribbonwood.

([Amendment made to draft guidelines](#)).

There is very little mention of establishing submerged species yet these can potentially have significant benefits for trapping sediment and contribute greatly to aquatic habitat and would be more suitable than *Lemna minor* which will self-introduce (courtesy of ducks). Reasonable water clarity is needed but we have successfully established *Potamogeton ochreatus* and *Myriophyllum propinquum* in CTS at Mangakaware.

([Amendment made to draft guidelines](#)).

Tim Martin and I have done a technical report for establishing submerged plants in stormwater ponds and wetlands which could be linked to...

<http://www.aucklandcouncil.govt.nz/EN/planspoliciesprojects/reports/technicalpublications/Documents/TR2013007comparativesuitabilityofnativesubmergedmacrophytesinconstructedstormwaterwetlands.pdf>

- *Baumea articulata* (pg 15) has had a name change – now *Macherina articulata*. ([Amendment made to draft guidelines](#)).
- It would be good to have some information on how to monitor sediment deposition so people know when to remove sediment. ([Amendment made to draft guidelines](#)).
- I haven't tried to design a CTS using the guidelines so not able to provide feedback on the usability of the site scoping and field sheets.
- Great diagrams!

Look forward to the guidelines being publicly available.

Paula

Guideline for Constructed Treatment Systems for Surface Water inflows to Shallow Lakes

General comments

Initially scanning through the document the clarity and aesthetic beauty of the diagrams is evident. Next my interest was drawn to the innovative designs of some of the systems proposed. Delving more deeply I found, however, that key bits of information and technical detail I would need to properly apply and assess the different systems was not present. I am aware that such information may be available to back-up these guidelines, but not yet be readily accessible. The somewhat unusual design of some of the systems compared to conventional engineering practice does require the design principles to be clearly explained and evidence provided to prove their efficacy and practicality.

[The designs require trials and monitoring to determine expected efficiencies. This is recommended in Section 8.1.](#)

This information needs to be readily available to support the approaches proposed. Farm drainage systems and waterways provide a range of basic services for farmers, land managers and the community- primarily drainage of excess ground and surface waters and avoidance of excessive flooding. Any guideline for modification of these systems to meet environmental goals needs to also provide for these functions.

[The designs require trials and monitoring to determine expected efficiencies. This is recommended in Section 8.1. Expanding the guidelines to include tools to determine subcatchment discharge/flow rates/ flow rating curves for small, predominantly ephemeral and/or artificial drainage catchments is also recommended in Section 8.2. This information would allow more accurate sizing of treatment systems relative to flows/discharge to ensure adequate hydraulic residence times.](#)

Also there are established engineering principles and procedures that have been developed and are commonly applied in the design and operation of drainage systems which must be addressed if the system is changed. For instance, if the roughness coefficient or resistance to flow of a drain constructed at a certain size and grade is significantly changed (e.g. by planting it with emergent vegetation) then it will not be able to transmit the same flow and will be more prone to flooding. As well as maintaining an equivalent cross-sectional area, additional provision must be made to overcome the additional flow resistance of the vegetation and allow for accumulation of trapped allochthonous sediment and autochthonous litter over time. Such practical issues do not seem to be adequately addressed in the guideline. [Agreed. \(Amendment made to draft guidelines\).](#)

The objectives and appropriate application of the various treatment options do not seem to be clearly delineated. Most of the options outlined appear to be primarily designed to remove sediment and associated contaminants. It is somewhat surprising that sediments should be the target given the low gradient of the catchments they are being deployed in and their low relative sediment yield compared to steeper more erodible catchments e.g. Waipa. Removal of nutrients is mentioned more broadly as a function of the open-water wetlands, but the means by which the systems proposed can address this is not specified. [\(Amendment made to draft guidelines\).](#) For removal of dissolved nutrients such as nitrate and phosphate different removal mechanisms operate and much larger treatment areas are generally required to achieve reasonable removal performance. [These treatments systems are designed for small catchments therefore concentrations and water volumes are not as large. The requirement for larger treatment systems as suggested may not be necessary, however, as stated, trials will determine efficiencies.](#) Special design may be required to avoid release of phosphorus bound to accumulated suspended solids. [This is a well known limitation of constructed wetlands. Removal of sediments with associated phosphorus, as instructed in the guidelines, is a method of minimising the risk of dissolved phosphorus release.](#)

Faecal microbes are another key contaminant concern which also needs to be addressed. [Faecal microbes were outside the scope of this research.](#)

A significant omission appears to be clear guidance on how the size of the various systems should be varied to account for different catchment areas, run-off coefficients (a function of soil type, slope, vegetation characteristics, landform etc.), or different types of receiving waters requiring removal of different target contaminants. General guidance is provided on appropriate catchment sizes for each option but the range is rather large (e.g. 5-25 ha) without clear guidance as to how the sizing of the system should be varied to account for this or other site characteristics. I could not find any information on expected treatment performance of the systems proposed either.

[The designs require trials and monitoring to determine expected efficiencies. This is recommended in Section 8.1. Expanding the guidelines to include tools to determine subcatchment discharge/flow rates/ flow rating curves for small, predominantly ephemeral and/or artificial drainage catchments is recommended also in Section 8.2. This](#)

[information would allow more accurate sizing of treatment systems relative to flows/discharge to ensure adequate hydraulic residence times.](#)

Provision of such information is likely to be important for farmers, land managers and regulators to be able to assess the benefits of such systems relative to the costs involved.

Some information on the costs (even relative costs and performance) would also be of value to enable such assessments. [To be established following trials.](#)

Specific comments

Title: Here there is an attempt to constrain the applicability of the guidelines to surface water inflows to shallow lakes. However, some relatively large shallow lakes with significant surface-water inflows occur in the Waikato (e.g. Whangape, Waahi and Waikare) which I would consider go beyond the scope and applicability of these guidelines.

[The scoping tools in the guidelines appendices would direct the user to the smaller subcatchments within the greater catchment of these larger lakes – that is the key point of difference these guidelines have with others currently available. The scoping tools help inform better decisions on the placement and location of such treatment systems so that they can be more easily and cost-effectively constructed, managed and maintained.](#)

Given the limitations mentioned above, it would probably be prudent to limit the guidelines (as done to some extent in the introductory sections) to small lowland lake catchments (such as the smaller peat lakes) and, given regional differences in plant distributions, to limit them to just the Waikato perhaps or at least to a more restricted area of New Zealand. [See comment above.](#)

2. Introduction: Some more appropriate recent references to diffuse pollution issues in New Zealand could be provided, such as the results of SPARROW modelling in the Waikato and nationally or assessment of national nutrient budgets, which enable the relative importance of different sources to be quantified. [I considered such information too technical for the target users.](#)

2.1 Best practice: more supporting information is required to support the contention that these guidelines represent “best practice” as outlined in this section. [\(Amendment made to draft guidelines\).](#)

4. Pre-design scoping considerations: I agree that the information requested is important for planning appropriate on-farm mitigations and management, but it is not clear how the information asked for in the guidance sheets provided in the Appendices is to be used to inform the design process, determine which system to use or size it appropriately. There seems to be a major disconnect here between the information being collected and its application to design of the treatment system. [The site data input spreadsheet in the appendix 4 collates this information together and provides direction as to the most appropriate treatment system for the site.](#)

5.1 and 6.1 Infiltration wetland: It is not clear how this approach (planting up an existing drain) will promote infiltration (passage of flow into the soil) apart from widening of the channel to expose a greater surface area. If the soils are heavy clays then infiltration will be not likely be significant during periods of moderate to high flow. [This treatment systems are only applicable to ephemeral watercourses, thus heavy clays are not the target. Fine suspended solids \(organic particulates of N and P\) and topsoil exposed from pugging for example are the target pollutants for this treatment system design.](#) This system could be loosely called a filtration wetland (or wetland filter) where passage through vegetation helps to filter out suspended solids. [\(Amendment made to draft guidelines\).](#) I have concerns though as to how drainage function (depth of drainage and outflow rate) will be adequately maintained. Although the cross-sectional area of the channel is maintained by widening the channel, the vegetated infiltration wetland will have greater resistance to flow than a clean drain, and the shallower depth with base set higher (as shown in the cross sectional view) will result in drainage to a shallower level (i.e. maintain a higher water table under saturated conditions). [\(Amendment made to draft guidelines\).](#)

Although I can appreciate the reasons behind doing a fish survey for this and other pond/wetland options, this requirement is likely to be relatively expensive and does not seem very practical for farmers- I’d predict it would be a major disincentive. Perhaps this requirement could be limited to large wetland developments. Do you require this for all drain-cleaning operations? [This is a consenting requirement for the Waikato Regional Council.](#)

5.2 and 6.2 Sedimentation ponds: The shape and form of these ponds proposed is interesting but rather different from established practice for settling basins. In the absence of a clear description of the design principles under which they operate, I was left to speculate that they are designed to operate as hydrocyclones, or vortex separators

using the principle of centrifugal sedimentation. [Enhances eddies and back-eddies, similar to those in a stream/river systems, encouraging settling of sediments when "trapped" in the circulating flow of the water.](#)

I can see how such a design might work well for coarse sediment, but I would need more evidence before I would accept they would work better across the range of flow rates and sediment characteristics likely to be experienced in land run-off situations. [Creating a number of sedimentation ponds in series is a method to target sediment particles of different size/shape/settling rates. The first pond captures heavier sediments and will require cleaning more frequently. The second pond would target sediment particles that settle more slowly. A third pond may include floating wetland rafts to target very fine clay particles which are inherently difficult to remove from suspension. A trial is currently examining the later at Lake Kaituna.](#)

Conventional design would suggest a broad shallow pond with a length to width ratio of at least 3:1 or greater would provide better settling characteristics^{1,2}.

[Flow from a channel 2 m wide into a pond 10 m diameter is a ratio of 5:1. A conventional pond would be sized 2 m by 6 m. It is expected the former would provide better settling characteristics. The trials recommended in Section 8.1 will address this uncertainty.](#)

The potential danger with having the inlet and outlet directly adjacent to each other (e.g. diagram on p 12) is that, particularly at low flows, the flow could short-circuit directly from the inlet to the outlet. [At low flows there are little/no suspended sediments in the water column of such watercourses \(refer Eivers et al. In prep; shallow lakes monitoring report from UoW\).](#)

A few other design details were questionable. Although steeper sidewalls are appropriate at depth in ponds such as these, vertical cut walls would be at risk of slumping and collapse. I would advise a 1:1 or 1:2 batter. [\(Amendment made to draft guidelines\).](#)

Also the way the crescent-shaped bund would function was not clear (although it is a nice design element). The cross-sectional view suggests the base of the bund would be at around the same relative height as the top of blocked drain and extend well up above it. It would seem that water would just flow over the filled in section of the drain rather than impound behind the crescent bund. Also the area provided for excavator maintenance (a good idea) looks somewhat constrained and would require the digger to traverse the drain.

5.3 and 6.3 Open water wetland: These are more commonly termed surface-flow or freewater surface wetlands³. [\(Amendment made to draft guidelines\).](#) Again no guidance is given on relative wetland size in relation to catchment size or run-off characteristics.

[The designs require trials and monitoring to determine expected efficiencies. This is recommended in Section 8.1. Expanding the guidelines to include tools to determine subcatchment discharge/flow rates/flow rating curves for small, predominantly ephemeral and/or artificial drainage catchments is recommended also in Section 8.2. This information would allow more accurate sizing of treatment systems relative to flows/discharge to ensure adequate hydraulic residence times.](#)

Some guidance relevant to NZ is however available elsewhere for tile_{drainage} which should perhaps be referenced⁴. [\(Amendment made to draft guidelines\).](#)

Matheson and Sukias (2010) as cited provide evidence for the importance of denitrification being the dominant nitrate removal mechanism, but there are more appropriate references available on the overall effectiveness of such wetlands in NZs. [\(Amendment made to draft guidelines\).](#)

The design example given (Fig 4) shows an oval wetland with a low length to width ratio and inlet and outlet placed directly adjacent to each other at 90 degrees, and a deep open water zone surrounded by a shallower marginal platform for emergent plants. Further information to justify these design choices relative to other approaches would be valuable. Although partial open-water zones can provide treatment benefits (e.g. increased solar exposure of faecal microbes) the extent of vegetation has generally been shown to be important for good treatment functioning for sediments and nutrients^{5,7}. [\(Amendment made to draft guidelines\).](#)

No real guidance is given as to outlet design, which can substantially modify the hydraulic performance of a wetland, for instance restricting outflow during flow events can buffer flows and increase residence times. [The outlet is placed at 90° to the inlet to encourage water flow in a circular motion \(as with eddies in a river\), thus increasing the residence time within the wetland. The level/height of the outlet relative to the inlet and the surrounding land will dictate the residence time. The inlet/outlet configuration is a practical way of constructing treatment systems off-line, finally creating a diversion from the watercourse into the completed system once planted etc. This approach has less impact on the in-stream environment as well as downstream as far less in-stream sediment is disturbed/re-suspended and mobilised downstream. Furthermore, retaining the existing channel as a by-pass channel which can be planted to assist with filtering particulates during flood events is a practical method of designing for flood events.](#)

5.4 and 6.4 Floating treatment wetland: Inclusion of this option is interesting and again I would be very interested to see the data you have to back up this choice in terms of effectiveness and cost. A recent comparison of FTW and conventional wetlands found FTWs to be relatively expensive.

[It would be useful to be able to read this information, clearly an important comparison. It would be good to understand the details of the site, the catchment characteristics, the watercourse, flow rates, nutrient/sediment loading etc.](#)

The need and practicalities of harvesting plants is an interesting question which would be good to provide some guidance on.

Also there is likely to be a need to maintain reasonable water depths for such systems to ensure plant survival and avoid roots becoming anchored into the bottom sediments. [Agreed. The trial of two FTW at Lake Kaituna has depths of 1.3 m below the rafts – hence the recommended depth range in the draft guidelines.](#)

7. Larger wetlands: I agree that it is good to require purpose-built designs for larger wetlands. The treatment train concept you propose is also good practice. However, for the specific example you give for Lake Ngaroto there would seem to be alternative design approaches that could result in a much greater proportion of wetland area connected to the dominant flow paths, and so greater residence time for treatment. The design proposed is rather attractive, but appears to be more focussed on biodiversity than water quality outcomes. It would be interesting to compare results from the systems proposed with those achievable with more conventional shallow densely vegetated surface-flow wetland designs.

Conclusions

These guidelines certainly present some fresh and innovative ideas. Proposing such approaches in a practical guideline document for widespread use usually does require a good evidence base to justify the recommendations being made and gain credibility. With good peer-reviewed evidence behind them the approaches outlined could indeed be promoted through such a guideline.

I would like to see more recognition in the guidelines of the important practical role the drainage ditches and channels being treated play from a farmer's perspective. Farmers and other landowners need to know that these systems will not detrimentally impact on their farming operations (e.g. pasture or crop productivity, soil pugging and health) or cause serious flooding issues. They also need to have some indication of the bang they can get for their buck in terms of contaminant reductions if they spend the money required to put them in on their farms.

[The designs require trials and monitoring to determine expected efficiencies as well as costs for construction.](#)

The Waikato Regional Council will of course want to ensure it is providing the best well-considered advice possible with the best outcomes for environmental quality. To ensure this I would recommend an expert working party is convened to review the content of the current guidelines and the evidence base behind them, and work constructively together to refine them further.

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NIWA

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