Diffuse-source agricultural sediment and nutrient attenuation by constructed wetlands

A systematic literature review to support development of guidelines

Prepared for DairyNZ

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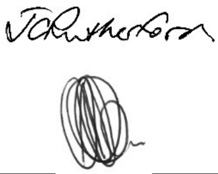	Reviewed by:	Kit Rutherford Neale Hudson
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Executive summary

Constructed wetlands (CWs) are contaminant mitigation tools with significant potential to help farmers meet water quality limits set under the National Policy Statement for Freshwater Management (NPS-FM). To support uptake and regulatory acceptance of CWs, accessible information is needed to guide their design and implementation, and define their contaminant attenuation efficacy (% contaminant reduction). To address these needs, DairyNZ and NIWA are collaborating on the INTERCEPTOR project – accelerating the uptake of constructed wetlands and riparian buffers. In part, this project aims to quantify CW contaminant attenuation efficacy across New Zealand to support their appropriate use and regulatory acceptance. This document systematically reviews the scientific literature to collate performance data and identify how contaminant attenuation efficacy is affected by design, climate and landscape. The information is then used to define appropriate CW performance expectations for the practical CW guidelines detailed in our companion report (Tanner et al. 2020). The catchment-scale effect of CW implementation on water quality and how to locate CWs within catchments for maximum water quality benefit are outside the scope of this report. The report contains four main sections:

(1) synthesis of previous New Zealand and international reviews

(2) systematic review of relevant quantitative field studies to identify key performance data and design characteristics

(3) derivation of CW performance guidelines for New Zealand conditions, and

(4) conclusions and directions for future CW research.

Synthesis of previous New Zealand and international reviews

In New Zealand, three reviews have previously provided estimates of CW performance in terms of attenuation efficacy of key contaminants in agricultural runoff:

- In the 'Stocktake Report', McKergow et al. (2007) estimated that CWs occupying between 1 and 2.5% of their contributing catchments would attenuate:
  - 60 to 80% of suspended sediment (TSS)
  - 30 to 50% of nitrogen (TN).
  - No P attenuation estimate was provided.
- When developing a wetland module for the Overseer™ farm nutrient budgeting tool, Rutherford et al. (2008) focused on nitrate attenuation in both natural and constructed wetlands. They based their attenuation estimates for CWs on water temperature and CW flow efficiency. Information was derived from modelling studies using performance information from 65 studies of New Zealand and international wetlands treating a range of nitrate-rich waters.
- Tanner et al. (2010) also targeted nitrate-N and provided design and performance advice for CWs intercepting tile drainage. Estimates of anticipated attenuation efficacy for CWs comprising up to 5% of their contributing catchment areas were provided.

Internationally, there are only two recent syntheses of performance data and design information for CWs receiving agricultural runoff, and neither address attenuation of the various species of N and P generated in agricultural run-off. The two studies are:

- A review by Kadlec and Wallace (2009) of CWs receiving event-driven flows from agricultural land. The variability in system design and contaminant loadings to which these CWs were subject was large; median attenuation efficacy (and range), based on concentrations, were:
  - 57% (12 to 87%) of sediment
  - 36% (-76 to 80%) of total phosphorus (TP)
  - 29% (-4 to 99%) of nitrate, and
  - 26% (-11 to 67%) of total nitrogen (TN).
- A review by Land et al. (2013) of TN and TP removal in restored or constructed wetlands used for treatment of urban, wastewater and agricultural runoff. For the relevant CWs in the review, median attenuations were:
  - 37% TN (95% confidence interval of 26 to 46%), and
  - 47% TP (95 % confidence interval of 37 to 56%).

We have documented CW performance in terms of **attenuation efficacy (% contaminant reduction)**, the difference in annual load measured in the CW outflow relative to the inflow load.

The flow paths intercepted by CWs have been categorised as: 1) **runoff**, which includes surface runoff and drains, 2) **drainage**, which includes all forms of subsurface flow including tile drainage, or 3) **mixed**, combinations of runoff and drainage. These different flow paths deliver contaminants in different forms, and as a consequence different attenuation should be anticipated.

Systematic review

Our review focused on field-scale CWs (not meso-scale devices or microcosms), occupying up to 6% of their contributing catchments, with data covering at least one year of hydrologic and contaminant loading (or concentration). For inclusion, these CWs had to be treating agricultural run-off, with the intercepted flow-path discernible. Sixteen studies of CWs were identified that satisfied these criteria; they represented 25 wetlands and a total period of 66 treatment years. Each treatment year provided a data point in our subsequent analysis. Median attenuation efficacies are summarised in Table i.

Table i: Median TN, TP and SS attenuation efficacies for field-scale New Zealand CWs intercepting runoff, drainage and a mixture of runoff and drainage identified in our review of the scientific literature. "na" indicates data not available.

Flow path intercepted	Median contaminant removal efficacy as proportion of inflow (inter quartile range)		
	TN	TP	SS
runoff and mixed	22% (16 to 30%)	41% (20 to 59%)	88% (83 to 89%)
drainage	30% (22 to 38%)	-52% (-105 to 1%)	na

The reported performance of CWs intercepting drainage in New Zealand was poorer than that derived from comparable international studies, but the performance of wetlands intercepting runoff was generally better than that reported in the international literature.

We used regression analysis to identify how contaminant loading and concentration, wetland size relative to contributing catchment (%) (hereafter termed 'relative size'), hydraulic loading rate, average annual temperature and average annual rainfall affect wetland efficacy in terms of contaminant attenuation. Three significant relationships were found:

- Attenuation efficacy increased as inflow TSS concentrations increased.
- TSS attenuation efficacy decreased as hydraulic loading rates increased.
- TN attenuation efficacy increased as CW relative size increased.

Guidelines

Based on our analysis of the available data on CWs in the New Zealand agricultural landscape, our review of overseas studies and expert opinion, we developed preliminary guidelines which are published separately (Tanner et al. 2020). Performance guidelines are given for:

- TSS attenuation in CWs whose catchment soils have <35% clay content,
- Nitrate-N and TN attenuation in CWs intercepting drainage and runoff and
- TP attenuation in CWs intercepting runoff or mixed runoff/drainage.

Variability across CW sizes and designs and their climates and landscape setting, combined with the relatively limited data and information available regarding CW performance in New Zealand agricultural landscapes, has limited our ability to quantify CW performance. Limitations and caveats apply to our estimates of CW attenuation efficacy for the contaminants investigated as discussed both in this document and in the companion CW guidelines report (Tanner et al. 2020).

Conclusions and implications for research

New Zealand studies furnished 16 estimates of annual attenuation efficacy which indicate that CWs intercepting runoff are highly effective at trapping TSS, moderately effective at trapping particulate P and N, but may act as sources of DRP. For CWs intercepting drainage, New Zealand studies show moderate attenuation of TN and nitrate-N but generally show some release of TP and DRP. Studies from other countries showed similar results for N attenuation but generally better performance for TP and DRP. Conversely, TSS attenuation was higher in New Zealand CWs than reported overseas, implying larger, faster settling particles enter New Zealand CWs.

New Zealand studies found high TN attenuation efficacy for large wetlands in small catchments (i.e., low loading per unit area of the wetland) but no significant relationship between nitrate-N attenuation efficacy and relative CW size, although this has been clearly shown in overseas studies. No statistically significant relationships were discernible between mass loading rate and attenuation efficacy for any of the contaminants investigated. However, attenuation efficacy for all contaminants investigated decreased with increasing hydraulic loading rate as a result of reducing contact and settling times.

Although numerous studies of CWs have been conducted, it is difficult to quantify attenuation efficacy. This is partly because key information is missing from much of the published literature. In addition, attenuation efficacy is affected by the interactions between internal CW characteristics (e.g., age, aspect ratio, vegetation type) with landscape (e.g., contaminant loading, particle size, soil type, slope) and climate variables (e.g., annual rainfall, rainfall variability, event rainfall, temperature). While studies have identified the processes that determine attenuation efficacy, attempts to quantify the effects of interactions have either not been attempted or have not been successful.

There have been relatively few studies of CWs in New Zealand and additional targeted studies would help improve design and our understanding of their contaminant attenuation efficacy and the processes that govern contaminant attenuation efficacy. Notably, additional research is required regarding:

- the attenuation and retention of TP and DRP
- what causes a wetland to become a source of DRP
- how N, P and sediment attenuation efficacy changes as CWs mature
- the form in which sediment (individual particles or aggregates) is transported to CWs, and
- the factors that affect sediment attenuation efficacy, particularly for fine particles (clays).

1 Introduction

Many landowners are in the process of identifying and implementing mitigations to reduce contaminants entering waterbodies under regional limit-setting processes required by the National Policy Statement for Freshwater Management (MfE 2017). Constructed wetlands (CWs) are key mitigation options available to farmers to reduce contaminant losses. Before landowners, agricultural industry bodies and councils commit to application of these mitigations to help improve the ecological health of water bodies and achieve attribute limits, more information is needed to quantify their capacity to improve water quality. To address this need, DairyNZ and NIWA are collaborating on the INTERCEPTOR project - accelerating the uptake of constructed wetlands and riparian buffers project which, among other things, aims to prepare preliminary guidance on CW design and contaminant attenuation performance. Scientifically defensible performance estimates are required to provide sufficient confidence for users to invest, and regulators to support, the uptake of CWs for mitigation of sediment and nutrient losses in farm runoff. In parallel, a range of on-the-ground multi-year case studies are being established to fill knowledge gaps and further refine attenuation rates and CW designs across New Zealand's agricultural landscapes and climate zones.

1.1 Project brief

The first step in this project was to review and summarise current knowledge on CW performance for attenuation of diffuse-source sediment, nitrogen (N) and phosphorus (P). Initially, faecal contaminants such as *E. coli* were included, however, it was agreed that due to a lack of published data these contaminants would be omitted from the project scope. The catchment-scale effect of CW implementation on water quality and specific guidance on location of CWs within catchments for maximum water quality benefit are also outside the scope of this report.

This review is focused on the field-scale performance of CWs. The review focuses on **inter-system variability** (viz., how the performance of similar CWs varies) and the factors (e.g., climate, landscape and design) that contribute to this variability. **Intra-system variability** (viz., how the performance of a wetland varies caused by seasonal and event-based factors) is outside the scope of this report.

The report contains four main sections:

- synthesis of previous New Zealand and international reviews
- systematic review of relevant quantitative field studies
- derivation of CW performance guidelines from the systematic review results, and
- a discussion that identifies how and where these guidelines should be used and provides direction for future CW research.

1.2 Definitions and key terminology

Constructed wetlands (CWs) are engineered systems that enhance biological, chemical and physical attenuation of a wide range of pollutants. Biological processes involving microbial communities and plants, as well as physical and chemical processes such as sedimentation, sorption and burial all contribute to pollutant attenuation in CWs. Four basic types of CW are potentially suitable for treatment of diffuse runoff, differentiated by their internal water regime and dominant macrophyte type: (1) surface-flow (SF; also known as free water surface; FWS), (2) subsurface flow (SSF), (3) floating emergent (FE); also known as floating treatment wetlands) and (4) hybrid systems. Surface-flow CWs are the focus of this report because they are the simplest and least expensive wetlands to create, they can cope with the variable flow characteristics of agricultural runoff, and they are not prone to clogging by sediment (unlike subsurface-flow CWs). Note that 'surface-flow CWs' refers to flow mechanisms within the wetland and not to the nature of the flow-path along which water moves to the wetland. Where the inflow pathway that delivers contaminants to the CW comprises overland flow, ephemeral surface flow and/or flow in ditches or drains we use the terminology 'runoff'. Where the inflow pathway comprises shallow subsurface flow (interflow), groundwater and/or tile drains the flow path is termed 'drainage'.

CW **attenuation efficacy** (%) can be calculated using either concentration or load (mass per unit time) in the outflow, relative to the inflow. Where reported, load or flow-proportional concentration multiplied by flow has been used to assess attenuation efficacy in this report. For CWs with defined inlet/outlet structures and lining, measurement of flow volume is straightforward and so concentration and load reduction estimates are readily available. For unlined systems, obtaining these values is less simple, but is possible provided both inflows and outflows are measured and estimates of volumes and concentrations of losses or gains to or from groundwater are known. The use of concentration changes alone to measure attenuation efficacy introduces problems because it may hide the effects of water losses (seepage out and evapotranspiration) or gains (seepage in and/or rainfall) on apparent CW performance. The use of load is preferable for inter-system comparisons as it reflects both the incoming contaminant and water volume variability.

A wetland's **hydraulic loading rate** (HLR; units $m^3\ m^{-2}\ d^{-1}$ or $m\ d^{-1}$) is the ratio between the rate of water flowing into a wetland ($m^3\ d^{-1}$) and the wetland area (m^3). The wetland water depth divided by the HLR gives you the **hydraulic residence time** (HRT; units d), which is defined as the theoretical time taken for a parcel of water to pass through a wetland assuming complete mixing. The HLR influences the HRT of water and contaminants in the wetland and so the time available for treatment processes to operate. HRT can be increased by increasing wetland depth however, this will not generally increase contaminant attenuation efficacy as increased depth does not increase the interactions between wetland features responsible for contaminant attenuation. **Hydraulic efficiency** is determined by the ratio between effective volume (how much of the available volume receives flow) and the uniformity of flow (or degree of dispersion) through the wetland. In practice it is determined from measurements during tracer tests or modelling. It may be reasonably approximated by dividing the time of the peak outflow concentration by the theoretical residence time.

The flow path intercepted by a CW has been categorised as either: 1) **runoff** which includes surface runoff and drains, 2) **drainage**, which includes all forms of subsurface including tile drainage, or 3) **mixed**, combinations of runoff and drainage. The justification for this categorisation is that these flow paths deliver contaminants in different forms. Runoff is dominated by particulates and the pollutants associated with them, drainage carries contaminants almost exclusively in dissolved form while mixed flow paths carry both particulates and dissolved contaminants.

2 Previous literature reviews

2.1 Introduction

Recent literature reviews were examined to summarise attenuation efficacy and identify critical factors that determine contaminant attenuation by CWs at the farm-scale. We utilised narrative reviews to identify key features that affect wetland performance (e.g. O'Geen et al. 2010). Data-based reviews allowed us to compare our results with those of others, potentially providing additional insights into CW performance. Information from existing New Zealand guidelines and tools (including models) have also been included.

2.2 Constructed wetland reviews

2.2.1 Existing New Zealand guidance on CW performance

Three New Zealand publications of nitrate-N removal exist – a guideline, a narrative review, and a published CW model. All of these documents build on the CW performance studies of Tanner et al. (2005) and subsequent modelling of nitrate removal based on the approach of Kadlec (2012), as outlined in Tanner and Kadlec (2013).

McKergow et al. (2007) summarised results for New Zealand CW research and estimated sediment and N attenuation performance for CWs treating either runoff or drainage and occupying either 1 or 2.5% of their contributing catchment area. No estimate of P attenuation was provided. Estimates for CWs intercepting runoff-dominated flows were extrapolated from drainage research and the professional judgement of the authors, informed by published international literature and local observations of contaminant attenuation by natural wetlands (Rutherford and Nguyen 2004, McKergow et al. 2006, Sukias and Nagels 2006). McKergow et al. (2007) used this information to identify the landscapes where CW would be useful and cost-effective. CWs treating drainage or runoff were most cost-effective for N removal ($15-120 per kg of N removed) in three scenarios: dairy/flat topography/moderately well-drained soils; dairy/flat topography/poorly drained soils; and intensive sheep and beef/rolling topography/heavy subsoil.

Rutherford et al. (2008) developed a CW module for OVERSEER® which estimated reduction of N load by a CW receiving drainage based on a calculated hydraulic loading rate, influent nitrate concentration, temperature and an estimated hydraulic flow efficiency class (flow path length: width ratio). Nitrate removal rates in the module are calculated using a first-order kinetic model (Kadlec 2012) based on mean removal rates and temperature coefficients from 65 New Zealand and overseas studies from mature wetland systems where inflow was nitrate-dominant (>80% of TN).

In 2010, NIWA published a set of design and performance guidelines for CW treatment of tile drainage, with a focus on sizing a wetland appropriately to maximise nitrate-N removal efficacy, as shown in Figure 2-1 (Tanner 2010). The guidelines, based on New Zealand farm-scale trials and modelling, combined with international data, allow users to estimate nitrate-N removal according to the relative size of a wetland (the ratio of wetland area to its contributing catchment area, expressed as a percentage value). The guidelines are specifically targeted at treating drainage (tile drainage) where N is predominantly in dissolved form (i.e., nitrate-N), rather than in particulate forms. The guidelines apply primarily to key dairying regions with rainfall ranges between 800 and 1400 mm/y. The N attenuation efficacy in the guidelines is lower than those derived by McKergow et al. (2007) from their scenario modelling, though they are similar to the *Upper performance band* in Figure 2-1.

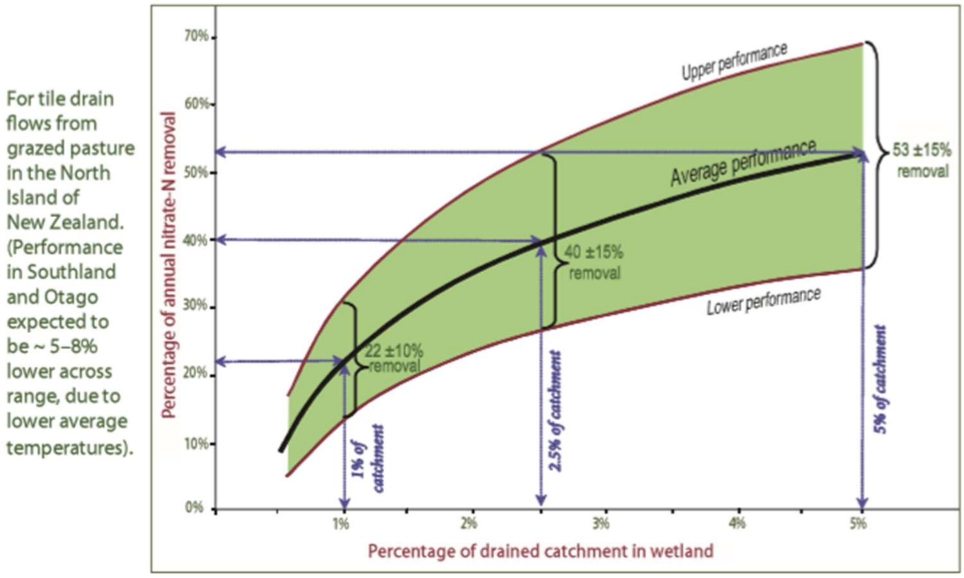

Figure 2-1: Nitrate-N removal by constructed wetlands treating tile drainage in relation to wetland size expressed as a percentage of its catchment (Tanner et al. 2010).

Table 2-1: Summary of New Zealand syntheses on contaminant attenuation performance of CWs.

Author & Title	Document type	Summary	Comments
McKergow et al. 2007	Cost-effectiveness evaluation	- previous New Zealand research summarised for scenario modelling of CWs treating surface and shallow sub-surface runoff: - CWs treating runoff removed 60%, 30% and 50-80% of sediment, N and P loads when 1% of catchment area, and 80%, 60% and 60-80% when 2.5% of catchment area. - CWs treating drainage removed 30-50% and 30% of sediment and N loads when 1% of catchment area, and 40-70% and 60% when 2.5% of catchment area. No removal of P was anticipated in these CWs without addition of P-sorbing material. - key features likely to enhance performance of wetlands treating drainage from both flow path types were: - low flow velocities and tortuous flow path - a mosaic of aerobic and anaerobic microenvironments, and - close contact between water, sediments, plants, detritus, and biofilms. - significant re-release of nutrients from plants with seasonal dieback unless harvested. - CW of all types (natural seepage, floating, constructed) and natural floodplain wetlands (low-lying paddocks on the floodplain managed to trap sediment and nutrients during over-bank flow events) were useful attenuation devices and most cost-effective (15-120 $/kg N) for three scenarios - dairy/flat topography/moderately well-drained; dairy/flat topography/poorly drained; and intensive sheep and beef/rolling topography/heavy subsoil.	-first synthesis of data; more farm-scale trial data are now available for New Zealand and internationally
Rutherford et al. 2008	Model	- CW drainage module for N in OVERSEER®. No sediment or P removal is included in the module. - OVERSEER® estimates the average nitrate-N concentration in drainage flow from annual nutrient yield and annual total drainage flows and estimates daily flow ($m^3\ d^{-1}$) and N load ($kg\ d^{-1}$). Daily flow/wetland area (m^2) gives hydraulic loading rate ($m\ d^{-1}$). - internal flow efficiency (based on length : width ratio) of the wetland is categorised as Type 1, 2 or 3 (lower number greater efficiency). - % nitrate reduction as a function of water temperature and flow efficiency based on 65 studies of New Zealand and international wetlands. - N load reduction is assumed to be the same as nitrate-N concentration reduction. - predictions are valid for hydraulic loading rates into the wetland of 0.05 to 0.8 $m\ d^{-1}$.	-designed for tile drain CW nitrate-N removal
Tanner et al. 2010	Guideline	- focus is CW-tile drain nitrate-N removal efficacy and provide estimates of anticipated performance for a range of relative wetland sizes when expressed as a % of their contributing catchment area. - provides advice on how to appropriately locate, size, design, and construct effective treatment wetlands including wetland planting, weed control, and maintenance.	-performance data provided in graphical format, including uncertainty (e.g., Figure 2-1 above) -for tile drain CW nitrate-N removal only

2.2.2 Recent international reviews

Kadlec and Wallace (2009) review data for CWs receiving event-driven flows from agricultural land. The variability in both system design (e.g., <0.1 to 20% of contributing catchment area) and contaminant loading to these CWs was large. Median attenuation efficacies (and range) of concentrations were 57% (12 to 87%) for sediment, 36% (-76 to 80%) for TP, 29% (-4 to 99%) for nitrate and 26% (-11 to 67%) for TN. Variability in each of the individual wetland performance summary values is due to event-to-event concentration variability, annual seasonal patterns and stochastic scatter. Kadlec and Wallace (2009) conclude that the very high contaminant and hydraulic loading rates may not allow time for interaction with the wetland biogeochemical cycle or microbial processes. Kadlec (2012) provides an update of available information on nitrate removal in CWs as part of his modelling review.

The most recent review, Land et al. (2013) considered TN and TP removal in restored or newly constructed wetlands of all types used for treatment of urban, wastewater and agricultural runoff. A significant proportion of the data included in this review was from small-scale mesocosm studies which are relevant for discerning factors affecting performance but less useful for determining real-world CW performance. Most studies included were surface-flow CWs treating mixed runoff or drainage from agricultural cropping land. For the CWs in the review, median removal efficacy was 37% for TN (95% confidence interval 26 to 46%) and 47% for TP (95 % confidence interval 37 to 56%) with median areal removal rates for all wetlands types of 93 g TN m^{-2} y^{-1} and 1.2 g TP m^{-2} y^{-1}. TN removal efficacy was negatively correlated with hydraulic loading rate (HLR) and positively correlated with annual average air temperature. For TP, removal efficacy was correlated with inlet TP concentrations (positive), HLR (negative), average air temperature (positive) and wetland area (positive). Land et al. (2016) did not find marked differences in performance between climate zones, wetland type, runoff type, wetland history (former land use) or water regime (type of hydraulic loading regime e.g., variable or steady inflow), although wetlands with precipitation-driven 'flashy' inflows ('hydrologic pulsing') showed significantly lower TP removal efficacy than wetlands with steady inflows. They make the distinction between mass removal rate (g m^{-2} yr^{-1}) and attenuation efficacy (% of the inflow removed). In general, high annual nutrient loading rates (g m^{-2} yr^{-1}) are associated with high removal rates (g m^{-2} yr^{-1}) while high hydraulic loading rates result in low attenuation efficacy (% removal). They also pointed out that no information regarding constructed wetland performance exists from long-term (>20 years) performance assessments.

3 Systematic review

We systematically reviewed existing scientific literature to find studies that could provide data to help answer the following question:

How effective are constructed wetlands at reducing loads of nitrogen, phosphorus and sediment, and how is this affected by landscape, climate and design characteristics?

3.1 Searches, eligibility criteria and data extraction

We considered only studies of surface-flow CWs, the most common type of CW used to treat agricultural drainage waters.

We used references within a recently published scientific review (Land et al. 2013) in combination with search results using ScienceDirect, Web of Science and Google Scholar. Searches were conducted using the following key words, either individually or in combination: wetland, constructed wetland, nitrogen, phosphorus, sediment, agriculture and diffuse pollution. Studies were included in the systematic review according to the following criteria:

- CW performance was reported or could be calculated as the change in sediment and/or nutrient loads before and after passage through the wetland.
- Constructed wetlands were treating agricultural run-off, not river or lake water or wastewater.
- The dominant flow-paths that the wetland intercepted were reported.
- The study investigated field-scale operational CWs (i.e., mesocosm studies were excluded).
- A minimum of 1-year of sediment and/or nutrient and hydraulic loading data were reported.
- CWs relative size was <6% of their contributing catchment area (greater than this is unrealistic in the New Zealand agricultural context).

Where available, the following information was extracted (or calculated) from the paper or research article:

- Hydrologic flow-paths transporting contaminants to the CW (e.g., runoff, drainage or a mixture of the two).
- Attenuation of contaminant concentration and/or load within the CW (i.e., relative – percentage reduction of incoming load or concentrations or specific retention – the rate of retention per unit area of the wetland).
- Hydraulic loading rate.
- Area of the CW as a proportion of the catchment that drains to the CW.
- Climate information, including type, annual average air temperature, precipitation during the study period where available, or annual average precipitation if no study period precipitation was reported.

Other factors that may have an effect on CW performance include wetland shape, the presence and type of macrophytes, the nature of the wetland sediment and the degree of aggregation of the soil particles entering the wetland. However, these aspects were not generally reported on and so were not considered in the systematic review. Each data point in our analysis represents one treatment year which was generally defined in each study as the hydrological year. So, a wetland that was studied for 3 years provided three data points in the analysis.

We categorised the study locations according to the Köppen-Geiger climate classification (Kottek et al. 2006). Studies from many countries are included in the Oceanic Climate classification (Cfb; Marine West Coast Climate; characterised as temperate, without marked dry season, warm summer), including New Zealand, Australia, France, Germany, Poland, the Netherlands, Denmark and the UK, but also southern parts of Norway, Sweden and Finland.

3.2 Analysis

The database search results and papers from the Land et al. (2013) review were screened to identify studies containing information that fitted our criteria. We then checked for linked articles to ensure that performance data from individual studies were not published in different forms and included multiple times in our analysis.

After applying the filtering rules, our CW systematic review yielded 16 studies, which investigated 25 constructed wetlands that had been studied for a total of 66 treatment years (i.e., some were studied for multiple years). Table 3-1 provides details on the data included in our analysis of each type of contaminant as not all studies investigate all our contaminants of interest. These constructed wetlands were distributed across eight countries with seven different climate types and had an average age of 1.8 years.

Only one of the published studies was based in New Zealand; it summarised the treatment performance of three wetlands in Northland, Southland and Waikato, for 3, 4 and 5 years respectively (Tanner and Sukias 2011). This built on the results of an earlier paper detailing the first 2 years performance of the Waikato CW (Tanner et al. 2005). One of the wetlands in the New Zealand study (Bog Burn in Southland) was set-up to restrict flow into the wetland under high flows. As this resulted in it not receiving the full flow from its contributing catchment, it was excluded from these analyses. Two further unpublished studies of constructed wetland performance in New Zealand carried out by NIWA were included, one in the Bay of Plenty studied over two years (Hudson and Nagels 2011) and one in the Waikato region with one year of data (Sukias et al. 2019). The Bay of Plenty CW was feed by a small stream, which violated one of our criteria, however, we felt the inclusion of a greater amount of data generated in New Zealand made this study relevant to our review. In total the New Zealand studies included 5 wetlands and furnished 16 annual performance estimates.

Table 3-1: A breakdown of the number of studies, wetlands and treatment years for each contaminant type. Flow path has been broken down into either runoff and mixed or drainage, and the number of treatment years for each contaminant provided. While the number of treatment years for each contaminant in New Zealand based studies is also detailed.

Contaminant	Studies	Constructed wetlands	Total number of treatment years	Runoff and mixed treatment years	Drainage treatment years	All New Zealand treatment years
TN	11	17	48	23	25	16
Nitrate-N	10	16	48	20	28	15
TSS	5	6	20	18	2	4
TP	14	22	59	25	34	16
DRP	9	13	42	17	25	15

We calculated the median and interquartile range of contaminant attenuation for studies from New Zealand, for other studies in oceanic climates, and for studies in non-oceanic climates across the three different flow path types. Regression analyses were used to explore relationships between contaminant attenuation and key landscape and design characteristics of CWs across all flow path types (runoff, drainage and mixed). The climate type and/or flow path intercepted were not used in the regression analysis because these data were not reported for all CWs and inclusion would have limited sample sizes excessively. However, to illustrate how these factors influenced performance, we have differentiated them on the graphs, where reported.

4 Systematic review results

4.1 Size, loading and performance summary

CWs in New Zealand generally occupy less of their contributing catchment areas and have higher hydraulic loading rates than CWs in other countries. The median relative size of overseas CWs was 1.25% (range 0.07 to 6%), larger than the New Zealand median of 0.75% (range 0.6 to 1.6%). The sizes of New Zealand CWs are below the recommended range of 1 to 5% of their contributing catchment area (Kadlec and Wallace 2009). The median hydraulic loading rate for New Zealand CWs was 34 m yr^{-1} (range 17 to 67) compared to 12 m yr^{-1} (range 1 to 660) in overseas CWs. Similarly, median N and P loading to New Zealand CWs were 252 g m^{-2} yr^{-1} (range 38 to 685) and 10.8 g m^{-2} yr^{-1} (range 1.0 to 21.9) respectively, greater than the loadings documented overseas. In the overseas studies median N and P loadings were 75 g m^{-2} yr^{-1} (range 10.9 to 2267), and 4.1 g m^{-2} yr^{-1} (range 0.1 to 191), respectively.

The median mass reduction (and interquartile range) for CWs in New Zealand (2 CWs and 4 treatment years) treating runoff waters was 88% (83 to 89%) for TSS, 22% (16 to 30%) for TN and 41% (20 to 59%) for TP. These performance values are higher than those found in the international studies (Table 4-1). The median mass reduction (and inter-quartile range) for New Zealand CWs treating drainage waters (3 CWs and 12 treatment years) was 30% (22 to 38%) for TN and -52% for TP (-105 to 1%). These performance values are lower than those derived from overseas studies (Table 4-1), particularly for TP.

There are relatively few studies of contaminant removal for New Zealand CWs treating agricultural runoff or drainage, with only 16 datapoints (annual attenuation efficacy estimates) available (four for CWs treating runoff and 12 treating drainage). There are no New Zealand or other oceanic climate zone studies of CWs intercepting mixed flow paths (Table 4-1). The maximum number of data points for a flow-path/climate grouping was 16 for “hot summer, humid continental” intercepting subsurface drainage – these fell into our non-oceanic grouping.

The limited results from New Zealand studies indicate that CWs intercepting runoff are highly effective at trapping TSS, moderately effective at trapping particulate P and N, but may act as sources of DRP (Table 4-1).

For CWs intercepting drainage, New Zealand studies show moderate attenuation of TN and nitrate-N, but generally show some release of TP and DRP. Studies from other oceanic and non-oceanic countries showed similar results for N attenuation but generally much better performance for TP and where measured, DRP (Table 4-1).

4.1.1 Factors affecting performance

We examined whether inflowing contaminant concentrations and loading rates, relative wetland size, annual average air temperature, average annual precipitation (or precipitation during the study period where available) and flow path affected contaminant attenuation efficacy. Our analyses for attenuation of TSS, TP and DRP were limited by data availability.

Only TSS attenuation was positively related to inflow concentration (Figure 4-1c). This relationship was likely related to an increase in the proportion of large particles (which subsequently fall out of suspension quickly) being transported into CW during periods when inflow TSS concentrations were high.

In general, high mass loading rates (g m^{-2} yr^{-1}) are associated with high areal attenuation rates (g m^{-2} yr^{-1}), while high hydraulic loading rates (m d^{-1}) are associated with reduced removal efficacy (%) (Land et al. 2013). However, no statistically significant relationships were discernible between mass loading rate and attenuation efficacy for any of the contaminants investigated (Figure 4-2). Thus, although visual examination of the data suggested that TN and nitrate-N attenuation increased as inflow concentrations increased, the relationships were not statistically significant even with outliers removed (Figure 4-1a).

TN treatment efficacy increased with relative CW size and declined with increasing wetland loading (Figure 4-2a and Figure 4-3a). Thus, in pasture systems with similar TN yields (kg TN ha^{-1} y^{-1}) one would expect higher efficacy for large wetlands in small catchments (i.e., low loading per unit area of the wetland). In contrast, no significant relationship was found between nitrate-N attenuation efficacy and relative CW size, although this has been clearly shown in other studies that considered larger wetland datasets (e.g., Kadlec 2012).

In terms of TSS, TP and DRP attenuation, the performance of New Zealand CWs differed from overseas CWs having similar relative sizes. For CWs in New Zealand, TSS attenuation was greater, implying larger, faster settling particles enter New Zealand CW compared to those overseas. However, TP and DRP attenuation efficacy in New Zealand CWs was lower than similar relative sized overseas CWs. Many CWs in New Zealand were sources of TP and DRP, a tendency which increased as hydraulic loading rates increased (Figure 4-4d, e). Similarly, TSS attenuation decreased as hydraulic loading increased (Figure 4-4c).

No statistically significant increases were found between TN or nitrate-N attenuation and air temperature (Figure 4-5a, b). TP and DRP attenuation tended to decrease as air temperature increased, but again no significant relationship was found. Annual precipitation was not found to be related to attenuation of any contaminant investigated (Figure 4-6).

CWs intercepting runoff demonstrated the most variable TN, nitrate-N and TP attenuation. CWs intercepting drainage had the most variable DRP attenuation (Figure 4-7d).

Table 4-1: Summary of constructed wetland contaminant attenuation efficacy by type of flow path intercepted and climate zone grouping. Values are medians (med) with interquartile range (IQR) and number of treatment years (n). na = no data available.

Type of flow path intercepted	Climate zone group	Attenuation of inflow contaminant load (%)														
		TSS			TP			DRP			TN			NO_3^-		
		med	IQR	n	med	IQR	n	med	IQR	n	med	IQR	n	med	IQR	n
Runoff	New Zealand	88	83 – 89	4	41	20 – 59	4	-12	-14 – 9	3	22	16 – 30	4	78	78 – 79	3
	Other oceanic	na			na			na			na			na		
	Non-oceanic	61	18 – 73	14	48	20 - 60	16	37	18-65	12	46	10 – 59	14	42	4 – 67	16
Drainage	New Zealand	na			-52	-105 – 1	12	-64	-170 – 10	12	30	22 – 38	12	45	32 – 54	12
	Other oceanic	35	33 – 37	2	42	28 – 43	13	43	34 – 50	4	na			na		
	Non-oceanic	na			15	-13 – 35	9	27	17 – 40	9	45	34 – 52	13	46	35 – 54	16
Mixed runoff and drainage	New Zealand	na			na			na			na			na		
	Other oceanic	na			na			na			na			na		
	Non-oceanic	na			40	37 – 44	5	43	38 – 47	2	15	14 – 32	5	37	34 – 40	2

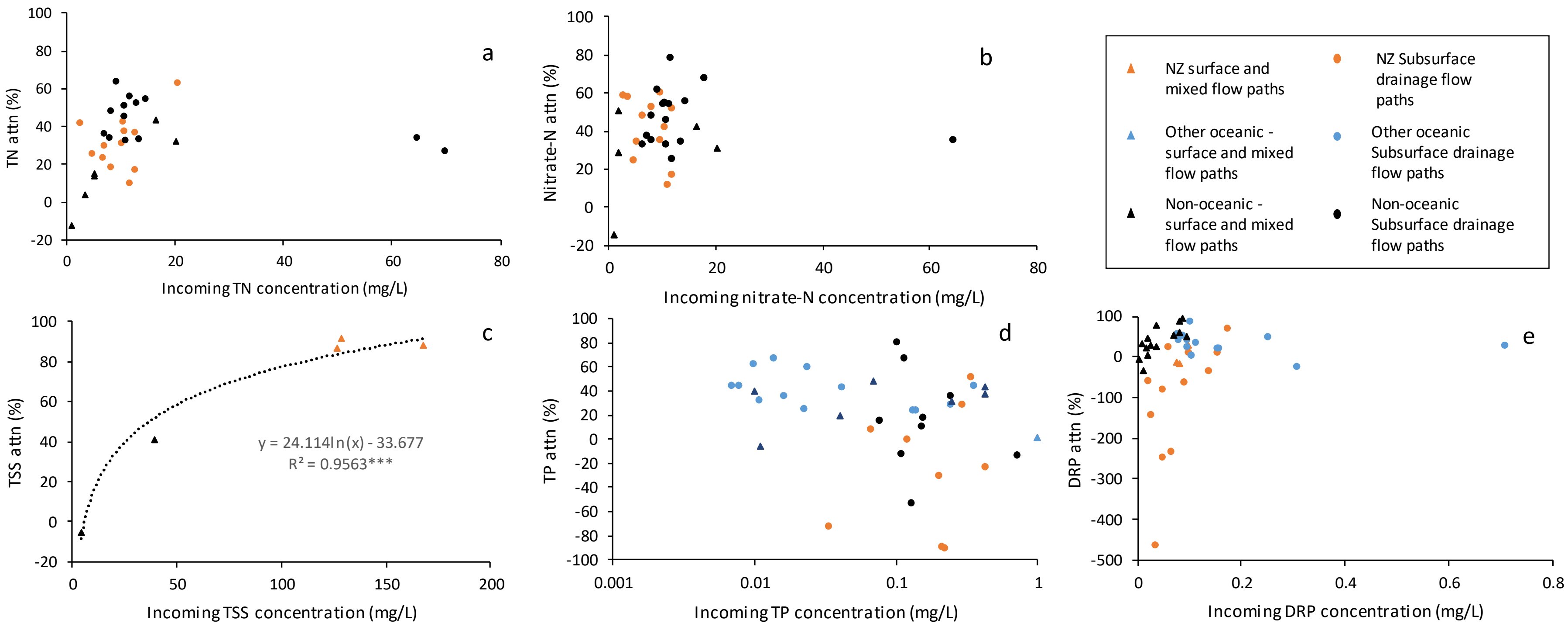

Figure 4-1: The effect of TN (a), nitrate-N (b), TSS (c), TP (d) and DRP (e) inflow concentration on respective CW attenuation efficacy. Significant relationships (across all data points) are indicated only where they were found (*p<0.05, **p<0.01, ***p<0.001). Note the logarithmic scale on the TP concentration axis.

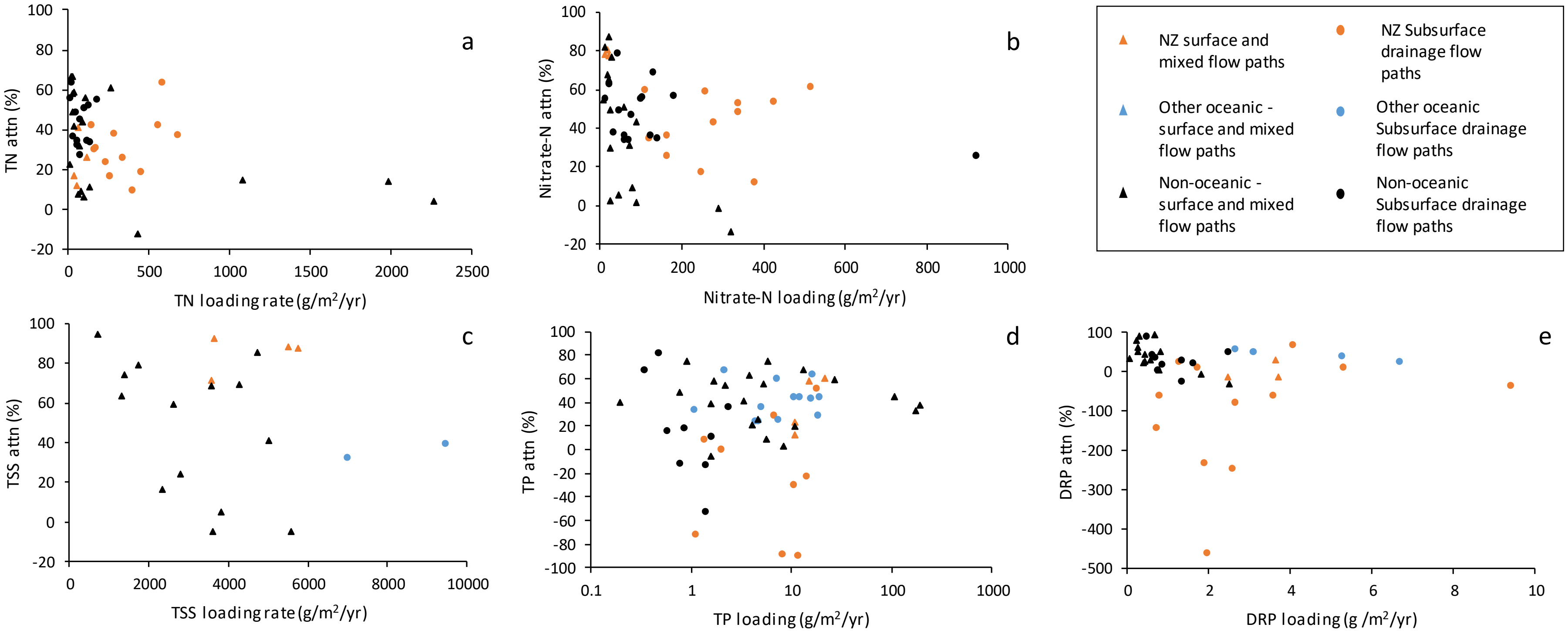

Figure 4-2: The effect of TN (a), nitrate-N (b), TSS (c), TP (d) and DRP (e) loading rate on CW attenuation efficacy. Significant relationships (across all data points) are indicated only where they were found (*p<0.05, **p<0.01, ***p<0.001). Note the logarithmic scale on the TP loading rate axis.

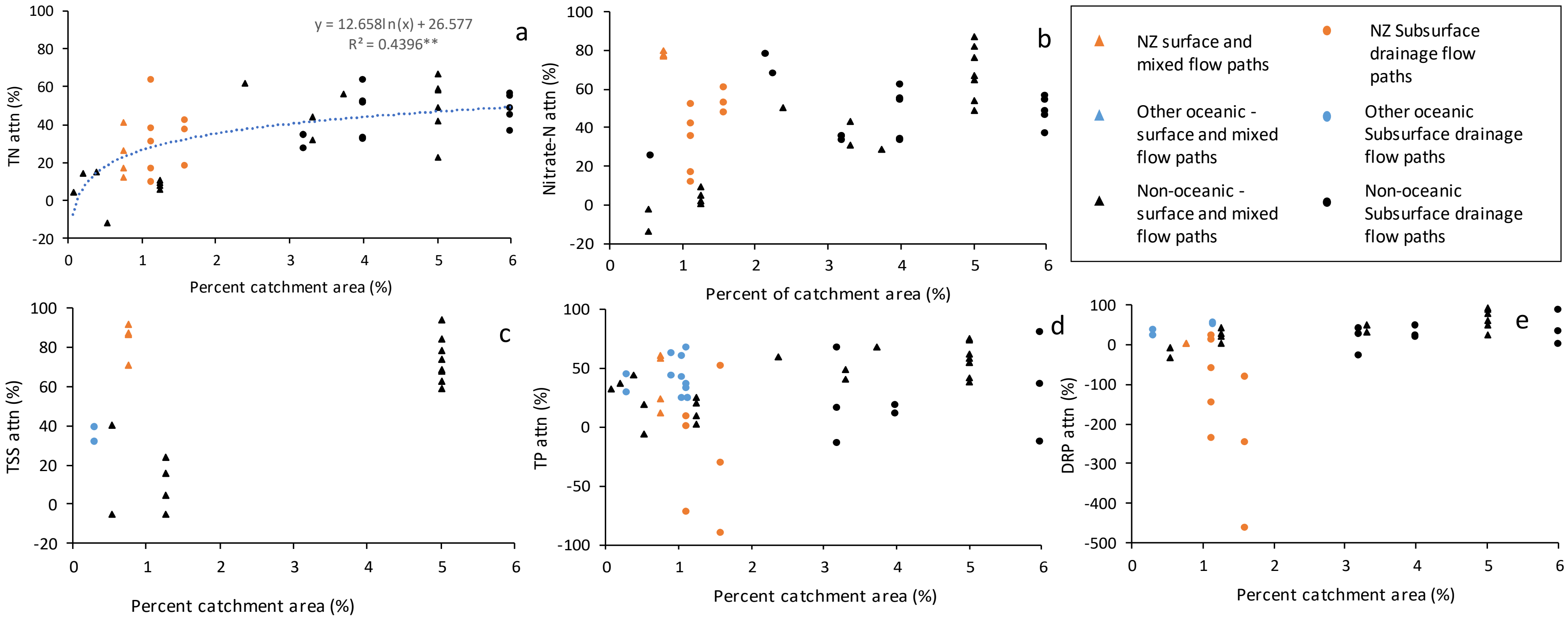

Figure 4-3: The effect of wetland size relative to its contributing catchment (%) on TN (a), nitrate-N (b), TSS (c), TP (d) and DRP (e) attenuation efficacy. Significant relationships (across all data points) are indicated only where they were found (*p<0.05, **p<0.01, ***p<0.001).

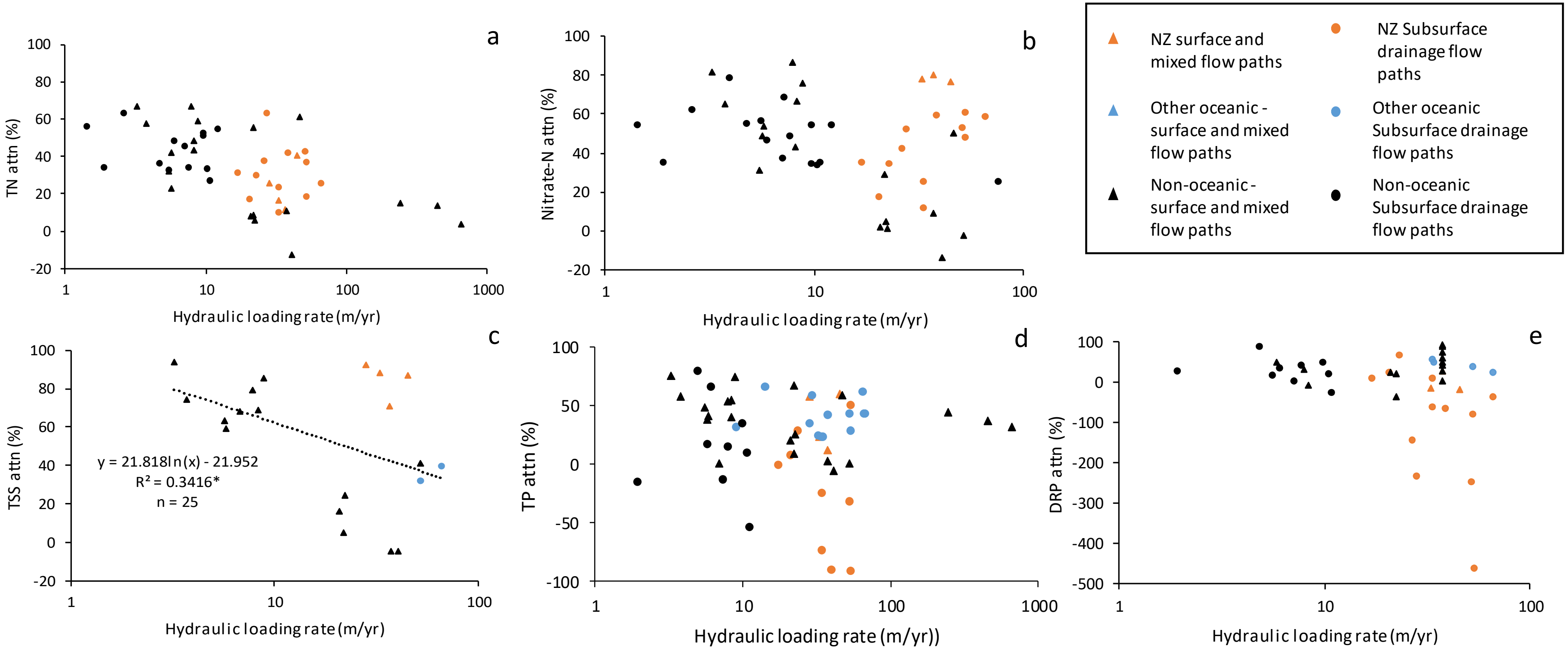

Figure 4-4: The effect of hydraulic loading rate on TN (a), nitrate-N (b), TSS (c), TP (d), and DRP (e) CW attenuation efficacy. Significant relationships (across all data points) are indicated only where they were found (*p<0.05, **p<0.01, ***p<0.001).

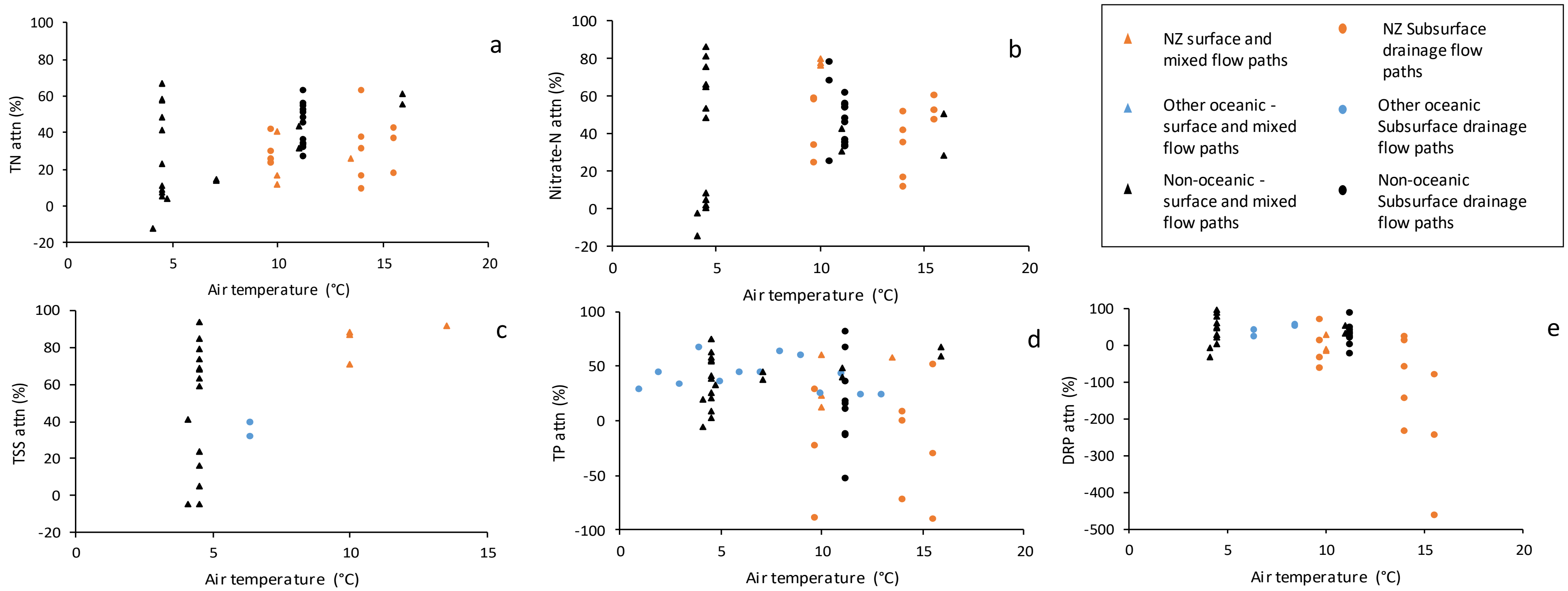

Figure 4-5: The effect of average annual air temperature on TN (a), nitrate-N (b), TSS (c), TP (d), and DRP (e) CW attenuation efficacy. Significant relationships (across all data points) are indicated only where they were found (*p<0.05, **p<0.01, ***p<0.001).

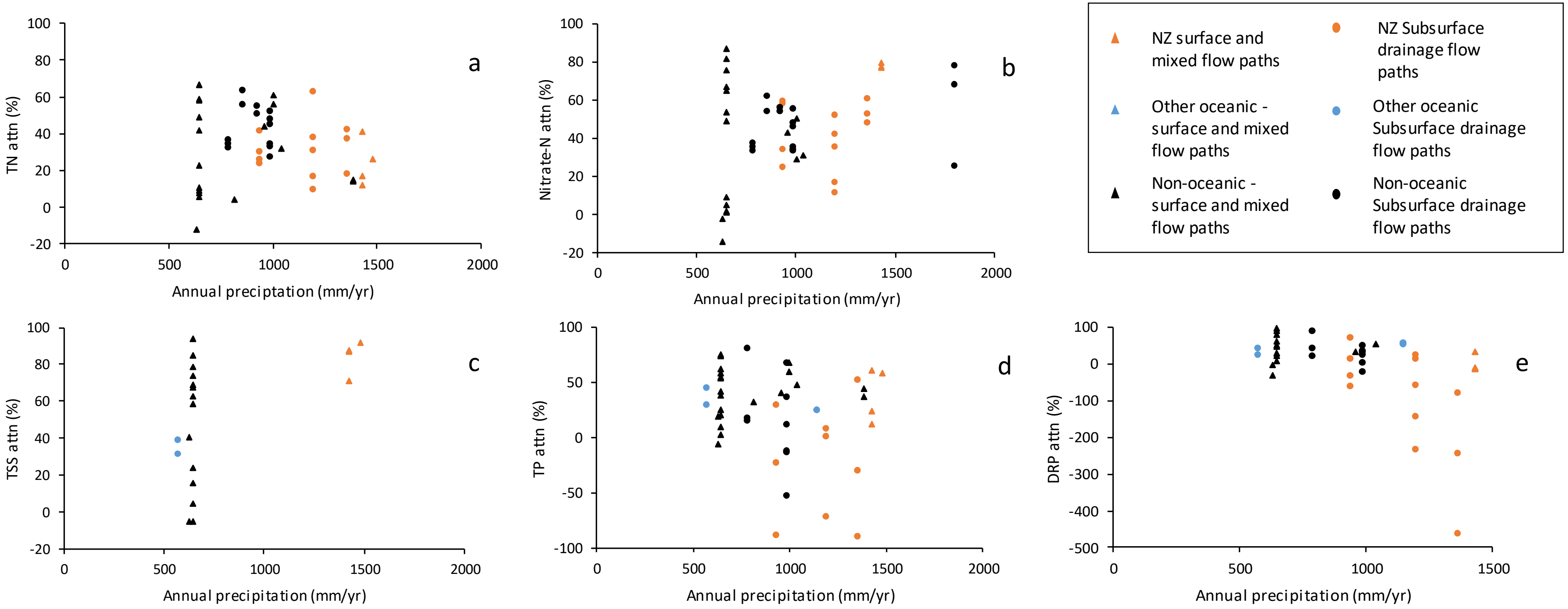

Figure 4-6: The effect of average annual precipitation (or precipitation during the study where reported) on TN (a), nitrate-N (b), TSS (c), TP (d) and DRP (e) CW attenuation efficacy. Significant relationships (across all data points) are indicated only where they were found (*p<0.05, **p<0.01, ***p<0.001).

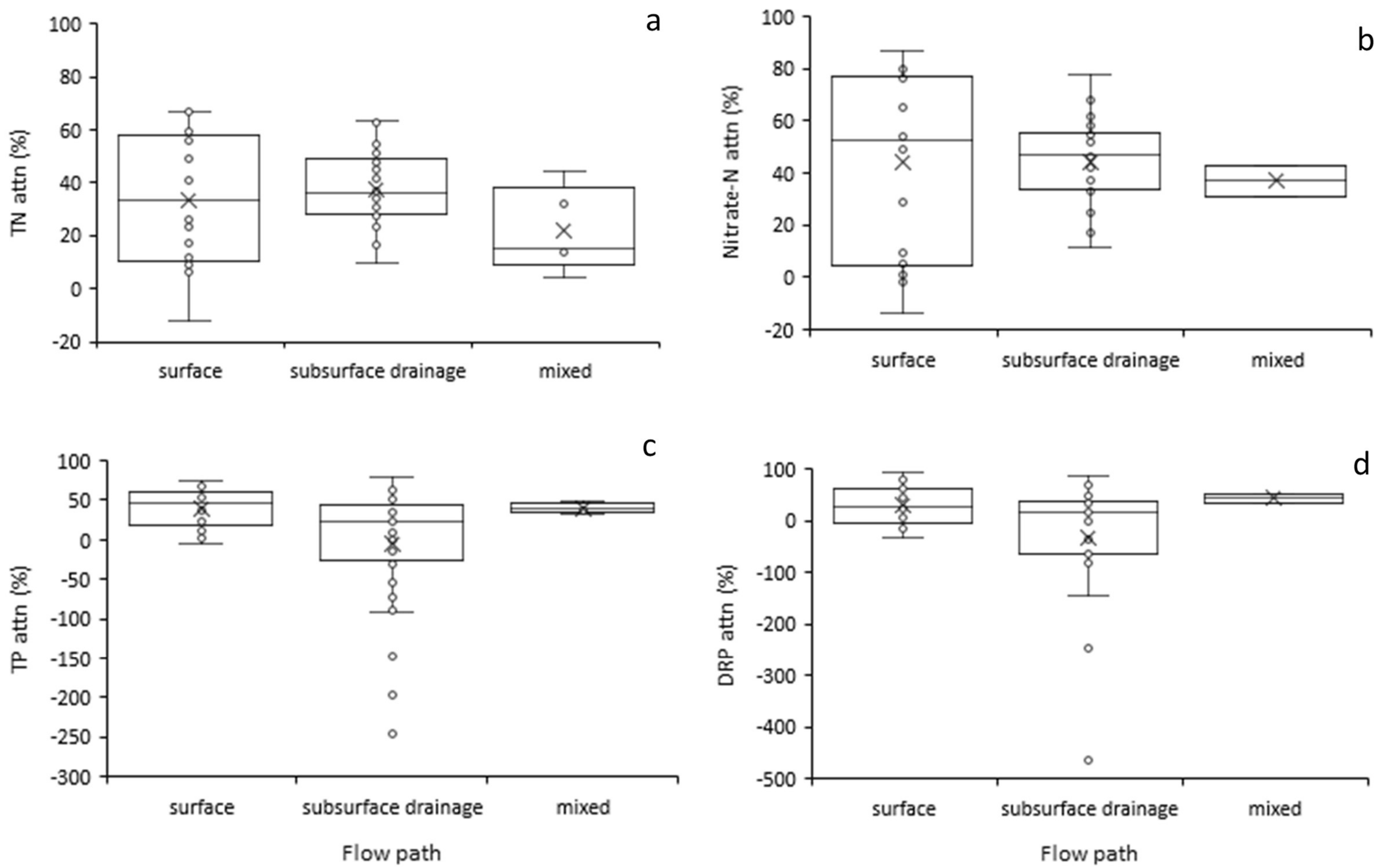

Figure 4-7: Box and whisker plots of TN, nitrate-N, TP and DRP CW attenuation efficacy expressed as a proportion of incoming load removed (%) for each different flow path intercepted. Plots show mean (x), median (line) and interquartile range. Data points are shown as open circles. Whiskers are the furthest data point within 1.5 x the interquartile range. Points outside this are considered outliers.

5 Discussion

5.1 Scale challenges with systematic reviews

Although CW research is maturing, summarising the many trials quantitatively is challenging. This is principally because key information is missing from much of the published literature. For example, annual average rainfall and temperature data are often reported whereas the temperature and rainfall during the study period would be more informative. There is also the disjunct between field or plot scale research which often investigates two or more similar CW to see how internal CW features affect attenuation efficacy, and landscape/climate scaled research which compares how (often internally dissimilar) CW attenuation efficacy is affected by different landscapes and climates. To our knowledge no studies have successfully reconciled the differences arising from these two scales of observation.

Predicting CW efficacy is further complicated by the interactions between internal CW characteristics, landscape and climate (and the different scales at which these factors interact). Controlled field- or plot-scale experimental studies with short timeframes or in a single location can demonstrate the effect of CW design characteristics on contaminant attenuation. However, when monitoring field-scale systems over multiple years, the relative influence of design characteristics on contaminant attenuation is increasingly dominated by climate and landscape. For example, because they cannot be controlled in field studies, inter-annual differences in precipitation and temperature cause substantial inter-year variability in CW performance (Kovacic et al. 2000, Tanner and Sukias 2011, Mendes et al. 2018). Additionally, landscape factors such as soil type or catchment slope can affect the performance of CWs even if they have the same design and experience the same climate. For example, differences in soil particle size distribution in a catchment will affect TSS and TP attenuation efficacy in CWs that are otherwise identical. Consequently, variability in reported CW performance (in terms of contaminant load attenuation) is high. While studies have identified the processes that determine attenuation efficacy, attempts to quantify the effects of differences in climate and landscape have not been successful. This makes it difficult to predict attenuation efficacy in any particular CW based on the available literature.

5.2 Factors affecting CW performance

5.2.1 Hydraulic loading rate

Our review found a negative relationship between hydraulic loading rate and SS attenuation efficacy. As HLR increases, residence time decreases and there is less time for SS to settle. However, in some Norwegian wetlands, located close to the sources of sediment they are intercepting, retention of sediment and particulate phosphorus was found increase as HLR increased. This arose because of the close proximity to the source of sediments allowing soil aggregates to be transported intact into the wetland during high rainfall events (high HLR), and settling very quickly (Braskerud et al. 2000, Braskerud 2003). Similar results would be expected in New Zealand if soil aggregates remain intact as they enter CWs.

For dissolved pollutants, in New Zealand pastoral systems, we would expect attenuation efficacy to decrease as HLR increased because higher HLR reduces the contact time and therefore the potential for interactions between wetland microbial and plant communities and sediment binding sites (Spieles and Mitsch 1999, Reinhardt et al. 2005, Woltemade and Woodward 2008, Johannesson et al. 2017). Although there are insufficient New Zealand data to confirm and quantify this relationship, we believe minimising HLR will increase attenuation efficacy. For DRP, the large scatter in the relationship between HLR and attenuation (including many results indicating a negative relationship) is likely due to anoxia developing within wetland sediments, which can cause release of DRP from internal TP stores (Díaz et al. 2012). DRP releases are most likely to occur when DRP sorption sites within sediments are at or close to saturation (Loeb et al. 2008, Meissner et al. 2008).

Modelling studies such as those of Tanner and Kadlec (2013) suggest that the variability of inflow to CW as influenced by climate and landscape characteristics will also impact overall treatment performance. Performance is likely to be best for CWs receiving steady flows that are evenly distributed across the CW (with long residence times), and worst during periodic high-inflow events when residence times are reduced. Highly variable or ephemeral flows may also fail to provide conditions able to sustain wetland plants or promote typical wetland treatment efficacies.

Limited space is available in agricultural landscapes, requiring optimisation of the ratio of wetland area to the area of its contributing catchment, to ensure efficient use of available land. If the ratio of wetland area to the area of its contributing catchment is small, the wetland is likely to have a high HLR, reducing the contact time between contaminants and active sites in the wetland, thereby reducing attenuation of dissolved nutrients. Conversely, if this ratio is large, gradual reduction of dissolved nutrient concentrations during passage through the wetland will reduce its apparent rate of areal mass removal. Generally, a wetland area of 1-5% of the catchment area is recommended for cost-effective contaminant removal (Kadlec and Wallace 2009).

5.2.2 Other factors affecting performance

In addition to the external factors that influence CW performance identified in our systematic review, internal factors such as shape, selection and density of macrophytes, and sediment accumulation affect the hydraulic efficiency of wetlands and their ability to attenuate contaminants. Wetland shape and depth profile affect the wetland's hydraulic residence time, water flow velocity, hydraulic efficiency and extent to which anoxia is likely to develop. Hydrological short-circuiting occurs when water flows preferentially from the inlet to the outlet via the shortest route, creating flow dead-zones. This decreases the interaction between water and wetland features responsible for contaminant attenuation and thereby impairs the ability of a wetland to attenuate dissolved nutrients. Even flow across the wetland can be promoted by providing appropriate shape, bathymetry, vegetation, position of inlets, outlets, and islands, and by avoiding embayments or deep channels oriented in the direction of flow (Persson et al. 1999, Su et al. 2009). Although, a long, thin wetland will theoretically have a greater hydraulic efficiency, it will also have higher flow velocities than a short, wide wetland of the same area, and anoxic conditions (which will promote nitrogen removal by microbial denitrification) are more likely to develop under the slower flow conditions of the wide wetland (Persson and Wittgren 2003).

Macrophytes play an important role in moderating short-circuiting, providing surfaces for deposition, reducing nutrient concentrations by biological uptake, providing carbon for denitrification and supplying oxygen to the root zone. Macrophytes may increase sedimentation rates by reducing re-suspension of previously deposited sediments (Kadlec and Knight 1996, Braskerud 2001) and reducing flow velocities and turbulence (Barko et al. 1991, Schmid et al. 2005). Evenly-distributed zones of macrophytes growing across the width of the wetland are ideal - patchily-distributed dense beds of macrophytes can in some cases promote flow channelization and short-circuiting (Fennessy et al. 1994). Biofilms attached to macrophytes can also accumulate suspended materials (Huang et al. 2008), adsorb dissolved nutrients and contribute to denitrification (Weisner et al. 1994). Net biological uptake by macrophytes is considered to be a relatively small nutrient attenuation process (Matheson and Sukias 2010), with the permanency of removal depending on either accumulation and burial of plant detritus or harvesting and removal from the wetland. Although the breakdown of plant biomass can release assimilated nutrients, the degraded material becomes a carbon source for many microbial processes, including denitrification (Bachand and Horne 1999, Kadlec 2008). Flooded organic soils and the epi-benthic layer of decaying plant detritus that accumulates on top of it has been shown to support high rates of denitrification (Fleming-Singer and Horne 2002, Hume et al. 2002). Oxygen released through macrophyte roots (Brix 1997) can create oxygenated micro-zones around their roots and oxic/anoxic boundaries where nitrification and denitrification can be coupled, enhancing transformation and removal of dissolved N from the aquatic system as nitrogen gas.

Accumulation of large amounts of sediment in wetlands can reduce internal wetland volumes (and associated hydraulic residence times) requiring sediment removal. Sediment deposition may also alter hydraulic properties of benthic materials in the wetland, reducing interaction between carbon sources, microbial populations and the dissolved nitrogen contaminant load. To facilitate sediment removal, inclusion of a sedimentation basin near the wetland entrance has been recommended (O'Geen et al. 2010).

5.3 Factors affecting CW performance in New Zealand agricultural landscapes

5.3.1 Inefficient DRP removal

The poor attenuation of P shown for New Zealand CWs is likely due to the fact that most of the systems were treating drainage flows in which DRP was the dominant fraction with minimal readily settleable, particulate-bound P content (Tanner and Sukias 2011). Furthermore, the sorption sites in the high P-status agricultural top-soils used as growth media in the wetlands were likely close to saturation with DRP (Loeb et al. 2008, Meissner et al. 2008), and so provided minimal additional DRP retention capacity. DRP binding and retention capacity of these soils is also likely to have been decreased under the anaerobic conditions present in the saturated organic-rich wetland soils (Reddy et al. 1999). Further investigation of methods to (a) reduce the DRP saturation of soils used in CWs (e.g., use of subsoils of lower P status) and/or (b) to increase the DRP-retention capacity of soils using amendments is required (e.g. Ballantine and Tanner 2010).

5.3.2 Temperature and N performance across New Zealand

Our systematic review did not identify a relationship between average annual air temperature and TN attenuation, although relationships between temperature and N attenuation have been described in other reviews (Kadlec and Wallace 2009, Land et al. 2013). Temperature is well known to regulate microbial rates of nitrogen cycling, including mineralisation, nitrification and denitrification, as well as rates of organic matter decomposition providing carbon sources to fuel denitrification (Kadlec and Reddy 2001). Temperature also influences rates of plant (including algal) growth and nutrient uptake (Pregitzer and King 2005). It is likely that inter-annual temperature variation during multi-year studies (which only report average annual temperature and annual average attenuation values) masked this relationship. For example, two CWs in Finland had an average annual temperature of 4.5°C. However, together they have 11 data points in our analysis with TN attenuation ranging from 6 to 67% (Koskiaho and Puustinen 2019). Such variability in TN attenuation for one average annual temperature, makes finding relationships between these variables difficult. The existing New Zealand guidelines for CW treatment of tile drainage (Tanner et al. 2010) suggests that nitrate-N removal performance in Otago and Southland is likely to be c. 5-8 percentage points lower than locations in the North Island. This analysis was based on data derived from New Zealand studies and expert opinion, so is more likely to reflect the effect of temperature on TN attenuation across New Zealand than say results from studies in Finland.

5.3.3 Climate and landscape effects on SS and TP performance

We might expect less effective attenuation of SS and TP in locations where there is more frequent rainfall and/or erodible soils with high clay content (CWs with higher hydraulic loading rate and contaminant concentrations). Larger CW area relative to catchment area wetlands are likely to be required, but the current New Zealand data are too limited to assess this. In addition to the propensity for larger storms to mobilise coarser materials more likely to settle (as noted previously), another complicating factor for SS and TP attenuation is whether or not soil aggregates remain intact when entering a wetland. Although soil aggregates are generally likely to settle more efficiently than dispersed fine particles, there is very limited information for New Zealand landscapes on the form in which particles are transported at edge-of-field and sub-catchment scale, and whether they remain intact or not during transport.

5.4 Derivation of CW performance guidelines

The availability of data and our mandate to develop contaminant attenuation performance guidelines appropriate for New Zealand, determined how we derived CW contaminant attenuation guidelines. For TN and TP sufficient data were available to create models using negative exponential equations. This type of equation is unable to utilise negative numbers, so any wetlands that released a contaminant (e.g., DRP) could not be represented in a model of this type. To create bounds around the performance predicted by these models, we used the 95% confidence interval. However, these models and confidence intervals were derived using the imperfect dataset (discussed above), and do not necessarily quantify how a CW of a given size relative to its catchment will attenuate contaminants in New Zealand. We reviewed the performance predicted by these models using the expert opinion of the project team members. This expert knowledge has been obtained from many years of CW performance assessment (both locally and overseas). This has enabled us to develop preliminary recommendations for contaminant attenuation by New Zealand CWs. We consider these suitable for regulatory purposes because they are based on conservative interpretation of available data, guided by expert opinion.

5.4.1 Total suspended solids

A significant negative relationship was found between hydraulic loading rate and TSS attenuation efficacy (Figure 4-4a). We did not find a corresponding positive relationship between CW relative size and TSS attenuation performance. This was surprising because as the relative size of a CW increases, we would expect hydraulic loading rate to decrease and, therefore, TSS attention to increase. As detailed in section 5.1, it is likely that variations in landscape (e.g., erosion rates and sediment yields) and climate features (e.g., rainfall and runoff) between CWs included in our analysis concealed this relationship. The most likely cause of this variability in performance is spatial differences in the soil particle size distributions and the possibility of soil aggregates being transported to CWs. There are several data points that sit below the lower bound of our recommended guidelines (Figure 5-1). Four of these points represent CWs comprising 1.25% of their contributing catchment area (the Rantamo-Seitteli CW in Finland). The poor apparent efficacy of this CW is the result of low TSS concentration entering the CW due to substantial forestation of the catchment, and the presence of bottom feeding fish that disturb sediments in the wetland (Koskiaho and Puustinen 2019). Another Finnish

CW, Alastaro, occupying 0.5% of its catchment fell below our expected lower bounds of TSS attenuation. In one year, this wetland released TSS and in another year attenuated 40% of its TSS load. This variability was attributed to differences in snow melt between the two years which may have affected hydraulic loading measurements (water entered the wetland as ice and snow, effectively diluting inflow measurements). Koskiaho et al. (2003) note that the size and shape of the wetland did not slow flow sufficiently in this CW to promote good settling and to avoid re-suspension of deposited clay soils previously mobilised in runoff from this catchment.

In deriving the guidelines, we have based our performance estimates of TSS CW attenuation efficacy on the relative size of CWs using expert opinion (Figure 5-1c). We have excluded areas where soils contain >35% clay. There was insufficient information given in most of the published studies to determine the soil characteristics of the catchments and the particle size distribution and degree of aggregation of the SS entering the wetlands. Clay soils contain a high proportion of fine, ultrafine and colloidal particles, which, consistent with their settling velocity (predicted by Stoke's Law), require considerable time and/or other processes such as flocculation, for deposition/adsorption to occur. The guidelines therefore focus on catchments where soils contain <35% clay content (viz., sand/silt/loam soils), which is the cut-off for clay soils used in New Zealand for classifying Land Use Capability (Lynn et al. 2009).

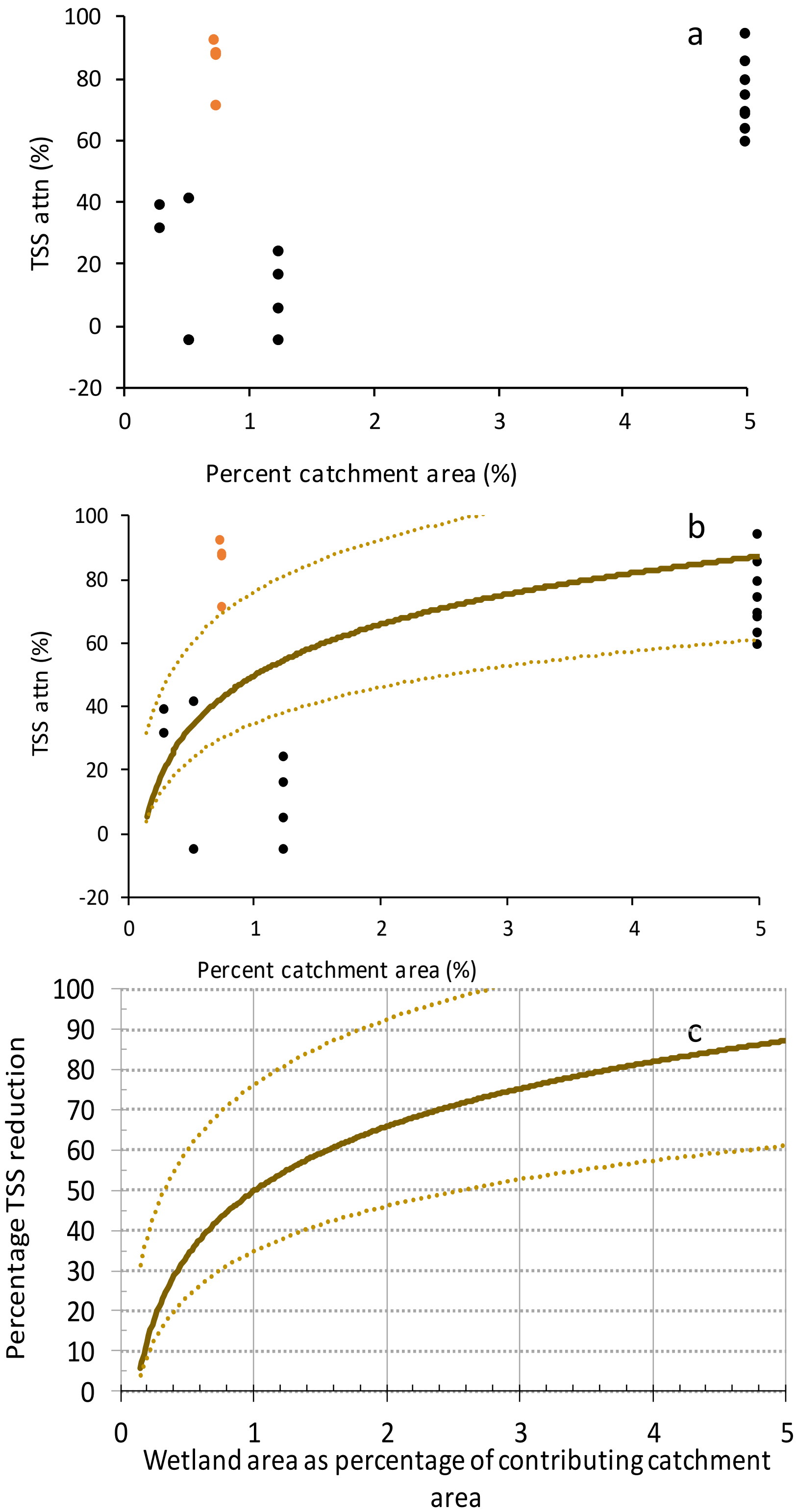

Figure 5-1: The derivation of predicted CW TSS attenuation efficacy. (a) review data from all climates and flow paths, (b) including expected long-term median annual performance and efficacy bounds fitted with expert opinion, (c) proposed final guideline relationship to wetland area. Orange dots in (a) and (b) represent New Zealand data and the black dots international data. These predictions do not apply to catchments with soil clay contents >35%.

5.4.2 Total phosphorus

No relationship between TP attenuation performance and relative CW size was found when CWs intercepting runoff and drainage were combined (Figure 4-3d). However, when this relationship was restricted to CWs intercepting runoff only, the relationship became significant ($p > 0.05$) (Figure 5-2a). Therefore, we have limited our guidelines to CWs intercepting runoff only. Figure 5-2b illustrates that the performance observed in New Zealand CWs is similar to our modelled data until a CW reaches approximately 1% of its catchment in size, from where observed performance is more conservative than modelled. There are 4 data points that lie outside our lower limits and 6 points above our upper limits. The 4 data points below our lower limits come from three CWs, two located in Finland and one in New Zealand. Factors common to these CWs include their small relative sizes, clay soil types and poorer performances during years when they were subject to high hydraulic loading rates. Despite observing lower efficacies for a number of systems, it is our expert opinion that when use of highly P-saturated soils is avoided during construction of CWs, the average TP attenuation efficacy will generally fall within the range indicated in our guidelines. It should be noted that all the wetlands studied were relatively "young" when their performance was monitored (i.e., their performance was assessed within a few years of construction). There is potential for gradual reduction in P reduction efficiency as wetlands mature, due to saturation of P-sorption sites and accumulation of P-rich sediments. In time, P-rich sediments have the potential to desorb bound P during seasonal wetting and drying or where conditions in the wetland become strongly anaerobic. (Reddy et al. 1999). For this reason, performance assessment based on data derived from longer-term studies are advised.

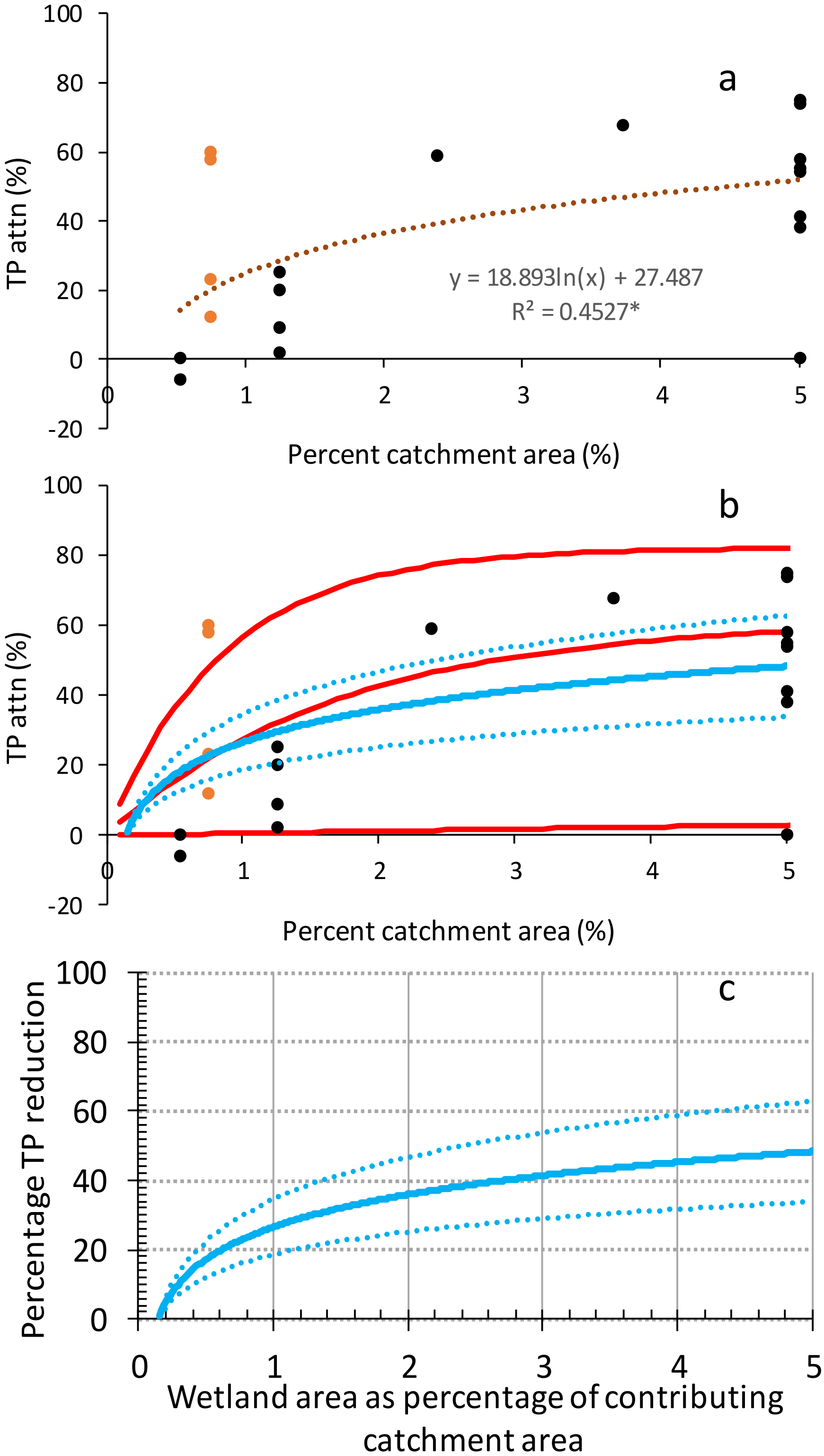

Figure 5-2: The derivation of predicted TP attenuation efficacy for CWs intercepting runoff/drainage. (a) data from the systematic review with fitted logarithmic curve (*p<0.05), (b) review data with (in red) modelled data with 95% confidence intervals and (in blue) expected long-term median annual performance and efficacy bounds fitted with expert opinion, and (c) proposed final guideline relationship to wetland area. The generation of the upper and lower bounds are based on expert opinion. Orange dots in (a) and (b) represent New Zealand data and the black dots international data, the data presented are from runoff only. This curve does not apply to CW intercepting either drainage or mixed surface and subsurface flow-paths.

5.4.3 Total nitrogen

Across all flow path types, a significant relationship between the TN attenuation efficacy and relative CW size was evident (Figure 4-3a). Our New Zealand contaminant attenuation performance guidelines include temperature, despite field data not showing that average annual temperature had a significant effect on TN attenuation efficacy. As discussed above, the relationship between temperature and TN and nitrate-N (which characteristically comprises 70-80% of TN) attenuation efficacy is well established in the scientific literature (Kadlec and Wallace 2009, Land et al. 2013). We have based our temperature-based predictions of TN removal performance on the results of simple dynamic kinetic models (e.g., Kadlec 2012; Tanner and Kadlec 2013) calibrated across multiple years of flow and attenuation for New Zealand CWs. Both the warm and the cold region models sit within the bounds of the modelled and 95% confidence interval based expected performances (Figure 5-3b). Although uncertainty is moderate, the draft guidelines provide the best available estimates of expected long-term average CW attenuation rates for TN under New Zealand conditions.

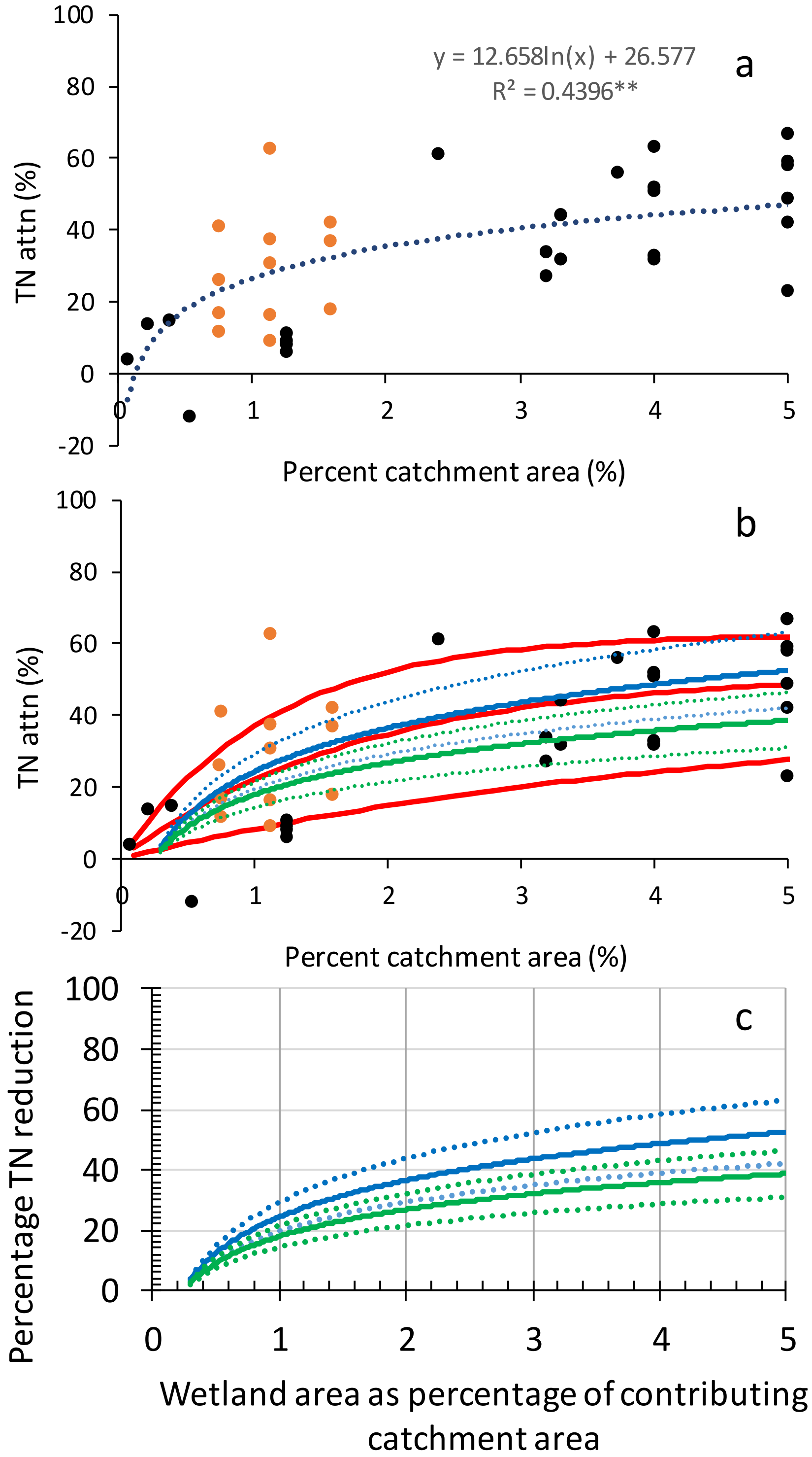

Figure 5-3: The derivation of predicted CW TN attenuation.. (a) data from the systematic review with fitted logarithmic curve (**p<0.01). (b) review data with (in red) modelled data and 95% confidence intervals, and (in blue and green) expected long-term median annual performance and efficacy bounds for warm and cool regions, respectively, based on expert opinion. (c) proposed final guidelines for warm (blue) and cold (green) regions of New Zealand. Orange dots in (a) and (b) represent New Zealand data and the black dots represent international data. The blue and green dotted lines in (c) represent expected long-term performance efficacy bounds for warm and cold climate zones.

6 Conclusions

6.1 CW performance and factors effecting performance

The objectives of this review were to collate published attenuation performance data from New Zealand and overseas for field-scale CWs subject to natural hydrological processes, to determine variability across CWs, to relate variability to different flow paths, and to derive CW performance guidelines for New Zealand conditions. We investigated the importance of inflowing contaminant concentrations and loading rates, wetland size relative to that of its contributing catchment, annual average air temperature, average annual precipitation (or precipitation during the study period, where available), intercepted flow path ,and design features for N, P and sediment attenuation.

- For nitrogen we found:
  - Median N attenuation rates of 22% for surface runoff/drainage and 30% for subsurface drainage in New Zealand CWs.
  - N attenuation was higher in CWs with low hydraulic and N loading rates (which correlated to wetland/catchment area ratio), and
  - less variable treatment of drainage waters than for runoff.
- For phosphorus we found:
  - Median attenuation efficacy for New Zealand CWs intercepting runoff flow paths was 41%.
  - Relatively poor performance of New Zealand CWs treating drainage in terms of P attenuation is probably related to DRP-release from high P soils used in the wetland.
  - It may be possible to improve P mitigation performance through use of soil with low P saturation, and inclusion of P-binding agents in wetland media selected during the design and construction phase.
- For sediment we found:
  - Median sediment attenuation efficacies of 88% for New Zealand CWs.
  - The limited data suggest that mass attenuation (kg ha^{-1} yr^{-1}) increases with increasing inflow concentration.
  - Sediment attenuation efficacy decreases with increasing hydraulic loading rate.
- Existing New Zealand CWs tend to have smaller wetland/catchment area ratios, and experience higher hydraulic and contaminant loading rates than CWs in other countries. Increasing wetland/catchment area ratios (e.g., by establishing multiple CWs within a sub-catchment) may be a useful strategy to improve catchment water quality outcomes.

Proposed CW performance curves for New Zealand pastoral farming conditions were derived from relevant local and international data, modelling studies and expert opinion. These have been included in preliminary CW guidelines (Tanner et al. 2020).

6.2 Implications for research

- Long-term performance of CWs is poorly investigated. Future work should include monitoring CWs as they mature because limited studies suggest attenuation efficacy for some contaminants decreases over time.
- Attenuation efficacy data currently focus on nitrate-N. There are data gaps for sediment in runoff and mixed runoff/drainage CWs and organic nutrient forms in all CWs. New CW trials in the Waikato, Taranaki and Golden Bay should provide additional data to support guideline development for runoff CWs for nitrate-N, TN and SS.
- Guidelines and tools are often for single contaminant/flow-path pairings. A challenge is to formulate tools that address multiple contaminants and trade-offs across scales. New tools are required that combine available spatial data resources (e.g., soils, topography, land cover) with ground survey and local knowledge to identify dominant contaminant flow pathways, and guide effective CW design and location within catchments.
- Identify P saturation levels (number of occupied soil P binding sites compared to the total number of potential sites) for different soils where CW DRP releases become likely, and/or how the addition of P-binding substrates (Ballantine and Tanner 2010) or soils with low P saturation may help to improve the P attenuation performance of New Zealand's CWs
- Investigate the factors affecting soil aggregate transportation and longevity after mobilisation to better-predict CW sediment removal and guide optimal CW location.

7 Acknowledgements

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