Seepage wetland protection review

Prepared for DairyNZ

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Executive summary

DairyNZ want to be able to provide clear guidance on the importance and management of seepage wetlands on farms, and to determine whether protection of the many seepage wetlands on pastoral lands is likely to be a useful water quality improvement strategy.

This report reviews the effectiveness of seepage wetlands for water quality protection, focusing on New Zealand studies but also referring to key overseas studies. Seepage wetland characteristics are summarised. The challenges and methods used in seepage wetland science are outlined and findings of New Zealand studies summarised. A simple conceptual model describes the relationships between wetland disturbance impacts, responses and outcomes. Wetland management options are explored and information gaps, many relating to wetland management are identified.

Seepage wetlands are mainly fed by subsurface flow via springs which emerge from a single point and seeps which emerge from the ground along a line or surface without a distinct origin. Their saturation status may range between temporary dryness and permanent saturation. Seepage wetlands commonly have mixed vegetation, including wetland grasses, rushes, and sedges. Seepage wetland soils are typically a dense mat of plant roots (up to 15 cm depth) overlying an unconsolidated saturated organic soil (porosities 0.6-0.9) and underlain by a less permeable soil layer, such as clay. They are depositional landforms – located at the change of slope where particulate solids, including mineral sediments and organic matter, accumulates. No New Zealand research has described seepage wetland genesis; some will be remnant wetlands, others will be more recent features that evolved in response to land clearance.

Seepage wetland science is challenging; inputs are diffuse, soils are unconsolidated, there are hot spots and hot moments of biogeochemical activity and key processes, such as denitrification are challenging to measure. The inability to measure and sample a wetland's diffuse inputs creates a gap in our understanding of wetland-scale water quality improvements. A variety of methods are used to overcome our inability to measure input information, including examining related systems, microcosm or mesocosm experiments, environmental tracers and modelling

Sub-wetland scale experiments can be used to identify transformations and estimate their rates under varying conditions. These experimental studies have shown:

1. rapid removal of nitrate from subsurface flow in seepage wetland soils, but postulated the potential for nitrogen to be transported across the wetland surface during rainfall events with minimal removal (Burns and Nguyen 2002)

2. seepage wetlands can remove nitrate from surface flow at high rates during dry weather (25% of added nitrate removed over a distance of 1.5 m) (Rutherford and Nguyen 2004)

3. seepage wetland soil properties vary with depth, for example hydraulic conductivity is higher in the top 10 cm (Rutherford and Nguyen 2004; Rutherford et al. 2000), and

4. wetland plants promote denitrification through rhizosphere oxidation of highly anoxic soils, discouraging dissimilatory reduction of nitrate to ammonium (Matheson et al. 2002).

Constructed wetlands provide guidance on the long-term areal removal rates for seepage wetlands. Mature constructed wetland removal rates vary with season and hydraulic loading, but systems have average areal nitrate removal rates[1] in the order of 230 to 280 mg N m^{-2} d^{-1}. Reported rates from studies of seepage wetlands in New Zealand are highly variable: 30 – 8100 mg N m^{-2} d^{-1}. The range of likely areal removal rates is therefore wide, and can be expected to vary with season and loading.

Within agricultural areas, drainage and grazing have the potential to significantly affect the water quality services of wetlands by altering soil-water contact, nutrient cycling, wetland vegetation and soils. When cattle enter wetlands water quality can be degraded directly through faecal and urine inputs and soil disturbance, and indirectly by altering soil physical properties (e.g., compaction) and damaging vegetation (e.g., by treading and herbivory). Large increases in *E. coli* concentration (Collins 2004) and high total and organic nitrogen exports (McKergow et al. 2012) have been measured from small, shallow seepage wetlands following livestock incursions. Cattle tend to be wary of deeper wetland zones and these zones are often left largely intact and may provide a water quality buffer to inflows and damaged shallow zones (Hughes et al. 2016).

Currently fencing is used to prevent cattle becoming entrapped in deeper wetland soils. For water quality improvement shallow wetlands are likely to benefit from livestock exclusion more than deeper wetlands. Cattle access shallow wetlands (< ~ 1 m depth) and tend to avoid deep (> ~1 m depth) wetlands. This “rule of thumb” may not apply to seepage wetlands with a channel of flowing water – exclusion of cattle from such wetlands would be beneficial regardless of depth.

Portable electric fences are often installed to prevent cattle access and offer a flexible solution. Permanent fencing is also used, and where the adjacent slopes are steep ‘benches’ are sometimes constructed. We do not recommend benches as they expose large areas of bare earth which may be vulnerable to erosion and they are likely to divert hillslope surface runoff away from wetlands, bypassing opportunities for contaminant buffering.

Planting seepage wetlands with large vegetation (e.g., flax, shrubs, trees) is not advised for water quality purposes alone. Vegetation cover plays an important water quality role by providing organic matter and to promote denitrification and trapping solids, however, in mature seepage wetlands with larger plants (e.g., flax, shrubs and trees) channels are common. Channels, similar to drains, reduce the contact time between the water and soil. Further work is required to assess whether the use of barriers, such as wood, are feasible and effective methods to reduce channelisation.

Identification and management of seepage wetlands will improve water quality outcomes, particularly livestock exclusion. However, experimental trials of seasonal livestock exclusion and vegetation management techniques have not been undertaken. Key information gaps include: prevention of channelisation, determining the ‘effective’ area of land draining to the wetland, seasonal grazing management and the importance of particulate and dissolved organic nitrogen and stabilising seepage wetlands during extreme events.

[1] Quoted removal rates have been corrected to a standard temperature of 20C for ease of comparison.

1 Introduction

1.1 Project brief

DairyNZ want to be able to provide clear guidance on the importance and management of seepage wetlands on farms, and to determine whether protection of the many seepage wetlands on pastoral lands is likely to be a useful water quality improvement strategy. Specifically, DairyNZ would like answers to the following questions:

1. What level of water quality improvement do existing unprotected seepage wetlands provide?
2. Does failure to protect seepage wetlands represent a threat to surface and groundwater quality?
3. What level of protection (in terms of stock exclusion, re-vegetation, establishment of extensive buffers, management of connectivity etc.,) is likely to be required to achieve measurable water quality improvements?
4. How do the likely water quality benefits arising from protection of seepage wetlands compare to those likely from other on-farm mitigation options that do not involve wetlands?
5. Can existing seepage wetlands be modified to enhance nutrient removal?
6. What gaps in knowledge regarding seepage wetlands need to be addressed before water quality benefits may be realised?

1.2 Project design

This project is a desktop study and reviews the existing literature, focusing on New Zealand studies where possible. A simple conceptual model describes the relationships between wetland disturbance impacts, responses and outcomes.

Our key tasks were to:

1. Review existing literature regarding the role of seepage wetlands in achieving water quality outcomes.
2. Review unpublished NIWA data and photographic records of the impact of livestock on monitored seepage wetlands and summarise the information derived from this assessment.
3. Build a simple conceptual model that describes wetland formation in pastoral landscapes.
4. Use a conceptual model to describe the role of key wetland characteristics on probable water quality outcomes from protection. This task will be undertaken at a workshop and developed using expert opinion.
5. Identify knowledge gaps and actions likely to fill them.

1.3 What is a seepage wetland?

The New Zealand Resource Management Act 1991 defines wetlands as ‘permanently or intermittently wet areas, shallow water, and land water margins that support a natural ecosystem of plants and animals adapted to wet conditions’. Natural wetlands have been called the “kidneys of the landscape” because of their ability to store, assimilate and transform contaminants lost from farmland before they reach waterways.

Many terms are used in the literature to describe areas with damp, organic soils in paddocks or along the edges of stream channels. They range from small areas at the head of stream channels to larger “swamps”. Such features may be known as: wet swales (Rutherford et al. 2009), riparian wetlands (Wantzen and Junk 2009), seeps (O'Driscoll and DeWalle 2010, Williams et al. 2015), groundwater seeps (Gold et al. 2001, Williams et al. 2015, Kaur et al. 2016), groundwater slope wetlands (Bullock and Acreman 2003, Mitsch and Gosselink 2007), headwater swamps (Hill and Waddington 1993), pastoral wetlands (Collins 2004), valley-bottom wetlands (Merot et al. 2006) and hillside wetlands (Wantzen and Junk 2009).

In this report we use the term **seepage wetland** to define small wetlands that are mainly fed by subsurface flow via springs and seeps and are connected to streams. Springs usually emerge from a single point, while seeps emerge or “ooze” from the ground along a line or surface without a distinct origin (WMO 2012, Toth 1971). Their saturation status may range between temporary dryness and permanent saturation.

Seepage wetlands are commonly located in surface and sub-surface flow convergence zones, with steep slopes fringing their margins. Water enters seepage wetlands via springs, seeps in through the banks of the wetland, and enters as surface runoff during rain events. Outflow rates are typically low and steady during dry weather, but short “flushes” occur during rainfall events driven by rain falling directly on the saturated soils plus surface runoff from the catchment. Generally surface water depth is shallow (or completely absent) although channels may form that take up a small proportion of total area but convey a large proportion of the outflow. Some seepage wetlands flow permanently, some cease to flow during droughts, while others cease to flow at the outlet in summer (Rutherford et al. 2009).

Wetlands are frequently called “sponges”, implying that they soak up rain and release it slowly (Newson 2010). This generic statement is not applicable to all wetlands as their hydrological inputs and functions vary. Seepage wetlands that are already saturated are unlikely to hold more water or attenuate flood flows (see Acreman and Holden 2013, Bullock and Acreman 2003).

Common characteristics of these wetlands include a shallow layer of organic soil, underlain by a less permeable soil layer, such as clay. The wetland “soil” is typically a small volume of unconsolidated organic material suspended in water; bulk densities are typically in around 0.1 – 0.3 g/cm^3 and porosities around 0.6-0.9 (Rutherford et al. 2000; Rutherford and Nyugen 2004, McKergow unpublished; Sukias unpublished). The top ~15 cm is typically a dense mat of vegetation roots.

Seepage wetlands commonly have mixed vegetation, including wetland grasses, rushes, sedges and raupo (Table 1). Wetland plants are capable of growing in soils that are often or constantly saturated during the growing season; dominant species are either obligate (occurs almost always in wetlands) or facultative wetland plants (occurs usually in wetlands; see Clarkson et al. 2013).

Table 1: **Summary of wetland characteristics.** After Rutherford et al. (2008).

Attribute	Characteristics
Location	Flow convergence zones, usually feed permanent streams.
Vegetation types	Wetland grasses (e.g., *Glyceria declinata, Holcus lantus* – usually introduced species), rushes (e.g., Juncus spp.), sedges (e.g., Carex spp.), raupo (*Typha orientalis*), with some larger shrubs/trees (e.g., Leptospermum spp., Salix spp.).
Surface soils	Organic, black, smelly, saturated, porous.
Sub-surface soils	Low permeability layer underlying surface soils, distinct change in porosity between surface and subsurface soils.
Standing water	Visible much of the time although may be dry in summer.
Response to rain	Rapid flow increase due to saturation of wetland, with rapid spilling (fill and spill).
Baseflow	Steady, low.
Hydroperiod	May dry up in summer.
Water source	Springs. Seeps (where water flows out of the ground). Surface runoff.

Due to their small size (10 to 5,000 m^2), seepage wetlands are rarely identified in regional wetland inventories or managed any differently from surrounding pasture. In some areas (e.g., northern edge of Lake Taupo) seepage wetlands cover 5% of catchment area (McKergow et al. 2007).

Wetland surface slopes vary - wetlands were research has been conducted have slopes ranging from flat (1°) to moderate (11°; see sections 2.2 and 2.3). Some wetlands have multiple levels, with steps or "knick points" between levels. For example, the RC wetland in the Tutaeuaua catchment has three distinct levels (McKergow et al. 2012).

1.4 Why do these wetlands form?

Seepage wetlands are depositional landforms – located at the change of slope where particulate solids, including mineral sediments and organic matter, accumulates (Figure 1). Groundwater (seeps or springs) entering these deposits provides conditions suitable for decomposition of accumulated organic matter. Organic soils develop over time composed primarily of the remains of plants in various stages of decomposition which accumulate as a result of anaerobic conditions (Mitsch and Gosselink 2007). Wetland vegetation enhances deposition of particulate organics and helps retain solids.

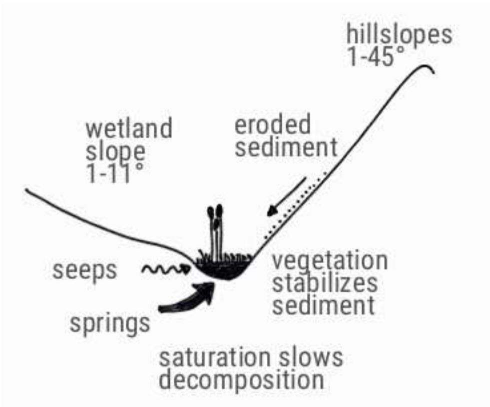


Figure 1: Seepage wetland formation.

To date no research has been published on the genesis of seepage wetlands in New Zealand. It is unlikely that all pasture seepage wetlands have the same landscape evolution; some will be relic wetlands formed before land clearance, while others will have formed more recently. Seepage wetlands have been reported in Australia and form when sediment, generated by catchment disturbance, is deposited on gully floors and colonised by emergent macrophytes (Zierholz et al. 2001). We hypothesise that some seepage wetlands in New Zealand have formed in a similar way.

Local evidence supporting the theory of deposition landforms includes cross sectional surveys and visual assessments of sedimentation. Cross sectional surveys of wetland soil depth (viz., the depth of unconsolidated, organic wetland soils) in the Tutaeuaua and Whatawhata catchments suggests that many are infilled channels, with large shallower areas and smaller deep areas (e.g., Figure 2). In parts of the Waikato, land clearance has transformed forested stream headwaters to headwater seepage wetlands (Bill Garland, per. comm., 29 July 2016).

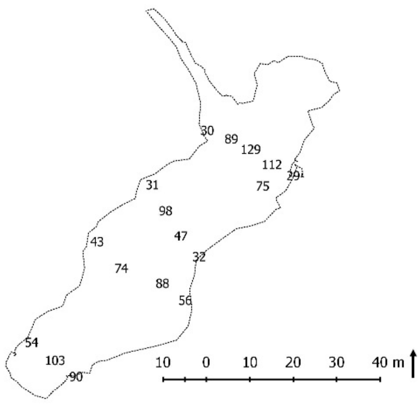


Figure 2: Depth (cm) below ground surface of unconsolidated, organic soils in LT wetland, Tutaeuaua catchment. Lucy McKergow, March 2006.

An unpublished long-term NIWA photographic record of mass-wasting (Figure 3) in the steep Pukemanga catchment (Whatawhata Research Station) shows deposition at the bottom of a hillslope in a low slope wetland (Figure 4). This is a likely mechanism for the formation of seepage wetlands elsewhere in the catchment. Wetland soils at Whatawhata are clays that have become organically enriched through the deposition of detritus from the catchment and the decay of wetland vegetation (Rutherford and Nguyen 2004).

Figure 3: Mass-wasting in the Pukemanga wetland catchment. Photo taken by Kerry Costley NIWA, 1 October 2014.

Figure 4: Sediment deposition on the Pukemanga wetland. Photo taken by Kerry Costley NIWA, 1 October 2014.

Wetland scour has been observed on hillslopes and in riparian zones as a mechanism that modified seepage wetlands. For example, the Barkers wetland (Whatawhata Research Station) was scoured out during a large storm in 2000 (Figure 5) although it re-established quickly. After a large storm, it was observed that fresh sediment had been deposited in the Barkers wetland, although it was not clear if it came from hillslope erosion or scouring within the wetland. In the Tutaeuaua catchment, a 150 year return period thunderstorm eroded riparian wetlands (see Figure 6, Figure 7).

Figure 5: Scoured Barker's wetland at Whatawhata. Photo taken by Rob Collins, NIWA

Figure 6: Riparian wetlands in the Tutaeuaua stream channel prior to an extreme rainfall event (compare with Figure 7). Photo taken by Lucy McKergow, 16 September 2005.

Figure 7: Tutaeuaua stream channel after an intense storm (compare with Figure 6). Photo taken by Lucy McKergow, 21 December 2005.

In other catchments, such as the low slope Kiwitahi wetland (Nguyen et al. 2002) and the Tutaeuaua catchment near Taupo, artificial structures (e.g., culverts and fences) may also contribute to wetland formation and stability.

2 Seepage wetlands and water quality

2.1 Key pollutants and attenuation

Nitrogen can be transported by water in several different forms, including dissolved inorganic N (nitrate, ammonium and nitrite), dissolved organic N and particulate associated N (e.g., particulate organic N and adsorbed ammonium, Table 2). Forms of nitrogen can change during transport due to biogeochemical transformations (e.g., mineralisation of organic N, nitrification of ammonium).

Pollutant		**Forms**			
	Total nitrogen (TN; excluding nitrogen gas)				
Nitrogen	organic nitrogen		inorganic nitrogen		
	dissolved (DON)	particulate (PON)	ammonium (NH_4)	nitrite (NO_2)	Nitrate (NO_3)
	Total phosphorus (TP)				
Phosphorus	dissolved/filterable/soluble(<0.45 µm)		particulate (PP)		
	reactive (DRP)	organic (DOP)	organic	inorganic	
	Suspended solids (SS)				
Suspended solids	organic		mineral		
			clay	silt	
	Faecal microbes				
Faecal microbes	viruses	bacteria	protozoa		
	Indicator: enteroviruses, phages	Indicator: *E. coli*	Indicator: *Clostridium perfringens* spores		
	Pathogens: human enteroviruses & adenoviruses, noroviruses, rotoviruses, hepatitis A	Pathogens: *E. coli, Campylobacter, Salmonella*	Pathogens: *Giardia, Cryptosporidium*		

Table 2: Forms of pollutants (nutrients after McCutcheon et al. 1993). (McKergow, unpublished).

Phosphorus may be transported in soluble and particulate forms, with particulate P including P sorbed (incorporated into or adhering to the surface) by soil particles and organic matter (Table 2). It is worth noting that dissolved or soluble P are imprecise terms, as the filtrate could be a mixture of dissolved forms and P attached to colloidal material that passes through the 0.45 µm filter.

Suspended solids consists of mineral or organic material (e.g., microbes, living organisms, organic particles; Droppo 2001; Table 2). Suspended sediment is operationally defined as fine-grained particles that are retained by an arbitrarily defined filter (typically 0.7 µm) but is mostly fine grained mineral particles (<63 µm) or low density organic particles (up to ~ 1 mm; Davies-Colley and Smith 2001). Organic and mineral suspended solids can act as a vector for sorbed nutrients (e.g., P, Haygarth et al. 1997) and faecal microbes (e.g., Oliver et al. 2007).

Faecal material is a source of enteric viruses, bacteria, cysts and oocysts and parasitic protozoa (Table 2). Faecal contamination is usually detected by testing for *indicator microorganisms*, such as *E.*

coli, that are consistently present in faecal wastes (Donnison and Ross 1999). It is assumed that if these organisms occur in stream water, then other more pathogenic micro-organisms such as *Campylobacter, Salmonella, Cryptosporidium* and *Giardia* are also likely to be present if a disease outbreak occurs. The key source of faecal contamination on farms is grazing livestock, although wild and feral animals can be an additional source. Faecal microbes may be introduced to freshwaters via "direct" (i.e., deposited directly into stream) or "indirect" pathways, such as the transmission of fresh or aged faecal matter in surface runoff, subsurface flows or drainage (Collins et al. 2007).

Attenuation is the transformation, temporary storage and/or permanent loss of pollutants between where they are generated (e.g., paddock, farm track) and where they impact water quality (e.g., stream, lake, estuary). Attenuation may be physical (e.g., flow attenuation, deposition), chemical (e.g., sorption, precipitation) or biological (e.g., assimilation, denitrification; Table 3).

Table 3: Wetland attenuation process definitions and constraints. (P = phosphorus; N = nitrogen; F = faecal microbes; after McKergow et al. 2008).

Attenuation process (and pollutant)	Description/definition	Constraints	References
Flow attenuation	Storage and attenuation of flood runoff.	Sufficient storage, location in landscape, flooding.	Mitsch 1992; Acreman and Holden 2013, Bullock and Acreman 2003.
Deposition	Settling of sediment, flocs, detritus, phytoplankton from the water column e.g., floodplains, soil deposits.	Low velocities promote settling; larger particles settle quickly, fines slowly. May be remobilised in subsequent events.	Knighton 1984.
Filtering	Sieving of coarse particles by plants or finer particles/microbes by soil matrix.	Porous barrier e.g., dense grass cover or soil.	Dosskey 2001; Oliver et al. 2005.
Assimilation (N & P)	Removal of dissolved inorganic or organic nutrients from water or soil water by plants, algae or microbial biomass.	Unless plants/algae are harvested and removed, nutrients will be released during senescence and decomposition. Microbial assimilation most significant when there is a large supply of carbonaceous material, e.g., plant litter, sawdust.	Vincent and Downes 1980; Howard-Williams et al. 1982; Matheson et al. 2002.
Denitrification (N)	Microbial production of nitric oxide (NO), nitrous oxide (N_2O) and nitrogen gas (N_2) from nitrate.	Denitrifying microbial community, plus sufficient contact time with low oxygen (suboxic) conditions, carbon source, nitrate available.	Matheson et al. 2002, Seitzinger et al. 2006, Wallenstein et al. 2006 ; Burgin and Hamilton 2007; Burgin et al. 2010, Rivett et al. 2008.
Precipitation (P)	Removal of components from solution by their mutual combination forming a new solid-phase compound.	P: soil chemistry (Fe III, Ca, Al), pH, redox potential	Reddy et al. 1999; Baldwin et al. 2002.

Attenuation process (and pollutant)	Description/definition	Constraints	References
Sorption (P and F)	Physical or chemical bonding of molecules to the surface of or within sediments, soil matrix or plants.	P: soil chemistry (FeO_x, AlO_x and clays), redox potential, solid type, pH, P concentration, organic matter; P may be released under anoxic conditions. Faecal microbes: presence of salts and organic matter, pH.	Holtan et al. 1988; Reddy et al. 1999 ; Baldwin et al. 2002; Ferguson et al. 2003.
Microbial inactivation (F)	Inactivation of faecal microbes by unfavourable environmental conditions.	Many stressors including temperature extremes, pH extremes, low soil water potential, high ammonia concentrations and organic matter contents, UV exposure, oxic conditions, and predation.	Ferguson et al. 2003; Oliver et al. 2005.

2.2 Wetland science

Wetlands are hotspots of hydrological and biogeochemical processes. However, quantifying these processes at the wetland scale is technically challenging for several reasons:

1. the diffuse water and nutrient inputs (seeps and springs) are challenging to locate, isolate and measure (Rutherford et al. 2009)
2. traditional physical soil measurement techniques are unsuitable for highly porous and "weak" wetland soils (Rutherford and Nguyen 2004)
3. small areas (hotspots) and brief periods (hot moments) often account for a high percentage of activity (Cooper 1990, McClain et al. 2003), and
4. biogeochemical processes, such as denitrification, are extremely challenging to measure (Groffman et al. 2006).

In an ideal world seepage wetland water quality performance would be assessed as the difference between inputs and outputs, similar to simple inlet and outlet comparisons made for constructed wetlands receiving piped flows (e.g., Tanner and Sukias, 2010). With seepage wetlands the challenge of measuring the diffuse inputs (and sometimes outputs, Rutherford et al. 2009) typically precludes this type of experimental design and hence the quantitative assessment of wetland contaminant removal. This inability to measure and sample a wetland's diffuse inputs creates a gap in our understanding of wetland-scale water quality improvements.

A variety of methods are used to overcome some of the gaps in input information, including examining related systems (e.g., constructed wetlands), microcosm or mesocosm experiments, environmental tracers and modelling (Figure 8). All of these methods give insights into the role of seepage wetlands as tools for improving water quality. Most seepage wetland studies estimate wetland removal (viz., inflow minus outflow) by outlet monitoring combined with inputs estimated by either small scale measurement of identifiable inputs, tracer experiments or modelling.

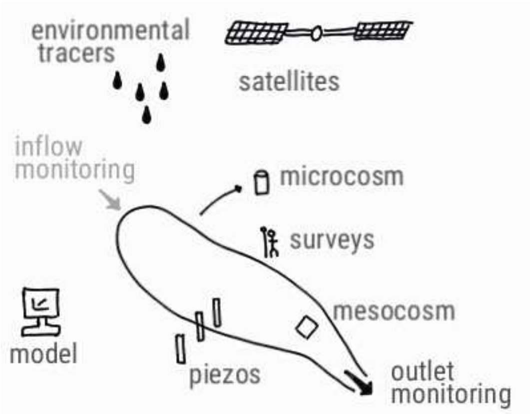


Figure 8: **Seepage wetland science methods and techniques.**

2.2.1 Wetland input and outflow monitoring

At some locations it possible to measure a major input source and outflows, and these studies provide information about potential water quality improvements at the wetland scale. Inputs measured include major springs, surface runoff and shallow groundwater. Varying degrees of quantification are possible; at some wetlands it is possible to measure input quantity and quality of some parameters, while at others only water quality (viz., inflow concentration) can be easily measured but inflow rate remains unknown.

The Pukemanga wetland (Whatawhata Research Station) is one example where inflow quantification is possible, assuming that the spring at the wetland head is the main water source; other springs and seeps may also contribute inputs. Three studies have contributed to our understanding of this wetland under different flow conditions (Table 4) and through time.

The first study by Nguyen et al. (1999) monitored spring inputs at the wetland head and outflows at the wetland base (Table 4). During this period the wetland was a nett sink for nitrate and DRP when summed over a range of flows. It was a sink for SS at low flows, but became a source at high flows (Nguyen et al. 1999). The linear relationship identified between nitrate and TN concentration and flow, suggests that contact times with the soil were inadequate for denitrification during storm events (> 75 m^3 d^{-1}). The proportion of surface runoff and groundwater in wetland outflows was determined using the environmental tracer oxygen-18. Overall >72% of outflows were contributed by old water (i.e., groundwater) and <27% was new water (i.e., surface runoff; Nguyen et al. 1999). During a large event (peak outflow 30 L s^{-1}; September 1997) the old:new water ratio varied from 64:34 to 46:54. For a smaller event (peak outflow 3 L s^{-1}; March 1998) the old:new water ratio was >91:<9.

A long-term monthly water quality concentration dataset (John Quinn, unpublished) is available for this wetland. Nitrate is the dominant N form (median 97% at seep and 86% at outlet) and the wetland has high nitrate removal capacity during summer and autumn (median 67%), and lower removal during winter (median 13%) and spring (median 30%; Figure 9). This dataset also demonstrates an increasing trend in nitrate concentration of spring water entering the wetland, thought to be caused by farm intensification. It is difficult to identify any trend in outflow concentration, which is affected by flow and temperature and is more variable than inflow concentration. Dissolved RP concentrations at the outlet are an order of magnitude lower than seep concentrations (Figure 9), but TP concentration reductions are more variable. The high pre-2004 TP data values for the seep are likely to be a sampling artefact; the sampling system was altered in August.

Stewart et al. (2009), using environmental tracers, estimated that: (1) the groundwater passing through the wetland has an mean age of 3-4 years, (2) 50% of the water leaving the catchment passes through the wetland, and (3) 74% of the wetland's flow is groundwater (Table 4).

This wetland therefore plays a role in improving the catchment's water quality, particularly removing nitrate during summer and autumn, but during winter nitrate removal is low. Monitoring to date has focused on nitrate, without detailed examination of the organic nitrogen components. There is also the possibility that high flows and floods entrain accumulated PON and DON and this is a major information gap.

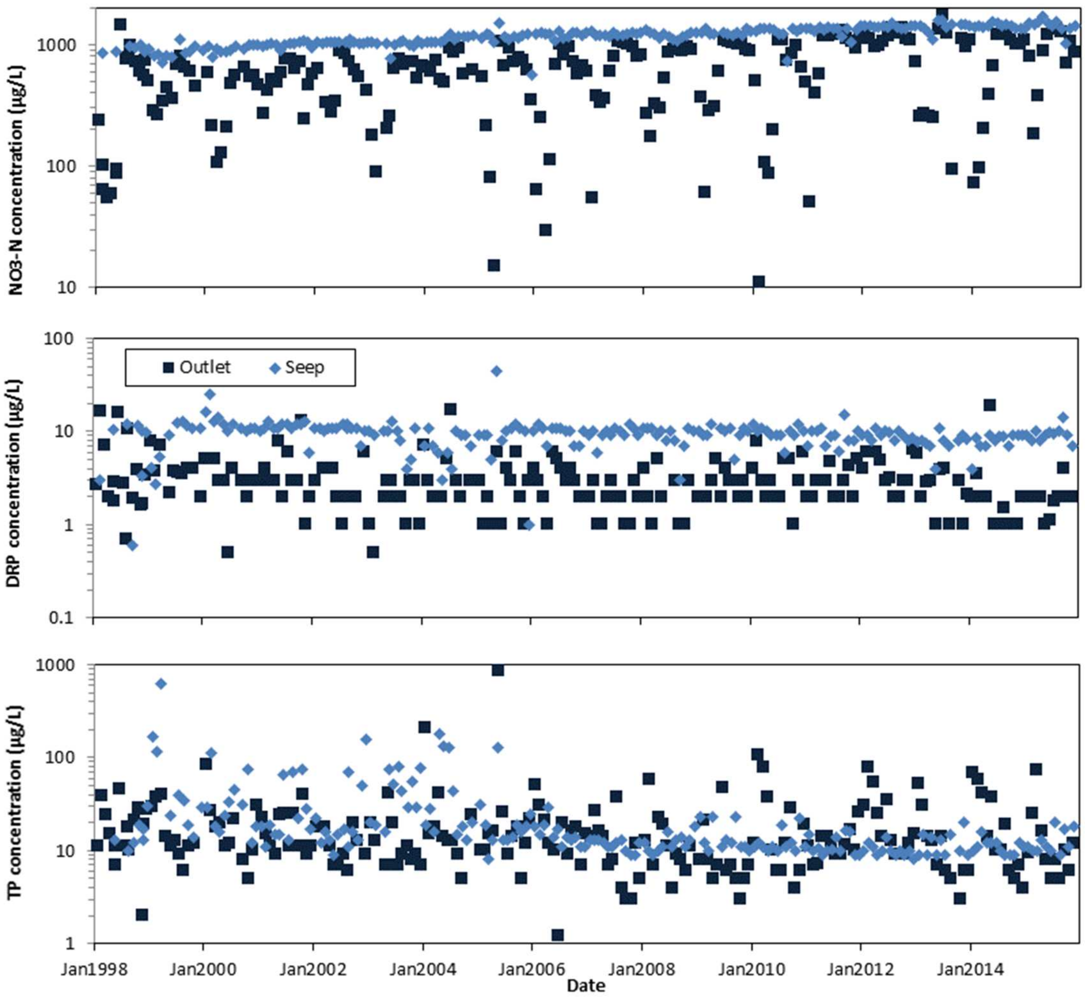


Figure 9: Pukemanga wetland monthly inflow and outflow nitrate, DRP and TP concentrations (1998-2015). Note high seep TP prior to August 2004 are likely to result from sampling set up. John Quinn, unpublished data.

The sediment trapping potential of a seepage wetland was demonstrated at the Kiwitahi wetland; the long, thin, deep wetland which was found to improve water quality by trapping solids entering from upslope dairy pasture. Surface runoff was measured at multiple locations and most entered through a gully at its head (Figure 9). Surface runoff suspended sediment concentrations during events were frequently greater than 100 mg L^{-1}, while the outlet concentrations were typically an order of magnitude lower (Figure 10).

Figure 10: Turbid surface runoff entering the upper weir at Kiwitahi wetland. Photo taken by Kerry Costley, 23 July 2012.

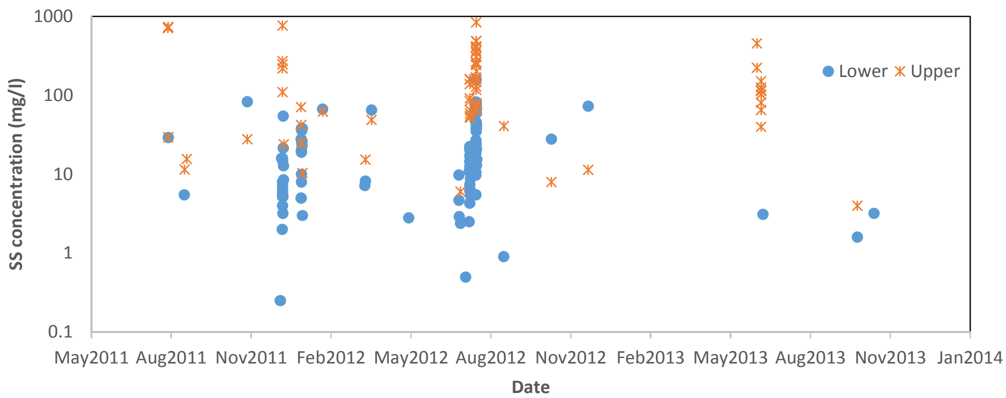


Figure 11: Suspended sediment concentration in surface runoff entering (upper) and leaving (lower) the wetland at Kiwitahi. Note log scale on y-axis. Andrew Hughes, unpublished data.

Table 4: Wetland water quality and hydrology studies in NZ. After McKergow, unpublished.

Location	Scale/land use	Interest	Experiment design & duration	Wetland description & condition	Key results	Reference
Scotsman's Valley, Waikato	Catchment with riparian wetlands, sheep & beef	N mass balance	12 baseflow snapshot surveys, 1 y	Depth 0.3 to 0.8 m, in total 1500 m^2 (1% of catchment)	Majority of inflow nitrate loss (56-100%) occurred in riparian organic soils (12% of stream border, 1% of catchment area); hotspots of denitrification activity near upslope edge of organic soils.	Cooper 1990
Scotsman's Valley, Waikato	As above	N removal rates in organic soils	Piezometers across riparian zones	As above	Nitrate flux reduced by 88-97% from range of 504-3427 mg N h^{-1}; one site 1.855 mg L^{-1} to 0.024 mg L^{-1} over 6 m of organic soil, adjacent site 2.06 mg L^{-1} to 0.01 mg L^{-1}.	Cooper 1990
Scotsman's Valley, Waikato	As above	DeN in-situ	Soil samples	As above	In-situ nitrate removal at 8100 mg N m^{-2} d^{-1} occurred at the upstream edge of the riparian zone where high nitrate (640 mg N m^{-3}) seepage flow first encountered organic rich and anoxic wetland soils. Close to the stream where nitrate concentrations had been reduced by denitrification (13 mg N m^{-3}) *in-situ* denitrification rates were very low (<2 mg N m^{-2} d^{-1}).	Cooper 1990
Scotsman's Valley, Waikato	Seepage soils, cattle	DeN in-situ	Soil samples, acetylene block		Potential deN activity 6480 mg N g^{-1} h^{-1}.	Cooke and Cooper 1988
Pukemanga, Whatawhata Research Station, Waikato	Wetland, sheep & beef, 17-30+°	SS, N and P removal	Monitoring, 6 mo including storms	62 m^2 unfenced headwater seepage wetland	Groundwater = baseflow + >75% stormflow; wetland TN (51%, inflow up to 20 mg d^{-1}); NO_3-N (54%, inflow up to 30 mg d^{-1}) and FRP (26%, inflow up to 2 mg d^{-1}) sink, but NH_4-N and PN source. Sediment sink at low flow and source at high flows. Suggest downstream buffer zone required.	Nguyen et al. 1999
Pukemanga, Whatawhata Research Station, Waikato	Wetland, sheep & beef, 17-30+°	runoff	Groundwater dating	62 m^2 headwater seepage wetland,	Mean residence time of water entering wetland 3-4 y; 74% of wetland water from deep groundwater; 50% of catchment water export passes through wetland.	Stewart et al. 2006

Location	Scale/land use	Interest	Experiment design & duration	Wetland description & condition	Key results	Reference
Pukemanga, Whatawhata Research Station, Waikato	Wetland, sheep & beef, 17-30+°	N, P and optical water quality	Monthly sampling for 20 years	62 m^2 unfenced headwater seepage wetland	Spring NO_3-N concentration increasing in time, from ~0.8 mg L^{-1} to ~1.5 mg L^{-1}. NO_3-N/TN concentrations reduction between spring and outlet; seasonal pattern in NO_3-N removal, with high NO_3-N removal in summer (median 66%) and low removal (13%) in winter. DRP concentrations reduced by ~1 order of magnitude. Particulate matter (TP, SS, turbidity) variable concentrations.	Quinn (unpublished data).
Barker's, Whatawhata Research Station, Waikato	Mescosm, sheep & beef	nitrate processing	DEA; Br & KNO3 tracer experiment, 24 d	350 m^2; 8-9°; maximum depth 1 m; top 20-30 cm organically enriched clay	1 m^2 mesocosm isolated, >90% NO_3-N over 1m; removal rate limited by NO_3-N supply; small storms likely to transport NO_3-N to streams; DEA 5.7 ± 1.8 mg kg^{-1} h^{-1}.	Burns and Nguyen 2002
Barker's, Whatawhata Research Station, Waikato	Mesocosm, sheep, 10-30° slopes	nitrate removal	Br & KNO3 tracer experiment	350 m^2; 8-9°; maximum depth 1 m; top 20-30 cm organically enriched clay; fenced wetland 2-3 years; flow over water surface 1-3 mm deep, increased to 5-10 mm after rain	Mesocosm (2.4 m x 1.06 m wide) in dry weather 24% NO_3-N added removed over 1.5 m; 1 day sufficient to remove NO_3-N; vertical mixing may be important to increase removal of upwelling water high in NO_3-N; low flow surface flow only 5% of tracer transport; DEA top 10 cm 4.1 mg kg^{-1} h^{-1}; Porosity ~80%, hydraulic conductivity decreases rapidly with depth (89 cm d^{-1} at 5 cm, to 3 cm d^{-1} at 25 cm); NO_3-N areal removal 120 ± 80 mg $m^{-2}d^{-1}$.	Rutherford and Nguyen 2004
Barkers, Whatawhata Research Station, Waikato	Microcosms, pasture	nitrate processing	^{15}N-nitrate tracer, 1 mo	400 m^2 wetland; soils bulk sampled, microcosms prepared	planted microcosms: denitrification (61-63%), immobilised (24-26%), plant assimilated (11-15%), DNRA <1%; unplanted microcosms: DNRA (49%), denitrification (29%), immobilised (22%).	Matheson et al. 2002
Kiwitahi Station Rd, Waikato	Wetland, dairy	N mass balance	Monitoring, 16 mo including storms	Grazing animals excluded 2+ years; flow constriction at outlet; 6817 m^2; shallow (<0.5 m) wetland soils; top 10 cm thick root mat, top 20 cm unconsolidated organics; organics +silt + clay to 0.5m;. Less permeable layer at 0.7 m	nitrate sink (70-95% reduction in concentration) under baseflow; NH_4-N, DON, PN frequently higher at outlet than inlet.	Nguyen et al. 2002

Location	Scale/land use	Interest	Experiment design & duration	Wetland description & condition	Key results	Reference
Kiwitahi Station Rd, Waikato	Mesocosm/dairy/ < 1°	N cycling	In-situ N_{15}, Br & SF_6 tracer, 48 h	Grazing animals excluded 4 years; flow constriction at outlet; 6817 m^2; shallow (<0.5 m) wetland soils; top 10 cm thick root mat, top 20 cm unconsolidated organics; organics +silt + clay to 0.5m;. Less permeable layer at 0.7 m	NO_3-N supply limited after 4 hours; mean removal rates when non-limited 15.7 mg L^{-1} d^{-1}; DeN accounted for 6-7% of NO_3-N removal; areal DeN rate 289 mg m^{-2} d^{-1}; nett NO_3-N removal rate 4094 mg m^{-2} d^{-1}	Zaman et al. 2008; Nguyen unpublished
Tutaeuaua, Taupo	wetland/ dry stock/ Taupo	runoff	monitoring	Range of wetlands	11-19% of streamflow comes through wetlands; permanent wetlands are baseflow dominated (80%), ephemeral wetlands are stormflow dominated (80%).	Rutherford et al. 2009
Tutaeuaua, Taupo	wetland/ dry stock/ Taupo	runoff	groundwater dating	Range of wetlands.	6 wetlands sampled. Groundwater ages 20-30 (CFC11) and 15-20 y (CFC12); 3 sites with large proportion of young water, 3 sites predominantly (20-40%) old groundwater.	Stewart (unpublished)
Tutaeuaua, Taupo	wetlands/ sheep & beef/	N spatial survey	Single samples at wetland seeps and springs and outlets	Many larger wetlands fenced, smaller ones unfenced.	Large reductions (at least 1 order of magnitude) in NO_3-N concentration at most wetlands between a seep (around 2 mg L^{-1}) and outlet (typically less than 0.02 mg L^{-1}). Some low inflow NO_3-N samples in survey.	McKergow (unpublished data); McKergow et al. 2007
RC wetland, Tutaeuaua, Taupo	wetland/ sheep & beef	N cycling	monthly monitoring piezos and weir	162 m long, 2200 m^2, depth 10-1.2 m, organic soil volume 890 m^3 (83% water).	Shallow groundwater upslope of wetland 2-3 mg L^{-1}. Average removal 70-90%. Areal removal rates 110-120 mg m^{-2} d^{-1} at 13C.	Sukias and Collins (unpublished)
JS wetland, Tutaeuaua, Taupo	mesocosm/ sheep & beef	N removal	Br & KNO3 tracer experiment , 28 days	200 m long, depth up to 1 m.	Pore water velocity 0.028-0.042 m h^{-1}. >95% removal of added NO_3-N over 160 m. Areal removal 60-135 mg m^{-2} d^{-1} at 7C.	Sukias and Collins (unpublished)
JS wetland, Tutaeuaua, Taupo	mesocosm/ sheep & beef	N removal	longitudinal surveys of wells along a transect	200 m long, depth up to 1 m.	DIN removal: 1-31 mg m^{-2} d^{-1} (deep wells), 3-346 (shallow). TN removal 4-31 mg m^{-2} d^{-1} (deep) & 10-468 (shallow).	Sukias and Collins (unpublished)

Location	Scale/land use	Interest	Experiment design & duration	Wetland description & condition	Key results	Reference
Kiwitahi, Waikato	wetland/ dairy	N, P, *E.coli* and optical water quality	Event and baseflow monitoring – 2 years	Large (1500m^2) headwater wetland, steep terrain with unlimited cattle access	NO_3-N, TN, *E.coli*, TSS concentrations at wetland outlet lower than upper weir and piezometer (assumed to be the major groundwater input location). For grab samples (n=13 to 17), nitrate median piezo 3.44 mg L^{-1} to outlet 0.023 mg L^{-1}; *E. coli* median piezo 16, upper weir 862, outlet 31 MPN/100 ml; TSS median upper weir 28 mg L^{-1}, outlet 6 mg L^{-1}.	Hughes et al. (unpublished), Hughes et al. 2012, Hughes et al. 2013;

DNRA = dissimilatory reduction of nitrate to ammonium.
DEA = denitrification enzyme activity.
DeN = in situ denitrification.

2.2.2 Seepage wetland: sub-wetland scale experimental research

Experimental studies have shown:

1. rapid removal of nitrate from subsurface flow in seepage wetland soils, but postulated the potential for nitrogen to be transported across the wetland surface during rainfall events with minimal removal (Burns and Nguyen 2002)
2. seepage wetlands can remove nitrate from surface flow at high rates during dry weather (25% of added nitrate removed over a distance of 1.5 m) (Rutherford and Nguyen 2004)
3. seepage wetland soil properties vary with depth, for example hydraulic conductivity is higher in the top 10 cm (Rutherford and Nguyen 2004; Rutherford et al. 2000), and
4. wetland plants promote denitrification through rhizosphere oxidation of highly anoxic soils, discouraging dissimilatory reduction of nitrate to ammonium (Matheson et al. 2002).

Sub-wetland scale experiments can be used to identify transformations and estimate their rates under varying conditions. These experiments may be run in microcosms (typically in the laboratory; e.g., Matheson et al. 2002) or in-situ mesocosms in small isolated areas of wetland soil (e.g., Rutherford and Nguyen 2004). Actual and potential denitrification rates are often determined by acetylene inhibition of N_2O reduction, and by measuring denitrifying enzyme activity (DEA). Transformation processes can also be investigated using isotopic tracers, such as ^{15}N-nitrate.

Published potential denitrification rates (DEA) for a number of New Zealand wetland soils range lie between 4 and 6 mg N kg^{-1} h^{-1} (Table 4; Burns and Nguyen 2002; Rutherford and Nguyen 2004). These numbers demonstrate that organic, anoxic wetland soils have the potential to remove significant quantities of nitrate. Burns and Nguyen (2002) demonstrated that ~24-48 hours contact time was sufficient for almost complete nitrate removal from seepage flow containing ~0.5 g N m^{-3} of added nitrate. DEA measures **maximum** potential denitrification – nitrate and often organic carbon are added to the soil samples during the test to ensure non-limiting conditions. These rates cannot, therefore, be applied to entire wetlands due to spatial and temporal variability – hot spots and hot moments – in nutrient removal.

Cooper (1990) measured high rates of permanent NO_3^- removal via denitrification within organic-rich riparian wetland soils. Removal rates varied spatially; they were highest where shallow sub-surface flow high in nitrate first came into contact with organic soils containing denitrifying bacteria (mean 1.35 mg N kg^{-1} h^{-1}). Further into the riparian wetland where nitrate concentrations had declined, removal rates were lower (Table 4).

DEA can be used to estimate the maximum likely nitrate removal rate. In the Barkers and Whakarewarewa wetlands hydraulic conductivity was highest in the top 10 cm of soil (Rutherford et al. 2000; Rutherford and Nguyen 2004). Assuming that the top 10 cm removes nitrate at the measured DEA rate then these wetlands would have an areal removal rate of 1500 ± 300 mg N m^{-2} d^{-1} (mean ± standard deviation). However, the top few cm may not be anoxic – no denitrification occurs in the presence of oxygen although nitrate may be removed by plant uptake. Nitrate is carried across these wetlands in surface flow and then mixes vertically (Rutherford and Nguyen 2004). High nitrate surface flow may not mix as deep as 10 cm in which case denitrification rate at this depth may be lower than the DEA. Thus 1500 ± 300 mg N m^{-2} d^{-1} is a likely upper bound estimate of nitrate removal rate.

Rutherford & Nguyen (2004) injected inert tracer onto the surface of Barkers wetland (Whatawhata Research Station) and knowing the flow and the time of passage of the tracer centroid inferred that the tracer mixed to a depth of 4-5 cm at low flows and 10 cm at high flows – comparable with the depth of soil in which hydraulic conductivity is high. However, as stated previously, the top few cm may not be anoxic. Assuming that 50% of the mixing depth is anoxic then the likely nitrate removal lies in the range 300-700 mg N m^{-2} d^{-1}.

This removal rate may not occur over the whole area of the wetland. As Cooper (1990) showed, as water flows across a wetland nitrate concentration drops and so removal rates drop. DEA measures the potential for nitrate removal but removal at this rate only occurs where nitrate is present at high concentrations.

Wetland plants play an important role in providing conditions suitable for denitrification. Using wetland soil microcosms and adding labelled nitrate (^{15}N-nitrate) at concentrations to mimic natural soil nitrate concentrations (~ 1.1 µg N g^{-1} soil), Matheson et al. (2002) demonstrated that the presence of plants (glaucous sweetgrass; *Glyceria declinata*) alters transformation pathways. In microcosms without plants the majority of nitrate was transformed to ammonium (DNRA; 49%) and the remainder was either denitrified (29%) or immobilised (22%). In contrast, in microcosms with glaucous sweetgrass denitrification was the dominant nitrate transformation pathway (~60%), with soil immobilisation (~24%) and plant uptake (11-15%) and DNRA (<1%) playing a small role. Elevated denitrification in the presence of glaucous sweetgrass was attributed to a higher degree of soil oxidation (mediated by the plant roots) which is considered to be the principal regulator of nitrate partitioning between denitrification and DNRA. Many denitrifiers are facultative anaerobes (i.e., prefer to use oxygen for metabolism if it is available) while microorganisms that undertake DNRA are typically obligate anaerobes (i.e., are unable to use oxygen).

2.2.3 Using related systems - constructed wetlands

Monitoring of constructed wetland performance can help fill the knowledge gap on seepage wetland attenuation performance. If we assume that seepage wetlands behave similarly to mature constructed wetlands (after organic soil has accumulated) we gain some appreciation of the likely performance of seepage wetlands.

Data from constructed wetlands receiving nitrate-rich tile drainage from intensive dairy pastures was collected in Northland (3 years), Waikato (5 years) and Southland (3 years), and for an array of small experimental wetlands in Waikato (Tanner and Sukias 2011). These studies show performance varies with year-to-year differences in seasonal drainage patterns. Nitrate removal performance is better in warm than in cold seasons and when residence times are extended (i.e., when influent flows are spread out relatively evenly over a period rather than arriving as a few large events).

Nutrient budgets over 5 years for the longest-running constructed wetland at Toenepi in the Waikato (~1% of a 2.6 ha drainage area, without supplementary irrigation) had an average annual areal nitrate removal of 280 mg N m^{-2} d^{-1} (annual averages ranged from 120 to 550 mg N m^{-2} d^{-1}). Typical annual TN removals of ~80-330 g m^{-2} d^{-1} (7-36% TN removal) were measured for the mature systems (2003/4-2005/6) receiving loads of ~490-1100 mg N m^{-2} d^{-1}. On a seasonal basis removal was more variable (see Figure 12) depending on climatic conditions and short-term variations in hydraulic loading. Winter TN reductions, when the main loading occurs, ranged widely from 50% decrease to 20% increase in N load.

At Bog Burn, Southland, the long term average areal nitrate removal was 230 mg N m^{-2} d^{-1}, varying annually from 150 to 420 mg N m^{-2} d^{-1}. Winter nitrate reductions ranged from ~20 to 63% and 200 and 600 mg N m^{-2} d^{-1}.

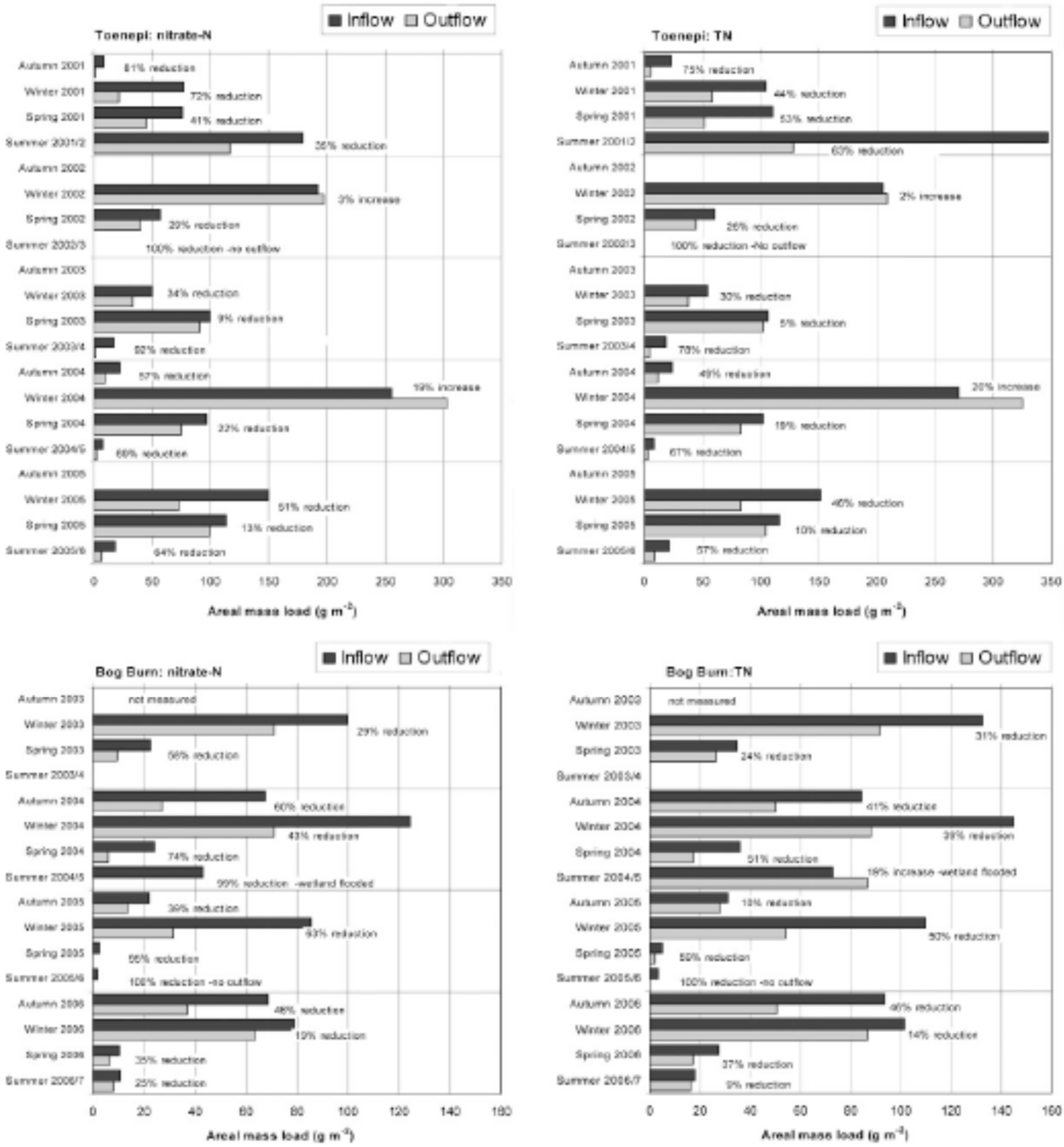


Figure 12: Summary of seasonal total areal loads of nitrate and TN in the inflows and outflows of constructed wetlands at Toenepi and Bog Burn. (Reproduced from Tanner and Sukias, 2011).

Constructed wetlands provide guidance on the long-term areal removal rates for seepage wetlands. Rutherford et al. (2007) selected a maximum removal rate of 250 mg N m^{-2} d^{-1} when temperature is 20C for the seepage wetland module in Overseer based on mature constructed wetlands and a limited number of seepage wetland studies. However, the variation of reported removal rates inferred from tracer experiments in seepage wetlands is very high 30-8100 mg N m^{-2} d^{-1}. Cooper (1990), at Scotsman's Valley, measured denitrification rates of 8100 mg m^{-2} d^{-1} where sub-surface flow first entered the wetland and NO_3 concentrations were high (640 mg m^{-3}), 6100 mg m^{-2} d^{-1} further downstream where NO_3 concentrations were lower (218 mg m^{-3}), and 30 mg m^{-2} d^{-1} at the stream edge where NO_3 concentrations were very low (13 mg m^{-3}). At Barkers wetland, Whatawhata, Rutherford and Nguyen (2004) measured NO_3 removal rates, and from these data Rutherford et al. (2008) estimated an average removal rate of 120 ± 80 mg N m^{-2} d^{-1}. Sukias and Collins (unpub. data) measured nitrogen concentrations flowing into and out of the RC wetland at Taupo. Rutherford (2017) re-examined their and estimated removal rates for DIN and TN of 49-55 mg m^{-2} d^{-1} at 13C (equivalent to 113-123 mg m^{-2} d^{-1} at 20C). At Walton Rd (Waikato) Zaman et al. (2008) measured a NO_3 removal rate of 4094 mg m^{-2} d^{-1} of which 6-7% (289 mg m^{-2} d^{-1}) was denitrification. Rutherford et al. (2008) summarise published denitrification rates from overseas studies of pasture wetlands in the range 11-288 mg NO_3-N m^{-2} d^{-1}. They also report nett removal rates measured by inflow/outflow studies, and small scale tracer experiments, in the range 7-29 and 26-270 mg NO_3-N m^{-2} d^{-1} respectively.

Constructed wetland mesocosm studies also provide insights into the predominant removal nutrient processes. Plant accumulation of 15-30 g N m^{-2} and uptake rates of 0.5-1 g N m^{-2} d^{-1} are possible during active growth periods (e.g., Tanner 1998). In-situ mesocosm studies in the Toenepi constructed wetland demonstrated that denitrification was the dominant N removal process, explaining 77% of NO_3 removal (Matheson and Sukias 2010). Zaman et al. (2008) also concluded that plant uptake could explain a high proportion of the observed NO_3 removal in the seepage wetland at Walton Rd, but in contrast with Matheson and Sukias (2010), they found that denitrification explained only 6-7% of NO_3 removal.

2.3 Wetland disturbance

Within agricultural areas, drainage and grazing have the potential to significantly affect the water quality services of seepage wetlands. Drainage and grazing can alter soil-water contact, nutrient cycling, wetland vegetation and soils (Figure 13) resulting in degraded water quality.

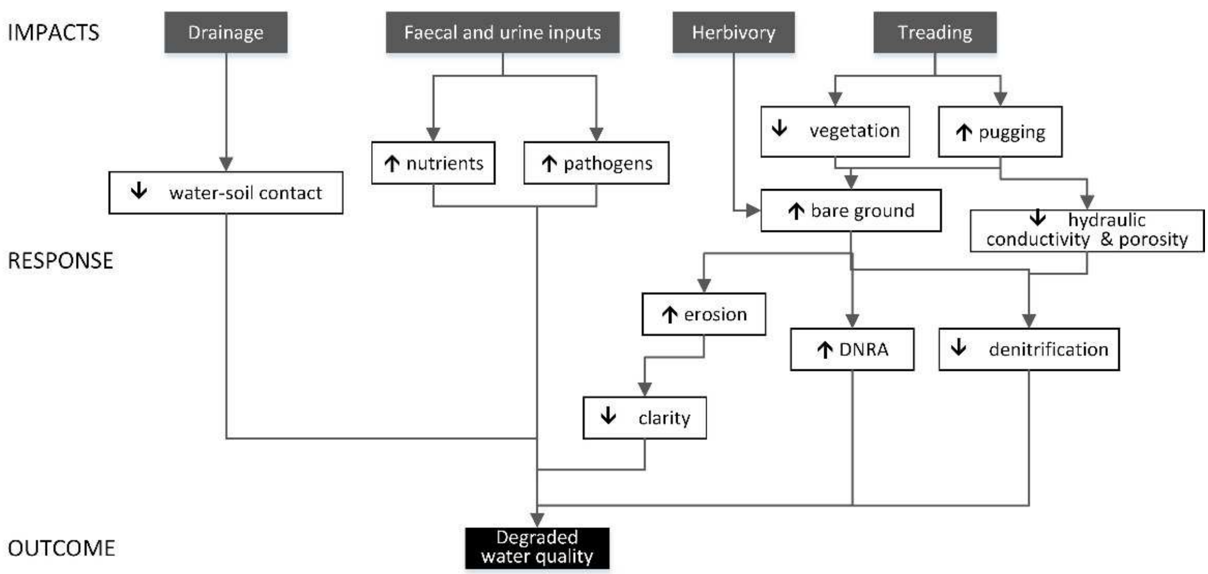


Figure 13: Conceptual diagram of the impact of drainage and grazing on seepage wetland water quality.

Drainage of seepage wetlands is widely practiced; sometimes at high cost to the farmer as drainage is "improved" year after year (e.g., Tanner et al. 2014). Artificial drainage encourages water to bypass the soil matrix reducing the nutrient processing and sediment trapping services provided by seepage wetlands.

When cattle access seepage wetlands the water quality can be degraded directly through faecal and urine inputs and soil disturbance, and indirectly by altering soil physical properties (e.g., compaction) and damaging vegetation (e.g., by treading and herbivory).

Large increases in *E. coli* concentration (Collins 2004) and high total and organic nitrogen exports (McKergow et al. 2012) have been measured from small, shallow seepage wetlands following livestock incursions (Table 5). Cattle tend to be wary of deeper wetland zones and these zones are often left largely intact (e.g., Figure 14d, Hughes et al. 2016).

Where larger wetlands have downstream deep zones (where cattle will not enter the water), these zones have buffering capacity. Although cattle may reduce nutrient removal in the upper (shallow) parts of the wetland, the downstream (deep) zones act as a sediment trap and nutrient processing buffer. For example, the configuration of Kiwitahi wetland means that there is always a downstream buffer which helps maintain water quality – when cattle access the upper wetland there is no water quality response at the outlet (Hughes et al. 2016). This is in contrast with the shallow RC wetland at Tutaeuaua when there was no buffer – when cattle had access to the RC wetland, water quality deteriorated at the outlet (McKergow et al. 2012).

Cattle can also damage wetlands by creating tracks on the wetland margins (

Figure 15). Some tracks become channels during rainfall events, with water bypassing the soil and moving rapidly through the wetland (Figure 16).

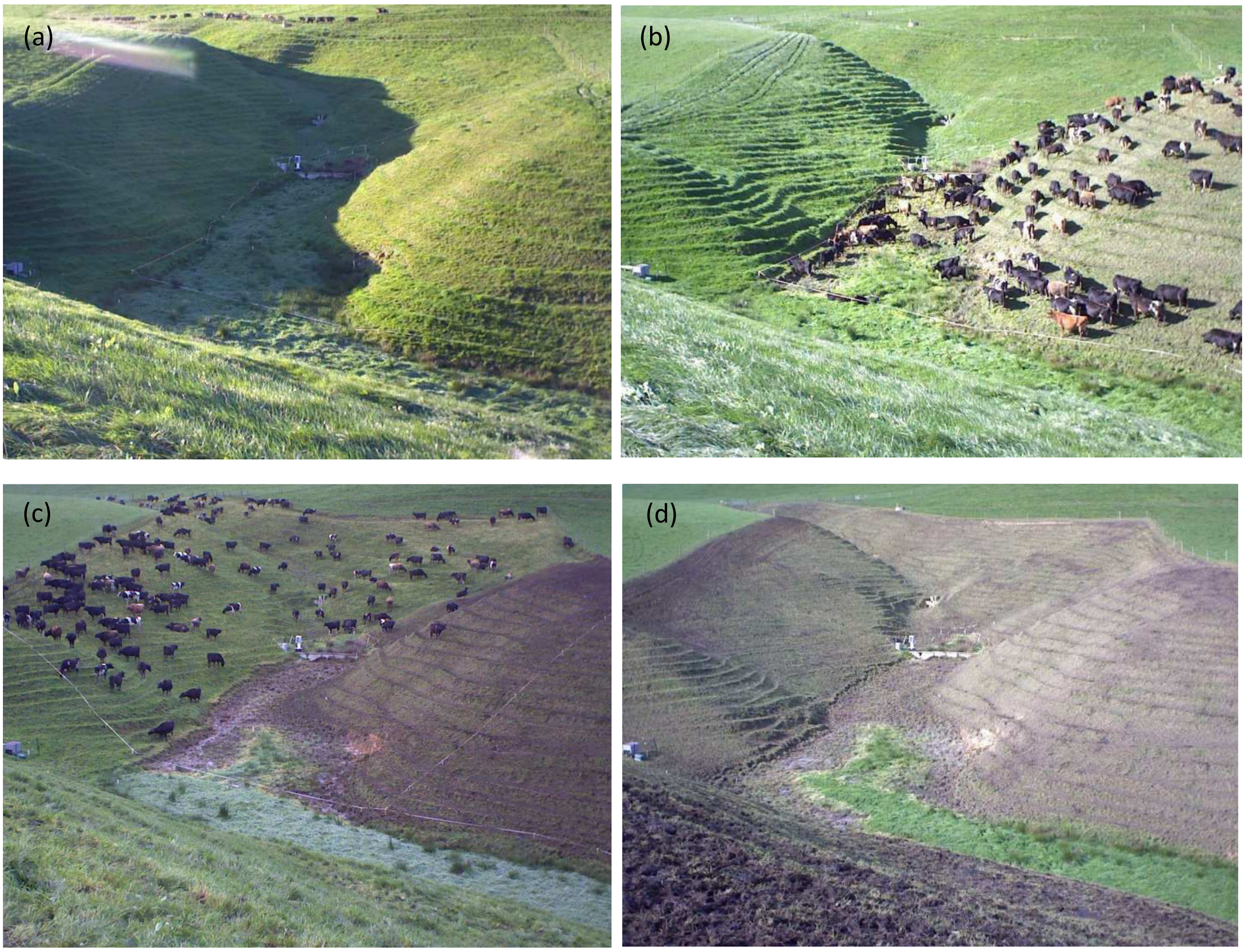
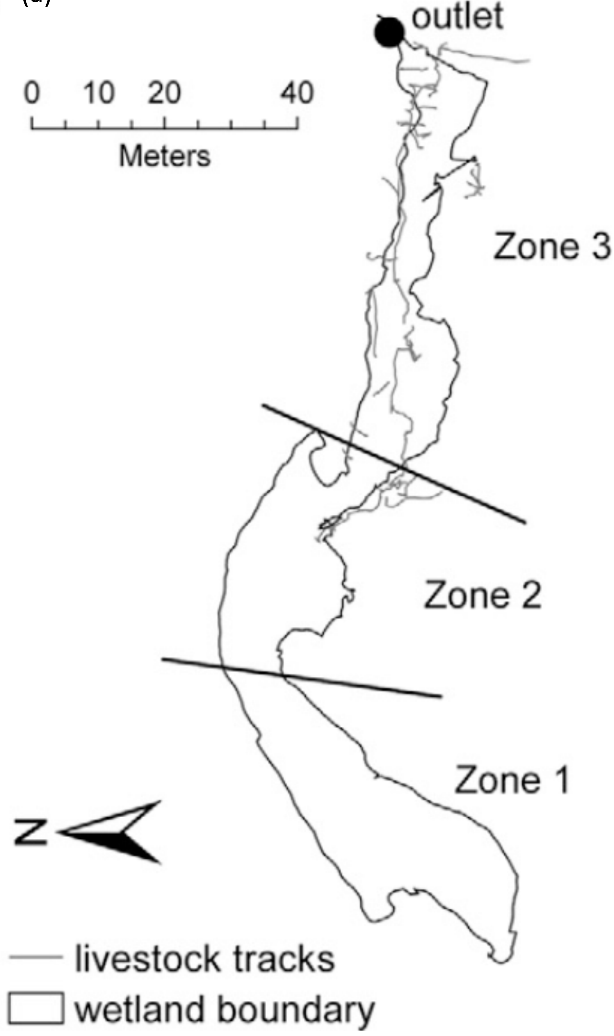


Figure 14: Time series of a grazing event 22-25 June 2013 at the upper Kiwitahi wetland. (a) just prior to grazing (22 June 08:53), (b) 3 hours after grazing started, note the entrapped cow (22 June 11:53), (c) the following morning (23 June 08:48), and (d) after the grazing event (25 June 13:28). Photos taken by a time-lapse camera, Andrew Hughes, NIWA.

(a)



(b)

Figure 15: (a) a map of cattle tracks (McKergow et al. 2012) and (b) channels caused by cattle in the RC wetland at Tutaeuaua. Note the differing orientation of (a) wetland viewed to east and (b) wetland viewed to west. Tracks mapped and photo taken by Lucy McKergow, NIWA.

Figure 16: Wetland flow bypassing the soil matrix in a bankside channel during a mid-winter rainfall event, Kiwitahi, Waikato. Photo taken by Kerry Costley, NIWA, 11 July 2011.

Table 5: Seepage wetland livestock & water quality studies in NZ. (After McKergow, unpublished)

Location	Scale &land use	Interest	Experiment design & duration	Wetland description & condition	Key results	Ref.
Barker's wetland Whatawhata Research Station, Waikato	Wetland, sheep & beef, 10-45°	*E. coli*	Monitoring, 3 mo	Wetland A – 32 m long x 3-7 m wide, max depth 1 m, wetland slope 8-11°.	Concentrations leaving the wetland. Baseflow 10^1 and 10^3 MPN/100mL; stormflow 10^3 to 10^6 MPN/100 mL.	Collins 2004
Whatawhata Research Station, Waikato	Wetland, sheep & beef, 10-45°	*E. coli*	Monitoring, 2 storms	Wetland B – 20 m long, 1-5 m wide, max depth 30 cm, slope 8-11°.	Cattle strongly attracted – 15% of defaecation occurs in wetland (22 cow pats in wetland + 19 in 2 m margin) over 3 day period; Event 1: peak 6 x 10^4 MPN/100mL in water leaving wetland 4 d after cattle removed; Event 2 - peak 10^3 MPN/100mL leaving wetland at peak flow of 2 L/s 15 days after cattle removed.	Collins 2004
Whatawhata Research Station, Waikato	Wetlands, sheep & beef, 10-45°	*E. coli*	snapshot survey	18 wetland outlets after rain moderate flow.	*E. coli* ranged from 0.5x10^1 to 2x10^4 MPN/100mL; concentrations at three small shallow wetlands exceeded 10^3 MPN/100 mL (all 3 had >10 recent cow pats).	Collins 2004
RC wetland, Tutaeuaua, Taupo	Wetland, sheep & beef, up to 25°	N export	outlet monitoring, 2 y	Grazed by beef and sheep; 1700 m^2; soil depth varied 0.1-1.2 m; 3 tiered wetland – top not channelized, mid-tier small channel, lower tier muddy pond.	Cattle grazed wetland 9% of time but contributed 32% of total N export over 11 months; Organic N dominant N form exported (median ~85% of TN).	McKergow et al. 2012
Kiwitahi, Waikato	Wetland, dairy, 20+°	N, SS, *E. coli* export	Monitoring/ 2 y	Grazed by dairy herd; 1500 m^2; soil depth 0.5-1 m within 1 m of margin, 1-2 m at centre of wetland; slope 3.5°.	Cattle grazed shallow margins and surrounding hillslopes heavily; deeper wetland a buffer on outlet water quality; only 1 of 18 cattle grazing days had a rise in turbidity at outlet (entrapped cow coincided with flow event) which equated to 20% of the TSS, TP, TN loads for the event and 5% of total *E. coli*.	Hughes et al. 2016

2.4 Seepage wetland protection and enhancement

Seepage wetlands on farms are typically not managed any differently to the surrounding pasture. Management options include drainage, livestock exclusion and vegetation and channel management.

2.4.1 Drainage management

Drains are commonly used to improve stock management (e.g., enable paddocks to be levelled and fenced into regular paddocks, reduce stock entrapment in boggy areas) and increase the productivity of pasture. On steeper land, wetlands may also be drained to reduce the risk of infrastructure damage if/when wetlands are scoured out (Hamish McMullin, pers. comm., 29 August 2016).

The science of wetland hydrological restoration is immature internationally; there is still little understanding of the effectiveness of restoration techniques or how they might change wetland soil and plant patterns. There is one New Zealand study on a unique wetland type in Westland (Sorrell et al. 2007). Tile drains and drainage ditches on a polje fen were blocked and soil redox potential and oxidation within the relic wetland rapidly became similar to those in control wetland areas. Further research is needed on how to reinstate drainage into wetlands and to determine: how much benefit could be gained by infilling or blocking drains through seepage wetlands, how long soil carbon stocks take to re-establish, how much cost/effort is required to remove drainage, and the potential adverse effects changing drainage patterns will have on infrastructure and pasture productivity.

2.4.2 Livestock exclusion

When cattle regularly become entrapped in deeper wetland soils, farmers tend to exclude them from seepage wetlands, either with portable or permanent fencing. Portable electric fences are often installed to prevent cattle access (Figure 17) and offer a flexible solution. Permanent fencing is also used, and where the adjacent slopes are steep, ‘benches’ are sometimes constructed by earth moving equipment (Figure 18). In the short-term, benching exposes large areas of bare earth adjacent to the wetland that may be vulnerable to erosion during rainfall events. Large inflows of sediment during major rainfall events can damage existing wetlands by burying vegetation. Benched areas adjacent to wetlands may also be used as preferential pathways or resting areas by livestock, potentially increasing faecal inputs and erosional losses beside the waterway. In the longer-term, it is likely that benches would also alter the hydrological connectivity and divert surface runoff away from wetlands, bypassing opportunities for contaminant buffering.

Figure 17: A portable electric fence installed along the edge of a pastoral wetland, Kiwitahi, Waikato. Photo: Lucy McKergow, NIWA, August 2010.

Figure 18: Newly constructed benches adjacent to a seepage wetland, Kiwitahi, Waikato. Photo: Ron Ovenden, NWA, April 2010.

From a water quality perspective shallow wetlands are likely to benefit from livestock exclusion more than deeper wetlands. Cattle readily graze shallow wetlands (< ~ 1 m depth) but generally avoid deep (> ~1 m depth) wetlands. This "rule of thumb" may not apply to seepage wetlands with a channel of flowing water – exclusion of cattle from such wetlands would be beneficial regardless of depth.

2.4.3 Vegetation management

Planting seepage wetlands with trees and shrubs is not recommended where the principal objective is to improve water quality – rather we recommend that grass cover is maintained or enhanced. Vegetation plays an important water quality role by providing: organic matter, by increasing rhizosphere aeration which promotes N removal by denitrification and discourages DNRA (Matheson et al. 2002) and by trapping solids (Hughes et al. 2013), including particulate nutrients. However, some types of vegetation increase the risk of channelization, and hence reduces the ability of the wetland to trap contaminants. For example, at Tutaeuaua, 'mature' riparian fenced areas often have channels that 'short circuit' some of the runoff through the wetland. This happens in wet areas with shrub/tree size native (e.g., flax, ti tree) or exotic (e.g., willows) plants.

Channelisation of wetlands may be reduced by several methods. Firstly, by summer grazing of ephemeral wetlands (viz., those wetlands, or parts of wetlands, that dry out in summer). Summer grazing of seasonally dry wetlands, particularly by sheep, can help maintain a healthy, even grass sward (Rutherford and Nguyen 2004). However, seasonal grazing of ephemeral channels may load the area with nutrients from dung (e.g., McDowell 2006) and/or cattle may cause channelization through treading damage. Secondly, by provision of grass buffers upslope of seepage wetlands that can be grazed or mowed. Mowing and/or grazing would best be undertaken in summer. However, the success of such measures has not been experimentally evaluated. Thirdly, in permanently wet seepage wetlands, using structures to disperse. In those seepage wetlands where preferential flow paths have developed (e.g., as the result of cattle ingress;

Figure 15) the exclusion of cattle may allow some rehabilitation of the wetland's attenuation capacity. However, planting of grassy species and/or the placement of barriers to flow (e.g., wood, tree fern trunks, rock bunds etc.,) could reduce the amount of water by-pass wetland soils and vegetation in channelized flow. Observations at Whatawhata revealed that grazing by sheep during summer (when the wetland was fairly dry) helped maintain a uniform grass cover and helped combat channelization – cattle would not have been suitable for maintaining uniform grass cover because of pugging damage. Further work is required to assess where and when such rehabilitation methods are feasible and effective.

Figure 19: Cattle trampling damage to the margin of a seepage wetland, Kiwitahi, Waikato. Photo: Andrew Hughes, NIWA, August 2011.

On steeper land, where there is a risk of wetland scour in extreme events, planting of trees to anchor the wetland to the underlying sub-soil is worth trialling. This is particularly important where considerable farm infrastructure is downstream of a wetland or series of wetlands. Suitable plants might include: Lake clubrush/Kapungawha (*Schoenoplectus tabernaemontani),* marsh clubrush/kukuraho (*Bolboschoenus fluviatilis*), swamp Cypress (*Taxodium distichum*) or swamp Maire (*Syzygium maire,* although it would require shade/shelter to establish). However, further investigation of the rooting structure of these plants is required.

2.5 Information gaps

Identification and management of seepage wetlands has the potential to improve water quality outcomes.

First, it is desirable to develop practical measures that promote the uniform distribution of flow across the wetland surface. These include: exclusion of cattle which cause pugging and channelization, management of vegetation to promote an even flow distribution, and building of structures that redistribute channelized flow. However, only a few experimental trials of livestock exclusion have been undertaken, although these clearly demonstrate the negative impacts that cattle can have in shallow wetlands. There have been no trials of vegetation management techniques using stock, although there is anecdotal evidence that grazing by sheep during summer helps maintain a uniform grass sward which helps distribute flow evenly across the wetland. There have been some trials using structures (e.g., rock gabions) to trap sediment and particulate phosphorus in ephemeral channels, but none on structures to redistribute flow and increase nitrogen removal in seepage wetlands.

Second, a key information gap relates to the effects of reversing artificial drainage, diverting runoff back into wetlands and managing catchment runoff and flooding upslope from the wetlands. This includes any dis-benefits (e.g., more and longer pasture inundation, the risk of flooding/damaging of farm infrastructure such as roads, culverts and fences).

Third, little is known about the bioavailability of particulate and dissolved organic nitrogen exports from wetlands under New Zealand conditions. While there is clear evidence that seepage wetlands are effective in reducing nitrate exports from pasture, it is less clear how effective they are in reducing the export of total bioavailable nitrogen. Thus, a key question that remains unanswered in New Zealand is whether seepage wetlands are nett sinks for nitrogen, or transformers of nitrogen from one form (viz., nitrate) to other forms (viz., labile dissolved and/or particulate nitrogen) that are bioavailable. While there is evidence of some permanent removal by denitrification, this does not explain 100% of nitrate removal in the few studies conducted in New Zealand.

Fourth, although uptake by wetlands plants has been identified as a major contributor to nitrate removal, the fate of that plant-nitrogen in New Zealand seepage wetlands is poorly understood. In particular, it is not clear what happens when plants die or are dislodged by floods into streams and lakes – because the bioavailability of the dissolved and particulate nitrogen derived from plant detritus has not been studied. Suggestions have been made that 'harvesting' of plant biomass from seepage wetlands might be beneficial because it would remove a potential source of nitrogen export. However, a balance is required between (a) ensuring sufficient organic matter accumulates in wetland soils so they remain carbon-rich (because this enhances the rate of denitrification) and (b) preventing the build up of excess detritus that could decay and release bioavailable nitrogen for export to streams, lakes and estuaries. Information gaps are summarised in

Table 6.

Table 6: Summary of key information gaps.

Information gap		Description
Managing hydrology	Reversing artificial drainage	Does the water-soil contact improve rapidly? Are there any unintended consequences? Does the drain need to be filled or can it be blocked at one/more locations? What are the likely impacts on pasture productivity and flood management?
	Approaches to reduce channelization in wetlands	Can timber structures redistribute channelized wetland flows? Field trials to investigate dimensions required.
Water quality	Organic nitrogen fractions	Monitoring to date has focused on nitrate, without examination of the organic nitrogen components. There is also the possibility that high flows and floods entrain accumulated PON and DON and this is a major information gap.
	SS, TP and *E. coli* retention	Research emphasis has been on nitrate. More data on removal of SS, TP and E. coli by filtering and deposition from inflowing surface runoff is needed.
Vegetation management	Seasonal grazing	Grazing of seepage wetlands that dry up in the summer by sheep. What is the impact of seasonal grazing on nutrient export? Comparison of grazed vs ungrazed seepage wetlands.
	Stabilising and enhancing seepage wetlands with plantings	Can pair planting of deep-rooting exotic or native tree species stabilise small seepage wetlands on steep slopes which may be vulnerable to blow out in high rainfall events? Can we enhance contaminant attenuation by densely planting seepage wetlands with native wetland sedges, rushes and tree species?

3 Conclusion

Seepage wetlands can provide water quality benefits on dairy farms, particularly nitrate and suspended solids removal. Seepage wetland science is challenging due to the diffuse nature of water and contaminant inputs which enter through springs, seeps and surface runoff. Consequently much is known about sub-wetland scale nitrate processing and removal rates, but there are few input-output studies. Whole wetland removal rates are challenging to produce but mature constructed wetlands provide an upper bound (~250 mg NO_3-N m^{-2} d^{-1-}) and demonstrate that rates will vary considerably with hydraulic loading and season.

Currently farmers fence deeper wetlands to prevent stock becoming entrapped. For water quality improvement, identifying and managing shallow seepage wetlands (that cattle are likely to enter) separately from paddocks will be beneficial. Wetland vegetation and soils, in the absence of channels (formed by livestock or vegetation), can provide a buffer to inflowing surface runoff laden with suspended solids and promote the conversion of nitrate to gaseous forms.

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