Assessment of Floating Treatment Wetlands for Remediation of Eutrophic Lake Waters – Maero Stream (Lake Rotoehu)

NIWA Client Report: HAM2010-104
December 2010

NIWA Project: BOP09205

Assessment of Floating Treatment Wetlands for Remediation of Eutrophic Lake Waters – Maero Stream (Lake Rotoehu)

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Prepared for

Environment BOP

NIWA Client Report: HAM2010-104
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NIWA Project: BOP09205

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Executive Summary

To address water quality problems in Lake Rotoehu, Environment Bay of Plenty has been assessing options for nutrient removal from inflowing stream using wetlands for nutrient removal. Prior investigations showed there were significant practical problems with developing conventional constructed wetlands at Lake Rotoehu. Large water level fluctuations and the porous nature of soils occurring at suitable sites around the lake limit opportunities for developing conventional constructed wetlands at Lake Rotoehu.

Floating Treatment Wetlands (FTWs) have been identified as a potential alternative to conventional land-based wetlands. Because FTWs are buoyant, they have the advantage that they can be deployed in deeper water such as lakes, and are not affected by changes in water level. Although they incorporate similar nutrient attenuation mechanisms to those found in conventional natural and constructed wetlands, little quantitative data exits regarding available on their treatment performance. A mesocosm trial was therefore undertaken to provide information on the efficacy of FTWs to reduce nutrients from stream inflows to Lake Rotoehu.

The trial was carried out in a modified steel shipping container split longitudinally into two separate tanks set-up adjacent to the Maero Stream on the southern shores of the lake. One tank received a low inflow rate (110 mm d^{-1}) and one a high inflow rate (270 mm d^{-1}) of water pumped from the stream. These rates corresponded to nominal hydraulic retention times of 9.4 and 3.6 days, respectively. Sediments from the lake were placed in the bottom of the tanks and pre-established FTWs planted with native sedges were placed on the water surface. The trial operated for a period of about a year, including a three month period of establishment followed by nine months of water quality sampling.

Mean total nitrogen (TN) removal rates of 157 and 239 mg m^{-2} d^{-1} (77% and 45% removal) were recorded in the low and high inflow FTW tanks respectively. Mean total phosphorus (TP) removal rates of 2.3 and 5.4 mg m^{-2} d^{-1} were recorded in the low and high inflow FTW tanks, corresponding to 35% and 32% removal. These data indicate their performance appears to be similar or greater than what would be expected for conventional surface-flow systems receiving similar loading rates.

Questions remain regarding optimal conditions for FTW systems in lake situations to optimise and sustain their nutrient removal performance. Based on measurements of plant biomass and nutrient content at the end of the trial, plant uptake appears to have been the dominant mechanism for phosphorus removal, and also accounts for a significant proportion of the nitrogen removal. Significant deoxygenation of stream waters occurred under the FTWs during passage through the tanks, creating conditions likely to be conducive to nitrogen removal by microbial denitrification. As the FTWs mature, plant uptake of nutrients is likely to become restricted unless ongoing harvesting of plant biomass is undertaken. Other nitrogen removal mechanisms, in particular microbial

denitrification, are likely to increase in importance as the plants reach maturity and provide an ongoing source of organic carbon in the form of plant litter and root exudates.

As a guide to the expected performance of FTWs in the lake treating the outflow of the Maero stream, we suggest assuming a conservative mean annual nutrient removal rate based on the results of the present tank study of ~125 mg TN m^{-2} d^{-1} and 20 mg TP m^{-2} d^{-1}. This allows for the fact that the study did not measure removal rates through most of the winter period, when plant uptake and microbial process rates would likely be depressed. Extrapolated over a year this equates to ~46 g TN m^{-2} y^{-1} and ~7.3 g TP m^{-2} y^{-1}, or 460 kg TN ha^{-1} y^{-1} and 73 kg TP ha^{-1}.

In conclusion, we consider that the results of the present study provide further evidence of the efficacy of FTWs for nutrient removal, particularly nitrogen, from stream waters entering Lake Rotoehu. Larger-scale application in the lake is warranted to further evaluate their field performance. Concurrent monitoring of physico-chemical and biological conditions beneath and in the vicinity of the FTWs is recommended to evaluate the key environmental factors influencing performance in the lake, and provide information to help optimise their layout and management. Resumption of monitoring of the lake-side experimental tanks would also provide valuable information on the longer-term performance of the FTWs and how best to manage them for sustained nutrient removal.

1. Introduction

Floating Treatment Wetlands (FTWs) are a novel ecotechnology for remediating polluted waters. They comprise emergent wetland plants supported on a buoyant mat. The plant roots grow through the mat, eventually forming a dense mass that hang down into the water below. FTWs share many attributes with surface-flow constructed and natural wetlands, harnessing a similar range of natural treatment processes, but because they float on the water surface they can tolerate fluctuating water levels and are not constrained to shallow waters. As the plant roots generally do not have access to nutrient sources in the sediment they develop larger root systems and must meet their nutrient requirements directly from the water column. Biofilms develop on the extensive surface area provided by the roots, increasing organic matter breakdown, nutrient adsorption and trapping of fine particulates. The floating mats shade the water column and create a quiescent environment that reduces algal growth and promotes settling of suspended solids and sloughed biofilms. Localised zones of anoxia can develop beneath the floating mats, which promote conversion of nitrate to nitrogen gases by denitrifying bacteria.

Because FTWs are a relatively new technology, only limited research has been undertaken to quantify their nutrient removal performance in lake environments (Headley & Tanner 2006, Park et al., 2008). In recent experimental mesocosm (~1 m^3) trials carried out for Environment BOP (Park et al., 2008), FTWs were found to remove ~0.7 g $N/m^2/d$ and ~0.055 g $P/m^2/d$ from synthetic eutrophic lake water. These nutrient removal rates were at least twice those commonly reported for constructed surface-flow wetlands receiving water of similar composition. However, the experimental trials were relatively short-term, small-scale trials undertaken during summer in controlled testing facilities, and so questions remain as to how they will interact with and function in the natural lake environment, and whether these removal rates will be sustained in the longer-term.

1.1 Background

Environment BOP are the lead agency managing the restoration and management of the Rotorua Lakes. Lake Rotoehu is one of the five lakes prioritised for remedial action in the Rotorua Lakes Protection and Restoration Action Programme. Targets have been set to reduce nutrient inflows and control trophic conditions in the lake (Environment BOP 2007). Nutrient control is necessary because the lake has experienced intermittent cyanobacterial blooms since the 1970s (Cassie 1978). These have continued into the past decade with increased severity and extent, raising environmental and human health concerns (Wilding 2000, Wood 2005). Wetlands are recognised as one of the key tools available to sustainably remove diffuse nutrient loads entering lakes and estuaries (Coveney et al., 2002, Kadlec & Wallace 2009, Mitsch et al., 2005). A large-scale wetland was constructed and its performance is currently being monitored at nearby Lake Okaro (Hudson et al., 2009). While constructed wetlands have also been investigated for some inflows to Lake Rotoehu, suitable sites are somewhat limited as there are issues in terms of land availability, and changes to stream channels affecting trout passage and spawning behaviour. There are also significant design uncertainties due to the pervious soils at the potential lake-side sites (requiring costly lining of the wetland basins) and the range of water level fluctuation that the lake experiences (± 1.5 m over the last 25 years).

FTWs have been identified as a promising alternative to surface-flow constructed wetlands, with potential application in Lake Rotoehu. They offer the potential to target key nutrient inputs to the lake and/or nutrients within the lake waters, whilst overcoming many of the constraints of conventional wetland treatment approaches. If the results of the present study are promising, Environment BOP intends to test the practical use of FTWs in Lake Rotoehu.

1.2 Brief

The current performance trial was proposed to better quantify nutrient removal rates from either Lake Rotoehu ora key streamflow entering it. The trial was to provide quantitative performance data to guide design and implementation of FTW systems at suitable sites in the lake.

2. Methods

2.1 Site set-up

The site selected in association with Environment BOP was adjacent to the Maero Stream (Figure 1), on the southern side of Lake Rotoehu. Water was pumped from the Maero Stream water for the FTW nutrient removal trial.

The trial was conducted in a large experimental mesocosm tank (a modified shipping container). The shipping container had its top removed and a wall placed down the centreline, dividing it into two elongated chambers (8.90 x 1.15 m, 10.24 m^2). An additional central wall divided each chamber into two equal compartments (Figure 2). Water was able to flow between the two compartments via two 50 mm holes located 500 mm from the base and 250 mm from either side (Figure 3). There was an outflow manifold with sampling port attached to the outer wall of each chamber that consisted of a 25 mm hole in the outer wall of the tank, 300 mm from the base, with a riser pipe attached in order to maintain a water depth of 0.8 m. The interior of the compartments were sand-blasted and primed with Altra-Prime 504-Light Grey, a multi-purpose epoxy primer and then coated with Altra-Build 536 general purpose, non-toxic, epoxy coating (Altex Coatings, Tauranga, NZ). The mesocosm tank was delivered on site by truck and positioned using a crane (Figure 4).

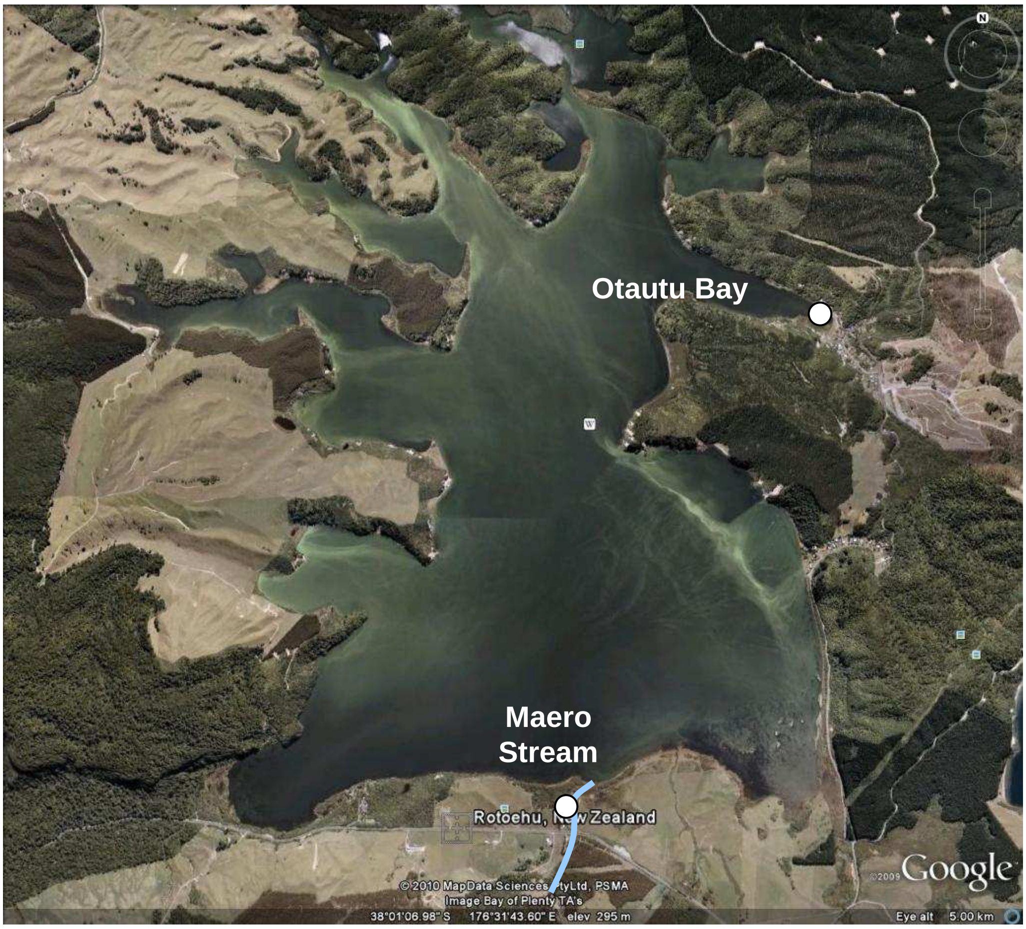

Figure 1: Lake Rotoehu, showing the location of the floating treatment wetland trial alongside the Maero Stream at the southern end of the lake. Sediments were collected in Otautu Bay on the north-eastern side of the lake.

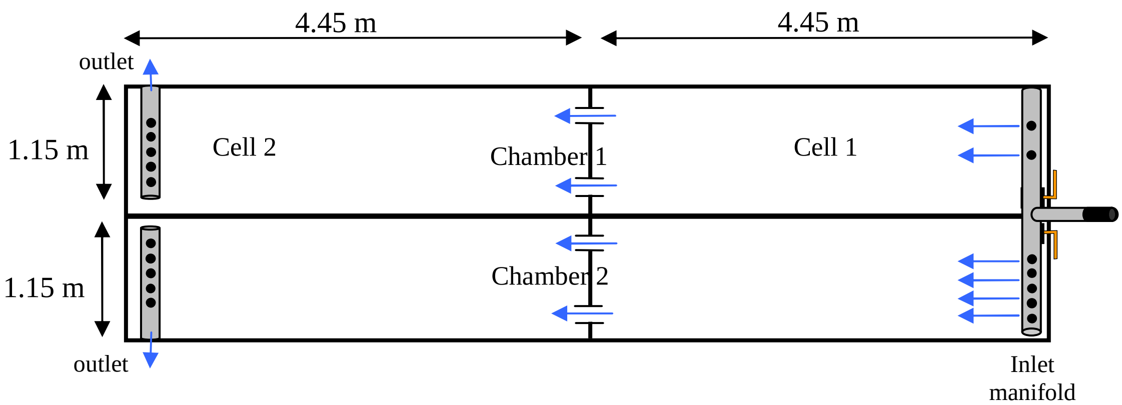

Figure 2: Plan view of the mesocosm tanks. The inflow enters on the right and exits on the left side of the diagram.

Figure 3: Internal view of an empty mesocosm tank, showing central wall dividing each chamber into two cells. The two holes through which water flows are also visible.

Figure 4: Positioning of mesocosm (a modified shipping container) at the experimental site.

Water was pumped from the adjacent Maero Stream into the mesocosm using a submerged pump controlled by an electronic timer. Stream water was distributed by a single inlet manifold which spanned both chambers (Figure 2). Five flow-restricting inlet holes on one side of the manifold provided 270 mm day^{-1} (2.6 m^3 day^{-1}), equivalent to a 3.6 day nominal Hydraulic Residence Time (HRT) to one side of the mesocosm. Two inlet holes on the opposite side of the manifold provided 110 mm day^{-1} (1 m^3 day^{-1}), equivalent to a 9.4 day nominal HRT, to the other side of the mesocosm.

Sediment was collected from Lake Rotoehu at Otautu Bay using a vacuum tanker truck. The 30 m inlet hose was directed by an operator standing in waist-deep water. The collected sediments were added equally to each of the mesocosm chambers. A large proportion of the sediment was sand, as more organic sediments were too deep in the lake to be reached with the length of hose available. The sediments were added to simulate in-lake conditions. They covered the base of the chambers to a depth of ~80 mm.

The floating mats used for the FTWs (Floating Islands, supplied by Kauri Park Ltd., Kaiwaka, Northland) comprised 150 mm thick sheets of intertwined polyester fibre injected with patches of polystyrene foam to provide buoyancy. The mats were divided into ~1.4 m x 1.15 m sections and covered with coconut fibre matting. Each section was planted at a density of 12 plants per m^{-2} with either *Carex virgata* or *Cyperus ustulatus* several months prior to starting the trial and maintained in a glass-house to promote rapid plant establishment. At the time of deployment (30/7/09), the plants were ~30-40 cm tall with roots extending 20–30 cm beneath the mats (Figure 5). Three FTW sections of each *Carex virgata* and *Cyperus ustulatus,* (total of six) were installed in each chamber to totally cover the water surface.

Figure 5: Mesocosm tanks with FTWs installed.

2.2 Sampling regime

2.2.1 Routine monthly water sampling

Routine sampling consisted of monthly manual collection of a sample from the inlet and from the outlet pipe of each tank. Sampling took place over a period of 9 months from 28/8/09 to 22/4/10. Samples were stored on ice until delivered to the NIWA Hamilton Water Quality Laboratory either the same or the following day.

Filtered samples (0.45 µm) were analysed for ammonium-N, measured colourimetrically using automated flow injection analysis (QuikChem 8000 FIA+, Lachat Instruments, Milwaukee, WI, method 31-107-06-1-1-B, Revision date 26 April 2001), and for combined nitrate and nitrite–N (oxidised nitrogen denoted as NO_x-N) after reduction to nitrite in a copperised cadmium column (QuikChem Method 31-107-04-1-A, Revision date 27 Feb 2001). Total N was analysed using the same method on an unfiltered sample which had first undergone digestion using an alkaline persulphate solution (modified from APHA, 2005, 4500N). Detection limits were 1 µg L^{-1} for oxidised nitrogen and ammonium-N and 10 µg L^{-1} for total N. Organic nitrogen (OrgN) was calculated by subtraction [TN – (NH_4-N + NO_X-N)].

Samples for dissolved reactive phosphorus (DRP) were filtered (0.45 µm) and analysed colorimetrically using QuikChem Method 31-115-01-1-I, while total phosphorus (TP) was analysed on an unfiltered sample using the same method after first undergoing acid hydrolysis using persulphate (modified from APHA, 2005, 4500P). Dissolved organic carbon (DOC) was analysed using high temperature catalytic oxidation with IR detection according to APHA (2005, method 5310B) with a detection limit of 0.2 µg L^{-1}.

pH, dissolved oxygen concentration (DO) and water temperature were measured approximately 200 mm below the water surface, and 150 mm above the bottom of the tank, within each mesocosm on each sampling day. pH was measured using a TPS™ WP-81 portable meter, while DO and temperature were measured using a TPS™ WP-82Y portable meter.

2.2.2 Sonde monitoring

During two periods (Dec 2009, May 2010), environmental conditions within each mesocosm were monitored using a selection of multiprobe sondes (either DataSonde 4a or Minisonde 4a, Hydrolab, Hach Environmental, Loveland CO, USA; RBR TDO-2050, Richard Brachner Research Ltd, Ontario, Canada). Sondes were placed near the inflow and outflows (deep, 20 cm above the bottom sediment; shallow, 20 cm beneath the water surface) as well as in the inflow pumping chamber situated in the stream. Temperature, pH, DO and electrical conductivity were measured at 15 minute intervals. At the time of the May 2010 monitoring, a non-toxic fluorescent dye (Rhodamine WT) was added to each side of the mesocosm and monitored with either a Hydrolab DataSonde3 (Hach Hydromet, Loveland CO, USA) with a Rhodamine fluorometer sensor attached, or a Seapoint Rhodamine Fluorometer (SFR, Seapoint Sensors Inc, Exeter NH, USA) connected to a Li-cor Li-1000 data logger (LI-COR

Environmental – GmbH, Lincoln NE, USA). These data were used to assess hydraulic mixing characteristics (mixed v. plug-flow).

2.2.3 Sediment sampling

A series of sediment cores were collected from within the chambers on two occasions: once shortly after sediment addition (November 2009), and once at the end of the trial (June 2010). Samples were collected at intervals along each chamber using a sediment sampler (Figure 6) inserted into the sediment through the gap between the FTWs. To retain the sample a valve at the top was sealed, and the tube withdrawn. The base cap was added immediately prior to the tube exiting the water. The depth of the sediments was recorded through the transparent sides of the sampling tubes. The contents were transported back to the laboratory. After drying (80 °C), dry weight was determined. The organic content of the sediments was determined after ashing (400 °C for 8 hrs).

Figure 6: Sediments collected in sampling tube from base of experimental tanks.

2.3 Final plant harvest

At the end of the trial an entire above-mat portion of a single plant was harvested from each FTW (Figure 7). This leafy material was placed in a paper bag and dried at 80 °C until bags were a constant weight (approximately 1 week). The plants selected for sampling were the more accessible outermost ones close to the outflow end of each mesocosm. In one instance an adjacent plant was selected as the original plant was noticeably atypical in size.

Figure 7: Plant stem-bases after harvest.

An area of root material (30 cm wide) was removed along the underside of each FTW (Figure 8 & Figure 9). This width was considered to represent that occupied by roots from one row of plants. In most cases, roots from individual plants could not be separated from those of their near neighbours. On some FTWs, however, the below-water root development was sufficiently restricted to demonstrate that the width selected was appropriate (Figure 10).

The harvested bulk root material from each FTW was placed in a paper bag and dried in the same manner as the above-mat material.

Figure 8: Extensive root development underneath FTW.

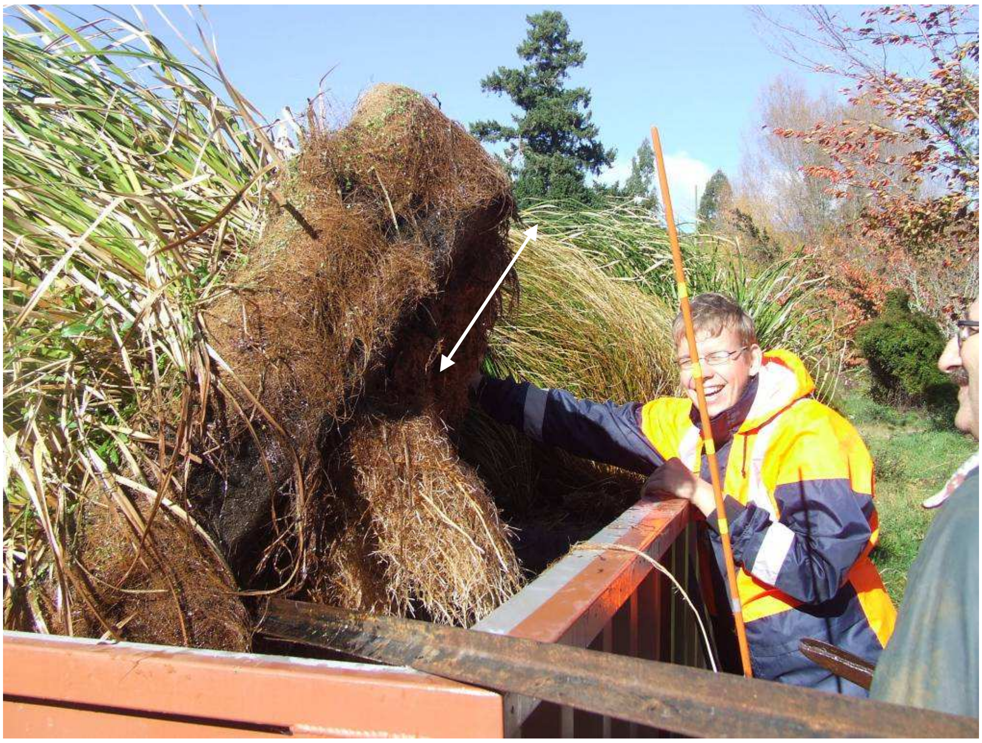

Figure 9: FTW with below-mat root material removed from outer edge (harvested area shown by arrow).

Figure 10: Section of FTW where root development was sufficiently restricted to demonstrate that the harvested area along one edge approximated the width of roots from the outer row of plants.

Plant material growing within the fibrous matrix of the floating mat could not be directly sampled. Previous studies indicated that this was likely to comprise 20-30% of the total plant biomass.

Subsamples of above- and below-FTW plant material were sent for nutrient analysis (mixed pasture nutrition) at a commercial/research laboratory (NZ Labs, Ruakura, Hamilton).

3. Results

3.1 Routine water quality monitoring

3.1.1 Phosphorus and nitrogen

Results from routine water quality monitoring are shown in Table 1. Data are expressed as inflow and outflow concentrations with associated percent removals, and as inflow loadings and associated areal mass removals.

Table 1: Summary of mean nutrient concentrations (± standard deviation), mass loadings/removals and percentage removal within the FTW chambers.

Variable		High loading			Low loading		
		inflow	**outflow**	**removal**	**inflow**	**outflow**	**removal**
DRP	Concentration (mg m^{-3})	51 (±12)	15 (± 11)	70%	51 (±12)	9 (± 7)	82%
	Areal mass loading/removal rate (mg m^{-2} day^{-1})	13	3.9	8.9	4.9	0.9	4.1
	Mass loading/ removal* (g m^{-2})	3.3	1.0	2.3	1.2	0.2	1.0
TP	Concentration (mg m^{-3})	67 (± 14)	46 (± 26)	32%	67 (± 14)	43 (± 37)	35%
	Areal mass loading/removal rate (mg m^{-2} day^{-1})	17	12	5.4	6.5	4.2	2.3
	Mass loading/removal* (g m^{-2})			1.4	1.6	1.1	0.6
NH4-N	Concentration (mg m^{-3})	9 (± 7)	32 (± 9)	increase	9 (± 7)	36 (± 25)	increase
	Areal mass loading/removal rate (mg m^{-2} day^{-1})	2.3	8.2	-5.8	0.9	3.5	-2.6
	Mass loading/removal* (g m^{-2})	0.6	2.1	-1.5	0.2	0.9	-0.7
NO3-N	Concentration (mg m^{-3})	1976 (± 60)	850 (± 293)	57%	1976 (± 60)	158 (± 134)	92%
	Areal mass loading/removal rate (mg m^{-2} day^{-1})	502	216	286	193	15	178
	Mass loading/ removal* (g m^{-2})	126.5	54.4	72.1	48.6	3.8	44.8
TN	Concentration (mg m^{-3})	2078 (± 57)	1136 (± 434)	45%	2078 (± 57)	473 (± 179)	77%
	Areal mass loading/removal rate (mg m^{-2} day^{-1})	528	289	239	203	46	157
	Mass loading/removal*(g m^{-2})	133.1	72.8	60.3	51.2	11.6	39.6

*Cumulative removal over the ~9 month (252 d) monitoring period.

Average inflow concentrations of DRP and TP were 51 and 67 mg m^{-3} respectively, indicating the majority of phosphorus was in a dissolved form. Similarly most of the inflowing nitrogen was dissolved, present mainly as nitrate-N, which comprised 95% of inflowing TN, with ammoniacal-N 0.4% and the remainder organic-N at around 4.6%.

At the higher hydraulic loading rate, DRP removal was 70% (8.9 mg m^{-2} d^{-1}), but much lower at 32% removal of TP (5.4 mg m^{-2} d^{-1}), indicating some uptake of dissolved nutrients and conversion into particulate organic forms. Similar behaviour was observed for nitrogen with 57% removal of nitrate-N (286 mg m^{-2} d^{-1}) at high loading, but less attenuation of TN (45%, or 239 mg m^{-2} d^{-1}). Some of the nitrate-N attenuation included conversion to ammoniacal-N. This increase in ammoniacal-N represented about 1.5% of inflow TN which is of minimal significance.

The reduction in P and N concentration from inlet to outlet at the lower hydraulic loading rate was 82% (4.1 mg m^{-2} d^{-1}) for DRP and 35% (1.4 mg m^{-2} d^{-1}) for TP. At this lower loading rate, nitrate-N removal was 92% (178 mg m^{-2} d^{-1}), but with some conversion to organic matter and ammoniacal-N, TN removal was lower at 77% (155 mg m^{-2} d^{-1}).

While the low loading rate treatment had higher removal rates in terms of concentration reduction, the high loaded system removed a larger total mass of nutrients. For DRP and TP, the high loaded tank attenuated twice as much as the low loaded tank. The difference between the two treatments for N removal was not quite as high as for P but the high loaded tanks still removed approximately 1.5 times more than the low loaded tanks.

Season had little apparent effect on DRP and TP removal, but DRP removal increased in the latter part of the monitoring period in the higher loaded tank, whilst TP removal declined (Figure 11). A seasonal effect on N removal was more evident for TN and NO_3-N. Outlet TN concentrations decreased for the higher loaded tank during the four warmer months of November 2009 to February 2010 (Figure 12). During these four months, the temperature within the tanks was an average of 4 °C warmer than during the other months monitored.

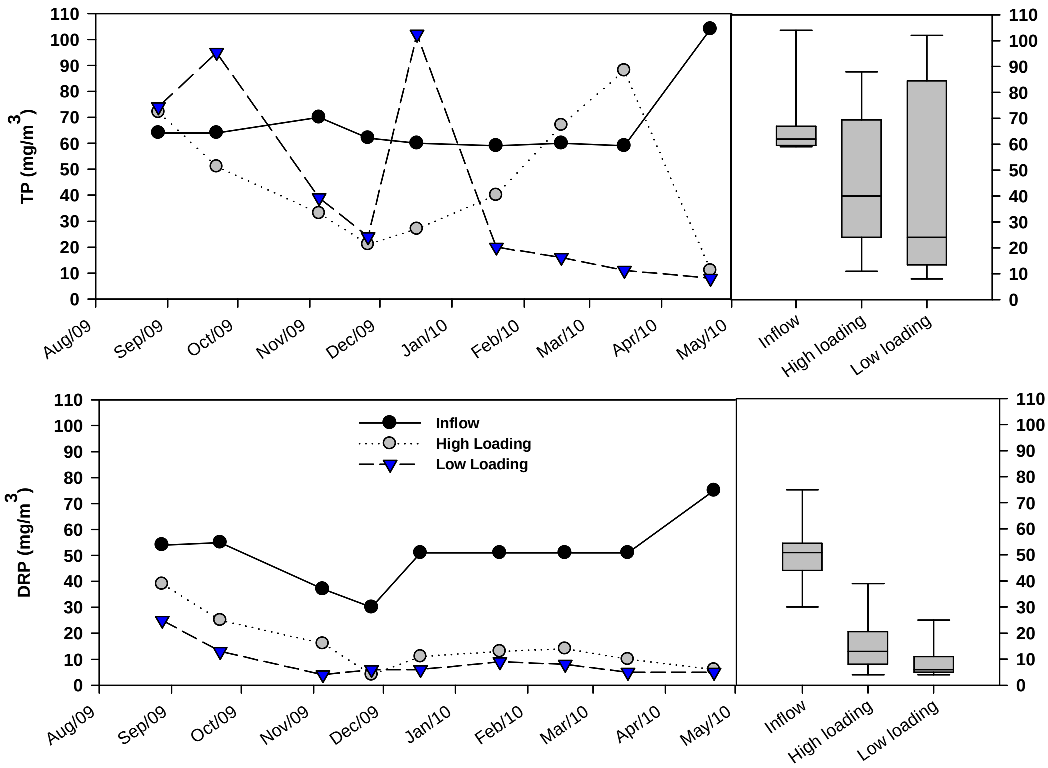

Figure 11: Concentration of inflow and outflow TP (top) and DRP (bottom) concentrations for high loaded (grey circles) and low loaded (blue triangles) tanks from August 2009 to April 2010 (left panels). Comparison of all inflow and outflow concentration data for TP and DRP are shown as box and whisker plots (right panels).

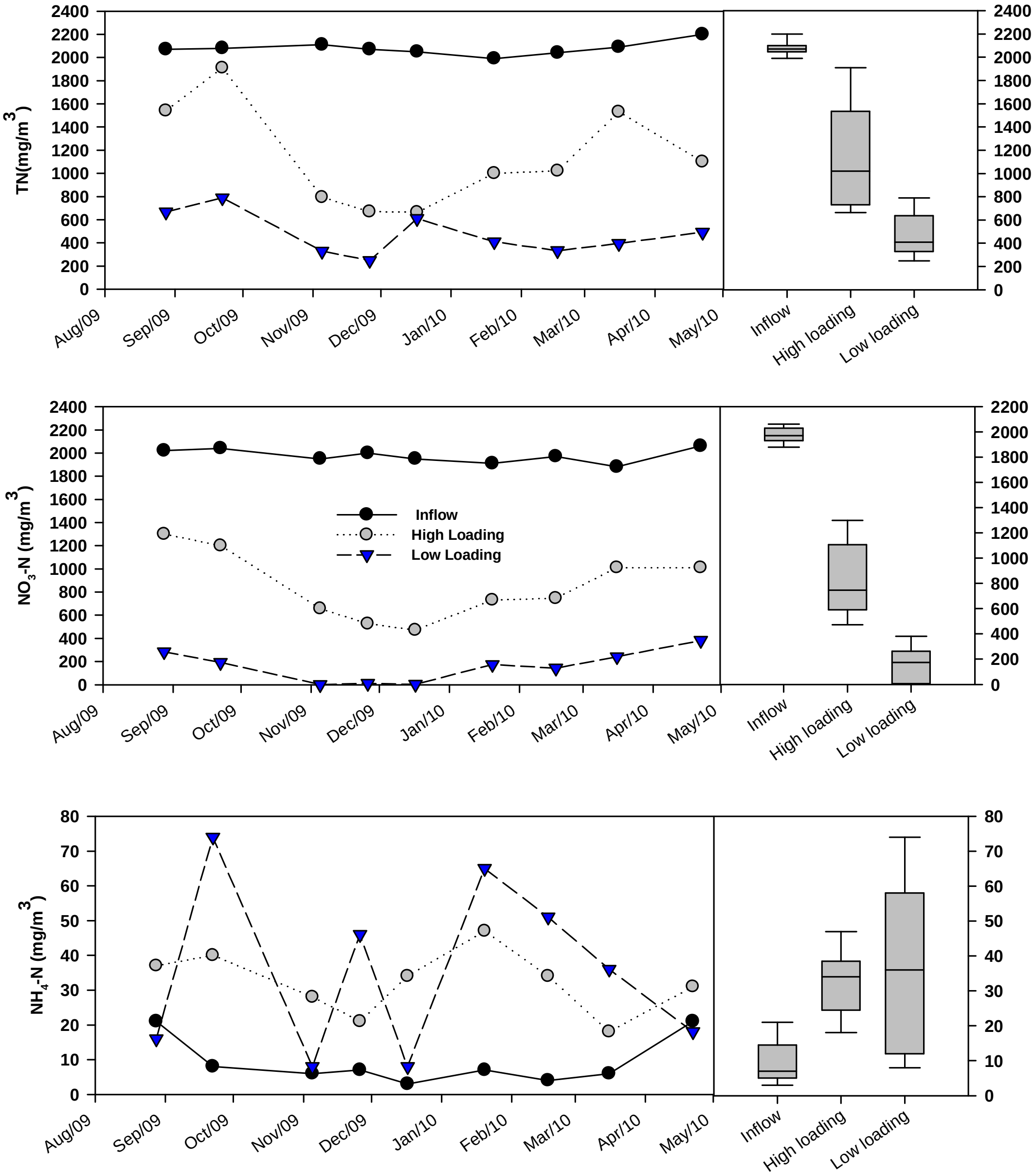

Figure 13: Concentration of inflow and outflow TN (top), NO_3-N (middle) and NH_4-N (bottom) for high loaded (grey circles) and low loaded (blue triangles) tanks from August 2009 to April 2010 (left panels). Comparisons of inflow and outflow concentration data are shown as box and whisker plots (right panels). Note different scale for ammoniacal-N.

The inflow from the stream was well oxygenated, with average dissolved oxygen of 8.8 g m^{-3} over the nine month study (Figure 14). Oxygen concentrations declined from the inflow to the outflow of the tank, indicating increasing anoxia from root, biofilm and benthic respiration. The oxygen consumption was higher in the tank with the longer HRT (lower loading), and conditions were also more anoxic closer to the sediment.

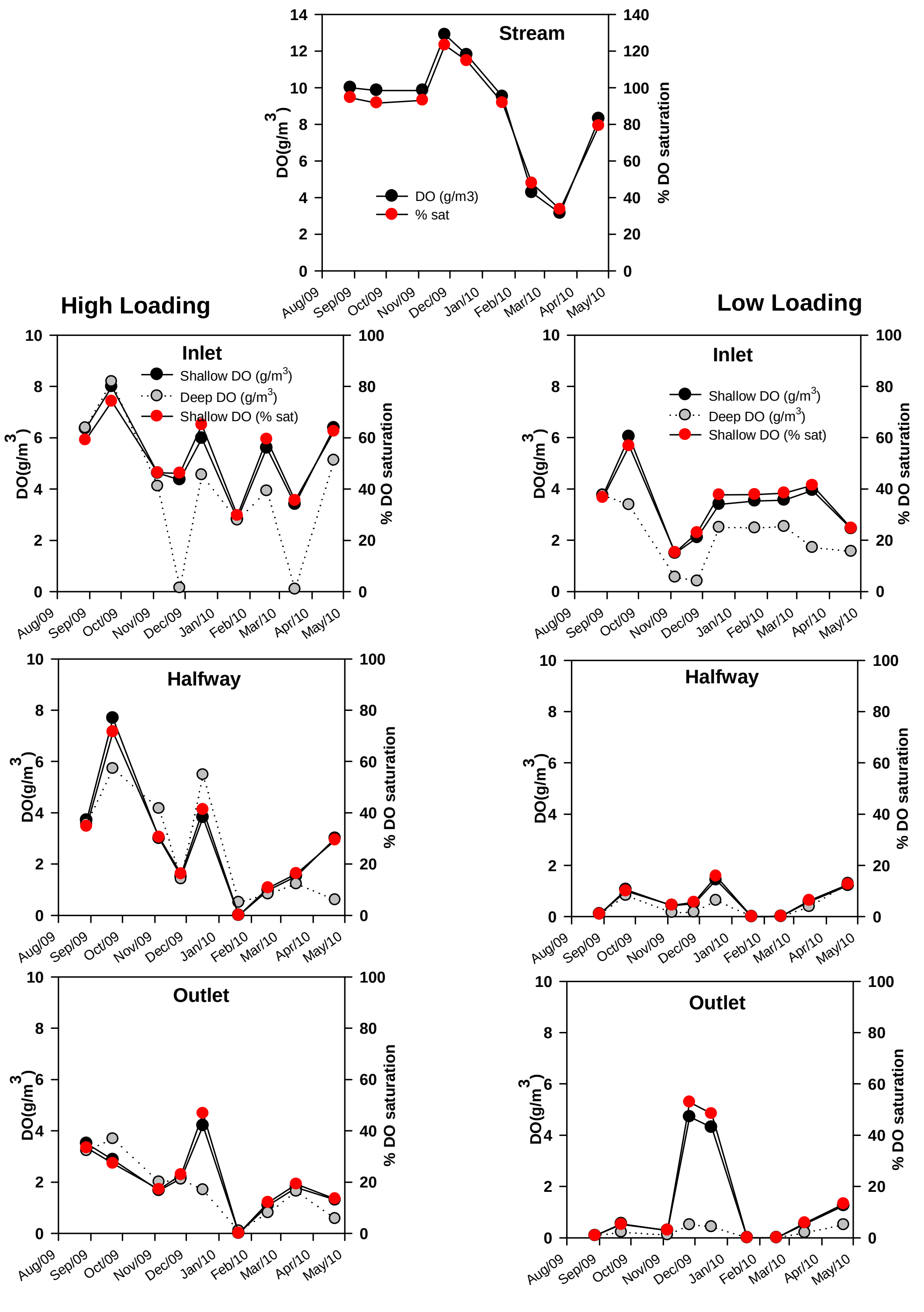

Figure 14: Dissolved oxygen data for stream (top), chamber water column at inlet, halfway and outlet positions (shallow and deep) for high loaded chamber (left) and low loaded chamber (right).

3.2 Sonde monitoring

3.2.1 Temperature

Temperature data for the various locations monitored during the December 2009 sonde deployment have been combined in Figure for the high loaded chamber and in Figure 13 for the low loaded chamber.

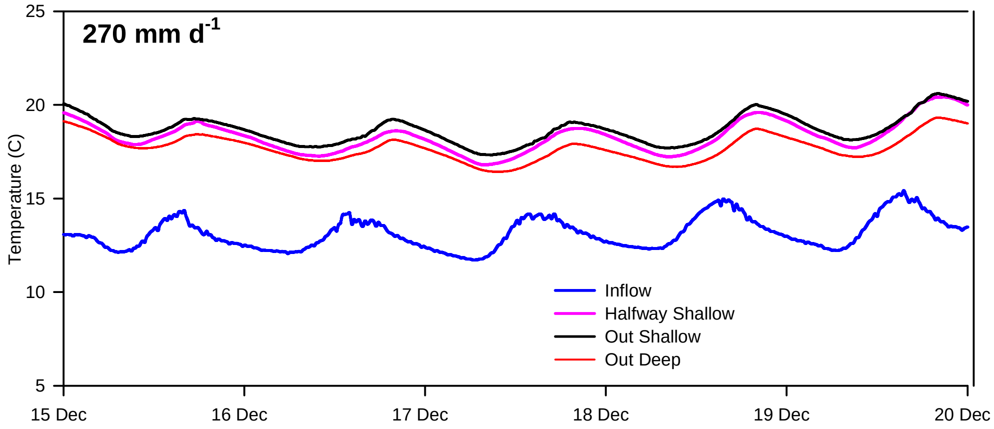

Figure 14: Temperatures measured in the high loaded chamber (Dec 2009).

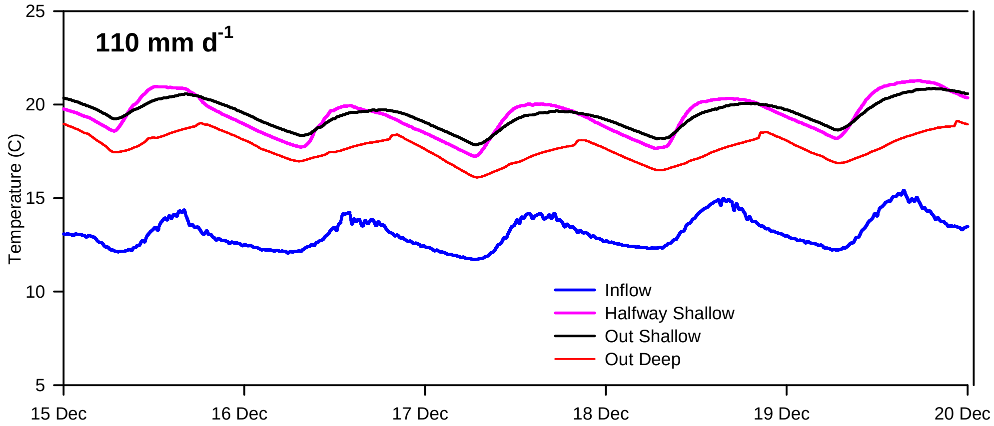

Figure 13: Temperatures measured in the low loaded chamber (Dec 2009).

Both graphs show a clear diurnal cycle of water temperature fluctuations, both in the inflow from the Maero Stream and within the tanks. The inflow fluctuated between 12.5 and 15.5 °C, in both chambers. Water temperatures rose by 3-5 °C during passage through the chambers, with lower temperatures near the base of the tanks than at the surface.

Temperature data for the various locations monitored during April 2010 have been combined in a similar fashion for the high loaded chamber (in Figure 14), and for the low loaded chamber (in Figure 15).

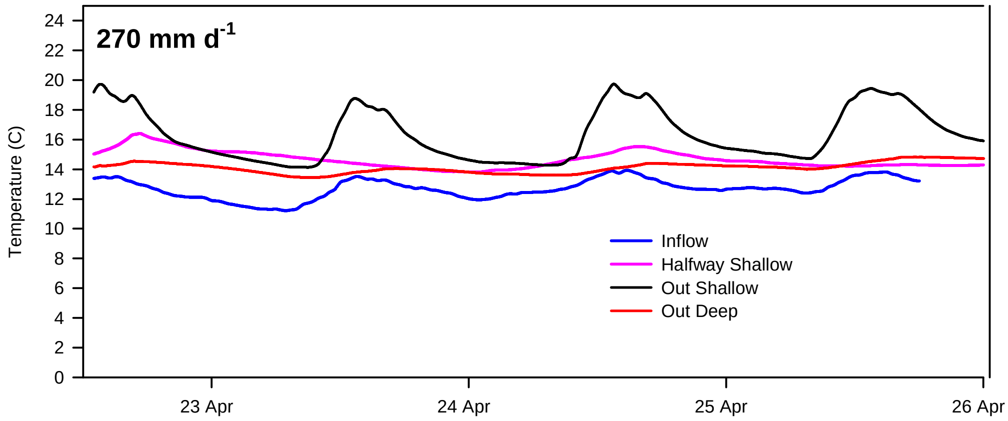

Figure 14: Temperatures measured at various locations within the high loaded chamber during April 2010.

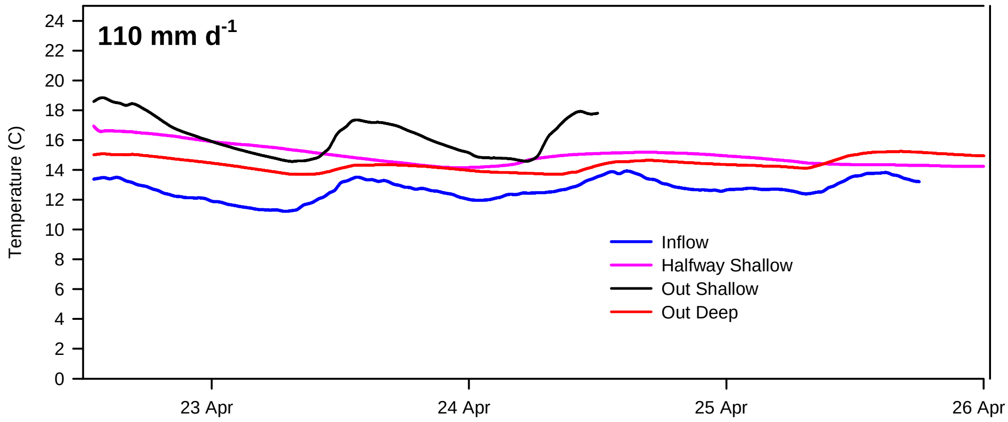

Figure 15: Temperatures measured within the low loaded chamber during April 2010.

Inflow temperatures fluctuated between 11.2 and 13.8°C with a clear diurnal cycle, similar to those measured in December 2009. Temperatures in the mesocosm were 5 to 7 °C above that of the inflow, and temperatures near the base of the tanks were cooler than at the surface.

3.2.2 Dissolved Oxygen Concentrations

Dissolved oxygen (DO) concentrations for the December 2009 (summer) and April 2010 (winter) monitoring periods are shown in Figure 16 through Figure 19.

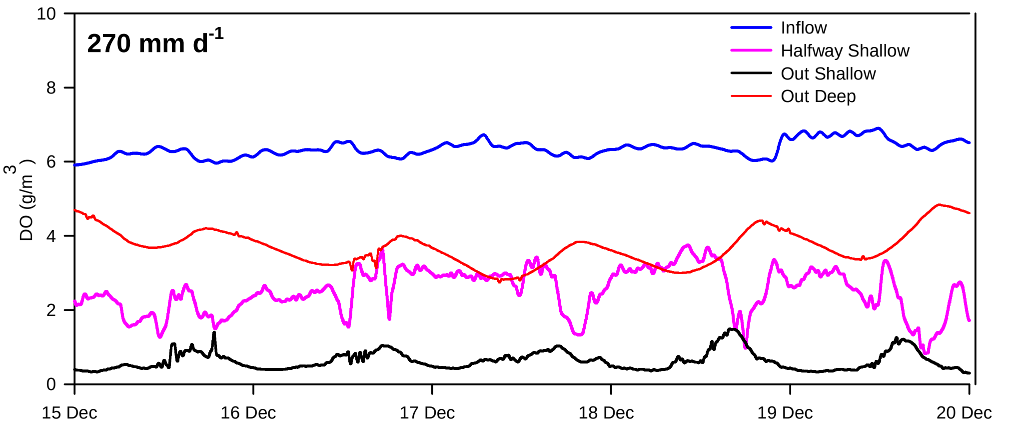

Figure 16: Dissolved oxygen concentrations measured within the high loaded chamber (Dec 2009).

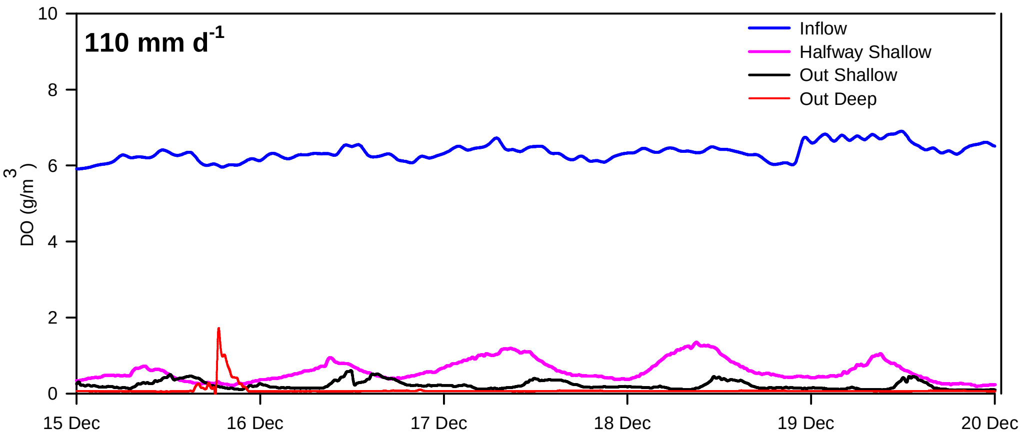

Figure 17: Dissolved oxygen concentrations measured within the low loaded chamber (Dec 2009).

The DO concentrations in the inflow were collected from a chamber set into the bank of the Maero Stream from which it was pumped into the mesocosms. The operation of the pump caused some spiking in the DO concentrations which have been removed from the data traces in Figure 16 and Figure 17 by plotting the lower bounds of the DO data (from the beginning and end of the pumping cycle). DO concentrations were consistently greater than about 6 g m^{-3} in the inflow chamber, and a diurnal pattern is also apparent.

DO concentrations became distinctly lower within the mesocosms, most notably in the longer hydraulic residence time (low loaded) treatment, where processes which could consume oxygen (e.g., organic matter degradation, sediment oxygen demand, net plant root and biofilm respiration) had more time to operate. Lower DO concentrations were occasionally recorded in surface waters than deep waters. This had also been noted occasionally during routine (monthly) monitoring and probably represents periods when root biomass (greatest near the surface) and associated respiration was causing DO depletion in the upper waters.

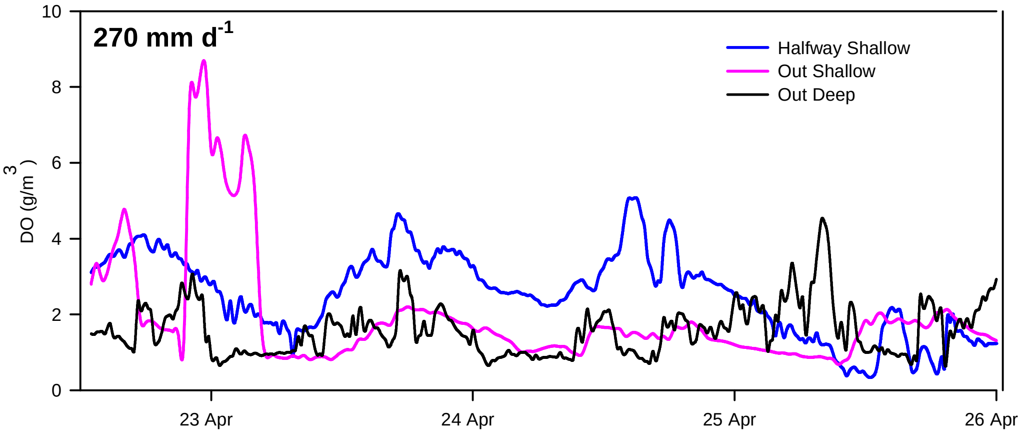

Figure 18: Dissolved oxygen concentrations measured within the high loaded chamber (April 2010).

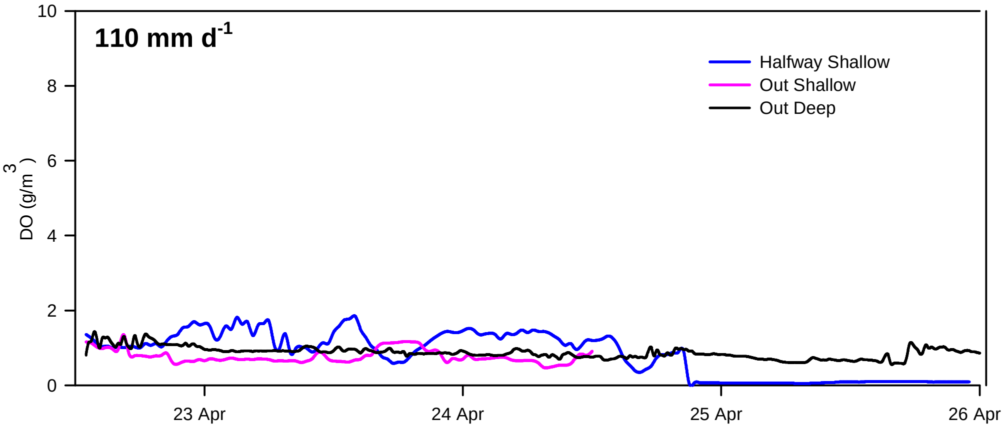

Figure 19: Dissolved oxygen concentrations measured within the low loaded chamber (April 2010).

The dissolved oxygen probe on the sonde in the inflow chamber gave unreliable results during the April monitoring period. These data have not been included in Figure 18 and Figure 19. In general it can be noted that the DO concentrations in the high loaded chamber are generally in the range of 1–4 g m^{-3}, occasionally reaching about 8 g m^{-3} at the surface. DO concentrations in the lower loaded chamber are consistently less than 2 g m^{-3}, following the trends seen during the summer monitoring.

3.2.3 pH

pH values for the December 2009 (summer) and April 2010 (winter) monitoring periods are shown in Figure 20 through Figure 23.

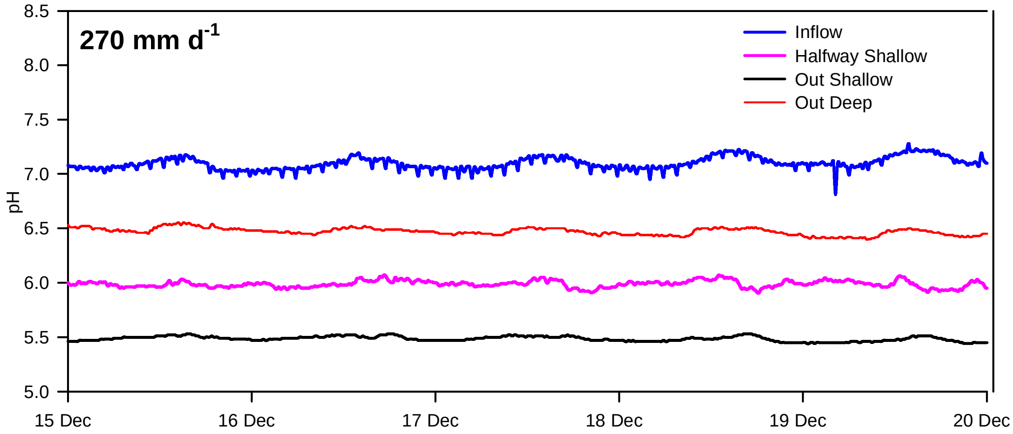

Figure 20: pH within the high loaded chamber (Dec 2009).

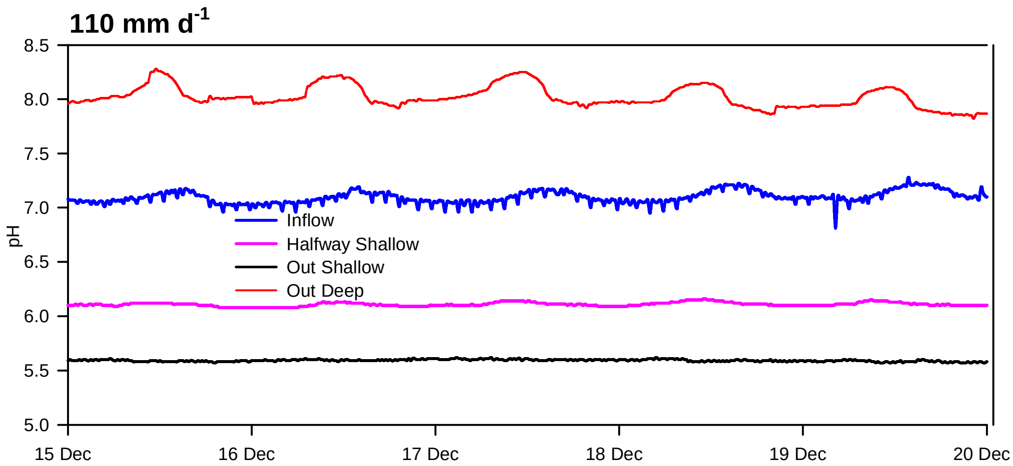

Figure 21: pH within the low loaded chamber (Dec 2009).

Inflow pH values were very stable, with only slight diurnal fluctuations between 7.0 and 7.2. Within the chambers, the near-surface sondes recorded lower pH values which were highly stable. At the half-way point, the pH was around 6.0 in both chambers, declining to around 5.6 at the outflow point. Values were higher close to the sediment at the base of the chambers, at around 6.5 in the high loaded chamber, and pH 8.0 in the low loaded chamber.

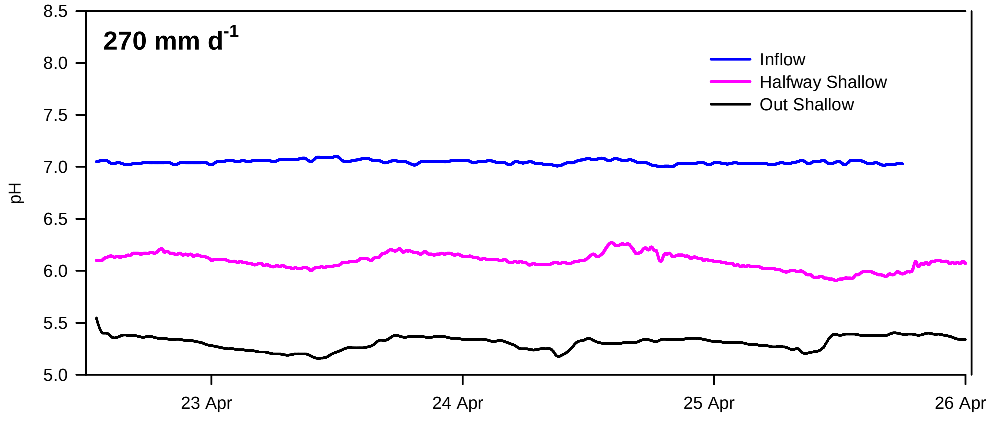

Figure 22: pH within the high loaded chamber (April 2010).

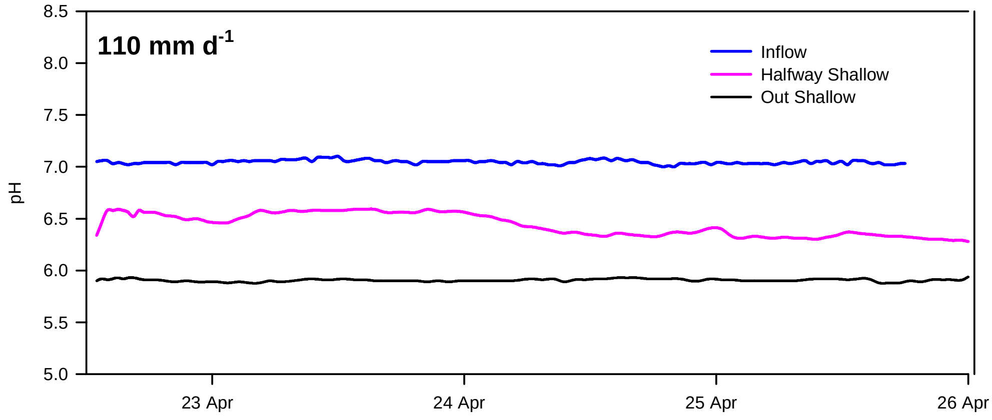

Figure 23: pH within the low loaded chamber (April 2010).

The inflow pH was again around 7.0 during the winter monitoring, with an even smaller diurnal variation than was apparent in the summer monitoring. The pH measured within the chambers during winter were similar to those measured during the summer, with lower pH at the half-way point in both chambers, which was reduced further at the outflow point. No pH values from near the base of the chambers are available for this period.

3.2.4 Electrical Conductivity

Conductivity data for the summer 2009–2010 summer is presented in Figure 24 and Figure 25.

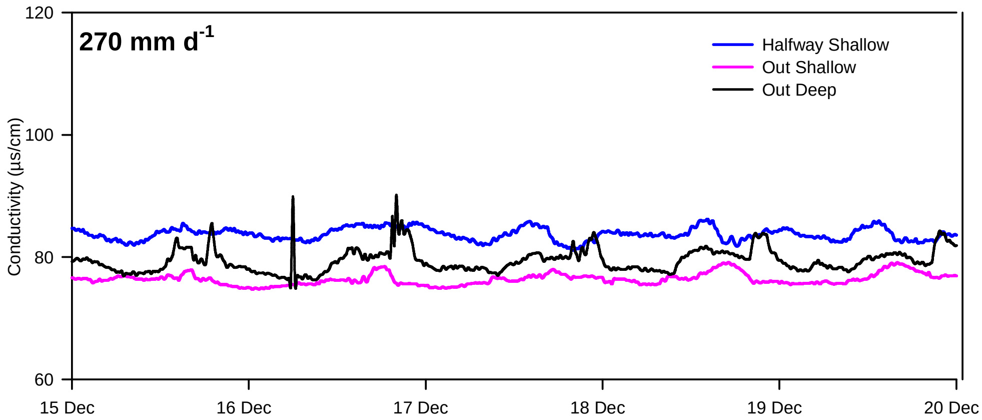

Figure 24: Electrical conductivity in the high loaded chamber (Dec 2009).

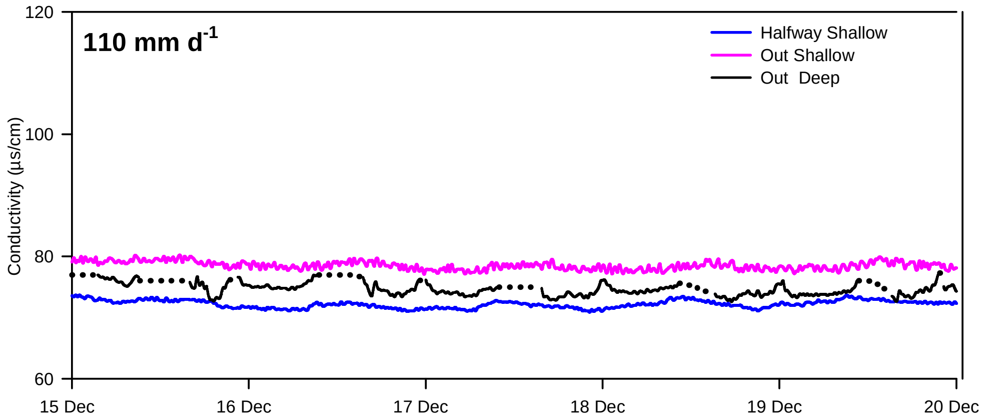

Figure 25: Electrical conductivity in the low loaded chamber (Dec 2009)[1].

In general the conductivity graphs are relatively stable and with only minor differences between the different sampling locations, indicative of a relatively fully-mixed system.

[1] Various peaks were evident in the HRT10 Outlet Deep trace. These appeared to be an artefact associated with sonde operation rather than actual changes in WQ in the mesocosm. Thus they have been removed. Dashed lines have been used to indicate where the data has been removed.

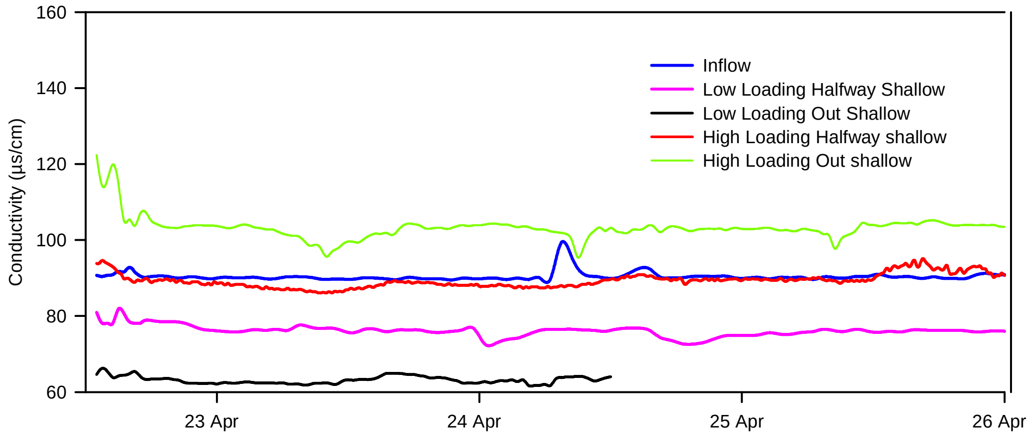

Figure 26: Combined conductivity graphs (April 2010).

Figure 26 demonstrates conductivity concentrations of a similar order to that seen in the summer monitoring, while also indicating that the inflow was stable, and around 90 µS cm^{-1}. Values within the chamber were also stable, but were sometimes higher, and sometimes lower than the inflow.

3.3 Dye tracer study

At the end of the trial, a tracer study test was conducted in chamber 1 (low hydraulic loading) of the mesocosm. These data enabled a comparison of the actual residence time with the nominal residence time of 9.6 d. The Rhodamine WT concentrations at the outflow point of the first and second cell of chamber 1 are shown in Figure 27. Sampling at multiple points allowed assessment of the degree of mixing within the tank.

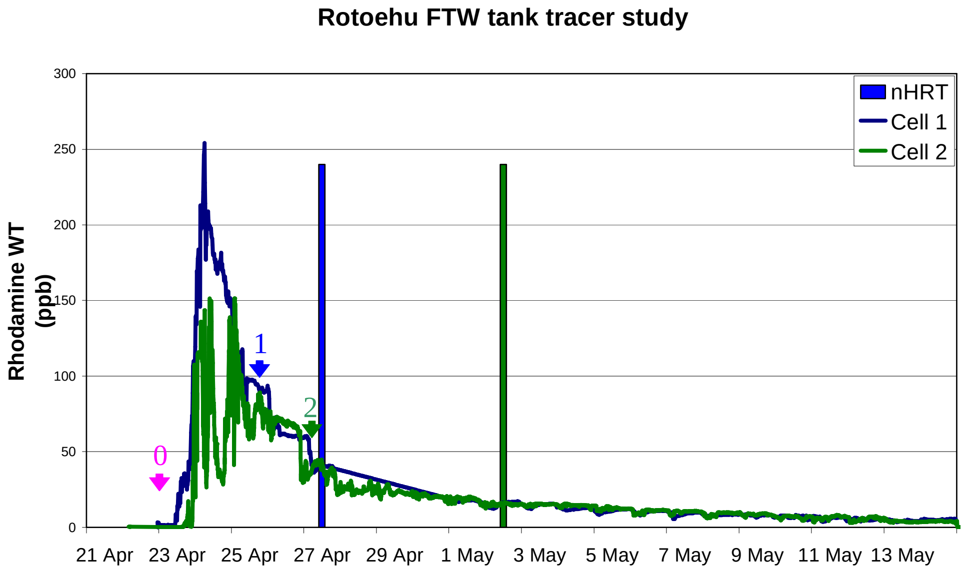

Figure 27: Rhodamine WT tracer from both cells of the high loaded chamber. Arrows indicate: time of tracer addition (0), tracer centroid for cell 1 (1) and cell 2 (2). The Nominal (or theoretical) retention times for each cell are shown as bars.

Figure 27 displays a typical trace obtained from a wetland tracer study, with the peak skewed to the left followed by a long tail. The nominal HRT (nHRT) of each cell is shown as a coloured bar, and is calculated according to the following formula (Kadlec, & Knight 1996, Eq 9-89):

$$nHRT = \frac{V}{Q}$$

Where V = volume of water in the chamber (m^3)

Q = water flow rate, ($m^3\ d^{-1}$)

The actual HRT is measured as the centroids for each trace (centre of the mass of tracer), which are shown as coloured arrows, and are calculated according to the following formula (Kadlec & Knight 1996, Eq 9-95):

$$\int_0^\infty tf(t)dt = HRT$$

Where t = time, d

f = Residence Time distribution function, 1 d^{-1}

The centroids are well to the left of, or shorter than, the nHRTs, indicating the chambers do not exhibit plug flow, exhibiting a pattern closer to a fully mixed chamber, thus have a much shorter actual HRTs than the theoretical HRTs.

Despite the elongated nature of the chambers (approximate length to width ration of 6:1), and the intermediate dividing wall, the tracer peak occurred after about 1 day in the first half of the tank and about 2 days in the second half, with an overall mean HRT of about 4 days, compared to a theoretical HRT of approximately 9.6 days.

From the actual HRT calculated above, it is possible to calculate the active volume of the system, assuming that the total and active flow rates are equal, i.e., $Q = Q_a$ which is generally true (Simi & Mitchell 1999, Smith et al., 2005).

$$V_a = \frac{Q_a \times HRT \times 100}{V_T}$$

Where V_a = active volume of wetland (%)
Q_a = active flow rate ($m^3\ d^{-1}$)
V_T = total volume of wetland (m^3)

Table 2: Summary of hydraulic characteristics for the low loaded tank.

Flow rate ($m^3\ d^{-1}$)	Volume (m^3)	Nominal HRT (days)	Actual HRT (centroid of cell 2)	Active volume (%)
1 m^3	9.2m^3	9.6 days	3.2 days	33 %

The large mass of roots growing beneath the FTWs likely explains the reduced active volume. As the root mass increased, the flow would have been shunted beneath it rather than through it (path of least resistance), resulting in a shorter actual hydraulic residence time than the nominal residence time.

3.4 Sediment characteristics

Accumulation of organic matter within the trial chambers was determined by measuring the sediment depth and sampling the sediments at the beginning (after a period of settling/consolidation) and the end of the trial. The organic carbon fraction of the sediments was estimated by Loss-on-Ignition; it ranged from 11–15% (Table 3). While these results show a small increase in sediment depth and organic carbon content, the differences are not likely to be statistically significant.

Table 3: Sediment depth and organic carbon content measured in cores taken from the experimental tanks at the beginning and end of the trial.

	Beginning of trial (August 2009)		End of trial (June 2010)*	
	Sediment depth (cm)	Average organic C % (± std dev.)	Sediment depth (cm)	Average organic C % (± std dev.)
Average of both tanks	5.8 (±1.7)	13.0% (±3.0%)	7.4 (±2.8)	13.4% (±3.4%)
Low loading	6.6	11.2%	5.8	13.1%
High loading	5.2	14.9%	9.6	13.4%

*Note: These dates relate to set-up and final harvest of the FTW trials, and do not coincide with the beginning and end of the water quality sampling period.

3.5 Plant biomass and nutrient uptake

The canopy height and root depth of the six FTWs were measured three times during the study (Figure 28). Both species exhibited good growth but the *Cyperus ustulatus* showed greater canopy height and shorter root length than the *Carex virgata*. By the end of the trial, roots had penetrated beneath the floating mats to depths of 90 mm (starting to grow into the sediment at the base of the tank) for the *Care*x species. Roots of *Cyperus* grew to a maximum length of ~60 cm by the end of the trial

The *Cyperus* had noticeably higher wet weight of shoot biomass than the *Carex*, (15 kg m^{-2} compared with 10 kg m^{-2}), although after drying the biomasses of the two species were essentially the same (4.8 kg m^{-2} and 5.9 kg m^{-2} respectively). In contrast, the *Carex* had much higher below-mat (root) biomass (both wet and dry). This difference may have important implications for plant selection where FTW stability is an issue. A species such as *Carex* may provide greater stability due to their lower relative above- to below-water biomass. More extensive root development also provides a greater surface area for plant nutrient uptake and biofilm growth.

The above and below-mat biomass and tissue nutrient contents are presented in Table 5 for nitrogen and phosphorus.

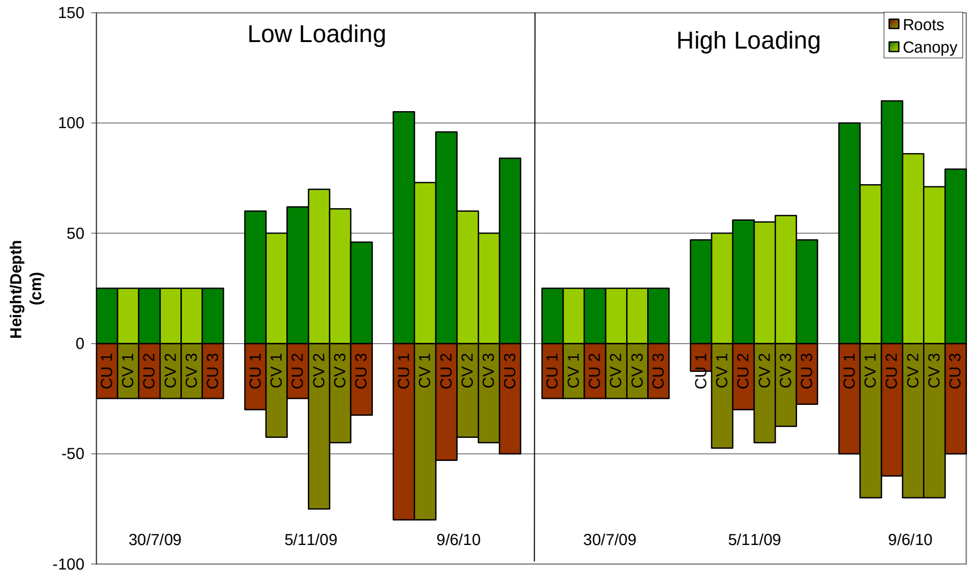

Figure 28: Plant canopy height and root depth development during the experimental trials. (CU = *Cyperus ustelatus*, CV = *Carex virgata*).

A summary table of plant macro-nutrient data is included in Appendix 1. Each chamber contained total FTW areas of 10.3 m^2, comprising 4.8 m^2 of *Cyperus* and 5.5 m^2 of *Carex*. Based on the above nutrient analyses, these equate to total storage of nitrogen in plant biomass of 36-44 g m^{-2} (low loading) and 40-64 g m^{-2} (high loading), and phosphorous uptake of 2.9-3.0 g m^{-2} (low loading) and 3.5-4.1 g m^{-2} (low loading).

Table 4: Average shoot and root biomass of the plants in the experimental tanks at the end of the study.

Sample source	Biomass	*Cyperus* kg m^{-2}	*Cyperus* per plant (g)	*Carex* kg m^{-2}	*Carex* per plant (g)
Above-mat	Wet	15	1367	10	950
(Shoots)	Dry	4.4	408	4.3	402
Below-mat	Wet	4.5	437	12	1225
(Roots)	Dry	0.4	38	1.6	160

Table 5: Plant height, biomass, nutrient content and uptake.

	Cyperus ustulatus					
	Above-mat		**Below-mat**		**Total**	
	Low loading	**High loading**	**Low loading**	**High loading**	**Low loading**	**High loading**
Height (cm)	95	96	61	53		
Dry wt (g m^{-2})	4230	4588	378	390	4608	5667
Plant [N] (g m^{-2})	32	35	4	5	36	40
%	0.75	0.75	1.14	1.27		
Plant [P] (g m^{-2})	2.5	3.0	0.4	0.5	2.9	3.5
%	0.06	0.06	0.11	0.14		
	Carex virgata					
Height (cm)	68	76	67	70		
Dry wt (g m^{-2})	3508	5176	1477	1753	4985	6929
Plant [N] (g m^{-2})	33	52	11	12	44	64
%	0.94	1.00	0.73	0.71		
Plant [P] (g m^{-2})	2.1	3.2	0.8	0.9	3.0	4.1
%	0.06	0.06	0.06	0.05		

4. Discussion

The hydraulic application rates (110 and 270 mm d^{-1} for the low and high inflows respectively) are relatively high compared to areal hydraulic loading rates commonly applied to conventional surface-flow constructed wetlands. However, because of the greater water depth beneath the FTWs (0.9 m cf. 0.3-0.4 m depth in surface-flow wetlands) their nominal hydraulic retention times (HRT, 9.4 and 3.6 d respectively) were comparable with the range commonly applied to surface-flow systems. In our earlier small-scale FTW trials (Park et al., 2008), performance was assessed in 0.7 m deep tanks during 7 d batches, which is equivalent to average areal hydraulic loadings of ~100 mm d^{-1} and similar to the low loading rate used here.

In the present study, incoming nutrients were mainly in soluble forms, with 95% of nitrogen in dissolved inorganic forms, comprising primarily nitrate, and 77% of incoming phosphorus present as DRP. When comparing the reduction in concentration from the inflow to the outflow, in every instance greater percentage removal occurred in the lower loaded (higher HRT) tank. This reflected the increased time available for attenuation via sorption, plant uptake and microbial processing.

The mean DRP concentration was reduced by 82% at low loading, declining to 70% at high loading. Attenuation of TP was lower at 35% and 32% respectively, indicating that a portion of the dissolved fractions removed were being incorporated within short duration storage pools and re-released as particulate P, most probably in organic forms such as algal matter or biofilm flocs. In the longer term as the vegetation matures, some release of dissolved and particulate organic matter from senescing and decaying plant litter is also likely.

The mean nitrate-N concentration was reduced by 92% at low loading, declining to 57% at high loading. While there were small increases in ammoniacal-N during passage through the tanks, outflow concentrations still remained low are not considered significant in the overall nitrogen budget. It is expected that some of the ammoniacal-N production will be from hydrolysis of organic matter, which was most probably produced within the mesocosm itself (as there was little organic matter in the stream inflow). In addition to the hydrolysis of organic matter, some ammoniacal-N could have been produced microbially via dissimilatory nitrate reduction to ammonium (DNRA, Tiedje 1988). Reduction of TN concentration was greater at the low loading than at the high loading (77% and 45% respectively).

When assessing nutrient removal performance, it is the areal mass removal rates (g m^{-2} day^{-1}) which are of most importance, rather than the concentration reduction. Areal mass removal rates for the FTWs were 2.3 and 5.4 mg TP m^{-2} d^{-1}, and 157 and 239 mg

TN m^{-2} d^{-1}, for the low and high loading respectively. Thus a greater overall quantity of nutrient was removed by passing a high flow of water and flux of nutrients beneath the FTWs. This trade-off between removal efficiency and load reduction is a fundamental performance principle that must be considered when using wetlands (and most other natural treatment systems) for nutrient control (Kadlec 2005).

The most comparable study undertaken of the attenuation potential of FTWs is that by Park et al (2008). This short-term (7d) batch study used smaller (0.7 m^3) mesocosms and artificial eutrophic lake water with concentrations of ~1000 mg N m^{-3} and ~180 mg P m^{-3}. This represents about half of the TN and 37% of the TP concentrations in Maero Stream water used in the present trial. Overall areal removal rates were higher than in the present study, ranging from 638-762 mg m^{-2} d^{-1} for TN, and 54–58 mg m^{-2} d^{-1} for TP. However in the earlier trial, the control tanks (in which shade cover equivalent to a FTW was provided) also showed significant levels of removal. The removal which could solely be attributed to the FTWs (i.e., omitting attenuation which may have been associated with algal uptake in the mesocosm tanks) was 339 mg m^{-2} d^{-1} of TN, and 30 mg m^{-2} d^{-1} of TP. In addition, the previous study was undertaken in summer when uptake rates would be expected to be at their highest. As a consequence we regard the rates measured in the present study to be broadly comparable with those measured by Park et al., (2008).

Finding directly comparable field data for conventional surface-flow wetlands is difficult, as the form of nutrients, and their concentration and loading rate greatly affect removal rates. In a review of 72 surface-flow wetlands receiving nitrate-rich inflows, Kadlec and Wallace (2009) report a median nitrate load removal of 140 g m^{-2} d^{-1}, which is slightly less than that recorded at lower nitrate loadings in the present study. Mitsch et al., (2005) summarised nitrate removal from river waters in the Mississippi River Basin for 12 wetland systems (total of 48 wetland years), defining an empirical relationship between the mass inflow and percentage nitrate-N mass removal. At equivalent loading rates, the percentage nitrate mass loads removed by the FTWs in our study were in the high range recorded for those systems. In comparison, attenuation rates in natural seepage wetlands are commonly much lower than those recorded for FTWs in the present study, with rates of 5-15 mg m^{-2} d^{-1} reported (Rutherford & Nguyen 2004). However, N loading rates also tend to be much lower in these natural systems. These comparisons suggest that the FTWs in the present study were meeting or exceeding the removal rates expected for conventional constructed wetland treatment systems.

Both of the plant species used in the present study showed grew well on the FTWs, accumulating more than 5 kg m^{-2} of live biomass dry weight. *Carex virgata* showed superior root development which is expected to improve its treatment efficacy and assist FTW stability in windy conditions. The final harvest (after approximately one years growth) indicated that roots from some of the plants had grown down into the sediments to a length of over 0.8 m. To avoid anchorage of roots into the bottom sediments, it is advisable that water depths of over 1 m are maintained beneath FTW. The high root biomass that developed below the floating mats also occupied a significant volume of the water column in the tanks, promoting short-circuiting of flow and significant reductions in hydraulic retention time. This was demonstrated by tracer testing which showed an active volume of only ~30% of the tanks. This reduced the contact time available for nutrient uptake and processing. This effect is likely to be more pronounced in this study than in an in-lake application due to the restricted volume of the mesocosms.

Plant biomass and nutrient content was not sampled at the start of the water quality monitoring period, so net plant uptake rates can not be directly measured and compared with measured reductions recorded from the through-flowing waters. However, if we assume that the unsampled plant material within the floating mat at the end of the trial is roughly equivalent to the plant material at the start of the trial (both estimated as ~20-30 % of biomass), and thus cancel each other out, we can make some approximations. These would suggest that all the P reductions and 66% or more of the N reductions measured in the experimental tanks could have been due to plant uptake. However, the N:P ratios measured in the plant tissues were in the range of 11-12, compared to measured N:P removal ratios of 43-66 from the tanks. This suggests that a significant proportion of the N removal must have occurred by means other than plant uptake (e.g., denitrification). The apparently greater accumulation of phosphorus in plant tissues than the amount removed from through-flowing waters suggests that high levels of nutrients were accumulated within the plants during their establishment prior to the trial, and/or nutrient release and subsequent uptake occurred from the lake sediments added to the experimental tanks that was subsequently assimilated by the plants.

As the FTWs mature, plant uptake of nutrients is likely to become restricted unless ongoing harvesting of plant biomass is undertaken. In practice, however, other removal mechanisms such as denitrification and sediment accumulation are likely to increase in importance as the plants reach maturity and provide an ongoing source of organic carbon in the form of plant litter and root exudates.

The sonde and routine field monitoring both highlighted the development of low oxygen conditions directly beneath the FTWs and at the water/sediment interface. Conditions became increasingly anoxic from inflow to the outflow of the mesocosms, and lower dissolved oxygen concentrations were measured in the lower loaded chamber with longer HRT. These low oxygen conditions are considered to result from root respiration, which probably also contribute to the observed reduction in pH in the mesocosms. Release of CO_2 as a result of such respiration shifts the carbonate equilibrium to production of carbonic acid. The relatively large reduction in pH of 1-1.5 units is consistent with the low alkalinity of many streams and water bodies in the wider Taupo Volcanic Zone (Timperley & Vigor-Brown 1986). While a low oxygen environment is essential for denitrification, it is not entirely clear to what extent this process was enhanced by containment of the water within a mesocosm. Where FTWs are deployed within a lake, water with a higher oxygen status would be able to move laterally under the FTWs, and may restrict development of localised anoxic conditions.

Park et al (2008) found algal settling beneath FTWs to be an important nutrient processing pathway. While accumulation of organic matter beneath the FTWs was not measurable in the present study, this process should not be discounted as some production of organic matter was beginning to occur. Plant and biofilm production from the FTWs themselves is an additional source of organic matter, which is likely to become more important as the system matures.

In studies of surface-flow wetland treatment of eutrophic lake waters Coveney et al., (2002) also found settling and retention of particulates (e.g., algae) was a key nutrient removal mechanism. The extent of deposition of these solids beneath the FTWs and the fate of associated nutrients is likely to depend on the size and orientation of the installations, and the situations in which they are deployed, and are likely to only be fully understood in full-scale trials.

5. Recommendations

We consider that the results of this study provide clear evidence of the efficacy of FTWs for nutrient removal, particularly nitrogen, from stream waters entering Lake Rotoehu. Larger-scale application in the lake is warranted to further evaluate their field performance. Concurrent monitoring of physico-chemical and biological conditions beneath and in the vicinity of the FTWs is recommended to evaluate the key environmental factors influencing performance in the lake. This will provide information to help optimise their layout and management, and evaluate their effects on aquatic life in the lake. The influence on aquatic life may include both positive and negative aspects; for example, fish and invertebrate avoidance may occur due to localised deoxygenation beneath the FTWs or conversely they may be attracted by the shelter and food-sources provided by the FTWs.

We recommend that a diversity of low-stature, hardy plant species are used in any larger-scale deployment to reduce the risks associated with reliance on a monoculture (e.g., rapid spread of disease or pests across the whole FTW). In addition to the two test species used here, we suggest inclusion of species such as *Juncus edgaria*e and *Schoenoplectus tabernaemontani*, that previous studies have demonstrated are likely to grow well and develop extensive submerged root systems in FTWs (Headley & Tanner 2007)

Important questions remain regarding about how best to set up and manage FTW systems in lake situations to optimise their nutrient removal performance. Because of the practical difficulties in measuring these processes in the lake, we recommend that a further period of monitoring should ideally be undertaken in the trial tanks with the more mature FTWs receiving lake waters from the vicinity of the Maero Stream mouth. This will provide:

- Longer-term data on FTW performance with mature vegetation
- Performance data for actual lake waters where a high proportion of nutrients are bound up in suspended algae
- Assessment of the influence of periodic harvesting of plant biomass on nutrient uptake rates.
- The ability to manipulate key factors driving nutrient removal (e.g. dissolved oxygen levels), to assist with optimisation of FTW use in the lake

As a guide to the expected performance of FTWs in the lake treating the outflow of the Maero stream, we suggest assuming a conservative mean annual nutrient removal rate based on the results of the present tank study of ~125 mg TN m^{-2} d^{-1} and 20 mg TP m^{-2} d^{-1}. This allows for the fact that the study did not measure removal rates through most of the winter period, when plant uptake and microbial process rates would likely be depressed. Extrapolated over a year this equates to ~46 g TN m^{-2} y^{-1} and ~7.3 g TP m^{-2} y^{-1}, or 460 kg TN ha^{-1} y^{-1} and 73 kg TP ha^{-1}.

6. Acknowledgements

This study was commissioned and funded by Environment BOP. We wish to acknowledge their support, particularly Andy Bruere and John McIntosh who provided oversight of the project and undertook significant liaison with affected land owners and lake-user groups. Various NIWA staff and visiting students assisted with the project at different times, particularly Joe Butterworth (routine sampling), John Nagels (sonde sampling), Jeremy Rutherford (site assistance), and Thomas Mann, Tim Wallis, Lex Brugger and Asaeli Tulagi.

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Appendix 1: Summary of FTW plant biomass and macro-nutrient content at the end of the trial.

	Cyperus ustulatus						***Carex virgata***					
	Above		Below		Total		Above		Below		Total	
	Low loading	High loading	Low loading	High loading	Low loading	High loading	Low loading	High loading	Low loading	High loading	Low loading	High loading
Shoot height / root length (cm)	95	96	61	53			68	76	67	70		
Dry wt (g m^{-2})	4824	5221	432	446	5256	5667	3989	5891	1688	2003	5677	7894
Nitrogen (g m^{-2})	38	38	5	5	43	43	38	63	12	14	50	77
%	*0.75*	*0.75*	*1.14*	*1.27*			*0.94*	*1.00*	*0.73*	*0.71*		
Phosphorus (g m^{-2})	3.1	3.2	0.5	0.5	3.6	3.6	2.5	4.0	0.9	1.0	3.4	5.0
%	*0.06*	*0.06*	*0.11*	*0.14*			*0.06*	*0.06*	*0.06*	*0.05*		
Potassium (g m^{-2})	81	104	3.1	6.5	84	111	58	76	25	29	83	105
%	*1.50*	*1.97*	*0.76*	*1.27*			*1.46*	*1.26*	*1.47*	*1.48*		
Sulphur (g m^{-2})	7.1	7.4	0.7	0.7	7.8	8.1	4.3	6.2	2.4	2.4	6.6	8.7
%	*0.14*	*0.14*	*0.17*	*0.17*			*0.10*	*0.10*	*0.14*	*0.12*		
Calcium (g m^{-2})	9.8	11.7	1.7	1.4	11.5	13.2	13.3	23.2	2.0	2.3	15.3	25.5
%	*0.24*	*0.25*	*0.43*	*0.42*			*0.37*	*0.42*	*0.14*	*0.13*		
Magnesium (g m^{-2})	4.3	4.5	0.9	0.9	5.1	5.4	15	17	2.9	3.2	17.7	20.3
%	*0.09*	*0.09*	*0.21*	*0.21*			*0.36*	*0.28*	*0.17*	*0.16*		
Sodium (g m^{-2})	17.4	12.96	1.05	1.6	18.5	14.6	1.00	0.57	5.1	4.5	5.5	5.1
%	*0.47*	*0.28*	*0.32*	*0.33*			*0.01*	*0.01*	*0.34*	*0.26*		