Aquatic Ecology of the Travis Wetland:

a reappraisal

Prepared for:
Christchurch City Council

AEL Report No. 76

Malcolm Main
Mark Taylor

Final Report

April 2010

Aquatic Ecology Ltd.
PO Box 3254
Christchurch 1
Telephone/ Facsimile (03) 366-4070
Email: mark@ael.org.nz

AEL
Aquatic Ecology Limited

Contents

1 Executive Summary

AEL undertook a survey of phytoplankton, invertebrates and fish at five sites within Travis Wetland on 27-28 February 2009. The purpose of this survey was to update previous resource surveys undertaken in 1996 and 1999. We also examined the available water quality data for the wetland, and in October 2009, collected dissolved oxygen concentration (D.O.) data.

The available water quality data are sparse, but they indicate that nutrient concentrations are often high, and highly variable throughout the wetland. The ammonia-nitrogen concentration was particularly elevated at one site, and nitrite-nitrogen at one other, although these observations were based on only one sample. Turbidity was high at most sites, and the lake had high pH.

Our dissolved oxygen measurements showed that the D.O. concentration, in October 2009, was also highly variable throughout the wetland, and concentrations at two sites were very low. It is likely that during the summer, when water temperatures are higher, those sites would at best be marginal for fish life, except perhaps for shortfin eels.

We found that the phytoplankton community had changed significantly since it was last sampled in 1996, which was prior to the lake excavation and elevation of the water level. These changes are probably not surprising, because of the significant modifications to the wetland since that time. In particular, the potentially toxic cyanobacterium *Anabaena,* which had been present during the previous surveys, was not recorded in 2009. However, at several sites another cyanobacterium, *Microcystis aerogenosa*, was present, which can also form toxic blooms. However, at the sites where we recorded it, *Microcystis* was not present in bloom proportions.

The invertebrate fauna collected in 2009 was slightly more diverse than that recorded in 1999, a year after the lake was excavated. To at least some extent, this is probably because the community was still developing in 1999. There was an indication that the invertebrate health had declined at one site (near the weir in Angela Stream). Of significance, was that we did not collect any mosquito larvae, which were a feature of the fauna at three of these sites in 1999 and in a quarter of the samples collected in 1996. They are probably limited by predation from the large population of shortfin eels in the wetland generally, and rudd in the lake.

Shortfin eels and rudd were the only two fish species that we collected. Only five rudd were caught by us (all from the lake), but the Department of Conservation has removed a large number of this pest fish from throughout the wetland. In contrast, our eel catch was substantial, especially at one of the sites. In addition to those two species, small numbers of inanga, common bully and a single smelt have been recorded by others from the wetland. However, the large number of predatory eels, poor access, and indications of seasonally low dissolved oxygen and high nitrite concentrations, probably prevent large populations of these fish developing in this habitat.

Poor water circulation throughout the wetland probably contributes to the low dissolved oxygen concentrations that we observed, and it could increase the probability of toxic cyanobacterial blooms. Weirs and other obstructions in the wetland that reduce water circulation through the wetland would also restrict access by migrating fish, other than eels.

We recommend that methods to improve water circulation be actively explored, and an initial meeting between ecologists, park rangers, and ecologists would be a good first step. Secondly, toxic bloom-forming algae are present in the wetland, and while we found no evidence of blooms taking place during our survey, conditions can become suitable in the warmer months, and sunny areas with poor water circulation need to be monitored for blooms. We raise the issue about the possible introduction of giant kokopu, partly as a biological control agent against pest fish, but these fish were likely to be present in the wetland, as they were in other swampy locations around Christchurch.

2 Historical Timeline Overview

The Travis Wetland is a partially restored fen wetland which has become an integral part of the natural landscape of Christchurch City. The wetland has an area of approximately 119 Ha, and was originally a wet sedge fen formed from an old (ca. 400 AD) estuary of the Avon River. Since then, it has undergone a series of ecological states, of which three distinct indigenous vegetation communities have been identified, induced by fire, deforestation, and drainage (McGlone 2009). A fen is characterised by having a neutral to slightly alkaline pH, with the water sourced from surface or groundwater flows, as opposed to a bog, which tends to be acidic and rainfall-fed.

By the latter half of the 20th century, the wetland had been extensively drained, grazed, and overrun with the exotic trees (esp. willow) and exotic grasses, and had become a target for the city's residential development. However, due to intense community pressure in the 1990s, the Christchurch City Council purchased the land from private developers in 1996. The purchase was based on the ecological potential of the wetland, and in part to its size, being the largest area of freshwater wetland in Christchurch City. The wetland is now classified as an Ecological Heritage Reserve in the City Plan. In 1998, modifications were made to the wetland, including the development of a small lake (Anderson, 1999).

The wetland is currently administered by the Christchurch City Council (CCC), with enduring management links with the Travis Wetland Trust, which provide assistance with weed control, native plantings, and other environmental enhancement and protection work.

3 Ecological Background

In the late 19th century, the wetland held value for Mãori for its mahinga kai, and at that time would have been a wetland fen. These values include tuna (eels) and other fish which would almost certainly have included inanga (adult whitebait – *Galaxias maculatus*), and possibly other galaxiid species such as giant kokopu (*Galaxias argenteus*).

Virtually nothing is known about the fish ecology in recent times. In the 1980s brief fish surveys were undertaken by MAF Fisheries Research Division in 1984 and 1988 in response to organisations concerned about development plans for the wetland, but only shortfin eels (*Anguilla australis*) were found (Eldon & Kelly 1992; NZFFDB 2009).

In 1996, an inventory of aquatic life – algae, invertebrates, fish - was prepared by NIWA (National Institute of Water and Atmospheric Research) on the more northern part of the area (Sagar *et al.* 1996). This report was commissioned by the CCC to assess both the actual and potential ecological importance of the northern part of the wetland. At the time, the report was limited to CCC-owned land in the northern part of the wetland, with the southern area (along Travis Road - now QEII Drive) still owned by Travis Country Estates Ltd. This inventory included the identification of an algal species which could form nuisance toxic blooms (*Anabaena*), as well as several other potential nuisance, but non-toxic species (*Spirogyra*, *Euglena*, and *Oedogonium*). A total of 24 aquatic invertebrate taxa were identified, all typical of wetland environments, and two freshwater fish species; the shortfin eel, which was moderately abundant, and several (4) inanga. The inanga, all adults approaching spawning condition, were recorded only from one central location in the wetland, in the vicinity of where the lake has since been excavated.

In summary, the 1996 inventory found little in the aquatic fauna which would rate the wetland, in its 1996 state, as ecologically important. However, it did indicate the potential of the wetland, which was relevant because of the Council's intention to purchase the wetland in its entirety. Cattle exclusion was considered important to reduce the risk of algal blooms by reducing the importation of nutrients into the wetland, with the risk to be reduced further by an increase in water depth, water circulation, and shading. These same three attributes were considered important to enhance the wetland's value as a habitat for freshwater fish, but with the added benefit of overhanging vegetation providing a source of invertebrate food for resident fish. It was considered at the time that it was possible some refuges could be made for Canterbury mudfish, an endangered species. However, the large population of predatory shortfin eels would have to be excluded from these habitats, and they would require ongoing netting to remove eels.

Another biological report was prepared with an emphasis on primary production and macroinvertebrates (Anderson 1999). The principal objectives of that report were to obtain baseline (pre-development) data on macroinvertebrate distribution, diversity and abundance, and to evaluate the variation of primary production (chlorophyll *a*) across the wetland. These data were obtained from unmodified aquatic habitats, and those which had been recently constructed at the time (e.g. the central pond was constructed in March 1998).

Like other waterbodies in the Christchurch area, the Travis Wetland has become a target for the illegal introduction of exotic fish, presumably for the purpose of establishing recreational fisheries. In recent years, a population of rudd, a pest fish, has been established in the central pond. This population is regularly culled through netting by the Department of Conservation (DoC) (DoC, Helen McCaughan, pers. comm.), but with future control possibly undertaken by the Christchurch City Council (John Skilton, CCC, pers. comm.).

4 Objectives

While effort has been expended on monitoring the developing flora and avifauna (birdlife), relatively little work has been spent on monitoring the indigenous fish life in the wetland. Some survey work has been undertaken on the potential of introducing mudfish into the wetland, but much of the fisheries work has been centred on culling what is now a significant population of rudd in the central pond. Incidental to that process, some native fishes were identified, but little information has been gathered on the spatial distribution of fishes in the wetland, or how the wetland fish community has changed with the restoration process. There appears to be no fish monitoring program or monitoring of values associated with primary or secondary production i.e. algae, or aquatic macroinvertebrates, although some invertebrate data are available from the time the wetland was first established (Anderson 1999). These organisms underpin the ecological processes in the wetland and form part of the fish diet, while the fish and invertebrates are likely to form an important food component for the bird life.

Moreover, since the wetland restoration, there was little information on the presence of potentially toxic or bloom-forming algae, and the current status of those species identified in the NIWA report (Sagar *et al.* 1996). Therefore, the purpose of this study was to at least partially redress this lack of information; specifically to:

1. Sample and identify algae (phytoplankton) from five sites in the wetland, and comment about their conservation status, or their potential to cause nuisance algal blooms or toxicity problems.

2. Sample and indentify macroinvertebrates from each of the same five sites, compute relevant health metrics, and compare the communities with those from the two studies undertaken in the 1990s (Anderson 1999; Sagar *et al.* 1996).

3. Re-survey some of the wetland fishing sites first fished in 1996 (Sagar *et al.* 1996), using similar gear, and at a similar time of year (February), and compare the fish community composition and condition.

5 Methods

5.1 Survey sites

The locations of the five surveyed sites and principal water features are shown in Figure 1, and the sites are illustrated in Appendix I. Sampling sites were chosen on the basis of re-surveying some of the sites from the 1996, and 1999 studies, but also those that were of interest to the Travis Wetland Trust (as outlined at a meeting on 18/11/08). The total number of sites was limited due to budget constraints, but the sites selected and the rationale for choosing them are presented below (Table 1).

Table 1. The survey sites, survey history, and re-survey rationale.

Site	Surveyed 1996 (NIWA; fish, algae invertebrates)	Surveyed 1999 (Anderson; Chlorophyll *a*, invertebrates)	Rationale
Control Site	Yes (fish only)	Yes	Consistency with 1996 and 1999 survey. Habitat still relatively unchanged since.
Lake Site	Yes (their site f4 approximately, but lake not excavated at the time)	Yes	Consistency with 1996 and 1999 surveys.
The Willows (Travis Stream)	No	Yes	Consistency with 1999 survey; also requested by Travis Wetland Trust.
The Open Site (Travis Stream)	Yes (fish; their site f5 approximately, but habitat changed)	Yes	Consistency with 1996 and 1999 surveys.
The Weir Site	No	Yes	Consistency with 1999 survey.

These five sites were diverse in physical form, and well dispersed around the wetland (Fig. 1). Two sites fished in 1996 were not re-surveyed. One site was the willow-choked channel along the north boundary of the wetland, which still exists (f1, Fig. 1 in Sagar *et al.* 1996). The second site that was a central drain, which was excavated in 1998 to form the northern section of the central pond, and has therefore disappeared.

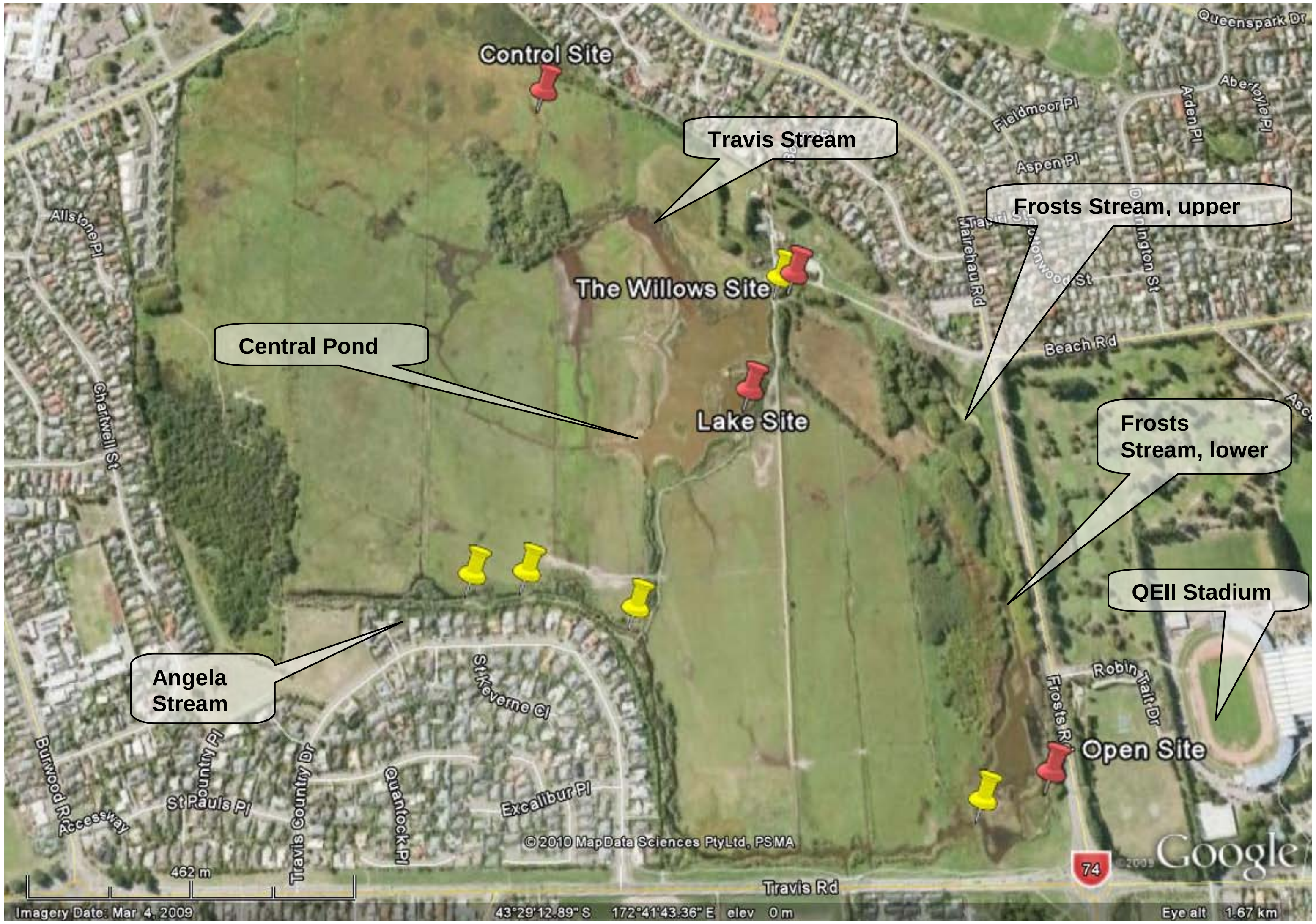

Figure 1. The five (red pins) principal sampling locations (Feb. 2009) (Aerial March 2009), and principal water features of Travis Wetland. Also noted are the location of five weirs in the wetland (yellow pins) (see section 7.5). The Lake Site was also the site for the Schott datalogger (water temperature and water conductivity).

5.2 Field methods

5.2.1 Dissolved Oxygen

We realised that dissolved oxygen (D.O.) concentrations might be useful to help interpret our results, although we did not propose to collect DO data during the February fieldwork. Therefore, dissolved oxygen concentrations were measured on 7 and 8 October 2009, using a WTW Oxi315i meter, calibrated immediately prior to use.

5.2.2 Data logger

A Schott datalogger was installed in the lake at the Lake Site from 27/2/10 (11:30 am), through to 28/3/2010 (7 pm). The sensor was fixed to a Waratah® at approximately mid-depth. This recorded water temperature and water conductivity at hourly intervals.

5.2.3 Phytoplankton sampling

Phytoplankton samples were obtained from the open water at each of the five sites in early March 2009. An approximately 300 ml sample was obtained from the mid-water from each of the five habitats, and allowed to settle for one or two days. Water samples were then removed by pipette, and identified by microscope (400x) to the lowest taxonomic level. Six slides were prepared for each site and scanned. Identification was made using keys (Prescott 1954; Pridmore *et al.* 1982), the NIWA on-line guide to the common genera of freshwater diatoms in New Zealand, and other on-line resources.

5.2.4 Macroinvertebrate collection

Samples were obtained on the 2-3/3/09, and sampling methods followed the guidelines and protocols established for sampling macroinvertebrates in soft-substrate habitats (Stark *et al.* 2001). In summary, invertebrates were sampled with a standard kicknet which was swept through aquatic vegetation, emergent rushes, and over woody debris. Where wood detritus was present, it was rinsed down with buckets of water through a series of 'Endecott' sieve plates (5 mm, to 500 microns). Invertebrate samples were field-preserved in isopropyl alcohol, and subjected to microscopic examination in the laboratory.

Versions of the macroinvertebrate community index (MCI-sb) and quantitative macroinvertebrate community index (QMCI-sb), were used to assess organic pollution impacts in soft-bottomed streams (Stark & Maxted 2007). These indices are based on tolerance scores of taxa that occur in soft-bottomed streams. That is, habitats with beds dominated by fine sediments, woody debris, and macrophytes. These indices rate the sites on the basis of subjective pollution tolerance scores for each taxon, ranked from 1 (most tolerant) to 10 (least tolerant). The MCI-sb is calculated by summing the taxon scores to obtain a site score and dividing by the number of taxa, then multiplying by 20, i.e.

$$MCI = \frac{site\ score}{no.of\ scoring\ taxa}\ X\ 20$$

An MCI-sb value ranges from 0 (when no taxa are present) to 200 (when all taxa score 10 points each). The QMCI-sb is derived by multiplying the number of a taxa present by its taxon score, summing them, and dividing by the total abundance. The QMCI-sb ranges from 0 to 10.

5.2.5 Fish sampling

The sampling occurred over 27-28 February, 2009. Fish survey work was undertaken with a fleet of fyke nets, and Gee minnow traps, all of which were baited with salmon feed pellets. Fyke nets were of standard design and construction (Fig. 2), but were of two sizes. The smaller nets had a wing of 2.1 m in length, and a hoop size of 0.45 m. The larger nets had a wing 3.3 m long, and a hoop size of 0.60 m. The stretched-mesh size of all of the nets was approximately 12 mm. These nets function by guiding fish along the wing into the tubular section of the net where they are prevented from swimming out by two non-return valves. Gee Minnow traps are another passive fishing device which superficially resemble a Mãori hinaki, and have a wire-mesh size aperture of approximately 6 mm.

All gear was set in the early evening and removed the following day as in the schedule outlined below (Table 2).

Table 2. Fishing effort schedule for the Travis Wetland fish Survey (27-28/2/09).

Site	Net type	Set time (27/2/09)	Raise time (28/2/09)
Willows	small fyke	6:30 p.m.	9:27 a.m.
Willows	Gee Minnow line (3 traps)	6:35 p.m.	12:55 p.m.
Willows	small fyke	6:35 p.m.	12:10 p.m.
Control	Gee Minnow line (3 traps)	6:05 p.m.	3:00 p.m.
Control	small fyke	6:05 p.m.	3:00 p.m.
Control	small fyke	6:05 p.m.	3:00 p.m.
Lake	large fyke	6:55 p.m.	1:20 p.m.
Lake	large fyke	6:58 p.m.	1:45 p.m.
Lake	Gee Minnow line (3 traps)	6:58 p.m.	1:35 p.m.
Open	Gee Minnow line (3 traps)	7:16 p.m.	9:05 a.m.
Weir	large fyke	7:45 p.m.	7:45 a.m.
Weir	Gee Minnow line (2 traps)	7:45 p.m.	8:40 a.m.
Weir	large fyke	7:45 p.m.	7:45 a.m.

Captured fish were removed from the fishing gear, anaesthetised to reduce stress, and then identified. The five rudd collected were measured. The original intention had been to measure all of the eels as well. However, the number of eels caught from two sites (the lake and the Willows) was too great to allow that, so a sub-sample of 380 individuals was measured to the nearest 2 mm, weighed, and the total numbers from those two sites were then estimated gravimetrically. That is, the total weight of eels was divided by the weight of a known number of eels to calculate the total numbers. At that stage, it was clear that the entirety of the eel catch was composed of the one species. After recovery from anaesthesia, all fish except the rudd were returned to their habitat.

Figure 2. A large fyke net (background), and a small fyke net (foreground). The nets incorporate two non-return valves.

6 Results

6.1 Water quality

The concentrations of dissolved oxygen measured on 7 and 8 October 2009 varied greatly from Site to Site (Table 3). The highest concentration (8.15 mgL^{-1}) was measured from the control site at 9.45 am on 7 October, and the lowest was only 2.8 mgL^{-1} measured at the Willows at 6.08 am on 8 October. The concentrations at the weir were also very low; only 3.7 – 3.9 mgL^{-1}. Dissolved oxygen concentrations tend to exhibit a diurnal variation, with lowest concentrations soon after dawn and the highest just after dusk, and these measurements were made soon after daybreak, when the concentrations would have been expected to be at their lowest.

Dissolved oxygen concentrations were also measured in the middle of the day on 29 January 2010 at two sites in the Willows (immediately above the bridge and 20 m upstream) and in the lake (green-shaded in Table 3). These summertime concentrations were higher than expected, given the higher water temperatures.

Table 3. Dissolved oxygen concentrations measured from five sites on 7 and 8 October 2009, and 29 January 2010. Green-shaded rows indicate values obtained in the summer.

Site	Date	DO (mg/L)	Temp (^{o}C)	Time	Comments
Angela Stream	7/10/09	3.70	10.6	8:24 am	Taken just u/s of low weir. Depth (> 1m)
Angela Stream	8/10/09	3.90	11.5	5:45 am	
The Open	7/10/09	6.10	9.0	8:47 am	Water 10cm deep overlying deep sludge
The Open	8/10/09	4.10	10.0	5:58 am	
The Willows	7/10/09	2.95	9.3	8:58 am	Taken where fykes set just d/s of path. Depth 0.5m. 2nd reading obtained 100 m further downstream, 20 cm depth, 2.72 mg/L

The Willows	8/10/09	2.80	10.6	6:08 am	
The Lake	7/10/09	7.08	10.4	9:10 am	Taken where fykes set. Depth 0.6m, southerly wind fetch approx. 70m
The Lake	8/10/09	5.78	11.2	6:16 am	
The Control	7/10/09	8.15	9.3	9:45 am	Turbid (geese and paradise duck activity), also choked with macrophytes (*Potamogeton crispus)*
The Control	8/10/09	4.86	10.2	6:30 am	
The Control, replicate #1	8/10/09	7.40	10.2	6:30 am	
The Control, replicate #2	8/10/09	7.29	10.2	6:30 am	
The Control, replicate #3	8/10/09	7.50	10.2	6:30 am	
The Willows replicate #1	29/1/10	9.8	21.3	12.25 pm	
The Willows replicate #2	29/1/10	10.6	21.2	12.30 pm	
The Lake	29/1/10	9.8	22.0	12.50 pm	

6.1.1 Logger Data

During late summer/early autumn water temperatures in the lake typically peaked in mid to late afternoon (3-5 pm), with a maximum on the 27 February (25.4 ^{o}C at 4:30 pm NZDT) (Appendix II, Fig a). Diurnal minima usually occurred between 6:30am and 8:30 am. A strong southerly event on the morning of the 11 March led to depressed water temperatures, leading to a monitoring period minimum of 13.3 ^{o}C at 8am-10 am on the 13 March.

Concurrent water conductivity measurements revealed sudden rises in conductivity which approximately coincided with spring-tide sequences occurring of the full moon (Appendix II, Fig. b). Specifically, after the spring-tide at the beginning of the month (1-3 March), conductivity in the lake increased sharply on the evening of the 4 March, and peaking around 10 March. Water conductivity slowly declined before increasing again around the time of the full tide on 28-30 March.

6.2 Phytoplankton

The phytoplankton assemblage that we recorded also varied amongst the sites, and is listed in Table 4. To the best of our knowledge, none of the phytoplankton taxa are of any conservation significance.

Table 4. Phytoplankton recorded from sites in Travis Wetland (× = present, ×× = abundant, ××× = dominant).

Class	Genera or species	Weir (Angela Stm)	Open	Control (spring)	Willows	Lake
Chlorophyta	*Cymbella*	×				
(green algae)	*Chlorococcum*		××	×××	××	××
	Chlorella		×			
	Cocconeis		×			
Bacillariophyta	*Navicula*		×			×
(diatoms)	*Fragilaria*		×		×	×
	Scenedesmus		×			
	Nitzchia				×	×
	Gomphonema				×	
	Ankistrodesmus					×
Dinophyta	*Gymnodinium*		×			×
(dinoflagellates)	*Peridinium*		×			×
Cyanobacteria	*Microcystis*		×××		×××	×××
(blue-green bacteria)	*aerogenosa*					
Euglenophyta	*Trachelomonas*			×	××	
(euglenoids)	*Euglena*				××	××
	Phacus				×	
	Lepocinclis					×
Chrysophyta (golden algae)	*Derepyxis*?				×	

Phytoplankton were uncommon in the sample from the site at the Angela Stream weir, which is not surprising, since it is a flowing-water site. Only low numbers of the chlorophyte *Cymbella* were recorded there. The control (spring) site also had only a few species present. It was dominated by the chlorophyte *Chlorococcum*, with the euglenophyte *Trachelomonas* also present. The remaining three sites all had greater species assemblages, but all three were dominated by the cyanobacterium *Microcystis aerogenosa*. *Chlorococcum* and the euglenophyte *Euglena* were also abundant at these sites.

There was a considerable difference in the composition of this phytoplankton community compared with that recorded by NIWA in 1996, before the wetland was developed. They did not supply a detailed breakdown, but (Sagar *et al.* 1996) noted that the chlorophyte *Spirogyra* was obvious to the eye then, whereas it was not recorded during this survey. In addition, a species of the bloom-forming cyanobacterium *Anabaena* was present during their survey, but it was not recorded during this survey; its place taken by another cyanobacterium, *Microcystis aerogenosa*. These changes in community composition are probably due to the physical changes that have occurred in the wetland between surveys.

6.3 Aquatic macroinvertebrates

Table 5 records the aquatic invertebrates collected from sites in Travis Wetland during this study. Invertebrates were recorded by Sagar *et al.* (1996) and then by Anderson in 1999. In both of those cases, the invertebrate fauna was considerably different than that recorded here. The previous fauna lists were less diverse. In particular, they did not record leeches, muscids, mites, amphipods, the damselfly *Austrolestes* or any Trichoptera (caddises), all of which were recorded in our survey. In contrast though, Sagar *et al.* (1996) did record *Culex* mosquito larvae, the tipulid (cranefly) *Zelandotipula*, and a diving beetle, *Rhantus pulverosus*, none of which were recorded during this survey. Anderson (1999) also recorded the mosquito larvae, as well as Sciritidae, Tanyderidae and Tanypodidae. As for algae, these differences in community composition recorded between their sampling and ours were probably associated with the physical changes to the habitat that have occurred between times.

The open site had the highest MCI-sb, and the Angela Stream site the lowest. This was caused by the presence at the open site of low numbers of high-scoring taxa. In contrast, Angela Stream had the highest QMCI-sb, and this anomaly is because the QMCI-sb is the better measure of invertebrate habitat quality, as takes into account the numbers of scoring taxa.

Sagar *et al.* (1996), aggregated the invertebrate data for all of the sites, so it was not possible to compare their individual site taxa lists with our data. Nevertheless, we calculated an overall MCI-sb using their data, but a QMCI-sb could not be calculated, because the calculation of this measure requires taxa abundances, which were not provided in the 1996 report. The calculated MCI-sb using the 1996 taxa was 58.2, which is similar to that from our open site, and slightly higher than that of the other four. MCI-sb indices were also calculated from Anderson's (1999) data. They ranged from 51.9 to 56.7, so they too were in a similar, though slightly narrower, range to that from our samples. One difference though, was that the site which scored the highest MCI-sb (56.7) on the basis of his data was the weir site, which scored the lowest on ours (48.38).

6.4 Fish Community composition

We collected two species of fish; shortfin eel (*Anguilla australis*) and rudd (*Scardinius erythrophthalmus*). Five rudd was captured, all of which were caught in the two fyke nets set in the lake. In contrast, the shortfin eel catch was large, especially at the Willows site (Figure 3; Table 6). The estimated total eel catch was 969, ranging in length from 264 to 795 mm T.L. (mean length 432 mm T.L.). A sub-sample of 380 eels, as listed in Table 6, was measured, with the length frequency distribution provided below (Fig. 4).

Figure 3. Lifting one of the fyke nets from the willows, showing the cod-end packed with shortfin eels. The second net had a similarly large catch.

Table 5. Aquatic invertebrate taxa, expressed as the number recorded and the percentage of the total abundance, from five sites in Travis Wetland. Locations of sampling sites may be found on Fig. 1. Predominant species are in red text.

Taxa groups	Family/genus/species	The Lake		The Open		The Control		The Willows		Travis Weir	
		No.	%	No.	%	No.	%	No.	%	No.	%
NEMATODA		0	0	136	0.73	0	0	0	0	8	0.12
PLATYHELMINTHES		0	0	8	0.04	0	0	20	0.16	0	0
HYDROZOA											
	Hydora	0	0	0	0	0	0	8	0.06	0	0
MOLLUSCA											
Gastropoda	*Gyraulis corinna*	103	1.00	40	0.21	178	16.70	128	1.00	24	0.35
	Physa acuta	106	1.03	76	0.41	189	17.73	156	1.22	80	1.16
	Potamopyrgus antipodarum	85	0.82	100	0.54	241	22.61	4	0.03	32	0.47
Bivalvia	Sphaeriidae	4	0.04	0	0	0	0	0	0	0	0
OLIGOCHAETA		0	0	236	1.26	2	0.19	136	1.07	1104	16.05
HIRUDINEA		0	0	4	0.02	0	0	4	0.03	0	0
CRUSTACEA											
Copepoda	Cycopoidea	239	2.31	1976	10.58	24	2.25	648	0	560	8.14
Cladocera	*Daphnia*	0	0	60	0.32	0	0	20	0.16	0	0
	Chydoridae	0	0	0	0	0	0	0	0	304	4.42
	Simocephalus	0	0	0	0	0	0	0	0	1304	18.95
Ostracoda		7750	75.06	14816	79.33	216	20.26	10864	85.25	3048	44.30
Amphipoda	*Paracalliope fluviatilis*	0	0	1	0.01	0	0	0	0	0	0
ARACNIDA	Acarina	30	0.29	16	0.09	19	1.78	24	0.19	96	1.40
INSECTA											
Diptera											
(flies)	Ceratopogonidae	28	0.27	32	0.17	0	0	28	0.22	0	0
	Chironominae	0	0	364	1.95	0	0	72	0.56	144	2.09
	Chironomus zealandicus	82	0.79	128	0.69	0	0	192	1.51	24	0.35
	Orthocladinae	53	0.51	84	0.45	26	2.44	100	0.78	8	0.12
	Scatella	1	0.01	0	0	0	0	0	0	0	0
	Ephydrella	0	0	32	0.17	0	0	0	0	0	0
	Muscidae	1	0.01	0	0	0	0	0	0	0	0
	Pyschodidae	0	0	80	0.43	0	0	0	0	0	0
Hemiptera											
(bugs)	*Anisops*	16	0.15	16	0.09	14	1.31	32	0.25	0	0
	Sigara arguta	1558	15.09	32	0.17	62	5.82	148	1.16	8	0.12
	Microvelia	259	2.51	348	1.86	59	5.53	124	0.97	0	0
Odonata											
(damselflies)	*Austrolestes colensonis*	0	0	4	0.02	0	0	0	0	0	0
	Xanthocnemis	2	0.02	28	0.15	0	0	4	0.03	0	0
	Zygoptera	0	0	32	0.17	10	0.94	16	0.13	112	1.63
Trichoptera											
(caddises)	Hydroptilidae	0	0	0	0	25	2.35	0	0	24	0.35
	Triplectides cephalotes	0	0	0	0	0	0	12	0.09	0	0
	Triplectides obsoletus	0	0	8	0.04	0	0	0	0	0	0
Coleoptera											
(beetles)	*Enochrus* (sp. A)	0	0	0	0	1	0.09	0	0	0	0
	Total taxa		17		26		14		22		16
	MCI-sb		54.59		58.85		52.00		55.73		48.38
	QMCI-sb		2.08		2.11		1.89		3.1		3.35

Table 6. Eel catch statistics for five sites in Travis Wetland. The numbers based on gravimetric measurements are shown in red.

Site	Number caught or estimated	Min length (mm)	Max length (mm)	Mean length (mm)
control	13	296	512	363.31
lake	170	291*	790*	449.78*
open site	19	264	524	373.58
weir	130	294	651	418.14
willows	637	304**	795**	450.26**
Totals	**969**	**264*** **	**795*** **	**432.15*** **

*Based on measurements of 85 eels.

**Based on measurements of 133 eels, ***Based on measurements of 380 eels.

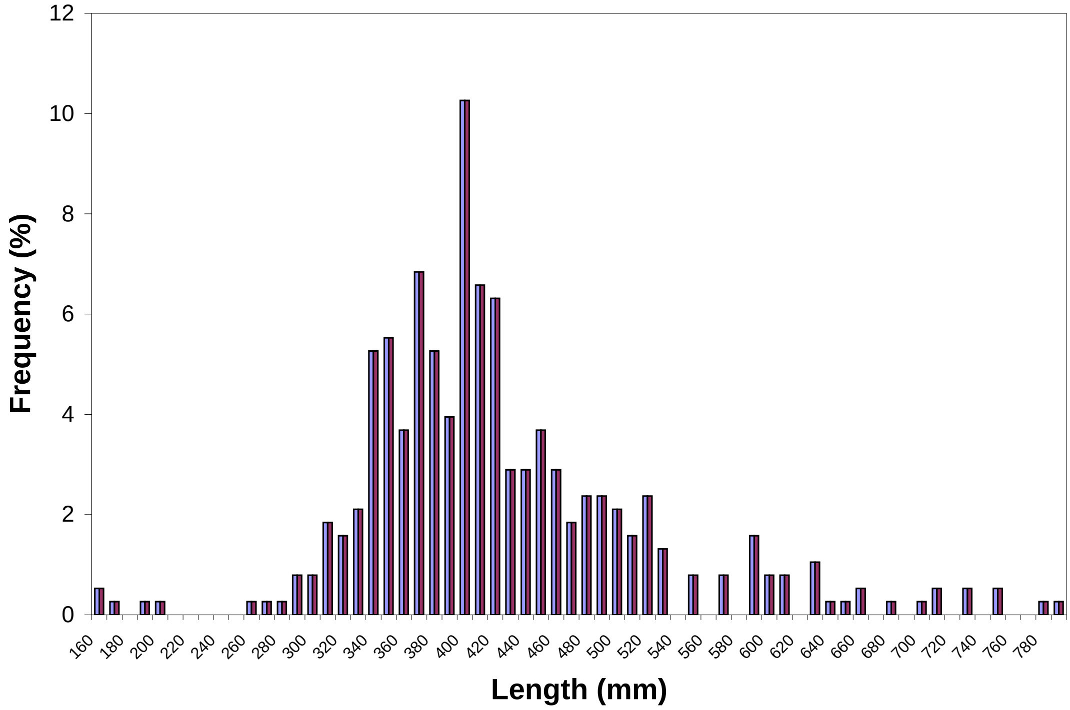

Figure 4. Length frequency of 380 shortfin eels measured from Travis Wetland.

The five rudd caught were all from the lake, and were all of a relatively uniform size (161-208 mm, mean length = 180.8 mm), presumably from the same cohort. DoC, using gill nets, has also recorded this species from the Open Site, Angela Stream, and a pond near the Information Centre. The uniformity in the size of the rudd is unusual; previous samples from other sites where rudd are present, for example at the Kaiapoi lakes, contained large numbers of small individuals.

7 Discussion

7.1 Water Quality

Water quality did not form part of the brief for this study, but we realised that water quality data could help to interpret the results of our survey. To this end, we examined the limited data available from other sources, and belatedly collected data for dissolved oxygen concentration and temperature, which were otherwise unavailable.

A limited set of nutrient data (one sample from three sites) was presented by Sagar *et al.* (1996). It showed that the concentrations of dissolved reactive phosphorus and ammonia-nitrogen were highly variable across the three sites (0.069-0.237 and 0.018-0.45 mgL^{-1}, respectively), whereas nitrate-nitrogen concentrations were similar to one another at the three sites, and very low (0.003-0.006 mgL^{-1}).

In March-April 2009, Travis Wetland Trust collected a set of water quality samples from the same five sites that we sampled (Travis Wetland Trust, raw data. 2009; Table 7).

Table 7. Travis Wetland Trust water quality data collected in March-April 2009

	pH	Turbidity (NTU)	Conductivity (μs cm^{-1})	Ammonia-N (mg L^{-1})	Nitrate-N (mg L^{-1})	Nitrite-N (mg L^{-1})	Dissolved reactive phosphorus (mg L^{-1})	*E. coli* (MPN 100 mL^{-1})
Control (3 April)	7.5	680	1410	2.2	0.01	<0.01	0.097	>24,000
Lake (3 April)	8.5	140	730	<0.01	0.16	<0.01	0.049	2,000
Open (3 April)	7.7	1600	1280	0.034	0.06	<0.01	0.084	780
Angela Stream (3 April)	7.4	5.3	200	0.049	0.25	0.014	0.033	520
Willows repl. 1 (11March)	7.7	110	778	0.054	<0.05	0.42	0.14	No data
Willows repl. 2 (11March)	7.7	110	779	0.050	<0.05	0.43	0.14	No data
Willows repl. 3 (11March)	7.7	110	772	0.052	<0.05	0.42	0.15	No data

It is difficult to draw definitive conclusions from such a dataset with little seasonal range, but the results exhibited a large variation in water quality determinds between the sites, with both some markedly low and some extremely high concentrations. For example, the ammonia-nitrogen concentration at the control site was extremely high (2.2 mgL^{-1}), but at the lake site this determinand was below the level of detection, and at the remaining sites it was similar to the range recorded by NIWA in 1996. The concentration at the control site was well above the normal range for lowland waterbodies in Canterbury recorded by (Meredith & Hayward 2002), and could be borderline for fish life, especially if the pH was higher (ammonia toxicity increases sharply at high pH). It was also well in excess of the current water quality guideline for ammonia (ANZECC 2000). The *E coli* concentration at the control site was also very high (Travis Wetland Trust, raw data). During our site visits we observed a number of waterfowl on this small waterbody. It has been shown that waterfowl can elevate indicator bacteria and nutrient concentrations on small waterbodies with limited circulation (Main 2001), so the high numbers of birds compared with the small size of the waterbody may be the cause of the poor water quality recorded at the control site.

The pH at most sites indicated alkaline waters, and these were within the range typically recorded from freshwaters in Canterbury (7.4-7.7, Travis Wetland Trust, raw data), and also were typical for a fen wetland. However, the pH in the lake was anomalously high (8.5).

Another interesting aspect of the water chemistry was the very high nitrite-nitrogen concentrations recorded from the Willows site on Travis Stream (0.47-0.48 mgL^{-1}). These are far higher than the nitrate-nitrogen concentrations recorded from the same site. Ordinarily, the reverse is the case in surface waters, because nitrite is rapidly oxidised to nitrate. However, this situation probably prevailed because the dissolved oxygen concentration at the site was very low, and under such reducing conditions, nitrate is instead reduced to nitrite. Nitrite is considerably more toxic to fish than nitrate. Its toxicity varies greatly with the chloride concentration, but in fresh water, rainbow trout (*Oncorhynchus mykiss*), were found to have a 96hr LC_{50} (50% mortality) of only 0.24 mg L^{-1} (Russo *et al.* 1974), or half the concentration measured from the Willows. Of more relevance it that the European eel (*Anguilla anguilla*) elvers can tolerate concentrations as high as 84 mgL^{-1} in fresh water (Saroglia *et al.* 1981), and shortfin eels would be equally-tolerant, so they would not be affected by the concentrations recorded there. The toxicity of nitrite to inanga, common bully, and other native fishes is unknown, but it is likely to be more similar to that of rainbow trout, and the high concentration of nitrite recorded there is likely to preclude them from this site.

Turbidity was low in Angela Stream (5.3 NTU), high in the lake, the Willows and the control site (110-680 NTU), and very high in the open site (1600 NTU; Travis Wetland Trust, raw data). The dissolved reactive phosphorus concentrations recorded by Travis Wetland Trust were in a similar range to those recorded by Sagar *et al.* in 1996. The nitrate-plus-nitrite nitrogen concentrations (0.06-0.47 mgL^{-1}) were relatively low, considering that groundwater, which tends to have high nitrate concentrations, is the source of water in the wetland.

We made several dissolved oxygen (D.O.) measurements in the spring (October 2009) with a view to aiding the interpretation of our results. These showed that at two sites (Travis Stream at the Willows, and Angela Stream at the weir), there was potential for D.O. to be a factor limiting the distribution of some fish species. This problem could be expected to be exacerbated in the summertime, when higher water temperatures could reduce D.O. levels lower than those that we recorded in the spring. More specifically, the springtime water temperature was only 10.6 ^{o}C at the Willows at 6.08 am on 8 October, when a low of 2.8 mg L^{-1} D.O. was measured. However, waters are much warmer during the summer, and the oxygen-carrying capacity would be expected to be lower. Shortfin eels have been known to survive when the D.O. concentration was less than 1.0 mgL^{-1} (Morgan & Graynoth 1978), but other species such as inanga are likely to be affected by such low levels. Experimental low-D.O. challenge tests indicated that at 1.0 mgL^{-1}, inanga survival declined to 39% after 48 hrs, whereas juvenile common bully did not survive exposure after just 4 hours (Dean & Richardson 1999). The large spatial variation in D.O. within the wetland is probably influenced, to least at some extent, by what we suspect is poor water circulation. In the summer, when the lake waters were discharging to Frost Stream, we found that the D.O. concentrations that we measured at the Willows site in the summertime (9.8 and 10.6 mgL^{-1}), were considerably higher than in the spring. We suspect that this was caused by a significant quantity of wind-mixed lake water entering Frosts Stream, and elevating the D.O.

The water temperature logger data over the late summer period we suspect is somewhat atypical, in that temperatures this high would normally be expected to occur in January, rather than late February as in 2010. However, New Zealand climate is quite variable, and while the maximum temperature is warm, it is not exceptional for an essentially enclosed sun-exposed water body.

The water conductivity data reveals sharp rises in the Lake's water conductivity coinciding approximately with the spring tide event, and then a gradual decline. This 'saw-tooth' pattern in conductivity (App. Fig. b) could be explained when the more conductive spring–tide water overtops the dam boards at the Lake Weir (Fig. 1) and enters the Lake. The lag between the conductivity rise and the spring tide sequences is probably due to a range of variables influencing the time lag between the logger and the entrance of tidal water from Travis Stream into the Lake. This would include tide height, and the speed at which the more conductive water arrives at the logger site. The time lag would explain why there was no sudden change in water temperature as the more conductive water arrived at the logger, as there had been adequate time for it to equilibrate with the surrounding water. The subsequent slow decline in conductivity over the weeks between spring tides may be caused by the ingress of lower-conductivity groundwater into the Lake, and possibly rainfall, along with the lack of normal tidal water entering the lake. Water circulation is discussed in the following section.

7.2 Wetland management issues regarding phytoplankton

There are no known conservation issues relating to the algal assemblage that we recorded, although there are potential nuisance issues. The toxic cyanobacterium *Anabaena,* which was recorded by Sagar et al. (1996) was not present during our sampling, but its place was taken by *Microcystis aerogenosa*, which was the dominant species recorded at several sites. *Microcystis* can form very dense populations, which are known as blooms. It typically thrives in warm, turbid, and slow-moving waters. *Microcystis* (and other cyanobacteria) tend to dominate the phytoplankton when the nitrogen:phosphorus ratio (N:P ratio) is less than 15 parts nitrogen (N) to 1 part phosphorus (P). These conditions give them a competitive advantage, because unlike other phytoplankton, they can fix atmospheric nitrogen. The data supplied by Travis Wetland Trust showed that this condition existed at all of the sites except at the Angela Stream weir. When the N:P ratio is greater than 20:1, other, non-nitrogen-fixing phytoplankton tend to predominate. As with other cyanobacteria, *Microcystis* blooms do not always produce toxins, but the specific trigger for the production of the hepatotoxin microcystin is not known. Our survey did not show that *Microcystis* was blooming as such, although it was the dominant phytoplankton species at three sites.

People swimming in dense *Microcystis* blooms have experienced irritations such as skin rashes, burns, and blistering of the mouth. Ingestion or inhalation of water containing dense bloom material may cause vomiting, nausea, headaches, diarrhoea, pneumonia, and fever. Ingestion of significant levels of the toxin microcystin can cause liver damage and dysfunction in humans and animals. No deaths from ingestion of microcystin have been reported in humans, but dogs, wildlife and livestock have died following exposure to this toxin (California Office of Environmental Health Hazard Assessment). Overseas, fish kills sometimes occur during bloom conditions because the decomposition of dead cells causes rapid deoxygenation of the water.

People are unlikely to swim at the site shown in the aerial photo, and dogs are excluded from the wetland, but fieldworkers would have to take care, although it would appear unlikely that they would accidently ingest water from the wetland. Perhaps of more concern is that *Microcystis* blooms have been associated with the deaths of waterfowl, especially diving ducks. For example, Matsunaga *et al*., (1999) found that microcystin was the probable cause of a mass death of spot-billed ducks in Japan.

It is possible that *Microcystis* will form blooms in larger areas of the wetland, in which case it could constitute a health risk, especially to waterfowl. There are no simple solutions to minimising the risk of cyanobacterial blooms. Given the available water quality data, it appears to be unrealistic to expect to reduce nutrient concentrations low enough such that they would be limiting. However, better water circulation would help to reduce the risk of blooms forming, since they usually only form in still water. Improving the water circulation through the wetland might be an achievable goal, and some ideas regarding this are discussed in a later section.

7.3 Macroinvertebrate Issues

None of the invertebrate species recorded by us are known to have any conservation value. They are mostly species which are adapted to still-water habitats with a soft substrate, and many can survive relatively low dissolved oxygen concentrations. The conservation status of freshwater invertebrates in New Zealand was reviewed by Collier (Collier 1993), but because of the incomplete knowledge of the invertebrate fauna, he listed only species of mayfly, stonefly and caddis which have restricted distributions. The three taxa of caddisfly that we recorded were not included in the list.

The invertebrate fauna that we identified was similar to, but not identical with, that found by Anderson in October 1999, a year after the wetland was modified (Anderson 1999). Exceptions were that he did not record *Physa*, *Daphnia*, *Austrolestes*, *Enochrus*, *Microvelia*, ceratopogonids, ephydrids, or pychopodids. In turn, the 1999 fauna was also more diverse than that recorded by Sagar *et al*. (1996), although a more limited range of habitats existed in 1996 when they collected their samples. In contrast, Anderson collected Sciritidae, Tanyderidae, and Tanypodidae, and Sagar *et al.* (1996) collected *Zelandotipula* larvae and *Rhantus pulverosus,* none of which were present in our samples.

Overall, there seems to have been an increase in the invertebrate diversity since Anderson collected his samples from the lake, because the community would still have been developing when he sampled it. We calculated MCI-sb indices from Anderson's (1999) data. The site which had the highest MCI-sb (56.7) on the basis of his data was at the Angela Stream weir, but it scored the lowest on ours (48.38). It is possible that this was caused by a reduction in the water quality at this site between his sampling and ours, although we have no comparable water quality data to support that idea. However, there is some conflict here with our QMCI data, as this site scored the highest on the basis of that index. The D.O. concentrations that we measured there in October were certainly very low (3.7-3.9 mgL^{-1}), and so could have been limiting to some species. In particular, we did not record *Triplectides* (a caddis) or Tanypodidae, two relatively high-scoring taxa that he collected from the site. In contrast though, we did collect *Triplectides* from the Willows, which also had a low dissolved oxygen concentration.

Taylor and Bradshaw (2009) sampled aquatic macroinvertebrates from two sites in Mimimoto Lagoon, a small coastal lake between Leithfield Beach and Amberley Beach. The invertebrate fauna in that lagoon was similar to that in Travis Wetland, but there were small differences in community composition between their two sites. This resulted in one site having a far higher MCI-sb than the other, whereas the QMCI-sbs for their sites were slightly reversed. Both of the QMCI-sb index values for Mimimoto Lagoon were very similar to that for the Angela Stream weir site, and higher than those for the other four Travis Wetland sites.

No mosquito larvae were collected by us, although they were present in a quarter of the NIWA samples collected in 1996, while Anderson (1999) reported them from the open, willows, and control sites, but not from the lake. Their presence tends to be highly seasonal, although our sampling took place at a time when they should have been present. However, it is probable that their numbers are now limited by predation from the large numbers of fish in the wetland, especially at the willows site. Mosquito larvae are highly vulnerable to fish predation because they occupy the water column, make frequent trips to the water surface to breathe, and are poor swimmers. Rudd were not present in 1996 and 1999, but they would also eat mosquito larvae. Thus, mosquito larave usually inhabit temporary bodies of water which lack fish. We therefore expect there will be temporary mosquito populations in isolated ponds and puddles in the wetland precinct which we did not record.

7.4 Fish Issues

7.4.1 The present fish community

It is clear that the creation of the lake at Travis Wetland has created a significant habitat for shortfin eels. We captured almost a 1000 shortfin eels in 8 nets and 14 traps, compared with only 84 caught by Sagar *et al*. in 1996 using a similar fishing effort (7 nets and 15 traps), which indicates that the eel population is now far larger than it was in 1996.

Fishes have been sampled nine times from the wetland; In May 1984 and October 1988 (by MAF Fisheries Research), February 1996 (by NIWA - (Sagar *et al.* 1996), April and December 2008, and January, April and October 2009 (by DoC) and February 2009 (by AEL – this study). Eldon and Kelly (1992) noted that only shortfin eels were recorded during the 1984 and 1988 MAF survey, and that their distribution was patchy. In 1996, 84 eels were caught in 7 fyke nets and 15 traps by Sagar *et al*., giving an overall catch-per-unit-effort (CPUE) of 3.8. They also caught four inanga. Our eel catch, using a similar sampling effort, gave a significantly greater CPUE of 44.

Since April 2008, DoC has captured 917 rudd, ranging from 85-251 mm in fork length, from the Lake alone, as well as a few from Angela Stream and the open site. They also recorded 44 common bully (*Gobiomorphus cotidianus*) and inanga (*Galaxias maculatus*), a few shortfin eels, and a solitary smelt (unpublished data courtesy of Helen McCaughan, DoC). Their low eel catch was a reflection of the type of fishing gear they were using, which was directed towards catching rudd, and largely unsuitable for catching eels.

The rudd, which has been introduced illegally into the wetland or else entered it from the Avon River, is a noxious fish in Canterbury. This fish is considered to be undesirable because small rudd compete for food with native fish and (in some habitats) with small trout (Hicks 2001). When they reach about 150 mm, they start to eat macrophytes, and by the time they exceed 200 mm, aquatic plants predominate in their diet. It was not mentioned by Hicks (2001), but large rudd are also sometimes piscivorous (fish-eating) (McDowall 1990).

The herbivory of large rudd has been implicated in the elimination of palatable plants from some lakes, both in New Zealand and overseas, resulting in domination by planktonic algae and resulting increases in turbidity. Water quality results collected by Travis Wetland Trust in April 2009 showed that the turbidity at the open site was extremely high (1600 NTU). We collected no rudd at that site, but they have been caught there by DoC. Rudd may have contributed to the turbidity recorded from there, as would the high concentration of algal cells, since Anderson (1999) found that this site had a fairly high chlorophyll *a* concentration.

The data for the rudd collected by DoC indicate that the minimum size of these fish within the lake has increased over time (Fig. 5). This suggests that there has been no juvenile recruitment into the population recently; so either there was no spawning, or else there is size-selective predation occurring, with all of the small fish being eaten by the large predatory eels.

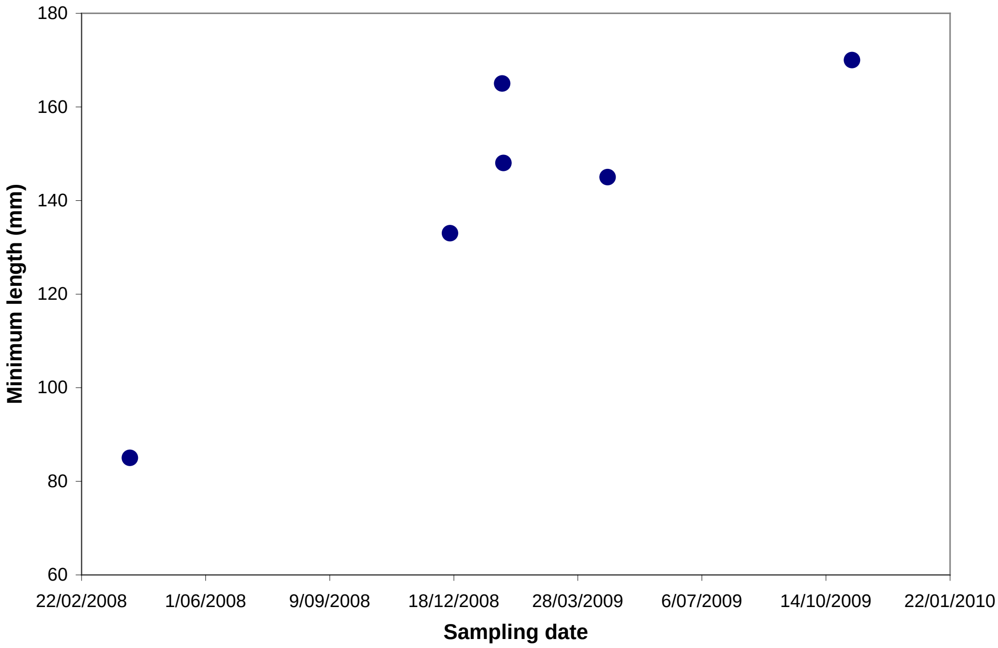

Figure 5. Minimum size (FL) of rudd collected by DoC from the lake between April 2008 and April 2009. Total sample size = 914.

During late summer and autumn, many eels migrate down-river to spawn in the sea. The migrants undergo distinct morphological changes; the eyes enlarge, the skin colour becomes silvery, the gonads develop, the gut shrinks, and they cease to feed (Todd 1980). Many of the large number of eels caught in the Willows area of Travis Stream appeared to be migrants (Fig. 6), probably because the survey timing coincided with the eel migration season. It is very likely that many of the captured eels were in the process of migrating either from the lake to the sea, or else they inadvertently entered the wetland whilst travelling from the Avon River to the sea. The length frequency distribution of a sample of 380 of the eels that we collected (Fig. 4), is similar to that of the Lake Ellesmere migrant eels recorded by Todd (1980). In that study, migrant males were predominantly around 440 mm in length, and females about 540 mm.

Figure 6. A migrant shortfin eel showing the silver belly coloration and enlarged eyes typical of this lifestage.

Small numbers of inanga and common bully have sometimes been recorded from the wetland, although we did not capture either species. They appear to be relatively rare in the system, which might be because of predation pressure. However, it might also have been that any of these which we caught were eaten by the eels in the nets. We did not examine the gut contents of any eels, so that cannot be confirmed. Another possible reason for their scarcity might be the low D.O that we recorded in parts of the wetland, and/or poor access from the sea. However, we did observe inanga whitebait in Angela Stream near the Corser Stream junction in October (Fig. 7), so they can at least enter that part of the wetland, but their survival rate might be low owing to predation.

Figure 7. Whitebait (some of them arrowed) in Angela Stream, October 2009

One of the aims of the restoration work in the wetland was to improve the fish diversity. To date, based on data available to us, it appears that this aim has not been achieved, although a significant shortfin eel habitat has been created, as well as a habitat for the noxious species rudd.

7.4.2 Future Fish introductions

As indicated in the ecological background section of this report, consideration has been given to introducing Canterbury mudfish into the wetland, but mudfish typically do not coexist with piscivorous species such as eels, so it is unlikely that a mudfish population would develop without intensive ongoing management to exclude predators. However, at the time of writing, it is understood that a release of Canterbury mudfish will occur in the Travis Wetland (D. Bradshaw, pers. comm.). The release of Canterbury mudfish into habitats isolated from eel predation was raised as a possibility in an early report (Sagar *et al.* 1996), and is probably of more urgency given the demise of several key Canterbury mudfish habitats.

Given the presence of large eel predators, other than inanga and common bully, the only native fish which is likely to find the wetland a suitable habitat is the giant kokopu (*Galaxias argenteus*). This species was almost certainly present in the wetland in pre-European times. It is almost extinct in Canterbury nowadays due to wetland habitat drainage, although a population survives in Horseshoe Lagoon in South Canterbury. Nevertheless, it is unlikely that giant kokopu would colonise the wetland of its own accord, because of the remoteness of the nearest population, and it would have to be re-introduced. If that was done, and water quality was improved, then giant kokopu could almost certainly live there.

However, it is likely that a sea-going population would not be self-sustaining , because giant kokopu whitebait are not known to home to their natal stream, so they would not necessarily find their way back there (R.M. McDowall, pers. comm.). This means that the population would become diluted and it would simply die-out over time. On the other hand, a population established from a land-locked stock such as that in Horseshoe Lagoon (South Canterbury) might be self-perpetuating. There will be consenting issues associated with the transfer of fish which would have to be overcome, but such a proposal is nevertheless worth entertaining. An attempt was made by NIWA staff in December 2009 to re-establish a giant kokopu population in an Auckland stream, but it is too early at this stage to establish whether that was successful.

Giant kokopu are a large and aggressive fish, and quite piscivorous (fish eating). In a diet study, while most of the diet was based on invertebrates, 26% of the fish examined contained at least some fish remains (Bonnett & Lambert 2002). Giant kokopu could potentially prey on small rudd should they be illegally re-introduced or should juvenile rudd become more predominant. The Lake habitat would be ideal for giant kokopu, owing to its depth, and already reasonable oxygen levels. Giant kokopu are terrestrial and benefit from abundant instream cover, to protect them from both fish predators, and in the case of the Travis Wetland, possible predation from birds. (Bonnett & Sykes 2002; Taylor 1988).

7.5 Water circulation and fish access

7.5.1 Water circulation

As indicated above, a lack of water circulation through at least parts of the wetland, in particular Frosts Stream, near the Willows, where the spring D.O. was very low. Low DO is not unusual in shallow wetlands with poor circulation and deep sediment. For example, Bottle Lake, near Bottle Lake Forest, historically a large open-water lake, now lacks virtually all its open water and is choked with Alder and other deciduous trees, and has a DO close to zero (Taylor & Sykes 1999).

AEL has been monitoring a former fen wetland on the Wairau River Plain, which has gradually become nutrient-loaded from willow leaf fall. This too has DO levels close to zero (Preece 2007; Taylor 2004). Under such nearly anoxic conditions in shallow coastal wetlands, shortfin eels are probably the only New Zealand species that can survive, using both physiological and behavioural adaptations. These include altering the blood chemistry (Forster 1981), or gulping air from the surface, and holding the air across the moist gill surfaces (Steen & Kruysse (1664) in Forster 1981).

We consider that the D.O. levels in Frosts Stream, in particular, would have to be higher if other fish species such as inanga and common bully were to become established. The higher DO could alleviate the high toxic nitrite levels. Both of these water quality determinands (low DO, high nitrite) could, at times, act as a barrier to migration into the wetland from the Avon River. However, as our summer records indicate, D.O. is not always low in Frosts Stream. The high D.O. level obtained in summer (29/1/10) in the Lake and in the northern end of Frosts Stream was obtained when the Lake was discharging along the Frost Stream course, as the weir boards were removed at the Lake Weir. We have no D.O. data along Frost Stream at the southern end. This was happening during a low tide event and it appeared the Wetland was discharging into the Avon River.

As originally envisaged, the circulation pattern through the wetland was that water would flow from the Avon River on the high tide, into the ecological corridor along ANZAC Drive, enter the wetland from the roundabout culvert, flow north along Frost Stream into the lake, and then discharge back into the Avon River via Angela and Corsers Stream (Paul Dickson, CCC, pers. comm.). This is depicted in Fig. 8. Currently this circulation appears to be impeded, at least in part, by the presence of weirs at several points in the wetland which are presumably in place to regulate the water level in the Lake. We consider that, as a result, water circulation is quite limited, which would explain the heterogeneous water quality results.

The poor water circulation could also allow the build-up of concentrations of nuisance algae in some parts of the wetland, which in the case of *Microcystis*, can become toxic to animal life. If there was better water movement through the wetland the algal cells would be flushed from the system decreasing the possibility of toxic algal blooms. One possible solution is to use the wind-mixed oxygenated water in the Lake to re-oxygenate the water in the inlets and outlets. There are probably several ways in which this could be achieved. One option is to lower the Lake more regularly by removing the dam boards at the Lake Weir so that it discharges into Frosts Stream. It is understood the Lake is lowered during the summer months to provide more wading bird habitat around its periphery. The Frosts Stream water level could be lowered prior to replenishment by discharging to the Avon River via Lake Kate Sheppard. This would also allow more fish access into the wetland. Another option is to allow more water exchange by the incorporation of holes in the dam boards, which could be bunged should it be desired to retain the lake, or tributaries, at an artificial level. Bung holes may be a suitable solution for the four weirs along Angela Stream, and facilitate fish passage along this waterway.

7.5.2 Fish recruitment

In addition to restricting circulation, the weirs (see Fig. 9) probably reduce fish recruitment to parts of the wetland by impeding access, except for eels, which are able to overcome minor obstacles. Removing the weir boards completely would allow better access for species such as inanga and common bully, although there would be a risk of allowing rudd to move from the wetland into the Avon River. Although it is possible they may already be present in the Avon. Juvenile rudd were identified from the Porritt Park loop of the Avon River in 1980, although further surveys have not yielded any more identifications (Eldon & Kelly 1992). Moreover, DoC is in the process of trying to eliminate the rudd by netting them from the wetland. There is some evidence that that is being successful – the minimum size of rudd the department is capturing has increased (Fig. 5) – which suggests that the recruitment of juvenile rudd is now becoming limited.

Figure 8. The five recorded weirs in the Travis Wetland (yellow pins). The blue line depicts the original flow design. The red pin indicates the location of the water ingress and egress to Lake Kate Sheppard and the Avon River.

In contrast, giant kokopu commonly occur in slowly-flowing sites and have a higher tolerance to high water temperatures than inanga (Main, 1988). We expect that they are likely to survive in waters with higher temperatures, and probably also with the associated lower dissolved oxygen concentrations. As mentioned above, giant kokopu can from lake-locked populations, and recruitment from the sea is not essential to maintain a population.

Figure 9. A weir on the outlet from the Open Site. This restricts the movement of water out of that area. It could also reduce access by migratory fish other than eels. Eels, with their ability to climb, could negotiate this obstruction.

8 Recommendations

We consider that the wetlands water quality and value as a fish habitat could be enhanced, in a series of steps. Firstly, the water quality, especially D.O. needs to be raised, with nitrite levels decreased.

AEL recommends that:

1. Further discussion is required on improving water quality and water circulation in the wetland with a wider representations of interests. We recommend that a meeting be convened between ecologists, engineers, and park rangers on practical ways of improving water quality in the wetland.

2. In parts of the wetland, the cyanobacterium *Microcystis aerogenosa* could form blooms which can be toxic. The growth of *Microcystis* populations, and the potential development of blooms, needs to be monitored over the summer, especially sunny locations where water circulation is poor.

3. The legalities and practicality of introducing giant kokopu be explored. An introduction of giant kokopu after the rudd have been fished out of the wetland may serve as a natural biological control technique against further introductions of pest fish.

4. Should water quality improve to the point where it could sustain more fish species, inanga whitebait could be netted and introduced to the wetland, although they require sea access to spawn. Inanga are known to spawn near Lake Shepherd within the ecological corridor. In contrast, given suitable water quality conditions, the common bully could form a self-sustaining population without assistance.

9 Acknowledgements

Thanks to EOS Ecology for processing the invertebrate samples and Helen McCaughan (DoC) for providing the DoC fish-sampling data. The Christchurch City Council provided funding for this study, and we acknowledge their patience while we prepared this report. We thank John Skilton (CCC, Travis Wetland Park Ranger) for providing vehicle access to the Travis Wetland, and discussions about ecological issues regarding the wetland. We would like to thank the Travis Wetland Trust for meeting with us, and their support and interest in ecological issues regarding the wetland. Ann Kennedy of the Trust provided valuable water quality data, and the pre-development report (Anderson 1999).

10 References

Anderson, A. 1999. Biological Monitoring of Travis Swamp. Lincoln University, *No*. 21 p.

ANZECC 2000. Australian Water Quality Guidelines for Fresh and Marine Waters. Australian and New Zealand Environment Council, Melbourne. *No*. p.

Bonnett, M. L.; Lambert, P. W. 2002: Diet of giant kokopu, *Galaxias argenteus*. *New Zealand Journal of Marine and Freshwater Research 36*: 361-369.

Bonnett, M. L.; Sykes, J. R. E. 2002: Habitat preferences of giant kokopu, *Galaxias argenteus*. *New Zealand Journal of Marine and Freshwater Research 36*: 13-24.

Collier, K. J. 1993: Review of the status, distribution, and conservation of freshwater invertebrates in New Zealand. *New Zealand Journal of Marine and Freshwater Research 27*: 339-356.

Dean, T. L.; Richardson, J. 1999: Responses of seven species of native freshwater fish and a shrimp to low levels of dissolved oxygen. *New Zealand Journal of Marine and Freshwater Research 33* (*1*): 99-106.

Eldon, G. A.; Kelly, G. R. 1992. Fisheries survey of the Avon River, 1991-1992. Freshwater Fisheries Centre, Christchurch. *New Zealand Freshwater Fisheries Report No*. *137*. 33 p.

Forster, M. E. 1981: Oxygen consumption and apnoea in the shortfin eel, *Anguilla australis schmidtii*. *New Zealand Journal of Marine and Freshwater Research 15* (*1*): 85-90.

Hicks, B. J. (2001): Biology and potential impacts of rudd (*Scardinius erythrophthalmus* L.) in New Zealand. *In:* Managing invasive freshwater fish in New Zealand. Hamilton, 10-12 May 2001. Pp. 49-58.

Main, M. R. 2001. The Influence of waterfowl on water quality of two small lakes in Canterbury. Environment Canterbury, *Technical Report No*. *U01/92*. 17 p.

McDowall, R. M. 1990: New Zealand Freshwater Fishes: A Natural History and Guide. Auckland, Heinemann Reed. 553 p.

McGlone, M. S. 2009: Postglacial history of New Zealand wetlands and implications for their conservation. *New Zealand Journal of Ecology 33* (*1*): 1-23.

Meredith, A. M.; Hayward, S. A. 2002. An overview of the water quality of the rivers and streams of the Canterbury region. Environment Canterbury, Christchurch. *Environment Canterbury Technical Report No*. *R02/25*. 32 p.

Morgan, D. R.; Graynoth, E. 1978. The influence of Forestry Practices on the Ecology of Freshwater Fishes in New Zealand; and introduction to the literature. Fisheries Research Division, Christchurch. *Occasional Publication No. 14*. 36 p.

NZFFDB 2009: Archives of the New Zealand Freshwater Fish Database.National Institute of Water and Atmospheric Research, Wellington.

Preece, J. 2007. The Rarangi Wetland Complex. WetlandsNZ, *No*. 21 p.

Prescott, G. W. 1954: How to Know the Fresh-water Algae. Dubuque, Wm. C Brown Co. 211 p.

Pridmore, R.; Hewitt, J.; Campin, M. E. 1982. A guide to the common freshwater algae in New Zealand. Ministry of Works and Development
Wellington. *Water and Soil Miscellaneous Publication No. 39*. 45 p.

Russo, R. C.; Smith, C. E.; Thurston, R. V. 1974: Acute toxicity of nitrite to rainbow trout (*Salmo gairdneri*). *Journal of the Fisheries Research Board of Canada 31*: 1653-1655.

Sagar, P. M.; Hawes, I.; Taylor, M. J.; Smith, R. 1996. Inventory of aquatic life in North Travis Wetland and an assessment of its ecology. National Institute of Water and Atmospheric Research, Christchurch. *NIWA Consultancy Report No. CCC 60511*. 10 p.

Saroglia, M. G.; Scarano, G.; Tibaldi, E. 1981: Acute toxicity of nitrite to sea bass (*Dicentrarchus labrax*) and European eel (*Anguilla anguilla*). *Journal of the World Mariculture Society 12*: 121-126.

Stark, J. D.; Boothroyd, I. K. G.; Harding, J. S.; Maxted, J. R.; Scarsbrook, M. R. 2001. Protocols for sampling macroinvertebrates in wadeable streams. Ministry for the Environment, Wellington. *Macroinvertebrate Working Group Report No. 1*. 65 p.

Stark, J. D.; Maxted, J. R. 2007: A biotic index for New Zealand's soft-bottomed streams. *New Zealand Journal of Marine and Freshwater Research 41*: 43-61.

Taylor, M. J. 1988. Features of freshwater fish habitat in South Westland, and the effect of forestry practices. Freshwater Fisheries Centre, MAFFish, *New Zealand Freshwater Fisheries Report No. 97*. 89 p.

Taylor, M. J. 2004. Preliminary survey of fish condition from the Hinepango Wetland, Marlborough, October 2004. Aquatic Ecology Limited, Christchurch. *AEL Report No. 26*. 11 p.

Taylor, M. J.; Bradshaw, D. 2009 Ecological aspects of the Mimimoto Lagoon. Aquatic Ecology Limited, *AEL Report*
No. 68. 15 p.

Taylor, M. J.; Sykes, J. R. E. 1999: Fish values in the Bottle Lake Region. Bottle Lake Wildlife: Preliminary Assessment. Christchurch City Council.

Todd, P. R. 1980: Size and age of migrating New Zealand freshwater eels (*Anguilla* spp.). *New Zealand Journal of Marine and Freshwater Research 14* (*3*): 283-293.

11 Appendix I. Travis wetland sampling sites

Figure i. Angela Stream

Figure ii. The open site.

Figure iii. Willows Stream.

Figure iv. The lake site.

Figure v. A panoramic view of the control site.

12 Appendix II. Water temperature and water conductivity logger data

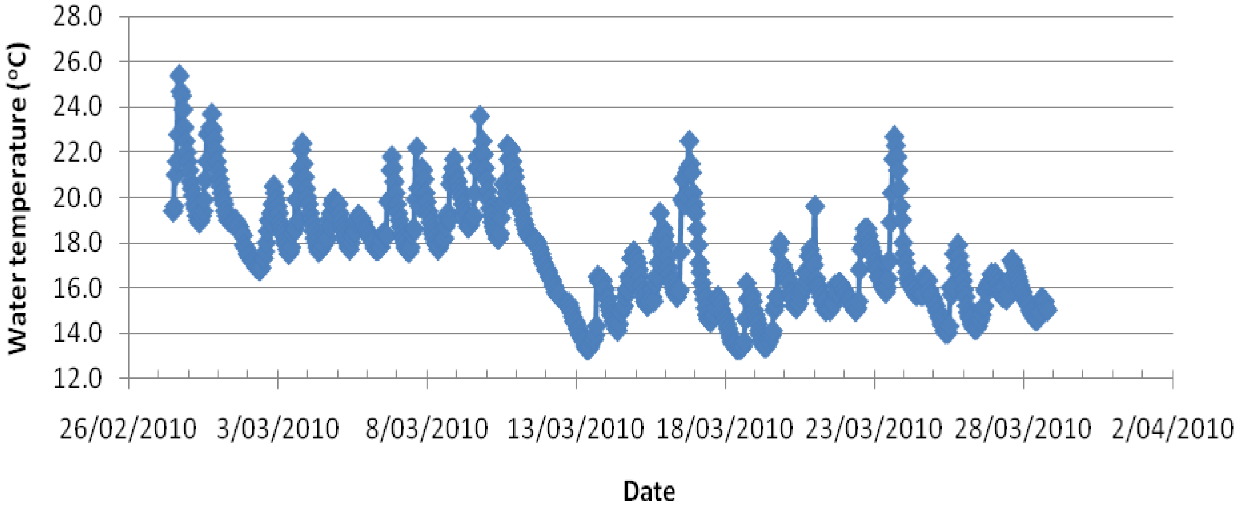

Figure a. Hourly water tempertaure records from the Travis Wetland Lake.

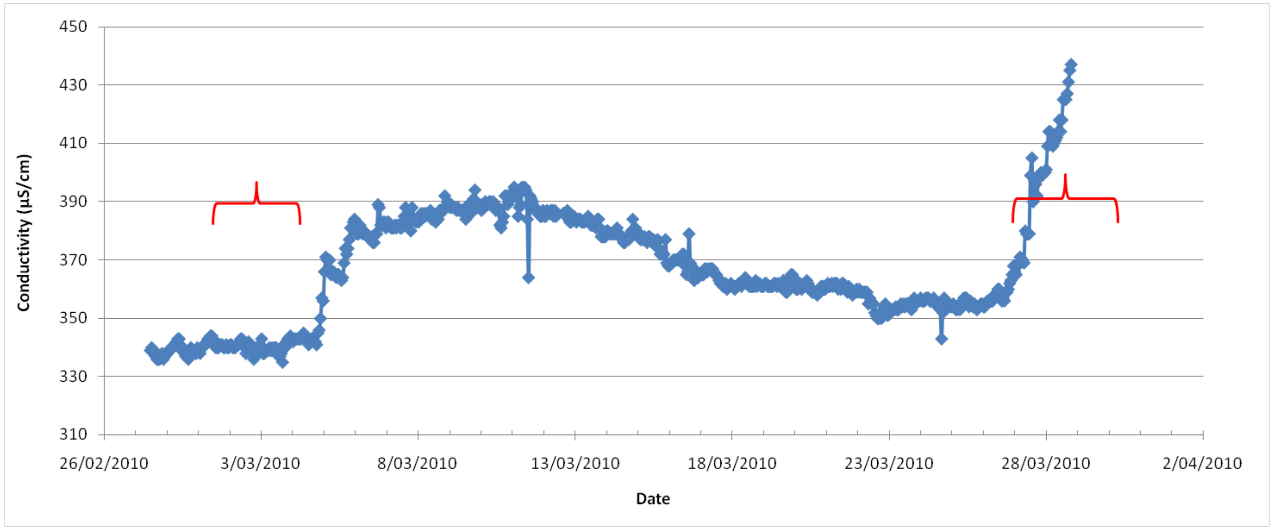

Figure b. Hourly conductivity from the Travis Wetland Lake. The brackets indicate the time of lunar spring tides.