Development of a South Island Wetland Macroinvertebrate Community Index score

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Executive Summary

1. Biotic indices are used to assess the condition of rivers and streams throughout New Zealand. In particular, the Macroinvertebrate Community Index (MCI) and its semiquantitative (SQMCI) and quantitative (QMCI) variants can be used to assess organic enrichment in stony bottomed streams. Recently, new indices, the soft bottomed MCI (MCI-sb) and its semiquantitative variant (SQMCI-sb), have been developed to assess the ecological condition of soft-bottomed streams.

2. Despite the widespread use of these invertebrate-based metrics to assess stream condition, no similar indices exist for assessing wetland condition in New Zealand. This deficiency is at odds with the demonstrable value of invertebrates as indicators of ecosystem condition. This report outlines the development of a wetland version of the macroinvertebrate community index based on presence-absence data (WMCI) and its quantitative variant (QWMCI), based on % abundance data, to assess wetland condition.

3. Invertebrates were collected from 82 lowland (< 250 m asl) wetlands throughout the South Island. Dominant land cover surrounding the wetlands varied greatly, from undisturbed native bush through to intensively farmed agricultural land and urban development. At each wetland, samples were collected from up to three open water bodies, giving a total of 240 samples. Samples were processed and invertebrates counted and identified down to levels commensurate with that used by the MCI with the exception of some groups of Crustacea, which were identified to Order instead of Subclass.

4. Water chemistry data (pH, conductivity, and nutrients) were obtained for each sampling location. Information on environmental factors (such as wetland area, distance to sea, elevation, land-use, and geology (both for the wetland itself, and for a 1 km buffer surrounding the wetland)) and climatic variables were extracted from GIS databases. Wetland condition was calculated using two independent landscape-based methods (Index of Ecological Integrity (IEI) & the Wetland Condition Index (WCI)) that assessed factors such as degree of hydrological modification, sedimentation, nutrient enrichment, and presence of exotic invasive species. A composite variable (WEED-AG) (based on the percentage of weedy species within the wetland, and agricultural development within a 1 km buffer around the wetland) was used to classify wetlands into nine condition classes.

5. A combination of ordination and TWINSPAN classification was used to determine which environmental variables were structuring invertebrate communities. This was to determine whether individual tolerance values (TVs) had to be developed for different wetland types (e.g., bogs, fens, and swamps), or for wetlands in different regions, or whether TVs could be derived from the entire South Island wetland invertebrate data set. These analyses suggested that TVs derived from the entire dataset was the preferred option.

6. A total of 141 taxonomic units were identified from the 82 wetlands. This was reduced to 122 taxonomic units following merging of adult and immature stages, and species into genera. The fauna was numerically dominated by the hydrobiid snail *Potamopyrgus antipodarum*, the midge *Tanytarsus*, micro-Crustacea (ostracods, Daphniidae, cyclopoid and harpacticoid copepods), oligochaetes and nematodes. Ordination showed that wetland invertebrates were influenced by factors including pH, wetland condition, the proportion of the wetland (or the 1km buffer) in agriculture or with tall exotic vegetation, and water chemistry (water nutrient concentrations and conductivity). Lack of strong consistent signals from factors such as pH or region meant that we created only single TVs for invertebrates from all wetlands.

7. We used an iterative rank correlation procedure to derive TVs for 122 invertebrate taxa. We tested the performance of the resulting WMCI and QWMCI indices by four methods:

   (i) determining the power of the WMCI, MCI and MCI-sb (and their quantitative variants) to discriminate between wetlands grouped according to WEED-AG;

   (ii) examining Spearman rank correlations between the different biotic metrics and environmental variables;

   (iii) examining correlations between the WMCI and the MCI and MCI-sb (including their quantitative variants), and other indices of wetland condition (IEI & WCI);

   (iv) creating specific numeric bands to assess wetland condition.

8. TVs for invertebrates ranged from 1 to 10 (the same as the original MCI), but taxa with low TVs were comparatively under-represented. Only 10 taxa had TVs of 3 or less, whereas 48 taxa had TVs of 8 or more.

9. Calculated WMCI and QWMCI scores displayed the strongest differences between wetlands when assigned to their WEED-AG class. The other two metrics (MCI, and MCI-sb, and their quantitative variants) did not differ as much between the WEED-AG classes. The WMCI and QWMCI displayed the strongest correlations to environmental variables when compared to the other biotic metrics. Both the QWMCI and WMCI had high negative correlations with water quality variables (pH, conductivity, and nutrients), and both were highly correlated to land-use conditions within a 1 km buffer of the wetland.

10. The WMCI and WQMCI had the highest correlation to each other, whereas correlations between the MCI and QMCI, and between the MCI-sb and QMCI-sb were much lower. The WMCI/QWMCI indices provided site rankings more similar to their corresponding soft-bottomed versions (MCI-sb and QMCI-sb) than to the original hard-bottomed versions (MCI and QMCI), possibly reflecting the similar habitat conditions (and associated invertebrate communities) in wetlands and soft-bottomed streams. The WMCI and QWMCI had the

strongest relationships to the two landscape-derived assessments of wetland condition (IEI and WCI).

11. The range of observed WMCI scores (84 – 154) was relatively lower than for the QWMCI scores (2.34 – 8.56), suggesting that the QWMCI was likely to provide a better basis for classifying wetlands into quality classes. We derived four quality classes based on values of the QWMCI from selected reference wetlands with the following values of the QWMCI: “Excellent” > 7.49, “Good” 6.25 – 7.49, “Fair” 5.0 – 6.24, and Poor < 5.00.

12. It is hoped that the existence of the WMCI (and QWMCI) will encourage resource managers and freshwater ecologists to use these biotic indices to assess wetland condition, in state of the environment monitoring, and as an aid to management of threatened wetland ecosystems.

1. Introduction

New Zealand is well endowed with a great variety of aquatic ecosystems, including lakes, rivers and streams, and wetlands. Although our lakes and rivers have been studied extensively for decades, wetlands have received much less attention from ecologists and managers (Sorrell & Gerbeaux 2004). Their plant communities may be well known, but their invertebrate communities have traditionally escaped scientific scrutiny (Suren & Sorrell 2010). Such lack of knowledge is at odds with the important role that invertebrates play in providing food for many native fish and wading birds (e.g., Goss-Custard 1970; Sanders 2000; Yates et al. 1993) contributing to carbon cycling in wetlands, and in adding to wetland biodiversity values. Wetlands now occupy only 10% of their original area in New Zealand (Ausseil et al. 2008; Campbell & Jackson 2004) because they often occupy low-lying, fertile land, much of which has been converted to agriculture. Of this 10%, only areas within the conservation estate can be considered safe from direct impacts associated with landuse changes such as alterations to hydrology through drainage, invasion by weedy species, elevated nutrient inputs and physical disturbance from actions of livestock. The impact of these continued stressors on macroinvertebrate communities in wetlands is unknown.

Two methods have been developed to assess wetland condition within New Zealand. The Wetland Condition Index (WCI: Clarkson et al. 2003) is based on field observations of five factors that affect wetland condition:

1) hydrological integrity;
2) physicochemical parameters;
3) ecosystem intactness;
4) browsing predation and harvesting regimes; and
5) dominance of native plants.

The second method, called the Index of Ecological Integrity (IEI: Ausseil et al. 2008) derives six spatial indicators of human activities known to degrade wetland biodiversity and function from national GIS databases. The six indicators are:

1) proportion of natural vegetation cover;
2) proportion of human-made impervious cover;
3) number of introduced fish;
4) percentage cover by woody weeds;
5) artificial drainage; and,
6) a surrogate measure of landuse intensity (nitrate leaching risk).

Both methods primarily assess the landscape and catchment factors that influence wetland plant communities. However, a complete assessment of wetland condition would ideally also include aquatic communities, since the functional and biodiversity values of invertebrates in freshwater ecosystems are widely recognised (e.g., Batzer et al. 1999; Mitsch and Gosselink 2000).

Aquatic invertebrates have been used for decades to assess the ecological condition of rivers and lakes (e.g., Chessman 1995; Hilsenhoff 1987; Plafkin et al. 1989; Stark 1985). Their wide usage reflects their relative ease of collection and identification, and the fact that their long life spans (weeks – months – years, which in many cases spans both aquatic and terrestrial habitats) allows them to act as integrators of antecedent environmental conditions over long time scales. This ability has been recognised by the almost universal uptake of the Macroinvertebrate Community Index (MCI), and its quantitative variant (QMCI) (Stark 1985, 1993, 1998), which is used by Regional Councils and other organisations throughout New Zealand to assess the biological condition of stony-bottomed rivers. Tolerance values (TVs) have recently been derived to permit the MCI to be applied to soft-bottomed streams as well, resulting in the creation of the MCI-sb and QMCI-sb (Stark & Maxted 2007a). These biotic indices rely on the assumption that different invertebrates respond to anthropogenic stressors in a consistent manner, and that invertebrate communities at a particular location reflect environmental conditions at that site (Stark and Maxted 2007a,b). The MCI and MCI-sb (and their quantitative variants) are arguably the most commonly used biotic indices for aquatic communities within New Zealand, reflecting their ease of use and interpretation, and apparent ability to accurately reflect ecosystem condition.

Despite the widespread use of the MCI throughout New Zealand to assess the ecological condition of rivers and streams, biotic indices specifically for assessing wetland condition using invertebrates have not been developed. This contrasts to the situation in Australia, where Chessman et al. (2002) developed a biotic score (called SWAMPS) for invertebrates in western Australian wetlands, which showed strong correlations with independent measures of cultural eutrophication and other anthropogenic disturbances. In North America, wetland ecologists have also produced a number of invertebrate indices to describe wetland health (Apfelbeck 2001; Helgen & Gernes 2001). In some cases, a multimetric approach has been taken scoring factors such as taxon richness, trophic structure and functional feeding groups, presence of exotic species, and measures of the relative abundance of different tolerant and intolerant taxa. Clearly, the use of such metrics relies on considerable *a priori* knowledge of the response of wetland invertebrates to anthropogenic disturbance – information that is currently lacking in New Zealand.

This report describes development of a wetland Macroinvertebrate Community Index (WMCI), which relies on presence-absence data, and its quantitative variant (QMCI), which relies on relative abundance data. These indices should enable managers to assess the biological health of wetlands based on their invertebrate communities. The WMCI currently applies to only low elevation wetlands (less than 250 m) throughout the South Island, because this is where most agricultural development has occurred, and where many remnant wetlands currently are threatened.

A potentially confounding factor in developing the WMCI (and QWMCI) was that wetlands encompass a range of types (e.g., bogs, fens, swamps) defined by natural gradients in water source, pH and nutrient concentrations (Johnson and Gerbeaux 2004). These different wetland types can support distinctive invertebrate communities (Suren and Sorrell 2010), with, for example, molluscs being absent from low pH wetlands. Invertebrate communities can also differ among wetlands affected by varying degrees of human impacts (Fennessy et al 2004; Helgen 2002). A strong geographic signal also is apparent for wetland invertebrates within the South Island (Suren and Sorrell 2010), reflecting amongst other things natural biogeographic patterns. For instance, wetlands in north-western Nelson supported 14 unique taxa, whereas those in South Westland and Southland supported seven and six unique taxa, respectively. Consequently, a wide range of low-elevation wetlands throughout the South Island was sampled to ensure strong environmental gradients were present in the data, an essential requirement when developing indices (Helgen 2002), and to ensure as great a coverage of different biogeographic areas as possible. TVs were derived for invertebrate taxa (mostly genera) encountered in the wetlands along a gradient of disturbance from wetlands in undisturbed catchments, to wetlands in modified landscapes. From these TVs, WMCI and QWMCI values were calculated.

We then assessed how well the WMCI and QWMCI could discriminate between wetlands of different condition, as assessed on the basis of catchment characteristics and presence of weeds within the wetland. Numerous studies have shown strong linkages between catchment land-use and stream hydrology (Dons 1987; Fahey & Rowe 1992) water chemistry (Close & Davies-Colley 1990; Cooper et al. 1987; McColl & Syers 1981; Quinn et al. 1997), energy inputs (DeLong & Brusven 1994; Duncan & Brusven 1985; Young & Huryn 1999) periphyton (Biggs 1995; Quinn et al. 1997) and invertebrate communities (Hall et al. 2001; Harding & Winterbourn 1995). While the effect of land-use on wetland hydrology, water chemistry, energy inputs, and biota are less well documented (Department of Sustainability and Environment 2005), methods to assess wetland condition do consider surrounding land-use (Ausseil et al. 2008; Clarkson et al. 2003; Helgen 2002), suggesting that it is also likely to be of great importance in determining wetland condition, as it does for streams.

It is hoped that the WMCI and QWMCI represent scientifically defensible ways to measure the condition of lowland wetlands by using aquatic invertebrates as indicators of ecosystem stress/health. We hope that this will assist managers to catalogue and describe remaining wetlands, commence wetland invertebrate state of the environment monitoring, set priorities for protection, and monitor effectiveness of restoration activities in order to better protect and manage these threatened indigenous ecosystems.

2. Methods

2.1. Field methods

Invertebrate samples were collected in spring or late summer between 2003 and 2007 from 82 wetlands throughout six regions of the South Island: Tasman, West Coast, Canterbury, Otago, Southland, and Stewart Island (Figure 1; Appendix 1). Stewart Island was considered a separate region to Southland, reflecting its geographic isolation from the mainland and unique freshwater biota (e.g., Chadderton 1990). Catchment land-use ranged from unmodified native forest / tussock through to pasture, dairy farming, and urban development. Wetlands were classified into fen, bog, swamp or shallow water, based on differences in their dominant vegetation, pH, nutrient status, and hydrological character (see Johnson and Gerbeaux, 2004). Wetlands were assigned to the appropriate class either in the field by experienced wetland ecologists, or classes were obtained from the wetland classification data as used by Ausseil et al 2008.

Within each wetland, duplicate invertebrate samples were collected semi-quantitatively (standardised 2 minute effort) from up to three discrete open-water bodies using a hand-held sweep net (300 µm mesh: Suren et al. 2008). The total number of water bodies (sites) sampled was 240. The location of each water body was recorded using a Garmin ® GPS (see Appendix 1). All invertebrate samples were preserved immediately in 100% isopropyl alcohol. Spot measurements of water chemistry (temperature, pH and conductivity) were made at each water body using a Horiba® multiprobe. Water samples were collected and filtered (GFF) for nutrient analysis, and frozen (-18°C) for subsequent analysis.

Independent assessments of wetland condition were made using two landscape-based methods (see Suren et al. 2010); the Wetland Condition Index (WCI: Clarkson et al., 2003), and the Index of Ecological Integrity (IEI: Ausseil et al. 2008).

2.2. Laboratory methods

Invertebrate samples were processed according to previous protocols (Suren et al., 2008; 2010). Briefly, this involved washing the sample through a series of nested

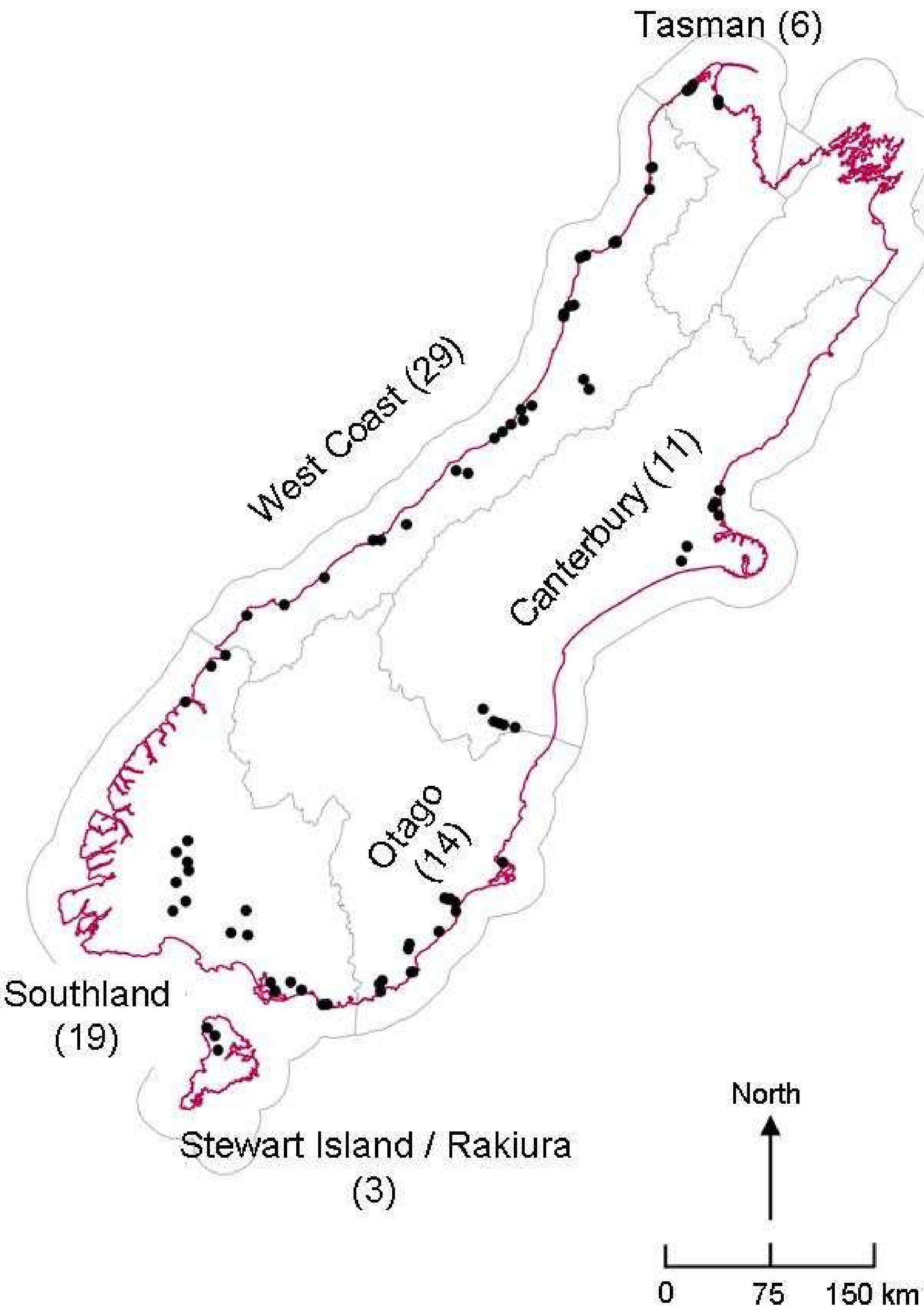

Figure 1: Map of the South Island showing locations of the 82 wetlands sampled in the six geographic regions, and the number of wetlands in each region (brackets).

sieves (2 mm, 1 mm, 0.5 mm, 0.25 mm, and 0.062 mm) to remove larger animals and plant fragments, and identifying and counting all invertebrates retained on the coarse (> 1 mm) and fine (0.062 – 1 mm) sieves. The contents of the coarse and fine material were each spread evenly across a small Bogorov tray (Winterbourn et al. 2006) and examined under a dissecting microscope (up to 40 times magnification) for invertebrates. Where the contents of the entire sieve could not be processed, a subsample (up to 1/16th) was selected. A minimum of 400 invertebrates from each sieve size was identified, and the rest of the sample or subsample was scanned for uncommon taxa (Duggan et al. 2003). All invertebrates were identified to as low a taxonomic resolution as possible given the availability of taxonomic keys and the practicality of identifying small taxa such as nematodes, tardigrades, and micro-crustaceans (Suren et al., 2008). Initially, we also recorded the adult and immature stages of insects as separate taxonomic units, given their widely different niches and ecological requirements. All invertebrate data were converted into percentages. The number of taxa in each sample was used to represent taxonomic richness.

Water samples were thawed and analysed for nutrients (NH_4-N, NO_3-N, dissolved reactive phosphorus (DRP), and total dissolved phosphorus and nitrogen (TDP and TDN, respectively)) using standard methods (APHA 1975; Diamond 2003).

2.3. GIS data extraction

The GPS co-ordinates of each sampling site were used to extract relevant data from two large-scale GIS databases: the New Zealand Land Cover Database (LCDB2, derived from satellite imagery), and the Freshwater Environments of New Zealand (FWENZ) database (Leathwick et al. 2007; Wild et al. 2005). The first step in this process was to create polygon boundaries around each wetland. This was done using a digital elevation model (DEM) with a 15 m resolution. Each polygon boundary was validated based on local knowledge and field observations. Originally we attempted to delineate catchment boundaries around each wetland. However, the fact that many wetlands are within flat, low lying areas combined with the relatively low height resolution of the DEM, meant that defining accurate catchment boundaries was problematic. Moreover, we could not fully justify our catchment definition for wetlands located near the lower reaches of large rivers such as the Waitaki (catchment area ~ 11,400 km^2), as we were not confident that individual wetlands would indeed be influenced by all upstream conditions. Instead of using information from the entire catchment, we delimited two buffers zones (100 m, and 1 km) around each wetland polygon within which we described land cover and geology. Information on land cover and underlying geology from the wetland and from these two buffer areas was extracted. However, correlation analysis of land cover and geological data showed significant redundancy in the wetland and 100 m buffer data. Therefore we only used data extracted from the wetland and 1 km buffer for all subsequent analyses. Wetland area and slope, as well as distance to sea from the centroid of each wetland were calculated also. Finally, we extracted variables describing climate, including temperature, rainfall, precipitation, evapotranspiration, and solar radiation (Table 1).

2.4. Statistical analysis

2.4.1. Investigation of regional or wetland differences

The aim of the analysis was to develop a WMCI based on TVs of individual taxa found in wetlands along a gradient of disturbance from pristine wetlands to wetlands in largely modified landscapes. However, we also sampled a range of wetlands from oligotrophic low pH bogs, to more mesotrophic higher pH swamps (Johnson & Gerbeaux 2004; Wheeler & Proctor 2000). Thus, the first stage of the analysis was focused on determining whether individual WMCI scores needed to be created for different wetland types, or for wetlands in different regions.

In an initial exploratory analysis, the invertebrate dataset was ordinated using Detrended Correspondence Analysis (DCA: McCune & Mefford 1997), of the percentage abundance data. Correlations between ordination scores and measured environmental parameters (log-transformed except for pH) and those extracted by the GIS analysis were calculated. All correlation coefficients were corrected using the Benjamini & Hochberg (1995) False Discovery Rate (FDR) to minimise the chance of rejecting the null hypothesis (i.e., to minimise the overall Type I error rate). The FDR is defined as the expected proportion of true hypotheses rejected out of the total number of rejections. It considers how many of the α-level rejections may be in error. The FDR correction was used for all correlation analyses where multiple correlations were undertaken.

All data were coded *a priori* to categorical variables describing the region that each wetland was in (viz., Tasman, West Coast, Canterbury, Otago, Southland, and Stewart Island; see Figure 1), the type of wetland (i.e., bog, fen, swamp, shallow water), and the type of water body collected (i.e., large pond, small pond, lead, channel and drain: see Suren and Sorrell 2010 for definitions). Analysis of similarity (ANOSIM: Clarke & Warwick 2001) was used on the similarity data to determine whether invertebrate communities differed according to region, wetland type, or water type.

The invertebrate data were then classified using TWINSPAN (McCune & Mefford 1997). All sites were allocated to their respective TWINSPAN groups after the 1st, 2nd and 3rd divisions. ANOSIM was used to examine how the invertebrate communities differed according to three biologically-derived TWINSPAN groups. Following this, ANOVA, with Bonferroni-corrected post-hoc testing, was used to determine which environmental variables differed between the TWINSPAN groups.

Table 1: Environmental variables obtained for all wetlands, including variables measured in the field (water quality, easting and northing), and variables derived from GIS databases. Note for clarity, we have shown the wetland, 100m and 1km buffer variables on the same line (Wet_, 100m_, and 1km_).

Variable type	Variable	Spatial size	Description
Wetland	Easting (X)	Sample site	GPS derived Easting (NZMS Series 260)
	Northing (Y)	Sample site	GPS derived northing (NZMS Series 260)
	Area	Wetland	Wetland area (km^2)
	DistSea	Wetland	Distance to sea (km)
	Elevat	Wetland	Mean wetland elevation (m asl)
	WCI	Wetland	Wetland Condition Index
	IEI	Wetland	Index of Ecological Integrity
Climate	TCold	Region	Average annual minimum temperature (^oC)
	TWarm	Region	Average annual maximum temperature (^oC)
	SolarSum	Region	Average annual summer solar radiation ($W\ m^{-2}$)
	SolarWin	Region	Average annual winter solar radiation ($W\ m^{-2}$)
	AnnRain	Region	Average annual rainfall (mm)
	PET	Region	Potential evapotranspiration ($mm\ y^{-1}$)
Geology	Alluv	Wetland (Wet_),100m_, and 1km_, buffer	% Alluvium
	Cong	Wetland (Wet_),100m_, and 1km_, buffer	% Conglomerate
	HSed	Wetland (Wet_),100m_, and 1km_, buffer	% Hard sedimentary rocks
	Loess	Wetland (Wet_),100m_, and 1km_, buffer	% Loess
	Meta	Wetland (Wet_),100m_, and 1km_, buffer	% Metamorphic rocks
	Mud	Wetland (Wet_),100m_, and 1km_, buffer	% Mudstones
	Peat	Wetland (Wet_),100m_, and 1km_, buffer	% Peat
	Sand	Wetland (Wet_),100m_, and 1km_, buffer	% Sandstones
	SSed	Wetland (Wet_),100m_, and 1km_, buffer	% Soft sedimentary rock
	Town	Wetland (Wet_),100m_, and 1km_, buffer	% Urban areas
	VolRck	Wetland (Wet_),100m_, and 1km_, buffer	% Volcanic rock
Landcover	agric	Wetland (Wet_),100m_, and 1km_, buffer	% agriculture (pasture, horticulture)
	exot	Wetland (Wet_),100m_, and 1km_, buffer	% tall exotic (e.g., pine trees, willows)
	exot_shrt	Wetland (Wet_),100m_, and 1km_, buffer	% short exotic (e.g,. gorse, broom, shrubland)
	ntv_shrt	Wetland (Wet_),100m_, and 1km_, buffer	% short native (e.g,. tussock, Matagouri, fernland)
	ntv_tall	Wetland (Wet_),100m_, and 1km_, buffer	% tall native (e.g., indigenous forest and hardwoods)
	urban	Wetland (Wet_),100m_, and 1km_, buffer	% urban
	wetland	Wetland (Wet_),100m_, and 1km_, buffer	% wetland plants
Water quality	pH	Sample site	Wetland water pH
	Cond	Sample site	Spot conductivity ($\mu S\ cm^{-1}$)
	Spot_Temp	Sample site	Spot water temperature (^oC)
	NH_4	Sample site	Ammonia concentration ($mg\ l^{-1}$)
	NO_3	Sample site	Nitrate-N concentration ($mg\ l^{-1}$)
	DRP	Sample site	Dissolved reactive phosphrous concentration ($mg\ l^{-1}$)
	TDP	Sample site	Total dissolved phosphorus ($mg\ l^{-1}$)
	TDN	Sample site	Total dissolved nitrogen ($mg\ l^{-1}$)

2.4.2. Development of tolerance values (TVs)

The second stage of the analysis was to develop TVs for the different taxa, either for different regions or wetland types, or for the entire dataset – depending on the results of the previous analyses. When developing the MCI using macroinvertebrate data from Taranaki Ringplain, North Island, streams, Stark (1985) assigned TVs using a weighting procedure based upon the relative percentage occurrence of taxa at three site groups differing in enrichment status. These groups were assigned *a priori* according to their location with regards to receiving nutrient inputs from dairy sheds, waste water discharges, or the degree of agricultural intensification in the catchment upstream. TVs calculated by this procedure for less commonly encountered taxa were unreliable, and so were assigned by professional judgement (Stark 1993, 1998), which was the usual method for allocating taxon TVs before 1985 (e.g., Chutter 1972; Hilsenhoff 1987, 1988).

In Australia, TVs for SIGNAL (the Australian equivalent of the MCI) were derived objectively from a taxa by site data matrix that covered a wide range of disturbance, or stream condition (Chessman et al. 1997). Chessman (2003) subsequently improved this procedure to produce a new biotic index called SIGNAL2, and this same procedure was used by Stark & Maxted (2007a) to develop biotic indices for soft-bottomed streams using data from the Auckland region. For brevity we refer to this procedure as the "Chessman process". A fundamental requirement of the Chessman process is to be able to rank sites initially according to a disturbance gradient. Furthermore, a fundamental assumption of this process (and of any biotic index) is that invertebrate taxa will respond to this gradient so that taxa common in least disturbed sites will be different to taxa common in heavily disturbed sites. Stark and Maxted (2007a) in their creation of the MCI-sb, ranked sites according to their original MCI score. However, no such equivalent biotic index exists for wetland invertebrates, and so we could not use the above procedure. Because we assessed wetland condition by the WCI and IEI, we could have used these rankings as our initial start point in the analyses. However, this would have meant that we could not use these condition scores subsequently for testing the performance of the created WMCI.

Bruce Chessman (pers. comm.) indicated to Stark & Maxted (2007a) that the final set of TVs derived by the iterative process is not overly sensitive to the starting condition if there is a strong environmental gradient in the data set, and if invertebrate communities do indeed respond to changes in environmental conditions along this gradient. Consequently, we used a random site order as the starting condition (thus avoiding any circularity, and freeing up both the IEI and WCI for subsequent testing). Sites were assigned values from 0 – 9 by reading the first 240 digits from a table of random numbers (Mendenhall & Ott 1980), and the iterative Chessman process was used to develop TVs for all the taxa encountered. The first step in this process involved calculating Spearman rank correlations (r_s) between the relative abundance of each taxon and the random-order starting condition of the wetland samples. Since it is mathematically impossible for rare taxa with few occurrences to achieve high correlations (Chessman et al. 1997), each r_s was expressed as a proportion of the maximum possible r_s for a taxon recorded from the same proportion of samples. The taxon with the highest adjusted positive r_s between its relative abundance and the random-order starting condition was assigned a TV of 10, and the taxon with the lowest adjusted negative r_s was assigned a TV of 1. The remaining taxa were assigned intermediate TVs (to the nearest integer) in proportion to their adjusted r_s values. The resulting TVs were pasted back into an Excel spreadsheet where a user-defined function (Stark & Maxted 2007b) calculated the WMCI for each site. The wetland samples were then re-ranked according to this first iteration of the WMCI, and then new rank correlations (r_s) calculated between the relative abundance of each taxon and this new ranked order. New TVs were thus calculated from this second set of correlations, which were then used to produce a new WMCI score which ranked the sites for a third time. New rank correlations were calculated, and this procedure was repeated until the TVs stabilised (i.e., did not change from one iteration to the next).

Once TVs were developed using a random-order start condition, the validity of this method was checked by comparing the derived TVs for each taxon to new values derived from ranking the wetlands initially using either the WCI or the IEI.

Following the creation of individual TVs for the different taxa, the WMCI and its quantitative variant, the QWMCI were calculated as follows:-

$$\text{WMCI} = 20 \times [\Sigma\, a_i / S]$$

$$\text{QWMCI} = \Sigma\, (n_i \times a_i) / N$$

where a_i = TV for the ith taxon, S = total number of taxa, n_i = the number of individuals in the i^{th} taxon, and N = the total number of individuals (Stark & Maxted 2007b).

2.4.3. Testing the performance of WMCI

We evaluated the performance of the WMCI and QWMCI by comparing them to the MCI, QMCI, MCI-sb, and QMCI-sb, which are well proven indicators of the condition of hard bottom and soft bottom streams. Our assessment of the performance of the WMCI was similar to that used by Stark and Maxted (2007a) in assessing the performance of the MCI-sb, namely:

I. Determining the ability of the WMCI to better discriminate between wetlands assigned to different land-use categories;

II. Examining Spearman rank correlations between index values and a range of environmental variables at both a regional, marginal (i.e., a 1 km buffer), wetland, and local scale;

III. Assessing relationships between the WMCI and the MCI and MCI-sb;

IV. Creating specific numeric bands to classify wetlands into health or quality classes.

(I) The correlation performed in the DCA ordination (Section 2.4.1) showed that the percentage of agricultural land use within a 1 km buffer was the most strongly correlated land-use variable to DCA acis 1 scores. The degree of agricultural modification in a catchment is also incorporated in wetland condition assessments such as the WCI and IEI. Based on this, a land cover variable (AG-CLASS) was created with five categories of land-use within the 1 km buffer such that:

1. = no agricultural land use;

2. = 1-25% agricultural land-use;

3. = 26-50% agricultural land-use;

4. = 51-75% agricultural land-use;

5 = >75% agricultural land-use.

The degree of weed infestation is an important factor in other methods for assessing wetland condition (e.g., WCI and IEI). This measure is a good surrogate of altered hydraulic conditions, as most weed species cannot tolerate anoxic soils associated with undisturbed wetlands (Clarkson et al. 2003; Sorrell et al. 2007a). . Given the importance of weeds in influencing wetland condition, we created another categorical variable (WEED-RANK) based on weed cover within a wetland such that:

1. = no weeds, >75% natural vegetation, < 10% modified;

2. = no weeds, >50% natural vegetation, > 10% modified vegetation;

3. = < 10% weeds, OR 25-50% modified vegetation, 0-5% urban;

4. = 10-50% weeds, OR 50-75% modified vegetation, 10% urban;

5. = >50% weeds, OR >75% modified vegetation OR >20% urban.

Finally, a composite categorical variable (WEED-AG = the sum of AG-CLASS & WEED-RANK) was used to assign wetlands into 9 categories of "condition" (ranging from 2 – 10), such that 2 = wetlands in unmodified catchments without weeds, and 10 = wetlands in catchments dominated by agriculture and/or urban development, and with weeds and/or modified vegetation or urban development within the wetland. ANOVA and box plots were used to compare differences in the biotic metrics between WEED-AG classes.

(II) Spearman rank correlations (r_s) were calculated to examine relationships between the WMCI, MCI, and MCI-sb (and their quantitative variants), and all measured or derived environmental parameters at the scale of the wetland, 1 km buffer, and region. As with Stark and Maxted (2007a), we used rank correlations because biotic indices most often are used to determine the condition of particular sites in comparison to others, which is essentially a ranking exercise. In addition, rank correlations are not affected by non-normal or skewed data distributions.

(III) Spearman rank correlations were calculated between the WMCI, MCI and MCI-sb (and their quantitative variants) to see whether these indices ranked wetlands in a similar manner. We also correlated the biotic indices to the IEI and WCI to see whether any relationships existed between invertebrate-derived indices and wetland condition as assessed by these landscape methods.

(IV) Stark & Maxted (2007a) defined four quality classes for the MCI-sb and QMCI-sb based on the statistical distribution of soft-bottomed stream biotic index values at reference sites – an approach that we wanted to use for the WMCI and QWMCI. The resulting quality classes for the soft-bottomed stream versions of the MCI, SQMCI, and QMCI were the same as those determined previously for the hard-bottomed stream versions (Stark 1998). Samples with MCI values > 119 were assigned to the Excellent quality class, with Good (100 – 119), Fair (80 – 99), and Poor (< 80) for the other three quality classes. Corresponding values for the SQMCI and QMCI were > 5.99, 5.00 – 5.99, 4.00 – 4.99, and < 4.00.

Several criteria must be met in order to use Stark & Maxted's (2007a) method for defining quality classes. Firstly, the data set must contain samples that cover the full range of environmental condition, from poor to excellent. Secondly, the biotic index values should cover as wide a range as possible to provide good discrimination between quality classes, and finally, a subset of reference condition sites (i.e., those in excellent health) must be identified. We were confident that our South Island dataset covered a full range of wetland health. Although the WMCI and QWMCI were significantly correlated (r_s = 0.796) and thus yielded very similar rankings of wetland health, we found that the WMCI had a much more restricted range. This suggested that the QWMCI was more likely to provide better discrimination between quality classes than the WMCI. Consequently, we derived quality classes for wetlands based only on the quantitative index.

Identifying wetlands that are in reference condition was not as straightforward as it was for rivers and streams. In the absence of strict definitions of reference wetlands, we used our composite variable (WEED-AG) that provided a measure of the degree to which the wetland catchment (within 1km) was modified by agricultural and urban activity, and the degree of invasion of the wetland itself by exotic vegetation. WEED-AG values ranged from 2 to 10, with low index values indicative of less modified wetlands. We selected WEED-AG = 2 as our independent measure of wetlands in reference condition, and calculated wetland condition classes based on the statistical distribution of the QWMCI in these reference wetlands.

3. Results

3.1. Physical conditions

Environmental conditions in the 240 sites across 82 wetlands varied greatly (Table 2). Calculated WCI scores ranged from 5.8 to 25.0, and IEI from 0.08 to 0.97. This range of scores reflected, amongst other things, the wide range of land-use surrounding each wetland. The most common land-use around each wetland was either short native vegetation (i.e., scrub, ferns or tussock) and agriculture, or wetlands plants, all with a mean cover of > 10% (Table 2). Land-cover differed between the wetland polygon, and the 1 km buffer. Within the 1 km buffer, agriculture and tall native vegetation were dominant (mean cover ~30%: Table 2), whereas wetland plants were dominant within the wetland polygon. There was no agricultural activity within the polygon boundaries of 19 wetlands, but only four wetlands had no agricultural activity within their 1 km buffer zones. Only four wetlands had some form of urban development within their boundaries, although 14 had some urban development in their 1 km buffer. One wetland (Travis Swamp in Christchurch) had *c.* 44% of its 1 km buffer urbanised, representing the extreme urban condition in this survey. Alluvium was the most common underlying geology in both the wetland and the 1 km buffer polygons.

Strong climatic gradients existed in the data, with a strong east-west gradient in rainfall, lowest average annual precipitation (less than 550 mm) in the Waitaki Valley, and greatest annual precipitation (greater than 4000 mm) on the West Coast and

Table 2: Summary statistics (minimum, mean and maximum) of measured or derived environmental variables collected from 82 wetlands throughout the South Island. Variable abbreviations as per Table 1.

Variable Type	Variable	Min	Mean	Max
Wetland	Area (ha)	1.0	378.0	7753.7
	DistSea (km)	0.2	10.0	51.0
	Elevat (m)	0.1	42.4	243.9
	WCI	5.8	18.7	25.0
	IEI	0.08	0.5	0.97
Climate	TCold (^{o}C)	3.2	6.0	7.9
	TWarm (^{o}C)	12.5	15.2	17.3
	SolarSum (W m^{-2})	1573.5	2197.8	2352.1
	SolarWin (W m^{-2})	280.5	461.6	593.0
	AnnRain (mm)	498.1	2017.2	5655.9
	PET (mm y^{-1})	562.0	648.0	766.8
Wetland Geology (%)	Wet_Alluv	0.0	54.1	100.0
	Wet_Cong	0.0	2.5	100.0
	Wet_HSed	0.0	0.2	8.6
	Wet_Loess	0.0	2.4	98.9
	Wet_Meta	0.0	0.2	17.2
	Wet_Mud	0.0	1.4	77.8
	Wet_Peat	0.0	26.1	100.0
	Wet_Sand	0.0	9.8	100.0
	Wet_SSed	0.0	1.2	72.4
	Wet_Town	0.0	0.0	0.8
	Wet_VolRck	0.0	0.1	2.9
Wetland landcover (%)	Wet_agric	0.0	21.5	100.0
	Wet_Exot	0.0	7.1	100.0
	Wet_exot_shrt	0.0	2.6	70.4
	Wet_ntv_shrt	0.0	22.0	98.4
	Wet_nte_tall	0.0	9.3	100.0
	Wet_urban	0.0	0.0	0.2
	Wet_Wetland	0.0	37.4	100.0
1 km buffer geology (%)	1km_Alluv	0.0	49.8	100.0
	1km_Cong	0.0	6.4	90.4
	1km_HSed	0.0	3.4	68.4
	1km_Loess	0.0	7.4	97.3
	1km_Meta	0.0	1.7	68.3
	1km_Mud	0.0	1.3	42.4
	1km_Peat	0.0	14.0	96.6
	1km_Sand	0.0	6.8	100.0
	1km_SSed	0.0	3.3	92.3
	1km_Town	0.0	0.8	47.3
	1km_VolRck	0.0	0.4	28.9
1 km buffer landcover (%)	1km_agric	0.0	35.7	97.2
	1km_exot	0.0	7.5	66.2
	1km_exot_shrt	0.0	3.2	36.3
	1km_ntv_shrt	0.0	11.5	93.2
	1km_ntv_tall	0.0	28.8	100.0
	1km_urban	0.0	0.8	43.6
	1km_Wetland	0.0	12.6	73.7
Water quality	pH	3.1	6.2	9.9
	Cond (µS cm^{-1})	20.3	484.3	15520.0
	NH_4 (mg l^{-1})	1.0	48.8	2300.0
	NO_3 (mg l^{-1})	0.5	51.2	1310.0
	DRP (mg l^{-1})	0.2	16.3	549.0
	TDP (mg l^{-1})	0.0	47.7	953.0
	TDN (mg l^{-1})	55.0	632.5	3560.0

Stewart Island. Most of the wetlands sampled were relatively small (average size = 378 ha), with 53 wetlands less than 100 ha. Only six wetlands were greater than 1000 ha.

As expected, water chemistry conditions varied greatly throughout the wetlands (Table 2). The large range of pH, conductivity, and nutrients encountered reflected the wide range of bogs, fens and swamps that were sampled.

3.2. The invertebrate community

A total of 141 taxa were identified from the 82 wetlands. The fauna was dominated numerically by the hydrobiid snail *Potamopyrgus antipodarum* (13.9%), the midge *Tanytarsus* (9.2%), ostracods (8.8%) Daphniidae (7.3%), oligochaetes (6.8%), cyclopoid and harpacticoid copepods (6.6 & 4.9 % respectively), and nematodes (5.6%). The most common non-chironomid insect was the damselfly *Xanthocnemis zealandicus* (2.6%). The most widespread taxa were cyclopoid copepods, oligochaetes, ostracods and nematodes, which were found at 90% or more of sites.

To align identifications with those used in the MCI, we merged some taxa identified to species, and also merged adult and immature forms of some taxa (e.g., beetle larvae and adults), because the presence of either stage of their life cycle indicates presence in the wetland habitat. However, to minimise potential loss of information for some more dominant wetland taxa (Suren and Sorrell 2010), we distinguished some Orders (e.g., Calanoida, Cyclopoida, Harpacticoida) whereas the MCI distinguishes them only to Subclass (i.e., Copepoda). Following this revision, 122 taxa remained

3.3. Dominant drivers of invertebrate community composition

Ordination of the revised taxonomic data showed that DCA axes 1 and 2 explained 46% and 31% respectively of the variation explained by the first three axes. The gradient lengths of axes 1 and 2 were relatively low (3.8 and 3.5 respectively), indicating that there was low taxonomic turnover along these two axes. Axis 1 scores were most strongly correlated positively with wetland pH ($r^2 = 0.688$), and negatively with both the WCI and IEI ($r^2 = -0.428$ and -0.481 respectively: Table 3). Other variables describing the proportion of either the wetland, or the 1km buffer in agriculture, or with tall exotic vegetation, as well as underlying geology, water nutrient concentration (NO_3-N, TDN, and TDP) and conductivity, also were significantly correlated to DCA axis 1 scores. Fewer environmental variables were correlated with the axis 2 scores, and these reflected mainly water chemistry and conductivity, and underlying catchment geology (Table 3), with swampy wetlands of high conductivity and NO_3-N having high axis 2 scores. These wetlands were also in areas of conglomerate or mud stone.

Table 3: Measured environmental variables and their correlations to DCA axis 1 and 2 (*p* < 0.05), following FDR correction. Variable abbreviations as per Table 1.

Axis 1		Axis 2	
Variable	**r^2 value**	**Variable**	**r^2 value**
pH	0.688	Cond	0.278
1km_agric	0.472	1km_Cong	0.215
Weed_Ag	0.399	NO3-N	0.171
WeedRank	0.35	1km_Mud	0.163
Easting (x)	0.321	Wet_Cong	0.142
wet_agric	0.314	pH	0.141
1km_exot_tall	0.284	Cond	0.278
wet_exot_tall	0.249	1km_Cong	0.215
NO3-N	0.234	Wet_Cong	0.142
TDN	0.234	NO3-N	0.171
TDP	0.227		
Cond	0.217		
1km_Loess	0.2		
Wet_Hsed	0.176		
1km_Hsed	0.165		
DRP	0.158		
AveTWarm	0.143		
wet_ntv_tall	-0.143		
wet_wetland	-0.164		
wet_ntv_shrt	-0.225		
Wet_Peat	-0.237		
Area	-0.243		
1km_Peat	-0.266		
MCI_sb	-0.28		
1km_ntv_tall	-0.289		
MCI	-0.316		
1km_wetland	-0.33		
1km_ntv_shrt	-0.358		
Rainfall	-0.387		
WCI	-0.427		
IEI	-0.481		

Four species of molluscs, Daphniidae, oligochaetes and the hydroptilid caddisfly *Paroxyethira* were indicative of sites with high axis 1 scores (i.e., high pH: Table 4), whereas taxa including ceratopogonids, mites, tanypodinid midges, *Tanytarsus*, and harpacticoid copepods were characteristic of sites with low axis 1 scores (i.e., low pH). Fewer taxa were significantly correlated to axis 2 scores, although *Potamopyrgus*, Amphipoda and Ostracoda were indicative of sites with high axis 2 scores (i.e., swamps), and micro-Crustacea and *Chironomus* were indicative of sites with low axis 2 scores (Table 4).

Table 4: Invertebrate taxa that had significant correlations with DCA axis 1 and 2 scores (*p* < 0.001).

Axis 1		Axis 2	
Invertebrate taxa	**r^2 value**	**Invertebrate taxa**	**r^2 value**
Potamopyrgus	0.632	*Potamopyrgus*	0.53
Daphniidae	0.313	Amphipoda	0.384
Sphaeriidae	0.276	Ostracoda	0.311
Gyraulus	0.255	Sphaeriidae	0.22
Oligochaeta	0.247	*Tenagomysis chiltoni*	0.208
Paroxyethira	0.204	Chironomus	-0.259
Physa	0.204	Cyclopoida	-0.523
Orthocladiinae	-0.201	Daphniidae	-0.64
Nematoda	-0.23		
Ceratopogonidae	-0.246		
Hydroptilidae	-0.304		
Acarina	-0.468		
Tanypodinae	-0.483		
Harpacticoida	-0.547		
Tanytarsus	-0.701		

ANOSIM showed the regional grouping performed poorly in discriminating between different invertebrate communities (Table 5). This, combined with the finding of low species turnover along the ordination axes, suggested that potential regional differences played only a small part in structuring invertebrate communities in the South Island. The wetland type classification also performed relatively poorly in discriminating between the different invertebrate communities, andthe habitat classification of the water body sampled performed the worst (Table 5). The largest difference between groups was found for the 2nd division of the TWINSPAN analysis (Table 5). ANOVAs of environmental data when all sites were coded to this division showed that the greatest difference between the four TWINSPAN groups was for water quality variables, and in particular pH (Table 6). Post-hoc testing showed that pH differed between three of the four the TWINSPAN groups (Figure 2), with wetlands in groups 5 and 7 having a similar, and intermediate pH than wetlands in groups 4 (high pH swamps) or group 6 (low pH bogs). These pH differences between the biologically based TWINSPAN groups suggested that pH in particular was a driving variable influencing invertebrate communities. We thus created a new group of wetlands based on pH groups. Analysis of the upper and lower 95th percentile of pH in each of the four TWINSPAN groups showed the following pH ranges of wetlands in each group:

- pH group 1 (TWINSPAN division 4) pH less than 5.5
- pH group 2 (TWINSPAN division 5 and 7) pH between 5.5 and 6.6
- pH group 3 (TWINSPAN division 6) pH greater than 6.6

Table 5: **Results of ANOSIM showing the strength of the different *a priori* classifications explaining variation to the invertebrate communities.**

Classification		ANOSIM Global *r*-value	ANOSIM *p*-value
Region		0.149	<0.001
Wetland type		0.219	<0.001
Habitat		0.085	0.001
TWINSPAN	1st division	0.444	<0.001
	2nd division	0.540	<0.001
	3rd division	0.011	0.001
pH group		0.486	<0.001

Table 6: **Results of ANOVA showing differences in measured or derived environmental parameters between four TWINSPAN groups. Variable abbreviations as per Table 1.**

Variable	Mean Squares	F-ratio	p-value
pH	0.24	66.40	<0.001
TDP	21.41	47.25	<0.001
1km_agric	26.14	38.01	<0.001
TDN	26.11	37.64	<0.001
DRP	11.91	34.00	<0.001
IEI	0.29	28.75	<0.001
wet_agric	22.46	26.56	<0.001
WCI	0.39	24.07	<0.001
Evapo-trans	0.17	19.31	<0.001
Rainfall	17.66	17.46	<0.001
1km_ntv_tall	19.19	16.97	0.001
Cond	27.12	16.30	0.002
WEED-AG	0.34	16.16	0.005
WeedRank	0.50	15.33	0.005
SolarRadWin	0.42	14.20	0.008
AveTWarm	0.02	13.08	0.008
1km_ntv_shrt	9.32	12.72	0.01
SolarRadSum	0.06	12.30	0.011
1km_exot_tall	8.24	11.13	0.011
Wet_Cong	3.49	10.75	0.012
wet_ntv_tall	9.25	10.60	0.021
1km_Cong	7.11	10.56	0.026
1km_wetland	6.95	10.19	0.027
1km_Loess	8.55	10.09	0.043
1km_urb	1.69	8.80	0.059

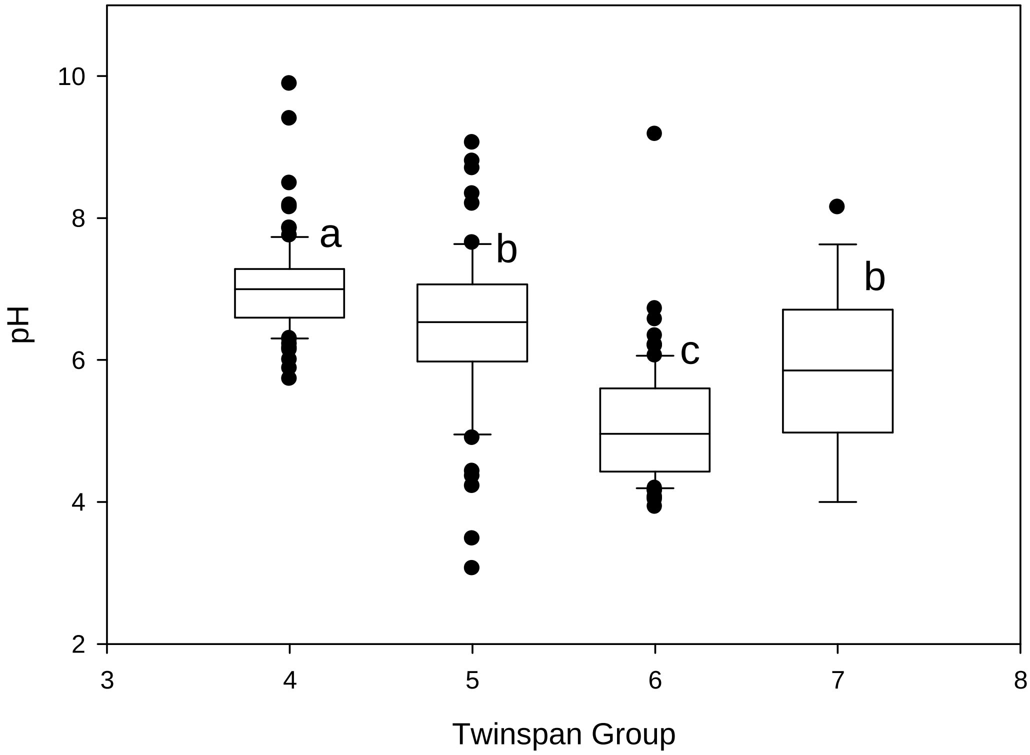

Figure 2: Differences in pH between the four TWINSPAN groups derived from the classification of the invertebrate data. Similar letters denote means that are not statistically different (Tukey's Post-hoc testing; *p* > 0.05)

All sites were assigned *a posteriori* to one of these three pH groups. ANOSIM showed that these pH derived groups did not explain as much of the variation in invertebrate communities as the biologically derived TWINSPAN 2nd division group (Table 5), suggesting that factors in addition to pH influenced the structure of invertebrate communities.

Based on these results, we regarded it as impractical to derive separate WMCI TVs for invertebrates from wetlands of particular pH, as it was evident that community composition was controlled by more than this single variable, despite it being a dominant driver. Moreover, we would be faced with the difficulty of creating TVs for invertebrates in a subset of the entire dataset, which may lead to difficulties for future sampling programs if new taxa not encountered in a particular pH-based wetland group were found. Also, we were concerned at the potential confusion that could arise should separate water bodies of different pH be found within an individual wetland. Under such a scenario, different TVs for the same taxa may have been found. Finally, how to decide which version of the WMCI to apply was likely to be a problem in situations where the pH of the wetland was unknown (or not able to be determined). Moreover, diurnal or seasonal variations in pH would have complicated the correct allocation of a wetland it its pH range were marginal between two classes. Our analysis also showed that regional differences were not particularly important in structuring invertebrate community composition. Consequently, we analysed all data from wetlands throughout the South Island to produce single WMCI TVs for each taxon.

3.4. Calculation of tolerance values (TVs)

TVs for the 122 taxa collected from the 82 wetlands stabilised after the 18th iteration of the Chessman process. These values are presented in Appendix 2, which also lists TVs for the MCI and MCI-sb where these exist for the taxa concerned[1]. Of the 122 taxa, 19 had no MCI TVs, and 25 had no MCI-sb TVs. TVs for invertebrates ranged from 1 to 10 (which is dictated by the Chessman process), but taxa with low TVs were comparatively under-represented, with only 10 taxa (8.2%) having TVs of 3 or less (Figure 3). At the other extreme, 48 taxa (39.3%) had TVs of 8 or more (Figure 3). Taxa with extreme TVs (i.e., TVs of 1 or 10) were not widely represented. The taxon with a TV of 1 (Moniidae) was recorded from only one site, and the 12 taxa with TVs of 10 were recorded at between 1 and 3 sites. In contrast, all taxa with TVs between 2 and 9 were recorded from at least one third (80+) of the sites.

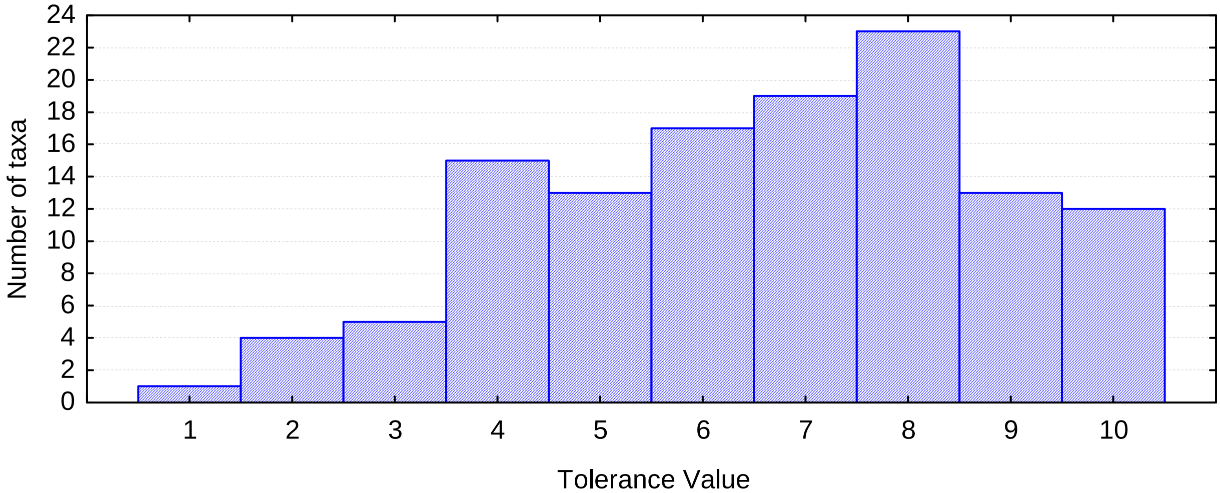

Figure 3: Histogram of taxa tolerance values.

Derived TVs assigned to each taxon from our random start were, with very few exceptions, the same as those derived by iterations started when wetlands were initially ranked either the IEI or WCI. This independent check confirmed that the Chessman process was not particularly sensitive to the initial site ranking, and produced TVs that reflected underlying gradients within the biological data.

[1] This list of TVs includes only taxa recorded from South Island wetlands. See Stark & Maxted (2007b) for a complete list of MCI and MCI-sb TVs.

Tolerance vales for 23 taxa were the same for both the WMCI and MCI, with a TVs for a further 29 taxa differing by ±1, and TVs for a further 30 taxa differing by ±2. Thus, TVs of nearly 80% of taxa in common to the WMCI and MCI were within ±2. Only 20% of the 103 taxa with TVs in common to the WMCI and MCI had TVs differing by more than ±2. The five taxa whose TVs differed the most between the WMCI and the MCI were *Cryptochironomus* (+6), Empididae (+5), Psychodidae (+5), Syrphidae (+5), and *Olinga* (-5) (Appendix 1).

Ninety-seven taxa with TVs assigned were common to the WMCI and MCI-sb (Appendix 1). However, direct comparison of WMCI and MCI-sb TVs is complicated by the fact that the latter are non-integers between 0.1 and 10.0 rather than integers between 1 and 10. Differences for 62 (64%) of them were within ±2. TVs for only 6 taxa differed by more than ±5. In all six of these cases, WMCI TVs were higher (viz., *Pycnocentrodes* +5.2, Chydoridae +5.3, *Harrisius* +5.3, *Aeshna* +5.6, *Hemicordulia* +5.6, *Onychohydrus* +7.6) (Appendix 1).

3.4.1. Discrimination of biotic scores between wetlands

WMCI scores differed significantly between WEED-AG categories, with highest WMCI values being found in wetlands with low WEED-AG scores and lowest WMCI values in wetlands with more than 75% agricultural activity within the 1 km buffer and > 50% weeds within the wetland. Smaller differences were observed between the MCI and MCI-sb and WEED-AG (Figure 4). A similar pattern was observed for the qualitative variants, where the QWMCI showed the biggest differences between WEED-AG categories, followed by the QMCI-sb. No differences were observed between the QMCI and WEED-AG (Figure 5).

3.4.2. Relationships with environmental variables

Some confirmation that biotic indices provided a realistic measure of ecosystem health was gained by correlating biotic index values with environmental variables that also measure enrichment (although strong relationships between biotic indices and spot water quality measurements were not always evident). The WMCI and QWMCI had statistically significant correlations with more environmental variables (34 and 30 respectively) than the MCI (30), MCI-sb (27), QMCI (17), or QMCI-sb (26) (Table 7). The QWMCI (especially) and the WMCI had higher inverse correlations (r_s) with wetland water quality variables such as pH, conductivity,

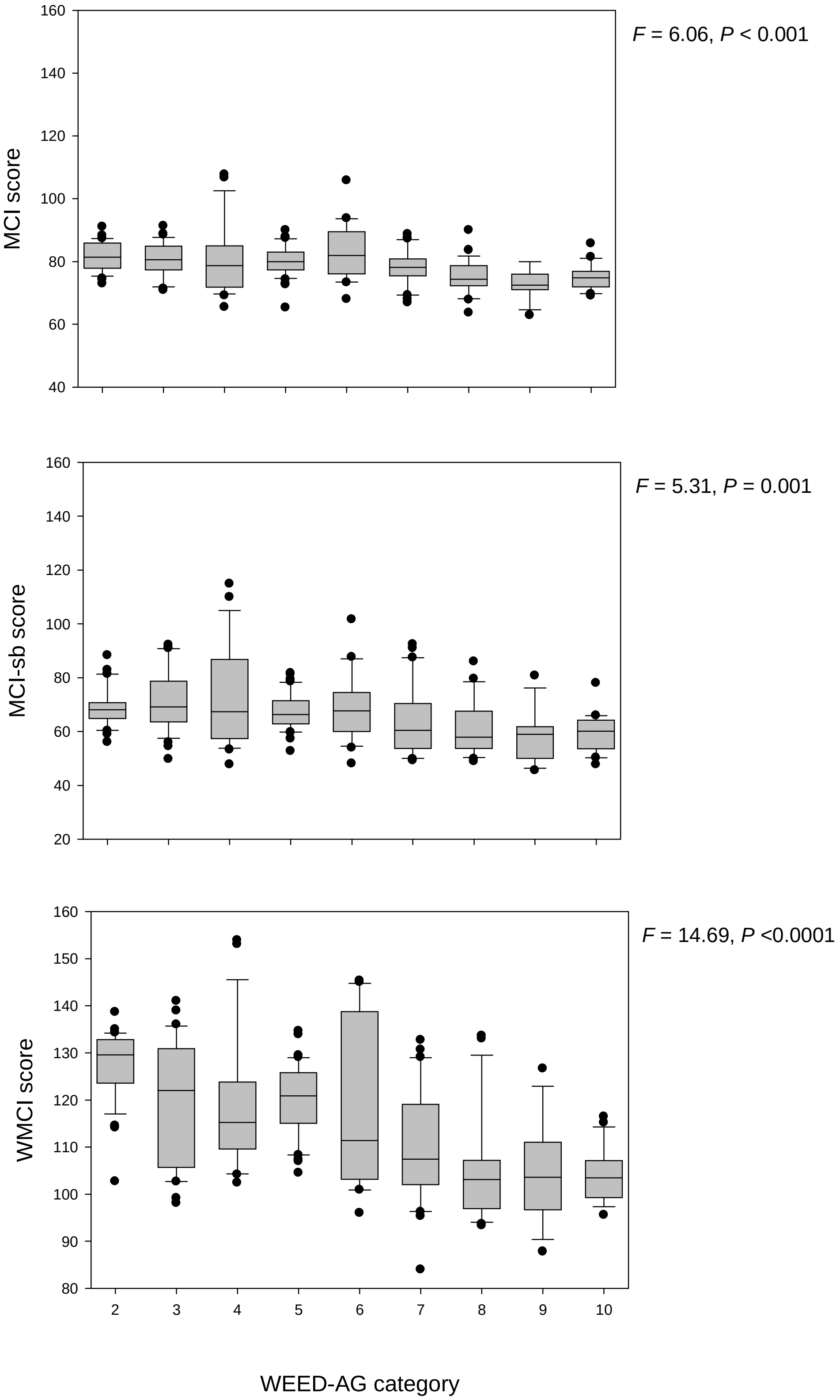

Figure 4: **Differences in calculated MCI, MCI-sb and WMCI scores between nine WEED-AG classes derived from the % of weeds within the wetland, and the % of agricultural activity in the 1 km buffer.**

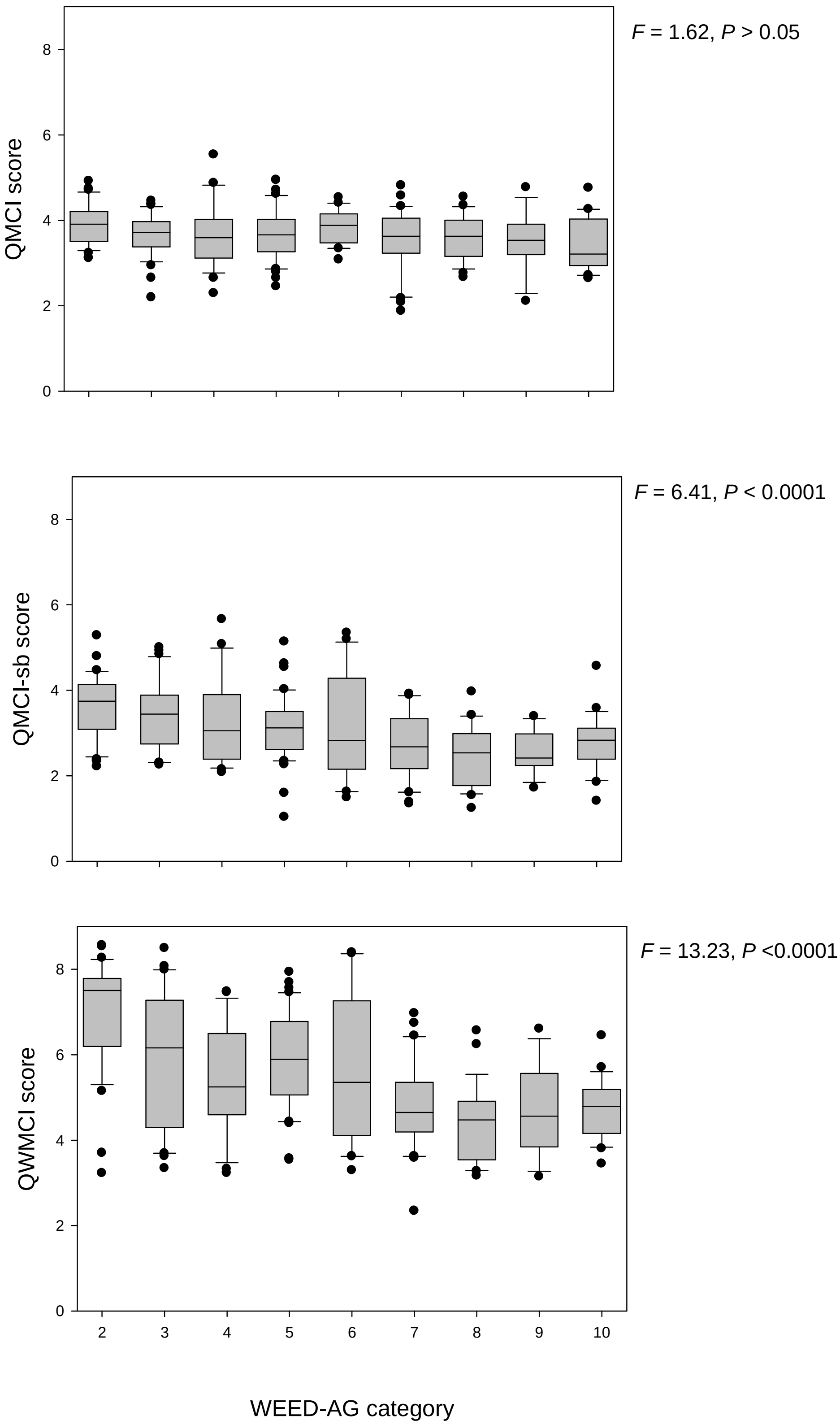

Figure 5: **Differences in calculated QMCI, QMCI-sb and QWMCI scores between nine WEED-AG classes derived from the % of weeds within the wetland, and the % of agricultural activity in the 1 km buffer.**

Table 7: List of Spearman rank correlation (r_s) values between biological metrics (MCI, MCI-sb, and the WMCI), and their qualitative variants (QMCI, QMCI-sb, QWMCI) and measured and derived environmental variables. The most powerful correlations are in bold. ns = not significant r_s values.

Variable Type	Variable	MCI	MCI-sb	WMCI	QMCI	QMCI-sb	QWMCI
Water Quality	pH	-0.444	-0.361	**-0.643**	ns	-0.505	**-0.655**
	Cond	-0.254	-0.246	**-0.515**	ns	-0.451	**-0.545**
	DRP	-0.221	-0.258	**-0.390**	ns	-0.327	**-0.425**
	NH_4	ns	ns	ns	**0.199**	ns	ns
	NO_3	ns	ns	**0.152**	ns	**0.194**	0.160
	TDN	ns	-0.251	**-0.324**	ns	-0.226	**-0.291**
	TDP	-0.231	-0.276	**-0.519**	ns	-0.467	**-0.562**
Wetland	DistSea	ns	ns	**-0.141**	**-0.154**	ns	ns
	Area	0.224	0.153	**0.307**	0.24	0.199	**0.306**
	Elevat	**0.212**	ns	ns	ns	ns	ns
	WCI	0.343	0.390	**0.549**	0.319	0.403	**0.506**
	IEI	0.357	0.347	**0.533**	0.159	0.376	**0.501**
Climate	TWarm	**-0.142**	ns	ns	**-0.207**	ns	ns
	TCold	ns	0.184	**0.208**	ns	ns	ns
	SolarSum	ns	ns	ns	**-0.151**	ns	ns
	SolarWin	ns	ns	ns	ns	ns	ns
	AnnRain	0.218	0.314	**0.493**	ns	0.317	**0.440**
	PET	ns	ns	ns	**-0.172**	ns	ns
Wetland Landuse	Wet_agric	-0.335	-0.301	**-0.507**	ns	-0.412	**-0.432**
	Wet_exot_tall	ns	-0.160	**-0.298**	ns	-0.198	**-0.31**
	Wet_exot_shrt	ns	ns	ns	ns	ns	ns
	Wet_ntv_shrt	**0.304**	0.224	0.302	**0.279**	ns	0.267
	Wet_net_tall	0.251	0.265	**0.329**	0.176	0.213	**0.257**
	Wet_urb	ns	ns	ns	ns	ns	ns
	Wet_wetland	0.142	0.172	**0.233**	ns	**0.192**	0.162
	WEED-RANK	0.326	-0.326	**-0.454**	-0.225	-0.304	**-0.376**
	AG-CLASS	ns	-0.242	**-0.269**	ns	-0.175	**-0.273**
	WEED-AG	-0.282	-0.363	**-0.466**	-0.190	-0.308	**-0.410**
1 km land-use	1km_agric	-0.367	-0.389	**-0.56**	ns	-0.43	**-0.502**

Variable Type	Variable	MCI	MCI-sb	WMCI	QMCI	QMCI-sb	QWMCI
	1km_exot_tall	-0.225	-0.237	**-0.402**	ns	-0.329	**-0.415**
	1lm_exot_shrt	ns	ns	**-0.145**	ns	-0.174	**-0.213**
	1km_ntv_shrt	0.357	0.314	**0.414**	0.265	0.208	**0.389**
	1km_ntv_tall	0.310	0.369	**0.441**	ns	0.309	**0.351**
	1km_urb	-0.279	-0.317	**-0.366**	ns	-0.271	**-0.319**
	1km_wetland	ns	0.157	**0.268**	ns	0.246	**0.252**
1km Geology	1km_Alluvium	**-0.208**	ns	ns	**-0.258**	ns	ns
	1km_Cong	0.179	**0.262**	0.180	ns	ns	ns
	1km_HSed	ns	ns	ns	ns	ns	ns
	1km_Loess	ns	-0.165	**-0.212**	ns	-0.258	**-0.268**
	1km_Meta	ns	ns	ns	ns	ns	ns
	1km_Mud	ns	ns	ns	ns	ns	ns
	1km_Peat	**0.329**	ns	0.311	**0.284**	0.171	0.265
	1km_Sand	ns	ns	ns	ns	ns	ns
	1km_SSed	0.142	**0.173**	ns	ns	ns	ns
	1km_SStone	**0.166**	ns	ns	ns	ns	ns
	1km_V_Rock	ns	ns	ns	ns	ns	ns
Wetland Geology	Wet_Alluv	**-0.23**	ns	ns	**-0.289**	ns	ns
	Wet_Cong	0.156	0.187	**0.192**	ns	ns	ns
	Wet_HSed	**-0.141**	ns	ns	ns	ns	ns
	Wet_Loess	-0.154	ns	**-0.208**	ns	**-0.250**	-0.247
	Wet_Meta	ns	ns	**-0.164**	ns	**-0.219**	-0.202
	Wet_Mud	ns	ns	ns	**0.148**	ns	ns
	Wet_Peat	**0.336**	ns	0.299	**0.256**	0.174	**0.256**
	Wet_Sand	ns	ns	ns	ns	ns	ns
	Wet_SSed	ns	ns	ns	ns	ns	ns
	Wet_SStone	ns	ns	ns	ns	ns	ns
	Wet_V_Rock	ns	ns	ns	ns	ns	ns
No. of significant correlations		30	27	34	17	26	30
No. of most powerful correlations		8	2	30	11	4	24

dissolved reactive phosphorus, total dissolved phosphorus, and total dissolved nitrogen (Table 7) than the other indices. In general, the correlations between biotic indices and forms of nitrogen (especially ammonium and nitrate) were weaker than for the other water quality variables tested (Table 7). Both the WMCI and QWMCI were highly correlated to land-use conditions within a 1 km buffer of the wetland. Significant negative correlations were observed between these indices and agriculture, exotic vegetation, and urban development, whereas positive correlations were observed with native vegetation and wetland vegetation. In addition, the WMCI and QWMCI had strong significant correlations with all land-use variables within the 1 km buffer, but not with all land-use variables within the wetland itself (Table 7). Few geological variables were correlated with any of the biotic indices. The MCI displayed the greatest number of significant correlations with geological variables (10), followed by the WMCI (7), QWMCI (5), QMCI (5), QMCI-sb (5), and MCI-sb (4). Climatic variables also generally had poor correlations with biotic indices although some of them were statistically significant (Table 7). The highest rank correlations were between rainfall and WMCI (r_s = 0.493) and QWMCI (r_s = 0.440), suggesting, not surprisingly, that the healthiest wetlands occur in wetter places.

Another measure of the performance of wetland biotic indices is their relationship to the IEI and the WCI. These indices (and the WMCI and QWMCI) are all indices of wetland condition and might be expected to be highly correlated with one another. The IEI and the WCI were significantly correlated with each other (r_s = 0.625) but they were not in perfect agreement. All the macroinvertebrate biotic indices evaluated had statistically significant rank correlations with the IEI and the WCI (Table 7), however the WMCI (r_s = 0.533 – 0.549) and the QWMCI (r_s = 0.501 – 0.506) had the strongest (positive) relationships with the IEI and the WCI. Of the two landscape-based wetland health indices, the WCI provided site rankings in better agreement to the WMCI and QWMCI than the IEI (Table 7).

The WMCI and QWMCI also performed better than other versions of the MCI when correlated to WEED-RANK, and AG-CLASS. Even stronger correlations were observed with WEED-AG, a compound variable of the two (Table 7). In all cases, the correlations were negative. That is, higher WMCI and QWMCI values were associated with wetlands that had lower levels of invasive exotic plants and lower levels of agricultural development in their surrounding catchments.

3.4.3. Relationships with other biotic indices

Spearman rank correlations presented in Table 8 compare the ability of the hard- and soft-bottomed versions of the MCI and QMCI, and the biotic indices developed specifically for wetlands (WMCI & WQMCI) to rank wetlands similarly on an environmental disturbance gradient. The highest Spearman rank correlation coefficient

(r_s) of 0.796 was between the presence-absence (WMCI) and quantitative (QWMCI) versions of the wetland MCI (in marked contrast to the much lower r_s (0.306) between the MCI and QMCI). The QMCI index seems least satisfactory at ranking wetlands on the basis of their health, given that it did not provide statistically similar site rankings to the MCI-sb, QMCI-sb, and the QWMCI (after FDR analysis). Overall, the wetland versions of the MCI provided site rankings that were more similar to their corresponding soft-bottomed versions (r_s = 0.742 & 0.762), than to the original MCI/QMCI (r_s = 0.658 & 0.135).

Table 8: Significant ($P < 0.05$) Spearman Rank Correlations (r_s) between biotic indices applied to 240 macroinvertebrate samples from wetlands in the South Island, New Zealand. ns = non-significant r_s values.

	MCI-sb	WMCI	QMCI	QMCI-sb	QWMCI
MCI	0.614	0.658	0.306	0.443	0.428
MCI-sb	X	0.742	ns	0.544	0.487
WMCI		X	0.207	0.687	0.796
QMCI			X	ns	ns
QMCI-sb				X	0.762

3.4.4. Creation of specific numeric bands to assess wetland health

Quality bands (i.e., a set of specific numeric values to assess wetland health) were developed based bands defined from the statistical distribution of QWMCI values at wetland reference sites (defined by WEED-AG = 2). We selected the QWMCI because it has a wider range of values than the WMCI, thus providing better discrimination over the disturbance gradient. QWMCI values for the 33 samples collected from these reference wetlands ranged from 3.2 to 8.6. The 25th percentile was 7.8, which, was selected as the division between the Excellent and Good quality classes (according to the method of Stark & Maxted (2007a)). The lowest observed QWMCI value was 2.3 and 50% of the range (2.3 – 7.8) was 5.05. This value, rounded to 5.0, was accepted as the division between the Fair and Poor quality classes. The division between Good and Fair was set at 6.4 (midway between 7.8 and 5.0). In summary, then, we have the following criteria for assigning wetlands to quality classes based on the QWMCI.

- Excellent: > 7.8
- Good: 6.4 – 7.8
- Fair: 5.0 – 6.4
- Poor: < 5.00

4. Discussion

4.1. Production of a single WMCI score

The aim of the study was to develop a biotic index to assess wetland condition based on the distribution of aquatic invertebrates in 82 wetlands (total of 240 sites) subject to different degrees of human pressure. The underlying assumptions of this work, and other studies using invertebrate communities to assess ecological condition, are that invertebrate taxa reflect the health of their habitats and can respond in predictable ways to changes in environmental condition.

Wetlands vary naturally, reflecting differences in hydrological conditions (e.g., hydro period, source of water), vegetation (e.g., emergent, shrub, scrub, native versus exotic), topography (e.g., riverine, domed), and soils. This variability was the rationale behind the wetland classification system of Johnson & Gerbeaux (2004), and meant that we first needed to determine whether separate biotic indices for different wetland types were required. Our results showed that although pH (and therefore wetland class) was responsible for explaining some of the variation in invertebrate communities, other variables such as wetland condition, land cover and even region also explained some of the variability in invertebrate communities. Because of the difficulty in assigning wetlands to specific classes based on wetland pH, and based on the absence of a strong classification of invertebrates based on pH, wetland type or region, we derived a single set of tolerance values (TVs) for invertebrate taxa collected from wetlands throughout the South Island. We felt that the advantages of a single scoring system far outweighed any potential losses of resolution arising from combining wetlands. A single scoring system also avoids issues such as:

- lack of confusion with multiple tables;
- difficulty in assigning wetlands to specific pH-based groups which are likely to have considerable overlap;
- problems of potentially finding water bodies within a single wetland that may differ in pH enough to require the use of two scoring tables;
- finding previously un-scored taxa if only a small number of wetlands had been used to develop TVs for subsets of the entire data set.

4.2. Performance of the WMCI

Stark and Maxted (2007a) developed the MCI-sb predominantly from surveys of Auckland soft bottoms streams, although they also used data obtained from soft bottom streams in other regions. They were able to compare the performance of the MCI-sb with that of the standard MCI, which was developed for stony bottomed streams. We developed the WMCI using the same procedure as Stark and Maxted (2007a), but we were unable to compare it with any other suitable biotic metric developed specifically for wetlands. In the absence of such comparable metrics, we compare the performance of the wetland-derived WMCI against the stream-derived MCI and the MCI-sb. Although we acknowledge that this is not the most robust comparison, it was done purely to see how our newly developed indices compared against those which have been developed for other freshwater ecosystems.

A major issue when developing biotic indices for wetland habitats is determining whether the resulting indices really do provide information on wetland condition or quality. We showed that, when applied to wetland invertebrate data, the WMCI and the QWMCI provided index values that were more similar to their respective soft-bottomed versions of the MCI than the original versions developed for assessing the health of hard-bottomed streams. This is not surprising, as habitat conditions in wetlands and soft-bottomed streams (e.g., presence of slow-flowing water, soft sediment, and accumulations of organic matter) are more similar than those in wetlands and hard-bottomed streams. Despite the apparent congruency between the WMCI and the MCI-sb, we found that the WMCI and QWMCI had stronger correlations with chemical measures of water quality or wetland enrichment (such as pH, conductivity, total dissolved phosphorus, and total dissolved nitrogen) than the MCI-sb. The importance of water chemistry, and particularly of nutrient enrichment, in influencing invertebrate communities was highlighted in a study by Suren et al (submitted), where invertebrate communities in wetlands on New Zealand’s west coast responded more to nutrient enrichment levels than either the IEI or WCI. The WMCI and QWMCI also had strong negative relationships to the degree of modified land in the 1 km buffer and the wetland, and to our composite variable WEED-AG, that combined both land-use and weediness - both of which show strong relationships to wetland condition (Ausseil et al. 2008; Clarkson et al. 2003). All these findings suggest that the WMCI and QWMCI are likely to provide useful measures of wetland health.

4.3. Comparison of WMCI and QWMCI

We found that the behaviours of both the WMCI and QWMCI to be relatively similar in differentiating between *a priori* WEED-AG categories. However, the WMCI had much stronger correlations with environmental variables and the two indices of wetland condition (WCI and IEI) than the QWMCI. Despite these stronger correlations, the WMCI had a more restricted range of scores than the QWMCI, suggesting it had a more limited ability to discriminate between wetlands of different condition. The underlying principle of the WMCI is that the existence of particular taxa at a site means it can tolerate the conditions there. Consequently, all taxa have equal influence on the result and score. However, the QWMCI relies on the principle that organisms which are most favoured by the conditions at a site will have the highest abundance, which is why the QWMCI had a greater range. Small taxa (meiofauna) such as micro- Crustacea (copepods, cladocerans, and ostracods), mites, and nematodes were more abundant at sites (both for mean and maximum density), and this greater abundance of meiofauna may have explained the greater range of the QWMCI. Meiofauna are not routinely found in biological surveys of rivers (Dole-Olivier et al. 2000; Robertson et al. 2000), reflecting either their actual absence from these environments, or the fact that they are overlooked during sample collecting and processing protocols (generally based on the use of 500 µm mesh collecting nets and processing sieves), which focus predominantly on macroinvertebrates (see Stark et al 2001). However, our sampling and processing protocols used a finer mesh (i.e., 300 µm) and examined each sample under a dissecting microscope (up to 40 × magnification), so that the meiofauna would be included. This method maximised the likelihood of encountering members of the meiofauna, which appear to make a major contribution to wetland invertebrate community biodiversity. However, examination of the TVs for these different meiofaunal taxa (see Appendix 1) showed that these small taxa had similar TVs to those of larger taxa. Thus, the abundant nature of the meiofauna in our samples would not explain the greater range of the QWMCI.

We do not recommend the WMCI for assigning wetlands to quality classes because our results showed that it had a more restricted range of values than the QWMCI (Figure 6). Given the formulae used to derive the WMCI and QWMCI, the maximum (WMCI = 180 & QWMCI =9) and minimum (WMCI = 40 & QWMCI =2) values on the axes of the graph are equivalent, so for the WMCI and QWMCI to cover equivalent ranges the regression line should pass through these points. The original MCI had a more restricted range than the QMCI (see Figure 2 in Stark 1993), although not as restricted as the WMCI. The ability of the WMCI and QWMCI to rank sites (r_s = 0.796) was slightly better than that for the MCI and QMCI (r_s = 0.70, Stark 1993).

Why does the WMCI have a more restricted range than the QWMCI? When calculating the WMCI, all taxa have equal weight, so the WMCI is effectively the average score of all taxa present at a site (multiplied by 20 – a scaling factor). To achieve a WMCI score less than 80, the average TV for the site (or sample) would need to be 4. Only 10 taxa had TVs of 3 or less, and some of these were not widely distributed (Table 10). Given that TVs are available for 122 taxa and that an average of 23.7 taxa (range 8 – 45) were recorded from South Island wetland sites, it is not surprising to find comparatively few WMCI values under 90 (Figure 6). The presence of the taxa listed on Table 9 in wetland samples (especially in high numbers) could therefore be considered indicative of wetlands of poorer than average health.

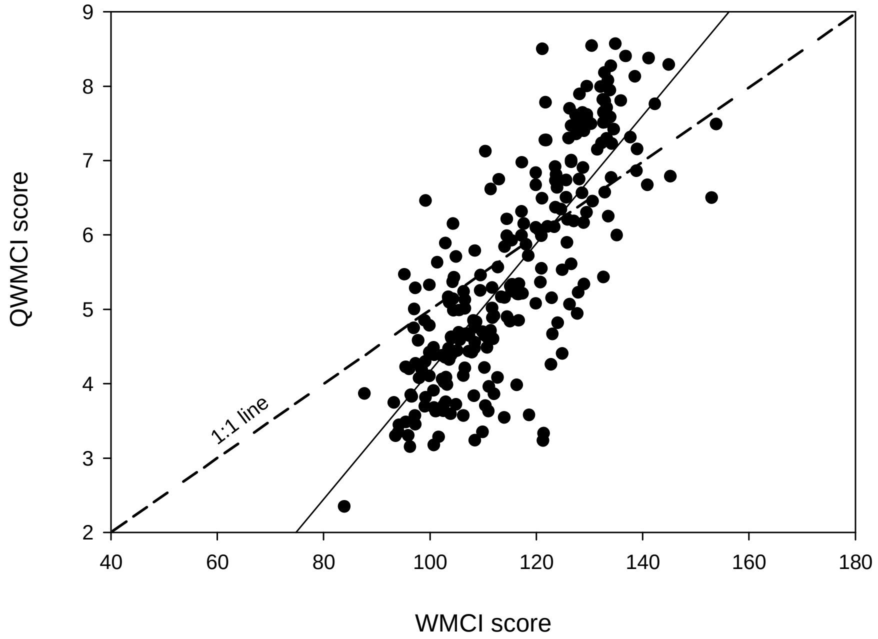

Figure 6: **Plot of WMCI and QWMCI scores from 240 wetland samples collected from the South Island. Also shown is the theoretical 1:1 line if the WMCI and QWMCI had the same range of scores.**

Table 9: **Macroinvertebrate taxa indicative of poor quality wetlands (WMCI TVs of 1 – 3). The average (Ave %) and maximum (Max %) percentage contribution to community composition and the number of sites (out of 240) they were recorded from are also shown. * = < 0.1%**

Taxon	TV	Ave %	Max %	No. of Sites
COELENTERATA				
Hydra	3	0.3	34.3	58
CRUSTACEA				
Daphniidae	3	6.4	82.8	124
Moinidae	1	0.2	51.5	1
INSECTA				
Hemiptera				
Mesoveliidae	2	*	1.5	2
Diptera				
Ephydridae	3	*	1.2	16
Paratanytarsus	2	*	0.6	1
MOLLUSCA				
Glyptophysa = Physastra	3	*	3.6	19
Gyraulus	2	1.0	32.6	74
Physa	2	1.1	49.0	101
Potamopyrgus	3	10.6	89.2	134

On the other hand, 48 taxa had TVs of 8 – 10 (Table 10). Many of these were not widely distributed – for example, 29 of them were recorded from 5 or fewer sites. The average of all TVs is 5.62, which is higher than the average TVs of both the MCI (5.17) and MCI-sb (4.72), so it is not surprising that the WMCI values were higher than either the MCI or MCI-sb. Given the fact that the theoretical WMCI scores could range from a minimum of 0 (no taxa present) to a maximum of 200 (1 taxa present with a score of 10) then the average WMCI score would be 100. The preponderance of taxa with above-average TVs makes it not surprising to find most WMCI values are higher than this average value of 100 (Figure 6).

The QWMCI, however, is biased towards the TV(s) of the dominant taxon (or taxa). Low QWMCI values (considered indicative of poor quality wetlands) less than 5.00 can be achieved when wetland invertebrate communities are dominated by taxa with TVs less than 5. Taxa such as Hydra, Daphniidae, Moiniidae, *Gyraulus*, *Physa*, and *Potamopyrgus* are all low-scoring taxa that can dominate community composition (Table 9) resulting in low QWMCI values.

At the other extreme, high QWMCI values arise when communities are dominated by taxa with high TVs. Of the 48 taxa with TVs of 8 – 10, there were several that dominated community composition at some sites, resulting in QWMCI values indicative of healthy wetlands. These taxa included Tardigrada, harpacticoid copepods, the chironomid *Tanytarsus*, and mites (Acarina). All of these taxa contributed more than 60% to community composition at some sites and most were also widely distributed (Table 10). The presence of the taxa listed in Table 10 in wetland samples (especially in high numbers) could be considered indicative of wetlands of better than average health.

4.4. Management implications

New Zealand wetlands traditionally have received less attention from freshwater ecologists than rivers and lakes (Sorrell & Gerbeaux 2004), with most work concentrating on plant communities and their role in nutrient cycling and the effects of alterations to hydrology (e.g., Sorrell et al. 2007b). Freshwater invertebrates have received much less attention (Suren et al. 2008; Suren & Sorrell 2010), despite their use in other countries to assess wetland condition. One reason for this may be the lack of protocols outlining collection and processing of wetland invertebrate samples (unlike national protocols for sampling soft and hard bottom streams (Stark et al. 2001)), logistical or practical difficulties with sample collection, as well as poor appreciation of the value of invertebrates to wetland ecosystems, and their potential role as ecological indicators of wetland health.

Table 10: Macroinvertebrate taxa indicative or good quality wetlands (WMCI TVs of 8 – 10). The average (Ave %) and maximum (Max %) percentage contribution to community composition and the number of sites (out of 240) they were recorded from are also shown. * = < 0.1%

Taxon	TV	Ave. %	Max. %	No. of Sites	Taxon	TV	Ave. %	Max.	No. of Sites
TARDIGRADA	9	0.4	62.9	34	**Diptera**				
CRUSTACEA					*Cryptochironomus*	10	*	0.1	1
Harpacticoida	9	3.8	63.0	125	*Dasyhelea*	9	0.2	13.1	36
Paranephrops	8	*	0.6	3	Empididae	9	*	1.2	6
Paratya	8	*	7.4	8	Eriopterini	8	*	0.4	4
INSECTA					Forcipomyiinae	8	*	0.7	6
Ephemeroptera					*Harrisius*	10	*	0.8	2
Austroclima	10	*	1.6	2	*Limonia*	9	*	1.0	11
Deleatidium	9	*	0.4	5	*Parachironomus*	8	*	2.0	9
Neozephlebia	8	*	4.0	10	*Paralimnophila*	9	*	1.9	14
Nesameletus	8	*	1.9	4	*Paucispinigera*	9	*	4.4	11
Zephlebia	9	*	5.7	5	Podonominae	8	*	1.4	20
Plecoptera					*Polypedilum*	8	0.8	23.2	50
Acroperla	8	*	0.2	1	Tanyderidae	8	*	*	2
Austroperla	10	*	0.1	1	Tanypodinae	8	3.0	29.9	195
Cristaperla	10	*	0.1	1	*Tanytarsus*	8	7.8	63.4	168
Megaleptoperla	8	*	0.3	4	*Zelandotipula*	8	*	3.9	16
Spaniocercoides	10	*	2.0	3	**Trichoptera**				
Odonata					*Ecnomina / Zelandoptila*	10	*	0.1	1
Aeshna	8	*	6.5	31	*Helicopsyche*	10	*	0.3	1
Hemiptera					*Hydrobiosis*	8	*	0.6	5
Saldidae	8	*	0.1	1	Oeconesidae	10	*	0.4	3
Coleoptera					*Psilochorema*	9	*	0.3	5
Berosus	8	*	0.1	1	*Pycnocentrella*	8	*	0.4	1
Elmidae	8	*	1.5	4	*Pycnocentria*	9	*	11.6	4
Hydraenidae	10	*	0.4	1	*Pycnocentrodes*	10	*	1.0	1
Onychohydrus	9	*	*	2	**ACARINA**	9	4.3	62.7	203
Ptilodactylidae	10	*	0.1	3	**MOLLUSCA**				
Scirtidae	8	0.2	17.1	33	*Hyridella*	8	*	1.2	1

As a result of this, to date, resource managers have had only two indices to assess wetland condition - the WCI and the IEI. Although these two methods appear complimentary in assessing overall wetland condition, their relevance in assessing the condition of the aquatic components of wetlands has been questioned (Suren et al 2010). We suggest that the WMCI and QWMCI provide another tool for managers to use when assessing wetland condition, or when monitoring the effects of wetland restoration activities

The creation of the MCI in the 1980s (Stark 1985) gave scientists and resource managers a simple to use biotic metric that was able to distil complex information describing invertebrate communities to a single number that reflected the nutrient status of a river. It could be argued that the widespread use of invertebrates to reflect the ecological condition of New Zealand streams has its origins with the development of the MCI, which despite development of new tools such as predictive modelling, continues to be used throughout the country. It is hoped that the existence of the WMCI will be a stimulus for resource managers and freshwater ecologists to do more work with wetland invertebrates. In particular, we encourage Regional Councils to consider monitoring wetland invertebrate communities as part of their state of the environment monitoring programmes and reporting on the results using the WMCI or QWMCI.

The results of this work were based on surveys throughout the South Island, and we are confident that the WMCI provides a useful tool to assess wetland condition throughout this area. Of the 248 samples collected from the 82 wetlands, 31% from bogs or fens, 6% from shallow water, and 62% from swamps. As part of a national survey of wetlands, a further 192 samples have been collected from 72 North Island wetlands. Only 11% of samples were from bogs or fens, while 31% were from shallow water and 56% from swamps. Examination of the invertebrate data showed that taxon richness was considerably higher in the South Island (141 taxa) than the North Island (99 taxa). Eight taxa were unique to the North Island, while 50 were were unique to the Sough Island. These faunistic differences combined with differences in wetland types, and environmental differences between the North and South Island suggest that the current WMCI scores may not work as well in the North Island. It is planned to undertake further analyses of the combined dataset to create a national WMCI score, and to see whether it outperforms the current, South Island based version.

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Appendix 1. The 82 wetlands sampled showing their region, wetland type, nature of the water body sampled, and grid references of each of the samples collected within each wetland

Region	Wetland	WetCode$	Type	Water	NZMS Easting	NZMS Northing
Canterbury	Coes Ford	CoesFd1	Swamp	Large Pond	2461991	5723665
		CoesFd2	Swamp	Large Pond	2462006	5723704
		CoesFd3	Swamp	Large Pond	2462008	5723633
	Lake Ellesmere	Ellesm1	Swamp	Lead	2458245	5712518
		Ellesm2	Swamp	Channel	2457376	5712998
		Ellesm3	Swamp	Small pond	2457897	5713213
	Otukaikino	Otukai1	Swamp	Large Pond	2481173	5752723
		Otukai2	Swamp	Large Pond	2481084	5752693
		Otukai3	Swamp	Large Pond	2481188	5752636
	Stonewall	Stonew1	Swamp	Large Pond	2335711	5590806
		Stonew2	Swamp	Large Pond	2335753	5590828
		Stonew3	Swamp	Large Pond	2335687	5590748
	Travis swamp	Travis1	Swamp	Large Pond	2485365	5746046
		Travis2	Swamp	Large Pond	2485526	5746875
		Travis3	Swamp	Large Pond	2485431	5746633
	Tutaepatu	Tutaep1	Shallow water	Large Pond	2486128	5764565
		Tutaep2	Shallow water	Large Pond	2486059	5764753
		Tutaep3	Shallow water	Large Pond	2485949	5764865
	Waimakariri wetland	Waimak1	Swamp	Large Pond	2482665	5755925
		Waimak2	Swamp	Large Pond	2482672	5755983
		Waimak3	Swamp	Large Pond	2482761	5756006
	Wetland 105	W_105S1	Swamp	Large Pond	2319895	5595258
		W_105S2	Swamp	Large Pond	2319911	5595215
	Wetland 39	W_39S1	Swamp	Large Pond	2312079	5604211
		W_39S2	Swamp	Large Pond	2312124	5604185
		W_39S3	Swamp	Large Pond	2311814	5604546
	Wetland 70	W_70S1	Swamp	Large Pond	2327140	5592996
		W_70S2	Swamp	Large Pond	2327053	5592518
		W_70S3	Swamp	Large Pond	2326994	5592511
	Wetland Francis	Franci1	Swamp	Large Pond	2323864	5593674
		Franci3	Swamp	Channel	2323711	5593680
Otago	Cannibal Bay Lower	CanLow1	Swamp	Drain	2260540	5411273
		CanLow2	Swamp	Large Pond	2260470	5411174
		CanLow3	Swamp	Drain	2260470	5411174
	Cannibal Bay Upper	CanUp1	Swamp	Large Pond	2259296	5411102
		CanUp3	Swamp	Lead	2259242	5411087
	Lake Wilkie	Wilkie2	Shallow water	Large Pond	2236828	5397120
		Wilkie3	Shallow water	Large Pond	2236813	5397190
	Longbeach	Longbe1	Swamp	Channel	2326761	5491853
		Longbe2	Swamp	Large Pond	2326470	5491719
		Longbe3	Swamp	Channel	2326506	5491662
	Lower Coutts Gully Swamp	Coutts1	Swamp	Channel	2292192	5455926
		Coutts2	Swamp	Channel	2292066	5455860
		Coutts3	Swamp	Channel	2292234	5455947
	Otanomono Wetland	Otanom1	Swamp	Large Pond	2257869	5431944
		Otanom2	Swamp	Large Pond	2258024	5431813
		Otanom3	Swamp	Large Pond	2258132	5431694

Region	Wetland	WetCode$	Type	Water	NZMS Easting	NZMS Northing
Otago	Pueroa Wetland	Pueroa1	Swamp	Large Pond	2257014	5428255
		Pueroa3	Swamp	Large Pond	2257229	5428192
	Sinclair Wetlands	Sincla1	Swamp	Channel	2284174	5465195
		Sincla2	Swamp	Large pond	2284070	5465244
		Sincla3	Swamp	Large pond	2283686	5465888
	Tahakopa Bay Podocarp Forest Wetland	TBPFW1	Swamp	Drain	2238005	5405136
		TBPFW2	Swamp	Drain	2238007	5405170
		TBPFW3	Swamp	Drain	2238005	5405064
	Tahakopa Peatbog	Tahako1	Bog	Large Pond	2235854	5402551
		Tahako2	Bog	Drain	2235837	5402451
		Tahako3	Bog	Drain	2235824	5402355
	Tairei Mouth	Tairei1	Swamp	Channel	2291485	5462960
		Tairei2	Swamp	lead	2291270	5462555
		Tairei3	Swamp	Small pond	2291345	5462310
	Tokomairiro River Swamp	Tokoma1	Swamp	Large Pond	2279789	5441051
		Tokoma2	Swamp	Channel	2279679	5441026
		Tokoma3	Swamp	Large Pond	2279580	5440962
	Waipori Boot Wetland	WaBoot1	Shallow water	Large pond	2287992	5465103
		WaBoot2	Shallow water	Large pond	2287898	5465026
	Waipori Wildlife Refuge	WaWild1	Swamp	Large pond	2286335	5464620
		WaWild2	Swamp	Large pond	2286235	5464705
		WaWild3	Swamp	Small pond	2286300	5464585
Southland	Awarua	Awarua1	Swamp	Small pond	2159434	5397414
		Awarua2	Swamp	Small pond	2159345	5397388
		Awarua3	Swamp	Large Pond	2159362	5397178
	Bayswater Bog	BayBog1	Bog	Drain	2126797	5439899
		BayBog2	Bog	Large Pond	2126803	5440170
		BayBog3	Bog	Channel	2126964	5440092
	Bluff Rd Wetland	Bluff1	Swamp	Large Pond	2155757	5403602
		Bluff2	Swamp	Large Pond	2155841	5403687
		Bluff3	Swamp	Large Pond	2155917	5403762
	Bog Burn	BogBrn1	Bog	Small pond	2137904	5456400
		BogBrn2	Bog	Drain	2137929	5456413
		BogBrn3	Bog	Small pond	2138008	5456390
	Borland Mire	Borlan1	Bog	Large Pond	2086710	5477364
		Borlan2	Bog	Large Pond	2086764	5477075
		Borlan3	Bog	Large Pond	2086266	5476933
	Drummond Wetland	Drummo2	Bog	Large Pond	2139463	5438265
		Drummo3	Bog	Large Pond	2139536	5438203
	Kepler Mire	Kepler1	Bog	Large Pond	2095194	5507435
		Kepler2	Bog	Large Pond	2095183	5507509
		Kepler3	Bog	Small pond	2095168	5507507
	Lake Cook	Cook2	Shallow water	Large Pond	2197509	5387398
		Cook1	Swamp	Drain	2197684	5387405
	LakeRakatu	Rakata1	Fen	Large Pond	2088073	5495860
		Rakata2	Fen	lead	2086198	5501282
		Rakata3	Fen	Small Pond	2086156	5501282
	Lillburn Wetland	Lillbu1	Swamp	Large Pond	2084151	5456061
		Lillbu2	Swamp	Lead	2084191	5456104
		Lillbu3	Swamp	Large Pond	2084093	5455866

Region	Wetland	WetCode$	Type	Water	NZMS Easting	NZMS Northing
Southland	Manuka Mire	Manuka1	Bog	Large Pond	2170860	5403908
		Manuka2	Bog	Large Pond	2170786	5403828
		Manuka3	Bog	Channel	2170458	5403833
	McKenzie	McKenz1	Fen	Large Pond	2112265	5635838
		McKenz2	Fen	Large Pond	2112237	5635844
		McKenz3	Fen	Small pond	2112219	5635845
	Mouat Wetland	Mouat1	Bog	Drain	2093378	5463066
		Mouat2	Bog	Small pond	2093698	5463120
		Mouat3	Bog	Drain	2093729	5463126
	Redcliffs	Redcli1	Shallow water	Large Pond	2095032	5491929
		Redcli3	Shallow water	Large Pond	2094432	5491786
		Redcli2	Swamp	Lead	2095013	5492116
	Te Koawa Wetland	Koawa1	Swamp	Large Pond	2095743	5485620
		Koawa2	Swamp	Lead	2095737	5485649
		Koawa3	Swamp	Large Pond	2095719	5485714
	TransitBay	Transi1	Fen	lead	2093328	5609404
		Transi2	Fen	lead	2093465	5609365
		Transi3	Fen	Large Pond	2093206	5609579
	Waipapa Point	Waipap1	Swamp	Drain	2194146	5387076
		Waipap2	Swamp	Drain	2194120	5387182
		Waipap3	Swamp	Drain	2194123	5387375
	Waituna Wetland	Waitun1	Bog	Channel	2178973	5398524
		Waitun2	Bog	Large Pond	2178959	5397809
		Waitun3	Bog	Large Pond	2178092	5397589
	Waiuna Lagoon	Waiuna1	Shallow water	Large Pond	2122759	5643882
		Waiuna3	Shallow water	Large Pond	2122952	5643966
		Waiuna2	Swamp	Channel	2122723	5643809
Stewart	Lake Sheila	Sheila1	Bog	Large Pond	2115130	5364497
		Sheila2	Bog	Small pond	2115107	5364467
		Sheila3	Bog	lead	2115136	5364442
	Mason	Mason1	Bog	Channel	2117059	5353988
		Mason2	Bog	Small pond	2117142	5353962
		Mason3	Bog	Small pond	2117226	5353904
	Ruggedy Flats	Rugged1	Bog	Small pond	2109412	5370250
		Rugged2	Bog	Small pond	2109396	5370261
		Rugged3	Bog	Small pond	2109433	5370270
Tasman	Deep Gully	DeepG1	Swamp	Channel	2464977	6060188
		DeepG2	Swamp	lead	2464955	6060054
		DeepG3	Swamp	lead	2464962	6060071
	LakeOtuhie	Otuhie1	Swamp	lead	2462135	6058203
		Otuhie2	Swamp	lead	2462164	6058278
		Otuhie3	Swamp	lead	2462164	6058297
	Longfords	Longfo1	Swamp	Channel	2484876	6051147
	Mangarakau	Mangar1	Swamp	Large Pond	2466270	6062270
		Mangar2	Swamp	Large Pond	2466080	6062330
		Mangar3	Swamp	lead	2466280	6062090
		Mangar4	Swamp	Small pond	2466200	6061970
		Mangar5	Swamp	lead	2466190	6062150
		Mangar6	Swamp	Channel	2466690	6062530

Region	Wetland	WetCode$	Type	Water	NZMS Easting	NZMS Northing
Tasman	Rotton Bog	Rotton1	Swamp	lead	2463103	6059071
		Rotton2	Swamp	Channel	2463323	6058519
		Rotton3	Swamp	lead	2463304	6058544
	SimonWalls	Simon1	Swamp	Channel	2485293	6048036
		Simon2	Swamp	Small pond	2485242	6047902
		Simon3	Swamp	Channel	2485377	6048071
Westcoast	BirchOne	BirOne1	Swamp	Drain	2410615	5948210
		BirOne2	Swamp	Drain	2410182	5947869
		BirOne3	Swamp	Large Pond	2409437	5947279
	BirchTwo	BirTwo1	Swamp	Channel	2409900	5946790
		BirTwo2	Swamp	Lead	2410002	5946875
		BirTwo3	Swamp	Lead	2409892	5946800
	BullMain	BulMai1	Fen	Drain	2378965	5900719
		BulMai2	Fen	Drain	2378940	5900895
		BulMai3	Fen	Large Pond	2378935	5900719
	BullThree	BulThr1	Swamp	Channel	2375825	5900177
		BulThr2	Swamp	Large Pond	2375806	5900082
		BulThr3	Swamp	Lead	2375776	5900039
	BullTwo	BulTwo1	Swamp	Large Pond	2376763	5900136
		BulTwo2	Swamp	Small Pond	2376762	5900136
		BulTwo3	Swamp	Large Pond	2376790	5900126
	Burmeister	Burmei1	Fen	Small pond	2166026	5680781
		Burmei2	Fen	Small pond	2166059	5680794
		Burmei3	Fen	Small pond	2166072	5680810
	Cape One	CapOne1	Swamp	Large Pond	2387763	5937284
		CapOne2	Swamp	Small Pond	2387763	5937284
		CapOne3	Swamp	Channel	2387786	5937207
	CapeTwo	CapTwo1	Swamp	Channel	2383322	5935402
		CapTwo2	Swamp	Lead	2383433	5935402
		CapTwo3	Swamp	Small Pond	2383627	5935490
	Cascade	Cascad1	Fen	Large Pond	2138564	5672935
		Cascad2	Fen	Large Pond	2138427	5673004
		Cascad3	Fen	lead	2138637	5672897
	Groves Swamp	Groves1	Swamp	Small pond	2341571	5816312
		Groves2	Swamp	Large Pond	2341572	5816404
		Groves3	Swamp	Small pond	2341516	5816396
	Heretaniwha	H_tani1	Fen	Large Pond	2231389	5728245
		H_tani2	Fen	Large Pond	2231312	5728234
		H_tani3	Fen	lead	2231384	5728222
	JonesCk	JonesC1	Fen	Large Pond	2408238	5946490
		JonesC2	Fen	Large Pond	2408310	5946545
		JonesC3	Fen	Small Pond	2408154	5946428
	Kakapotahi	Kakapo1	Fen	Channel	2320325	5803022
		Kakapo2	Fen	Channel	2320371	5803285
		Kakapo3	Fen	Small pond	2320389	5803179
	Kaniere	Kanier1	Swamp	Lead	2347926	5826920
		Kanier2	Swamp	Lead	2347906	5826942
		Kanier3	Swamp	Large Pond	2347997	5826969
	KaraOne	KarOne1	Swamp	Channel	2435692	6001321
		KarOne2	Swamp	Drain	2435642	6001342
		KarOne3	Swamp	Large Pond	2435255	6001190

Region	Wetland	WetCode$	Type	Water	NZMS Easting	NZMS Northing
Westcoast	Kongahu Swamp	Kongah1	Swamp	Drain	2434918	5986223
		Kongah2	Swamp	Drain	2434617	5985805
		Kongah3	Swamp	Drain	2434079	5985893
	Lake Kini	Kini1	Bog	lead	2237289	5728433
		Kini2	Bog	Large Pond	2237176	5728258
		Kini3	Bog	Channel	2236547	5728076
	Mahers	Mahers1	Swamp	Small Pond	2371340	5892290
		Mahers2	Swamp	Channel	2371430	5892372
		Mahers3	Swamp	Drain	2371565	5892338
	Mahinapua	Mahina1	Fen	Small pond	2340208	5823868
		Mahina2	Fen	Small pond	2340218	5823864
		Mahina3	Fen	Channel	2340258	5823910
		Mahina4	Fen	Channel	2340167	5823815
	MaoriLakes	MaoriL1	Fen	lead	2195340	5700370
		MaoriL2	Fen	Channel	2195310	5700571
		MaoriL3	Fen	Channel	2196188	5701410
	McCullum	McCull1	Fen	Lead	2436664	6001917
		McCull2	Fen	Lead	2436632	6001956
		McCull3	Fen	Lead	2436624	6001971
	Moana	Moana1	Swamp	Small pond	2386188	5846207
		Moana2	Swamp	Lead	2386205	5846167
		Moana3	Swamp	Channel	2386072	5846232
	Nikau	Nikau1	Swamp	Large Pond	2371803	5894576
		Nikau2	Swamp	Lead	2371770	5894594
		Nikau3	Swamp	Large Pond	2371610	5894114
	Ohinetamatea	Ohinet1	Swamp	Channel	2256088	5739808
		Ohinet2	Swamp	lead	2256108	5739839
		Ohinet3	Swamp	Small pond	2255542	5739833
	Rotokino	Rkino1	Swamp	Large Pond	2300827	5777366
		Rkino2	Swamp	Lead	2301008	5777522
		Rkino3	Swamp	Drain	2301165	5777154
	Rotomanu	Rmanu1	Swamp	Drain	2390095	5838935
		Rmanu2	Swamp	Drain	2390076	5838970
		Rmanu3	Swamp	Large Pond	2390027	5839102
	Shearer	Sheare1	Fen	Large Pond	2326465	5807782
		Sheare2	Fen	Large Pond	2326411	5807742
		Sheare3	Fen	Small pond	2326503	5807674
	Totara	Totara1	Swamp	Channel	2332843	5813438
		Totara2	Swamp	Channel	2332586	5813208
		Totara3	Swamp	Channel	2332205	5812880
	Waitangitaona	Wtaona1	Swamp	Large Pond	2291890	5779768
		Wtaona2	Swamp	Lead	2292279	5779197
		Wtaona3	Swamp	Large Pond	2292566	5778992

Appendix 2: Invertebrate taxa encountered in the 82 wetlands sampled throughout the South Island, and their WMCI tolerance values. Also shown are the TVs derived for the MCI (Stark 1985) and the MCI-sb (Stark and Maxted 2007). Taxa that were not encountered in the wetland dataset are not listed. See Stark & Maxted (2007) for the complete list of MCI and MCI-sb tolerance values.

Order	Taxa	WMCI	MCI	MCI-sb
Odonata	*Aeshna*	8	5	1.4
	Austrolestes	5	6	0.7
	Hemicordulia	7	5	0.4
	Procordulia	7	6	3.8
	Xanthocnemis	6	5	1.2
Ephemeroptera	*Austroclima*	10	9	6.5
	Deleatidium	9	8	5.6
	Neozephlebia	8	7	7.6
	Nesameletus	8	9	8.6
	Oniscigaster	7	10	5.1
	Zephlebia	9	7	8.8
Plecoptera	*Acroperla*	8	5	5.1
	Austroperla	10	9	8.4
	Cristaperla	10	8	-
	Megaleptoperla	8	9	7.3
	Spaniocercoides	10	8	-
	Taraperla	7	7	8.3
	Zelandobius	6	5	7.4
Hemiptera	*Anisops*	5	5	2.2
	Corixidae	7	-	-
	Diaprepocoris	7	5	4.7
	Mesoveliidae	2	-	-
	Microvelia	6	5	4.6
	Saldidae	8	5	3.9
	Sigara	4	5	2.4
Trichoptera	*Ecnomina/Zelandoptila*	10	8	7
	Helicopsyche	10	10	8.6
	Hudsonema	6	6	6.5
	Hydrobiosis	8	5	6.7

Order	Taxa	WMCI	MCI	MCI-sb
Trichoptera	Hydroptilidae	6	-	-
	Oecetis	7	6	6.8
	Oeconesidae	10	9	6.4
	Olinga	5	9	7.9
	Oxyethira	6	2	1.2
	Paroxyethira	5	2	3.7
	Plectrocnemia	6	8	6.6
	Polyplectropus	7	8	8.1
	Psilochorema	9	8	7.8
	Pycnocentrella	8	9	-
	Pycnocentria	9	7	6.8
	Pycnocentrodes	10	5	3.8
	Triplectides	5	5	5.7
	Triplectidina	6	5	-
Lepidoptera	*Hygraula*	5	4	1.3
Coleoptera	*Antiporus*	4	5	3.5
	Berosus	8	5	-
	Elmidae	8	6	7.2
	Hydraenidae	10	8	6.7
	Hydrophilidae	6	5	8
	Lancetes	6	-	-
	Liodessus	5	5	4.9
	Onychohydrus	9	5	0.4
	Ptilodactylidae	10	8	7.1
	Rhantus	6	5	1
	Scirtidae	8	8	6.4
	Staphylinidae	4	5	6.2
Diptera	*Austrosimulium*	6	3	3.9
	Ceratopogonidae	7	3	6.2
	Chironominae	6	2	3.8
	Chironomus	4	1	3.4
	Cladopelma	5	-	-
	Corynocera	4	-	-
	Corynoneura	4	2	1.7

Order	Taxa	WMCI	MCI	MCI-sb
Diptera	*Cryptochironomus*	10	3	-
	Culicidae	4	3	1.2
	Dasyhelea	9	-	-
	Empididae	9	3	5.4
	Ephydrella	3	4	1.4
	Eriopterini	8	9	7.5
	Forcipomyiinae	8	-	-
	Harrisius	10	6	4.7
	Kiefferulus	6	-	-
	Limonia	9	6	6.3
	Molophilus	7	5	6.3
	Muscidae	6	3	1.6
	Neolimnia	5	3	5.1
	Orthocladiinae	7	2	3.2
	Parachironomus	8	-	-
	Paradixa	6	4	8.5
	Paralimnophila	9	6	7.4
	Paratanytarsus	2	-	-
	Paucispinigera	9	6	7.7
	Podonominae	8	8	6.4
	Polypedilum	8	3	8
	Psychodidae	7	1	6.1
	Stratiomyidae	4	5	4.2
	Syrphidae	7	1	1.6
	Tabanidae	5	3	6.8
	Tanyderidae	8	4	5.9
	Tanypodinae	8	5	6.5
	Tanytarsus	8	3	4.5
	Zelandotipula	8	8	7
COLLEMBOLA		7	6	5.3
Crustacea	AMPHIPODA	7	5	5.5
	Calanoida	5	5	2.4
	Chydoridae	7	5	0.7
	Cyclopoida	4	-	-

Order	Taxa	WMCI	MCI	MCI-sb
Crustacea	Daphniidae	3	-	-
	Harpacticoida	9	-	-
	Ilyocryptidae	7	-	-
	ISOPODA	6	5	4.5
	Macrothricidae	7	-	-
	Moinidae	1	-	-
	Ostracoda	5	3	1.9
	Paranephrops	8	5	8.4
	Paratya	8	5	3.6
	Tenagomysis	5	-	-
Acarina	Acarina	9	5	5.2
Gastropoda	*Ferrissia*	4	3	2.4
	Glyptophysa	3	5	0.3
	Gyraulus	2	3	1.7
	Lymnaea	4	3	1.2
	Physa[1]	2	3	0.1
	Potamopyrgus	3	4	2.1
Bivalvia	Echyridella[2]	8	3	6.7
	Sphaeriidae	4	3	2.9
Oligochaeta		4	1	3.8
Hirudinea		4	3	1.2
Platyhelminthes		4	3	0.9
Nematoda		7	3	3.1
Hydra		3	3	1.6
Tardigrada		9	-	-

[1]according to the checklist available at http://www.molluscs.otago.ac.nz/, *Physella* is back to being *Physa* in NZ.

[2] *Hyridella* is now *Echyridella.*