THE IMPACT OF URBAN DEVELOPMENT AND HABITAT FRAGMENTATION ON AQUATIC INVERTEBRATE COMMUNITIES IN REMNANT WETLANDS: A CHRISTCHURCH CASE STUDY

A thesis presented in partial fulfillment of the requirements for the degree of Master of Applied Science in Natural Resource Management at Massey University, Palmerston North

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Abstract

The coastal plains that are now occupied by Christchurch City, in the South Island of New Zealand, were once dominated by palustrine and estuarine wetland systems. These wetlands were almost completely drained over a 100-year period in order to allow the construction of the city and to provide arable land for farming. However, remnants of the original wetlands have been preserved and are scattered throughout the present metropolitan area. Most of these are small riparian wetlands associated with Christchurch's many streams and three major river systems. In addition, there are also several large remnant wetland reserves that each cover many hectares. These remnant wetlands experience a range of environmental pressures from adjacent urban development, including stormwater discharge, landscaping, flood control, the presence of dense housing, pressure from introduced plants and insects, and more recently, wetland enhancement programmes.

This study investigated the impact of urban development and habitat fragmentation on remnant urban riparian wetlands primarily by comparing the aquatic invertebrate communities that they support, with the same communities in three unmodified 'natural' wetlands associated with lowland streams flowing through native tussock and scrubland. A range of physical parameters (water clarity, conductivity, pH, temperature) were also measured. Three wetlands of a similar type and size located in pastoral grazing areas, and three artificially constructed urban wetlands, were also assessed to provide additional points of reference.

The unmodified wetlands exhibited slightly higher species richness and abundance when compared to the remnant urban wetlands. However, this difference was not statistically significant ($p>0.05$). The unmodified wetlands showed significantly higher species richness than both the constructed and pastoral wetlands ($p<0.01$). Both of these highly modified wetland types contained large numbers of dipterans and molluscs, whereas the unmodified and remnant wetlands contained higher proportions of coleoptera and hemiptera. Significant differences were also detected between some of the pH, water clarity and temperature levels measured in the various wetland types.

The effect of wetland size was also measured by comparing the invertebrate faunas in small, medium and large remnant fragments. Although lower macroinvertebrate abundance and species richness was observed in the small fragments, no statistically significant difference was detected between the three fragment sizes ($p>0.05$). There was also no significant difference between the unmodified wetlands and the remnant fragments.

It was concluded that fragment size did not have a significant effect on the aquatic invertebrate communities in remnant urban wetlands, and adjacent urban development did not have a significant adverse impact on remnant urban wetlands when compared to natural wetland systems. Intensive pastoral grazing had a significant and quite severe effect on wetland systems, probably due to eutrophication and sediment wash-off. Artificially constructed wetlands contained significantly lower species richness than natural wetland systems. Remnant urban fragments appear to be resistant to the effects of urbanisation, and are considered to be suitable habitats for preserving native aquatic biodiversity in urban areas.

Acknowledgements

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I wish to dedicate this report to my amazing, wonderful wife Jennifer. Not only was she a very capable field assistant, she has never once complained about the many hours I have recently spent tapping on a keyboard, or peering down a microscope. She has been a constant source of love, strength and support.

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1.0 Introduction

1.1 *Importance and Function of Wetlands*

Wetlands have been described as one of the world's most biologically productive and diverse environments (Hemond and Benoit 1988). They provide transitional zones between land and water environments, and their biological productivity is comparable to that of tropical rainforests (Whigham 1999). They are havens for biodiversity, providing the water and primary productivity on which many highly adapted species of plants and animals depend for survival. Wetlands perform many useful functions, including mitigating flood events, recharging aquifers, and improving water quality by trapping sediment and filtering nutrients and pollutants (Makaloff 1988; Mitsch *et al.* 1998).

There are numerous different definitions for the diverse range of wetland types. The definition used in this study is the same as that used in the New Zealand Resource Management Act (1991):

> "*permanently or intermittently wet areas, shallow water, and land water margins that support natural ecosystems of plants and animals that are adapted to wet conditions*" (New Zealand Government 1991).

Another useful definition is given by Johnson and Brooke (1989) in *Wetland Plants in New Zealand*:

> "*A collective term for permanently or intermittently wet land, shallow water and land-water margins. Wetlands may be fresh, brackish or saline, and are characterised in their natural state by plants and animals that are adapted to living in wet conditions.*"

1.2 *New Zealand's wetland history and loss*

Wetlands once covered extensive areas of New Zealand. They accounted for more than 20% of New Zealand's total land area and covered more than 670,000 hectares (McLay 1976; Ministry for the Environment 1997). A wide range of wetland types were present, including coastal lagoons, estuaries, bogs, fens, marsh and riparian sedges. Coastal and lowland wetlands contained more biodiversity, and were more productive, than almost any other New Zealand ecosystem with the exception of some native forests (Ministry for the Environment 2000a). They supported a prolific range of plant species, large bird populations, and were a vital part of the life cycle of many fish species.

Because of the resources provided by wetlands, many Maori tribes settled primarily around coastal estuaries and lagoons (Stephenson 1983). Flax (*Phormium* sp.) was used for weaving and the abundant populations of waterfowl were hunted for food. Early European colonisers also exploited wetlands for food and developed farms on the adjacent fertile plains. However, as the rate of European settlement increased rapidly at the end of the 19th century, many wetland areas were drained to provide pasture for farming, flood control and urban development (McLay 1976; Ministry for the Environment 1997). These wetland areas yielded exceptionally fertile soil when drained. The greatest wetland losses occurred mostly between 1920 and 1980 as New Zealand was transformed from a diverse mosaic of native forest, wetlands and flood plains, into a pastoral farming ecosystem (Ministry for the Environment 2000a). Vast areas of native forest were also cleared to provide additional area for agricultural production.

This dramatic modification of the New Zealand environment was accompanied by an influx of exotic animals and plants. Many of these have since become significant pests (Parliamentary Commissioner for the Environment 2001). Drainage, dam construction and flood control have completely altered the natural character and behaviour of most of New Zealand's large waterways (Ministry of Agriculture and Forestry 1993; Stephenson 1983). Many of the natural processes that originally formed wetlands have ceased. Rainfall now rapidly enters a network of drains and ditches and then flows into river systems, and out to sea. Significant groundwater abstraction for irrigation has also lead to a lowering of the water table in many areas, which has contributed to wetland degradation and loss (Ministry for the Environment 1997; McLay 1976).

As a result, more than 90% of New Zealand's original wetland area has been completely drained. The total area of freshwater wetland that remains is estimated to be less than 100,000 hectares (Cromarty and Scott 1996; Stephenson 1983) (Figure 1). In addition, most of the remaining wetlands have been significantly modified. Unmodified wetlands are now mostly restricted to alpine tarns and high country fens. This process has continued until quite recently. Between 1978 and 1983 approximately 15 percent (3,175 hectares) of the remaining wetland areas had been drained (Ministry for the Environment 1997). Until the mid-1980s farmers were encouraged to drain wetlands by Government subsidies (Stephenson 1983).

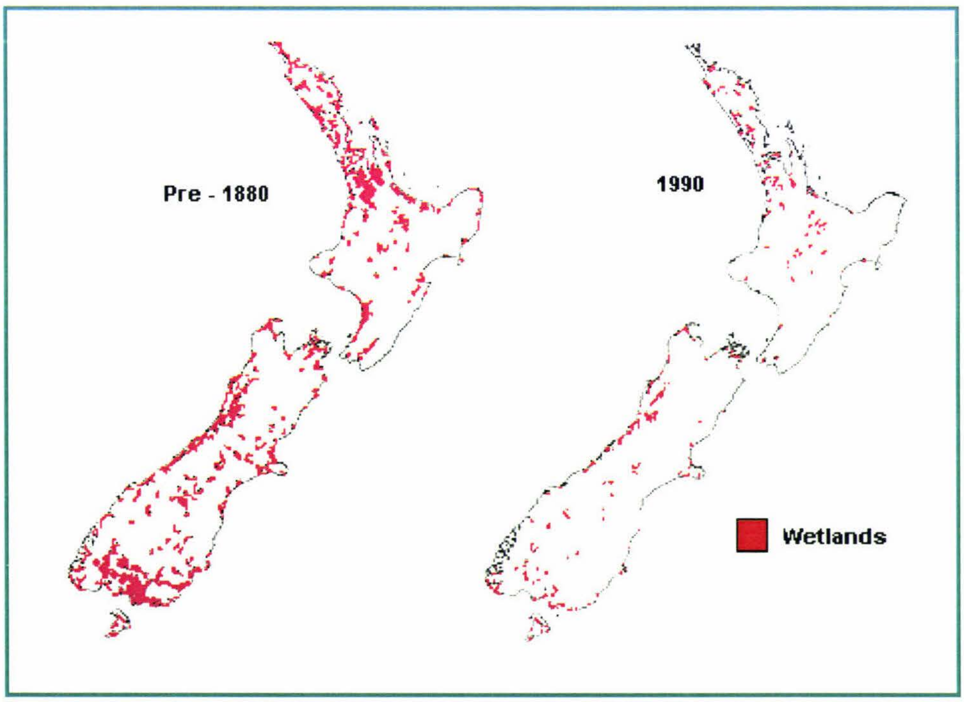

Figure 1: New Zealand wetland loss 1880* - 1995
(source: New Zealand State of the Environment Report, 1997)
(* inferred from soil types)

Although thousands of individual wetlands remain, they are highly fragmented and 75% are less than one hectare in area (Stephenson 1983). Wetlands are now widely regarded as one of New Zealand's most threatened ecosystems (Ministry for the Environment 1997). Due to this habitat loss and degradation, many native plants and animals in wetlands are threatened.

However, in recent years there has been a growing awareness of the crisis facing New Zealand's wetlands, and their value as reservoirs of biodiversity and remnant natural ecosystems. New Zealand became a signatory to the RAMSAR Convention on Wetlands of International Significance in 1976 and six of the largest remnant wetlands have been designated as 'wetlands of international significance' and have been afforded special protection (Ministry for the Environment 1997).

A recent renaissance in Maori culture has also contributed to this new ecological awareness. According to Maori traditional values, water is perceived as having spiritual power and is the 'life-force' of the land. Waterways and wetlands are regarded as spiritual entities, as well as a source of food and other resources. Hence any destruction or degradation of wetlands is regarded as violating a sacred area (Ministry for the Environment 2000a). The Resource Management Act, introduced in 1991, gave regional councils responsibility over water resources in their jurisdiction, and required Maori values and concerns to be taken into account in the management of these resources. Maintaining the natural character of remnant wetlands is identified in the Resource Management Act as a matter of national importance (New Zealand Government 1991).

1.3 *Remnant wetland areas in Christchurch*

Christchurch is the largest city in the South Island of New Zealand (Appendix A - figure 1). The metropolitan area covers more than 14,000 hectares, of which more than 3,000 hectares consists of parks and reserves (Christchurch City Council 1999). This includes 13 major metropolitan parks and more then 350 small urban parks and reserves. The population of the city at the beginning of 2002 was estimated to be 332,000, and this has been growing by approximately 0.8% per year over the past decade (Christchurch City Council 2002b). Recently there has been an increase in urban development, with many new subdivisions being developed on the edge of the city.

The city is sited on the eastern edge of a series of merged alluvial fans formed by large braided shingle rivers that flow from the Southern Alps. The different layers of shingle deposited by these rivers have formed a series of aquifers under the Canterbury Plains (Environment Canterbury 2001). Rainfall enters these aquifers in the foothills around the Alps and flows eastward until it discharges through a large number of coastal springs. As a result, most of the area now occupied by Christchurch was dominated by vast palustrine wetlands associated with springs and streams, and estuarine wetlands around the coastal lagoons and river mouths (Environment Canterbury 2002) (Figure 2). The wetlands were interspersed with large stands of indigenous forest and sand dunes. An early settler, Dr A. C. Barker, described the area in 1850:

"This great plain is copiously watered with rivers, two of which the Heathcote and Avon are small, but deep, larger than the Avon at Rugby but as clear as crystal. The banks are everywhere bordered with luxuriant growth of flax. Near the swamps there is a great deal of grass called `toi-toi' (Cordyline indivisa), very handsome, with its flowers often twelve feet high. The woods are full of beautiful birds, especially the bronze pigeon (Hemiphaga novaeseelandiae), which is very plentiful. Fish in the rivers are few and the best are the eels and flounders which are excellent." Dr A. C. Barker, 1850 (Christchurch City Council 2003).

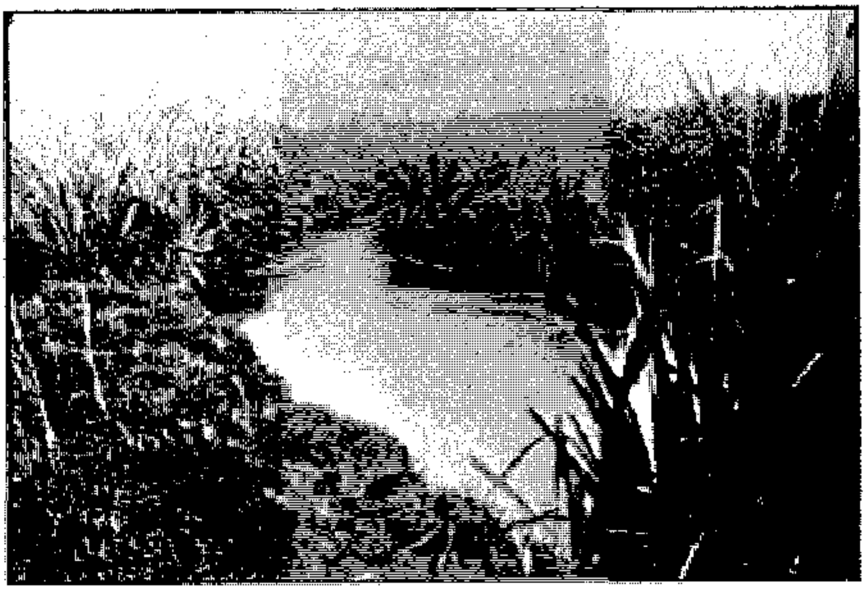

Figure 2: Avon River in 1860. (source: CCC website)

The original township was also subject to frequent flooding from the nearby Waimakariri River and other smaller waterways such as the Avon, Heathcote and Styx Rivers (Christchurch City Council 2000). One of the primary objectives of the early European settlers was to straighten and channel most of the local waterways, and to drain and remove as much water from wetland areas as possible. This resulted in the complete removal of more than 95% of the wetlands within the area now occupied by the city (Christchurch City Council 2003). More than 400 kilometres of drains and pipes flow from the suburbs, primarily into the three local urban rivers (Christchurch City Council 1999). In addition to straightening and channelling, many small streams have been piped underground.

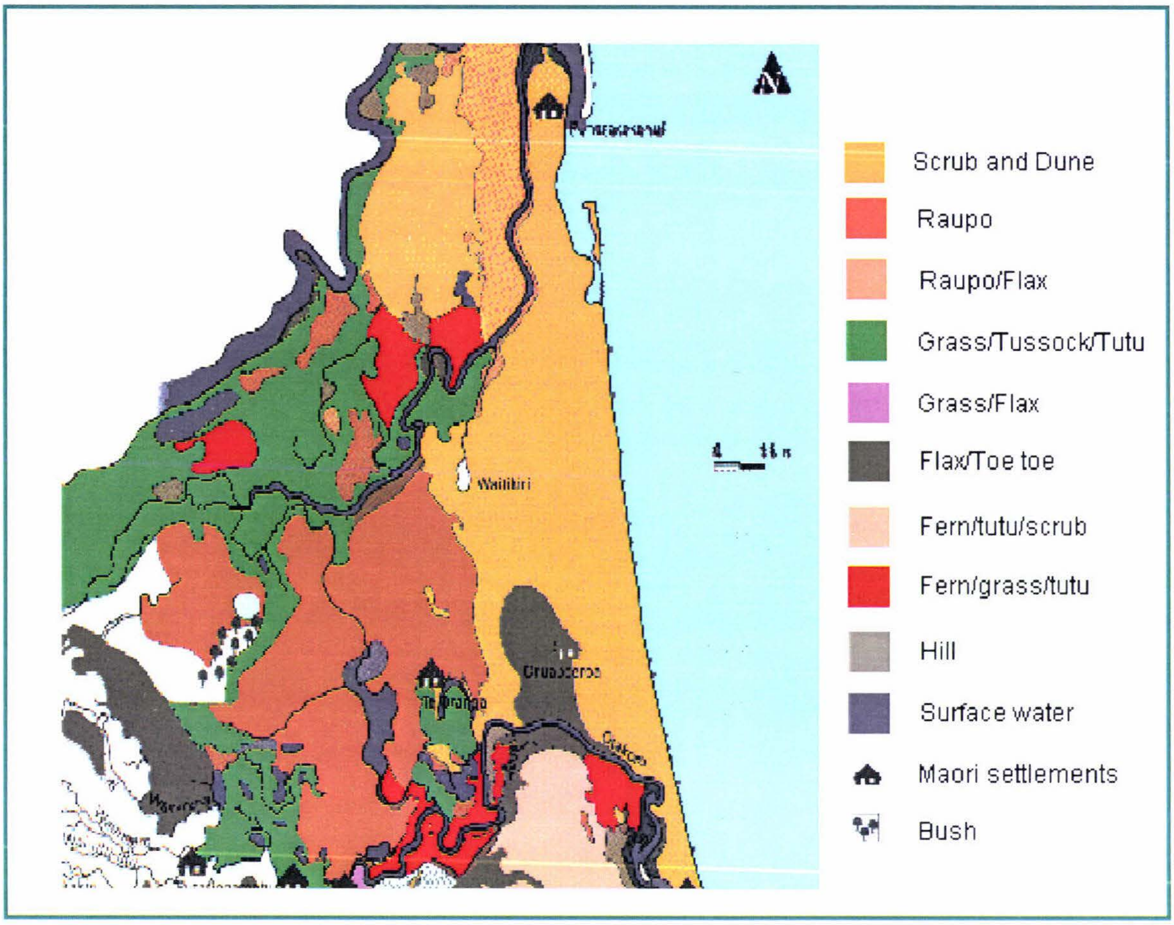

Figure 3: Pre-european vegetation in Christchurch (source: CCC website)
(Raupo and flax areas were mostly likely wetland systems)

The removal of most wetland habitat and associated waterways has had a catastrophic effect on wetland species in the area. Native wetland plants, such as raupo (*Typha orientalis*), matai (*Prumnopitys taxifolia*), houhere (*Hoheria populnea*), which formerly dominated the landscape are now restricted to a few remnant wetland reserves and stream margins (Christchurch City Council 2003). Wetland bird populations are also a fraction of their estimated numbers prior to European settlement (Environment Canterbury 2001). Pukeko (*Porphhyrio porphyrio melanotus*) were once abundant throughout the area but now only one significant population remains in Travis Wetland. Other birds, such as the White-faced Heron (*Ardea novahollandiae*), and native Kingfisher (*Halcyon sancta vagans*), are now rarely seen (MacFarlane *et al.* 1999).

However, several large wetland areas, and many smaller fragments, have escaped complete destruction. Although most of these systems have been significantly modified by urbanisation they still provide a refuge for many specialised wetland species (Christchurch City Council 2003). Travis Wetland covers an area of 116 hectares and is considered to be the remaining wetland fragment that most closely resembles the original ecosystem in the region, and contains many nationally rare plants and invertebrates (Christchurch City Council 1990). Almost 80% of pre-European wetland plant species and 77% of the native wetland birds of lowland Canterbury have been recorded in this wetland seen (MacFarlane *et al.* 1999).

1.4 *Christchurch wetland restoration initiatives*

Recently there has been a growing realisation of the historical, ecological and recreational value of Christchurch's wetland areas, and the adverse impact that anthropogenic pressures have on the native flora and fauna that they support (Christchurch City Council 2000; Environment Canterbury, 2001). Christchurch City Council has produced a draft Biodiversity Strategy for the city in which they state one of their primary environmental objectives is to "*protect and restore ecological heritage areas that still contain elements of Canterbury's original natural character and provide habitats for our unique and precious biodiversity*" (Christchurch City Council 2003). A total of 300 sites within the city have been identified by the City Council as 'Ecological Heritage Sites' because they contain remnants of Christchurch's original ecosystem (Christchurch City Council 2002a). These are mainly stands of native bush but also include a range of historical wetland sites of various sizes.

The Council has developed an active programme of protection and restoration of remnant wetland sites, aimed primarily at the largest remaining fragments. Travis Wetland was purchased by the Council in 1989 at market value to prevent the area being sold to a property developer. It has since been recognised as an 'Ecological Site of National Importance' under Section 7 of the Reserves Act 1977 and has been renamed as the Travis Wetland Nature Heritage Park (MacFarlane *et al.* 1999). An intensive replanting and naturalisation programme has been conducted with the aim of re-establishing wetland forest habitats in order to re-introduce locally extinct native birds (Christchurch City Council 1990). Other large remnant wetland areas include Styx Mill Reserve, Bexley Wetland, Heathcote Wetland Reserve, Horseshoe Lake and Otukaikino Wetland. These areas all have a continuous wetland history but have been impacted by degraded water quality, landscaping, flood control, and invasive pests, to a greater or lesser extent. Wetland management plans have been developed for many of these sites (Christchurch City Council 1992; Christchurch City Council 1993).

There has also been a recent trend to construct artificial wetlands, often as water features in new subdivisions, in areas where wetlands have not previously existed. However, many studies have shown that these wetlands often lack the hydrological cycles required to establish a healthy wetland ecosystem and can quickly turn into stagnant bogs (e.g. Malakoff 1998; Zedler 2000; Zedler & Callaway 1999).

2.0 Land-use effects on wetlands and associated waterways

2.1 *Introduction*

The global human population has more than doubled over the past fifty years, which has lead to a dramatic increase in human-induced pressures on the environment (Rotheram 1999; Booth, 1997; Ministry for the Environment 2000b). There has also been an associated wide-spread and accelerating replacement of unmodified natural systems by anthropogenic systems (Schall 1978). This has been particularly evident in third-world countries where indigenous forests and wetlands have been cleared and converted into farmland in order to feed their burgeoning populations. In New Zealand, agricultural and forestry production systems now cover almost 60% of the countries total land area (Figure 4 and Appendix A - figure 3). Urban landscapes account for a further 8% (Ministry for the Environment 1997). These modified ecosystems have a range of direct and indirect impacts on adjacent wetland areas. The intensity of these impacts will depend on the relative size and proximity of the wetland, and the amount of interaction between the two systems.

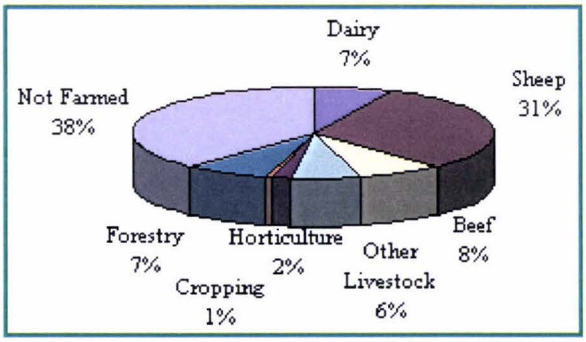

Figure 4: Agricultural land use in New Zealand (MfE 1997)

2.1.1 *Impacts of agriculture on wetlands*

Approximately 50% of New Zealand's land area is used for pastoral agriculture, and hence due to the scale of this activity, it is widely considered to be the greatest source of environmental pressure on wetlands (Ministry of Agriculture and Forestry 1993;

Hamill and McBride 2003; Wilcock 1986). The acquisition of arable land was the driving force behind the draining of more than 500,000 hectares of New Zealand's wetlands (Ministry for the Environment 1997). Flood control works to protect pastoral land have also had significant adverse effect on wetlands. Large scale drainage schemes, which continued well into the 1970's, have almost ceased following the release of a 'National Wetlands Policy' by the Government in 1986 which provided protection to most remaining wetlands areas (Ministry for the Environment 1997).

However, agriculture continues to have a significant adverse impact on the water quantity and quality in surface waterways and groundwater (Ministry of Agriculture and Forestry 1993; Hickey and Rutherford, 1986). Recent conversions from sheep (*Ovis aries*) and beef (*Bos* sp.) grazing to intensive dairy farming have exacerbated these impacts. The organic wastes produced by agricultural livestock are approximately 40 times greater than those produced by New Zealand's human population (Ministry of Agriculture and Forestry 1993). Much of this excreta is washed off pasture into waterways, which may intermittently support wetlands, or drain into wetlands or lagoons. Livestock grazing adjacent to wetland areas can also have adverse effects on unprotected wetland systems. Large animals can severely damage soil and vegetation by grazing and trampling.

Sediment from soil erosion, fertilisers and pesticides are also washed off or leached into waterways. This has resulted in the eutrophication of many streams that pass through farmland, and rising nitrate and phosphate concentrations in groundwater (Hamill and McBride 2003; Burden 1982). In turn, the water entering many wetlands contains increasing amounts of organic nutrients and sediments. This can cause many effects including rapid increases in the populations of pollution-tolerant invertebrates (e.g. mosquito (Culicidae) larvae) which can displace other species, and algal blooms that can deoxygenate the water resulting in a putrid anaerobic bog (Hedmond and Benoit 1988). Suspended sediments entering wetlands can smother aquatic plants and invertebrates, and reduce the amount of light reaching benthic vegetation.

Duggan *et al.* (2002) compared lowland streams in Waikato, an area that has been highly modified and intensively farmed, with similar streams in Westland, a region that has undergone much less anthropogenic modification and still retains large areas of native forest. They found that the Westland streams had lower nutrient concentrations, conductivity, turbidity, and suspended sediment loading. The authors concluded that land-use intensity on an ecoregion scale is an important factor influencing the chemistry of streams and the composition of aquatic invertebrate communities.

Hamill and McBride (2003) measured six water quality parameters at 29 sites throughout Southland, New Zealand from 1995 to 2001. This region experienced a dramatic increase in the conversion from sheep grazing farms to dairy farms over this period. The results of this study showed that waterways in catchments converted to dairying experienced significant increases in dissolved nitrogen and phosphorous and a decrease in dissolved oxygen. Quinn and Stroud (2002) found that total suspended solids, nitrogen and phosphorous exports from streams located in agricultural areas were between 3 and 7-fold higher than streams draining from native forest over a five year period. Streams in pasture were also shown to have much greater variation in water quality attributes than forested streams, presumably due to sudden run-off of nutrients caused by heavy rainfall.

As streams pass from high-country areas through farmland to the sea, the water quality has been shown decline measurably and cumulatively (Collier 1995a; Quinn and Stroud 2002). A large number of remnant wetland areas are located in lowland and coastal areas, and hence they are at greater risk of eutrophication than wetlands in other areas. This could be reflected by the results of the 1993-1994 New Zealand Mosquito Survey conducted Laird (1995) which found that lowland stream ponding was the second most utilised mosquito breeding habitat, and were often the dominant species in pastoral wetlands.

2.1.2 *Impacts of urban development on wetlands*

Most of the largest urban populations in New Zealand are sited in areas where extensive wetland systems previously existed (e.g. Auckland, Wellington, Christchurch, Nelson, Invercargill) (Ministry for the Environment 1997). Within many of these cities the remnants of the original wetlands can still be found. Urban development around wetland areas has a range of direct and indirect impacts. These impacts include pollution from stormwater, drainage, stream channelling, pest and weed invasion, land reclamation, infilling, road construction, and landscaping (McConchie 1992; Finkenbine *et al.* 2000; Booth and Jackson 1997).

New Zealand's metropolitan area has been increasing steadily during the past several decades (Figure 5), and much of this urban development has involved the infilling of open spaces previously used as reserves or for farming and horticulture, and the construction of new subdivisions of the fringes of cities.

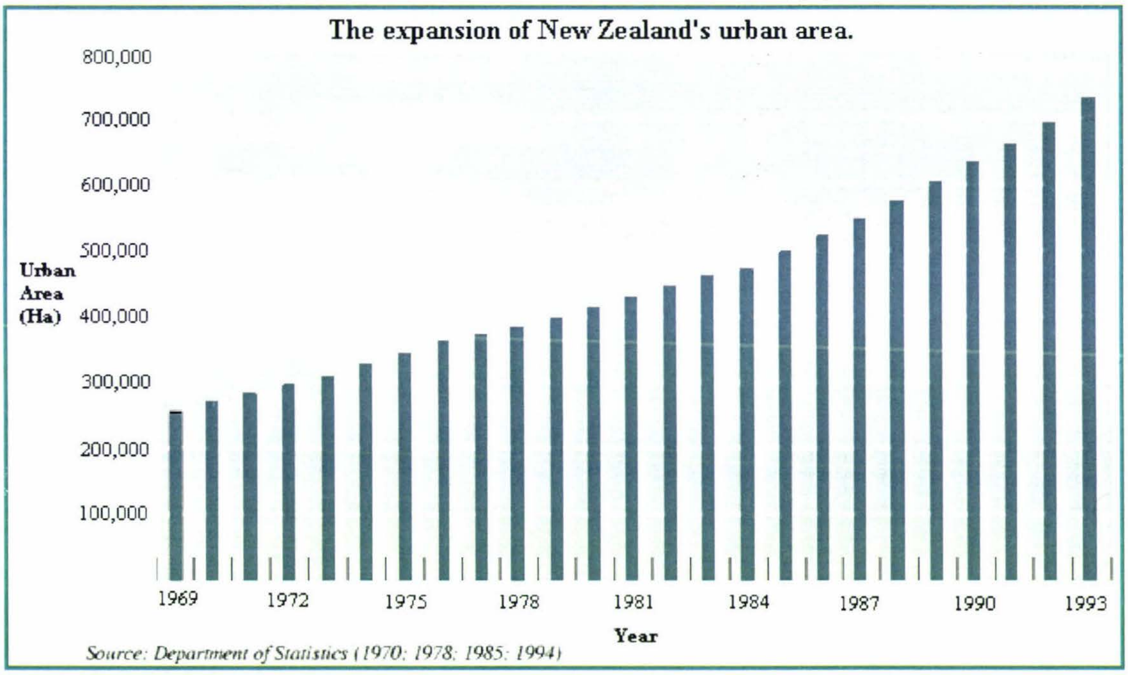

Figure 5: Increase in New Zealand's urban area 1969 - 1993. (source: MfE 1997)

A study by Holland *et al.* (1995) investigated the impact of rapid urban development around Portland, Oregon on nearby wetland areas. Aerial photographs and monitoring records were analysed over a 10 year-period (1982-1992). During that period, approximately 40% of the original 233 wetlands in the study area had been completely drained in order to construct new roads and subdivisions. Of the 141 remaining wetlands 25% had been severely degraded by human activities. The authors suggest that because wetlands are among the most sensitive ecosystems, wetland protection and planning must be provided before a natural area is developed or the wetland systems will inevitably suffer serious, if not catastrophic, damage.

Wetlands are particularly vulnerable to toxins that can be carried by stormwater running off roadways. This stormwater can contain a range of pollutants including oil, lead, heavy metals, contaminated dust, organic matter, and debris. Because wetlands are semi-closed systems they tend to accumulate these pollutants, particularly in their sediments, and over time these can reach levels that make the wetland uninhabitable by some species (Booth and Jackson 1997). Many urban wetlands are also adversely impacted by the discharge of gutters and drains, and in some cases, are used as final drainage areas. Stormwater that runs off urban catchments will typically carry high levels of sediment and nutrients, particularly if there is active construction occurring in the immediate vicinity (e.g. land clearance for new subdivisions) (Booth and Jackson 1997).

A study by Hall *et al.* (2001) compared the macroinvertebrate communities in streams draining native bush, agricultural and urban catchments. They found significant differences between the three stream types. The urban and agricultural streams were dominated by pollution-tolerant species (e.g. Oligochaeta, Mollusca) but contained very few pollution-sensitive species (e.g. Plecoptera, Trichoptera). In contrast, the native bush streams displayed the opposite pattern of abundance and were dominated by Ephemeroptera, Plecoptera and Trichoptera. Riparian and drainage wetlands have been described as "a reflection of their catchments" and are likely to be affected by different land-uses in a similar way as the streams in this study.

Urban wetlands are often under considerable pressure from exotic plants and animals, particularly from introduced aquatic plants such as *Egeria* and *Elodea* that can completely displace existing vegetation and dramatically lower dissolved oxygen levels (Ministry for the Environment 2000a). Terrestrial plants, including large trees, can also adversely impact wetland areas by displacing native plants that some species rely on and alter the amount of shading the wetland experiences. A study of the effects of introduced willow (*Salix* sp.) trees on small streams (Lester *et al.* 1994) showed that invertebrate densities and biomass was significantly lower in willow-lined stream sections when compared to adjacent open sections. The authors concluded that that observed effect was a result of an 80% decrease in incident stream illumination. A similar effect could be anticipated if willows were grown around a wetland area. Introduced fish, such as Koi Carp (*Cyprinus carpio*), can also have a rapid and devastating effect on native wetland fauna and flora.

2.1.3 *Correlation between wetland condition and land-use*

Several studies have attempted to determine if there is a relationship between the physical and biological condition of a wetland and the surrounding land-use matrix within which it is located. Tangen *et al.* (2002) compared the macroinvertebrate communities in 24 wetlands located in different land-uses ranging from 100% grassland to 100% cropland in the Prairie Pothole region of the United States. They detected a weak, but statistically significant correspondence between land-use and invertebrate abundance and composition. However, they found a much stronger relationship between the presence or absence of fish within a wetland system and the composition of the macroinvertebrate assemblages. The researchers concluded that the weak correspondence was most likely due to the low diversity of resilient taxa in wetlands and the dominant influence of predatory fish.

Anderson and Vondracek (1999) investigated whether flying insects could be used as indicators of land-use in the Prairie Pothole region. They collected insects from 11 different land-use types, including intensive agriculture, woodlands and five different wetland types. They found significant relationships between the insects collected for many of the land-use types. The strongest relationship detected was for insects collected around riparian wetland systems, which differed significantly from those collected from agricultural areas. A study by Aznar *et al.* (2002) investigated the relationship between macrophyte communities in selected French wetlands and the density of canals and drains around the wetlands. They found a positive relationship between this indicator of anthropogenic modification and macrophyte composition and diversity.

Lundkvist *et al.* (2001) compared the abundance and diversity of diving beetles (Dytiscidae) in wetlands located in agricultural, wooded and grassland wetlands in

Sweden. They found a significant difference in beetle diversity between the different wetland types, with the woodland wetlands containing the highest diversity. On a similar theme, Euliss and Mushet (1999) evaluated the influence of intensive agriculture on macroinvertebrate communities in ephemeral Prairie Pothole wetlands. They sampled 19 wetlands in cropland-dominated areas and 19 wetlands in natural grassland areas. Significant differences were found between the two different land-types, and most of the cropland wetlands were missing large sections of the taxa found in natural wetlands. The authors noted that their study corroborates the findings of other researchers that intensive agriculture has a significant and adverse effect on the macroinvertebrate populations in adjacent wetlands.

2.1.4 *Related New Zealand stream studies*

There have been few studies that have comparatively assessed the characteristics of New Zealand wetlands located in different land-use types (Ministry for the Environment 2000a; Stephenson 1983). Much more attention has been paid to the relationship between catchment land-use and the physical and biological characteristics of streams. Although they are not directly comparable, wetland systems, especially riparian wetlands, are likely to experience many of the same pressures and disturbances that small waterways experience in anthropogenically modified environments. In fact researchers have suggested that wetlands are more sensitive to environmental disturbances because they tend to accumulate sediment and pollutants, and are more seriously affected by changes in drainage patterns and hydrology (Holland *et al.* 1995; Stephenson 1983).

Quinn *et al.* (1997) compared the invertebrate communities in 11 streams in Waikato flowing through native forest, pine (*Pinus radiata*) plantations and pasture. They found that suspended solids, nutrients and temperature were all significantly higher in pastoral streams than the forest streams. Although the overall invertebrate taxa richness was not significantly different between the three land-uses, there were significant differences in the composition of the assemblages. Chironomids and molluscs dominated streams in pasture, whereas mayflies, stoneflies, and caddisflies dominated native forest streams.

Harding and Winterbourn (1995) performed a similar study in which they assessed streams passing through pasture, scrubland, *Pinus radiata*, and native forest land-uses. The authors describe an 'ecological gradient' where both diversity and abundance decreased as the land-use changed from natural (native forest), to slightly modified (scrubland) and finally to highly modified (agricultural). Strong correspondence was reported between land-use and aquatic invertebrate diversity. Streams flowing primarily through agricultural land lacked entire groups of invertebrates (mayflies, stoneflies, caddisflies) that dominated native forest streams.

The authors of both of the studies described above suggest that physical changes caused by the different land-uses, such as differences in shading, temperature, stream bank erosion, and vegetation clearance are likely to exert a stronger controlling influence on invertebrate communities than differences in water chemistry. Scarsbrook and Halliday (1999) assessed the change in aquatic invertebrate abundance and diversity in streams flowing through open pasture and then entering remnant fragments of native forest. They found that the composition of the invertebrate community had changed significantly when the stream was sampled just 50 metres into the forest fragment, and had reached a 'steady state' within 300 metres. The results of this study supports the theory that aquatic invertebrate communities are more strongly affected by stream shading and channel morphology than the nutrient loading within the stream.

Boulton *et al.* (1997) also found strong positive correlations between catchment land-use and the physical characteristics of streams. The strongest effects were higher water temperature and decreased levels of dissolved oxygen in streams flowing through pasture compared to streams flowing through exotic and native forests. Streams draining native forest were found to have a diverse and abundant aquatic invertebrate fauna, whereas pastoral streams were much less productive and dominated by relatively few taxa, mainly dipterans and molluscs. The authors concluded that severe erosion problems in the area, the clearance of riparian vegetation, stream narrowing and livestock wastes entering the streams were the main causes of the contrasting invertebrate communities.

Another study investigating the effect of land-use on invertebrate populations was performed by Collier *et al.* (1997). However, this time the researchers captured and compared the abundance and diversity of adult Trichoptera alongside a range of streams in native forest, pine forest and pasture catchments using light traps. Significant differences were found in the species captured at each stream type. Surprisingly, approximately the same number of insects were captured besides the native forest and pasture streams. Pine forest streams had significantly lower catch rates than the other two stream types.

Scott *et al.* (1994) surveyed the benthic invertebrate communities in three streams flowing intermittently through developed and undeveloped pasture. They found that invertebrate diversity and abundance was significantly affected by pasture type even though no point sources of pollution are present. Mollusca and worms increased within sections of developed pasture but beetles and some hemipterans decreased. The authors struggled to explain some of their findings but suggested that dips and drenches used in developed pastures may have been an additional source of pollutants that adversely affected certain aquatic invertebrates.

Other studies have investigated the effect of land-use on larger inhabitants of New Zealand streams. Hicks and McCaughan (1997) assessed the abundance of fish and freshwater crayfish (*Paranephrops planifrons*) in streams located in native forest, pine forest and pasture catchments. Fish densities were significantly higher in pastoral streams than the forest streams. However most of these fish were longfin eels (*Anguilla* sp.) that are easily able to tolerate streams with a high nutrient and sediment loading. More sensitive species, such as the banded kokopu (*Galaxias fasciatus*), were only found in the forest streams. No difference was detected in crayfish densities between the three stream types. This concurs with other studies that have found that crustaceans are moderately tolerant of eutrophied aquatic environments (e.g. Rowe 1998).

2.2 Materials and Methods

The aim of this study is to determine:

- How closely the abundance and species richness of aquatic macroinvertebrate communities in remnant wetlands in the Christchurch metropolitan area resemble those found in wetlands with similar characteristics located in relatively unmodified environments.

2.2.1 *Site selection*

Wetlands from each of four different wetland types were selected for aquatic invertebrate assessment. These different wetland types are described in Table 1. Two wetland types are differentiated solely by their surrounding land-use (**pastoral** and **natural tussock/scrub**) while the other two types are differentiated by both their surrounding land-use and their origin (**remnant urban** and **constructed urban**).

In addition to their surrounding land-use, the wetlands used in this study were selected depending on their physical characteristics. All wetlands selected were:

- medium size (between 5,000m^2 and 10,000m^2 total wetland area)
- freshwater (supporting freshwater wetland flora only)
- riparian (closely associated with and sharing water with an adjacent waterway)
- permanently wet
- still water (flow less than 0.01m/s)
- moderate depth (30-60 cm deep)
- mixture of emergent and benthic vegetation

Table 1: Wetland Type Definitions

remnant urban	There are still several large areas of remnant wetlands located within the city boundaries. There are also numerous smaller remnant wetlands scattered throughout the city. The majority of these wetlands are associated with the margins of rivers and streams that flow through the city. Although the wetlands have been continuously present since settlement most have been modified to some extent (eg. landscaping, drainage, planting). ***selection criteria:*** *continuous wetland history*
constructed urban	Many of the wetlands within Christchurch have been constructed by diverting streams and drains into shallow artificial basins. This practise has become very popular around new subdivisions and are often termed "wetland reserve". ***selection criteria:*** *created in area with no previous wetland history*
pastoral grazing	Many areas of remnant wetland are associated with streams that flow through beef and sheep grazing pastures on the margins of Christchurch. ***selection criteria:*** *within 50m of grazing paddocks, continuous wetland history*
natural tussock	There are numerous areas on the margins of Christchurch, in the valleys of the Port Hills, and on Banks Peninsula, where lowland streams flow through native tussock and scrub. Many of these streams have associated wetland areas. ***selection criteria:*** *located in native tussock/scrub with minimal human influence*

Experimental design

Three replicates were assessed from each wetland type. Aquatic macroinvertebrates were collected (see section 2.2.3) and physical parameters were measured (see section 2.2.4) during December 2003.

Subject: *Remnant urban*: Primary objective of this study was to determine the impact of urban development pressures on this wetland type.

Control: *Natural tussock*: The baseline against which the remnant urban wetlands were measured.

Reference point 1: *Constructed urban*: The health of remnant urban wetlands were compared to constructed urban wetlands in order to determine which type more closely resembles the baseline condition.

Reference point 2: *Pastoral grazing*: The health of remnant urban wetlands were compared to another highly anthropogenically modified ecosystem - pastoral grazing.

2.2.2 *Site descriptions*

Table 2: Remnant Urban Wetlands (medium)

Wetland Name	Associated waterway	Description
Cockayne Reserve	Avon River	Cockayne Reserve is a remnant area of native raupo swamp adjacent to the Avon river. The wetland forms two distinct areas, the lower subtidal section and an upstream section dominated by freshwater plants such as raupo and mikimiki (*Carpodetus linariifolia*). The sampling site is located in the middle of the upstream freshwater section.
Unamed Wetland (Horseshoe Lake Reserve)	Waikarikari Stream	An 11 hectare area surrounding Horseshoe Lake was set aside as a nature reserve in 1904, primarily as a wild-fowl sanctuary. The wetland sampled is located alongside Waikarikari Stream which feeds into Horseshoe Lake, and is approximately 1.8 km upstream from the lake.
Bexley Wetland Reserve	Unamed drains	This wetland reserve is located immediately adjacent to the Avon river and north of estuary. There are saltmarsh areas alongside the river and a remnant freshwater wetland, fed by three drains and springs, located beside Bexley Road.

Table 3: Constructed Urban Wetlands

Wetland Name	Associated waterway	Description
Charlesworth Reserve	Linwood Ave Canal	This area of ponds and wetlands was constructed adjacent to the Linwood Ave Canal. The wetland sampled is located north of a series of small ponds.
Anzac Drive wetland	Drains and Avon River	This recently constructed wetland is located next to Anzac drive and is fed by a series of drains and flows into the adjacent Avon River.
Regents Park Reserve	Regents Drive drain	This wetland was recently constructed as a water feature in a new subdivision. The drain running through the subdivision has been 'naturalised'using a series of ponds and wetlands. The sampling site is located approximately 500m upstream of the main pond.

Table 4: Pastoral Grazing Wetlands

Wetland Name	Associated waterway	Description
Unamed Wetland (Springston South)	L2 Stream	This wetland is located in paddocks used to graze sheep and cattle opposite Wolfes Road near Springston South. The wetland is in a small depression basin next to the L2 River and is surrounding by cattle and sheep paddocks.
Unamed Wetland (Motukarara)	Motukarara Stream	This wetland in cattle paddocks located beside the Christchurch-Akaroa Highway. Sheep and cattle are able to graze on both sides of the wetland area.
Unamed Wetland (Ataahua)	Waikoko Stream	This wetland area is located in paddocks used to intensively graze cattle adjacent to the Christchurch-Akaroa Highway. Although the wetland is fenced off cattle are able to graze very close to the wetland edge and can cross the Waikoko Stream approximately 300m upstream.

Table 5: Natural tussock/scrub Wetlands

Wetland Name	Associated waterway	Description
Unamed Wetland (Kaituna Valley)	Kaituna River	This wetland area is located at the head of the Kaituna Valley in an area surrounded by native scrubland. The wetland has been formed by ponding alongside the Kaituna River which originates in native tussock in the hills above.
Unamed Wetland (Ahuriri)	Halswell River	This wetland is located near the end of the Ahuriri Road. The river arises in the nearby from an area of native bush near Ahuriri Reserve.
Unamed Wetland (Little River Valley)	Unamed tributary of Little River	This wetland is located approximately 15 minutes drive up the Western Valley Road. The wetland is formed by ponding in a depression next to the stream.

2.2.3 *Collection and identification of aquatic invertebrates*

Aquatic macroinvertebrates were collected using a D-framed hand-net (30 cm base; 0.5mm mesh) in accordance with a modified version of *Protocol C2 for soft-bottomed habitats* described in Stark *et al.* 2001. The modified protocol is supplied in Appendix B of this report. Samples units of approximately 0.3m^2 were collected from 2 different areas of benthos around emergent vegetation and 2 different areas of submerged benthic vegetation in each wetland by sweeping the net through the benthos for a distance of 1 metre, giving a total sample area of 1.2 m^2. Samples were then passed through a 0.5 mm sieve and transferred to a sample container where they were preserved using a 75% ethanol solution.

Samples were later screened using *Protocol P3: Full Count with Subsampling Option* as in Stark *et al.* 2001 (Appendix B). This involved sequentially passing each sample though a series of sieves (0.5mm, 2mm, 4mm) onto a white tray into order to separate the organisms into different size fractions. Each size fraction was then placed on a gridded tray (10x10) and individuals from each taxon were removed using tweezers and placed on a petri dish. After morphological examination to confirm that all individuals appeared to belong to the same taxa they were placed in sealed bottles, and preserved using 70% ethanol, for subsequent microscopic examination, identification and enumeration. Where particular taxa were present in very high numbers (>500 per sample unit) individuals were collected from 3 squares (of ten). This result was cleared noted on the sample bottle and the final count was multiplied by 3.33 to give an estimated full count.

The aquatic invertebrates were primarily identified using a 20-40x binocular microscope and the keys in MacFarlane (1999), Winterbourn *et al.* (2000), Winterbourn (1973), and Chapman and Lewis (1973). Certain groups of aquatic organisms such as mites, nematodes, bryozoans, protozoa, platyhelminthes and coelenterates were not included in this study because they were either very rare, too small to identify, or fall outside the range of macroinvertebrates traditionally used for assessment of freshwater ecosystems.

2.2.4 *Water quality assessment*

In addition to the abundance and diversity of aquatic macroinvertebrate communities contained within the wetlands, a number of physical water quality parameters were measured in the field.

Table 6: Water quality assessment methods and equipment.

Parameter	Unit	Method
temperature	C	Hach digital thermometer
conductivity	mS/cm	Hach conductivity probe
pH	pH units	Hach pH probe
clarity	cm	Secchi disk

The Hach probes were calibrated and operated in accordance with the *Water Analysis Handbook* (Hach Company, 1997). These physical parameters were measured at three different locations within each wetland.

2.2.5 *Data analysis*

The data collected was analysed using 1-way ANOVA, and where a significant difference was detected, Tukey's Multiple Comparison Test was performed. The results were also analysed by graphing the means, and standard errors, of each wetland type for each parameter measured.

Figure 6: Sampling at Cockayne Reserve.

Figure 7: Sampling at Charlesworth Wetland.

Figure 8: Sampling at Bexley Wetland.

Figure 9: Ataahua wetland.

Figure 10: Kaituna Valley wetland sampling site.

Figure 11: Sampling at Regents Park Wetland.

Figure 12: Motukarara wetland sampling site.

Figure 13: Motukarara stream.

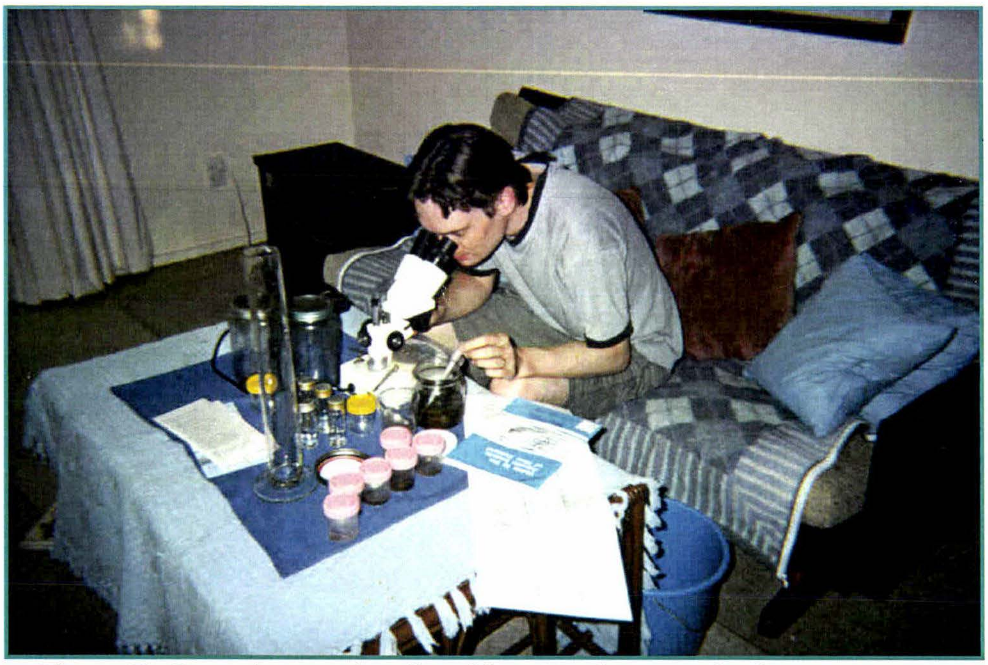

Figure 14: Processing sample collected from Motukarara wetland.

2.3 Results

A total of 15,177 individual organisms, comprising 52 taxa, were collected from the twelve wetlands (3 x 4 land-use types) sampled in this study. The average macroinvertebrate density across all sites was 1054 organisms/m^2 (S.E = 230). The land-use type with the greatest density was pastoral (1723 organisms/m^2) followed by unmodified (959 organisms/m^2), urban remnant (886 organisms/m^2) and constructed urban (646 organisms/m^2). There was a high degree of variability in the densities measured between the three replicates for each land-use. These were most pronounced for pastoral and unmodified wetlands (Figure 15).

The most abundant taxonomic groups collected were insects (55%), Crustacea (18%) and Mollusca (17%). The insects were dominated by Diptera (61%), Hemiptera (19%), and Coleoptera (16%). Crustacea were dominated by Amphipods (40%) and Cladocerans (26%). The absence of significant numbers of pollution-sensitive groups (e.g. Trichoptera, Emphemeroptera, Plecoptera) from all habitats sampled suggests that the water quality is quite poor, which is to be expected in a wetland system.

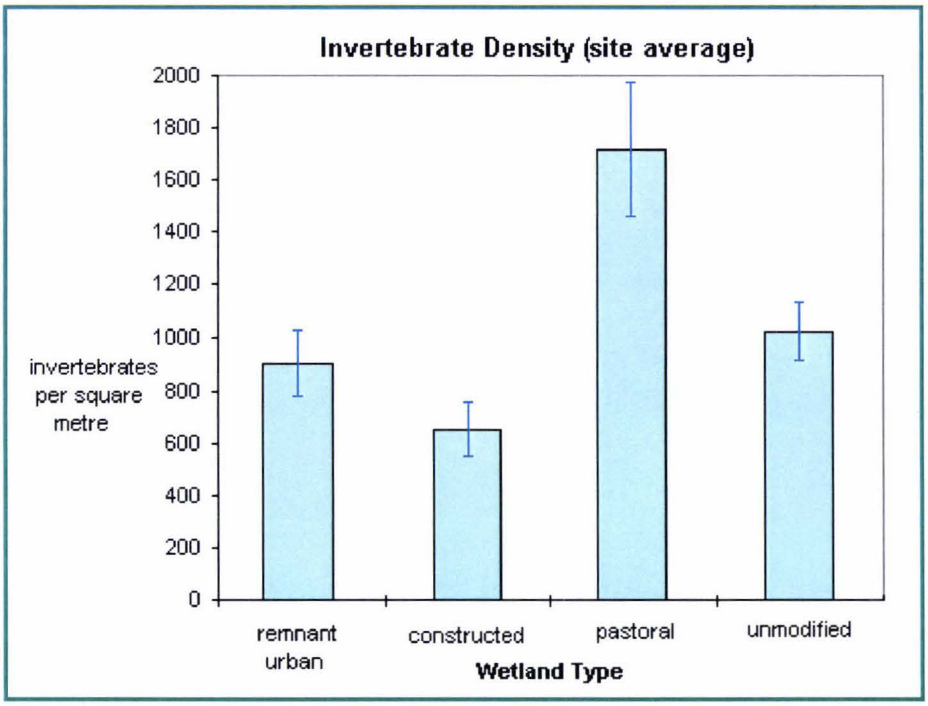

Figure 15: Average macroinvertebrate density at each wetland type (mean ± 1 SE).

2.3.1 *Total Abundance*

Higher macroinvertebrate populations were observed in wetlands located adjacent to pasture used for livestock grazing than in any of the other wetland types. Although unmodified wetlands contained higher populations than the remnant urban wetlands no significant difference was detected ($p>0.05$). Constructed wetlands supported lower populations than the pastoral and unmodified wetlands but not the remnant wetlands (Table 7 & Figure 16).

	Remnant urban	Constructed urban	Pastoral	Unmodified
Coleoptera	119	55	62	222
Hempitera	101	72	132	250
Crustacea	359	156	156	335
Diptera	311	261	988	161
Mollusca	116	200	427	143
Oligochaeta	55	86	305	43
Other	23	14	1	70
sum	**1085**	**844**	**2071**	**1224**

Table 7: Average total abundance in each wetland type.

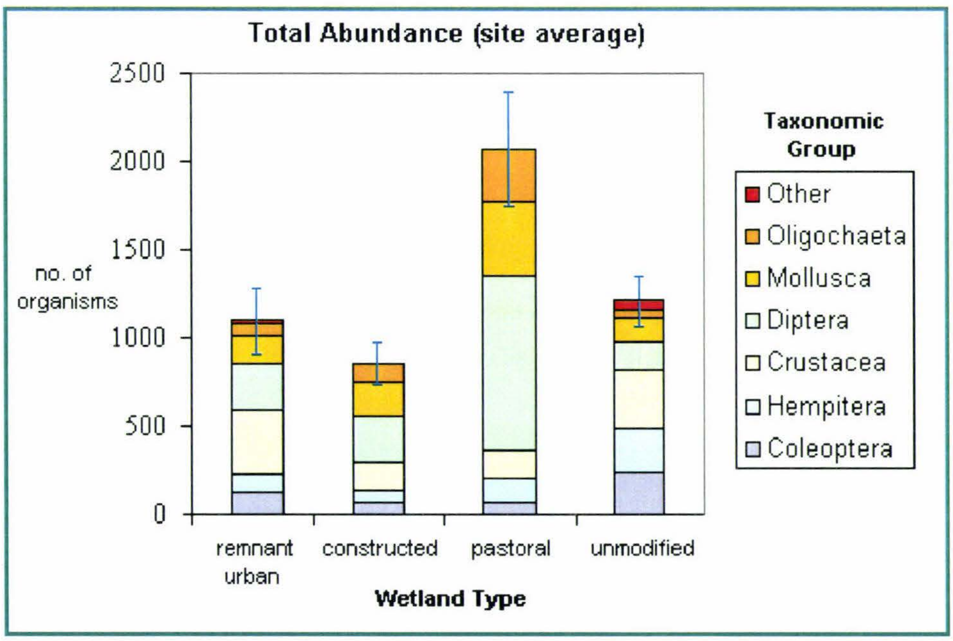

Figure 16: Average total abundance in each wetland type (mean ± 1 SE).

2.3.2 *Relative Abundance*

Pastoral and constructed wetlands were dominated by diptera, snails and worms. These groups accounted for more than 70% of the organisms collected from these habitats. The remnant wetlands were dominated by crustacea and diptera, while the unmodified wetlands showed a more even distribution of organisms across the major taxonomic groups (Table 8 & Figure 17).

	Remnant urban	Constructed urban	Pastoral	Unmodified
Coleoptera	7	7	3	18
Hempitera	8	9	6	20
Crustacea	40	18	8	27
Diptera	22	31	48	13
Mollusca	10	24	21	12
Oligochaeta	8	10	15	3
Other	4	2	0	6
sum	**100**	**100**	**100**	**100**

Table 8: Relative abundance at each wetland type.

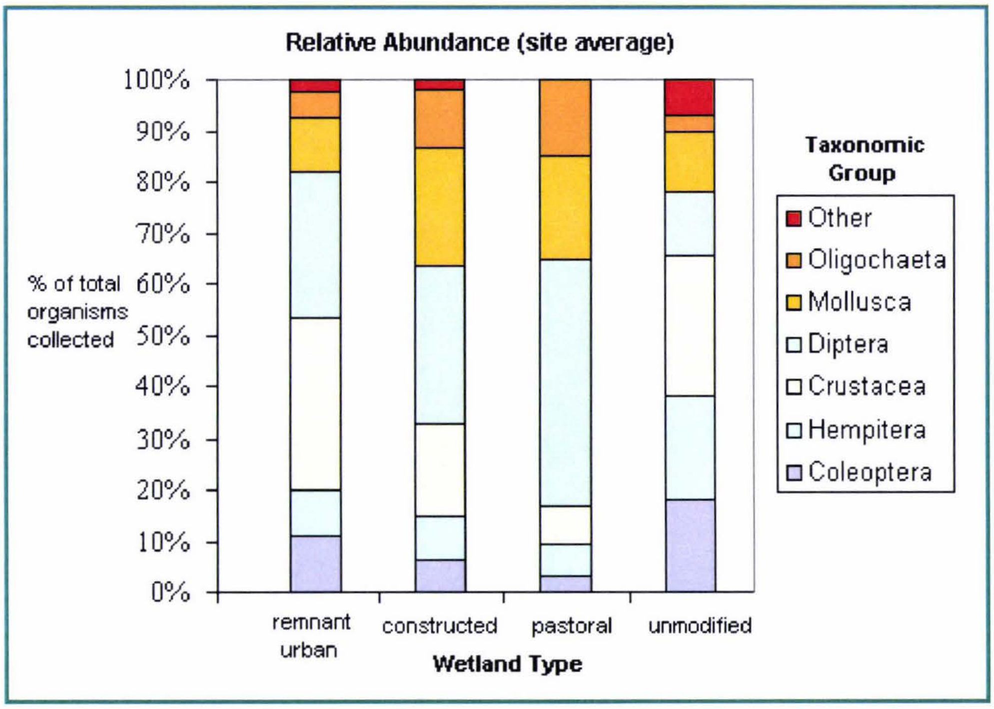

Figure 17: Relative abundance at each wetland type.

2.3.3 *Species richness*

Although the pastoral wetlands contained larger macroinvertebrate populations, a lower number of species were recorded compared to the unmodified and remnant urban wetlands. In general, the more highly modified wetlands (pastoral and constructed) contained fewer distinct RTU across all the major taxonomic groups except for oligochaetes (Table 9 & Figure 18).

2.3.3.1 Recognisable Taxonomic Units

	Remnant urban	Constructed urban	Pastoral	Unmodified
Coleoptera	3.33	2.33	2.00	5.67
Hempitera	4.33	2.67	2.33	5.33
Crustacea	8.67	6.33	3.00	8.33
Diptera	8.33	6.33	6.33	8.33
Mollusca	4.00	3.00	2.67	5.00
Oligochaeta	2.67	1.67	2.00	2.67
Other	2.67	1.00	1.00	4.33
	34.00	**24.33**	**19.33**	**39.67**

Table 9: Average number of RTU collected from each wetland type.

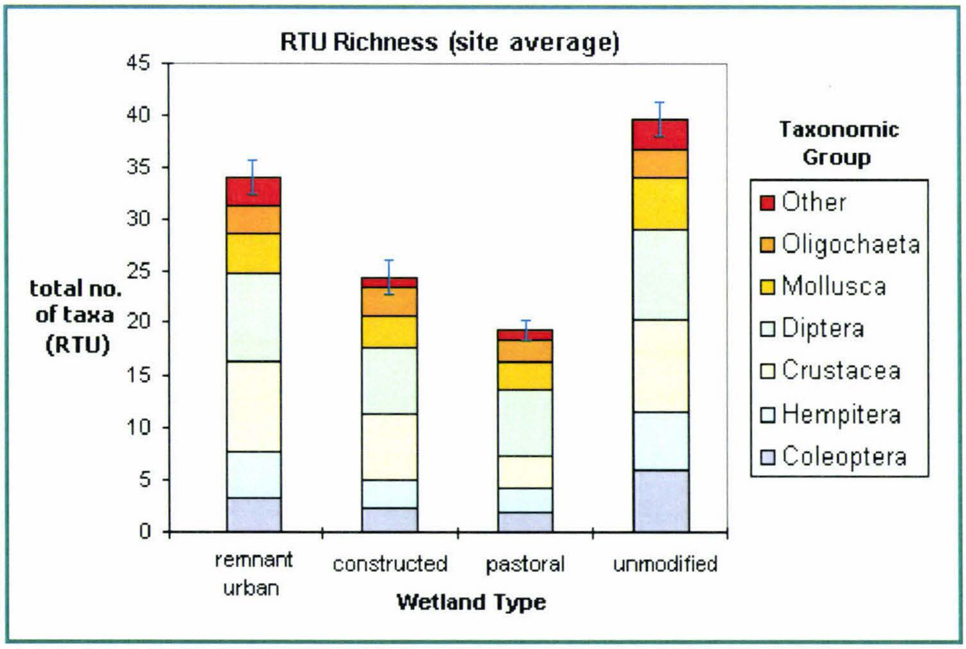

Figure 18: Average number of RTU collected from each wetland type (mean ± 1 SE).

2.3.3.2 Families

Relatively few distinct taxa for certain commonly-occurring groups (e.g. Oligocheata, Mollusca) were collected. This is likely to be a result of the lack of taxonomic knowledge and resources required to distinguish between closely related taxa. This is also true for the larval stages of most groups, especially for early instars. Hence, a more accurate measure of taxa richness is probably given by comparing the number of families collected. The pastoral and constructed wetlands contained fewer families than the other two land-use types. No significant differences were observed between pastoral and constructed wetlands (Table 10 & Figure 19).

	Remnant urban	Constructed urban	Pastoral	Unmodified
Coleoptera	2.33	1.00	1.33	3.00
Hempitera	3.67	2.67	2.00	4.67
Crustacea	4.67	5.00	4.00	4.67
Diptera	6.00	5.00	5.33	6.67
Mollusca	4.00	3.00	2.67	5.00
Oligochaeta	2.67	1.67	2.00	2.67
Other	2.33	1.00	0.33	3.67
sum	**25.67**	**18.33**	**16.67**	**30.33**

Table 10: Average number of families collected from each wetland type.

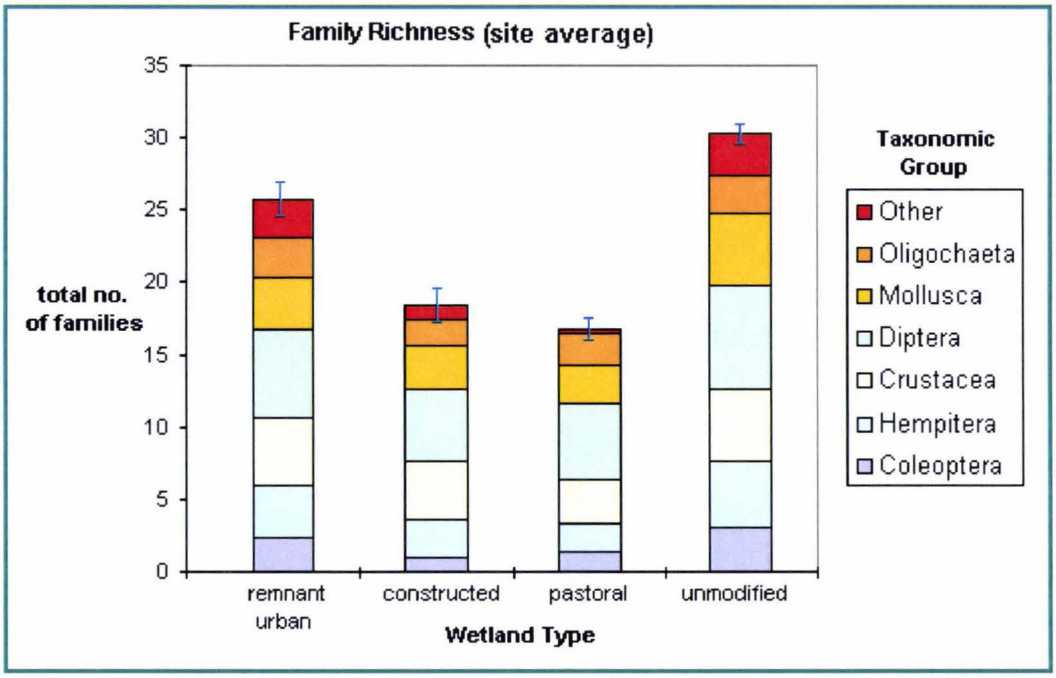

Figure 19: Average number of families collected from each wetland type (mean $\pm$ 1 SE).

2.3.4 *Relative species richness*

2.3.4.1 Recognisable Taxonomic Units

All wetlands in this study contained a higher proportion of distinct dipteran and crustacean taxa (>40% of all taxa). Coleoptera and hemiptera accounted for 22% of the total RTU identified in all wetland types. However, these two groups accounted for more than 28% of the RTU identified in remnant urban and unmodified wetlands (Table 11 & Figure 20).

	Remnant urban	Constructed urban	Pastoral	Unmodified
Coleoptera	10	10	10	15
Hempitera	13	11	12	14
Crustacea	25	26	16	22
Diptera	25	26	33	22
Mollusca	12	12	14	13
Oligochaeta	8	11	10	7
Other	8	4	5	8
	100	**100**	**100**	**100**

Table 11: Relative RTU richness in each wetland type.

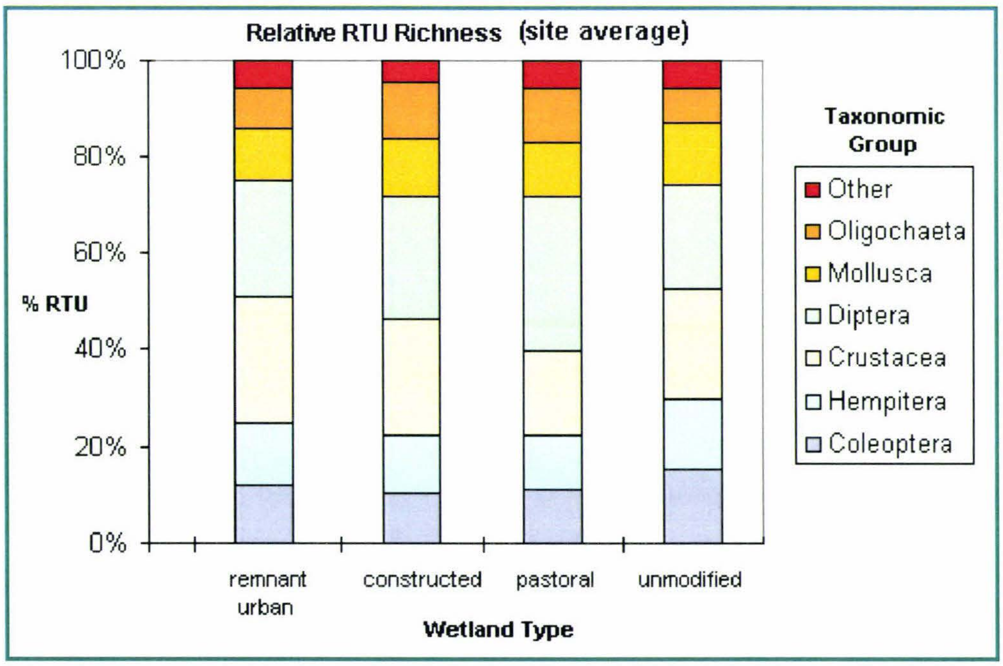

Figure 20: Relative RTU richness in each wetland type.

2.3.4.2 Families

When the relative proportions of the different families collected are compared crustacean and dipteran families once again dominate all the wetland types. Coleopteran and crustacean families represent a smaller proportion of taxa when relative taxa count is compared at the family level than at the RTU level (Table 12 & Figure 21).

	Remnant urban	Constructed urban	Pastoral	Unmodified
Coleoptera	9	5	8	10
Hempitera	14	15	12	15
Crustacea	18	22	18	16
Diptera	23	27	32	23
Mollusca	14	16	16	16
Oligochaeta	10	9	12	9
Other	10	5	2	10
	100	**100**	**100**	**100**

Table 12: Relative family richness in each wetland type.

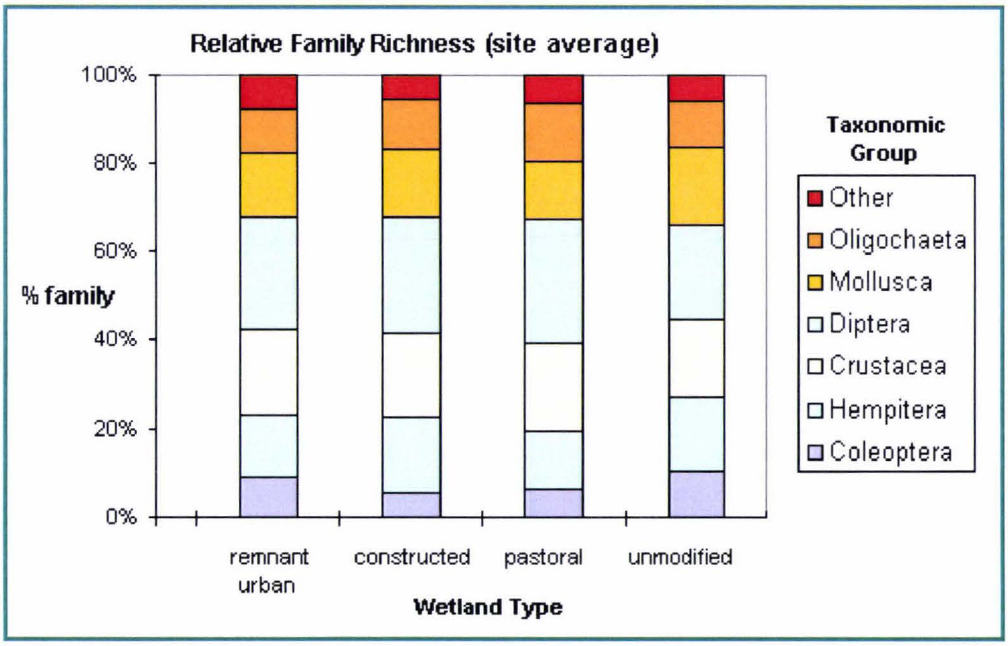

Figure 21: Relative family richness in each wetland type.

2.3.5 *Percentage of total taxa collected at each land-use*

2.3.5.1 Recognisable Taxonomic Units

Representatives from all of the crustacean and oligochaetan taxa collected in this study were present in at least one of the unmodified wetlands, and at least one of the urban remnant wetlands. Approximately half of the hemipteran and coleopteran taxa were not present in any of the constructed or pastoral wetlands sampled (Table 13 & Figure 22).

	Remnant urban	Constructed urban	Pastoral	Unmodified
Coleoptera	85.71	42.86	42.86	100.00
Hempitera	83.33	50.00	33.33	100.00
Crustacea	100.00	77.78	33.33	100.00
Diptera	64.71	58.82	58.82	82.35
Mollusca	83.33	66.67	50.00	83.33
Oligochaeta	100.00	66.67	66.67	100.00
Other	83.33	16.67	16.67	100.00

Table 13: Proportion of total RTU found in each wetland type.

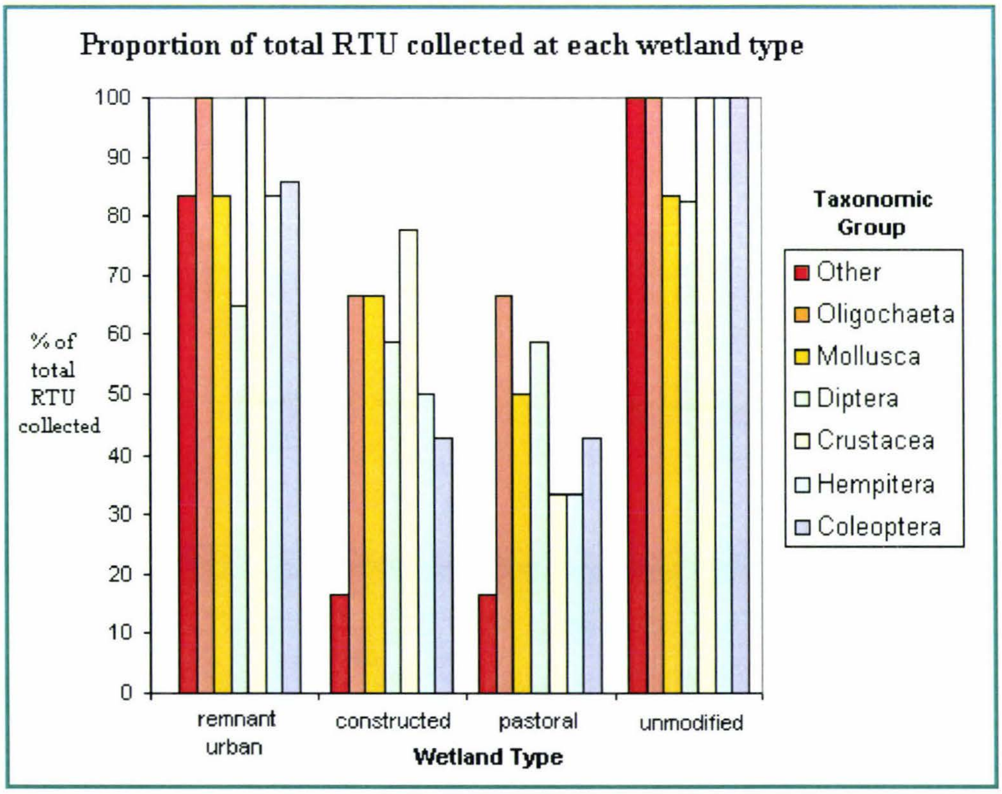

Figure 22: Proportion of total RTU found in each wetland type.

2.3.5.2 Families

A similar pattern was detected when family-level proportions are compared. All coleopteran, crustacean and oligochaetan families were found in at least one of the unmodified wetlands, and one of urban remnant wetlands. Pastoral and constructed wetlands lacked a significant number of families in most taxonomic groups, particularly coleopteran, hemipteran, and 'other' families (Table 14 & Figure 23).

	Remnant urban	Constructed urban	Pastoral	Unmodified
Coleoptera	100.00	25.00	50.00	100.00
Hempitera	80.00	60.00	40.00	100.00
Crustacea	100.00	80.00	60.00	100.00
Diptera	75.00	66.67	66.67	75.00
Mollusca	83.33	66.67	50.00	83.33
Oligochaeta	100.00	66.67	66.67	100.00
Other	80.00	20.00	20.00	100.00

Table 14: Proportion of total families found in each wetland type.

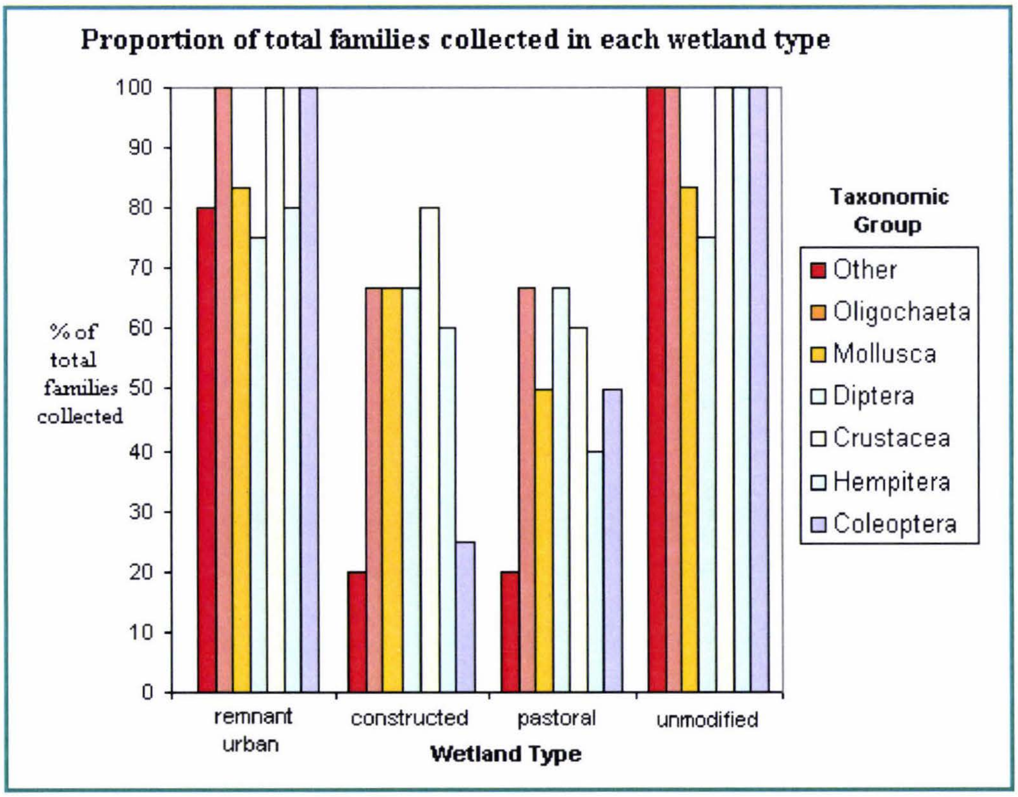

Figure 23: Proportion of total families found in each wetland type.

2.3.6 *Water Quality Parameters*

No significant difference was detected between the conductivity measured in the four wetland types (Figure 24). Pastoral wetlands and constructed wetlands exhibited significantly lower **water clarity** than the unmodified and remnant wetlands but only a small difference was observed between these two latter types (Figure 25).

Only small differences were observed between the **pH** levels in four wetland types. Pastoral wetlands exhibited the lowest pH, while the unmodified wetlands showed the highest average pH (Figure 26). Likewise, only small water **temperature** differences were detected between the various wetlands. However, the temperatures recorded are likely to be affecting by a number of confounding factors, including diurnal variation and the source of the waterway. Pastoral wetlands had the highest average temperature, followed by the constructed wetlands (Figure 27).

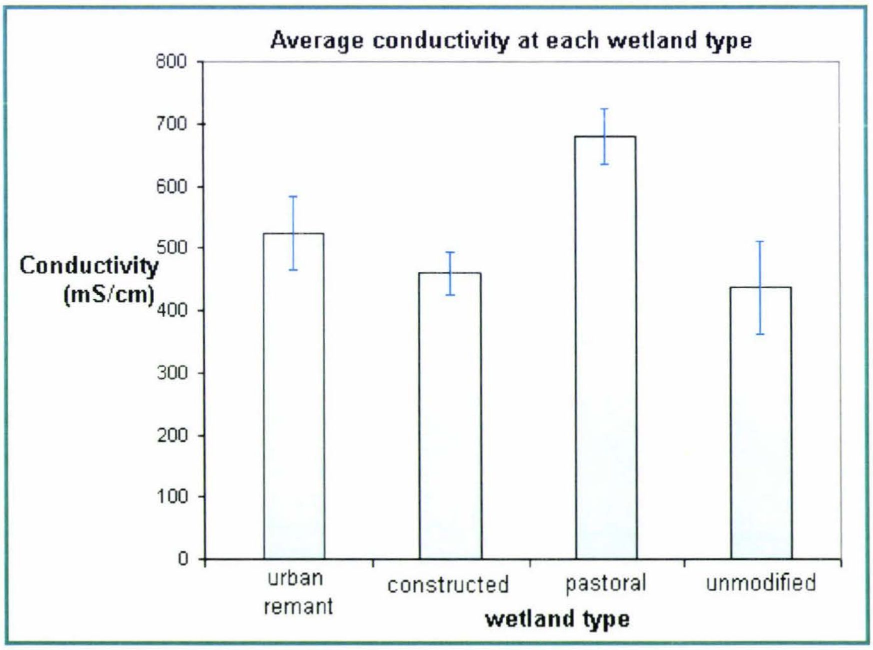

Figure 24: Average conductivity in each wetland type (mean ± 1 SE).

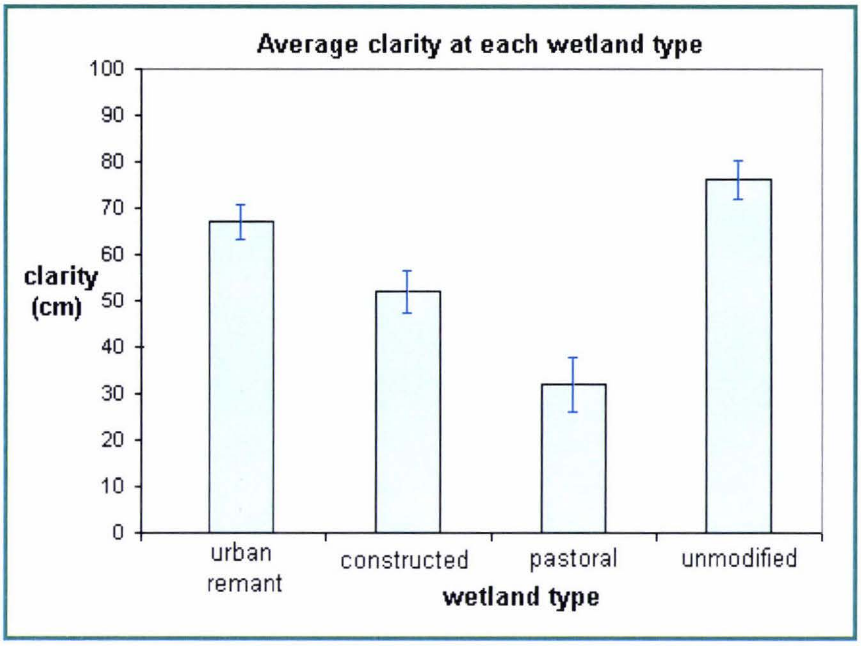

Figure 25: Average water clarity in each wetland type (mean ± 1 SE).

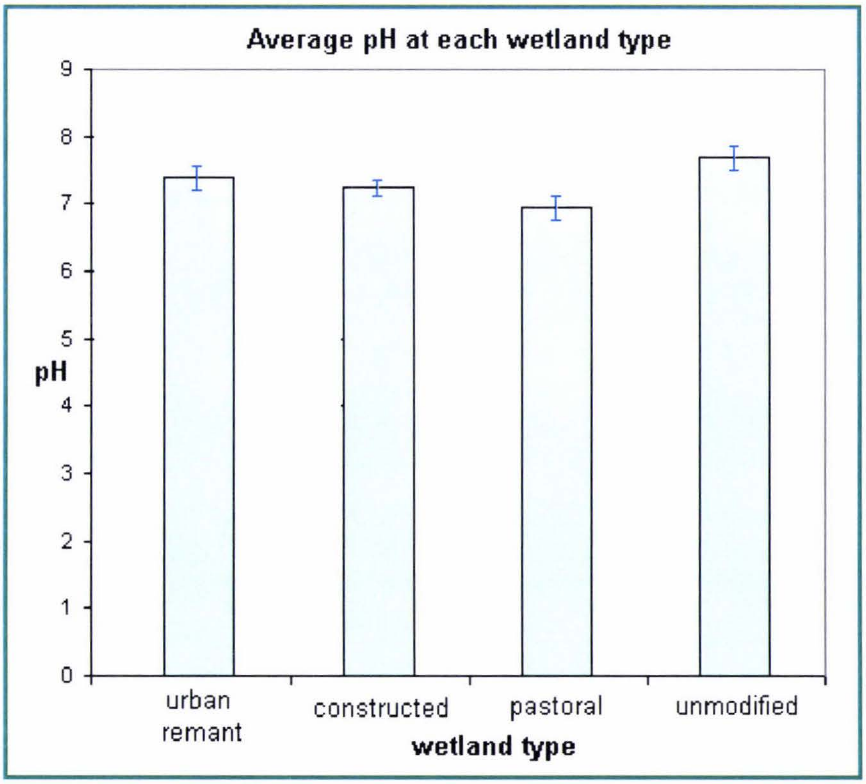

Figure 26: Average pH in each wetland type (mean ± 1 SE).

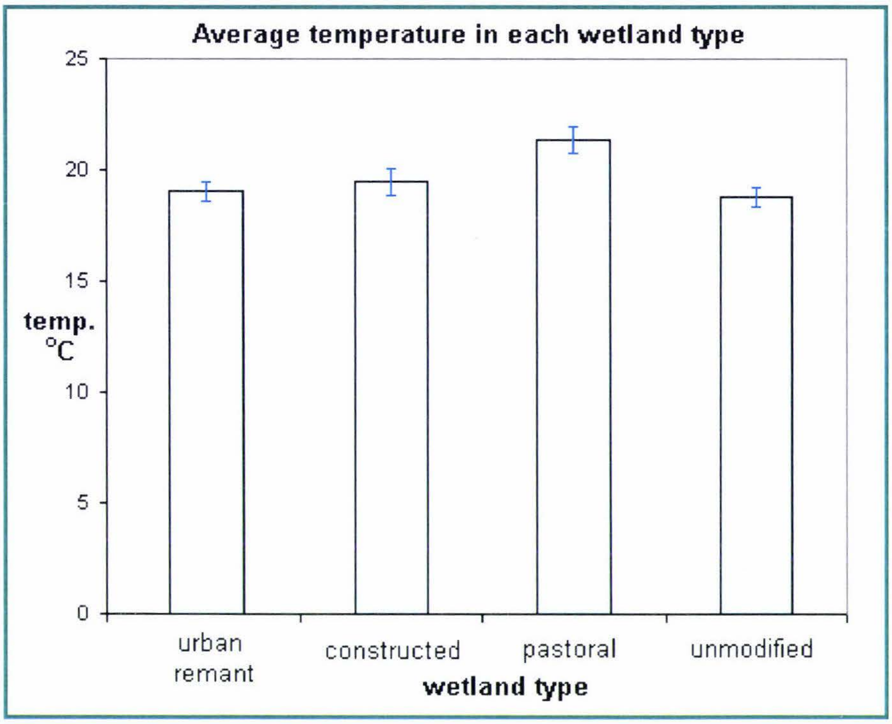

Figure 27: Average temperature in each wetland type (mean ± 1 SE).

2.3.7 *Statistical analysis summary*

Table 15: Land-use significance: Tukey's Multiple Comparison Test.

Land Use	abundance	species richness (rtu)	species richness (family)	pH	temperature	conductivity	clarity
remnant vs constructed	N	N	N	N	N	N	N
remnant vs pastoral	Y	Y	Y	N	Y	N	Y
remnant vs unmodified	N	N	N	N	N	N	N
constructed vs pastoral	Y*	N	N	N	Y	N	N
constructed vs unmodified	N	Y*	Y*	N	N	N	N
pastoral vs unmodified	N	Y*	Y*	Y	Y*	N	Y*

Y = significant difference detected ($p<0.05$)
Y* = strong significant difference detected ($p<0.01$)
N = no significant difference detected ($p>0.05$)

3.0 Habitat fragmentation effects on wetlands

3.1 *Introduction*

As discussed in the previous chapter, the majority of New Zealand's wetland areas have been drained in order to facilitate agricultural production and urban development (Ministry for the Environment 1997). However, in addition to a 90% decrease in total wetland area, most of the remnant natural wetlands have also been divided into smaller fragments (Stephenson 1983; Ministry for the Environment 2000a). This process of habitat fragmentation has been defined as 'the division of a natural habitat into progressively smaller patches of smaller total area isolated from each other by a matrix of habitats unlike the original' (Debinkski and Holt 2000; Andren 1994) (Figure 28). Wetland fragmentation can have numerous adverse effects including the direct removal of total suitable habitat, a reduction in the size of individual fragments and the isolation of remnant fragments (Andren 1994; Silva *et al.* 2003).

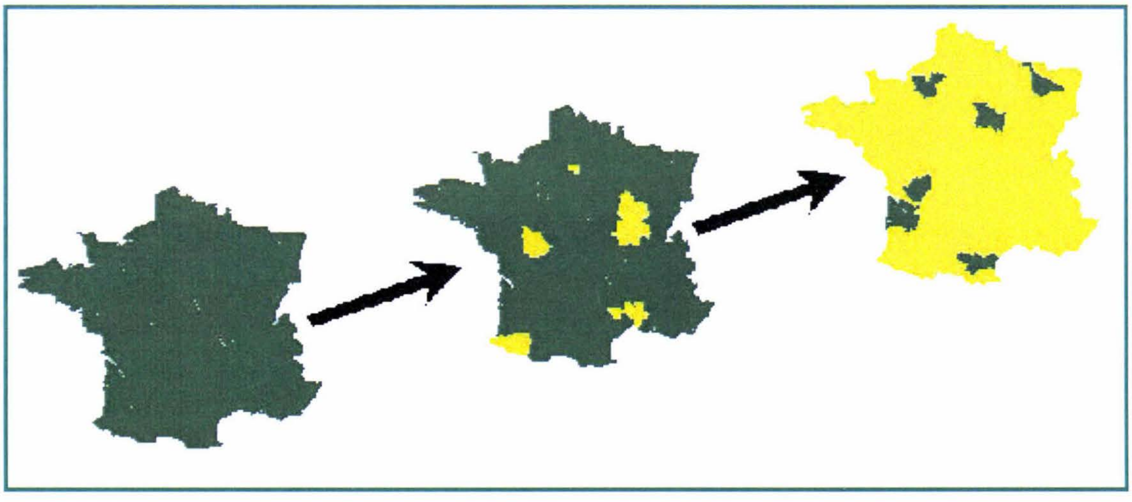

Figure 28: Fragmentation process (source: Wu and Loucks 1995).

3.1.1 *Fragmentation of wetlands*

The transformation of approximately 65% of New Zealand's land area into an agricultural and forestry production system has removed most of the complex ecological gradients that once merged different habitats, and has removed, or dramatically reduced, many natural ecosystems (Ministry for the Environment 1997). These transitional areas between different ecosystems, especially aquatic and terrestrial environments, supported an abundant, and often specialised, flora and fauna. This has resulted in many isolated 'islands' of native forest or wetland areas that are located within a surrounding matrix of pasture or pine forest. Although thousands of wetlands remain most are very small and are remnant fragments of much larger historical wetlands (Cromarty and Scott 1996).

Many lowland and coastal wetlands appear to have been irretrievably fragmented and modified when compared to their original condition, with the exception of several large isolated coastal lagoons (Ministry for the Environment 2000a). Almost all of New Zealand's healthiest wetland systems are now located in undisturbed habitats within the conservation estate where the wetlands are sequestered within native forest. Ecological fragmentation and habitat loss is widely recognised as one of the most serious threats to indigenous biodiversity in New Zealand's remnant wetland areas (Ministry for the Environment 2000a).

An unmodified, integrated, large wetland is more likely to maintain long-term viability and contain greater biodiversity because the basic ecological functions are robust enough to enable the system to resist the adverse impacts of human activities (Andren 1994; Bender *et al.* 1998). Wetlands that have been fragmented or significantly reduced from their historical size will have modified ecological cycles, and may be below the minimum area thresholds to protect against environmental disturbances (Jules and Shahani 2003). The fragmentation of a wetland system into isolated islands will also adversely affect aerial or overland dispersal of some species. The most viable remnant fragments appear to be those within dispersal range of other fragments (Andren 1994).

Some researchers have found that the nature of the matrix separating isolated wetland or forest fragments strongly influences the effect that fragmentation will have on the population contained within the fragment (Wu and Loucks 1995; Jules and Shahani 2003). For example, Lominolo *et al.* (1989) found that indigenous forest fragments separated by woodlands had greater indigenous diversity that fragments separated by desert scrub. The authors suggest that the adverse impacts of fragmentation can be correlated to how similar the surrounding matrix is to the original habitat. If the matrix is similar, or exhibits a high degree of heterogeneity and contains numerous habitat types, then it is also more likely to facilitate the movement of organisms between fragments. The matrix of dense housing surrounding urban wetlands is also more likely to inhibit the movement and dispersal of native species between wetland fragments than a surrounding matrix of bush, scrub or pasture.

Jules and Shahani (2003) reviewed a number of recent studies that have investigated the movement of pollinators, herbivores, and seed dispersers through various types of matrices between indigenous forest fragments. These matrices included clear-cut forest, pasture and dense urban areas. They concluded *"that the nature, characteristics and uniformity of the matrix separating habitat fragments has a far more profound influence on within-fragment processes than had been previously thought"*. It appears that the composition of invertebrate and bird populations within areas of remnant vegetation is altered by significant changes in the surrounding matrix.

When remnant wetland fragments are truly isolated from each other the long-term survival of species that depend on the ability to disperse and colonise other similar habitats is threatened (Andren 1994). The likelihood that another wetland fragment will be discovered during dispersal will decrease as fragments become smaller and further apart. Isolated populations will eventually lose genetic integrity and diversity because of forced inbreeding (Jenkins *et al.* 2003). This will be particularly evident in small wetland fragments that can only supported very limited population numbers (Wu and Loucks 1995).

In addition to the isolation of wetland fragments, the size and shape of the fragment will also affect the viability of the populations contained within (Andren 1994;

Debinski and Holt 2000). For many species there will be a critical population threshold, or habitat size threshold, that will be required for survival and reproduction. Different species, depending on their size, feeding habits and dispersal ability, will be affected by fragmentation in different ways (Tscharntke *et al.* 2002). Functional diversity within an isolated wetland fragment relies on the right functional groups of organisms (e.g., herbivores, decomposers, predators) being present in the correct abundance (Andren 1994). When wetlands are reduced in size to the extent that shifts in populations occur then the basic ecological functions occurring within the wetland will also change, allowing non-adapted species to invade and displace specialised species (Wu and Loucks 1995; Tscharntke *et al.* 2002).

Several researchers have even suggested that small fragmented ecosystems are doomed and the inhabitants will inevitably succumb because the underlying ecological processes have broken down (Debinski and Holt 2000; Tscharntke *et al.* 2002). Researchers have described a 'lag period' between habitat degradation and the subsequent extinction of dependant species (Ministry for the Environment 2000a; Wu and Loucks 1995). The large number of threatened species that are contained in New Zealand's wetlands suggests that these systems are not robust enough to buffer against stochastic or 'chance' events (e.g. disease outbreaks, poor breeding, natural disasters) (Ministry for the Environment 1997).

3.1.2 *Edge effects of fragmentation*

In addition to the reduction in total habitat area and fragment isolation, one of the underlying dynamics of wetland fragmentation is an increase in the 'edge to area' ratio. This phenomenon is known as the "edge effect" (Debinski and Holt 2000). Small fragments have a greater amount of edge relative to their area, and a smaller area of core, interior habitat that is buffered from the surrounding matrix by the wetland/matrix transition zone (Moorhead 1999). The centre of the wetland is also closer to an edge.

Ecological processes have been shown to accelerate at habitat edges because many species are attracted by the foraging, predation and cover opportunities provided

(Debinski and Holt 2000; Moorhead 1999). Species that are adapted to live in the interior of a wetland will be particularly vulnerable when they find themselves living in the transitional zone between a wetland fragment and the surrounding matrix. Wetland species most at risk are likely to be ground nesting birds, native lepidoptera and amphibians that are vulnerable to predation (Debinski and Holt 2000; Tscharntke *et al.* 2002). Wetland flora will also face considerable pressure from invasive plants and windblown seeds. Wetland edges are often 'less wet' than the interior which is another factor that facilitates the establishment of non-wetland plants.

The physical reduction or removal of wetland transitional zones can significantly exacerbate this problem. Many wetlands located in pastoral areas are characterised by 'hard edges' where the natural transitional zone between the wetland and pasture has been removed and animals are allowed to graze close to the edge of the wetland (Ministry for the Environment 2001). This loss of natural buffer zones also exacerbates the problem of animal wastes and other pollutants being washed into wetland systems (Collier 1995b).

3.1.3 *Correlation between fragmentation and wetland biodiversity*

There has been much recent interest in the effects of habitat fragmentation on biodiversity, and the related literature has been growing rapidly (e.g. Debinski and Holt 2000; Tscharntke *et al.* 2002). Much of this research has been focussed on the ecological impact of fragmentation on old-growth forests. However, a number of recent studies have also investigated fragmentation impacts on wetland systems. Many of these studies have occurred in the mid-west of the United States where hundreds of thousands of hectares of ancient ephemeral wetlands were drained to provide land for agriculture. Peat samples from the remnants of these vast wetlands have been aged at over 10,000 years old, and yet 85% were drained within a 50-year period (Jenkins *et al.* 2003).

Jenkins *et al.* (2003) investigated the impact of this severe fragmentation on the microcrustaceans in remnant wetlands in Illinois, USA. Surprisingly they found that only 10% of the crustacean species thought to have been present before land drainage had become locally extinct. However, cellular automata simulations showed that many of the remaining species lacked the genetic diversity that has been found in unfragmented wetlands.

On a similar theme, Hooftman *et al.* (2003) compared the 'genetic fitness' of two common wetland plants, *Carex davalliana* and *Succisa pratensis*, in 18 wetlands of differing sizes and at various distances from other fragments. They found that plants in small isolated fragments had significantly less biomass, flowers and germination rates than plants from larger fragments. Transplanting of species into large wetlands did not affect their reproductive fitness, which suggests that the differences observed were due to inbreeding rather than environmental conditions.

Lienert *et al.* (2002) also measured the reproductive fitness of a specialised wetland plant, *Swertia perennis*, in Swiss wetlands of differing size and spatial isolation. Populations in the smallest, most isolated fragments were shown to have a 78% lower density of reproductive adults and significantly lower seedling survival than in the largest fragments. The authors concluded that the viability of specialised plant populations in isolated fragments is reduced due to forced inbreeding.

Invertebrates, especially those with a low dispersal potential, are likely to adversely impacted by genetic isolation in a similar to isolated plant communities. Tscharntke *et al.* (2002) reviewed the literature relating to the impact of habitat fragmentation on insect populations. The authors argue that the species most negatively impacted would be those that are rare and specialised, and those that are characterised by high natural population variability and have a high trophic position. The authors also suggest that a large number of small reserves scattered across an area are superior to a few large reserves.

This contradicts the view of Debinski and Holt (2000) who argued that while small, numerous fragments might be more resistant to disease outbreaks and may be suitable for highly mobile taxa such as birds and mammals, they are more susceptible to generalist species, predators and invasive species, primarily due to edge effects. The authors of this review noted that there is a remarkable lack of consistency in results across 20 fragmentation studies assessed, especially with regard to species richness and abundance relative to fragment size. They also suggest that short-term studies, and those that focus only on a few taxa, are likely to be strongly influenced by species-specific responses to fragmentation, and overlook more subtle and fundamental ecosystem changes.

In a large fragmentation study Bolger *et al.* (2000) assessed the invertebrate populations in 40 urban habitat fragments, composed mainly of indigenous vegetation, located in Southern California. The diversity and abundance of arthropods in each individual fragment was correlated against the age, area and edge length of the fragment. Interestingly invertebrate diversity and abundance positively correlated with fragment area and negatively correlated with fragment age. These findings suggest that the fragments are under ongoing pressure from the surrounding urban matrix and as fragments age they are unable to support the same level of biodiversity. Smaller fragments also showed significantly lower numbers of higher trophic level insects, although spider populations were not affected.

Another large study was performed by Wettstein and Schmid (1999) in which the arthropod diversity in 24 wetland fragments in Switzerland was assessed and correlated against the size, altitude, vegetation structure, and isolation of each fragment. Overall the diversity of the invertebrate populations decreased with decreasing fragment size. However the authors noted that the response to habitat fragmentation was highly species-specific. Specialist wetland butterflies were found to be the most sensitive to both fragment size and isolation.

3.1.4 *Rehabilitation of degraded fragments*

In response to the recent increased understanding of the ecological role and value of wetlands, and the catastrophic impact that human activities have had on these ecosystems world-wide over the last 100 years, many governments have established policies and programmes to rehabilitate and restore habitat functions in badly damaged sites (Streever 1997; Makaloff 1998). The international recognition of the significance and importance of wetlands is reflected in the 'Convention on Wetlands of International Importance', an intergovernmental agreement adopted on 2 February 1971 which provided for the protection and management of wetlands. By 2001 the Convention had 123 signatories, and provided international protection to 1050 sites, which totalled more than 78.7 million hectares (Ministry for the Environment 1997). New Zealand became a party to the Convention on 13th December 1976 and now has six sites, covering almost 40,000 hectares, designated under the Convention as 'Wetlands of International Importance' (Ministry for the Environment 2000a).

Many territorial authorities and government agencies within New Zealand have put considerable effort into restoring degraded wetlands, often with good success (Ministry for the Environment 1997). However wetland restoration is still a relatively new science and there has been considerable debate in the literature regarding the ecological integrity of restored systems. Malakoff (1998) suggested that many restored wetlands fail to exhibit the same ecological functions, or support similar flora and fauna, as natural wetlands. Zedler (2000) discussed many of the factors that can inhibit the restoration process including the absence of former the hydrological regime, appropriate nutrient supplies and seed banks. He concluded that "it takes more than water to restore a wetland" and the rehabilitation process will usually require careful management over several decades.

Brown (1998) suggested that analysing the remnant seed bank at a restoration site might be a useful predictor of the probability of recreating natural wetland vegetation. However, field experiments showed very poor correlation between the abundance and diversity of seeds and subsequent restoration success. Experiments using soil transplanted from a natural wetland into a restoration site have shown more success (Brown 1997). The author concluded that restoring the hydrology of a wetland system is likely to be the dominant factor that allows the re-establishment of wetland fauna and flora.

An increasingly common restoration technique is the provision of 'green corridors' between wetland fragments that 'softens' the matrix surrounding the fragments and allows organisms to move between them more easily (Streever 1997, Lomolino *et al.* 1989). Christchurch City Council has a strategy to construct an green corridor that links the major urban wetland areas with other urban waterways eventually links up to Canterbury lakes and rivers (Christchurch City Council 2003).

3.2 Materials and Methods

The aims of this study are:

- To determine if there is any relationship between the size of remnant wetland fragments within Christchurch and the abundance and species richness of aquatic macroinvertebrates that they support.
- To provide recommendations on which size remnants would be the best candidates for restoration and enhancement in order to preserve urban biodiversity.

3.2.1 *Site selection*

Urban remnant wetland fragments of three different sizes were selected for aquatic invertebrate assessment. The criteria for these three size types was:

small fragment:	$50m^2$ - $500m^2$ total fragment size
medium fragment:	$5{,}000m^2$ – $10{,}000m^2$ total fragment size
large fragment:	greater than $50{,}000m^2$ total fragment size

The data obtained for medium sized remnant urban wetlands in Chapter 2 was used for the medium sized fragments in this study. In addition to their size, the wetlands were also selected depending on their physical characteristics. All wetlands selected were:

- able to be accessed and sampled at the centre of the fragment
- freshwater (supporting freshwater wetland flora only)
- riparian (closely associated with and sharing water with an adjacent waterway)
- permanently wet
- still water (flow less than 0.01m/s)
- shallow-moderate depth (20-60 cm deep)
- mixture of emergent and benthic vegetation

3.2.2 Site descriptions

Table 16: Large Remnant Urban wetlands.

Wetland Name	**Associated waterway**	**Description**
Travis Wetland	Travis Stream	Travis Wetland Nature Heritage Park is a lowland freshwater wetland covering 116 hectares, and has been designated as a site of National Importance because of the high diversity of native plants, birds, invertebrates that it supports. The reserve consists of a mixture of fens, streams, ponds and riparian wetlands. The sampling site is located adjacent to Travis Stream.
Styx Mill Reserve	Styx River	Styx Mill Reserve is a low-lying swampy conservation reserve covering 53 hectares of remnant wetlands, ponds and streams. It is the second largest remnant wetland area remaining in Christchurch and has been the subject of a major protection and enhancement programme. The sampling site is an extensive wetland area adjacent to the Styx river near the centre of the reserve.
Otukaikino Wetland Reserve	Otukaikino Stream	Until recently the Otukaikino wetland was known as Wilson's Swmap. It is one of the few original raupo wetland remnants in Christchurch. The current owners, in conjunction with the Department of Conservation and local Iwi, have undertaken a significant enhancement programme which involved removing Willow trees and planting natives. The sampling site is along a boardwalked section near the middle of the wetland.

Table 17: Medium Remnant Urban wetlands.

Wetland Name	Associated waterway	Description
Cockayne Reserve	Avon River	Cockayne Reserve is a remnant area of native raupo swamp adjacent to the Avon river. The wetland forms two distinct areas, the lower subtidal section and an upstream section dominated by freshwater plants such as raupo and mikimiki. The sampling site is located in the middle of the upstream freshwater section.
Horseshoe Lake stream	Waikarikari Stream	An 11 hectare area surrounding Horsehoe Lake was set aside as a nature reserve in 1904, primarily as a wild-fowl sanctuary. The wetland sampled is located alongside Waikarikari Stream which feeds into Horseshoe Lake, and is approximately 1.8 km upstream from the lake.
Bexley Wetland Reserve	Unamed drains	This wetland reserve is located immediately adjacent to the Avon river and north of estuary. There are saltmarsh areas alongside the river and a remnant freshwater wetland, fed by three drains and springs, located beside Bexley Road.

Table 18: Small Remnant Urban wetlands.

Wetland Name	Associated waterway	Description
Thistledown Reserve	Steamwharf Stream	This small waterway flows from a natural spring in Thistledown reserve. The stream has been enhanced by the planting of native vegetation along the banks. The sampling site is an area of standing water and wetland plants south east of Thistledown Reserve.
Unamed wetland	Cashmere Stream	Cashmere Stream arises in the Hoon Hay Valley and flows into the Heathcote River approximately 8km downstream. The sampling site is an area of standing water and wetland plants near the junction of Hendersons Road and Cashmere Road.
Corsers Stream Reserve	Corsers Stream	Corsers Stream flows south from Travis Wetland approximately 2.5 km to the Avon River. The banks of the stream have been enhanced by native plantings. There are several patches of standing water that have formed small riparian wetland areas. The sampling site is an area of rushes near the second foot-bridge.

3.2.3 *Collection and identification of aquatic invertebrates*

Aquatic macroinvertebrates were collected, sorted and identified using the same methods and protocols described in **section 2.2.3**.

3.2.4 *Water quality assessment*

Water conductivity, pH, temperature and clarity were measured using the same methods and equipment described in **section 2.2.4**.

3.2.5 *Data analysis*

The data collected was analysed measured using the same methods described in **section 2.2.5**.

Figure 29: Sampling at Corsers Stream wetland.

Figure 30: Sampling at Bexley Wetland.

Figure 31: Sampling at Cockayne Reserve wetland.

Figure 32: Sampling at Thistledown Reserve.

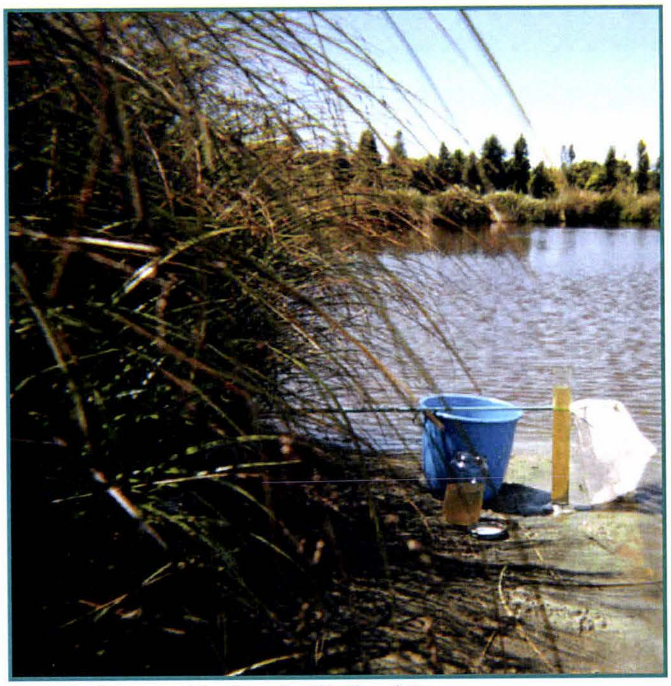

Figure 33: Travis Wetland Reserve.

3.3 Results

A total of 8,834 individual organisms, comprising 52 taxa, were collected from the nine urban remant fragments sampled in this study. The average macroinvertebrate density across all sites was 818 organisms/m^2 (S.E = 92) (Figure 34). The most abundant taxonomic groups collected were Crustacea (34%), Insects (44%) and Mollusca (11%). The insects were dominated by Diptera (52%), Hemiptera (25%), and Coleoptera (23%). As in the first study, pollution-sensitive groups (Emphemeroptera, Plecoptera and Trichoptera) were almost completely absent from all wetlands sampled.

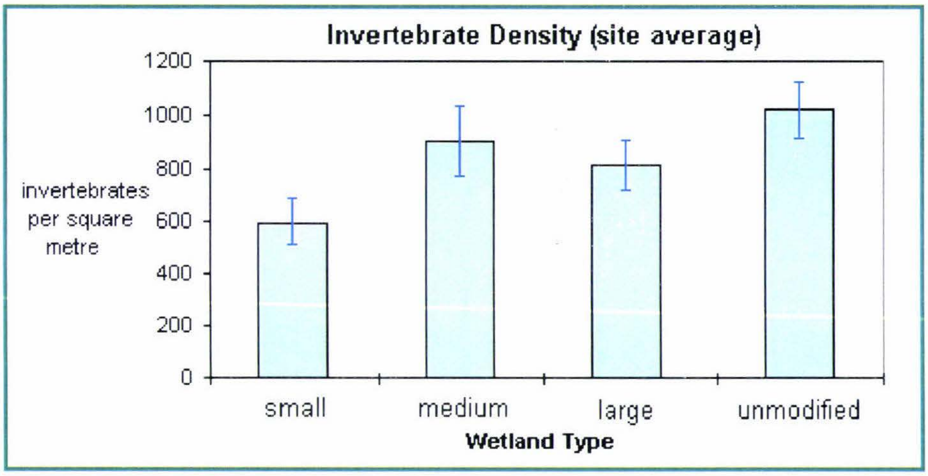

Figure 34: Average macroinvertebrate density in each fragment size (mean ± 1 SE).

3.3.1 *Total Abundance*

Small remnant wetland fragments supported smaller populations of aquatic invertebrates than the other two fragment sizes. However, no significant difference was detected between the three fragment sizes wetlands, and no significant difference was detected when the abundance in the remnant fragments are compared to the abundance found in unmodified wetlands ($p>0.05$) (Table 19 & Figure 35).

	Small urban	**Medium urban**	**Large urban**	**Unmodified***
Coleoptera	51	119	111	222
Hempitera	59	101	145	250
Crustacea	286	359	349	335
Diptera	160	311	164	161
Mollusca	73	116	116	143
Oligochaeta	56	55	53	43
Other	26	23	34	70
site average	**710**	**1085**	**972**	**1224**

Table 19: Average total abundance in each fragment size.

(*unmodified results included as a point of reference)

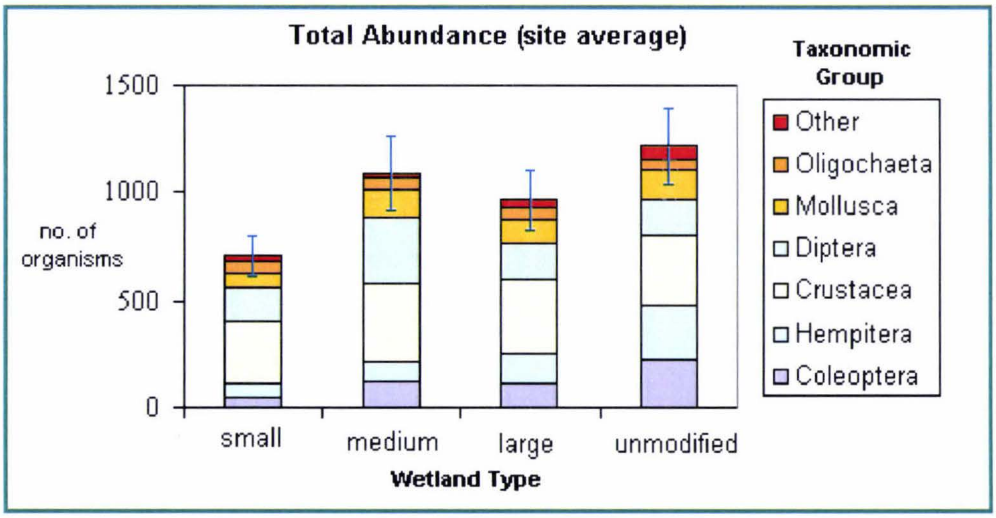

Figure 35: Average total abundance in each fragment size (mean ± 1 SE).

3.3.2 *Relative Abundance*

Small fragments were dominated by crustaceans and dipterans (67%) and contained a higher proportion of oligochaetes. Medium and large fragments were also dominated by crustacea but showed a slightly more even distribution of organisms across the major taxonomic groups than the small fragments. The unmodified wetlands had a much more even distribution and had higher proportions of coleoptera and hemiptera than any of the urban remnant wetlands (Table 20 & Figure 36).

	Small urban	Medium urban	Large urban	Unmodified *
Coleoptera	7	11	11	18
Hempitera	8	9	15	20
Crustacea	40	33	36	27
Diptera	22	29	17	13
Mollusca	10	11	12	12
Oligochaeta	8	5	5	3
Other	4	2	3	6
sum	**100**	**100**	**100**	**100**

Table 20: Relative abundance in each fragment size.

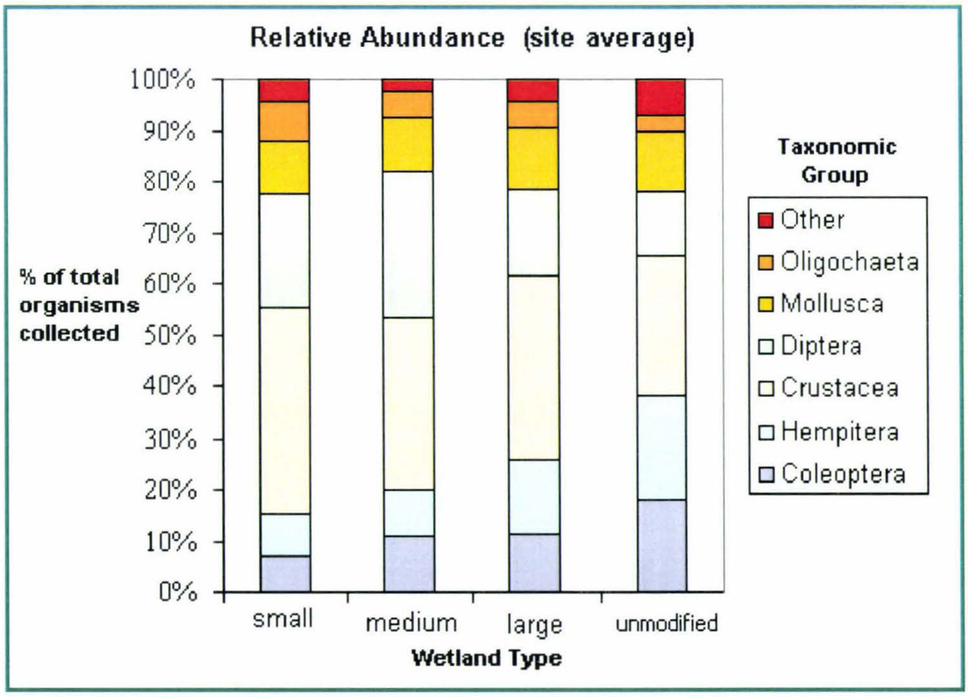

Figure 36: Relative abundance in each fragment size.

3.3.3 *Species richness*

3.3.3.1 Recognisable Taxonomic Units

No significant differences were detected between the number of distinct RTU identified in the medium and large fragments, or when compared to the number found in the unmodified reference wetlands. However, less species richness was observed in the small fragments (Table 21 & Figure 37).

	Small urban	Medium urban	Large urban	Unmodifie d
Coleoptera	2.67	3.33	2.67	5.67
Hempitera	3.33	4.33	4.33	5.33
Crustacea	6.33	8.67	9.00	8.33
Diptera	5.67	8.33	8.00	8.33
Mollusca	3.33	4.00	4.00	5.00
Oligochaeta	2.00	2.67	2.67	2.67
Other	2.67	2.67	1.67	4.33
sum	**27.00**	**34.00**	**32.33**	**39.67**

Table 21: Average number of RTU collected from each fragment size.

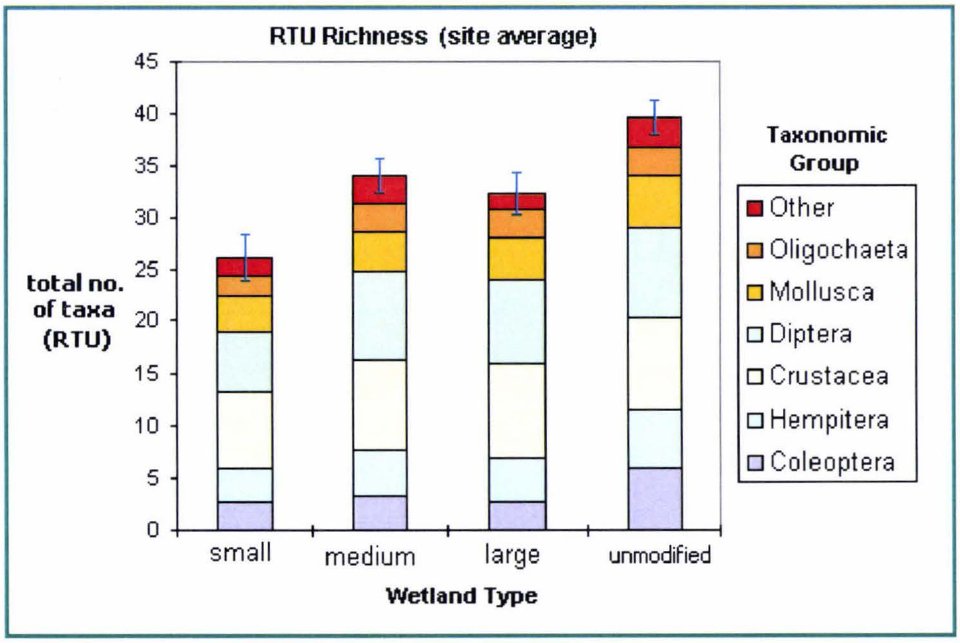

Figure 37: Average number of RTU collected from each fragment size (mean ± 1 SE).

3.3.3.2 Families

When the number of distinct number of families identified is compared a similar pattern occurs as was found with RTU. Small fragments contained a lower number of families than the medium and large fragments the difference was not statistically significant ($p>0.05$). More distinct families were collected from the unmodified wetlands than all the remnant wetlands but this was also not statistically significant ($p>0.05$) (Table 22 & Figure 38).

	Small urban	Medium urban	Large urban	Unmodified
Coleoptera	1.33	2.33	2.67	3.00
Hempitera	3.00	3.67	4.00	4.67
Crustacea	4.33	4.67	5.00	5.00
Diptera	4.67	6.00	6.00	7.00
Mollusca	3.33	4.00	4.00	5.00
Oligochaeta	2.00	2.67	2.67	2.67
Other	2.00	2.33	2.33	3.00
sum	**20.67**	**25.67**	**26.67**	**30.33**

Table 22: Average number of families collected from each fragment size.

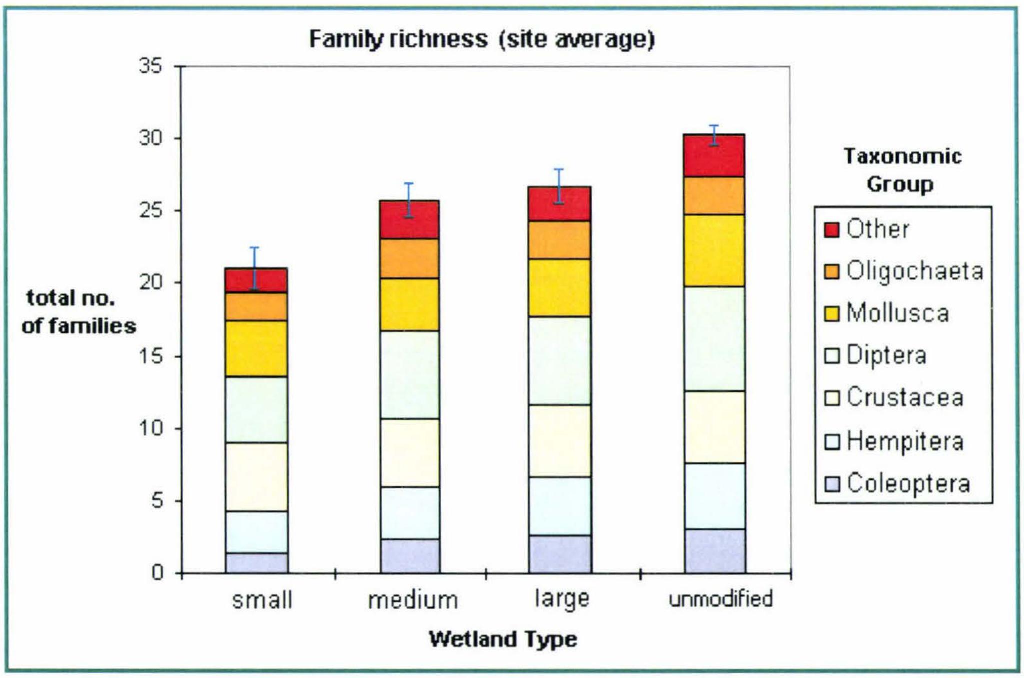

Figure 38: Average number of families collected from each fragment size (mean $\pm$ 1 SE).

3.3.4 *Relative species richness*

3.3.4.1 Recognisable Taxonomic Units

All wetlands exhibited similar proportions of RTU's across the major taxonomic groups. Crustacean and dipteran taxa contained the highest relative proportion of taxa (>45%), followed by hemiptera and coleoptera. Slight differences in the relative proportion of coleopteran taxa between small fragments and unmodified wetlands were observed (Table 23 & Figure 39).

	Small urban	Medium urban	Large urban	Unmodified
Coleoptera	10	10	8	15
Hempitera	13	13	13	14
Crustacea	28	25	28	22
Diptera	22	25	25	22
Mollusca	13	12	12	13
Oligochaeta	8	8	8	7
Other	6	8	5	8
sum	**100**	**100**	**100**	**100**

Table 23: Relative RTU richness in each fragment size.

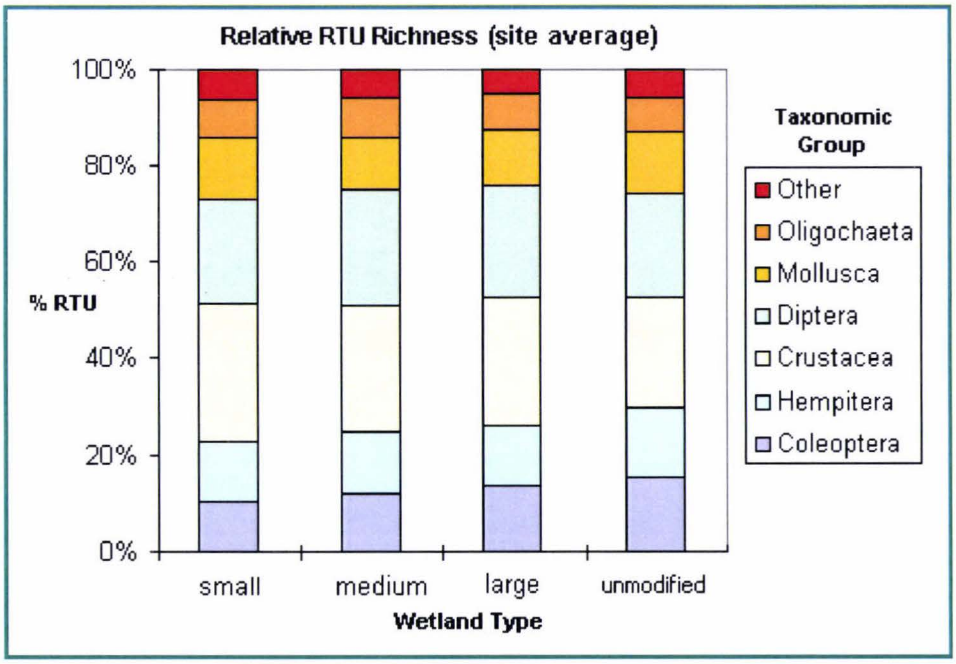

Figure 39: Relative RTU richness in each fragment size.

3.3.4.2 Families

When the relative proportions of family-level taxa are compared similar patterns of distribution were once again found. As in the first study, coleopteran and crustacean families represent a smaller proportion of taxa when relative richness is compared at the family level than at the RTU level (Table 24 & Figure 40).

	Small urban	Medium urban	Large urban	Unmodified
Coleoptera	6	9	10	10
Hempitera	14	14	15	15
Crustacea	22	18	19	16
Diptera	22	23	23	23
Mollusca	17	14	15	16
Oligochaeta	10	10	10	9
Other	8	10	9	10
sum	**100**	**100**	**100**	**100**

Table 24: Relative family richness in each fragment size.

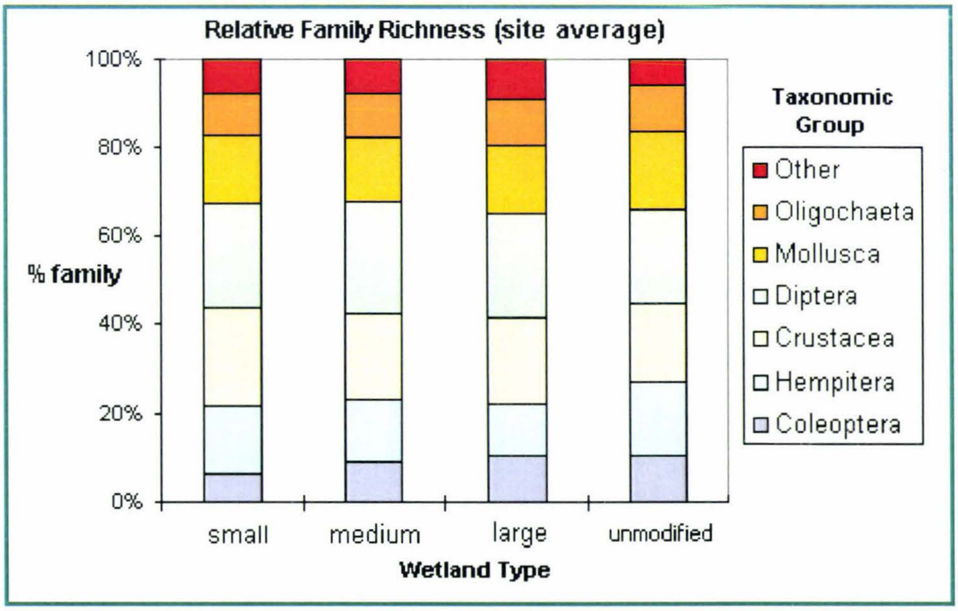

Figure 40: Relative family richness in each fragment size.

3.3.5 *Percentage of total taxa collected at each land-use*

3.3.5.1 Recognisable Taxonomic Unit

The unmodified and large urban remnant wetlands contained very similar percentages of taxa collected across all wetlands sampled. Representatives from all the coleopteran, hemipteran, crustacean, and oligochaetan taxa collected were present in at least one of the unmodified wetlands, and at least one of the large urban remnant wetlands. Approximately 25-50% of the coleopteran, oligochaetan, molluscan and 'other' taxa were absent from all of the small wetland fragments sampled (Table 25 & Figure 41).

	Small urban	Medium urban	Large urban	Unmodified
Coleoptera	57.14	85.71	100.00	100.00
Hempitera	83.33	83.33	100.00	100.00
Crustacea	88.89	100.00	100.00	100.00
Diptera	47.06	64.71	64.71	82.35
Mollusca	66.67	83.33	100.00	83.33
Oligochaeta	66.67	100.00	100.00	100.00
Other	66.67	83.33	83.33	100.00

Table 25: Proportion of total RTU found in eac fragment size.

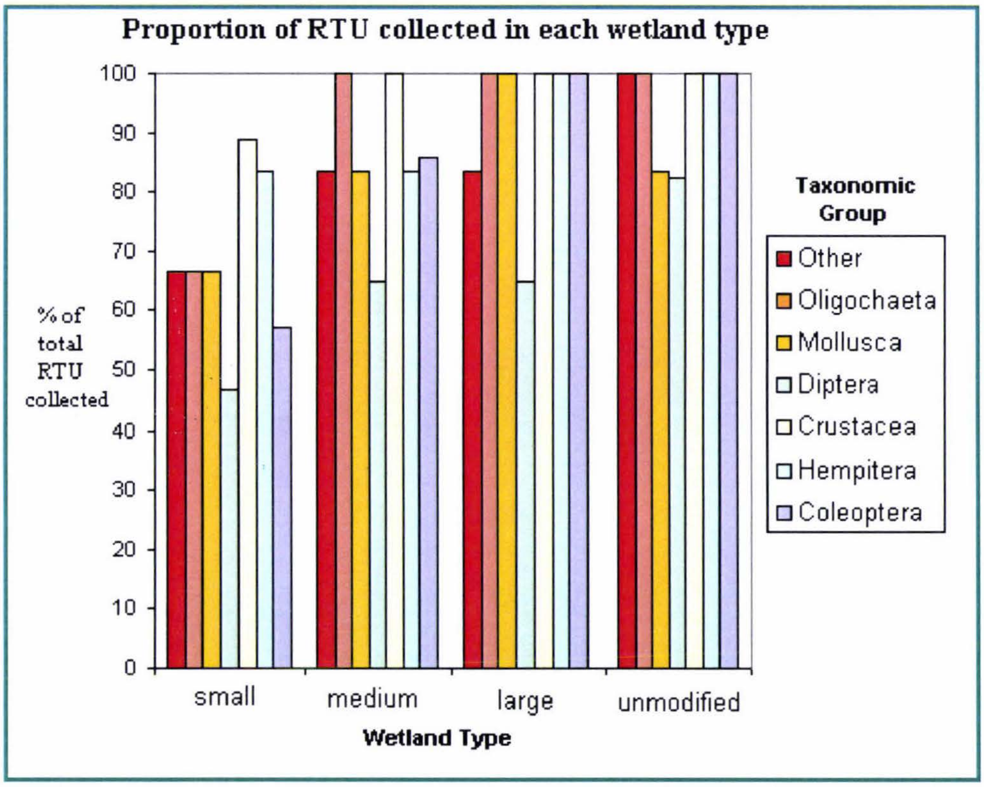

Figure 41: Proportion of total RTU found in each fragment size (mean ± 1 SE).

3.3.5.2 Families

The large wetland fragments contained all the families identified for all the taxomonic groups except for diptera. Unmodified wetlands also contained almost all families, with the exception of some dipteran and molluscan families. Several families were absent from all the small fragments for all taxonomic groups except crustacea (Table 26 & Figure 42).

	Small urban	Medium urban	Large urban	Unmodified
Coleoptera	50.00	100.00	100.00	100.00
Hempitera	80.00	80.00	100.00	100.00
Crustacea	100.00	100.00	100.00	100.00
Diptera	58.33	75.00	66.67	75.00
Mollusca	66.67	83.33	100.00	83.33
Oligochaeta	66.67	100.00	100.00	100.00
Other	80.00	80.00	80.00	100.00

Table 26: Proportion of total families found in each fragment size.

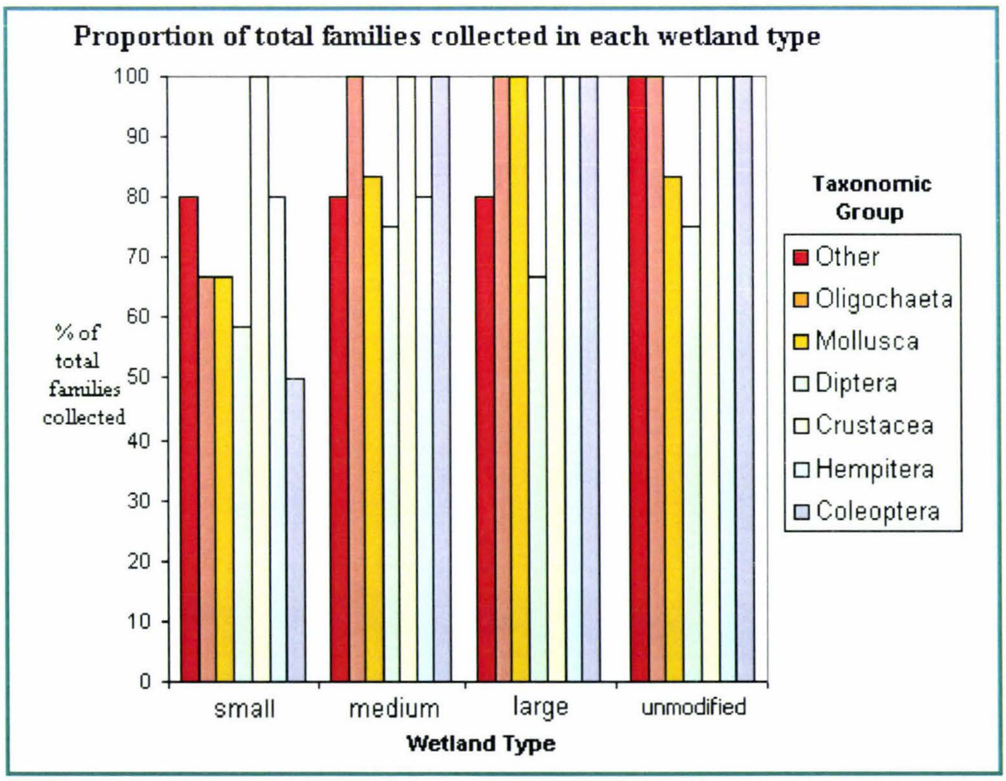

Figure 42: Proportion of total families found in each fragment size (mean ± 1 SE).

3.3.6 *Water Quality Parameters*

No significant differences were detected between levels of **conductivity** in the three fragment sizes. There was also no significant difference between the conductivity measured in the remnant and unmodified wetlands (Figure 43). Small fragments also showed higher **water clarity** than the other wetlands but this difference was not statistically significant (p>0.05) (Figure 44).

No significant differences were detected between the **pH** levels in three fragment sizes (Figure 45). Likewise, only very small water **temperature** differences were detected between different fragments (Figure 46). However, the temperature recorded are likely to be affecting by a number of confounding factors, including diurnal variation and the source of the waterway.

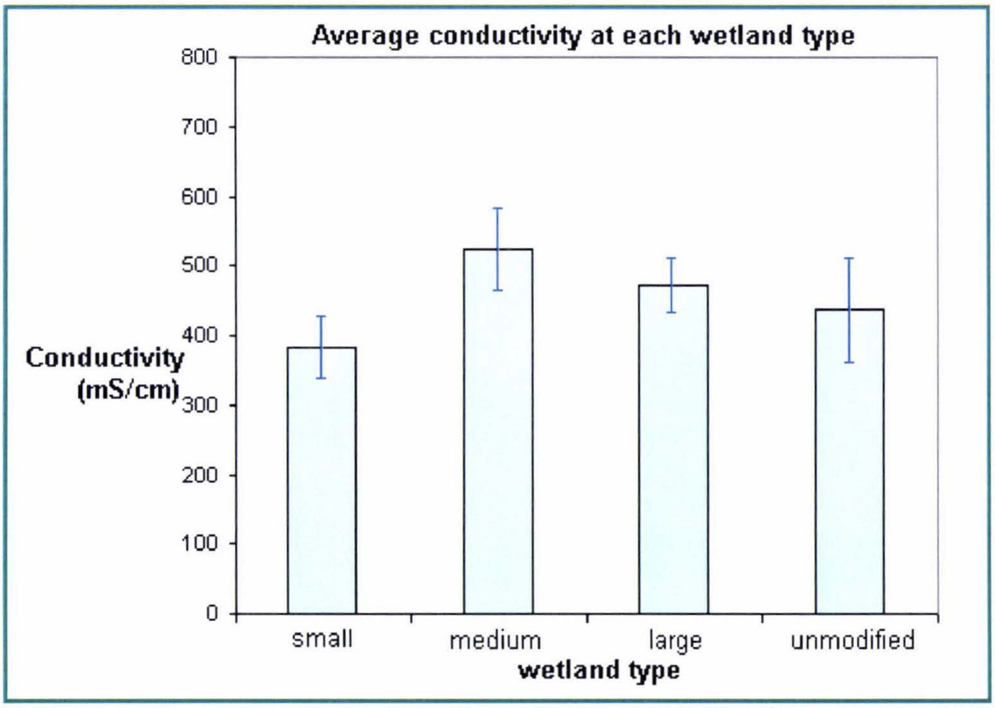

Figure 43: Average conductivity in each fragment size (mean ± 1 SE).

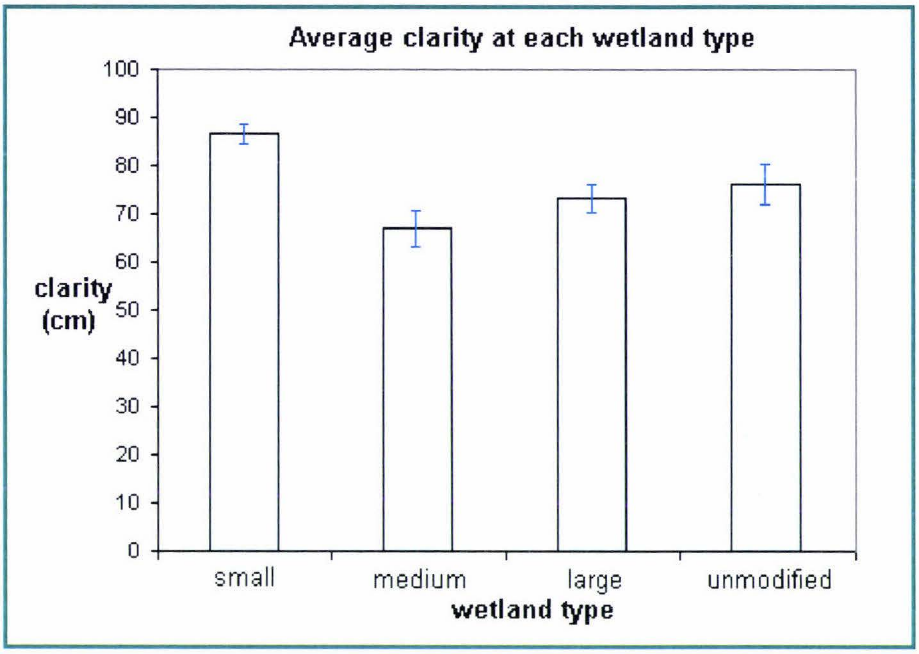

Figure 44: Average water clarity in each fragment size (mean ± 1 SE).

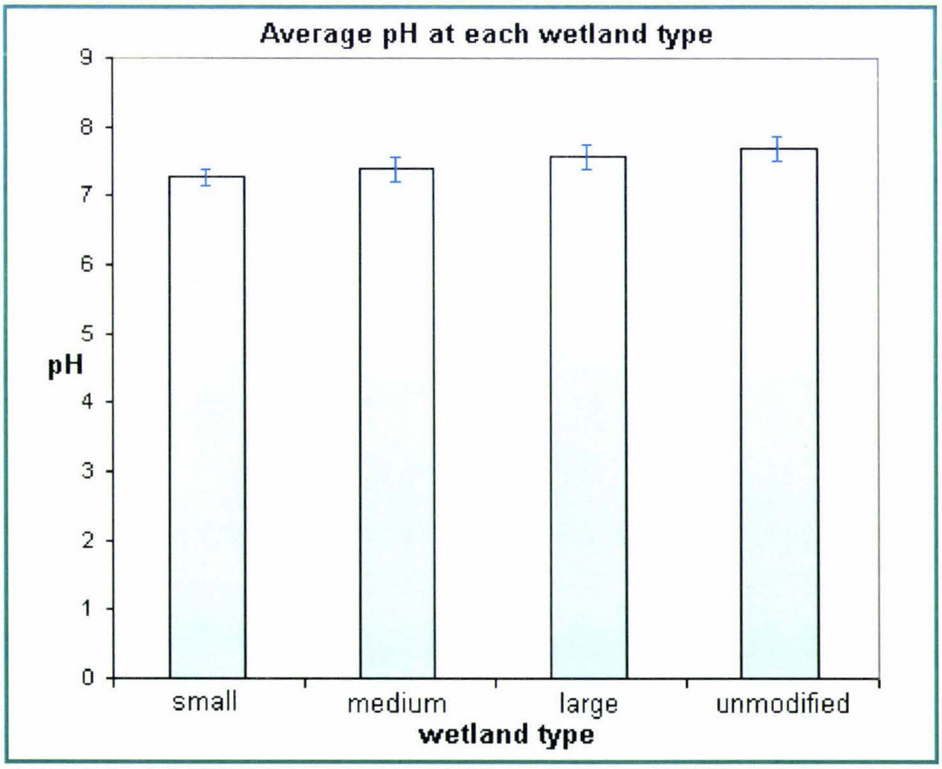

Figure 45: Average pH in each fragment size (mean ± 1 SE).

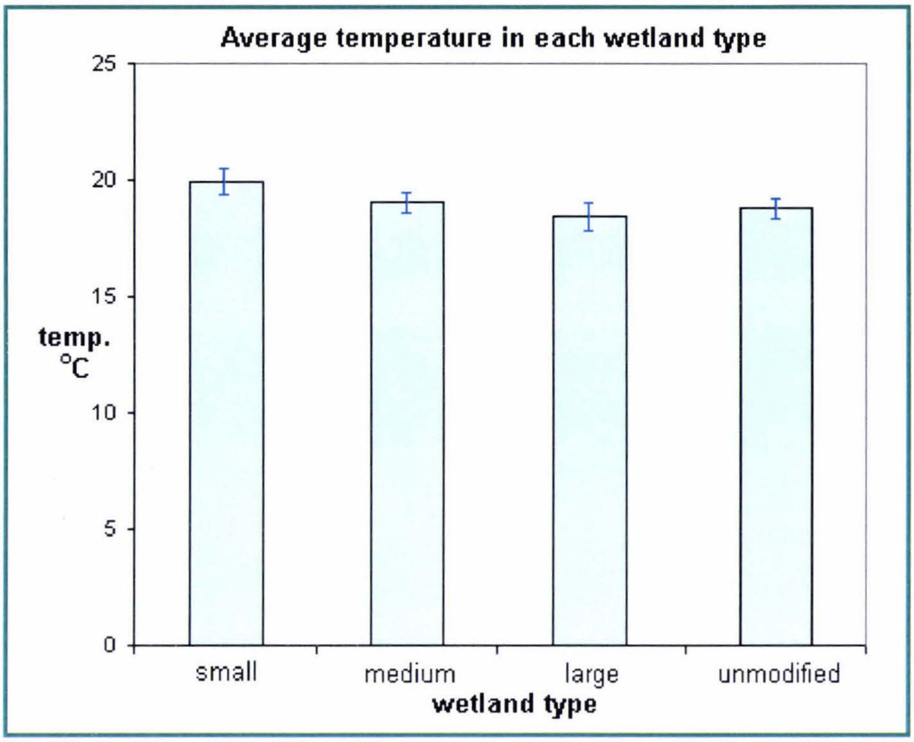

Figure 46: Average temperature in each fragment size (mean ± 1 SE).

3.3.7 Statistical analysis summary

Table 27: Fragment size significance: Tukey's Multiple Comparison.

Fragment Size	abundance	species richness (rtu)	species richness (family)	pH	temperature	conductivity	clarity
small vs medium	**N**	**N**	**N**	**N**	**N**	**N**	**N**
small vs large	**N**	**N**	**N**	**N**	**N**	**N**	**N**
medium vs large	**N**	**N**	**N**	**N**	**N**	**N**	**N**

N = no significant difference detected ($p>0.05$)

4.0 Discussion

4.1 *Effect of land-use on aquatic macroinvertebrates*

The average macroinvertebrate densities measured in the land-use study (650 - 1726 individuals/m^2; mean = 1,075 ± 460 SD). Unfortunately there are very few published reports of the macroinvertebrate densities found in other New Zealand wetlands with which to compare the results of this study. The average density is lower than those recorded for most surveys of New Zealand hard-bottomed streams (e.g. 2,784 individuals /m^2 recorded in hard-bottomed streams throughout New Zealand by Scarsbrook *et al.* 2000), but is similar to the 1,334 individuals/m^2 reported by Collier *et al.* (1998) in 20 lowland, soft-bottomed Waikato streams. The mean density, and community composition, is also comparable to a survey of the macroinvertebrates found in Lake Roxburgh (Contact Energy, 2000) which recorded a mean density of 1,588 individuals/m^2.

However, the mean density is significantly lower than those that have been reported in studies of wetlands in the United States. These typically report macroinvertebrate densities ranging from 4,000 - 12,000 individuals /m^2 (e.g. Anderson and Vondracek 1999; Tangen *et al.* 2002; Jenkins *et al.* 2003) This difference could be attributed to the fact that the riparian wetlands used in this study have faster water replacement than other wetland types, and hence accumulate lower levels of nutrients which could limit total productivity. The lower densities could also be due to the fact that the wetlands were sampled relatively early in the summer. Mosquito surveillance conducted by the Crown Public Health has shown that mosquito and midge populations usually peak in late February (Crown Public Health 2002) (Figure 47).

Although differences were observed between many of parameters measured, 1-way ANOVA did not detect a significant difference between the wetland types in a large number of cases. This could, in part, be due to the fact that only having three replicates for each wetland type did not provide the statistical power to detect small, but still significant, differences.

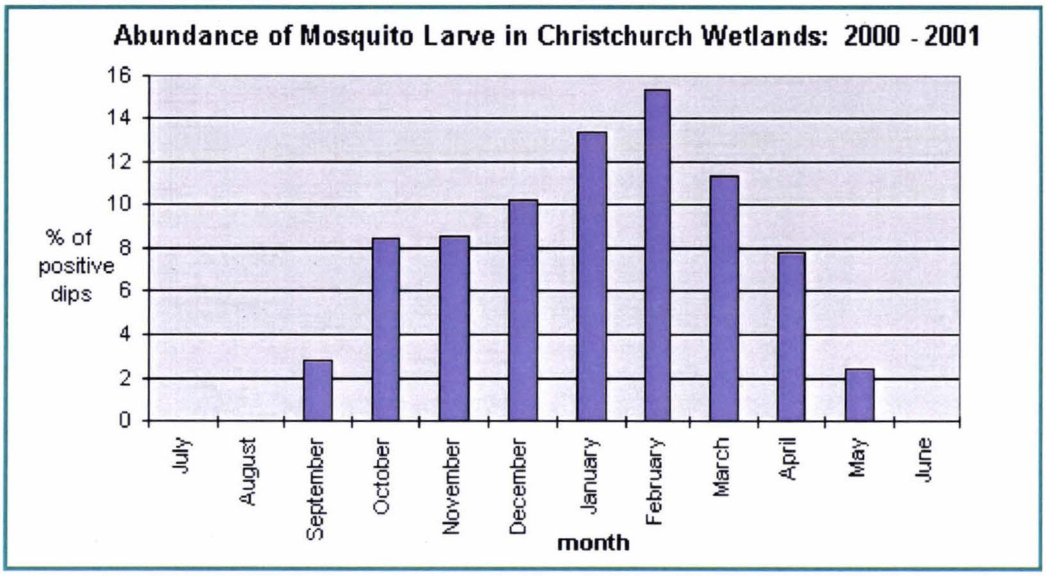

Figure 47: Seasonal fluctuations in mosquito populations in Christchurch. (source: Crown Public Health 2002)

The total number of taxa identified across all of the wetlands sampled is lower, and is a subset, of the taxa identified in many New Zealand stream surveys, and lacks many of the major groups that have been commonly found in soft-bottomed stream habitats. For example Collier *et al.* (2000) recorded 188 taxa from the Mangaotama catchment in Waikato, and Cowie (1983) recorded 182 taxa from Devils Creek in Westland. The most striking feature of the wetland invertebrate fauna, when compared to these stream studies, is the almost complete absence of Trichoptera, Ephemeroptera, Plecoptera, and Megaloptera. In similar fashion, the macroinvertebrates found in the pastoral and constructed wetlands appear to be a subset of those found in the unmodified and urban remnant wetlands. The most notable groups absent from pastoral and constructed wetlands are copepods, Hydrophilidae, Staphylinidae, and Hydrometridae.

These results are similar to those found in studies that have compared invertebrate abundance in pastoral streams and natural streams (e.g. Hamill and McBride 2003). Pastoral wetlands showed significantly higher macroinvertebrate populations and significantly lower species richness than the natural wetlands. Livestock were able to graze close to the edge of all the pastoral wetlands, and in the case of the Ataahua wetland, were able to cross the stream flowing into the wetland. The high level of macroinvertebrate productivity is almost certainly due to animal wastes and fertilisers being washed into adjacent waterways. This would be exacerbated by the absence of riparian buffer zones along the banks of the streams flowing into these wetlands. Livestock passing through streams would also stir up sediment releasing nutrients and increasing the suspended solids in the water column. This would directly increase the turbidity and conductivity, and indirectly increases the temperature by increasing absorption of solar radiation.

The elevated nutrient inputs appear to have skewed the composition of the macroinvertebrate community, allowing high productivity of pollution tolerant cosmopolitan species. This is a common phenomenon that has been observed in many polluted aquatic ecosystems (Figure 48) (Quinn *et al.* 1997). In addition to very high populations of midges, mosquitoes, snails and worms, highly eutrophied wetlands often experience 'algal blooms' that can dramatically lower dissolved oxygen levels and make the habitat even more unsuitable for sensitive groups.

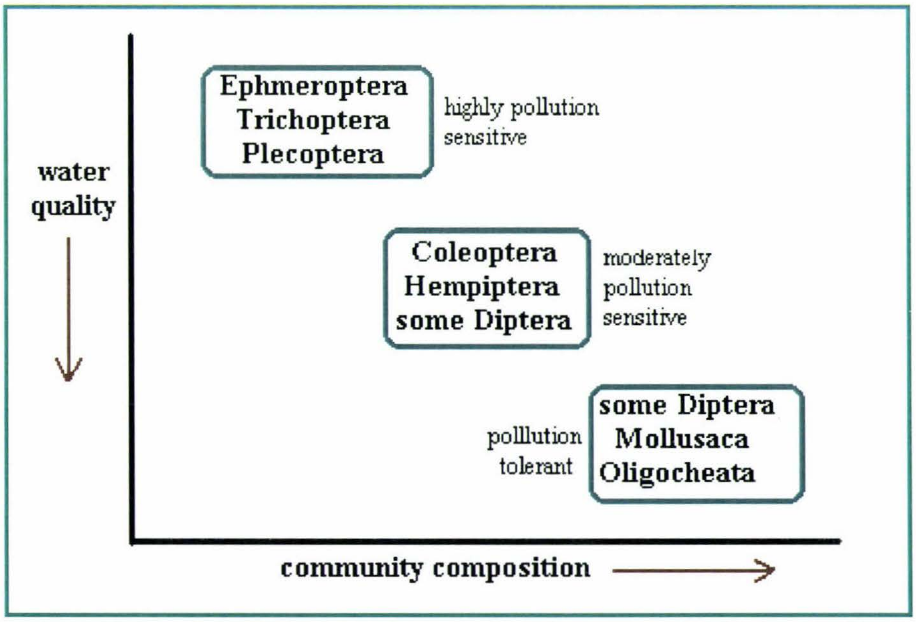

Figure 48: Response of aquatic invertebrate community to decreasing water quality. (original diagram)

Explosions in the populations of dipterans have also been correlated with the absence of coleopteran and hemipteran groups from the invertebrate wetland fauna (Euliss and Mushet 1999). Many of these species prey on midge and mosquito larvae and act as population regulators in a healthy balanced wetland system.

Constructed wetlands had a high diptera:coleoptera ratio (4.7:1) and had less species richness when compared to the remnant urban and unmodified wetlands. However, these wetlands did not exhibit the high productivity observed in pastoral wetlands. This is likely to due to the fact that most of these constructed wetlands were formed by the 'naturalisation' of drains and ditches, whereas the urban remnant wetlands are almost always associated with streams, most of which originate from natural springs.

Naturalised drains are drains that once flowed underground, or through narrow concrete culverts, but have since been allowed to flow more slowly through wider soil surface channels, often with intermittent ponds and small wetland areas, and many have had native vegetation planted along their 'banks'. Stormwater is a significant water source for most of these drains and hence they are likely to carry high sediment loadings from debris and soil washed off roads and gardens, and a range of toxins and hydrocarbons. The fact that the constructed wetlands are dominated by pollution tolerant snails, worms and midges but do not support large populations of these species suggests that the water quality is low and is relatively oligotrophic.

Urban remnant wetlands had similar levels of macroinvertebrate abundance and species richness when compared to the unmodified wetlands. This suggests that these remnant fragments have not been significantly impacted by adjacent urban development and they are still able to function as healthy, balanced habitats when compared to the natural wetlands. However the composition of taxonomic groups within urban remnants appears to have been somewhat modified with higher levels of crustacea and diptera, and slightly lower levels of coleoptera and hemiptera than the unmodifed wetlands.

4.2 Effect of land-use on wetland water quality

Conductivity

The high level of conductivity measured in the pastoral wetlands is most probably due to a high level of suspended and dissolved solids in the water column (also reflected in the low water clarity) and high concentrations of nutrients (e.g. NO_3^-, PO_4^{2-}). The elevated sediment loading in the streams associated with the pastoral wetlands is almost certainly a result of soil being washed off adjacent pasture and livestock passing through the streams.

Moderate levels of conductivity were measured in remnant and unmodified wetlands and this appears to be result from natural biological processes (e.g. breakdown of litterfall and detritus) and sediment input from associated streams and drains.
The conductivity measured in the constructed wetlands is probably due to nutrient and sediment inputs from urban stormwater.

pH

The unmodified wetlands had higher pH levels than the three other wetland types. This is probably due to higher rates of photosynthesis occurring in these habitats. These unmodified wetlands had visibly higher levels of floating pond-weed, emergent and benthic vegetation. High rates of photosynthesis increases the dissolved oxygen in aquatic environments, which in turn increases the alkalinity. The geology of the surrounding environment will also have a significant influence on the pH of each wetland.

The pastoral wetlands had lower pH levels than the other wetland types which is likely to be a result of anaerobic oxidation occurring in the sediments of the wetlands and high rates of heterotrophic microbial activity. These wetlands did have large amounts of filamentous green algae but this is probably a result of eutrophication rather than high rates of photosynthesis. The remnant and constructed wetlands had pH levels that are typical of lowland streams, and hence the pH of the wetlands is likely to be a result of the pH of the feeding waterways rather than the biological processes occurring within the wetlands.

Clarity

The low water quality measured in the pastoral wetlands is almost certainly due to high sediment loadings in the streams feeding the wetland areas. The source of these sediments is likely to be from soil washed off pasture and stream banks, and stream bank collapse caused by livestock. This problem would be exacerbated by the lack of riparian buffers that would exclude livestock and filter sediment from run-off. The constructed wetland also had reduced water clarity when compared to the remnant and unmodified wetlands and this is most probably due to sediment being washed off impervious surfaces (e.g. roads) into streams and drains, and then into the wetlands.

The remnant wetlands generally had greater amounts of riparian vegetation than the constructed wetlands and this would help to abate sediment being washed directly into the wetlands. The remnant wetlands appeared to fed by natural streams, whereas the constructed wetlands were generally fed by 'naturalised' drains.

Temperature

Both the pastoral and constructed wetlands had higher water temperatures than the other two wetland types. This is likely to be a result of the direct relationship between turbidity and water temperature, where water with higher turbidity absorbs more solar radiation and increases in temperature. The unmodified wetlands had a slightly lower water temperature than the remnant wetlands and this could be due to shading of the wetland and their associated streams by native bush.

The temperatures recorded will also be affected by a number of confounding factors such as the ambient air temperature, cloud cover and time of the day that they were sampled. They will also be affected by the depth of water and the size of the wetland area compared to the associated waterway. Wetlands with small feeding streams will have lower flushing rates, allowing standing water to heat up more.

4.3 *Implications for wetland management*

Pastoral Wetlands

From these results it appears that the pastoral wetlands support significantly different macroinvertebrate faunas than the natural wetlands. If natural wetlands in areas that are being converted to intensive agricultural are to survive in a form that resembles their original condition then they will require stringent protection and management. A range of techniques have been used to mitigate the adverse impacts of non-point nutrient run-off on waterways. These include the fencing off the waterways to exclude livestock, planting riparian buffer zones, raising stream banks, and placing sediment traps in drains. Riparian buffer zones have been shown to be very effective at filtering sediment and nutrients from agricultural run-off (Ministry for the Environment 2001).

Another useful management strategy would be to encourage less intensive land-uses (e.g. sheep grazing rather than dairy farming). An example of this is Lake Taupo in the North Island of New Zealand where farmers have received substantial payments from the local regional council if they are willing to convert to farming practises that result in lower nutrient inputs into the lake.

Constructed Wetlands

The constructed wetlands in this study appear to lack the natural hydrology required to form a healthy, balanced and fully functioning wetland system. This is consistent with the findings of other studies that have compared constructed and natural wetlands (Streever *et al.* 1997). Many researchers have suggested that healthy wetland systems cannot be created on sites that lack the hydrological cycles found in natural systems and such wetlands on such sites will inevitably form ponds unable to support specialised wetlands fauna and flora (Malakoff 1998). The constructed wetlands sampled in this study appear to be of less value for protecting native aquatic wetland biodiversity in an urban environment than the remnant wetlands.

Many of these constructed wetlands are fed by urban drains, which will almost always carry higher concentrations of suspended solids and toxins than a natural stream or spring. The catchments for these drains are usually large impervious areas (e.g. roads, parking areas) that quickly collect large volumes of stormwater that is rapidly discharged through the wetland. Hence, during heavy rainfall, the macroinvertebrate community within a constructed wetland fed by stormwater will experience sudden increases in water flow and inundation with sediment and toxins (e.g. oil, detergents).

The recent trend of placing artificial wetlands and ponds in new subdivisions has been implicated with an increase in the number of mosquito complaints received by the Christchurch City Council. Monitoring of these wetlands by Crown Public Health found significantly higher numbers of mosquitoes and midges in these wetlands than in naturally occurring wetland areas (Crown Public Health 2002). Under the Local Government Act 2003, local authorities must *"ensure that public amenities do not pose a hazard or nuisance to the local community"*. In response to these mosquito complaints the City Council has had to institute a regular spraying programme to control larvae number in several constructed wetland areas. Hence, it is very important that artificial wetlands are planned and constructed in manner that allows the wetland to support a diverse invertebrate community. This would prevent the habitat being dominated by a few nuisance species. Some researchers have found that restoring wetlands in sites that have previously been drained is a more successful strategy than constructing a wetland in a site that has no previous wetland history (Brown 1998).

Remnant Wetlands

According to the results of this study the aquatic invertebrate communities in remnant urban wetlands do not appear to have been significantly adversely impacted by adjacent urban development and appear to be just as capable of supporting a diverse and abundant aquatic invertebrate community as the unmodified reference wetlands. Hence, these remnant urban wetlands are of high ecological and heritage value as the represent the original character of the area. If these wetlands are given the right protection, and are monitored to ensure that the protection programmes are working, they should be able to act as urban reservoirs of indigenous biodiversity and valuable recreational amenities.

In contrast to the constructed wetlands, the remnant wetlands are almost always fed by a spring or a natural stream. Hence, they are much less adversely affected by urban run-off and stormwater than the constructed wetlands. They also tend to be located within reserve areas and the surrounding vegetation provides some protection from direct run-off and alien species. If the remnant wetlands are to continue functioning as healthy, balanced ecosystems, capable of supporting a wide range of invertebrate taxonomic groups, then it is important that these wetlands are protected as much as possible from the adverse impacts of urban stormwater. It is also important to protect these ecosystems from the adverse impacts of alien invasive plant species, especially aquatic weeds. Many native aquatic invertebrate species have a strong relationship with certain native aquatic plants (MacFarlane *et al.* 1998), and hence it is very important that these native species are not displaced by exotic species.

Unmodified wetlands

The unmodified (natural) wetlands used in this study are all located on the Banks Peninsula immediately adjacent to the main metropolitan area of Christchurch City. This area was chosen because it contains large areas of native bush and is reasonably close to the city. However, the whole Peninsula has been anthropogenically modified to some extent, including the removal of more than 85% of the original native vegetation to facilitate farming, and the recent establishment of several large *Pinus radiata* plantations (Environment Canterbury 2001). This has lead to severe erosion problems on some parts of the Peninsula and has caused changes in the flow and drainage patterns of many streams. This raises the question of how 'natural' and unmodified these wetlands really are. A useful comparison would be to compare the wetlands used in the study with truly isolated wetlands (e.g. on the West Coast). However, the invertebrate communities in these wetlands would also be affected by regional and climatic differences.

If the unmodified wetlands are to retain their current level of aquatic invertebrate biodiversity then it is important that any urban, agricultural or forestry developments in the immediate vicinity are planned in such a way that they do not impair the functioning of the wetland system. In particular, it is important that the upstream catchment is protected from erosion and eutrophication problems that are often associated with farming and forestry activities.

4.4 *Effect of fragment size on aquatic macroinvertebrate community*

Average invertebrate densities in the three different fragment sizes sampled (592 - 904 individuals/m^2) exhibited less variability than the densities measured in the first study of this report. No significant differences were detected between the abundance and species richness of aquatic invertebrates supported by the remnant medium fragments, remnant large fragments and natural wetlands in this study. However, the urban remnant wetlands showed definite shifts in the realtive abundance of the various taxanomic groups, with higher numbers of crustacea and diptera, and lower numbers of coleoptera and hemiptera. Small remnant fragments contained smaller populations and less species richness than the other two wetland sizes.

This suggests that the environmental pressures from adjacent urban development have not had a measurable effect on the invertebrate faunas present within medium and large fragments. Conversely, the differences observed in small fragments may be a result of changes in response to urban environmental pressures, or they may be a directly related to the smaller size of these wetlands. In order to determine if the effect is solely size-related, the abundance and species richness of small natural wetlands would have to be measured and compared to the medium natural wetlands.

The observed differences could also be caused by the fact that small riparian wetlands are more greatly influenced by their associated waterway than larger areas where the actual wetland is usually much larger than the size of the stream channel. There is also likely to be less nutrient accumulation due to stream flushing, and higher rates of organism drift.

4.5 *Effect of fragment size on wetland water quality*

Conductivity

Conductivity was higher in the medium size fragments than the small and large fragments. This may be due to the fact that the Bexley and Cockayne wetlands could have slight saltwater intrusion doing high tides as their associated waterways flow into the nearby Christchurch estuary. The higher 'flushing rate' and water exchange in the small fragments could also result in lower concentrations of nutrients and dissolved solids which contribute to conductivity.

pH

Periphyton and benthic vegetation was denser in medium and large fragments than in the small fragments and this is may explain the slightly higher pH levels measured in these wetlands. However the observed differences were very small and not statistically significant. The range of pH's measured (7.2 - 7.8) are ideal for the growth of most specialised wetlands flora and fauna and should not be a significant controlling factor on community abundance or composition.

Clarity

Generally these wetlands exhibited good water clarity levels (67 - 87cm). In fact, the water clarities were higher than those that have been measured in many stream studies. This is probably due to the fact that there was negligible water flow in the wetlands and suspended sediments quickly settle out of the water column. Small wetland fragments had higher water clarity than the two other wetland sizes or the unmodified wetlands. This could be due to higher rate of water exchange between the small fragments and their associated streams, which reduces that build-up of sediments.

Temperature

Water temperatures were slightly higher in the smaller fragments then the other two fragment sizes. This is likely to be due to the fact that the smaller fragments were generally shallower and had less benthic vegetation. The source of the feeding stream is also likely to have a strong influence on the temperature in each wetland. The streams feeding the large and medium fragments (e.g. Travis Wetland, Horseshoe Lake, Styx Mill) arose from nearby streams and may not have reached temperature equilibrium by the time they entered the wetlands. In contrast the small fragments (except Thistledown) tended to be located further downstream their associated waterways. The temperatures recorded will also be affected by the climatic conditions and time of day that they were sampled.

4.6 *Implications for urban biodiversity*

To date the wetland protection and enhancement programmes conducted by the Christchurch City Council have concentrated mainly on the largest remnant wetlands (e.g. Travis Wetland, Styx Mill Reserve, the Groynes, Otukaikino Wetland Reserve, Bexley Wetland). All of these areas have been granted legal protection and are zoned as 'Protected Natural Areas'. Travis Wetland Reserve and Styx Mill Reserve both have active community groups that arrange regular 'native planting days' and act as volunteer wardens (Christchurch City Council 2002a).

However, according to the results of this study, medium-sized urban remnant fragments (5,000m^2 - 10,000m^2) are able to support aquatic invertebrate populations of similar abundance and species richness as those found in large urban remnant fragments (>50,000m^2). This suggests that these medium-sized wetlands are just as valuable, and should be offered the same level of protection as the larger fragments from an aquatic invertebrate perspective. Many medium-sized fragments are located on the edges of the city, particularly around the Styx and Halswell Rivers, where a large number of new subdivisions are under construction or are planned. If, as this study suggests, these fragments are capable of providing habitats and refuge for the same abundance and diversity of native aquatic macroinvertebrates as the large wetland reserves then they should be afforded the same level of protection and management.

As part of their wetland restoration and enhancement programme the Christchurch City Council has 'naturalised' a large number of urban streams, ditches and drains by planting native vegetation, and removing the strict channelisation regime that previously existed. This has allowed the streams to pond in certain areas and form small riparian wetlands (Christchurch City Council 2002). However, the results of this study suggest that small riparian wetlands are able to support slightly lower invertebrate populations and less species richness than the other wetland types. Although no significant differences were detected between the three remnant urban fragment sizes, there is likely to be some threshold area, which is required to provide enough habitat to allow a fully-functioning wetland system, or is large enough to buffer against the impacts of adjacent urban development.

Rutledge (2003) discusses the concept of a 'threshold effect' in detail. He concluded that "*a species will persist until particular amount of habitat is reached and then exhibit relatively quick declines in population size or persistence*" (Figure 49). The threshold area will vary depending on the species involved (Tscharntke *et al.* 2002).

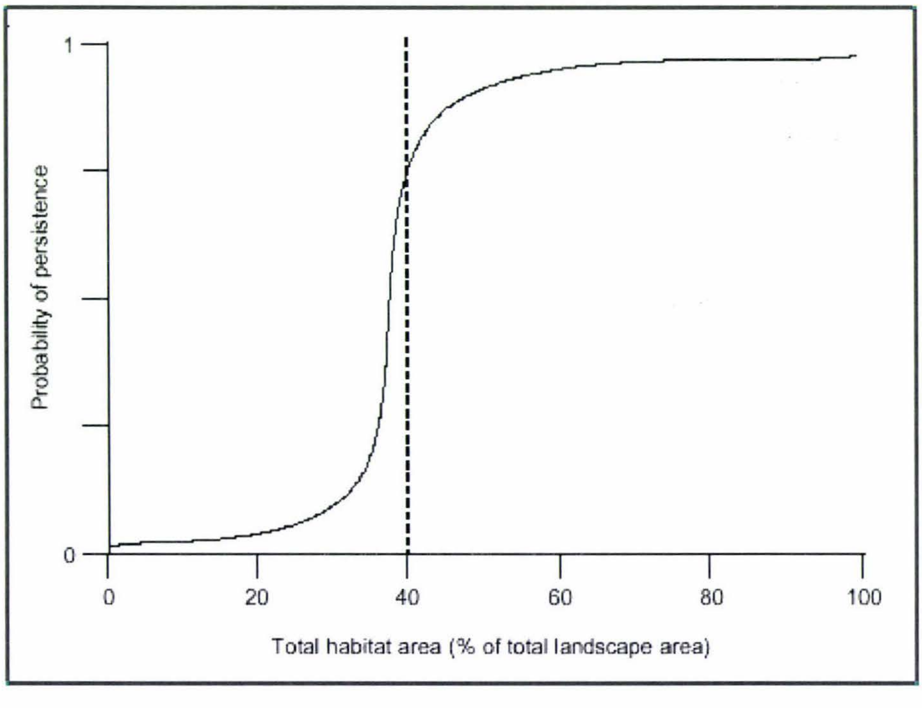

Figure 49: Habitat threshold effect (source: Rutledge 2003).

Hence, a large number of small wetlands scattered throughout Christchurch may not provide suitable habitat for some rare or native invertebrates. However, a small number of very large wetland reserves may also not provide the best solution for protecting native invertebrate biodiversity in an urban environment, as these areas may be geographically isolated from each other, preventing the exchange of genetic diversity, and may be susceptible to 'chance' events (e.g. disease outbreaks, flooding).

The slight differences observed between small and larger remnant fragments may also be attributable to the 'edge effects' discussed in section 3.1.2. The small wetlands have a much smaller proportion of interior core compared to the large wetlands, and the area of this core habitat may not be large enough to support a diverse range of specialised wetland fauna and flora. Predatory and generalist species will have much easier access to the interior of the fragment and will be more able to prey on and displace specialised species. This effect may explain the fact that the small fragments contained higher proportions of generalist aquatic species (e.g. *Daphnia* sp., *Paracalliope fluviatalis*, midges) and smaller populations of more wetland-typical species (e.g. *Rhantus* sp.).

4.7 *Future Studies*

This study was conducted on a relatively small scale with only a limited number of wetlands in each 'land-use type' and 'fragment size' sampled and assessed. This small sample size, and the high level of natural variability within these ecosystems, makes it difficult to detect statistically significant differences between the various wetland types. Small differences were observed in many of the parameters measured but these were subsequently not found to be statistically significant. Also, due to the lack of access to a high-powered compound microscope, and time constraints, it was not possible to identify many of the invertebrates collected to genus or species level. If this had been possible then important shifts in the composition within each taxonomic group may have been detected.

In addition, only a limited range of water quality parameters could be measured due to the lack of access to the required analytical equipment. Ideally, parameters such as nitrate and phosphate concentration, turbidity, suspended solids, and the level of environmental toxins could have been measured. This information could be have helped to explain some of the significant differences observed between wetland types. (e.g. is high invertebrate abundance and low species richness in pastoral wetlands is related to high nutrient levels?).

Hence, the results of this study could be verified, and elucidated, by conducting a similar study that included more wetlands within each wetland type and measured a wider range of parameters. A more sophisticated system could be developed to classify wetlands into different 'land-use types'. For example, a Geographic Information System (GIS) could be used to more accurately determine the characteristics of the catchment area for each wetland. The impact of land-use and fragment size on other components of wetlands ecosystems could also be assessed (e.g. aquatic vegetation, terrestrial vegetation, water chemistry, waterfowl, periphyton etc). Some of these parameters may be more sensitive to changes in land-use and fragment size, and hence, may be better indicators of anthropogenic disturbance.

Possible future projects related to this study:

Land-use

- Do the unmodified wetlands in this study resemble truly natural isolated wetland systems? Compare with wetlands on the West Coast.

- To what extent is water quality a controlling factor in these wetlands? Measure and compare water chemistry. Are there higher levels of environmental toxins in constructed wetlands and are these at a level that could affect invertebrate abundance and diversity?

- Determine the effects of land-use on wetland vegetation, terrestrial invertebrates, water chemistry, and avian abundance and diversity.

Fragment size

- Determine if there is a threshold wetland area above which no significant differences in macroinvertebrate population and diversity are observed.

- Determine if there are any significant differences between small, medium and large size unmodified wetlands.

- Determine the effects of fragment size on wetland vegetation, terrestrial invertebrates, water chemistry, and bird abundance and biodiversity.

- As the area of a wetland decreases which species are affected first and could this be used as an indicator of disturbance?

- Do areas on the edge of a wetland area support different assemblages than areas in the centre of a wetland? Perform gradient sampling across a large wetland.

5.0 Conclusions and Recommendations

5.1 *Conclusions*

1. Christchurch's remnant wetland areas have historical, cultural and ecological links extending over many centuries. These ecosystems are some of the few remaining areas that represent the original landscape and character of the Canterbury Plains and have been shown to contain some rare indigenous species.

2. Aquatic invertebrate faunas in remnant urban wetlands do not appear to have been significantly impacted by adjacent urban development and, given the right protection and management, they should be able to act as valuable urban reservoirs of indigenous biodiversity.

3. Medium and large urban remnant fragments both closely resemble unmodified wetlands and support a similar abundance of aquatic invertebrates and species richness.

4. Small urban remnant fragments contained smaller populations and less species richness than medium and large fragments. Small urban fragments are not considered to be as useful for protecting native biodiversity as larger fragments as they are likely to be more vulnerable to invasion by exotic species and may be not provide enough habitat to support certain specialised wetland species.

5. Constructed wetlands contain significantly less species richness than natural wetlands and appear to lack the natural hydrology required to form a 'fully functioning wetland system'. These wetlands would be of less value than remnant wetlands for protecting native invertebrate biodiversity.

6. Pastoral wetlands appear to have been negatively impacted by adjacent intensive agriculture and are possibly degraded beyond rehabilitation.

5.2 *Management recommendations*

1. The City Council should include the dozen or so 'medium sized' remnant wetlands areas in their protection and enhancement programmes, in addition to the large wetland fragments.

2. Larger integrated wetland areas are likely to be more resilient than a string of small wetlands alongside a stream or river, and hence the City Council should concentrate their wetland enhancement and protection on medium and large remnant wetlands rather than restoring small wetland areas along streams.

3. Comprehensive hydrological, drainage and soil testing should be conducted in order to determine if a proposed site is suitable to construct a wetland. Ideally, artificial wetlands should be constructed on sites that previously supported wetland ecosystems.

4. Constructed wetlands should be planned in a manner that minimises the potential for large populations of nuisance species (e.g. mosquitoes, midges). This could include installing sediment traps.

5. Remnant wetlands should be protected from the potential adverse impacts of urban stormwater. This could include ensuring the appropriate riparian buffers are in place, and drains do not discharge into the wetlands or within a certain distance upstream.

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Appendix A

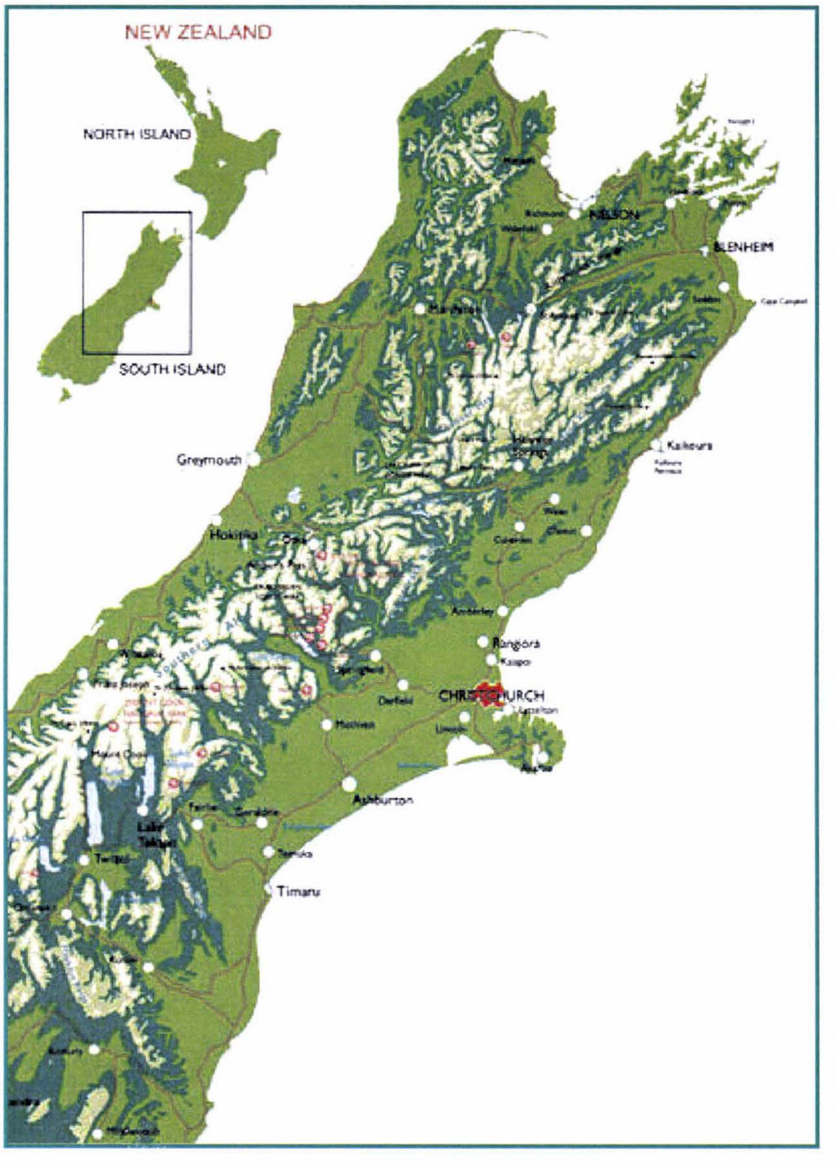

Figure 1: Map of South Island & Christchurch (source: wises.co.nz)

Figure 2: Major Christchurch waterways (source: www.ccc.govt.nz)

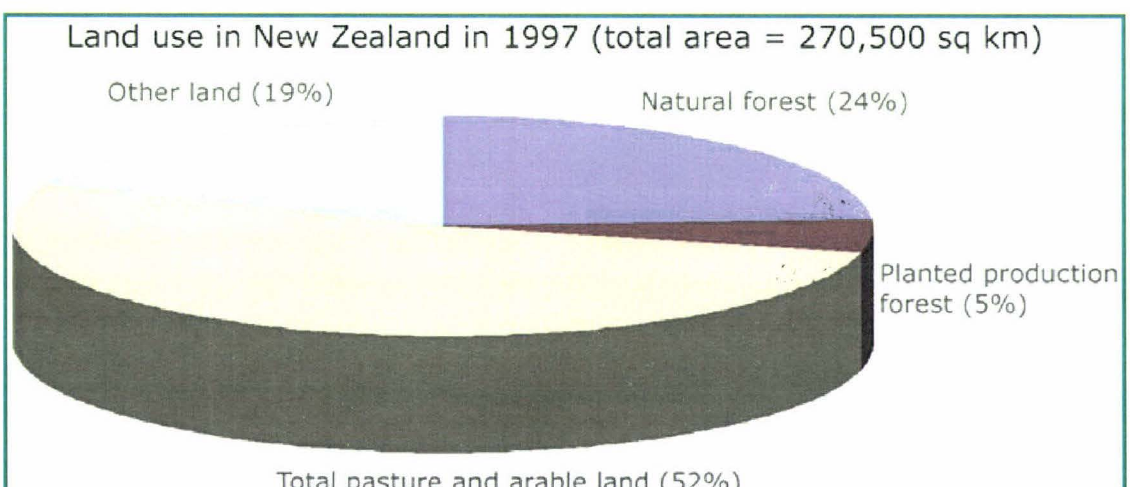

Figure 3: Land use in New Zealand (source: Ministry of Agriculture and Forestry 2002a)

Appendix B

Modified Aquatic Invertebrate Collection Protocol C2: Soft-bottomed, Semi-quantitative habitats

Aquatic Macrophytes - Macrophyte beds are sampled by jabbing the net in submerged plants for a distance of approximately 1-metre to dislodge organisms, followed by 2 - 3 cleaning sweeps. Place clumps of plants that have been disturbed in the net (do not uproot) and shake and brush by hand to dislodge organisms. Collection is done off the bottom of the wetland to avoid the collection of fine detritus, sand, and mud. Filamentous algae are avoided where possible. Each unit collection effort represents an area of approximately 0.3 m^2.

1. Sample 2 different areas of emergent vegetation and 2 different areas of submerged benthic vegetation. Remove sample material to a bucket or sieve bucket after each collection to avoid clogging the net. The bucket or sieve bucket should now contain one entire sample comprising material dislodged from 1.2 m^2 of emergent and benthic vegetation.

2. Fill the bucket with water and rinse and remove any unwanted large debris items (e.g., sticks, leaves) that may not fit into the sample container or will absorb and diminish the effectiveness of the preservative.

3. Transfer the sample to the sample container via a 0.5 mm sieve if a sieve bucket is not used. Two containers may be needed; each container should be no more than 2 /3 full with sample material. Inspect the sieve or sieve bucket and return any macroinvertebrates to the sample container.

4. Add preservative. Aim for a preservative concentration in the sample container of 70-80% (i.e., allowing for the water already present). Be generous with preservative for samples containing plant material (leaves, fine detritus, algae, moss, and macrophytes).

5. Place a sticky label on the side of the sample container and record the site code/name, date, and replicate number (if applicable) using a permanent marker. Write on the label when it is dry and do not rely on a label on the pottle lid! Place a waterproof label inside the container. Screw the lid on tightly.

6. Note the sample type (e.g., D-net), collector's name and preservative used on the field data sheet.

7. Record notes on the field data sheet describing the proportion of habitat units sampled.

Sorting and Screening Protocol P3: Full Count with Subsampling Option

1. Sieve and place the sample in grided sorting trays.

2. Starting with the largest size fraction, work systematically across each tray removing all of the organisms in the sample. Normal eyesight should be precise enough to detect organisms > 1mm in total length. Do not use magnification.

3. Place the organisms of each taxon encountered into separate Petri dish to confirm identifications by microscopic examination (if necessary). Place sorted animals into vials or pottles containing 70% alcohol for storage and QC.

4. Place a label in the vial or pottle noting the site code/name, date, sample type, and collector's name. Label multiple containers (e.g., "1 of 2, 2 of 2).

5. On completion of sample processing, there should be (1) labelled vials or pottles containing sorted organisms, and (2) the preserved sample residue in its original plastic pottle with the original label.

Subsampling Option:

(Note: Only very abundant taxa should be subsampled. Full counts should be made for all other taxa).

1. Subsampling of very abundant taxa (> 500 individuals) can save considerable time.

2. Count the number individuals of each very abundant taxon from a fixed fraction (between 10% and 50% recommended) of the sample grids for each sorting tray. Estimate the total abundance for that taxon by multiplying the number counted by between 10 (for 10% fraction) and 2 (for 50% fraction) according to the fraction of the sample that was counted.

3. Record the count estimate on the bench data sheet and note that the value is a subsampling estimate (e.g., 25% fraction)

4. Remove 10-20 representatives of each taxon subsampled and store in a separate vial or pottle from that containing the other sorted organisms.