Drivers of Wetland Zooplankton Community Structure in a Rangeland Landscape of the Southern Interior of British Columbia

by

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ABSTRACT

Zooplankton play a vital role in aquatic ecosystems and communities, demonstrating community responses to environmental disturbances. Surrounding land use practices can impact zooplankton communities indirectly through hydrochemistry and physical environmental changes. This study examined the effects of cattle disturbance on zooplankton community structure in wetlands of the Southern Interior of British Columbia. Zooplankton samples were obtained from fifteen morphologically similar freshwater wetlands in the summer of 2009. Physical, chemical and biological characteristics of the wetlands were also assessed. Through the use of Cluster Analysis and Non-metric Multidimensional Scaling (NMDS), differences in community assemblages were found amongst wetlands. Correlations of environmental variables with NMDS axes and multiple regression analyses indicated that both cattle impact (measured by percent of shoreline impacted by cattle) and salinity heavily influenced community structure (species richness and composition). *Leptodiaptomus* was the dominant copepod genus, *Hexarthra* and *Keratella* the dominant rotifer genera, and *Ceriodaphnia* and *Daphnia* the dominant cladoceran genera. Species richness decreased with both increasing cattle presence and increasing salinity, with copepods and cladocerans dominating. Least-impacted, least-saline wetlands were characterized by diverse rotifer-dominated assemblages. Impacted wetlands pertained to the highest salinity readings, thus both salinity and cattle impact appeared to have similar confounding effects on community structure. This study supports the use of zooplankton in monitoring programs in B.C. wetlands as there was a significant response to intensifying cattle impact and considerable community differences within small geographical ranges; however, future research is required which examines effects of cattle without confounding results of environmental parameters such as salinity.

Keywords: Zooplankton; Wetlands; British Columbia; Cattle; Salinity; Non-Metric Multidimensional Scaling; Land Use; Species Richness; Community Structure

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CHAPTER 1
INTRODUCTION

With the escalation of the world's population and the escalation of the exploitation of natural resources, human land use practices and their effects on the environment are issues in the forefront of scientific research. Understanding the pathways of impacts due to anthropogenic activities is vital for appropriate management of the earth's ecosystems and resources (Tilman, 1999). Loss of biodiversity is a well-known effect from many types of disturbances in natural systems. In freshwater environments, biodiversity make up a precious natural resource, economically and scientifically (Dudgeon et al., 2006). Although contributing to barely 0.01% of the earth's water resources, freshwater environments support almost 6% of all known aquatic species (Dudgeon et al., 2006). Land use practices such as urbanization, industry and agriculture are having particularly detrimental effects on freshwater ecology and biodiversity (Hughes et al., 2000). As consequences of human land use are becoming ever more apparent, it is vitally important to discover ways to decrease the negative impacts of these practices and understand what controls diversity.

In the scientific community, ecological disturbance is broadly accepted as a leading factor which negatively impacts species diversity (Mackey & Currie, 2001). Land use practices have been shown to contribute to local extinctions and ultimately a reduction of biological diversity in both aquatic and terrestrial systems (Brönmark & Hansson, 2002; Foley et al., 2005). Declines in freshwater biodiversity as a result of human land use practices are much greater compared to terrestrial systems (Dudgeon et al., 2006). Aquatic biodiversity has vast economic and aesthetic values and is principally responsible for the sustainment of "healthy" habitats (United States Environmental Protection Agency, 2010). Aside from species loss, a reduction in biodiversity can potentially have severe consequences for ecosystem functions. The abundance and composition of species assemblages determine traits of the organisms present, ultimately influencing ecosystem processes (Brönmark & Hansson 2002). As preservation of species diversity is central for ecosystems to function, protection of freshwater biodiversity is an over-riding conservation priority (Dudgeon et al., 2006) and a common goal in wetland conservation studies.

Wetlands are unique ecosystems that are vitally important to the surrounding landscapes and the countless organisms which depend on them (Mitsch & Gosselink, 2007). While wetlands can vary greatly in hydrology, chemistry, size and geography, they can generally be defined by shallow waters or saturated soils at or near the soil surface, and the presence of vegetation that is adapted to saturated or "hydric" soils (Committee on Characterization of Wetlands, 1995; Mitsch & Gosselink, 2007). The unique hydrological, vegetative and trophic characteristics of wetlands contribute to their high compositional dissimilarity (Williams et al., 2004; De Meester et al., 2005). Playing a large role in metapopulation dynamics (Gibbs, 1993), they are among some of the best environments for studying metacommunity ecology (Soininen et al., 2007).

Globally, there is urgent need for wetland conservation efforts. While the need for conservation measures is highlighted by measurable species and habitat loss, wetlands remain under heavy threat (Turner et al., 2000). It is estimated that over fifty percent of the world's wetlands have been lost, with some regions approaching 99% (Van der Valk, 2006). One of the principal reasons for continued large-scale wetland loss is the enduring perception of their low worth and value (Turner et al., 2000). Historically perceived as harbouring disease, impeding agriculture and obstructing settlement and travel, wetlands in the United States have a long history of being eliminated (Dahl & Allord, 1997). Canadian wetlands share a similar unfortunate past, traditionally considered as unnecessary obstacles impeding land development; many have been actively eliminated due to these reasons (Zoltai & Pollett, 1983).

Wetlands are both susceptible yet adaptive systems that provide boundless services important to humans (Turner et al., 2000). Essential features within their surrounding landscapes, wetlands provide environmental stability on both local and global scales. In localized systems, wetlands are extremely important regulators of water, acting as sponges that both retain and restore groundwater which also moderates the effects of floods (Bardecki, 1984). They function as environmental kidneys by improving water quality through absorbing and retaining pollutants, excess nutrients and sediments from the landscape - many of which are further broken down into non-harmful components (Mitsch & Gosselink, 2007). Globally, wetlands assist in stabilizing atmospheric nitrogen, sulphur, methane and carbon dioxide levels, and have a significant yet underappreciated role in the global carbon cycle (Mitsch & Gosselink, 2007).

Wetlands support millions of people worldwide, offering goods and services far beyond the local wetland environment (Barbier, 1997). From an economic perspective, wetlands provide benefits to both individuals and society through a range of goods and services, some of which include timber, furs, fish, ducks, peat, carbon sequestration, flood reduction, purification of water, species habitat and recreational activities (Woodward & Wui, 2001). Extrapolating the calculated value of wetlands from Costanza (1997), environmental economists have estimated that the total value of wetland goods and services in British Columbia yield a potential value of over 100 billion dollars a year (Cox & Cullington, 2009). A recent study estimated that just a single hectare of wetland in B.C.'s Fraser Valley region yields a goods and services value in the range of approximately $6,000 – $25,000 per year (Olewiler, 2010).

Biologically, wetlands are considered some of the most diverse ecosystems in the world and are hot spots of plant and animal biodiversity (Williams, 1999). These biological "supermarkets" provide unique habitats for an extensive variety of aquatic and terrestrial organisms (Mitsch & Gosselink 2007). Wetlands are host to numerous large and small mammals, amphibians, reptiles and invertebrates. In British Columbia, wetlands make up merely 7% of the province's total land area; however, many bird, mammal, amphibian and reptile species depend on wetlands for survival, including a large number of species at risk (Delesalle, 1998; MacKenzie & Shaw, 2000; Wetland Stewardship Partnership, 2010).

In arid and semi-arid climates, wetlands are arguably even more imperative to the health of the surrounding landscape compared to less arid regions (Williams, 1999). The Southern Interior of British Columbia is a semi-arid environment hosting some of the hottest and driest zones in Canada (Environment Canada, 1997). In this region, wetlands are relatively uncommon and as such, are vital sources of water and refuge for the numerous organisms that depend on them. Land use practices in this region which contribute to the destruction, isolation and alteration of wetlands include urbanization, livestock grazing, forestry, crop production and hydrological alterations (MacKenzie & Shaw, 2000). Due to the ever increasing urbanization in the Southern Interior region, natural well-functioning wetlands have been drastically reduced (Wetland Stewardship Partnership, 2010). The deterioration and loss of these wetlands has significantly contributed to the numbers of species at risk, many of which are critically dependent on interior wetlands for survival (Delesalle, 1998; Wetland Stewardship Partnership, 2010).

Worldwide, wetlands provide important habitat for innumerous bird species. In North America, approximately a third of waterfowl species are dependent upon wetlands for feeding, breeding and shelter (Kroodsma, 1979). Certain species have adapted to specific types of wetlands in such a way that without them, they cannot survive (Stewart, 2001). Because of their importance to waterfowl, the destruction and degradation of wetlands has been directly attributed to declines in the abundance of many bird species (Stewart, 2001).

In the Southern Interior region of British Columbia, wetlands are essential for almost a quarter of the province’s waterfowl which utilize wetland habitats as imperative resting, feeding and breeding grounds (Delesalle, 1998; Trochlell & Bernthal, 1998). Human activities within the wetland environments are having negative impacts on these environments and their waterfowl. In these predominantly grassland environments where livestock cattle ranching has historically been a prominent industry (BC Ministry of Agriculture, Food and Fisheries, 2004), wetlands are used as a major source of available water for rangeland livestock, particularly during the hot and dry summer months.

Provincial Acts and Regulations allow cattle free access to natural sources of water under the requirements that Best Management Practices be followed and water bodies are not polluted (BC Cattleman's Association, 2012); however, these guidelines are ultimately the livestock owner's responsibility.

Cattle may potentially have severe impacts on their environments, both directly and indirectly (Steinman et al., 2003). Direct impacts include vegetation removal, aquatic nutrient input and sediment trampling. Trampling, or pugging, exposes soil to increased erosion, decrease soil water retention and disrupts succession of plants (BC Forest Practices Board, 2002). Indirect impacts can occur through trophic reactions which may ultimately affect ecosystem productivity in not only terrestrial, but aquatic environments to which cattle are exposed (Dodson et al., 2005). Grazing cattle have been shown to negatively interfere with wetland waterfowl populations through the reduction of vegetation available for nesting and shelter. As a result, decreased numbers of pairs and broods for species of ducks prevalent in the area have been observed (Bruce Harrison, pers. comm.)

In British Columbia, wetland loss has been severe in areas of developmental interest to humans. In the 20th century, wetlands province-wide were drained for large-scale agricultural and industrial purposes, water diversion projects and urbanization (Wetland Stewardship Partnership, 2010). Particular areas have been targeted more than others; for example, it is estimated that in the South Okanagan alone, 85% of the natural wetlands have vanished (Wetland Stewardship Partnership, 2010). Despite efforts to counteract the long perceived notion of wetlands being of low value and worth, this view persists, particularly in developing countries. Because wetland conservation is of low priority in these regions, vast destruction continues at an unknown economic and social cost (Turner et al., 2000). Wetland degradation, through pollutant and nutrient loading is expected to continue in nations that are developing most rapidly (Brinson & Malvárez, 2002); therefore, it becomes increasingly more pressing to initiate further research to promote the conservation of these ecosystems.

Human activity is the biggest threat to health of wetlands, and as such, studies examining the effects of various land use practices are at the forefront of wetland research. Disturbance studies frequently examine the effect of land development, deforestation, agricultural practices and various recreational activities on the physical and biological components of aquatic systems. In many of these studies, ecological indicators are used to quantify habitat conditions (Stemberger et al., 2001). Indicators can be any physical, chemical or biological parameter; however, specific groups of organisms are better indicators of environmental conditions (Holt & Miller, 2011). Biological indictors, or “bioindicators”, can give excellent indications of ecosystem health and can serve as early warning systems of potential threats to aquatic systems such as pollution and degradation (Holt & Miller 2011; United States Environmental Protection Agency, 2011). In wetland systems, bioindicators are a trusted method for monitoring wetland changes, and can be a potentially invaluable tool for land managers (Seilheimer et al., 2009).

Invertebrates are excellent indicators of aquatic health and are commonly used biodindicator organisms (Hodkinson & Jackson, 2005). Invertebrates can indicate environmental change through their responses both as individuals and as a community (Hodkinson & Jackson, 2005). In wetland systems, aquatic and terrestrial insects, freshwater crustaceans (e.g., amphipods), aquatic annelids (e.g., aquatic earthworms) and zooplankton are commonly studied indicator organisms and have been proven to react strongly to changes in trophic conditions (Gannon & Stemberger, 1978; Adamus & Brandt, 1990).

Zooplankton, the free-swimming and principally microscopic invertebrates of aquatic environments (Harris et al., 2000), are familiar and frequently used study organisms in laboratory and field studies (Hoffmann & Dodson, 2005). Both the consumers and the consumed, zooplankton play an intermediary trophic role, reflecting both top-down and bottom-up processes (Zimmer et al., 1999). They have become increasingly relied upon as simple, robust indicators due to their ease of sampling and identification (Whitman et al. 2004), quick response time to environmental change

(Gannon & Stemberger, 1978; Holt & Miller, 2011) and predictable variability in wetland systems (Lougheed & Chow-Fraser, 2002). Indices of biological integrity, scientific tools using various bioindicators to assess overall water body health, have been applied to zooplankton; however, these indices are usually regional specific and require rigorous testing usually over long time periods in order to provide valid results (Fore & Conquest, 1994; U.S. Environmental Protection Agency, 2012).

Examining zooplankton in wetland systems in B.C. rangelands could give important insight into how livestock affect community structure. In the Southern Interior of British Columbia, implementing and controlling sustainable cattle ranching practices is necessary to sustain healthy grassland and wetland ecosystems, not only for the multitude of organisms that depend on these habitats for survival, but also for the economic and recreational benefits, and aesthetic appeal for the people of British Columbia.

Understanding the connections between zooplankton community structure and cattle grazing may provide some of the necessary information required to create effective management strategies in order to promote the sustainability of the cattle ranching industry whilst conserving ecosystem function and biodiversity into the future. The literature is saturated with regards to zooplankton dynamics in lake ecosystems; however, there is much less knowledge regarding zooplankton communities in wetland ecosystems (Lougheed & Chow-Fraser, 1998). Amongst agriculture-focused ecological disturbance studies, the effect of livestock impact on wetlands remains relatively unexplored (Steinman et al., 2003).

The objectives of this study were to characterize wetlands by examining their individual zooplankton assemblages and to determine which measured environmental and anthropogenic factors were most related to differences in zooplankton community structure. The overall hypothesis was that community structure would be most influenced by cattle activity within the watershed compared with the influences of various other measured biological, chemical and wetland morphological parameters. Anthropogenic activities such as land development and agriculture have been shown to have significant direct and indirect influences on the way zooplankton communities are structured in freshwater wetlands (Beaver et al, 1998; Dodson et al., 2005; Hoffmann & Dodson, 2005). Therefore, it could also be expected that cattle presence within wetlands, through habitat degradation and nutrient input, will have measurable effects upon zooplankton.

My field study was conducted at fifteen cattle-accessed wetlands in the interior of British Columbia and involved sampling and analysis of zooplankton, hydrochemistry and morphological parameters. Patterns in community data as well as relationships between zooplankton community structure and measured environmental variables of the wetland environment were analyzed. Wetlands were categorized into impacted and non-impacted categories, and these were compared to examine differences in biological and hydro-chemical variables. Potential connections between cattle impact, measured environmental variables and patterns in zooplankton community structure were explored.

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CHAPTER 2

Drivers of Wetland Zooplankton Community Structure in a Rangeland Landscape of the Southern Interior of British Columbia

Introduction

Wetlands are essential features of many types of landscapes worldwide, having important functions such as water storage, providing essential habitats, and absorption of excess nutrients and pollutants. Known as hot spots of biodiversity, wetlands serve as vital habitat for the survival of many species, including many species at risk (MacKenzie & Shaw, 2000). Supporting millions of people across the globe through the goods and services they provide, wetlands are also very important from an economic perspective (Barbier, 1997; Turner et al., 2000). Under-protected and fragile to the effects of overexploitation, wetland ecosystems and the multitude of benefits provided by them are under considerable danger (Turner et al., 2000; Mitsch & Gosselink, 2007).

The Southern Interior region of British Columbia hosts one of the hottest and most arid climates in Canada (Environment Canada, 1997). Within this grassland-dominated landscape, wetlands are major sources of water for wildlife and livestock (British Columbia Ministry of Environment, 2004). Due to their significant water retention abilities and support of local ecosystem biodiversity, wetlands play a vital role in B.C.'s grassland ecosystems. Human population increase and associated anthropogenic activities such as urban and rural development, intensive recreational use and agricultural practices have resulted in the degradation, isolation and even disappearance of many wetlands in B.C.'s interior (Bradford & Irvine, 2000; MacKenzie & Shaw, 2000). Large areas of land have been dredged and drained for conversion to croplands, vineyards, orchards and rangelands (British Columbia Ministry of Environment, 2004).

Livestock ranching is a prevalent industry in the Southern Interior region. In a dry landscape, cattle naturally exhibit a preference for wetlands due to the availability of water, thermal cover and the increased quality of forage in the riparian zone, particularly during the dry summer months (Warner & Hendrix, 1984). Under unrestricted grazing regimes wetlands can rapidly show signs of the damaging effects of overgrazing (Kauffman & Kreuger, 1984). Improper range practices can have a host of potential impacts on wetlands, some of which include the removal of plant biomass, trampling of vegetation, nutrient input (through urination and fecal deposition), increased turbidity, and reduced soil quality (Kauffman & Kreuger, 1984; Fleischner, 1994). Trampling and removal of vegetation directly impacts wetland-associated organisms by eliminating habitat and food sources (Reeves & Champion, 2004), and can change riparian areas into exposed patches void of vegetation (Kauffman & Kreuger, 1984). Trampling can also destabilize wetland banks and compact the soil, leading to large potential alterations in wetland hydrology (Belsky et al., 1999). Fecal and urine inputs can alter natural nutrient fluxes in the surrounding terrestrial and aquatic environments, affecting water quality and productivity and potentially causing changes up the trophic levels which can result in changes in animal and plant species composition (Chase, 2003; Steinman et al., 2003). Increased turbidity caused by livestock entering the aquatic environment can interfere with predator-prey relationships as well as affect feeding abilities of filter-feeding organisms (Abrahams & Kattenfield, 1997; Lougheed & Chow-Fraser, 1998). These effects can be further compounded during the summer months, when high temperatures drive cattle to increase their water uptake and frequent wetlands more regularly. Because of the potential effects caused by the usage of wetlands by cattle, it is important to try to quantify potential disturbance effects in order to mitigate hazardous practices and justify wetland protection measures.

One way to detect disturbance impact upon wetlands is to analyze changes in communities of organisms that are particularly sensitive to environmental change. Zooplankton, consisting of the microcrustacean groups Copepoda and Cladoceran, as well as all individuals from the phylum Rotifera, have been assessed as indicators in freshwater ecological disturbance studies (Holt & Miller, 2011), examining individual groups and their combined communities as a whole. Zooplankton communities are affected by local environmental parameters (Dodson et al., 2009) including physical variables (e.g., wetland depth and area), chemical variables (e.g., salinity, pH, nutrients) and biological variables (e.g., predator/prey interactions, presence of vegetation) (O'Brien, 1979; Arnott & Vannie, 1993; Cottenie et al., 2001). Primary producers also have a significant influence on zooplankton dynamics (Carpenter et al., 1985; Canfield & Jones, 1996). The tight coupling between primary productivity, measured by phytoplankton chlorophyll-*a*, and zooplankton, is well documented for freshwater aquatic systems (Carpenter et al., 1985; McQueen et al., 1989) and has been correlated with changes in the composition of zooplankton communities (Allen et al., 1999).

Livestock grazing may potentially influence zooplankton in a variety of ways. Nitrogenous wastes are deposited directly into the aquatic environments by cattle (Bagshaw, 2002), potentially causing aquatic nutrient spikes, depending on stocking density and size of water bodies. Increases in nutrients can trigger trophic cascades which lead to changes in algal biomass and aquatic invertebrate composition (Carpenter et al., 1985). Zooplankton are heavily influenced by invertebrate predators (Havens 1990; Herwig & Schindler, 1996) as well as vertebrate predators (Hanazato & Yasuno, 1989; Frisch et al., 2007). As heavy livestock trampling of the wetland riparian environment depletes habitat required by organisms that feed on zooplankton species (e.g., invertebrates, waterfowl, and amphibians) (Reeves & Champion, 2004), changes in predatory organism communities caused by cattle disturbance of their wetland habitats should also have a measurable effect on zooplankton assemblages. The depletion of wetland vegetation by cattle also affects zooplankton directly. Many zooplankton species are directly associated with aquatic vegetation (Wetzel, 2001). Agricultural land use, which can lessen riparian and aquatic vegetation, is known to be associated with lower zooplankton diversity (Dodson et al., 2005). Cattle activity may also contribute to increased turbidity, reducing zooplankton diversity (Lougheed & Chow-Fraser 1998; Cottenie et al., 2001).

The availability of literature investigating the impacts of anthropogenic activities on wetland environments is significantly less compared to the information available targeting lake ecosystems (Hoffmann & Dodson, 2005). There have been few studies examining the effects of cattle on larger wetland invertebrates (Reeves & Champion, 2004); however, results indicate that cattle most often impact population densities (Kostecke et al., 2005) and contribute to a decrease in species richness and diversity of the specific invertebrates that pertain to each study (Hornung & Rice, 2003; Steinman et al., 2003; Foote & Hornung, 2005). There is also evidence suggesting that livestock presence have no effect upon invertebrate communities (Steinman et al., 2003) while other studies found that it may actually increase wetland invertebrate diversity (Pyke & Marty, 2005; Davis & Bidwell, 2008). The contradictory nature of these studies is most likely due to lack of direct quantification of cattle impact (intensities vary amongst studies) and differing measures of impact. The present study seeks to quantify a measurable impact variable so that the intensity is more acutely addressed.

While research examining the effects of livestock on wetland zooplankton community structure has been limited, there have been considerable efforts to examine the effects of various other forms of anthropogenic activities on zooplankton communities in shallow lakes and wetlands. A prevalent trend in disturbance studies that investigate and compare zooplankton, between impacted and non-impacted sites, is that species richness significantly decreases as disturbance increases (Dodson & Lillie, 2001; Stemberger et al., 2001; Dodson et al., 2005; Dodson et al., 2007; Lougheed et al., 2008). In addition to decreased richness, studies have shown that disturbance favours species more adaptable to the impacted environment, causing the community composition of an aquatic system to change (Gannon & Stemberger, 1978; Gulati, 1983; Rellstab et al., 2011).

In the present study, measured local parameters such as water chemistry and chlorophyll-*a* concentration were expected to have measurable effects on zooplankton community structure. Additionally, as wetlands are located within a rangeland landscape, parameters linked to the presence of cattle such as increased nutrients and turbidity were expected to exert a strong influence.

Study Goals

The hypothesis of this study was that cattle impact, measured by percentage of trampled wetland perimeter, will influence wetland zooplankton community structure through a decrease in species richness and a change of species to those most tolerant of impacted conditions. The specific study goals were to determine the relationships between measured chemical, physical or morphological parameters and zooplankton community structure, and determine of these relationships were consistent with the hypothesis. My aim was to test if cattle impact, measured as a single environmental variable, had a primary role in shaping the structure of zooplankton communities, or if other environmental parameters were more important.

Materials and Methods

Site Description

Fifteen fishless wetlands located in cattle-grazed rangelands surrounding Kamloops British Columbia were selected for analysis (Figure 2.1). Kamloops is characterized by a semi-arid climate with an annual precipitation of 279 mm with generally mild short winters and hot, mostly dry summers (Environment Canada, 2011). Wetlands were located in four main geographical locations and were named with a letter corresponding to location: Lac de Bois Provincial Park (Bachelor Heights area: B11, B9.1, B9, B6, B3; general area: L6.3 L4.1, L4, L3, L2), Campbell Range (C6), Rose Hill (R17, R12) and Hamilton Commonage (H6, H4.2). The areas of Lac de Bois, Campbell range and Rose Hill are located within twenty kilometers of Kamloops at elevations ranging from 770 – 1070 meters. Hamilton Commonage is located approximately 115 kilometers south of Kamloops at a slightly higher elevation of 1189 – 1240 meters. Wetland areas ranged from just under one hectare to 6.3 hectares (mean 3.4 hectares), with depths ranging from 0.6 – 5.6 meters, with the majority of wetlands (80%) under 3 meters in depth.

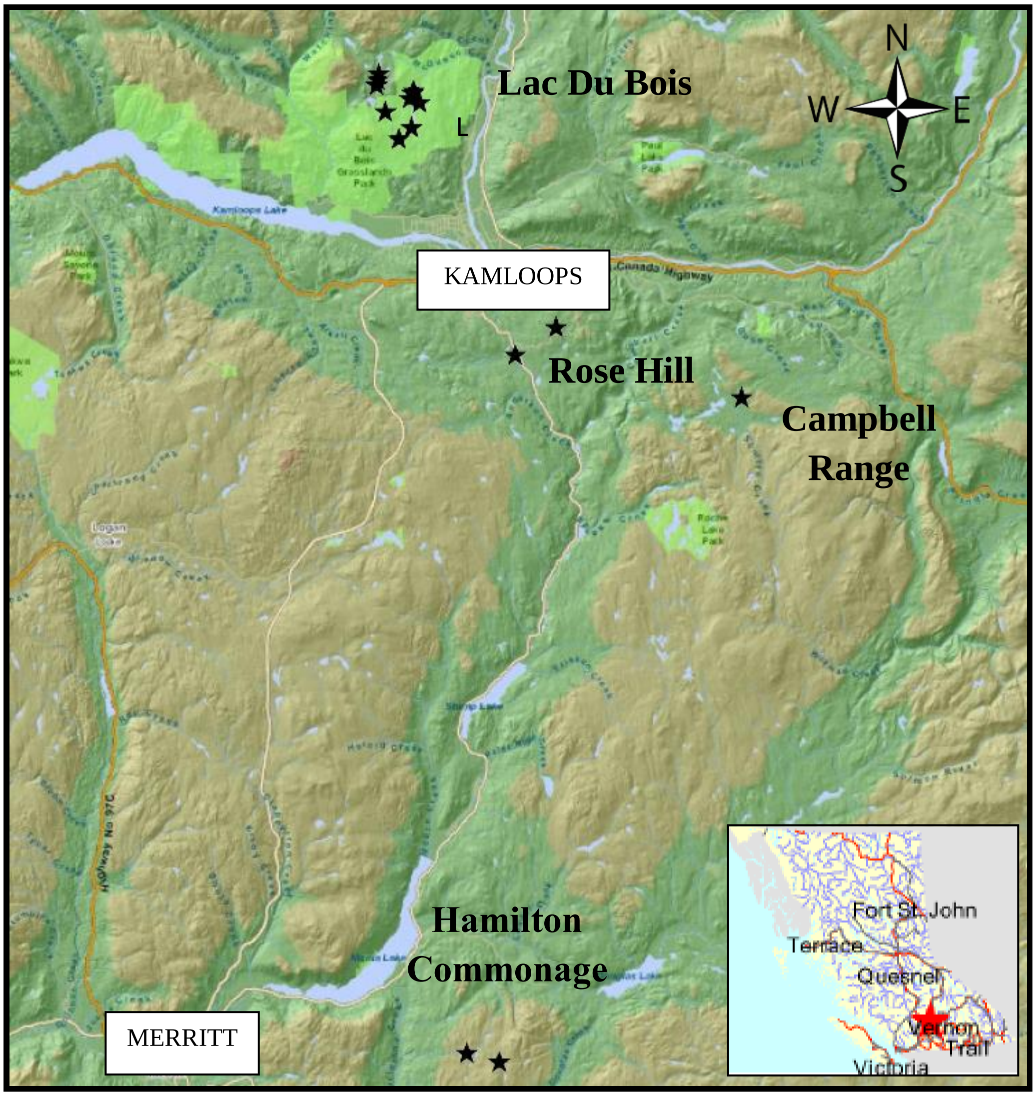


Figure 2.1. Location of study wetlands. Stars denote site locations. Scale 1: 350,000 (iMapBC, 2010).

Sampling Design

Wetlands were sampled monthly in July, August and September of 2009. Sampling was conducted during daylight hours (7 am – 3 pm) from an inflatable two-person raft. As described in details below, samples of zooplankton (crustaceans and rotifers), chlorophyll-*a* and water chemistry were obtained for analyses. Multimeter probe measurements and assessment of cattle impact were determined simultaneously with zooplankton samples. Elevation and surface area were determined after sampling using the iMapBC online measurement tools (iMapBC, 2010).

Cattle Impact Analysis

Cattle impact at each wetland was determined by measuring the amount of cattle-trampled/pugged wetland perimeter, within one meter of the shoreline. Pugging refers to the deep hoof prints left by cattle in fine soils, Cattle impact was chosen as it is a direct measure of impact, as opposed to alternate measures. As the density of trampling ranged, each portion that was heavily, moderately or lightly trampled was calculated separately. In order to calculate total trampling length, the portions that were moderately and lightly trampled were modified to accurately portray total trampling amount, taking into account differing intensities. As lightly trampled areas were approximately one third as dense as heavy areas, these lengths were divided by three, moderate areas divided by two (moderate trampling was approximately half as dense as heavy trampling) and heavily trampled lengths were unmodified. Totals were converted to percentage values that ranged from zero to one hundred percent impact. When classifying wetlands for comparison analyses, wetlands were classified as either “least-impacted” or “impacted” (Table 2.1). These groups corresponded with either ≤25% or >25% cattle impact.

A threshold of 25% cattle impact was chosen as it best separated wetlands that visually either did or did not appear impacted by cattle. Additionally, least-impacted wetland shorelines were almost completely dominated by undisturbed vegetation, whereas all impacted wetlands were almost completely devoid of shoreline vegetation (trampled/bare). Furthermore, there was a distinct decrease in species richness at approximately 25-27% cattle impact percentage (Figure 2.9), which further justified using 25% as a cut-off for comparisons. A cut-off of 25% is further supported by a study by the BC Forest Practices Board (2002), which examined how cattle pugging affected the functionality of the riparian zone. Pugging was also quantified as percent occurrence of the wetland perimeter (riparian zone). This study found that the riparian area of wetlands with cattle pugging greater than 25% of the wetland perimeter were determined as "non-functional". This study evaluated proper-functioning wetlands as being able to withstand floods, filter runoff and safely store and release water.

Table 2.1 Wetlands with corresponding cattle impact % rating and impact category assignment (LI=low impact, I=impacted).

Wetland	**% Cattle Impact**	**Impact Group:** (LI=≤25%, I=>25%)
B11	0.1	LI
B9.1	3.2	LI
B9	0.1	LI
B6	42.4	I
B3	28.9	I
L6.3	38.5	I
L4.1	19.4	LI
L4	0.1	LI
L3	21.6	LI
L2	30.4	I
C6	80.6	I
R17	22.5	LI
R12	51.3	I
H4.2	100	I
H6	64.8	I

Sampling – Abiotic Variables

To incorporate environmental variables that were potentially important drivers of zooplankton community structure, physical and chemical variables were measured. Wetland depths and chlorophyll-*a* were obtained during each sampling event. Multimeter probe measurements (salinity, conductivity, pH, dissolved oxygen and total dissolved solids) were obtained for the August and September sessions. Additional water chemistry variables were obtained for the August session, these included total phosphorus, total nitrogen, chloride, ammonia, phosphate, sulphate and alkalinity. Samples were refrigerated until analysis, which was completed within 24 hours of sampling. Chemical analyses were conducted by Ecotech Laboratories in Kamloops, BC, following the methodology from Lenore et al. (1998). During August and September sampling, salinity, conductivity, total dissolved solids, temperature, dissolved oxygen and pH were measured *in situ* using a YSI 556 Multi-Probe. Note that salinity readings are not true representations of the salts found in these particular wetlands, but calculated by the multimeter probe based on both conductivity and temperature.

Depth was determined by averaging measurements of three of the deepest perceived points of each wetland. Rough estimations of area and perimeter were measured once from 2004 online aerial photos of the wetlands using the iMapBC web-based mapping tool (GeoBC 2010).

<u>Sampling – Biotic Variables</u>

Zooplankton collections were made monthly (July – September) from a tube-sampler constructed from a one meter by 3 cm PVC tube, sealable on one end by a rubber ball connected to heavy-duty twine (design obtained from Jonathan Shurin, personal communication, 2009). Tube-samplers are ideal for sampling shallow water wetlands, are cost effective and have been proven to give reliable samples in the field (Paggi et al., 2001). Five replicate zooplankton samples were collected per site. Each replicate was a composite of two tube volumes (0.35 L total) obtained from the wetland centre on alternating sides of the raft, with care to stay at least 30 cm above the sediment and avoiding aquatic vegetation. Samples were immediately filtered through 64-µm-mesh netting, chosen to minimize the loss of rotifers yet optimize filtration rates (Bottrell et al., 1976). Mesh contents were washed into jars containing 250 ml of 70% ethanol solution.

Prior to laboratory analysis, each sample was thoroughly washed of alcohol and reconstituted into approximately 50 ml of water. Most sample densities did not warrant sub-sampling; however, in rare cases when densities greatly exceeded those observed in most other samples, three 10 ml sub-samples were taken using a Hensen Stempel pipette. Samples were transferred to a gridded Petri dish and individuals were identified and enumerated under a Leica MZ6 dissecting microscope at 40X magnification**.** Crustacean zooplankton (calanoid copepods, cyclopoid copepods and cladocerans) and rotifers were identified to lowest possible taxonomic level using the keys of Thorp & Covich (1991) and Pennak (1978). For calculation of biomass and confirmation of zooplankton identification, one sample from each wetland was sent to BSA Environmental Services in Beachwood, OH. BSA performed estimates of biomass according to established length/weight regressions (Dumont et al., 1975; McCauley, 1984; Lawrence et al., 1987), and zooplankton identifications (species level) using the taxonomic references of Ruttner-Kolisko (1974) and Pennak (1978).

Zooplankton total abundance and abundances of cyclopoids, calanoids, cladocerans and rotifers were calculated as individuals per litre. Species richness was calculated as the sum of adult taxa (excluding nauplii and immature individuals). Diversity was calculated using the Shannon-Wiener index (Pielou, 1975) and Simpson's index (Simpson, 1949).

Three replicate surface water samples were obtained using one litre amber jars obtained during zooplankton sampling for subsequent chlorophyll-*a* analysis. Samples were obtained at the same location as zooplankton samples. Prior to chlorophyll extraction, the samples were filtered through 1.5 μm pore 934-AH Whatman glass microfibre filters in the field. Filters were transferred into vials containing 20 ml of methanol and held for 24 hours to ensure pigment extraction. Vials were dried under a fume hood (4 – 6 days). Upon evaporation, vials were reconstituted with 500 μL of methanol which, after being thoroughly washed over the inside of the vials to ensure reconstitution of any methanol-containing solid, was extracted into microcentrifuge tubes, spun for one minute, and subjected to spectrophotometric analysis at wavelengths of 650 nm and 665 nm. Chlorophyll-*a* was calculated using the following equation specified by ICES standard operating procedures (Aminot & Rey, 2000):

$$\mu g\ Chlorophyll/ml = 16.5(A_{665}) - 8.3(A_{650})$$

The three measurements per site were averaged to give a single chlorophyll-*a* measured in μg/L.

Statistical Analysis

All statistical analyses were conducted using R statistical software (R Development Core Team, 2009). To characterize zooplankton community structure, both hierarchical clustering and Non-Metric Multidimensional Scaling (NMDS) were used to analyse presence/absence species lists and abundance data. Linear regression and generalized linear regression were also used to assess possible relationships among zooplankton parameters and environmental descriptors. To compare physical, chemical, biological and community differences between impact groups, parametric and non-parametric ANOVAs and t-tests were used. To detect temporal community changes throughout the summer, species data matrices were compared using both Multiple Response Permutation Procedures (MRPP) and Analysis of Dissimilarities (ADONIS).

Community Structure

Cluster analysis was chosen to explore zooplankton community groupings, similarities and dissimilarities amongst wetlands. Analyses were performed using a genus level presence-absence distance matrix, Euclidean distance measure and Ward's linkage method, which is recommended to avoid data distortion (McCune and Grace, 2002). Presence-absence data were used as using these data resulted in the fewest clusters and clusters that could be attributed to measurable parameters in this study. Species data for all sampling events were combined for this analysis.

To visualize patterns in community structure, abundance data for each month's sampling event were subjected to NMDS separately. NMDS seeks to position similar sites together based on similarities in species composition and is ideal for community data, which most often violate the assumptions of normality that is required for parametric tests. NMDS is becoming increasingly popular in community ecology due to its high degree of flexibility and is commonly considered the most effective ordination method of community data sets (McCune and Grace, 2002; Borcard et al., 2011). NMDS analysis was the method of choice over analyses such as correspondence analysis, detrended correspondence analysis and canonical correspondence analysis due to serious faults these methods can impose on ecological community data (McCune and Grace, 2002). NMDS was performed using the *metamds* function in the Vegan R package (Oksanen et al., 2010), with random starting configurations and 1000 runs with real data. Abundance data were Wisconsin double standardized and converted to a Bray-Curtis similarity matrix (Bray & Curtis, 1957). For each of the three sampling months, two-dimensional configurations with respective stresses of 14.40, 14.58 and 15.54 were chosen as the optimal models.

Community Change Through Time

To test for overall differences in communities through time (from July – September), monthly community data were assessed using MRPP and Adonis. MRPP is a non-parametric permutation procedure that tests the null hypothesis that there are no significant differences between two or more groups of multivariate observations (McCune & Grace, 2002). In this study, MRPP compared the differences within and amongst months based on their species composition. The test statistic, *delta*, gives the significance of the disparity between distributions, and a second output statistic, *A*, describes the effect size, or the “chance-corrected within group agreement” (McCune & Grace, 2002). Ranging from negative values to 1, an *A* value of 0 would suggest that groups are no less different than expected by chance (agreement with $H0$), a value of 1 would indicate that units within each sampling group are homogenous (in perfect agreement) and negative values indicate groups are heterogeneous (disagreement with $H0$). MRPP tests used the Sørensen distance measure with 1000 permutations.

Most MRPP models can also be assessed with the Adonis function, which is considered a more robust alternative (Oksanen et al., 2010) and was used in conjunction with MRPP in this study. Adonis functions to partition sums-of-squares using semi-metric and metric distance matrices (Oksanen et al., 2010). Adonis is similar to a non-parametric ANOVA test and was performed in R using the Bray-Curtis distance measure using 1000 permutations.

Environmental Variables Influencing Community Structure

Environmental variables most influencing the structure of the ordination (the location of sites and species) were overlain into the NMDS figure, to create a biplot diagram. Only significant correlations between the environmental variables and the ordination were fit onto the biplot, with arrows to show both the strength (arrow length) and direction of the environmental gradient.

Environmental drivers were explored using stepwise linear regression. The following were used as the dependent variables: total abundance, group abundances (cladocerans, copepods (total, calanoid and cyclopoid), and rotifers), biomass, richness and diversity. A full set of physical and chemical variables were assessed as predictor variables for the August sampling session; however, for the July and September sessions, a reduced set of variables were obtained and analyzed. For the August sampling session, colinearity of the predictor variables was assessed prior to running regressions, and covariates were eliminated. Conductivity and TDS were removed due to high correlation with salinity, and ammonia was omitted due to a high correlation with total nitrogen. Variables were first assessed for normality using the Kolmogorov-Smirnov test. Non-normal variables were transformed using Box-Cox power transformation (powers rounded to the nearest 0.25) to maximize the normality of the output distribution (Table 2.2). The best models were chosen by assessment of adjusted R^2 values and the Akaike's information criterion (Quinn & Keough, 2002). Generalized linear models (GLMs) were performed using the equivalent dependent and independent variables (untransformed), using poisson or quasipoisson linkages where appropriate. GLMs were used to compare with results from their linear regression counterparts. Additionally, GLMs using binomial presence/absence data of individual genera (logistic regressions) were run to test if measured parameters had significant impacts on species occurrence in wetland groups.

Cattle Impact Groups

Differences in physical, chemical and biological variables were analysed using one-way ANOVA and t-tests. The two wetland groups used in the analyses corresponded to the classifications of non-impacted (≤25% cattle impact) and impacted (>25% cattle impact). Homogeneity of variance was checked prior to analysis, and if requirements were not met, the equivalent non-parametric Kruskal-Wallis rank sum test or Mann-Whitney-U test was performed. Significant t-test results were subsequently checked with the Fligner-Killeen test of homogeneity of variance.

Table 2.2 Continuous variables with non-normal distributions (August 2009). Shown are untransformed Kolmogorov-Smirnov (K-S) results, Box-Cox power transformation value, and K-S test result post-transformation to normality.

Variable	**K-S Test (raw data)**	**Box-Cox Power Transformation**	**K-S Test (post transform)**
Abundance	0.02	0.25	0.71
Biomass	0.007	0 (log)	0.71
Copepod Abundance	0.002	0.25	0.44
Rotifer Abundance	0.001	0.25	0.80
Shannon's Index		-0.5	0.67
Chlorophyll-*a*	0.08	0 (log)	0.92
Salinity	0.002	-0.25	0.58
pH	0.02	-0.25	***0.02***
Chloride	.001	0 (log)	0.63
TP	4.882e-05	0 (log)	0.60
Sulfate	0.0001	0 (log)	0.79
Alkalinity	0.03	-0.25	0.88

Results

Wetland Zooplankton Characteristics

Sampling sessions combined, a total of 32 crustacean and rotifer taxa were identified (Table 2.3): 4 species of copepods, 6 species of cladocerans and 22 species of rotifers. Twenty-one of these occurred in all three months, with a decrease in species through time from 32, 28 and 26 for July, August and September, respectively. A summary of all biological variables calculated can be found in Tables 2.5 – 2.7.

For August (focus month), a total of 28 crustacean and rotifer taxa were identified (Table 2.4). Species richness varied from 3 to 10 species per wetland, and was highly dominated by rotifers. Cladocerans and copepods were represented by a total of 4 species each, with each site containing between 1 and 3 cladoceran species, and up to 2 copepod species. The most commonly found taxa, appearing in 11 out of 15 wetlands (73%), were the copepod *Leptodiaptomus* spp. and the rotifers *Hexarthra* spp. and *Keratella* spp. Intermediately found genera, occurring in 5 to 8 wetlands, were *Ceriodaphnia* spp., *Daphnia* spp., *Brachionus* spp., *Monostyla* spp., *Platyias* spp. and *Polyarthra* spp. The remaining taxa, occurring in four wetlands or less (<26%), were *Alona* spp., *Acanthocyclops* spp., *Diacyclops* spp., *Asplanchna* spp., *Filinia* spp., *Lecane* spp., *Lepadella* spp., *Notholca* spp., *Synchaeta* spp., *Testudinella* spp. and *Trichocerca* spp.

Table 2.3 List of all zooplankton found and identified during sampling. Check denotes species presence; dash denotes species absence in sampling month.

	July	August	September
CLADOCERA			
Alona guttata	√	√	√
Ceriodaphnia spp.	√	√	√
Chydorus sphaericus	√	--	--
Daphnia magna	√	√	√
Daphnia pulex	√	√	√
Simocephalus spp.	√		√
COPEPODA			
Acanthocyclops vernalis	√	√	√
Diacyclops thomasi	√	√	√
Leptodiaptomus connexus	√	√	√
Leptodiaptomus sicilus		√	
ROTIFERA			
Asplanchna spp.	√	√	√
Brachionus plicatilus	√	√	√
B. quadridentatus f. brevispinus	√	√	√
B. urceolaris	√	√	√
Conochilus unicornis	√	√	√
Euchlanis calpidia	√	--	√
Hexarthra mira	√	√	√
Filinia longiseta	√	√	√
Keratella hiemalis	--	√	--
K. quadrata	√	√	√
Lecane mira	√	√	√
Lepadella ovalis	√	√	√
Monostyla lunaris	√	√	--
Monostyla bulla	√	√	--
Monostyla quadridentata	√	√	√
Mytilina ventralis	√	--	√
Notholca squamula	√	√	√
Platyias quadricornus	√	√	--
Polyarthra vulgaris	√	√	√
Synchaeta spp.	√	√	√
Testudinella patina	√	√	√
Trichocerca spp.	√	√	√

Table 2.4 Distribution and abundance of zooplankton in study wetlands (August 2009).

	WETLANDS														
SPECIES	B11	B9.1	B9	B6	B3	L6.3	L4.1	L4	L3	L2	C6	R12	R17	H4.2	H6
CLADOCERA															
Alona guttata			*				*		*						
Ceriodaphnia spp.	*					**	*	*		**			**		
Daphnia magna															**
Daphnia pulex	*							**	*	*	**	**		**	
COPEPODA															
Cyclopoid Copepods															
Acanthocyclops vernalis												*	**		
Diacyclops thomasi														*	*
Calanoid Copepods															
Leptodiaptomus connexus						***		*		****	*	****	*		**
Leptodiaptomus sicilus	*		*	**	***										
ROTIFERA															
Asplanchna spp.			*		*					*					
Brachionus plicatilus			*		*			*					*		
B. quadridentatus f. brevispinus		*													
B. urceolaris														*	**
Conochilus unicornis		**													
Hexarthra mira	*	****	****	***	*	**	*	**	****		****		**		
Filinia longiseta															****
Keratella hiemalis							*								
K. quadrata	***	****	*****	*		*		*	*		**		******		****
Lecane mira							*		*						
Lepadella ovalis	*	*					*						*		
Monostyla bulla															
Monostyla lunaris			*				*								
Monostyla quadridentata		*							*	*					
Notholca squamula			*												
Platyias quadricornus	*	*	*				*		*						
Polyarthra vulgaris	**		**					**	*				****		
Synchaeta spp.	*										*				
Testudinella patina	*								*						
Trichocerca spp.		*													

Code to abundance - *=<10 organisms/L, **=10-100 organisms/L,***=100-200 organisms/L, ****=200-500 organisms/L, *****=500-1000 organisms/L, ******=1000+ organisms/L

Table 2.5 Biological variables calculated for July 2009. The following abbreviations were used: Tot Abund= total zooplankton abundance per litre, Rich= Species Richness, Sha=Shannon's Diversity index, Simp=Simpson's Diversity Index, Clad Abund= abundance of cladoceran taxa per litre, Rot Abund=abundance of rotifer taxa per litre, Cope Abund=abundance of copepod taxa per litre, Cala Abund=abundance of calanoid copepods per litre, Cyclo Abund=abundance of cyclopoid copepods per litre, Chl*a*=Chlorophyll-*a* concentration.

Wetl -and	Tot Abund	Bioma ss	Rich	Sha	Simp	Clad Abund	Rot Abund	Cope Abund	Cala Abund	Cyclo Abund	Chl*a*
B11	1660.8	43.5	9.0	0.9	0.5	8.2	2052.3	1.1	0.0	1.6	0.2
B9.1	1209.8	68.1	13.0	0.7	0.3	4.4	1630.5	2.6	0.0	2.6	0.8
B9	1336.6	130.3	11.0	1.1	0.5	75.3	853.3	0.0	0.0	0.0	0.5
B6	198.7	292.3	5.0	0.8	0.5	0.4	179.5	43.7	113.7	0.0	0.0
B3	380.9	520.6	4.0	0.8	0.5	0.0	147.5	185.9	63.9	0.0	0.0
L6.3	395.2	732.5	4.0	0.3	0.1	198.9	3.8	172.6	158.4	0.0	0.1
L4.1	135.6	115.2	13.0	1.3	0.6	20.1	19.8	5.9	0.0	0.7	0.3
L4	189.8	70.2	6.0	1.3	0.7	15.2	80.8	27.2	2.2	0.6	02
L3	287.1	30.6	9.0	1.3	0.7	4.1	67.8	0.7	0.0	0.0	0.1
L2	246.4	265.7	7.0	1.3	0.7	53.9	1604.0	100.4	35.2	0.0	0.0
C6	1809.6	69.7	6.0	0.8	0.5	31.0	1608.6	0.0	0.0	0.0	0.2
R17	523.0	32.9	4.0	1.0	0.5	58.6	7.9	11.2	0.0	1.8	0.3
R12	280.1	394.6	4.0	0.5	0.3	23.3	71.5	153.0	44.3	0.0	0.2
H4.2	1638.4	2509.0	5.0	1.2	0.6	144.4	130.3	373.6	0.8	4.6	1.5
H6	644.0	168.5	5.0	1.0	0.6	13.0	59.1	74.1	8.8	3.4	0.4

Table 2.6 Biological variables calculated for August 2009. The following abbreviations were used: Tot Abund= total zooplankton abundance per litre, Rich= Species Richness, Sha=Shannon's Diversity index, Simp=Simpson's Diversity Index, Clad Abund= abundance of cladoceran taxa per litre, Rot Abund=abundance of rotifer taxa per litre, Cope Abund=abundance of copepod taxa per litre, Cala Abund =abundance of calanoid copepods per litre, Cyclo Abund =abundance of cyclopoid copepods per litre, Chl*a*=Chlorophyll-*a* concentration.

Wetl -and	Tot Abund	Biomass	Rich	Sha	Simp	Clad Abund	Rot Abund	Cope Abund	Cala Abund	Cyclo Abund	Chl *a*
B11	376.5	26.6	10.0	0.8	0.4	4.6	230.9	0.0	0.0	0.0	0.4
B9.1	821.8	41.8	7.0	0.7	0.5	0.0	845.4	0.0	0.0	0.0	1.2
B9	1277.4	201.7	10.0	0.9	0.5	0.9	1298.6	10.8	0.0	2.8	0.8
B6	220.9	78.9	3.0	0.8	0.5	0.0	124.9	81.9	10.9	0.0	0.1
B3	227.9	198.4	4.0	0.4	0.2	0.0	9.1	169.3	48.1	0.0	0.2
L6.3	206.1	270.9	4.0	0.6	0.3	18.0	14.1	161.1	11.5	1.3	0.2
L4.1	567.3	108.7	8.0	1.8	0.8	20.4	214.1	7.0	3.3	2.8	0.3
L4	112.4	116.1	7.0	1.4	0.7	0.0	55.8	14.3	0.0	0.6	0.2
L3	308.5	23.1	9.0	0.4	0.2	3.8	206.3	2.4	0.0	2.5	0.1
L2	308.9	92.2	5.0	0.6	0.3	43.9	3.9	220.4	23.8	0.0	0.2
C6	426.6	543.4	5.0	0.7	0.4	25.6	332.1	1.4	0.0	1.1	0.2
R17	2196.2	648.2	8.0	0.9	0.5	21.6	1645.7	226.3	0.0	120.7	0.4
R12	498.7	745.0	3.0	0.6	0.4	61.8	0.0	354.8	102.8	2.0	0.3
H4.2	21.8	81.7	3.0	0.5	0.3	16.4	2.4	2.2	0.0	0.0	1.0
H6	1119.5	845.7	6.0	1.2	0.7	23.4	961.9	78.5	0.0	14.7	0.7

Table 2.7 Biological variables calculated for September 2009. The following abbreviations were used: Tot Abund= total zooplankton abundance per litre, Rich= Species Richness, Sha=Shannon's Diversity index, Simp=Simpson's Diversity Index, Clad Abund= abundance of cladoceran taxa per litre, Rot Abund=abundance of rotifer taxa per litre, Cope Abund=abundance of copepod taxa per litre, Cala Abund=abundance of calanoid copepods per litre, Cyclo Abund =abundance of cyclopoid copepods per litre, Chl*a*=Chlorophyll-*a* concentration.

Wetland	**Tot Abund**	**Biomass**	**Rich**	**Sha**	**Simp**	**Clad Abund**	**Rot Abund**	**Cope Abund**	**Cala Abund**	**Cyclo Abund**	**Chl *a***
B11	261.7	51.8	7.0	1.1	0.6	4.1	96.0	4.6	0.0	0.0	0.7
B9.1	1053.4	36.3	8.0	0.3	0.1	2.1	878.7	0.0	0.0	0.0	0.8
B9	1078.1	37.5	12.0	0.4	0.2	1.7	594.8	0.9	0.0	0.0	0.5
B6	52.8	49.1	4.0	1.1	0.6	0.0	24.9	27.9	1.8	0.0	1.3
B3	309.8	342.2	5.0	0.6	0.4	0.0	218.3	81.6	2.8	0.1	1.2
L6.3	203.5	76.1	4.0	1.0	0.6	1.6	83.1	110.7	5.5	0.0	0.4
L4.1	730.7	30.0	12.0	1.8	0.8	5.5	23.0	13.3	0.0	11.1	0.4
L4	94.2	95.0	7.0	1.5	0.7	12.4	43.7	6.2	0.7	0.2	0.4
L3	147.3	83.6	9.0	1.4	0.7	8.7	64.5	4.1	0.0	3.9	2.5
L2	241.0	208.2	6.0	0.3	0.1	4.1	6.0	225.4	31.1	0.0	0.8
C6	1342.3	49.0	8.0	0.3	0.1	29.0	1291.7	0.0	0.0	0.0	1.9
R17	9354.5	1370.8	8.0	0.8	0.5	24.1	8423.6	460.0	0.0	384.4	8.0
R12	215.2	248.8	4.0	0.7	0.4	40.8	6.0	156.1	32.2	0.0	0.5
H4.2	140.7	622.2	3.0	0.7	0.5	48.1	0.5	53.0	0.0	25.1	1.2
H6	572.1	1461.0	3.0	0.7	0.4	18.4	0.0	93.2	4.8	8.1	6.9

Wetland Hydrochemistry

A summary of wetland water chemistry including measured physical variables can be found in Tables 2.8 – 2.10. Wetlands fell into the range of fresh (<1 ppt), subsaline (1 – 3 ppt) and hyposaline (3 – 20 ppt) waters (Last & Ginn, 2009). Examining differences between impacted and low-impact wetlands, analysis of variance, Kruskal-Wallace rank sum and permutation multivariate ANOVA tests indicated that there were significant differences between impact categories with respect to nutrient concentrations and salinity (Table 2.11). There were no significant differences in physical wetland characteristics (depth, perimeter) between the two categories. Impacted wetlands had significantly higher concentrations of both total nitrogen and total phosphorus. Average salinity also happened to be much higher in highest impact wetlands (Table 2.11).

Table 2.8 Physical variables measured in July 2009. Chemical variables were not measured. The following abbreviations were used: Elev=elevation, SA=surface area.

Wetland	Elev (m)	SA (Ha)	Depth (cm)
B11	940.0	2.9	287.0
B9.1	920.0	0.9	250.0
B9	910.0	2.5	140.0
B6	880.0	4.9	470.0
B3	780.0	3.6	180.0
L6.3	810.0	2.0	226.0
L4.1	770.0	1.0	73.0
L4	780.0	3.0	312.0
L3	760.0	6.3	574.0
L2	750.0	1.9	134.0
C6	1070.0	4.8	197.5
R17	860.0	2.5	112.0
R12	1000.0	5.4	320.0
H4.2	1180.0	3.3	95.5
H6	1240.0	5.6	272.0

Table 2.9 Physical-chemical variables measured in August 2009. The following abbreviations were used: Elev=elevation, SA=surface area, ST=surface temperature, Sal=salinity, O_2=oxygen, Alk=alkalinity, TotN=total nitrogen, TotP=total phosphorus, Sulf=sulfate, Chlor=chloride, Amm=ammonia.

Wetland	Elev (m)	SA (Ha)	Depth (cm)	ST (°C)	Sal (ppt)	O_2 (mg l^{-1})	pH	Alk (mEq l^{-1})	TotN (mg l^{-1})	TotP (mg l^{-1})	Sulf (mg l^{-1})	Chlor (mg l^{-1})	Amm (mg l^{-1})
B11	940.0	2.9	200.0	23.4	0.7	108.0	9.4	177.3	3.31	0.0	14.0	91.0	0.2
B9.1	920.0	0.9	105.0	23.5	0.6	119.4	10.2	360.0	5.51	0.0	14.0	79.0	0.0
B9	910.0	2.5	109.0	23.1	1.1	110.1	9.6	322.7	1.31	0.0	230.0	96.0	0.0
B6	880.0	4.9	429.0	24.6	12.9	155.6	8.9	286.7	6.41	0.0	19200.0	212.0	1.2
B3	780.0	3.6	172.0	24.0	19.1	14.8	9.0	307.0	5.47	0.1	35400.0	165.0	4.7
L6.3	810.0	2.0	209.0	23.5	10.1	128.3	9.6	102.3	3.39	0.1	17600.0	23.3	0.6
L4.1	770.0	1.0	63.0	21.2	0.5	76.4	8.9	461.7	2.57	0.1	250.0	4.3	0.0
L4	780.0	3.0	299.0	23.1	0.6	94.7	9.5	93.3	2.65	0.1	370.0	6.3	0.0
L3	760.0	6.3	564.0	23.2	2.1	67.0	9.4	170.0	2.80	0.0	16300.0	44.6	0.0
L2	750.0	1.9	113.0	22.2	10.0	30.5	8.6	135.0	4.04	0.1	2420.0	11.9	0.3
C6	1070.0	4.8	191.0	24.1	1.6	20.0	9.4	191.3	4.18	1.7	640.0	60.5	0.6
R17	860.0	2.5	99.0	24.9	2.1	130.0	9.4	146.7	4.77	0.1	930.0	662.0	0.3
R12	1000.0	5.4	400.0	25.4	5.5	115.5	8.7	70.0	3.80	0.7	7220.0	115.0	0.1
H4.2	1180.0	3.3	88.0	20.8	3.0	3.6	8.5	920.0	7.27	0.4	2760.0	60.5	3.0
H6	1240.0	5.6	277.0	22.1	1.2	123.8	9.4	150.0	5.14	0.2	610.0	36.0	0.0

Table 2.10 Physical-chemical variables measured in September 2009. The following abbreviations were used: Elev=elevation, SA=surface area, ST=surface temperature, Sal=salinity, O_2=oxygen..

Wetland	Elev (m)	SA (Ha)	Depth (cm)	ST (°C)	Sal (ppt)	O_2 (mg l^{-1})	pH
B11	940.0	2.9	115.0	23.4	0.7	126.2	9.4
B9.1	920.0	0.9	97.0	23.5	0.7	172.0	10.2
B9	910.0	2.5	104.0	23.1	1.1	170.0	9.7
B6	880.0	4.9	420.0	24.6	13.5	185.7	9.2
B3	780.0	3.6	161.0	24.0	21.1	59.0	8.9
L6.3	810.0	2.0	207.0	23.5	11.2	87.3	9.6
L4.1	770.0	1.0	69.0	21.2	0.6	44.1	8.7
L4	780.0	3.0	287.0	23.1	0.7	74.3	9.6
L3	760.0	6.3	552.0	23.2	2.2	83.0	9.5
L2	750.0	1.9	98.0	22.2	10.8	57.4	8.7
C6	1070.0	4.8	182.0	24.1	1.7	68.0	9.4
R17	860.0	2.5	82.0	24.9	2.6	86.4	8.7
R12	1000.0	5.4	375.0	25.4	5.7	52.5	8.7
H4.2	1180.0	3.3	64.0	20.8	3.4	44.1	8.7
H6	1240.0	5.6	253.0	22.1	1.3	66.9	9.3

Table 2.11 Significant ANOVA results for chemical parameters differing between impacted (>25%) and least-impacted (≤25%) wetlands in August 2009.

					Impacted (8)		Least-Impacted (n=7)	
Variable	**TEST**	**F**	**Df**	**p-val**	**Mean**	**Range**	**Mean**	**Range**
Total nitrogen	ANOVA	5.5	1	.036	4.9	3.39 - 7.3	3.3	1.3 - 5.5
Total phosphorus	ANOVA	6.7	1	.022	0.4	0.02 - 1.7	0.1	0.01 - 0.1
Salinity	ANOVA	17.3	1	.001	7.9	1.2 - 19.1	1.1	0.5 - 2.1

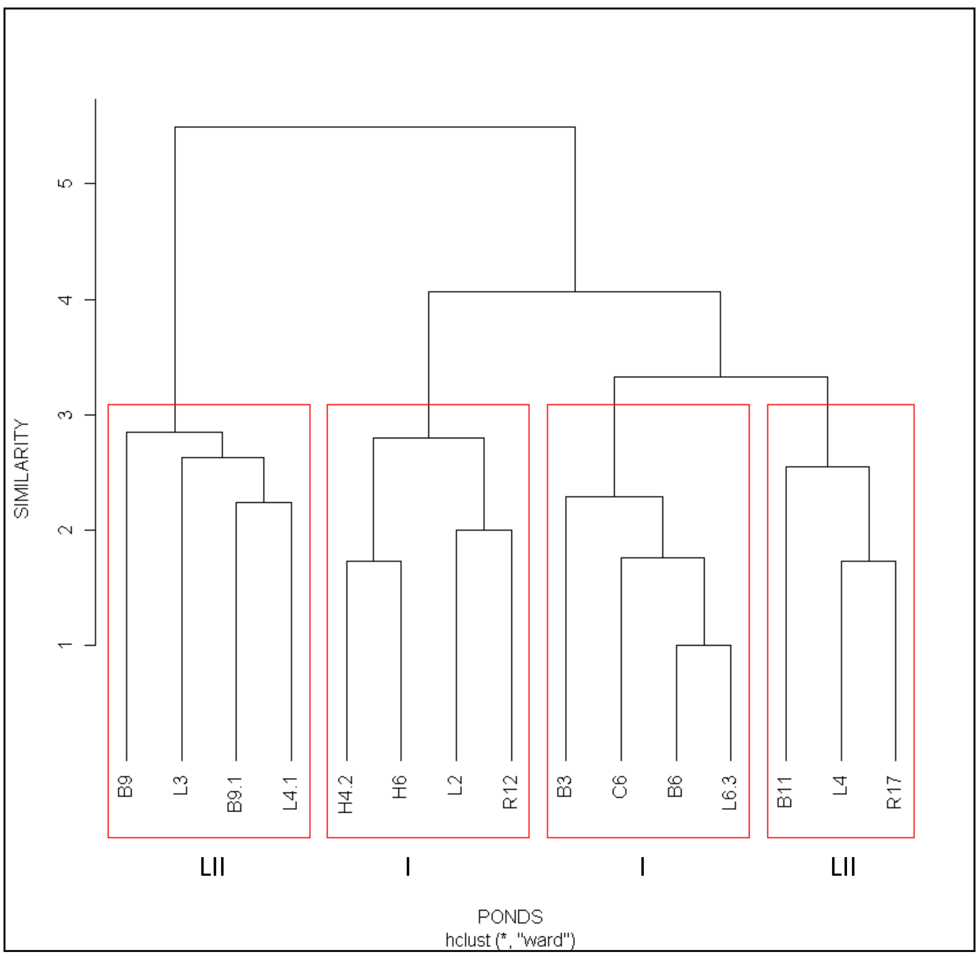
Cattle Impact Assessment

Wetlands were quantified for percent cattle impact of shoreline perimeter. Cattle impact ratings ranged from 0 – 100%, with 0% showing no visible signs of cattle pugging, and 100% pertaining to a wetland where the entire perimeter was surrounded by dense pugging. Pugging resulted in visible soil disturbance, vegetation removal, aquatic sediment destabilization (where cattle loitered within the water) and increased turbidity (in near-shore areas that were visibly recently accessed by cattle). The cattle impact variable was not found to correlate with any measured hydrochemistry variables.

Community Structure

Both cluster and ordination analyses were conducted using genus-level data. Previous studies have consistently found limnetic communities most often contain just one dominant species of cladoceran, copepod and rotifer at any one time (Pennak, 1957), thus using species-level data often does not give better results. Four distinct groups were revealed from the cluster dendrogram for the August sampling session (Figure 2.2); two of the groups contained wetlands characterized by least cattle impact (≤25% of the perimeter trampled) and lowest salinities, with the other two groups characterized as having higher cattle impact (>25% perimeter trampled) and higher salinities.

Figure 2.2 Results of hierarchical cluster analysis using presence/absence species data for all sampling events combined. Euclidean distance and Ward's linkage method were applied. Groups labeled LI and I correspond to those wetlands belonging to the ≤ 25% and >25% impact categories.



Due to limited environmental data available for the July sampling session, only NMDS ordinations for August and September will be reported (Figs. 2.3-2.4). Based on the characteristic taxa present in August (Fig. 2.3), a clear separation of sites can be seen which also corresponds to cattle impact categories. Both wetlands and the species most common to closest wetlands on the diagram are included. Triangles connecting low-impact wetlands (≤25% impact) can be found on the left of the diagram. All wetlands with over 25% cattle impact were considered “impacted” and are found on the right side of the diagram. Allocation of sites by impact category is reasonably demonstrated along both axes, in which there is a shift beginning with least-impacted wetlands developing to the most impacted on axis 1.

As overlap between impact groups is relatively minimal on the ordination plot, results suggest the presence of two potential community types: rotifer-dominated communities corresponding to lowest impact wetlands, and crustacean dominated communities of higher impact wetlands. A diverse rotifer-dominated assemblage characterized least-impacted wetlands. The two most dominant low-impact wetland-associated rotifers were *Platyias quadricornus* and *Polyarthra vulgaris.* Impacted wetlands, those found on the right hand side of the ordination diagram, are much less diverse. Wetland impact intensity increased from those wetlands dominated by calanoid copepods found in the bottom right of the diagram, to highest impact wetlands dominated by cladocerans and low-diversity rotifers found in the top right of the diagram.

The September NMDS ordination (Figure 2.4) shows a grouping pattern very similar to August 2009. Again, when looking from the perspective of cattle impact, impact categories are clearly defined along axis 1.

Adonis results confirm that wetlands appear to significantly differ from each other based on their zooplankton communities when comparing low impact and impacted wetland groups (Table 2.16). Average distance to the centroid was significantly less in impacted compared to low impact wetlands, indicating that those classified as impacted contained very different assemblages compared to those least-impacted.

Figure 2.3 Site-by-genus NMDS ordination of zooplankton community structure for August 2009. Wetlands are represented by their respective names and are connected based on percent impact to demonstrate groupings. LI –location least-impacted wetlands (≤25%). I –location of impacted wetlands (>25%).

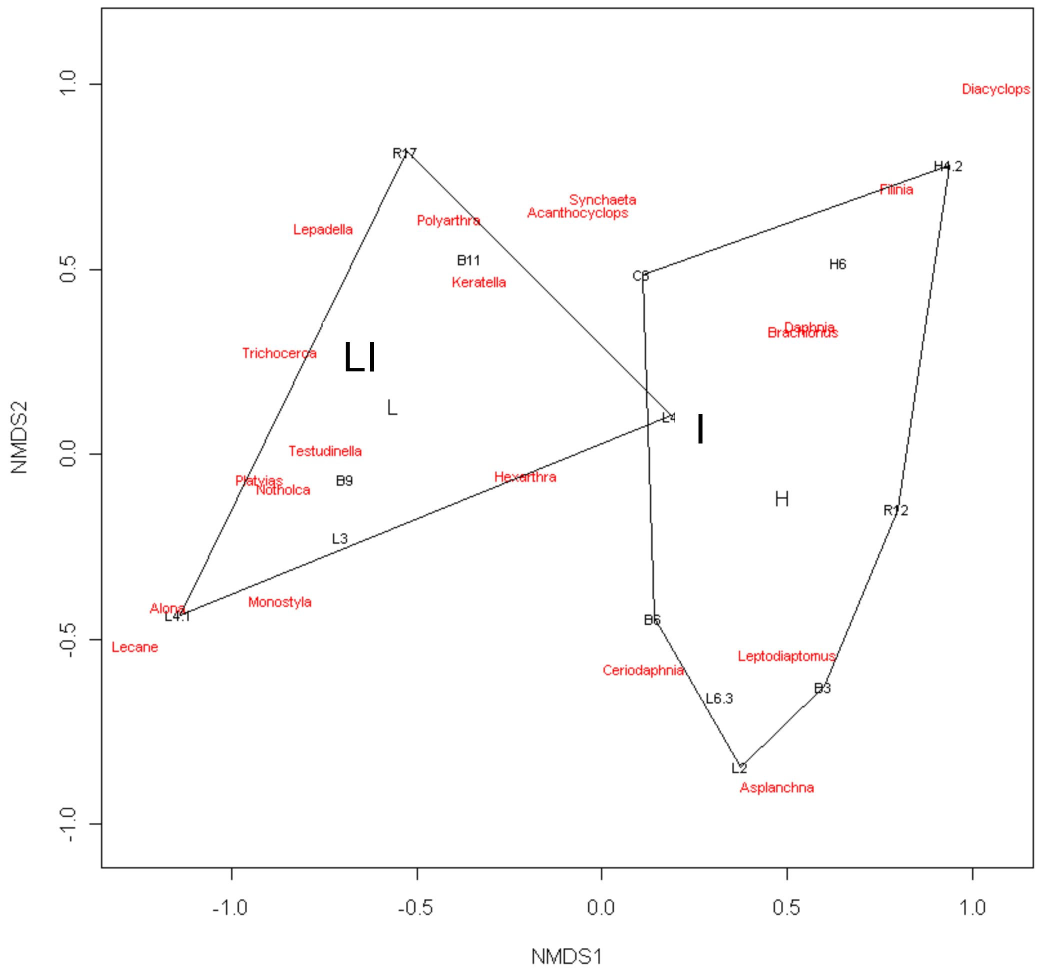
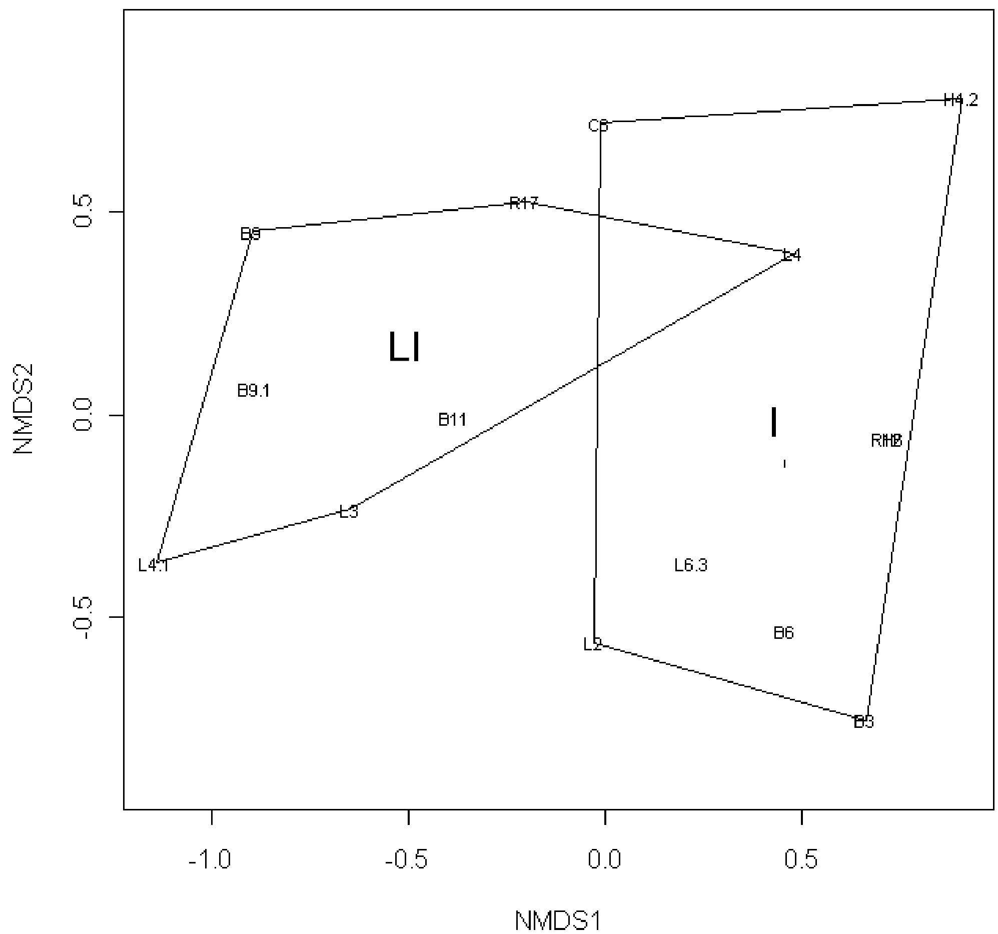


Figure 2.4 Site-by-genus NMDS ordination of zooplankton community structure for September 2009. Species not shown for simplification. Wetlands are represented by their respective names and are connected based on percent impact to demonstrate groupings. LI –location of least-impacted wetlands (≤25%). I –location of impacted wetlands (>25%).



Community Structure in Relation to Environmental Variables

Correlation tests between measured environmental variables and genus data in the NMDS ordination showed that both salinity and cattle impact were the only two ($p < 0.05$) environmental variables that were significantly related to zooplankton community structure in both August and September (Tables 2.12 and 2.13). This test was not possible for the July sampling session due to limited parameters measured.

The ordination biplots (Figures 2.5 – 2.7) illustrate that these variables increased towards wetlands of increasing cattle impact in both sampling events. For the August ordination, impacted wetlands were correlated with the cattle impact and salinity (Figure 2.5). There were no measured variables that were found correlated significantly with the ordination axes influencing low impact wetlands. In a second analysis for August, biological variables (group abundances, richness, biomass) were also included with the previously correlated environmental data, and the identical correlation tests were run with significantly correlated variables included on the biplot (Figure 2.6). Results demonstrate which major zooplankton groups are associated with different assemblages in cattle impact groups. Least-impacted wetlands cluster together similarly due to their high number of rotifer species and highest species richness overall, while highest impact wetlands are grouped similarly in part due to common crustacean zooplankton, particularly cladocerans.

Ordination results for the September sampling session were very similar. The ordination biplot (Figure 2.7) shows the combination of the significant environmental parameters (salinity and cattle impact) as well as the significant environmental variables contributing to wetland groupings in a simplified diagram. Comparable to August, low impact wetlands are species rich and driven by a diverse assemblage of rotifer species, and highest impact wetlands are characterized by cladoceran zooplankton. In slight contrast to August, the highest impacted wetlands in September appear correlated with the highest concentrations of salinity, whereas in August highest salinities appeared to be driving wetlands associated with more intermediary levels of cattle impact.

Table 2.12 Correlations of environmental variables with NMDS ordination axes for August 2009. Significant p-values in bold.

Variable	Axis 1	Axis 2	R^2	Pr(>r)
Chlorophyll-*a*	0.14	0.99	0.238	0.20
Temperature	0.64	-0.77	0.003	0.98
Salinity	0.52	0.86	0.653	**0.0008 *** **
Oxygen	-0.99	0.16	0.080	0.60
pH	-0.88	0.47	0.304	0.12
Chloride	-0.33	0.94	0.150	0.41
TN	0.86	0.50	0.344	**0.08**
TP	0.64	0.77	0.175	0.32
Ammonia	0.97	-0.26	0.259	0.16
Alkalinity	0.17	0.98	0.068	0.71
Area	0.93	0.38	0.177	0.31
Depth	0.55	-0.84	0.054	0.72
Impact	0.90	0.43	0.515	**0.01 * **

Table 2.13 Correlations of environmental variables with NMDS ordination axes for September 2009. Significant p-values in bold.

Variable	Axis 1	Axis 2	R^2	Pr(>r)
Chlorophyll-*a*	0.37	0.93	0.07	0.68
Temperature	-0.87	-0.50	0.21	0.24
Salinity	0.44	-0.89	0.66	**0.0004 *** **
pH	-0.89	0.45	0.16	0.35
Area	0.99	0.01	0.23	0.22
Depth	0.51	-0.86	0.13	0.44
Impact	0.86	0.50	0.46	**0.02 * **

Figure 2.5 Wetland-by-species NMDS ordination with overlay of significantly correlated (p=0.05) environmental variables for August 2009. Sal=salinity, impact=percent cattle impact, LI – general location of least-impacted wetlands. I – general location of impacted wetlands.

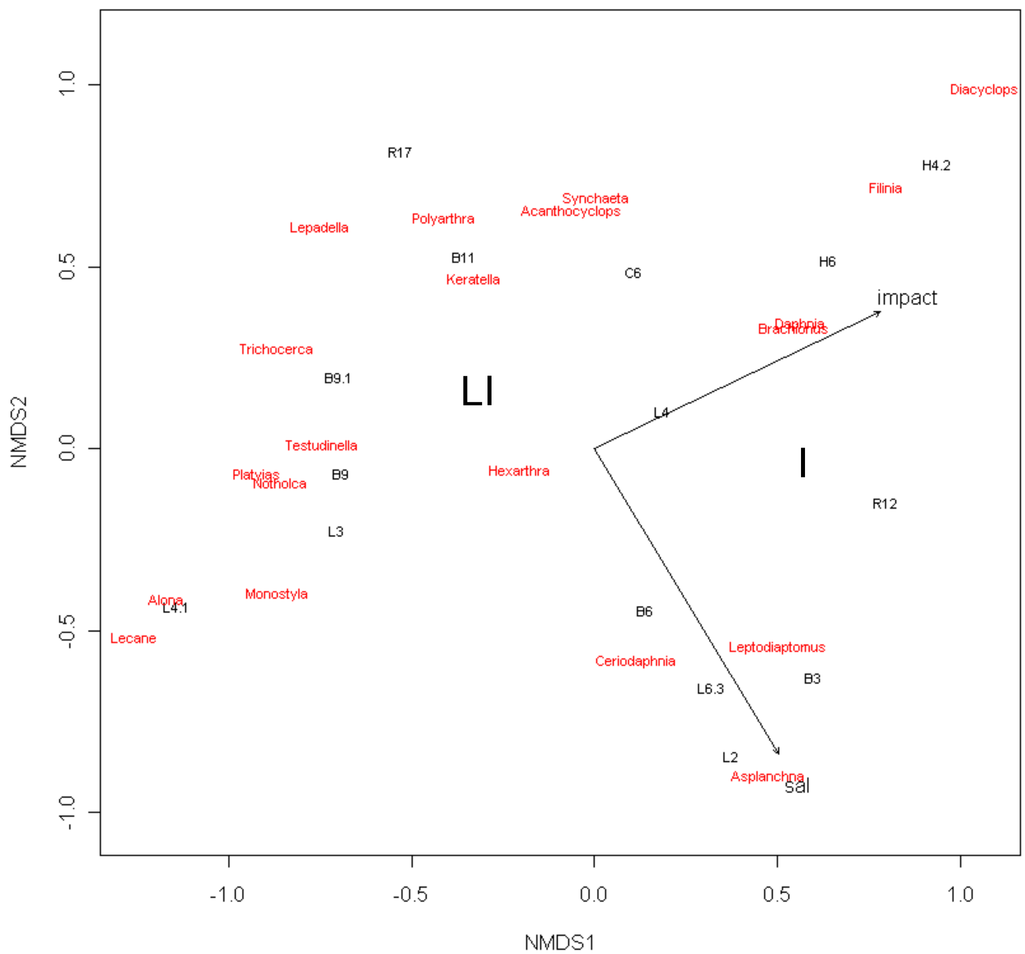


Figure 2.6 NMDS ordination with overlay of significantly correlated environmental and biological variables for August 2009. Sal=salinity, impact=percent cattle impact, Rot=rotifer abundance, naup=nauplii abundance, clad=cladoceran abundance, cope=all copepod abundance, calacope= calanoid copepod abundance.

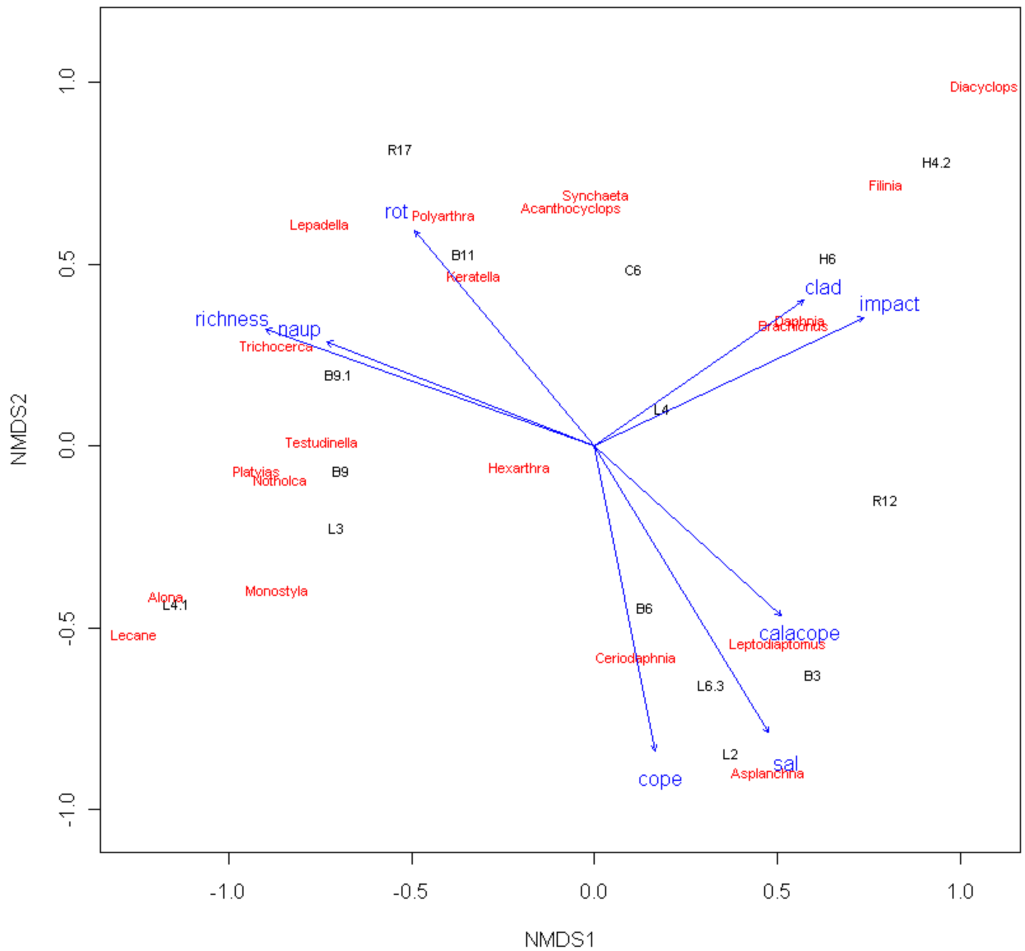
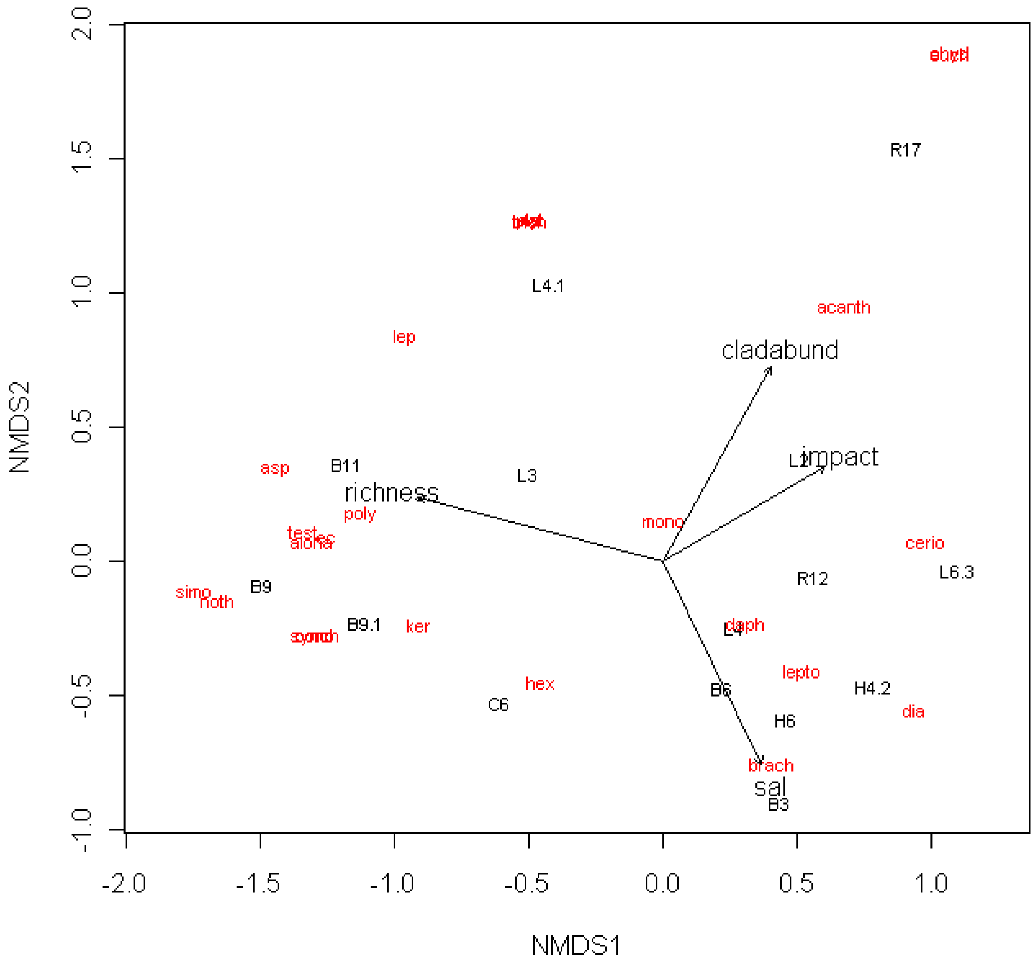


Figure 2.7 Wetland-by-species NMDS ordination with overlay of significantly correlated (p=.05) environmental variables and biological variables for September 2009. Cladabund=cladoceran abundance, Sal=salinity.



Community Change Through Time

A comparison of main taxa groups (cladocerans, calanoid copepods, cyclopoid copepods and rotifers) amongst sampling months revealed that zooplankton communities did not differ significantly over time. The same outcome was found when comparing the entire species matrices amongst sampling months. (Table 2.14 and 2.15).

Table 2.14 Summary statistics for MRPP analyses testing for significant differences in zooplankton taxa composition amongst sampling months.

Test	Testing:	Chance corrected within-group agreement A	Significance of delta:
MRPP	Main Taxa Groups	-0.010	0.72
MRPP	Species Composition	-0.009	0.78

Table 2.15 Summary statistics for Adonis analyses testing for significant differences in zooplankton taxa composition amongst sampling months.

Test	Testing:	Pr(>F)
Analysis of Dissimilarities	Main Taxa Groups	0.80
Analysis of Dissimilarities	Species Composition	0.80

Species Richness and Diversity

Most impacted sites had significantly lower species richness in August. Diversity measures were not found to differ significantly between impact categories. Overall species richness was significantly lower in the impacted categories compared to the least-impacted category (Table 2.16). Rotifer species richness was also significantly less in impacted wetlands decreasing on average from 6 to 2 species in this group (Table 2.16).

Table 2.16 Significant ANOVA, Kruskal-Wallace rank sum and Adonis tests comparing impacted (>25%) and least-impacted (≤25%) wetlands for August 2009.

					Impacted (8)		**Low Impact (n=7)**	
Variable	**TEST**	**F**	**Df**	**p-val**	**Mean**	**Range**	**Mean**	**Range**
Species Richness	ANOVA	48.4	1	1.001e-05	4.1	3.0 – 6.0	8.4	7.0 – 10.0
Rotifer Abundance	ANOVA	6.5	1	.024	181.0	0.0 - 961.9	642.4	55.8 - 1645.7
Rotifer Species Richness	K-W Rank Sum	-	1	.002	2.0	0.0 - 3	6.0	3.0 – 8.0
Cladoceran Abundance	ANOVA	3.6	1	.082	23.6	1.0 - 61.8	7.3	0.00 - 21.6
Copepod Abundance	ANOVA	4.9	1	.045	133.7	1.41 - 354.8	37.3	0.0 - 226.3
Calanoid Abundance	K-W Rank Sum	-	1	.036				
Distance matrices	Adonis	2.8	1	.019	-	-	-	-

Salinity appeared to cause a significant decline in zooplankton species richness for both August (Figure 2.8) and September. Multiple regression analyses indicated that for August, the decline in species richness was correlated with salinity, cattle impact percentage, total phosphorus and total nitrogen (Table 2.17). In a second significant model for August, salinity and cattle impact explained 68% of the variance in species richness (p=0.0004) (Table 2.16). In September, salinity and cattle impact were found to be the two most important parameters explaining 42% of the variance for species richness among wetlands. Figure 2.9 demonstrates the decreasing trend of species richness with impact. There is a noticeable change or possible threshold shown at approximately the 20-25% cattle impact level, in which species richness begins to markedly decrease.

Salinity was the single most important variable explaining variance in both Shannon-Weiner Diversity and Simpson’s Diversity in August (adjusted R^2=0.41, p=0.006; adjusted R^2=0.35, p =0.012, respectively). There were no significant measured variables correlated with diversity in September.

Figure 2.8 Regression plot of zooplankton species richness as a function of salinity (transformed) with locally weighted scatterplot smoothing line for August 2009.

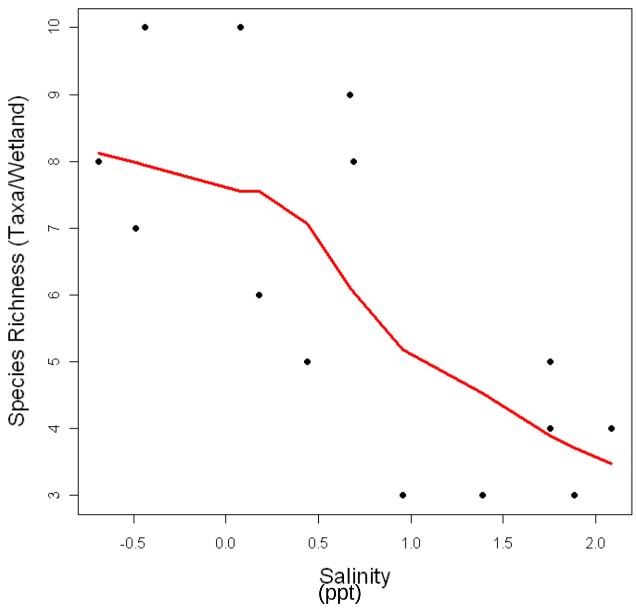
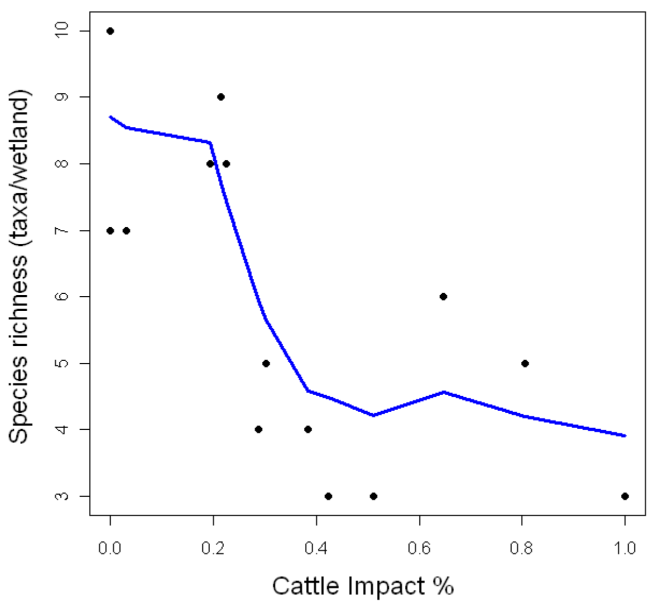


Figure 2.9 Regression plot of zooplankton species richness as a function of cattle impact % (transformed) with locally weighted scatterplot smoothing line for August 2009. Note marked decrease at ~ 25-27% impact.



Abundance

Zooplankton total abundance in August ranged from 22 to 2196 individuals per litre and was not significantly different between disturbance groups. Abundances of all three major groups of zooplankton (cladocerans, copepods and rotifers) were found to differ significantly between the two impact categories (Table 2.16). In high impact wetlands, rotifer abundance was significantly lower (adjusted R^2=0.28, p=0.024) and copepod abundance was significantly higher (adjusted R^2=0.48, p=0.045). Although not quite statistically significant, higher cladoceran abundances were found in most impacted wetlands (p=0.08).

Despite limited measured environmental parameters in July, one significant regression model demonstrates that 23% of total zooplankton abundance was explained by both chlorophyll-*a* and the cattle impact variable (adjusted R^2=0.23, p=0.04) (Table 2.17). For August, although not quite significant (p=0.07), total nitrogen and chloride combined to explain 26% of variation in total abundance (Table 2.17). Salinity played a major part in driving both copepod and rotifer abundance, (Table 2.17) but, did not appear to influence cladoceran abundance which appeared to be correlated with increasing phosphorus (total phosphorus explaining 33% of the model variance). In September, copepod abundance was explained in part by salinity (adjusted R^2=0.32, p=0.02) with a near-significant model demonstrating rotifer abundance among wetlands being partially explained by cattle impact (adjusted R^2 =0.17, p=0.07) (Table 2.17).

Biomass

In August, biomass ranged from 0.07 – 845.74 mg dry weight/m^3 (mean 268.2) and was not significantly different between wetland disturbance groups. The best regression model predictor was total phosphorus, accounting for 50% of the variance in the model (p=0.002) (Table 2.16).

Chlorophyll-a

ANOVA indicated that there were no significant differences in chlorophyll-*a* concentrations between wetland impact groups. As a dependent variable, chlorophyll-*a* was an influential variable for total zooplankton abundance in July and biomass in September (as described in the previous section).

Table 2.17 Summary statistics of significant linear models for all sampling sessions (alphabetical order by month). Abbreviations include clad=cladoceran, cope=copepod and rot=rotifer. Where there are more than one significant model using the same dependent variable, (a) and (b) are indicated.

Sampling Month	Dependent Variable	Independent Variable/s	Estimates	F	SE	p	df	R^2	Adj R^2
July	Total Abundance	*Intercept*	3.29	5.119		0.041	13	0.28	0.23
		Chlorophyll-*a*	0.10						
		Impact	5.97						
August	Biomass	*Intercept*	6.51	15.02	0.83	0.002	13	0.54	0.50
		TP	0.60						
August	Clad Abundance	*Intercept*	35.45	7.91	14.84	0.015	13	0.38	0.33
		TP	7.75						
August	Cope Abundance	*Intercept*	2.50	13.79	3.44	.0025	13	0.51	0.48
		Salinity	4.15						
August	Rot Abundance (a)	*Intercept*	2.14	7.60	5.42	0.007	12	0.56	0.49
		Chloride	2.89						
		Salinity	-5.89						
August	Rot Abundance (b)	*Intercept*	12.73	6.36	6.43	0.024	13	0.33	0.28
		Salinity	-4.52						
August	Shannon Index	*Intercept*	-0.14	10.78	0.38	0. 006	13	0.45	0.41
		Salinity	-0.35						
August	Simpson’s Index	*Intercept*	0.51	8.57	0.15	0.012	13	0.40	0.35
		Salinity	-0.13						
August	Species Richness (a)	*Intercept*	8.88	32.99	0.92	$8.546e\text{-}^{06}$	11	0.90	0.87
		Salinity	-1.70						
		TP	-0.45						
		TN	-0.59						
August	Species Richness (b)	*Intercept*	8.39	15.79	1.42	.0004	12	0.72	0.68
		Salinity	-1.38						
		Impact	-4.04						
August	Total Abundance	*Intercept*	12.59	3.418	4.00	0.067	12	0.36	0.26
		Chloride	2.01						
		TN	-1.55						
September	Biomass	*Intercept*	2.17	4.00	0.17	0.050	12	0.40	0.30
		Chlorophyll-*a*	0.11						
		Salinity	0.07						
September	Copepod Abundance	*Intercept*	2.67	7.45	3.92	0.020	13	0.36	0.32
		Salinity	3.38						
September	Rot Abundance	*Intercept*	5.61	3.83	3.45	0.070	13	0.23	0.17
		Impact	5.97						
September	Species Richness	*Intercept*	8.98	6.06	2.22	0.020	12	0.50	0.42
		Impact	-4.06						
		Salinity	-1.26						

Discussion

The goal of this study was to assess wetland zooplankton community structure in cattle grazed rangelands of the Southern Interior of British Columbia, and to relate community patterns to measured environmental variables. Sampling occurred over a period of three months, with most environmental variables assessed during August 2009, the focus sample month. No significant difference was found in species composition through time, resulting in similar NMDS ordination results for all months. Two-dimensional ordinations showed distinct separation of wetlands based on their communities. Amongst sampled wetlands there was a gradient of change in community composition from diverse rotifer assemblages found in least-impacted wetlands, to species-poor crustacean assemblages found with increasing cattle impact and salinity. As predicted, species richness was less in impacted wetlands; however, salinity appeared to have confounding effects.

Zooplankton Community Structure in Relation to Cattle Impact

<u>Impacted Wetlands</u>

Specific zooplankton taxa appeared to be associated with impacted wetlands. Zooplankton associated with most impacted wetlands in my study included the cladocerans *Diacyclops* and *Daphnia*, and the rotifers *Brachionus* and *Filinia*. Impacted wetlands on the lower end of the spectrum (50-75% impact) were primarily represented by the cladoceran genera *Ceriodaphnia*, *Daphnia* and the rotifer *Asplanchna*. In a long-term study examining the ability of zooplankton to be indicators of water quality in wetlands, Lougheed & Chow-Fraser (2002) found that *Daphnia*, *Brachionus*, *Filinia*, and *Asplanchna* typically dominated most impacted wetlands. As zooplankton taxa found in most impacted wetlands of this study also coincide with indicator zooplankton found by Lougheed & Chow-Fraser (2002), they are potential candidates as bioindicators of Southern Interior wetland water quality.

Cladocerans and copepods dominated wetlands most heavily accessed by cattle, with cladocerans associated with the most eutrophic sites (characterized by higher total phosphorus concentrations), and copepods associated with wetlands of highest salinities. My findings are consistent with other studies finding crustacean zooplankton linked with eutrophic water bodies (Pinto-Coelho et al., 2005; Van Egeren et al., 2011), with copepods specifically salinity-associated, and cladocerans linked with eutrophic conditions (Kagalou et al., 2010).

In August, half of the impacted wetlands were dominated by cyclopoid copepods. No cyclopoids were found in least-impacted wetlands in any sampling session. This observation is consistent with other studies finding the replacement of calanoids with cyclopoids with increasing wetland disturbance (Gannon & Stemberger, 1978; Pace, 1986). The presence of cyclopoids in impacted wetlands of this study could be an indication of environmentally impacted conditions induced by the presence of cattle; however, due to the limited number of wetlands containing cyclopoids, the sample size to support this claim is not optimal.

Least-Impacted Wetlands

A species rich assemblage of zooplankton was found in least-impacted wetlands. Aquatic vegetation is known to increase zooplankton habitat heterogeneity in wetlands in undeveloped landscapes (Lougheed et al., 2008), and has been shown to affect rotifer community composition (Duggan et al., 2001). Lougheed & Chow-Fraser (2002) found that pristine wetlands with submerged aquatic vegetation and more complex, heterogeneous environments contained diverse rotifer genera that were specific to these types of wetlands. Therefore, the high rotifer diversity observed in the least-impacted wetlands of my study could be attributed to the higher degree of habitat heterogeneity.

Based on the NMDS ordination biplot, there were no significant correlations between any measured environmental variable in least-impacted wetlands on the NMDS ordination. Variables associated with least-impacted zooplankton communities were either not measured, or the limited sample size was not large enough to uncover significant relationships. However, these results could also indicate that zooplankton communities can simply be more species rich in the absence of environmentally impacted conditions.

Zooplankton Community Structure in Relation to Salinity

It is important to note that the measurement of salinity in my study does not quantify and represent the various salts in the wetlands, but is calculated based on conductivity and temperature readings within the multimeter probe which are estimated based on seawater salt concentrations. However, as I am not addressing effects of individual ions on zooplankton, I can still use the salinity readings obtained from the multimeter in statistical analyses to infer salinity effects on zooplankton. Even relatively small changes in salinity concentrations have been shown to have substantial impacts on freshwater zooplankton assemblages (Schallenberg et al., 2003). A study has shown that there is a strong relationship between occurrence of zooplankton taxa and salinity within the Southern Interior of B.C. (Bos et al., 1996). In my study, salinity appeared to be heavily influencing the position of impacted wetlands in NMDS diagrams, based on community composition.

Salinity is a very important driver influencing aquatic invertebrate communities (Zimmer et al., 1999), and can exert significant effects on trophic dynamics. Salt concentration affects an organism's ability to osmoregulate (Wetzel, 1975), thus aquatic organisms have specific tolerance thresholds. Compared to freshwater lakes and wetlands, considerably different aquatic communities are found in brackish and saline water bodies (Brock et al., 2005; Jensen et al., 2010). In this study, salinity ranged from fresh to subhaline, thus differences among communities in water bodies differing in salinity classifications would be expected.

High salinity lakes and wetlands are associated with lower aquatic biodiversity (Brock et al., 2005; Jensen et al., 2010). With increasingly saline wetlands, zooplankton communities become less diverse as species reach their maximum salt tolerance and are unable to exist in more extreme conditions (Nielsen et al., 2008). Results from my study agree with this in that both species richness and diversity decreased with increasing salinity. Jeppesen et al. (1994) and Jensen et al. (2010) also found zooplankton richness and diversity to decrease as a result of increasing salinity in shallow temperate lakes. Not only was salinity a strong driver influencing species richness (August and September) and diversity (August), the abundance of copepods and rotifers individually decreased with increasing salinity concentrations in August and September. These results have also been found in studies of Danish and Canadian freshwater and brackish lakes (Hammer, 1993; Jeppesen et al., 1994).

Other studies examining zooplankton in wetlands of increasing salinities found that communities are commonly dominated by small-bodied cladocerans such as the cladoceran genus *Ceriodaphnia*, calanoid copepods and salt tolerant rotifers (Jeppesen et al., 1994, Brucet et al., 2009). My results support those findings, with *Ceriodaphnia* and the calanoid copepod *Leptodiaptomus* frequently occurring in wetlands of higher salinity. Calanoid copepods were found in six wetlands of which five are considered cattle impacted. These five wetlands were also the most saline. The rotifer *Asplanchna*, found to increase in abundance with increasing salinity in this study, is frequently observed in wetlands of both high salinity and nitrogen concentration (Angeler et al., 2010), and is commonly found in rotifer assemblages associated with saline and eutrophic wetlands (Nielsen et al., 2008).

A possible explanation for the observed range of salinities in seemingly similar wetlands is the varying geology of the Southern Interior region. This region contains many soil types and ionic compositions and concentrations in lakes and wetlands within close proximity (Topping & Scudder, 1977). Saline lakes are found in abundance in this region, with a wide-range of chemical compositions (Renaut & Long, 1989). Wetlands sharing similar salinity concentrations in this study could possibly share common soil types.

Climate change could also be another reason for higher salinity wetlands found in the Southern Interior region. Reduced annual precipitation and increasing dry-periods cause wetlands to retain less water, therefore increasing aquatic salinity concentrations (Nielson & Brock, 2009). Evaporative processes were found to be the primary factors influencing high salinities of certain lakes found in close proximity to lakes of very low salinities within B.C.'s interior region (Barjaktarovic & Bendell-Young, 2001). Alternately, salinity differences could be attributed to changes in surrounding land use practices, which are known to interfere with natural hydrology and regulation of salinity in freshwater wetlands (Brock et al., 2005). Many wetlands assessed with a high-impact rating also pertained to higher salinity readings, and this could be related to soil and sediment disturbance by cattle. Disturbance and mixing of the soil and sediments could release ions into the aquatic environment that would normally be retained within the sediments under undisturbed conditions.

Zooplankton Abundance, Richness and Biomass

<u>Abundance</u>

My prediction that total abundance would be highest in impacted wetlands was not supported. I had predicted that with greater nutrient input from cattle, there would be an increase in primary productivity and subsequently an increase in total zooplankton abundance. In July, total abundance was explained in part by chlorophyll-*a* and cattle impact. In August, total nitrogen was an influential parameter explaining variance in the model for total abundance. Aquatic nitrogen can drastically affect crustacean zooplankton communities (Bagella et al., 2010). This may have also been the case in September; however, water chemistry variables were not obtained and analysed.

With regards to zooplankton sub-groups, rotifer abundance in August was significantly lower in impacted wetlands compared to least-impacted, contrasting several studies finding that rotifer abundance (not richness) typically increased in eutrophic and impacted wetland environments (Beaver et al., 1999; Kagalou et al., 2010). One theory as to why rotifers are thought to dominate zooplankton abundance in impacted wetlands is due to increased turbidity. Kirk & Gilbert (1990) found rotifers dominating over cladoceran abundance in lakes of high turbidity. As my results contrast with many typical findings, there may be an undetermined mechanism governing abundances in these study wetlands. As the highest impacted wetlands are also the most saline, salinity may be a contributing factor in the abundance of rotifers. It is known that rotifer abundances can be significantly lower in wetlands of high salinity (Nielsen et al., 2003). In concurrence with the previous studies, results from September shown an increase of rotifer abundance with increased cattle impact.

Copepods and cladocerans were found to be most abundant in impacted wetlands. This is an uncommon finding in zooplankton-anthropogenic disturbance literature. In addition, it is in distinct contrast with that of Beaver et al., (1999) who found that both cladoceran and copepod abundance were much higher in wetlands non-impacted by land use. Therefore, due to the lack of data correlating crustacean abundance with eutrophic or impacted water bodies, my results indicate that environmental variables, most likely salinity, are influencing copepod and cladoceran abundances in high impact/high saline wetlands. Regression results support this, finding a positive trend in cladoceran abundance with increasing wetland salinity for both August and September (July salinities not measured). In August, total phosphorus was a highly important driver for cladoceran abundance. It is not common to find studies which link cladoceran abundance with total phosphorus; however, total phosphorus has been shown to dramatically influence cladoceran species composition (Jeppesen et al., 2000).

Species Richness

My prediction that impacted wetlands would harbour lower species richness was supported. These wetlands averaged less than half as many species compared to least-impacted wetlands. Species richness decreased with cattle impact for all three sampling sessions; however, again, this could also be due to impacted wetlands commonly having highest salinities. In August, total phosphorus and total nitrogen were shown to negatively influence species richness in a highly significant regression model. Aquatic nutrients are proven surrogates for productivity in lakes (Carpenter et al., 1985), and increased productivity commonly corresponds with lower diversity (Jeppesen et al., 2000).

Low zooplankton richness is a typical trait of hypereutrophic, impacted systems (Dodson et al., 2005; Kagalou et al., 2010). Similar results are reported from studies analyzing various land use disturbances on wetland zooplankton species richness. Dodson & Lillie (2001) found that taxon richness in agricultural sites was significantly less compared to least-impacted wetland types in palustrine settings across Wisconsin, USA. In a comparison of sixteen isolated wetlands, Lougheed et al. (2008) found that wetlands within developed environments were nutrient-rich and contained significantly lower zooplankton species richness. In addition, Dodson et al., (2005, 2007) found that zooplankton species richness is indirectly associated with land use in shallow lakes, with significantly less taxa appearing in sites impacted by watershed development.

Studies which show no impact on zooplankton richness include those of Dodson et al. (2009), which found that watershed usage did not have an effect on species richness in impacted sites, Beaver et al. (1999), finding zooplankton richness differing insignificantly amongst categories of disturbance and Van Egeren et al. (2011), which found that richness actually increased with watershed agricultural use. The results of these studies contradict my findings; however, as they all examined larger water bodies (lakes) with much greater depth and surface area, there could be a "dilution effect" resulting in undetectable impacts from anthropogenic activities. Due to the fact that the contrasting studies of Dodson et al. (2009) and Beaver et al. (1999) examine land use practices other than livestock grazing (i.e., watershed housing development and crop fertilizer contamination, respectively), the mechanisms driving these impacts cannot directly be compared to cattle disturbance effects on species richness.

Due to their small size, smaller water bodies are more sensitive to influences of water and sediment quality compared to larger bodies of water that have an enhanced buffer capacity to resist environmental change (Müller et al., 1998). This could be why impacted wetlands contained lower zooplankton diversity in this study. In addition, many similar studies did not have the overarching issue of interfering salinity effects. The varying results from zooplankton disturbance studies emphasize the many possible ways in which land disturbance practices can influence wetland ecosystems.

Biomass

The cattle impact variable was not found to influence zooplankton biomass; however, total phosphorus positively increased biomass in August. It is well accepted that total phosphorus positively influences zooplankton biomass (Conde-Porcuna et al., 2002; Gyllström et al., 2005). Additionally, it has been found to play a major role in the connection between land development and increased zooplankton biomass (Gélinas & Pinel-Alloul, 2008).

Conclusion

My study indicated that although wetland environmental disturbance by cattle could very possibly be affecting zooplankton community structure in terms of both species richness and species composition, salinity also proved to be a very strong factor limiting species richness and influencing abundance. My results supporting the significant influence of cattle on zooplankton community structure reflect previous field based studies that examined zooplankton in freshwater wetlands under disturbed and undisturbed conditions (Dodson & Lillie, 2001; Dodson et al., 2005; Hoffmann & Dodson 2005; Lougheed et al., 2008; Dodson et al., 2009). These studies, which examined the impacts of land use on zooplankton communities in freshwater environments, consistently find a decreasing trend in zooplankton species richness under disturbed conditions.

Due to the plethora of ecological, recreational and economic benefits wetlands provide, conservation of these essential habitats through protective measures is vitally important. As one of the first analyses relating zooplankton community structure to environmental disturbance in the Interior of British Columbia, this study provides insight into how zooplankton are structured within rangeland environments. Through examining communities and relating patterns to cattle impact intensity, there are possible effects from cattle disturbance; however, as there is evidence to suggest that other environmental parameters such as salinity and nutrients influence the way zooplankton communities are structured, study results are unable to directly implicate cattle with having direct or indirect influences on zooplankton communities. Nevertheless, this study reveals that communities do differ dramatically even within a small geographical region.

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CHAPTER 3

Conclusion and Management Implications

As urbanization, agricultural practices, and industrial growth increase with the world's swelling human population, strain on natural resources is naturally intensifying. Research on impacts of anthropogenic activities is critical if we wish to preserve healthy ecosystems and biodiversity. Human land use practices have caused reduction of biodiversity in both aquatic and terrestrial ecosystems (Brönmark & Hansson, 2002; Foley et al., 2005), the consequences of which include severe impacts on ecosystem functioning and sustainability (Tilman et al., 1996). Certain ecosystems, particularly wetland systems, are very susceptible to impacts from anthropogenic disturbances. Because wetlands are some of the most biologically diverse ecosystems in the world (Williams, 1999) and provide humans with innumerable beneficial services (Woodward & Wui, 2001), protection and preservation of these environments is critical.

Wetlands in British Columbia are an essential part of the landscape and provide vital habitat for many species (Mackenzie & Shaw, 2000). Due to their high rates of production, strong selection pressures and unique assemblages (Gibbs, 2000; Scheffer et al., 2006), biodiversity is generally very high in these systems. In the Southern Interior of B.C., activities such as urban development, agriculture and recreation have large impacts on the natural environment, disrupting trophic dynamics and ultimately threatening biodiversity (MacKenzie & Shaw, 2000).

Rangeland cattle ranches are prevalent in British Columbia's interior region (Fraser, 2009); however, they pose a potential threat to wetland ecosystems under poor management practices. In wetlands located in rangeland environments, cattle grazing can both indirectly and directly affect biodiversity, potentially triggering trophic cascades and ultimately causing a shift in populations of aquatic organisms (Dodson et al., 2005). As Canadian legislation controlling cattle access to wetlands is not strongly enforced (MacKenzie & Shaw, 2000), cattle are often given unrestricted access to wetlands that are located within their ranges. Livestock overuse of wetlands can cause substantial effects on the health of wetlands and riparian areas (Kauffman & Krueger, 1984), highlighting the necessity to improve management strategies that incorporate the promotion of sustainable wetland practices.

To assess disturbance impacts upon wetland ecology, monitoring and assessment of certain groups of organisms, particularly zooplankton, can be a valuable method of detecting environmental stress. The study of zooplankton in small, shallow wetland environments has proven to be very successful in demonstrating responses to anthropogenic activities within the wetland watershed (Dodson et al., 2005); therefore, examining zooplankton community responses to environmental conditions could give helpful insight into community dynamics in response to cattle presence.

Research Summary

My research was a field-based survey of zooplankton community structure in wetlands of varying cattle impact in the Southern Interior of British Columbia. Study sites were selected for homogeneity of physical attributes, sampling ability (minimum depth greater than that of the tube-sampler length) and range of apparent exposure to rangeland cattle. Hydrochemical, chlorophyll-*a* and morphological data were obtained from all sites.

Zooplankton community data was evaluated by the use of multivariate methods to visualize wetland similarities based on their assemblage structure, as well as to determine the variables that were most responsible for observed patterns of species occurrence and abundance. Individual zooplankton parameters (species richness, diversity, biomass and abundance) were additionally examined in linear analyses to detect the strength of influencing variables.

Overall, both salinity and cattle impact percentage appeared to have an influence on how wetland communities were structured. Comparing wetlands classified into either “impacted” or “least-impacted” cattle-disturbance categories yielded significant differences in zooplankton community structure with respect to species present and richness and abundances of total zooplankton and individual taxonomic groups for all wetlands. Zooplankton communities from impacted sites included specific zooplankton taxa typically associated with eutrophic water bodies, and showed markedly decreased numbers of species compared to less impacted sites. Communities in least-impacted wetlands yielded a much more diverse assemblage of zooplankton taxa. Nutrients (total phosphorus and total nitrogen) were significant drivers of increased zooplankton richness and abundance. Chlorophyll-*a* was found to influence zooplankton abundance and biomass, although results were inconsistent through the summer season.

It is difficult to conclude what is truly driving community structure in these specific wetlands with limited measured variables. Other known processes shown to exert an influence upon zooplankton communities include predation and dispersal processes, among others. This study demonstrated the significant influence of salinity on zooplankton assemblages, a finding that is supported by many other multivariate studies confirming the dominance of salinity relative to other environmental parameters in aquatic environments (Attayde & Bozelli, 1998, Swadling et al., 2000; Schallenberg et al., 2003). Since both salinity and the cattle impact variable were significant in influencing zooplankton in this study, differences in communities of impacted wetlands compared to least-impacted are most likely due to a combination of conditions induced by salinity concentrations and possibly indirect and direct effects of cattle presence.

Study Significance

The information gained from this study is important in that it provides a survey of wetland zooplankton of the Southern Interior region and demonstrates the capabilities of using zooplankton as bioindicators in cattle-impacted environments. The use of zooplankton as indicator organisms in future monitoring studies could lead to the development of a regional zooplankton biotic index of integrity, which would enable the exploitation of the indicator properties of zooplankton assemblages to monitor aquatic health in British Columbia’s interior. The promotion and implementation of sustainable practices that benefit grasslands and wetlands is critically necessary if these ecosystems are to endure. As cattle rely upon healthy wetlands for water resources in dry grassland landscapes, it is in the best interest of ranchers to work compatibly with law-makers and environmentalists to investigate and implement sustainable ranching practices.

This study is important globally to wetland research as it supports the use of zooplankton as indicator organisms for assessment of overall wetland health, which is still a relatively new concept. In concordance with previous studies, I observed that under increasingly stressful conditions, there is a decline in species richness and a shift in zooplankton community composition; however, the role of salinity in this process is undetermined. Although relatively small-scale, my study applied methods and analyses commonly used in recent land use/zooplankton studies to gain insight into how zooplankton communities are constructed within a potentially impacted environment. Zooplankton-disturbance studies continue to offer a wealth of information both in providing further support for the use of zooplankton as indicators of environmental health, and in demonstrating possible effects of anthropogenic activities on the often overlooked but vastly important plankton communities of seemingly resilient ecosystems.

Management Implications

Management of wetland ecosystems requires combining scientific-based wetland knowledge with the realities of economic, legal and governmental capabilities (Mitsch & Gosselink, 2007). In an attempt to protect grassland environments and wetland riparian areas from the effects of livestock overgrazing in Canada, Best Management Practices (BMPs) are currently in place (Agriculture and Agri-Food Canada, website). BMPs directed specifically at wetland management are much less detailed, and although guidelines for suggested rangeland practices do exist (Ducks Unlimited Canada, 2009), there are limited formal protection measures. The lack of strict regulations in place to effectively regulate the use of wetlands by cattle in B.C.'s rangelands may lead to inappropriate range practices, resulting in some areas experiencing increased disturbance impacts.

It is difficult to impose regulations that benefit both the ranching industry and those interested in wetland management. Conflicts of interest may arise between landowners interested in continuing potentially unsustainable grazing practices, and environmentalists focused on curbing degrading land use activities (Pyke & Marty, 2005). An approach to land management that encourages a collaborative, mutually beneficial effort between land managers, including private landowners and ranchers is recommended (Euliss & Mushet, 1999). Ranching operations that have made efforts to balance cattle access amongst available wetlands have been able to limit nutrient loading in surface waters (Sigua et al., 2006). These efforts have resulted in significantly less impact on wetland ecosystems compared with operations that allow unrestricted access. As this example demonstrates, alterations in policies and management strategies can lead to a reduction of harmful environmental impacts.

Limitations and Future Research Possibilities

As a survey-based field study, my study presents some shortcomings. As there were no controlled experiments, it was impossible to control for interfering influences of other environmental parameters. Secondly, focusing primarily on limited sampling events is a "snapshot approach" that ignores long-term community dynamics. However, there have been a large number of studies based on single sampling events which have proven to be effective and useful in representing freshwater wetland zooplankton communities' response to anthropogenic impacts (Stemberger, et al., 2001; Dodson et al., 2005). Lastly, the measurement of percent cattle impact was a self-determined variable. As cattle accessing wetland environments can produce a range of potential impacts, it was impossible to quantify to what extent cattle "disturb" the wetlands; however, measuring the percentage of pond perimeter trampled appeared to be a decent indication as to how frequently wetlands were accessed by cattle.

Further research is required in order to fully comprehend the effects of cattle on zooplankton and to properly characterize disturbance-indicator species. This can be accomplished through addressing a wider range of influential factors (including effects of dispersal and predators), and sampling a larger range of chemical, physical and biological variables. Using zooplankton to develop indices of biological integrity would be a significant step towards a region-specific long-term monitoring program. As these indices require long-term and consistent monitoring of zooplankton communities for specific environments, preliminary studies that assess assemblage structure and species presence are required. In the Southern Interior region, future research should carefully select wetlands with a more homogenous salinity range, or ensure that the two gradients are not correlated amongst study sites prior to analysis. In addition, controlled experiments, as opposed to purely field-based surveys, should be conducted in conjunction with field research to help support study findings.

While complex relationships between environmental variables, land use practices and zooplankton community structure are difficult to assess, it is essential to understand their mechanisms in order to provide critical information for their future evaluation. Early detection of changes in zooplankton community structure may provide necessary indications of ecosystem disturbance that are required in part for effective habitat management.

Concluding Remarks

Zooplankton communities are incredibly dynamic, and it is therefore important to exercise caution when invoking causal relationships between zooplankton community compositions and wetland conditions (Gannon & Stemberger, 1978). Despite being unable to directly connect cattle impact with zooplankton in this study, I recommend that livestock should still be controlled in a way that avoids overgrazing and over access as physical disturbances are apparent in many wetlands. This can be achieved either by herd rotation or a physical restriction of entry (fencing). Fencing could be implemented temporarily, or could be in place to restrict cattle access from a specified portion of the wetland. Additionally, providing alternate water sources for cattle can aid to reduce destructive impacts in wetlands.

It is possible to offset wetland losses through protective measures, restoration campaigns and greater public understanding of the values of wetlands (Brinson & Malvárez, 2002). An understanding of the complex abiotic and biotic processes controlling lower trophic level dynamics in wetlands is vital to assist in preventative and remedial measures targeted at saving aquatic habitats (Lougheed & Chow-Fraser, 1996). Thus, ecological research examining the consequences of anthropogenic activities affecting wetlands is crucial. When wetland rehabilitation methods are designed to address the environmental setting, geological and hydrological past, and usage history of the particular location in question, wetland diversity can be restored (Bedford, 1999; Dodson & Lillie, 2001).

With British Columbia's growing population, environmental pressures from urbanization, agriculture, forestry, resource harvesting and recreational activities are escalating. With the mounting strain from these land use practices, it is essential to establish wetland management policies that mitigate environmental effects induced by these practices. Policies based on current scientific studies are critical to preserve healthy ecosystems while maintaining and promoting the growth of resource-based communities (Delesalle, 1998; MacKenzie & Shaw, 2000). In the Southern Interior of B.C., zooplankton field surveys should be promoted in environmental research as the information on community structure can provide a wealth of valuable information for long-term monitoring programs.

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APPENDIX A

Wetland Photos
(In order of cattle impact, in order of highest – lowest)


H4.2


C6


H6


R12


B6


L6.3

L2

B3

R17

L3


L4.1


L9.1

L4

L9.0

B11