EFFECTS OF LIVESTOCK GRAZING ON AQUATIC MACROINVERTEBRATES IN SOUTHERN INTERIOR WETLANDS OF BRITISH COLUMBIA, CANADA

by

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ABSTRACT

Grasslands in the southern interior of British Columbia are extensively grazed by free ranging livestock. Water sources are limited in these grassland landscapes and wetlands are commonly used by livestock for drinking water and forage. My study examined the impacts of livestock disturbance on the abundance, biomass and community composition of aquatic macroinvertebrates residing in these wetlands. Aquatic macroinvertebrates were collected in the spring and summer of 2008 from 17 wetlands with a range of grazing disturbance. Three sweep and core samples were collected from each wetland and grazing intensity was determined by the amount of bare ground at each site. Spring sweep total abundance (r^2=0.464, p=0.003) and biomass (r^2=0.728, p<0.001) were negatively correlated with livestock disturbance as were spring abundance and biomass of zygopterans (r^2=0.593, p<0.001; adj. r^2=0.513, p=0.001). Spring sweep family richness (r^2=0.462, p=0.003), Shannon's family diversity (r^2=0.569, p<0.001) and Simpson's family diversity (r^2=0.385, p=0.008) also decreased as livestock disturbance increased. Resource managers should consider Zygoptera (damselflies) as a potential indicator of wetland water quality and livestock impact. Range plans should adopt only light grazing in wetland areas and limit livestock access to sustain the biodiversity and productivity of these valuable aquatic ecosystems.

Keywords: aquatic macroinvertebrates, grassland wetlands, livestock grazing, British Columbia, Zygoptera

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GLOSSARY OF TERMS

Benthic invertebrates: Aquatic invertebrates living on, or in, the bottom substrates of an aquatic habitat.

Bioturbation: The restructuring of sedimentary deposits, as in a lake bottom or seabed, by living organisms (e.g., worms, clams) by activities such as burrowing, or ingestion or defecation of sediment grains.

Depressional wetlands: Wetlands that occur in topographical depressions which allow the accumulation of surface waters.

Diversity: A quantitative measure that reflects how many different types (e.g., invertebrate families) there are in a dataset and simultaneously takes into account the proportions of individuals or how evenly the individuals are distributed among those types.

Endophytic ovipositor: An organism (e.g., damselfly) that uses a specialized abdominal organ (ovipositor) to insert its eggs into plant tissue.

Lentic systems: Aquatic habitats situated in still fresh water.

Lotic systems: Aquatic habitats situated in flowing fresh water.

Nektonic invertebrates: Aquatic invertebrates living in the water column of an aquatic habitat, including those mobile organisms along the nekton/benthic (epi-benthic) boundary and those found on aquatic plants (epi-phytic).

Richness: A quantitative measure that reflects how many different types (e.g., invertebrate families) there are in a dataset.

Latin and common names for the aquatic invertebrates examined in this study are listed in Appendix C.

CHAPTER 1. INTRODUCTION

Wetlands throughout the world are being lost at an alarming rate (Zedler and Kercher 2005; Mitsch and Gosselink 2007). As wetland function and productivity are extremely valuable to society, conservation of these ecosystems is of utmost importance. In British Columbia's (BC) southern interior, ranching operations use grassland and wetland areas to provide abundant forage and drinking water for domestic livestock. Impacts of livestock disturbance on wetland aquatic invertebrate communities have not been thoroughly examined in Canada (Hornung and Rice 2003; Foote and Rice Hornung 2005; Silver and Vamosi 2012), and research in BC is lacking from the primary literature. Concern has been expressed by government agencies, Ducks Unlimited Canada, local researchers and ranching operators over the effectiveness of best management practices and sustainable levels of livestock use (Bruce Harrison, pers. comm. 2007). Southern interior wetland biodiversity and productivity can be conserved by developing a better understanding of this complex relationship between wetlands and ranching practices. This chapter will discuss southern interior grassland and wetland ecosystems, livestock disturbance impacts on wetlands, aquatic invertebrates and how aquatic invertebrates can be used to assess wetland health. I will finish with an outline of my thesis objectives.

Grassland Ecosystems

British Columbia grasslands are rare on the landscape and occupy less than one percent of the land base (Wikeem and Wikeem 2004). Located in areas of the province where summers are hot and dry with little precipitation, provincial grasslands are in the northern reaches of the Great Basin shrub-steppe grasslands found in the western United States and Mexico (GCCBC 2011). Almost 90% of BC grasslands lie within the hot semi-arid southern interior region. Grasslands provide many recreational opportunities and are economically important to the beef cattle industry, which relies on them for forage. Water is limiting in BC grasslands (Tisdale 1947), and wetland habitats are often primary sources of water for both domestic and wild animals.

Wetland Ecosystems

Wetlands are integral components of grassland ecosystems and critical to the survival of many organisms (Mitsch and Gosselink 2007). Generally defined as areas with hydric soils and hydrophytic vegetation, wetlands are highly variable in size, hydrology, water chemistry and geographic area (Rader 2001; MacKenzie and Moran 2004). Wetland ecosystems provide many important functional contributions to the environment and, thus, to society. Wetlands mitigate water quality by filtering sediments, nutrients and pollutants flowing from surface water, streams, rivers, lakes and ground water. They recharge ground water, provide relief from flood waters, reduce soil erosion, sequester carbon from the atmosphere, and provide habitat and forage for many wildlife species (Delesalle 1998, MOE 2004). Economically, wetlands provide many recreation opportunities such as fishing, hunting, and bird watching. They supply irrigation for agriculture operations, and water, shade and forage for livestock. Wetlands are under increasing pressures from development and, it is more important now than ever that we understand how to manage them for multiple users while maintaining the integrity of the resource for those wildlife dependent upon them.

The depressional wetlands of BC's southern interior occur as a mosaic throughout the grassland landscape and are similar to prairie pothole wetlands in terms of their ecology, climate and hydrology. They rely on rainwater or snow melt, have cycles of wet and dry years, exist in semi-arid climates with warm summers and cold winters and are usually fishless (Wikeem and Wikeem 2004). These shallow open water wetlands are dynamic productive ecosystems and can be further classified based on duration of flooding (Stewart and Kantrud 1971). Different durations of flooding create different vegetation communities, which in turn provide a variety of habitats and food resources for aquatic invertebrates and both terrestrial and aquatic vertebrates (Murkin and Ross 2000). In BC, 32 species at risk are among the 30% of wildlife dependent upon southern interior wetlands for survival (Delesalle 1998; Wikeem and Wikeem 2004). Many studies have examined the physical and chemical attributes and aquatic invertebrate communities in BC interior wetlands (e.g., Topping and Scudder 1977; Cannings and Scudder 1978; Cannings et al. 1980; Cannings and Cannings 1987); however, the effects of livestock grazing on these ecosystems has yet to be explored. Poor management of livestock could potentially decrease habitat quality and quantity for all wetland users.

Livestock Grazing

Livestock ranching is a widespread land use within BC grasslands, and if poorly managed can be detrimental to wetland structure and function (Collins et al. 1998; Steinman et al. 2003). Livestock spend a disproportional amount of time in wetlands versus upland areas grazing submergent, emergent and shoreline vegetation, drinking, loitering and cooling off (Adams and Fitch 1998; Nader et al. 1998; Ganskopp 2001). Effects of livestock disturbance in wetland ecosystems can be both direct and indirect. Direct impacts include vegetation trampling and removal, and fecal and urine inputs which decrease water quality and reduce habitat availability (Coffin and Lauenroth 1988; Collins et al. 1998; Steinman et al. 2003). Cattle feces and urine decrease dissolved oxygen, vascular plant richness and percent cover and increase algal productivity in vernal pool mesocosm experiments (Croel and Kneitel 2011). Indirect effects result from shifts in vegetation communities which induce changes throughout higher trophic levels and affect wetland productivity (Rader and Richardson 1994; Dodson et al. 2005). Livestock-induced changes to wetlands have caused reductions in available cover and nesting habitat and food resources for waterfowl (Ryan et al. 2006).

Aquatic Invertebrates

As an important ecological link between primary production and higher trophic levels, aquatic invertebrates are a critical component of wetland food webs. In wetland ecosystems invertebrates are primary consumers affecting primary production through consumption of living vegetation as herbivores or consumption of plant and animal litter as detritivores (van der Valk 2012). They also are secondary consumers or predators, feeding on zooplankton, other aquatic invertebrates and small vertebrates (Hershey and Lamberti 2001; White and Roughley 2008). Aquatic invertebrates provide abundant food for secondary consumers and many vertebrates such as waterfowl are dependent on them. Several studies have demonstrated that waterfowl select wetland habitats based on invertebrate densities (e.g., Murkin et al. 1982; Murkin and Kadlec 1986). Wetland invertebrates provide waterfowl with a crucial source of dietary protein (Krapu 1974) and inadequate invertebrate densities during the breeding season are known to reduce clutch size, egg viability and brood survivorship (Krapu 1981; Cox et al. 1998). Changes in abundance and diversity of aquatic macroinvertebrate communities can be useful in determining livestock impacts on wetland productivity and function due to their important contributions to wetland food webs.

Our knowledge of livestock grazing impacts on aquatic invertebrates in wetland ecosystems in Canada is limited. Examinations of aquatic invertebrate communities are a well-documented method of assessing and monitoring aquatic resources. Biomonitoring studies have traditionally used aquatic invertebrates to examine the effects of anthropogenic activities on water and habitat quality in rivers and streams in North America (e.g., Hilsenhoff 1988; Rosenberg and Resh 1993; Barbour et al. 1999). The use of biomonitoring practices are wide ranging, examining the before and after effects of projects, activities or release of toxicants (e.g., Mackay and Heise 2007; Thomson et al. 2005; Johnson et al. 2015), compliance monitoring for release of pollutants into aquatic systems (e.g., Lowell and Culp 2002), and larger-scale studies to compare impacts of land use practices against unstressed reference condition sites (e.g., Bailey et al. 1998). Aquatic invertebrates have also been used to examine agricultural impacts on lotic systems including those of cattle grazing (e.g., Moore and Palmer 2005; Carlisle et al. 2008). More recently, biological assemblages have been employed to assess wetland function; however, relatively few studies have applied this technique to determine the impacts of livestock grazing (e.g., Steinman et al. 2003; Davis and Bidwell 2008). Canadian studies on wetland invertebrate assemblage response to grazing disturbance are rare. Alberta research has revealed that odonates (damselflies and dragonflies) could potentially be used as indicators of wetland health due to their close association with and dependence on wetland vegetation and the relative ease of their collection and identification (Hornung and Rice 2003, Foote and Rice Hornung 2005).

Thesis Objectives and Format

The primary objective of this research was to examine the effects of livestock disturbance on BC's southern interior wetland invertebrate communities. Wetland aquatic invertebrate community density and structure were analyzed in response to a gradient of livestock disturbance levels and environmental parameters. Overall predictions were that because of habitat loss and degradation 1) heavy livestock disturbance would decrease the richness and diversity of wetland aquatic macroinvertebrate communities and 2) zygopteran (damselfly) abundance and biomass would decrease with heavy levels of livestock grazing because these taxa have been described in Alberta studies as particularly sensitive to grazing-induced reductions in wetland vegetation. A secondary project objective was to characterize and provide baseline data on aquatic invertebrate densities and community composition in wetlands near Kamloops, BC. By examining a range of grazing intensities within four different areas on private and public lands, I expected to provide regionally-specific recommendations for sustainable wetland use to local land managers. Chapter 2 describes the study design and the results of livestock grazing on wetland aquatic invertebrate communities. Chapter 3 concludes the thesis with a discussion on the study's limitations, future directions for research and recommendations for resource managers.

Literature Cited

Adams B., and L. Fitch. 1998. Caring for the green zone – riparian areas and grazing management. 2nd edition. Riparian Habitat Management Project, Lethbridge, AB, Pub. No. I-581.

Bailey, R. C., M. G. Kennedy, M. Z. Dervish, and R. M. Taylor. 1998. Biological assessment of freshwater ecosystems using a reference condition approach: comparing predicted and actual benthic invertebrate communities in Yukon streams. Freshwater Biology 39(4):765-774.

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, 2nd edition. United States Environmental Protection Agency; Office of Water, Washington, DC. EPA 841-B-99-002.

Cannings, R. A., S. G. Cannings, and R. J. Cannings. 1980. The distribution of the genus *Lestes* in a saline lake series in central British Columbia, Canada (Zygoptera: Lestidae). Odonatologica 9(1):19-28.

Cannings, R. A., and S. G. Cannings. 1987. The Odonata of some saline lakes in British Columbia, Canada: Ecological distributions and zoogeography. Odonatologica 3:7-21.

Cannings, R. A., and G. G. E. Scudder. 1978. The Littoral Chironomidae (Diptera) of saline lakes in central British Columbia. Canadian Journal of Zoology 56:1144-1155.

Carlisle, D. M., C. P. Hawkins, M. R. Meador, M. Potapova, and J. Falcone. 2008. Biological assessments of Appalachian streams based on predictive models for fish, macroinvertebrate, and diatom assemblages. Journal of the North American Benthological Society 27(1):16-37.

Coffin, D. P. and W. K. Lauenroth. 1988. The effects of disturbance size and frequency on a shortgrass plant community. Ecology 69:1609-1617.

Collins, S. L., A. K. Knapp, J. M. Briggs, J. M. Blair, and E. M. Steinauer. 1998. Modulation of diversity by grazing and mowing in native tallgrass prairie. Science 280:745-747.

Cox, R. R., M. A. Hanson, C. C. Roy, N. H. Euliss, D. H. Johnson, and M. G. Butler. 1998. Mallard duckling growth and survival in relation to aquatic invertebrates. Journal of Wildlife Management 62(1):124-133.

Croel, R. C. and J. M. Kneitel. 2011. Cattle waste reduces plant diversity in vernal pool mesocosms. Aquatic Botany 95(2):140-145.

Davis, C. A. and J. R. Bidwell. 2008. Response of aquatic invertebrates to vegetation management and agriculture. Wetlands 28(3):793-805.

Delesalle, B. 1998. Understanding Wetlands: A Wetland Handbook for British Columbia's Interior. Ducks Unlimited Canada, Kamloops, BC.

Dodson, S. I., R. A. Lillie, and S. Will-Wolf. 2005. Land use, water chemistry, aquatic vegetation, and zooplankton community structure of shallow lakes. Ecological Applications 15:1191–1198.

Foote, A. L., and C. L. Rice Hornung. 2005. Odonates as biological indicators of grazing effects on Canadian prairie wetlands. Ecological Entomology 30:273-283.

Ganskopp, D. 2001. Manipulating cattle distribution with salt and water in large arid-land pastures: a GPS/GIS assessment. Applied Animal Behaviour Science. 73:251-262.

Grassland Conservation Council of British Columbia (GCCBC). 2011. Grasslands of BC. Available from http://www.bcgrasslands.org. [accessed August 2011].

Hershey, A. E. and G. A. Lamberti. 2001. Aquatic Insect Ecology. Pages 733-775 *in* Thorp, J. H., and A. P. Covich (editors). Ecology and classification of North American freshwater invertebrates, 2nd Edition. Academic Press, San Diego, CA.

Hilsenhoff, W. L. 1988. Rapid field assessment of organic pollution with a family-level biotic index. Journal of North American Benthological Society. 7:65-68.

Hornung, J. P., and C. L. Rice. 2003. Odonata and wetland quality in southern Alberta, Canada: a preliminary study. Odonatologica 32:119-129.

Johnson, B. R., P. C. Weaver, C. T. Nietch, J. M. Lazorchak, K. A. Struewing and D.H. Funk. 2015. Elevated major ion concentrations inhibit larval mayfly growth and development. Environmental Toxicology and Chemistry 34(1):167-172.

Krapu, G. L. 1974. Feeding ecology of pintail hens during reproduction. Auk 91:278-290.

Krapu, G. L. 1981. The role of nutrient reserves in mallard reproduction. Auk 98:29-38.

Lowell, R. B. and J. M. Culp. 2002. Implications of sampling frequency for detecting temporal patterns during environmental effects monitoring. Water Quality Research Journal of Canada 37(1):119-132.

Mackay, C.A., and B.A. Heise. 2007. Effects of Logging on Export of Organic Matter from Headwater Streams. Forest Science Program Final Technical Report. Executive Summary available from https://www.for.gov.bc.ca/hfd/library/FIA/2007/FSP_Y073367a.pdf [accessed January 2015].

MacKenzie, W. H. and J. R. Moran. 2004. Wetlands of British Columbia: a guide to identification. Research Branch, BC Ministry of Forests, Victoria, BC, Land Management Handbook No. 52.

Ministry of the Environment (MOE). 2004. Wetlands in BC. Available from http://www.env.gov.bc.ca/wld/wetlands.html#first [accessed August 2011].

Mitsch W.J., and J. G. Gosselink. 2007. Wetlands. 4th Edition. John Wiley & Sons Inc., Hoboken, NJ.

Moore, A. A., and M. A. Palmer. 2005. Invertebrate biodiversity in agricultural and rural headwater streams: implications for conservation and management. Ecological Applications 15(4):1169-1177.

Murkin, H. R. and J. A. Kadlec. 1986. Responses by benthic macroinvertebrates to prolonged flooding of marsh habitat. Canadian Journal of Zoology 64:65-72.

Murkin, H. R., R. M. Kamkinski, and R. D. Titman. 1982. Responses by dabbling ducks and aquatic invertebrates to an experimentally manipulated cattail marsh. Canadian Journal of Zoology 60:2324-2332.

Murkin, H. R. and L. C. M. Ross. 2000. Invertebrates in Prairie Wetlands. Pages 201-247 *in* H. R. Murkin , A. G. van der Valk, and W. R. Clark (editors). Prairie Wetland Ecology: The Contribution of the Marsh Ecology Research Program. Iowa State University Press, Ames, IA.

Nader, G., K. W. Tate, R. Atwill, and J. Bushnell. 1998. Water quality effect of rangeland beef cattle excrement. Rangelands 20(5):19-25.

Rader, R. B. 2001. An introduction to wetland bioassessment and management. Pages 339-354 *in* R. B. Rader, D. P. Batzer and S. A. Wissinger (editors). Bioassessment and Management of North American Freshwater Wetlands. John Wiley and Sons, Inc., New York, NY.

Rader, R. B. and C. J. Richardson. 1994. Response of macroinvertebrates and small fish to nutrient enrichment in the northern Everglades. Wetlands 14:134–146.

Rosenberg D. M., and V. H. Resh. 1993. Freshwater biomonitoring and benthic macroinvertebrates. Routledge, Chapman and Hall Inc., New York, NY.

Ryan, M. R., R. A. Pierce, K. M. Suedkamp-Wells, and C. K. Kerns. 2006. Assessing bird population responses to grazing. Pages 8-24 *in* W. Hohman (editor). Migratory bird responses to grazing. United States Department of Agriculture, Washington, DC. Natural Resources Conservation Service, Technical Report 190-54

Silver, C. A., and S. M. Vamosi. 2012. Macroinvertebrate community composition of temporary prairie wetlands: a preliminary test of the effect of rotational grazing. Wetlands 32(1):185-197.

Steinman, A. D., J. Conklin, P. J. Bohlen, and D. G. Uzarski. 2003. Influence of cattle grazing and pasture land use on macroinvertebrate communities in freshwater wetlands. Wetlands 23(4):877-889.

Stewart, R. E., and H. A. Kantrud. 1971. Classification of natural ponds and lakes in the glaciated prairie region. Bureau of Sport Fisheries and Wildlife, U.S. Fish and Wildlife Service, Washington, DC. Jamestown, ND: Northern Prairie Wildlife Research Center Resource Publication 92, Available from http://www.npwrc.usgs.gov/resource/wetlands/pondlake/index.htm [accessed April 1998].

Thomson, J. R., D. D. Hart, D. F. Charles, T. L. Nightengale, and D. M. Winter. 2005. Effects of removal of a small dam on downstream macroinvertebrate and algal assemblages in a Pennsylvania stream. Journal of North American Benthological Society 24:192-207.

Tisdale, E. W. 1947. The grasslands of the Southern Interior of British Columbia. Ecology 28:346-382.

Topping, M. S., and G. G. E. Scudder. 1977. Some physical and chemical features of saline lakes in central British Columbia. Syesis 10:145-149.

van der Valk, A. G. 2012. The biology of freshwater wetlands. 2nd edition. Oxford University Press Inc., New York, NY.

White, D. S. and R. E. Roughley. 2008. Aquatic Coleoptera. Pages 571-671 *in* Merritt, R. W., K. W. Cummins, and M. B. Berg (editors). An Introduction to the Aquatic Insects of North America. 4th edition. Kendall/Hunt Publishing Co., Dubuque, IA.

Wikeem B. and S. Wikeem. 2004. The Grasslands of British Columbia. Grassland Conservation Council of British Columbia. Kamloops, BC.

Zedler, J. B. and S. Kercher. 2005. Wetland resources: status, trends, ecosystem services and restorability. Annual Review of Environment and Resources 30:39-74.

CHAPTER 2. EFFECTS OF LIVESTOCK GRAZING ON AQUATIC MACROINVERTEBRATES IN SOUTHERN INTERIOR WETLANDS OF BRITISH COLUMBIA, CANADA

Introduction

Depressional wetlands are common in the grasslands of British Columbia's (BC) southern interior and are often used by free ranging livestock for forage and fresh water. Livestock spend a disproportional amount of time in wetlands versus upland areas grazing submergent, emergent and shoreline vegetation and drinking, loitering and cooling off (Adams and Fitch 1998; Nader et al. 1998). These livestock activities can decrease water quality with fecal and urine inputs, increase turbidity and decrease biodiversity by altering habitats and food resources for insects, amphibians, reptiles, waterfowl and other wildlife (Coffin and Lauenroth 1988; Collins et al. 1998; Steinman et al. 2003). Heavy livestock use may have negative consequences on the ecological condition and sustainability of these important ecosystems.

Aquatic invertebrates play an important role in the trophic dynamics of aquatic systems. In wetlands they function as primary consumers, acting as detritivores on litter and herbivores on algae, as well as secondary consumers preying on zooplankton, other aquatic invertebrates, and small vertebrates (Pip 1978; Caldwell et al. 1980; Travis et al. 1985; Nelson et al. 1990; Wen 1992). Many benthic taxa also contribute to mixing of sediment particles and nutrient flux through bioturbation (Brönmark and Hansson 2002). Many vertebrates are drawn to wetlands for the abundant invertebrate food resources there. Aquatic invertebrates account for a large proportion of waterfowl diets (Krapu 1974), particularly during the breeding season (Swanson et al. 1985), and are crucial for waterfowl brood survival (Cox et al. 1998). Aquatic invertebrate communities are important contributors to the productivity and functioning of wetland ecosystems and compositional and density changes in them can be indicative of wetland impairment.

The use of aquatic invertebrates as bioindicators of water and habitat quality in lotic systems has long been in practice (e.g., Hilsenhoff 1988; Rosenberg and Resh 1993; Barbour et al. 1999) with many studies examining cattle impacts (e.g., Moore and Palmer 2005; Carlisle et al. 2008). The application of this technique in lentic wetlands has been a more recent development with relatively few studies focusing on livestock grazing as the disturbance (e.g. Steinman et al. 2003; Ausden et al. 2005; Davis and Bidwell 2008). Canadian studies examining invertebrate response to livestock disturbance in wetlands are rare and have been conducted only in Alberta and Manitoba. Manitoba studies examined the effects of cattail mowing (Neckles et al. 1990) and hydrologic changes to wetland aquatic invertebrates (Murkin and Ross 1999; Murkin and Ross 2000; Wrubleski 2005). Alberta studies have either focused on the effects of timing of grazing or specific taxon responses to cattle grazing and the disturbance it causes. Effects of rotational wetland grazing were examined by Silver and Vamosi (2012), who found that early grazed wetlands had lower abundance and diversity of invertebrates, as well as different common taxa than late grazed pastures. Cattle grazing caused a significant decrease in odonate abundance and reproductive effort by reducing vegetation height both within and adjacent to wetlands and a reduction in odonate species richness and diversity with complete vegetation removal (Hornung and Rice 2003; Foote and Rice Hornung 2005). To my knowledge, no published studies have examined livestock effects on BC's southern interior depressional wetland invertebrate communities.

The primary objective of this research was to clarify the ecological links between livestock disturbance and aquatic macroinvertebrate abundance, biomass and community composition in grassland wetlands of the southern interior of BC. I predicted that 1) heavy livestock disturbance would decrease the richness and diversity of wetland aquatic macroinvertebrate communities and 2) zygopteran abundance and biomass would decrease with heavy levels of livestock grazing. The results of my study will contribute to the primary literature, provide crucial regionally specific data and promote the implementation of effective management of these wetland resources.

Methods

Study Area

Seventeen wetlands were examined in four grassland areas near Kamloops (50˚40'N, 120˚20'W) in the southern interior of British Columbia, Canada: Campbell Range, Rose Hill, Hamilton Commonage and Lac Du Bois (Figure 2.1). These southern interior wetlands are situated in the Thompson Very Dry Warm Bunchgrass Variant (BGxw2) biogeoclimatic zone (Meidinger and Pojar 1991), and have an average annual precipitation of 270 mm (MOFR 2007). Wetlands ranged in area (0.35 to 2.30 ha), perimeter length (0.22 to 0.86 km), elevation (764 to 1227 m) and pH (7.66 to 10.64) across study areas (Appendix A). Conductivity values were highly variable across sites (950 to 11820 µS/cm) due to the seasonality and semi-permanence of these waterbodies. Study wetlands were selected based on: 1) an absence of salt tolerant plants; 2) their inclusion in the Ducks Unlimited Canada annual waterfowl survey routes (Bruce Harrison pers. comm. 2007); 3) that the wetlands examined provided a full range of grazing intensities. To control and limit effects not due to livestock grazing, efforts were made to avoid high salinity wetlands as the invertebrate communities are known to be different than those in wetlands with lower salinity (e.g., Cannings and Scudder 1978; Cannings et al. 1980). My highest conductivity value was below the 15000 - 45000 µS/cm range at which wetlands are considered subsaline or saline in the Alberta Wetland Classification System (AESRD 2014). High disturbance wetlands coincided with the high elevation wetlands. To account for elevational differences in emergency timing (i.e., temperature difference), sampling sessions began with low elevation sites and concluded with those wetlands found at higher elevations.

Two of the study areas, Campbell Range and Rose Hill, are privately owned while the other two areas, Hamilton Commonage and Lac du Bois (2 sub-areas: Lac du Bois Bachelor and Lac du Bois Long Lake), are grazed under a Crown land lease agreement. Cattle are rotated from pasture to pasture in all study areas depending on the season, range

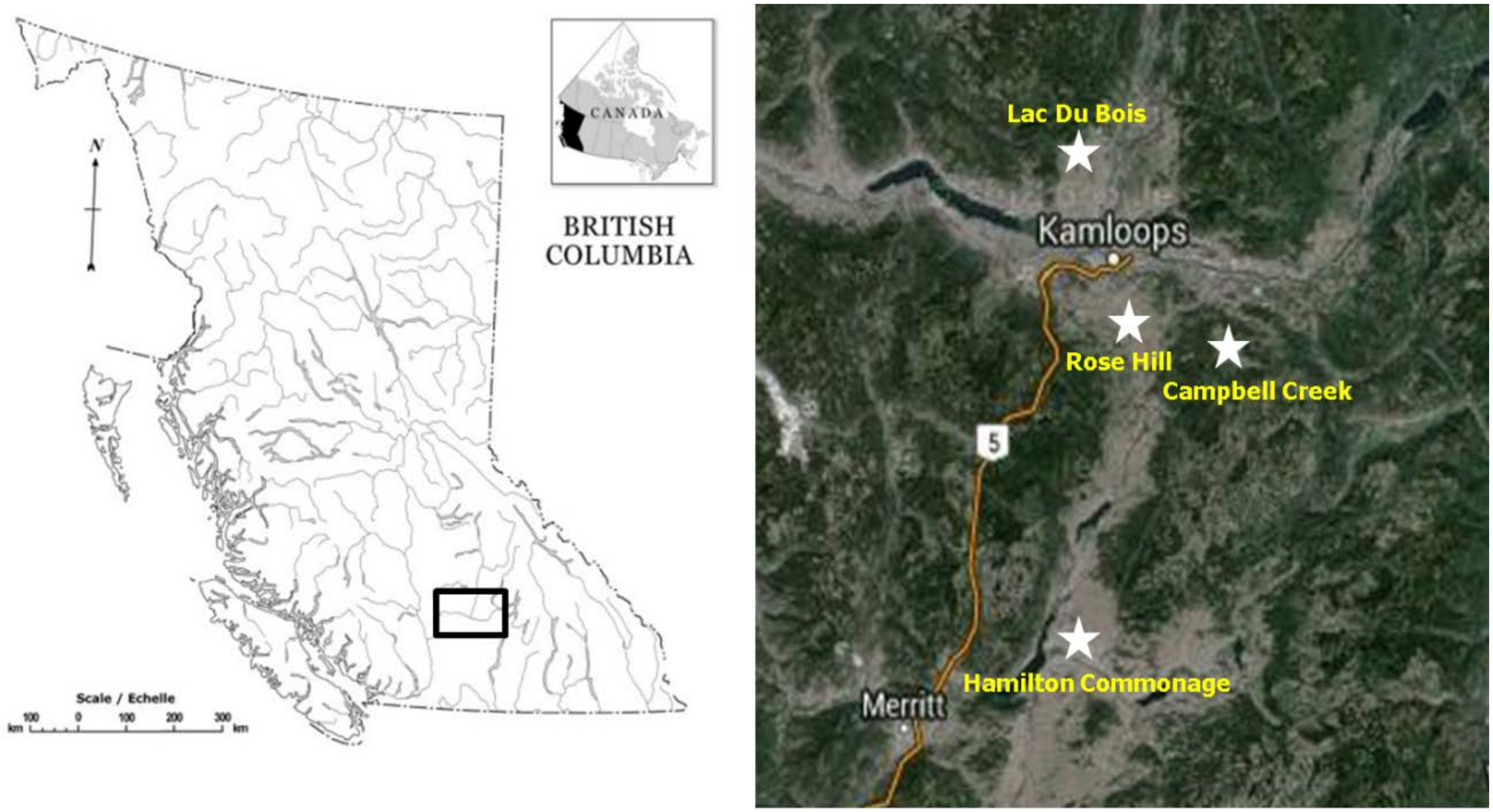


Figure 2.1. Location of the four study areas (stars) near Kamloops, BC. (Map sources: Natural Resource Canada 2004; Google Maps 2014).

conditions and land parcel size. Ranches in the southern interior are mostly cow-calf operations; however, some yearling cattle and horse operations exist in some areas. Cattle are generally Herefords although other breeds occur in smaller numbers.

Estimates of Livestock Disturbance

A concurrent study examining impacts of livestock on wetland vegetation provided a grazing intensity or livestock disturbance gradient. The mean number of quadrat corners that intersected bare ground was measured at each wetland site and used as a surrogate for grazing intensity (Jones et al. 2011). Bare ground measurements were negatively correlated with vegetation biomass and positively correlated with soil bulk density and were an efficient way to quantify livestock disturbance. Photographs of wetlands with low and high disturbance levels are shown in Appendix B.

Aquatic Invertebrate Sampling and Processing

Aquatic invertebrates were sampled in 2008 over a two-week period in both early and mid-summer to correspond with waterfowl nesting and brood rearing periods. All wetlands sampled were class 4 (Stewart and Kantrud 1971) based on the duration of flooding; this allowed the examination of wetlands with similar attributes and varying grazing pressures. Seventeen wetlands were sampled in spring (May/June) and 12 in the summer (July). Five dried up before the second sampling session in July. Conductivity and pH were recorded concurrently with invertebrate sampling at three locations on each wetland using an YSI multi-probe (YSI Inc., Yellow Springs, Ohio).

Wetland invertebrate sampling sites were chosen by measuring perimeter of each wetland on a provincial online map provider (BCGov 2008) and then selecting random sampling points along the perimeter using a random number generator. Sample locations were plotted on the online map and GPS coordinates were obtained for each site. Three sweep and core samples were collected from each wetland during each sampling session. Sweep net samples of the nektonic community (including mobile epi-benthic and epi-phytic organisms) were collected 2 metres from the wetted edge of the wetland using a 500 μm sweep net lowered to just above the substrate surface and rapidly pulled vertically to the water surface (Rader 2001; Merritt et al. 2008b). Water depth was measured at sweep sites and used in conjunction with the net area to determine the volume of water sampled; the number of organisms per cubic metre was then calculated. Core samples (5.1 cm diameter by 10.2 cm deep) were collected 2 metres from the wetland edge using a benthic hand corer (Swanson 1983; Rader 2001). Samples were placed in Whirlpak® bags filled with 70% ethanol for later processing. Invertebrates were sorted using a 3X power magnifying lamp. Microinvertebrates (e.g., cladocerans) were ignored in samples as the primary focus of this study was on the macroinvertebrate community. Aquatic invertebrates were identified using keys and descriptions from Merritt et al. (2008a) and Thorp and Covich (2001). Taxa collected and their common names are listed in Appendix C. Aquatic invertebrates were identified to the family level for insects (Orders Ephemeroptera, Trichoptera, Diptera, Coleoptera, Hemiptera, and Odonata), molluscs (Classes Gastropoda and Bivalvia) and macro-crustaceans (Order Amphipoda), while other aquatic groups were only identified to order or higher taxonomic levels. The large number of immature insect specimens prevented identification to genus or species. Identification keys usually require mature larvae for identification beyond the family level.

Biomass was determined using taxon body length and length-mass regressions (Appendix D) from the literature where available, or specimens were dried and weighed. Dry mass was determined using $M = a\ L^b$, where M = mass, L = body length and a and b are constants (Smock 1980; Benke et al. 1999; Johnston and Cunjak 1999). When chironomid larval densities exceeded 100 individuals they were volumetrically subsampled in 500 ml of water using a Folsom plankton splitter (McEwen et al. 1954; Glozier et al. 2002). The subsampled chironomids were measured and multiplied by the proportion of the sample examined to obtain total chironomid biomass. Those taxa to be dried and weighed were separated into one of three categories: Gastropoda, Aquatic Others and Terrestrial Others. The Aquatic Others category consisted of aquatic Hydrachnidiae, Oligochaeta, Nematoda, Hirudinea, Bivalvia, Ostracoda, Collembola and pupae of various insect taxa. Terrestrial Others included Lepidoptera, Homoptera, terrestrial and semi-terrestrial Hemiptera, Araneae, terrestrial Coleoptera, Hymenoptera, Thysanoptera, and adult Diptera, Ephemeroptera and Trichoptera. Taxa grouped into these three categories were transferred to pre-weighed aluminum pans and placed in a drying oven at 60°C for 24 h (Johnston and Cunjak 1999). The pans were removed from the oven and placed in a desiccator to cool (minimum of 1 hour) and then weighed to a constant mass (± 0.0001 g) on a microbalance. Dry weights were conservative estimates as the ethanol preservative results in some loss of body mass (Howmiller 1972; Johnston and Cunjak 1999).

Richness and diversity indices were calculated at the family level for insects, molluscs, and macro-crustaceans and at higher taxonomic levels for all other aquatic invertebrate groups. Therefore, calculations were conservative and should be considered an underestimate of actual richness and diversity in the study wetlands. Shannon’s diversity index was used to determine the proportional diversity of wetland taxa (Shannon and Weaver 1949), and Simpson’s index was used as a measurement of equitability or evenness (Simpson 1949).

Statistical Analysis

All statistical analyses were conducted using the R statistical software program (R Development Core Team 2011). Only dominant and widespread taxa were used in analyses. Taxa that had >5% relative abundance/biomass (Steinman et al. 2003; Corcoran et al. 2009) in at least one wetland site and occurred in >50% of wetlands were used in analyses (Batzer et al. 2004). Dominant taxa were determined for abundance and biomass separately; they were also selected separately for the two different sampling methods (sweeps and cores). The inclusion of uncommon taxa in these analyses proved statistically problematic because of the presence of many zero values.

Nonmetric multidimensional scaling (NMDS), an indirect ordination technique, was used to summarize associations among wetland invertebrate community composition and environmental variables. NMDS analysis groups similar sites based on dissimilarities in community composition and is robust enough to handle numerous zero values and nonnormal multivariate data (Clarke 1993). Ordinations were performed using a Bray-Curtis dissimilarity matrix using the metaMDS function in the vegan community ecology package for the R statistical software (Oksanen et al. 2011; R Development Core Team 2011). Analyses used random starting configurations and the number of dimensions used was determined by stress reduction using a scree plot (McCune and Grace 2002). Significant environmental variables were plotted on the ordinations as vectors so that their relationship with the invertebrate communities could be readily visualized.

Stepwise linear regressions were explored in the examination of the relationship between the response variables (total abundance, total biomass, richness, diversity, and abundance and biomass of the most common taxa) and the explanatory variables (disturbance, wetland perimeter length, pH and conductivity). Data were transformed where required to meet the test assumptions of normal errors and homoscedasticity.

Results

Aquatic Invertebrate Community Structure

The study documented 37 higher taxa of aquatic macroinvertebrates from nine classes, eleven orders and thirty-one families (Appendix C). This estimate is conservative as identification of insects was to order or family level and non-insect groups to order, class or phylum. Both spring sweep abundance and biomass had ten dominant taxa (>50% occurrence in all wetlands and >5% relative abundance in at least one wetland) (Table 2.1 and Table 2.2). Summer core abundance taxa contained fewer widespread taxa with only Oligochaeta, Ceratopogonidae and Chironomidae meeting the above criteria. Across all wetlands, sweep abundance was primarily dominated by dipterans and ostracods in the spring and dipterans in the summer (Appendix E, Figure E.1). Core abundance followed a similar pattern, with spring samples consisting primarily of dipterans and ostracods while summer core abundance had a large dipteran presence (Appendix E, Figure E.2). Spring sweep biomass had high proportions of zygopterans, whereas zygopterans, dipterans and gastropods were most prominent in summer sweeps (Appendix E, Figure E.3). Both spring and summer core biomass was primarily composed of dipterans; however, when present, gastropods, trichopterans and zygopterans greatly contributed to the total biomass (Appendix E, Figure E.4).

Abundance, biomass, richness and diversity were inconsistent among wetlands and seasons and variance amongst samples was often high (Tables 2.3-2.6). No consistent pattern of abundance, biomass, richness or diversity was found with livestock disturbance at a wetland site level. Summer sweeps had the greatest overall mean abundance (7.98 ± 5.09 organisms/m^3) and biomass (3.13 ± 1.66 mg/m^3) densities whereas spring sweeps had the lowest mean abundance (5.61 ± 2.75 organisms/m^3) and spring cores the lowest mean biomass (1.40 ± 0.80 mg/m^3) densities. Mean family richness (9.35 ± 1.59) and diversity (Shannon's 1.27 ± 0.23; Simpson's 0.58 ± 0.09) was greatest in spring sweep samples. Summer cores had the lowest family richness (3.56 ± 0.84) and diversity (Shannon's 0.68 ± 0.17; Simpson's 0.37 ± 0.09).

Table 2.1. Dominant taxa for sweep and core abundance during spring and summer sampling sessions. Taxa were selected based on their presence in >50% of all wetlands and having >5% relative abundance in at least one wetland site.

Sweep Abundance		**Core Abundance**	
Spring	**Summer**	**Spring**	**Summer**
Ostracoda	Ostracoda	Ostracoda	Oligochaeta
Oligochaeta	Oligochaeta	Oligochaeta	Ceratopogonidae
Gastropoda	Ceratopogonidae	Nematoda	Chironomidae
Collembola	Chironomidae	Ceratopogonidae	
Ceratopogonidae	Dytiscidae	Chironomidae	
Chironomidae	Lestidae	Lestidae	
Dytiscidae	Lymnaeidae		
Aeshnidae			
Lestidae			
Corixidae			

Table 2.2. Dominant taxa for sweep and core biomass during spring and summer sampling sessions. Taxa were selected based on their presence in >50% of all wetlands and having >5% relative biomass in at least one wetland site.

Sweep Biomass		**Core Biomass**	
Spring	**Summer**	**Spring**	**Summer**
Gastropoda	Hemiptera	Coleoptera	Coleoptera
Ceratopogonidae	Gastropoda	Ceratopogonidae	Ceratopogonidae
Chironomidae	Ceratopogonidae	Chironomidae	Chironomidae
Dytiscidae	Chironomidae	Lestidae	Ephydridae
Hydrophilidae	Dytiscidae		
Libellulidae	Aeshnidae		
Aeshnidae	Coenagrionidae		
Coenagrionidae	Lestidae		
Lestidae			
Limnephilidae			

Table 2.3. Summary of mean spring sweep abundance, biomass and diversity measures. Column heading abbreviations are defined as Disturb=livestock disturbance gradient (low values are least disturbed), Abu=abundance (organisms/m^3), Bio=biomass (mg/m^3), S=family richness, H'=Shannon's Diversity Index, and D=Simpson's Diversity Index. Values in parenthesis represent ± 1 S.E..

Study Area	Site	Disturb	Abu	Bio	S	H'	D
Campbell Range	7	1.62	3.79 (1.44)	0.69 (0.17)	13.33 (2.33)	1.50 (0.46)	0.60 (0.18)
	8	0.98	9.65 (3.69)	0.86 (0.18)	12.67 (0.88)	1.24 (0.26)	0.51 (0.11)
	Mean	1.30	6.72 (2.57)	0.78 (0.17)	13.00 (1.61)	1.37 (0.36)	0.55 (0.14)
Hamilton Commonage	4.3	0.14	6.25 (2.81)	9.17 (5.60)	11.67 (1.86)	1.69 (0.19)	0.73 (0.05)
	5	1.21	0.71 (0.44)	0.65 (0.63)	5.33 (2.60)	0.95 (0.50)	0.45 (0.23)
	7.1	1.88	0.83 (0.16)	0.17 (0.05)	5.33 (0.88)	1.05 (0.03)	0.58 (0.01)
	9	2.22	2.36 (0.85)	0.23 (0.12)	6.00 (1.53)	0.72 (0.27)	0.39 (0.16)
	Mean	1.36	2.54 (1.07)	2.56 (1.60)	7.08 (1.72)	1.10 (0.25)	0.54 (0.11)
Lac Du Bois Batchelor	4	0.25	4.05 (1.62)	1.36 (0.57)	13.00 (1.00)	1.38 (0.13)	0.61 (0.04)
	4.1	0.46	6.27 (4.32)	2.38 (0.22)	9.00 (1.53)	1.11 (0.13)	0.55 (0.07)
	5	0.27	19.98 (12.25)	9.69 (4.10)	11.33 (1.20)	1.23 (0.15)	0.61 (0.08)
	6.1	0.33	4.58 (2.65)	1.49 (1.13)	9.67 (1.20)	1.40 (0.22)	0.64 (0.09)
	10	0.09	3.29 (1.76)	1.69 (0.86)	11.00 (1.00)	1.90 (0.12)	0.79 (0.04)
	Mean	0.28	7.63 (4.52)	3.32 (1.37)	10.80 (1.19)	1.40 (0.15)	0.64 (0.07)
Lac Du Bois Long Lake	4.1	0.23	9.53 (4.05)	2.71 (1.16)	13.00 (3.51)	1.80 (0.04)	0.80 (0.01)
	6.1	1.32	3.84 (1.70)	0.80 (0.25)	9.00 (2.08)	1.20 (0.17)	0.56 (0.07)
	7	0.06	14.20 (6.89)	1.35 (0.33)	10.33 (1.20)	1.51 (0.21)	0.71 (0.06)
	Mean	0.54	9.19 (4.21)	1.62 (0.58)	10.78 (2.27)	1.50 (0.14)	0.69 (0.04)
Rose Hill	13.1	2.84	0.53 (0.32)	0.20 (0.13)	3.33 (0.88)	0.94 (0.13)	0.57 (0.03)
	14	1.50	4.43 (1.30)	0.71 (0.29)	9.67 (2.19)	0.72 (0.36)	0.30 (0.18)
	19	1.05	1.17 (0.50)	1.45 (1.06)	5.33 (1.20)	1.18 (0.45)	0.55 (0.20)
	Mean	1.79	2.05 (0.70)	0.79 (0.49)	6.11 (1.42)	0.94 (0.31)	0.47 (0.14)
Mean Across All Sites		0.97	5.61 (2.75)	2.09 (0.99)	9.35 (1.59)	1.27 (0.23)	0.58 (0.09)

Table 2.4. Summary of mean summer sweep abundance, biomass and diversity measures. Column heading abbreviations are defined as Disturb=livestock disturbance gradient (low values are least disturbed), Abu=abundance (organisms/m^3), Bio=biomass (mg/m^3), S=family richness, H`=Shannon's Diversity Index, and D=Simpson`s Diversity Index. Values in parentheses represent ± 1 S.E..

Study Area	Site	Disturb	Abu	Bio	S	H'	D
Hamilton Commonage	9	2.22	2.89 (1.95)	1.78 (1.10)	9.33 (2.33)	1.41 (0.16)	0.65 (0.08)
Lac Du Bois Batchelor	4	0.25	2.79 (0.77)	3.44 (1.49)	6.67 (2.19)	0.72 (0.23)	0.34 (0.12)
	4.1	0.46	22.27 (11.64)	8.80 (2.63)	8.33 (2.03)	0.63 (0.28)	0.29 (0.15)
	5	0.27	20.94 (18.35)	5.53 (4.72)	8.33 (2.40)	0.52 (0.03)	0.23 (0.03)
	6.1	0.33	1.20 (0.27)	1.06 (0.66)	7.33 (1.76)	1.25 (0.08)	0.61 (0.01)
	10	0.09	2.80 (1.36)	2.51 (1.03)	6.33 (0.67)	1.04 (0.18)	0.52 (0.11)
	mean	0.28	10.00 (6.48)	4.27 (2.11)	7.40 (1.81)	0.83 (0.16)	0.40 (0.08)
Lac Du Bois Long Lake	4.1	0.23	6.78 (1.89)	1.93 (0.86)	12.33 (0.67)	1.58 (0.13)	0.68 (0.05)
	6.1	1.32	2.02 (0.65)	2.09 (0.90)	10.33 (1.45)	1.38 (0.28)	0.61 (0.11)
	7	0.06	4.03 (1.46)	0.94 (0.17)	7.67 (1.20)	1.22 (0.27)	0.58 (0.12)
	mean	0.54	4.28 (1.33)	1.66 (0.64)	10.11 (1.11)	1.39 (0.23)	0.62 (0.09)
Rose Hill	13.1	2.84	26.28 (21.91)	8.12 (5.78)	7.67 (2.85)	0.83 (0.36)	0.38 (0.15)
	14	1.50	1.59 (0.16)	0.54 (0.24)	10.00 (1.53)	1.42 (0.17)	0.65 (0.06)
	19	1.05	2.20 (0.63)	0.85 (0.38)	8.33 (1.86)	1.13 (0.06)	0.55 (0.05)
	mean	1.79	10.02 (7.57)	3.17 (2.13)	8.67 (2.08)	1.12 (0.20)	0.53 (0.08)
Mean Across All Sites		0.89	7.98 (5.09)	3.13 (1.66)	8.56 (1.74)	1.09 (0.19)	0.51 (0.09)

Table 2.5. Summary of mean spring core abundance, biomass and diversity measures. Column heading abbreviations are defined as Disturb=livestock disturbance gradient (low values are least disturbed), Abu=abundance (organisms/m^3), Bio=biomass (mg/m^3), S=family richness, H`=Shannon's Diversity Index, D=Simpson`s Diversity Index. Values in parentheses represent ± 1 S.E..

Study Area	**Site**	**Disturb**	**Abu**	**Bio**	**S**	**H'**	**D**
Campbell Range	7	1.62	4.39 (1.46)	3.52 (3.22)	4.67 (0.33)	1.04 (0.03)	0.53 (0.04)
	8	0.98	8.90 (4.00)	0.52 (0.29)	6.00 (0.00)	1.48 (0.02)	0.74 (0.01)
	Mean	1.30	6.64 (2.73)	2.02 (1.76)	5.33 (0.17)	1.26 (0.03)	0.64 (0.03)
Hamilton Commonage	4.3	0.14	4.59 (2.13)	0.38 (0.13)	5.00 (2.31)	0.91 (0.49)	0.46 (0.24)
	5	1.21	3.69 (1.28)	1.19 (0.90)	5.00 (0.58)	1.37 (0.08)	0.72 (0.02)
	7.1	1.88	4.55 (1.05)	0.20 (0.05)	5.33 (0.88)	1.21 (0.11)	0.63 (0.04)
	9	2.22	5.33 (2.70)	4.52 (4.25)	5.00 (1.15)	1.31 (0.14)	0.68 (0.03)
	Mean	1.36	4.54 (1.79)	1.57 (1.33)	5.08 (1.23)	1.20 (0.21)	0.62 (0.08)
Lac Du Bois Batchelor	4	0.25	1.60 (0.43)	0.11 (0.03)	4.33 (0.33)	1.21 (0.04)	0.64 (0.04)
	4.1	0.46	8.61 (4.73)	0.23 (0.16)	3.33 (0.88)	0.57 (0.34)	0.29 (0.19)
	5	0.27	10.70 (0.62)	1.72 (0.56)	7.67 (1.20)	1.17 (0.02)	0.57 (0.04)
	6.1	0.33	6.11 (0.42)	0.30 (0.12)	4.33 (0.88)	1.00 (0.19)	0.54 (0.10)
	10	0.09	8.32 (5.43)	0.93 (0.35)	6.00 (0.58)	1.30 (0.31)	0.61 (0.15)
	Mean	0.54	7.07 (2.33)	0.66 (0.25)	5.13 (0.78)	1.05 (0.18)	0.53 (0.10)
Lac Du Bois Long Lake	4.1	0.23	19.39 (8.53)	1.93 (0.93)	6.33 (1.20)	0.99 (0.31)	0.47 (0.16)
	6.1	1.32	7.17 (1.21)	0.44 (0.22)	4.33 (0.88)	0.92 (0.11)	0.52 (0.07)
	7	0.06	5.33 (2.02)	0.56 (0.22)	4.67 (0.33)	0.88 (0.23)	0.45 (0.14)
	Mean	1.79	10.63 (3.92)	0.98 (0.46)	5.11 (0.81)	0.93 (0.22)	0.48 (0.12)
Rose Hill	13.1	2.84	5.53 (0.93)	5.44 (0.86)	3.33 (0.33)	0.82 (0.09)	0.48 (0.08)
	14	1.50	2.66 (0.29)	0.57 (0.34)	3.33 (0.33)	0.76 (0.19)	0.42 (0.12)
	19	1.05	7.05 (0.64)	1.24 (0.94)	4.67 (0.88)	0.95 (0.09)	0.53 (0.00)
	Mean	1.79	5.08 (0.62)	2.42 (0.72)	3.78 (0.52)	0.85 (0.13)	0.48 (0.07)
Mean Across All Sites		0.97	6.70 (2.23)	1.40 (0.80)	4.90 (0.77)	1.05 (0.17)	0.55 (0.09)

Table 2.6. Summary of mean summer core abundance, biomass and diversity measures. Column heading abbreviations are defined as Disturb=livestock disturbance gradient (low values are least disturbed), Abu=abundance (organisms/m^3), Bio=biomass (mg/m^3), S=family richness, H`=Shannon's Diversity Index, D=Simpson`s Diversity Index. Values in parentheses represent ± 1 S.E..

Study Area	Site	Disturb	Abu	Bio	S	H'	D
Hamilton Commonage	9	2.22	3.61 (0.98)	8.70 (1.80)	5.00 (1.00)	1.13 (0.24)	0.56 (0.10)
Lac Du Bois Batchelor	4	0.25	2.99 (1.05)	1.07 (0.89)	2.33 (0.33)	0.69 (0.05)	0.47 (0.02)
	4.1	0.46	7.83 (2.24)	2.38 (1.00)	2.33 (0.88)	0.37 (0.21)	0.21 (0.11)
	5	0.27	4.39 (1.17)	1.94 (0.50)	3.00 (0.58)	0.69 (0.11)	0.39 (0.04)
	6.1	0.33	14.84 (6.00)	5.58 (3.86)	3.00 (0.58)	0.18 (0.04)	0.07 (0.02)
	10	0.09	5.41 (3.20)	0.62 (0.23)	3.33 (0.33)	0.90 (0.06)	0.54 (0.04)
	mean	0.28	7.09 (2.73)	2.32 (1.30)	2.80 (0.54)	0.57 (0.10)	0.33 (0.05)
Lac Du Bois Long Lake	4.1	0.23	13.57 (6.75)	2.81 (1.72)	6.00 (2.00)	0.86 (0.19)	0.43 (0.10)
	6.1	1.32	2.75 (0.95)	1.20 (0.56)	3.33 (1.20)	0.68 (0.35)	0.36 (0.18)
	7	0.06	5.74 (1.61)	2.24 (1.00)	3.67 (0.88)	0.67 (0.25)	0.35 (0.13)
	mean	0.54	7.35 (3.10)	2.08 (1.10)	4.33 (1.36)	0.74 (0.26)	0.38 (0.14)
Rose Hill	13.1	2.84	7.34 (2.17)	2.25 (0.92)	4.00 (1.00)	0.61 (0.09)	0.34 (0.06)
	14	1.50	1.43 (0.23)	0.20 (0.10)	3.00 (0.58)	0.92 (0.26)	0.53 (0.14)
	19	1.05	9.55 (3.46)	2.45 (1.26)	3.67 (0.67)	0.40 (0.20)	0.20 (0.12)
	mean	1.79	6.11 (1.95)	1.63 (0.76)	3.56 (0.75)	0.64 (0.18)	0.36 (0.11)
Mean Across All Sites		0.89	6.62 (2.48)	2.62 (1.15)	3.56 (0.84)	0.68 (0.17)	0.37 (0.09)

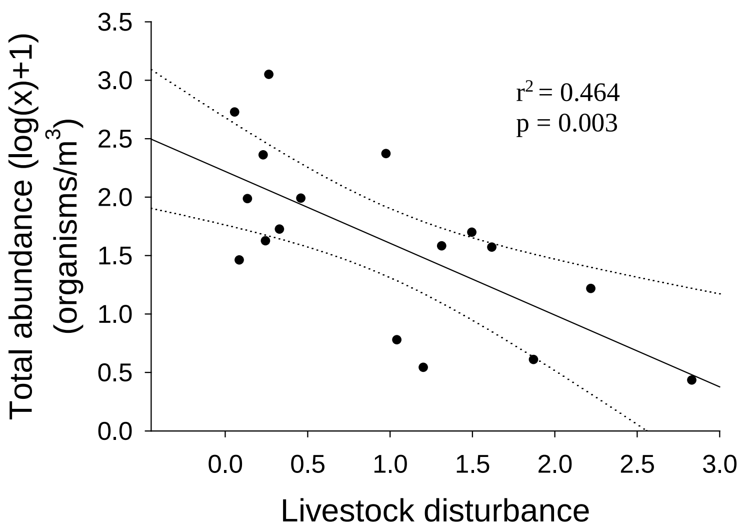
Community Response to Disturbance

The main impacts of livestock disturbance on wetland macroinvertebrate communities occurred primarily in the spring and affected those taxa found within the nektonic community. Livestock disturbance was significantly negatively correlated with spring sweep total abundance and total biomass (Figure 2.2). Significant negative associations were also found between livestock disturbance and spring sweep richness, Shannon's diversity and Simpson's diversity (Figure 2.3). The only significant benthic community response to livestock disturbance also included conductivity; both were negatively correlated with spring core richness (Table 2.9).

In a few instances, regression models including both disturbance and conductivity were significantly correlated with wetland taxa (Tables 2.7-2.9). As most of these taxa also had significant relationships with disturbance alone, the models that included both environmental parameters were not confounded by the effects of conductivity but rather improved the model by explaining more of the variation.

NMDS ordinations demonstrated distinct groupings of higher disturbance ponds versus lower disturbance ponds in cases where livestock disturbance was significantly correlated to the community composition. See Appendix F for NMDS ordination plots that include all environmental variables and Appendix G for NMDS correlation values for the environmental variables. The NMDS plot of spring sweep abundance (2 dimensional solution, stress of 10.3%) included disturbance (r^2=0.486, p=0.008) as an environmental feature related to abundance of macroinvertebrates (Figure 2.4). The influence of livestock disturbance was evident as indicated by site separation within the plot. However, the effect of conductivity on the community composition, although not significant (r^2=0.322, p=0.063), was apparent as disturbed site groupings were not always grouped according to disturbance alone. The ordination of summer sweep abundance (2 dimensional solution, stress of 13.3%; Figure 2.5) and summer core abundance communities (2 dimensional solution, stress of 1.7%; Figure 2.6) showed a clear site separation based on livestock disturbance (r^2=0.714, p=0.006; r^2=0.534, p=0.035 respectively).

a)



b)

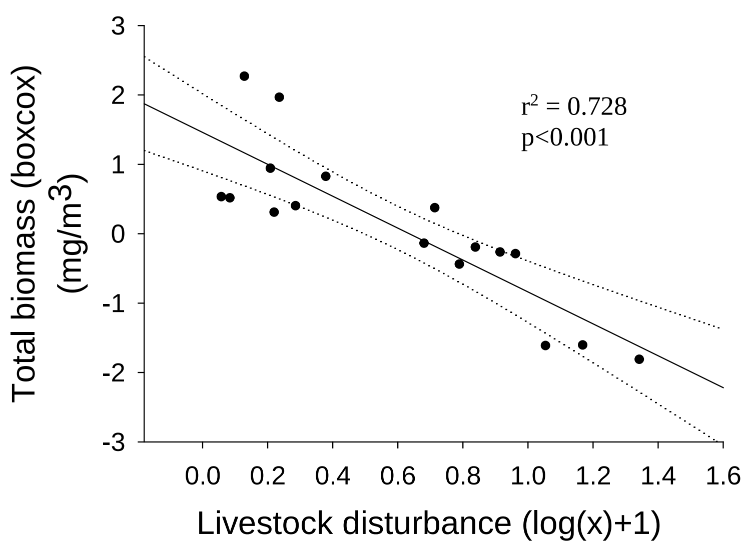


Figure 2.2. Linear relationship between livestock disturbance and a) spring sweep total abundance and b) spring sweep total biomass. Livestock disturbance represents the mean # of quadrat corners that intersected bare ground at each wetland (Jones et al. 2011). Dotted lines represent 95% confidence intervals.

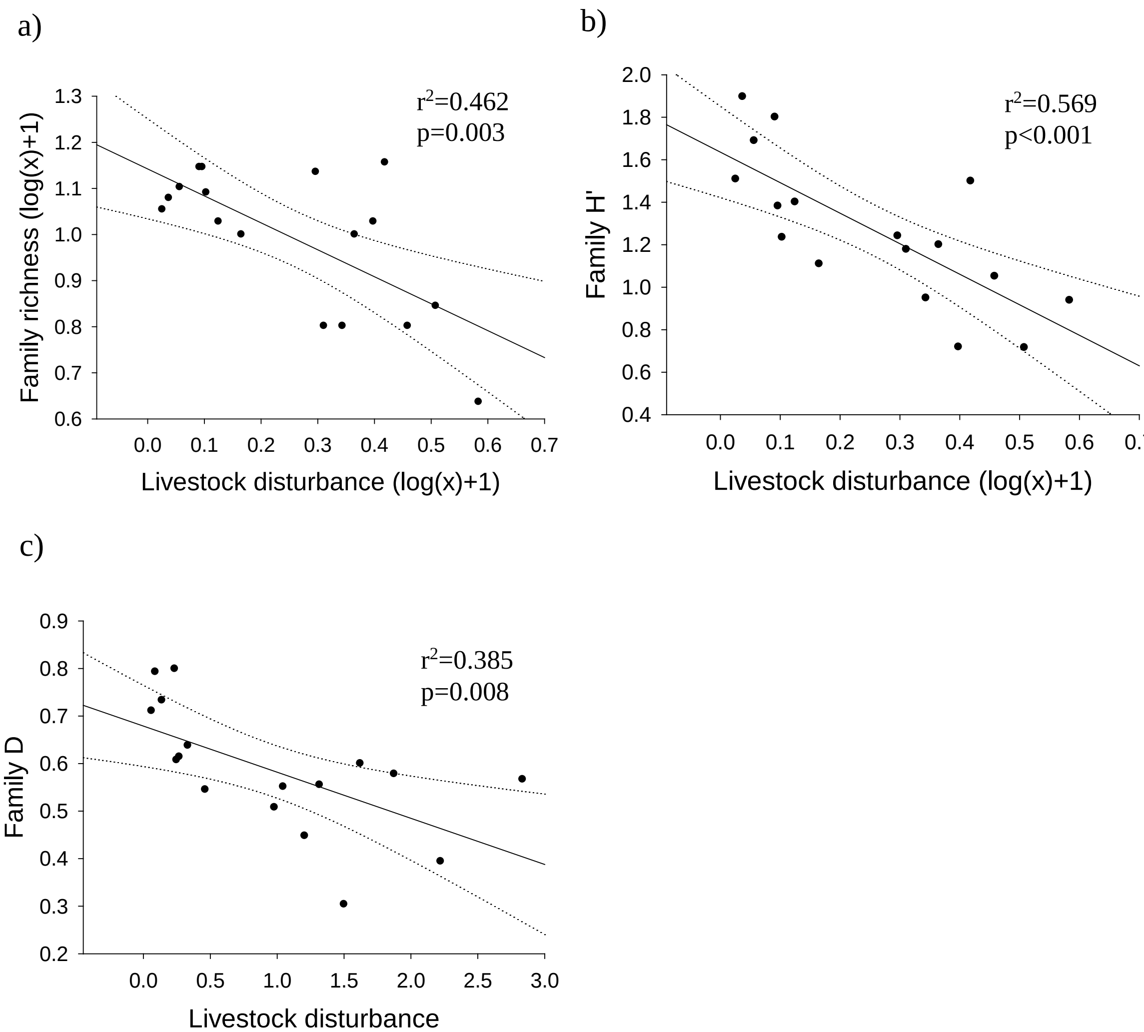


Figure 2.3. Linear relationship between livestock disturbance and spring sweep a) family richness, b) Shannon's family-level diversity (H') and c) Simpson's family-level diversity (D). Livestock disturbance represents the mean # of quadrat corners that intersected bare ground at each wetland (Jones et al. 2011). Dotted lines represent 95% confidence intervals.

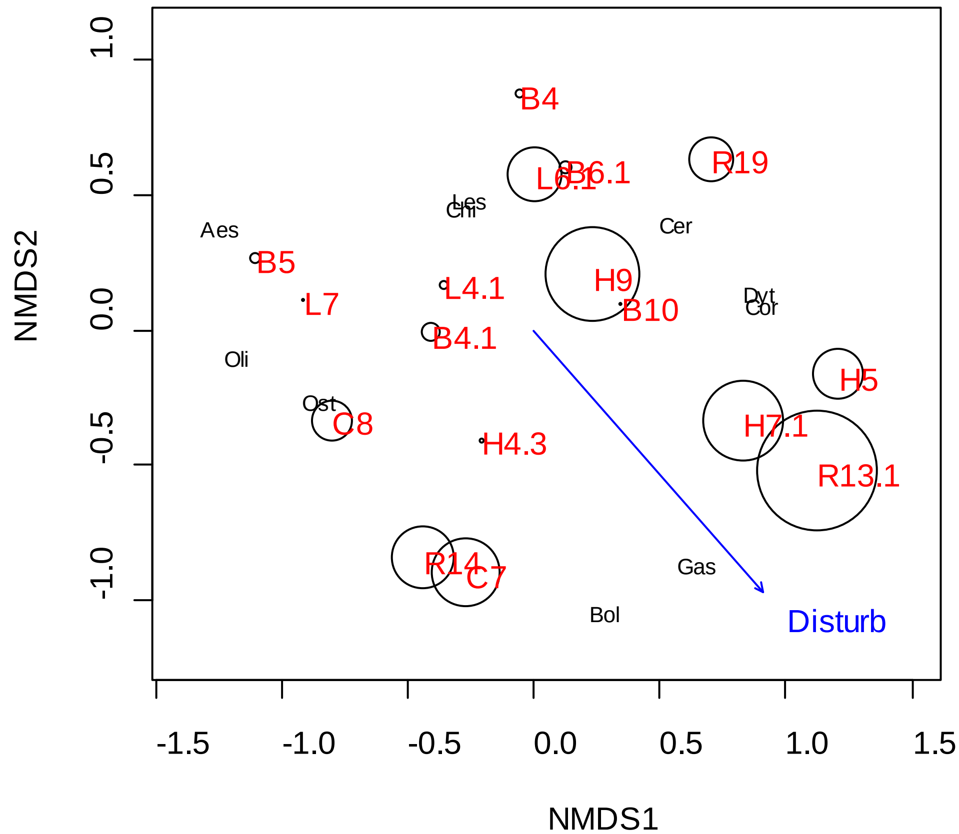


Figure 2.4. Nonmetric multidimensional scaling (NMDS) ordination (stress=10.3%) of spring sweep abundance dominant taxa community structure with overlay of fitted vector representing the significant (r^2=0.486, p=0.008) environmental variable livestock disturbance (Disturb). Wetland site (in red) disturbance level is represented by a continuum with large circles being most disturbed and dots representing least disturbed sites. Taxa (in black) are as follows: Aes=Aeshnidae, Bol=Collembola, Cer=Ceratopogonidae, Chi=Chironomidae, Cor=Corixidae, Dyt=Dytiscidae, Gas=Gastropoda, Les=Lestidae, Oli=Oligochaeta, and Ost=Ostracoda.

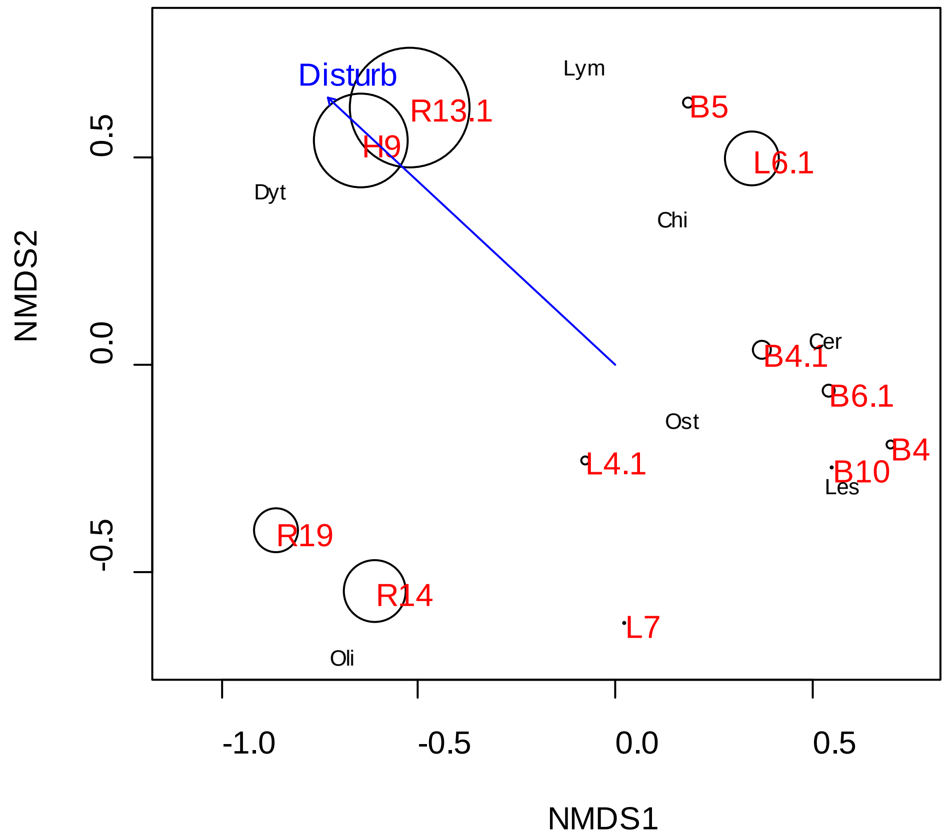


Figure 2.5. Nonmetric multidimensional scaling (NMDS) ordination (stress=13.3%) of summer sweep abundance dominant taxa community structure with overlay of fitted vector representing the significant (r^2=0.714, p=0.006) environmental variable livestock disturbance (Disturb). Wetland site (in red) disturbance level is represented by a continuum with large circles being most disturbed and dots representing least disturbed sites. Taxa (in black) are as follows: Cer=Ceratopogonidae, Chi=Chironomidae, Dyt=Dytiscidae, Les=Lestidae, Lym=Lymnaeidae, Oli=Oligochaeta, and Ost=Ostracoda.

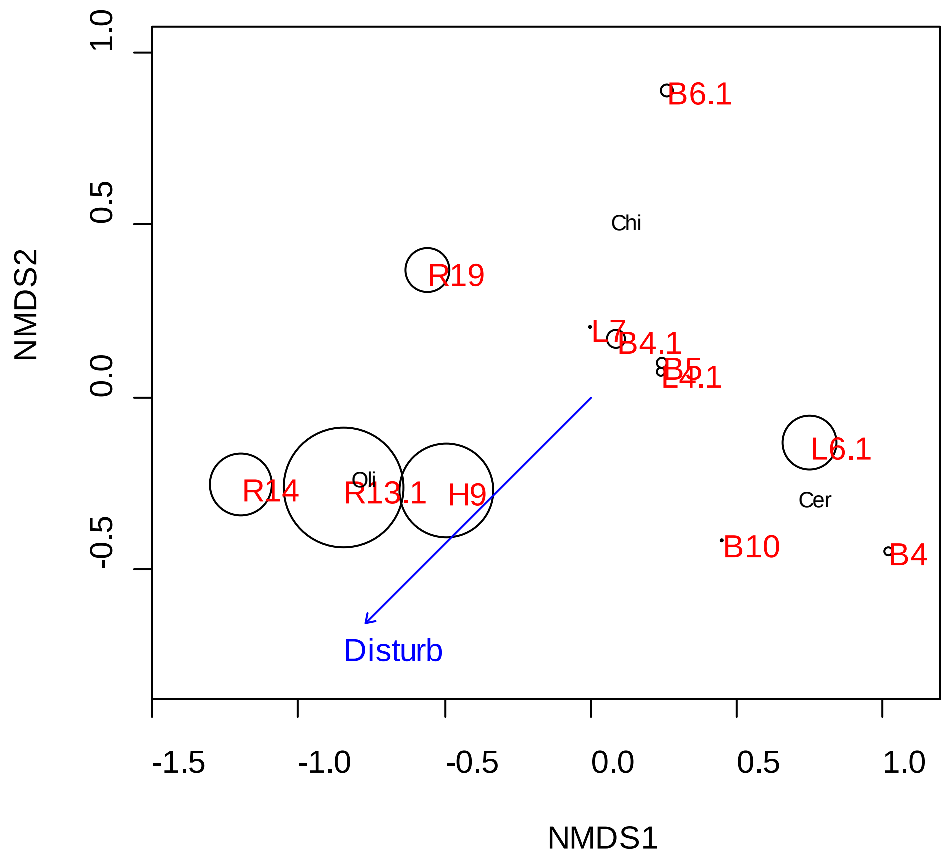


Figure 2.6. Nonmetric multidimensional scaling (NMDS) ordination (stress=1.74%) of summer core abundance dominant taxa community structure with overlay of fitted vector representing the significant (r^2=0.534, p=0.034) environmental variable livestock disturbance (Disturb). Wetland site (in red) disturbance level is represented by a continuum with large circles being most disturbed and dots representing least disturbed sites. Taxa (in black) are as follows: Cer=Ceratopogonidae, Chi=Chironomidae, Oli=Oligochaeta.

Individual Taxa Response to Livestock Disturbance

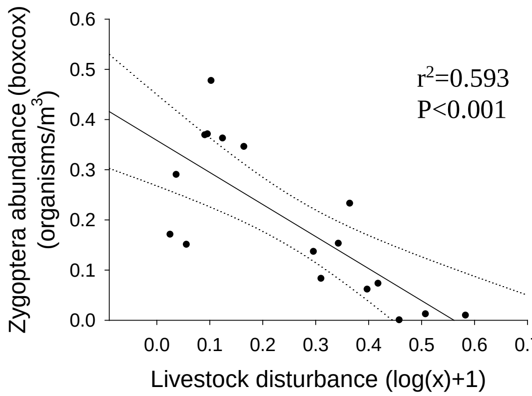
Livestock disturbance was an important factor influencing Odonata at the suborder and family levels. Zygoptera (damselfly) spring sweep abundance and biomass were both significantly negatively correlated with livestock disturbance (Figure 2.7). One of the two zygopteran families found in spring sweeps, the Lestidae, followed a similar trend with a significant decrease in abundance and biomass as disturbance levels increased (Figure 2.7). Both spring sweep Lestidae abundance and biomass had significant negative associations with a combination of livestock disturbance and perimeter length (Table 2.7-2.8). Both Zygoptera and Lestidae spring core abundance were negatively correlated with livestock disturbance (Table 2.9). In spring sweeps, Aeshnidae (a dragonfly family) abundance was the only positive association with livestock disturbance.

Sweep samples containing the families Dytiscidae (Coleoptera), Chironomidae (Diptera), and Ceratopogonidae (Diptera) were negatively affected by heavy levels of livestock disturbance (Table 2.7-2.8). Dytiscid abundance was negatively correlated with livestock disturbance in spring but positively associated in summer (Table 2.7). Spring sweep chironomid abundance decreased as livestock disturbance increased (Table 2.7). Ceratopogonid summer abundance was negatively correlated with livestock use (Table 2.7). Spring biomass of both Chironomidae and Ceratopogonidae had negative relationships with livestock disturbance (Table 2.8).

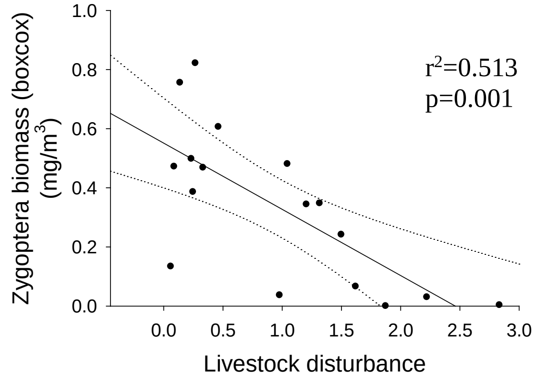
Core samples had few significant relationships with livestock disturbance (Table 2.9). Summer core Ceratopogonidae abundance and biomass were negatively associated with high levels of grazing. In summer, livestock disturbance significantly increased Oligochaeta core abundance.

Significant linear relationships with environmental variables other than livestock disturbance can be found in Appendix H.

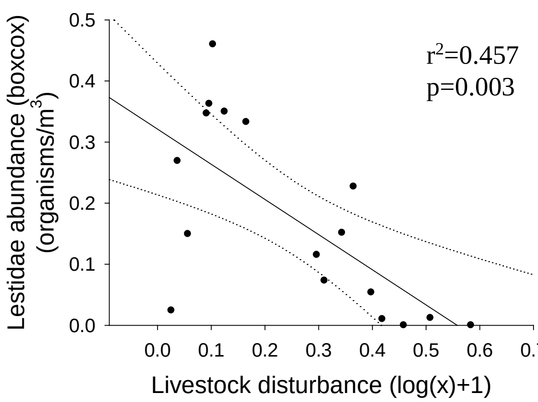
a)



b)



c)



d)

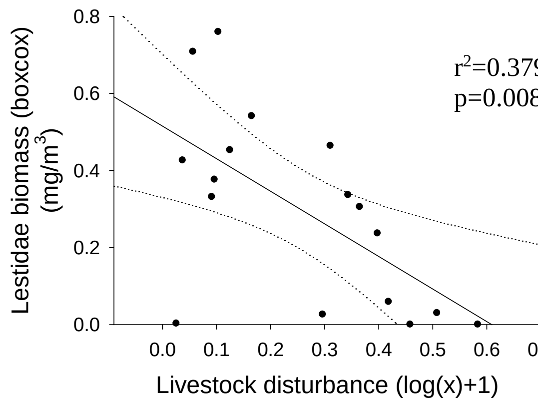


Figure 2.7. Linear relationship between livestock disturbance and spring sweep a) Zygoptera abundance b) Zygoptera biomass c) Lestidae abundance and d) Lestidae biomass. Livestock disturbance represents the mean # of quadrat corners that intersected bare ground at each wetland (Jones et al. 2011). Dotted lines represent 95% confidence intervals.

Table 2.7. Significant spring and summer sweep abundance regressions with livestock disturbance as the independent variable alone or in combination with other environmental variables. Regression slopes are described as either positive (+) or negative (-). Only relationships with p <0.05 are shown.

Sampling Period	Dependent Variable	Independent Variable(s)	Slope	F	df	p	r^2	Adj. r^2
Spring	Coleoptera	Disturbance	-	8.60	15	0.010	0.364	0.322
	Dytiscidae	Disturbance	-	9.13	15	0.009	0.378	0.337
	Lestidae	Disturbance + Perimeter	- & -	10.29	14	0.002	0.595	0.537
	Aeshnidae	Disturbance	+	12.51	15	0.003	0.455	0.418
	Hemiptera	Disturbance + Conductivity	- & -	10.17	14	0.002	0.592	0.534
	Hemiptera	Disturbance	-	5.56	15	0.032	0.271	0.222
	Corixidae	Disturbance + Conductivity	- & -	10.17	14	0.002	0.592	0.534
	Diptera	Disturbance	-	9.99	15	0.006	0.400	0.360
	Chironomidae	Disturbance	-	6.77	15	0.020	0.311	0.265
Summer	Dytiscidae	Disturbance	+	10.20	10	0.010	0.505	0.456
	Ceratopogonidae	Disturbance	-	8.49	10	0.015	0.459	0.405

Table 2.8. Significant spring sweep biomass regressions with livestock disturbance as the independent variable alone or in combination with other environmental variables. Regression slopes are described as either positive (+) or negative (-). Only relationships with $p < 0.05$ are shown.

Sampling Period	Dependent Variable	Independent Variable(s)	Slope	F	df	p	r^2	Adj. r^2
Spring	Diptera	Disturbance	-	17.50	15	<0.001	0.538	0.508
	Ceratopogonidae	Disturbance+Conductivity	- & -	9.79	14	0.002	0.583	0.524
	Ceratopogonidae	Disturbance	-	10.08	15	0.006	0.402	0.362
	Chironomidae	Disturbance	-	15.49	15	0.001	0.508	0.475
	Aeshnidae	Disturbance+Conductivity	- & +	7.27	14	0.007	0.510	0.440
	Lestidae	Disturbance+Perimeter	- & -	7.89	14	0.005	0.530	0.463

Table 2.9. Significant spring and summer core abundance and summer core biomass regressions with livestock disturbance as the independent variable alone or in combination with other environmental variables. Regression slopes are described as either positive (+) or negative (-). Only relationships with $p < 0.05$ are shown.

Sampling Period	Dependent Variable	Independent Variable(s)	Slope	F	df	p	r^2	Adj. r^2
CORE ABUNDANCE								
Spring	Zygoptera	Disturbance	-	18.86	15	<0.001	0.557	0.528
	Lestidae	Disturbance	-	6.13	15	0.026	0.290	0.243
	Richness	Disturbance + Conductivity	- & -	3.79	14	0.048	0.351	0.259
Summer	Ceratopogonidae	Disturbance	-	8.62	10	0.015	0.463	0.409
	Oligochaeta	Disturbance	+	10.92	10	0.008	0.522	0.474
	Oligochaeta	Disturbance + Conductivity	+ & -	7.97	9	0.010	0.639	0.559
CORE BIOMASS								
Summer	Ceratopogonidae	Disturbance	-	8.65	10	0.015	0.464	0.410

Discussion

Aquatic Invertebrate Community Response

The results of this study indicate that high levels of livestock grazing negatively affects the aquatic macroinvertebrate assemblages in local wetlands by reducing the total abundance, total biomass, taxa richness and diversity of these communities. Strongest relationships were observed within nektonic communities during the spring, although benthic and summer patterns also emerged. NMDS ordinations showed significant associations amongst macroinvertebrate community composition and livestock disturbance while regression analyses indicated heavy disturbance to be an important factor in reducing both total abundance and biomass, and family richness and diversity. The literature both supports and refutes these findings and shows the spatial and temporal variability of aquatic invertebrate response to livestock disturbance.

Wetland richness and diversity were anticipated to decrease with increasing livestock disturbance because of the reduction of submerged and emergent vegetation and the elevation of nutrients resulting in increased primary production. Higher productivity is often associated with decreased diversity (Jeppesen et al. 2000). My results are corroborated by Ausden et al. (2005), who found species richness was significantly reduced by cattle grazing in United Kingdom fens. Furthermore, a Kansas study found family richness was greater in control than in grazed treatments although no difference in diversity was detected (Kostecke et al. 2005). In sharp contrast to my study, Marty (2005) reported higher aquatic invertebrate richness in grazed versus ungrazed Californian vernal pools. Similarly, Davis and Bidwell (2008) found both benthic and nektonic richness and benthic diversity were greatest in grazed treatments in Nebraska. Steinman et al. (2003) discovered that although invertebrate richness and diversity varied in the two vegetation types present in their south-central Florida wetlands, cattle grazing did not have any effect on the aquatic invertebrate community. Similar patterns of geographic heterogeneous response to livestock disturbance were also evident with total abundance and biomass of macroinvertebrates.

Total macroinvertebrate abundance and biomass decreased in the presence of high livestock disturbance. I had expected a compositional shift in wetland taxa under heavy grazing pressure but had surmised this would not necessarily reduce the number or biomass of wetland macroinvertebrates present. McAbendroth et al. (2005) found high invertebrate biomass was correlated with high vegetation complexity and suggested habitat complexity may control invertebrate biomass in wetlands. In my study, decreased wetland vegetation complexity and structure resulting from heavy grazing is supported by a concurrent study examining grazing impacts on wetland vegetation (Jones et al. 2011). Other projects have recorded varying abundance and biomass response to livestock disturbance. In Florida, simulated grazing in depressional freshwater marshes decreased macroinvertebrate abundance (Morrison and Bohlen 2010). In contrast, Davis and Bidwell (2008), found total biomass was greater in grazed than in reference wetlands; however, the opposite was true the following year, with reference wetlands having greater biomass of macroinvertebrates than the grazed treatments. The high degree of disagreement across, and even within, studies suggests that regional differences, such as local climate, geology and elevation may prevent detailed comparison of wetland communities (Batzer et al. 2005).

Zygoptera Response

Spring sweep biomass was dominated by zygopterans and heavy levels of livestock grazing reduced Zygoptera biomass and abundance at the suborder and family (Lestidae) level. These results were no surprise due to the connection between wetland vegetation and Zygoptera habitat requirements (Hornung and Rice 2003). *Lestes* is the only genus of Lestidae found in British Columbia and many species of *Lestes* are specialists of temporary wetlands (Cannings 2002). Most Zygoptera are obligate endophytic ovipositors that lay their eggs in emergent as in the case of *Lestes*, and submergent vegetation within wetlands (Duffy 1994; Cannings 2002). Macrophytes are utilized by larval stages for foraging, refugia and emergence. This reliance on vegetation makes them susceptible to livestock grazing within, and adjacent to, aquatic habitats. Also, *Lestes* eggs laid in emergent stems may be eaten along with the plants in spring and summer. Vegetation in heavily grazed sites was completely lacking or, if present, was sparse and/or reduced in height and did not have the visual habitat features adult zygopterans prefer and require (Corbet 1999; Bernath et al. 2002). Foote & Hornung (2005) proposed that vegetation height is also important for wind protection, as zygopterans are not strong flyers and wind refugia may be a critical structural habitat requirement. My study supports this as plant communities within my study wetlands shifted from tall and rhizomatous species to shorter-lived, smaller species in the presence of increased grazing (Jones et al. 2011).

Other Taxa Response to Disturbance

Not surprisingly, Diptera, because of their rapid reproductive rate, a terrestrial adult stage, and the ability of the larvae of many species to extract oxygen from the atmosphere, are a common and often a dominant taxon present in wetlands (King and Richardson 2002). Within the Diptera, the family Chironomidae usually makes the highest contribution to invertebrate abundance in wetlands (Wrubleski 1987; Batzer et al. 2001). This is supported in my study by both the benthic and nektonic communities. Diptera, primarily family Chironomidae, and Ostracoda were the most abundant macroinvertebrate orders in spring; in summer, the Diptera were the most abundant. Morrison and Bohlen (2010) found that Diptera abundance increased when vegetation was clipped and removed to simulate grazing. Contrary to their results, I found that spring nekton Diptera and chironomid abundance and biomass decreased with heavy grazing. However, chironomids in the spring benthic community responded positively to higher levels of grazing. Given that my taxonomic resolution was to family only, I suggest that different genera or species residing in the benthos versus in the nekton were responsible for this difference. Benthic taxa are more tolerant of anoxic conditions resulting from eutrophication and increased turbidity (Campbell et al. 2009), and would be expected to be less affected by livestock presence. The absence of relationships between summer sweep and core dipteran abundance and richness might be due to the absence of chironomid collected because of emergence timing.

The relationship between Dytiscidae abundance and livestock disturbance was negative in the spring but positive in the summer. Dytiscid beetles collected in spring were primarily larvae which depend upon wetland vegetation for hunting prey and require contact with the surface to obtain atmospheric oxygen (Resh et al. 2008). During this time, larval dytiscids would be more vulnerable to vegetation removal and trampling, and agitation of the water surface by livestock. In the absence of fish and birds, Odonata (Suborders Anisoptera and Zygoptera), Dytiscidae and Hemiptera are the top predators of macroinvertebrates in semi-permanent wetlands, and are usually speciose and numerous in these wetland ecosystems (Batzer and Wissinger 1996). I speculate that as livestock disturbance increases and zygopterans declined as a result, dytiscid abundance may have increased due to reduced competition for prey and resources.

Study limitations

As invertebrate communities are highly variable in time and space, my study, because of low sampling frequency and intensity, may have failed to find some potential relationships caused by livestock disturbance. Miller et al. (2008) suggest that within-year temporal shifts in the community dynamics may result in misconstrued and unreliable data. Two sampling sessions may have been insufficient to accurately portray all effects of livestock on macroinvertebrates. Within-wetland abundance and biomass variability was often high. Downing (1991) noted that the structural heterogeneity of wetland habitats causes patchy aggregations of macroinvertebrates. To alleviate this sampling problem, King and Richardson (2002) suggest that aggregate samples from more than one habitat are most accurate in quantifying the community and recording rare taxa. More intensive sampling of each wetland would give a better picture of the invertebrate community and help to determine whether taxa collected were in fact rare or only captured in low numbers due to low sampling effort. More intensive sampling would also potentially allow for genera or species analyses, which probably would provide more insight into livestock effects. Family level analyses may not be sufficient to detect most responses to livestock impacts (King and Richardson 2002). I was unable to conduct statistical analyses using genera or species because of the many immature specimens (difficult or impossible to identify past family level) and the many zero observations in my data set.

Management Recommendations

Sweep sampling appears to give clearer results than core sampling. For resource managers, this collection method is usually a more efficient technique, requiring less sample processing time than core samples. Although more expensive, the analysis of the biomass of aquatic macroinvertebrates, and not just abundance alone, should be explored in these wetland systems. Biomass characterizes food web energy flow of these important prey items more accurately than abundance alone, thus providing better insight into wetland trophic dynamics and the effects of disturbance.

Resource managers should recommend range use plans that include only light grazing regimes and limit livestock access to wetlands. My data indicate that aquatic macroinvertebrates could potentially be used to indicate heavy levels of livestock disturbance; such information would assist resource managers in monitoring for wetland impairment. Specifically, damselflies (Odonata: Zygoptera) at the suborder level or family level (Zygoptera: Lestidae) show promise as a bioindicator of heavy levels of livestock use in British Columbia`s southern interior wetlands. Resource managers would be prudent to include zygopterans in any index developed to assess wetland condition.

Conclusion

High intensity livestock grazing in southern interior wetlands is reducing total abundance, biomass, richness and diversity of aquatic macroinvertebrates. My study is the first to examine the impact of free-range livestock practices on macroinvertebrates in BC southern interior wetlands, but further study is required to fully understand the consequences of this wide spread disturbance. Future work should examine annual patterns of abundance, biomass, richness and diversity to produce a better understanding of the temporal variability of these systems. This will be increasingly important to our understanding of the resilience of aquatic invertebrate communities with predicted climate change and the increasing pressure placed on wetland resources in these dry grassland regions.

Resource managers should implement sweep sampling as an effective, inexpensive method of monitoring wetland water quality and livestock impact. Consideration should be given to producing a local bioassessment index using Zygoptera as an indicator taxon. Range plans should adopt only light grazing in wetland areas and limit livestock access to sustain the biodiversity and productivity of these valuable aquatic ecosystems.

Literature Cited

Adams B., and L. Fitch. 1998. Caring for the green zone – riparian areas and grazing management. 2nd edition. Riparian Habitat Management Project, Lethbridge, AB, Pub. No. I-581.

Alberta Environment and Sustainable Resource Development (AESRD). 2014. Alberta wetland classification system. Water Policy Branch, Policy Division. Edmonton, AB.

Ausden, M., M. Hall, P. Pearson, and T. Strudwick. 2005. The effects of cattle grazing on tall-herb fen vegetation and molluscs. Biological Conservation 122:317-326.

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, 2nd edition. U.S. Environ. Protection Agency; Office of Water, Washington, DC. EPA 841-B-99-002.

Batzer, D. P., S. E. Dietz-Brantley, B. E. Taylor, and A. E. DeBiase. 2005. Evaluating regional differences in macroinvertebrate communities from forested depressional wetlands across eastern and central North America. Journal of the North American Benthological Society 24:403–14.

Batzer, D. P., B. J. Palik, and R. Buech. 2004. Relationships between environmental characteristics and macroinvertebrate communities in seasonal woodland ponds of Minnesota. Journal of the North American Benthological Society 23(1):50-68.

Batzer, D. P., A. S. Shurtleff, and R. B. Rader. 2001. Sampling invertebrates in wetlands. Pages 339-354 *in* R. B. Rader, D. P. Batzer and S. A. Wissinger (editors). Bioassessment and Management of North American Freshwater Wetlands. John Wiley and Sons, Inc., New York, NY.

Batzer, D. P. and S. A. Wissinger. 1996. Ecology of insect communities in nontidal wetlands. Annual Review of Entomology 41:75-100.

Benke, A. C. 1993. Concepts and patterns of invertebrate production in running waters, Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie 25:15-38.

Benke, A. C., A. D. Huryn, L. A. Smock, and J. B. Wallace. 1999. Length-mass relationships for freshwater macroinvertebrates in North America with particular reference to the southeastern United States. Journal of the North American Benthological Society 18(3):308-343.

Bernath, B., G. Szednic, G. Wildermuth, and G. Horvath. 2002. How can dragonflies discern bright and dark waters from a distance? The degree of polarization of reflected light as a possible cue for dragonfly habitat selection. Freshwater Biology 47:1707-1719.

British Columbia Government (BCGov). 2008. Geographic Services, DataBC, iMapBC. Available from http://maps.gov.bc.ca/ess/sv/imapbc/ [accessed April 2008].

Brönmark, C. and L. Hansson. 2002. The Biology of Lakes and Ponds. Oxford University Press Inc. New York, NY.

Caldwell, J. P., J. H. Thorp, and T. O. Jervey. 1980. Predator-prey relationships among larval dragonflies, salamanders and frogs. Oecologia 46:285-289.

Campbell, B. D., R. J. Haro and W. B. Richardson. 2009. Effects of agricultural land use on chironomid communities: comparisons among natural wetlands and farm ponds. Wetlands 29(3):1070-1080.

Cannings, R. A. 2002. Introducing the Dragonflies of British Columbia and the Yukon. Royal British Columbia Museum, Victoria, BC.

Cannings, R. A., S. G. Cannings, and R. J. Cannings. 1980. The distribution of the genus *Lestes* in a saline lake series in central British Columbia, Canada (Zygoptera: Lestidae). Odonatologica 9(1):19-28.

Cannings, R. A., and G. G. E. Scudder. 1978. The Littoral Chironomidae (Diptera) of saline lakes in central British Columbia. Canadian Journal of Zoology 56:1144-1155.

Carlisle, D. M., C. P. Hawkins, M. R. Meador, M. Potapova, and J. Falcone. 2008. Biological assessments of Appalachian streams based on predictive models for fish, macroinvertebrate, and diatom assemblages. Journal of the North American Benthological Society 27(1):16-37.

Clarke, K. R. 1993. Nonparametric multivariate analyses of changes in community structure. Australian Journal of Ecology 18:117-143.

Coffin, D. P. and W. K. Lauenroth. 1988. The effects of disturbance size and frequency on a shortgrass plant community. Ecology 69:1609-1617.

Collins, S. L., A. K. Knapp, J. M. Briggs, J. M. Blair, and E. M. Steinauer. 1998. Modulation of diversity by grazing and mowing in native tallgrass prairie. Science 280:745-747.

Corbet, P. S. 1999. Dragonflies: Behavior and Ecology of Odonata. Cornell University Press, Ithaca, NY.

Corcoran, R. M., J. R. Lovvorn, and P. J. Heglund. 2009. Long-term change in limnology and invertebrates in Alaskan boreal wetlands. Hydrobiologia 620(1):77-89.

Cox, R. R., M. A. Hanson, C. C. Roy, N. H. Euliss, D. H. Johnson, and M. G. Butler. 1998. Mallard duckling growth and survival in relation to aquatic invertebrates. Journal of Wildlife Management 62(1):124-133.

Davis, C. A. and J. R. Bidwell. 2008. Response of aquatic invertebrates to vegetation management and agriculture. Wetlands 28(3):793-805.

Downing, J. A. 1991. The effect of habitat structure on the spatial distribution of freshwater invertebrate populations. Pages 87-119 *in* S. S. Bell, E. D. McCoy, and H.R. Mushinsky (editors). Habitat structure: the physical arrangement of objects in space. Chapman and Hall, New York, NY.

Duffy W. G. 1994. Demographics of *Lestes disjunctus disjunctus* (Odonata: Zygoptera) in a riverine wetland. Canadian Journal of Zoology 72:910-17.

Eaton, K. E. 1983. The life history and production of *Chaoborus punctipennis* (Diptera: Chaoboridae) in Lake Norman, North Carolina, U.S.A. Hydrobiologia 106:247-252.

Foote, A. L., and C. L. Rice Hornung. 2005. Odonates as biological indicators of grazing effects on Canadian prairie wetlands. Ecological Entomology 30:273-283.

Glozier, N. E., J. M. Culp, and D. Halliwell. 2002. Review of cycle 2 benthic processing protocols and QA/QC issues. National Water Research Institute, Environment Canada, Saskatoon, SK.

Google Maps. 2014. TerraMetrics Map data, City of Kamloops Satellite Imagery – map source [online]. Available from: https://www.google.ca/maps/ [accessed April 2014]

Hilsenhoff, W. L. 1988. Rapid field assessment of organic pollution with a family-level biotic index. Journal of the North American Benthological Society 7:65-68.

Hornung, J. P., and C. L. Rice. 2003. Odonata and wetland quality in southern Alberta, Canada: a preliminary study. Odonatologica 32:119-129.

Howmiller, R. P. 1972. Effects of preservatives on weights of some common macrobenthic invertebrates. Transactions of the American Fisheries Society 101:743–746.

Jeppesen, E., J.P. Jensen, M. Søndergaard, T. Lauridsen, T. and F. Landkildehus, 2000. Trophic structure, species richness and biodiversity in Danish lakes: changes along a phosphorus gradient. Freshwater Biology 45: 201–218.

Johnston, T.A., and R.A. Cunjak. 1999. Dry mass-length relationships for benthic insects: a review with new data from Catamaran Brook, New Brunswick, Canada. Freshwater Biology 41:653-674.

Jones, W. M., L. H. Fraser, and P. J. Curtis. 2011. Plant community functional shifts in response to livestock grazing in intermountain depressional wetlands in British Columbia, Canada. Biological Conservation 144:511-517.

King, R. S. and C. J. Richardson. 2002. Evaluating subsampling approaches and macroinvertebrate taxonomic resolution for wetland bioassessment. Journal of the North American Benthological Society 21(1):150-171.

Kostecke, R. M., L. M. Smith and H. M. Hand. 2005. Macroinvertebrate response to cattail management at Cheyenne Bottoms, Kansas, USA. Wetlands 25(3):758-763.

Krapu, G. L. 1974. Feeding ecology of pintail hens during reproduction. Auk 91:278-290.

Marchant, R., and H. B. N. Hynes. 1981. Field estimates of feeding rate of *Gammarus pseudolimnaeus* (Crustacea: Arthropoda) in the Credit River, Ontario. Freshwater Biology 11:27-36.

Marty, J. T. 2005. Effects of cattle grazing on diversity in ephemeral wetlands. Conservation Biology 19(4):1626-1632.

McAbendroth, L., P. M. Ramsay, A. Foggo, S. D. Rundle and D. T. Bilton. 2005. Does macrophyte fractal complexity drive invertebrate diversity, biomass and body size distributions? Oikos 111:279-290.

McCune, B. and J. B. Grace. 2002. Analysis of ecological communities. MjM Software Design, Gleneden Beach, Oregon, USA.

McEwen, G. F., M. W. Johnson, and T. R. Folsom. 1954. A statistical analysis of the performance of the folsom plankton sample splitter, based upon test observations. Meteorology and Atmospheric Physics 7(1):502-527.

Meidinger, D., and J. Pojar. 1991. Ecosystems of British Columbia. Research Branch, Ministry of Forests, Victoria, BC. Special Report Series Number 6.

Merritt, R. W., K. W. Cummins, and M. B. Berg (editors). 2008a. An Introduction to the Aquatic Insects of North America, 4th edition. Kendall/Hunt Publishing Co., Dubuque, Iowa, USA.

Merritt, R. W., K. W. Cummins, Resh V. H. and D. P. Batzer. 2008b. Sampling aquatic insects: collection devices, statistical considerations and rearing procedures. Pages 15-37 *in* R. W. Merritt, K. W. Cummins, and M. B. Berg (editors.). An Introduction to the Aquatic Insects of North America, 4th Edition. Kendall/Hunt Publishing Co., Dubuque, Iowa, USA.

Miller, A. T., M. A. Hanson, J. O. Church, B. Palik, S. E. Bowe and M. G. Butler. 2008. Invertebrate community variation in seasonal forest wetlands: implications for sampling and analyses. Wetlands 28(3):874-881.

Ministry of Forests and Range (MOFR). 2007. Protection Branch, Kamloops Climate. Available from http://bcwildfire.ca/AboutUs/Organization/Kamloops/Climate/ [accessed October 2007].

Moore, A. A., and M. A. Palmer. 2005. Invertebrate biodiversity in agricultural and rural headwater streams: implications for conservation and management. Ecological Applications 15(4):1169-1177.

Morrison III, W. R., and P. J. Bohlen. 2010. Influence of vegetation on invertebrate communities in grazed freshwater wetlands in South-Central Florida. Southeastern Naturalists 9(3):453-464.

Murkin H. R., and L. C. M. Ross. 1999. Northern prairie marshes (Delta Marsh, Manitoba). I. Macroinvertebrate responses to a simulated wet/dry cycle. Pages 543-570 *in* D. P. Batzer, R. B. Rader and S. A. Wissinger SA (editors). Invertebrates in freshwater wetlands of North America: ecology and management. John Wiley and Sons, New York, NY.

Murkin, H. R. and L. C. M. Ross. 2000. Invertebrates in Prairie Wetlands. Pages 201-248 *in* H. R. Murkin, A. G. van der Valk, and W. R. Clark (editors.). Prairie Wetland Ecology: The Contribution of the Marsh Ecology Research Program. Iowa State University Press, Ames, IA.

Nader, G., K. W. Tate, R. Atwill, and J. Bushnell. 1998. Water quality effect of rangeland beef cattle excrement. Rangelands 20(5):19-25.

Natural Resources Canada. 2004. BC line-map source. Available from www.nrcan.gc.ca [accessed February 2015].

Neckles, H. A., H. R. Murkin, and J. A. Cooper. 1990. Influences of seasonal flooding on macroinvertebrate abundance in wetland habitats. Freshwater Biology, 23:311–322.

Nelson, J. W., J. A. Kadlec, and H. R. Murkin. 1990. Responses by macroinvertebrates to cattail litter quality and timing of litter submergence in a northern prairie marsh. Wetlands 10:47-60.

Oksanen, J., F. G. Blanchet, R. Kindt, P. Legendre, P. R. Minchin, R. B. O'Hara, G. L. Simpson, P. Solymos, M. H. H. Stevens, and H. Wagner. 2011. Vegan: community ecology package version 2.0-1. Available from http://vegan.r-project.org/ [accessed April 2011].

Pavlov A. M., and I. M. Zubina. 1990. Relationship of weight to body dimensions in dragonfly larvae. Hydrobiological Journal 26:99-103.

Pip, E. 1978. A survey of the ecology and composition of submerged aquatic snail-plant communities. Canadian Journal of Zoology 56:2263-2279.

R Development Core Team. 2011. R: language and environment for statistical computing. Version 2.12.2 R Foundation for Statistical Computing, Vienna. Available from http://cran.r-project.org/. [accessed April 2011].

Rader, R. B. 2001. An introduction to wetland bioassessment and management. Pages 339-354 *in* R. B. Rader, D. P. Batzer and S. A. Wissinger (editors). Bioassessment and Management of North American Freshwater Wetlands. John Wiley and Sons, Inc., New York, NY.

Resh, V. H., D. B. Buchwalter, G. A. Lamberti, and C. H. Eriksen. 2008. Aquatic insect respiration. Pages 39-54 *in* R. W. Merritt, K. W. Cummins, and M. B. Berg (editors.). An Introduction to the Aquatic Insects of North America, 4th Edition. Kendall/Hunt Publishing Co., Dubuque, IA.

Rosenberg D. M., and V. H. Resh. 1993. Freshwater biomonitoring and benthic macroinvertebrates. Routledge, Chapman and Hall Inc., New York, NY.

Shannon, C. E. and W. Weaver. 1949. The mathematical theory of communication. The University of Illinois Press, Urbana, ILs.

Silver, C. A., and S. M. Vamosi. 2012. Macroinvertebrate community composition of temporary prairie wetlands: a preliminary test of the effect of rotational grazing. Wetlands 32(1):185-197.

Simpson, E. H. 1949. Measurement of diversity. Nature 163:688-688.

Smock, L. A. 1980. Relationships between body size and biomass of aquatic insects. Freshwater Biology 10:375-383.

Steinman, A. D., J. Conklin, P. J. Bohlen, and D. G. Uzarski. 2003. Influence of cattle grazing and pasture land use on macroinvertebrate communities in freshwater wetlands. Wetlands 23(4):877-889.

Stewart, R. E., and H. A. Kantrud. 1971. Classification of natural ponds and lakes in the glaciated prairie region. Bureau of Sport Fisheries and Wildlife, U.S. Fish and Wildlife Service, Washington, D.C. Jamestown, ND: Northern Prairie Wildlife Research Center Resource Publication 92, Available from http://www.npwrc.usgs.gov/resource/wetlands/pondlake/index.htm [accessed April 1998].

Swanson, G. A. 1983. Benthic sampling for waterfowl foods in emergent vegetation. Journal of Wildlife Management 47:821-23.

Swanson G. A., M. I. Meyer, and V. A. Adomaitis. 1985. Foods consumed by breeding mallards on wetlands of south-central North Dakota. Journal of Wildlife Management 49(1):197-203.

Thorp, J. H., and Alan P. Covich. 2001. Ecology and classification of North American freshwater invertebrates. 2nd Edition. Academic Press, San Diego, CA.

Travis, J., W. H. Keen, and J. Juilianna. 1985. The role of relative body size in a predator-prey relationship between dragonfly naiads and larval anurans. Oikos 45:59-65.

Wen, Y. H. 1992. Life history and production of *Hyalella azteca* (Crustacea:Amphipoda) in a hypereutrophic prairie pond in southern Alberta. Canadian Journal of Zoology 70:1417-1424.

Wrubleski D. A. 1987. Chironomidae (Diptera) of peatlands and marshes in Canada. Memoirs of the Entomological Society of Canada 140:141-161.

Wrubleski D. A. 2005. Chironomidae (Diptera) responses to the experimental flooding of prairie marshes. Wetlands 25:200–209.

CHAPTER 3. CONCLUSION

Research Summary

My research objectives were to characterize wetland macroinvertebrate communities in the southern interior of British Columbia and to determine their response to a gradient of livestock grazing disturbance. As predicted, 1) macroinvertebrate richness and diversity decreased with higher levels of grazing as did 2) zygopteran, specifically Lestidae, abundance and biomass. In addition, total macroinvertebrate abundance and biomass decreased as grazing intensity increased. As a vital link between producers (algae and macrophytes) and higher trophic links in wetlands (Batzer and Wissinger 1996), invertebrate response to heavy livestock use may have large-scale consequences not only for the aquatic macroinvertebrate assemblages, but the entire ecosystem.

Challenges of Wetland Invertebrate Research and Future Research Directions

My study provides baseline knowledge of southern interior wetland macroinvertebrate assemblages and contributes to the primary literature on the response of these communities to livestock grazing disturbance. As with most ecological studies, there were limitations to my study due to finite resources and time constraints. Low sampling effort and intensity, and the lack of identifications at genus and species levels, may have prevented the discovery of more detailed associations between livestock grazing and wetland invertebrate communities.

Wetland macroinvertebrate distributions vary spatially and temporally. The complexity of wetland habitat structure results in patchy taxon distributions (Downing 1991) and requires intensive sampling to thoroughly sample invertebrate communities. Sampling wetland invertebrates is relatively easy; however, extraction of specimens from sediments and debris can be laborious, time consuming and consequently expensive. Accurate identification of specimens is the most difficult task of all. To adequately compare livestock effects across a range of grazing intensities required examination of a large number of wetlands. Due to the length of time required to process samples, I was able to collect only three samples per sampling device in each wetland. This restricted the statistical analyses of many taxa due to their scarcity in samples. I could not determine whether some taxa were rare in the study wetlands or were simply not captured. Future research should address this problem by collecting aggregate samples from many different habitats within a wetland and subsampling (King and Richardson 2002) to improve the diversity of taxa collected and to determine which taxa are, in fact, rare.

Although I sampled during the spring and summer, temporal differences in community assemblages may have been missed. Miller et al. (2008) caution that within-year temporal differences in invertebrate community response to gradients of disturbance may be unreliable if sampling has occurred over periods of greater than 15 days. Budgetary limitations prevented us from sampling more than twice. The inclusion of more sampling sessions would potentially have allowed genus or species level identifications, because larger numbers of mature larvae (more readily identified than immature ones) probably would have been collected. King and Richardson (2002) suggest that family level identification may not be adequate as wide ranging disturbance tolerances of species within families may result in misinterpretation of family level data. Future studies should attempt to sample over many intervals throughout the year to provide a temporal baseline of wetland invertebrate densities and composition. This would help alleviate concerns regarding the interpretation of environmental disturbance impacts on invertebrate communities.

Despite some sampling inadequacies and only moderate taxonomic resolution, I was able to show clear effects of livestock disturbance on wetland macroinvertebrates. These differences were detected at both the community level and, in the case of zygopterans, at the family level.

Management Implications

Range and wildlife managers face challenges in balancing economic feasibility with best management practises for wetland and upland areas. In BC’s grasslands there is no formal legislation for protection of depressional wetlands; however, livestock grazing is managed through the use of best management practices (BMPs) (MFLRNO 2015). Grazing within Crown range is allocated through the province's Range Program which requires tenure and lease holders to develop prescriptive Range Use Plans (RUP) or Range Stewardship Plans. Livestock grazing on private land relies on landowners to employ BMPs and conduct their operations with consideration for what best suits their livelihood and the environment. This can often be difficult for smaller operators as indicated by the range of grazing disturbance within my study. With one exception, study wetlands located within Crown land in Lac du Bois Provincial Park were all grazed at the low end of the livestock grazing gradient whereas many of the wetlands on private land were in the middle or in the upper end of the gradient.

The negative effects of livestock grazing can be avoided by following an adaptive management plan that includes the four principals of range management: distribution, use level, rest and time and duration of grazing (Fraser 2013). Should adverse livestock grazing effects become evident, RUPs can be adapted to effectively mitigate further impacts. Tenure and lease holders are encouraged to assess their land management practices, and are given online brochures and training by the provincial ministry (BCMFR 2006). Overgrazing of riparian areas is often an issue of livestock distribution due to livestock's affinity for wetland versus upland areas (Ganskopp 2001). In many cases, it is more cost-effective for managers either to completely fence off wetland areas and provide off-site water or use fencing to limit livestock access, rather than reducing stocking densities to compensate for over-usage (Stillings et al. 2003).

Aquatic invertebrate research is important to the understanding and management of wetland ecosystems in southern interior grasslands. North America studies show highly variable responses of ecosystems to livestock disturbance, suggesting regionally specific studies are necessary to characterize communities and their response. My study provides regionally specific data on macroinvertebrate community assemblages and shows that aquatic invertebrates respond to high levels of livestock disturbance in wetlands. Managers should consider aquatic macroinvertebrates as an assessment tool for determining sustainable levels of livestock grazing in wetlands. The development of regional indices of biological integrity would allow the establishment of long-term monitoring programs. Zygoptera and perhaps Lestidae should be included as a metric as they respond strongly to the removal of vegetation during heavy grazing.

Recommendations

Based on the findings of this study, I propose two recommendations that will improve monitoring and management of southern interior wetlands to sustain and conserve them for wildlife and the ranching community.

My first recommendation is for provincial ministries to consider developing an index of biological integrity to monitor and prevent impairment of regional wetlands. Nektonic communities showed a stronger response to heavy grazing than benthic communities, and if rapid assessment is the goal of land or wildlife managers, research should focus on sweep sampling as a monitoring method. Due to their clear response to high livestock disturbance, the damselfly suborder Zygoptera should be used as a possible indicator taxon.

Secondly, southern interior ranchers should adopt only light grazing in local wetlands. This can be achieved by limiting the time livestock spend at each wetland and associated riparian area with rest rotation grazing cycles. Fencing and/or off-site watering stations should also be implemented to limit livestock access to wetlands. Crown land managers should ensure that ranchers are closely adhering to the principles outlined in their RUPs and should monitor for compliance to ensure plans are effective at preventing or reducing livestock damage to grasslands and wetland areas. Regulatory agencies should establish a working relationship with private ranches and farms to promote stewardship and assist in the development of sustainable livestock grazing management plans that protect wetlands found on private properties.

Literature Cited

Batzer D. P. and S. A. Wissinger. 1996. Ecology of insect communities in non-tidal wetlands. Annual Review of Entomology 41:75–100.

British Columbia Ministry of Forests and Range (BCMFR). 2006. A methodology for monitoring Crown range. Range Branch, Kamloops, BC. Rangeland Health Brochure [10] draft.

Downing, J. A. 1991. The effect of habitat structure on the spatial distribution of freshwater invertebrate populations. Pages 87-119 *in* S. S. Bell, E. D. McCoy, and H.R. Mushinsky (editors). Habitat structure: the physical arrangement of objects in space. Chapman and Hall, New York, NY.

Fraser, D. 2013. The four principles of range management. BC Ministry of Forests, Lands and Natural Resource Operations, Range Branch. Rangeland Health Brochure 13. Available from http://www.for.gov.bc.ca/hra. [accessed Jan 2015].

Ganskopp, D. 2001. Manipulating cattle distribution with salt and water in large arid-land pastures: a GPS/GIS assessment. Applied Animal Behaviour Science. 73:251-262.

King, R. S. and C. J. Richardson. 2002. Evaluating subsampling approaches and macoinvertebrate taxonomic resolution for wetland bioassessment. Journal of the North American Benthological Society 21(1):150-171.

Miller, A. T., M. A. Hanson, J. O. Church, B. Palik, S. E. Bowe and M. G. Butler. 2008. Invertebrate community variation in seasonal forest wetlands: implications for sampling and analyses. Wetlands 28(3):874-881.

Ministry of Forests, Lands and Natural Resource Operations (MFLNRO). 2015. Range Branch, BC Government Rangeland Management. Available from http://www.for.gov.bc.ca/hra. [accessed Jan 2015].

Stillings, A. M., J. A. Tanaka, N.R. Rimbey, T. Delcurto, P. A. Momont and M. L. Porath. 2003. Economic implications of off-stream water developments to improve riparian grazing. Journal of Range Management 56:418-424.

APPENDIX A. Physical Data

Table A.1. UTM coordinates, elevation, area, perimeter length, mean pH and mean conductivity values for wetlands in spring (May/June) and summer (July), 2008 (n=3).

Study Area	Wetland	UTM Coordinates (NAD 83)	Elevation (m)	Area (ha)	Perimeter Length (km)	pH		Conductivity (µS/cm)	
						June	July	June	July
Campbell Range	7	10: 708561, 5604710	1076	1.20	0.43	7.68	wetland dry	2401	wetland dry
	8	10: 707444, 5606570	1099	1.30	0.47	9.59	wetland dry	1384	wetland dry
Hamilton Commonage	4.3	10: 683727, 5553621	1168	0.85	0.42	8.87	wetland dry	2906	wetland dry
	5	10: 685113, 5552865	1196	2.30	0.86	9.94	wetland dry	3343	wetland dry
	7.1	10: 683129, 5550460	1197	1.40	0.63	10.52	wetland dry	3152	wetland dry
	9	10: 683646, 5549430	1227	1.50	0.73	8.65	10.35	1842	2481
Lac Du Bois Bachelor	4	10: 681244, 5627504	784	0.38	0.34	8.67	9.15	9144	11820
	4.1	10: 681094, 5627892	798	0.44	0.38	8.60	8.81	5604	8571
	5	10: 681013, 5628073	816	0.35	0.23	8.39	9.40	3098	3686
	6.1	10: 680559, 5628828	861	0.43	0.28	8.67	9.70	4044	4705
	10	10: 679933, 5632183	936	0.37	0.22	9.00	10.64	1947	1613
Lac Du Bois Long Lake	4.1	10: 682899, 5631165	764	1.10	0.50	8.63	9.06	1088	952
	6.1	10: 683235, 5630800	811	0.31	0.22	8.66	9.15	6581	9538
	7	10: 683804, 5630309	855	1.50	0.70	8.13	8.77	3179	3491
Rose Hill	13.1	10: 692569, 5611180	1009	0.43	0.25	8.10	9.46	2392	2721
	14	10: 694156, 5611476	1029	1.00	0.46	7.66	8.78	2140	2220
	19	10: 692334, 5605109	891	0.53	0.32	8.06	7.91	3083	4174

APPENDIX B. Photos of livestock disturbance at study wetlands.

Figure B.1. July 2008 photos of study wetlands with a) low (LDBL 7) and b) high (RH 13.1) levels of livestock grazing disturbance.

APPENDIX C. Aquatic macroinvertebrate taxa list

Table C.1. List of aquatic macroinvertebrate taxa found in wetlands sampled May/June (Spring) and July (Summer), 2008 near Kamloops, British Columbia, Canada. An "X" denotes taxon presence in sample type (sweeps or cores) and season.

		Spring (n = 17)		Summer (n = 12)	
Macroinvertebrate Taxon	**Common Name**	**Sweeps**	**Cores**	**Sweeps**	**Cores**
EPHEMEROPTERA	Mayflies				
Baetidae	Small Minnow Mayflies	X	X	X	
TRICHOPTERA	Caddisflies				
Limnephilidae	Northern Case-maker Caddisflies			X	
DIPTERA	True Flies				
Chironomidae	Non-biting Midges	X	X	X	X
Ceratopogonidae	Biting Midges, No-See-Ums	X	X	X	X
Tipulidae	Crane Flies	X	X	X	X
Chaoboridae	Phantom Midges	X	X	X	X
Culicidae	Mosquitoes	X	X	X	X
Dixidae	Dixid Midges, Meniscus midges	X		X	X
Psychodidae	Moth and Sand Flies			X	
Sciomyzidae	Marsh Flies, Snail-killing Flies	X	X		
Stratiomyidae	Soldier Flies	X	X	X	
Tabanidae	Horse Flies, Deer Flies	X	X	X	X
Ephydridae	Shore and Brine Flies	X	X		X
Empididae	Dance Flies			X	
Dolichopodidae	Longlegged Flies		X	X	
HEMIPTERA	True Bugs				
Corixidae	Water Boatmen	X		X	X
Notonectidae	Backswimmers	X		X	X
Gerridae	Water Striders	X	X	X	

COLEOPTERA	Beetles				
Dytiscidae	Predaceous Diving Beetles	X	X	X	X
Hydrophilidae	Water Scavenger Beetles	X	X	X	X
Haliplidae	Crawling Water Beetles	X	X	X	X
ODONATA (ZYGOPTERA)	Damselflies				
Lestidae	Spreadwings	X	X	X	X
Coenagrionidae	Pond Damsels	X	X	X	X
ODONATA (ANISOPTERA)	Dragonflies				
Aeshnidae	Darner Dragonflies	X	X	X	X
Libellulidae	Skimmer Dragonflies	X	X	X	
COLLEMBOLA	Springtails	X		X	
AMPHIPODA	Scuds, Side-swimmers				
Hyalellidae	no common name	X	X	X	X
Gammaridae	no common name			X	
GASTROPODA	Snails and Limpets				
Planorbidae	Ram's horn Snails	X	X	X	X
Lymnaeidae	Pond Snails	X	X	X	X
Physidae	Bladder Snails	X		X	
OSTRACODA	Seed Shrimps	X	X	X	X
BIVALVIA	Freshwater Clams and Mussels				
Sphaeriidae	Fingernail Clams	X		X	
HIRUDINEA	Leeches	X		X	
OLIGOCHAETA	Aquatic Earthworms	X	X	X	X
NEMATODA	Roundworms	X	X	X	X
ACARINA	Water Mites	X	X	X	X

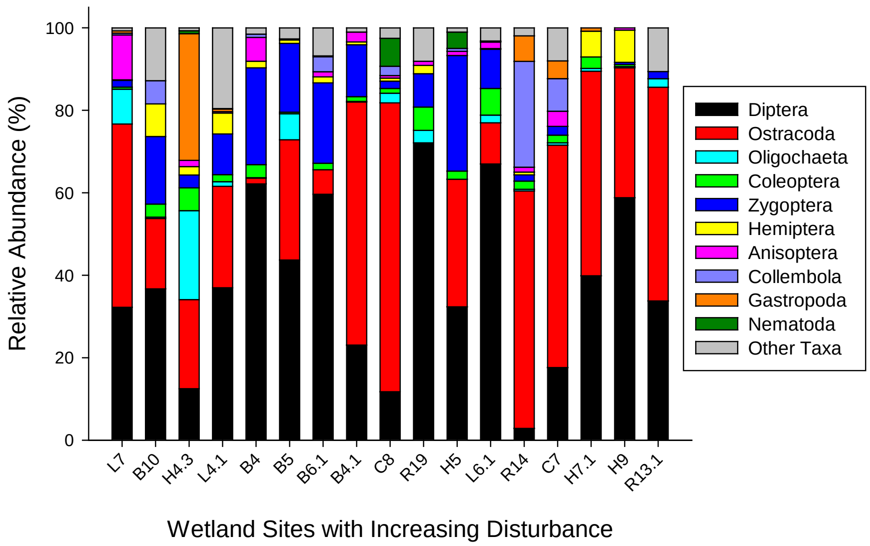
APPENDIX D. Length-mass regression equations

Table D.1. Length-mass regression equations used to determine biomass from taxon body lengths. Dry mass was determined using $M = a L^b$, where M = mass, L = body length and a and b are constants. Reference sources can be found in the Literature Cited section of Chapter 2.

Taxon	***a***	***b***	**Reference**
EPHEMEROPTERA			
Baetidae	0.0068	2.72	Benke 1993
TRICHOPTERA			
Limnephilidae	0.0052	2.832	Smock 1980
DIPTERA			
Chironomidae	0.0051	2.322	Smock 1980
Ceratopogonidae	0.0039	2.144	Smock 1980
Tipulidae	0.0054	2.463	Smock 1980
Chaoboridae	0.000453	2.43	Eaton 1983
Culicidae	0.000453	2.43	Eaton 1983
Dixidae	0.0051	2.322	Smock 1980
Psychodidae	0.0051	2.322	Smock 1980
Sciomyzidae	0.0066	2.436	Smock 1980
Stratiomyidae	0.0066	2.436	Smock 1980
Tabanidae	0.0050	2.591	Smock 1980
Ephydridae	0.0066	2.436	Smock 1980
Empididae	0.0066	2.436	Smock 1980
Dolichopodidae	0.0066	2.436	Smock 1980
HEMIPTERA			
Corixidae	0.0031	2.904	Smock 1980
Notonectidae	0.0031	2.904	Smock 1980
Gerridae	0.0150	2.596	Smock 1980
COLEOPTERA			
Dytiscidae (Adult)	0.0618	2.502	Smock 1980
Dytiscidae (Larva)	0.0111	2.490	Smock 1980
Hydrophilidae (Adult)	0.0618	2.502	Smock 1980
Hydrophilidae (Larva)	0.0111	2.490	Smock 1980
Haliplidae (Adult)	0.0271	2.744	Smock 1980
Haliplidae (Larva)	0.0111	2.490	Smock 1980
ODONATA			
Zygoptera			
Lestidae	0.00745	2.97	Pavlov and Zubina 1990
Coenagrionidae	0.0086	2.666	Smock 1980
Anisoptera			
Aeshnidae	0.0082	2.813	Smock 1980
Libellulidae	0.0072	2.618	Benke 1993
AMPHIPODA			
Hyalellidae	0.0049	3.001	Marchant and Hynes 1981
Gammaridae	0.0049	3.001	Marchant and Hynes 1981

APPENDIX E. Relative abundance and biomass (%) of spring and summer aquatic macroinvertebrates.

a)



b)

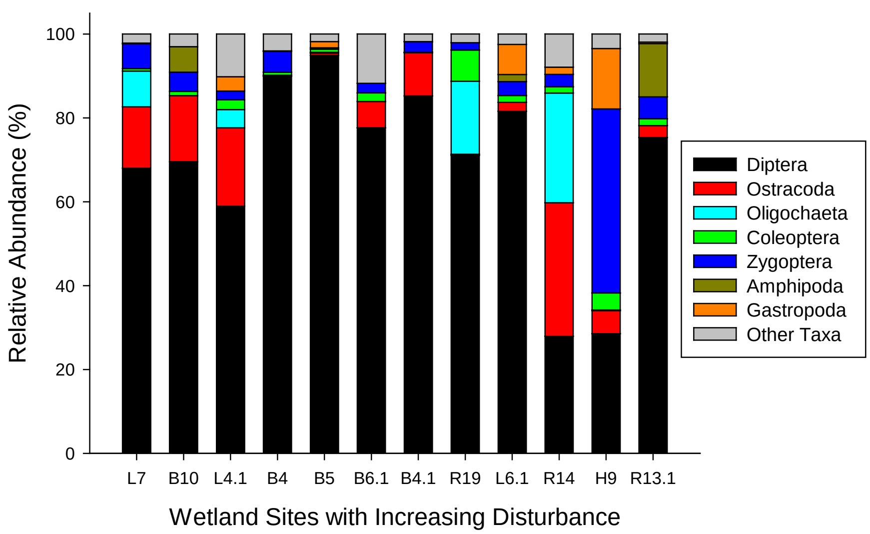
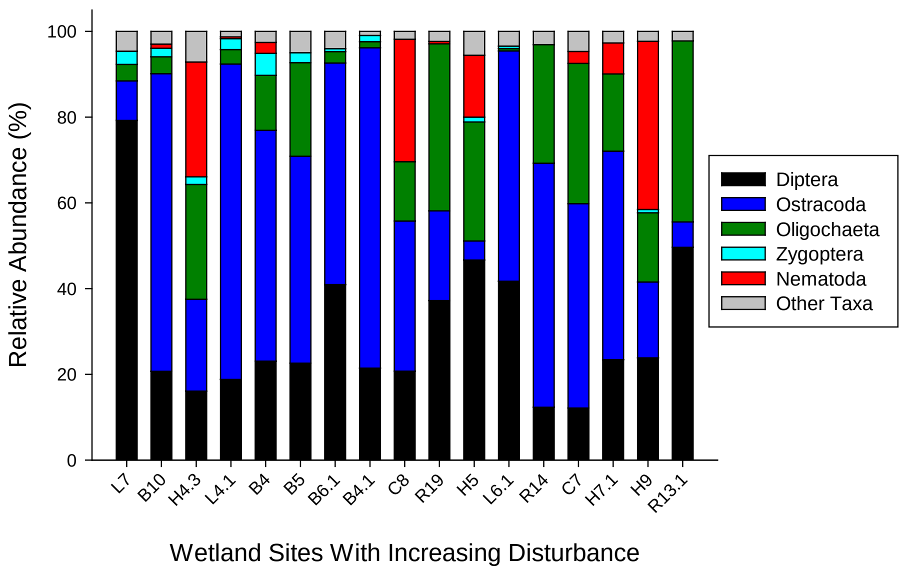


Figure E.1. Relative aquatic macroinvertebrate abundance (%) of a) spring and b) summer nektonic communities. To better illustrate the community composition, taxa that were present in >25% of the wetlands and had >5% relative abundance in at least one wetland are shown. Note that this cut-off protocol is different than the criteria used for statistical analyses (see Chapter 2, Methods Section).

a)



b)

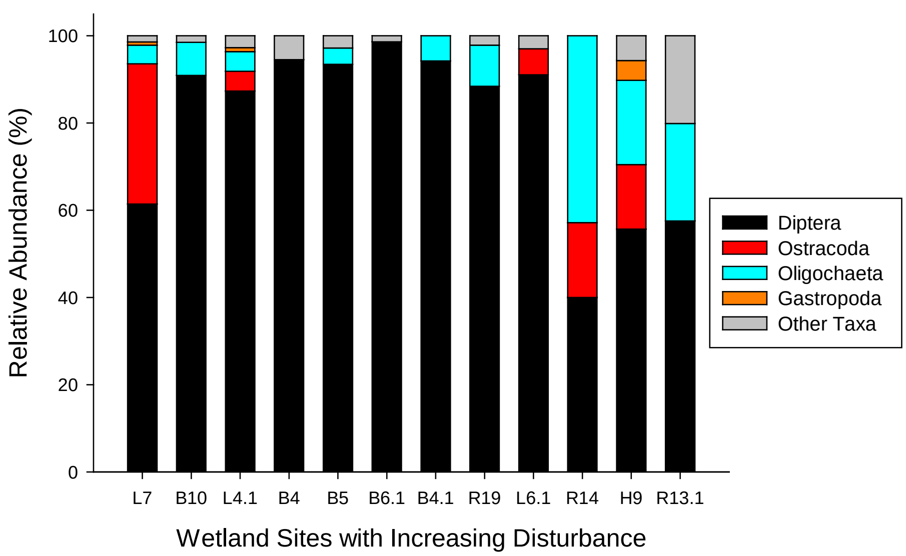
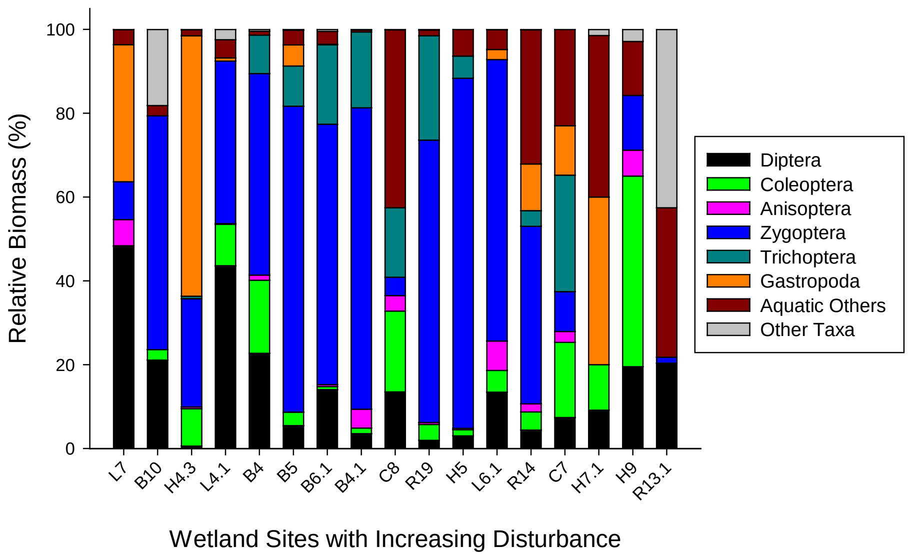


Figure E.2. Relative aquatic macroinvertebrate abundance (%) of a) spring and b) summer benthic communities. To better illustrate the community composition, taxa that were present in >25% of the wetlands and had >5% relative abundance in at least one wetland are shown. Note that this cut-off protocol is different than the criteria used for statistical analyses (see Chapter 2, Methods Section).

a)



b)

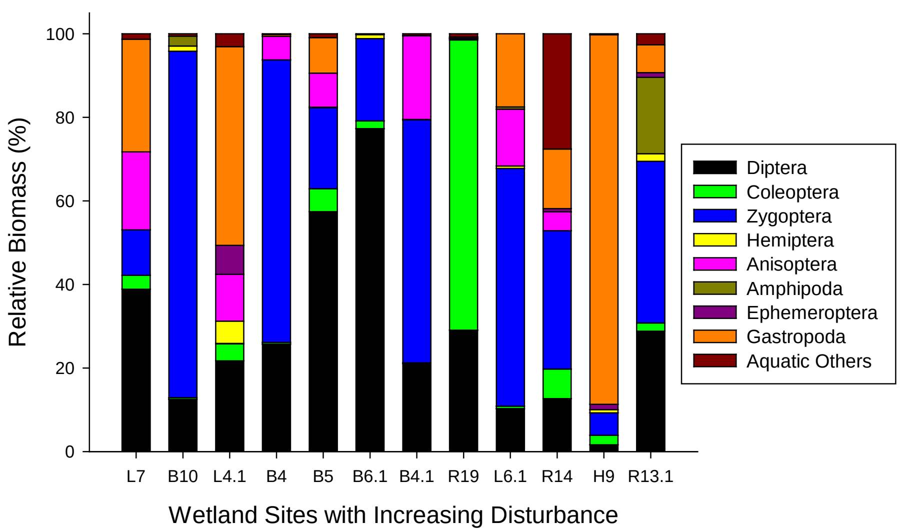
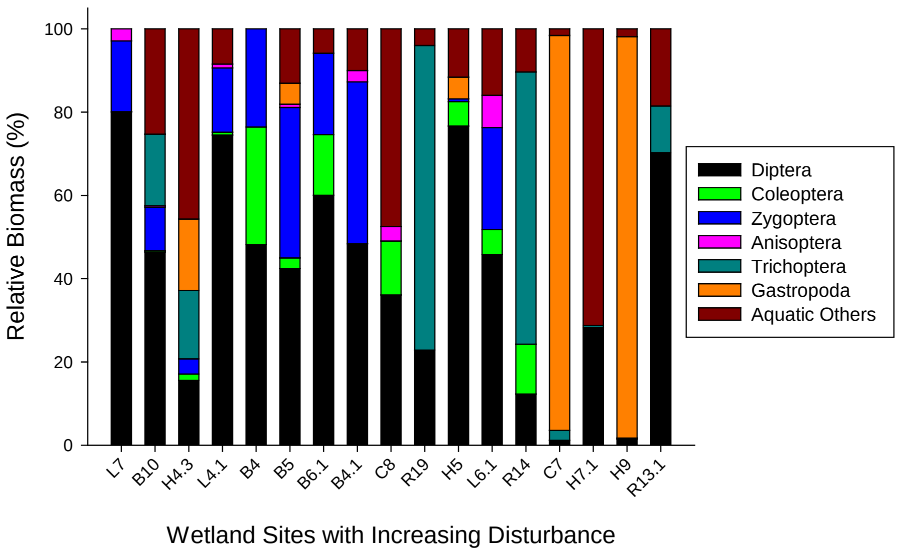


Figure E.3. Relative aquatic macroinvertebrate biomass (%) of a) spring and b) summer nektonic communities. To better illustrate the community composition, taxa that were present in >25% of the wetlands and had >5% relative abundance in at least one wetland are shown. Note that this cut-off protocol is different than the criteria used for statistical analyses (see Chapter 2, Methods Section).

a)



b)

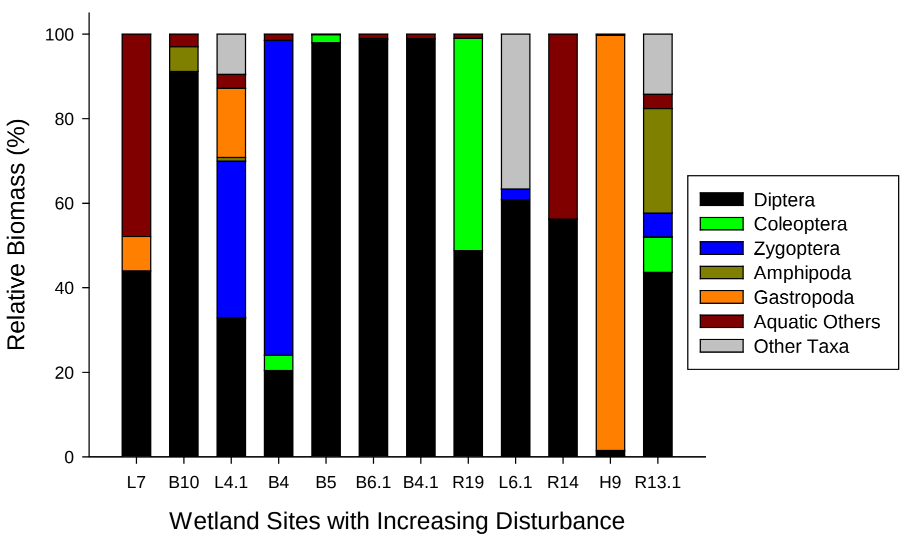


Figure E.4. Relative aquatic macroinvertebrate biomass (%) of a) spring and b) summer benthic communities. To better illustrate the community composition, taxa that were present in >25% of the wetlands and had >5% relative abundance in at least one wetland are shown. Note that this cut-off protocol is different than the criteria used for statistical analyses (see Chapter 2, Methods Section).

APPENDIX F. Nonmetric multidimensional scaling (NMDS) ordinations of dominant taxa community structure and environmental variables.

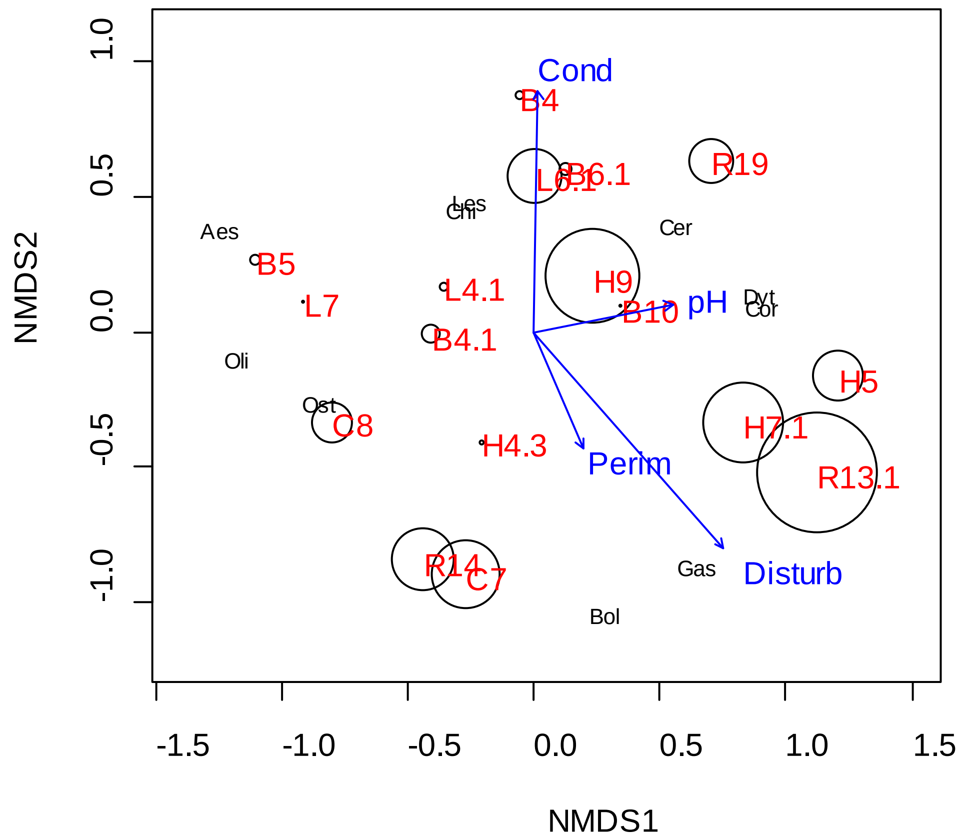


Figure F.1. Nonmetric multidimensional scaling (NMDS) ordination (stress=10.3%) of spring sweep abundance dominant taxa community structure. The environmental variables of livestock disturbance (Disturb), conductivity (Cond), perimeter length (Perim) and pH are shown as fitted vectors with the length of the arrow corresponding to the strength of the relationship. Livestock disturbance was significantly correlated with the ordination (r^2=0.486, p=0.008). Wetland site (in red) disturbance level is represented by a continuum with large circles being most disturbed and dots representing least disturbed sites. Taxa (in black) are as follows: Aes=Aeshnidae, Bol=Collembola, Cer=Ceratopogonidae, Chi=Chironomidae, Cor=Corixidae, Dyt=Dytiscidae, Gas=Gastropoda, Les=Lestidae, Oli=Oligochaeta, and Ost=Ostracoda.

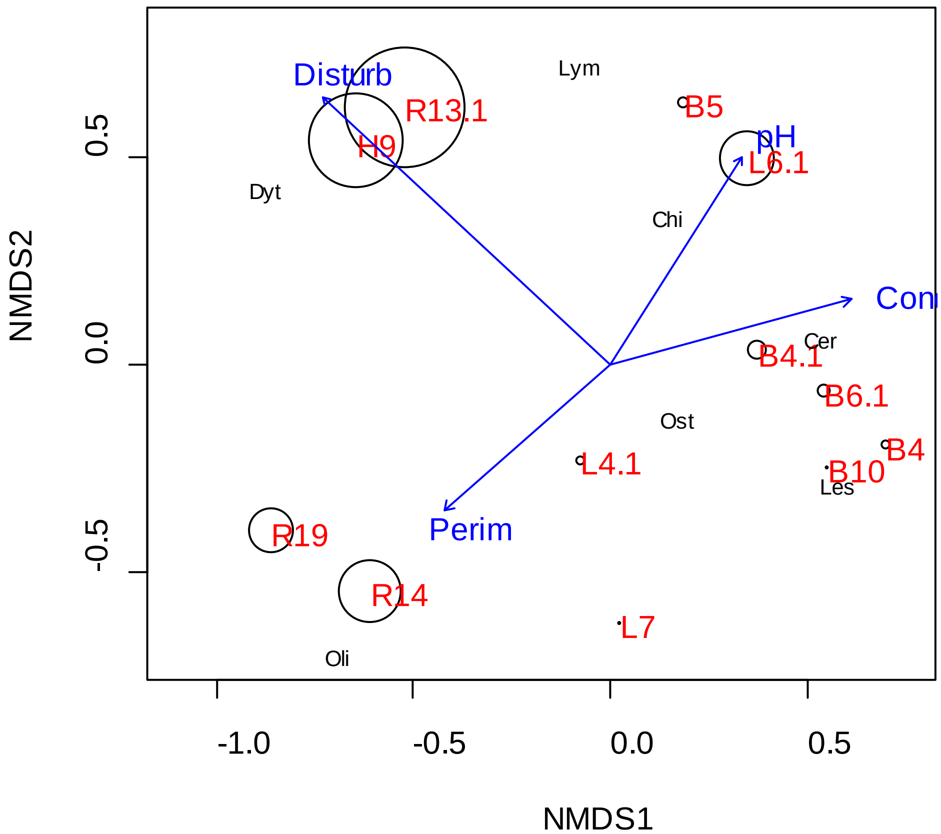


Table F.2. Nonmetric multidimensional scaling (NMDS) ordination (stress=13.3%) of summer sweep abundance dominant taxa community structure. The environmental variables of livestock disturbance (Disturb), conductivity (Cond), perimeter length (Perim) and pH are shown as fitted vectors with the length of the arrow corresponding to the strength of the relationship. Livestock disturbance was significantly correlated with the ordination (r^2=0.714, p=0.006). Wetland site (in red) disturbance level is represented by a continuum with large circles being most disturbed and dots representing least disturbed sites. Taxa (in black) are as follows: Cer=Ceratopogonidae, Chi=Chironomidae, Dyt=Dytiscidae, Les=Lestidae, Lym=Lymnaeidae, Oli=Oligochaeta, and Ost=Ostracoda.

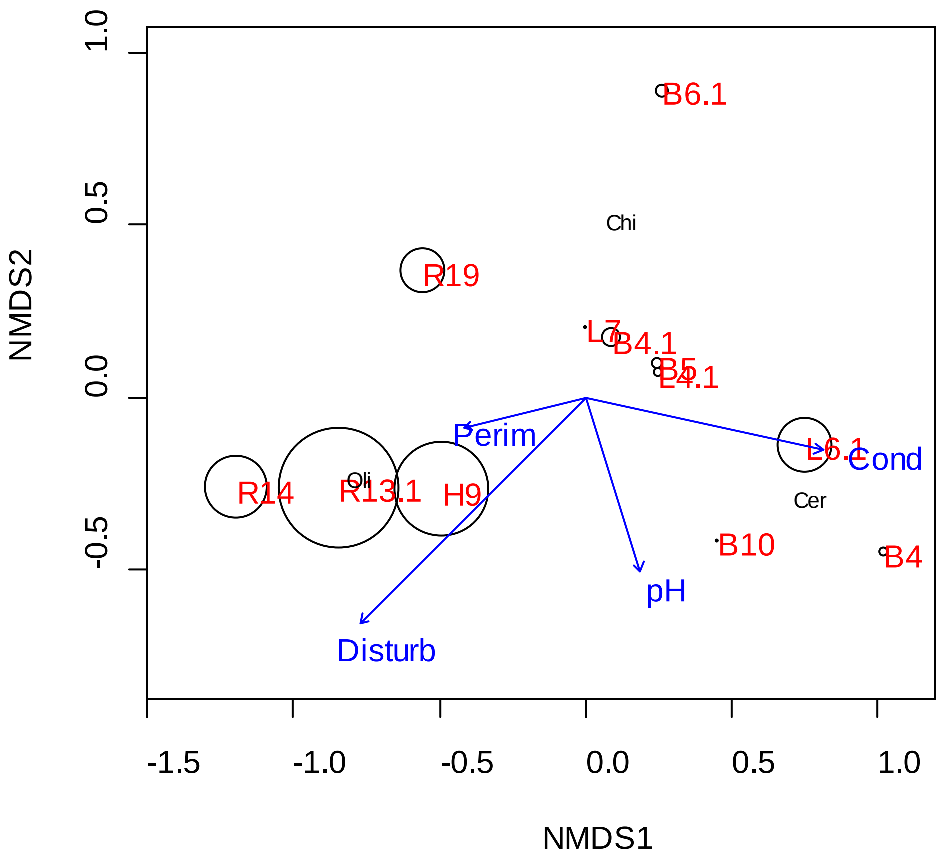


Figure F.3. Nonmetric multidimensional scaling (NMDS) ordination (stress=1.74%) of summer core abundance dominant taxa community structure. The environmental variables of livestock disturbance (Disturb), conductivity (Cond), perimeter length (Perim) and pH are shown as fitted vectors with the length of the arrow corresponding to the strength of the relationship. Livestock disturbance was significantly correlated with the ordination (r^2=0.534, p=0.034). Wetland site (in red) disturbance level is represented by a continuum with large circles being most disturbed and dots representing least disturbed sites. Taxa (in black) are as follows: Cer=Ceratopogonidae, Chi=Chironomidae, Oli=Oligochaeta.

APPENDIX G. Environmental variable correlation values from nonmetric multidimensional scaling (NMDS) ordinations.

Table G.1. Environmental variable correlation values from nonmetric multidimensional scaling (NMDS) ordinations. Only ordinations with at least one significant association with an environmental variable are shown. Significant variables ($p \leq 0.05$) are in bold.

Sample	Environmental Variables	r^2	p value
Spring Sweep Abundance	**Disturbance**	0.486	**0.008**
	Perimeter	0.089	0.529
	pH	0.126	0.426
	Conductivity	0.322	0.063
Summer Sweep Abundance	**Disturbance**	0.721	**0.005**
	Perimeter	0.229	0.307
	pH	0.275	0.238
	Conductivity	0.300	0.187
Summer Core Abundance	**Disturbance**	0.534	**0.035**
	Perimeter	0.094	0.627
	pH	0.152	0.455
	Conductivity	0.357	0.133

APPENDIX H. Significant linear regressions with macroinvertebrates and environmental variables other than livestock disturbance.

Table H.1. Significant linear regressions with macroinvertebrate communities and environmental variables other than livestock disturbance. Regression slopes are described as either positive (+) or negative (-). Only relationships with p <0.05 are shown.

Sampling Period	Dependent Variable	Independent Variable(s)	Slope	F	df	p	r^2	Adj. r^2
SWEEP ABUNDANCE								
Spring	Lestidae	Perimeter	-	4.71	15	0.046	0.239	0.188
Summer	Oligochaeta	pH + Conductivity	- & -	21.82	9	0.000	0.829	0.791
	Oligochaeta	pH	-	8.65	10	0.015	0.464	0.410
SWEEP BIOMASS								
Spring	Aeshnidae	Conductivity	+	7.89	15	0.013	0.345	0.301
Summer	Hemiptera	Conductivity	-	4.89	10	0.051	0.328	0.261
CORE ABUNDANCE								
Spring	Shannon Diversity	pH	+	8.12	15	0.012	0.351	0.308
	Simpson's Diversity	pH	+	6.69	15	0.021	0.308	0.262
Summer	Oligochaeta	Conductivity	-	6.62	10	0.028	0.398	0.338
	Richness	Conductivity	-	11.38	10	0.007	0.532	0.486
CORE BIOMASS								
Summer	Coleoptera	pH	-	5.52	10	0.041	0.356	0.291