

**FACTORS AFFECTING THE COMMUNITY ECOLOGY OF
PREDACIOUS DIVING BEETLES (COLEOPTERA: DYTISCIDAE)
IN BOREAL AND PRAIRIE PONDS ACROSS
SOUTHERN MANITOBA**

A Thesis

Submitted to the Faculty

of

Graduate Studies

by

Michael Alperyn

In Partial Fulfillment of the

Requirement for the Degree

of

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THE UNIVERSITY OF MANITOBA
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**Factors Affecting the Community Ecology of Predacious Diving Beetles
(Coleoptera: Dytiscidae) in Boreal and Prairie Ponds Across Southern Manitoba**

BY

Michael Alperyn

**A Thesis/Practicum submitted to the Faculty of Graduate Studies of The University
of Manitoba in partial fulfillment of the requirements of the degree
of**

MASTER OF SCIENCE

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Abstract

Alperyn, M., M.Sc., University of Manitoba, 2004.

Factors affecting the community ecology of predacious diving beetles (Coleoptera: Dytiscidae) in boreal and prairie ponds across southern Manitoba.

Major professor: R. E. Roughley

Bottle-traps and volume-sampling were effective techniques for collecting dytiscids. A total of 17 060 dytiscids were collected in two years among 30 ponds, using both sampling methods. Conclusions about the factors that organize dytiscid communities were similar, based on the results obtained from both methods of sampling. However, bottle-traps collected elusive, rare, or large species effectively whereas volume-sampling was better at collecting small species. Recommendations for the sampling regimes of future studies are provided.

A primary objective of this study was to relate and rank the environmental variables of boreal and prairie ponds that influence dytiscid diversity. This objective was accomplished by measuring various environmental variables for each pond including pH, conductivity, chlorophyll *a* density, pond permanence, pond area, macrophyte density, emergent vegetation density, and presence of fish and tiger salamanders. The environmental profiles of ponds were compared with their respective dytiscid communities. Multivariate analysis was used to determine the affinities of pond communities and species to the respective environmental variables. Environmental variables important for discriminating between boreal and prairie ponds included macrophyte diversity, pH, water conductivity, chlorophyll *a* density, and pond

permanence. Among these variables, macrophyte diversity was considered a particularly good indicator of dytiscid communities. Ponds that were high in conductivity or that varied in hydroperiod also produced a number of strong associations. Adults of *Hygrotus salinarius* Wallis and *Hygrotus masculinus* Crotch were found only in the most conductive of the ponds examined. Ponds with unstable hydroperiods were inhabited by *Agabus bifarius* Kirby, *Agabus canadensis* Fall, *Agabus punctulatus* Aubé, and *Rhantus consimilis* Motschulsky.

An additional objective was to examine the extent that predatory guilds, including odonate larvae, fish, and grey tiger salamanders (*Ambystoma tigrinum diaboli* Dunn), contributed to the community structure of predacious diving beetles. Boreal ponds had higher densities of odonate larvae than prairie ponds, although predatory pressure by odonates in the boreal ecozone could not be demonstrated. Brook sticklebacks (*Culaea inconstans* Kirtland) and fathead minnows (*Pimephales promelas* Rafinesque) were the two most abundant fish species. Tiger salamanders were found in more than half of the prairie ponds examined. Ponds that were inhabited by tiger salamanders or fish could not be differentiated, based on dytiscid assemblages, from ponds that had neither of these vertebrates.

The final purpose of this study was to compare and contrast the species of dytiscids in boreal and prairie ponds. This objective was addressed by sampling 30 ponds among eight regional localities along a transect across southern Manitoba each month, May through September, over two years. Ponds in boreal plains and boreal ecozones shared more species than boreal plains and prairie communities. Boreal plains ponds were thus treated as boreal ponds. Dytiscid communities in prairie ponds were made up

of ecological generalists. Species found in 25 or more of the 30 ponds included *Hygrotus sayi* Balfour-Browne, *Laccophilus maculosus* Say, *Liodessus obscurellus* LeConte, *Ilybius fraterculus* LeConte, *Coptotomus longulus* LeConte, *Rhantus sericans* Sharp, and *Graphoderus perplexus* Sharp. Boreal ponds had a higher frequency of ecological specialists, including *Desmopachria convexa* Aubé, *Hygrotus farctus* LeConte, *Graphoderus liberus* Say, and *Uvarus granarius* Aubé. Prairie ponds contained more species in greater abundance than boreal ponds.

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INTRODUCTION

Predacious diving beetles (Coleoptera: Dytiscidae) are a diverse family with members adapted to many different types of water bodies. The species richness and abundance of dytiscids within lentic ecosystems are thought to be dependent on a number of variables, including predation, permanence and stability, macrophyte diversity, emergent vegetation, pH, and salinity (Galewski 1971; Larson 1985; Cuppen 1986; Lancaster and Scudder 1987; Larson 1990; Juliano 1991; Nilsson *et al.* 1994; Nilsson and Söderberg 1996; Fairchild *et al.* 2003).

Characteristics of water bodies, such as water chemistry, are generally attributed to the surrounding region. A region can be broadly categorized as an ecozone and defined as an area with characteristic physical features and biota (Wilken 1986). Within its provincial boundary, Manitoba contains a total of five ecozones. A latitudinal transect through the southern portion of the province cuts through three of these ecozones including (from east to west) boreal shield, boreal plains, and prairie (Wilken 1986). In the prairie ecozone, lakes and ponds are typically, ecologically productive waters that are nutrient rich, alkaline, and have varying salinities. Such water also may have unstable hydroperiods due to the arid climate. In contrast, boreal waters are generally less productive, poorer in nutrient content, and acidic. The hydroperiod of boreal waters is more stable than their prairie counterparts. The boreal plains is a transitional ecozone, thus a mixture of environmental features of either adjacent ecozones may influence associated waters. In addition, the proximity of water bodies to neighbouring prairie or boreal ecozones will dictate how close their environmental features resemble the waters of each ecozone.

Interestingly, some of the same environmental variables noted by authors to affect dytiscid communities vary between prairie and boreal waters (Larson 1985). Therefore, the existing environmental variation between prairie and boreal water bodies presents an excellent opportunity to investigate the ecology of these water beetles. The major objective of this study was to compare the composition of dytiscid communities in boreal and prairie ponds. A second goal was to relate and rank the influence of environmental variables within boreal and prairie ponds on dytiscid diversity. The last objective was to examine the extent that predatory guilds, including odonate larvae, fish, and tiger salamanders contributed to community ecology of predacious diving beetles.

LITERATURE REVIEW

Introduction

Ponds are fascinating and important ecosystems. These standing waters serve not only as wildlife habitats but also have significant hydrologic functions (Reimold 1994). Murkin (1998) outlined four functions of standing water in the maintenance and renewal of freshwater supplies, including: (1) control and storage of surface water; (2) recharge of groundwater supplies; (3) sinks for excess nutrients; and (4) filters for sediments and a wide variety of chemicals. Ponds also are recognized as being centres of biological activity (Gerald *et al.* 1984). Ecologists have capitalized on their manageable size and used ponds as models to analyze the interactions of abiotic and biotic factors on community structure (Heino 2000). However, before delving into the ecology of ponds, it is important to consider a definition and classification of these habitats.

What is a pond? A “small lake” is one definition (Reid 1961), referring to the common element of size. Roughley and Larson (1991) also based their definition on surface area, to account for the effect of wave action on the shoreline. Ponds have been defined as bodies of water in which the water temperature does not dramatically vary from the surface to the bottom (Gerald *et al.* 1984). Another definition points out a ponds’ typically shallow depth, “...a body of water that has rooted plants growing across it...” (Reid 1961; p. 13). The littoral zone, or the zone occupied by rooted vascular aquatic plants, is a dominant feature of ponds (Reid 1961).

Ponds also can be classified according to the way they were formed. Potholes or sloughs in the prairie region are examples of water-filled hollows formed by retreating glaciers (Winter 1989). Flowing waters can be converted to ponds. For example, ox-bow

ponds are formed by areas that are cut off from rivers and streams. Ponds can be man-made, as is the case of dug-outs for such purposes as aquaculture or to supply water for fire fighting (Gerald *et al.* 1984).

Zoltai and Vitt (1995) classified ponds based on three gradients important to the ecological processes of these habitats: pH, nutrients (particularly nitrogen and phosphorus), and water fluctuation. Zoltai and Vitt's classification implied that an aquatic community is predictable and that by understanding its constituents, the organization of these communities can be realized. Southwood (1977) adopted the notion of the habitat as an ecological templet, where the most prominent features of the templet would dictate the nature of the community. Thinking in this format poses interesting questions. What are the most important factors of the pond templet? What about other factors that contribute to organizing a community, such as predation producing top-down effects, or limits in primary production producing bottom-up effects? Can some of these factors be generalized and applied to a wider ecological scale, perhaps such as across an entire ecozone?

To answer such questions, ecologists generally use a study group from which patterns can be extrapolated based on the presence, absence, and abundance of certain members of the study group. The Dytiscidae (Coleoptera) is a family that has caught the attention of pond ecologists for their potential utility in understanding some of these questions. Predacious water beetles, or dytiscids, are numerically dominant predators and are taxonomically well known (Larson *et al.* 2000). Sampling in a given aquatic habitat, it is possible to find many species of varying sizes (Larson *et al.* 2000). The co-

occurrence of similar size classes of water beetles also is cause for contemplation into the organizational forces that structure these communities.

The purpose of this literature review is for the reader to gain an understanding of: (1) dytiscid biology, life history and ecology as it applies to their community ecology; (2) to review some of the factors (water chemistry, aquatic macrophytes, predation by fish and odonate larvae) that determine dytiscid community structure; and (3) to predict and examine factors [salinity, algal density, fathead minnows (Cypriniformes: Cyprinidae: *Pimephales promelas* Rafinesque), and grey tiger salamanders (Caudata: Ambystomatidae: *Ambystoma tigrinum diaboli* Dunn)] specific to lentic waters of southern Manitoba that determine aquatic invertebrate communities and also may influence dytiscid communities.

Dytiscid biology, life history, and ecology

Dytiscids inhabit various aquatic habitats (Larson *et al.* 2000). In standing waters, they can often be found in very shallow water, partly because larvae and adults respire at the air-water interface. Adults store air beneath the elytra where the abdominal spiracles are located. They use the air for respiration and sometimes to control their buoyancy (Larson *et al.* 2000). Larvae store air throughout their tracheal trunks. In addition, larvae obtain oxygen through cuticular exchange, and members of one genus, *Coptotomus* Say, possess gills (Larson *et al.* 2000).

Most adult dytiscids have an oval body form that is dorsal-ventrally flattened. This form is streamlined and well suited for swimming. In some species, the body takes on a more globular form, presumably adapted for manoeuvrability, rather than speed

(Ribera *et al.* 1997). In other groups, the body is further dorsal-ventrally flattened, to permit beetles access among vegetation or under rocks (Larson *et al.* 2000).

Adults swim by moving their hind legs in unison. The legs are equipped with setae that increase the surface area of the power stroke. On the return stroke, setae are retracted to reduce drag. Species that are not powerful swimmers have reduced setation on their hind legs, and the overall hind leg is reduced in size. Many adult dytiscids are buoyant and, therefore, need to hold onto substrate or swim to stay submerged. In contrast, most larvae have a specific gravity greater than water, although larvae of species of Dytiscinae are neutrally buoyant (Larson *et al.* 2000).

Many dytiscids are early colonists of aquatic habitats and/or may inhabit temporary habitats, so dispersal by flight is an important aspect of their existence. Wings may not always be functional, depending on the condition of the flight muscles. Muscles may be absorbed depending on the species, populations, and season (Larson *et al.* 2000). Dytiscids cannot fly directly out of water, but have to exit the water before attempting flight (Larson *et al.* 2000).

The life histories of dytiscids vary (Larson *et al.* 2000). Typically, temperate species are univoltine, adults overwinter, mate, oviposit in the spring, and larvae develop during the summer, pupating, and emerging as adults in late summer. Oviposition strategies among dytiscids are diverse. Eggs may be scattered, attached to the substrate, deposited into crevices, or inserted into plant tissue. Female dytiscids that deposit eggs into plants possess a serrated ovipositor (some species of *Agabus* Leach, *Coptotomus* Say, *Cybister* Curtis, *Dytiscus* Linnaeus, *Ilybius* Erichson, *Laccophilus* Leach, and

Thermonectus Dejean) used for cutting (Larson *et al.* 2000). Larvae develop through three larval instars before crawling out of the water to pupate on land.

Predacious diving beetles are carnivorous as larvae and adults. The larvae seize their prey and inject pre-oral digestive enzymes through their large mandibles (Larson *et al.* 2000). The mouth of most dytiscid larvae is sealed. Their liquefied meal is sucked through the mandibles and into the digestive tract (Larson *et al.* 2000). The larvae feed on a variety of prey items, limited by what is available and what they can overpower (Larson *et al.* 2000). There are examples of specialized feeding in dytiscid larvae. For example, some members of Hydroporinae feed on microcrustacea (Matta 1983), and some species of *Dytiscus* are specialized for feeding on caddisfly larvae and tadpoles (Leclair *et al.* 1986). The larvae tend to be stealthy hunters, but adult dytiscids operate more as scavengers than predators, with dead or injured prey making up the greater portion of their diet (Hicks 1994).

Adult beetles are the life stage least affected by predators (Larson *et al.* 2000). Adults are fast swimmers, have hardened bodies, cryptic colouration, and chemical defences that, combined, help them evade predation (Larson *et al.* 2000). Despite these defences, adults are still heavily preyed on by insectivorous fish, which often results in reductions or the absence of dytiscids in waters where fish are present (Larson *et al.* 2000). The larvae of dytiscids have fewer defences and as a result are the food items of a variety of different predators, particularly dragonfly larvae (Larson 1990). Leaving the water makes dytiscids a target of terrestrial predators (e.g. ants and carabid beetles) and parasites (e.g. mites and parasitic wasps) (Larson *et al.* 2000).

Water chemistry in southern Manitoba

Prairie wetlands are unique in their levels of pH and salinity (LaBaugh 1989). These wetlands are typically alkaline with pH values greater than 7.4. The chemical factor responsible for elevating pH in this region is salinity (Hammer 1986). In contrast, in the southeastern portion of the province, in the boreal plains and boreal shield ecozones, waters are more acidic (McKillop 1985). Wetlands in this region typically are low in conductivity (McKillop 1985). Conductivity in prairie wetlands is typically high, but values may range from as low as $42 \mu\text{S cm}^{-1}$ to as high as $472\,000 \mu\text{S cm}^{-1}$ (LaBaugh 1989). In a survey of Manitoban wetlands, Barica (1978) found conductivity to range from 220 to $12\,070 \mu\text{S cm}^{-1}$. Dissolved salts, principally chloride and bicarbonate-carbonate salts of sodium tend to be responsible for elevating conductivity values in prairie ponds (Euliss *et al.* 1999). Cowardin *et al.*'s (1979) classification of saline wetlands, proposed the following categories: fresh ($< 800 \mu\text{S cm}^{-1}$), oligosaline (800 - $8000 \mu\text{S cm}^{-1}$), mesosaline (8 000 - $30\,000 \mu\text{S cm}^{-1}$), polysaline (30 000 - $45\,000 \mu\text{S cm}^{-1}$), eusaline (45 000 - $60\,000 \mu\text{S cm}^{-1}$) and hypersaline ($>60\,000 \mu\text{S cm}^{-1}$).

Aquatic insect response to salinity and pH

Some aquatic insects are able to exploit novel habitats by tolerating adverse hydrological conditions, such as extremes in pH and salinity. These habitats result in low intensities of predation, parasitism, and/or competition (Ward 1992a). For example, *Cenocorixa expleta* Uhler (Hemiptera: Corixidae) was found in saline waters but was excluded from lakes of low salinity (Scudder 1983). Scudder (1983) suspected that this species' preference for saline lakes was an avoidance of parasites, which were excluded

from these habitats. Coping mechanisms for surviving adverse conditions may include a combination of behavioural (e.g. burrowing into substrate), physiological (e.g. increased absorption of water across midgut wall), and structural (e.g. waterproof secretions) adaptations (Scudder 1987). The distribution of species along pH or saline gradients is not always caused by a physiological response (Ward 1992a) but may represent a response to habitat stability, microhabitats, food resources, and/or the availability of oviposition sites (Ward 1992a).

Species richness is typically low in saline habitats but abundance of tolerant taxa may be high (Ward 1992a). Insect orders that possess saline specialists include Hemiptera (especially Corixidae), Coleoptera, Odonata, Trichoptera, and Diptera (Ward 1992a). Rawson and Moore (1944), in a survey of Saskatchewan saline lakes, found that flies and beetles possessed the greatest number of saline specialists. Dytiscids are especially adapted to tolerate saline waters as indicated by their high species richness (Rawson and Moore 1944). A number of authors have indicated that salinity is a good predictor of dytiscid composition (Balfour-Browne 1940; Galewski 1971; Larson 1985; Cuppen 1986). In fact, there is often very little overlap of dytiscid species between fresh and saline waters (Rawson and Moore 1944; Larson 1985). *Hygrotus salinarius* Wallis is the most notable example of a saline specialist in which larvae and adults can live in water more saline than seawater (Tones 1978).

There are few studies of aquatic insects in naturally acidic lentic waters (Ward 1992a). The response of aquatic insect communities to pH is quite similar to salinity: there is an increase in the abundance of a reduced number of taxa toward the endpoints of the pH continuum (Ward 1992a). Galewski (1971) noted that *Hydroporus* Clairville

species seem well adapted for inhabiting acidic *Sphagnum* bogs where the pH may be as low as 4 to 5. Other selected species from the genera *Laccophilus*, *Agabus*, *Ilybius*, *Rhantus* Dejean, *Acilius* Leach, and *Dytiscus* also inhabit acidic bog pools (Galewski 1971). Among pond sites in The Netherlands, pH was a better predictor than conductivity of *Hydroporus* species distribution (Cuppen 1986). However, the response of invertebrate communities to pH is often not direct (Ward 1992a). Associated factors of acidic water, which may have a direct effect on the insect fauna, include changes in physicochemistry (e.g. oxygen deficiency, see Jewell and Brown 1929) and food resources (e.g. reduced algae, see Sutcliffe and Carrick 1973). This phenomenon is probably equally true for dytiscid species. For example, in lab experiments, larvae of *Dytiscus verticalis* Say were unaffected in acidic waters as low as pH 3.0 (Frisbie and Dunson 1987). This species is known to inhabit acidic bogs, but it typically is found in ponds with a rich diversity of macrophytes (Larson *et al.* 2000). This pattern of occurrence suggests that *Dytiscus verticalis* may be responding to factors such as macrophyte diversity in its habitat selection.

Aquatic vegetation

Aquatic macrophytes are vascular plants that grow principally on or below the surface of the water. They are differentiated from emergent vegetation, which grows around the periphery of aquatic habitats (Sharitz and Batzer 1999). Rooted macrophytes grow at a variety of depths although they are restricted to the photic zone (Sharitz and Batzer 1999). Aquatic macrophytes and other aquatic vegetation such as green algae, liverworts, and mosses, may play an equal ecological role in contributing to the structural

complexity of the aquatic habitat. Therefore any reference to aquatic vegetation will also include these non-vascular plant groups.

Dytiscids and aquatic vegetation

A number of authors have commented on the positive correlation between dytiscid diversity and aquatic vegetation diversity (Larson 1985; Nilsson *et al.* 1994; Fairchild *et al.* 2003). Indirectly, aquatic vegetation attracts other invertebrates, which supply dytiscids with a variety of prey items. Macrophytes are attractive to invertebrates for a number of reasons: (1) they serve as a refuge from predators (Gilinsky 1984); (2) they serve as a substrate, allowing invertebrates to use a greater portion of the water column (Rabe and Gibson 1984); (3) they provide an indirect food source for invertebrates that feed on periphyton that grows on the plants (Rabe and Gibson 1984); and (4) they provide a direct food source for a small proportion of invertebrates (Newman 1991). In addition, macrophytes are tapped for oxygen by invertebrates such as mosquitoes, in the genus *Mansonia* Blanchard (Diptera: Culicidae), and the aquatic syrphid, *Chrysogaster* sp. Meigen (Diptera: Syrphidae) (Ward 1992b).

Exactly how dytiscids benefit from aquatic vegetation has not been examined, but dytiscids probably gain shelter from predators within aquatic vegetation. In addition, aquatic vegetation satisfies an important role in the life history of many dytiscid species by serving as oviposition sites (Larson *et al.* 2000).

Algae and aquatic communities

Wetlands are extremely productive habitats (Crumpton 1989). An important contributor to this productivity is algae. It is becoming clear that algae are a far more important food source for primary consumers than macrophytes (Neill and Cornwell 1992; Euliss *et al.* 1999). The low importance of macrophytes in food webs has been known for some time; macrophytes function primarily as substrate. Shelford (1918, p. 47) suggested that one could probably, “substitute glass structures of the same form and surface texture without greatly affecting the immediate food relations”. There are four algal assemblages: plankton, metaphyton, epiphyton and epipelon. Planktonic algae are often referred to as phytoplankton; these algae are suspended in the water column. Periphyton is made up of diatoms, cyanobacteria, flagellates, and unicellular, colonial, and some filamentous algae. Metaphyton is unattached, but loosely associated with the substrate. These algae are the most conspicuous, growing in floating mats and are made up of filamentous algae and cyanobacteria. Periphyton (attached algae) are either epiphytic or epipelic. Epiphytic algae grow attached on plants, whereas epipelic algae grow on sediment.

The production of algae in fertile lakes may be equal to or even surpass that of macrophytes in the littoral zones (Jones 1984; Allen 1971). Unlike macrophytes, algae can respond quickly to nutrient fluctuations. Without root systems, algae obtain their nutrients directly from the surrounding water. Nutrient pulses can lead to an explosive population increase in algal abundance, the so-called “algal bloom”. This can be extensive enough to shade out macrophytes (Hann and Goldsborough 1997). In the prairie region, the seasonal pattern of algae generally involves as many as three blooms,

occurring in the spring, summer, and fall. The spring bloom occurs when temperature and light conditions become ideal, resulting in a surge in the production of diatoms.

Chlorophytes and cyanobacteria give rise to blooms in the mid-summer months. Algal blooms are often exacerbated by agricultural practices which contribute nutrients in the form of run-off (Crumpton 1989).

Algal density is a function of nutrient availability and it is correlated with invertebrate communities (Murkin *et al.* 1994; Hann and Goldsborough 1997; Sandilands *et al.* 2000). Algal density has been positively correlated with invertebrate abundance (Campeau *et al.* 1994). As predators, dytiscids would be terminal units in many of the associated food webs. Although the relationship between dytiscids and algal density has not been examined, it is probable that elevated primary production is met with increases in abundance at higher trophic levels (Campeau *et al.* 1994; Hann and Goldsborough 1997).

Predator guilds

Dragonfly larvae

Odonates are divided into two suborders: Zygoptera (damselflies) and Anisoptera (dragonflies) (Walker 1953). Within littoral zones, dragonflies are often the dominant predators (Benke and Benke 1975) and are taxonomically well known for Canada (Walker 1958, Walker and Corbet 1975). One-hundred and forty-three species of adult odonates are known for Canada; the immature, aquatic stages (larvae) are known for 133 species (Corbet 1979). Ninety-five species are known from Manitoba (Walker 1958, Walker and Corbet 1975, Conroy and Kuhn 1978; Hughes and Duncan 2003).

The immature stages of dragonflies advance through 10-15 instars and are exclusively aquatic during this period (Westfall and Tennesen 1996). Dragonflies are uni- and semi-voltine, although dragonflies in northern latitudes tend to be semi-voltine. Some larvae may take more than five years to complete their life cycle (Westfall and Tennesen 1996). Thus, dragonfly larvae tend to be excluded from habitats that are subject to seasonal drying (Larson and House 1990).

Dragonfly larvae are often the most diverse predacious insects ($>1000/m^2$; 10-20 coexisting species) in the littoral zone of permanent lentic habitats (Wissinger 1989). In the absence of fish, dragonfly larvae may be keystone predators (Larson and House 1990). Several authors have examined the change in community ecology of various invertebrate groups as a result of odonate predation (Pritchard 1964; Benke 1978; Thorp and Cothran 1984; Wissinger 1989; Larson and House 1990; McPeck 1990). Predation by odonates generally results in a decline in species abundance rather than species richness within invertebrate communities (Larson and House 1990; Larson 1990; Thorp and Cothran 1984).

Odonates are regarded as keystone predators in fishless, permanent ponds, but dytiscids occupy this ecological role in temporary water bodies (Larson 1990). It is suspected that this habitat partitioning has evolved, in part, by the predatory pressure by odonates on dytiscids (Larson 1990). As a result, dytiscids have adapted to colonize unstable habitats that do not favour odonate survivorship. However, ecological relationships are rarely sharply defined; thus dytiscids do co-occur with odonates to varying degrees in various habitats. To what extent does the predatory pressure exerted by odonate larvae affect the community ecology of dytiscids? This question was posed by

Larson and Colbo (1983) who suggested that odonates may be important predators of dytiscids, and the question was further investigated by Larson (1990). Larson's (1990) examined Newfoundland bog pools and found that: (1) dytiscid and odonate densities were inversely correlated; (2) size of adult dytiscids was positively correlated to pool size, in which odonates were speculated to exclude small dytiscids from large pools; and (3) at least one specimen of all species of dragonfly larvae examined had dytiscid larvae in their guts. These findings provide support for the hypothesis that odonate larvae may alter dytiscid communities.

Fish

Freshwater fish are generally the keystone predator in aquatic habitats and thus are the major organizing force within a community (Connell 1975). However, the predatory force that fish exert on an invertebrate community varies with the species (both fish and invertebrate), the season and habitat (Gilinsky 1984).

The ecological role of various fish species in relationship to invertebrates (at least in the higher trophic levels) probably represents a continuum, from predator to competitor to prey. Insectivory may approach 100% in some species of trout (Duffield and Nelson 1993). Brook sticklebacks (Gasterosteiformes: Gasterosteidae: *Culaea inconstans* Kirtland) compete with many predacious aquatic insects for prey items such as chironomid larvae, an important dietary component of sticklebacks (Tompkins and Gee 1983). Even the early life stages of large game fish, such as northern pike (Esociformes: Esocidae: *Esox lucius* Linnaeus) are often susceptible to predation by

aquatic insects (notonectids, naucorids, odonate larvae, and dytiscid adults) (Louarn and Cloarec 1997).

The predatory pressure that fish exercise tends to be seasonal. In northern latitudes fish may be piscivorous in the winter and then feed almost exclusively on invertebrates in the summer (Jacobsen *et al.* 2002). This relationship is a result of a decrease in availability of invertebrates during winter months (Jacobsen *et al.* 2002).

Habitat complexity plays an important factor governing predation efficiency and also the type of prey that is seized. In aquatic habitats, macrophytes increase spatial complexity (Crowder and Cooper 1982; Gilinsky 1984). Crowder and Cooper (1982) found that bluegill sunfish (Perciformes: Centrarchidae *Lepomis macrochirus* Rafinesque) predation on invertebrates at low and intermediate macrophyte densities was greater than in high densities of macrophytes. In addition, bluegills were more likely to capture larger (odonate larvae) or more mobile prey (corixids) in high-density treatments.

Fish in southern Manitoba

There are over 95 species of freshwater fishes in Manitoba (Stewart and Watkinson 2004). However, conditions in shallow pond environments in southern Manitoba, generally, do not favour fish (Euliss *et al.* 1999). Many fish are prevented from establishing permanent populations in Manitoba ponds due to frequent drying, nonintegrated watersheds, and harsh winter conditions (Euliss *et al.* 1999). Brook sticklebacks (*Culaea inconstans*) and fathead minnows (*Pimephales promelas*) are the most common, naturally occurring species in prairie regions (Peterka 1989). Fatheads are

also introduced deliberately into ponds in this region for the baitfish industry (Hanson and Riggs 1995).

The southeastern portion of Manitoba is a transition into the southern reaches of the boreal plains and, further east, into the boreal ecozone. In comparison to the prairie region, boreal wetlands are equivalently harsh habitats for fish, being characteristically acidic and oxygen-limited during winter months. The fish assemblage in these shallow water bodies includes central mudminnows [Esociformes: Umbridae: *Umbra limi* (Kirtland)], Iowa darter [Perciformes: Percidae: *Etheostoma exile* (Girard)], and redbelly dace (Cypriniformes: Cyprinidae: *Phoxinus eos* Cope), along with brook stickleback and fathead minnows (Tompkins and Gee 1983). Brook sticklebacks and fathead minnows appear to be the dominant fish in the wetlands of this region (M. Alperyn, personal observation).

In habitats where fathead minnows and brook sticklebacks co-occur, fatheads are usually dominant in terms of numbers of individuals and biomass and are often the keystone predator (Zimmer *et al.* 2000). A number of authors have examined the effects of fatheads minnows on invertebrate communities (Price *et al.* 1991; Hanson and Riggs 1995; Duffy 1998; Zimmer *et al.* 2001). Fathead minnows affect wetlands because of their diverse diets and dense populations (Duffy 1998; Zimmer *et al.* 2000). Fathead minnows alter a wetland directly through their consumption of prey and indirectly through the cascading effects of their excrement and nutrient transport (Zimmer *et al.* 2001). Wetlands supporting fathead minnow populations show increases in turbidity, chlorophyll *a* and phosphorus and have lowered aquatic macrophyte abundance (Zimmer *et al.* 2001). Fathead minnows reduce the abundance of invertebrates and may play a

major role in structuring invertebrate communities (Zimmer *et al.* 2000, Zimmer *et al.* 2001).

The effect of other small-bodied fish, such as brook sticklebacks and mudminnows, on invertebrate communities is less clear. In comparison to fathead minnows, brook sticklebacks, feed on smaller invertebrates, such as *Daphnia* (Cladocera: Daphnidae), and appear to be more prey selective (Winn 1960; Laurich *et al.* 2003). Mudminnows are similar to fatheads, feeding as generalists on a wide variety of invertebrates (Peckham and Dineen 1957). In addition, mudminnows are tolerant of harsh conditions, surviving hypoxic and extreme temperature conditions (Peckham and Dineen 1957; Rahel and Nutzman 1994). These characteristics predispose the mudminnow to exert a predatory force on wetland communities.

Wilson (1923) extensively examined the influence of fish predation on dytiscid communities in fishponds. He found that species richness and abundance were negatively impacted by fish predation (Wilson 1923). This finding agrees with other authors who have observed reductions in dytiscid diversity from large water bodies (Galewski 1971; Larson 1990). A possible explanation for this phenomenon is that large bodies of water are more likely to be permanent and thus more likely to contain fish than smaller, less hydrologically stable habitats. Wilson (1923) examined predation by bluegills (*Lepomis macrochirus*), black crappie [Perciformes: Centrarchidae: *Pomoxis nigromaculatus* (Lesueur)], largemouth bass (Perciformes: Centrarchidae: *Micropterus salmoides* (Lacépède), and other large carnivorous fish on aquatic invertebrates. Such species can hardly be compared to smaller-bodied fish such as fathead minnows, brook sticklebacks, and central mudminnows, most common in wetlands of southern Manitoba. Central

mudminnows and probably fathead minnows consume dytiscids (Peckham and Dineen 1957; K. D. Zimmer, North Dakota State University, personal communication 2004). Zimmer *et al.* (2000) examined invertebrate communities and found that dytiscid community structure was affected by the presence of fathead minnows. However, Zimmer *et al.* (2000) and other studies (Price *et al.* 1990; Mallory *et al.* 1994; Zimmer *et al.* 2001) were designed to examine the gross effects of fish predation, and therefore to examine the inhabitants of aquatic communities at a low taxonomic resolution. Therefore, potential shifts of taxa in the community structure within a particular family of invertebrates, such as Dytiscidae, have been overlooked.

Tiger salamanders

The grey tiger salamander (*Ambystoma tigrinum diaboli*) is the more common of the two subspecies occurring in southern Manitoba (USGS 2003; Larson 1968). The grey tiger salamander is distributed across much of southwestern Manitoba, while the eastern tiger salamander, *Ambystoma tigrinum tigrinum* (Green) occupies a small portion of the southeastern corner of the province (USGS 2003).

Metamorphic (terrestrial) tiger salamanders overwinter in surrounding forested areas of nearby aquatic habitats (Whiteman *et al.* 1994). In late spring, they migrate to permanent and semipermanent ponds where they mate and remain until autumn before returning to overwintering sites. Eggs are deposited on aquatic vegetation and hatch in mid-July. Development of the larvae into metamorphic and sexually mature salamanders may take two to five growing seasons (Wissinger *et al.* 1999). However, some

individuals forgo metamorphosis completely and become sexually mature as larvae (i.e. become paedomorphic adults) (Whiteman *et al.* 1994).

Tiger salamanders are nocturnal, floating up from their benthic retreats to feed on invertebrates in the water column (Anderson and Graham 1967). They are opportunistic predators (Norris 1989). Wissinger *et al.* (1999) found that almost all pond invertebrates sampled with aquatic nets were represented in the stomach contents of tiger salamanders. When describing their feeding habits, Zaret (1980, p. 44) called tiger salamanders “fish in amphibian garments”. In Manitoba, their ecological role has been compared with dabbling ducks (Benoy *et al.* 2002) and rainbow trout (Salmonidae: Salmoniformes: *Oncorhynchus mykiss* Walbaum; Olenick and Gee 1981). Maximum annual production of tiger salamanders was estimated at $565 \text{ kg}\cdot\text{ha}^{-1}$ in prairie lakes of North Dakota (Deutschman and Peterka 1988). This production is comparable to the production estimates of fathead minnows ($144 - 482 \text{ kg}\cdot\text{ha}^{-1}$), which are known to alter invertebrate communities (Duffy 1998). Similar to fish and ducks, tiger salamanders are capable of affecting invertebrate communities through predation (Dobson 1970; Sprules 1972; Holomuzki *et al.* 1994). Tiger salamander densities were correlated negatively with mayfly and odonate larvae abundance (Holomuzki *et al.* 1994). In enclosure experiments, tiger salamanders reduced the density of zooplankton (Holomuzki *et al.* 1994).

Dytiscid and tiger salamander predator-prey interactions have been reported (Holomuzki 1986). Tiger salamander larvae altered their microhabitat usage in the presence of *Dytiscus dauricus* Gebler larvae (Holomuzki 1986). This behaviour was suspected to be an attempt to avoid predation by beetle larvae (Holomuzki 1986). However, large dytiscids such as *D. dauricus* make up a small percentage of a typical

community (Larson 1985). In addition, numerous authors have reported adults and larvae of water beetles in the guts of tiger salamanders (Dobson and Dobson 1971; Olenick and Gee 1981; Collins and Holomuzki 1984; Holomuzki and Collins 1987; Benoy *et al.* 2002). Therefore, it is more likely that the predatory pressure is enforced by tiger salamanders. Studies involving tiger salamanders, like fish, have been focused on the predatory effects at higher taxonomic levels. Therefore, a potential alteration in the community structure of Dytiscidae due to tiger salamander predation is probable, but has been overlooked.

Summary

Predacious diving beetles communities in ponds are governed by various environmental factors. Salinity and pH are factors that primarily affect species richness. Ponds at the extremes of these variables have a modified or selected fauna. Predation and primary productivity mainly regulate dytiscid species abundance by affecting the community through top-down and bottom-up interactions, respectively. Environmental factors such as diversity in aquatic vegetation are positively correlated with dytiscid species richness and abundance.

The factors that influence dytiscid communities can be examined on a broader ecological scale. What are the factors influencing dytiscid communities in ponds across southern Manitoba? Brook sticklebacks and fathead minnows occur in ponds across southern Manitoba and, therefore, so does their potential to impact dytiscid communities. The influence of other factors may be more localized. Odonate larvae, for example, can be found in high densities in ponds with stable hydroperiods. Such ponds are typically found in the boreal region of the province, so the impact of odonate as predators of dytiscids would be expected to be more pronounced in this ecoregion than in other ecoregions. Finally, factors such as pH may be dichotomously represented across the province and correspondingly affect dytiscid communities. For example, ponds in the eastern, boreal region are generally more acidic, favouring acidophilic species, whereas western, prairie ponds have higher pH values favouring species adapted to such waters.

Ponds can be viewed as experimental replications due to their differences and similarities in environmental variables. This variability allows the experimenter to select ponds that are similar in many respects but differ in relation to a few environmental

variables. By sampling the dytiscid communities of ponds and comparing them with particular environmental variables, maybe it is possible to account for differences in species diversity. Southern Manitoba is an excellent region for such a study due to its large number of ponds, dytiscid diversity, and contrasting boreal and prairie ecozones.

MATERIALS AND METHODS

Survey design

Dytiscid communities were examined in ponds across southern Manitoba. The study was designed to examine some of the environmental factors within a given pond that influence dytiscid diversity. Environmental factors that were evaluated were pond size, pH, conductivity, aquatic vegetation diversity, emergent vegetation density, algal density, presence of certain fish species, presence of tiger salamanders, and odonate larval density.

Description of the general study area

The sampling was conducted in 2001 and 2002 in the southern region of Manitoba between latitudes 49°17'N and 49°43'N. A transect across southern Manitoba was subdivided into eight regional localities. In this study, a locality was designated as an area in which a number of ponds were: (1) close in proximity (all ponds within a 5 km radius) and (2) varied in terms of size and/or the presence/absence of fish. Each locality consisted of three to five ponds for a total of 30. Two localities were located within the boreal ecozone, one in the boreal plains ecozone and five localities were located in the prairie ecozone of southern Manitoba (Fig. 1).

Pond selection

A pond was defined by Roughley and Larson (1991; p. 27) as a “small permanent body of water; with small enough surface area that wave action does not significantly affect shoreline so that fringing organic matter and emergent plants occur; generally with an open-water central area”. By this definition aquatic habitats that are typically known as ‘small lakes’ (which will subsequently be referred to as ponds) were included for comparison purposes to smaller water bodies (traditionally known as ponds). A large variety of water bodies are classified as ponds using this definition, including reservoirs (e.g. Paterson and Fernando 1969), aestival ponds (Daborn 1974), and saline ponds (Hammer *et al.* 1975).

In May, 2001, survey trips located and assessed ponds for inclusion in the study. The initial selection of localities was made with Energy, Mines and Resources Canada 1:50 000 maps to find areas with clusters of neighbouring ponds. A global positioning system (GPS; Garmin[®] 12 XL model) was used to find their locations. Over 60 ponds were considered for the study, and from these, 30 ponds among eight localities were selected (Table 1). Ponds within a locality were selected so that they could be compared within a locality and compared to ponds in other localities. Criteria for ponds included accessibility, size, distance to other ponds within a given locality, minimal impact by agricultural practices and/or other anthropogenic activities, and presence/absence of fish (seven of the eight localities included at least one pond in which at least one fish species was present).

Pond specifications

The geographical co-ordinates and elevation of each pond were recorded using the GPS. Pond surface area was calculated by walking the perimeter of the pond, entering waypoints around the pond margin and using the “calculate area” function contained in the GPS. Surface area of ponds >500 m² was extrapolated from maps. Water samples were taken from each pond each month, from May through September, and measured in the laboratory for conductivity and pH. A Hanna Instruments model 9812[®] portable pH meter was used to measure pH. Conductivity was measured using a Bach-Simpson model CDM3[®] conductivity metre.

Algal collection and quantification

Water samples were collected and filtered for algae each month, from May through September in 2001 only. Samples were collected by dipping a one-litre Nalgene[™] bottle near the surface in open areas of the ponds. Generally 0.4 L of water provided a sufficient biomass, although on occasions when algal densities were low, greater volumes of water were needed. Water samples were filtered in the field through 47 mm, glass fiber filters (Whatman[®] GFC) using a Nalgene[™] vacuum filtering apparatus with a hand operated Nalgene[™] vacuum pump at approximately 100 mm Hg vacuum. Filtered volumes were recorded. After filtration, filters were folded and placed into 20

mL glass vials. Filtered samples were stored in the dark and kept frozen. These filtered samples were transferred to 20 x 150 mm test tubes and extracted in 6 mL of 90% methanol. Test tubes were covered with Parafilm M™ and refrigerated at 4-5 °C in the dark for at least 12 hours. Chlorophyll extracts were then filtered through 25 mm diameter glass fibre filters. Glass fibre sample filters were rinsed with 4 mL of 90% methanol, combined with the extract and the final volume was adjusted to 10 mL with methanol. An aliquot of each filtered extract was pipetted into a 10 cm path length cuvette and the optical density (OD) was read using a Jenway® model 6300 spectrophotometer at three different wavelengths [650, 665, and 750 nanometers (nm)]. Any interference due to sample extract turbidity was corrected by subtracting the OD at 750 nm from the OD readings at 650 and 665 nm, respectively. Algal density was expressed as micrograms of chlorophyll *a* per litre (µg chlorophyll *a*/L) and calculated from the OD measured readings using the following equation and extinction coefficients (MacKinney, 1941):

$$\mu\text{g chlorophyll } a/\text{L} = \frac{(1.65 \times \text{OD}_{665} - 0.83 \times \text{OD}_{650})(\text{vol. of extract, mL})}{(\text{vol. of sample, L}),}$$

where **OD₆₆₅** is the optical density reading at 665 nm and **OD₆₅₀** is the optical density reading at 650 nm. The extinction coefficients 1.65 at 665 nm and 0.83 at 650 nm were according to MacKinney (1941). The **vol. of extract, mL** equalled 10 mL for all the samples and the **vol. of sample, L** was the volume of water, in litres, that was initially filtered in the field.

Between samples, the cuvette was rinsed with 90% methanol to avoid cross contamination. In addition, the spectrophotometer was re-calibrated after every five samples using a blank sample (cuvette filled with only methanol).

Pond permanence

Efforts were made to include only permanent ponds in the study; that is, ponds that would contain water throughout the entire year, for at least both study years. However, some ponds dried out at some point in the sampling period, either in the first or the second study year or both. Therefore, pond permanence was ranked on a 6 point scale (0-5). A point was given for each sampling month that each pond contained water. Ponds were ranked separately for each study year.

Survey of submergent vegetation and emergent vegetation

Submergent vegetation was defined as macro-vegetation that was partially or completely submerged. This definition was not limited to vascular plants, so macroalgae and liverworts also were considered. Emergent vegetation was defined as rooted plants that had parts extending above the water surface, such as *Typha latifolia* Linnaeus (Typhales: Typhaceae). Representative samples of aquatic plants were pressed with a plant-press and returned to the laboratory for identification. All aquatic plants were identified to species. Submergent vegetation and emergent vegetation were surveyed in all 30 ponds once in August, 2001.

Submergent vegetation was assessed on a 6-point scale (0-5) to determine its diversity for the entire pond. For example, ponds that had either one or two dominant aquatic plant species were scored a "1", ponds that had either three or four dominant plant species were scored a "2" and so forth. The entire pond was visually scanned and scored for the number of dominant aquatic plant species present. For ponds that were $>500 \text{ m}^2$ and, therefore, sub-sampled, only the sub-sampled portion was surveyed for submergent vegetation. Dominant aquatic plant species were classified as plants that were present throughout the entire pond and in relatively high densities.

Emergent vegetation was assessed on an 11-point scale (0-10). The entire pond shoreline was visually scanned and the proportion of emergent vegetation bordering the pond was scored. For ponds that were $>500 \text{ m}^2$ and, therefore, sub-sampled, only the sub-sampled portion was surveyed for emergent vegetation.

Faunal sampling techniques

Ponds were divided into four sectors (A, B, C, and D) corresponding to compass points (N, E, S, and W) prior to sampling (Fig. 2). Ponds that were $>20\,000\text{ m}^2$ in size were sub-sampled in such a way that a smaller, representative portion of the pond was selected and divided into the same four sectors. Larger ponds were sampled only in these smaller sectors.

Bottle-trap sampling

Bottle-trap sampling was conducted each month, May through September, for years 2001 and 2002. Bottle-traps were used to collect adult dytiscids. These bottle-traps were constructed out of two, two-litre soft drink bottles (Fig. 3). Each bottle-trap was weighted with a piece of lead, and floating cork markers were attached with fishing line to aid in its retrieval. One bottle-trap was placed into each sector (A to D) of the pond at a depth of $\sim 20\text{ cm}$ and oriented parallel with the shoreline (Aiken and Roughley, 1985). The bottle-trap contents were collected after a 24 h period. Dytiscid catches were preserved in a 70% ethanol and returned to the laboratory for identification.

This method samples large, mobile species that may be elusive and in low abundance when sampled with discrete volumetric methods. However, bottle-trapping has several drawbacks when compared to volumetric methods: the abundance of guilds of large-sized dytiscids are overestimated; small, less mobile dytiscid species are underestimated; and odonate larvae are sampled ineffectively (Gundersen *et al.* 2002). To overcome these shortcomings, volumetric sampling was employed in conjunction with bottle-trap sampling.

Volumetric sampling

Volume-sampling was conducted each month, May through September, for year 2001 only. This method was used to obtain accurate, quantitative samples of adult dytiscids and odonate larvae. Volumetric samples were taken with an aluminium garbage can (diameter = 42 cm; length = 90 cm), the bottom of which was cut out and reinforced with a steel band. When the bottom of the apparatus was placed firmly into the substrate of the pond it created a cylinder of defined volume (Fig. 4). With the volume-sampler in

place, the contents were swept with a D-frame aquatic net (400 x 800 µm mesh size) and placed into a sorting tray. This procedure was repeated until no more dytiscids and odonate larvae were collected. Dytiscids and odonate larvae were then sorted, preserved in a 70% ethanol and returned to the laboratory for identification. These samples provided a comparison to the data provided by the bottle-traps. Two volume-samples were taken from each pond sector (A to D) at different depths, following a stratified random approach. One sample was taken at the pond margin and the second was taken at an approximate water depth of 60 cm (+/- 15 cm).

Seine net sampling

Seine net samples were used to determine the relative abundance of fish and tiger salamanders in each of the study ponds. Seine net sampling was done once in August, 2001 for all 30 sites. The seine net used was 1.8 m by 12.24 m and the mesh size was 1.27 cm. Seine netting was conducted with two people, in which one person at the bank would be unravelling the net while another person in snorkelling gear would tow the net out. The leading end of the net was then brought back to the same bank, forming a horseshoe, the net was then brought to shore, and the catch was transferred to a holding bin. Specimens were killed by a tricaine methane sulfonate (MS-222) overdose (2 g/L) (University of Manitoba Animal Care permit #F01-028). Salamanders were also pithed to ensure humane death. All specimens were preserved in 70% ethanol and returned to the laboratory for identification.

Identifications

All organisms were identified in the laboratory, using a Heerbrugg Wild M3B (40x) dissecting microscope. Adult dytiscids were identified to species using Larson *et al.* (2000). Odonate larvae were identified to family level using Westfall and Tennesen (1996). Salamanders and fish were identified to species. Fish were identified using Fedoruk (1971). A voucher collection of dytiscids, odonate larvae, fish, and salamanders is deposited in the J. B. Wallis Museum, Department of Entomology, University of Manitoba.

Data analysis

Dytiscid species diversity index and evenness

Dytiscid diversity for each pond was examined to understand how it varied in ponds among selected (boreal, boreal plains and prairie) ecozones. Among the three ecozones, boreal ponds and boreal plains ponds were the most similar based on dytiscid species composition. Therefore, in all subsequent analyses in which ecozone was treated as a grouping variable, boreal and boreal plains ponds were treated as one, and compared to prairie ponds. Dytiscid diversity was calculated for each pond and each sampling technique using Hill's reformulation for Renyi's entropy of order two:

$$N2 = \left[\sum_{i=1}^s p_i^2 \right]^{-1},$$

where p_i is the proportion of individuals belonging to the i th species and \sum denotes the summation from $i = 1$ to $i = s$, with s being the number of species in the sample (Kvålseth 1991).

Dytiscid evenness was calculated for each pond and each sampling technique using the evenness measure that was used was proposed by Kvålseth (1991):

$$E = (N2-1)/(N1-1),$$

where $N1$ is calculated by the following formula,

$$N1 = \exp(-\sum p_i \log_e p_i).$$

To quantify and qualify the similarities and differences between boreal and prairie ponds based on dytiscid diversity and evenness, two group t-tests were performed on all three data sets using SYSTAT version 10.

Cluster Analysis

Cluster analysis was used to determine how ponds grouped together based on similarity in dytiscid species abundance, richness, and composition. Cluster analysis was done using SYN-TAX 2000 software, within the Hierclus module. The clustering method used was sum of squares, based on chord distance, performed on all three logarithmically transformed species data sets.

Multivariate Analysis

A multivariate approach, using CANOCO version 4.5 for Windows (ter Braak and Šmilauer 1998), was used to determine: (1) how specific environmental variables (e.g. pH, macrophyte diversity, etc.) in each pond influenced the corresponding dytiscid communities, and (2) how attributes of the ecozone influenced the dytiscid communities of each pond.

Redundancy analysis (RDA) was done on all three logarithmically transformed dytiscid data sets. The environmental variables that were used with the volume and bottle-trap dytiscid data sets in 2001, included pH, \log_{10} (conductivity), \log_{10} (chlorophyll

a density), pond permanence, pond area, macrophyte density, emergent vegetation density, and presence of fish and tiger salamanders. Fish and tiger salamanders were considered present if they were collected either by volume, bottle-trap, or seine net sampling. Presence of fish and of tiger salamanders were the only environmental variables that were not treated as continuous variables. All environmental variables, except chlorophyll *a* density, that were used with the 2001 data sets, were also used in the single 2002 dytiscid data set. In the RDA, three environmental variables were forward selected to determine which variables best explained the trends in the species data. Monte Carlo permutation tests were used to determine the significance of the relationships among the three environmental variables and the species data. For each Monte Carlo test, 199 permutations were used (ter Braak and Šmilauer, 1998).

Redundancy analysis was also used to examine the potential effect of odonate larvae on dytiscid communities in prairie and boreal ponds. The odonate larval data set was logarithmically transformed, treated as an environmental variable, and included the abundances of six families of Odonata collected: Lestidae, Coenagrionidae, Gomphidae, Aeshnidae, Corduliidae, and Libellulidae.

RESULTS

Dytiscid species data set

Species of adult dytiscids collected by volume-sampling were combined (eight samples per pond) for all four months of collecting (May – August). Collections by bottle-trap sampling were also combined (four samples per pond) for all five months of collecting (May – September). The result was two dytiscid data sets obtained from 2001, corresponding to collecting techniques, bottle-traps (Appendix I) and volume-sampling (Appendix II) from 30 ponds. The collections by bottle-trap sampling for 2002 were also combined (four samples per pond) for all five months of collecting (May – September) (Appendix III).

Dytiscid diversity

Dytiscids collected using volume-sampling and bottle-traps in 2001 and bottle-traps in 2002 are summarized in Table 2. A total number of 17 060 dytiscids were collected representing 74 species during this study. The ten most common species made up 79.5 % of the total number of dytiscids collected. These ten species included (in descending order of abundance): *Hygrotus sayi* Balfour-Browne, *Laccophilus biguttatus* Kirby, *Rhantus sericans* Sharp, *Agabus antennatus* Leech, *Laccophilus maculosus* Say, *Liodesus obscurellus* (LeConte), *Agabus ajax* Fall, *Neoporus superioris* (Balfour-Browne), *Coptotomus longulus* LeConte, and *Neoporus undulatus* (Say).

There were 26 species found only in prairie ponds and 13 species that were only in boreal ponds (Table 3). Among the set of species that was unique to boreal ponds was *Hydrovatus pustulatus* (Melsheimer). This find is the first published occurrence of this

species in Manitoba. For the three boreal plains and eight boreal ponds, a total of 2285 dytiscids were collected. The top ten species of that total made up 87.4% and included species, in descending order: *Hygrotus sayi*, *Coptotomus longulus*, *Laccophilus maculosus*, *Acilius semisulcatus* Aubé, *Liodessus obscurellus*, *Desmopachria convexa* (Aubé), *Hygrotus farctus* (LeConte), *Neoporus undulatus*, *Uvarus granarius* (Aubé), and *Graphoderus liberus* (Say). In prairie ponds, a total of 14775 dytiscids were collected. The top ten species of that total made up 82.8% and included (in descending order of abundance): *Laccophilus biguttatus*, *Hygrotus sayi*, *Rhantus sericans*, *Agabus antennatus*, *Laccophilus maculosus*, *Liodessus obscurellus*, *Agabus ajax*, *Neoporus superioris*, *Hygrotus impressopunctatus* (Schaller), and *Neoporus undulatus*.

Volume-sampling of dytiscids, 2001

A total of 5080 dytiscid specimens was collected using volume-sampling for all 30 ponds in 2001. The ten most common species made up 81.2 % of the total number of dytiscids collected. The ten most common species collected included (in descending order of abundance): *Hygrotus sayi*, *Liodessus obscurellus*, *Neoporus undulatus*, *Laccophilus biguttatus*, *Rhantus sericans*, *Agabus antennatus*, *Coptotomus longulus*, *Neoporus superioris*, *Laccophilus maculosus*, and *Hygrotus impressopunctatus*. A total of 1273 dytiscids was collected in the boreal ponds. The top ten species of that total made up 91.4% and included (in descending order of abundance): *Hygrotus sayi*, *Coptotomus longulus*, *Desmopachria convexa*, *Liodessus obscurellus*, *Laccophilus maculosus*, *Hygrotus farctus*, *Uvarus granarius*, *Neoporus undulatus*, *Acilius semisulcatus*, and *Hydroporus dentellus* Fall. A total of 3807 dytiscids was collected from the prairie ponds.

The top ten species of that total made up 84.2% and included the following species (in descending order of abundance): *Liodesus obscurellus*, *Hygrotus sayi*, *Laccophilus biguttatus*, *Neoporus undulatus*, *Agabus antennatus*, *Rhantus sericans*, *Neoporus superioris*, *Laccophilus maculosus*, *Hygrotus impressopunctatus*, and *Hydroporus fuscipennis* Schaum.

Species richness was averaged for boreal and prairie ponds. The two groups of ponds were compared using a two-group t-test with a confidence of 95%. Boreal ponds, on average, had fewer species ($s=14 \pm 4.3$) than prairie ponds ($s=17 \pm 5.0$), although the difference was not significant ($t=-1.7$, $df=24$, $p=0.10$) (Fig. 5). The Hill's effective species richness diversity index (N_2) was calculated for each pond. Prairie ponds, on average, had an N_2 value of 6.0 ± 1.8 , whereas boreal ponds on average were less diverse, $N_2 = 4.0 \pm 1.6$. Prairie ponds were significantly more diverse than boreal ponds ($t=-3.1$, $df=23$, $p=0.0051$) (Fig. 6). Prairie ponds, on average, had a Kvålseth evenness value of 0.67 ± 0.066 compared to 0.59 ± 0.12 in boreal ponds. The difference between these two groups of ponds was not significant ($t=-2.1$, $df=14$, $p=0.051$) (Fig. 7).

Bottle-trap sampling of dytiscids, 2001

A total of 5391 dytiscid specimens was collected using bottle-traps for all 30 ponds in year 2001. The ten most common species made up 81.3 % of the total dytiscids collected. The ten most collected species using bottle-traps included (in descending order abundance): *Hygrotus sayi*, *Laccophilus biguttatus*, *Rhantus sericans*, *Agabus ajax*, *Agabus antennatus*, *Laccophilus maculosus*, *Ilybius fraterculus* (LeConte), and *Graphoderus perplexus* Sharp, *Neoporus superioris*, and *Coptotomus longulus*. A total of

608 dytiscids were collected in the boreal ponds. The top ten species of that total made up 89.1% and included the following species (abundances in descending order): *Hygrotus sayi*, *Coptotomus longulus*, *Acilius semisulcatus*, *Liodesus obscurellus*, *Laccophilus maculosus*, *Graphoderus liberus*, *Graphoderus perplexus*, *Neoporus undulatus*, *Ilybius subaeneus* (Erichson), and *Rhantus binotatus* (Harris). A total of 4783 dytiscids was collected from the prairie ponds. The top ten species of that total made up 85.5% and included the following species (in descending order of abundance): *Laccophilus biguttatus*, *Rhantus sericans*, *Hygrotus sayi*, *Agabus ajax*, *Agabus antennatus*, *Laccophilus maculosus*, *Ilybius fraterculus*, *Graphoderus perplexus*, *Neoporus superiorus*, and *Graphoderus occidentalis* Horn.

Boreal ponds had an average species richness of 9 ± 4.0 compared to an average species richness of 20 ± 3.6 in prairie ponds. This difference was significant ($t=-7.3$, $df=20$, $p>0.001$) (Fig. 8). Prairie ponds on average had an N_2 value of 6.2 ± 2.3 , whereas boreal ponds on average were less diverse, $N_2=4.2 \pm 1.7$. Prairie ponds were significantly more diverse than boreal ponds ($t=-2.7$, $df=27$, $p=0.013$) (Fig. 9). Prairie ponds, on average, had an evenness value of 0.62 ± 0.12 compared to 0.70 ± 0.12 in boreal ponds. This difference was not significant ($t=1.8$, $df=21$, $p=0.080$) (Fig. 10).

Bottle-trap sampling of dytiscids, 2002

A total of 5666 dytiscid specimens was collected using bottle-traps for all 30 ponds in 2002. The ten most common species made up 85.9 % of the total dytiscids collected. The ten most frequently collected species using bottle-traps included (in descending order of abundance): *Laccophilus biguttatus*, *Hygrotus sayi*, *Rhantus sericans*, *Agabus antennatus*, *Laccophilus maculosus*, *Neoporus superiorus*, *Hygrotus*

impressopunctatus, *Coptotomus longulus*, *Hygrotus sellatus* (LeConte), and *Graphoderus perplexus*. A total of 404 dytiscids was collected in the boreal ponds. The top ten species of that total made up 86.6% and included the following species (in descending order of abundance): *Coptotomus longulus*, *Hygrotus sayi*, *Laccophilus maculosus*, *Acilius semisulcatus*, *Graphoderus liberus*, *Graphoderus perplexus*, *Rhantus wallisi* (Hatch), *Ilybius fraterculus*, *Dytiscus harrisii* Kirby, and *Agabus anthracinus* Mannerheim.

Prairie ponds had a significantly greater species richness ($s=17 \pm 6.8$) than boreal ponds (9.1 ± 4.3) ($t=-4.1$, $df=28$, $p>0.001$) as with bottle-trap sampling in 2002 (Fig. 11). Boreal ponds on average were less diverse ($N2 = 4.8 \pm 2.4$) than prairie ponds ($N2 = 5.0 \pm 2.3$), but the difference was not significant ($t=-0.44$, $df=27$, $p=0.66$) (Fig. 12). Boreal ponds on average had a significantly higher evenness value (0.74 ± 0.16) than prairie ponds (0.60 ± 0.11) ($t=2.6$, $df=17$, $p=0.018$) (Fig. 13).

Dytiscid species abundance

To determine whether there were differences in productivity between prairie and boreal ponds, ponds were examined based on their dytiscid abundances for all three data sets. To quantify the differences between these two groups, group t-tests were performed within all three data sets. An average of 200 ± 204 specimens of dytiscids were collected using volume-sampling in prairie ponds, in comparison to 116 ± 92 specimens of dytiscids in boreal ponds. The difference in abundance between groups of ponds was not significant ($t=-1.6$, $df=27$, $p=0.13$) (Fig. 14). An average of 252 ± 195 dytiscid specimens was collected in bottle traps in 2001 in prairie ponds, and 55 ± 89 dytiscid specimens in boreal ponds. The difference in abundance between groups of ponds was significant

($t=-3.8$, $df=27$, $p<0.001$)(Fig. 15). In 2002, an average of 326 ± 278 dytiscids was collected in bottle traps in prairie ponds and 37 ± 46 dytiscids collected in boreal ponds. The difference in abundance between them was significant ($t=-4.4$, $df=20$, $p<0.001$)(Fig. 16).

Cluster analysis of ponds based on the dytiscid data sets

The clustering method used was sum of squares, based on chord distance. All three data sets were transformed logarithmically. Cluster analysis of the 30 ponds based on similarity of occurrences of dytiscids produced two well-defined clusters. These two representing ponds located in prairie sites and boreal sites (including boreal plains sites). Ponds represented in both clusters had short pectinate runs suggesting that the first division represented two discrete groups.

Volume-sampling of dytiscids, 2001

The dendrogram formed by the communities of dytiscids sampled using volume-sampling in 2001 formed two main clusters, representing prairie ponds and boreal ponds (Fig. 17). The prairie ponds clustered together at a lower level of similarity than the boreal ponds, which suggested that the prairie ponds had more variable dytiscid communities than boreal ponds. There were some regional locality affinities among boreal ponds. Examples of ponds from the same general locality clustered together, included all three ponds from the Richer locality (2E, 3E, 4E), two of three ponds from the East Braintree locality (19E, 21E), and four of five ponds from the Prawda locality

(8E, 9E, 12E, 14E). In contrast, prairie ponds showed less of an affinity to regional locality, except for the Miami locality, which formed a unique cluster (2W, 4W, 5W).

Bottle-trap sampling of dytiscids, 2001

The dendrogram formed by the communities of dytiscids sampled using bottle-traps in 2001 also formed two main clusters consisting of prairie ponds and boreal ponds (Fig. 18). The clustering of boreal ponds showed less affinity to their respective localities than in Fig. 17. One prairie pond (2W) clustered together with the boreal ponds. All the ponds near Baldur (12We, 12Ww, 13We, 13Ww) formed a cluster that also included ponds 23W and 25Wn.

Bottle-trap sampling of dytiscid specimens, 2002

The cluster diagrams of dytiscids collected using bottle-traps in 2002 showed more variability in the boreal ponds than either of the previous two dendrograms (Fig. 19). All three ponds from the Miami region (2W, 4W, 5W) grouped within the predominately boreal pond cluster. Prairie ponds 30W, 31W, 32W, and 103W were closely associated and are from the same region. Ponds 12We (Baldur) and 24W (Dand) formed a group separate from all the other prairie ponds.

Redundancy analysis of dytiscid species

Volume-sampling of dytiscids, 2001

The redundancy analysis (RDA) of the dytiscid species collected using volume-sampling in 2001 produced an ordination diagram in which 60.2 % of the variation in the

species data was explained by axis 1, and 19.6 % was explained by axis 2 (Fig. 20). The species-environment relation for axis 1 was 52.2% and 17.0% for axis 2.

Forward selection was used to determine which variables were most significant in explaining the trends in dytiscid communities. Three variables were forward selected and included in the ordination diagram. The first axis separated the ponds based on macrophyte diversity and chlorophyll *a* density. Using a Monte-Carlo test, the relationship between these two environmental variables and the species data was significant (chlorophyll *a* density, $F=6.72$, $p=0.005$; macrophyte diversity, $F=1.66$, $p=0.08$). The second axis separated the ponds based on pH. A Monte-Carlo test showed that the relationship between pH and the species data was significant ($F=2.22$, $p=0.02$). Most of the prairie ponds were positively correlated with conductivity and chlorophyll *a* density. Ponds that were the exceptions to this trend included prairie ponds 12We, 20W, and 23W, which grouped closer to boreal and boreal plains ponds. Boreal and boreal plains ponds showed limited displacement along axis 2, which is correlated to pH. Boreal and boreal plains ponds that were most correlated with high and low pH values were ponds 21E and 4E, respectively. Ponds 12We, and 103W were at the positive end of the pH environmental vector, while ponds 30W, 5W, and 4W were at the negative end. As a group, dytiscid species showed a bipolar association with axis 1, being either correlated positively or negatively with chlorophyll *a* and macrophyte diversity. Some species, such as *Hygrotus sayi*, *H. picatus* (Kirby), *Neoporus undulatus*, and *Agabus phaeopterus* (Kirby), were exceptions and were negatively correlated with pH values, whereas *H. sellatus* was positively correlated with pH.

Bottle-trap sampling of dytiscid specimens, 2001

The RDA of the dytiscid species collected using bottle-trap sampling in 2001 produced an ordination diagram in which 59.9% of the variation in the species data is explained by axis 1, and 11.2% is explained by axis 2 (Fig. 21). The species-environment relation for axis 1 is 76.6% and 14.8% for axis 2.

The same set of environmental variables used for the volume-sampling 2001 ordination was included in the bottle-trap sampling 2001 ordination. Three environmental variables were again forward selected. Conductivity, macrophyte, and pond permanence were the most significant in explaining the trends in the dytiscid communities. The first axis separated the ponds based on conductivity and macrophyte diversity. Using a Monte-Carlo test, the relationship between these two environmental variables and the species data was significant (conductivity, $F=8.01$, $p=0.05$; macrophyte diversity, $F=4.7$, $p=0.005$). The second axis separated the ponds based on permanence and using a Monte-Carlo test, this relationship was significant ($F=3.89$, $p=0.025$). Boreal and boreal plains ponds were all located at the negative end of axis 1 and were negatively correlated with conductivity and macrophyte diversity. These ponds were not evenly distributed along axis 2. Most prairie ponds were positively correlated with conductivity and macrophyte diversity, except for ponds 20W and 23W. *Hygrotus punctilineatus* (Fall), *H. masculinus* (Crotch), and *Hydroporus columbianus* Fall exclusively co-occurred within prairie pond 12We. Prairie ponds as a whole also do not show a strong correlation to permanence. Pond 24W was an outlier and was negatively correlated to permanence. A number of dytiscids also were negatively correlated to permanence and closely associated with pond 24W, including, *Agabus punctulatus* Aubé, *Agabus canadensis* Fall, and *Agabus bifarius*

(Kirby). Others species of dytiscids that were negatively correlated with permanence include *Hygrotus infuscatus* (Sharp), *Hygrotus turbidus* (LeConte), *Agabus ajax*, *Hygrotus sellatus*, *Hydroporus tenebrosus* LeConte, *Liodessus obscurellus*, *Hygrotus patruelis* (LeConte), *Hygrotus marklini* (Gyllenhal), *Hygrotus impressopunctatus*, *Colymbetes exaratus* LeConte, *Graphoderus occidentalis*, and *Ilybius angustior* (Gyllenhal).

Bottle-trap sampling of dytiscids, 2002

The RDA of dytiscid species collected using bottle-traps in 2002 produced an ordination diagram in which, 65.5% of the variation in the species data was explained by axis 1, and 3.9% was explained by axis 2 (Fig. 22). The species-environment relation for axis 1 was 90.2% and 5.3% for axis 2.

Environmental variables used for bottle-trap sampling in the 2002 ordination were similar to those used for volume-sampling of the 2001 ordination, except chlorophyll *a* data were unavailable. Macrophyte diversity, pH, and bordering vegetation were forward-selected and were most significant in explaining trends in the species data. The first axis separated the ponds based on macrophyte diversity and pH. Using a Monte-Carlo test, the relationship between the species data and macrophyte diversity ($F=5.83$, $p<0.002$) and pH was significant ($F=4.98$, $p<0.004$). Bordering vegetation separated the ponds along axis 2 and using a Monte-Carlo test this relationship was significant ($F=5.76$, $p<0.002$). Boreal, boreal plains, and prairie ponds 24W, 13Ww, 12We, 2W, and 4W were negatively correlated with pH and macrophyte diversity. The remaining prairie ponds were positively correlated with both pH and macrophyte diversity. Approximately half of

the boreal, boreal plains, and prairie ponds were positively correlated with bordering vegetation. Ponds 13Ww, 24W, and 12We were outliers and negatively correlated with bordering vegetation. *Hygrotus salinaris* (Wallis) and *Hygrotus punctilineatus* were only found in pond 12We.

Boreal ponds, dytiscids and odonates from volume-sampling, 2001

The RDA of boreal ponds using the dytiscid species and odonate larvae (Table 4) collected by volume-sampling in 2001 produced an ordination diagram in which 36.1 % of the variation in the dytiscid species data was explained by axis 1, and 17.8 % was explained by axis 2 (Fig. 23). The dytiscid species-odonate relationship for axis 1 was 56.0 % and 83.5 % for axis 2.

Forward selection was used to determine which odonate families were most significant in explaining trends in dytiscid species. Three odonate families (Aeshnidae, Libellulidae and Gomphidae) were forward-selected and included in the ordination. However, using a Monte-Carlo test, the relationship between these three odonate families and the dytiscid species data was not significant (Aeshnidae $F=1.45$, $p=1.54$; Libellulidae $F=1.17$, $p=0.348$; Gomphidae $F=1.60$, $p=0.078$).

Prairie ponds, dytiscids and odonate specimens from volume-sampling, 2001

The RDA of prairie ponds using the dytiscid species and odonate larvae (Table 5) collected by volume-sampling in 2001 produced an ordination diagram in which 36.3 % of the variation in the species data was explained by axis 1, and 13.2 % was explained by

axis 2 (Fig. 24). The dytiscid species-odonate relationship for axis 1 was 60.7 % and 83.0 % for axis 2.

Forward selection with the six odonate families included revealed that Gomphidae, Lestidae, and Coenagrionidae were the most significant in explaining trends in dytiscid species. Using a Monte-Carlo test, the relationship between only Gomphidae and the species of dytiscid was significant (Gomphidae $F=3.51$, $p=0.002$; Lestidae $F=1.20$, $p=0.284$; Coenagrionidae $F=1.48$, $p=0.134$). Ponds 2W and 4W from the Miami region were positively correlated with Gomphidae. Dytiscid species correlated with Gomphidae included *Hygrotus acaroides*, *Neoporus undulatus*, *Colymbetes sculptilis*, *Coptotomus longulus*, *Liodessus obscurellus*, *Hydaticus aruspex*, *Neoporus superioris*, *Laccophilus maculosus*, *Ilybius fraterculus*, *Hygrotus turbidus*, *Hygrotus patruelis*, *Colymbetes exaratus*, and *Hygrotus impressopunctatus*.

Table 1: Reference number, locality, water body name, ecozone, and selected environmental variables of ponds selected for sampling in 2001 and 2002 across southern Manitoba. Ponds are arranged east to west.

Ref. #	Locality	GPS coordinates	Water body name	Ecozone (boreal plains=BPL; boreal=BOR; prairie=PRA)	Area (m ²)	pH		Conductivity (µS/cm)		Chlorophyll a density 2001	Permanence scale 0-5		Macrophyte diversity scale 0-5		Emergent vegetation scale 0-10		Tiger salamanders presence=1 absence=0	Fish presence=1 absence=0
						2001	2002	2001	2002		2001	2002	2001	2002	2001	2002		
16E	East Braintree	49°38'05.8"N, 95°31'17.7"W	n/a	BOR	120	8.4	7.7	331	402	2.0	5	5	2	2	3	3	0	1
19E	East Braintree	49°37'54.4"N, 95°30'42.1"W	Strawberry Pond	BOR	111	8.3	8.1	331	377	1.9	5	5	2	2	6	6	0	1
21E	East Braintree	49°38'12.6"N, 95°27'33.3"W	Saskatoon Pond	BOR	119	8.5	8.0	243	312	2.4	5	5	1	1	4	4	0	0
8E	Prawda	49°38'48.5"N, 95°45'51.4"W	n/a	BOR	43	7.8	7.8	545	609	1.8	5	5	1	1	10	10	0	1
9E	Prawda	49°38'49.6"N, 95°45'49.4"W	n/a	BOR	74	8.2	7.9	328	337	1.5	5	5	1	1	10	10	0	1
12E	Prawda	49°38'50.9"N, 95°45'32.7"W	n/a	BOR	42	8.1	7.9	338	361	1.3	5	5	2	3	8	8	0	0
13E	Prawda	49°38'50.8"N, 95°45'33.4"W	n/a	BOR	49	8.3	8.1	324	354	1.6	5	5	1	1	10	10	0	1
14E	Prawda	49°38'48.1"N, 95°44'48.8"W	n/a	BOR	27	8.1	8.3	343	399	1.2	5	5	1	1	10	10	0	0
2E	Richer	49°43'28.4"N, 95°15'35.9"W	n/a	BPL	57	8.3	8.2	286	365	2.8	5	5	2	3	5	5	0	0
3E	Richer	49°43'43.7"N, 96°16'02.4"W	Reynolds Pond	BPL	141	8.0	7.8	318	340	4.1	5	5	1	3	3	3	0	1
4E	Richer	49°43'01.9"N, 96°14'25.9"W	n/a	BPL	28	7.7	8.0	255	298	1.7	5	5	2	2	6	7	0	0
2W	Miami	49°17'03.4"N, 98°15'21.1"W	n/a	PRA	85	8.0	8.0	817	790	14.9	5	5	3	3	7	7	0	1
4W	Miami	49°17'20.1"N, 98°14'18.1"W	n/a	PRA	44	7.7	7.6	890	846	37.6	5	5	3	3	10	10	1	0
5W	Miami	49°17'18.9"N, 98°14'11.2"W	n/a	PRA	43	7.5	8.3	1192	1356	23.7	5	5	2	2	10	10	0	1
12We	Baldur	49°25'30.7"N, 99°04'51.3"W	Cobbes Lake (east side)	PRA	3500	9.1	9.1	9500	10572	3.8	5	5	1	1	1	1	1	1
12Ww	Baldur	49°25'44.7"N, 99°15'18.6"W	Cobbes Lake (west side)	PRA	625	8.7	8.7	6075	7000	16.0	5	5	2	2	8	8	1	1
13We	Baldur	49°26'24.1"N, 99°14'48.8"W	Taylor's Lake (east side)	PRA	80	8.6	8.7	3094	3321	28.6	5	5	3	3	4	5	1	1
13Ww	Baldur	49°26'24.3"N, 99°14'49.5"W	Taylor's Lake (west side)	PRA	625	8.7	8.7	3250	3499	24.4	5	5	1	3	0	0	1	1
14W	Baldur	49°27'45.8"N, 99°14'43.9"W	n/a	PRA	625	8.8	9.0	1218	1306	36.6	5	5	3	3	2	4	0	1

Table 1: continued

Ref. #	Locality	GPS coordinates	Water body name	Ecozone (boreal plains=BPL; boreal=BOR; prairie=PRA)	Area (m ²)	pH		Conductivity (μ S/cm)		Chlorophyll <i>a</i> density 2001	Permanence scale 0-5		Macrophyte diversity scale 0-5		Emergent vegetation scale 0-10		Tiger salamanders presence=1 absence=0	Fish presence=1 absence=0
						2001	2002	2001	2002		2001	2002	2001	2002	2001	2002		
18W	Ninette	49°24'47.2"N, 99°37'56.3"W	Grass Lake	PRA	10000	8.5	8.5	781	880	9.2	5	5	3	3	10	10	0	1
19W	Ninette	49°21'38.5"N, 99°39'11.7"W	n/a	PRA	206	8.1	8.3	987	1137	15.6	5	5	2	3	10	10	1	1
20W	Ninette	49°20'38.7"N, 99°39'15.4"W	n/a	PRA	225	8.0	8.1	900	954	3.1	5	5	2	3	10	10	1	1
23W	Dand	49°20'26.6"N, 100°27'37.6"W	n/a	PRA	24	8.3	9.0	883	1053	3.6	5	5	1	1	10	10	1	0
24W	Dand	49°20'25.8"N, 100°28'21.8"W	n/a	PRA	42	9.0	9.1	862	1286	92.5	3	4	1	1	10	0	1	0
25Wn	Dand	49°21'51.3"N, 100°29'29.3"W	Chain Lakes (north side)	PRA	3000	8.1	8.4	754	700	21.2	5	5	4	4	4	6	1	1
25Ws	Dand	49°21'47.6"N, 100°29'32.4"W	Chain Lakes (south side)	PRA	80	8.2	8.5	843	998	6.0	5	2	3	3	9	9	0	1
30W	Reston	49°31'43.3"N, 101°09'54.2"W	n/a	PRA	25	7.6	7.8	2043	3000	28.3	5	4	2	3	10	10	0	0
31W	Reston	49°33'47.1"N, 101°10'01.8"W	n/a	PRA	52	8.4	9.3	1132	1553	21.2	5	4	3	3	3	4	1	0
32W	Linklater	49°35'48.4"N, 101°12'47.1"W	n/a	PRA	64	8.5	8.2	1493	1898	5.4	5	5	4	4	8	9	1	0
103W	Linklater	49°36'38.5"N, 101°12'49.4"W	n/a	PRA	80	9.1	8.9	1114	1422	2.0	5	5	4	4	5	7	0	0

Table 2: Dytiscid species (Coleoptera: Dytiscidae) collected using volume-sampling in 2001, bottle-traps in 2001, and bottle-traps in 2002, and the sum of dytiscids collected by all three sampling methods at 30 ponds across southern Manitoba. Taxa are arranged in accordance with Larson *et al.* (2000).

Species name	Total collected in 2001 using volume-sampling	Total collected using bottle-traps		SUM
		2001	2002	
<i>Laccophilus biguttatus</i> Kirby	398	865	1462	2725
<i>L. maculosus</i> Say	250	268	621	1139
<i>Desmopachria convexa</i> (Aubé)	69	1	12	82
<i>Liodessus cantralli</i> (Young)	2	0	0	2
<i>L. fuscatus</i> (Crotch)	1	0	0	1
<i>L. obscurellus</i> (LeConte)	718	117	74	909
<i>Uvarus granarius</i> (Aubé)	49	0	1	50
<i>Hygrotus acaroides</i> (LeConte)	4	0	1	5
<i>H. farctus</i> (LeConte)	51	7	6	64
<i>H. impressopunctatus</i> (Schaller)	136	63	235	434
<i>H. infuscatus</i> (Sharp)	0	3	2	5
<i>H. laccophilinus</i> (LeConte)	4	1	1	6
<i>H. marklini</i> (Gyllenhal)	57	24	75	156
<i>H. masculinus</i> (Crotch)	0	1	0	1
<i>H. patruelis</i> (LeConte)	102	34	52	188
<i>H. picatus</i> (Kirby)	22	8	7	37
<i>H. punctilineatus</i> (Fall)	0	1	5	6
<i>H. salinarius</i> (Wallis)	0	0	1	1
<i>H. sayi</i> Balfour-Browne	1148	923	1067	3138
<i>H. sellatus</i> (LeConte)	26	32	125	183
<i>H. tumidiventris</i> (Fall)	5	4	18	27
<i>H. turbidus</i> (LeConte)	50	20	81	151
<i>Neoporus superioris</i> (Balfour-Browne)	253	134	317	704
<i>N. undulates</i> (Say)	413	26	24	463
<i>Hydroporus columbianus</i> Fall	4	1	1	6
<i>H. dentellus</i> Fall	15	0	0	15
<i>H. fuscipennis</i> Schaum	102	9	11	122
<i>H. larsoni</i> Nilsson	50	6	1	57
<i>H. notabilis</i> LeConte	5	1	10	16
<i>H. pervicinus</i> Fall	2	0	3	5
<i>H. rectus</i> Fall	1	0	0	1
<i>H. striola</i> (Gyllenhal)	2	0	0	2
<i>H. tartaricus</i> LeConte	0	0	1	1
<i>H. tenebrosus</i> LeConte	33	6	2	41
<i>H. tristis</i> (Paykull)	7	0	0	7
<i>Laccornis conoideus</i> (LeConte)	9	1	1	11
<i>Hydrovatus pustulatus</i> (Melsheimer)	0	1	1	2
<i>Agabus ajax</i> Fall	48	644	71	763
<i>A. ambiguus</i> (Say)	1	0	0	1
<i>A. antennatus</i> Leech	272	307	647	1226
<i>A. anthracinus</i> Mannerheim	7	8	10	25
<i>A. bifarius</i> (Kirby)	2	1	0	3
<i>A. canadensis</i> Fall	0	2	18	20
<i>A. erichsoni</i> Gemminger & Harold	0	2	0	2
<i>A. phaeopterus</i> (Kirby)	6	4	2	12
<i>A. punctulatus</i> Aubé	2	2	6	10

Table 2: continued

Species name	Total collected in 2001 using volume-sampling	Total collected using bottle traps		SUM
		2001	2002	
<i>Ilybius angustior</i> (Gyllenhal)	5	23	2	30
<i>I. fraterculus</i> (LeConte)	53	174	98	325
<i>I. picipes</i> (Kirby)	0	1	2	3
<i>I. pleuriticus</i> (LeConte)	1	1	2	4
<i>I. subaeneus</i> (Erichson)	10	45	16	71
<i>Colymbetes exaratus</i> LeConte	29	69	24	122
<i>C. sculptilis</i> Harris	7	5	6	18
<i>Rhantus binotatus</i> (Harris)	2	9	2	13
<i>R. consimilis</i> Motschulsky	1	1	1	3
<i>R. sericans</i> Sharp	280	772	928	1980
<i>R. sinuatus</i> (LeConte)	0	1	0	1
<i>R. suturellus</i> (Harris)	3	0	1	4
<i>R. wallisi</i> (Hatch)	8	4	13	25
<i>Coptotomus longulus</i> LeConte	256	123	139	518
<i>Dytiscus alaskanus</i> Balfour-Browne	4	60	53	117
<i>D. circumcinctus</i> Ahrens	0	1	0	1
<i>D. cordieri</i> Aubé	4	90	12	106
<i>D. dauricus</i> Gebler	0	7	8	15
<i>D. harrisii</i> Kirby	1	8	10	19
<i>D. hybridus</i> Aubé	0	0	2	2
<i>D. verticalis</i> Say	2	3	1	6
<i>Hydaticus aruspex</i> Clark	8	38	7	53
<i>H. piceus</i> LeConte	4	26	39	69
<i>Acilius semisulcatus</i> Aubé	25	94	54	173
<i>A. sylvanus</i> Hilsenhoff	0	2	2	4
<i>Graphoderus liberus</i> (Say)	10	13	23	46
<i>G. occidentalis</i> Horn	23	122	47	192
<i>G. perplexus</i> Sharp	18	172	125	315
Total species abundance	5080	5391	6589	17060
Total species richness	60	60	62	74

Table 3: Dytiscid species (Coleoptera: Dytiscidae) collected using volume-sampling in 2001, bottle-traps in 2001, and bottle-traps in 2002 from boreal and prairie ponds and ecozone preference. Ecozone preference is denoted for species that were only collected in either boreal or prairie ponds for all samples. Boreal preference represents species that were collected predominantly in either boreal plains or boreal ponds.

Species name	Volume-sampling		Bottle-trap sampling				Ecozone preference
	2001		2001		2002		
	Ponds						
	Boreal	Prairie	Boreal	Prairie	Boreal	Prairie	
<i>Laccophilus biguttatus</i> Kirby	1	397	0	865	2	1460	-
<i>L. maculosus</i> Say	54	196	49	219	48	573	-
<i>Desmopachria convexa</i> (Aubé)	68	1	1	0	5	7	-
<i>Liodessus cantralli</i> (Young)	2	0	0	0	0	0	boreal
<i>L. fuscatus</i> (Crotch)	1	0	0	0	0	0	boreal
<i>L. obscurellus</i> (LeConte)	67	651	56	61	5	69	-
<i>Uvarus granarius</i> (Aubé)	48	1	0	0	1	0	-
<i>Hygrotus acaroides</i> (LeConte)	0	4	0	0	0	1	prairie
<i>H. farctus</i> (LeConte)	51	0	7	0	6	0	boreal
<i>H. impressopunctatus</i> (Schaller)	3	133	2	61	0	235	-
<i>H. infuscatus</i> (Sharp)	0	0	0	3	0	2	-
<i>H. laccophilinus</i> (LeConte)	4	0	1	0	1	0	boreal
<i>H. marklini</i> (Gyllenhal)	1	56	0	24	0	75	-
<i>H. masculinus</i> (Crotch)	0	0	0	1	0	0	prairie
<i>H. patruelis</i> (LeConte)	2	100	0	34	0	52	-
<i>H. picatus</i> (Kirby)	9	13	2	6	2	5	-
<i>H. punctilineatus</i> (Fall)	0	0	0	1	0	5	prairie
<i>H. salinarius</i> (Wallis)	0	0	0	0	0	1	prairie
<i>H. sayi</i> Balfour-Browne	582	566	219	704	78	989	-
<i>H. sellatus</i> (LeConte)	0	26	0	32	0	125	prairie
<i>H. tumidiventris</i> (Fall)	0	5	0	4	0	18	prairie
<i>H. turbidus</i> (LeConte)	0	50	0	20	1	80	-
<i>Neoporus superioris</i> (Balfour-Browne)	2	251	0	134	0	317	-
<i>N. undulates</i> (Say)	46	367	8	18	4	20	-
<i>Hydroporus columbianus</i> Fall	0	4	0	1	0	1	prairie
<i>H. dentellus</i> Fall	15	0	0	0	0	0	boreal
<i>H. fuscipennis</i> Schaum	0	102	0	9	0	11	-
<i>H. larsoni</i> Nilsson	3	47	0	6	0	1	-
<i>H. notabilis</i> LeConte	1	4	0	1	1	9	-
<i>H. pervicinus</i> Fall	0	2	0	0	0	3	prairie
<i>H. rectus</i> Fall	0	1	0	0	0	0	prairie
<i>H. striola</i> (Gyllenhal)	0	2	0	0	0	0	prairie
<i>H. tartaricus</i> LeConte	0	0	0	0	0	1	prairie
<i>H. tenebrosus</i> LeConte	2	31	0	6	0	2	-
<i>H. tristis</i> (Paykull)	7	0	0	0	0	0	boreal
<i>Laccornis conoideus</i> (LeConte)	0	9	0	1	0	1	boreal
<i>Hydrovatus pustulatus</i> (Melsheimer)	0	0	1	0	1	0	boreal
<i>Agabus ajax</i> Fall	0	48	0	644	0	71	prairie
<i>A. ambiguus</i> (Say)	0	1	0	0	0	0	prairie
<i>A. antennatus</i> Leech	0	272	0	307	0	647	prairie
<i>A. anthracinus</i> Mannerheim	7	0	2	6	9	1	-
<i>A. bifarius</i> (Kirby)	0	2	0	1	0	0	prairie
<i>A. canadensis</i> Fall	0	0	0	2	0	18	prairie
<i>A. erichsoni</i> Gemminger & Harold	0	0	0	2	0	0	prairie
<i>A. phaeopterus</i> (Kirby)	1	5	0	4	0	2	-
<i>A. punctulatus</i> Aubé	0	2	0	2	0	6	prairie
<i>Ilybius angustior</i> (Gyllenhal)	0	5	1	22	0	2	-
<i>I. fraterculus</i> (LeConte)	12	41	6	168	10	88	-

Table 3: continued

Species name	Volume-sampling		Bottle-trap sampling				Ecozone preference
	2001		2001		2002		
			Ponds				
	Boreal	Prairie	Boreal	Prairie	Boreal	Prairie	
<i>I. picipes</i> (Kirby)	0	0	0	1	0	2	prairie
<i>I. pleuriticus</i> (LeConte)	1	0	1	0	2	0	boreal
<i>I. subaeneus</i> (Erichson)	3	7	8	37	2	14	-
<i>Colymbetes exaratus</i> LeConte	2	27	2	67	1	23	-
<i>C. sculptilis</i> Harris	5	2	0	5	0	6	-
<i>Rhantus binotatus</i> (Harris)	1	1	8	1	1	1	-
<i>R. consimilis</i> Motschulsky	0	1	0	1	0	1	prairie
<i>R. sericans</i> Sharp	8	272	6	766	2	926	-
<i>R. sinuatus</i> (LeConte)	0	0	1	0	0	0	boreal
<i>R. suturellus</i> (Harris)	3	0	0	0	1	0	boreal
<i>R. wallisi</i> (Hatch)	8	0	0	4	11	2	-
<i>Coptotomus longulus</i> LeConte	208	48	90	33	106	33	-
<i>Dytiscus alaskanus</i> Balfour-Browne	0	4	0	60	0	53	prairie
<i>D. circumcinctus</i> Ahrens	0	0	0	1	0	0	prairie
<i>D. cordieri</i> Aubé	1	3	1	89	3	9	-
<i>D. dauricus</i> Gebler	0	0	7	0	8	0	boreal
<i>D. harrisii</i> Kirby	1	0	8	0	10	0	boreal
<i>D. hybridus</i> Aubé	0	0	0	0	0	2	prairie
<i>D. verticalis</i> Say	2	0	3	0	1	0	boreal
<i>Hydaticus aruspex</i> Clark	1	7	5	33	1	6	-
<i>H. piceus</i> LeConte	1	3	4	22	3	36	-
<i>Acilius semisulcatus</i> Aubé	24	1	84	10	42	12	-
<i>A. sylvanus</i> Hilsenhoff	0	0	1	1	0	2	-
<i>Graphoderus liberus</i> (Say)	10	0	11	2	19	4	-
<i>G. occidentalis</i> Horn	2	21	4	118	0	47	-
<i>G. perplexus</i> Sharp	3	15	9	163	17	108	-
SUM	1273	3807	608	4783	404	6185	Total # of species unique to boreal = 13; prairie ponds = 26.

Table 4: Numbers of odonate larvae (Insecta: Odonata) arranged by family collected in boreal ponds using volume-sampling, 2001. Site locations as in Table 1.

Odonate family	16E	19E	21E	8E	9E	12E	13E	14E	2E	3E	4E	SUM
Lestidae	0	0	1	4	2	19	0	21	7	0	172	226
Coenagrionidae	81	77	116	76	108	187	26	46	134	50	134	1035
Gomphidae	9	5	9	0	0	0	4	0	2	3	0	32
Aeshnidae	29	24	25	41	54	34	19	12	6	6	26	276
Corduliidae	4	27	8	5	7	24	18	14	0	2	2	111
Libellulidae	71	179	420	102	211	165	170	223	135	90	186	1952

Table 5: Numbers of odonate larvae (Insecta: Odonata) arranged by family collected in prairie ponds using volume-sampling, 2001. Site locations as in Table 1.

Odonate family	2W	4W	5W	12We	12Ww	13We	13Ww	14W	18W	19W	20W	23W	24W	25Wn	25Ws	30W	31W	32W	103W	SUM
Lestidae	35	35	0	0	3	0	0	0	1	52	13	18	134	2	8	3	45	82	13	445
Coenagrionidae	294	13	6	50	63	275	108	219	22	21	19	6	29	224	37	45	177	19	73	1816
Gomphidae	18	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	30
Aeshnidae	62	31	0	0	3	2	0	0	4	2	10	2	13	17	14	0	10	8	4	207
Corduliidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8
Libellulidae	16	54	1	0	9	0	0	0	5	33	6	12	96	3	20	1	42	74	47	839

Table 6: Pond reference number, dytiscid species richness, Hill's effective species richness (N2), and Kvålseth evenness, for volume-sampling 2001, bottle-trap sampling 2001, and bottle-sampling 2002 for each of the 30 pond sampled in southern Manitoba. Site locations as in Table 1.

Ponds	Volume-sampling 2001			Bottle-trap sampling					
	Species richness	Hill's effective species richness (N2)	Kvålseth evenness	2001			2002		
				Species richness	Hill's effective species richness (N2)	Kvålseth evenness	Species richness	Hill's effective species richness (N2)	Kvålseth evenness
14E	12	6.362	0.786	11	5.261	0.532	8	5.158	0.818
16E	13	3.810	0.571	9	2.977	0.565	5	3.846	0.849
19E	13	3.256	0.571	4	2.613	0.748	7	6.400	0.943
8E	19	2.121	0.371	7	2.738	0.684	11	4.670	0.641
9E	11	3.331	0.636	11	2.895	0.540	11	7.024	0.764
12E	21	3.195	0.407	10	5.507	0.763	11	6.443	0.797
13E	7	2.109	0.626	5	2.750	0.744	6	5.261	0.929
2E	16	4.744	0.674	18	4.414	0.616	19	4.547	0.609
3E	13	3.219	0.510	6	4.000	0.797	5	1.914	0.534
4E	10	4.839	0.694	6	4.765	0.866	5	3.522	0.796
21E	19	7.087	0.614	11	8.036	0.868	12	3.320	0.474
2W	19	4.314	0.637	14	8.640	0.759	13	6.000	0.680
4W	20	6.026	0.592	21	10.394	0.743	13	9.959	0.875
5W	16	5.210	0.631	16	8.605	0.767	22	6.503	0.583
12We	5	3.024	0.769	16	3.974	0.553	11	3.220	0.489
12Ww	12	5.618	0.718	18	7.822	0.764	9	2.632	0.531
13We	17	7.257	0.654	18	6.382	0.720	19	3.515	0.499
13Ww	11	3.151	0.662	18	2.063	0.401	9	2.855	0.568
14W	20	5.262	0.621	21	3.548	0.501	24	2.135	0.404
18W	20	8.074	0.741	21	8.096	0.668	16	4.892	0.625
19W	25	6.396	0.589	25	6.826	0.607	22	4.038	0.615
20W	25	8.285	0.666	22	5.625	0.520	20	8.012	0.750
23W	19	7.220	0.673	14	2.962	0.468	31	8.502	0.654
24W	16	8.547	0.786	25	2.559	0.375	1	1.000	n/a
25Wn	18	8.126	0.763	17	6.431	0.651	20	2.558	0.476
25Ws	12	3.438	0.529	18	5.140	0.552	19	7.224	0.694
30W	23	7.640	0.705	27	9.009	0.619	22	3.863	0.482
31W	16	5.459	0.681	19	5.943	0.634	23	6.415	0.599
32W	12	3.973	0.655	22	5.763	0.669	20	4.052	0.586
103W	16	6.560	0.702	19	7.069	0.737	18	4.568	0.680

Figure 1: Distribution of the five ecozones within the Province of Manitoba: taiga shield (TSH), Hudson Bay plains (HPL), boreal (BOR), boreal plains (BPL), and prairie (PRA) (Wilken 1986). Inset map indicates the general location (●) of all eight localities which represent groups of ponds: Reston/Linklater (1), Dand (2), Ninette (3), Baldur (4), Miami (5), Richer (6), Prawda (7), and East Braintree (8).

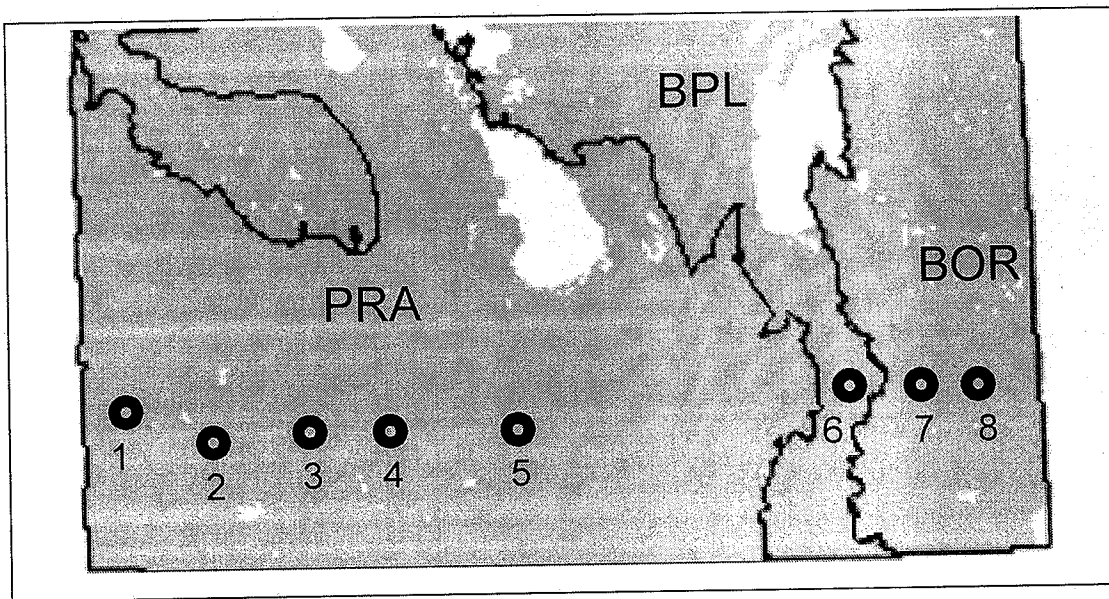
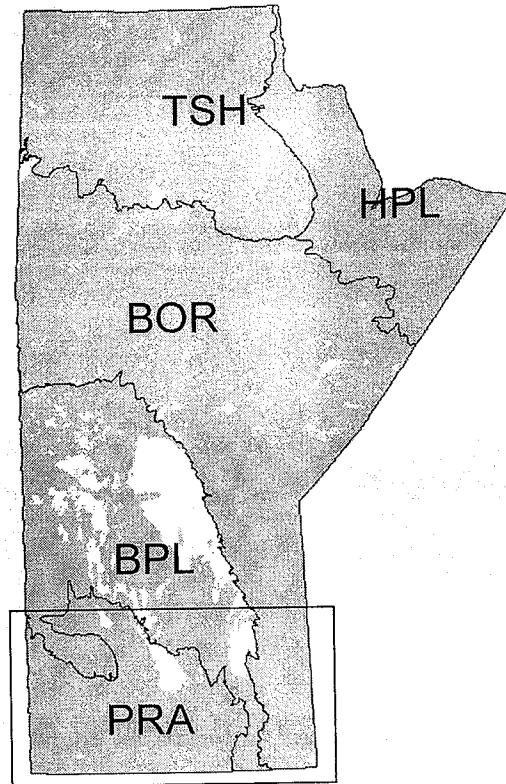






Figure 2: Sampling protocol in ponds across southern Manitoba illustrating the placement of bottle-traps and where volume-samples were taken within the four sectors of a hypothetical pond.

-  volume-sample
-  bottle-trap
-  pond depth 0 – 60 cm
-  pond depth >60 cm

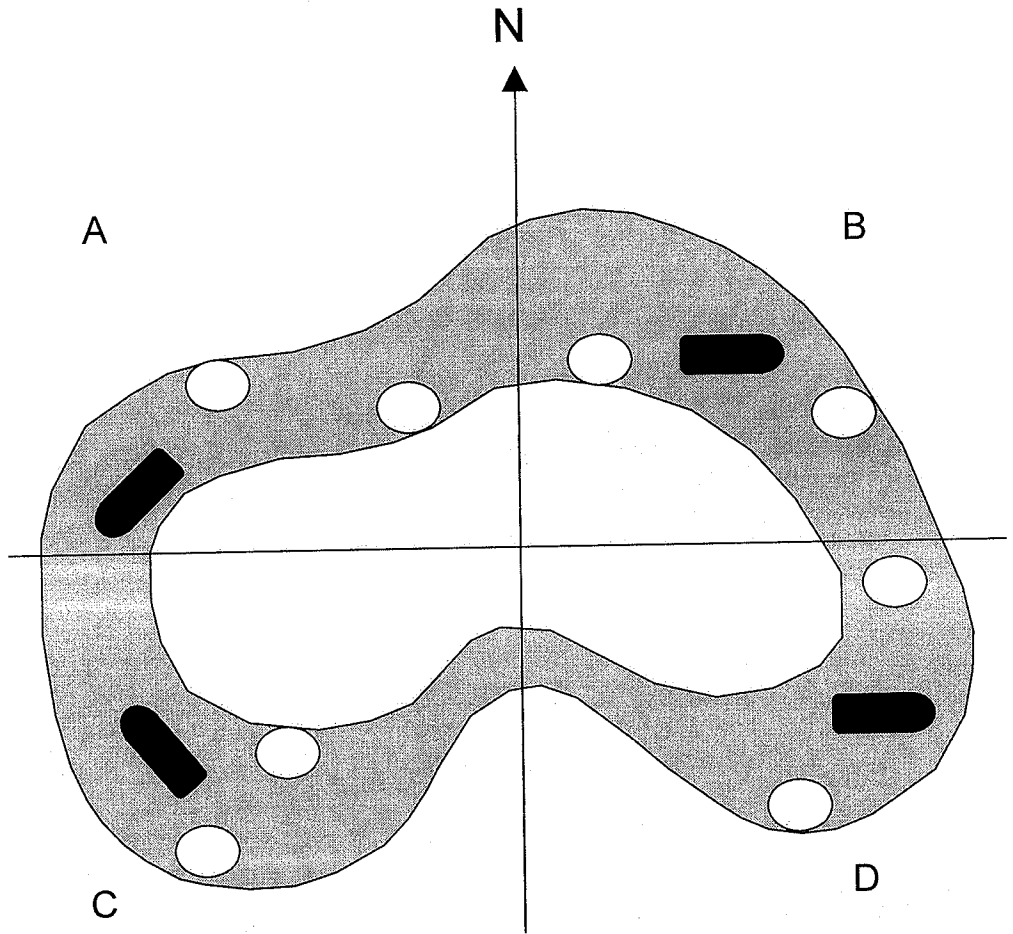


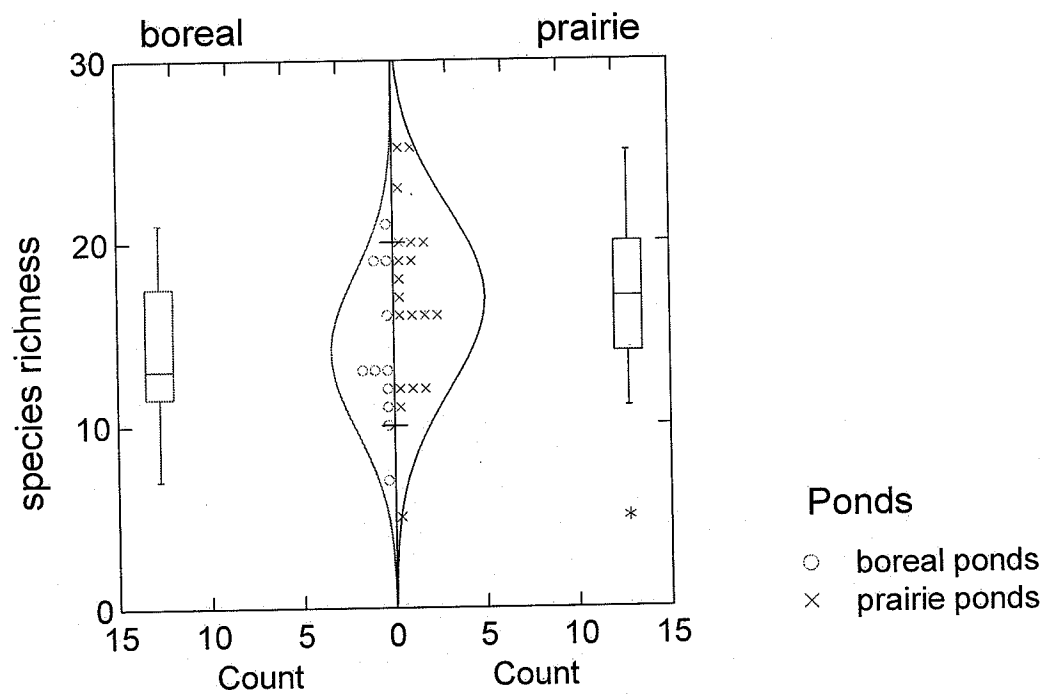
Figure 3: A bottle-trap (inset) being submerged in a pond for sampling of invertebrates. See Figure 2 for placement of bottle traps within the pond.



Figure 4: Use of a volume-sampler and a D-net for sampling invertebrates from a pond. See Figure 2 for placement of samples within a pond.

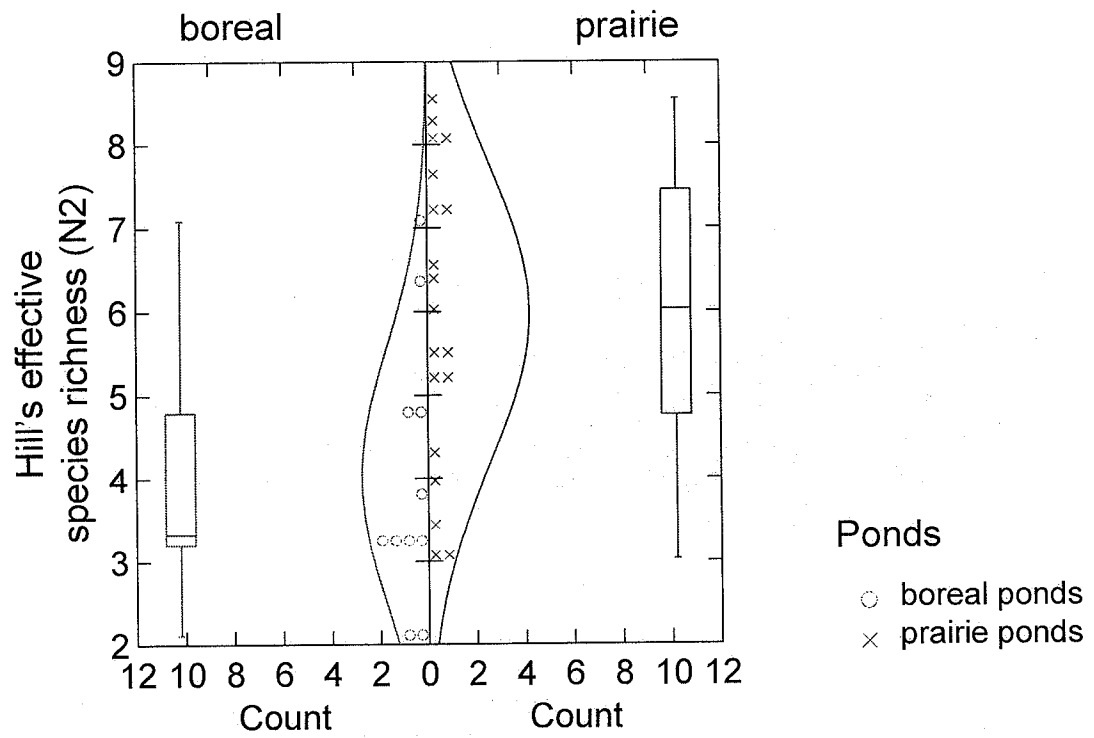


Figure 5: Group t-test comparing species richness of dytiscids (Coleoptera: Dytiscidae) collected by volume-sampling in 2001 in boreal ponds and prairie ponds across southern Manitoba with a confidence of 95%. Box and whisker plots indicate species richness in boreal and prairie ponds. Upper and lower ends of boxes represent 75th and 25th percentiles. The median is depicted with a solid line between the upper and lower end of the boxes. Whiskers represent 90th and 10th percentiles, and the stars (*) represent any outliers.



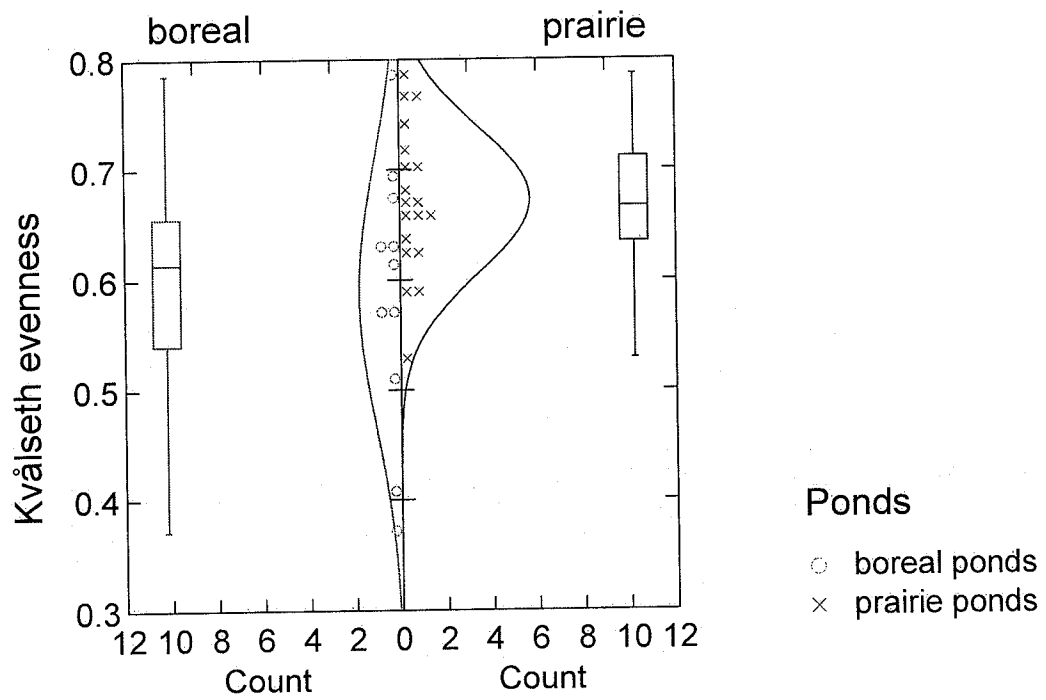
$t=-1.7$ $df=24$ $p=0.10$

Figure 6: Group t-test comparing Hill's effective species richness (N_2) of dytiscids (Coleoptera: Dytiscidae) collected by volume-sampling in 2001 in boreal ponds and prairie ponds across southern Manitoba with a confidence of 95%. Box and whisker plots indicate Hill's effective species richness (N_2) in boreal and prairie ponds. Upper and lower ends of boxes represent 75th and 25th percentiles. The median is depicted with a solid line between the upper and lower end of the boxes. Whiskers represent 90th and 10th percentiles, and the stars (*) represent any outliers.



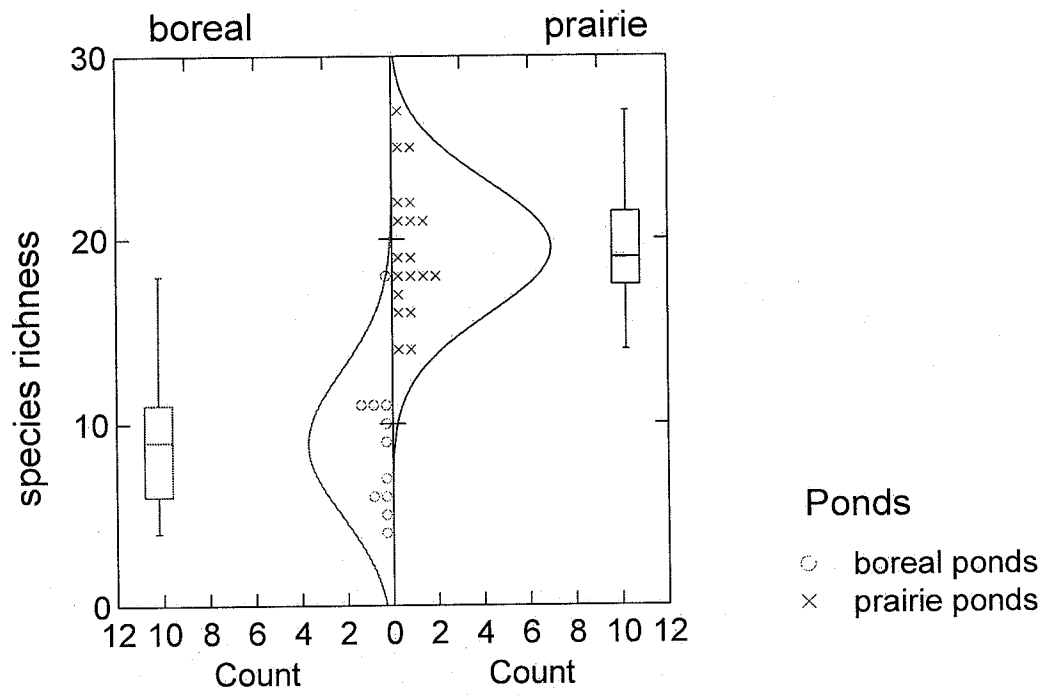
t=-3.1 df= 23 p=0.0051

Figure 7: Group t-test comparing Kvålseth evenness index of dytiscids (Coleoptera: Dytiscidae) collected by volume-sampling in 2001 in boreal ponds and prairie ponds across southern Manitoba with a confidence of 95%. Box and whisker plots indicate Kvålseth evenness index in boreal and prairie ponds. Upper and lower ends of boxes represent 75th and 25th percentiles. The median is depicted with a solid line between the upper and lower end of the boxes. Whiskers represent 90th and 10th percentiles, and the stars (*) represent any outliers.



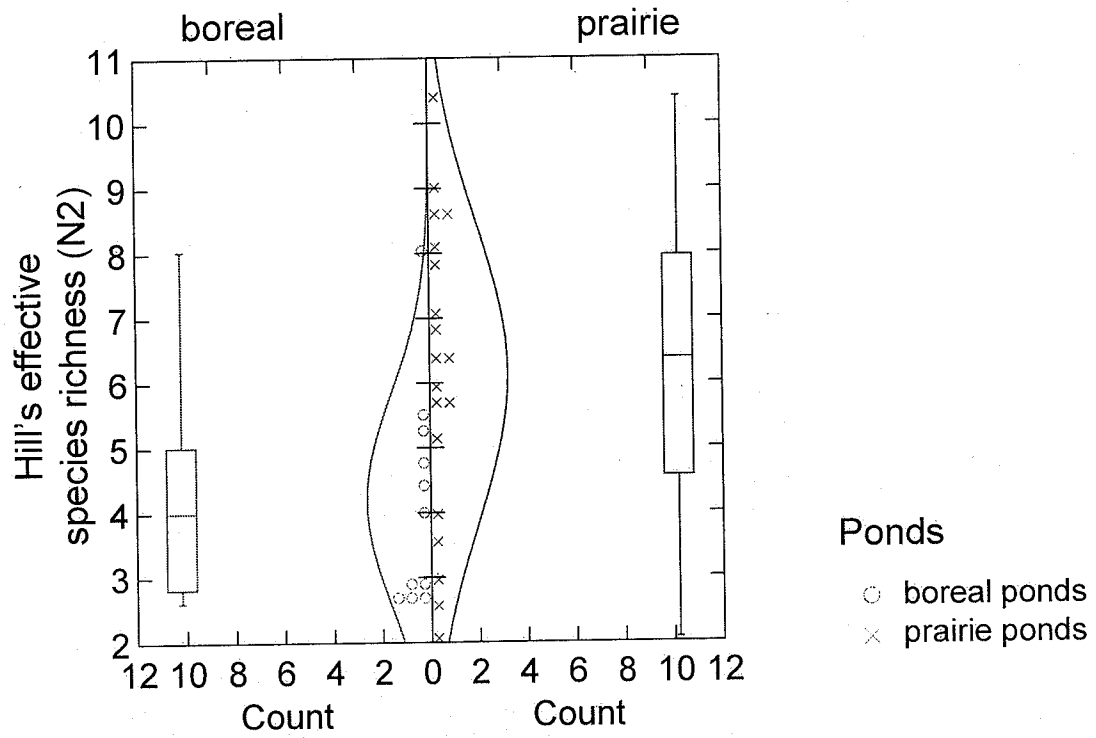
$t=-2.1$ $df=14$ $p=0.051$

Figure 8: Group t-test comparing species richness of dytiscids (Coleoptera: Dytiscidae) collected by bottle-trap sampling in 2001 in boreal ponds and prairie ponds across southern Manitoba with a confidence of 95%. Box and whisker plots indicate species richness in boreal and prairie ponds. Upper and lower ends of boxes represent 75th and 25th percentiles. The median is depicted with a solid line between the upper and lower end of the boxes. Whiskers represent 90th and 10th percentiles, and the stars (*) represent any outliers.



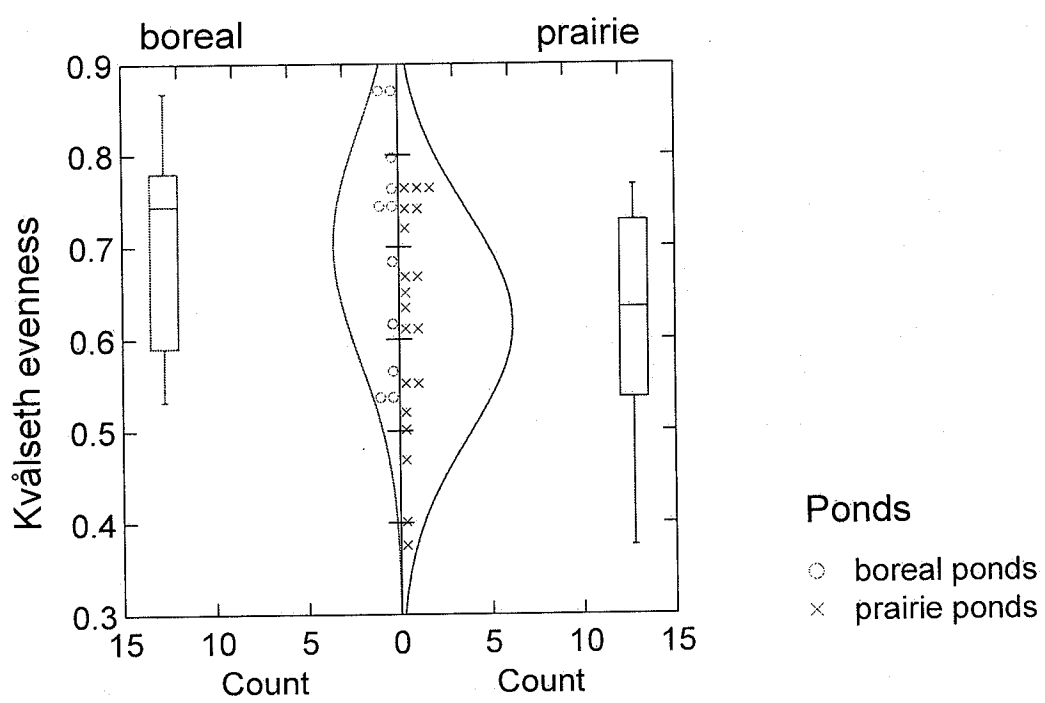
t=-7.3 df= 20 p<0.001

Figure 9: Group t-test comparing Hill's effective species richness (N2) of dytiscids (Coleoptera: Dytiscidae) collected by bottle-trap sampling in 2001 in boreal ponds and prairie ponds across southern Manitoba with a confidence of 95%. Box and whisker plots indicate Hill's effective species richness (N2) in boreal and prairie ponds. Upper and lower ends of boxes represent 75th and 25th percentiles. The median is depicted with a solid line between the upper and lower end of the boxes. Whiskers represent 90th and 10th percentiles, and the stars (*) represent any outliers.



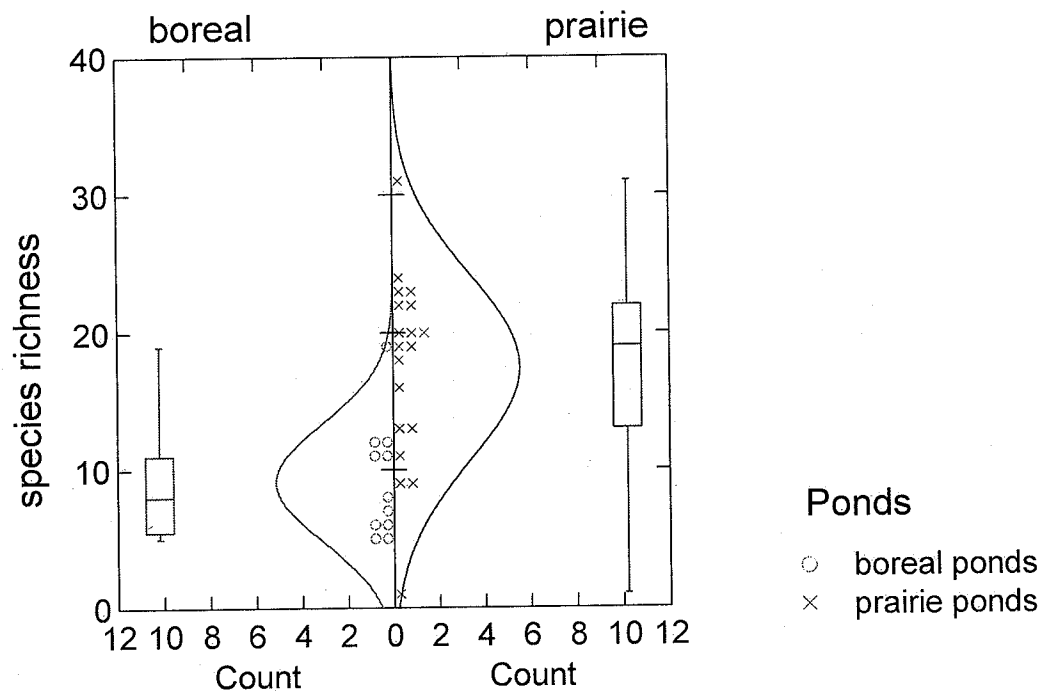
$t=-2.7$ $df=27$ $p=0.013$

Figure 10: Group t-test comparing Kvålseth evenness index of dytiscids (Coleoptera: Dytiscidae) collected by bottle-trap in 2001 in sampling in boreal ponds and prairie ponds across southern Manitoba with a confidence of 95%. Box and whisker plots indicate Kvålseth evenness index in boreal and prairie ponds. Upper and lower ends of boxes represent 75th and 25th percentiles. The median is depicted with a solid line between the upper and lower end of the boxes. Whiskers represent 90th and 10th percentiles, and the stars (*) represent any outliers.



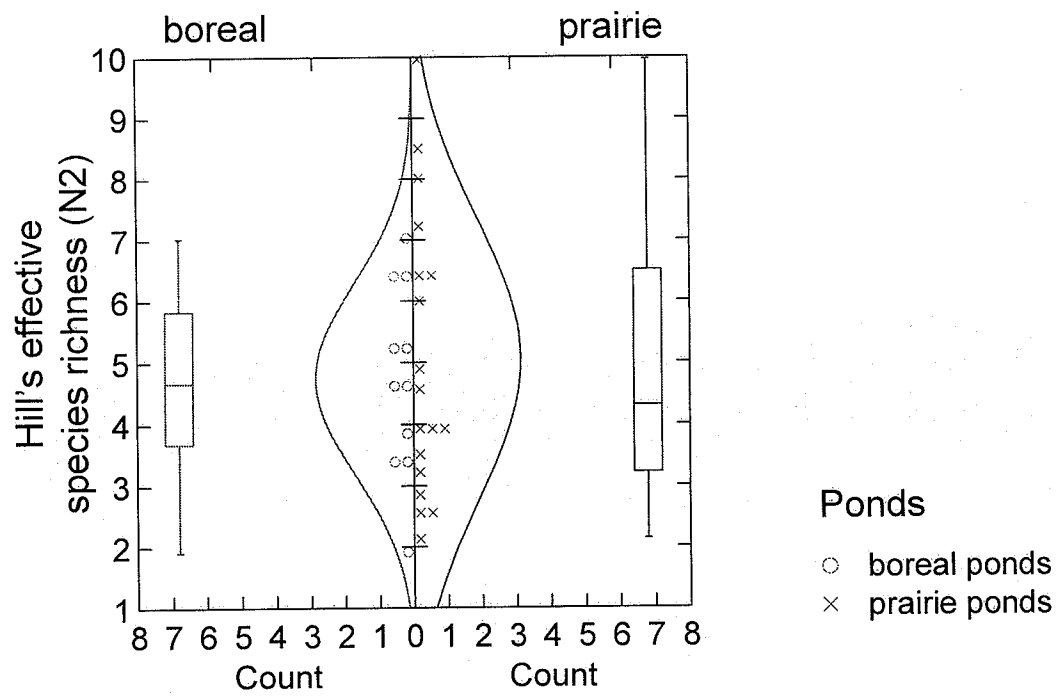
t=1.8 df=21 p=0.080

Figure 11: Group t-test comparing species richness of dytiscids (Coleoptera: Dytiscidae) collected by bottle-trap sampling in 2002 in boreal ponds and prairie ponds across southern Manitoba with a confidence of 95%. Box and whisker plots indicate species richness in boreal and prairie ponds. Upper and lower ends of boxes represent 75th and 25th percentiles. The median is depicted with a solid line between the upper and lower end of the boxes. Whiskers represent 90th and 10th percentiles, and the stars (*) represent any outliers.



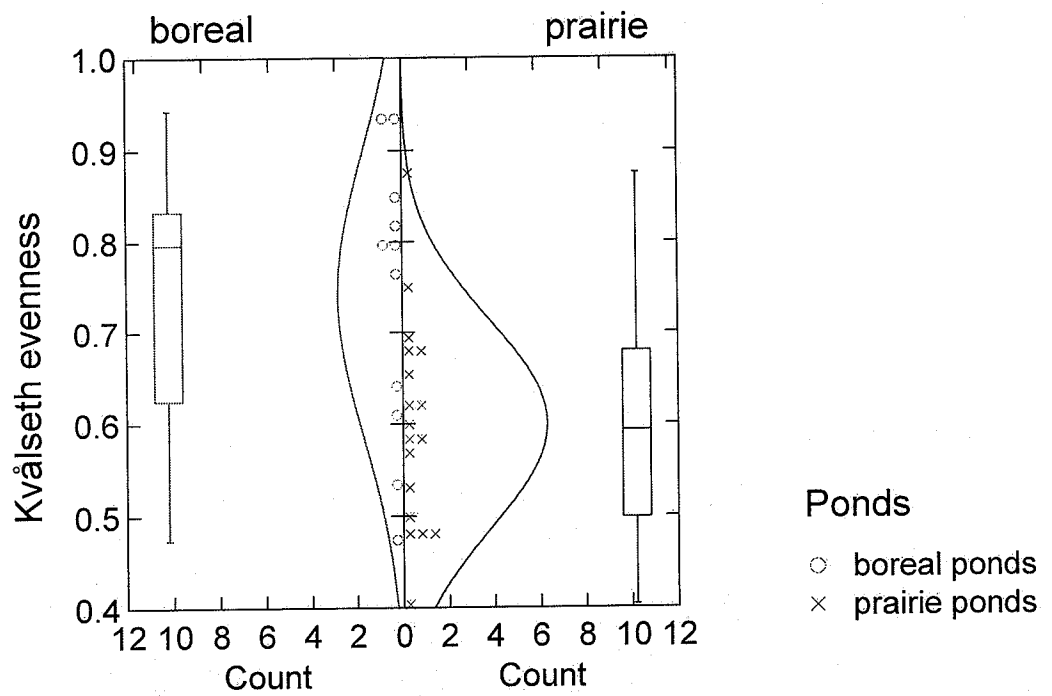
$t=-4.1$ $df=28$ $p<0.001$

Figure 12: Group t-test comparing Hill's effectiveness species richness (N2) of dytiscids (Coleoptera: Dytiscidae) collected by bottle-trap sampling in 2002 in boreal ponds and prairie ponds across southern Manitoba with a confidence of 95%. Box and whisker plots indicate Hill's effectiveness species richness (N2) in boreal and prairie ponds. Upper and lower ends of boxes represent 75th and 25th percentiles. The median is depicted with a solid line between the upper and lower end of the boxes. Whiskers represent 90th and 10th percentiles, and the stars (*) represent any outliers.



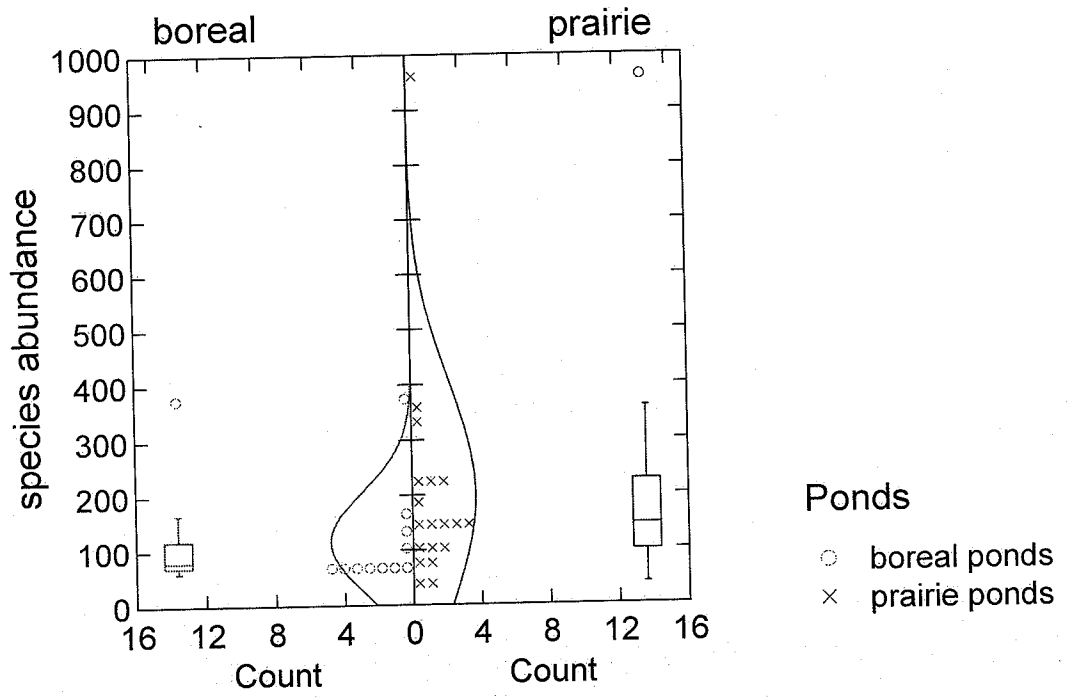
$t=-0.44$ $df=27$ $p=0.66$

Figure 13: Group t-test comparing Kvålseth evenness index of dytiscids (Coleoptera: Dytiscidae) collected by bottle trap in 2002 sampling in boreal ponds and prairie ponds across southern Manitoba with a confidence of 95%. Box and whisker plots indicate Kvålseth evenness index in boreal and prairie ponds. Upper and lower ends of boxes represent 75th and 25th percentiles. The median is depicted with a solid line between the upper and lower end of the boxes. Whiskers represent 90th and 10th percentiles, and the stars (*) represent any outliers.



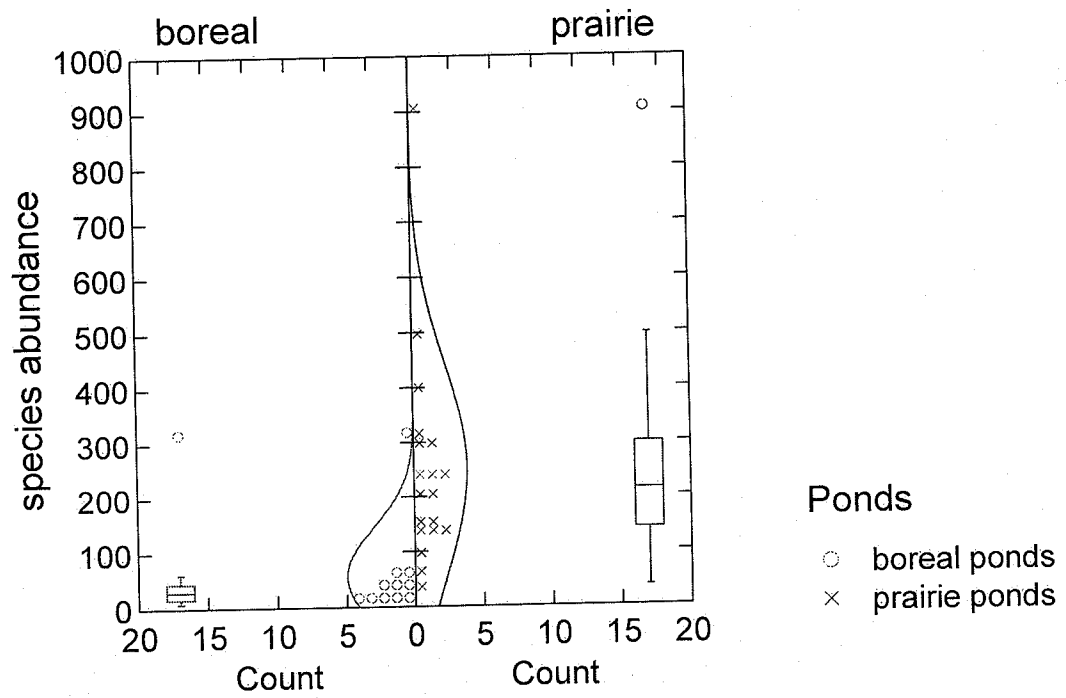
$t=2.6$ $df=17$ $p=0.018$

Figure 14: Group t-test comparing species abundance of dytiscids (Coleoptera: Dytiscidae) collected by volume-sampling in 2001 in boreal ponds and prairie ponds across southern Manitoba with a confidence of 95%. Box and whisker plots indicate species abundance in boreal and prairie ponds. Upper and lower ends of boxes represent 75th and 25th percentiles. The median is depicted with a solid line between the upper and lower end of the boxes. Whiskers represent 90th and 10th percentiles, and the stars (*) represent any outliers.



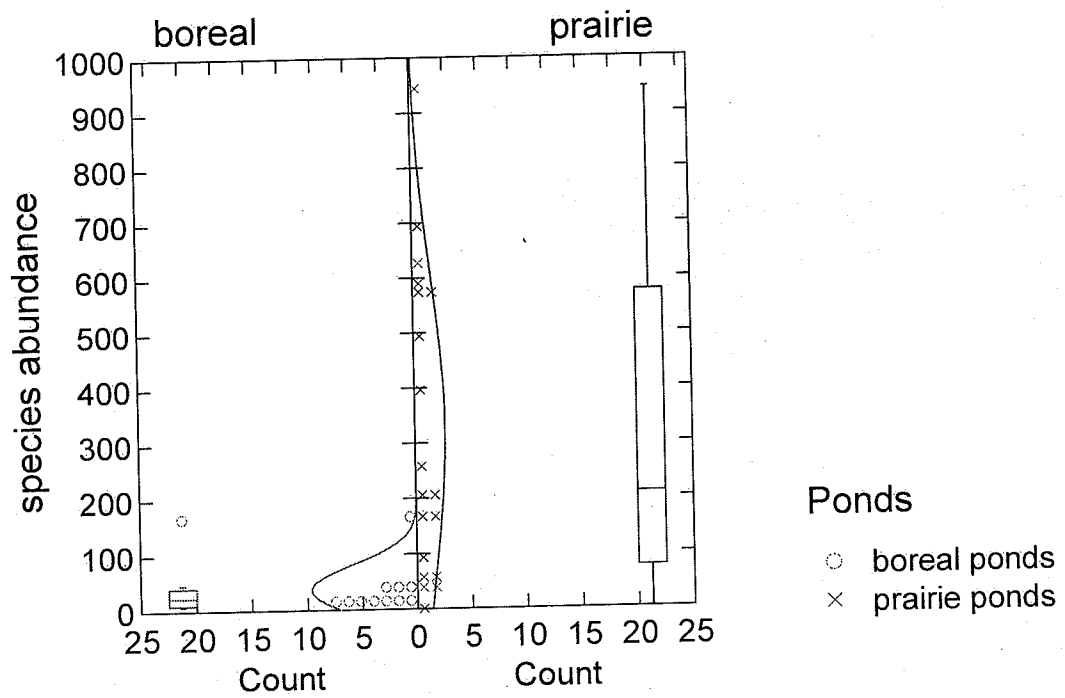
$t=-1.6$ $df=27$ $p=0.13$

Figure 15: Group t-test comparing species abundance of dytiscids (Coleoptera: Dytiscidae) collected by bottle trap sampling in 2001 in boreal ponds and prairie ponds across southern Manitoba with a confidence of 95%. Box and whisker plots indicate species abundance in boreal and prairie ponds. Upper and lower ends of boxes represent 75th and 25th percentiles. The median is depicted with a solid line between the upper and lower end of the boxes. Whiskers represent 90th and 10th percentiles, and the stars (*) represent any outliers.



$t=-3.8$ $df=27$ $p<0.001$

Figure 16: Group t-test comparing species abundance of dytiscids (Coleoptera: Dytiscidae) collected by bottle-trap sampling in 2002 in boreal ponds and prairie ponds across southern Manitoba with a confidence of 95%. Box and whisker plots indicate species abundance in boreal and prairie ponds. Upper and lower ends of boxes represent 75th and 25th percentiles. The median is depicted with a solid line between the upper and lower end of the boxes. Whiskers represent 90th and 10th percentiles, and the stars (*) represent any outliers.



$t=-4.4$ $df=20$ $p<0.001$

Figure 17: Sum of squares (chord) cluster analysis dendrogram of ponds across southern Manitoba based on dytiscids (Coleoptera: Dytiscidae) collected by volume-sampling in 2001. Site locations as in Table 1.

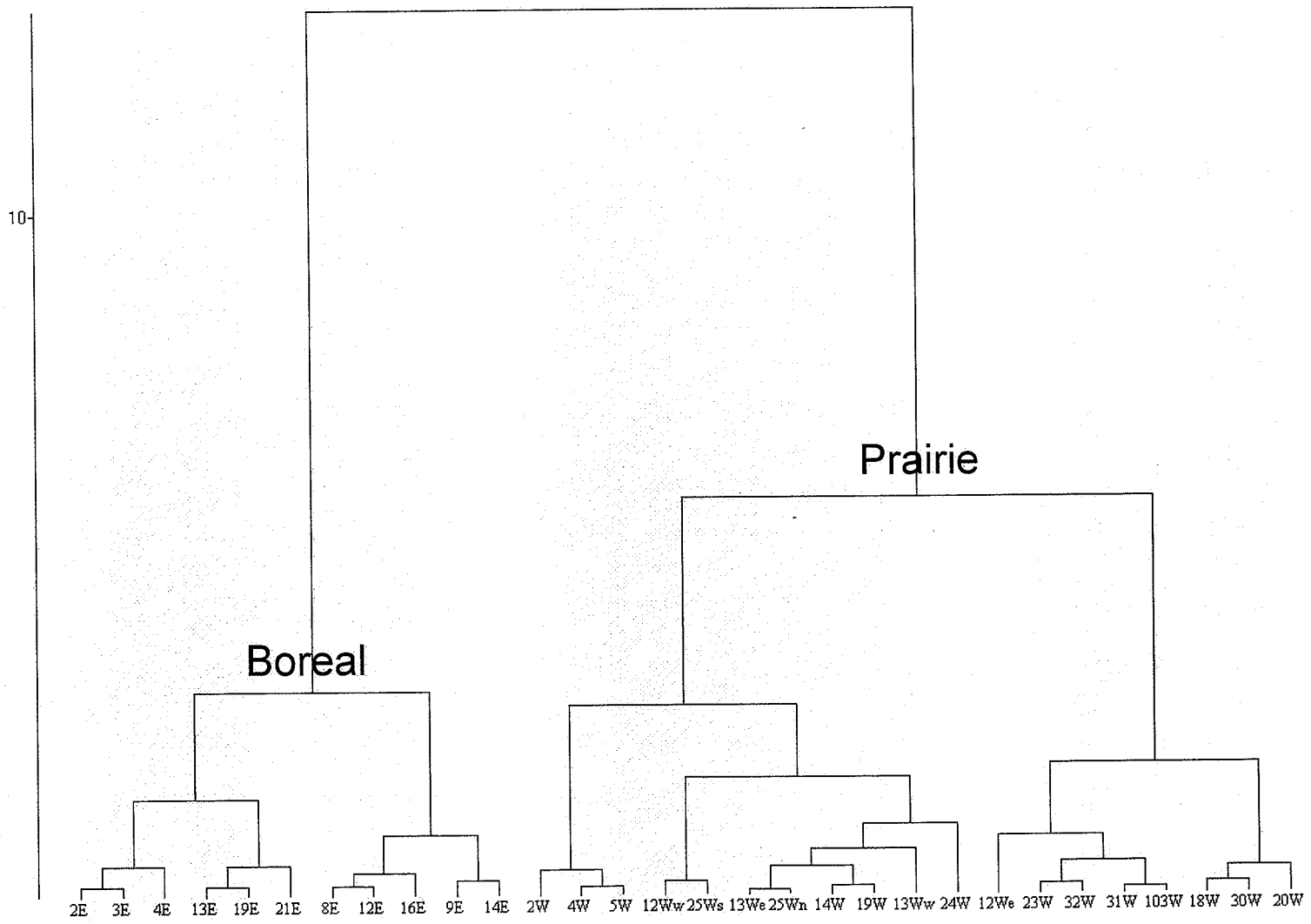


Figure 18: Sum of squares (chord) cluster analysis dendrogram of ponds across southern Manitoba based on dytiscids (Coleoptera: Dytiscidae) collected by bottle-trap sampling in 2001. Site locations as in Table 1.

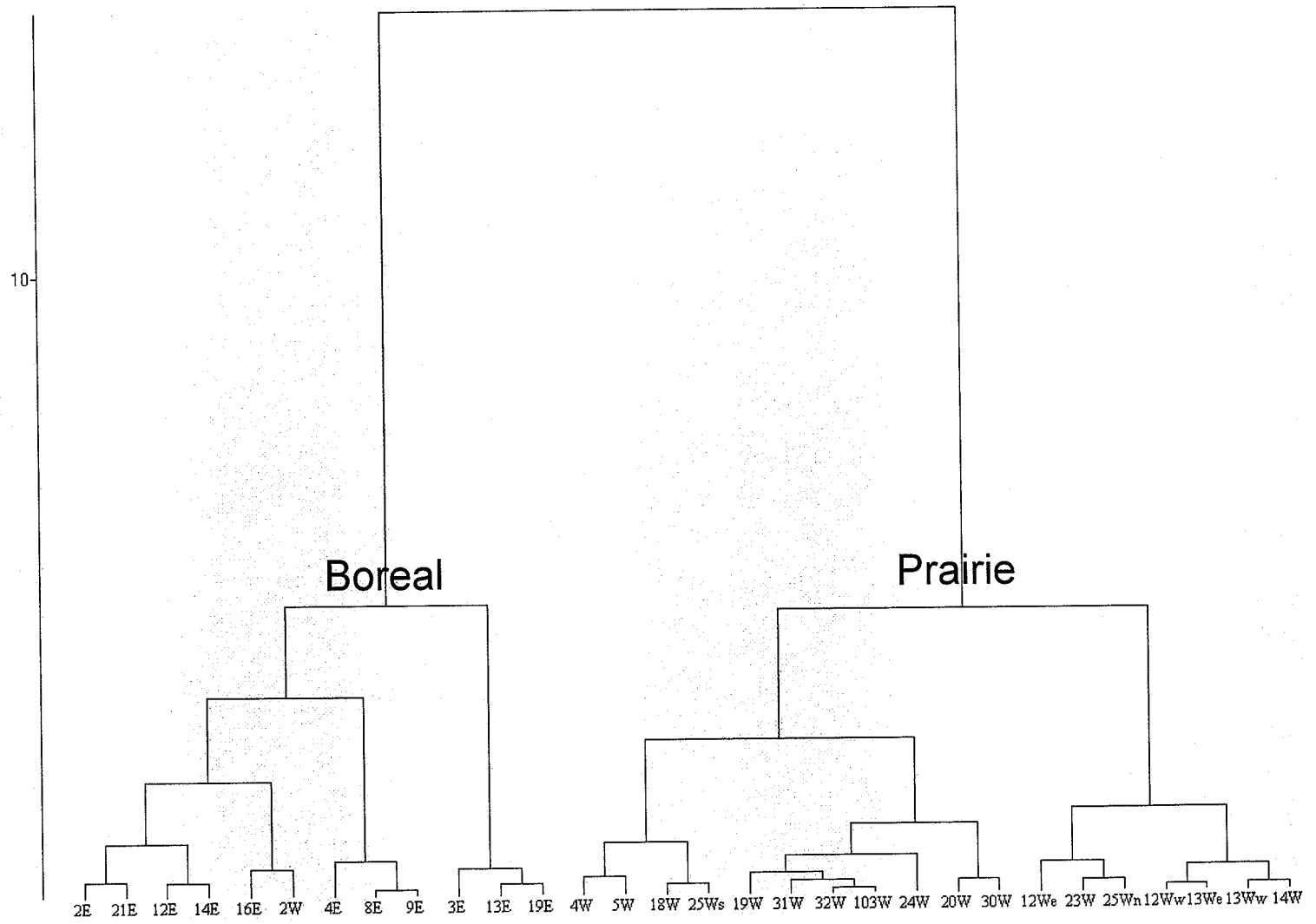


Figure 19: Sum of squares (chord) cluster analysis dendrogram of ponds across southern Manitoba based on dytiscids (Coleoptera: Dytiscidae) collected by bottle-trap sampling in 2002. Site locations as in Table 1.

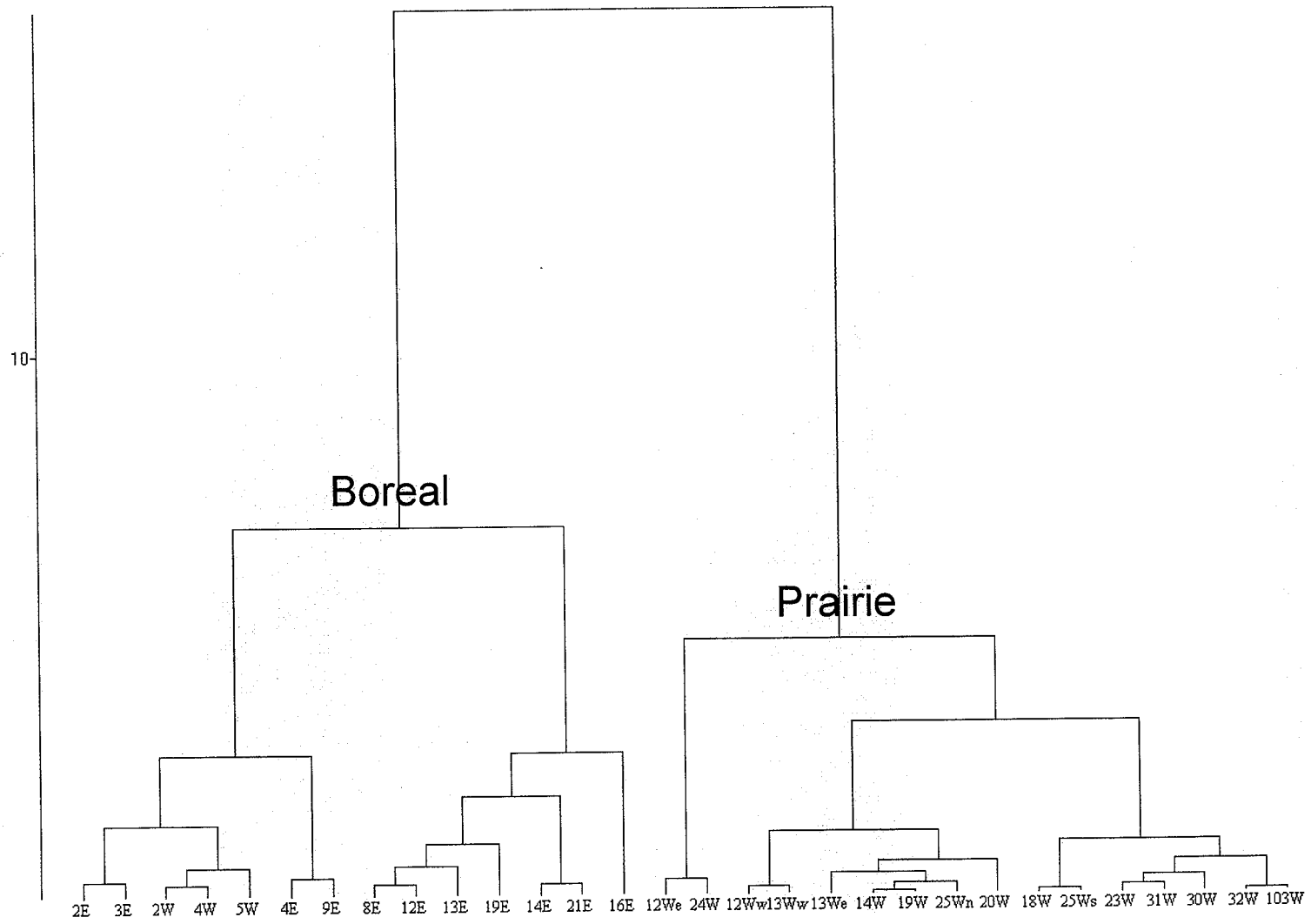


Figure 20: Redundancy analysis ordination diagram with site scores (●), species scores (▶), and environmental vectors (▶). The first axis (horizontal) has an eigenvalue of 0.239=60.2%, and the second axis (vertical) has an eigenvalue of 0.078=19.6%. Site locations as in Table 1. Acronyms for dytiscid species (Coleoptera: Dytiscidae): specimens were collected from ponds across southern Manitoba using volume-sampling for 2001.

Aci_sem = <i>Acilius semisulcatus</i>	Hyd_den = <i>Hydroporus dentellus</i>	Hyg_tur = <i>Hygrotus turbidus</i>
Aga_aja = <i>Agabus ajax</i>	Hyd_fus = <i>Hydroporus fuscipennis</i>	Ily_ang = <i>Ilybius angustior</i>
Aga_amb = <i>Agabus ambiguus</i>	Hyd_lar = <i>Hydroporus larsoni</i>	Ily_fra = <i>Ilybius fraterculus</i>
Aga_ant = <i>Agabus antennatus</i>	Hyd_not = <i>Hydroporus notabilis</i>	Ily_ple = <i>Ilybius pleuriticus</i>
Aga_bif = <i>Agabus bifarius</i>	Hyd_per = <i>Hydroporus pervicinus</i>	Ily_sub = <i>Ilybius subaeneus</i>
Aga_can = <i>Agabus canadensis</i>	Hyd_pic = <i>Hydaticus piceus</i>	Lac_big = <i>Laccophilus biguttatus</i>
Aga pha = <i>Agabus phaeopterus</i>	Hyd_rec = <i>Hydroporus rectus</i>	Lac_con = <i>Laccornis conoideus</i>
Aga_pun = <i>Agabus punctulatus</i>	Hyd_str = <i>Hydroporus striola</i>	Lac_mac = <i>Laccophilus maculosus</i>
Agaanth = <i>Agabus anthracinus</i>	Hyd_ten = <i>Hydroporus tenebrosus</i>	Lio_can = <i>Liodessus cantralli</i>
Col_exa = <i>Colymbetes exaratus</i>	Hyd_tri = <i>Hydroporus tristis</i>	Lio_fus = <i>Liodessus fuscatus</i>
Col_scu = <i>Colymbetes sculptilis</i>	Hyg_aca = <i>Hygrotus acaroides</i>	Lio_obs = <i>Liodessus obscurellus</i>
Cop_lon = <i>Coptotomus longulus</i>	Hyg_far = <i>Hygrotus farctus</i>	Neo_sup = <i>Neoporus superioris</i>
Des_con = <i>Desmopachria convexa</i>	Hyg_imp = <i>Hygrotus impressopunctatus</i>	Neo_und = <i>Neoporus undulatus</i>
Dyt_ala = <i>Dytiscus alaskanus</i>	Hyg_lac = <i>Hygrotus laccophilinus</i>	Ran_bin = <i>Rhantus binotatus</i>
Dyt_cor = <i>Dytiscus cordieri</i>	Hyg_mar = <i>Hygrotus marklini</i>	Ran_con = <i>Rhantus consimilis</i>
Dyt_har = <i>Dytiscus harrisii</i>	Hyg_mas = <i>Hygrotus masculinus</i>	Ran_ser = <i>Rhantus sericans</i>
Dyt_ver = <i>Dytiscus verticalis</i>	Hyg_pat = <i>Hygrotus patruelis</i>	Ran_sut = <i>Rhantus suturellus</i>
Gra_lib = <i>Graphoderus liberus</i>	Hyg_pic = <i>Hygrotus picatus</i>	Ran_wal = <i>Rhantus wallisi</i>
Gra_occ = <i>Graphoderus occidentalis</i>	Hyg_pun = <i>Hygrotus punctilineatus</i>	Uva_gra = <i>Uvarus granarius</i>
Gra_per = <i>Graphoderus perplexus</i>	Hyg_say = <i>Hygrotus sayi</i>	
Hyd_aru = <i>Hydaticus aruspex</i>	Hyg_sel = <i>Hygrotus sellatus</i>	
Hyd_col = <i>Hydroporus columbianus</i>	Hyg_tum = <i>Hygrotus tumidiventris</i>	

Figure 21: Redundancy analysis ordination diagram with site scores (●), species scores (→), and environmental vectors (→). The first axis (horizontal) has an eigenvalue of 0.338=59.9%, and the second axis (vertical) has an eigenvalue of 0.063=11.2%. Site locations as in Table 1. Acronyms for dytiscid species (Coleoptera: Dytiscidae): specimens were collected from ponds across southern Manitoba using bottle-trap sampling for 2001.

Aci_sem = <i>Acilius semisulcatus</i>	Hyd_den = <i>Hydroporus dentellus</i>	Hyg_tur = <i>Hygrotus turbidus</i>
Aga_aja = <i>Agabus ajax</i>	Hyd_fus = <i>Hydroporus fuscipennis</i>	Ily_ang = <i>Ilybius angustior</i>
Aga_amb = <i>Agabus ambiguus</i>	Hyd_lar = <i>Hydroporus larsoni</i>	Ily_fra = <i>Ilybius fraterculus</i>
Aga_ant = <i>Agabus antennatus</i>	Hyd_not = <i>Hydroporus notabilis</i>	Ily_ple = <i>Ilybius pleuriticus</i>
Aga_bif = <i>Agabus bifarius</i>	Hyd_per = <i>Hydroporus pervicinus</i>	Ily_sub = <i>Ilybius subaeneus</i>
Aga_can = <i>Agabus canadensis</i>	Hyd_pic = <i>Hydaticus piceus</i>	Lac_big = <i>Laccophilus biguttatus</i>
Aga_pha = <i>Agabus phaeopterus</i>	Hyd_rec = <i>Hydroporus rectus</i>	Lac_con = <i>Laccornis conoideus</i>
Aga_pun = <i>Agabus punctulatus</i>	Hyd_str = <i>Hydroporus striola</i>	Lac_mac = <i>Laccophilus maculosus</i>
Agaanth = <i>Agabus anthracinus</i>	Hyd_ten = <i>Hydroporus tenebrosus</i>	Lio_can = <i>Liodessus cantralli</i>
Col_exa = <i>Colymbetes exaratus</i>	Hyd_tri = <i>Hydroporus tristis</i>	Lio_fus = <i>Liodessus fuscatus</i>
Col_scu = <i>Colymbetes sculptilis</i>	Hyg_aca = <i>Hygrotus acaroides</i>	Lio_obs = <i>Liodessus obscurellus</i>
Cop_lon = <i>Coptotomus longulus</i>	Hyg_far = <i>Hygrotus farctus</i>	Neo_sup = <i>Neoporus superioris</i>
Des_con = <i>Desmopachria convexa</i>	Hyg_imp = <i>Hygrotus impressopunctatus</i>	Neo_und = <i>Neoporus undulatus</i>
Dyt_ala = <i>Dytiscus alaskanus</i>	Hyg_lac = <i>Hygrotus laccophilinus</i>	Ran_bin = <i>Rhantus binotatus</i>
Dyt_cor = <i>Dytiscus cordieri</i>	Hyg_mar = <i>Hygrotus marklini</i>	Ran_con = <i>Rhantus consimilis</i>
Dyt_har = <i>Dytiscus harrisii</i>	Hyg_mas = <i>Hygrotus masculinus</i>	Ran_ser = <i>Rhantus sericans</i>
Dyt_ver = <i>Dytiscus verticalis</i>	Hyg_pat = <i>Hygrotus patruelis</i>	Ran_sut = <i>Rhantus suturellus</i>
Gra_lib = <i>Graphoderus liberus</i>	Hyg_pic = <i>Hygrotus picatus</i>	Ran_wal = <i>Rhantus wallisi</i>
Gra_occ = <i>Graphoderus occidentalis</i>	Hyg_pun = <i>Hygrotus punctilineatus</i>	Uva_gra = <i>Uvarus granarius</i>
Gra_per = <i>Graphoderus perplexus</i>	Hyg_say = <i>Hygrotus sayi</i>	
Hyd_aru = <i>Hydaticus aruspex</i>	Hyg_sel = <i>Hygrotus sellatus</i>	
Hyd_col = <i>Hydroporus columbianus</i>	Hyg_tum = <i>Hygrotus tumidiventris</i>	

