**University of Alberta**

# Effects of Stocked Trout on Native Fish and Littoral Invertebrates in Boreal Foothills Lakes

by

Leslie Emily Nasmith



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# Abstract

In Alberta, fisheries managers stock non-native trout species into lakes to enhance angling opportunities. I assessed ecological effects of this management practice by comparing forage fish density, size structure, and habitat use, and littoral invertebrate community composition, abundance, and size structure, in five stocked and six unstocked lakes in the boreal foothills over three summers (2005-2007). Although altered size structure and changes in habitat use of forage fish were evident in stocked lakes, trout did not decrease forage fish densities in stocked relative to unstocked lakes. Similarly, few invertebrate taxa differed in abundance or size structure, and community composition did not differ between treatments. These productive lakes had abundant refuges for forage fish and invertebrates, and while trout preyed primarily on invertebrates, trout impacts were not detectable over the background forage fish impact. Thus, impacts of introduced trout on forage fish and littoral invertebrates in boreal foothills lakes appeared limited.

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Remember, "Fish are friends, not food!"

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# Chapter 1. General Introduction

*Sport Fish Stocking and Sport Fishing*

Wilderness stocking of sport fish in high elevation lakes of North America began in the late nineteenth century to create, enhance, or diversify regional recreational fishing opportunities (Wiley 2003). Initially, stockings of desirable fish were carried out by private citizens with no regulation, monitoring, or record keeping (Whittier and Kincaid 1999; Pister 2001). Stocking of fish was so pervasive that we do not know if many contemporary populations are natural or were established from stocked populations. In some areas, such as Wyoming, all fish populations in high-elevation lakes are the result of introductions (Wiley 2003) and this sort of stocking was especially widespread in western North America. Nevertheless, it is a contemporary management tool that is used across all ecoregions (Whittier and Kincaid 1999; Cowx and Gerdeaux 2004).

Modern commercial fishing can remove large amounts of biomass from aquatic systems (Pauly et al. 1998; Jackson et al. 2001; Myers and Worm 2003); however, the efficiency of recreational angling should not be discounted (Post *et al.* 2002; Cooke and Cowx 2004). Recreational fisheries can contribute to alterations in system function and quality (Cooke and Cowx 2004). Although a single angler might not have a measurable effect, the impact in recreational fisheries is cumulative and may be problematic because angling is simultaneously wide spread across landscapes and focused on particular habitats (Cooke and Cowx 2004). In fact, high angler pressure has been associated with recreational fisheries collapses. In western Canada, 78% of walleye (*Sander vitreus*) populations show signs of having collapsed, and Northern pike (*Esox lucius*) populations are also in trouble (Post *et al.* 2002). As angler densities increase and stocks decrease, catch rates begin to decline (Post *et al.* 2002). One way to deal with the high demand for fishing opportunities is to create new fishing locations through stocking of fishless lakes or lakes that lack game fish (Ashley and Nordin 1999). Stocking is a common method of enhancing or creating recreational fisheries in areas where naturally occurring sport fish are sparse or nonexistent (Wiley 2003).

Stocking can have negative side effects on recreational fisheries. Managers often stock fingerlings in areas of anticipated high angler pressure but high exploitation rates lead to stocks of small, short-lived fish (Bailey and Hubert 2003). Furthermore, stocking is often used to create put-and-take fisheries where fish are stocked at high densities. This practice artificially inflates yields, and creates high angler expectations and demand for further stocking (Van Zyll De Jong *et al.* 2004).

*Sport Fish Stocking in Alberta*

The popularity of recreational fishing in Alberta is demonstrated by the annual participation of almost 300,000 anglers (Park 2007). Alberta, however, has only about 2% of Canada's freshwater area (National Research Council of Canada 2004) and estimates suggest that only 800 lakes in the province can naturally support game fish (Park 2007). Alberta lakes are not only relatively scarce, they are also generally small in size. For example, Alberta has only 3 lakes larger than $400km^2$, whereas other prairie provinces, Saskatchewan and Manitoba, have 15 and 14 lakes of that size, respectively (National Research Council of Canada 2004). This scarcity of lakes and high number of anglers results in a high average angler density of 400 anglers per lake (Park 2007).

Alberta Sustainable Resource Development (ASRD) and the Alberta Conservation Association (ACA) have created several successful recreational fisheries by stocking brook *(Salvelinus fontinalis)*, brown (*Salmo trutta*), and rainbow (*Oncorhynchus mykiss*) trout into a number of lakes throughout the province. In the boreal foothills, for example, some lakes have been stocked since the early 1950s (Miller and Thomas 1957; S. Herman, Alberta Sustainable Resource Development, personal communication). Trout are stocked at sizes of 5 to 15 cm (S. Herman, Alberta Sustainable Resource Development, personal communication) and can grow rapidly, even in the first season following introduction (C. MacLeod, Alberta Conservation Association, personal communication). However, these stocked fish do not establish self-sustaining populations in boreal foothills lakes due to the trout's inability to reproduce, or survive over-winter without aeration. Consequently, populations are artificially maintained by annual or biennial stockings.

A major, natural limiting factor of sport fish populations in Alberta lakes is dissolved oxygen levels in winter. Alberta lakes are typically shallow and naturally productive, often leading to oxygen depletion over the course of the long winter, causing winterkill of fish. An efficient means of overcoming this problem is through the use of aerators. Aerators facilitate the transfer of atmospheric oxygen to the water by increasing turbulence and the surface area of water in contact with the air (Boyd 1998). In winter, this also creates an ice-free area around the aerator. Aeration is most often used in commercial aquaculture (e.g., Arctic charr; Summerfelt *et al.* 2004) and in habitat restoration and rehabilitation (Boyd 1998). Aeration will permit over-winter survival of fish, allowing them to grow older and larger, and thus improve fishing opportunities (Ashley and Nordin 1999).

*Effects of Stocked Sport Fish*

As lakes are stocked, the native, resident fauna face an uncertain future. Negative effects of stocked sport fish on native fauna is well documented in the literature, although research has primarily been conducted in naturally fishless, oligotrophic, high altitude lakes (Knapp *et al.* 2001a; Dunham *et al.* 2004). Research has not been conducted in productive, forage-fish bearing lakes in the boreal zone, nor in aerated lakes.

Ross (1991) described potential responses of native fish to fish invasions: no effect, a change in population size or age structure, and/or shifts in resource use. Generally, sport fish introductions do not favour native fish, with estimates as high as 77% of cases showing a reduction or elimination of native fish after introduction (Ross 1991). Research in Quebec suggests that piscivory by introduced fish is responsible for many local extinctions (Chapleau and Findlay 1997), and in the Adirondacks, lakes with introduced piscivores, forage fish richness was only one-third as high as in lakes without top piscivores (Findlay *et al.* 2000).

Non-native trout can have complex interactions with other salmonids (e.g., Ruzycki *et al.* 2003; Hasegawa *et al.* 2004; Morita *et al.* 2004), whereas their relationship with native small-bodied fish is often that of predator and prey. Predation directly affects forage fish populations by reducing densities (Tonn 1985) and altering prey size distributions (Tonn and Paszkowski 1986; Tonn *et al.* 1992). Even where no direct effect on forage fish populations are seen, behavioural responses may still be present (Ross 1991; Pink *et al.* 2007) often in the form of altered habitat use (Tonn and Paszkowski

1987; Museth *et al.* 2002), and decreased activity (Bryan *et al.* 2002). It is unlikely that all prey taxa will be equally affected by presence of trout. Responses may vary both among and within taxa, since predation risk can change with differences in behaviour (Abrahams 1994), resource use (Price *et al.* 1991), and size (Magoulick 2004).

Changes in fish populations, including the introduction of sport fish into lakes where they did not previously exist, can have significant impacts on invertebrate communities (Post and Cucin 1984; Tonn *et al.* 2004). High predation rates can lead to shifts in size structure of benthic macroinverebrates towards smaller sizes (Post and Cucin 1984; Blumenshine *et al.* 2000). In naturally fishless alpine lakes, distribution and abundance of conspicuous invertebrates decreased after fish introductions (Knapp *et al.* 2001b). Similarly, invertebrate composition in a small boreal plains lake shifted from large conspicuous to smaller inconspicuous taxa after the introduction of northern pike (Venturelli and Tonn 2005). These changes may not be permanent, as some invertebrates are able to repopulate a lake after fish removal (Knapp *et al.* 2001b; Wissinger *et al.* 2006). As well, it can not be assumed that the presence of trout will always lead to the decimation of invertebrate taxa (Bolger *et al.* 1990; Wissinger *et al.* 2006).

Contrary to some of the above examples, invertebrate communities in boreal foothills lakes have been under predation pressure by native forage fish prior to stocking. Forage fish populations consist of different combinations of fathead minnow (*Pimephales promelas*), brook stickleback (*Culaea inconstans*), Iowa darter (*Etheostoma exile*), and mixtures of pearl (*Margariscus margarita*), northern redbelly (*Phoxinus eos*), finescale (*P. neogaeus*) dace, and dace hybrids (*P. eos* x *P. neogaeus*). Forage fish alone can impact invertebrates (e.g., Zimmer *et al.* 2001). It is not known if introduced trout can impact invertebrate communities in lakes with forage fish similar to those in fishless systems (e.g., Knapp *et al.* 2001b; Dunham *et al.* 2004; Sarnelle and Knapp 2005; Venturelli and Tonn 2005), or if the presence of native fish obscures effects of trout.

*Objectives*

Given the position of sport fish at the top of a lake's food web, management activities involving these species are whole ecosystem manipulations that could advance our understanding of lake communities. Too often, however, insufficient attention has been paid to the impact of fisheries management practices and thus we have missed opportunities to advance our knowledge of these practices (Cowx and Gerdeux 2004). For example, there is a paucity of systematic monitoring of impacts of stocking initiatives and when monitoring does occur, the information gathered is often poorly disseminated.

The primary objective of my study is to assess the impact of trout stocking on native populations of forage fish and co-occurring aquatic invertebrate communities in boreal foothill lake ecosystems. More specifically, in Chapter 2, I examine taxa-specific differences in forage fish abundance, size-structure, and habitat use. In Chapter 3, I investigate differences in abundance, community composition, and size structure of invertebrates in the presence and absence of stocked trout.

This research was conducted with the cooperation and assistance of fisheries managers from ACA and ASRD. Findings will be communicated to these managers and will thus help inform the implementation of Alberta's province-wide stocking and aeration program.

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## Chapter 2. Effects of stocked trout on native fish in boreal foothills lakes

### Introduction

Stocking sport fish is a long-standing and popular management tool used to create or enhance fisheries (Wiley 2006). Historically, many introductions of sport fish were unintentional, illegal or otherwise undocumented, such that no record of their impact exists (Pister 2001). In Alberta, many lakes have been stocked for decades and initial monitoring focused on the survival and growth of the sport fish (Miller and Thomas 1957) and not on how they affected the receiving ecosystem. More recently, it has been recognized that sport-fish stocking is often an introduction of non-native species. Thus, while stocking remains a successful fisheries management tool, impacts that introduced sport fish can have on ecosystems are of increasing interest (Cowx and Gerdeaux 2004).

Many commonly stocked sport fishes are piscivorous. Predation by piscivorous fish has long been recognized as significant for structuring prey populations. The abundance of many small-bodied prey fish is generally lower when coexisting with predators (Tonn 1985; Tonn and Paszkowski 1986; Persson *et al.* 1996), and species richness of the prey assemblage can also decrease (Ross 1991; MacRae and Jackson 2001), sometimes resulting in local extinctions (Chapleau and Findlay 1997; Findlay *et al.* 2000). Brown trout (*Salmo trutta*) has been directly linked to reductions in the abundance of many members of the Galaxiidae family in New Zealand (Townsend 1996) and to extirpations of prey species in Poland (Penczak 1999). Elsewhere, however, trout introductions have not decreased prey abundance (Pink *et al.* 2007) or caused other noticeable, negative effects (Whittier and Kincaid 1999).

Size-biased predation by predators can alter the size distribution of the prey. For example, smaller fathead minnows (*Pimephales promelas*) were found in lakes with northern pike (*Esox lucius*), which selectively eats larger minnows (Duffy 1998). Alternately, if predation is size-limited, then the prey population will be dominated by larger individuals (Tonn and Paszkowski 1986). Predation can also indirectly bring about changes in the size structure of prey populations. Predation can decrease density-dependent competition, thereby increasing growth rates of surviving prey (Heibo and Magnhagen 2005). Alternately, presence of a predator can reduce foraging activities of vulnerable prey, thereby reducing their growth (Tonn *et al.* 1992).

Alterations in habitat use, especially increasing use of bottom habitat, macrophytes, and other structures as refuge, are common behavioural responses to predation risk of many fishes (Bryan *et al.* 2002; Stuart-Smith *et al.* 2007a). Such habitat shifts may have population-level consequences in terms of growth and survival that develop and persist over longer time scales (Biro *et al.* 2003). As well, small-bodied taxa may alter their vertical distribution, for example, perch use the littoral zone in the presence of pike, and are not found below the thermocline (Persson *et al.*1996). Similarly, juvenile sunfish (*Lepomis* spp.) and mudminnows (*Umbra limi*) congregate inshore (Tonn and Paszkowski 1987; Harvey 1991), and dace (*Phoxinus* spp.) increase shoal size (Pink *et al.* 2007) in the presence of predators. Furthermore, behavioural responses to predators are often size-dependent even within a taxon (Holopainen *et al.* 1991; Magoulick 2004), since individuals of different sizes differ in their vulnerability (Tonn and Paszkowski 1986; Nannini and Belk 2006).

In some instances, the introduction of a predator will have no or only weak impacts on the native prey assemblage. Physiological constraints and environmental preferences of a predator may determine its distribution, limiting the contact it has with its prey (Rowe and Chisnall 1995; Dockray *et al.* 1996; Barwick *et al.* 2004; Leprieur *et al.* 2006). Prey can also avoid predation through behavioural changes and size refuge as discussed above. As well, a specialized predator may not pose an equal predation risk to all taxa (East and Magnan 1991; Macchi *et al.* 1999), while a generalist predator may prey on aquatic invertebrates as well as forage fish.

Introduced piscivores can also affect the productivity and nutrient cycling of the system into which they are introduced (Carpenter *et al.* 1985; Schindler *et al.* 2001). A piscivore can reduce the biomass of planktivores (such as forage fishes), which, in turn, can increase the biomass of herbivores, leading to a decrease in phytoplankton biomass (Carpenter *et al.* 1985). Piscivore-free lakes may have higher algal production because the planktivores decrease the herbivore biomass and therefore there is less suppression of the phytoplankton biomass (Carpenter and Kitchell 1988).

The relatively shallow and productive lakes of Alberta are prone to winterkill, which can cause large declines in forage fish abundance (Danylchuk and Tonn 2003), and is also a serious threat to trout, which have higher oxygen demands than forage fish (Moyle and Cech 2004). To protect stocked trout populations from over-winter mortality, some stocked lakes in Alberta are aerated in the critical winter months. Aeration allows for extended survival and increased growth of trout, and can also increase oxygen concentrations in parts of the lake, thereby expanding trout distribution (Prepas *et al.* 1997; Muller and Stadelman 2004). Effects that aeration may have on forage fish and their responses to introduced trout are unknown.

Sport fish stocking in Alberta boreal foothills lakes introduces a potential predator to native fish. Since trout are piscivorous (East and Magnan 1991) and can reduce prey populations in a number of regions (e.g., Townsend 1996; Penczak 1999), I expected abundances of forage fishes to be lower in lakes with trout than in lakes without. Because trout are size-limited predators (e.g., McIntosh 2000), I also expected to find fewer small forage fish in lakes with trout relative to unstocked lakes. Finally, I also expected changes in forage fish behaviour and activity, reflecting attempts by forage fishes to avoid trout. To test these predictions, I compared forage fish abundance, population size-structure, and habitat use in stocked and unstocked lakes over the course of 3 years.

## Methods

### *Study Area*

This research was conducted on 11 lakes in the boreal foothills of Alberta, Canada in the areas of Rocky Mountain House (52°22'38.94"N & 114°54'36.87"W) and Caroline (52° 5'35.69"N & 114°45'28.47"W) between May and August, 2005-2007. Lakes were placed into categories concordant with fisheries management practices: stocked (n=5; 2 of which were aerated), and unstocked (n=6). Lakes ranged in size and depth, but were generally small with various combinations of native forage fishes, including fathead minnow, brook stickleback, Iowa darter, and mixtures of pearl, northern redbelly, finescale dace, and dace hybrids (Table 2-1). Finescale and northern redbelly dace commonly hybridize in Alberta lakes (Das and Nelson 1989; Nelson and

Paetz 1992). External identifying features of the hybrids are variable, and accurate identification relies on meristic counts that require sacrificed fish (J. Mee, University of British Columbia, personal communication). Pearl dace were observed in few lakes, but were not caught consistently across years or in great numbers. As a result, dace populations were predominantly *Phoxinus*; consequently, all dace taxa were placed into one group for the purposes of this study (hereafter called "dace"). Brook, brown, and rainbow trout were present in the stocked lakes (Table 2-1), although taxa, stocking history, and angling regulations differed (Table 2-2). All brood stocks were from Alberta lakes, but rainbow and brook trout originated from California or Washington sources, and brown trout came from a strain of trout originally stocked in Banff National Park, Alberta; however, the ultimate origins of the stocked fish are not known (R. Konynenbelt, Alberta Sustainable Resource Development, personal communication).

The aerated lakes (Ironside and Mitchell) were treated with one ½-hp or two 1-hp floating aerators, respectively, from mid October to early April. Initial aeration on Ironside began in the fall of 2005, following my first field season, whereas Mitchell had been aerated since 2003 (C. Rasmussen, Alberta Conservation Association, personal communication). Two lakes (Strubel and Teal) had a small amount of shoreline development, but all other lakes had undeveloped shorelines. Shorelines and riparian zones were generally wooded, but in some cases the riparian zone was used for cattle grazing (e.g., Dog Leg, Picard). Most lakes were also associated with wetlands and thick beds of *Typha latifolia* were common along the shorelines. The dominant macrophytes in the study lakes were *Potamogeton* spp., *Sparganium angustifolium*, and *Nuphar variegatum* (C. Schank and L. Nasmith, personal observation).

*Water Chemistry*

Epilimnetic water samples were collected during the summer from each lake, once in 2005, four times in 2006, and twice in 2007. Water was collected at a depth of ca. 1 m from the center of a lake; 100-200mL samples were filtered on site through GFF filters, and frozen for later chlorophyll-*a* analysis. The rest of the unfiltered sample was refrigerated until processed within 7 days by the Biogeochemical Analytical Laboratory at the University of Alberta, Edmonton, Alberta. Variables measured by the Laboratory were total nitrogen (TN), total dissolved nitrogen (TDN), total phosphorous (TP), total dissolved phosphorous (TDP), and fluormetric (2005 only) or spectrophotometric chlorophyll-*a* (Chl-*a*). Procedures followed the guidelines of the Canadian Association for Environmental Analytical Laboratories (M. Ma, Biogeochemical Analytical Laboratory, personal communication).

Dissolved oxygen (DO) and temperature profiles were taken with an OxyGuard International Handy Mark II meter in June and August 2005, and monthly (May- August) in 2006 and 2007. Readings were recorded at 1 m intervals on the up- and down-cast and the average of the two readings was used for the profile. At the time of these profiles, single measures of epilimnetic pH and conductivity were also taken. Temperature loggers were deployed in the lakes in all years from June-August in 2005, May-August in 2006, and May/June-September in 2007. Two data loggers were set in each lake: one above and one below the thermocline, as determined by the May or June temperature profiles. Loggers were anchored and floated in the water column ca. 15cm off the bottom. Mean depths of deep data loggers in the 3 years (2005-2007) were 4.0, 4.7, and 5.5m, respectively. Data loggers above the thermocline were generally in the littoral zone, at mean depths of 0.8, 1.2, and 1.0 m for the 3 years (2005-2007, respectively).

*Forage Fish Population Estimates*

Mark-recapture population estimates of forage fishes were conducted between early May and early July in each of 3 years: on 2 lakes in 2005 (1 stocked, 1 unstocked), 8 lakes in 2006 (4 stocked, and 4 unstocked), and 3 lakes in 2007 (1 stocked, 2 unstocked). All lakes were sampled on 4-6 consecutive days. Between 20 and 30 unbaited Gee minnow traps (2 cm opening, 5 mm mesh) were set in randomly chosen locations (both in- and offshore) within the 3m isobath; preliminary sampling on these lakes indicated that most individuals were caught in waters less than 3m (L. Nasmith, personal observation). Random locations were determined by overlaying a grid on maps of each lake and using a random number generator to select trap locations, which changed daily. In lakes with low catch-per-unit-effort (CPUE), traps were set overnight (12-16h); lakes with high CPUE had 2-3h sets.

Once caught, forage fish were anaesthetized in clove oil (ca. 30 mg $l^{-1}$) and given a partial fin clip. Each forage fish population in a lake, including the combined "dace," was assessed separately, and fathead minnows were further separated as males, females, and juveniles based on sexual dimorphism (Danylchuk and Tonn 2001). Fish from different populations received different clips to facilitate faster identification of recaptures. After clipping, fish were placed in a bucket of clean lake water to recover from the anesthetic. After fish were observed to be swimming upright and behaving normally, they were released at least 5m away from the remaining traps. At each trap, the number of newly marked fish and previously marked fish were recorded.

Population estimates were made using the Schnabel method (Ricker 1975). Standard error (SE) for population estimates, however, were calculated from the range of daily Peterson estimates for days with 4 or more recaptures. This method was chosen because it provides an estimate of error based on observed variability, whereas other methods produce only a theoretical error value (Ricker 1975).

Mark-recapture calculations assumed a closed population in which all samples were randomly collected and all individuals had the same probability of being caught; it also assumed that marks were retained, observable, and did not affect catchability (Seber 1973). All but one lake (Gun Range, unstocked) contained closed populations, and to meet this assumption, an outlet to a nearby wetland complex was blocked with a seine net at the lake end of the outlet for the course of the sampling period, stopping fish from migrating to the wetland. Trapping behind the seine net on each day of sampling did not yield any clipped fish, so I assumed the blocking net was successful. The partial fin clips used were highly recognizable and were retained over the course of the mark-recapture period, as significant regrowth does not occur over 4-6 days. The effect of fin clips on forage fish survival was not directly studied, but similar studies that used clipping reported no unexpected mortality (e.g., Noraker *et al.* 1999; Wootton and Smith 2000; Danylchuk and Tonn 2003).

Catch-per-unit-effort values were calculated from sampling on all lakes in May-July of 2005 and 2006. On lakes that were not used for mark-recapture sampling, minnow trapping was conducted over 2 days. Trapping procedures followed those used for mark-recapture. For all lakes, CPUE was calculated for each trap in a lake as the number of fish caught per hour. These values were then averaged across traps to get a value for each taxon in a lake. Population sizes in lakes that were not determined by mark-recapture procedures were estimated based on a relationship between mark-recapture estimates and CPUE from the mark-recapture lakes (see "Statistical analyses" below). Population estimates based on CPUE are from the same sampling periods previously mentioned, with the exception of Ironside Pond (2005), and Teal and Picard lakes (2006), which were sampled in mid-late August.

*Forage Fish Size Structure*

For all sampling years, I measured total length (TL, mm) when possible, for a minimum of 100 fish for each taxon in each lake. In 2006, TL of male, female, and juvenile fathead minnows were recorded separately.

*Spatial Distribution and Temporal Activity*

Depth-stratified sampling was conducted over 24-h periods on 8 lakes in July/August 2006 (5 stocked, 3 unstocked) and 4 lakes in July 2007 (2 stocked, 2 unstocked). In 2006, 30 traps were set, 10 each at the bottom, surface, and midwater. Surface traps were suspended just below the surface of the water, whereas midwater traps were suspended in the column 1-1.5m below the surface. Of the bottom and surface traps, half were inshore (depths <1.5m) and half were offshore (2-4m). All midwater traps were offshore. Inshore and offshore distinctions were based on macrophyte densities, which typically began to thin in waters deeper than 2m. Trap locations were chosen randomly within the desired isobaths and trap locations remained fixed throughout the 24-h sampling period. Traps were set initially at 0500 and checked every 6 hours, with the last trap check being 0500 the following day. Every time a trap was checked, the number of individuals of each taxon was recorded, and a subsample were measured for TL (mm).

In 2007, the midwater trap was eliminated and the trap number was reduced to facilitate a higher sampling frequency. Twelve traps were set per lake, 6 at the surface and 6 on the bottom. For each stratum, half were inshore and half were offshore. Traps were checked every 2 hours, with check times bracketing sunrise and sunset. Illuminance levels (lux) were also recorded once every 2 hours with a Type 217 General Electric Light Meter. For both years, results were grouped into four sampling periods: Morning (0500-1100), Afternoon (1100-1700), Evening (1700-2300), and Night (2300-0500).

*Before and After Stocking and Aeration*

Ironside Pond was stocked for the first time in June 2005, and aerated for the first time the following fall and winter. I was able to compare certain metrics before and after stocking, and before and after aeration, to assess short-term impacts of these changes on forage fish size structure and population density (see "Statistical analyses" below).

*Statistical Analysis*

To avoid pseudoreplication (Stewart-Oaten *et al.* 1986), my unit of replication for assessing differences between the stocked and unstocked treatments was a lake. For all tests, results were considered marginally significant if $0.1 \geq p > 0.05$ and significant if $p \leq 0.05$. All statistical computations were performed using SPSS 16.0 (SPSS for Windows, Rel. 16.0.1. 2007).

To examine differences between treatments for environmental variables, CPUE, population densities, and mean length, two-factor repeated measures mixed model ANOVAs were performed with Lake as a subject within Treatment, and Year as the repeated factor. Main effects and interactions were tested for Treatment and Year.

To determine the relationship between CPUE and mark-recapture population estimates, linear regression was performed separately for each taxon, with population size as the dependent variable. Dace data were untransformed, and fathead minnow CPUE and population estimates were $\log_{10}(x+1)$ transformed. Regression analysis was also used to examine relationships between total catch and illuminance during the diel sampling in 2007. In that case, both data sets were $\log_{10}(x+1)$ transformed.

To assess size distributions, the mean proportion of "small" individuals was compared between treatments using the ANOVA procedure described above. The cutoff points (50mm for dace and fathead minnow, 45mm for brook stickleback) were chosen based on inspection of size distributions from my study systems. Proportion data were arcsine-square root transformed before analysis. I also examined quantile-quantile (QQ; *sensu* Post and Evans 1989) plots for differences in length distributions between treatments. A QQ plot transforms a length frequency distribution into a linear function by graphing the length at each quantile of the distribution. This linear representation of the distribution can then be compared to other similarly transformed distributions. The mean length distributions in stocked and unstocked lakes were compared at quantiles: 1, 5, 10, 25, 35, 50, 65, 75, 90, 95, and 99 for each taxon. These plots can be used to identify changes in shapes of distributions that might be due to size-limited predation (Post and Evans 1989).

To test for significant vertical and horizontal movement of individuals, I compared catches among the four sampling periods within treatments. This was done separately for the proportions of fish caught in bottom traps, and those in inshore traps. Proportions were arcsine square root transformed and compared using Kruskall Wallis tests. To test for a treatment effect on the transformed proportion of fish in bottom traps, I used two-factor repeated measures mixed model ANOVAs with Lake as the subject within Treatment, and Sampling period as the repeated factor. This test was also performed on the inshore trap data.

To look at the impact of trout stocking and aeration on Ironside Pond, two separate Before-After-Control-Impact (BACI) analyses were conducted. These analyses were computed as replicated, two-factor ANOVAs, with Before-After, and Control-Impact as the main effects. A significant F-ratio for the interaction term (Before-After*Control-Impact) in this test indicates that there is a difference between the Control and Impact lakes that varies with the Before and After time periods (Downes *et al.* 2002). To look at trout impact, forage fish total length data from Ironside Pond (Impact) in May 2005 (Before) and August 2005 (After) were compared to data from the same sampling periods in Dog Leg Lake (Control). Dog Leg was chosen because it is of similar size to Ironside, and it was sampled around the same period. To look at aeration impact on length, TL data from Ironside Pond (Impact) in August 2005 (Before) and August 2006 (After) were compared to data from the same sampling periods in Strubel Lake (Control). To look at the impact of aeration on density, population density estimates from 2005 (Before) and 2006 (After) were used. Strubel was chosen for these latter analyses because its forage fish assemblage was most similar to Ironside's, and both lakes were stocked with rainbow trout.

## Results

*Water Chemistry*

Mitchell (stocked) and Picard (unstocked) lakes were generally isothermal but all other lakes showed stratification. Thermal stratification often prevented DO mixing below 3-6m, and metalimnetic DO peaks were seen between June and early August in both stocked (Ironside, Strubel, Birch) and unstocked (Gun Range and Fiesta) lakes. Otherwise, DO was well-mixed in the top 1-4m of the water column (Appendix A).

Stocked and unstocked lakes differed in all variables, except conductivity and shallow temperature (Table 2-3). An effect of Year was seen in deep and shallow logger values, TDN, TP, Chl-*a*, and marginally in TDP. Deep temperature, DO, and pH were higher in stocked lakes, whereas TN, TDN, TP, TDP, and Chl-*a* were all higher in unstocked lakes. Interaction between Year and Treatment was significant for deep temperature and Chl-*a*. Chlorophyll-*a* and TP values indicated that the stocked lakes were oligo-mesotrophic and unstocked lakes meso-eutrophic (Carlson 1977).

Stocked lakes with and without aeration did not differ for most variables measured. However, both TDN and TDP were marginally greater in lakes with aeration ($F_{1,9}$=3.4, p=0.09, and $F_{1,8}$=4.9, p=0.06, respectively) and pH was marginally greater in lakes without aeration ($F_{1,9}$=5.0, p=0.06).

*Relative and Absolute Abundances*

Dace were the most commonly captured forage fish in most lakes in 2005 and 2006, though cyprinid populations varied greatly both among lakes and years (Table 2-4). Catch-per-unit-effort was also highly variable between treatments and years (Table 2-5). Catch-per-unit-effort was highest in stocked lakes, marginally for dace ($F_{1,14}$ = 3.3, p=0.09) and significantly for fathead minnow ($F_{1,9}$ = 5.3, p=0.05). Brook stickleback CPUE did not differ between treatments (p>0.6). There were no significant Year effects, or Treatment*Year interactions on CPUE for any of the forage fish.

The relationship between dace CPUE and abundance estimate was significant and positive and characterized as: Population estimate = 1165.5(CPUE) + 30,193 ($r^2$=0.46, $F_{1,11}$=8.74, p=0.017). Although fathead minnows were initially sorted into different life history groups, sample sizes were small and variability was high, preventing development of separate relationships between CPUE and abundance estimates. After grouping the life stages together the relationship was: Population estimate = 0.836 ([$\log_{10}$(x+1)]CPUE) + 3.7 ($r^2$=0.79, $F_{(1,5)}$ = 18.62, p=0.008). Due to insufficient recaptures, no relationship could be documented between the brook stickleback CPUE and population estimates.

*Densities*

Dace densities ranged from 617 fish/ha (stocked, Yellowhead, 2006) to 18,000 fish/ha (stocked, Ironside, 2006; Figure 2-1A,B). Fathead minnow was present in 8 of 11 study lakes. Densities were lowest in stocked lake Birch (194 fish/ha) and highest in unstocked lake Gun Range (3713 fish/ha), both in 2006 (Figure 2-1C,D). There was no effect (p>0.1) of either Treatment or Year on dace or fathead minnow densities. Because of the predominance of dace in all the lakes, combined cyprinid densities give the same results as those of dace.

For 3 lakes I was able to study changes in dace density over 3 years (Figure 2-2A). The stocked lake had similar densities during 2005 and 2006, but dace increased significantly in 2007 (Kruskal-Wallis, $\chi^2$=7.4, df=2, p=0.03). Dace increased in the unstocked lake Dog Leg, between 2005 and 2006 ($\chi^2$=7.5, df=2, p=0.02), but density did not change in 2007. This was similar to the other unstocked lake, Fiesta, in which density increased between 2005 and 2006 ($\chi^2$=24.1, df=2, p<0.001), but did not change in 2007.

Fathead minnow populations were observed for 3 years in 2 unstocked lakes (Figure 2-2B). During this time, fathead minnow density in Dog Leg Lake did not change (Kruskal-Wallis , $\chi^2$=3.82, df=2, p=0.15), but in Fiesta, the density was marginally higher in 2006 ($\chi^2$=5.34, df=2, p=0.069), compared to 2005 and 2007.

*Size Structure*

Total length of dace was larger in lakes with trout ($F_{1,12}$=7.0, p=0.02; Table 2-6); there was no effect of Year, nor a significant (Treatment*Year) interaction. This analysis could only be made using 2005 and 2006 data, as the data collected in 2007 were from a subset of the study lakes. In 2007, average lengths of dace did not differ between stocked and unstocked lakes (Mann-Whitney, U=0, p=0.2).

In 2005 and 2006, fathead minnows were slightly larger in stocked lakes than unstocked lakes (Table 2-6) but the difference was not significant ($F_{1,8}$=2.1, p=0.2). There was also no effect of Year on fathead minnow TL, nor an interaction effect. In 2006, I analyzed the life history groups separately. Males, females, and juveniles each tended to be larger in the stocked than in unstocked lakes; however, comparisons for each group, as well as the combined probability test (Sokal and Rohlf 1981), were not significant (p>0.1). Brook stickleback TL did not differ between treatments, or years, and there was no interaction (Table 2-6).

Lakes with trout generally had lower proportions of small dace (<50mm) than unstocked lakes (Figure 2-3A), but this was not significant ($F_{1,13}$=2.4, p=0.15). Fathead minnow populations had marginally lower proportions of small individuals (<50mm) in stocked lakes ($F_{1,11}$=5.4, p=0.06; Figure 2-3B). Proportions of small brook stickleback (<45mm) did not differ between stocked and unstocked lakes ($F_{1,11}$=0.8, p=0.4; Figure 2-3C). There were neither Year (p>0.4) effects or interactions (p>0.3) for any of the taxa.

The quantile-quantile regression plots of transformed forage fish distributions in stocked and unstocked lakes were highly significant for all taxa (p<0.001; Figure 2-4). The slopes for fathead minnows and brook stickleback did not differ from 1 (t-test, p>0.2), indicating that the length distributions were not different in stocked and unstocked lakes (Post and Evans 1989). If there is selection of small individuals by predators in stocked lakes, the slope will be <1, and quantiles will deviate above the 1:1 line (Post and Evans 1989). The slopes for dace were significantly >1 (p=0.002). The plots suggest that dace in stocked lakes were larger than those in unstocked lakes.

*Spatial Distribution and Temporal Activity*

In both years, dace and fathead minnows were found most often in bottom traps in lakes with trout, but were more often in the water column (surface and midwater) in unstocked lakes (Figure 2-5A-C). The effect of treatment on the proportion of fish in bottom traps was significant for dace in 2006 ($F_{1,20}$=32.6, p<0.001) and 2007 ($F_{1,7}$=30.4, p=0.001), and fathead minnow in 2006 ($F_{1,11}$=17.8, p=0.001). Brook stickleback were found predominantly in bottom traps, regardless of treatment (p>0.3; Figure 2-5D,E). There were no significant changes in the proportion of fish in bottom traps among the 6h sampling periods (p>0.2), within treatments for any taxa.

There was no difference (p>0.2) in the size of fish caught in the different strata within treatments for fathead minnow and brook stickleback in both years, and dace in 2007 (Table 2-7). In 2006, there was no difference in dace size among strata in unstocked lakes (p=0.5), but there was a difference in stocked lakes ($\chi^2$=6.8, df=2, p=0.03). Post hoc pairwise tests showed that dace in midwater traps did not differ from those in surface (p=0.2) or bottom (p=0.4) traps, but that dace in stocked lakes were significantly larger in bottom traps relative to surface traps (U=4.0, p=0.008).

Dace were generally found more often inshore for all treatments and sampling periods (Figure 2-6A,B). There was no effect of Treatment or Time period on inshore/offshore distribution of dace in either year, or for fathead minnows (2006). In 2006, brook stickleback in unstocked lakes were offshore, whereas about half were found inshore in stocked lakes (Figure 2-6). There was a significant effect of Treatment for that year ($F_{1,13}$=5.63, p=0.03). However, in 2007, brook stickleback were found mostly inshore in unstocked lakes, but were mostly offshore in stocked lakes, except in the afternoon (Figure 2-6). This relationship could not be tested because only one lake with trout and brook stickleback was sampled that year. The proportion of fish inshore did not differ (p>0.4) among sampling periods within treatments for either dace or fathead minnow in either year. In 2006, a marginally smaller proportion of brook stickleback in stocked lakes were inshore at night relative to other sampling periods (t=1.395, df=8, p=0.1), but no change was seen in unstocked lakes (p>0.4). In 2007, stocked lakes could not be tested, but in unstocked lakes, a smaller proportion of sticklebacks were inshore at night relative to other sampling periods (t=4.53, df=5, p=0.001).

Forage fish activity (all taxa) in 2006 peaked during the morning and decreased throughout the day and was lowest at night when often no fish were caught (Figure 2-7). Higher sampling frequencies in 2007 allowed for a closer look at activity. Dace in stocked lakes had a large peak of activity at 0630, with much smaller peaks between 1230 and 1430 and again at 2300. Catches were generally higher in unstocked than in stocked lakes throughout the day, 0830 - 1900. Catches were minimal at night in both treatments.

In 2007, dace catch correlated significantly to light meter readings and light explained 42% of the variation in catch in unstocked lakes (Figure 2-8; $F_{1,23}$= 16.6, p<0.001), and 34% of variation in stocked lakes ($F_{1,22}$=11.17, p=0.003). Brook stickleback catches in 2007 were generally low. Nevertheless, light explained 87% of the variation in catch in stocked lakes ($F_{1,10}$=66.9, p<0.001) and 43% of variation in unstocked lakes ($F_{1,23}$=17.47, p<0.001).

*Before and After Stocking and Aeration*

To examine short-term effects of trout stocking on size, the mean TL of dace was compared before and after stocking in Ironside Pond relative to an unstocked control lake (Dog Leg Lake). In both lakes, mean TL increased over time, and there was no interaction between before-after and control-impact (p=0.6; Figure 2-9A). A similar test was performed to look at the effect of aeration on size, using an unaerated, stocked control lake (Strubel Lake). The mean length of dace after aeration was smaller than before aeration, but this was true in both lakes and there was no interaction ($p=0.2$; Figure 2-9B). However, there was a significant increase in the dace density in Ironside after stocking-aeration, and a decrease in density in the control, and the interaction term was significant ($F_{1,8}=25.9$, $p=0.001$; Figure 2-9C).

**Discussion**

The forage fishes in my study responded to the presence of trout in a variety of ways. Neither size nor habitat use of brook stickleback was affected by trout presence, nor did cyprinid densities differ between stocked and unstocked lakes. However, there was evidence of possible size-limited predation in the trend towards larger mean sizes and smaller proportions of small cyprinids in stocked lakes relative to unstocked lakes. In the presence of trout, cyprinids stayed inshore, close to the bottom. This contrasted with cyprinids in unstocked lakes, which mostly used the water column, and were as likely to be inshore as offshore. Forage fish activity correlated with illuminance levels in stocked and unstocked lakes.

Forage fish metrics in aerated lakes did not differ from those seen in lakes that were stocked but not aerated. Furthermore, a BACI test of aeration in Ironside Pond indicated no change in dace size after aeration commenced in Ironside, but population density actually increased in the year after stocking-aeration began.

In lakes with trout, indicators of productivity (e.g., Chl-*a*, TP) were significantly lower than in unstocked lakes. In stocked lakes that were aerated, there were greater concentrations of TDN and TP, and a lower pH. Otherwise, aeration did not affect the water quality variables that I measured.

*Trout predation*

The trout species stocked in my lakes are all considered piscivorous (Lacasse and Magnan 1992; Kahilainen and Lehtonen 2001; Mazur and Beauchamp 2003), and in other systems, forage fish have been stocked to provide a prey base for trout (Van Zyll de Jong *et al.* 2004). However, the piscivorous nature of trout is highly variable, both within (L'Abée-Lund *et al.* 1992) and among (Rowe *et al.* 2003) species, among lakes (East and Magnan 1991), and temporally within lakes (Museth *et al.* 2003). Important factors affecting the degree of piscivory in trout, including density and size structure of their populations (East and Magnan 1991), were not studied directly.

As piscivory is a function of fish size, lakes in which trout grow faster and attain larger sizes (i.e., a trophy fishery) would presumably have more piscivorous individuals, which should affect the forage fish population more strongly. This was not the case, however, in Ironside Pond, which in fact had the highest density of dace in the study. In that lake, forage fish made up 5% of trout diet, based on numerical abundance (stomach content data, J. Hanisch, University of Alberta, personal communication). Yellowhead Lake (stocked) had the most piscivorous trout population (brook trout; J. Hanisch, University of Alberta, personal communication), as well as the lowest dace density, but also supported one of the highest densities of fathead minnow, a species exceptionally sensitive to predation (Moody *et al.* 1983; Robinson 1989; Savino and Stein 1989).

*Forage Fish Abundance and Density*

Although I did not know population sizes in my lakes prior to the initial stockings, over the time of my study the densities of forage fish in stocked and unstocked lakes were not systematically different. This is contrary to many examples of negative effects of trout (Townsend and Crowl 1991; Townsend 1996; Penczak 1999) or other piscivores (Tonn 1985; MacRae and Jackson 2001) on small-bodied species. However, in some instances trout do not have strong negative effects on forage fish (Macchi *et al.* 1999; MacRae and Jackson 2001).

Increases and decreases in forage fish densities were seen in both treatments over the course of my study. Similarly, an increase in minnows (*P. phoxinus*) co-existing with brown trout was documented in a Norway subalpine lake (Museth *et al.* 2002). Similarly, Pink *et al.* (2007) saw increases in forage fish abundance (including fathead minnow and northern redbelly dace) over the course of 2 years in the presence of brook trout in northern Ontario lakes. The latter authors suggested that a longer time period (>2 years) could be needed to observe population impacts on forage fishes, or that brook trout were feeding mainly on invertebrates. Since some of my lakes have been stocked for >50 years, the former does not seem likely.

Ironside Pond provided an opportunity to observe short-term impacts on densities following stocking. The population density of dace increased in 2006, the first summer following stocking. The lake was not sampled in 2007, but density estimates from 2008 were nearly identical to 2006 (>17,000 dace/ha; J. Hanisch, University of Alberta, personal communication). If predation does not decrease densities, it is likely that trout are relying primarily on other prey sources.

A number of my density estimates were based on the relationship I found between abundance and CPUE. I found a relationship for fathead minnow similar to that found by Danylchuk (2003) for fathead minnows in piscivore-free Boreal Plains lakes. While not as strong, my relationship for dace was still significant. These relationships should be tested with more populations, but my data further indicate that CPUE reflects absolute abundance of small-bodied fish both in the presence and absence of trout.

*Size Structure*

Prey response to potential predators can be size-dependant (Magoulik 2004). Prey populations in the presence of trout are often dominated by larger size classes (McIntosh *et al.* 1994; Johnson and Belk 1999). Optimum prey size for trout in my study system is not known, but preliminary stomach content analysis put forage fish prey sizes (TL) at 13-21mm for trout <350mm, and 42-53mm for trout >400mm (J. Hanisch, University of Alberta, personal communication). This range of prey size for smaller trout suggests use of fish smaller than those reported for similarly sized trout (40-60 and 48-94mm; East and Magnan 1991, and McIntosh 2000, respectively). Thus, the trout in my system may be capable of consuming larger prey but are selecting smaller prey, as was seen with brown trout in a boreal lake in Finland (Hyvarinen and Huusko 2006). I found trends towards lower proportions of small fish in stocked lakes, and correspondingly greater proportions of large fish, relative to unstocked lakes, suggesting predation is focused on smaller individuals.

Predators can positively affect growth rate by releasing prey from competition; i.e., surviving prey will have more resources available per capita and therefore should grow faster (Van Buskirk and Yurewicz 1998; Peacor 2002; Vandenbos *et al.* 2006). In some populations, decreased densities and subsequent increased growth rate can be more important in structuring prey size than the selective consumption of smaller individuals (Craig *et al.* 2006). For the 2 years of my study, fish in stocked lakes with low forage fish densities were larger (TL) than forage fish in high-density stocked lakes, but this trend was not significant. Also, there was relatively little variation in the proportions of small bodied dace among stocked lakes. All this suggests that thinning is not likely contributing to the observed structure of prey populations, and that direct predation is likely more important in structuring populations.

*Interspecific Competition*

Aside from predation, trout can negatively affect forage fish by competing with them, either for resources or habitat. McDowall (2006) includes competitive displacement among plausible mechanisms for detrimental effects of trout on populations of native New Zealand Galaxiidae (see also McIntosh *et al.* 1992). In some cases, competitive interactions between trout and small-bodied potential prey can be detrimental to the trout (Magnan 1988; Museth *et al.* 2007). However, other research has shown little diet overlap between brown trout, three-spine stickleback, and *Phoxinus* minnows (Bolger *et al.* 1990) or between northern redbelly dace and brook trout (Scott and Crossman 1985).

When the stocking program in Alberta commenced in 1950, it was believed that no significant competition occurred between introduced trout and native forage fish, and that the latter did not affect survival or growth of the trout (Miller and Thomas 1957). Competition with trout would likely result in smaller sizes and/or lower densities of forage fish in stocked relative to unstocked lakes (Aday *et al.* 2003), but this was not seen in my lakes. Although this study was not focused on competition, the evidence available suggests that competition in these lakes is not strong.

*Spatial Distribution and Temporal Activity*

Cyprinds in stocked lakes were found in significantly greater proportions in bottom traps, whereas most cyprinids in unstocked lakes were found in the water column. Although cyprinids in stocked lakes tended to be predominantly inshore, their inshore/ offshore distribution did not differ from cyprinids in unstocked lakes. In contrast to cyprinids, brook stickleback in both treatments were found on the bottom, and in stocked lakes they were found more inshore than in unstocked lakes. Many studies document diel offshore migration in *Phoxinus* species (Naud and Magnan 1988; Gauthier and Boisclair 1997; Comeau and Boisclair 1998; Gaudreau and Boisclair 2000), but I was only able to detect movement offshore at night by brook stickleback in stocked lakes (2007).

A dramatic increase in use of bottom habitat by forage fishes in the presence of trout represents a common response of fish to predators. Seeking refuge on the bottom, and/or shifting resource use in the presence of a predator is seen in many species, such as sunfish (*Lepomis sp.*; Harvey 1991), Japanese dace (*Tribolodon hakonensis*; Katano *et al.* 2003), fathead minnow (Savino and Stein 1989), Little Colorado spinedace (*Lepidomeda vittata*; Bryan *et al.* 2002), and the European minnow (Museth *et al.* 2002). Although some fishes may be naturally benthic, dace (*P. eos* and *P. neogaeus*) were not strongly bottom oriented in lakes without predators (He and Lodge 1990). By concentrating their distribution near the bottom in stocked lakes, cyprinids in my study were responding to trout, in terms of their habitat distribution, in a classic fashion.

Activity levels of forage fishes can be affected by a number of factors, including the presence of a predator (Jacobsen and Berg 1998). Indeed, many prey species appear to simply decrease activity in the presence of a predator, rather than alter the periodicity of that activity (Reebs 2002). This could explain why the forage fish in my system did not become more active at night to avoid predation by trout, which are visual predators. Although the cyprinids in stocked lakes changed their habitat, the basic patterns of activity were the same in stocked and unstocked lakes, and all forage fish were apparently responding similarly to light cues in lakes regardless of the presence of trout.

*Prey Refuge*

My lakes contain abundant populations of forage fishes and these were largely unaffected by trout. Aside from a possible preference for invertebrate prey (Radke *et al.* 2003), and the predation avoidance afforded by changes in size and distribution already discussed, other factors could limit predation on forage fish. East and Magnan (1991) posited that lake-specific biotic and abiotic factors, including predator size, fish community structure, structural complexity of the littoral zone, and refuge for prey species, can dictate the piscivorous behaviour of brook trout.

MacRae and Jackson (2001) suggested that effects of salmonids on prey populations can be limited by thermal characteristics of a lake in summer. This barrier would effectively protect prey during crucial periods of reproduction and growth. Rowe and Chisnall (1995) found that temperature was the main variable determining vertical distribution of rainbow trout in New Zealand lakes in summer months, if oxygen was not limiting. Similarly, thermal stratification of a deep, eutrophic reservoir had significant impact on the distribution of brown trout (Radke *et al.* 2003). Preferred temperature for rainbow trout varies from 8.3 to13.4 $^{o}$C (Barwick *et al.* 2004), but can be as high as 16$^{o}$C (Schurmann *et al.* 1991). This is similar to optimal temperature for brown trout (16$^{o}$C; Forseth and Jonsson 1994) but cooler than that of brook trout (20$^{o}$C; Peterson *et al.* 1979). In my lakes, the mean summer (May-August) epilimnetic water temperature in stocked lakes ranged from 19 to 22$^{o}$C, with maximum temperatures between 23 and 31 $^{o}$C. Thus, thermal stratification seen in many stocked lakes (Appendix A) may be sufficient in some of the summer months to keep trout away from the majority of forage fish that inhabit the inshore, shallow water.

Complex habitat, specifically macrophytes, can offer refuge for prey species. The presence of structural refuge can reduce predator attacks and mortality of small-bodied fish (East and Magnan 1991; Stuart-Smith *et al.* 2007a). Although macrophyte presence does not guarantee the safety of forage fishes (Savino and Stein 1989; Stuart-Smith *et al.* 2007a) and may reduce their feeding (Diehl 1992; Stuart-Smith *et al.* 2007b), it is likely that macrophytes in these systems serve as functional refuges, and if they are being used in my system, their use may reduce predation risk experienced by the forage fish.

*Lake Productivity and Aeration*

Stocked lakes that were aerated had elevated concentrations of TDN, TDP, and a lower pH than unaerated stocked lakes. Increased nutrient concentrations following aeration has been seen in pothole lakes (Taggart and McQueen 1981) and tank experiments (Sengupta and Jana 1987). Although aeration elevated some measures of productivity, unstocked (and unaerated) lakes had the highest concentrations of nutrients in this study, perhaps because they lacked piscivores (Carpenter *et al.* 1985).

Increases in population densities following aeration have been seen in large-bodied fish (Aku and Tonn 1997). Aeration might indirectly affect forage fish though expansion of trout habitat, or directly through increased survival of forage fish. It is not uncommon for small-bodied fish to be winterkilled in small lakes in boreal Alberta (Danylchuck and Tonn 2003) and since aeration is sufficient to sustain trout with their higher oxygen demands, aeration should also benefit small-bodied fish. Some of the highest densities of forage fish were observed in stocked and aerated lakes, and the BACI test indicated that the increase in density in Ironside Pond after aeration was significant. However, my BACI analysis was based on only one pre-aeration sample, and did not account for the natural variation of forage fish populations in Ironside. Dace did not differ in size or distribution in aerated vs. unaerated stocked lakes. If aeration is altering the nutrient concentrations of lakes, or increasing suitable habitat for trout, there is no evidence of a corresponding alteration in the relationship between trout and forage fish.

*Conclusion*

Trout in this study are occasionally eating forage fish and, although predation does not appear to be strong enough to cause differences in density, I have associated the presence of trout with some differences in the size structure and habitat use of forage fishes. Lack of an effect on population densities of cyprinids reflect a combination of behavioural changes, a body size refuge, use of physical refuge by forage fish, summer thermal characteristics, and an abundance of alternative macroinvertebrate prey for trout. Although trout predation was likely focused on the smaller size classes of forage fish, it did not negatively affect forage fish recruitment over the 3 years of the study. Aeration is clearly not exacerbating the effect of trout, but may, in fact, be also benefiting forage fish. These results are not altogether typical of native fish responses to introduced trout (McIntosh *et al.* 1994; Townsend 1996; Penczak 1999), but responses are not globally uniform, and will depend on the unique characteristics of the species involved and of the ecosystems in which they are found.

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**Table 2-1** Morphometry and fish species present in the 11 study lakes.

| Treatment | Lake[1] | Area (ha) | Max. depth (m) | Forage fish Populations[2] | Trout Populations[3] |
|---|---|---|---|---|---|
| Stocked | Ironside | 3.3 | 12.5 | Dace | RNTR |
| | Mitchell | 15 | 6 | Dace | RNTR, BNTR |
| | Strubel | 25.9 | 12.5 | Dace, BRST | RNTR |
| | Birch | 28.8 | 8.5 | Dace, BRST, FTMN | BKTR |
| | Yellowhead | 24.5 | 12.2 | Dace, BRST, FTMN, IWDR | BKTR |
| Unstocked | Dog Leg | 6.7 | 5 | Dace, BRST, FTMN | n/a |
| | Fiesta | 7.1 | 6.6 | Dace, BRST, FTMN | n/a |
| | Gun Range | 5.9 | 13.4 | Dace, BRST, FTMN | n/a |
| | Gas Plant | 17.5 | 3.9 | Dace, BRST, FTMN | n/a |
| | Teal | 16.5 | 9 | Dace, BRST, FTMN | n/a |
| | Picard | 8.7 | 5.4 | BRST, FTMN | n/a |

[1]Mitchell, Strubel, and Birch are official names, all other lakes are unofficial names used by Alberta Sustainable Resource Development's Clearwater District office.

[2]Dace: finescale dace (*Phoxinus neogaeus*), northern redbelly dace (*P. eos*), finescale x northern redbelly hybrids (*P. eos* x *P. neogaeus*), and pearl dace (*Margariscus margarita*); FTMN: fathead minnow (*Pimephales promelas*); BRST: brook stickleback (*Culaea inconstans*); IWDR: Iowa darter (*Etheostoma exile*).

[3]RNTR: rainbow trout (*Oncorhynchus mykiss*); BNTR: brown trout (*Salmo trutta*); BKTR: brook trout (*Salvelinus fontinalis*).

**Table 2-2:** Stocking data for the 5 trout-bearing lakes in the study. Numbers and stocking sizes are data from the 2005 stocking (S. Herman, Alberta Sustainable Resource Development, personal communication).

| Lake | Year of initial stocking; current frequency | Numbers and species[1] | Stocking Sizes (cm) | Adult sizes (mm; mean ± SE (n))[2] | Regulations |
|---|---|---|---|---|---|
| Ironside | 1977;<br>Stopped 1987;<br>Resumed 2005;<br>biennially | 500 RNTR | >15 | 432 ± 7 (94) | Mid April - late October;<br>Trout limit 0 |
| Mitchell | 1950; annually | 1000 BNTR<br>4000 RNTR | 5-10<br>10-15 | 266 ± 5 (120) | Open all year; Trout limit 5 |
| Birch | 1983; biennially | 15800 BKTR | 5-10 | unknown | Open all year; Trout limit 5 |
| Strubel | 1950; annually | 19000 RNTR | 10-15 | 250 ± 4 (142) | Open all year; Trout limit 5 |
| Yellowhead | 1983; biennially | 17400 BKTR | 5-10 | 374 ± 5 (18) | Open all year; Trout limit 5 |

[1] RNTR: rainbow trout; BNTR: brown trout; BKTR: brook trout.

[2]2007 data: J. Hanisch, University of Alberta, unpublished data

**Table 2-3** Summary of the physical, chemical, and biological properties of the study lakes for 2005-2007. All values are the mean of the lakes in the treatment category (values for each lake are means of 4 months) ± SE. Shallow and deep refer to temperature logger positions above and below the thermocline, respectively. Dissolved oxygen values are metalimnion means. TN: total nitrogen; TDN: total dissolved nitrogen; TP: total phosphorous; TDP: total dissolved phosphorous, Chl-*a*: fluormetric chlorophyll-*a* (2005 only) and spectrophometric chlorophyll-*a* (2006 and 2007). ANOVA results are from two-factor repeated measures mixed model ANOVAs (see text) performed on all years, with the exception of chlorophyll-*a*, for which only 2006 and 2007 data were used.

| **Treatment (n)** | **Shallow logger °C** | **Deep logger °C** | **Dissolved oxygen $mgL^{-1}$** | **TN $\mu gL^{-1}$** | **TDN $\mu gL^{-1}$** | **TP $\mu gL^{-1}$** | **TDP $\mu gL^{-1}$** | **Chl- *a* $\mu gL^{-1}$** | **pH** | **Conduct-ivity $\mu Scm^{-1}$** |
|---|---|---|---|---|---|---|---|---|---|---|
| **2005** | | | | | | | | | | |
| Stocked (5) | 19.2 ± 0.3 | 14.5[1] | 5.8 ± 1.1 | 862 ± 98 | 772 ± 60 | 18.6 ± 4.8 | 5.8 ± 0.9 | 0.7 ± 0.3 | 8.3 ± 0.1 | 130.1 ± 35 |
| Unstocked (6) | 18.3 ± 0.3 | 14.9 ± 0.6 | 4.0 ± 0.6 | 1029 ± 33 | 1010 ± 58 | 45.0 ± 4.3 | 19.9 ± 2.9 | 0.9 ± 0.3 | 7.7 ± 0.2 | 140.4 ± 26 |
| **2006** | | | | | | | | | | |
| Stocked (5) | 20.2 ± 0.9 | 14.9 ± 1.5 | 7.1 ± 0.9 | 785 ± 111 | 663 ± 61 | 15.0 ± 1.8 | 5.8 ± 0.7 | 3.6 ± 0.6 | 8.0 ± 0.2 | 135.2 ± 37 |
| Unstocked (6) | 20.5 ± 0.3 | 12.2 ± 1.3 | 4.7 ± 0.8 | 992 ± 30 | 884 ± 54 | 36.7 ± 3.9 | 13.7 ± 1.9 | 9.7 ± 1.1 | 7.9 ± 0.2 | 150.5 ± 28 |
| **2007** | | | | | | | | | | |
| Stocked (5) | 19.2 ± 0.3 | 14.0 ± 1.2 | 7.2 ± 1.1 | 732 ± 68 | 634 ± 60 | 14.2 ± 1.8 | 5.8 ± 0.4 | 2.8 ± 0.7 | 8.0 ± 0.07 | 137.6 ± 38 |
| Unstocked (6) | 18.5 ± 0.4 | 7.3 ± 1.1 | 4.2 ± 0.4 | 906 ± 36 | 787 ± 25 | 29.2 ± 3.6 | 12 ±1.4 | 5 ± 0.9 | 7.8 ± 0.1 | 151.2 ± 26 |
| **ANOVA results** | | | | | | | | | | |
| Treatment | NS | $F_{1,19}$=9.66, p=0.006 | $F_{1,24}$=10.6, p=0.003 | $F_{1,24}$=11.0, p=0.003 | $F_{1,24}$=21.7, p<0.001 | $F_{1,22}$=50.9, p<0.001 | $F_{1,19}$=48.4p <0.001 | $F_{1,18}$=21.1, p<0.001 | $F_{1,19}$=5.8, p=0.03 | NS |
| Year | $F_{2,16}$=7.4, p=0.005 | $F_{2,14}$=7.39, p=0.007 | NS | NS | $F_{2,19}$=6.1, p=0.009 | $F_{2,20}$=3.6, p=0.045 | $F_{2,17}$=2.7, p=0.09 | $F_{1,18}$=9.3, p=0.007 | NS | NS |
| Treatment*Year | NS | $F_{2,14}$=5.6, p=0.02 | NS | NS | NS | NS | NS | $F_{1,18}$=4.5, p=0.05 | NS | NS |

[1] Only one deep temperature logger was retrieved in 2005

**Table 2-4** Population estimates (mean ± SE) of dace and fathead minnow for all sampling years. All estimates generated from mark-recapture sampling unless otherwise noted (see text for methods).

| Treatment | Lake | Year | Dace | Fathead minnow |
|---|---|---|---|---|
| Stocked | Ironside | 2005 | 40,408 ± 4584[1] | - |
| | | 2006 | 58,629 ± 4391 | - |
| | Mitchell | 2005 | 50,431 ± 931 | - |
| | | 2006 | 51,293 ± 2249 | - |
| | | 2007 | 88,055 ± 3926 | - |
| | Birch | 2005 | 40,236 ± 2399[2] | 21,620 ± 2899[2] |
| | | 2006 | 31,488 ± 405[2] | 5594 ± 101[2] |
| | Strubel | 2005 | 69,863 ± 6876[2] | - |
| | | 2006 | 60,486 ± 6718 | - |
| | Yellowhead | 2005 | 32,474 ± 1818[2] | 29,112 ± 11496[2] |
| | | 2006 | 15,119 ± 4702 | 49,054 ± 5757 |
| Unstocked | Dog Leg | 2005 | 8491 ± 377 | 5773 ± 110 |
| | | 2006 | 28,025 ± 3758 | 9720 ± 2853 |
| | | 2007 | 32,821 ± 2163 | 5812 ± 4055 |
| | Fiesta | 2005 | 31,627 ± 955[2] | 5439 ± 165[2] |
| | | 2006 | 76,639 ± 5269 | 14,004 ± 1997 |
| | | 2007 | 69,390 ± 9623 | 5482 ± 1838 |
| | Gas Plant | 2005 | 33,896 ± 1724[2] | 7547 ± 1395[2] |
| | | 2006 | 57,906 ± 5530 | 5467 ± 59[2] |
| | Gun Range | 2005 | 31,245 ± 426[2] | 16,218 ± 4740[2] |
| | | 2006 | 14,431 ± 505 | 21,910 ± 3032 |
| | Teal | 2005 | 37,821 ± 2186[2] | 8763 ± 1663[2] |
| | | 2006 | 54,807 ± 5305[2] | 8137 ± 1049[2] |
| | Picard | 2005 | n/a | 5153 ± 1[2] |
| | | 2006 | 31,212 ± 19[2] | 8830 ± 1682[2] |

[1]Ironside estimates made from a small sample size of CPUE values collected in late August 2005.

[2]Values estimated from CPUE from sampling May-mid July of each year (see text).

**Table 2-5**: Mean ± SE catch-per-unit-effort (fish/hour/trap) by taxon for stocked and unstocked lakes in 2005 and 2006. Means include only those lakes within a treatment that contained the taxon, numbers in brackets are the numbers of lakes within the treatment containing that taxon. - : taxon not present in the treatment.

| Treatment | Year | Dace (n lakes) | Fathead minnow (n lakes) | Brook stickleback (n lakes) | Iowa darter (n lakes) |
|---|---|---|---|---|---|
| Stocked (n=5) | 2005 | 19.4 ± 6.8 (5) | 7.1 ± 2.2 (2) | 1.4 ± 0.9 (3) | 0.3 ± 0.1 (1) |
| | 2006 | 15.3 ± 6.8 (5) | 3.7 ± 3.6 (2) | 0.3 ± 0.1 (3) | 0.07 ± 0.02 (1) |
| Unstocked (n=6) | 2005 | 3.39 ± 1.7 (5) | 1.1 ± 0.5 (5) | 1.4 ± 1.0 (5) | - |
| | 2006 | 12.7 ± 4.7 (5) | 2.2 ± 1.3 (5) | 1.1 ± 0.3 (5) | - |

**Table 2-6** Mean total lengths (mm) ± SE (n lakes) of each forage fish taxon by treatment and year. n/a : no measurements were collected for that sampling period.

| Taxa/Treatment | 2005[1] | 2006 | 2007 |
|---|---|---|---|
| **Dace** | | | |
| Stocked | 59.9 ± 2.0 (5) | 60.6 ± 0.7 (5) | 58.3 ± 0.7 (2) |
| Unstocked | 52.7 ± 2.8 (5) | 57.4 ± 1.8 (4) | 63.7 ± 2.8 (3) |
| **All Fathead minnows** | | | |
| Stocked | 56.8 ± 3.0 (2) | 60.8 ± 4.3 (2) | n/a |
| Unstocked | 54.5 ± 1.3 (6) | 56.7 ± 0.8 (4) | 59.4 ± 1.1 (2) |
| **Male Fathead minnow** | | | |
| Stocked | n/a | 64.0 ± 6.4 (2) | n/a |
| Unstocked | n/a | 63.5 ± 1.8 (4) | 66.3 ± 1.2 (2) |
| **Female Fathead minnow** | | | |
| Stocked | n/a | 60.8 ± 0.2 (2) | n/a |
| Unstocked | n/a | 57.2 ± 1.3 (4) | 61.7 ± 1.8 (2) |
| **Juvenile Fathead minnow** | | | |
| Stocked | n/a | 52.0 ± 1.7 (2) | n/a |
| Unstocked | n/a | 49.5 ± 1.5 (4) | 50.5 ± 0.2 (2) |
| **Brook stickleback** | | | |
| Stocked | 49.4 ± 1.4 (3) | 50.3 ± 2.2 (3) | 47.2 (1) |
| Unstocked | 48.5 ± 2.9 (6) | 50.2 ± 1.2 (4) | 49.7 ± 2.7 (3) |
| **Iowa darter** | | | |
| Stocked | 49.3 ± 0.5 (1) | 53.0 ± 0.8 (1) | n/a |

**Table 2-7** Forage fish total length (mm; mean ± SE) by strata for sampling conducted in 2006 (3 strata) and 2007 (2 strata). n/a : no measurements were collected for that stratum/taxon.

| Year/Taxa | Treatment (n) | Bottom | Midwater | Surface |
|---|---|---|---|---|
| **2006** | | | | |
| Dace | Stocked (5) | 55.9 ± 0.5 | 54.2 ± 1.9 | 51.8 ± 1.3 |
| | Unstocked (3) | 66.1 ± 1.9 | 63.8 ± 1.2 | 60.2 ± 3.0 |
| Fathead minnow | Stocked (2) | 53.5 ± 2.7 | 55.7 ± 1.3 | 55.9 ± 0 |
| | Unstocked (3) | 60.4 ± 0.8 | 56.2 ± 2.9 | 58.4 ± 1.1 |
| Brook stickleback | Stocked (3) | 48.1 ± 0.2 | 50.5 ± 2.4 | 50.0 ± 1.0 |
| | Unstocked (3) | 51.3 ± 1.7 | 51.6 ± 1.5 | 48.9 ± 1.3 |
| **2007** | | | | |
| Dace | Stocked (2) | 56.7 ± 0.9 | | 55.4 ± 7.8 |
| | Unstocked (2) | 61.9 ± 0.3 | | 59.6 ± 0.9 |
| Fathead minnow | Unstocked (1) | 56.3 ± 2.2 | | 55.8 ± 1.3 |
| Brook stickleback | Stocked (1) | 47.2 ± 1.5 | | n/a |
| | Unstocked (2) | 48.2 ± 2.7 | | 44.5 ± 1.9 |

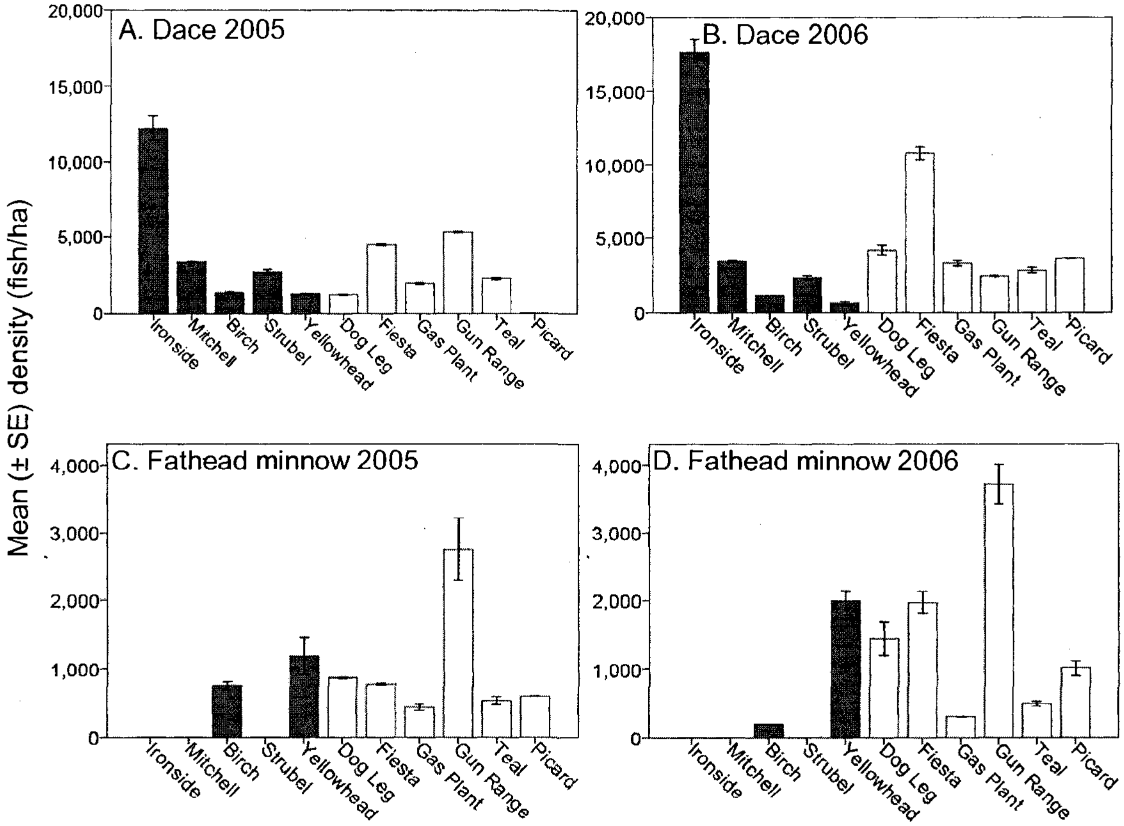


**Figure 2-1** Density estimates (±SE) for dace in 2005 (A) and 2006 (B), and for fathead minnow in 2005 (C) and 2006 (D), for stocked (gray), and unstocked (white) lakes. SE estimates based on the range of daily Peterson estimates for each lake/year (see text; Table 2-4).

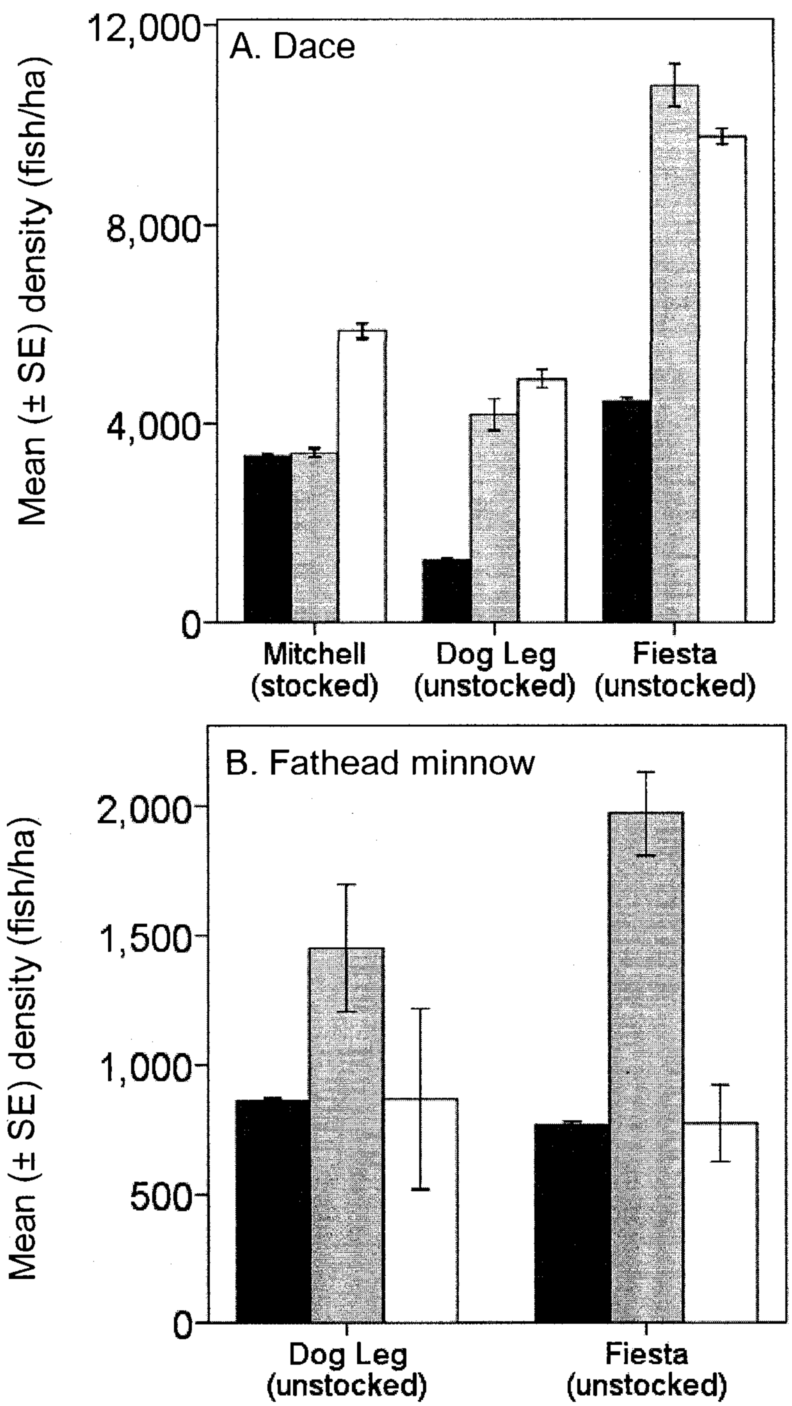


**Figure 2-2** Population densities (±SE) for dace in three lakes (A), and fathead minnow in two lakes (B) over the 3 sampling years: 2005 (black), 2006 (gray), and 2007 (white). All means estimates derived from mark-recapture sampling except the 2005 estimates for Fiesta Lake, which was derived from CPUE regression; SE estimates based on the range of daily Peterson estimates for each lake/year (see text; Table 2-4).

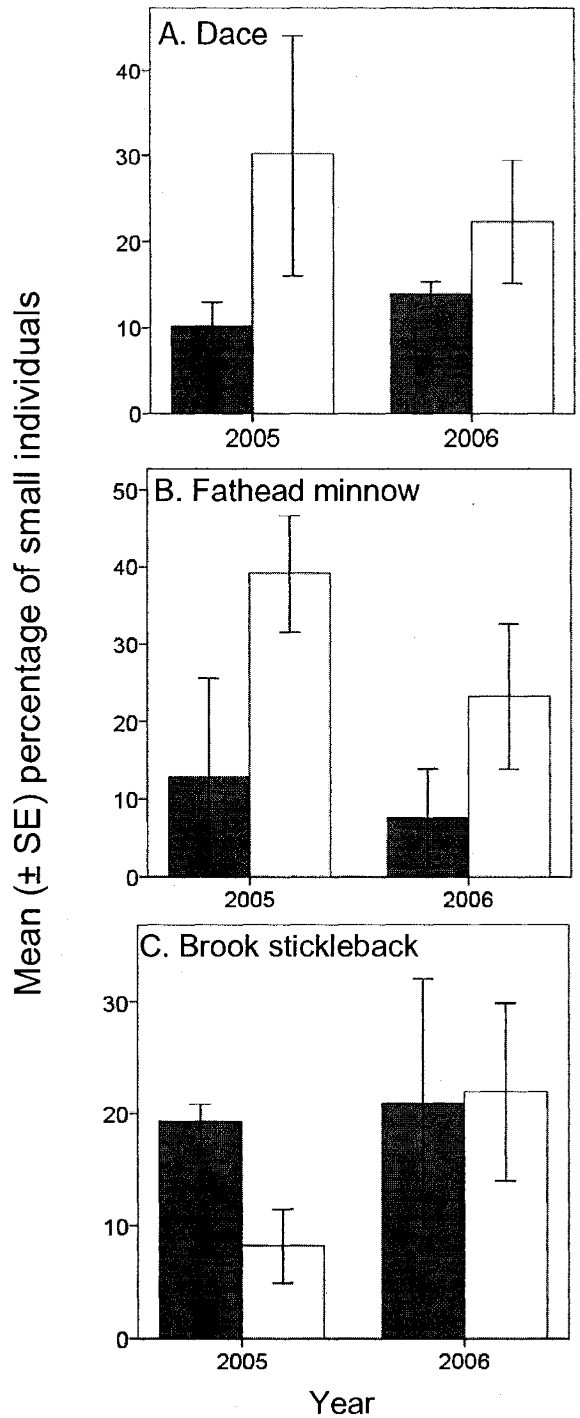


**Figure 2-3** Mean (±SE) percentage of dace <50mm (A), fathead minnow <50mm (B), and brook stickleback <45mm (C), over 2 years, in stocked lakes (gray bars; n=5 for dace, n=3 for brook stickleback, and n=2 for fathead minnow) and unstocked lakes (white bars; n=5 for dace, n=6 for fathead minnow and brook stickleback).

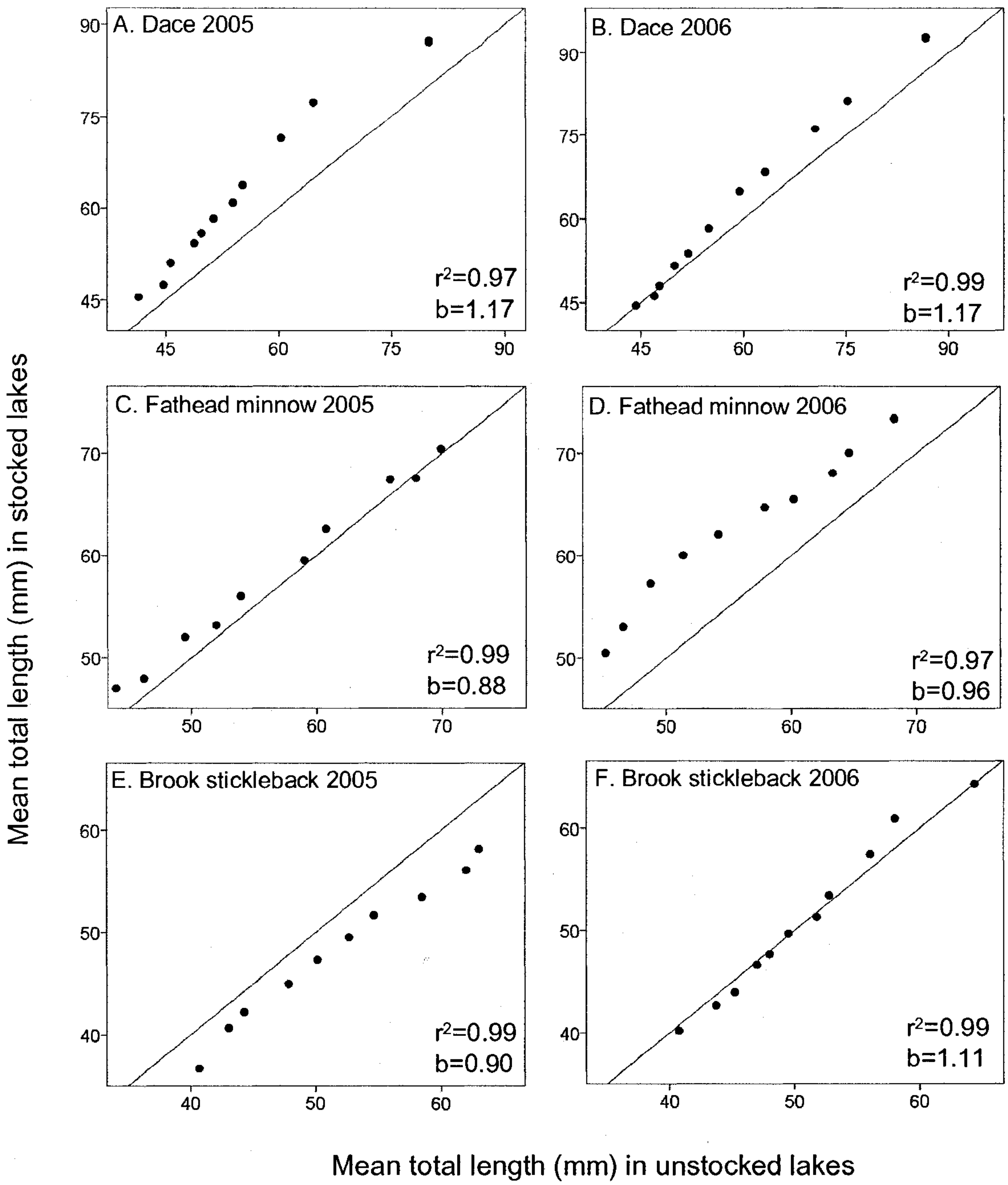


**Figure 2-4** Quantile-quantile plots comparing mean lengths between stocked and unstocked lakes at 11 quantiles in 2005 and 2006 for dace (A,B), fathead minnow (C,D) and brook stickleback (E,F). Plots include coefficients of determination ($r^2$) and slopes (b). The solid line represent the 1:1 ratio.

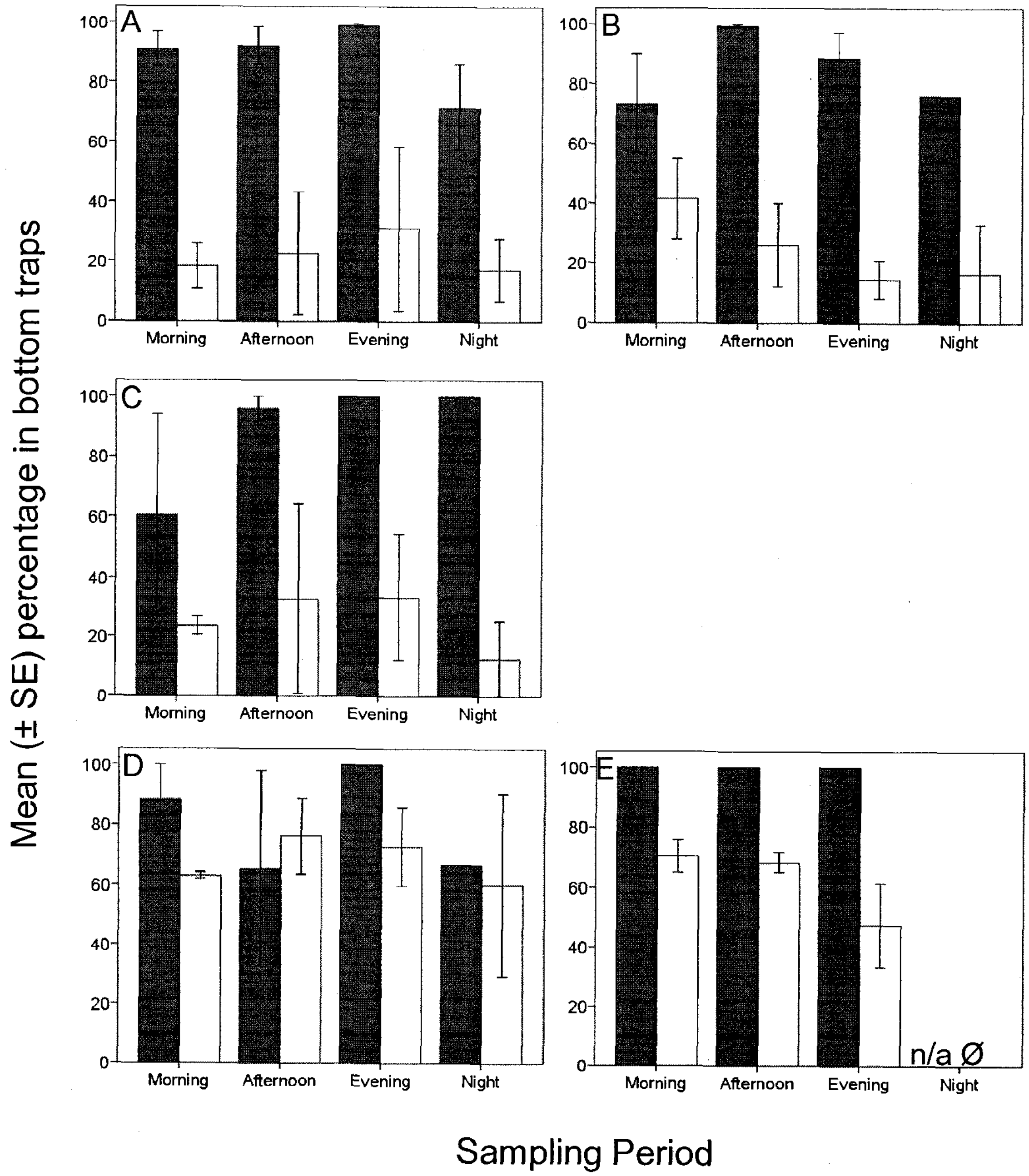


**Figure 2-5** Mean (± SE) percentage of total catch per lake found in bottom traps for (A) dace in 2006 (stocked n=5, unstocked n=3) and (B) 2007 (stocked n=2, unstocked n=2); fathead minnow in (C) 2006 (stocked n=2, unstocked n=3;no stocked lakes with fatheads were sampled in 2007); and (D) brook stickleback in 2006 (stocked n=3, unstocked n=3) and (E) 2007 (stocked n=1, unstocked n=2). Gray: stocked lakes; white: unstocked lakes. Ø indicates that all fish were caught in the water column; n/a indicates that no fish were caught for that period-treatment.

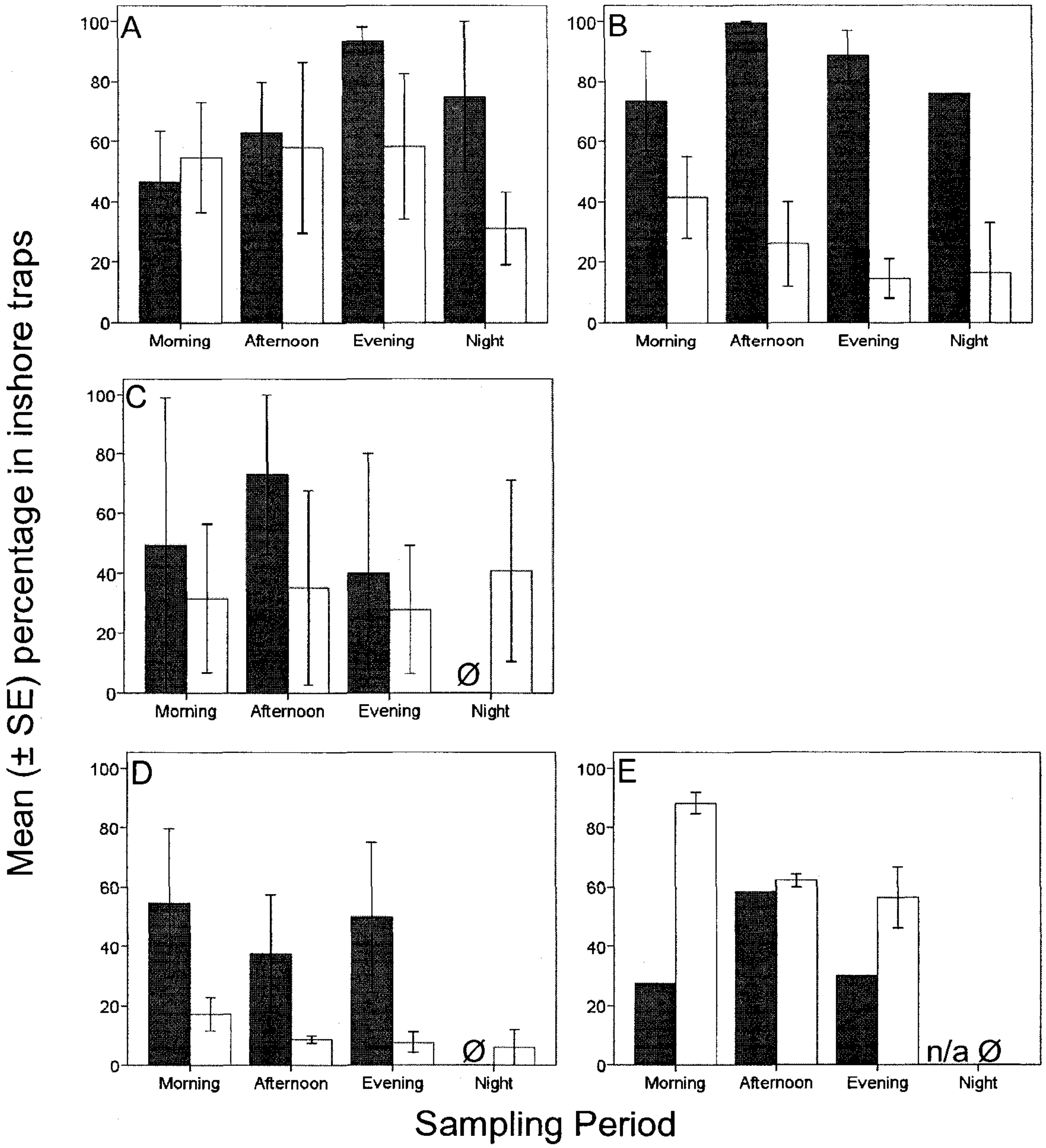


**Figure 2-6** Mean (± SE) percentage of total catch per lake found in inshore traps for (A) dace in 2006 (stocked n=5, unstocked n=3) and (B) 2007 (stocked n=2, unstocked n=2); fathead minnow in (C) 2006 (stocked n=2, unstocked n=3;no stocked lakes with fatheads were sampled in 2007); and (D) brook stickleback in 2006 (stocked n=3, unstocked n=3) and (E) 2007 (stocked n=1, unstocked n=2). Gray: stocked lakes; white: unstocked lakes. Ø indicates that all fish were caught in offshore traps; n/a indicates that no fish were caught for that period-treatment.

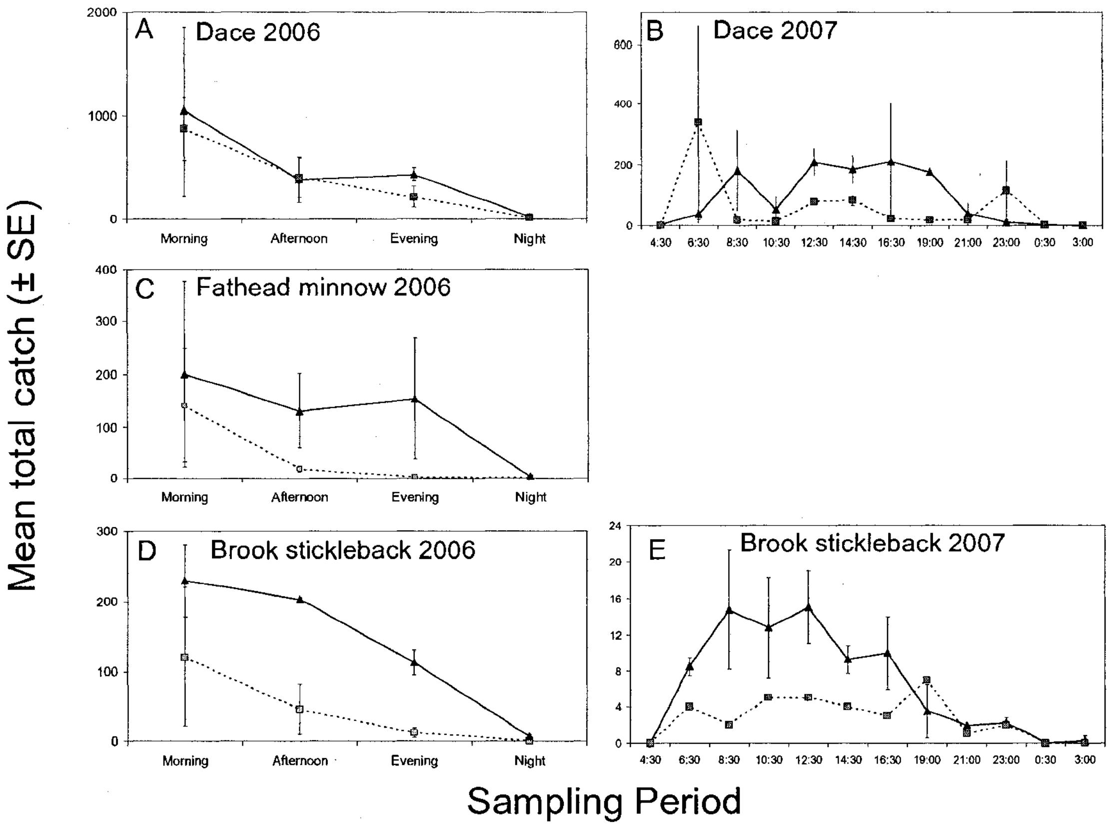


**Figure 2-7** Mean (± SE) total catch per lake for dace in 2006 (A) and 2007 (B), fathead minnow in 2006 (C; no stocked lakes with fatheads were sampled in 2007), and brook stickleback in 2006 (D) and 2007 (E). Gray square and dotted line: stocked (n=5 for dace, n=2 for fathead minnow and n=3 for brook stickleback in 2006; n=2 for dace, n=1 for brook stickleback in 2007); black triangle and solid line: unstocked (n=3 in 2006, n=2 in 2007).

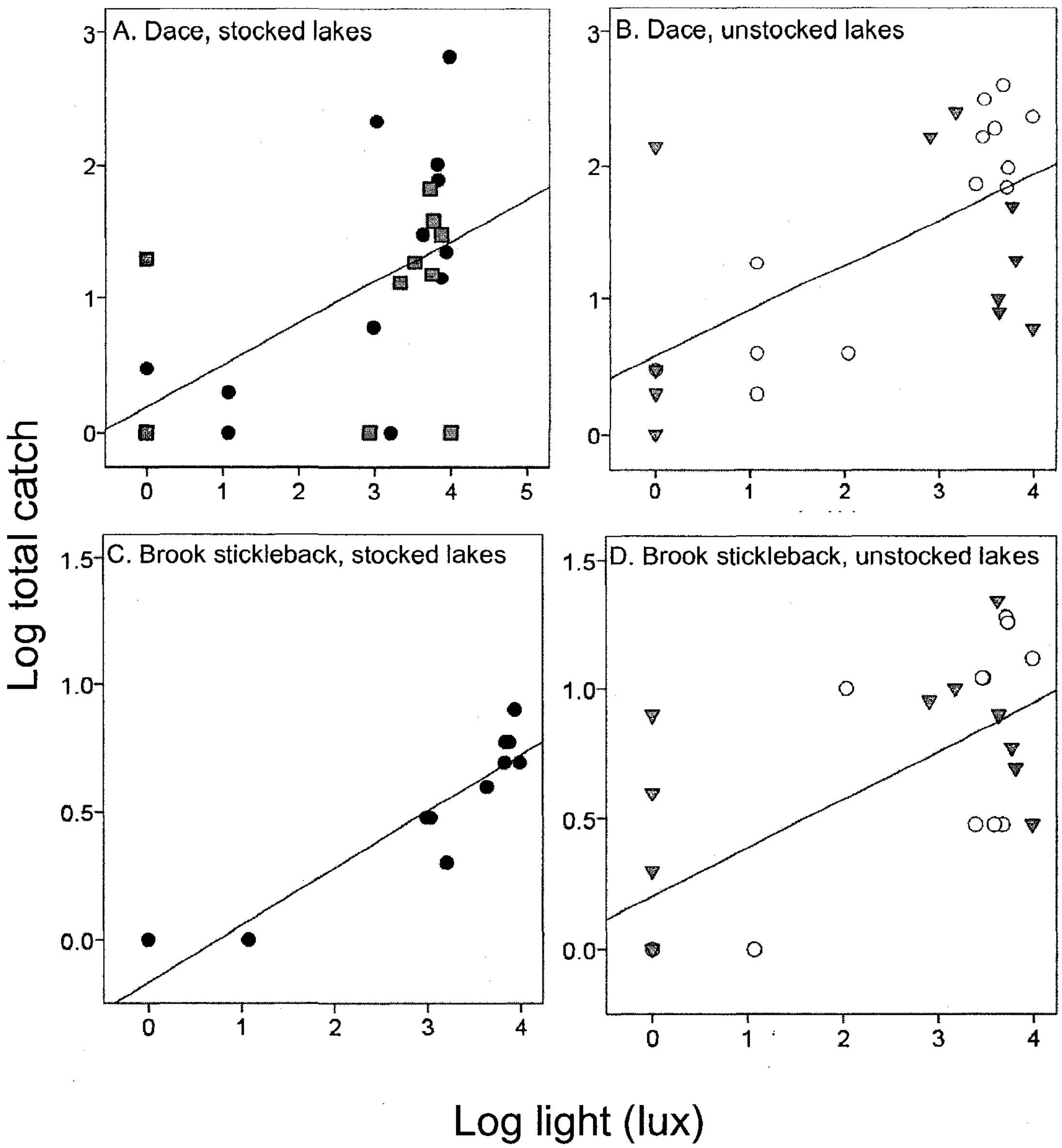


**Figure 2-8**: Scatter plots of log light (lux) and log total catch for dace in (A) stocked lakes, and (B) unstocked lakes; and for brook stickleback in (C) stocked, and (D) unstocked lakes. Black circles: Strubel (stocked); Squares: Mitchell (stocked); Open circles: Fiesta (unstocked); and triangles: Gas Plant (unstocked).

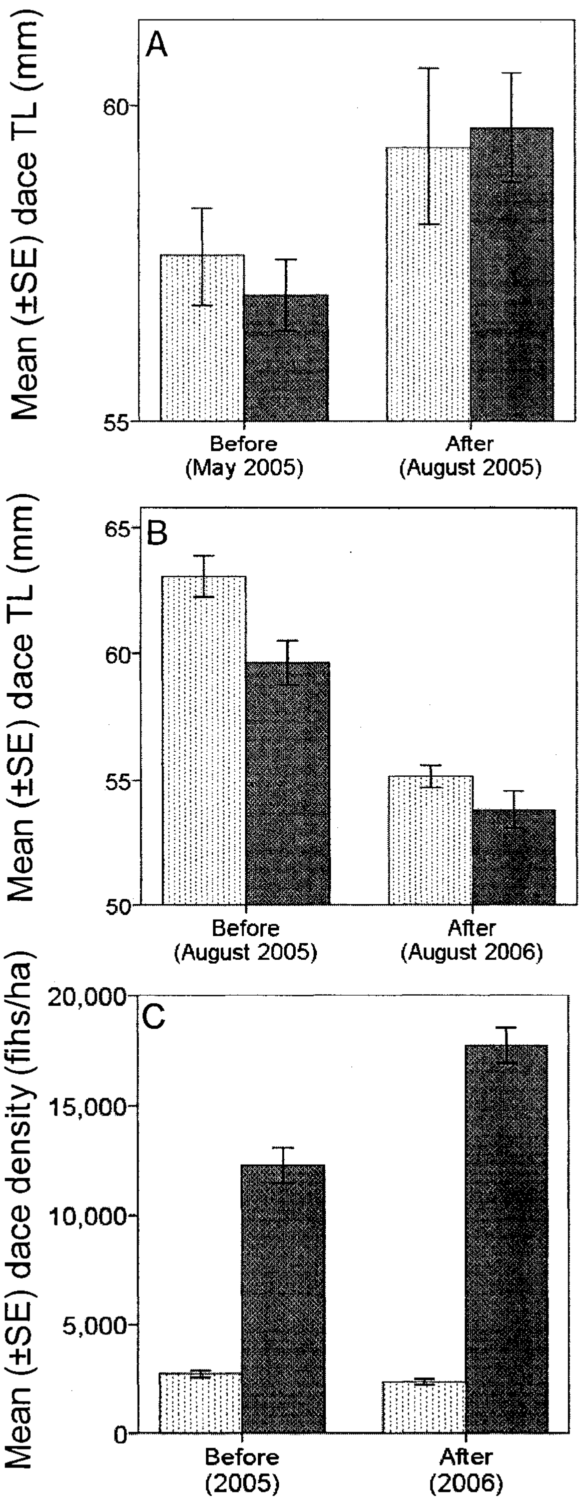


**Figure 2-9**: Before-After-Control-Impact comparisons in Ironside Pond for (A) dace total length before and after stocking (control lake Dog Leg); (B) dace total length before and after aeration (control lake Strubel) and (C) dace density before and after aeration (control lake Strubel). Light bars: control lakes, dark bars: Ironside Pond (Impact)

# Chapter 3. Effects of stocked trout on littoral invertebrates in boreal foothills lakes

## Introduction

In Alberta's boreal foothills, several lakes lacking game fish have been stocked with trout. This popular management practice creates successful recreational fisheries. However, trout are predators that can have major, but often selective, impacts on their invertebrate prey in systems after introduction (e.g., Knapp *et al.* 2001). Therefore, the impacts of stocking on the resident macroinvertebrate fauna are of interest, especially if stocking additional lakes is to occur.

Zooplanktivory by fish usually affects larger-bodied taxa and large individuals of a species more strongly than smaller ones, resulting in changes to zooplankton community composition and size structure (Brooks and Dodson 1965; Carpenter and Kitchell 1988; Elser *et al.* 1995). There is less evidence of similar effects in littoral and benthic habitats, and application of planktonic models to macroinvertebrates may not always be relevant (Strayer 1991). On the other hand, predation by fish is often linked to strong effects on benthic invertebrates (e.g., Gilinsky 1984; Post and Cucin 1984; Northcote 1988; Diehl 1992), and some studies have documented specific effects on benthic invertebrate biomass (Ball and Hayne 1952; Dermott 1988; Karjalainen *et al.* 1999), community composition (Hanson and Butler 1994; Zimmer *et al.* 2001; Venturelli and Tonn 2005), and size structure (Crowder and Cooper 1982; Morin 1984; Blumenshine *et al.* 2000). However, effects are not universal because other studies have found little or no effect of fish predation on benthic invertebrate biomass (Hanson and Leggett 1986; Brönmark 1994; Pierce and Hinrichs 1997; Cobb and Watzin 1998; Michaletz *et al.* 2005), density (Thorp and Bergey 1981; Hershey 1985), or size structure (Pierce and Hinrichs 1997) in either single-predator or multiple-predator assemblages.

Evidence for impacts of predation by trout on benthic invertebrates is also mixed. Brown (*Salmo trutta*), rainbow (*Oncorhynchus mykiss*), cutthroat (*O. clarki*), and lake (*Salvelinus namaycush*) trout can be size-selective predators on macroinvertebrates (Luecke 1990; Merrick *et al.* 1992; Nyström *et al.* 2001; Wissinger *et al.* 2006), however, this does not necessarily translate into measurable size effects on their prey (Carlisle and Hawkins 1998). The strongest negative impacts of trout on invertebrate abundance have been observed after introductions into high alpine, low-productivity, fishless lakes (Carlisle and Hawkins 1998; McNaught *et al.* 1999; Knapp *et al.* 2001; Knapp *et al.* 2005). In contrast, introduced brown and rainbow trout did not negatively affect benthic invertebrates in New Zealand lakes with naturally occurring populations of small-bodied fishes (Wissinger *et al.* 2006).

Based on current literature, it is difficult to predict responses of benthic invertebrate communities in boreal foothills lakes to trout presence. These are relatively low elevation, productive lakes with populations of native forage fish, and thus the results of introductions are not likely to mirror those in unproductive and fishless alpine lakes (e.g., Carlisle and Hawkins 1998; McNaught *et al.* 1999; Knapp *et al.* 2001; 2005). Circumstances in foothills lakes are more similar to introductions in New Zealand, where receiving systems had natural populations of small-bodied fish prior to trout introduction

(Townsend 1996; Wissinger *et al.* 2006). In these examples, piscivory is thought to reduce trout impact on invertebrates (e.g., Townsend and Crowl 1991).

Because stocked trout are not native to (Nelson and Paetz 1992), and cannot often survive the hypoxic winter conditions typical of, small, shallow Alberta lakes, some stocked lakes are also aerated over the winter. Aeration can impact invertebrates irrespective of trout, and responses to aeration may also vary among taxa and lakes (Lackey 1973; Wilhm and McClintock 1978; Cowell *et al.* 1987). All of these previous studies, however, were from continuous aeration, not seasonal applications. It is unknown if over-winter aeration will affect invertebrates, either directly, or indirectly, through altered habitat use or prolonged survival of fish.

To examine effects of introduced trout on littoral invertebrates, I collected littoral samples from stocked and unstocked lakes in the boreal foothills of western Alberta. Invertebrate community composition, abundance, and size structure were examined to assess effects of the presence and absence of trout. Since trout in this system were not predominantly piscivorous (J. Hanisch, University of Alberta, personal communication). I expected to find a stronger impact on invertebrates than that seen in New Zealand lakes. I also assessed the impacts of aeration on benthic invertebrates.

## Methods

*Study Area*

Littoral macroinvertebrates were sampled in 11 lakes in the boreal foothills of Alberta, Canada near Rocky Mountain House (52°22'39"N & 114°54'37"W) and Caroline (52° 5'36"N & 114°45'28"W) during May and August 2005-2006. Lakes were either stocked with trout (n=5; 2 of which also received over-winter aeration), or unstocked (n=6). All lakes have been stocked for ca. 20 years, with the exception of Ironside Pond, which was stocked for the first time in June 2005, after my first invertebrate sampling (Table 2-1). In addition to being stocked, 2 lakes were aerated from mid October to early April using one ½-hp (Ironside) or two 1-hp (Mitchell) floating aerators. Aeration of Ironside Pond was initiated in the fall of 2005, following my first season of data collection, whereas Mitchell Lake had been aerated since 2003 (C. Rasmussen, Alberta Conservation Association, personal communication).

Lakes were generally small, and all supported populations of native forage fish (Table 2-1) with densities ranging from 500-15,000 fish/ha (Chapter 2). Native forage fishes included fathead minnow, brook stickleback, Iowa darter, and mixtures of pearl, northern redbelly, and finescale dace, and dace hybrids (Table 2-1). Brook, brown, and/or rainbow trout have been introduced into stocked lakes (Tables 2-1, 2-2).

*Water Chemistry and Macrophytes*

Water chemistry variables were measured from epilimnetic samples collected once in 2005 and monthly May-August 2006 and 2007 (see Chapter 2 for methods). Profiles of temperature and dissolved oxygen were taken monthly for June-August 2005 and May-August 2006 and 2007. A qualitative survey of littoral-zone macrophytes (percent cover) was conducted on 10-1x1m quadrats, on all lakes in late August 2005.

*Invertebrate Collection and Laboratory Processing*

Littoral macroinvertebrate communities were sampled from two lakes (Ironside and Gas Plant) in early May 2005, and from all lakes between 15-26 August 2005, and 04-07 May 2006, and 19-23 August 2006. Sites at each lake were chosen randomly along the 1m depth contour and invertebrates were sampled with a 30x30x30cm triangular sweep net of 500μm mesh. The net was inserted ca. 5cm into the lake sediment and dragged quickly through ca. 1m of the sediment then upwards through the water column and/or macrophyte bed. Ten samples were collected for each lake during each sampling period. Samples were stored in containers and preserved in 80% ethanol, and transferred to the laboratory for subsequent processing. Three samples were lost due to insufficient preservation.

To rinse away silt and facilitate recovery of invertebrates, samples were washed through sequential sieves (2mm and 0.5mm). Samples contained large detritus and macrophytes, which retained invertebrates smaller than 2mm, causing taxa <2mm to be represented in the 2mm size fraction. Items that passed through the 2mm mesh, and were retained by the 0.5mm sieve (0.5mm size fraction), were also preserved in 80% ethanol.

The 2mm size fractions were sorted under a dissecting microscope. Individuals were identified to lowest feasible taxonomic level (LFTL; generally family or genus), and a subset of each taxon was measured (n=20-30 per sample). Only intact specimens were measured. Measurements were typically dorsal body length, with some exceptions. Amphipods were straightened before measurements were taken. Appendages such as cerci and antennae were not included in length measurements. Widths, as well as lengths, were measured for gastropods (width only for limpets). Measurements were made using calibrated ocular micrometers.

All individuals in the 2mm fraction were counted. In the case of damaged individuals, only partial bodies with heads were counted, to avoid counting an individual twice. Some taxa could not be accurately counted and were recorded as presence/absence. These included Bryozoan statoblasts from *Cristatella mucedo*, as well as oligochaetes and nematodes that were difficult to enumerate because they were most often fragmented in the samples. Because my sampling method included a 1-m sweep of the water column, some predominantly pelagic taxa were also present and included in my analyses.

Due to time constraints, the 0.5mm size fractions for most samples were not sorted. To determine the extent to which omission of this size fraction altered the results I processed, counted, and measured 22 randomly selected samples. Inclusion of individuals from the 0.5mm size fraction rarely changed the relative abundance of taxa. As well, for most of the taxa, the inclusion of the length data from the 0.5mm size fraction did not significantly change the mean length, or size distribution as determined by One-way ANOVA and Kolmogorov-Smirnov tests, respectively (L. Nasmith, unpublished data). Further support for omitting this size fraction was provided by studies that excluded small size classes when looking at effects of (primarily size-selective) fish predation on invertebrate abundance (e.g., Batzer *et al.* 2000) and size distribution (Blumenshine *et al.* 2000). I concluded that omitting the 0.5mm size class would not alter my results.

*Before and After Stocking and Aeration*

Ironside Pond underwent two major manipulations over the course of the study. In June 2005 it was stocked for the first time in 30 years; the following winter, it was aerated for the first time. One round of invertebrate sampling (following the procedures outlined above) was conducted on Ironside before stocking began (May 2005) and one round after stocking but before aeration (August 2005). Two rounds of invertebrate sampling were conducted after stocking and aeration (May and August 2006).

*Statistical Analysis*

All multivariate analyses of community composition and univariate analyses of size structure were conducted at the LFTL. For univariate tests of abundance, invertebrates were grouped by class or order.

Relative abundance was calculated as the percentage of individuals of each taxon (LFTL) in a sweep. Estimates from the 10 sweeps within a lake were averaged for each LFTL unit within each sampling period and then means were arcsine-squareroot transformed before further analyses.

To compare patterns of community composition in stocked and unstocked lakes, I used Nonmetric Multidimensional Scaling (NMS) ordinations on relative abundance data. Ordinations were performed using PC-ORD version 5 (McCune and Mefford 1999). NMS was chosen because of its wide use in ecology and ability to handle non-normal data (McCune and Grace 2002). Before ordinations were performed, rare (occurring in $<2$ lakes) and outlier ($>2$SD from the mean relative abundance) taxa were deleted or merged with another taxa when possible (e.g., pooled according to genus). For all ordinations, I used the Sorensen (Bray-Curtis) distance measure. I began runs with a random starting point and included 600 runs with real data, 3 axes, and 15 iterations to determine stability. Monte Carlo tests (performed with 300 randomized runs) were used to assess the suitability of the ordination. Number of axes was determined by permutation tests with 300 randomizations and scree plots (McCune and Grace 2002). If a 3-axis solution was recommended, the two axes that represented the most variation were used in graphs. Biplots were accepted for taxa if $r^2>0.5$ and for environmental variables if $r^2>0.3$. Multiple Response Permutation Procedures (MRPP; McCune and Grace 2002) were performed on raw matrices to test for differences between treatments.

To assess the proportion of variance in invertebrate community composition that could be explained by the environmental variables and the presence of trout, I performed a two-way Variance Partitioning Analyses (VPA; Borcard *et al.* 1992; Hall *et al.* 1999). Analyses were performed using the program CANOCO for Windows, version 4.5 (ter Braak and Smilauer 2006). Using a preliminary detrended correspondence analyses (DCA) of the species matrix to calculate the length of the dominant axis, it was determined that a linear model (redundancy analysis – RDA) was appropriate for the VPA (ter Braak and Smilauer 2006). All my environmental variables (Chapter 2) were assessed with forward stepwise selection ($p<0.09$; ter Braak and Smilauer 2006) to determine which were associated with significant variation in the invertebrate data set. These identified variables were included in the VPA.

Shannon-Weiner diversity, richness, and evenness (% of maximum diversity; Legendre and Legendre 1998) were calculated at the family level for stocked and unstocked lakes. To assess differences in abundance (mean number of individuals per sweep) of taxa at the class or order level, I used two-way fixed factor ANOVAs with "treatment" and "sampling period" as the main effects.

Analysis of size structure between treatments was limited to taxa most likely to be affected by trout. Assessments of vulnerability were based on a combination of a taxon being sufficiently abundant in my samples for analysis, and on its frequency (>3%) in trout diets (J. Hanisch, University of Alberta, personal communication). For each taxon analyzed, I compared body lengths between treatments using a two-way ANOVA as above. The values used in this test were the mean values from each lake with sampling period as the repeated measure. A sampling period was excluded from analyses if either of the treatments had values for <2 lakes.

For taxa that had sufficient sample sizes (>50 lengths/lake/sampling period), I analyzed proportions of large individuals in a population. The length cut off for "large" taxon included most lengths in the $4^{th}$ quartile of the length distribution for the taxon across all lakes. These proportions were transformed, and compared among treatments and sampling periods with two-way fixed factor ANOVAs (described above).

To examine if trout predation was size-selective, I examined quantile-quantile (QQ; *sensu* Post and Evans 1989) plots of the mean length distributions in stocked and unstocked lakes. A QQ plot transforms a length frequency-distribution into a linear function by graphing the length at each quantile, which can then be directly compared to other transformed distributions. Distributions in stocked and unstocked lakes were compared at the following quantiles: 1, 5, 10, 25, 35, 50, 65, 75, 90, 95, and 99. These plots can be used to identify differences in distributions (Post and Evans 1989).

To study the impact of trout stocking and aeration on macroinvertebrates in Ironside Pond, I conducted two separate Before-After-Control-Impact (BACI) analyses. Paucity of "Before" samples restricted the tests possible for my data, so a one-before one-after analysis was chosen (Downes *et al.* 2002). This analysis is calculated as a replicated, two-factor ANOVA, with Before-After, and Control-Impact as main effects. A significant F-ratio for the interaction term (Before-After*Control-Impact) in this test indicates a difference between the Control and Impact sites that varies with the Before and After conditions. However, a significant result in a BACI analyses can not necessarily be attributed to the perturbation being studied (Stewart-Oaten *et al.* 1992). To look at trout impact, size and abundance data from Ironside Pond (Impact) in May 2005 (Before) and August 2005 (After) were compared to data from the same sampling periods in Gas Plant Lake (Control). This comparison is adequate since both lakes were unstocked at the beginning of the study, and sampled in the same time period. Data of size and abundance from Ironside Pond (Impact) in August 2005 (Before) and August 2006 (After) were compared to data from the same sampling periods in stocked but unaerated Strubel Lake (Control). Strubel Lake was chosen for this comparison because its forage fish assemblage and densities were more similar to Ironside than those of other stocked lakes (Chapter 2). As well, both Ironside and Strubel contain only rainbow trout.

Statistical computations were performed using SPSS 16.0 (SPSS for Windows, Rel. 16.0.1. 2007). Results for all statistical tests were considered significant if $p<0.05$ and marginally significant for $p<0.1$.

**Results**

Over the three main sampling periods (August 2005, and May and August 2006), 50,163 invertebrates were collected from 347 (2mm fraction) samples. I measured 48%

of the individuals collected. Average number of individuals per sample was lowest for both stocked and unstocked lakes in August 2005; highest values occurred in May 2006 for stocked lakes and in August 2006 for unstocked lakes (Table 3-1). There was no difference between treatments in family-level richness, but unstocked lakes had marginally greater diversity and evenness, and stocked lakes had a marginally greater number of individuals per sample (Table 3-1).

*Water Chemistry and Macrophytes*

Water temperatures were highest in the stocked lakes, whereas dissolved oxygen and pH were lowest in unstocked lakes. Concentrations of TN, TDN, TP, TDP, and Chl-*a* were highest in the unstocked lakes. Stocked lakes were oligo-mesotrophic and the unstocked lakes were meso-eutrophic, as indicated by Chl-*a* and TP concentrations (Carlson 1977; Chapter 2, Appendix A).

The maximum depth of macrophytes was ca. 3m. This depth corresponded with the photic zone, which in most lakes was also ca. 3m deep. In the macrophyte survey, greater than 85% of the areas of all quadrats were covered with macrophytes (C. Schank and L. Nasmith, personal observation). The dominant macrophytes in the study lakes were *Potamogeton spp.*, *Sparganium angustifolium*, and *Nuphar variegatum*.

*Invertebrate Abundance*

The abundance (mean number of individuals in a taxon per sample) of higher-level taxa varied both between stocked and unstocked lakes and among sampling periods within treatments (Table 3-2). For all treatments and sampling periods, nematocerans were most abundant, ranging from 30-130 individuals/sample. Other highly abundant groups were Amphipoda, Gastropoda, Ephemeroptera, and Bivalvia.

Abundances of most taxa did not differ between stocked and unstocked lakes (Table 3-2). Mites and copepods were marginally more abundant in unstocked lakes, whereas abundances were highest in stocked lakes for Ephemeroptera, Brachycera, Nematocera, and marginally for Odonata. Although the abundance of many taxa varied among sampling periods, there were no significant interactions between treatment and sampling period.

*Community Composition*

There was no difference in littoral invertebrate community composition between stocked and unstocked lakes for any of the 3 sampling periods (MRPP; $p>0.2$).

The ordination from August 2005 represented 86% of variation in the data set (93 taxa), at a stress level of 9.6 (Monte Carlo permutation, $p=0.01$; Figure 3-1). Taxa that were correlated with the two axes ($r^2>0.5$) were gastropods (*Armiger crista* and *Helisoma*), Acari (*Hydrozetes*), cladocerans (Sididae), bivalves (*Sphaerium*), Anisoptera (Aeshnidae), and chironomid larvae and pupae. Environmental variables correlated with the two axes ($r^2>0.3$) were dissolved oxygen, pH, and TDN.

The ordination from May 2006 represented 76% of variation in the data set (80 taxa), at a stress level of 9.3 (Monte Carlo permutation, $p=0.04$; Figure 3-2). Taxa that were correlated with the two axes ($r^2>0.5$) were Acari (*Arrenurus*), Hirudinea (*Helobdella stagnalis*), Trichoptera (Molannidae), Zygoptera ( *Ischnura/Enallagma/*

*Coenagrion* species complex), and chironomid larvae. Environmental variables correlated with the two axes ($r^2>0.3$) were Chl-*a*, and TP.

The ordination for August 2006 represented 89% of variation in the data set (95 taxa), at a stress level of 8.9 (Monte Carlo permutation, p=0.003; Figure 3-3). Taxa that were correlated with the two axes ($r^2>0.5$) were gastropods (*Planorbula* and *Helisoma*), Acari (*Neumania*), and ceratopogonid and chironomid larvae. Environmental variables associated with the two axes ($r^2>0.3$) were dissolved oxygen, TN, and pH. The environmental variables chosen for variance partitioning analyses through forward stepwise selection were TDN and TP for August 2005, Chl-*a* and Conductivity for May 2006, and TN, TP, and TDP for August 2006. These environmental variables explained between 24-62% of the variation in the invertebrate data, whereas the presence of trout explained 5-8% of the variation. The variation explained by the combination of environment and trout was 0.1-9.5% (Figure 3-4).

*Invertebrate Size Distribution*

Representative lower-level taxa were chosen from the major groups based on adequate sample sizes in both treatments for analyses of size distribution. There were no significant differences in size between treatments, except that *Planorbula* (Gastropoda) in unstocked lakes had wider shells, and Phryganeidae (Trichoptera) were larger in unstocked lakes (Table 3-3). There was a significant sampling period effect for *Unionicola*, Phryganeidae, and chironomid pupae. The data for Daphnidae showed a significant interaction between treatment and sampling period, although the main effects were not significant.

For taxa with sample sizes >50 individuals per lake (*Caenis*, *Hyalella azteca,* Chironomidae, and Ceratopogonine), I compared the proportion of large body size classes between stocked and unstocked lakes. There was no effect of trout for any of the taxa (Figure 3-5), but there was a significant effect of sampling period for Chironomidae ($F_{2,27}=4.5$, p=0.02) and *H. azteca* ($F_{2,18}=28.1$, p<0.001), with May 2006 having greater proportions of large individuals.

In QQ plots, a slope of 1.0 indicates no difference in the distribution of two populations. If there is selection of large prey, the slope will be <1, and quantiles that deviate will fall below the 1:1 line (Post and Evans 1989). The average slopes for *H. azteca* and *Caenis* over three sampling periods did not differ from 1 (Figure 3-6). In contrast, Chironomidae (*t*-test, $t_4=6.9$, p=0.002) and Ceratopogoninae ($t_4=5.62$, p=0.005) had an average slope <1 (Figure 3-6). Both of these taxa showed a tendency for deviations below the 1:1 line at the largest sizes, suggesting size-selective predation of larger individuals in stocked lakes.

*Before and After Stocking and Aeration*

A number of taxa in Ironside decreased in abundance after the commencement of stocking; notably, Hemiptera were only present in pre-stocking samples (Table 3-4). Between May and August 2005, the number of individuals per sample decreased for most taxa in Ironside. The BACI comparison with Gas Plant Lake identified taxa that had a significant (Copepoda, Ephemeroptera, Odonata, and Trichoptera) or marginally significant (Nematocera, Brachycera) interaction between Before-After and Control-Impact (Table 3-5). These taxa also decreased in abundance in Gas Plant, with the exceptions of Copepoda which showed no change; however, the decrease in abundance was larger for Odonata and Trichoptera in Ironside (Table 3-5). One taxon, Brachycera, showed a marginally significant interaction and increased in abundance in Ironside between May and August, while decreasing in Gas Plant. These results are consistent with the community analyses (above) in which brachycerans were more abundant in stocked lakes.

To examine potential effects of stocking on invertebrate sizes in Ironside Pond, taxa were chosen based on known trout diet in Ironside (J. Hanisch, University of Alberta, personal communication), and on results from the among-lake comparisons (above). Of the taxa that increased in mean length between the sampling periods in Ironside, the interaction term was significant for Chironomidae, and marginally significant for both *Pisidium* and Ceratopogoninae (Table 3-5). The mean length of these three taxa in Gas Plant either did not change or decreased (Table 3-6). There was a significant interaction term for *H. azteca*, which decreased in length in both lakes.

Most taxa in Ironside increased in abundance between August 2005 and 2006, following the initiation of aeration (Table 3-4). Of these taxa, the BACI analysis with Strubel Lake identified those that had a significant (Bivalve, Cladocera, Odonata) or marginally significant (Ostracoda, Amphipoda, Ephemeroptera, Nematocera) interaction (Table 3-5). Abundance increased for Nematocera, Odonata, Ostracoda, and Amphipoda in Ironside but decreased in Strubel. Abundances in both lakes increased for Bivalvia, Cladocera, and Ephemeroptera between August 2005 and August 2006.

The same taxa discussed above were used to study effects of aeration on length in Ironside Pond. Few of the taxa examined showed changes in length between August 2005 and August 2006 (Table 3-6). The interaction terms of the BACI with Strubel Lake were significant for *Chrysops* and *H. azteca*, and marginally significant for Ceratopogoninae (Table 3-5). Mean length of Ceratopogoninae and *H. azteca* decreased in Ironside while increasing or not changing in Strubel. Mean length of *Chrysops* increased in Ironside while decreasing in Strubel between sampling periods.

**Discussion**

Negative impacts of trout on benthic invertebrates in boreal foothills lakes were no more than modest, occurring at the taxon, and not community, level. While unstocked lakes had greater diversity and evenness, community composition did not differ between stocked and unstocked lakes. In fact, the presence of trout explained only a small percentage of the variation in communities relative to environmental variables. Two small-bodied, pelagic taxa were marginally more abundant in unstocked lakes, whereas several larger-bodied, more active taxa were actually more abundant in lakes with trout. There was evidence of size differences between stocked and unstocked lakes in 2 invertebrate taxa, although there was weak evidence for selective predation by trout on of larger Chironomidae and Ceratopogoninae in stocked lakes.

Short-term changes following stocking and aeration using BACI analyses identified taxa that had significant interactions between Control-Impact and Before-After, indicating differences between Ironside and a control lake that were influenced by the Before-After condition. After stocking, a number of taxa decreased in both Ironside and Gas Plant. Only brachycerans increased in Ironside while decreasing in Gas Plant. Mean length increased after stocking for *Pisidium*, Cetatopogonidae, and Chironomidae while not changing or decreasing in Gas Plant Lake. Following aeration in Ironside, the abundance of Odonata, Ostracoda, Amphipoda, and Nematocera all increased, but decreased over the same sampling time in Strubel Lake. Few taxa showed effects of aeration on size, but a post-aeration increase was seen in Ironside for *Chrysops*, while decreasing in length in Strubel.

*Community Composition and Abundance*

Some taxa are more sensitive to trout introductions than others. For example, BACI analyses showed that abundances of Odonata and Trichoptera in Ironside decreased to a greater extent than in the control lake over the summer following stocking, a time period when taxa generally increase in abundance (Ball and Hayne 1952). Decreases in the abundance and occurrence of predatory (e.g., Odonata and Coleoptera) and grazing (e.g., Ephemeroptera and Trichoptera) taxa in the presence of trout is often seen (Knapp *et al.* 2001; Nyström *et al.* 2001). However, in among-lake comparisons across three sampling periods, Ephemeroptera and Odonata showed positive responses to trout, while those of Coleoptera and Trichoptera were neutral.

Differences between the among-lake comparisons and the Ironside BACI analysis suggest that any short-term prey community responses may not persist. Invertebrate communities have been known to recover following the removal of trout (Knapp *et al.* 2005), and perhaps in my study lakes the communities were able to recover after the initial impact of introduction. This possibility should be further examined on another lake with longer before and after sampling periods. An increased timeline could establish the natural temporal variation in invertebrate communities in the absence of trout, and provide a more complete assessment of the changes that come after stocking.

Many taxa demonstrated no response to trout presence or addition in my study. It is not uncommon for some taxa to be impacted while others are not. Even in studies that found more wide-spread negative effects (Knapp *et al.* 2001; Nyström *et al.* 2001), there were some taxa (e.g., Bivalvia, Gastropoda) that showed no response. Elsewhere, trout have no short-term impact on invertebrate abundance and/or community composition, both in lakes that were fishless prior to stocking (Dahl and Greenberg 1998), or that contained populations of small-bodied fish prior to stocking (Wissinger *et al.* 2006). Trout impacts are also not found over long time periods (Luecke 1990).

*Invertebrate Size Structure*

In addition to limited community-level effects, I found few examples of size-selective predation on littoral invertebrates. Among potential prey taxa, mean size was larger in unstocked lakes for two of 11 taxa tested, while QQ plots suggested size-selective predation on two of four groups. No taxa (of four tested) showed differences in the proportion of large individuals between stocked and unstocked lakes. Only *Hyalella azteca* (Amphipoda) showed a decrease in body size after stocking in Ironside, however, size of this species also decreased in the control lake.

Size-selective predation can play a role in determining size-structure of invertebrate populations (Merrick *et al.* 1992), although it has been suggested that such impacts are less common for benthic than for pelagic species (Diehl 1992). Even though size-selective predation is well documented in trout, it does not necessarily have noticeable effects on prey populations. In my study, the taxa that showed negative size responses to trout were generally large and/or active, and therefore more likely to encounter, and be seen by, trout. Carlisle and Hawkins (1998) and Luecke (1990) both supported the idea that trout (brook and cutthroat) will feed selectively on the largest prey taxa and individuals, although only Luecke found that this selectivity resulted in differences in size distribution. While it is possible that trout show no size selection (Dahl and Greenberg 1998), it is most likely that size selective predation is occurring in my lakes, but that such predation is apparently not strong enough to elicit consistent, significant responses in size structure of prey taxa.

Invertebrate growth, and therefore size structure of populations, can also be affected by competition, which should be correlated with abundance (Constable 1999). In my "control-impact" among-lake comparison, taxa with reduced abundance in the presence of trout showed no size response, possibly due to offsetting responses to size-selective predation and competitive release. In contrast, Nematocera abundance in the BACI analysis decreased marginally following stocking, whereas the mean length of the most ubiquitous nematocerans, Chironomidae and Ceratopogoninae, increased relative to the control lake. In the case of Ironside, it is possible this is an indirect effect of trout, with decreased competition brought about by reductions in abundance affecting the size distribution (Carlisle and Hawkins 1998; Cobb and Watzin 1998).

*Forage Fish Effects*

The forage fish populations in my study varied in density among lakes and between years, and assemblage composition also varied among lakes or between treatments. Forage fish can directly affect littoral macroinvertebrates through predation, and are capable of altering community composition (Vinebrooke *et al.* 2001; Zimmer *et al.* 2001). In this role, forage fish may "condition" the prey community to predators, thereby reducing their susceptibility to additional predation from novel predators such as stocked trout (Wellborn and Robinson 1991). In the present study, similarities in littoral macroinvertebrate community composition between lakes with and without trout suggest that any differences attributable to the presence or absence of trout were not strong relative to the (pre-existing) effects of forage fish.

Predation by forage fish can be sufficient to elicit effects on invertebrate abundance (e.g., Zimmer *et al.* 2002). A mixed assemblage of fathead minnow and brook stickleback in Alberta's Aspen Parkland decreased the abundance of small leeches and gastropods relative to a reference wetland (McParland and Paszkowski 2006). In my study, Acari were marginally less abundant in stocked lakes, an unexpected result since Acari only made up a small proportion (0-4%) of trout diet in stocked lakes (J. Hanisch, University of Alberta, personal communication). Furthermore, in other cases of trout introduction, Acari increased (Knapp *et al.* 2001; Ortubay *et al.* 2006).

I suggest that the observed decrease in Acari abundance could be caused by forage fish switching to Acari in the presence of trout. Preliminary stomach content analysis of dace showed that Acari were found in 27% of dace stomachs in a stocked lake (Strubel), but in only 1% of stomachs in an unstocked lake (Fiesta; L. Castro, University of Alberta, personal communication). Stable isotope analyses of food webs in these lakes also suggest that a shift to a food web based more on littoral sources of primary productivity occurs in stocked relative to unstocked lakes (J. Hanisch, University of

Alberta, personal communication). Forage fish could be shifting their diets to avoid competition with trout, or restricting foraging to low-risk areas in the presence of a potential predator. In stocked lakes, forage fish are active near the bottom rather than in the water column (Chapter 2), and most of the dace consuming Acari were caught on the lake bottom. Thus, the decrease in Acari could have been directly due to increased consumption by forage fish and thus was an indirect effect of trout presence.

Presence of forage fish may have obscured the impacts of trout on invertebrate abundance, but may not have affected size structure of invertebrate populations. While there may be some diet overlap between trout and forage fish (e.g., Rennie and Jackson 2005), forage fish likely eat smaller prey than trout, and are less likely to reduce the occurrence of large individuals within the population. A limited sample of stomach contents from dace in two of my study lakes revealed that invertebrate prey ranged in size from 0.3 to 4.0mm, and prey size increased with dace total length (L. Nasmith, unpublished data). Brown trout, in contrast, have been found to rarely consume prey <2mm, and generally ate prey >3.5mm (Rincon and Lobon-Cervia 1999). Little evidence suggests that forage fish can significantly alter macroinvertebrate size structure, and Zimmer *et al.* (2001) suggested that size structure of invertebrates in wetlands was relatively consistent regardless of forage fish presence. If this is the case, size differences seen among study lakes are likely attributable to direct and/or indirect effects of trout.

Ideally, this study would have included a suite of fishless lakes to characterize littoral macroinvertebrate community in the absence of fish predators. With a fishless treatment, I could have determined the impact of forage fish alone on invertebrate composition, abundance, and size structure. I suspect that forage fish had a strong impact on invertebrate abundance that was relatively consistent across all lakes. The impact of trout, in addition to forage fish, was not strong enough to cause significant differences in community composition or abundance. However, it is unlikely that forage fish predation was responsible for the few observed differences in invertebrate size structure.

*Lake Productivity*

The structure of aquatic communities is influenced by both top-down and bottom-up effects (Carpenter *et al.* 2001; Menge *et al.* 2003). In my lakes, however, interactions between trout presence and environmental variables explained little variation invertebrate community composition, and most variation was explained by environment alone.

Lakes without piscivores tend to have higher phytoplankton productivity than lakes with piscivores (Carpenter and Kitchell 1988), and introduced piscivores can affect the productivity of receiving lakes (Carpenter *et al.* 1985; Schindler *et al.* 2001). Although productivity was not measured directly in my lakes, indices such as Chl-*a* and nutrient concentrations suggested that unstocked lakes were more productive than stocked lakes. As there were no before-stocking nutrient samples, however, I do not know if levels actually decreased following trout introductions, or if there is another reason for lower productivity indices in the stocked lakes.

In my BACI analysis of aeration, increases in abundance of many taxa were observed, but few taxa showed any size effects. More sampling periods both before and after the impact should be applied to better assess the effects of aeration. It is possible that aeration may mask the effects of trout; if aeration positively affects invertebrate abundance, it may not allow the true impact of trout on abundance to be resolved.

Although environmental variables, including water quality, explained much of the variation in invertebrate community composition, I could not study the effect of those variables on abundance or size structure. It is most likely that productivity would most strongly influence the abundance of specific taxa, and not overall abundance, biomass, or size (Michaletz *et al.* 2005). Ephemeroptera and Odonata were most abundant in the lower-productivity stocked lakes of my study. In impoundments, Ephemeroptera and Odonata became more abundant with decreasing productivity (Michaletz *et al.* 2005), although over a larger range of productivity than observed in my study. However, invertebrates might respond differently to productivity in boreal lakes than in temperate impoundments (Michaletz *et al.* 2005) or arctic lakes, where there is almost no response (Hershey 1992). To begin to answer these questions, more direct measures of productivity are needed, as well as further BACI comparisons before and after stocking.

Primary productivity is also linked to aquatic vegetation, which plays an important role in freshwater systems. All of the lakes in my study contained submerged and emergent aquatic vegetation, oftentimes comprising dense beds, although differences in macrophyte communities between stocked and unstocked lakes were not quantified. Aquatic vegetation can influence size structure (Hanson 1990) and abundance (Carlisle and Hawkins 1998; Rennie and Jackson 2005) of invertebrates. Invertebrate prey can use macrophyte beds as refugia from predators, and such complex habitat can reduce predation intensity (Crowder and Cooper 1982; Gilinsky 1984; Diehl 1992). It is possible that the impact of trout on littoral invertebrates was somewhat reduced by the presence of dense beds of macrophytes, although they may not have provided protection from forage fish, which also utilize macrophyte cover (Chapter 2). The effects of macrophytes on the trout-invertebrate interaction could be explored further. Sampling specific habitats (macrophytes present vs. open substrate) or gradients of macrophyte cover in both stocked and unstocked lakes would elucidate the effect of macrophytes on macroinvertebrates, and reveal whether that relationship changes in stocked lakes.

*Conclusions*

I detected no negative effects of introduced trout on invertebrate community composition in boreal foothills lakes. Consistent with this, there were few taxa-specific examples of decreased abundance . Furthermore, if size-selective predation is occurring, it is not altering size structure of invertebrate populations. Indirect effects were suggested by increased abundances. It is also likely that forage fish are impacting the invertebrates, as there were indications that forage fish were changing their diet in the presence of trout and exploiting new prey. Thus, invertebrates were not naïve to fish predation and already exhibited impacts of fish predation prior to trout introduction. All of these fish effects may be influenced by lake productivity, which may influence invertebrate abundance and support macrophyte beds as refugia from predation pressure. Abundant populations of forage fish, dense macrophyte cover, aeration, and the productive nature of these lakes all likely interact to allow invertebrates to withstand the impact of trout. Effects of introduced trout in boreal foothills lakes were weak.

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**Table 3-1.** Mean (± SE) number of individuals per sample (n), Family-level Shannon-Weiner diversity index (H), Family richness (q) and eveness (J; %) for each sampling period and treatment: stocked (n=5) and unstocked lakes (n=6). Also presented are results of two-way fixed factor ANOVAs for each metric, with main effects, Treatment (T) and Sampling Period (SP), and interaction (T*SP). Degrees of freedom for main effects and error are given in parentheses.

| | August 2005 | | May 2006 | | August 2006 | | ANOVA F-statistic | | |
|---|---|---|---|---|---|---|---|---|---|
| **Metric** | **Stocked** | **Unstocked** | **Stocked** | **Unstocked** | **Stocked** | **Unstocked** | **T (n=1,27)** | **SP (n=2,27)** | **T*SP (n=2,27)** |
| n | 107 ± 66.1 | 83.6 ± 16.5 | 210 ± 53.0 | 129 ± 21.8 | 221 ± 63.5 | 179 ± 52.4 | F=3.5* | F=6.2** | F=0.43 |
| H | 1.79 ± 0.1 | 1.81 ± 0.1 | 1.53 ± 0.2 | 1.77 ± 0.1 | 1.94 ± 0.1 | 1.93 ± 0.2 | F=3.8* | F=1.4 | F=0.19 |
| q | 32.6 ± 2.9 | 27.3 ± 2.6 | 36.8 ± 2.8 | 29.7 ± 1.9 | 42.2 ± 2.1 | 33.8 ± 2.1 | F=0.01 | F=5.4** | F=0.006 |
| J | 51.9 ± 3.3 | 55.2 ± 3.6 | 42.6 ± 4.9 | 52.5 ± 3.1 | 51.7 ± 3.0 | 54.5 ± 4.9 | F=4.3** | F=1.2 | F=0.38 |

*0.1>p>0.05, **p<0.05

**Table 3-2**. Mean (± SE) number of individuals per sample of major taxa in each sampling period for each treatment: stocked lakes (n=5) and unstocked lakes (n=6). Also presented are results of two-way fixed factor ANOVAs for each taxa with main effects, Treatment (T) and Sampling Period (SP), and interaction (T*SP). Degrees of freedom for main effects and error are given in parentheses. - : Taxa not present in that treatment/sampling period; n/a: analyses not conducted.

| | **August 2005** | | **May 2006** | | **August 2006** | | **ANOVA F-statistic** | | |
|---|---|---|---|---|---|---|---|---|---|
| **Taxa** | **Stocked** | **Unstocked** | **Stocked** | **Unstocked** | **Stocked** | **Unstocked** | **T (n=1,27)** | **SP (n=2,27)** | **T*SP (n=2,27)** |
| Hirudinea | 1.5 ± 0.5 | 2.5 ± 1.1 | 0.8 ± 0.3 | 1.3 ± 0.3 | 0.5 ± 0.2 | 2.6 ± 1.6 | F=2.7 | F=0.6 | F=0.4 |
| Bivalvia | 8.0 ± 2.5 | 10.3 ± 2.8 | 10.7 ± 4.9 | 10.8 ± 2.9 | 19.0 ± 7.3 | 7.9 ± 1.5 | F=0.9 | F=0.6 | F=1.7 |
| Gastropoda | 9.3 ± 3.5 | 10.4 ± 3.8 | 13.5 ± 3.4 | 11.9 ± 5.8 | 20.5 ± 6.8 | 25.4 ± 10.6 | F=0.08 | F=2.2 | F=0.1 |
| Acari | 1.0 ± 0.5 | 0.8 ± 0.3 | 3.5 ± 0.9 | 5.9 ± 1.9 | 3.5 ± 0.9 | 5.8 ± 0.7 | F=3.0* | F=8.3** | F=0.9 |
| Conchostraca | 0.06 ± 0.02 | 0.07 ± 0.1 | 0.6 ± 0.3 | 1.2 ± 0.6 | 1.5 ± 1.1 | 1.4 ± 0.5 | F=0.1 | F=3.2* | F=0.2 |
| Cladocera | 1.5 ± 1.2 | 1.2 ± 0.6 | 0.2 ± 0.1 | 0.1 ± 0.03 | 2.6 ± 1.1 | 6.4 ± 2.7 | F=1.0 | F=5.2** | F=1.5 |
| Copepoda | 0.08 ± 0.04 | 0.3 ± 0.2 | 1.3 ± 0.5 | 2.3 ± 0.8 | 0.2 ± 0.08 | 0.9 ± 0.5 | F=3.1* | F=7.6** | F=0.4 |
| Ostracoda | 0.2 ± 0.1 | 0.9 ± 0.7 | 1.0 ± 0.7 | 0.6 ± 0.3 | 0.6 ± 0.5 | 1.0 ± 0.8 | F=0.2 | F=0.1 | F=0.4 |
| Amphipoda | 17.4 ± 9.5 | 17.4 ± 9.5 | 23.6 ± 11.6 | 8.9 ± 4.9 | 24.7 ± 10.9 | 33.8 ± 25.6 | F=0.03 | F=0.5 | F=0.4 |
| Ephemeroptera | 8.7 ± 4.6 | 1.7 ± 0.7 | 17.1 ± 5.2 | 5.3 ± 1.5 | 34.9 ± 10.1 | 16.2 ± 4.9 | F=8.9** | F=8.4** | F=0.6 |
| Odonata | 1.9 ± 0.8 | 0.7 ± 0.2 | 3.3 ± 1.0 | 0.9 ± 0.3 | 7.1 ± 2.7 | 4.4 ± 2.1 | F=3.0* | F=5.2** | F=0.14 |
| Hemiptera | 0.1 ± 0.1 | 0.2 ± 0.06 | 0.1 ± 0.08 | 0.1 ± 0.03 | 0.06 ± 0.04 | 0.4 ± 0.2 | F=2.7 | F=0.9 | F=2.5 |
| Lepidoptera | - | 0.02 ± 0.02 | - | 0.02 ± 0.02 | - | 0.08 ± 0.1 | n/a | n/a | n/a |
| Trichoptera | 1.5 ± 0.6 | 1.4 ± 0.7 | 5.9 ± 1.3 | 4.9 ± 2.8 | 6.7 ± 1.1 | 4.9 ± 2.2 | F=0.4 | F=3.8** | F=0.12 |
| Coleoptera | 0.8 ± 0.3 | 0.7 ± 0.2 | 0.04 ± 0.03 | 0.2 ± 0.1 | 0.5 ± 0.3 | 0.5 ± 0.1 | F=0.003 | F=5.3** | F=0.3 |
| Nematocera | 52.6 ± 18.8 | 30.4 ± 6.9 | 133 ± 19.3 | 71.1 ± 13.9 | 96.5 ± 22.9 | 68.8 ± 19.0 | F=7.1** | F=6.5** | F=0.8 |
| Brachycera | 0.8 ± 0.09 | 0.3 ± 0.1 | 0.7 ± 0.2 | 0.23 ± 0.1 | 0.5 ± 0.1 | 0.2 ± 0.1 | F=19.9** | F=1.8 | F=0.13 |

*0.1>p>0.05, **p<0.05

**Table 3-3** Mean ± SE length (mm) of selected taxa in each sampling period for each treatment: stocked lakes (n=5) and unstocked lakes (n=6). Also presented are results of two-way fixed factor ANOVAs for each taxon, with main effects, Treatment (T) and Sampling Period (SP), and interaction (T*SP). -- : Taxa not present in that treatment/sampling period.

| | | **August 2005** | | **May 2006** | | **August 2006** | | **ANOVA F-statistic** | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Taxa** | | **Stocked** | **Un-stocked** | **Stocked** | **Un-stocked** | **Stocked** | **Un-stocked** | **T** | **SP** | **T*SP** |
| Hirudinea | *Helobdella stagnalis* | 4.2 ± 0.2 | 4.9 ± 0.7 | 5.2 ± 0.4 | 5.5 ± 0.6 | 3.7 ± 0.3 | 4.7 ± 0.6 | $F_{1,23}$=1.9 | $F_{2,23}$=1.9 | $F_{2,23}$=0.2 |
| Gastropoda | *Planorbula* (length) | 2.8 ± 0.4 | 3.1 ± 0.1 | 2.7 ± 0.2 | 2.8 ± 0.2 | 2.4 ± 0.2 | 2.9 ± 0.3 | $F_{1,23}$=1.96 | $F_{1,23}$=0.4 | $F_{1,23}$=0.3 |
| | *Planorbula* (width) | 1.2 ± 0.2 | 1.4 ± 0.04 | 1.1 ± 0.07 | 1.3 ± 0.1 | 1.1 ± 0.08 | 1.3 ± 0.1 | $F_{1,23}$=63** | $F_{1,23}$=0.4 | $F_{1,23}$=0.9 |
| Acari | *Hydrozetes* | 0.49 | - | 0.48 ± 0.02 | 0.48 ± 0.01 | 0.48 ± 0.02 | 0.52 ± 0.02 | $F_{1,18}$=1.0 | $F_{2,18}$=0.9 | $F_{2,18}$=0.9 |
| | *Unionicola* | 0.7 | - | 0.9 ± 0.08 | 0.92 ± 0.06 | 0.8 ± 0.03 | 0.81 ± 0.07 | $F_{1,16}$=0.2 | $F_{2,16}$=2.9* | $F_{2,16}$=0.04 |
| Cladocera | Daphnidae | 1.1 ± 0.1 | 1.4 ± 0.08 | 0.8 ± 0.04 | - | 1.4 ± 0.09 | 1.2 ± 0.06 | $F_{1,15}$=0.0 | $F_{2,15}$=11** | $F_{2,15}$=4** |
| Odonata | *Somatochlora* | - | - | 8.8 ± 1.6 | 7.6 ± 4.1 | 7.6 ± 0.5 | 7.3 ± 1.1 | $F_{1,10}$=0.2 | $F_{1,10}$=0.2 | $F_{1,10}$=0.1 |
| Trichoptera | Limnephilidae | 7.1 ± 2.3 | 7.2 ± 0.9 | 5.4 ± 0.5 | 6.3 ± 0.9 | 4.4 ± 0.3 | 4.9 ± 0.5 | $F_{1,20}$=0.4 | $F_{2,20}$=4** | $F_{2,20}$=0.1 |
| | Phryganeidae | 7.1 ± 0.9 | 6.3 ± 0.8 | 9.5 ± 1.2 | 10.8 ± 1.3 | 5.4 ± 0.9 | 11.8 ± 2.0 | $F_{1,22}$=4.1* | $F_{2,22}$=3.1* | $F_{2,22}$=4** |
| Nematocera | Chironomidae pupae | 3.1 ± 0.3 | 3.5 ± 0.5 | 5.5 ± 0.6 | 6.8 ± 0.9 | 3.6 ± 0.3 | 3.6 ± 0.7 | $F_{1,27}$=0.9 | $F_{2,27}$=10** | $F_{2,27}$=0.5 |
| Brachycera | *Chrysops* | 11.9 ± 1.5 | 14.1 ± 0.9 | 12.8 ± 1.5 | 11.4 ± 1.2 | 10.9 ± 1.3 | 7.8 ± 2.0 | $F_{1,18}$=0.4 | $F_{2,18}$=2.6 | $F_{2,18}$=1.5 |

*0.1>p>0.05, **p<0.05

**Table 3-4**: Mean (± SE) number of individuals per sweep for major taxa groups in Ironside Pond (unstocked in May 2005, stocked in August 2005, then stocked-aerated). Also shown are values for Strubel (stocked) and Gas Plant (unstocked) Lakes used in the BACI analyses. - : Taxa not present in that treatment/sampling period.

| | **Ironside Pond** | | | **Gas Plant Lake** | | **Strubel Lake** | |
|---|---|---|---|---|---|---|---|
| **Taxa** | **May 2005** | **August 2005** | **August 2006** | **May 2005** | **August 2005** | **August 2005** | **August 2006** |
| Hirudinea | 0.3 ± 0.2 | 0.5 ± 0.2 | 0.2 ± 0.1 | 5.4 ± 1.3 | 4.4 ± 1.7 | 2.6 ± 0.9 | 0.9 ± 0.3 |
| Bivalvia | 16.5 ± 10.5 | 5.8 ± 1.3 | 42.7 ± 9.2 | 21.3 ± 4.7 | 10.4 ± 4.1 | 16.6 ± 4.1 | 27.0 ± 7.1 |
| Gastropoda | 23.1 ± 8.9 | 6.1 ± 1.4 | 34.4 ± 5.7 | 16.3 ± 5.1 | 9.5 ± 5.9 | 22.8 ± 3.1 | 38.7 ± 6.3 |
| Acari | 3.8 ± 1.4 | 0.4 ± 0.3 | 4.2 ± 1.3 | 6.3 ± 2.0 | 0.6 ± 0.3 | 1.1 ± 0.5 | 4.3 ± 1.5 |
| Conchostraca | 10.0 ± 4.3 | 0.1 ± 0.1 | 5.8 ± 1.9 | 1.5 ± 0.8 | - | - | 0.8 ± 0.5 |
| Cladocera | 12.2 ± 8.9 | 0.4 ± 0.2 | 7.1 ± 2.1 | 0.5 ± 0.2 | 1.3 ± 0.8 | 0.6 ± 0.3 | 1.3 ± 0.6 |
| Copepoda | 7.3 ± 2.3 | 0.2 ± 0.1 | 0.5 ± 0.3 | 0.3 ± 0.2 | 0.2 ± 0.2 | 0.1 ± 0.1 | 0.1 ± 0.1 |
| Ostracoda | 1.2 ± 0.5 | 0.5 ± 0.3 | 2.4 ± 1.0 | 0.3 ± 0.2 | 0.7 ± 0.4 | 0.5 ± 0.4 | 0.3 ± 0.2 |
| Amphipoda | 16.2 ± 5.5 | 6.9 ± 5.0 | 20.2 ± 5.1 | 22.2 ± 6.3 | 11.8 ± 6.5 | 1.9 ± 1.2 | 1.0 ± 0.6 |
| Ephemeroptera | 9.9 ± 2.4 | 2.3 ± 1.3 | 27.2 ± 8.0 | 25.8 ± 5.9 | 2.8 ± 1.6 | 11.8 ± 3.4 | 70.1 ± 18.2 |
| Odonata | 29.1 ± 10.4 | 0.6 ± 0.3 | 12.5 ± 4.0 | 1.8 ± 0.6 | 0.2 ± 0.1 | 4.9 ± 1.6 | 3.6 ± 0.5 |
| Hemiptera | 0.1 ± 0.1 | - | - | - | 0.2 ± 0.1 | - | - |
| Lepidoptera | - | - | - | - | 0.1 ± 0.1 | - | - |
| Trichoptera | 12.0 ± 3.6 | 0.3 ± 0.3 | 4.2 ± 1.4 | 2.5 ± 0.7 | 0.2 ± 0.1 | 1.0 ± 0.3 | 4.4 ± 0.9 |
| Coleoptera | 0.7 ± 0.3 | 1.1 ± 0.5 | 1.5 ± 0.5 | 0.4 ± 0.2 | 0.7 ± 0.4 | 0.1 ± 0.1 | 0.1 ± 0.1 |
| Nematocera | 81.9 ± 16.2 | 34.4 ± 11 | 112.8 ± 42.2 | 128.4 ± 31 | 15.4 ± 5.6 | 125.2 ± 34.9 | 88.1 ± 20.2 |
| Brachycera | 0.4 ± 0.2 | 0.8 ± 0.3 | 0.9 ± 0.3 | 0.4 ± 0.2 | 0.1 ± 0.1 | 0.6 ± 0.3 | 0.7 ± 0.3 |

**Table 3-5**: F-statistics of interaction terms for Before-After-Control-Impact tests of stocking and aeration on abundance of major taxa groups, and lengths of selected taxa at the lowest feasible taxonomic level (see text for details about the tests and taxa selection). Tests for stocking effects compare Ironside Pond and Gas Plant Lake between May and August 2005. Tests for aeration effects compare Ironside Pond and Strubel Lake between August 2005 and August 2006. – :Taxa not present in that treatment/sampling period in one/both of the study lakes.

| | **Abundance (n=1,36)** | | | **Length** | |
|---|---|---|---|---|---|
| **Taxa** | **Stocking** | **Aeration** | **Taxa** | **Stocking** | **Aeration** |
| Hirudinea | F=0.3 | F=1.9 | | | |
| Bivalvia | F=0.0 | F=5.6** | *Pisidium* | $F_{1,249}$=3.39* | $F_{1,342}$=0.03 |
| | | | *Sphaerium* | $F_{1,108}$=0.56 | $F_{1,310}$=0.2 |
| Gastropoda | F=0.7 | F=1.9 | *Planorbula* (L) | $F_{1,132}$=0.63 | $F_{1,318}$=0.1 |
| | | | *Planorbula* (W) | $F_{1,132}$=1.1 | $F_{1,310}$=0.003 |
| Acari | F=0.9 | F=0.09 | | | |
| Conchostraca | - | - | | | |
| Cladocera | F=1.9 | F=7.7** | | | |
| Copepoda | F=8,9** | F=0.82 | | | |
| Ostracoda | F=2.4 | F=3.7* | | | |
| Amphipoda | F=0.01 | F=3.8* | *Hyalella azteca* | $F_{1,359}$=8.4** | $F_{1,85}$=20.4** |
| Ephemeroptera | F=6.2** | F=2.7* | *Caenis* | $F_{1,61}$=0.9 | $F_{1,468}$=0.32 |
| Odonata | F=6.7** | F=9.2** | *Ischnura/ Enallagma/ Coenagrion* | - | $F_{1,12}$=2.8 |
| Hemiptera | - | - | | | |
| Trichoptera | F=6.5** | F=0.9 | Phryganeidae | $F_{1,8}$=3.36 | - |
| Coleoptera | F=0.02 | - | | | |
| Nematocera | F=3.2* | F=3.8* | Chironomidae | $F_{1,657}$=15.6** | $F_{1,886}$=1.4 |
| | | | Ceratopogoninae | $F_{1,183}$=2.87* | $F_{1,128}$=3.1* |
| Brachycera | F=2.9* | F=0 | *Chrysops* | - | $F_{1,23}$=5.9** |

*0.9>p>0.05, **p<0.05

**Table 3-6** Mean (± SE) lengths (mm) for selected taxa in Ironside Pond for all sampling periods (unstocked in May 2005, stocked in August 2005, then stocked-aerated). Also shown are values for Strubel (stocked) and Gas Plant (unstocked) Lakes used in the BACI analyses. n/a: no measurements for that taxa/sampling period.

| Taxa | Lake | May 2005 | August 2005 | August 2006 |
|---|---|---|---|---|
| *Sphaerium* | Ironside | 1.6 ± 0.02 | 1.9 ± 0.2 | 1.8 ± 0.07 |
| | Gas Plant | 6.1 ± 0.7 | 5.4 ± 1.0 | 7.7 ± 0.4 |
| | Strubel | n/a | 5.5 ± 0.3 | 4.9 ± 0.3 |
| *Pisidium* | Ironside | 2.0 ± 0.07 | 2.4 ± 0.1 | 2.5 ± 0.05 |
| | Gas Plant | 2.6 ± 0.08 | 2.6 ± 0.1 | 2.5 ± 0.1 |
| | Strubel | n/a | 2.6 ± 0.1 | 2.7 ± 0.1 |
| Planorbula L | Ironside | 2.2 ± 0.06 | 2.6 ± 0.1 | 2.2 ± 0.2 |
| | Gas Plant | 2.6 ± 0.09 | 3.2 ± 0.6 | 3.4 ± 0.2 |
| | Strubel | n/a | 3.5 ± 0.1 | 3.3 ± 0.1 |
| Planorbula W | Ironside | 1.0 ± 0.03 | 1.2 ± 0.1 | 1.1 ± 0.03 |
| | Gas Plant | 1.1 ± 0.04 | 1.5 ± 0.2 | 1.5 ± 0.1 |
| | Strubel | n/a | 1.5 ± 0.04 | 1.4 ± 0.04 |
| *Hyalella azteca* | Ironside | 5.3 ± 0.08 | 4.2 ± 0.1 | 3.7 ± 0.06 |
| | Gas Plant | 6.1 ± 0.07 | 4.6 ± 0.09 | 4.4 ± 0.08 |
| | Strubel | n/a | 3.1 ± 0.2 | 3.2 ± 0.2 |
| *Caenis* | Ironside | 3.0 ± 0.1 | 2.5 ± 0.2 | 2.8 ± 0.07 |
| | Gas Plant | 3.0 ± 0.1 | 2.5 ± 0.09 | 2.7 ± 0.08 |
| | Strubel | n/a | 2.6 ± 0.1 | 2.8 ± 0.07 |
| *Ischnura/ Enallagma/ Coenagrion* | Ironside | 7.2 ± 0.3 | 6.9 ± 3.3 | 9.4 ± 1.4 |
| | Gas Plant | 12.7 ± 1.1 | 7.7 | 11.5 ± 1.5 |
| | Strubel | n/a | 8.7 ± 1.4 | 3.6 ± 1.5 |
| Phryganeidae | Ironside | 10.5 ± 1.1 | 3.3 ± 1.1 | 7.1 ± 0.6 |
| | Gas Plant | 12.9 ± 1.9 | 18.0 | 14.8 ± 2.2 |
| | Strubel | n/a | 6.0 ± 1.2 | 6.5 ± 0.6 |
| Chironomidae | Ironside | 4.6 ±0.1 | 5.5 ± 0.2 | 5.2 ± 0.1 |
| | Gas Plant | 5.1 ± 0.1 | 4.8 ± 0.2 | 5.9 ± 0.2 |
| | Strubel | n/a | 5.1 ± 0.1 | 4.5 ± 0.1 |
| Ceratopogoninae | Ironside | 8.2 ± 0.5 | 10.5 ± 1.8 | 9.0 ± 0.7 |
| | Gas Plant | 8.2 ± 0.3 | 8.3 ± 0.3 | 9.6 ± 0.4 |
| | Strubel | n/a | 8.9 ± 0.3 | 10.0 ± 0.4 |
| *Chrysops* | Ironside | 10.5 ± 3.3 | 11.7 ± 2.0 | 13.6 ± 1.5 |
| | Gas Plant | 17.4 ± 2.5 | - | - |
| | Strubel | n/a | 13.8 ± 3.3 | 6.6 ± 0.6 |

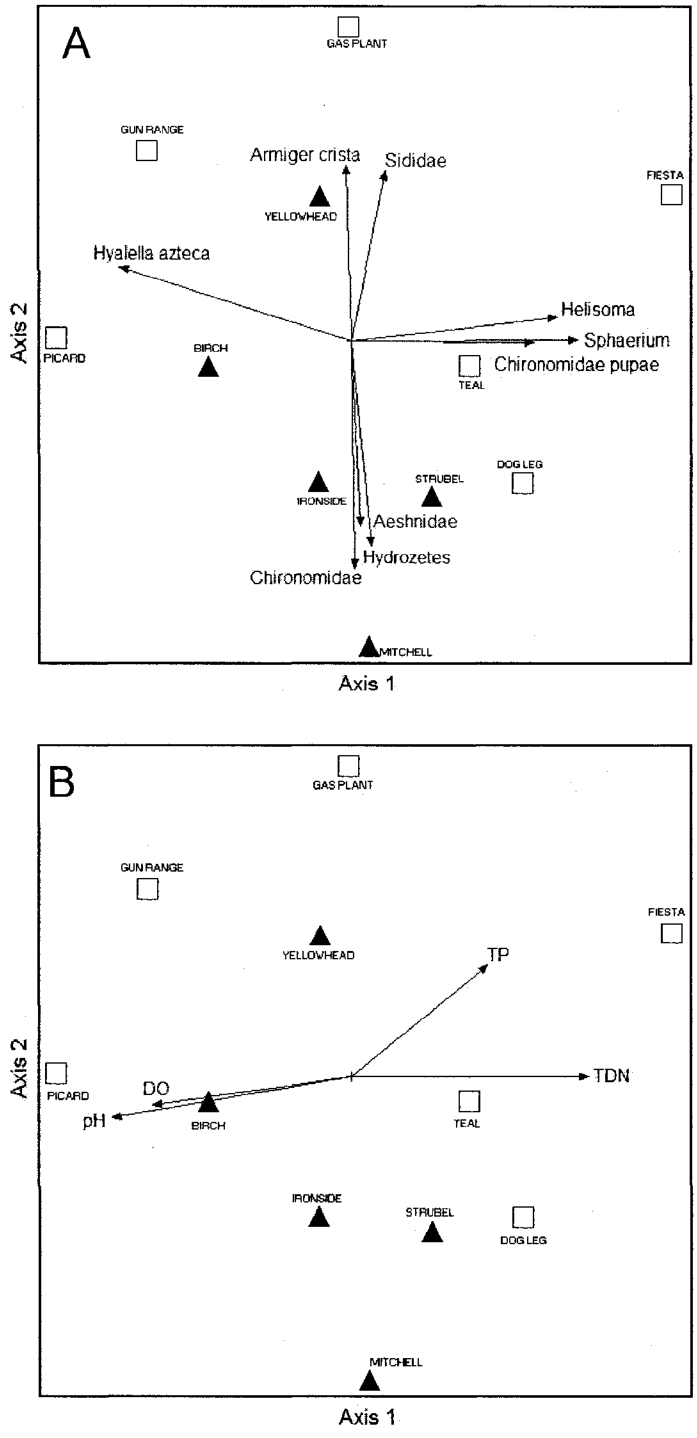


**Figure 3-1** Non-metric multidimensional scaling (NMS) joint plots of littoral macroinvertebrate communities in study lakes for August 2005. Vectors point in the direction of increasing (A) abundance ($r^2$>0.5) and (B) levels of environmental variables ($r^2$>0.3), and the length of a line indicates the strength of the relationship. Black triangles: stocked lakes (n=5); open squares: unstocked lakes (n=6).

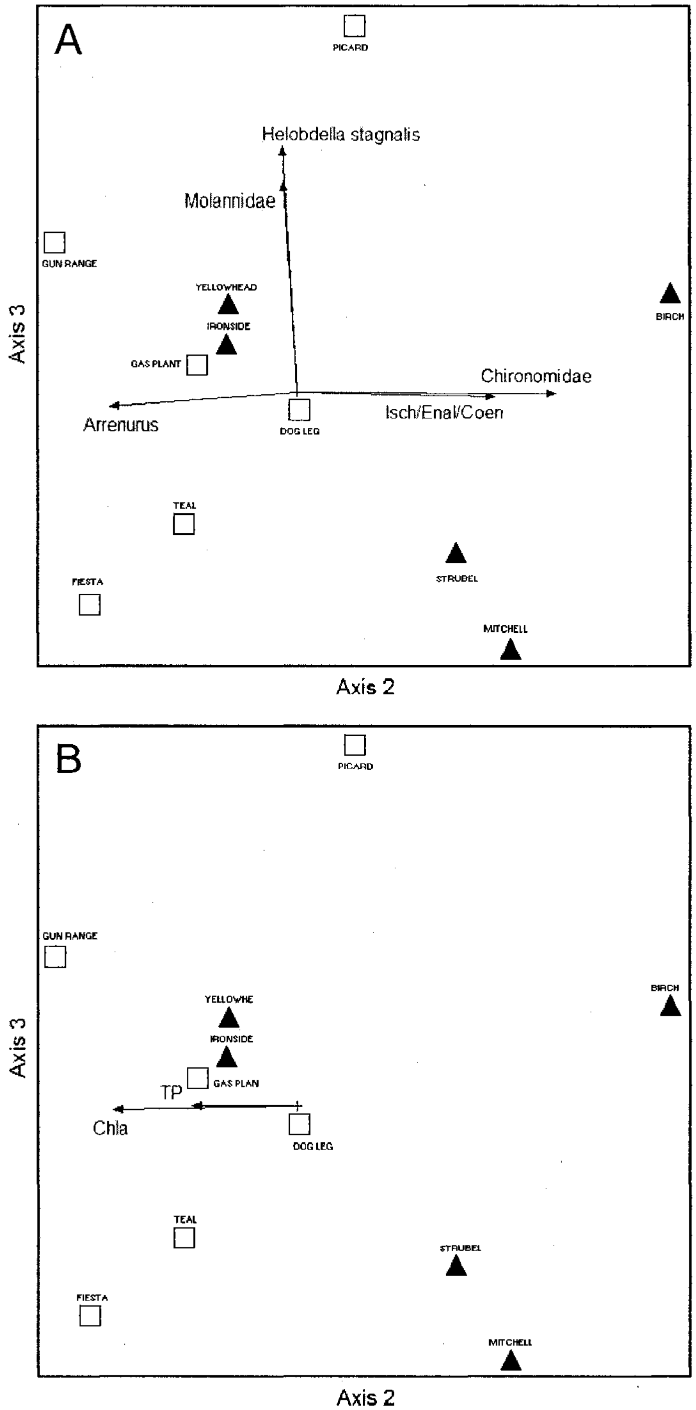


**Figure 3-2** Non-metric multidimensional scaling (NMS) joint plots of littoral macroinvertebrate communities in study lakes for May 2006. Vectors point in the direction of increasing (A) abundance ($r^2$>0.5) and (B) levels of environmental variables ($r^2$>0.3), and the length of a line indicates the strength of the relationship. Black triangles: stocked lakes (n=5); open squares: unstocked lakes (n=6).

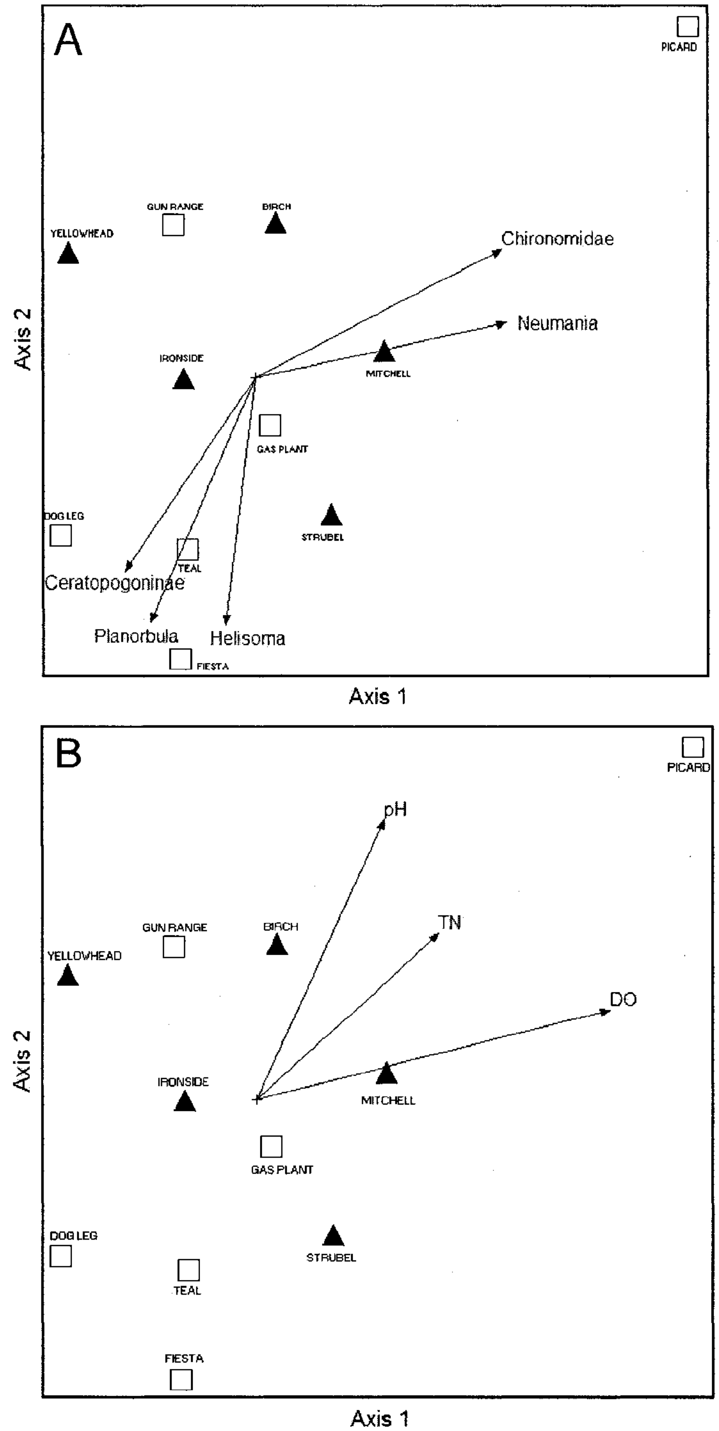


**Figure 3-3** Non-metric multidimensional scaling (NMS) joint plots of littoral macroinvertebrate communities in study lakes for August 2006. Vectors point in the direction of increasing (A) abundance ($r^2$>0.5) and (B) levels of environmental variables ($r^2$>0.3), and the length of a line indicates the strength of the relationship. Black triangles: stocked lakes (n=5); open squares: unstocked lakes (n=6).

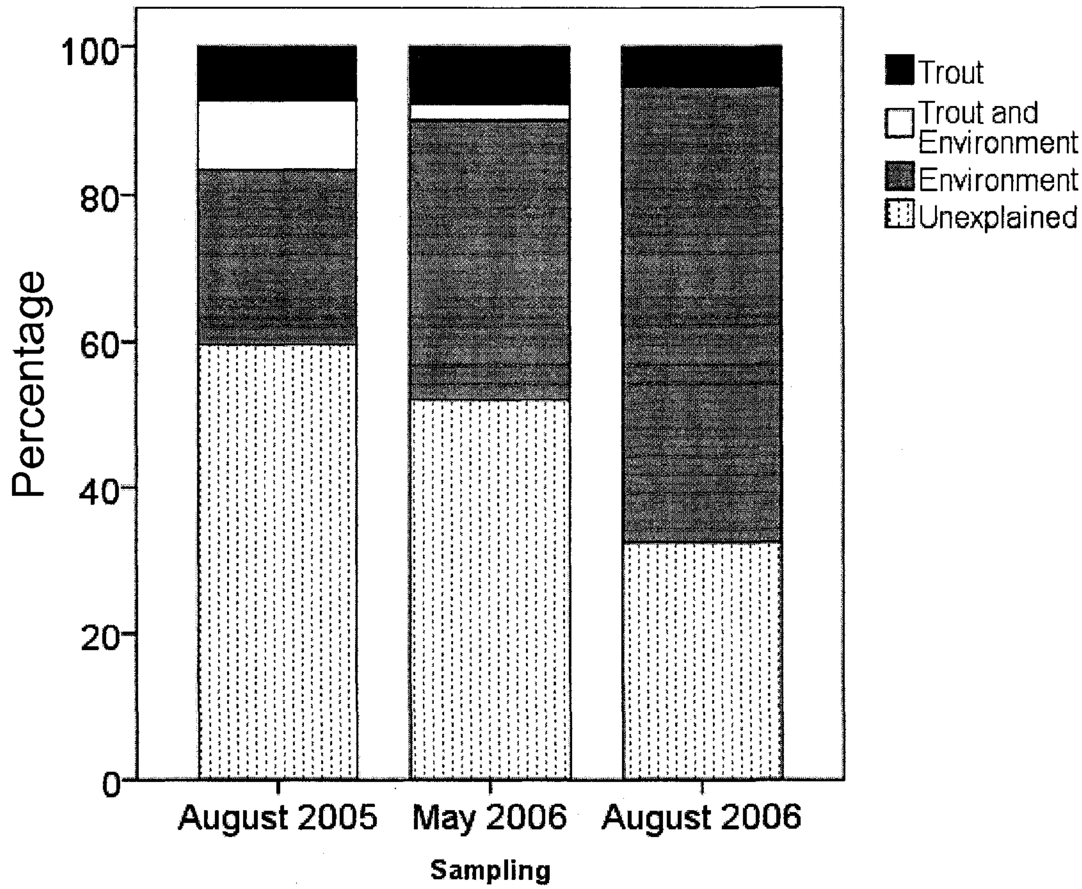


**Figure 3-4** Results of the two-way variance partitioning analysis for each sampling period. Values represent the percentages of variation in the invertebrate communities explained independently by environmental variables and trout taxa, the percentage shared by environment and trout, and variation unexplained by environment and trout.

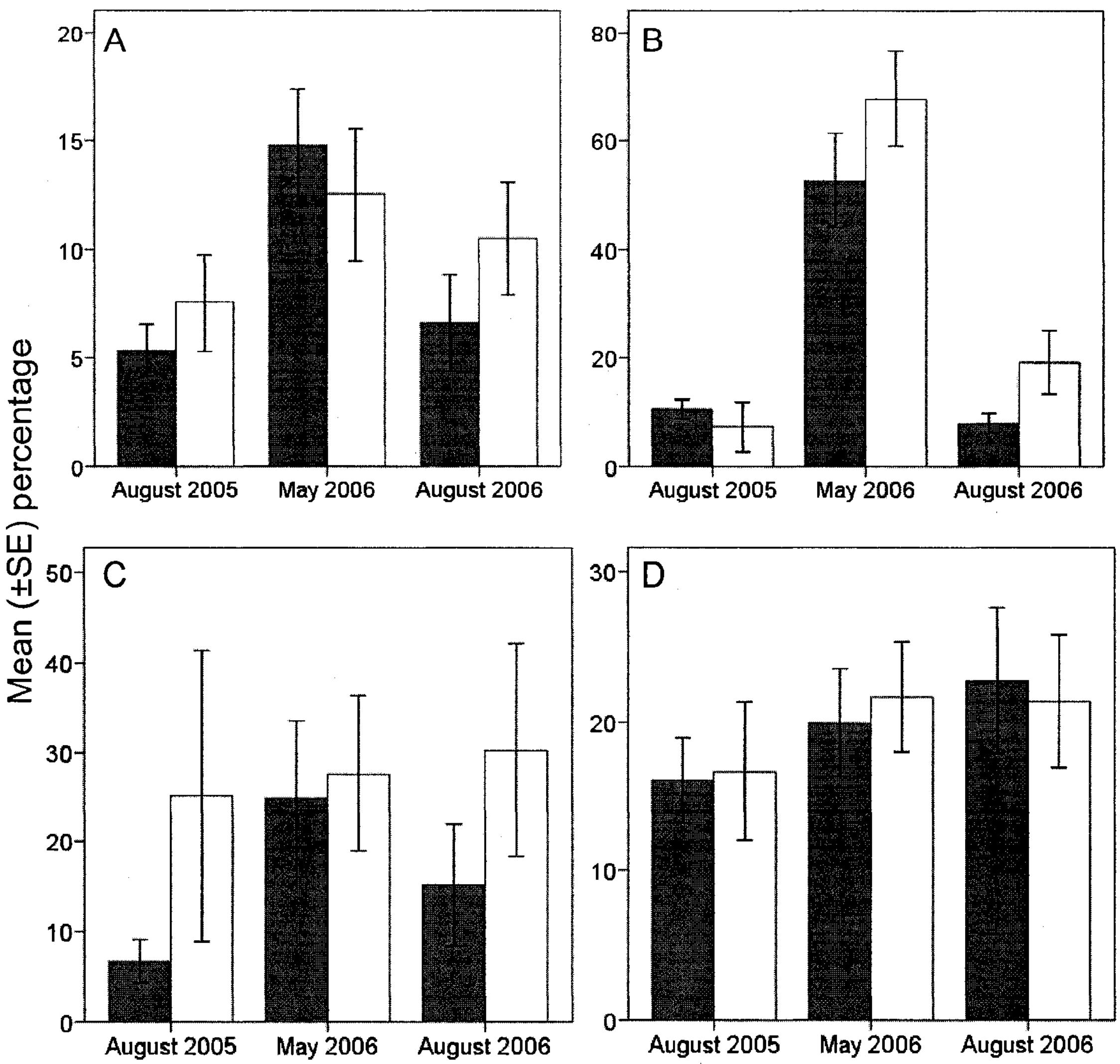


**Figure 3-5** Mean percentage (± SE) of large individuals in stocked and unstocked lakes, over the three sampling periods for A) Chironomidae larvae >9mm; B) *Hyalella azteca* >5mm; C) *Caenis* >3.5mm; and D) Ceratopogoninae larvae >12mm. Dark bars: stocked lakes (n=5), white bars: unstocked lakes (n=5).

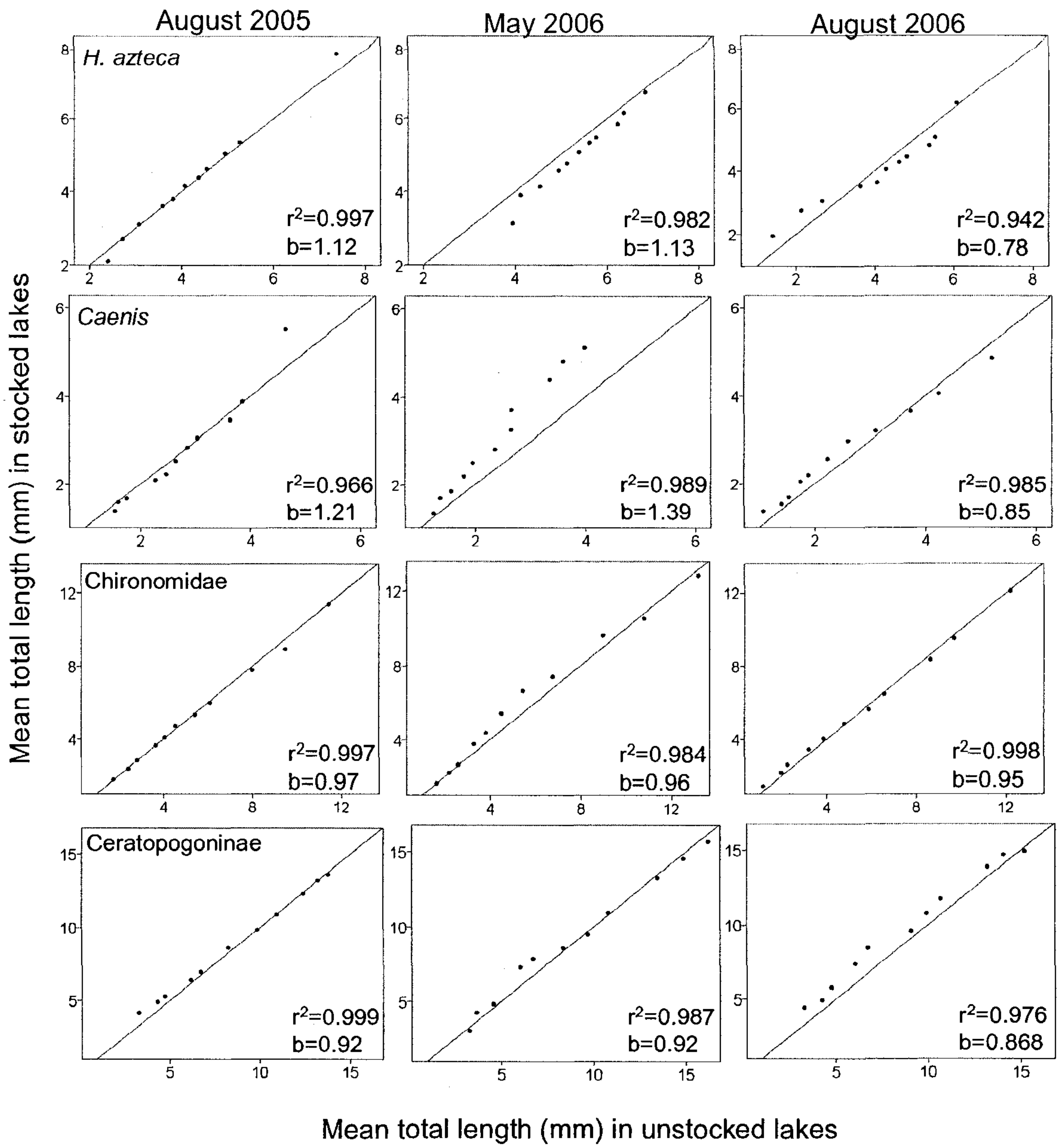


**Figure 3-6** Quantile-quantile plots comparing mean lengths at 11 quantiles (see text) between stocked and unstocked lakes for three sampling periods and four taxa. Plots include coefficients of determination ($r^2$) and slopes (b) from a least-squared linear regression. The solid line represent the 1:1 relationship.

# Chapter 4. General Discussion

Conventional wisdom suggests that introduction of a non-native species into what is considered a pristine ecosystem can result in ecological changes, including displacement or extirpation of native species. In fact, not all non-native species are harmful, and not all introductions have negative results (Gozlan 2008). Effects of an introduced species are unpredictable, and impacts vary depending on the system into which it is introduced (e.g., Eby *et al.* 2006).

Various trout are commonly introduced as sport fish in North America (Dunham *et al.* 2004) and across the globe (Macchi *et al.* 1999; Penczak 1999; Jackson *et al.* 2004); consequently, they are non-native in much of their contemporary ranges. The ubiquity of trout, however, does not allow for generalizations about their effects on ecosystems. Their impact on small-bodied fish varies. In New Zealand rivers, introduction of trout has been strongly linked to reduced abundance and extirpation of native small-bodied fishes (McInstosh *et al.* 1994; Townsend 1996). In other systems, such as subalpine Norway (Museth *et al.* 2002) and northern Ontario (Pink *et al.* 2007), trout coexist with, and appear to have no negative impact on, small bodied fish.

Trout impacts on invertebrates also vary. In alpine lakes devoid of small-bodied fish, trout can decimate invertebrate populations (Knapp *et al.* 2001; Parker and Schindler 2006). In other systems, however, such as New Zealand lakes (Wissinger *et al.* 2006) and Minnesota streams (Zimmerman and Vondracek 2007) trout seem to have no discernable impact on invertebrate communities. In fact, recent analyses of effects of introduced fish show that salmonids, in general, have a low incidence of ecological impact (ca. 5%; Gozlan 2008). These findings suggest that diverse responses to trout should be expected when they are introduced into diverse habitats and communities.

My research in the boreal foothills revealed another example of minimal trout impact on native forage fish and invertebrates. I observed no difference in forage fish densities in the presence of trout relative to lakes without trout. Forage fish in my lakes altered habitat use in the presence of trout, possibly to avoid predation or competition. One consequence of this habitat shift may be increased predation by forage fish on prey items not usually favoured. Few invertebrate taxa appeared to be particularly vulnerable to trout, and no effect of trout was observed at the community level. There was evidence of size-selective predation by trout on both forage fish and some invertebrate taxa, but in neither case was predation strong enough to affect abundance. Similarly, a study of amphibian populations on these lakes concurrent with my research found no negative effect of trout on relative abundance, adult size, or other life history traits (Schank 2008). Forage fish may avoid predation by changing their habitat in the presence of trout, and by using available refuges. Invertebrates were not likely naïve to the introduced predator since forage fish in these lakes fed on invertebrates prior to stocking. Overall, native fauna of boreal foothills lakes are minimally impacted by introduced trout.

*Recommendations & Conclusion*

In Alberta, recreational fishing is a popular pastime, and the majority of Alberta anglers believe the trout stocking program is important (Park 2007). The majority of Alberta anglers also believe that maintenance of healthy aquatic ecosystems and science-based management of fisheries are important (Park 2007). Fisheries managers are therefore faced with the joint tasks of continuing to stock sport fish while maintaining healthy aquatic systems. Because of this, management no longer focuses only on sport fish, but seeks a more complete knowledge of entire communities, which is important in the creation of ecologically successful game fishing opportunities (Wiley 2006). Inherent in these goals are proper selection of lakes for stocking, and continued monitoring post-stocking, to insure that the fishery is good for both anglers and the ecosystem.

Given the success of the current stocking and aeration program in the boreal foothills region, and the continually increasing pressure for more angling opportunities, more lakes will be stocked. A pre-stocking assessment of lakes targeted to be stocked would ideally follow the sampling protocol outlined in Chapters 2 and 3, and would include at least two summers to establish natural temporal variation. However, financial resources for such intense sampling may not always be available for provincial departments or non-governmental organizations. At a minimum, pre-stocking assessments for future stocked lakes in the boreal foothills should establish that the lakes have properties similar to those seen in this study. The stocked lakes in this study varied in a number of respects (e.g., size, depth, thermal characteristics, and forage fish populations) but no single lake stood out as being more severely impacted than the others, suggesting a range of biotic and abiotic characteristics capable of absorbing trout impacts. It is likely that many boreal foothill lakes fit into this range and will be minimally impacted by the introduction of trout. However, these results are only applicable to introductions to similar systems and should not be extrapolated to other piscivorous taxa or introductions into streams, alpine lakes, plains lakes, or wetlands.

Post-stocking assessments should be conducted to detect changes in forage fish densities, invertebrate community composition and abundance, and the trout population. Collecting information on the trout is easier and less time consuming (e.g., creel surveys), but monitoring the potential prey populations is also essential to meet long-term management objectives. Again, sampling procedures outlined in Chapters 2 and 3, or some modification thereof, would be ideal, although intense and time-consuming. Monitoring assesses the current status of particular lakes, documents changes over time, and allows managers to adjust and improve the stocking and aeration program.

This study showed, through 3 years of monitoring, that stocked trout are having little effect on native forage fishes or littoral invertebrates of boreal foothills lakes. Continued monitoring will produce a long-term data set that will be an invaluable source of information on both within- and among-lake variation over time, in both manipulated and natural lakes. Work such as this provides insight into the potential effects that human resource management practices can have, and in understanding our impacts, we can work to mitigate them and improve management practices.

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## Appendix A: Water Chemistry and Dissolved Oxygen and Temperature Profiles

**Table A-1**: Summary of the physical, chemical, and biological properties of the study lakes in 2005. Shallow and deep refer to temperature logger positions above and below the thermocline, respectively. Dissolved oxygen values are metalimnion means; all values for each lake are means of 2 months ± SE. TN: total nitrogen; TDN: total dissolved nitrogen; TP: total phosphorous; TDP: total dissolved phosphorous, Chl-*a*: fluormetric chlorophyll-*a*.

| Treatment/ Lake | Shallow logger °C | Deep logger °C | Dissolved oxygen $mgL^{-1}$ | TN $\mu gL^{-1}$ | TDN $\mu gL^{-1}$ | TP $\mu gL^{-1}$ | TDP $\mu gL^{-1}$ | Chl- *a* $\mu gL^{-1}$ | pH | Conductivity $\mu Scm^{-1}$ |
|---|---|---|---|---|---|---|---|---|---|---|
| **Stocked** | | | | | | | | | | |
| Ironside | 13.9 ± 0.03 | - | 3.7±0.7 | 789.99 | 811.68 | 18.40 | 7.90 | .23 | 7.9±0.05 | 231.5 ± 10.5 |
| Mitchell | 19.6 ± 0.06 | - | 6.9 ± 0.9 | 1071.00 | 920.57 | 13.40 | 6.60 | .53 | 8.3 ± 0.3 | 40.2 ± 0.4 |
| Birch | 18.3 ± 0.04 | 14.5 ± 0.02 | 9.4 ± 0.6 | 1092.00 | 760.45 | 34.50 | 5.10 | 1.90 | 8.6 ± 1.1 | 72.7 ± 0.8 |
| Strubel | 19.1 ± 0.03 | - | 7.1 ±0.6 | 576.15 | 555.13 | 7.60 | 2.90 | .24 | 8.4 ± 0.01 | 179.6 ± 3.5 |
| Yellowhead | 19.0 ±0.05 | - | 5.1 ±0.8 | 785.46 | 811.87 | 19.10 | 6.60 | .53 | 8.1 ± 0.1 | 126.1 ± 2.7 |
| **Unstocked** | | | | | | | | | | |
| Dog Leg | 19.0 ± 0.05 | 13.3 ±0.02 | 4.1 ± 0.8 | 1008.40 | 1182.83 | 41.30 | 26.90 | 1.38 | 7.1 ± 0.1 | 86.1 ± 6.2 |
| Fiesta | 17.5 ± 0.04 | 16.1 ± 0.05 | 2.8 ±0.7 | 1050.71 | 1053.90 | 47.70 | 17.90 | .59 | 7.4 ± 0.3 | 161.5 ± 3.5 |
| Gas Plant | 18.3 ± 0.06 | 15.8 ± 0.04 | 5.3 ± 0.8 | 909.96 | 887.34 | 53.40 | 26.20 | 1.73 | 7.3 ± 0.04 | 95.7 ± 7.0 |
| Gun Range | 18.8 ±0.04 | 15.5 ±0.03 | 4.0 ± 0.7 | 1074.73 | 874.90 | 30.00 | 11.70 | .29 | 8.1 ± 0.1 | 241.5 ± 7.5 |
| Teal | 17.9 ± 0.04 | 13.9 ± 0.03 | 4.4 ± 0.7 | 1099.36 | 1050.28 | 52.80 | 16.90 | .78 | 7.6 ± 0.2 | 175.3 ±2. 7 |
| Picard | - | - | 10.3 ±0.5 | | | | | | 8.60 | 82.50 |

**Table A-2.** Summary of the physical, chemical, and biological properties of the study lakes for 2006. Shallow and deep refer to temperature logger positions above and below the thermocline, respectively. Dissolved oxygen values are metalimnion means; all values for each lake are means of 4 months ± SE. TN: total nitrogen; TDN: total dissolved nitrogen; TP: total phosphorous; TDP: total dissolved phosphorous, Chl-*a*: spectrophometric chlorophyll-*a*.

| Treatment/ Lake | Shallow logger °C | Deep logger °C | Dissolved oxygen $mgL^{-1}$ | TN $\mu gL^{-1}$ | TDN $\mu gL^{-1}$ | TP $\mu gL^{-1}$ | TDP $\mu gL^{-1}$ | Chl- *a* $\mu gL^{-1}$ | pH | Conduct-ivity $\mu Scm^{-1}$ |
|---|---|---|---|---|---|---|---|---|---|---|
| **Stocked** | | | | | | | | | | |
| Ironside | - | 14.7 ± 0.1 | 6.6 ± 1 | 635 ± 74 | 661 ± 80 | 16 ± 3 | 7 ± 3 | 3 ± 1 | 7.6 ± 0.5 | 249 ± 6 |
| Mitchell | 22.1 ± 0.1 | 20.8 ± 0.1 | 8.2 ± 1 | 1072 ± 92 | 812 ± 20 | 16 ± 1 | 7 ± 1 | 3 ± 1 | 7.7 ± 0.5 | 44 ± 1 |
| Birch | 17.9 ± 0.3 | 12.8 ± 0.2 | 6.5 ± 1 | 480 ± 23 | 441 ± 34 | 8 ± 1 | 3 ± 0 | 5 ± 1 | 8.4 ± 0.2 | 122 ± 2 |
| Strubel | 20.8 ± 0.8 | 13.4 ± 0.6 | 6.9 ± 1 | 998 ± 217 | 687 ± 49 | 19 ± 1 | 6 ± 1 | 5 ± 1 | 7.8 ± 0.7 | 77 ± 1 |
| Yellowhead | 20.1 ± 0.2 | 12.9 ± 0.0 | 10 ± 0.4 | 741 ± 37 | 715 ± 49 | 16 ± 1 | 6 ± 1 | 2 ± 0 | 8.3 ± 0.3 | 184 ± 3 |
| **Unstocked** | | | | | | | | | | |
| Dog Leg | 19.2 ± 0.1 | 11.2 ± 0.0 | 4.8 ± 1 | 929 ± 79 | 815 ± 38 | 37 ± 9 | 16 ± 3 | 13 ± 4 | 8.4 ± 0.1 | 190 ± 12 |
| Fiesta | 20.5 ± 0.1 | 14.4 ± 0.0 | 6.7 ± 1 | 1096 ± 85 | 947 ± 143 | 51 ± 4 | 20 ± 4 | 11 ± 2 | 7.9 ± 0.3 | 117 ± 2 |
| Gas Plant | 20.9 ± 0.1 | 7.7 ± 0.0 | 6.7 ± 1 | 1005 ± 98 | 794 ± 73 | 23 ± 3 | 6 ± 1 | 7 ± 4 | 7.3 ± 1.2 | 248 ± 9 |
| Gun Range | 20.9 ± 0.1 | 14.3 ± 0.1 | 4.6 ± 1 | 889 ± 77 | 725 ± 43 | 32 ± 7 | 12 ± 2 | 8 ± 4 | 7.8 ± 0.2 | 188 ± 2 |
| Teal | 20.7 ± 0.1 | - | 9.4 ± 0.2 | 1022 ± 315 | 1090 ± 129 | 42 ± 3 | 14 ± 1 | 7 ± 1 | 8.2 ± 0.5 | 78 ±2 |
| Picard | - | 14.7 ± 0.1 | 6.6 ± 1 | 635 ± 74 | 661 ± 80 | 16 ± 3 | 7 ± 3 | 3 ± 1 | 7.6 ± 0.5 | 249 ± 6 |

**Table A-3** Summary of the physical, chemical, and biological properties of the study lakes for 2007. Shallow and deep refer to temperature logger positions above and below the thermocline, respectively. Dissolved oxygen values are metalimnion means; all values for each lake are means of 4 months ± SE. TN: total nitrogen; TDN: total dissolved nitrogen; TP: total phosphorous; TDP: total dissolved phosphorous, Chl-*a*: spectrophometric chlorophyll-*a*.

| **Treatment/ Lake** | **Shallow logger °C** | **Deep logger °C** | **Dissolved oxygen $mgL^{-1}$** | **TN $\mu gL^{-1}$** | **TDN $\mu gL^{-1}$** | **TP $\mu gL^{-1}$** | **TDP $\mu gL^{-1}$** | **Chl- *a* $\mu gL^{-1}$** | **pH** | **Conduct-ivity $\mu Scm^{-1}$** |
|---|---|---|---|---|---|---|---|---|---|---|
| **Stocked** | | | | | | | | | | |
| Ironside | 19.2 ± 0.1 | 12.3 ± 0.1 | 5.7 ± 0.6 | 658 ± 27.5 | 574 ± 92 | 13 ± 2 | 5 ± 2 | 2 ± 1 | 8.1 ± 0.1 | 255 ± 2 |
| Mitchell | 19.7 ± 0.2 | 16.1 ± 0.1 | 8.5 ± 0.5 | 930 ± 90 | 828 ± 69 | 15 ± 1 | 7 ± 1 | 3 ± 0 | 7.9 ± 0.2 | 46 ± 1 |
| Birch | 19.1 ± 0.2 | 16.5 ± 0.1 | 10.2 ± 0.4 | 523 ± 75 | 468 ± 48 | 8 ± 1 | 6 ± 4 | 0.8 ± 1 | 8.3 ± 0.2 | 193 ± 6 |
| Strubel | 19.8 ± 0.2 | 10.2 ± 0.1 | 7.7 ± 0.6 | 796 ± 103 | 624 ± 74 | 19 ± 3 | 5 ± 1 | 5 ± 1 | 8.0 ± 0.2 | 81 ± 1 |
| Yellowhead | 18.4 ± 0.2 | 15.0 ± 0.1 | 5.7 ± 0.7 | 754 ± 150 | 680 ± 93 | 16 ± 3 | 6 ± 1 | 3 ± 0 | 7.9 ± 0.3 | 113 ± 3 |
| | | | | | | | | | | |
| Dog Leg | 17.8 ± 0.1 | 4.9 ± 0.02 | 4.5 ± 0.7 | 803 ± 45 | 707 ± 76 | 25 ± 3 | 12 ± 3 | 3 ± 1 | 7.8 ± 0.3 | 177 ± 4 |
| Fiesta | 18.8 ± 0.2 | 10.3 ± 0.1 | 6.7 ± 0.9 | 956 ± 104 | 804 ± 112 | 46 ± 3 | 17 ± 1 | 8 ± 0 | 7.9 ± 0.2 | 123 ± 6 |
| Gas Plant | - | 5.6 ± 0.01 | 5.2 ± 0.8 | 881 ± 12 | 767 ± 55 | 22 ± 6 | 7 ± 1 | 6 ± 2 | 7.9 ± 0.2 | 218 ± 7 |
| Gun Range | - | 6.2 ± 0.02 | 4.3 ± 0.7 | 828 ± 50 | 741 ± 55 | 26 ± 4 | 14 ± 1 | 4 ± 1 | 7.8 ± 0.1 | 223 ± 3 |
| Teal | - | - | 8.9 ± 0.7 | 1040 ± 40 | 875 ± 67 | 25 ±2 | 11 ± 0 | 2 ± 1 | 7.9 ± 0.5 | 70 ± 9 |
| Picard | 19.2 ± 0.1 | 12.3 ± 0.1 | 5.7 ± 0.6 | 658 ± 27.5 | 574 ± 92 | 13 ± 2 | 5 ± 2 | 2 ± 1 | 8.1 ± 0.1 | 255 ± 2 |

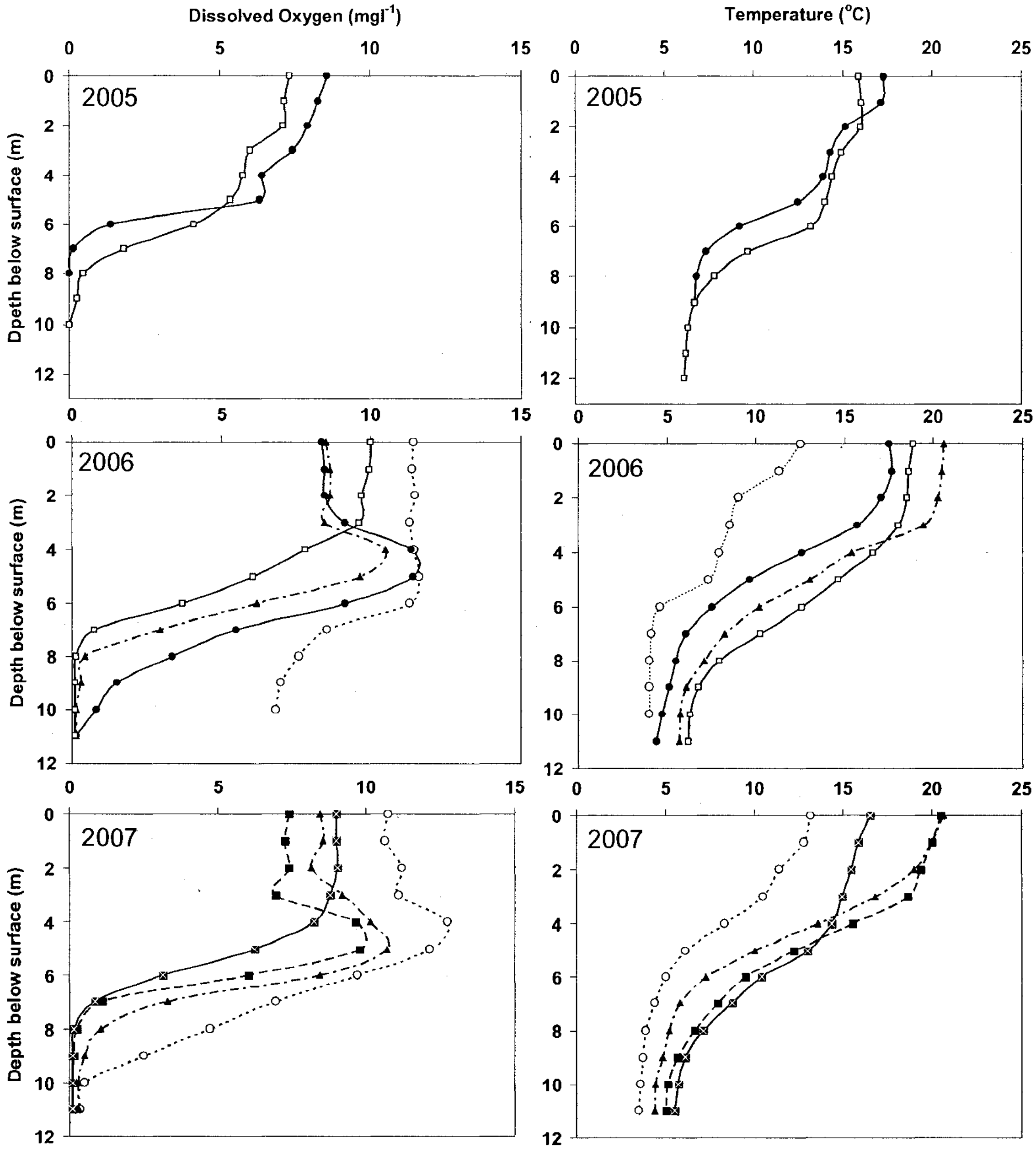


**Figure A-1** Ironside Pond (stocked and aerated) dissolved oxygen and temperature profiles for the summer months of 2005-2007. May: open circle; June: solid circle; July: solid triangle; August open square. For 2007 only: early August: solid square with broken line; late August: black square with solid line.

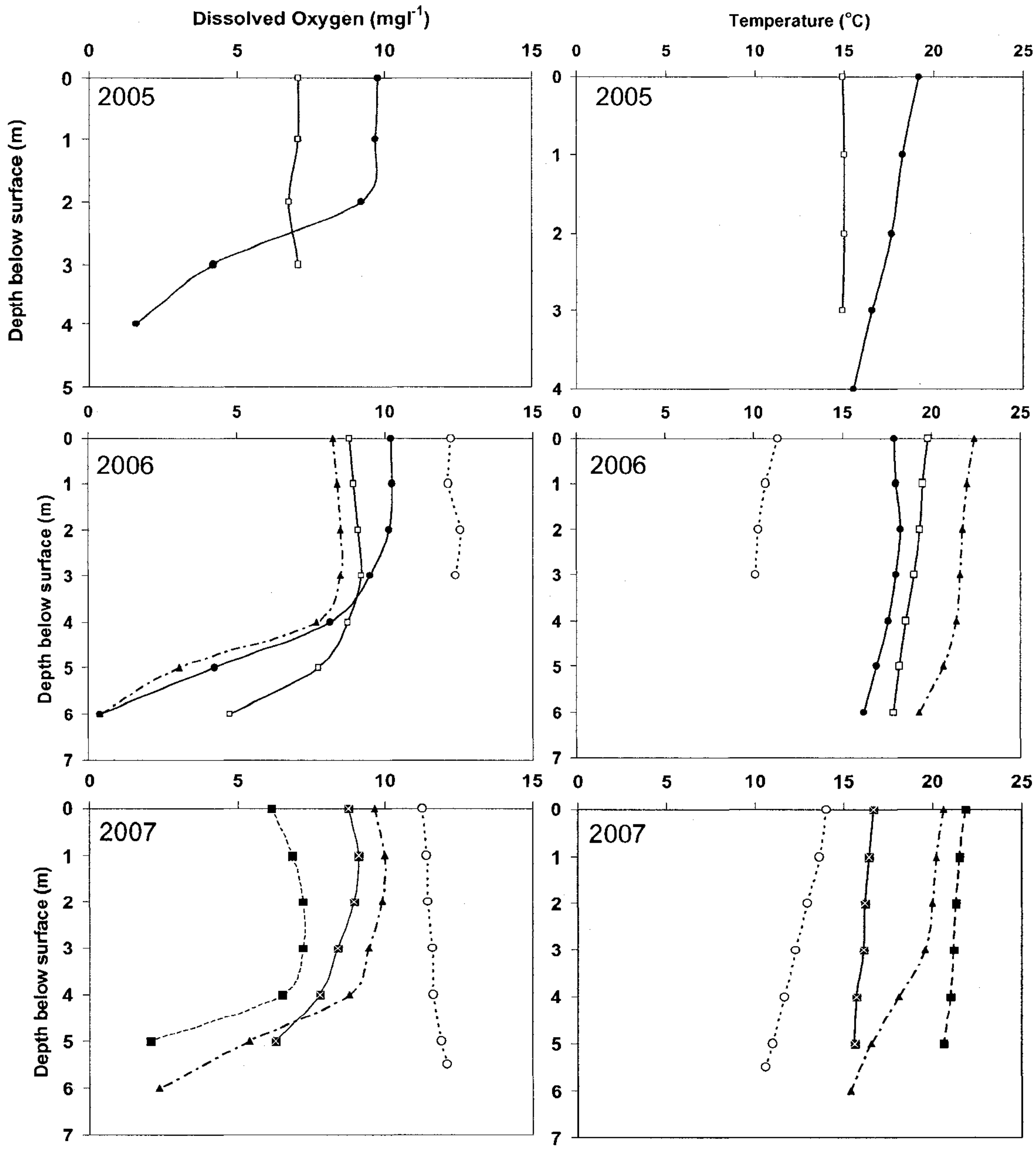


**Figure A-2** Mitchell lake (stocked and aerated) dissolved oxygen and temperature profiles for the summer months of 2005-2007. May: open circle; June: solid circle; July: solid triangle; August open square. For 2007 only: early August: solid square with broken line; late August: black square with solid line.

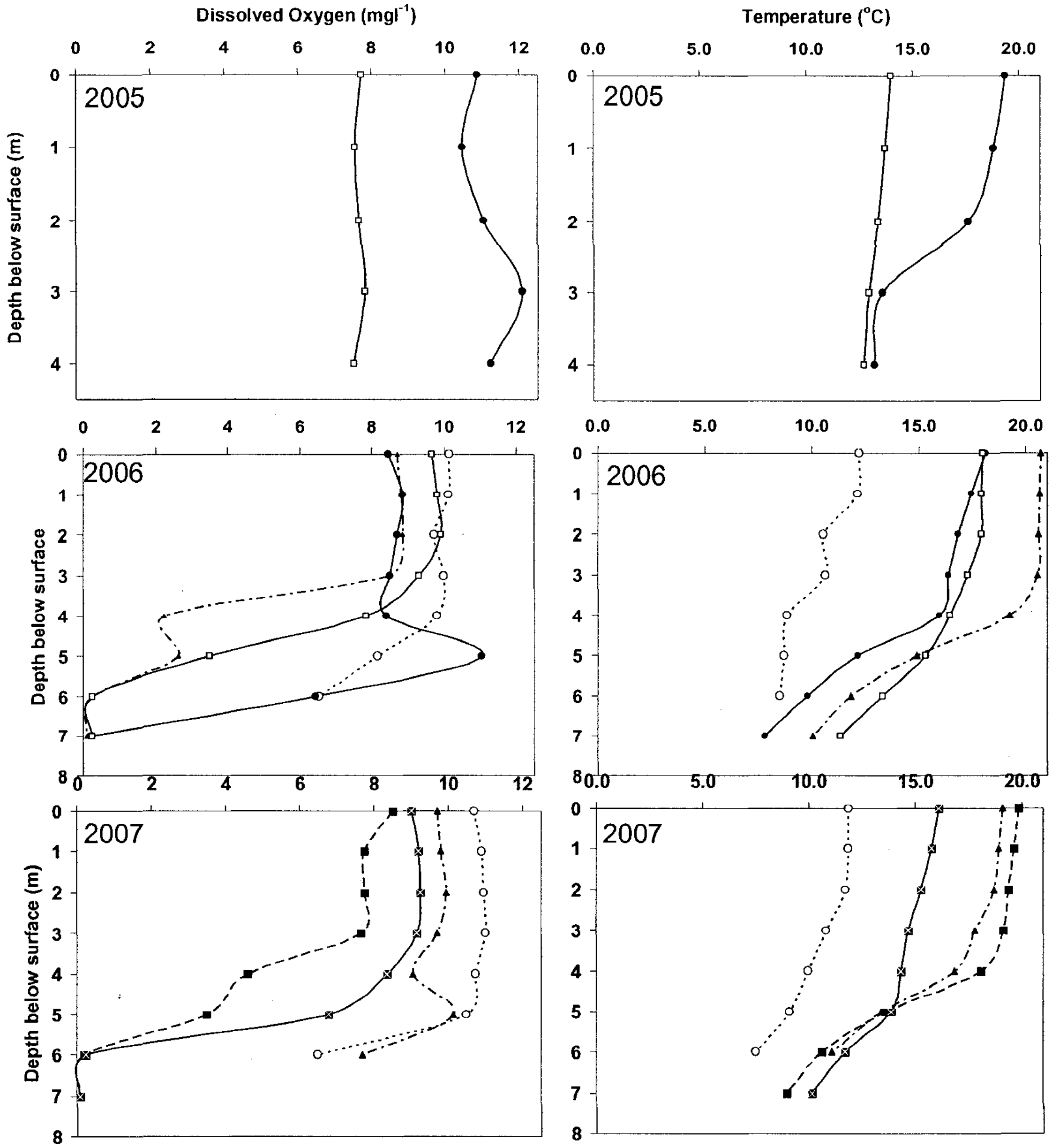


**Figure A-3** Birch Lake (stocked) dissolved oxygen and temperature profiles for the summer months of 2005-2007. May: open circle; June: solid circle; July: solid triangle; August open square. For 2007 only: early August: solid square with broken line; late August: black square with solid line.

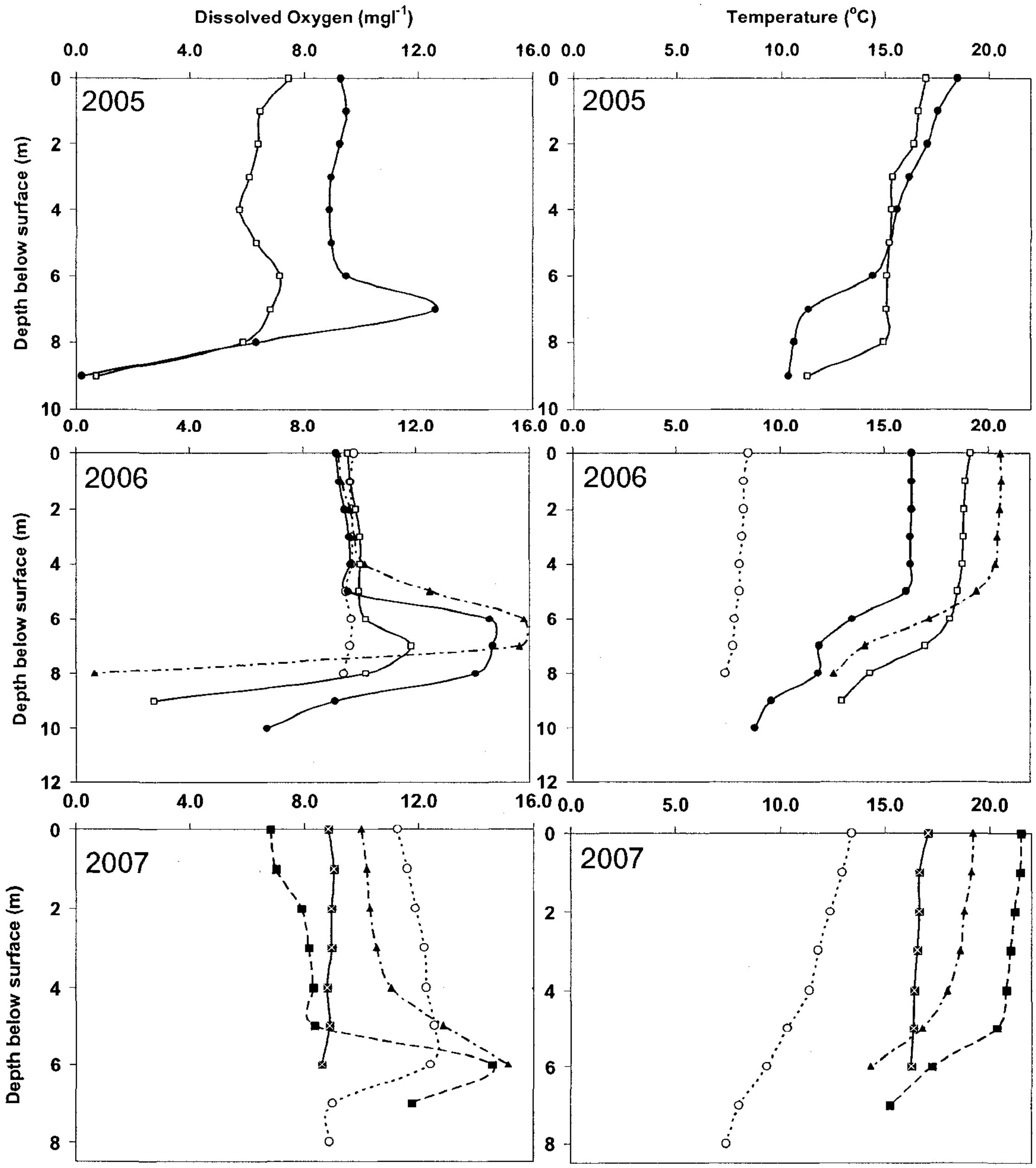


**Figure A-4** Strubel Lake (stocked) dissolved oxygen and temperature profiles for the summer months of 2005-2007. May: open circle; June: solid circle; July: solid triangle; August open square. For 2007 only: early August: solid square with broken line; late August: black square with solid line.

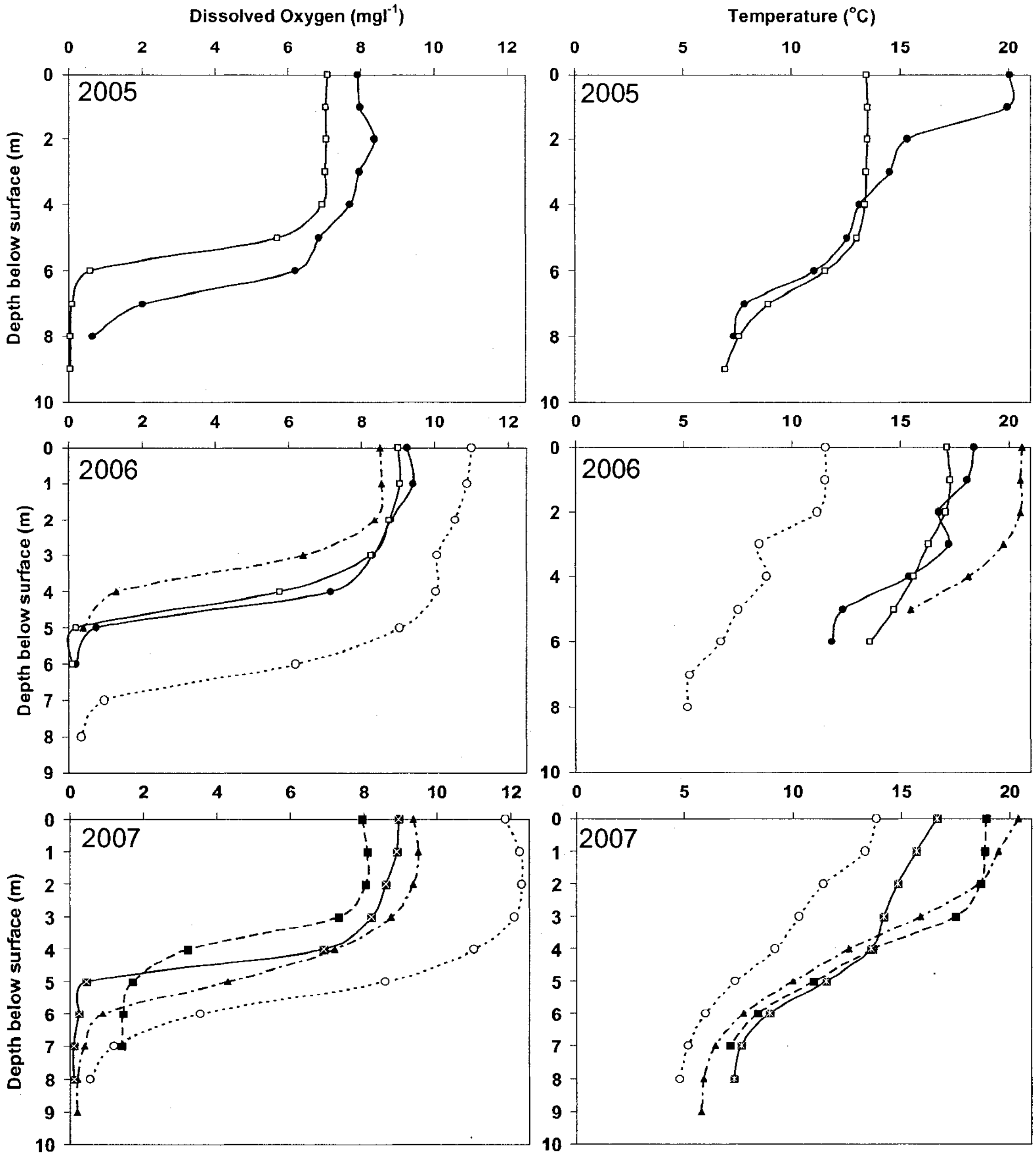


**Figure A-5** Yellowhead Lake (stocked) dissolved oxygen and temperature profiles for the summer months of 2005-2007. May: open circle; June: solid circle; July: solid triangle; August open square. For 2007 only: early August: solid square with broken line; late August: black square with solid line.

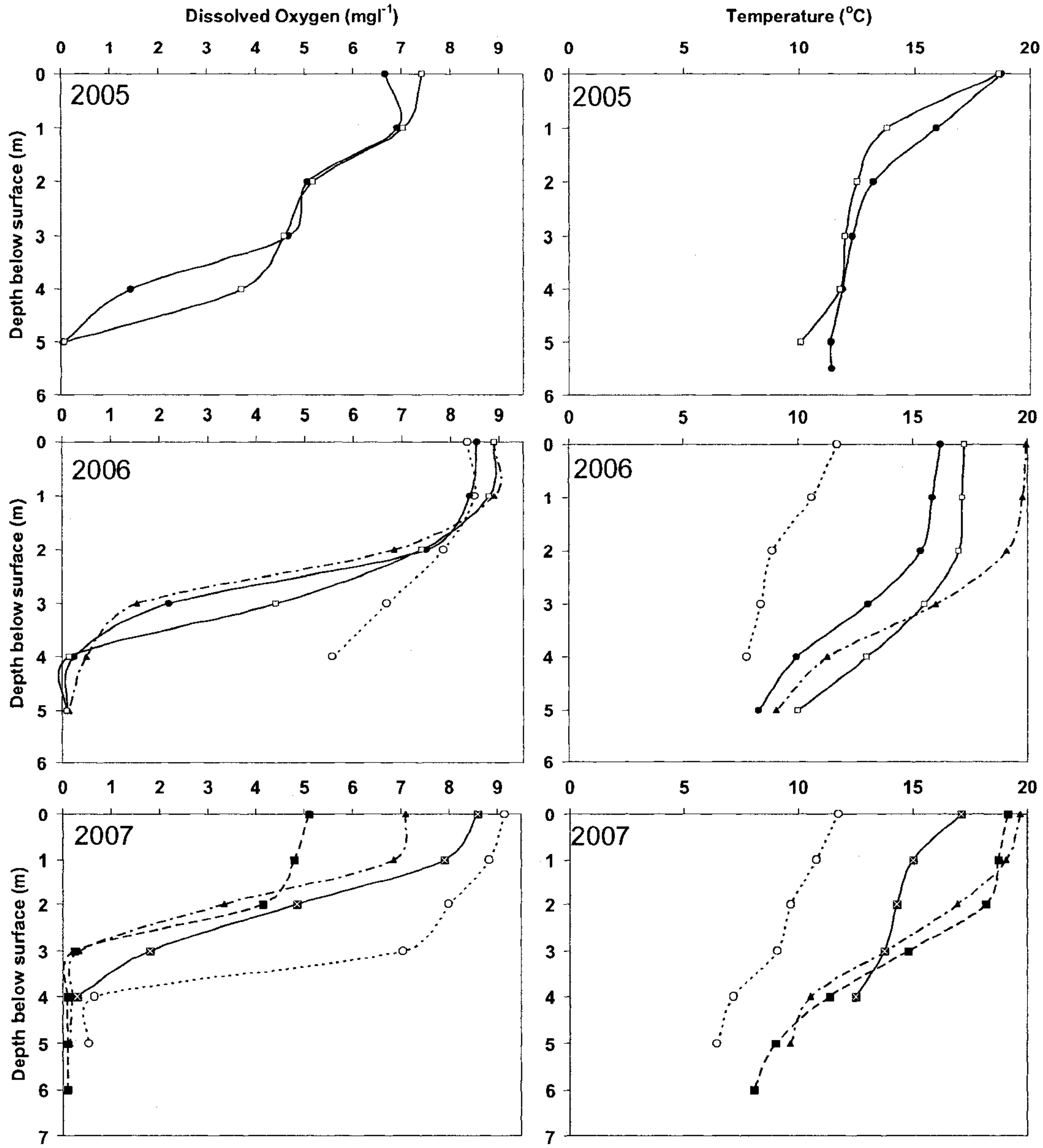


**Figure A-6** Dog Leg Lake (unstocked) dissolved oxygen and temperature profiles for the summer months of 2005-2007. May: open circle; June: solid circle; July: solid triangle; August open square. For 2007 only: early August: solid square with broken line; late August: black square with solid line.

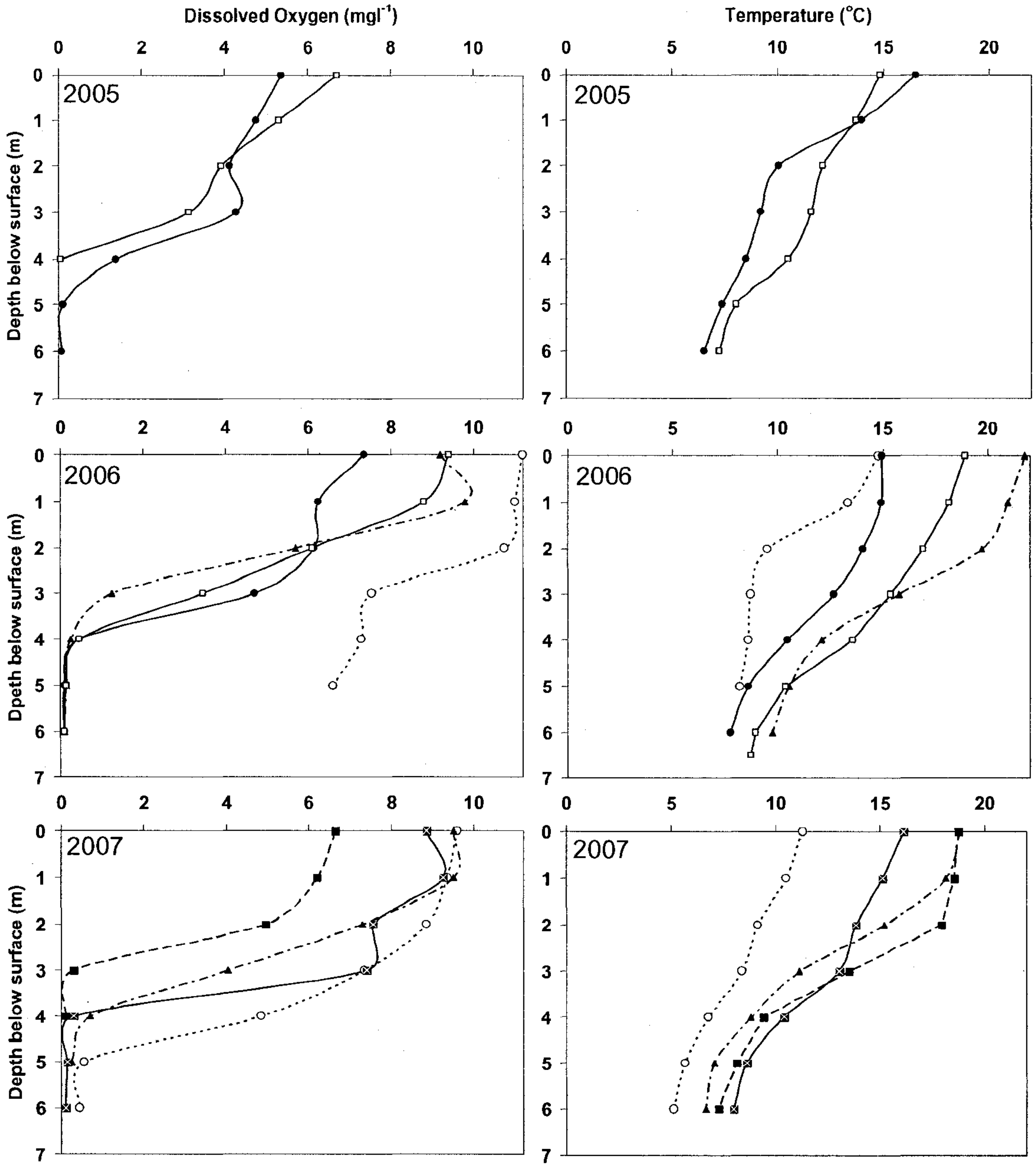


**Figure A-7** Fiesta Lake (unstocked) dissolved oxygen and temperature profiles for the summer months of 2005-2007. May: open circle; June: solid circle; July: solid triangle; August open square. For 2007 only: early August: solid square with broken line; late August: black square with solid line.

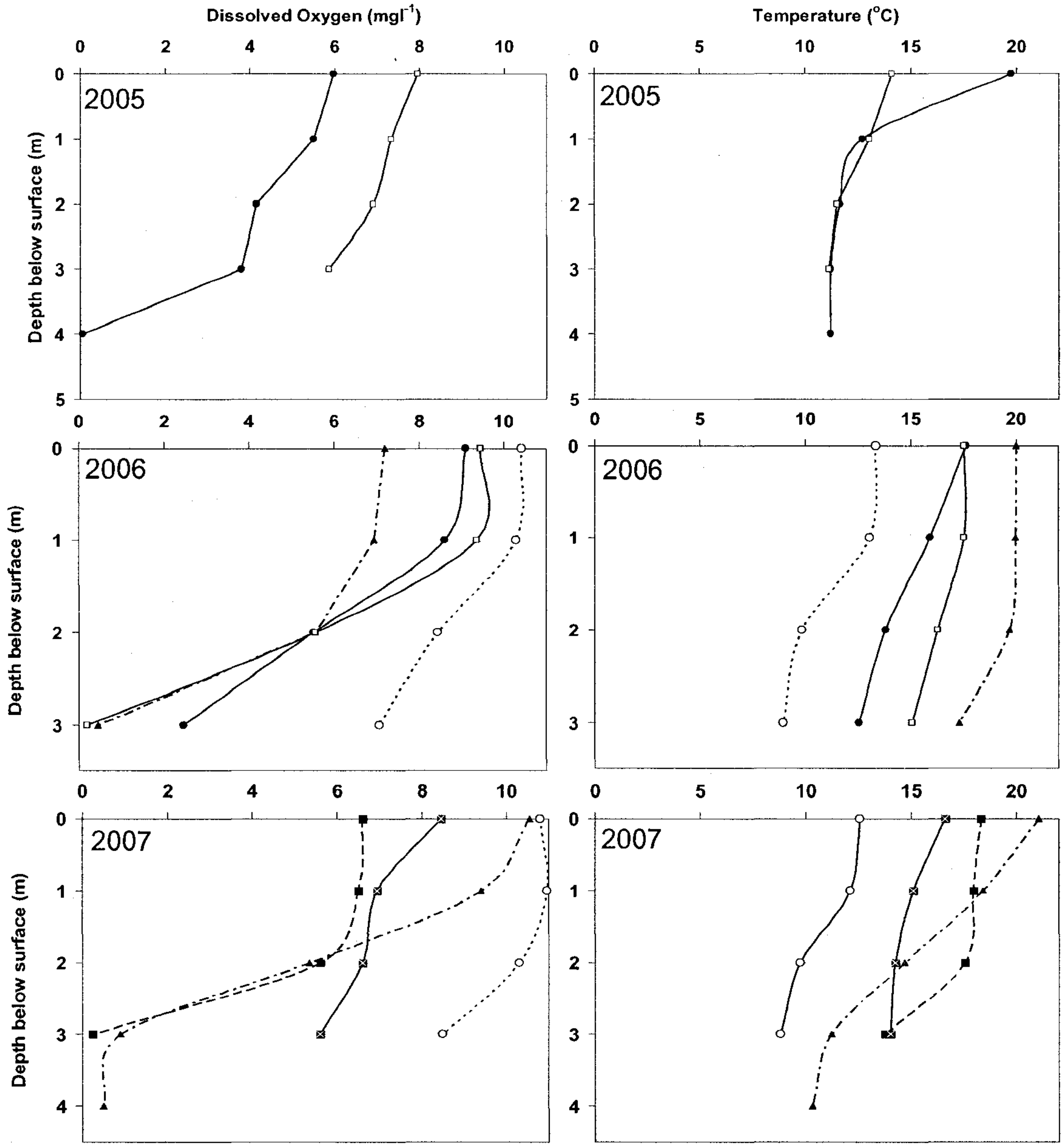


**Figure A-8** Gas Plant Lake (unstocked) dissolved oxygen and temperature profiles for the summer months of 2005-2007. May: open circle; June: solid circle; July: solid triangle; August open square. For 2007 only: early August: solid square with broken line; late August: black square with solid line.

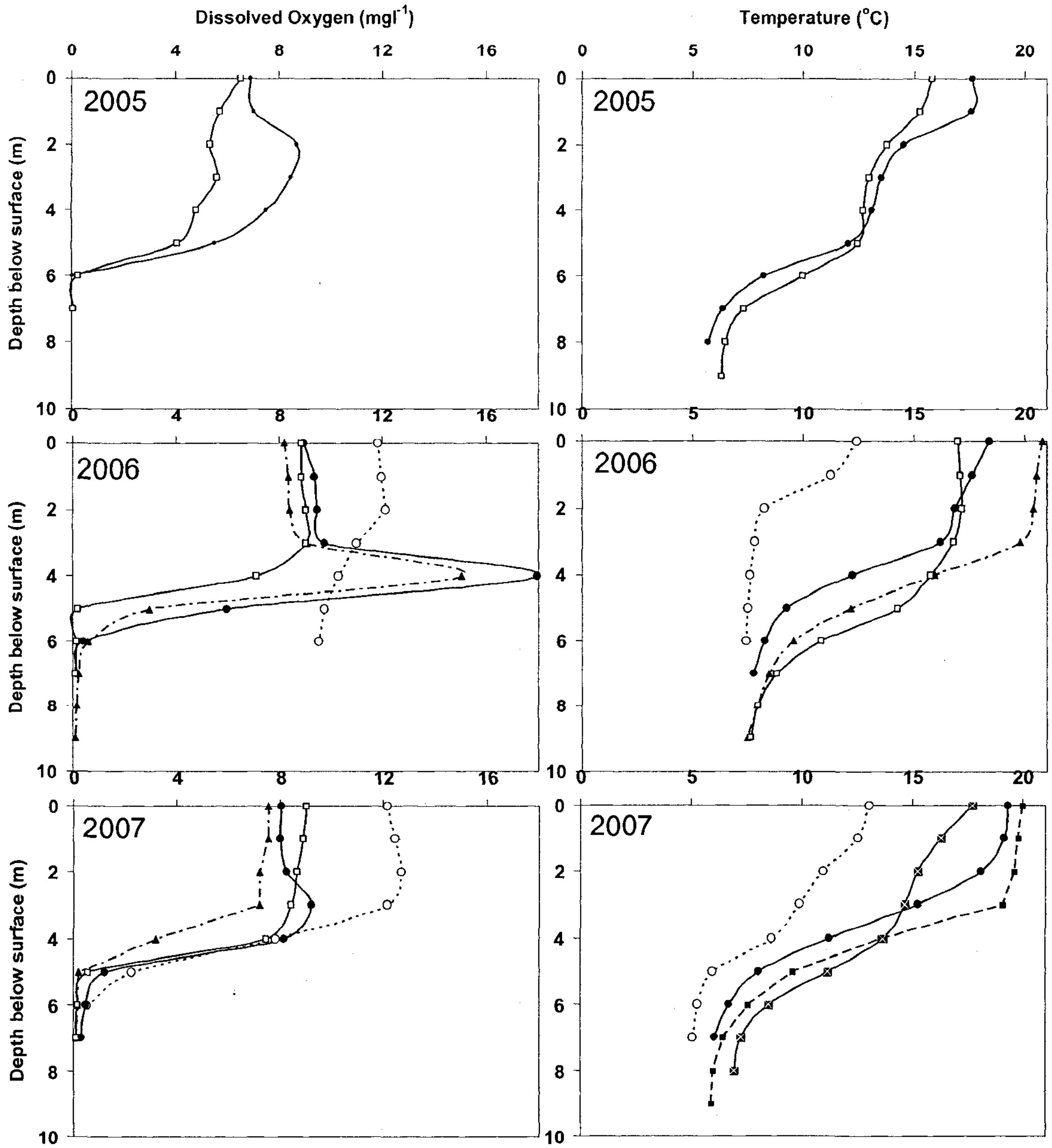

**Figure A-9** Gun Range Lake (unstocked) dissolved oxygen and temperature profiles for the summer months of 2005-2007. May: open circle; June: solid circle; July: solid triangle; August open square. For 2007 only: early August: solid square with broken line; late August: black square with solid line.

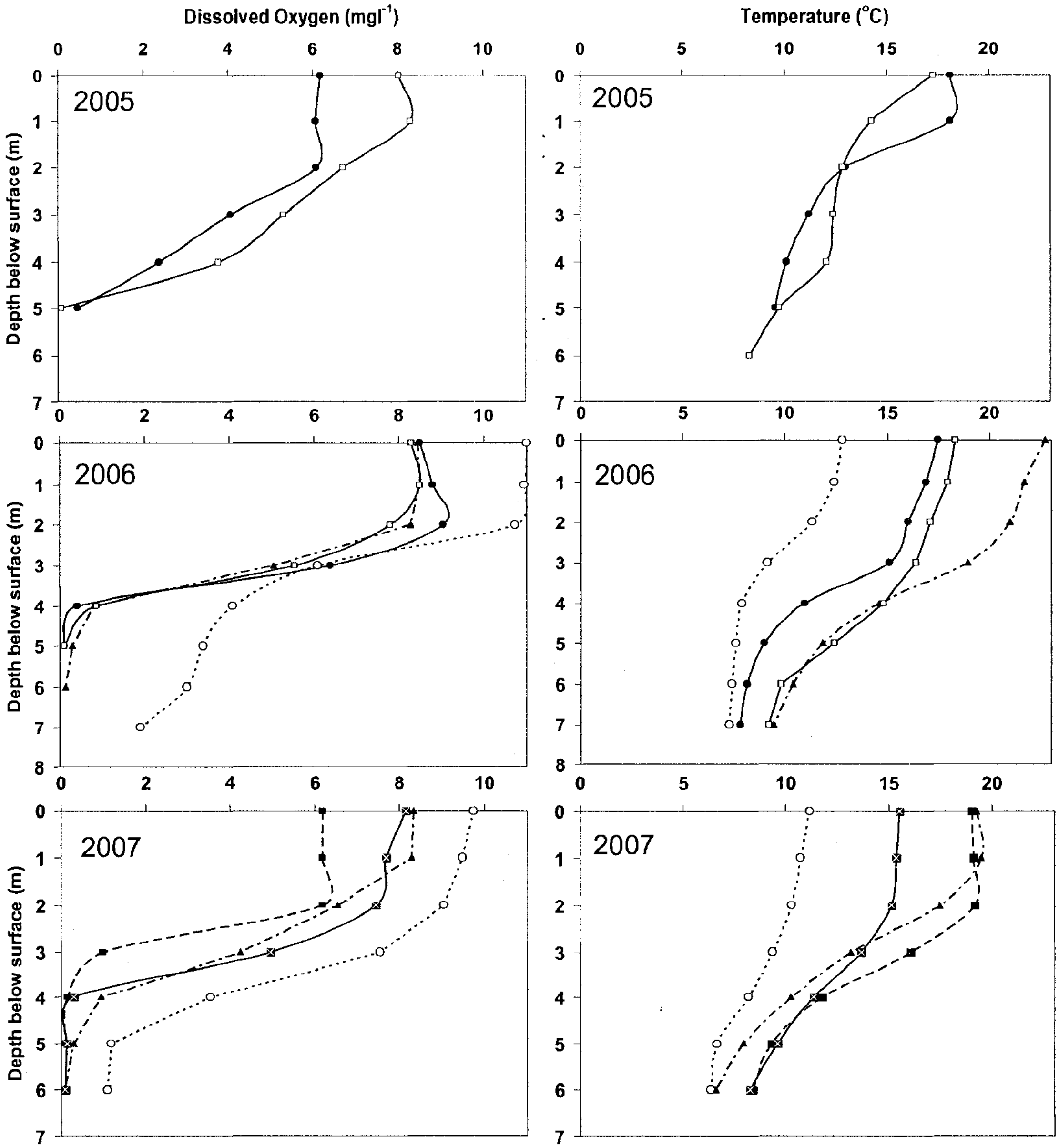


**Figure A-10** Teal Lake (unstocked) dissolved oxygen and temperature profiles for the summer months of 2005-2007. May: open circle; June: solid circle; July: solid triangle; August open square. For 2007 only: early August: solid square with broken line; late August: black square with solid line.

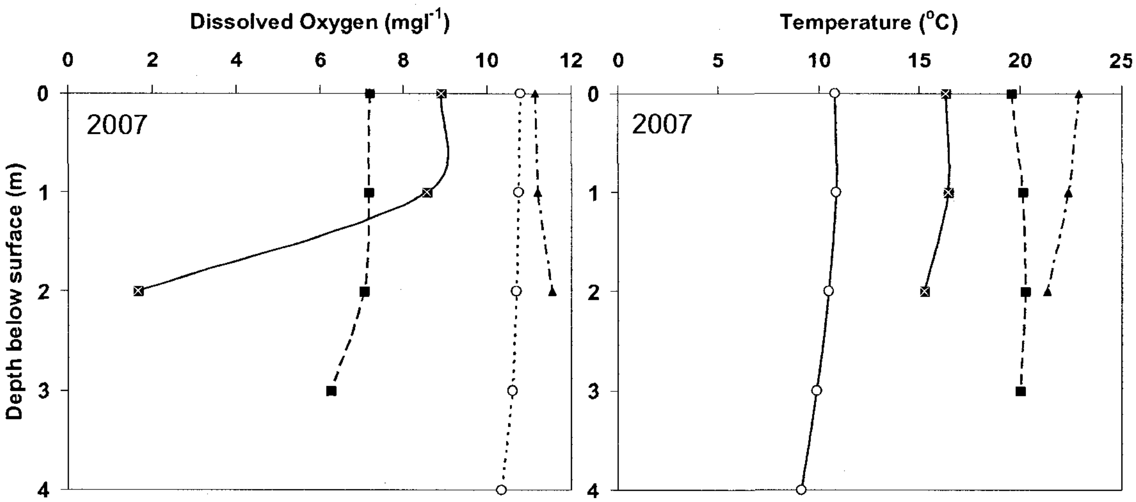


**Figure A-11** Picard Lake (unstocked) dissolved oxygen and temperature profiles for the summer months of 2007 May: open circle; July: solid triangle; early August: solid square with broken line; late August: black square with solid line.

## Appendix B. Forage Fish

**Table B-1**. Yearly mean (± SE) dace catch-per-unit-effort (fish/hour) for each lake containing dace. Fish were caught in unbaited Gee minnow traps (2 cm opening, 5 mm mesh). n/a: lake not sampled.

| Treatment | Lake | 2005 | 2006 | 2007 |
|---|---|---|---|---|
| Stocked | Ironside | n/a | 36.5 ± 4.1 | n/a |
| | Mitchell | 24.6 ± 2.3 | 16.6 ± 2.0 | 39.2 ± 4.7 |
| | Birch | 17.2 ± 3.5 | 1.1 ± 0.3 | n/a |
| | Strubel | 34.0 ± 5.9 | 22.1 ± 2.7 | n/a |
| | Yellowhead | 1.9 ± 1.6 | 0.45 ± 0.1 | n/a |
| Unstocked | Dog Leg | 1.6 ± 0.3 | 1.0 ± 0.2 | 0.88 ± 0.1 |
| | Fiesta | 1.2 ± 0.8 | 24.4 ± 2.2 | 3.6 ± 0.4 |
| | Gas Plant | 3.2 ± 1.5 | 2.7 ± 0.4 | n/a |
| | Gun Range | 0.90 ± 0.4 | 14.4 ± 1.8 | n/a |
| | Teal | 10.0 ± 2.7 | 20.7 ± 4.5 | n/a |

**Table B-2**. Yearly mean (± SE) fathead minnow catch-per-unit-effort (fish/hour) for each lake containing fathead minnow, life history stages combined. Fish were caught in unbaited Gee minnow traps (2 cm opening, 5 mm mesh). n/a: lake not sampled.

| Treatment | Lake | 2005 | 2006 | 2007 |
|---|---|---|---|---|
| Stocked | Birch | 4.9 ± 0.9 | 0.11 ± 0.02 | n/a |
| | Yellowhead | 9.3 ± 4.8 | 7.3 ±0.4 | n/a |
| Unstocked | Dog Leg | 0.89 ± 0.2 | 0.26 ± 0.1 | 0.16 ± 0.04 |
| | Fiesta | 0.07 ± 0.04 | 2.7±0.4 | 0.22 ± 0.03 |
| | Gas Plant | 0.69 ± 0.4 | 0.08 ±0.2 | n/a |
| | Gun Range | 3.6 ± 1.7 | 8.4 ± 0.9 | n/a |
| | Teal | 1.0 ± 0.5 | 0.82 ± 0.3 | n/a |
| | Picard | .0002 ± 0.5 | 1.0 ± 0.2 | n/a |

**Table B-3**. Yearly mean (± SE) brook stickleback catch-per-unit-effort (fish/hour) for each lake containing brook stickleback. Fish were caught in unbaited Gee minnow traps (2 cm opening, 5 mm mesh). n/a: lake not sampled.

| Treatment | Lake | 2005 | 2006 | 2007 |
|---|---|---|---|---|
| Stocked | Birch | 3.2 ± 0.7 | 0.13 ± 0.03 | n/a |
| | Strubel | 0.37 ± 0.08 | 0.39 ± 0.04 | n/a |
| | Yellowhead | 0.54 ± 0.2 | 0.28 ±0.05 | n/a |
| Unstocked | Dog Leg | 0.07 ± 0.02 | 1.25 ±0.2 | 0.19 ± 0.03 |
| | Fiesta | 0.22 ± 0.06 | 1.4 ±0.1 | 0.66 ± 0.06 |
| | Gas Plant | 6.3 ± 1.1 | 0.51 ± 0.05 | n/a |
| | Gun Range | 0.57 ± 0.1 | 2.46 ± 0.2 | n/a |
| | Teal | 1.2 ± 0.2 | 0.57 ± 0.1 | n/a |
| | Picard | 0.0006 ± 0.1 | 0.36 ± 0.1 | n/a |

**Table B-4**. Yearly mean ± SE (n) dace total length (mm) for each lake containing dace. n/a: lake not sampled.

| Treatment | Lake | 2005 | 2006 | 2007 |
|---|---|---|---|---|
| Stocked | Ironside | 57.0 ± 0.5 (167) | 62.1 ± 0.5 (648) | n/a |
| | Mitchell | 60.7 ± 1.1 (1170 | 61.8 ± 0.5 (787) | 57.7 ± 0.5 (417) |
| | Birch | 57.7 ± 1.1 (109) | 61.1 ± 0.7 (353) | n/a |
| | Strubel | 56.9 ± 0.6 (200) | 58.2 ± 0.4 (650) | 59.0 ± 0.8 (160) |
| | Yellowhead | 67.6 ± 0.6 (100) | 59.6 ± 0.5 (331) | n/a |
| Unstocked | Dog Leg | 57.6 ± 0.8 (98) | 60.2 ± 0.4 (597) | 66.3 ± 1.0 (94) |
| | Fiesta | 48.1 ± 0.6 (102) | 58.2 ± 0.5 (687) | 60.6 ± 0.5 (581) |
| | Gas Plant | 61.4 ± 0.8 (100) | 59.1 ± 0.4 (526) | 61.2 ± 0.7 (149) |
| | Gun Range | 48.6 ± 0.5 (89) | 52.2 ± 0.3 (377) | n/a |
| | Teal | 48.0 ± 0.5 (107) | n/a | n/a |

**Table B-5**. Yearly mean ± SE (n) fathead minnow total length (mm) for each lake containing fathead minnows, all life history stages combined. n/a: lake not sampled.

| Treatment | Lake | 2005 | 2006 | 2007 |
|---|---|---|---|---|
| Stocked | Birch | 53.8 ± 0.6 (117) | 56.5 ± 0.6 (101) | n/a |
| | Yellowhead | 59.9 ± 0.6 (102) | 65.1 ± 0.3 (632) | n/a |
| Unstocked | Dog Leg | 53.2 ± 2.2 (32) | 58.3 ± 0.3 (398) | 60.5 ± 0.5 (298) |
| | Fiesta | 56.0 ± 6.1 (6) | 57.2 ± 0.3 (361) | 58.2 ± 0.4 (302) |
| | Gas Plant | 55.8 ± 1.0 (53) | 56.6 ± 0.7 (228) | n/a |
| | Gun Range | 52.8 ± 0.5 (139) | 54.7 ± 0.3 (381) | n/a |
| | Teal | 59.1 ± 0.5 (104) | n/a | n/a |
| | Picard | 50.0 (1) | n/a | n/a |

**Table B-6**. Mean ± SE (n) total length (mm) for the different life history stages of fathead minnow in 2006.

| Treatment | Lake | Male | Female | Juvenile |
|---|---|---|---|---|
| Stocked | Birch | 57.6 ± 0.5 (51) | 60.6 ± 1.1 (24) | 50.3 ± 0.9 (26) |
| | Yellowhead | 70.4 ± 0.2 (300) | 61.1 ± 0.3 (300) | 53.7 ± 0.8 (32) |
| Unstocked | Dog Leg | 62.5 ± 0.3 (128) | 58.4 ± 0.3 (181) | 52.4 ± 0.2 (89) |
| | Fiesta | 63.1 ± 0.3 (116) | 56.0 ± 0.3 (169) | 51.0 ± 0.3 (76) |
| | Gas Plant | 68.4 ± 0.6 (47) | 60.2 ± 0.5 (98) | 45.6 ± 0.4 (83) |
| | Gun Range | 59.8 ± 0.3 (138) | 54.2 ± 0.3 (135) | 48.8 ± 0.3 (108) |

**Table B-7**. Mean ± SE (n) total length of different life history stages of fathead minnow in 2007.

| Treatment | Lake | Male | Female | Juvenile |
|---|---|---|---|---|
| Unstocked | Dog Leg | 67.5 ± 0.4 (98) | 63.5 ± 0.4 (100) | 50.7 ± 0.4 (100) |
| | Fiesta | 65.2 ± 0.4 (101) | 59.9 ± 0.4 (101) | 50.2 ± 0.4 (100) |

**Table B-8**. Yearly mean ± SE (n) brook stickleback total length (mm) for each lake containing brook stickleback. n/a: lake not sampled.

| Treatment | Lake | 2005 | 2006 | 2007 |
|---|---|---|---|---|
| Stocked | Birch | 52.2 ± 0.7 (100) | 54.7 ± 0.5 (102) | n/a |
| | Strubel | 48.2 ± 0.7 (58) | 47.6 ± 0.6 (126) | 47.2 ± 0.9 (38) |
| | Yellowhead | 47.7 ± 0.5 (57) | 48.6 ± 0.4 (153) | n/a |
| Unstocked | Dog Leg | 49.2 ± 1.5 (19) | 48.6 ± 0.3 (301) | 54.0 ± 0.5 (100) |
| | Fiesta | 48.1 ± 0.6 (102) | 48.1 ± 0.3 (247) | 50.2 ± 0.4 (184) |
| | Gas Plant | 54.3 ± 1.0 (69) | 53.1 ± 0.3 (220) | 44.7 ± 0.9 (69) |
| | Gun Range | 56.4 ± 0.8 (59) | 50.9 ± 0.3 (171) | n/a |
| | Teal | 48.0 ± 0.5 (107) | 45.7 ± 0.4 (91) | n/a |
| | Picard | 35.3 ± 8.7 (3) | 45.3 ± 1.1 (23) | n/a |

**Table B-9** Mean ± SE (n) total length (mm) by strata for dace in 2006 and 2007. n/a: no fish in that stratum

| Year/Treatment | Lake | Bottom | Midwater[1] | Surface |
|---|---|---|---|---|
| **2006** | | | | |
| Stocked | Ironside | 54.3 ± 0.8 (192) | 49.0 ± 0.9 (6) | 48.5 ± 0.8 (11) |
| | Mitchell | 57.2 ± 0.5 (270) | 54.0 (1) | 54.2 ± 1.1 (56) |
| | Birch | 56.6 ± 1.9 (8) | 57.6 ±0.5 (33) | 53.3 ± 8.4 (3) |
| | Strubel | 55.1 ± 0.6 (210) | 56.1 ± 0.9 (57) | 53.9 ± 0.9 (39) |
| | Yellowhead | 59.0 ± 1.7 (48) | n/a | 48.9 ± 1.4 (14) |
| Unstocked | Dog Leg | 64.4 ± 1.4 (40) | 65.7 ± 2.4 (26) | 65.7 ± 0.8 (137) |
| | Fiesta | 69.9 ± 1.1 (80) | 64.1 ± 1.5 (81) | 55.3 ± 0.7 (149) |
| | Gas Plant | 63.8 ± 0.9 (108) | 61.7 ± 0.9 (93) | 59.6 ± 1.2 (49) |
| **2007** | | | | |
| Stocked | Mitchell | 55.8 ± 0.7 (106) | | 47.5 ± 1.0 (11) |
| | Strubel | 57.6 ± 0.9 (120) | | 63.2 ± 1.2 (40) |
| Unstocked | Fiesta | 61.6 ± 0.8 (198) | | 58.7 ± 0.6 (182) |
| | Gas Plant | 62.2 ± 1.1 (61) | | 60.5 ± 1.0 (88) |

[1]There were no midwater traps used in 2007.

**Table B-10** Mean ± SE (n) total length (mm) by strata for fathead minnow in 2006, 2007. n/a: no fish in that stratum

| Year/Treatment | Lake | Bottom | Midwater[1] | Surface |
|---|---|---|---|---|
| **2006** | | | | |
| Stocked | Birch | 45.2 ± 1.1 (22) | 54.4 ± 2.7 (24) | n/a |
| | Yellowhead | 61.8 ± 0.9 (109) | 57.0 (1) | 55.9 ± 2.7 (14) |
| Unstocked | Dog Leg | 61.1 ± 0.9 (44) | 58.0 ± 1 (4) | 60.4 ± 0.9 (35) |
| | Fiesta | 61.2 ± 1.1 (65) | 60.1 ± 0.8 (140) | 56.9 ± 0.7 (137) |
| | Gas Plant | 58.9 ± 1.0 (92) | 50.6 ± 2.8 (9) | 57.8 ± 3.0 (13) |
| **2007** | | | | |
| Unstocked | Fiesta | 56.3 ± 2.2 (8) | | 55.8 ± 1.3 (23) |

[1]There were no midwater traps used in 2007.

**Table B-11** Mean ±SE (n) total length (mm) by strata for brook stickleback in 2006, 2007. n/a: no fish in that stratum

| Year/Treatment | Lake | Bottom | Midwater[1] | Surface |
|---|---|---|---|---|
| **2006** | | | | |
| Stocked | Birch | 47.8 ± 0.5 (77) | 52.9 ± 0.9 (29) | 51.0 ± 3 (2) |
| | Strubel | 47.9 ± 0.6 (53) | n/a | n/a |
| | Yellowhead | 48.5 ± 1 (15) | 48.0 (1) | 49.0 (1) |
| Unstocked | Dog Leg | 48.9 ± 0.5 (78) | 49.4 ± 0.7 (53) | 46.6 ± 0.4 (12) |
| | Fiesta | 50.5 ± 0.6 (54) | 50.9 ± 0.5 (51) | 49.0 ± 1.1 (17) |
| | Gas Plant | 54.5 ± 0.6 (75) | 54.5 ± 0.6 (76) | 51.2 ± 1.2 (19) |
| **2007** | | | | |
| Stocked | Strubel | 47.2 ± 0.5 (38) | | n/a |
| Unstocked | Fiesta | 50.9 ± 0.6 (55) | | 46.4 ± 1.3 (28) |
| | Gas Plant | 45.5 ± 1.0 (50) | | 42.6 ± 1.7 (19) |

[1]There were no midwater traps used in 2007.

## Appendix C: Littoral invertebrates

**Table C-1** Mean (± SE) number of individuals per sample of Hirudinea in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 0.3 ± 0.2 | 0.5 ± 0.2 | 0.7 ± 0.1 | 0.2 ± 0.1 |
| | Mitchell | n/a | 0.2 ± 0.2 | - | 0.2 ± 0.2 |
| | Birch | n/a | 2.0 ± 0.6 | 0.6 ± 0.3 | 1.0 ± 0.5 |
| | Strubel | n/a | 2.6 ± 0.9 | 0.9 ± 0.3 | 0.9 ± 0.3 |
| | Yellowhead | n/a | 2.0 ± 0.5 | 1.8 ± 0.7 | 0.4 ± 0.2 |
| Unstocked | Dog Leg | n/a | 0.7 ± 0.5 | 1.8 ± 0.6 | 0.7 ± 0.3 |
| | Fiesta | n/a | 1.6 ± 0.5 | 0.3 ± 0.2 | 2.5 ± 0.6 |
| | Gas Plant | 5.4 ± 1.3 | 4.4 ± 1.7 | 0.7 ± 0.4 | 10.2 ± 7.1 |
| | Gun Range | n/a | 6.9 ± 2.5 | 2.9 ± 0.9 | 1.0 ± 0.4 |
| | Teal | n/a | 1.0 ± 0.4 | 0.8 ± 0.3 | 0.4 ± 0.2 |
| | Picard | n/a | 0.6 ± 0.3 | 1.3 ± 0.4 | 0.5 ± 0.2 |

**Table C-2** Mean (± SE) number of individuals per sample of Bivalvia in each sampling period for each lake. n/a: lake not sampled.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 16.5 ± 10.5 | 5.8 ± 1.3 | 28.8 ± 7.0 | 42.7 ± 9.2 |
| | Mitchell | n/a | 4.5 ±1.8 | 5.6 ± 1.6 | 8.4 ± 1.7 |
| | Birch | n/a | 2.5 ± 0.5 | 0.9 ± 0.5 | 1.0 ± 0.4 |
| | Strubel | n/a | 16.6 ± 4.1 | 5.9 ± 2.3 | 27.0 ± 7.1 |
| | Yellowhead | n/a | 10.7 ± 6.7 | 12.3 ± 4.4 | 16.0 ± 2.4 |
| Unstocked | Dog Leg | n/a | 10.0 ± 4.4 | 16.3 ± 7.4 | 8.1 ± 3.1 |
| | Fiesta | n/a | 8.1 ± 3.5 | 21.8 ± 7.4 | 8.6 ± 3.1 |
| | Gas Plant | 21.3 ± 4.7 | 10.4 ± 4.1 | 7.0 ± 1.6 | 8.7 ± 4.1 |
| | Gun Range | n/a | 22.8 ± 8.1 | 8.0 ± 3.3 | 8.2 ± 2.7 |
| | Teal | n/a | 7.9 ± 2.9 | 9.5 ± 3.7 | 12.5 ± 4.6 |
| | Picard | n/a | 2.4 ± 0.7 | 2.2 ± 0.7 | 1.1 ± 0.5 |

**Table C-3** Mean (± SE) number of individuals per sample of Gastropoda in each sampling period for each lake. n/a: lake not sampled.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 23.1 ± 8.9 | 6.1 ± 1.4 | 21.9 ± 6.7 | 34.4 ± 5.7 |
| | Mitchell | n/a | 9.1 ± 3.5 | 16.1 ± 3.0 | 16.4 ± 4.6 |
| | Birch | n/a | 3.2 ± 1.2 | 3.0 ± 1.3 | 6.7 ± 1.2 |
| | Strubel | n/a | 22.8 ± 3.1 | 18.1 ± 3.4 | 38.7 ± 6.3 |
| | Yellowhead | n/a | 5.6 ± 2.0 | 8.3 ± 4.2 | 6.3 ± 3.9 |
| Unstocked | Dog Leg | n/a | 6.6 ± 2.4 | 6.1 ± 1.8 | 49.7 ± 16.6 |
| | Fiesta | n/a | 19.4 ± 5.5 | 33.5 ± 8.2 | 65.7 ± 22.5 |
| | Gas Plant | 16.3 ± 5.1 | 9.5 ± 5.9 | 3.7 ± 1.3 | 13.9 ± 7.4 |
| | Gun Range | n/a | 2.7 ± 0.8 | 1.8 ± 0.8 | 12.2 ± 4.3 |
| | Teal | n/a | 23.9 ± 9.0 | 26.1 ± 4.5 | 10.3 ± 2.2 |
| | Picard | n/a | 0.6 ± 0.2 | 0.5 ± 0.2 | 0.3 ± 0.2 |

**Table C-4** Mean (± SE) number of individuals per sample of Acari in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 3.8 ± 1.4 | 0.4 ± 0.3 | 3.5 ± 1.1 | 4.2 ± 1.3 |
| | Mitchell | n/a | 0.1 ± 0.1 | 0.6 ± 0.3 | 1.3 ± 0.3 |
| | Birch | n/a | 0.4 ± 0.2 | 2.9 ± 0.9 | 6.2 ± 1.9 |
| | Strubel | n/a | 1.1 ± 0.5 | 6.0 ± 2.0 | 4.3 ± 1.5 |
| | Yellowhead | n/a | 2.8 ± 1.5 | 4.4 ± 2.6 | 1.4 ± 0.6 |
| Unstocked | Dog Leg | n/a | 1.0 ± 0.4 | 5.7 ± 1.4 | 7.4 ± 2.8 |
| | Fiesta | n/a | 0.6 ± 0.4 | 3.8 ± 2.0 | 8.1 ± 1.4 |
| | Gas Plant | 6.3 ± 2.0 | 0.6 ± 0.3 | 2.4 ± 1.1 | 5.1 ± 0.8 |
| | Gun Range | n/a | 0.8 ± 0.6 | 4.6 ± 2.1 | 6.3 ± 2.7 |
| | Teal | n/a | 1.9 ± 1.0 | 15.2 ± 5.9 | 3.2 ± 0.7 |
| | Picard | n/a | - | 3.5 ± 1.1 | 4.6 ± 2.0 |

**Table C-5** Mean (± SE) number of individuals per sample of Conchostraca in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 10.0 ± 4.3 | 0.1 ± 0.1 | 1.6 ± 0.9 | 5.8 ± 1.9 |
| | Mitchell | n/a | 0.1 ± 0.1 | 0.2 ± 0.1 | 0.5 ± 0.2 |
| | Birch | n/a | - | 0.1 ± 0.1 | 0.1 ± 0.1 |
| | Strubel | n/a | - | 0.2 ± 0.2 | 0.8 ± 0.5 |
| | Yellowhead | n/a | 0.1 ± 0.1 | 1.1 ± 1.1 | 0.1 ± 0.1 |
| Unstocked | Dog Leg | n/a | - | 0.1 ± 0.1 | 0.5 ± 0.3 |
| | Fiesta | n/a | - | 2.4 ± 1.9 | 3.5 ± 1.3 |
| | Gas Plant | 1.5 ± 0.8 | - | - | 0.8 ± 0.4 |
| | Gun Range | n/a | 0.4 ± 0.4 | 1.4 ± 0.8 | 1.5 ± 0.7 |
| | Teal | n/a | - | 3.2 ± 1.4 | 1.9 ± 0.8 |
| | Picard | n/a | - | - | - |

**Table C-6** Mean (± SE) number of individuals per sample of Cladocera in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 12.2 ± 8.9 | 0.4 ± 0.2 | 0.3 ± 0.2 | 7.1 ± 2.1 |
| | Mitchell | n/a | 0.1 ± 0.1 | - | 1.9 ± 0.7 |
| | Birch | n/a | 0.2 ± 0.2 | 0.4 ± 0.3 | 1.5 ± 0.9 |
| | Strubel | n/a | 0.6 ± 0.3 | 0.5 ± 0.5 | 1.3 ± 0.6 |
| | Yellowhead | n/a | 6.2 ± 4.5 | - | 1.0 ± 0.6 |
| Unstocked | Dog Leg | n/a | 0.6 ± 0.3 | 0.1 ± 0.1 | 14.7 ± 7.0 |
| | Fiesta | n/a | 0.6 ± 0.4 | - | 4.2 ± 2.1 |
| | Gas Plant | 0.5 ± 0.2 | 1.3 ± 0.8 | 0.2 ± 0.2 | 2.2 ± 1.3 |
| | Gun Range | n/a | 0.6 ± 0.2 | 0.1 ± 0.1 | 15.0 ± 13.2 |
| | Teal | n/a | 3.9 ± 2.4 | 0.1 ± 0.1 | 2.2 ± 0.7 |
| | Picard | n/a | - | 0.2 ± 0.1 | 0.3 ± 0.2 |

**Table C-7** Mean (± SE) number of individuals per sample of Copepoda in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 7.3 ± 2.3 | 0.2 ± 0.1 | 2.8 ± 0.8 | 0.5 ± 0.3 |
| | Mitchell | n/a | - | 0.7 ± 0.4 | 0.1 ± 0.1 |
| | Birch | n/a | - | 2.0 ± 1.1 | 0.2 ± 0.1 |
| | Strubel | n/a | 0.1 ± 0.1 | 0.6 ± 0.2 | 0.1 ± 0.1 |
| | Yellowhead | n/a | 0.1 ± 0.1 | 0.6 ± 0.4 | 0.1 ± 0.1 |
| Unstocked | Dog Leg | n/a | 0.5 ± 0.3 | 3.6 ± 1.5 | 3.1 ± 1.6 |
| | Fiesta | n/a | - | 3.5 ± 3.1 | 1.1 ± 0.6 |
| | Gas Plant | 0.3 ± 0.2 | 0.2 ± 0.2 | 0.9 ± 0.6 | 0.6 ± 0.2 |
| | Gun Range | n/a | 0.1 ± 0.1 | 1.3 ± 0.6 | 0.1 ± 0.1 |
| | Teal | n/a | 1.0 ± 0.6 | 4.7 ± 1.4 | 0.5 ± 0.2 |
| | Picard | n/a | - | - | - |

**Table C-8** Mean (± SE) number of individuals per sample of Ostracoda in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 1.2 ± 0.5 | 0.5 ± 0.3 | 3.7 ± 1.4 | 2.4 ± 1.0 |
| | Mitchell | n/a | - | - | - |
| | Birch | n/a | 0.1 ± 0.1 | - | - |
| | Strubel | n/a | 0.5 ± 0.4 | 0.2 ± 0.1 | 0.3 ± 0.2 |
| | Yellowhead | n/a | 0.1 ± 0.1 | 1.0 ± 0.6 | 0.5 ± 0.3 |
| Unstocked | Dog Leg | n/a | 0.2 ± 0.2 | 0.1 ± 0.1 | 0.1 ± 0.1 |
| | Fiesta | n/a | 0.1 ± 0.1 | 1.8 ± 0.6 | 0.4 ± 0.2 |
| | Gas Plant | 0.3 ± 0.2 | 0.7 ± 0.4 | 0.3 ± 0.2 | 0.1 ± 0.1 |
| | Gun Range | n/a | 4.3 ± 1.7 | 0.2 ± 0.1 | 5.2 ± 1.5 |
| | Teal | n/a | - | 1.0 ± 0.8 | 0.2 ± 0.1 |
| | Picard | n/a | - | 0.2 ± 0.1 | - |

**Table C-9** Mean (± SE) number of individuals per sample of Amphipoda in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 16.2 ± 5.5 | 6.9 ± 5.0 | 60.3 ± 25.8 | 20.2 ± 5.1 |
| | Mitchell | n/a | 0.9 ± 0.5 | 0.3 ± 0.2 | 1.9 ± 1.2 |
| | Birch | n/a | 26.5 ± 4.2 | 18.1 ± 5.2 | 51.3 ± 13.9 |
| | Strubel | n/a | 1.9 ± 1.2 | 0.5 ± 0.3 | 1.0 ± 0.6 |
| | Yellowhead | n/a | 50.8 ± 26.0 | 38.6 ± 12.2 | 49.2 ± 7.4 |
| Unstocked | Dog Leg | n/a | 0.4 ± 0.3 | 0.9 ± 0.3 | 8.6 ± 2.3 |
| | Fiesta | n/a | 2.2 ± 1.2 | 4.3 ± 2.1 | 2.5 ± 0.8 |
| | Gas Plant | 22.2 ± 6.3 | 11.8 ± 6.5 | 10.8 ± 4.9 | 27.7 ± 7.4 |
| | Gun Range | n/a | 59.0 ± 18.8 | 32.6 ± 9.7 | 160.3 ± 41.7 |
| | Teal | n/a | 1.6 ± 0.7 | 1.2 ± 0.6 | 1.7 ± 0.6 |
| | Picard | n/a | 29.6 ± 9.7 | 3.3 ± 1.2 | 2.5 ± 0.9 |

**Table C-10** Mean (± SE) number of individuals per sample of Ephemeroptera in each sampling period for each lake. n/a: lake not sampled.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 9.9 ± 2.4 | 2.3 ± 1.3 | 19.6 ± 4.6 | 27.2 ± 8.0 |
| | Mitchell | n/a | 3.0 ± 1.2 | 27.9 ± 4.8 | 44.0 ± 13.6 |
| | Birch | n/a | 0.6 ± 0.3 | 1.2 ± 0.5 | 16.4 ± 9.2 |
| | Strubel | n/a | 11.8 ± 3.4 | 27.5 ± 7.3 | 70.1 ± 18.2 |
| | Yellowhead | n/a | 25.7 ± 12.0 | 9.2 ± 6.1 | 17.0 ± 6.3 |
| Unstocked | Dog Leg | n/a | 0.8 ± 0.4 | 6.6 ± 2.0 | 24.0 ± 9.4 |
| | Fiesta | n/a | 0.1 ± 0.1 | 5.1 ± 2.9 | 15.4 ± 6.1 |
| | Gas Plant | 25.8 ± 5.9 | 2.8 ± 1.6 | 1.9 ± 1.1 | 15.9 ± 5.0 |
| | Gun Range | n/a | 1.2 ± 0.3 | 6.8 ± 5.2 | 34.8 ± 13.3 |
| | Teal | n/a | 4.5 ± 2.2 | 10.9 ± 4.6 | 6.1 ± 1.9 |
| | Picard | n/a | 0.8 ± 0.3 | 0.3 ± 0.2 | 0.9 ± 0.3 |

**Table C-11** Mean (± SE) number of individuals per sample of Odonata in each sampling period for each lake. n/a: lake not sampled.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 29.1 ± 10.4 | 0.6 ± 0.3 | 2.8 ± 1.2 | 12.5 ± 4.0 |
| | Mitchell | n/a | 2.6 ± 1.3 | 4.9 ± 1.1 | 14.9 ± 4.4 |
| | Birch | n/a | 1.0 ± 0.6 | 6.2 ± 4.0 | 3.1 ± 0.7 |
| | Strubel | n/a | 4.9 ± 1.6 | 1.9 ± 0.4 | 3.6 ± 0.5 |
| | Yellowhead | n/a | 0.6 ± 0.3 | 0.8 ± 0.4 | 1.4 ± 0.7 |
| Unstocked | Dog Leg | n/a | 1.1± 0.6 | 0.9 ± 0.3 | 13.8 ± 2.8 |
| | Fiesta | n/a | 0.7 ± 0.3 | 1.5 ± 1.1 | 3.0 ± 1.5 |
| | Gas Plant | 1.8 ± 0.6 | 0.2 ± 0.1 | 0.5 ± 0.2 | 2.1 ± 1.4 |
| | Gun Range | n/a | 0.9 ± 0.4 | 0.3 ± 0.2 | 6.3 ± 1.7 |
| | Teal | n/a | 1.4 ± 0.7 | 2.4 ± 0.9 | 1.1 ± 0.4 |
| | Picard | n/a | 0.2 ± 0.1 | 0.1 ± 0.1 | 0.3 ± 0.2 |

**Table C-12** Mean (± SE) number of individuals per sample of Hemiptera in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 0.1 ± 0.1 | - | - | - |
| | Mitchell | n/a | - | 0.1 ± 0.1 | - |
| | Birch | n/a | 0.1 ± 0.1 | 0.1 ± 0.1 | 0.1 ± 0.1 |
| | Strubel | n/a | - | - | - |
| | Yellowhead | n/a | 0.5 ± 0.3 | 0.4 ± 0.2 | 0.2 ± 0.1 |
| Unstocked | Dog Leg | n/a | 0.3 ± 0.2 | 0.2 ± 0.2 | 0.6 ± 0.3 |
| | Fiesta | n/a | - | - | 0.3 ± 0.2 |
| | Gas Plant | - | 0.2 ± 0.1 | - | 0.3 ± 0.2 |
| | Gun Range | n/a | - | 0.1 ± 0.1 | - |
| | Teal | n/a | 0.2 ± 0.1 | 0.2 ± 0.1 | 0.2 ± 0.1 |
| | Picard | n/a | 0.4 ± 0.3 | 0.1 ± 0.1 | 1.2 ± 0.8 |

**Table C-13** Mean (± SE) number of individuals per sample of Lepidoptera in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | - | - | - | - |
| | Mitchell | n/a | - | - | - |
| | Birch | n/a | - | - | - |
| | Strubel | n/a | - | - | - |
| | Yellowhead | n/a | - | - | - |
| Unstocked | Dog Leg | n/a | - | 0.1 ± 0.1 | 0.2 ± 0.2 |
| | Fiesta | n/a | - | 0.3 ± 0.3 | 0.3 ± 0.3 |
| | Gas Plant | - | 0.1 ± 0.1 | - | - |
| | Gun Range | n/a | - | - | - |
| | Teal | n/a | - | - | - |
| | Picard | n/a | - | - | - |

**Table C-14** Mean (± SE) number of individuals per sample of Trichoptera in each sampling period for each lake. n/a: lake not sampled.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 12.0 ± 3.6 | 0.3 ± 0.3 | 9.1 ± 1.9 | 4.2 ± 1.4 |
| | Mitchell | n/a | 0.6 ± 0.3 | 8.9 ± 1.9 | 9.8 ± 3.5 |
| | Birch | n/a | 2.1 ± 0.7 | 4.1 ± 1.9 | 7.5 ± 2.0 |
| | Strubel | n/a | 1.0 ± 0.3 | 2.5 ± 0.9 | 4.4 ± 0.9 |
| | Yellowhead | n/a | 3.3 ± 1.0 | 5.1 ± 1.2 | 7.5 ± 3.9 |
| Unstocked | Dog Leg | n/a | 1.5 ± 0.4 | 2.2 ± 0.7 | 8.8 ± 3.5 |
| | Fiesta | n/a | 0.6 ± 0.3 | 2.6 ± 1.8 | 2.2 ± 0.7 |
| | Gas Plant | 2.5 ± 0.7 | 0.2 ± 0.1 | 0.9 ± 03 | 1.6 ± 0.3 |
| | Gun Range | n/a | 4.8 ± 1.5 | 18.7 ± 7.5 | 14.4 ± 2.9 |
| | Teal | n/a | 0.9 ± 0.5 | 2.3 ± 1.0 | 1.4 ± 0.4 |
| | Picard | n/a | 0.7 ± 0.3 | 2.9 ± 1.1 | 1.2 ± 0.4 |

**Table C-15** Mean (± SE) number of individuals per sample of Coleoptera in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 0.7 ± 0.3 | 1.1 ± 0.5 | - | 1.5 ± 0.5 |
| | Mitchell | n/a | 1.9 ± 1.8 | 0.1 ± 0.1 | 0.1 ± 0.1 |
| | Birch | n/a | 0.1 ± 0.1 | - | 0.1 ± 0.1 |
| | Strubel | n/a | 0.1 ± 0.1 | - | 0.1 ± 0.1 |
| | Yellowhead | n/a | 0.8 ± 0.4 | 0.1 ± 0.1 | 0.7 ± 0.4 |
| Unstocked | Dog Leg | n/a | 1.2 ± 0.4 | 0.1 ± 0.1 | 0.9 ± 0.3 |
| | Fiesta | n/a | 0.9 ± 0.3 | - | 0.3 ± 0.2 |
| | Gas Plant | 0.4 ± 0.2 | 0.7 ± 0.4 | 0.5 ± 0.3 | 0.4 ± 0.2 |
| | Gun Range | n/a | 0.1 ± 0.1 | - | 0.1 ± 0.1 |
| | Teal | n/a | 0.8 ± 0.4 | 0.2 ± 0.1 | 0.7 ± 0.3 |
| | Picard | n/a | 0.3 ± 0.2 | 0.4 ± 0.3 | 0.3 ± 0.2 |

**Table C-16** Mean (± SE) number of individuals per sample of Nemaotocera in each sampling period for each lake. n/a: lake not sampled.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 81.9 ± 16.2 | 34.4 ± 10.8 | 108.8 ± 25.3 | 112.8 ± 42.2 |
| | Mitchell | n/a | 20.3 ± 5.9 | 133.6 ± 30.9 | 172.4 ± 65.5 |
| | Birch | n/a | 31.9 ± 8.4 | 138.2 ± 57.0 | 75.8 ± 30.9 |
| | Strubel | n/a | 125.2 ± 34.9 | 200.5 ± 33.2 | 88.1 ± 20.2 |
| | Yellowhead | n/a | 51.2 ± 16.1 | 84.9 ± 44.5 | 33.1 ± 10.9 |
| Unstocked | Dog Leg | n/a | 44.9 ± 12.1 | 11.6 ± 26.0 | 58.4 ± 12.1 |
| | Fiesta | n/a | 11.0 ± 3.4 | 58.6 ± 19.8 | 31.8 ± 9.5 |
| | Gas Plant | 128.4 ± 30.5 | 15.4 ± 5.6 | 53.4 ± 13.0 | 110.4 ± 55.4 |
| | Gun Range | n/a | 35.1 ± 12.3 | 55.1 ± 21.4 | 140.4 ± 52.9 |
| | Teal | n/a | 53.3 ± 28.7 | 115.3 ± 22.3 | 23.0 ± 4.9 |
| | Picard | n/a | 22.4 ± 10.0 | 32.7 ± 7.3 | 48.6 ± 16.5 |

**Table C-17** Mean (± SE) number of individuals per sample of Brachycera in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 0.4 ± 0.2 | 0.8 ± 0.3 | 1.3 ± 0.4 | 0.9 ± 0.3 |
| | Mitchell | n/a | 0.7 ± 0.3 | 0.6 ± 0.2 | 0.3 ± 0.2 |
| | Birch | n/a | 0.7 ± 0.3 | 0.6 ± 0.3 | 0.2 ± 0.1 |
| | Strubel | n/a | 0.6 ± 0.3 | 0.9 ± 0.6 | 0.7 ± 0.3 |
| | Yellowhead | n/a | 1.1 ± 0.6 | 0.2 ± 0.1 | 0.5 ± 0.3 |
| Unstocked | Dog Leg | n/a | 0.5 ± 0.2 | 0.5 ± 0.2 | 0.2 ± 0.1 |
| | Fiesta | n/a | 0.6 ± 0.4 | 0.3 ± 0.2 | 0.7 ± 0.3 |
| | Gas Plant | 0.4 ± 0.2 | 0.1 ± 0.1 | 0.2 ± 0.2 | 0.1 ± 0.1 |
| | Gun Range | n/a | 0.1 ± 0.1 | - | - |
| | Teal | n/a | 0.7 ± 0.3 | 0.1 ± 0.1 | 0.1 ± 0.1 |
| | Picard | n/a | 0.1 ± 0.1 | 0.3 ± 0.2 | 0.1 ± 0.1 |

**Table C-18** Mean ± SE (n) length (mm) of *Helobdella stagnalis* (Hirudinea) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 3.8 (1) | 4.5 ± 1.7 (2) | 5.2 ± 0.2 (2) | - |
| | Mitchell | n/a | 2.7 (1) | - | 3.9 ± 0.4 (2) |
| | Birch | n/a | 4.6 ± 0.3 (13) | 5.0 ± 0.5 (4) | 3.8 ± 0.2 (7) |
| | Strubel | n/a | 3.8 ± 0.2 (15) | 6.1 ± 0.4 (3) | 4.4 ± 0.5 (5) |
| | Yellowhead | n/a | 3.9 ± 0.3 (11) | 4.4 ± 0.4 (7) | 2.8 ± 0.4 (2) |
| Unstocked | Dog Leg | n/a | 3.2 ± 0.3 (3) | 4.1 ± 0.3 (8) | 3.4 ± 0.1 (2) |
| | Fiesta | n/a | 3.4 ± 0.5 (5) | 6.3 ± 3.2 (2) | 4.2 ± 0.6 (10) |
| | Gas Plant | 3.8 ±0.2 (32) | 5.8 ± 0.7 (11) | 4.1 ± 0.6 (4) | 4.3 ± 0.3 (27) |
| | Gun Range | n/a | 5.6 ± 0.3 (41) | 6.3 ± 0.4 (19) | 6.8 ± 1.0 (8) |
| | Teal | n/a | 7.9 ± 3.5 (4) | 4.2 ±1.5 (2) | 4.9 ± 0.4 (2) |
| | Picard | n/a | 4.4 ± 0.9 (3) | 7.8 ± 1.0 (8) | 5.7 (1) |

**Table C-19** Mean ± SE (n) length (mm) of *Sphaerium* (Bivaliva) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 1.6 ± 0.08 (53) | 1.9 ± 0.2 (8) | 1.6 ± 0.06 (80) | 1.8 ± 0.07 (117) |
| | Mitchell | n/a | 2.8 ±0.7 (14) | 5.3 ± 0.7 (23) | 6.8 ±0.5 (34) |
| | Birch | n/a | 2.8 ± 0.7 (6) | 1.3 (1) | - |
| | Strubel | n/a | 5.5 ± 0.3 (62) | 5.3 ± 0.5 (26) | 4.9 ± 0.3 (127) |
| | Yellowhead | n/a | 1.9 ± 0.2 (44) | 2.5 ± 0.4 (48) | 3.8 ± 0.5 (52) |
| Unstocked | Dog Leg | n/a | 3.3 ± 0.3 (36) | 3.3 ± 0.3 (38) | 2.8 ±0.2 (29) |
| | Fiesta | n/a | 7.7 ± 0.4 (49) | 7.1 ± 0.3 (118) | 8.4 ± 0.5 (44) |
| | Gas Plant | 6.1 ± 0.7 (39) | 5.4 ± 1.0 (12) | 7.6 ± 0.6 (44) | 7.7 ± 0.4 (66) |
| | Gun Range | n/a | 1.6 ± 0.2 (21) | 2.6 ±1.2 (10) | 2.6 ± 1.0 (10) |
| | Teal | n/a | 6.8 ± 0.9 (16) | 5.1 ±0.6 (50) | 5.2 ± 0.4 (40) |
| | Picard | n/a | 4.4 (1) | 2.3 ± 0.3 (4) | - |

**Table C-20** Mean ± SE (n) length (mm)of *Pisidium* (Bivalvia) in each sampling period for each lake. n/a: lake not sampled.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 2.0 ± 0.1 (55) | 2.4 ± 0.1 (36) | 2.1 ± 0.1 (152) | 2.5 ± 0.1 (173) |
| | Mitchell | n/a | 2.7 ± 0.2 (22) | 3.0 ± 0.2 (17) | 2.8 ± 0.1 (47) |
| | Birch | n/a | 2.7 ± 0.2 (15) | 3.7 (1) | 1.9 ± 0.2 (4) |
| | Strubel | n/a | 2.6 ± 0.1 (53) | 2.5 ± 0.2 (27) | 2.7 ± 0.1 (84) |
| | Yellowhead | n/a | 2.0 ± 0.1 (29) | 2.4 ± 0.1 (48) | 2.4 ± 0.1 (90) |
| Unstocked | Dog Leg | n/a | 2.4 ± 0.1 (35) | 2.2 ± 0.1 (43) | 2.3 ± 0.2 (38) |
| | Fiesta | n/a | 3.0 ± 0.2 (7) | 2.0 ± 0.1 (29) | - |
| | Gas Plant | 2.6 ± 0.1 (116) | 2.6 ± 0.1 (46) | 2.9 ± 0.2 (14) | 2.5 ± 0.1 (6) |
| | Gun Range | n/a | 2.0 ± 0.1 (67) | 1.8 ± 0.1 (47) | 2.1 ± 0.1 (52) |
| | Teal | n/a | 2.9 ± 0.2 (19) | 2.8 ± 0.2 (28) | 3.3 ± 0.1 (46) |
| | Picard | n/a | 2.5 ± 0.3 (17) | 2.5 ± 0.2 (6) | 2.6 ± 0.1 (2) |

**Table C-21** Mean ± SE (n) length (mm) of *Planorbula* (Gastropoda) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 2.2 ± 0.1 (88) | 2.6 ± 0.1 (6) | 2.0 ± 0.1 (100) | 2.2 ± 0.2 (148) |
| | Mitchell | n/a | 2.1 ± 0.2 (21) | 3.0 ± 0.2 (23) | 2.4 ± 0.1 (39) |
| | Birch | n/a | 2.3 ± 1.3 (2) | 2.5 ± 0.2 (10) | 2.0 ± 0.1 (32) |
| | Strubel | n/a | 3.5 ± 0.1 (80) | 3.4 ± 0.1 (90) | 3.3 ± 0.1 (89) |
| | Yellowhead | n/a | 2.4 ± 0.1 (26) | 2.9 ± 0.2 (47) | 2.1 ± 0.2 (23) |
| Unstocked | Dog Leg | n/a | 2.9 ± 0.5 (8) | 3.4 ± 0.4 (13) | 2.8 ± 0.1 (94) |
| | Fiesta | n/a | 3.3 ± 0.3 (8) | 2.6 ± 0.1 (44) | 2.6 ± 0.1 (96) |
| | Gas Plant | 2.6 ± 0.1 (38) | 3.2 ± 0.6 (5) | 2.7 ± 0.2 (12) | 3.4 ± 0.2 (35) |
| | Gun Range | n/a | 3.1 ± 0.4 (7) | 1.9 (1) | 2.0 ± 0.1 (63) |
| | Teal | n/a | 2.7 ± 0.1 (52) | 4.1 ± 1.5 (2) | 3.7 ± 0.2 (34) |
| | Picard | n/a | - | - | - |

**Table C-22** Mean ± SE (n) width (mm) of *Planorbula* (Gastropoda) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 1.0 ± 0.03 (87) | 1.2 ± 0.1 (6) | 0.9 ± 0.02 (100) | 1.1 ± 0.03 (148) |
| | Mitchell | n/a | 0.9 ± 0.08 (21) | 1.3 ± 0.07 (23) | 1.2 ±0.05 (39) |
| | Birch | n/a | 1.2 ± 0.5 (2) | 1.1 ± 0.05 (10) | 0.9 ± 0.06 (32) |
| | Strubel | n/a | 1.5 ± 0.04 (80) | 1.3 ± 0.04 (90) | 1.4 ± 0.04 (88) |
| | Yellowhead | n/a | 1.0 ± 0.06 (26) | 1.2 ± 0.06 (47) | 0.98 ± 0.07 (23) |
| Unstocked | Dog Leg | n/a | 1.3 ± 0.1 (8) | 1.6 ± 0.1 (13) | 1.3 ± 0.05 (94) |
| | Fiesta | n/a | 1.4 ± 0.1 (8) | 1.2 ± 0.08 (44) | 1.1 ± 0.04 (96) |
| | Gas Plant | 1.1 ± 0.04 (38) | 1.5 ± 0.2 (5) | 1.2 ± 0.08 (12) | 1.5 ± 0.1 (35) |
| | Gun Range | n/a | 1.5 ± 0.2 (7) | 0.9 (1) | 0.9 ± 0.03 (63) |
| | Teal | n/a | 1.2 ± 0.05 (52) | 1.7 ± 0.7 (2) | 1.6 ± 0.1 (34) |
| | Picard | n/a | - | - | - |

**Table C-23** Mean ± SE (n) length (mm) of *Hydrozetes* (Acari) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 0.52 ± 0.02 (11) | - | 0.52 ± 0.01 (9) | 0.52 ± 0.02 (23) |
| | Mitchell | n/a | 0.47 (1) | 0.44 ± 0.08 (3) | 0.46 ± 0.03 (10) |
| | Birch | n/a | - | 0.51 ± 0.02 (12) | 0.51 ± 0.01 (44) |
| | Strubel | n/a | 0.49 ± 0.03 (5) | 0.47 ± 0.02 (44) | 0.42 ± 0.02 (31) |
| | Yellowhead | n/a | - | 0.46 ± 0.02 (23) | 0.53 ± 0.02 (5) |
| Unstocked | Dog Leg | n/a | - | 0.47 ± 0.04 (10) | 0.57 ± 0.05 (26) |
| | Fiesta | n/a | - | 0.54 ± 0.02 (5) | 0.56 ± 0.03 (43) |
| | Gas Plant | 0.54 ± 0.02 (13) | - | 0.47 ± 0.06 (3) | 0.52 ± 0.01 (26) |
| | Gun Range | n/a | - | 0.50 ± 0.01 (21) | 0.50 ± 0.01 (21) |
| | Teal | n/a | - | 0.45 ± 0.02 (28) | 0.50 ± 0.02 (13) |
| | Picard | n/a | - | 0.46 ± 0.01 (32) | 0.47 ± 0.01 (42) |

**Table C-24** Mean ± SE (n) length (mm) of *Unionicola* (Acari) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 0.74 ± 0.06 (9) | 0.49 (1) | 0.92 ± 0.04 (7) | 0.71 ± 0.04 (6) |
| | Mitchell | n/a | - | - | - |
| | Birch | n/a | - | 1.1 (1) | 0.85 ± 0.1 (3) |
| | Strubel | n/a | - | 1.0 ± 0.1 (2) | 0.81 ± 0.1 (3) |
| | Yellowhead | n/a | 0.74 ± 0.05 (16) | 0.77 ± 0.07 (10) | 0.68 ± 0.07 (5) |
| Unstocked | Dog Leg | n/a | 1.06 (1) | 1.0 ± 0.09 (6) | 0.81 ± 0.02 (27) |
| | Fiesta | n/a | - | 0.74 ± 0.03 (18) | 0.68 ±0.07 (8) |
| | Gas Plant | 0.91 ± 0.03 (44) | 0.59 (1) | 0.98 ± 0.06 (12) | 1.1 ± 0.1 (18) |
| | Gun Range | n/a | - | 1.0 ± 0.02 (23) | 0.73 ± 0.03 (32) |
| | Teal | n/a | 0.85 ± 0.05 (6) | 0.82 ± 0.02 (63) | 0.77 ± 0.05 (9) |
| | Picard | n/a | - | - | 0.65 (1) |

**Table C-25** Mean ± SE (n) length (mm) of Daphnidae (Cladocera) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 1.7 ± 0.05 (20) | 0.85 ± 0.2 (3) | 0.86 ± 0.2 (3) | 1.5 ± 0.07 (30) |
| | Mitchell | n/a | 2.1 (1) | - | 1.4 ± 0.1 (11) |
| | Birch | n/a | 1.3 ± 0.2 (2) | 1.2 (1) | 1.5 ± 0.2 (4) |
| | Strubel | n/a | 1.1 ± 0.1 (4) | 0.80 ± 0.01 (5) | 1.6 ± 0.3 (2) |
| | Yellowhead | n/a | 1.2 ± 0.1 (5) | - | 1.1 ± 0.1 (6) |
| Unstocked | Dog Leg | n/a | 1.1 ± 0.2 (4) | 1.0 (1) | 1.2 ± 0.04 (78) |
| | Fiesta | n/a | 1.5 ± 0.04 (2) | - | 1.1 ± 0.06 (29) |
| | Gas Plant | 1.3 ± 0.2 (2) | 1.5 ± 0.1 (11) | 0.75 ± 0.01 (2) | 1.5 ± 0.1 (11) |
| | Gun Range | n/a | - | - | 1.2 ± 0.05 (32) |
| | Teal | n/a | 1.4 ± 0.04 | 0.80 (1) | 1.1 ± 0.08 (13) |
| | Picard | n/a | - | - | 0.71 (1) |

**Table C-26** Mean ± SE (n) length (mm) of *Hyalella azteca* (Amphipoda) in each sampling period for each lake. n/a: lake not sampled.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 5.7 ± 0.5 (116) | 4.2 ± 0.1 (41) | 4.9 ± 0.05 (172) | 3.7 ± 0.06 (153) |
| | Mitchell | n/a | 3.4 ± 0.3 (8) | 3.9 ± 1.1 (3) | 2.8 ± 0.1 (19) |
| | Birch | n/a | 4.2 ± 0.07 (199) | 5.3 ± 0.06 (127) | 4.1 ± 0.05 (206) |
| | Strubel | n/a | 3.1 ± 0.2 (19) | 4.2 ± 0.6 (5) | 3.2 ± 0.2 (10) |
| | Yellowhead | n/a | 4.0 ± 0.09 (170) | 4.9 ± 0.06 (144) | 4.1 ± 0.05 (217) |
| Unstocked | Dog Leg | n/a | 2.6 ± 0.9 (2) | 4.9 ± 0.2 (9) | 3.7 ± 0.1 (79) |
| | Fiesta | n/a | 3.1 ± 0.2 (7) | 4.9 ±0.08 (29) | 3.3 ± 0.3 (17) |
| | Gas Plant | 6.1 ± 0.1 (133) | 4.5 ± 0.08 (73) | 5.9 ±0.08 (78) | 4.4 ± 0.08 (154) |
| | Gun Range | n/a | 3.9 ± 0.07 (184) | 5.4 ± 0.05 (171) | 4.4 ± 0.06 (262) |
| | Teal | n/a | 3.1 ± 0.2 (14) | 4.9 ± 0.3 (4) | 3.0 ± 0.3 (8) |
| | Picard | n/a | 3.9 ± 0.08 (174) | 5.3 ± 0.2 (28) | 4.4 ± 0.2 (19) |

**Table C-27** Mean ± SE (n) length (mm) of *Caenis* (Ephemeroptera) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 3.0 ± 0.1 (81) | 2.5 ± 0.2 (18) | 2.8 ± 0.1 (118) | 2.8 ± 0.07 (159) |
| | Mitchell | n/a | 2.3 ± 0.2 (29) | 3.2 ± 0.06 (190) | 2.4 ± 0.04 (204) |
| | Birch | n/a | 2.3 ± 0.3 (6) | 2.1 ± 0.2 (11) | 2.1 ± 0.07 (91) |
| | Strubel | n/a | 2.6 ± 0.1 (97) | 2.7 ± 0.08 (192) | 2.8 ± 0.07 (198) |
| | Yellowhead | n/a | 2.9 ± 0.06 (114) | 3.3 ± 0.2 (44) | 3.2 ± 0.09 (99) |
| Unstocked | Dog Leg | n/a | 2.9 ± 0.7 (5) | 2.0 ± 0.06 (66) | 2.0 ± 0.05 (129) |
| | Fiesta | n/a | - | 2.1 ± 0.06 (50) | 2.9 ± 0.2 (106) |
| | Gas Plant | 3.0 ± 0.1 (161) | 2.5 ± 0.09 (28) | 3.1 ± 0.2 (19) | 2.7 ± 0.08 (112) |
| | Gun Range | n/a | 2.5 ± 0.1 (11) | 2.7 ± 0.2 (34) | 2.6 ± 0.08 (145) |
| | Teal | n/a | 2.8 ± 0.1 (44) | 2.7 ± 0.1 (78) | 2.2 ± 0.09 (53) |
| | Picard | n/a | 4.7 ± 0.4 (8) | 3.7 ± 1.3 (2) | 5.1 ± 0.6 (4) |

**Table C-28** Mean ± SE (n) length (mm) of *Somatochlora* (Odonata) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 3.8 ± 1.0 (7) | - | 7.0 ± 4.3 (2) | 6.9 ± 0.9 (17) |
| | Mitchell | n/a | - | 6.3 ± 0.6 (13) | 6.5 ± 0.4 (28) |
| | Birch | n/a | 7.5 (1) | 8.5 ± 3.9 (5) | 8.5 ± 3.5 (6) |
| | Strubel | n/a | 7.2 (1) | 13.3 ± 2.8 (4) | 8.3 ± 0.8 (5) |
| | Yellowhead | n/a | 3.0 (1) | - | 4.0 (1) |
| Unstocked | Dog Leg | n/a | - | - | 5.4 ± 0.4 (27) |
| | Fiesta | n/a | 10.0 (1) | 2.7 (1) | 8.3 (1) |
| | Gas Plant | - | - | 11.7 ± 0.7 (3) | 10.5 ± 0.7 (4) |
| | Gun Range | n/a | - | - | 6.5 ± 0.5 (26) |
| | Teal | n/a | 10.6 (1) | 3.4 ± 0.4 (3) | 6.9 ± 3.6 (2) |
| | Picard | n/a | - | - | - |

**Table C-29** Mean ± SE (n) length (mm) of *Ischnura/Enallagma/Coenagrion* (Odonata) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 7.2 ± 0.3 (86) | 6.9 ± 3.3 (2) | 12.1 ± 2.4 (4) | 9.4 ± 1.4 (9) |
| | Mitchell | n/a | 10.0 (1) | 10.5 ± 2.1 (6) | 9.4 ± 0.8 (9) |
| | Birch | n/a | - | 13.7 ± 1.9 (19) | 16.9 (1) |
| | Strubel | n/a | 8.7 ± 1.4 (4) | 5.2 ± 1.4 (2) | 3.6 ± 1.5 (2) |
| | Yellowhead | n/a | - | 3.9 (1) | - |
| Unstocked | Dog Leg | n/a | 9.4 ± 1.3 (4) | 8.3 (1) | 9.9 ± 1.1 (18) |
| | Fiesta | n/a | 8.0 (1) | 12.7 ± 3.1 (4) | 3.7 ± 0.1 (2) |
| | Gas Plant | 12.7 ± 1.1 (3) | 7.7 (1) | 20.0 (1) | 11.5 ± 1.5 (9) |
| | Gun Range | n/a | 9.2 ± 0 (2) | - | 5.8 ± 0.2 (2) |
| | Teal | n/a | 14.4 (1) | 13.6 ± 3.5 (3) | - |
| | Picard | n/a | 10.2 (1) | - | - |

**Table C-30** Mean ± SE (n) length (mm) of Limnephilidae (Trichoptera) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 7.1 ± 0.2 (88) | 11.6 ± 3.4 (2) | 4.6 ± 0.3 (20) | 4.3 ± 1.0 (8) |
| | Mitchell | n/a | 3.6 (1) | 4.6 ± 0.1 (35) | 4.3 ± 0.2 (33) |
| | Birch | n/a | 3.9 ± 0.3 (13) | 6.8 ± 2.4 (5) | 3.5 ± 0.3 (26) |
| | Strubel | n/a | - | - | 4.2 ± 0.6 (9) |
| | Yellowhead | n/a | 6.1 ± 2.8 (2) | 5.8 ± 0.7 (13) | 5.4 ± 0.6 (34) |
| Unstocked | Dog Leg | n/a | 9.1 ± 3.3 (5) | 9.3 ± 4.4 (5) | 4.6 ± 0.09 (6) |
| | Fiesta | n/a | - | - | 4.1 ± 1.4 (3) |
| | Gas Plant | 6.0 ± 0.5 (4) | 2.7 (1) | 7.6 ± 2.9 (3) | 4.0 ± 0.1 (2) |
| | Gun Range | n/a | 6.7 ± 0.9 (14) | 5.1 ±0.4 (59) | 6.3 ± 0.3 (33) |
| | Teal | n/a | - | 5.1 ± 1.2 (2) | 6.4 ± 2.6 (4) |
| | Picard | n/a | 5.8 ± 1.0 (4) | 4.6 ± 0.5 (3) | 3.8 ± 0.3 (4) |

**Table C-31** Mean ± SE (n) length (mm) of Phryganeidae (Trichoptera) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 11.2 ± 0.9 (7) | 8.7 (1) | 9.3 ± 0.9 (10) | 7.1 ± 0.6 (10) |
| | Mitchell | n/a | - | 5.4 ± 1.7 (4) | 2.4 ± 0.5 (9) |
| | Birch | n/a | 8.7 ± 1.1 (3) | 12.8 ± 0.4 (2) | 4.6 ± 1.4 (11) |
| | Strubel | n/a | 6.0 ± 1.2 (6) | 9.7 ± 2.7 (5) | 6.5 ± 0.6 (4) |
| | Yellowhead | n/a | 6.5 ± 0.7 (15) | 10.5 ± 2.4 (11) | 6.5 ± 0.5 (18) |
| Unstocked | Dog Leg | n/a | 7.4 ± 1.6 (6) | 12.3 ± 0.5 (11) | 11.3 ± 0.9 (24) |
| | Fiesta | n/a | 5.1 ± 2.5 (3) | 10.0 ± 1.0 (4) | 6.3 (1) |
| | Gas Plant | 12.9 ± 1.9 (3) | 18.0 (1) | 14.7 ± 6.1 | 10.8 ± 1.4 (10) |
| | Gun Range | n/a | 8.0 ± 0.8 (7) | 10.3 ± 2.0 (5) | 9.0 ± 1.3 (7) |
| | Teal | n/a | 4.7 ± 1.9 (7) | 12.1 ± 2.0 (7) | 19.6 ± 1.9 (3) |
| | Picard | n/a | - | 5.5 ± 1.5 (5) | 8.3 ± 0.3 (2) |

**Table C-32** Mean ± SE (n) length (mm) of Chironomidae pupae (Nematocera) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 3.7 ± 0.2 (42) | 3.4 (1) | 7.5 ± 0.4 (18) | 3.9 ±0.4 (5) |
| | Mitchell | n/a | 3.2 ± 0.6 (2) | 5.4 ± 0.1 (47) | 4.2 ±0.5 (6) |
| | Birch | n/a | 2.4 ± 0 (2) | 3.9 ± 0.3 (2) | - |
| | Strubel | n/a | 3.8 ± 0.2 (6) | 4.9 ± 0.5 (14) | 3.2 ± 0.5 (2) |
| | Yellowhead | n/a | 3.0 ± 0.2 (10) | 5.4 ± 1.3 (5) | 3.1 ± 0.9 (2) |
| Unstocked | Dog Leg | n/a | 3.6 ± 0.3 (4) | 6.9 ± 0.4 (2) | 2.8 ± 0.4 (6) |
| | Fiesta | n/a | 4.2 ± 0.5 (2) | 7.1 ± 1.1 (7) | 2.3 ± 0.6 (4) |
| | Gas Plant | 4.0 ± 0.4 (16) | - | 7.9 ±0.2 (39) | 3.4 ± 0.3 (4) |
| | Gun Range | n/a | - | 7.0 ± 0.4 (16) | 6.2 ± 0.7 (16) |
| | Teal | n/a | 2.6 ± 0.4 (7) | 9.3 ± 0.3 (23) | 3.4 ± 0.3 (7) |
| | Picard | n/a | 5.7 (1) | 2.4 ± 0.7 (2) | - |

**Table C-33** Mean ± SE (n) length (mm) of Chironomidae larvae (Nematocera) in each sampling period for each lake. n/a: lake not sampled.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 4.6 ± 0.1 (212) | 5.5 ± 0.2 (167) | 6.1 ± 0.2 (217) | 5.2 ± 0.1 (228) |
| | Mitchell | n/a | 5.3 ± 0.2 (111) | 6.6 ± 0.2 (227) | 6.0 ± 0.2 (261) |
| | Birch | n/a | 4.4 ± 0.2 (164) | 5.0 ± 0.2 (196) | 4.8 ± 0.1 (219) |
| | Strubel | n/a | 5.1 ± 0.1 (244) | 5.4 ± 0.2 (258) | 4.5 ± 0.1 (251) |
| | Yellowhead | n/a | 4.8 ± 0.1 (181) | 5.8 ± 0.2 (139) | 5.4 ± 0.2 (127) |
| Unstocked | Dog Leg | n/a | 4.6 ± 0.2 (176) | 4.5 ± 0.1 (211) | 4.1 ± 0.1 (217) |
| | Fiesta | n/a | 6.2 ± 0.3 (59) | 5.9 ± 0.3 (154) | 4.4 ± 0.2 (141) |
| | Gas Plant | 5.1 ± 0.1 (197) | 4.8 ± 0.2 (85) | 4.9 ± 0.2 (194) | 5.6 ± 0.2 (224) |
| | Gun Range | n/a | 5.8 ± 0.2 (137) | 6.0 ± 0.2 (168) | 5.7 ± 0.2 (216) |
| | Teal | n/a | 3.9 ± 0.2 (86) | 4.6 ± 0.1 (233) | 4.0 ± 0.2 (136) |
| | Picard | n/a | 5.1 ± 0.2 (106) | 5.5 ± 0.2 (166) | 6.5 ± 0.2 (181) |

**Table C-34** Mean ± SE (n) length (mm) of Ceratopogoninae (Nematocera) in each sampling period for each lake. n/a: lake not sampled.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 8.2 ± 0.5 (25) | 10.5 ± 1.8 (6) | 10.1 ± 0.9 (16) | 9.0 ± 0.7 (12) |
| | Mitchell | n/a | 10.7 ± 0.7 (14) | 9.0 ± 0.5 (31) | 10.0 ±0.5 (34) |
| | Birch | n/a | 6.2 ± 1.9 (6) | 10.0 ± 0.7 (17) | 11.1 ± 0.6 (29) |
| | Strubel | n/a | 8.9 ± 0.3 (63) | 8.4 ± 0.5 (54) | 10.0 ± 0.4 (51) |
| | Yellowhead | n/a | 8.6 ± 0.4 (26) | 9.3 ± 0.3 (51) | 7.6 ± 0.4 (38) |
| Unstocked | Dog Leg | n/a | 8.5 ± 0.3 (65) | 9.6 ± 0.3 (110) | 7.1 ± 0.2 (111) |
| | Fiesta | n/a | 10.3 ± 1.0 (12) | 7.6 ± 0.5 (38) | 9.1 ± 0.6 (38) |
| | Gas Plant | 8.2 ± 0.3 (99) | 8.3 ± 0.3 (57) | 8.7 ± 0.5 (54) | 9.6 ± 0.4 (53) |
| | Gun Range | n/a | 8.7 ± 0.5 (44) | 8.7 ±0.5 (48) | 10.2 ± 0.6 (19) |
| | Teal | n/a | 8.1 ± 0.5 (12) | 6.9 ± 0.3 (104) | 10.8 ± 0.6 (9) |
| | Picard | n/a | 10.8 ± 0.6 (13) | 10.0 ±0.5 (34) | 11.0 ± 0.4 (20) |

**Table C-35** Mean ± SE (n) length (mm) of *Chrysops* (Brachycera) in each sampling period for each lake. n/a: lake not sampled; - :Taxa not present.

| Treatment | Lake | May 2005 | August 2005 | May 2006 | August 2006 |
|---|---|---|---|---|---|
| Stocked | Ironside | 10.5 ± 3.3 (4) | 11.7 ± 2.0 (8) | 14.3 ± 1.3 (13) | 13.6 ± 1.5 (8) |
| | Mitchell | n/a | 13.0 ± 2.3 (7) | 15.1 ± 2.6 (5) | 9.6 ± 4.6 (3) |
| | Birch | n/a | 6.2 ± 1.9 (6) | 6.8 ± 0.04 (2) | 12.9 ± 7.1 (2) |
| | Strubel | n/a | 13.8 ± 3.3 (4) | 13.8 ± 1.0 (8) | 6.6 ± 0.6 (7) |
| | Yellowhead | n/a | 14.9 ± 2.0 (8) | 14.1 ± 7.9 (2) | 12.1 ± 3.1 (5) |
| Unstocked | Dog Leg | n/a | 13.5 ± 3.0 (5) | 11.0 ± 1.5 (5) | 5.7 ± 0 (2) |
| | Fiesta | n/a | 15.9 ± 2.9 (5) | 8.2 ± 1.4 (3) | 9.8 ± 3.6 (4) |
| | Gas Plant | 17.4 ± 2.5 (3) | - | 13.9 ± 3.5 (2) | - |
| | Gun Range | n/a | - | - | - |
| | Teal | n/a | 13.0 ± 2.1 (4) | 13.5 (1) | 5.7 (1) |
| | Picard | n/a | 4.4 (1) | 12.5 ± 3.5 (3) | 4.0 (1) |