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Final Report

Study I
Segments 6-10

Biotic Communities in Nebraska Sandhill Lakes

1 March 2003 through 28 February 2009

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COMPLETION REPORT

State: Nebraska

Study Number: I

Project Title: Sandhill Lakes Fisheries Management Project

Project No: F-118-R

Segment Number: 6-10

Period Covered: 1 March 2003 through 28 February 2009

Study Title: Biotic Communities in Nebraska Sandhill Lakes

Objective:

- 1) To determine long-term cohort dynamics of age-0 bluegills and yellow perch in Pelican Lake, Nebraska, and relate the extent of reproduction to environmental conditions (i.e., habitat, biota, climatology).
- 2) To assess adult yellow perch age structure and age-0 cohort dynamics in Marsh Lake, Nebraska.
- 3) To determine under-ice food habits of northern pike in Pelican Lake, Nebraska.
- 4) To assess invertebrate communities in fishless Sandhill lakes, and compare those data to estimates collected from lakes containing fish communities.

Introduction

Previous research indicated that bluegill *Lepomis macrochirus* population size structure in Nebraska Sandhill lakes was related to abundance of predators, especially largemouth bass (Paukert et al. 2002b). However, that work was primarily correlative in nature, and we could not separate the effects of environmental regulation of bluegill recruitment from predator regulation of recruitment. We are uncertain about the reproductive biology, and more specifically the duration and extent of bluegill spawning, in Nebraska Sandhill lakes. The extended spawning season commonly assumed for bluegill (Cargnelli and Gross 1996; Garvey et al. 2002) was

documented in Crane Lake, Indiana, where larval bluegills were collected from early June to early September) (Werner 1969). Beard (1982) reported bluegills spawning season lengths as short as 31 d (four spawning periods) and as long as 112 days (11 spawning periods) in three Wisconsin lakes. Hartmann et al. (1978) reported another half dozen such documentations in various sources of grey literature. Garvey et al. (2002) reported two peaks in bluegill nesting, once in late May and again in mid-June, in Lake Opinicon, Ontario. They reported that inshore densities of larval bluegills peaked once in late May and again in early July. Santucci and Wahl (2003) found that predation was an important mechanism regulation bluegill recruitment in Ridge Lake, Illinois, and that early cohorts produced during protracted bluegill spawning periods were subjected to higher mortality than later-produced cohorts.

Chvala (2000) evaluated the reproductive biology of bluegills in two Nebraska Sandhill lakes. While larvae were first collected in both lakes during June, the spawning season was somewhat extended (7 weeks) at one lake, but quite brief at the other lake. Newly hatched (i.e., 4-6 mm) larvae were collected at Cozad Lake between June 5 and July 24, while newly hatched larvae were only collected from Pelican Lake between June 25 and July 9. Egg-diameter distributions from bluegill ovaries in both lakes had multiple modes, indicating multiple-spawning (i.e., fractional spawning) capabilities. The purpose of this part of the study was to better understand the influence of predation and environment on age-0 bluegill dynamics in Sandhill lakes. While some predatory effects limiting bluegill recruitment have been inferred (Paukert et al. 2002b), questions about the extent of predatory influence (DeBates et al. 2003; Paukert et al. 2003) create the need to further assess environmental influence.

Variable recruitment commonly occurs in percid populations (Koonce et al. 1977; Willemsen 1977; Hackney and Holbrook 1978; Newsome and Aalto 1987) and numerous studies

have documented yellow perch *Perca flavescens* recruitment patterns (e.g., Forney 1971; Koonce et al. 1977; Mills et al. 1989). Researchers commonly incorporate the concept of a “critical period” in describing the recruitment process, a stage characterized by mortality events that greatly influence year-class strength (Marr 1956). Research indicates that this period occurs early in age-0 yellow perch development (Forney 1971; Clady 1976; Anderson et al. 1998). Alternatively, findings reported for walleyes *Sander vitreus* in Oneida Lake (Forney 1976) and for largemouth bass *Micropterus salmoides* in southern reservoirs (Novinger 1988; Sammons and Bettoli 1998) suggest that although larval mortality events may greatly alter cohort size, actual year-class strength is determined later in development.

Describing the type and relative importance of mortality factors functioning during this critical period is difficult (Marr 1956; Braum 1978). Critical periods have been associated with catastrophic declines in larval abundance caused by physiological shifts during development (Hjort 1914); however, Houde (1989) suggested that fluctuations in recruitment may be regulated more by subtle shifts in daily mortality and growth. Several studies have related variability in yellow perch recruitment to environmental variables (Craig et al. 1979; Kallemeyn 1987; Treasurer 1989; Pope et al. 1996) but agreement among researchers as to the influence of these factors is low (Koonce et al. 1977), suggesting that recruitment regulation varies among systems or that regulation is affected by more complex factors. Size-selective overwinter mortality of yellow perch has been reported (Post and Prankevicius 1987; Post and Evans 1989; Johnson and Evans 1991), but the effect of overwinter mortality on perch recruitment has not been well analyzed. Post and Evans (1989) predicted that overwinter starvation mortality of yellow perch could greatly influence year-class strength.

Initial work indicated that Marsh Lake on the Valentine National Wildlife Refuge contained an unexploited and high quality yellow perch population (Paukert and Willis 2000). This population was unique in that growth was very fast (3.5 years to reach 25 cm), size structure was quite high as evidenced by 31% of the trap-net sample being 30 cm or longer in total length, and fish out to age 12 (based on scales) were collected. Thus, this population provides a view of a high quality yellow perch population in the absence of angler exploitation. However, initial samples involved aging of yellow perch from scales, and more precise ages could be obtained by aging sagittal otoliths (Robillard and Marsden 1996; Niewinski and Ferri 1999). An accurate population structure could be assessed to determine environmental influence on year-class strength (Maceina 1997; Maceina and Stimpert 1998). The Marsh Lake fish community also was unique in that it contained a relatively simple fish community, with no predatory fishes except for the small-sized grass pickerel (Paukert and Willis 2000). Evaluation of the reproductive and recruitment processes in Marsh Lake thus would provide information on yellow perch recruitment in the face of limited predation, but in a population that would be considered quite high quality by anglers.

Common carp *Cyprinus carpio* are typically viewed by biologists as a detriment in waters managed for waterfowl production. The direct cause-and-effect relationship between common carp abundance and submerged aquatic plants has not been clearly established, and the concept does remain a point of discussion among fishery and wetland biologists. However, there is a substantial body of evidence indicating that common carp, and perhaps black bullheads *Ameiurus melas*, can reduce the abundance of aquatic plants (Ricker and Gottschalk 1940; Chamberlain 1948; Anderson 1950; Cahoon 1953; Threinen and Helm 1954; Tryon 1954; Robel 1961; King and Hunt 1967; McCrimmon 1968; Crivelli 1983; Fletcher et al. 1985; Kolterman

1990). Some of these studies were observational in nature and simply reported the improvement in aquatic plant abundance after removal of common carp from a water body. Others were experimental in nature, and involved the study of common carp effects on vegetation in enclosures. Together, these studies provide a compelling body of evidence implicating common carp in the reduction of aquatic plant communities, at least at high levels of carp abundance.

The northern pike *Esox lucius* population in Pelican Lake is currently managed under a 71-cm (i.e., 28-inch) maximum length limit, where anglers can harvest only fish less than the length limit, and all larger fish must immediately be released. U.S. Fish and Wildlife Service biologists believe that this regulation has helped to limit common carp recruitment in Pelican Lake (U.S. Fish and Wildlife Service 2001). DeBates (2003) assessed seasonal diets of northern pike in Pelican Lake, and found that the pike only fed on common carp (age-0 individuals) in September, but carp made up 63.6% by weight of the northern pike diet for 71-cm and longer pike (carp were only consumed by 71-cm and longer pike). Paukert et al. (2003) used a bioenergetics model to predict that northern pike likely consumed over 76,000 yellow perch, but only 1,310 common carp, in Pelican Lake during April-September of 2001. Sammons et al. (1994) found that large northern pike consumed juvenile common carp during the winter in Lake Thompson, South Dakota. Thus, DeBates (2003) suggested that northern pike diets be determined during the winter in Pelican Lake to examine the extent of common carp consumption during that time period.

Overly abundant fish planktivore populations can negatively influence size structure and abundance of invertebrates (Mills and Schiavone 1982; Mills et al. 1987). Overly abundant yellow perch reduced size structure of zooplankton and macroinvertebrates in eastern South Dakota lakes compared with lakes with lower density, higher quality yellow perch populations

(Lott 1991; Lott et al. 1996; Lott et al. 1998). In Lake Cochrane, South Dakota, a lake with a high density yellow perch population, the overly abundant perch also cropped the size structure of chironomids (Lott 1991). Gilinsky (1984) found that the effects of bluegill presence on species of Chironomidae were mixed, with some species increasing in abundance and others decreasing. In addition, Crowder and Cooper (1982) found that the presence of bluegills reduced total benthic biomass because the bluegills primarily preyed upon larger macroinvertebrates (i.e., odonates and amphipods). However, Diel (1992) suggested that fish and macroinvertebrates could coexist in vegetated habitats. In his study, Eurasian perch *Perca fluviatilis* foraging efficiency on macroinvertebrates was reduced in vegetation, allowing the invertebrates to remain in relatively high abundance.

Fishes can compete with waterfowl, especially broods, for invertebrates (Swanson and Nelson 1970; Bouffard and Hanson 1997). For example, Hanson and Riggs (1995) found that fathead minnows *Pimephales promelas* reduced abundance and biomass of insects and crustaceans in wetlands to the extent that the suitability of these habitats was reduced as seasonal foraging areas for waterfowl.

Fish predation can also influence other organisms living in palustrine wetlands. For example, some fishes prey upon amphibians (Bouffard and Hanson 1997). Similarly, some predatory fishes, such as northern pike, often are implicated as predators that can reduce abundance of young waterfowl, such as duckling broods. However, little direct evidence of substantial mortality is available. Lagler (1956) assessed food habits of northern pike in a Michigan marsh, and found that waterfowl were only present in stomachs of three of 1,218 northern pike (35 cm and longer) during the waterfowl brooding season. Bouffard and Hanson (1997) found no evidence that waterfowl avoid lakes in response to predation risk by fishes.

There is some evidence that productive Sandhill lakes may not see as much negative effect from fish predation on invertebrates as would less-productive lakes. Swanson and Nelson (1970) reported the following information. Benson (1964) presented evidence that increased productivity could reduce the negative effects on waterfowl. In a study of 30 wetlands, he included two that were fish rearing ponds. One pond was used to rear "bass" and was not fertilized. The other was used to rear walleye and was fertilized regularly. Thirty-three percent of the duck breeding pair use on the entire study area was on the fertilized walleye pond compared to only 9% use on the unfertilized bass pond. Brood use was also greater on the fertilized pond, accounting for about 35% of the use in the entire study area. Bottom samples from the fertilized pond contained an abundant invertebrate population. "There is some evidence, therefore, that some fishery management practices could have a favorable influence on waterfowl."

A recently completed study in Minnesota indicated that walleye fry stockings could suppress fathead minnow populations in wetlands (Ward et al. 2008). The invertebrate communities in walleye-dominated wetlands were more abundant than those in fathead-minnow-dominated wetlands, primarily because fathead minnow annual production can be so high that tremendous amounts of invertebrates are consumed (Duffy 1998; Zimmer et al. 2000, 2001). Thus, biologists in Minnesota are discussing the potential for using walleye stocking programs to improve invertebrate communities in wetlands dominated by fathead minnows.

Paukert and Willis (2003a) summarized abundance estimates for zooplankton and macroinvertebrates in 30 Nebraska Sandhill lakes that contained fish communities. A comparison between the 30-lake summaries and zooplankton and macroinvertebrate

communities in fish-less lakes on the Valentine National Wildlife Refuge should provide an initial understanding of the potential effect of fishes on these productive water bodies.

(1) JOB OBJECTIVE 1:

Determine long-term cohort dynamics of age-0 bluegills and yellow perch in Pelican Lake, Nebraska, and relate the extent of reproduction to environmental conditions (i.e., habitat, biota, climatology).

(2) JOB OBJECTIVE 2:

To assess adult yellow perch age structure and age-0 cohort dynamics in Marsh Lake, Nebraska.

Introduction

Previous research indicated that bluegill *Lepomis macrochirus* population size structure in Nebraska Sandhill lakes was related to abundance of predators, especially largemouth bass (Paukert et al. 2002b). However, that work was primarily correlative in nature, and the effects of environmental regulation of bluegill recruitment from predator regulation of recruitment could not be separated. We are uncertain about the reproductive biology, and more specifically the duration and extent of bluegill spawning, in Nebraska Sandhill lakes.

Although yellow perch recruitment has been correlated to environmental factors (Craig et al. 1979; Kallemeyn 1987; Treasurer 1989; Pope et al. 1996; Ward et al. 2004), agreement regarding the relative importance of these factors is low (Koonce et al. 1977), suggesting that recruitment regulation is variable among systems or is regulated by more complex factors. Size-selective overwinter mortality has been documented for yellow perch (Post and Prankevicius 1987; Post and Evans 1989; Johnson and Evans 1991), but the effect of overwinter mortality on perch recruitment has not been well analyzed. However, Post and Evans (1989) predicted that

overwinter starvation mortality of yellow perch, based on simulation modeling, could greatly influence year-class strength in Southern Ontario.

We investigated recruitment dynamics of bluegill and yellow perch and potential relationships with abiotic and biotic variables in two Sandhills lakes. Bluegill and yellow perch were studied in Pelican Lake, Nebraska over 5 years and yellow perch were also studied in Cameron Lake, Nebraska over two years. Our objectives were to describe and compare several life-stages of bluegill and yellow perch. I indexed the larval, juvenile, spring age 1, and adult populations each year and concurrently assessed abiotic and biotic variables that may affect each life stage. I used otoliths to determine hatch date and daily growth rates for age-0 bluegill and yellow perch. Exploratory analyses were performed to identify potential abiotic and biotic variables that may be related to recruitment.

Study areas

Pelican Lake is a 332-ha, shallow (mean depth = 1.3 m) natural lake in Cherry County of the Sandhills region of north central Nebraska within the Valentine National Wildlife Refuge (McCarragher 1977). Total vegetation coverage (emergent and submergent) ranged from 40 to 57% from 2004 to 2007. The fish community was relatively simple and comprised primarily of bluegill, largemouth bass, yellow perch, northern pike, black bullhead, common carp, and fathead minnow. The lake watershed is mostly mid and tall-grass prairie and limited livestock grazing is allowed. A detailed description of all study lakes are given in Appendix 1.

Cameron Lake was used as a replacement for Marsh Lake because drought conditions had dewatered the lake to a point where fish sampling was not feasible. Cameron Lake is a 39-ha, shallow (mean depth = 1.8 m), a private, natural lake in Rock County of the Sandhills region of north central Nebraska (McCarragher 1977). Submergent and emergent vegetation coverage

was low (< 17% total coverage) in 2004 and 2005. The fish community was simple, primarily composed of yellow perch, green sunfish *Lepomis cyanellus*, black bullhead, common carp, fathead minnow, and golden shiner *Notemigonus crysoleucas*. The lake was closed to angling for over 12 months preceding my study, with little fishing effort for the previous three years (personal communication, R. Lackaff, property owner). However, the lake is used for commercial harvest of yellow perch (TL > ~ 76 mm). The lake watershed is mostly mid and tall-grass prairie and primarily used for livestock grazing (personal communication, R. Lackaff, property owner).

Larval fish

Larval trawling was conducted at Pelican Lake from 2004 to 2008 and at Cameron Lake from 2004 to 2005. Larval yellow perch were obtained from Pelican and Cameron lakes and larval bluegill were obtained from Pelican Lake using a surface trawl with a 0.76-m diameter and 1,000- μ m mesh (bar measure) towed in large, oblong circles. Isermann et al. (2002) found no difference in yellow perch density estimates by comparing a 500 and 1,000- μ m mesh trawl, and the 1,000- μ m size was less likely to become fouled with algae and zooplankton. Trawl duration was 3-5 min at an estimated speed of 1.75 m/sec. Larval sampling occurred approximately every 10 d from late April to early September in Pelican Lake and from late April to mid-June in Cameron Lake. In Pelican Lake, five randomly chosen locations within the lake were trawled on each occasion. Each random location was paired as a nearshore (<100 m from emergent, littoral vegetation) and an offshore (>100 m from emergent, littoral vegetation) site to account for potential migrational patterns (Post and McQueen 1988) for a total of 10 trawling locations per occasion. In Cameron Lake, four randomly chosen locations within the lake were trawled on

each occasion and paired as described above for a total of eight trawling locations per occasion. The number of recently hatched (e.g., TL \leq 13 mm) bluegill and yellow perch larvae in the lakes was indexed using a flowmeter (Ocean Test Equipment, Inc.) in the mouth of the trawl, which allowed determination of water volume towed. All larval samples were preserved in 70% ethanol and returned to laboratory for identification and further analysis.

Larval fishes were identified to genus using identification keys (Auer 1982; Holland-Bartels et al. 1990). All larvae were counted and up to 200 fish per sample were measured (mm TL) from each site. Larvae of TL \leq 13 mm were used to calculate larval abundances; fish of TL $>$ 13 mm were considered juvenile fish. Based on our observations, trawl catches generally declined for fish $>$ 13 mm indicating a likely migration to littoral habitats as well as gear avoidance. Garvey et al. (2002) characterized bluegill larvae as those $<$ 15 mm TL. Bluegills generally begin a migration to littoral habitats between 12 and 20 mm TL (Werner and Hall 1988; Garvey et al. 1998; Santucci and Wahl 2003) and yellow perch begin migrating between 8 and 40 mm TL (summarized by Urho 1996).

Larval fish density of each species was analyzed with a Wilcoxon sum of ranks test to determine if differences in density between inshore and offshore sites existed within a lake on a given date (NPAR1WAY procedure; SAS 9.1 [2002]). If no difference between strata was detected then density was reported for the entire lake on each sampling date. Peak density of larval bluegill and yellow perch was examined for differences within lakes, among years using one-way analysis of variance (ANOVA; SAS 9.1 2002). Post-hoc comparisons were performed using the Student-Newman Keuls multiple range test.

Sagittal otoliths were removed from 30 fish of each species per sampling date when adequate numbers were collected from each lake to obtain estimates of hatching date and daily

growth (Taubert and Coble 1977). Otoliths were wiped clean and mounted on microscope slides with cyanoacrylic cement. Prepared otoliths were viewed under a 400X magnification microscope that projected images to a monitor to aid in the enumeration of daily growth increments. Daily growth increment counts were conducted by two independent readers; if counts were within 5 d the counts were averaged. When disparity was greater than 5 d, a third experienced reader was consulted and the otolith was read in concert until consensus was reached. When all readers did not come to a consensus the otolith was removed from the data set. Taubert and Coble (1977) reported that the first growth increment occurred at swim-up for bluegill; Garvey et al. (2002) confirmed this and reported that swim-up occurred approximately 3 d post hatching. Therefore, hatching date for individual bluegill in our study was calculated by adding 3 d to the growth increment count. Length at time of hatching for yellow perch varies among studies, but can be as short as 4–5 mm (Heidinger and Kayes 1986; Fisher et al. 1998); length at time of swim-up varies from 6 to 10 mm (Whiteside et al. 1985; Powles and Warlen 1988). Powles and Warlen (1988) reported that the first increment formation in yellow perch larvae occurred 1–3 d posthatch; therefore, hatch date for yellow perch was corrected by adding 2 d to the estimates. The corrected age (days) estimate was then subtracted from the collection date to determine hatching date. To estimate the number of fish hatched throughout the spawning season, the abundance estimate (number/100 m³) was multiplied by the hatch date estimates, similar to an age-length key (DeVries and Frie 1996). Finally, average daily growth was calculated by dividing the total length at capture by the number of days since hatch and expressed as mm/d. Mean daily growth of larval bluegill and yellow perch was examined for differences within lakes, among years using one-way analysis of variance (ANOVA; SAS 9.1 [2002]). For bluegill, mean larval daily growth rates were examined for differences between

hatching period and among years with two-way ANOVA. Hatching period was categorized as early (i.e., June) and late (i.e., July and August). Post-hoc comparisons were performed using the Student-Newman Keuls multiple range test. Significant interaction terms were further assessed using one-way ANOVA with a Bonferroni correction applied.

Juvenile fish

Juvenile bluegill and yellow perch were collected from Pelican Lake in August or September 2004-2008 and juvenile yellow perch were collected from Cameron Lake in August using cloverleaf traps in the same years as trawling collections were made. The same cohorts were sampled the following April or May as age-1 fish. Each three-lobed cloverleaf trap was constructed of galvanized 6.4-mm bar mesh, with three 12.7-mm wide openings between lobes to accommodate entrance of small yellow perch (Brown and St. Sauver 2002). Each lobe was 50-cm in diameter with 41-cm height. Juvenile fish were preserved in 90% ethanol and enumerated in the laboratory. All fish were identified and measured (TL in mm), mean total length was calculated for each sampling period, and length-frequency histograms were constructed. Size-selective overwinter mortality was investigated by comparing length-frequency histograms between fall and spring samples. Size-selective overwinter mortality will increase the mean total length and decrease the variance in size of the survivors (in the absence of growth, Ricker 1969; Post and Evans 1989). I used these criteria to investigate the occurrence of size-selective overwinter mortality. Mean total length and coefficient of variation were compared between samples to test the significance of changes in mean size and variation through winter. In addition, the occurrence of size-selective mortality may be difficult to detect if a threshold exists above which mortality rates are constant and a subset of fish in the population in a given year are

larger than that size. To this end, I truncated the length-frequency histograms by removing bluegill greater than 60 mm and yellow perch greater than 90 mm TL, representing the upper tail of the length-frequency distributions but not removing a significant portion of the distribution. Finally, to separate size-selective mortality from potential growth between samples I employed empirical quantile-quantile (QQ) plots (Chambers et al. 1983). Total length at quantiles 1, 5, 10, 25, 50, 75, 90, 95, and 99 were determined for fall and spring samples. The difference between fall and spring total lengths at each quantile was determined and plotted. A plot with a negative slope would be evidence for size-selective overwinter mortality. A larger magnitude of difference between the smallest quantiles would indicate that the smallest fish are underrepresented.

Adult fish

Adult bluegill (i.e., age 2 and older) and yellow perch (i.e., age 1 and older) were collected in late-May or early June and aged using otoliths to track and compare the age-0 abundances and examine if these year classes recruited to the adult population. Age-frequency histograms were constructed and examined for relative year-class strength. Finally, peak larval density was related to fall catches of age-0 fish, spring catches of age-1 fish, and adult collections of age-2 bluegill and age-1 yellow perch using correlation analysis.

Zooplankton

Zooplankton was collected at the time of each trawling sample. Zooplankton was collected during the daytime as two replicates at each site using a 2-m long tube sampler (Rabeni 1996). Samples were filtered through a 65- μ m mesh net and stored in 90% ethanol. Benthic

macroinvertebrates were sampled at the same time and sites using a 231-cm² Ekman grab sampler.

Zooplankton were enumerated and identified to family for cladocerans (i.e. Bosminidae, Chydoridae, and Daphnidae), and as cyclopoid or calanoid copepods, copepod nauplii, ostracods, and rotifers. Other uncommon taxa were also noted. Each sample was diluted with water to a measured volume of 30 mL. Three subsamples were then taken with a 5-mL Hensen-Stempel pipette and placed in a Ward counting wheel. Up to 20 individuals of each category were measured (mm TL) and all individuals were counted. The total number of zooplankton of each taxon in a sample was calculated by dividing the number of organisms counted by the proportion of the sample volume processed. Density was then calculated by dividing the number of zooplankters of each taxon by the volume of the water filtered with the tube sampler. I then used taxon-specific, length-dry weight equations (Dumont et al. 1975; Mason 1977; McCauley and Kalff 1981; Culver et al. 1985; Lynch et al. 1986) to convert length to biomass. Potential differences in mean density within months, among years were assessed with a one-way ANOVA. Post-hoc comparisons were performed using the Student-Newman Keuls multiple range test. Alpha levels were adjusted using the Bonferonni correction to account for year*month interactions.

Benthic macroinvertebrates

Benthic macroinvertebrates were collected at the time of each trawling sample. Two replicate benthic macroinvertebrate samples were collected at each site, strained through a 583- μ m mesh sieve in the field, and stored in 90% ethanol. Replicate samples were collected and processed separately. Macroinvertebrates were identified to order and enumerated using a

dissecting microscope in the laboratory. The most common macroinvertebrate taxa were reported and less common taxa were pooled into an “other” category. Up to 20 individuals of each taxon were measured (mm TL) and all individuals were counted. Density was then calculated by dividing the number of benthic invertebrates of each taxon by the area sampled with the Ekman grab (i.e., 231-cm²). I then used taxon-specific, length-dry weight equations (Eckblad 1971; Dumont and Balvay 1979; Smock 1980; Benke et al. 1999) to convert length to biomass. Biomass was not estimated for oligochaetes due to a lack of reliable length-weight conversions, but their abundance was low. Potential differences in mean density within months, among years were assessed with a one-way ANOVA. Post-hoc comparisons were performed using the Student-Newman Keuls multiple range test. Alpha levels were adjusted using the Bonferroni correction to account for year*month interactions.

Phytoplankton

Phytoplankton biomass was indexed using chlorophyll-*a* estimated from replicate water samples at each site using a 2-m long tube sampler. Water (100 mL) was filtered through filters (Whatman glass fiber filters, 0.45 µm) in the field and extracted in the laboratory following the methods described by Lind (1985). Two samples of unfiltered lake water were collected at each site and frozen for later determination of total phosphorous following the methods of Wetzel and Likens (1991). From the same sampling locations, total alkalinity and pH were measured using a water chemistry analysis kit (Hach Company, Loveland, CO), and total dissolved solids (TDS) were recorded 0.5 m below the surface using an electronic meter (Hach Company, Loveland, CO). Secchi disk transparency was measured at each site during daytime and dissolved oxygen (DO) was measured 0.5 m below the surface using an electronic DO meter (Fisher Scientific,

Pittsburgh, PA). Water temperature was measured hourly with an *in situ* automatic temperature logger (Onset Computer Corporation, Pocasset, MA) and expressed as the mean temperature/24-h period.

Vegetation

Submergent and emergent vegetation coverage was quantified annually during July. Sampling was conducted following the transect methods outlined by Paukert et al. (2002a) and values were expressed as percent coverage of different vegetation types (i.e., emergent or submergent). Total percent vegetation coverage was examined for differences within lakes, among years using a Chi-square multiple comparison technique (PROC GENMOD and MULTTEST procedures, SAS 9.1 2002; Zar 1984).

Exploratory analyses

Exploratory analyses of variables related to different indices of fish abundance and fish growth were performed for bluegill and yellow perch in Pelican Lake. The abundance indices that were examined were peak larval abundance, fall age-0 abundance, spring age-1 abundance, age-2 bluegill abundance, and age-1 yellow perch abundance. The growth indices that were examined were mean daily growth rate of early- and late-hatched bluegill, mean daily growth rate of yellow perch, and mean TL of juvenile fish captured in the fall. Relationships were investigated for these indices with various measures of biotic and abiotic variables (described above). Factor analysis (principal components analyses [PCA] with varimax rotation) was employed to reduce the dimensionality of the data sets because of the large number of independent variables that were measured. The PCA was used to identify meaningful combined

(i.e., created) variables. The PCA was completed using SAS software (SAS 2002). Independent variables and the subsequent principal component (PC) groupings were placed into five categories: benthic macroinvertebrate variables, zooplankton variables, climate variables, habitat variables, and physicochemical variables. Benthic macroinvertebrate and zooplankton variables included measures of mean monthly density of the most dominant taxa. Climate variables included various measures of air temperature, precipitation, wind, and winter severity. Habitat variables included measures of DO, water temperature, and vegetation coverage. Physicochemical variables included measures of Secchi depth, conductivity, total dissolved solids, alkalinity, and pH. Principal components were retained based on a combination of the eigenvalue-one criteria (Kaiser 1960) and the proportion of variance method. Principal components with eigenvalues > 1.0 were retained until the cumulative proportion of variance accounted for by the principal components was greater than or equal to 0.70 (Stevens 2002). Variables that loaded (greater than 40%) on more than one factor were omitted in interpretation (Stevens 2002) because it is unclear which dimension these complex variables describe. The PC scores were then related to the described fish indices using correlation analysis. I accepted an α of 0.10 as a useful target for this exploratory analysis. Because my unit of observation is 1 year, this analysis was hampered by a low number of observations (i.e., 5 years). Therefore, plots of significant correlations were examined for the potential undue influence of data pairs.

(a) Target Date for Achievement:

Sampling and data collection for this objective was completed in September 2008. All data analysis is complete.

(b) Date of Accomplishment:

Activities are proceeding on schedule.

(c) Significant Deviations:

Marsh Lake was replaced as a study site by Cameron Lake, as previously described.

(d) Remarks:

None.

Results

Trawl collections of bluegill larvae in Pelican Lake indicated protracted spawning in all years (Figure 1-1). Larvae appeared in the trawls in mid- to late June in all years. The spawning season was generally 2 months long. Larvae appeared the earliest in 2006; they were first captured on June 8 in Pelican Lake (Appendix 2). Larval bluegills were collected into August in all years.

Larval fish densities did not differ between inshore and offshore strata for bluegill on any date (Wilcoxon two-sample test, $P > 0.05$, Appendix 2) with one exception. Inshore density of larval bluegill was greater on 7 July 2008 than the offshore density (Wilcoxon two-sample test, $P = 0.04$, Appendix 2). Peak bluegill larval density was highly variable among years and was primarily unimodal, with peaks consistently occurring in mid-July (Figure 1-1). Mean peak larval density ranged from 116 to 2,294 larvae/100 m³. In 2004 there were two peaks in larval bluegill density; the second peak in density occurred in mid-August. The highest density was

2,294 larvae/100 m³ in 2008 and lowest density was 116 larvae/100 m³ in 2004 (Figure 1-1).

Results of the one-way ANOVA indicated that density varied by year ($P < 0.05$, $F = 4.89$, $df = 4$).

Larval density was highest in 2008, moderate in 2005, 2006, and 2007, and lowest in 2004 (Figure 1-1).

Trawl collections of yellow perch larvae in Cameron and Pelican lakes indicated a truncated spawning period in all years (Figure 1-2). Larvae appeared in the trawls in late April or early May in all years and lakes. The spawning season was generally 2.5 weeks long. Larvae appeared earliest in Cameron Lake in 2004 (30 April) and latest in Pelican Lake in 2006 (9 May). Larval yellow perch densities did not differ between inshore and offshore strata on any date (Wilcoxon two-sample test, $P > 0.05$, Appendix 3). Peak larval density of yellow perch was highly variable between lakes and among years, and was unimodal; peaks occurred in mid-May but peak date varied by several weeks among years (Figure 1-2). Mean peak larval density ranged from 5 to 1,685 larvae/100 m³. The highest density was 1,685 larvae/100 m³ in Cameron Lake in 2004 and lowest density was 5 larvae/100 m³ in Pelican Lake in 2005 (Figure 1-2), a potential year-class failure or at least a weak year class. Water temperatures in Pelican and Cameron lakes in 2005 rose to nearly 15 C but then dropped to nearly 5 C over a period of 2 weeks in late April (Figure 1-2) which coincided with the yellow perch hatching period (Figure 1-4).

Results of the one-way ANOVA indicated that density varied by year in Cameron Lake ($P < 0.01$, $F = 10.14$, $df = 1$) and in Pelican Lake ($P < 0.01$, $F = 6.41$, $df = 4$). In Cameron Lake, density was higher in 2004 than 2005. In Pelican Lake, density was higher in 2008 than in 2004, 2005, 2006, or 2007 (Figure 1-2).

Hatching Dates

Bluegill hatching generally began in early to mid-June (Figure 1-3) coinciding with sustained water temperatures of 20 C or greater. Hatching duration was the longest in Pelican Lake in 2004 and 2008 (56 and 60 d); in 2004 water temperatures were cooler in early summer and did not rise and remain over 20 C until early July (Figure 1-2). In addition, there were two peaks in larval abundance in 2004 accompanied by a comparatively extended hatching duration (Figure 1-3). Water temperatures dropped by approximately 8 C in late July for an extended time period before rising again, which corresponds with the time period between the two peaks in larval abundance (Figure 1-1). Bluegill began hatching earliest in 2006 when early summer water temperatures were comparatively warmer than the other years.

Yellow perch hatching began in April in all years typically when water temperatures rose consistently above 7 C (Figure 1-4). Yellow perch began hatching in early April in Pelican Lake in 2007, the year with the comparatively warmer spring (Figure 1-1). Yellow perch began hatching in late April/early May in Pelican and Cameron lakes in 2005. Finally, the hatching period was somewhat extended in Cameron Lake in 2004, lasting for 47 d (Figure 1-4). Water temperature data were not available for Cameron Lake in 2004.

Mean average daily growth rate of larval bluegill ranged from 0.16 to 0.50 mm/day over the 4 years of study. Results of the two-way ANOVA of mean daily growth rate indicated that growth varied by year ($P < 0.01$, $F = 23.90$, $df = 4$) and by hatching period ($P < 0.01$, $F = 63.50$, $df = 1$). In addition, the year*hatching period interaction was significant ($P < 0.01$, $F = 23.65$, $df = 4$). The later-hatched bluegill (mean = 0.34 mm/d) grew faster than earlier-hatched bluegill (mean = 0.25 mm/d) with the exception of 2005 (Figure 1-5). For early hatched bluegill, differences were detected in growth rates among years (Bonferonni correction, $\alpha = 0.025$,

$P < 0.01$, $F = 64.48$, $df = 4$, Appendix 4). Specifically, larvae in 2005 grew fastest followed by 2006 larvae, while 2004, 2007, and 2008 larvae did not differ from each other (Figure 1-5). Differences in mean daily growth rates for late hatched larvae were also detected among years (Bonferonni correction, $\alpha = 0.025$, $P < 0.01$, $F = 3.97$, $df = 4$). Specifically, larvae in 2006 grew faster than those in 2007. Larval bluegill growth in 2004, 2005, and 2008 was similar to all other years (Figure 1-5).

Mean average daily growth rate of larval yellow perch ranged from 0.27 to 0.35 mm/d in Cameron Lake and from 0.17 to 0.37 in Pelican Lake. Results of the one-way ANOVA of average daily growth rate indicated that growth varied by year in Cameron Lake ($P < 0.01$, $F = 7.50$, $df = 1$) and in Pelican Lake ($P < 0.01$, $F = 88.40$, $df = 4$, Appendix 4). In Cameron Lake, larval growth was faster in 2005 than 2004 (Figure 1-5). In Pelican Lake, growth was fastest in 2004, moderate in 2005 and 2006, slower in 2007, and slowest in 2008 (Bonferonni correction; $\alpha = 0.01$; Figure 1-5).

Juvenile fish

Collections of juvenile bluegill and yellow perch in cloverleaf traps in Pelican Lake were variable among years in spring and fall (Table 1-1). Insufficient sample sizes precluded comparison of overwinter mortality for some pairs of years. For bluegill and yellow perch in Pelican Lake, I compared fall and spring length-frequency histograms in 2004-2005, 2006-2007, and 2007-2008. Yellow perch samples in Cameron Lake were inadequate in 2005 to examine potential size-selective overwinter mortality.

Mean total length of juvenile bluegill ranged from 29 to 35 mm in fall and ranged from 36 to 38 mm in spring (Table 1-2). All t -tests indicated significant differences in total length

from fall to spring while variability (i.e., coefficient of variation [CV]) in total length was similar or increased from fall to spring. Visual inspection of length-frequency histograms indicate that growth likely occurred between sampling occasions (Figure 1-6). Quantile-quantile (QQ) plots revealed size differential growth overwinter but no size-selective mortality occurred (Figure 1-7). In the 2004/2005 and 2006/2007 winters, the incremental change in length was greater for larger individuals than for smaller individuals and the slopes of the QQ regression plots were significantly different from the 1:1 line ($P < 0.01$). This explains the length-frequency distributions spanning a greater length range in spring compared to the previous fall (Figure 1-6). The slope of the QQ regression plot for the 2007/2008 winter was not different than a 1:1 line ($P = 0.23$) indicating that growth occurred between sampling occasions but the incremental change in length was the same for all sizes and no size-selective overwinter mortality occurred (Figure 1-7).

Mean total length of juvenile yellow perch ranged from 45 to 62 mm in fall and from 63 to 73 mm in spring (Table 1-2). All t -tests indicated significant differences in total length from fall to spring while variability in total length decreased from fall to spring. Visual inspection of length-frequency histograms indicated that growth likely occurred between sampling occasions in 2004/2005 and 2007/2008 (Figure 1-8). Quantile-quantile (QQ) plots revealed that growth occurred overwinter but no size-selective overwinter mortality occurred in 2004/2005 and 2006/2007 (Figure 1-9). The slope of the QQ regression plots were not different than the 1:1 line for 2004/2005 ($P = 0.13$) and 2006/2007 ($P = 0.16$) indicating that growth occurred between sampling occasions but the incremental change in length was the same for all sizes and no size-selective overwinter mortality occurred. The slope of the QQ regression plot for the 2007/2008 winter was different than the 1:1 line ($P < 0.05$) indicating that growth and some size-selective

overwinter mortality occurred between sampling occasions (Figure 1-9). The proportion of smaller fish decreased from fall 2007 to spring 2008 even though growth occurred (Figure 1-8).

Adult fish

Age-frequency histograms of adult bluegill were qualitatively examined revealing relatively consistent recruitment (i.e., no missing year classes) within my study period (Figure 1-10). Histograms were examined for 2006, 2007, and 2008; histograms were not examined in 2005 due to inadequate sample size. Catch of age-1 bluegill was not examined because these fish are likely not of a size to fully recruit to the modified-fyke nets. Thus, I was able to examine potential recruitment of the 2004, 2005, and 2006 year classes. The 2004 year class appears to have recruited in relatively higher numbers and was evident in the population over the three years of collection. The 2005 year class was evident in 2007 and 2008 but the numbers were relatively lower in the 2008 sample. From the 2008 sample it appears that the 2006 year class has also recruited, at least to age 2.

Age-frequency histograms for adult yellow perch revealed recruitment of the 2006, 2005, and 2004 year classes by presence of fish from those years (Figure 1-11). Catches of adult yellow perch were low in all years making interpretation of the age structure challenging. In fact, no yellow perch were collected in 2006 and collections in the other years were <30 fish. Age-1 fish are likely recruited to the modified-fyke nets and it appears that the 2006 year class was relatively strong as these fish were collected in greater numbers in 2007 and 2008. At least some of the bluegill and yellow perch survived to recruit to the adult population in each year studied.

Finally, none of my indices of life stage abundance were related, within or among species, with one exception (Appendix 5). Peak density of larval bluegill was not correlated to juvenile fall catches ($r = -0.11$, $P = 0.87$, $N = 5$), juvenile spring catches ($r = 0.67$, $P = 0.33$, $N = 4$), but was correlated to adult catches of age-2 bluegill ($r = 0.93$, $P = 0.07$, $N = 4$). Peak density of larval yellow perch was not correlated to juvenile fall catches ($r = -0.15$, $P = 0.74$, $N = 7$), juvenile spring catches ($r = -0.04$, $P = 0.95$, $N = 6$) or adult catches of age-1 yellow perch ($r = -0.21$, $P = 0.73$, $N = 5$).

Invertebrates

The zooplankton community of Pelican Lake displayed varying biomass dynamics from 2004 to 2007 (Figure 1-12). Generally, larger daphnids dominated the community early in the season and declined by early to mid-June in all years with the exception of 2005 when they remained abundant throughout the season. Copepods were generally most abundant in mid-summer. *Bosmina* populations were most abundant later in the season, typically coinciding with a decreased daphnid population. Other taxa (i.e., chydorids and copepod nauplii) were less abundant in most years (Figure 1-12). Results of the two-way ANOVA of density (Table 1-3) for the different zooplankton categories by year and month (i.e., May-August) indicated varying densities in every instance as well as significant year*month interactions ($P < 0.0001$). *Post-hoc* one-way analyses of variance were performed for each zooplankton category within each month to examine yearly differences (Bonferroni correction, $\alpha = 0.0125$). Of 24 different statistical examinations, 21 of those had significant differences in density within months, among years (Table 1-3). Individual-year, among-month differences were not examined due to the expected dynamic nature of zooplankton populations. Notably, July *Bosmina* density was over five-fold

higher in 2006 than the other years and July and August cladoceran density was over five-fold higher in 2007 than the other years.

The zooplankton community of Cameron Lake was largely dominated by daphnids, *Chydorus*, copepods, and copepod nauplii in 2004 and 2005 (Figure 1-13). Daphnid abundance declined in early June 2004 but remained abundant throughout my 2005 sampling dates. Results of the two-way ANOVA of density for the different zooplankton categories by year and month (i.e., May-June) indicated varying densities in many instances as well as significant year*month interactions ($P < 0.05$). Rotifer density did not vary among months or years in Cameron Lake ($P > 0.05$, $df=1$). *Post-hoc* one-way analyses of variance were performed for each zooplankton category within each month to examine yearly differences (Bonferroni correction, $\alpha = 0.025$). Of 12 different statistical examinations, eight exhibited significant differences in density within months, among years (Table 1-4). Individual-year, among-month differences were not examined due to the expected dynamic nature of zooplankton populations. Notably, May *Bosmina* density was over 30-fold higher in 2004 than in 2005.

For Pelican Lake, results of the two-way ANOVA for density (n/m^2) of the different macroinvertebrate categories by year and month (i.e., May-August) indicated varying densities of oligochaetes, chironomids, other taxa, and total macroinvertebrates as well as significant year*month interactions ($P < 0.05$). *Post-hoc* one-way analyses of variance were performed for these macroinvertebrate categories within each month to examine yearly differences (Bonferroni correction, $\alpha = 0.01$). Density of ephemeropterans, odonates, amphipods, and gastropods were similar among seasons and years ($P > 0.05$). Of 16 different statistical examinations, 15 of those had significant differences in density within months, among years (Table 1-5). Notably,

chironomids were over twice as abundant in June of 2004 than in any other year. Biomass of the major invertebrate taxa in Pelican Lake and Cameron Lake is summarized in Appendix 6.

For Cameron Lake, results of the two-way ANOVA for density (number/m²) of the different macroinvertebrate categories by year and month (i.e.; May-June) indicated varying densities of oligochaetes, chironomids, other taxa, and total macroinvertebrates as well as significant year*month interactions ($P<0.05$). *Post-hoc* one-way analyses of variance were performed for these macroinvertebrate categories within each month to examine yearly differences (Bonferroni correction, $\alpha = 0.025$). Density of ephemeropterans, amphipods, and hirudineans were similar among seasons and years ($P>0.05$). Of eight different statistical examinations, six of those had significant differences in density within months, among years (Table 1-6). Notably, chironomids and oligochaetes were over twice as abundant in 2004 compared to 2005.

Phytoplankton (as indexed by chlorophyll-*a*) was lower in spring and increased in concentration throughout the season in Pelican Lake, ranging from 1.3 to 253.7 $\mu\text{g/L}$ (Figure 1-14). Results of the two-way ANOVA for chlorophyll-*a* concentration ($\mu\text{g/L}$) by year and month (i.e.; May-June) indicated varying phytoplankton concentrations as well as significant year*month interactions ($P<0.05$). *Post-hoc* one-way analyses of variance were performed within each month to examine yearly differences (Bonferroni correction, $\alpha = 0.01$). Among-year differences were detected within all months. In general, 2006 and 2007 exhibited higher phytoplankton concentrations than 2004, 2005, and 2008 (Figure 1-14). Chlorophyll-*a* in Cameron Lake ranged from 5.3 to 184.4 $\mu\text{g/L}$ in May and June in 2004 and 2005. An increasing trend over time was again observed (Figure 1-15). Results of the two-way ANOVA for chlorophyll-*a* concentration ($\mu\text{g/L}$) by year and month (i.e.; May-June) indicated varying

phytoplankton concentrations as well as significant year*month interactions ($P < 0.05$). *Post-hoc* one-way analyses of variance were performed within each month to examine yearly differences (Bonferroni correction, $\alpha = 0.025$). Yearly differences were detected within both months. Phytoplankton concentration in May was higher during 2005 while the concentration was lower during June in 2005 (Figure 1-13).

Total phosphorous concentrations in Pelican Lake ranged from 18.4 to 24.4 $\mu\text{g/L}$ in May through August from 2004 to 2008 (Figure 1-16). Results of the two-way ANOVA for total phosphorous concentration ($\mu\text{g/L}$) by year and month (i.e.; May-August) indicated no differences in concentration as well as no year*month interactions ($P > 0.05$). Total phosphorous concentrations in Cameron Lake ranged from 18.4 to 28.3 $\mu\text{g/L}$ in May and June in 2004 and 2005 (Figure 1-17). Results of the two-way ANOVA for total phosphorous concentration ($\mu\text{g/L}$) by year and month (i.e.; May-June) indicated varying phosphorous concentrations as well as a significant year*month interaction ($P < 0.05$). *Post-hoc* one-way analyses of variance were performed within each month to examine yearly differences (Bonferroni correction, $\alpha = 0.025$). Total phosphorous concentrations were higher in May and June of 2005 than in 2004 ($P < 0.025$).

Other mean physical and chemical measurements for Pelican Lake and Cameron Lake are summarized in Appendix 7. Secchi depth was variable and ranged from 12.9 to 171.0 cm in Pelican Lake and from 6.4 to 12.5 cm in Cameron Lake. Secchi depth generally decreased throughout each season. Dissolved oxygen ranged from 4.8 to 9.5 mg/L in Pelican Lake and from 7.0 to 11.0 mg/L in Cameron Lake. Conductivity ranged from 342 to 880 $\mu\text{S/cm}$ in Pelican Lake and from 210 to 600 $\mu\text{S/cm}$ in Cameron Lake. Alkalinity ranged from 119.7 to 324.9 mg/L in Pelican Lake and from 85.5 to 171.0 mg/L in Cameron Lake. Finally, pH ranged from 7.5 to 10.0 in Pelican Lake and from 7.5 to 8.5 in Cameron Lake.

Total vegetation coverage ranged from 40 to 57 % in Pelican Lake and from 15.0 to 16.7% in Cameron Lake (Table 1-7). Percent coverage was lower in Pelican Lake in 2004 compared with 2005 through 2008 ($X^2= 32.73, P < 0.01$) and similar in 2004 and 2005 in Cameron Lake ($X^2= 0.52, P > 0.05$).

Principal Components Groupings

Principal component analyses were conducted for 2004-2008 with the exception of the climate variables, which were examined for 2004-2007 due to a complete dataset. For the invertebrate variables, my PCA indicated that the first two principal components explained 73% of the variability (Table 1-8). Invertebrate PC 1 was interpreted primarily as an index of overall invertebrate density in May and June (Table 1-9). Principal component 2 was primarily an index of chironomid and other invertebrate density in May and July.

For the zooplankton variables, PCA indicated that the first two principal components explained 73% of the variability (Table 1-8). Zooplankton PC 1 was interpreted primarily as an index of *Bosmina*, copepod, and cladoceran density in May and June (Table 1-9). Principal component 2 was primarily an index of July cladoceran density and August rotifer density.

For the climate variables, PCA indicated that the first two principal components explained 78% of the variability (Table 1-8). Climate PC 1 was primarily an index of wind speed from March to August (Table 1-9). Principal component 2 was negatively loaded by April air temperature and positively loaded by August air temperature suggesting that these variables are inversely related to an underlying dimension.

For the habitat variables, PCA indicated that the first two principal components explained 88% of the variability (Table 1-8). Habitat PC 1 was an index of dissolved oxygen while PC 2 was loaded by water temperature throughout the summer (Table 1-9).

For the physicochemical variables, PCA indicated that the first two principal components explained 81% of the variability (Table 1-8). Physicochemical PC 1 was primarily an index of Secchi depth and pH throughout the summer months. Principal component 2 was an index of a mixture of conductivity and total dissolved solids measures (Table 1-9). All bivariate correlations are summarized in Appendix 8.

Correlations with bluegill life-stage and growth indices

Age 2 bluegill CPUE was positively correlated with bluegill peak larval density ($r = 0.93$, $P = 0.07$) and negatively correlated with fall CPUE of juvenile yellow perch ($r = -0.99$, $P = 0.01$; Figure 1-18). Mean CPUE of juvenile bluegill in the fall was negatively correlated with climate PC 2 ($r = -0.99$, $P < 0.01$, Figure 1-19) which was loaded by April and August air temperature in an inverse fashion. Mean CPUE of age 2 bluegill was negatively correlated with physicochemical PC 1 in the hatching year ($r = -0.99$, $P = 0.06$, Figure 1-19). The correlation is based on three data pairs because age 2 fish from the 2007 and 2008 year class will not be collected until 2009 and 2010. Peak larval density of bluegill was not significantly correlated to any of the principal components ($P > 0.10$).

Main daily growth of early-hatched bluegill was negatively correlated with zooplankton PC 2 ($r = -0.82$, $P = 0.09$; Figure 1-20). Mean daily growth of late-hatched bluegill was correlated with mean CPUE of age 1 yellow perch ($r = 0.91$, $P = 0.09$; Figure 1-20). Finally,

mean TL of bluegill in fall was positively correlated with mean peak abundance of bluegill larvae ($r = 0.82$, $P = 0.09$; Figure 1-20).

Correlations with yellow perch life-stage and growth indices

Peak larval density of yellow perch was correlated with habitat PC 1 ($r = 0.84$, $P = 0.08$) although one data point may be placing undue influence on this relation (Figure 1-19). Peak larval density was also correlated to zooplankton PC 1 ($r = 0.97$, $P = 0.01$) indicating a positive relationship with zooplankton density in the May and June. No other indices of yellow perch abundance were correlated to any of the principal components ($P > 0.10$).

Mean daily growth of yellow perch was correlated with invertebrate PC 2 ($r = 0.90$, $P = 0.04$; Figure 1-21) and was negatively correlated with zooplankton PC 1 ($r = -0.87$, $P = 0.06$; Figure 1-21). Mean total length of juvenile yellow perch collected in the fall was positively correlated with invertebrate PC 1 ($r = -0.91$, $P = 0.03$; Figure 1-21). Mean daily growth of yellow perch was negatively correlated with spring CPUE of age-1 yellow perch ($r = -0.78$, $P = 0.07$, Figure 1-19). Mean daily growth of yellow perch was negatively correlated with yellow perch peak larval density ($r = -0.92$, $P = 0.03$, Figure 1-21). No other yellow perch growth indices were correlated to indices of abundance or any of the principal components ($P > 0.10$, Appendix 8).

Discussion

A single peak of larval bluegill abundance was observed in all years with one exception. Two peaks in larval abundance were observed in 2004, but overall abundance was low that year. Bluegills are typically assumed to be synchronous colonial spawners that exhibit a protracted

spawning season, although a small percentage (4.5 – 7.0%) will nest solitarily (Gross and MacMillan 1981; Neff et al. 2004). A mid-summer temperature drop may have interrupted the bluegill spawn in 2004 and thus resulted in an apparent bimodal appearance when overall synchrony may not be occurring. Beard (1982) indicated that fluctuations in water temperature provided stimuli for repeated spawning and were thus related to the overall length of the spawning season. Although there are many examples of multiple peaks in larval bluegill abundance or direct observations of multiple spawning bouts, examples of years with a single peak in larval density also are well-documented. The frequency of sampling in my study (i.e., 7-10 d) was likely adequate to identify multiple peaks in abundance and the use of a 1,000- μ m mesh trawl is likely effective at capturing newly hatched fish (Isermann et al. 2002). Elevated turbidity precluded direct observation of bluegill spawning or nesting activity in my study. Beard (1982) reported a range of 4-11 spawning bouts in three Wisconsin lakes. Dominey (1981) reported that bluegill breeding synchrony was greater within colonies than among colonies. He noted that even neighboring colonies may cycle out of phase. Consequently, the ability to detect distinct spawning events may be limited as a result of potential asynchronous colony spawning events in a water body as large as Pelican Lake. In addition, daily age estimates have a margin of error of approximately 7 d. A loss of resolution in the identification of specific hatching days may have occurred. Nonetheless, I contend that potential multiple peaks in hatching were identifiable with our sampling method.

Larval bluegill densities in my study lake differed among years and were relatively low when compared to those reported in other studies. Thus, a plausible hypothesis is that a longer spawning season at lower latitudes may lead to increased larval densities. Beard (1982) reported that longer spawning seasons (mediated by appropriate water temperatures) resulted in a greater

number of individual spawning periods in Wisconsin. I found several instances of larval densities from lower latitudes up to 20-fold higher than our observations (Partridge and DeVries 1999; Kim and DeVries 2000; Santucci and Wahl 2003), but most observations were within two to three times of my estimates (Mayhew 1976; Sammons and Bettoli 2002). In addition, several studies at higher latitudes reported similar or lower densities than we encountered (Bryan 1989; Garvey et al. 2002; Jolley et al. 2009). Generally, bluegill populations at lower latitudes may have the potential to reach relatively high larval densities in some years but may also exhibit lower densities more comparable to our study (Kolar et al. 2003). A suite of factors may interact in complex ways, leading to variable larval densities and subsequent recruitment. As a result, direct comparisons to previous research are challenging. Interacting factors may include abiotic factors such as physical habitat, temperature, and weather (Beard 1982; Pope et al. 1996; Jackson and Noble 2000; Casselman et al. 2002) and biotic factors such as food availability and competition (Partridge and DeVries 1999; Rettig and Mittelbach 2002), predation (Houde 1987; Gray et al. 1998; Santucci and Wahl 2003), and lake productivity (Latta and Merna 1977).

The larval bluegill duration (as a surrogate of spawning season) in my study ranged from 1 to 2 months in Pelican Lake. Jolley et al. (2009) found that latitude and larval bluegill duration were generally negatively correlated. The expected, extended spawning season was observed in Crane Lake, Indiana, where larval bluegills were collected from early June to early September (Werner 1969). Beard (1982) reported bluegill spawning durations from 31 d to 112 d in three Wisconsin lakes. Chvala (2000), who evaluated the reproductive biology of bluegill in two Nebraska Sandhills lakes, found that while larvae were initially collected in both lakes during June, the spawning season was relatively extended in one lake compared with the other. Newly hatched (i.e., 4–6 mm) larvae were collected at Cozad Lake between June 5 and July 24, while

newly hatched larvae were only collected from Pelican Lake between June 25 and July 9. Egg-diameter distributions from bluegill ovaries in both lakes had multiple modes, indicating multiple-spawning (i.e., fractional spawning) capabilities.

An inverse relation between latitude and frequency of bluegill spawning bouts is often assumed but was not observed by Jolley et al. (2009). While three or more spawning bouts are common in more southerly waters, such occurrences are less frequent at northern latitudes, although Jolley et al. (2009) reported evidence of three spawning bouts in one South Dakota impoundment. However, even at this more northerly latitude, the influence of latitude could not be discerned as substantial inter-annual variation in the number of spawning bouts within a water body was found. Thus, geographic location alone certainly does not explain the frequency of bluegill spawning and this topic certainly warrants further investigation.

Growth rates of larval bluegill were variable in my study with later-hatched bluegill growing faster than early-hatched fish in most years. Garvey et al. (2002) reported that later-hatched bluegill grew more rapidly than earlier-hatched fish through September but that growth rates were similar by October. Growth rates of fish typically increase with temperature, given adequate prey, to some optimum (Jones 2002). Increased growth rates of later-hatched fish in my study may simply have been influenced by warmer summer temperatures. Breck (1993) suggested that warmer temperatures may have led to a relatively larger size of age-0 bluegill in Michigan ponds. Although my estimated mean daily growth rates were variable, they fell within the range reported by other researchers (i.e., 0.14-0.70 mm/d; Werner 1969; Beard 1982; Breck 1993; Welker et al. 1994; Cargnelli and Gross 1996; Partridge and DeVries 1999; Santucci and Wahl 2003). In addition, mean total length of juvenile bluegill by fall in Pelican Lake (range 29-35 mm) was within the reported range (23-70 mm TL) for populations in Clear Lake, Iowa

(Ridenhour 1960), Michigan ponds (Latta and Merna 1977) and northern Wisconsin lakes (Beard 1982). In addition, mean TL of bluegill captured in the fall of 2008 in five nearby Sandhills lakes ranged from 31 to 38 mm (J.C. Jolley, unpublished data), slightly higher than those observed in Pelican Lake. Reported mean TL of juvenile bluegill in the fall from Lake Opinicon, Ontario (Garvey et al. 2002) and several lakes in Central Wisconsin (Toneys and Coble 1979) were higher than those observed in my study.

My exploratory correlation analysis indicated that growth of early-hatched bluegills was negatively related to abundance of juvenile yellow perch in the fall. Yellow perch typically hatch 2 months prior to bluegill and thus likely have a competitive size-advantage over bluegill larvae. Early-hatched bluegills often have higher survival than later-hatched bluegills (Beard 1982; Cargnelli and Gross 1996) likely due to the ability to attain larger sizes by fall thereby increasing overwinter survival. In contrast, early-hatched bluegills may be subject to variable environmental factors (e.g., temperature) and elevated predation mortality, thereby reducing survival to the fall (Garvey et al. 2002; Santucci and Wahl 2003). Later-hatched fish may compensate for a shorter growing season with increased growth rates owing to more conducive water temperatures. This may be the situation in Pelican Lake as growth of late-hatched bluegill was related to mean CPUE of juvenile bluegill in the fall and following spring. Faster-growing, late-hatched bluegills may contribute more to year-class strength than their earlier-hatched counterparts. Hatch date estimates of juvenile bluegill collected in the fall are necessary to confirm this contention, and otoliths are difficult to age (i.e., daily increment counts) by this time of year.

Although densities of yellow perch larvae varied considerably (>300%) among years it appears that recruitment of yellow perch was relatively consistent in Pelican and Cameron Lake

over the years examined. Most other Sandhills lakes studied exhibited relatively consistent recruitment with the exception of Marsh Lake. Large annual variation in yellow perch year-class strength is common (Hamley et al. 1983; Henderson 1985) although recruitment patterns may vary among water bodies within a localized region (Lott 1991; Lucchesi 1991; Isermann et al. 2007). The early life stages are commonly reported as the time when year-class strength is formed (Clady 1976; Forney 1971; Mills and Forney 1981). My observed densities of larval yellow perch were generally higher than reported values of density in six South Dakota glacial lakes monitored for over eight years (Jansen 2008). Densities were similar in two years to Long Lake, Michigan (Hoffman et al. 2001) and generally similar to or higher than reported densities from southern Lake Michigan (Dettmers et al. 2005). My observed densities were lower than those reported in two embayments of Lake Ontario (Mason and Brandt 1996) and also lower than estimates of larval Eurasian perch in Scotland (Treasurer 1989).

The hatching period of yellow perch was relatively truncated in all years with the exception of 2004 in Cameron Lake (47 d). This is consistent with previous findings of a narrow spawning period and hatching window for yellow perch (Clady 1976; Weber and Les 1982; Powles and Warlen 1988; Fisher 1996; Isermann and Willis 2008). Yellow perch may also protract their spawning under suitable environmental conditions. Fitzgerald et al. (2001) inferred a spawning period >9 weeks in Lake St. Clair, Michigan and Lake Opinicon, Ontario mediated by a strong El Niño climatic phenomenon and Sandström et al. (1997) documented protracted spawning of Eurasian perch mediated by a heated effluent plume, although these examples appear to be the exception. Isermann and Willis (2008) suggested that factors prior to, during, or immediately following hatching may play a critical role in the recruitment of yellow perch under the constraint of a narrow spawning window. In 2005, yellow perch hatched over a very narrow

time frame in Pelican Lake (4 d) and larvae were only collected in low numbers on one day, likely indicating a relatively weak initial year class. Concurrently, Pelican Lake experienced a drop in water temperature from nearly 17 C to nearly 6 C over a period of 2 weeks in late April. This corresponded to the time period when yellow perch eggs would have been incubating and hatching. It is unclear whether the eggs or newly hatched larvae were negatively affected by this cold front. Jansen (2008) simulated the effect of a cold front on yellow perch eggs and found no decrease in egg survival suggesting that the newly hatched larval stage may be more susceptible than eggs to these extreme weather events. Longhenry (2006) attributed the lack of larval yellow perch abundance in semi-permanent wetlands in South Dakota to a cold-front that decreased the water temperature from 12 C to 8 C in 24 h. In general, a higher abundance of yellow perch swim-up larvae, with fewer abnormalities, are produced during a shorter hatching period when water temperatures gradually increase (Hokanson and Kleiner 1974). Treasurer (1989) reported that newly hatched Eurasian perch larvae were more susceptible to sudden temperature changes than were embryos. Sandhills lakes are shallow and windswept; they thus are susceptible to erratic temperature changes and can warm and cool quickly. We are uncertain why the yellow perch hatching period was more protracted in Cameron Lake in 2004.

The abundance indices of yellow perch life stages were not correlated to the abiotic and biotic variables (i.e., principal components) with the exception of peak larval density and invertebrate PC 2. This PC was loaded by July and August densities of chironomids and hirudineans. If the correlation is not spurious, it is possible that elevated densities of larval yellow perch may consequently depress later season abundance of these invertebrates through predation. Alternatively, larval abundance and invertebrate PC 2 may be related to another, unmeasured variable. For example, due to the voltinism in chironomid populations, density

alone is insufficient to fully characterize the population dynamics of this taxon. There are typically multiple overlapping generations of chironomids present in an ecosystem and identification of specific instar stage is necessary to describe these dynamics (Tokeshi 1995), but beyond the scope of this study. No other correlations were significant, although these analyses were based on a low number of observations ($N \leq 6$). The additional years of study accomplished through Segments 11-15 of this project will allow better determination of the veracity of such correlations.

I did not detect differences in larval bluegill or yellow perch densities between inshore and offshore strata as has been reported by others (Werner 1969; Post and McQueen 1998). Pelican Lake is almost entirely littoral with patchily distributed submergent and emergent vegetation throughout the lake (Paukert 2001). This feature of Sandhills lakes likely precludes larvae from becoming spatially stratified within the lake.

Growth rates of larval yellow perch were variable in my study. First-year growth seems to be quite variable both spatially and temporally (e.g., Post and McQueen 1994). Although previously reported data on daily growth rates of yellow perch are scarce, Whiteside et al. (1985) reported rates of approximately 0.57 – 0.66 mm/d in Lake Itasca, Minnesota and Fisher and Willis (1997) reported daily growth rates of 0.45 – 0.65 mm/d in two eastern South Dakota lakes. Mean total lengths of juvenile yellow perch in my fall samples (range 45-62 mm) may have been slightly smaller than those reported elsewhere. Some other ranges reported were 71-89 mm in Clear Lake, Iowa (Ridenhour 1960), 78-90 mm in Lake Winnebago, Wisconsin (Weber and Les 1982), 61-70 mm in Ontario (Post and Evans 1989), and 60-90 mm in Lake Oneida, New York (Irwin et al. 2009). In addition, mean total length of yellow perch captured in the fall in four nearby Sandhills lakes ranged from 60 to 83 mm (J.C. Jolley, unpublished data). Growth of

poikilothermic animals is largely mediated by temperature, but other factors may override or mask the effects of temperature on growth. Some researchers did not find thermal effects on age-0 yellow perch growth and recruitment (Weber and Les 1982; Henderson 1985; Post and McQueen 1994) while others have found the converse (Kallemeyn 1987; Ward et al. 2004). Various measures of the thermal regime may correlate to year-class strength only in certain instances (Koonce et al. 1977; Henderson and Nepszy 1988). Post and McQueen (1994) attributed the majority of the variation in growth of age-0 yellow perch to the availability of benthic and planktonic prey. Some of my observations suggest that temperature may be important to the early life-stages of yellow perch although correlational evidence was not found. Thermal effects may only become evident under extreme conditions in localized areas (Eshenroder 1977; Henderson 1985; Newsome and Aalto 1987). Other factors such as predation rates and availability of prey may override temperature effects (Forney 1971; Crowder 1980; Mason and Brandt 1996). In addition, small changes in a single factor affecting mortality may consequently affect recruitment (Houde 1987).

My estimates of larval density for both species were a poor predictor of later season catches or recruitment to the adult population although supportive evidence was stronger for bluegill than yellow perch. Abundance of larval bluegill may not be related to juvenile survival (and resulting recruitment) in lakes that contain piscivore populations (Cargnelli and Gross 1996; Partridge and DeVries 1999) suggesting that events occurring after the larval life-stage may determine year-class strength. Previous researchers have found indices of larval abundance useful in predicting later life stage abundance. Anderson et al. (1998) reported that larval yellow perch abundance was positively correlated with fall juvenile abundance and Fisher and Willis (1997) reported that August juvenile abundance was positively correlated with larval abundance

in south Dakota lakes, although continued research indicated a weaker relation (Isermann 2003; Jansen 2008). Forney (1971) reported that high trawl catches of age-0 yellow perch corresponded to subsequent dominant year classes in Lake Oneida. Sammons and Bettoli (1998) also reported that larval abundances of crappie and white bass *Morone chrysops* were accurate predictors of later year-class strength. The lack of correlation between larval abundance and later life-stage abundance in my study should be interpreted cautiously as analyses were based on a low number of observations. Significant events may be occurring after the larval stage in my study lakes that directly contribute to year-class formation.

I found little evidence of size-selective overwinter mortality for bluegill or yellow perch in Pelican Lake. Differences in mean total length of bluegill and yellow perch from fall to spring indicate that some growth likely occurred between sampling occasions, which has also been reported by Santucci and Wahl (2003) for bluegill. In one year in Pelican Lake, larger bluegill may have grown more than smaller bluegill over the winter time period. Toney and Coble (1979) only documented size-selective overwinter mortality of bluegill in one out of three lakes and several hatchery ponds. Santucci and Wahl (2003) found little evidence for size-selective overwinter mortality of bluegill although they documented high overall mortality through the first winter. Size-selective mortality of yellow perch may have occurred in the winter of 2007/2008. Overwinter size-selective mortality has been documented experimentally for yellow perch (Johnson and Evans 1989) and in two lakes in Ontario (Post and Prankevicius 1987; Post and Evans 1989). Size-selective overwinter mortality may not be ubiquitous to bluegill and yellow perch in Nebraska Sandhills lakes. Relatively severe winters did not occur over the course of my study.

Vegetation coverage in Pelican Lake was higher (range 40-57%) than that previously reported (i.e., 36%, Paukert and Willis 2000). Although higher vegetation coverage in lakes may lead to higher panfish abundance (Lucchesi 1991; Hinch and Collins 1993) and relatively lower numbers of larger fish (Colle et al. 1987), Paukert et al. (2002b) reported that vegetation coverage was not linked to panfish quality in Nebraska Sandhills lakes. Schneider (1999) also suggested that increased submergent vegetation may not be detrimental to Michigan bluegill populations when harvest is low (1-3% exploitation) and there is an adequate food supply, which may be the situation in Nebraska Sandhills lakes (Paukert 2001).

My study provides some of the first system-wide information on the dynamics of age-0 bluegill and yellow perch. The baseline environmental dataset paired with the abundance and growth indices of the different life stages will continue to be assembled (n = 10 years). As more data are available, the effects of specific abiotic and biotic variables on recruitment may be identified. My exploratory analyses indicated that temperature, winter severity, and growth rates may affect survival and ultimate recruitment to the adult population.

Table 1-1. Mean catch per unit effort (CPUE; number per trap night) for bluegill and yellow perch sampled with cloverleaf traps in Pelican and Cameron lakes, 2004-2008. Values in parentheses are SE, N.

Lake	Species	Year	Season	Mean CPUE
Pelican	Bluegill	2004	Fall	52 (21, 6)
		2005	Spring	11 (1, 60)
			Fall	44 (11, 16)
		2006	Spring	0 (0, 65)
			Fall	83 (16, 26)
		2007	Spring	35 (7, 26)
			Fall	9 (2, 26)
	2008	Spring	0.5 (0.1, 99)	
		Fall	6 (1, 12)	
	Yellow perch	2004	Fall	26 (5, 23)
		2005	Spring	1 (0.2, 60)
			Fall	3 (1, 16)
		2006	Spring	0 (0, 65)
			Fall	17 (3, 26)
2007		Spring	17 (3, 26)	
		Fall	25 (5, 26)	
2008		Spring	39 (15, 99)	
	Fall	9 (3, 12)		
Cameron	Yellow perch	2004	Fall	4 (1, 14)
		2005	Spring	0 (0, 12)
			Fall	0 (0, 16)
		2006	Spring	3 (0.5, 36)

Table 1-2. Mean total length with associated standard errors and N in parentheses, for bluegill and yellow perch sampled with cloverleaf traps in Pelican and Cameron Lakes, 2004-2008. Results of fall to spring comparisons of mean TL are given by *t* statistic, degrees of freedom (*df*), and *P* value.

Lake	Species	Year	Season	Mean TL	CV	<i>t</i>	<i>df</i>	<i>P</i>
Pelican	Bluegill	2004	Fall	29 (0.1, 508)	11	-32.6	1,159	<0.0001
		2005	Spring	36 (0.2, 668)	13			
		2006	Fall	31 (0.1, 705)	11	-	-	-
			Spring	-	-	-	-	-
		2007	Fall	35 (0.1, 1983)	15	-16.7	1,509	<0.0001
			Spring	38 (0.2, 857)	15			
	2008	Fall	32 (0.3, 184)	12	-4.4	206	<0.0001	
		Spring	36 (0.9, 24)	13				
		Fall	33 (1.3, 77)	37				
	Yellow perch	2004	Fall	45 (0.2, 564)	11	-43.6	134	<0.0001
		2005	Spring	73 (0.6, 108)	9			
		2006	Fall	49 (0.2, 51)	3	-	-	-
			Spring	-	-	-	-	-
		2007	Fall	62 (0.4, 433)	12	-2.1	862	0.04
Spring			63 (0.3, 431)	11				
2008		Fall	61 (0.2, 555)	9	-24.6	1,999	<0.0001	
		Spring	67 (0.1, 1446)	8				
	Fall	59 (0.6, 104)	9					
Cameron	Yellow perch	2004	Fall	54 (0.4, 104)	8	-	-	-
		2005	Spring	-	-	-	-	-
			Fall	-	-	-	-	-
		2006	Spring	86 (1, 12)	4	-	-	-

Table 1-3. Mean density (number/L) and associated standard errors of zooplankton taxa by month in Pelican Lake, Nebraska, 2004-2008. Mean density in columns followed by different letters were significantly different (within months, among years; $P < 0.05$); those without letters were not different.

Year	Month	<i>Bosmina</i>	Copepoda	Cladocera	Copepod nauplii	Rotifera	Total
2004	May	1.2 ^b	22.1 ^b	64.1 ^b	6.5 ^a	12.9	116.1 ^b
		(0.1)	(4.7)	(11.7)	(1.7)	(10.5)	(16.6)
2005		60.4 ^b	16.3 ^b	256.8 ^b	24.4 ^{bc}	12.8	384.4 ^b
		(10.0)	(3.2)	(28.3)	(5.1)	(3.9)	(44.3)
2006		1.1 ^b	20.1 ^b	42.2 ^b	37.0 ^{ab}	2.9	109.6 ^b
		(0.1)	(2.4)	(4.8)	(5.4)	(0.5)	(9.2)
2007		107.3 ^b	15.7 ^b	88.5 ^b	17.1 ^a	2.2	240.6 ^b
		(14.1)	(2.0)	(7.6)	(2.4)	(0.3)	(21.6)
2008		640.8 ^a	39.3 ^a	1226.7 ^a	47.9 ^a	12.6	1332.2 ^a
		(103.7)	(3.4)	(151.5)	(5.5)	(2.6)	(154.3)
2004	June	1.1 ^b	21.8	4.7 ^c	19.7 ^a	235.0 ^a	306.0 ^b
		(0.1)	(4.6)	(0.8)	(3.8)	(64.7)	(64.9)
2005		3.3 ^b	14.1	57.4 ^{bc}	8.4 ^b	4.2 ^b	97.7 ^c
		(0.6)	(1.3)	(6.9)	(1.1)	(0.8)	(7.8)
2006		26 ^b	24.7	9.6 ^c	21.8 ^a	184.7 ^a	288.6 ^b
		(6.3)	(4.1)	(1.4)	(3.0)	(27.4)	(31.6)
2007		181.5 ^a	22.9	157.8 ^b	26.7 ^a	19.0 ^b	423.9 ^{ab}
		(31.2)	(2.7)	(25.1)	(3.3)	(5.5)	(61.4)
2008		200.0 ^a	26.5	463.6 ^a	25.6 ^a	42.0 ^b	565.2 ^a
		(38.4)	(3.5)	(67.5)	(3.2)	(11.9)	(68.9)
2004	July	278.9 ^{bc}	43.8 ^a	60.1 ^b	28.8 ^c	150.6 ^a	593.4 ^{dc}
		(65.9)	(5.7)	(11.5)	(4.2)	(19.5)	(87.9)
2005		18.1 ^c	13.9 ^c	94.6 ^b	10.8 ^c	1.3 ^b	140.0 ^d
		(2.6)	(1.8)	(7.0)	(2.6)	(0.1)	(9.2)
2006		1,907.5 ^a	46.8 ^a	255.9 ^b	83.9 ^b	204.7 ^a	2,548.4 ^a
		(316.4)	(7.5)	(50.2)	(10.1)	(45.3)	(365.8)
2007		353.2 ^{bc}	50.7 ^b	1,363.1 ^a	67.8 ^b	43.6 ^b	1,901.1 ^{ab}
		(37.7)	(6.5)	(274.3)	(7.3)	(10.2)	(294.0)
2008		705.5 ^b	77.0 ^a	907.5 ^a	127.8 ^a	128.3 ^a	1256.4 ^{bc}
		(131.6)	(7.6)	(158.2)	(10.4)	(20.0)	(168.3)

Table 1-3 continued.

Year	Month	<i>Bosmina</i>	Copepoda	Cladocera	Copepod nauplii	Rotifera	Total
2004	August	753.0 ^a (100.2)	48.6 ^a (3.5)	175.7 ^c (14.1)	56.8 ^a (4.0)	44.0 ^b (8.7)	1,101.3 ^{bc} (102.2)
2005		92.6 ^b (8.4)	11.7 ^b (1.2)	180.4 ^c (16.8)	7.6 ^c (1.2)	1.0 ^b (0.0)	292.2 ^d (20.4)
2006		687.4 ^a (140.8)	33.5 ^a (5.0)	165.7 ^c (16.3)	25.9 ^{bc} (5.3)	4.5 ^b (1.3)	942.8 ^c (143.4)
2007		404 ^{ab} (64.6)	39.6 ^a (7.4)	1,015.1 ^b (70.0)	38.9 ^{ab} (5.2)	144.5 ^a (38.6)	1,658.0 ^a (113.0)
2008		512.5 ^a (82.4)	38.9 ^a (3.8)	1293.8 ^a (142.1)	57.0 ^a (10.1)	50.0 ^b (8.8)	1447.8 ^{ab} (149.1)

Table 1-4. Mean density (number/L) and associated standard errors of zooplankton taxa by month in Cameron Lake, Nebraska, 2004-2005. Mean density in columns followed by the same letter were significantly different (within months, among years; $P < 0.05$); those without letters were not different.

Year	Month	<i>Bosmina</i>	Copepoda	Cladocera	Copepod nauplii	Rotifera	Total
2004	May	151.7 ^a (13.0)	37.3 ^b (4.6)	1,155.5 ^a (114.3)	22.9 ^b (4.7)	69.4 (18.0)	1,440.3 ^a (139.3)
2005		4.9 ^b (1.2)	75.1 ^a (11.4)	550.8 ^b (58.9)	57.6 ^a (6.3)	61.1 (12.3)	758.8 ^b (78.5)
2004	June	14.1 (2.6)	20.4 ^b (4.1)	233.0 ^b (35.3)	16.3 (1.8)	45.7 (5.6)	330.9 ^b (40.2)
2005		19.2 (3.0)	78.3 ^a (7.7)	497.9 ^a (51.1)	16.7 (3.0)	51.1 (10.2)	666.1 ^a (71.5)

Table 1-5. Mean density (number/m²) and associated standard errors of macroinvertebrate taxa by month in Pelican Lake, Nebraska, 2004-2008. Mean density in columns followed by the same letter were significantly different (within months, among years; $P < 0.05$); those without letters were not different.

Year	Month	Order Ephemeroptera	Order Odonata	Order Amphipoda	Class Hirudinea	Class Gastropoda	Class Oligochaeta	Family Chironomidae	Other	Total
2004	May	8.3	2.1	26.9	28.0	0.0	1,012.5a	671.9ab	228.8b	1963.9
		(4.4)	(1.4)	(22.8)	(14.9)	(0.0)	(188.2)	(72.6)	(61.5)	(265.0)
2005		3.6	2.2	60.1	72.5	31.9	134.8b	949.9a	27.5b	1274.0
		(2.4)	(1.6)	(20.0)	(24.4)	(20.8)	(30.9)	(149.5)	(9.7)	(152.3)
2006		8.0	2.2	47.8	39.1	2.2	515.2b	1107.3a	23.2b	1742.8
		(4.6)	(1.6)	(23.8)	(9.5)	(1.2)	(114.6)	(149.1)	(7.7)	(223.3)
2007		2.2	0.0	5.1	32.6	2.9	258b	910.2a	68.1b	1271.1
		(1.2)	(0.0)	(2.3)	(12.3)	(2.7)	(64.8)	(148.3)	(18.2)	(184.4)
2008		0.7	2.2	2.2	5.2	0.0	376.7b	244.0b	469.7a	1093.0
		(0.7)	(1.7)	(1.7)	(2.6)	(0.0)	(186.1)	(53.7)	(121.4)	(248.9)
2004	June	5.0	0.8	92.9	77.5ab	3.9	750.2a	1,582.6a	256.4a	2,710.5a
		(2.8)	(0.8)	(62.7)	(14.9)	(2.0)	(171.9)	(308.8)	(68.9)	(339.3)
2005		4.0	3.4	44.6	87.5ab	4.0	97.3b	513.8b	6.9b	759.8b
		(2.5)	(2.1)	(13.8)	(25.5)	(2.4)	(22.7)	(111.2)	(2.3)	(118.4)
2006		2.9	2.2	135.5	76.8ab	3.6	79.0b	701.5b	46.4b	1,043.5b
		(1.4)	(1.2)	(37.8)	(20.5)	(2.1)	(20.3)	(132.7)	(9.8)	(174.1)
2007		10.1	0.0	45.7	120.3a	1.4	166.7b	697.9b	63.0b	1,092.8b
		(6.7)	(0.0)	(30.0)	(28.6)	(1.0)	(60.2)	(141.0)	(12.4)	(171.4)
2008		0.7	0.0	0.0	1.5b	0.0	221.8b	368.9b	158.7ab	734.1b
		(0.7)	(0.0)	(0.0)	(1.5)	(0.0)	(78.5)	(81.7)	(73.9)	(176.4)
2004	July	13.0	0.0	368.1	166.7a	3.6	226.1	1,191.4a	52.2	2,012.4a
		(11.0)	(0.0)	(284.5)	(44.4)	(1.8)	(62.1)	(195.5)	(20.8)	(417.8)
2005		4.3	0.0	54.4	46.7ab	0.0	141.3	1,410.9a	2.2	1,659.9ab
		(3.0)	(0.0)	(32.6)	(16.0)	(0.0)	(28.5)	(235.5)	(2.2)	(261.2)
2006		0.0	1.1	30.4	109.8ab	3.3	81.5	280.5b	21.7	529.4b
		(0.0)	(1.1)	(24.2)	(41.6)	(1.8)	(21.0)	(56.1)	(8.3)	(90.5)
2007		0.7	2.9	5.8	34.1ab	0.7	210.2	360.9b	73.2	680.5b
		(0.7)	(2.9)	(5.1)	(9.2)	(0.7)	(61.4)	(52.4)	(14.2)	(104.9)
2008		0.0	0.0	3.6	19.6b	0.0	176.1	608.0b	197.8	997.9ab
		(0.0)	(0.0)	(3.6)	(18.8)	(0.0)	(46.6)	(130.3)	(87.7)	(165.3)

Table 1-5 continued.

Year	Month	Order Ephemeroptera	Order Odonata	Order Amphipoda	Class Hirudinea	Class Gastropoda	Class Oligochaeta	Family Chironomidae	Other	Total
2004	August	23.3 (22.5)	6.2 (4.0)	15.5 (9.7)	82.3ab (28.9)	6.2 (4.2)	168.5 (22.2)	485.3b (75.8)	70.7b (25.6)	844.8ab (134.5)
2005		1.4 (1.0)	5.1 (5.1)	165.2 (151.1)	113.8a (25.8)	1.4 (1.0)	108.0 (20.9)	953.7a (163.2)	29.7b (7.8)	1,365.3a (227.4)
2006		1.1 (1.1)	0.0 (0.0)	10.9 (10.9)	46.7ab (12.0)	1.1 (1.1)	80.4 (19.7)	331.5b (76.2)	18.5b (6.6)	490.2b (85.1)
2007		0.0 (0.0)	8.7 (7.6)	0.0 (0.0)	30.4ab (17.5)	0.0 (0.0)	130.4 (27.4)	168.5b (72.3)	282.6a (123.7)	600.0b (161.3)
2008		0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0b (0.0)	0.0 (0.0)	160.9 (49.3)	239.1b (36.1)	26.1b (23.9)	423.9b (61.3)

Table 1-6. Mean density (number/m²) and associated standard errors of macroinvertebrate taxa by month in Cameron Lake, Nebraska, 2004-2005. Mean density in columns followed by the same letter were significantly different (within months, among years; $P < 0.05$); those without letters were not different.

Year	Month	Order Ephemeroptera	Order Odonata	Order Amphipoda	Class Hirudinea	Class Oligochaeta	Family Chironomidae	Other	Total
2004	May	1.0	0.0	1.0	1.0	2,152.3 ^a	898.6 ^a	369.6 ^a	3,299.3 ^a
		(1.0)	(0.0)	(1.0)	(1.0)	(259.9)	(135.0)	(80.4)	(351.3)
2005		0.0	0.0	6.8	0.0	818.0 ^b	388.6 ^b	111.4 ^b	1,270.4 ^b
		(0.0)	(0.0)	(6.8)	(0.0)	(110.2)	(58.2)	(35.2)	(137.0)
2004	June	0.0	2.7	2.7	2.7	1,728.3 ^a	394.0	347.8 ^a	2,320.8
		(0.0)	(2.7)	(2.7)	(2.7)	(337.1)	(140.7)	(74.7)	(442.0)
2005		0.0	0.0	0.0	0.0	584.3 ^b	652.2	54.5 ^b	1,263.6
		(0.0)	(0.0)	(0.0)	(0.0)	(106.9)	(117.2)	(48.4)	(164.4)

Table 1-7. Percent vegetation coverage by type, and mean and maximum water depths in Pelican Lake and Cameron Lake, Nebraska, 2004-2008.

Lake	Year	N	Total vegetation	Sparse emergent	Moderate emergent	Dense emergent	Sparse submergent	Moderate submergent	Dense submergent	Mean depth (cm)	Maximum depth (cm)
Pelican	2004	132	40.2	2.3	0.8	4.5	0.0	21.2	11.4	117.1	221.0
	2005	85	51.8	5.9	9.4	8.2	8.2	8.2	11.8	143.7	210.0
	2006	89	53.9	14.6	9.0	10.1	5.6	7.9	6.7	120.1	245.0
	2007	79	57.0	5.1	10.1	31.6	1.3	0.0	8.9	123.3	236.0
	2008	57	59.6	15.8	5.3	24.6	3.5	0.0	10.5	133.2	250.0
Cameron	2004	20	15.0	0.0	0.0	15.0	0.0	0.0	0.0	83.2	110.0
	2005	30	16.7	0.0	0.0	16.7	0.0	0.0	0.0	84.4	124.0

Table 1-8. Eigenvalues, proportion of variance explained, and cumulative variance explained by principal components (PC) analyses for independent variables measured in Pelican Lake, Nebraska from 2004 to 2008. Components were separated into benthic macroinvertebrate, zooplankton, climate, habitat, and physicochemical categories.

Component	PC	Eigenvalue	Difference	Variance	
				Proportion explained	Cumulative
Benthic macroinvertebrates	1	7.534	0.472	0.377	0.377
	2	7.062	3.788	0.353	0.730
Zooplankton	1	11.914	6.221	0.496	0.496
	2	5.693	1.963	0.237	0.734
Climate	1	10.240	4.051	0.488	0.488
	2	6.189	1.617	0.295	0.782
Habitat	1	4.691	1.428	0.521	0.521
	2	3.263	2.571	0.364	0.884
Physicochemical	1	9.991	3.820	0.500	0.500
	2	6.171	2.703	0.308	0.808

Table 1-9. Results of factor analysis (i.e., principal components analysis [PCA] with varimax rotation) of independent variables measured in Pelican Lake, Nebraska from 2004 to 2007. Factor scores with a “*” were those selected for the factor (loadings greater than 40). Variables that loaded (greater than 40) on more than one factor were omitted in interpretation.

Component	Variable	PC 1	PC 2
Benthic macroinvertebrates	May chironomid density	5	-100*
	June chironomid density	96*	-4
	July hirudinean density	96*	-20
	May oligochaete density	91*	9
	June oligochaete density	87*	39
	May other density	2	99*
	July other density	-26	88*
	May total density	93*	-32
	June total density	96*	9
Zooplankton	May <i>Bosmina</i> density	97*	25
	May copepod density	93*	0
	May cladoceran density	99*	8
	June cladoceran density	92*	40
	July cladoceran density	27	94*
	August rotifer density	-12	99*
	May total density	98*	12
Climate	April air temperature	11	-98*
	July air temperature	98*	21
	August air temperature	-21	92*
	March wind	100*	4
	April wind	99*	8
	May wind	99*	12
	June wind	100*	5
	July wind	99*	7
	August wind	100*	0
	Total annual precipitation	16	92*

Table 1-9 continued

Component	Variable	PC 1	PC 2
Habitat	June dissolved oxygen	95*	0
	July dissolved oxygen	98*	-21
	August dissolved oxygen	95*	15
	May water temperature	-92*	16
	June water temperature	-34	80*
	July water temperature	-20	94*
	August water temperature	13	79*
Physicochemical	June Secchi depth	95*	32
	July Secchi depth	92*	14
	May conductivity	36	-91*
	July conductivity	32	94*
	August conductivity	88*	39
	May total dissolved solids	33	-93*
	July total dissolved solids	28	95*
	May pH	-94*	4
	June pH	-90*	-16
	July pH	-96	12

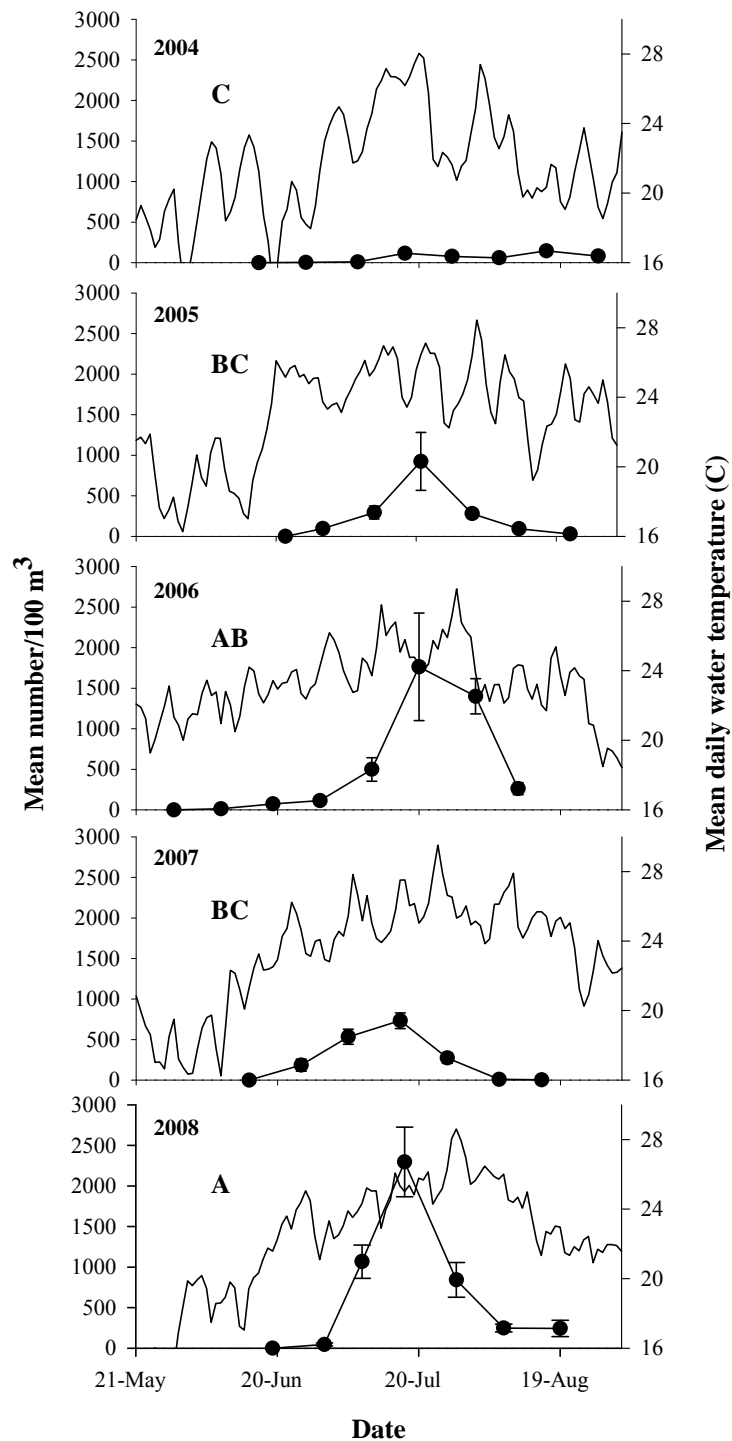


Figure 1-1. Mean larval bluegill density (\pm SE) and mean daily water temperature (C) in Pelican Lake, Nebraska from 2004 to 2008. Panels with different letters had significantly different peak densities ($\alpha = 0.05$).

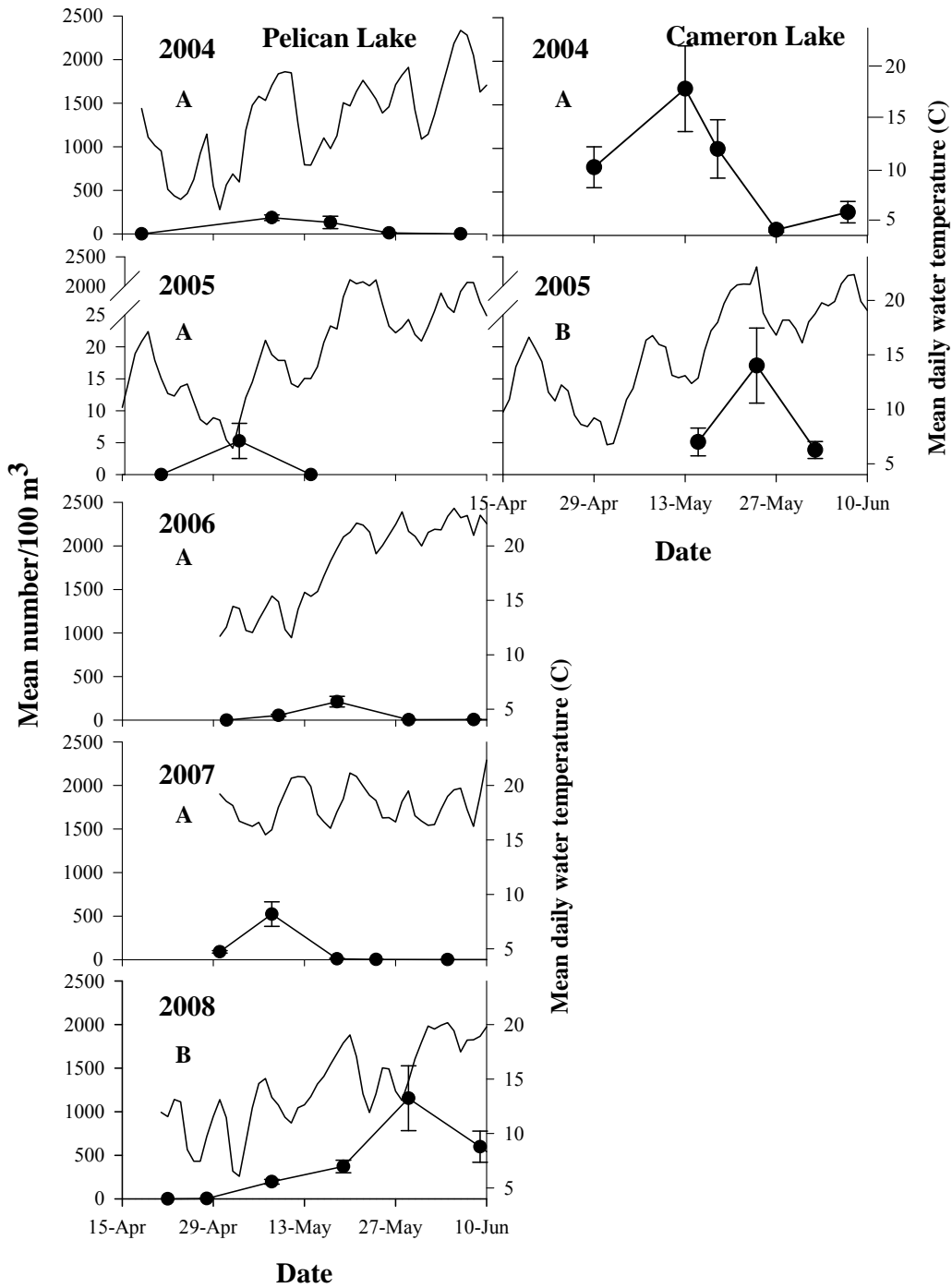


Figure 1-2. Mean larval yellow perch density (\pm SE) and mean daily water temperature (C) in Pelican Lake and Cameron Lake, Nebraska from 2004 to 2008. Panels with different letters within lakes had significantly different peak densities ($\alpha = 0.05$). The temperature logger was not recovered in Cameron Lake in 2004.

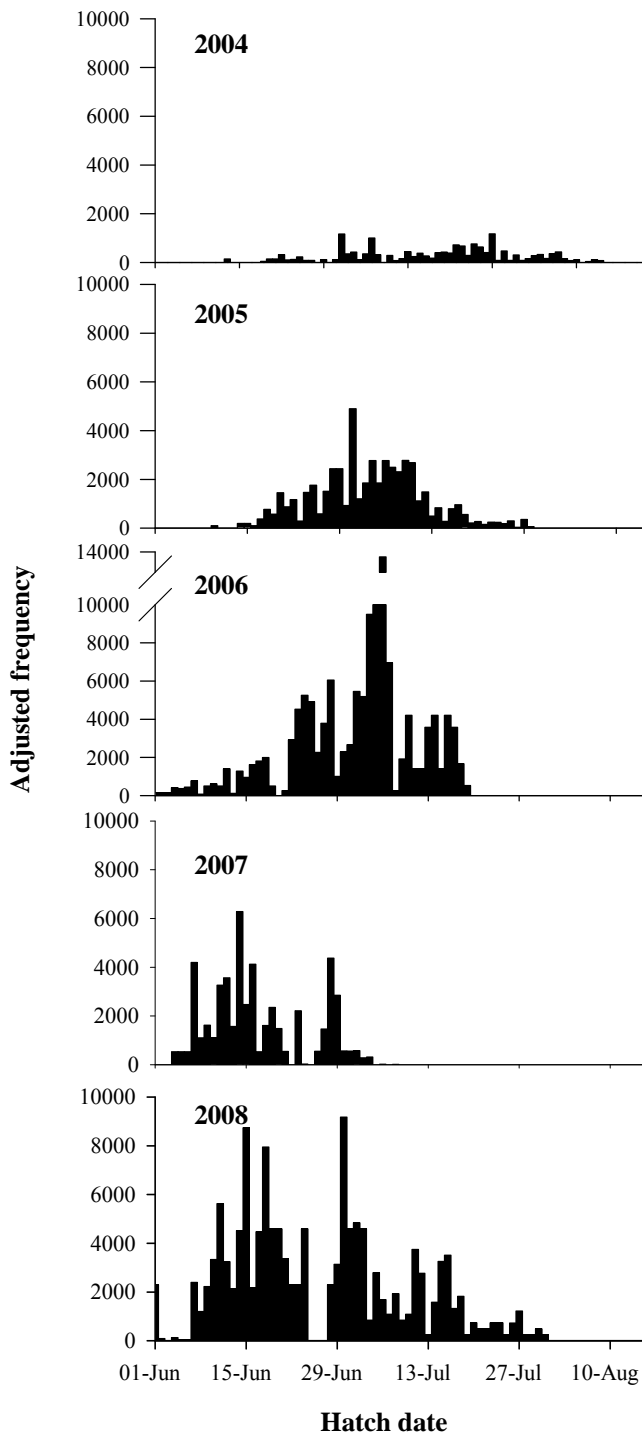


Figure 1-3. Bluegill hatch date frequency in Pelican Lake, Nebraska from 2004 to 2008. Bluegill hatch dates were weighted by corresponding larval abundances.

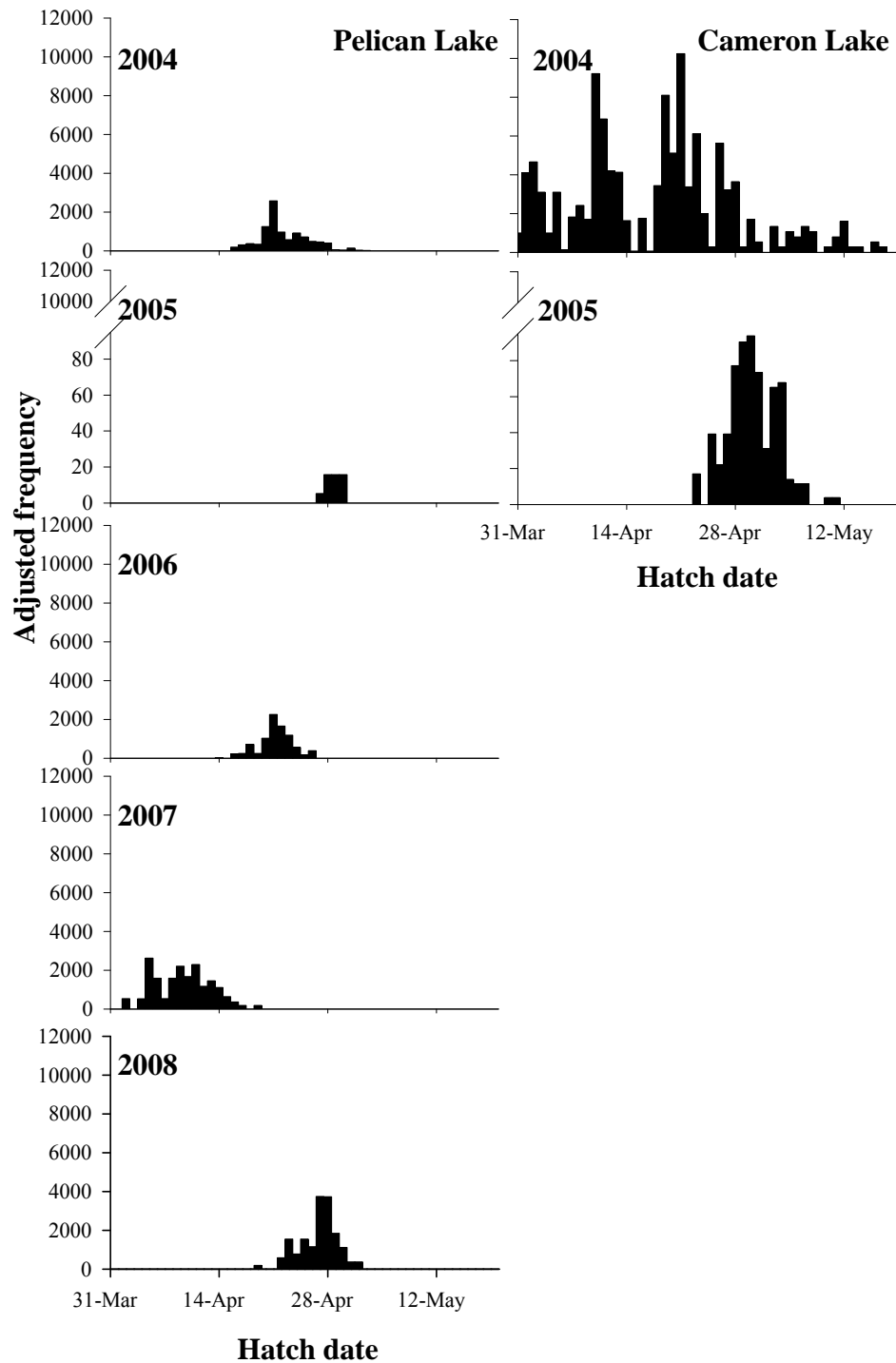


Figure 1-4. Yellow perch hatch date frequency in Pelican Lake and Cameron Lake, Nebraska from 2004 to 2008. Yellow perch hatch dates were weighted by corresponding larval abundances.

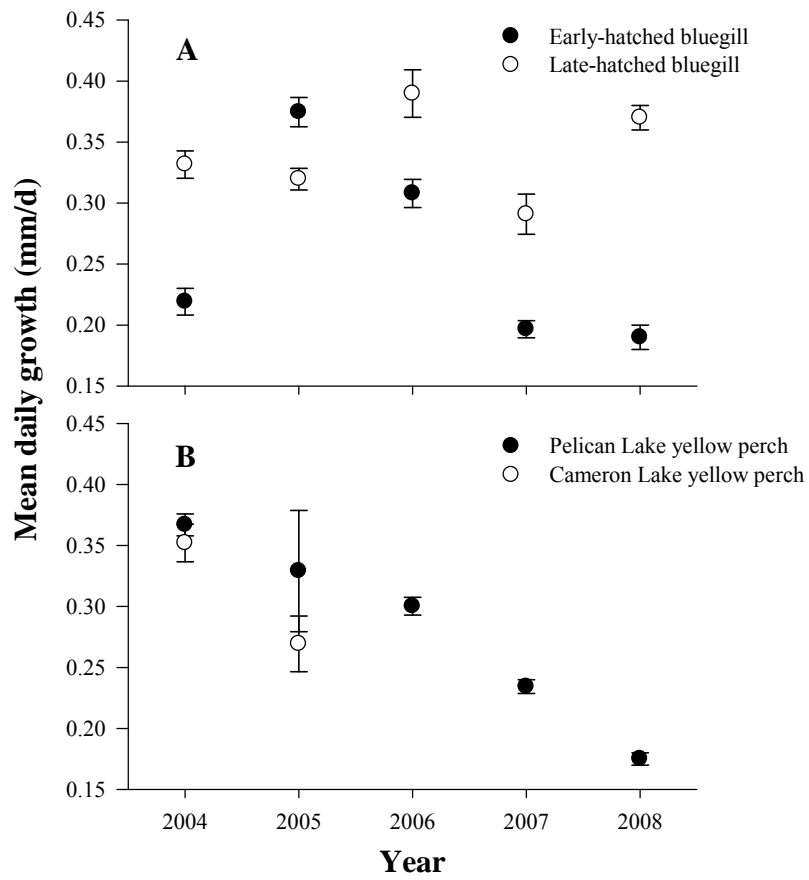


Figure 1-5. Mean daily growth rates mm/d of early- and late-hatched bluegill in Pelican Lake (A) and yellow perch in Pelican Lake and Cameron Lake, Nebraska (B) 2004-2008.

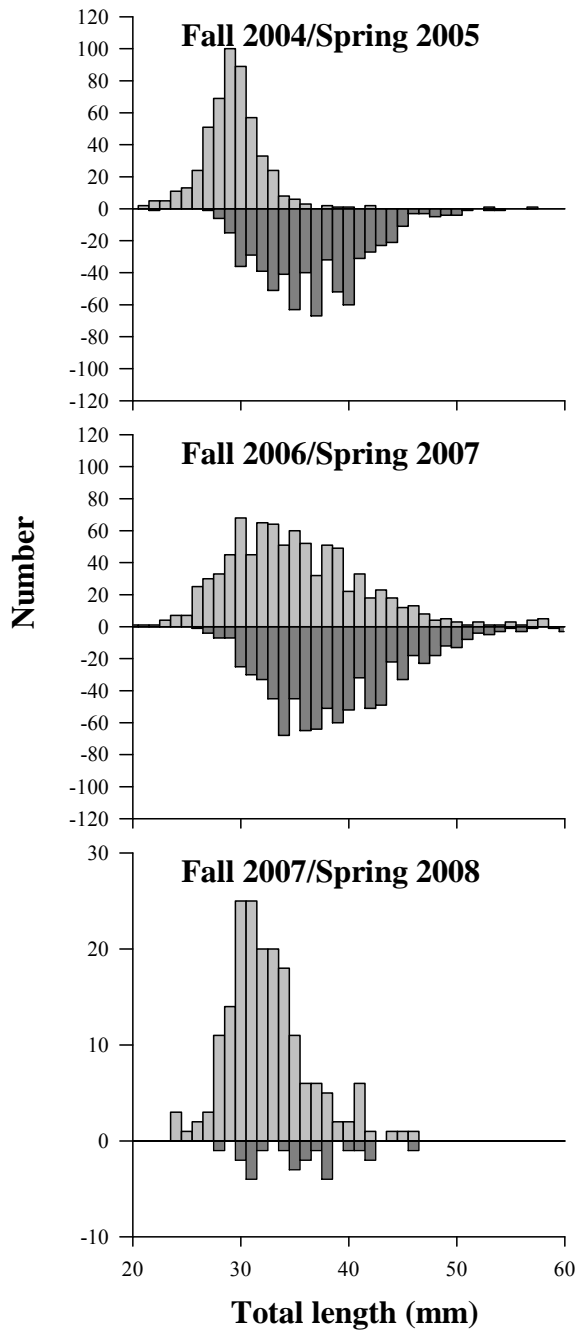


Figure 1-6. Fall and spring size distributions of bluegill from the 2004, 2006, and 2007 year classes captured in cloverleaf traps at Pelican Lake, Nebraska. Values above the x-axis are from the fall and values below the x-axis are from the following spring.

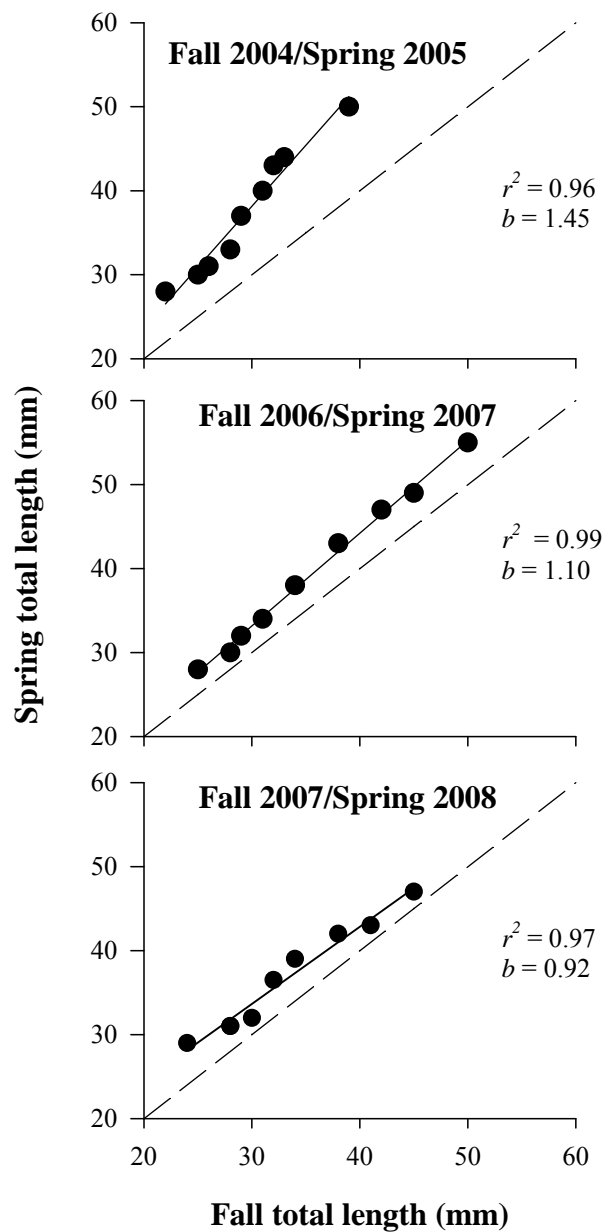


Figure 1-7. Empirical quantile-quantile plots of the length-frequency data for the 2004, 2006, and 2007 bluegill cohorts in Pelican Lake, Nebraska. The broken line is the 1:1 relationship and solid line is the least squares regression for each cohort. Coefficients of determination (r^2) and regression slopes (b) are indicated.

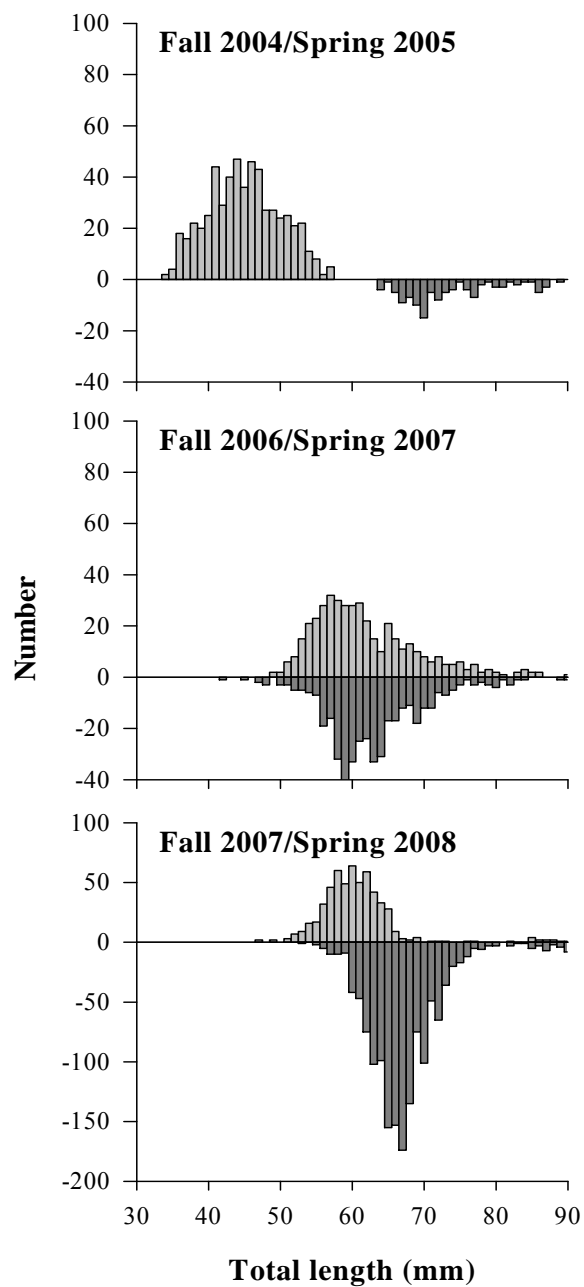


Figure 1-8. Fall and spring size distributions of yellow perch from the 2004, 2006, and 2007 year classes captured in cloverleaf traps from Pelican Lake, Nebraska. Values above the x-axis are from the fall and values below the x-axis are from the following spring.

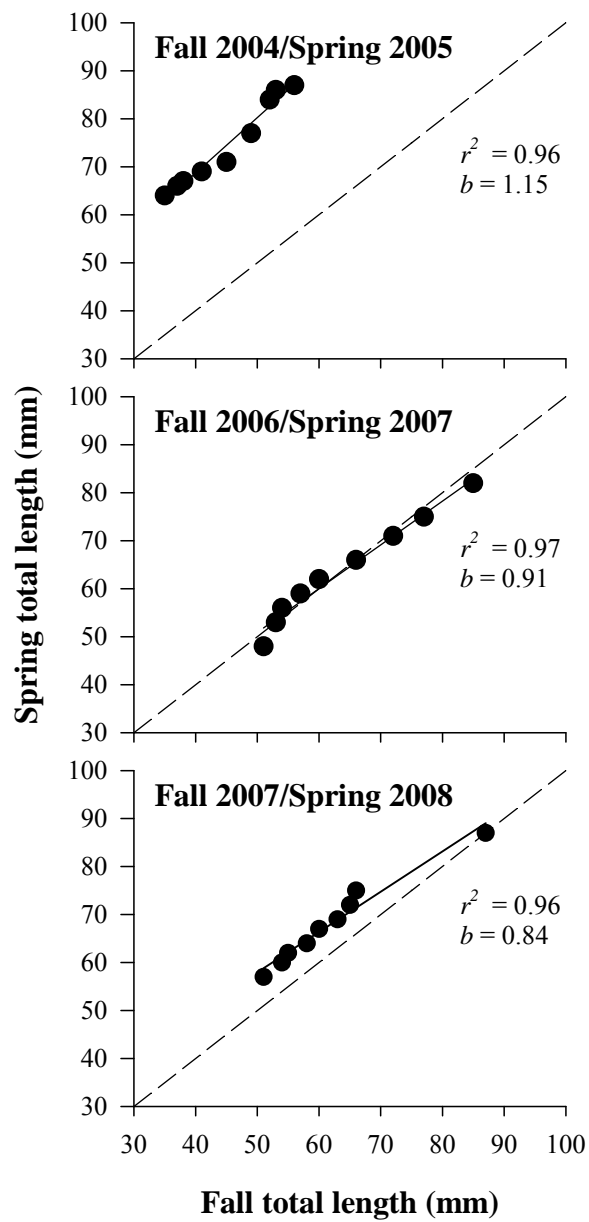


Figure 1-9. Empirical quantile-quantile plots of the length-frequency data for the 2004, 2006, and 2007 yellow perch cohorts in Pelican Lake, Nebraska. The broken line is the 1:1 relationship and solid line is the least squares regression for each cohort. Coefficients of determination (r^2) and regression slopes (b) are indicated.

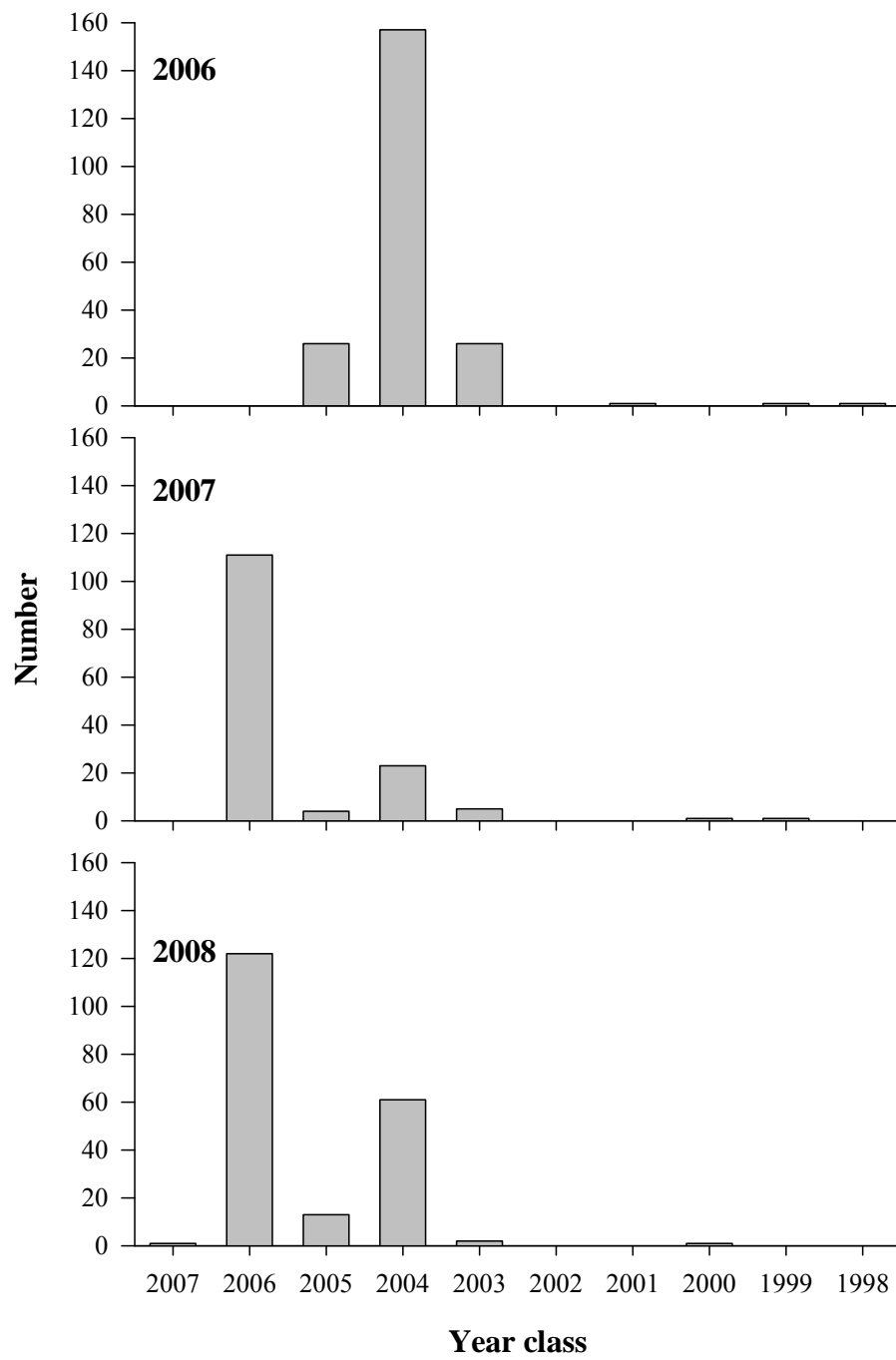


Figure 1-10. Age-frequency histograms of adult bluegill collected with modified-fyke nets from Pelican Lake, Nebraska, 2005-2008.

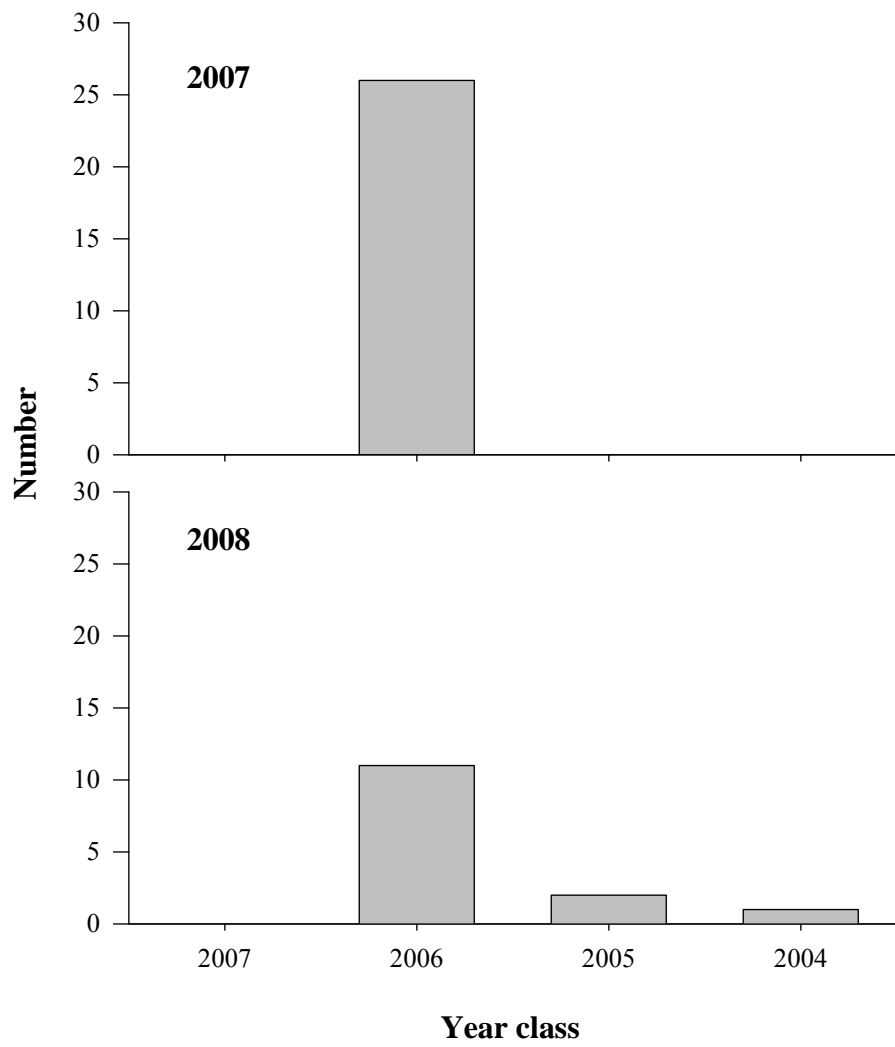


Figure 1-11. Age-frequency histograms of adult yellow perch collected with modified-fyke nets from Pelican Lake, Nebraska collected in 2005-2008.

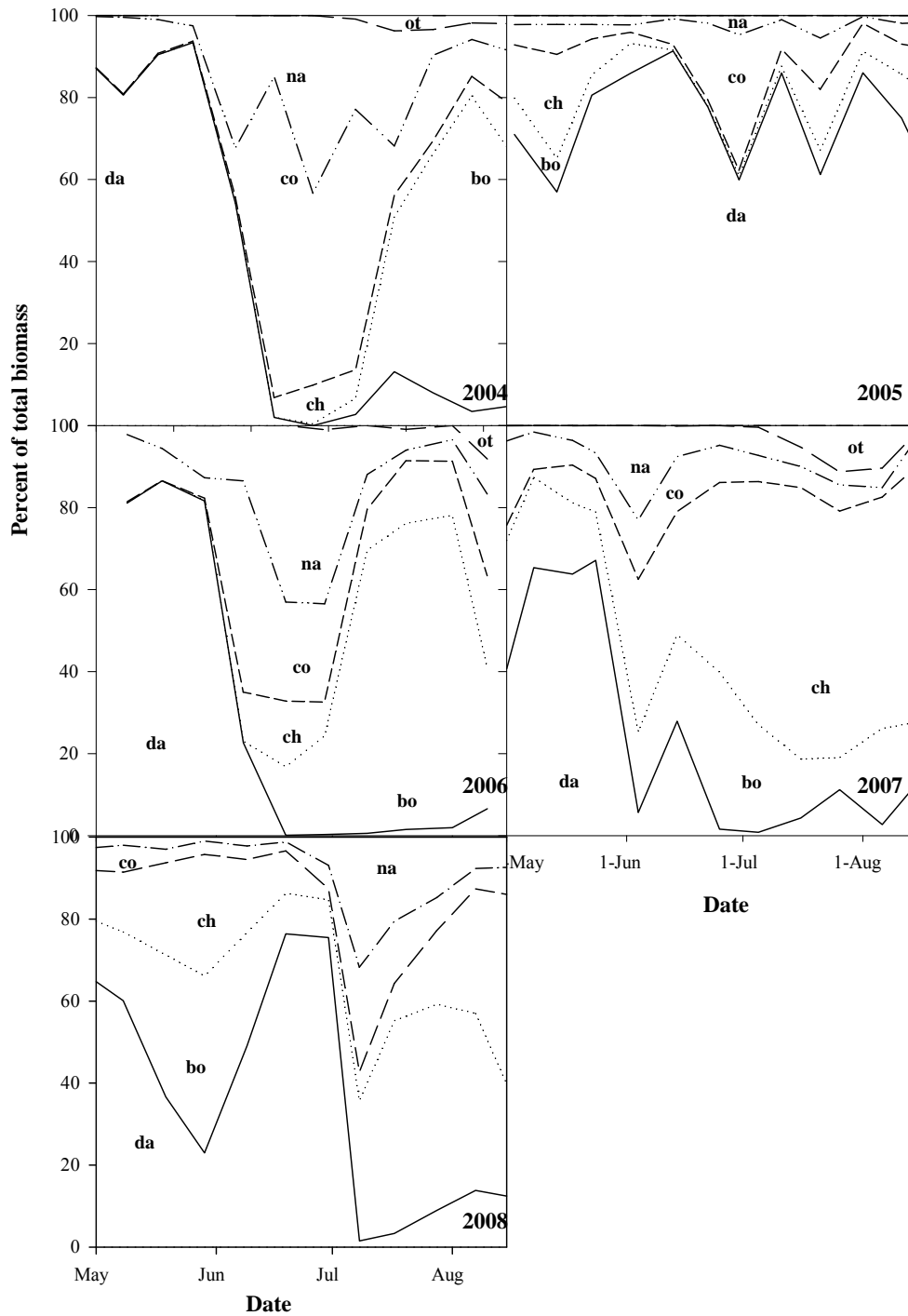


Figure 1-12. Percent of biomass for the zooplankton taxa in Pelican Lake, Nebraska, 2004-2008. Taxa are: da = daphnid (*Daphnia* and *Ceriodaphnia*), bo = *Bosmina*, ch = *Chydorus*, co = calanoid and cyclopoid copepod, na = copepod nauplii, and ot = other zooplankton.

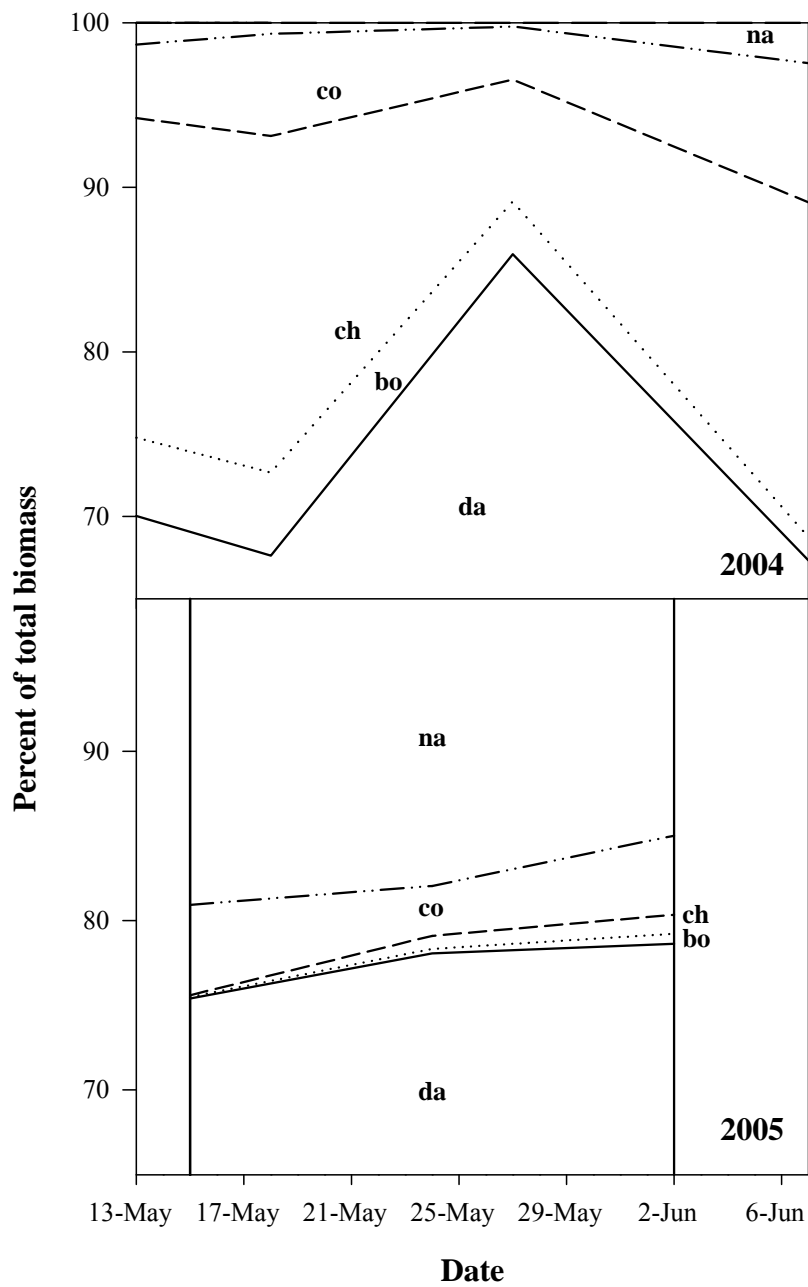


Figure 1-13. Percent of biomass for the zooplankton taxa in Cameron Lake, Nebraska, 2004-2005. Taxa are: da = daphnid (*Daphnia* and *Ceriodaphnia*), bo = *Bosmina*, ch = *Chydorus*, co = calanoid and cyclopoid copepod, and na = copepod nauplii.

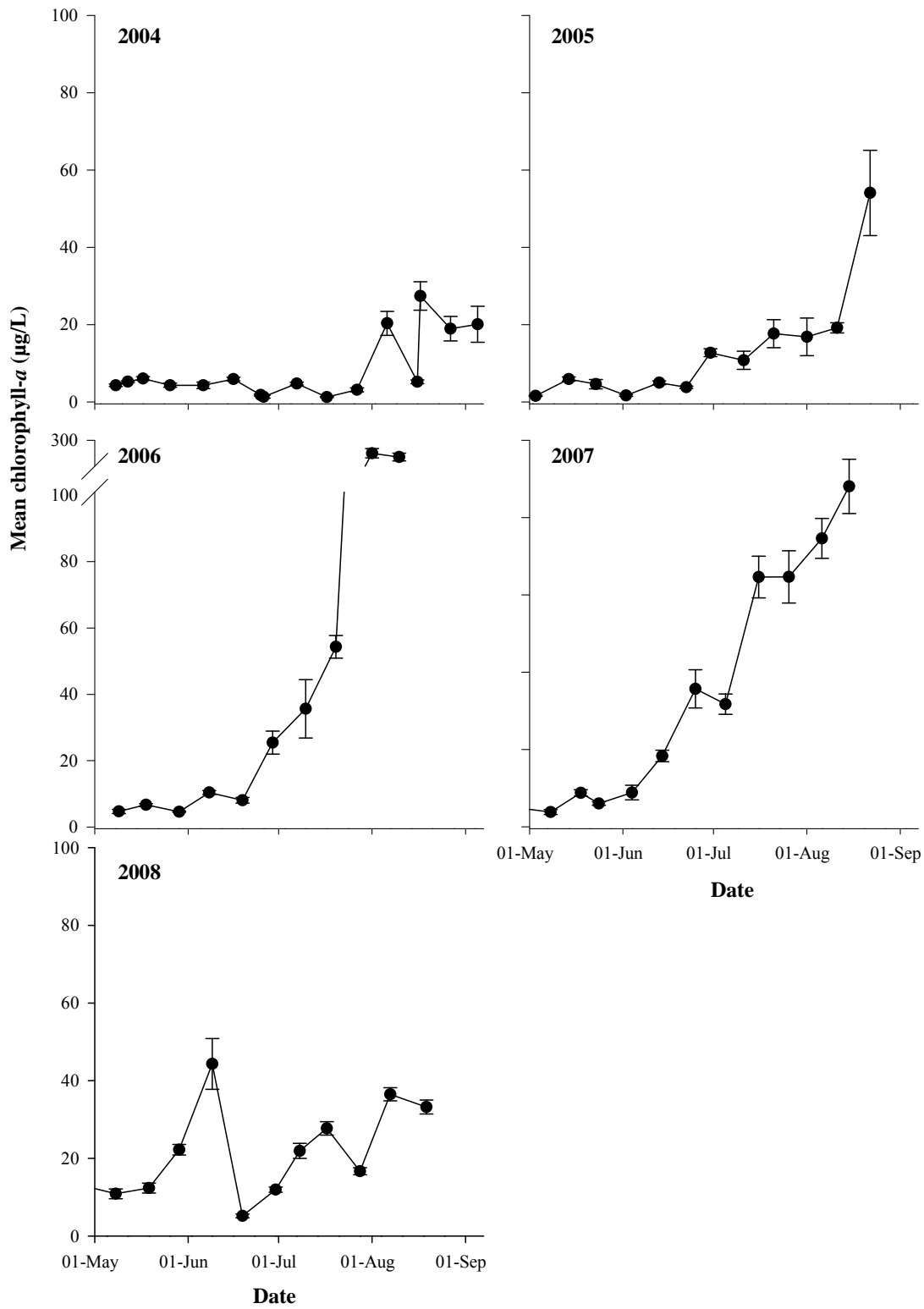


Figure 1-14. Mean (\pm SE) chlorophyll-a concentration ($\mu\text{g/L}$) from phytoplankton samples collected from Pelican Lake, Nebraska, 2004-2008.

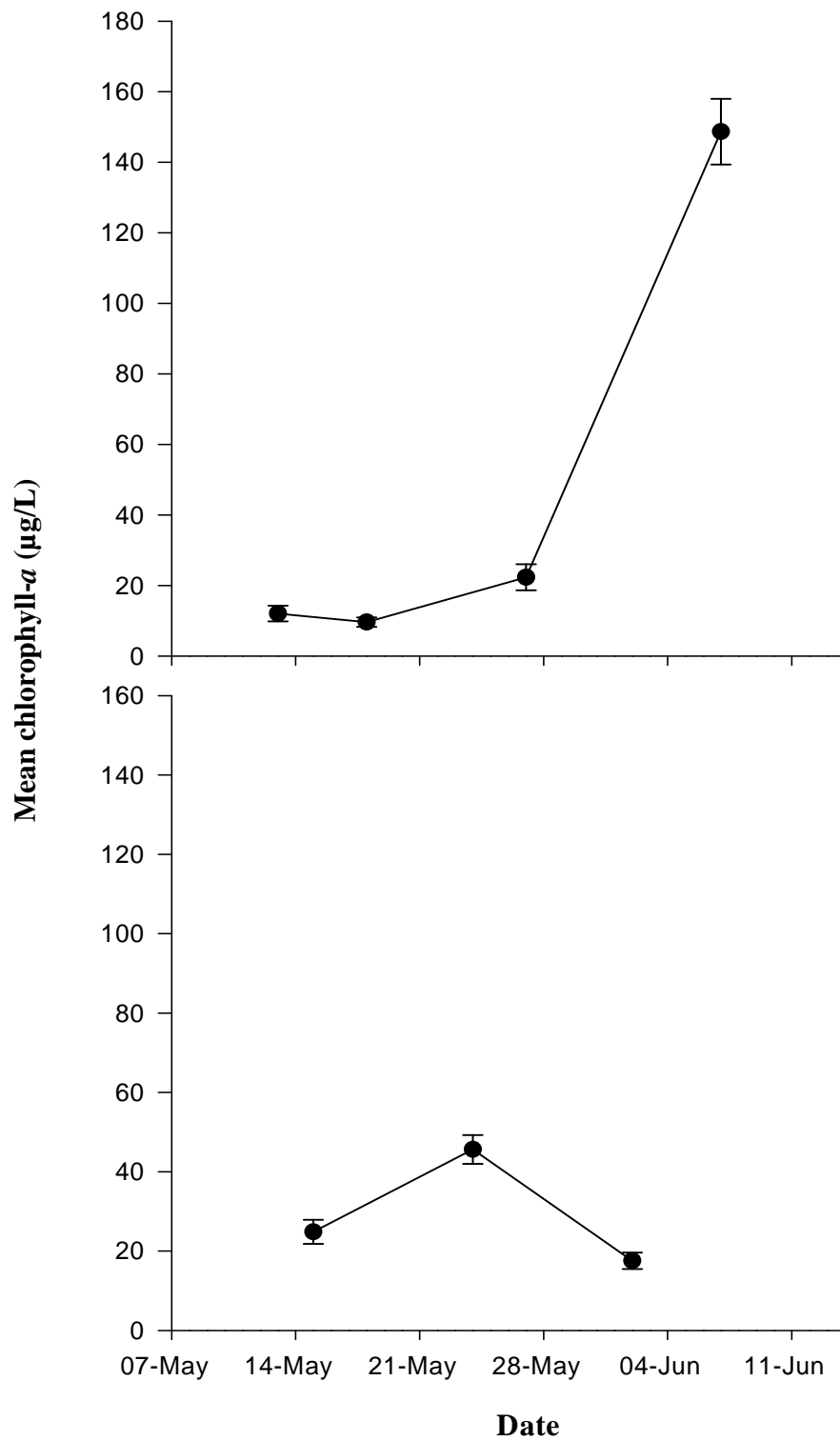


Figure 1-15. Mean (\pm SE) chlorophyll-*a* concentration ($\mu\text{g/L}$) from phytoplankton samples collected from Cameron Lake, Nebraska, 2004-2005.

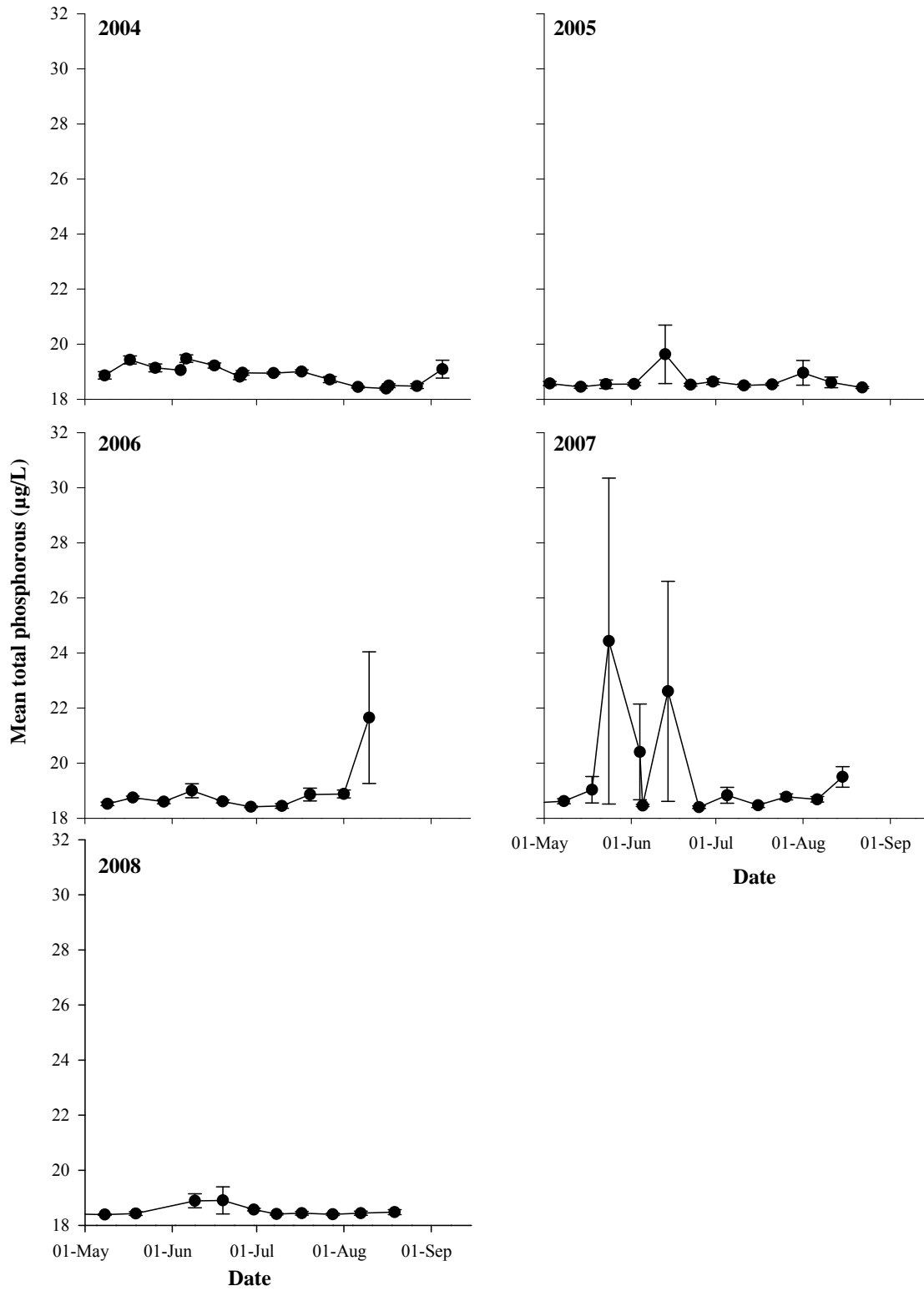


Figure 1-16. Mean (\pm SE) total phosphorous ($\mu\text{g/L}$) concentration for water samples collected from Pelican Lake, Nebraska, 2004-2008.

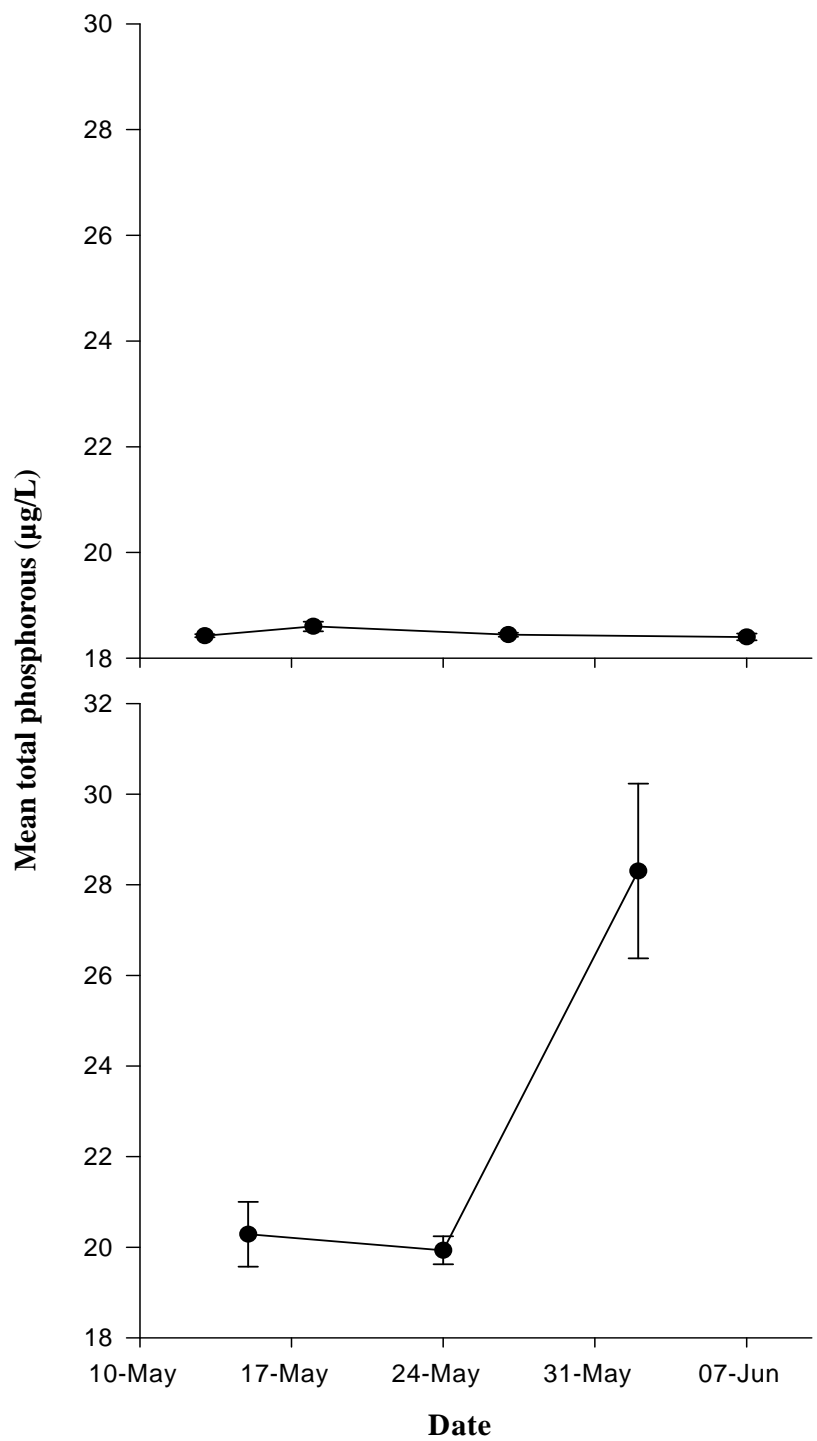


Figure 1-17. Mean (\pm SE) total phosphorous ($\mu\text{g/L}$) concentration for water samples collected from Cameron Lake, Nebraska, 2004-2005.

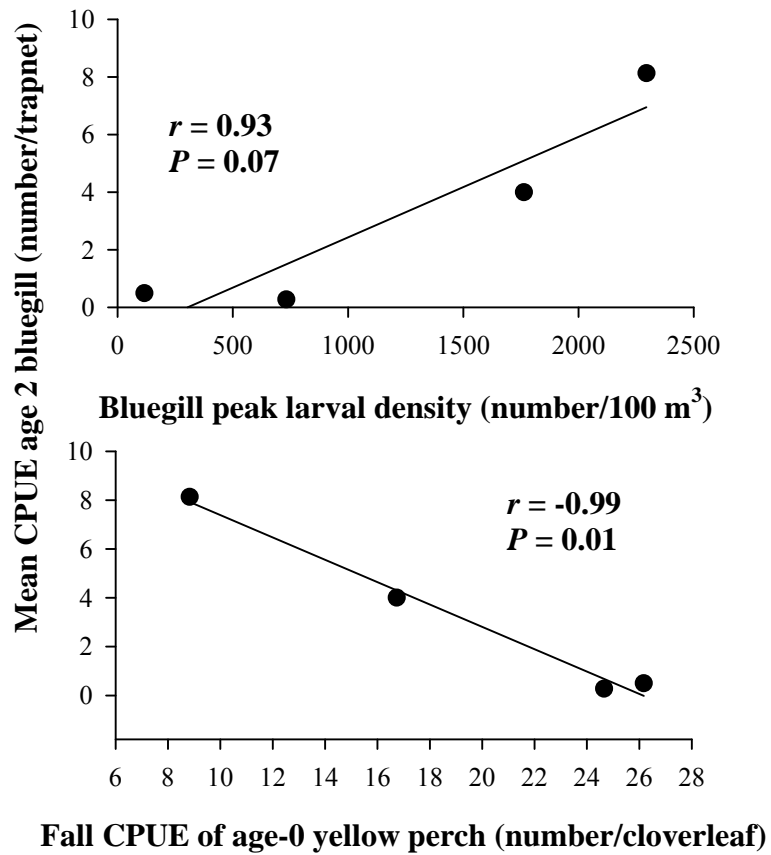


Figure 1-18. Correlations between selected bluegill and yellow perch abundance indices. CPUE = catch per unit effort.

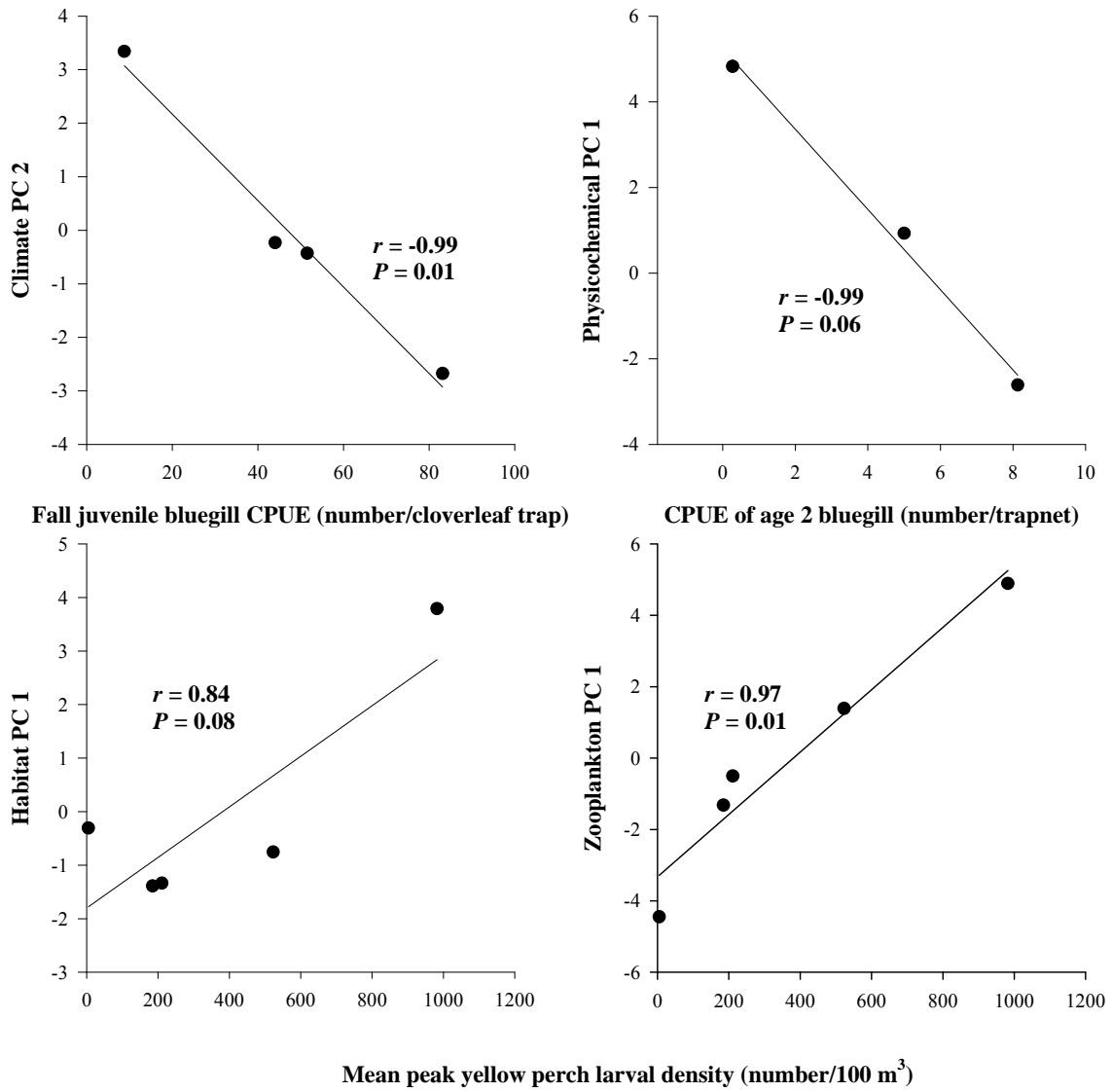


Figure 1-19. Correlations between selected bluegill and yellow perch abundance indices (mean catch per unit effort [CPUE]) and principal components (PC).

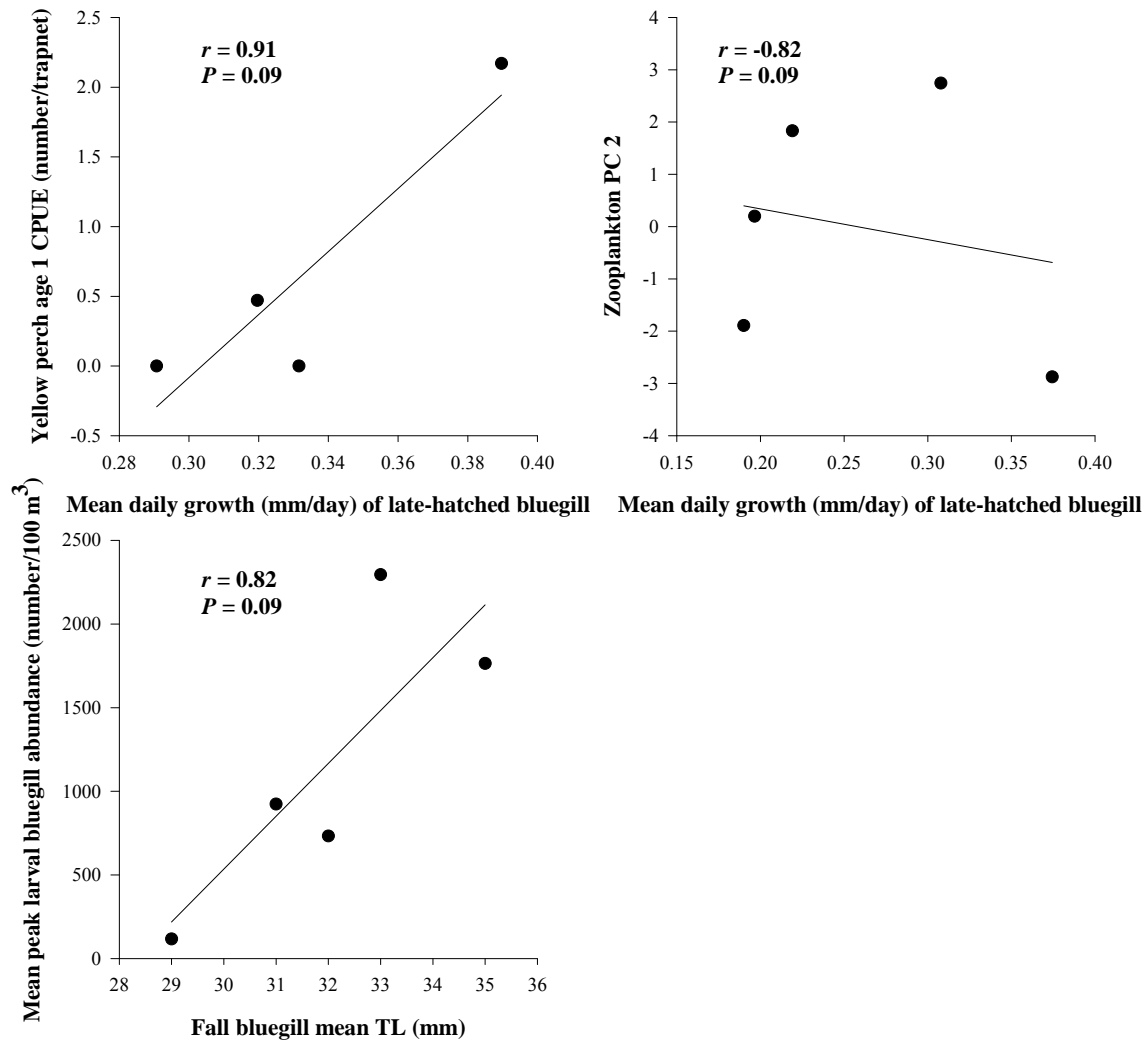


Figure 1-20. Correlations between selected bluegill abundance and growth indices and principal components (PC).

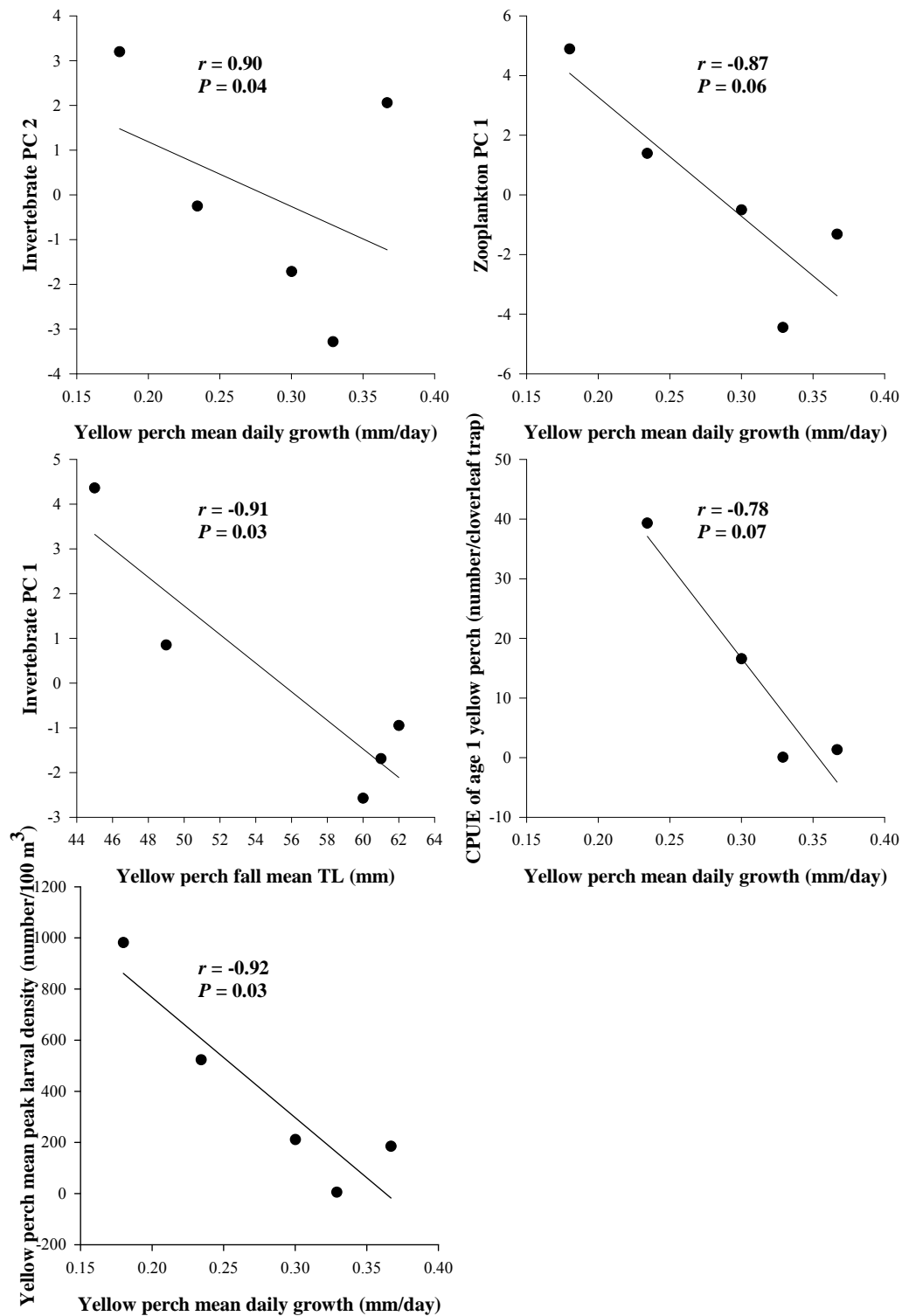


Figure 1-21. Correlations between selected yellow perch abundance and growth indices and principal components (PC).

(3) JOB OBJECTIVE:

To determine under-ice food habits of northern pike in Pelican Lake, Nebraska

Introduction

Common carp have been shown to alter abiotic (Lougheed et al. 1998; Angeler et al. 2002; Parkos et al. 2003) and biotic (Forester and Lawrence 1978; Crivelli 1983; Zambrano and Hinojosa 1999; Parkos et al. 2003) conditions of lakes. Because common carp can have substantial impacts on aquatic ecosystems, understanding basic population characteristics is a necessary first step to determine appropriate management strategies. This information, however, is lacking for populations in Nebraska Sandhill lakes.

The northern pike population in Pelican Lake is currently managed under a 71-cm (i.e., 28-inch) maximum length limit, where anglers can harvest only fish less than the length limit, and all larger fish must immediately be released. U.S. Fish and Wildlife Service biologists believe that this regulation has helped to limit common carp recruitment in Pelican Lake (U.S. Fish and Wildlife Service 2001). Therefore, our objectives were to 1) describe under-ice food habits of northern pike in Pelican Lake, Nebraska; 2) describe the size and age structure, growth, and recruitment patterns of common carp in Marsh and Pelican lakes, Nebraska; and 3) describe size and age structure, growth rates, and recruitment patterns and the relationship to climatological variables of northern pike in Hackberry Lake, Nebraska.

Methods

Northern pike were collected with angling methods from December 2003 to late-February 2004. Tip-ups baited with frozen rainbow smelt *Osmerus mordax* and equipped with treble-hooks were utilized. Recreational-angler-caught northern pike were also obtained from willing participants. Northern pike were measured (mm TL) and stomachs were removed and preserved in 10% formalin.

Food habits were determined separately by northern pike length group (TL \leq 711 mm and TL $>$ 711 mm). Diet items were identified to lowest practical level (species for fish, order for invertebrates), enumerated, measured (mm TL), and weighed (g wet weight). Diet data were summarized by frequency of occurrence, percent by number, and percent by weight.

The availability of appropriately sized common carp (i.e., 2003 year-class) during the time of food-habits sampling was determined through examination of length-frequency histograms from annual standardized sampling conducted on Pelican Lake by USFWS personnel in 2003 and 2004. To further substantiate the presence of a 2003 year class of common carp in Pelican Lake, an adult common carp population sample was obtained in 2006. In addition, a sample of adult common carp was obtained in 2006 from Marsh Lake where no top-level piscivores (i.e., northern pike, largemouth bass) were present. This information was used to compare common carp populations in a lake with a piscivore-heavy fish community (Pelican) to a lake with low piscivory (Marsh).

Common carp in Pelican Lake were obtained via pulsed-DC (200-250 V, 3-6 A) boat electrofishing in July 2006, while common carp were obtained from Marsh Lake in April 2006 using modified fyke nets (16-mm bar mesh, 1.1- by 1.5-m frames, and 22-m leads). Captured common carp were placed on ice and returned to the laboratory. Total length

(mm) and weight (g) were recorded, and length-frequency histograms were generated to describe common carp population size structure in each lake. Proportional stock density (PSD; percentage of 28-cm and longer fish that also exceeded 41 cm) and relative stock density of preferred-length fish (RSD-P; the percentage of 28-cm and longer fish that also exceeded 53 cm) to quantify size structure (Gabelhouse 1984) were calculated. The 95% confidence intervals (CI) for PSD and RSD-P were calculated as suggested by Gustafson (1988). Catch per unit effort (CPUE), an index of population abundance, was calculated as the mean number of fish \geq stock length (i.e., 28 cm) captured per net night/h of electrofishing (Hubert 1996).

Asteriscus otoliths, the only validated aging structure for common carp (Brown et al. 2004), were removed from common carp to determine age and growth. Otoliths were mounted in epoxy and sectioned transversely through the nucleus using a low-speed Isomet saw. Two independent readers determined age in years with the aid of a microscope. Discrepancies in age assignments were resolved by examining the otoliths in unison. Age-frequency histograms were created to describe the age structure and assess recruitment patterns in both lakes. Growth was described as mean length by cohort at time of capture and von Bertalanffy growth parameters were determined for each population (Van Den Avyle and Hayward 1999).

Reproductive status and gender of pre-spawn common carp from Marsh Lake were recorded by determining sexual maturity through direct examination of gonads. We determined 50% maturity at length and age using probit analysis (Welch and Foucher 1988). Reproductive characteristics were not recorded for carp from Pelican Lake because they were sampled post-spawn (i.e., July).

Northern pike were collected from Hackberry Lake using four overnight experimental gill net sets in June and July of 2004. Experimental gill nets were of two different sizes. One net size was 36-m long with five 7.6-m panels of 1.9-, 2.5-, 3.8-, 5.1-, and 6.4-cm bar mesh. The other net used was 43-m long with six 7.6-m panels of 1.9-, 2.5-, 3.8-, 5.1-, 6.4-, and 7.6-cm bar mesh. Collected fish were placed on ice and returned to the laboratory for further processing.

Total length (TL; mm) and weight (g) were recorded from each specimen prior to removal of sagittal otoliths for aging. Some northern pike were damaged due to predation by snapping turtles *Chelydra serpentina* while in the gill nets; lengths and/or weights were not recorded from these individuals although otoliths were removed when possible. Although scales and cleithra have been used in the past to age northern pike (Casselman 1990; Laine et al. 1991), the use of otoliths has gained recent favor over many other bony structures to age fish (Casselman 1987; Secor et al. 1995). Sagittal otoliths were removed, polished, and dried. Otoliths were mounted in epoxy and sectioned through the focus using an Isomet low-speed saw. Several sections were mounted on a slide to ensure that annuli were not missed. Annuli were counted by two independent readers and disagreements were resolved by reading in concert.

Length- and age-frequency histograms were generated to describe size and age structure of northern pike. Size structure was further indexed using proportional stock density (PSD; the percentage of 35 cm and larger fish that also exceeded 53 cm) and relative stock density of preferred-length fish (RSD-P; the percentage of 35 cm and larger fish that also exceeded 71 cm; Gabelhouse 1984). Northern pike growth was described as mean length by cohort at time of capture.

A catch-curve analysis (Ricker 1975) was computed by regressing the \log_e of number-at-age as a function of age for age 2-11 northern pike. Residuals produced from the catch-curve analysis were used to index year-class strength (Maceina 1997). Positive residuals represented relative strong year classes while negative residuals represented relative weak year classes. Bi-variate relationships between year-class strength (i.e. residual) and selected climatological variables (i.e. temperature, precipitation, wind, and winter severity) that corresponded with key time periods in northern pike spawning, larval, and juvenile stages were explored by conducting correlation analyses. Climatological data were obtained from the National Oceanic and Atmospheric Administration weather station in Valentine, Nebraska, located approximately 32 km from Hackberry Lake and operated by the National Weather Service. Climate variables examined were mean daily air temperature ($^{\circ}\text{C}$), cumulative daily precipitation, mean daily wind speed (km/h), and winter severity (cumulative number of days where temperature was $\leq 0^{\circ}\text{C}$) in the first winter for an age-0 northern pike. Statistical analyses were conducted using the Statistical Analysis System (SAS 2002). Statistical significance was set *a priori* at $\alpha = 0.10$ because this was an exploratory study with a low number of observations (i.e. year-classes).

Results

A total of 80 northern pike $\text{TL} \leq 711$ mm and 13 northern pike $\text{TL} > 711$ mm were captured from Pelican Lake and examined for food habits. Diet items were found in 83% ($n=66$) of fish ≤ 711 mm TL and 54% ($n=7$) of fish > 711 mm TL. Eight prey types were

identified for smaller northern pike while four prey types were identified for larger pike. Fish dominated the diets of both size groups of northern pike. Lepomid sunfish, rainbow smelt *Osmerus mordax*, and unidentified fish made up the majority of diet items (Table 2-1). It is unknown if the smelt observed in diets were scavenged from the lake bottom or were consumed from angling activities (i.e., “bait robbing”). No common carp were observed in northern pike stomachs.

It appears that age-0 common carp were produced in 2003. Catches during routine surveys by the USFWS were dominated by age-0 (based on total length) fish although it is uncertain if these fish recruited to the fishery (USFWS 2005). Nevertheless, these fish should have been available to northern pike as prey at the time of our sampling.

A total of 560 common carp was sampled from Marsh Lake and mean CPUE was 2.6 stock-length fish/trap net night (Coulter et al. 2008). Total length ranged from 87 mm to 709 mm. The size structure was dominated by smaller individuals between 80 mm and 140 mm (Figure 2-1). Proportional stock density was 51 (95% CI = ± 20) and RSD-P was 23 (95% CI = ± 16). Marsh Lake demonstrated relatively consistent recruitment through age 7 as no missing year classes were detected (Figure 2-2). Maximum age was 12, although no specimens ages 8-11 were captured. An abundant 2005 cohort dominated the sample (n=281). Growth of common carp in Marsh Lake was slow (Figure 2-3). The von Bertalanffy growth coefficient (K) was 0.173 and ultimate length (L_{inf}) was 709 mm.

We observed 100 percent maturity for common carp of both sexes in Marsh Lake by 360 mm. The 50% total length at maturity (both sexes combined) was 312 mm (PROBIT, $\chi^2 = 40.27$, $P < 0.0001$) and the 50% age at maturity was 3.4 years (PROBIT,

$\chi^2 = 31.43, P < 0.0001$). The smallest mature male was 171 mm while the smallest mature female was 348 mm.

A total of 63 common carp was collected from Pelican Lake and mean CPUE was 12.2 stock-length fish/h electrofishing. Total length ranged from 90 mm to 865 mm. The size structure of common carp in Pelican Lake was dominated by individuals 600 mm and longer (Figure 2-1). Proportional stock density for Pelican Lake was 98 (95% CI = ± 6) and RSD-P was 96 (95% CI = ± 7). The Pelican Lake population exhibited erratic recruitment with several missing year classes not included or not sampled. The population sample was dominated by the age-4 and -5 cohorts (2001 and 2002 year classes), with a maximum observed age of 16 years (Figure 2-2). Growth was rapid up to age 4, where they reached asymptotic lengths (Figure 2-3). The von Bertalanffy growth coefficient (K) was 0.295 and L_{inf} was 758 mm in Pelican Lake. Common carp exceeded 600 mm by age 4. We were unable to analyze maturity of common carp from Pelican Lake because they were collected after spawning.

A total of 139 northern pike was collected from Hackberry Lake between 16 June and 16 July 2004 (Rydell et al. 2008). Total length ranged from 490 mm to 860 mm with a mean of 679 mm (Figure 2-1). Proportional stock density was 98 (95% CI = ± 3) and RSD-P was 40 (95% CI = ± 9) indicating a population dominated by larger individuals. One memorable-length (TL ≥ 86 cm) and no trophy-length (TL ≥ 112 cm; Gabelhouse 1984) northern pike were collected.

Ages ranged from 2 to 11, years corresponding to the 2002-1993 year classes (Figure 2-2). Age-1 northern pike (2003 year class) were absent from our sample; however, those may not have been sampled effectively with our gill nets. Northern pike

exhibited relatively consistent recruitment from 1993 to 2002, as no missing year classes were detected. However, year-class strength was variable as indicated by the erratic pattern in the age-frequency histogram (Figure 2-2). The 2000-2002 year classes were weaker than expected in comparison with the more abundant, older age groups.

Growth was rapid through the first two years but slowed thereafter (Figure 2-3). The mean length for age-3 fish (i.e., approximately 3.5 years of age) was 644 mm (SE = ± 10) while the mean length was 677 mm (SE = ± 12) for age-6 fish.

Mean combined wind speed for April, May, and June was negatively correlated with northern pike year-class strength ($r = -0.57$, $P = 0.09$; Figure 2-4) while various measures of annual precipitation were not correlated with year-class strength. Combined March, April, and May mean daily air temperature and year-class strength were curvilinearly related ($r = 0.81$, $P = 0.02$; Figure 2-5). Winter severity (cumulative number of days $\leq 0^{\circ}$ C) was not significantly correlated with year-class strength ($r = -0.40$, $P = 0.26$).

Discussion

Although common carp were not observed in the diet of northern pike in this study, sub-stock length (TL < 280 mm) were observed in the diets of northern pike in fall of 2004 (USFWS 2006). Few northern pike exceeding 70 cm were collected in 2003-2004 winter efforts, and previous research (DeBates 2003) indicated that only these larger pike preyed upon juvenile common carp in the fall. Although predation on age-0 common carp by northern pike may be a rare event, it could have important implications for suppressing recruitment into the common carp population. Certainly the size

structure of common carp was dominated by larger individuals in Pelican Lake (PSD=98, RSD-P=96) which could be the result of predation on small ages by northern pike and largemouth bass. Phelps et al. (2008a) did find size-selective overwinter mortality of age-0 common carp in South Dakota glacial lakes, but did not identify the mechanism for the differential survival.

Common carp population samples from our two study lakes were obtained with different gears and at different times of year, which may preclude direct comparisons between the two samples. However, while we suspect that length-related differences may occur between gears, the use of distinct gears does not explain the difference in recruitment patterns observed in the two lakes. The Pelican Lake population demonstrated erratic recruitment while common carp in Marsh Lake showed consistent recruitment, suggesting that carp recruitment patterns in Nebraska Sandhill lakes may be influenced by factors specific to individual lakes. In contrast, Phelps et al. (2008b) found that common carp recruitment patterns in 18 eastern South Dakota lakes exhibited highly synchronous, erratic recruitment, presumably due to large-scale climatic factors.

We did not determine population abundance for common carp in our two study lakes. However, Pelican Lake likely had only a moderate population abundance based on the mean CPUE of 12.2 stock-length fish/h of electrofishing. Paukert and Willis (2000) collected common carp by electrofishing in nine Sandhill lakes, and four had a mean CPUE that exceeded the value for Pelican Lake; maximum CPUE was 77.1 stock-length fish/h at Home Valley Lake. We had no comparison data for trap-net CPUE, so we cannot assess population abundance of common carp in Marsh Lake based on that index.

Pelican and Hackberry lakes are regulated with a 71-cm (28-in) maximum length limit for northern pike in an attempt to increase the abundance of large piscivores and reduce common carp abundance. Sammons et al. (1994) found that northern pike preyed on common carp in midsummer and winter in a South Dakota glacial lake and DeBates (2003) found that pike in Pelican Lake preyed upon age-0 carp in September and October. Furthermore, Pelican Lake has a 38-cm (15-in) minimum length limit for largemouth bass, another piscivore. The common carp population in Pelican Lake was characterized by larger, older, and faster-growing individuals than that of Marsh Lake, which lacked top-level piscivores. A lower overall population abundance in a predator-dominated fish community likely explains the faster growth of common carp in Pelican Lake. In contrast, the common carp in Marsh Lake had no predators; thus, the slower growth rate suggests a population with slower, density-dependent growth.

The Hackberry Lake northern pike population had faster early growth rates (ages 2-4) than most populations in North America and Europe (Willis 1989; Neumann et al. 1994; Margenau et al. 1998). Paukert and Willis (2003a) reported similar growth of northern pike in a study of 30 Sandhill lakes. Out of the 139 northern pike sampled in Hackberry Lake, only one memorable length and no trophy length fish were collected, suggesting that northern pike at the southern boundary of their natural geographic range may not attain large sizes due to physiological environmental limitations. Neumann et al. (1994) found that northern pike growth in Lake Thompson, South Dakota was restricted during the high water temperature periods of summer.

The high observed PSD and RSD-P are indicative of a northern pike population dominated by larger individuals. The maximum size limit (71 cm) for anglers likely

contributed to the observed size structure. The angling restrictions were established to protect larger northern pike that may control common carp recruitment through predation, Sammons et al. (1994) found that northern pike in a South Dakota lake preyed on common carp in midsummer and winter. Our gill net sizes may not have captured smaller (i.e. age 1) northern pike effectively, as no substock size (TL < 350 mm; Gabelhouse 1984) fish were captured. Neumann and Willis (1994) caught no northern pike <340 mm in a South Dakota lake in a 25-mm (bar measure) gill net; the smallest gill net mesh size used in our study was 19 mm. Thus, we likely did not catch small (i.e., TL < 350 mm) northern pike because fish were not fully recruited to the gear. The Hackberry northern pike population exhibited relatively consistent but still variable recruitment. Paukert and Willis (2003b) also reported variable northern pike recruitment, but missing year classes were rare. They suggested that among-lake variability in recruitment patterns may be attributable to lake-specific factors.

Northern pike year-class strength was curvilinearly related to mean combined March, April, and May air temperature, corresponding to the time period when northern pike are spawning, hatching, and entering their juvenile life stages. Because northern pike in our study lake are near the southwestern edge of their natural geographical range (Crossman 1996), warmer years may depress recruitment. Previous research has found positive relationships between temperature and recruitment but most of these studies occurred at more northerly latitudes (Kipling and Frost 1970; Casselman and Lewis 1996). Although water levels have been cited as important for spawning and nursery habitats (Casselman and Lewis 1996), thereby potentially affecting year class strength, precipitation was not related to year-class strength in Hackberry Lake over the range of

years in our study. We suspect that the abundance of aquatic vegetation for spawning habitat in Sandhill lakes provided adequate resources for northern pike recruitment in most lakes. Mean combined wind speed for April, May, and June was weakly correlated with northern pike year-class strength. Wind has been reported to affect recruitment of other fishes through physical destruction of eggs, transport of eggs to unfavorable locations, and sedimentation on eggs. Hassler (1970) reported that higher mortality was associated with wind-induced siltation of embryos. Some combination of wind-related factors is likely to detrimentally affect northern pike recruitment in Hackberry Lake.

Finally, although winter severity has been implicated in high overwinter mortality of other fishes (Oliver et al. 1979; Adams et al. 1982; Miranda and Hubbard 1994), we found no evidence that recruitment of northern pike in Hackberry Lake was affected by severity of winter.

Population characteristics of common carp differed in the two populations examined during this study; therefore, recruitment patterns and factors influencing recruitment patterns in Nebraska Sandhill lakes may need to be assessed on a lake-by-lake basis. Given the substantial influence that this organism can have on entire aquatic ecosystems, we recommend further research to better understand population characteristics of both common carp and northern pike dynamics. This study also provided detailed information on a northern pike population in one Sandhill lake. However, we recommend further research in other lakes to fully understand the recruitment patterns of northern pike in Nebraska Sandhill lakes.

Table 2-1. Percent by number, weight, and occurrence of diet items consumed by length group (mm) for northern pike from Pelican Lake, Nebraska, winter 2003-2004. Unid = unidentified.

Prey type	Length group					
	≤711 (n = 66)			>711 (n = 7)		
	Percent by number	Percent by weight	Frequency of occurrence	Percent by number	Percent by weight	Frequency of occurrence
Plant/detritus	2.0	3.4	2.5	0.0	0.0	0.0
Amphipoda	22.0	0.2	1.3	0.0	0.0	0.0
Rainbow smelt	18.0	39.3	6.3	66.7	34.4	15.4
Largemouth bass	4.0	5.0	5.0	0.0	0.0	0.0
Lepomid sunfish	20.0	12.6	7.5	19.0	62.6	15.4
Yellow perch	2.0	1.6	1.3	0.0	0.0	0.0
Unid fish	29.0	27.8	36.3	14.3	3.0	23.1
Ranidae	3.0	10.2	3.8	0.0	0.0	0.0

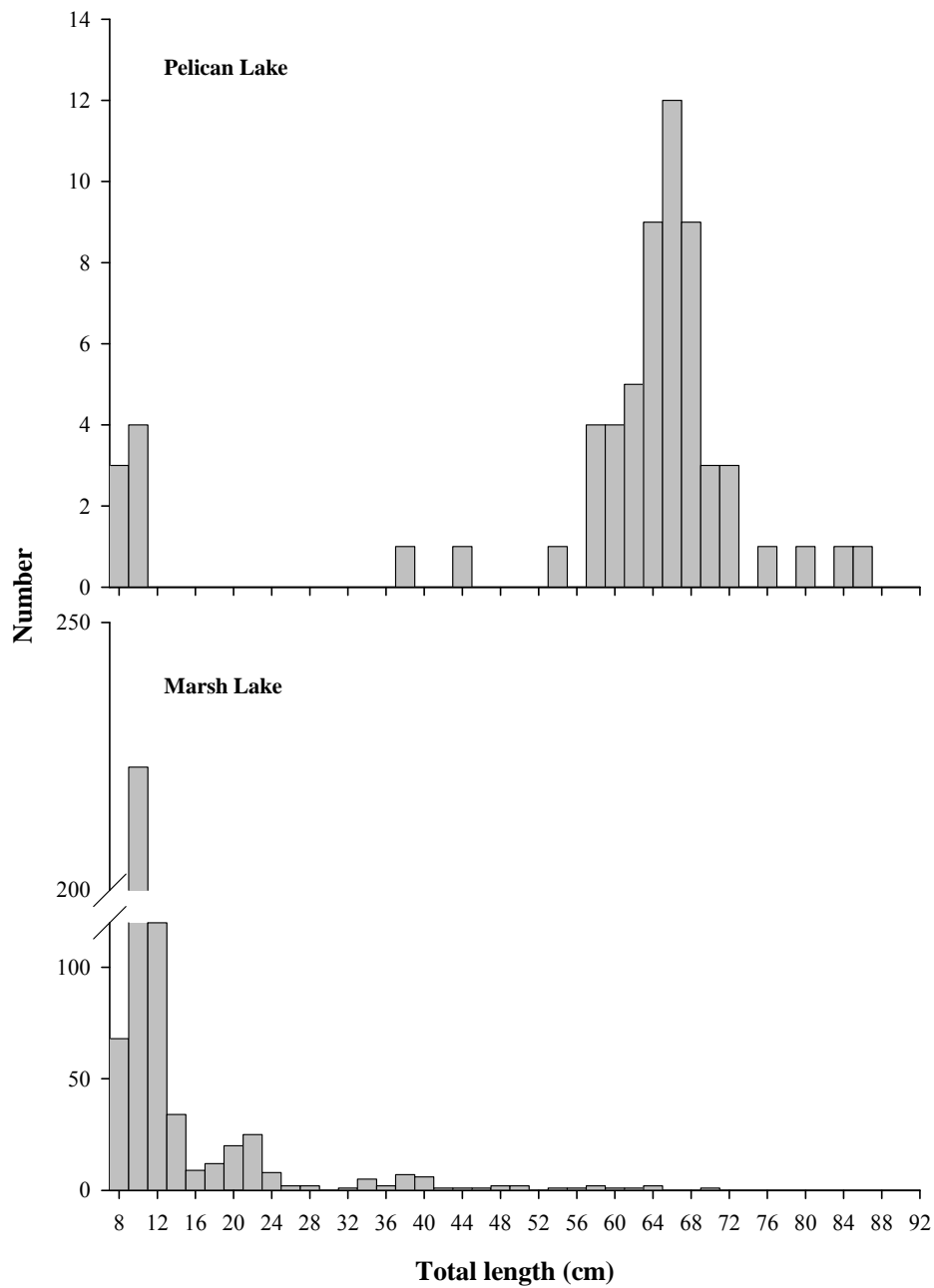


Figure 2-1. Length frequency by 2-cm length groups for common carp sampled from Pelican and Marsh lakes, Nebraska in April and July of 2006, respectively.

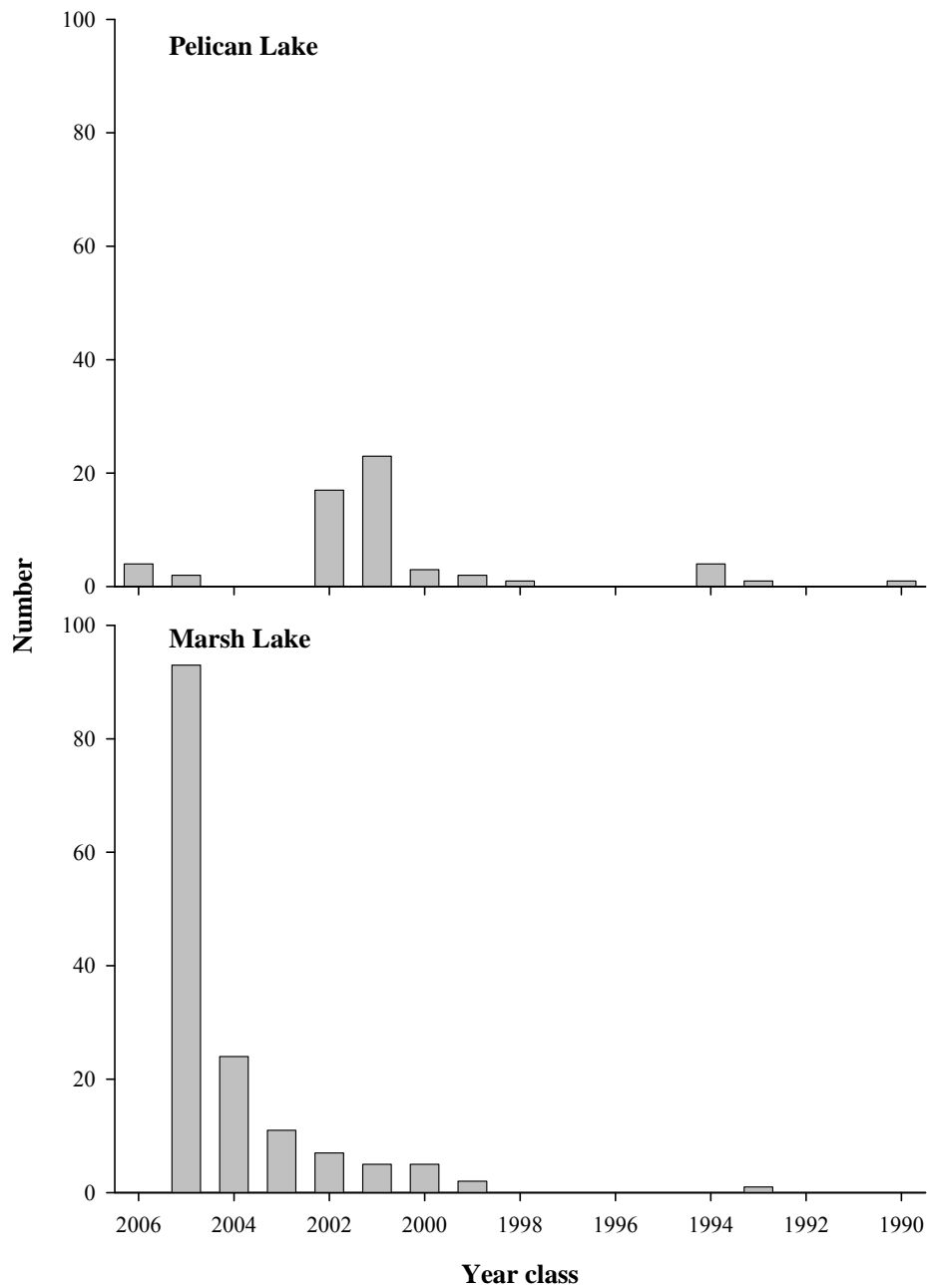


Figure 2-2. Age frequency for common carp sampled from Pelican and Marsh lakes, Nebraska in April and July of 2006, respectively.

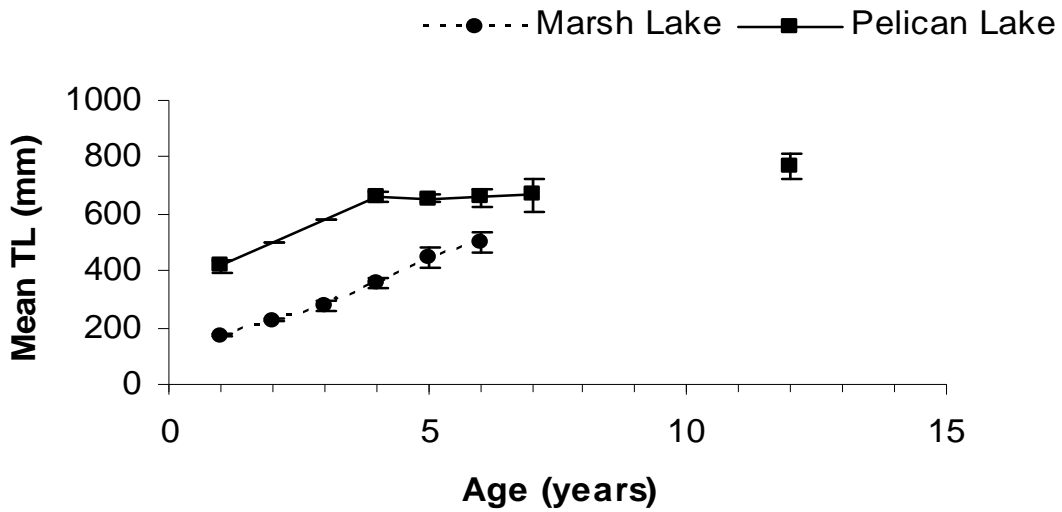


Figure 2-3. Mean total length (TL) (\pm SE) by cohort at time of capture for common carp in Marsh and Pelican lakes, Nebraska in 2006.

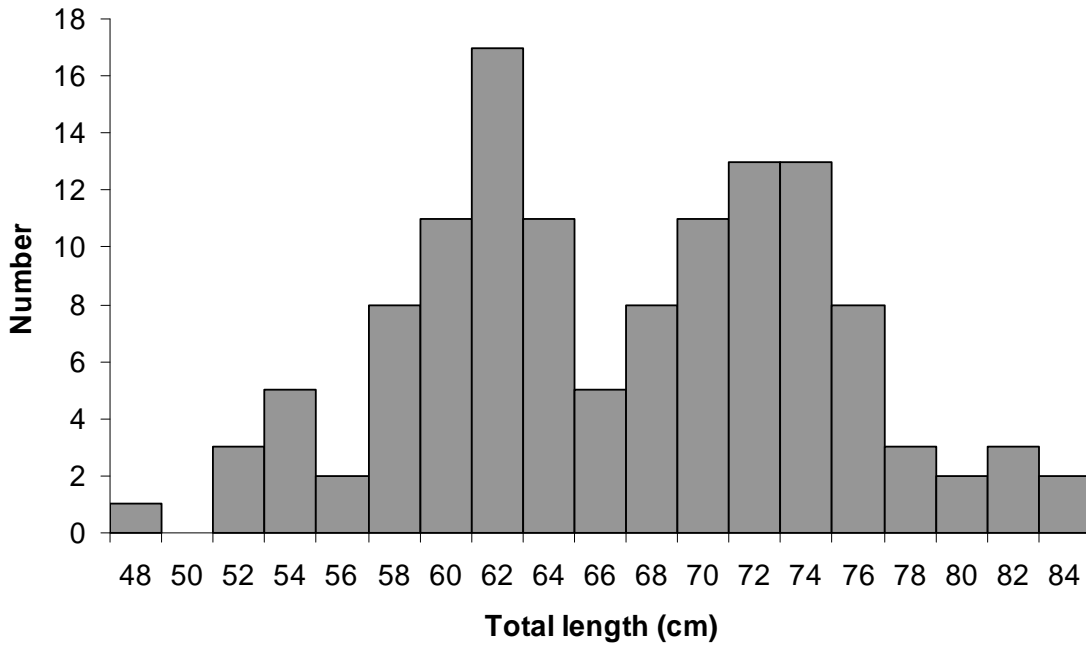


Figure 2-4. Length frequency for northern pike sampled from Hackberry Lake, Nebraska in June and July 2004.

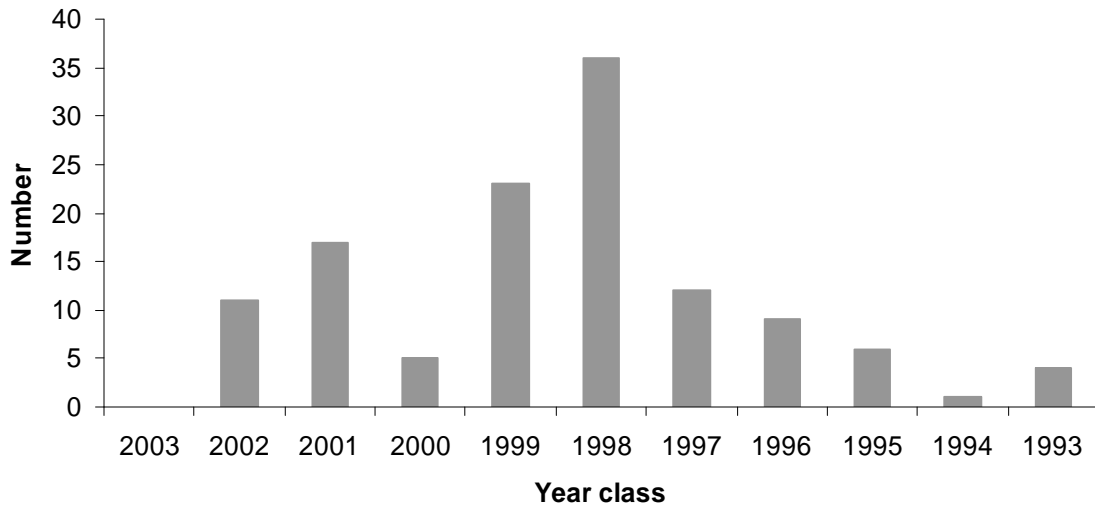


Figure 2-5. Age frequency for northern pike sampled from Hackberry Lake, Nebraska in July 2004.

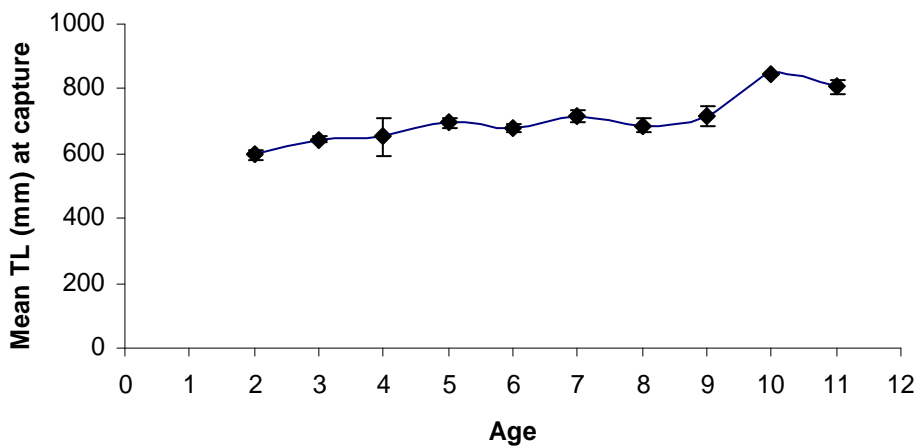


Figure 2-6. Mean observed total length (\pm SE) at time of capture by cohort for northern pike in Hackberry Lake, Nebraska in 2004.

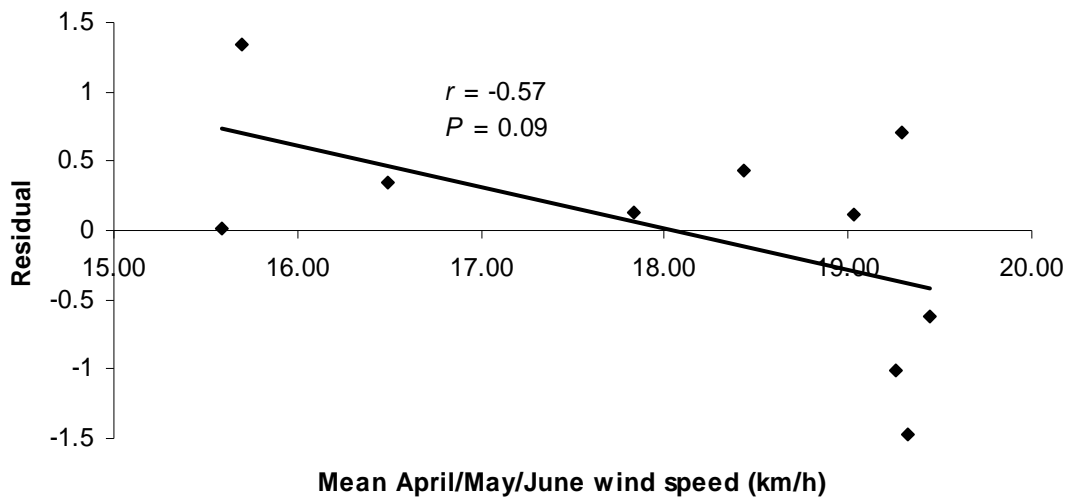


Figure 2-7. Plot of mean cumulative wind speed (km/h) for April, May, and June as a function of year-class strength (i.e., residual) for northern pike from Hackberry Lake, Nebraska.

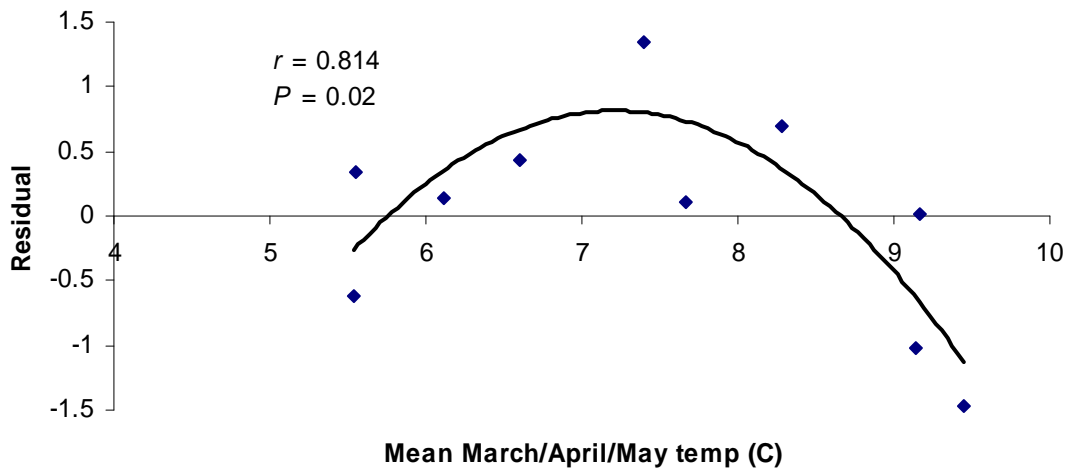


Figure 2-8. Plot of mean cumulative air temperature for March, April, and May as a function of year class strength (i.e., residual) for northern pike from Hackberry Lake, Nebraska.

(4) JOB OBJECTIVE 4:

To assess invertebrate communities in fishless Sandhill lakes, and compare those data to estimates collected from lakes containing fish communities.

Introduction

Fish communities dominated by overly abundant planktivores can negatively influence size structure and abundance of invertebrates (Mills and Schiavone 1982; Mills et al. 1987). Paukert and Willis (2003a) summarized abundance estimates for zooplankton and macroinvertebrates in 30 Nebraska Sandhill lakes that contained fish communities. A comparison between the 30-lake summaries and zooplankton and macroinvertebrate communities in fish-less lakes on the Valentine National Wildlife Refuge should provide an initial understanding of the potential effect of fishes on these productive water bodies. The original objective for this study proved to be unattainable as fishless study waters could not be located, which we reported in the annual performance report for this study. The objective for this study was modified to compare invertebrate populations in a manipulated Sandhill lake with those in a lake with a fish community dominated by piscivores.

Study areas

Pony Lake is a 64-ha, shallow (mean depth = 1.0 m) natural lake in Cherry County of the Sandhills region of north central Nebraska (McCarragher 1977). Submergent and emergent vegetation coverage was low (< 23% total coverage) in 2004 through 2008. The fish community was simple, primarily composed of common carp,

black bullhead, fathead minnow, golden shiner, and grass pickerel. Notably the lake contained no top-level piscivores (i.e., largemouth bass and northern pike). The lake is closed to angling.

West Long Lake is a 25-ha, shallow (mean depth = 1.3 m) natural lake in Cherry County of the Sandhills region of north central Nebraska within the Valentine National Wildlife Refuge (McCarragher 1977), and served as a “control” lake. Total vegetation coverage (emergent and submergent) was high (>90% coverage). The fish community was relatively simple and comprised primarily of bluegill, largemouth bass, yellow perch, northern pike, black bullhead, common carp, and fathead minnow. Piscivores were abundant and control planktivorous fish communities (DeBates et al. 2003; Jolley et al. 2008). The lake watershed is mostly mid and tall-grass prairie and limited livestock grazing is allowed. A detailed description of the study lakes is provided in Appendix 1.

Methods

Benthic macroinvertebrates

Two replicate benthic macroinvertebrate samples were collected at each site, seasonally (i.e., April, July, and October) strained through a 583- μ m mesh sieve in the field, and stored in 90% ethanol. Replicate samples were collected and processed separately. Macroinvertebrates were identified to order and enumerated using a dissecting microscope in the laboratory. The most common macroinvertebrate taxa were reported and less common taxa were pooled into an “other” category. Up to 20 individuals of each taxon were measured (mm TL) and all individuals were counted. Density was then calculated by dividing the number of benthic invertebrates of each

taxon by the area sampled with the Ekman grab (i.e., 231-cm²). I then used taxon-specific, length-dry weight equations (Eckblad 1971; Dumont and Balvay 1979; Smock 1980; Benke et al. 1999) to convert length to biomass. Biomass was not estimated for oligochaetes due to a lack of reliable length-weight conversions. Potential differences in mean density within months, among years were assessed with a one-way ANOVA. Post-hoc comparisons were performed using the Student-Newman Keuls multiple range test. Alpha levels were adjusted using the Bonferroni correction to account for year*month interactions.

Zooplankton

Zooplankton was collected during the daytime seasonally (i.e., April, July, and October) as two replicates at each site using a 2-m long tube sampler (Rabeni 1996). Samples were filtered through a 65- μ m mesh net and stored in 90% ethanol. Benthic macroinvertebrates were sampled at the same time and sites using a 231-cm² Ekman grab sampler.

Zooplankton were enumerated and identified to family for cladocerans (i.e. Bosminidae, Chydoridae, and Daphnidae), and as cyclopoid or calanoid copepods, copepod nauplii, ostracods, and rotifers. Other uncommon taxa were also noted. Each sample was diluted with water to a measured volume of 30 mL. Three subsamples were then taken with a 5-mL Hensen-Stempel pipette and placed in a Ward counting wheel. Up to 20 individuals of each category were measured (mm TL) and all individuals were counted. The total number of zooplankton of each taxon in a sample was calculated by dividing the number of organisms counted by the proportion of the sample volume

processed. Density was then calculated by dividing the number of zooplankters of each taxon by the volume of the water filtered with the tube sampler. We then used taxon-specific, length-dry weight equations (Dumont et al. 1975; Mason 1977; McCauley and Kalff 1981; Culver et al. 1985; Lynch et al. 1986) to convert length to biomass. Potential differences in mean density within months, among years were assessed with a one-way ANOVA. Post-hoc comparisons were performed using the Student-Newman Keuls multiple range test. Alpha levels were adjusted using the Bonferonni correction to account for year*month interactions.

Phytoplankton

Phytoplankton biomass was indexed using chlorophyll-*a* estimated from replicate water samples at each site using a 2-m long tube sampler seasonally (i.e., April, July, and October). Water (100 mL) was filtered through filters (Whatman glass fiber filters, 0.45 μm) in the field and extracted in the laboratory following the methods described by Lind (1985). Two samples of unfiltered lake water were collected at each site and frozen for later determination of total phosphorous following the methods of Wetzel and Likens (1991). From the same sampling locations, total alkalinity and pH were measured using a water chemistry analysis kit (Hach Company, Loveland, CO), and total dissolved solids (TDS) were recorded 0.5 m below the surface using an electronic meter (Hach Company, Loveland, CO). Secchi disk transparency was measured at each site during daytime and dissolved oxygen (DO) was measured 0.5 m below the surface using an electronic DO meter (Fisher Scientific, Pittsburgh, PA). Water temperature was measured hourly with

an *in situ* automatic temperature logger (Onset Computer Corporation, Pocasset, MA) and expressed as the mean temperature/24-h period.

Vegetation

Submergent and emergent vegetation coverage was quantified annually during July. Sampling was conducted following the methods outlined by Paukert et al. (2002a) and values were expressed as percent coverage of different vegetation types (i.e., emergent or submergent). Total percent vegetation coverage was examined for differences within lakes, among years using a Chi-square multiple comparison technique (PROC GENMOD and MULTTEST procedures, SAS 9.1 2002; Zar 1984).

Fish removal

The fish community in Pony Lake was removed in October 2007 with a rotenone treatment. Rotenone was delivered at a target rate of 5 ppm using powder and liquid formulations. Population estimates were obtained for all species occurring in the lake. Shoreline transects (100 m) were walked and all fish were identified, counted, and a subsample (n=150) were measured (TL in mm). The mean number per transect was extrapolated to the entire shoreline to obtain population estimates. The mean weight of each species was estimated using the species-specific standard weight equation and then extrapolated to the population estimate to obtain the estimated population biomass. Age was estimated for 100 common carp and all grass pickerel collected using sagittal otoliths to examine the age structure. A Peterson mark-recapture estimate was also conducted for common carp; 100 common carp were initially collected via electrofishing prior to the

application and marked with a pectoral fin clip. The recovery period for the population estimate then occurred after the treatment when dead fish were collected and examined for the fin clip.

(e) Target Date for Achievement:

Sampling and data collection for this objective was completed in October 2008, although the study is ongoing through Segments 11-15. All data analysis through 2008 is complete.

(f) Date of Accomplishment:

Activities are proceeding on schedule.

(g) Significant Deviations:

The original objective for this study proved to be unattainable as fishless study waters could not be located, which we reported in the annual performance report for this study. The objective for this study was modified to compare invertebrate populations in a manipulated Sandhill lake with those in a lake with a fish community dominated by piscivores.

(h) Remarks:

None.

Results

Benthic invertebrates

The benthic invertebrate community of Pony Lake was relatively homogenous among years and generally contained seven categories of dominant taxa (Table 3-1) although chironomids and amphipods were the only taxa occurring in substantial numbers. All other taxa were very low in density. In contrast, the benthic invertebrate community of West Long Lake was quite diverse and dense (Table 3-2). Amphipods, chironomids, and trichopterans were generally the most abundant categories. Results of the two-way ANOVAs on mean density by lake revealed that West Long had more dense populations of chironomids ($P < 0.01$, $F = 8.69$, $df = 1$), amphipods ($P < 0.01$, $F = 17.77$, $df = 1$), and trichopterans ($P < 0.01$, $F = 43.30$, $df = 1$).

The invertebrate community indexed by sweep nets in Pony Lake was also low in abundance (Table 3-3). Hemipterans, amphipods, and gastropods were primarily captured in sweep nets. Qualitatively, it appeared that the abundance of several taxa of invertebrates in sweep net samples began to increase in 2008 following the fish community removal. For example, amphipods and trichopterans were rarely present in substantial numbers prior to 2008. In addition, live gastropods were rarely caught until 2008 when they were relatively plentiful (Table 3-3). Hemipterans in the form of Notonectids and Corixids began to increase in number. However, fishes were still present as the rotenone treatment was not successful (i.e., several fish species remained after the treatment).

The invertebrate community of West Long Lake indexed by sweep nets was much more diverse (Table 3-3). Mean catch per sweep of amphipods ranged from 8 to nearly

700 individuals per sweep and of hemipterans over 150 individuals per sweep (Table 3-3).

Zooplankton

The zooplankton community of Pony Lake was largely dominated by copepods in the spring and cladocerans (i.e., *Daphnia*, *Ceriodaphnia*, and *Bosmina*) in the summer and fall. Results of the two-way ANOVAs of density (Table 3-4) for the different zooplankton categories by year and season (i.e., spring, summer, and fall) indicated varying densities in every instance as well as significant year*month interactions ($P < 0.01$). *Post-hoc* one-way analyses of variance were performed for each zooplankton category within each month to examine yearly differences (Bonferroni correction, $\alpha = 0.01$). Of 18 different statistical examinations, 17 of those had significant differences in density within seasons, among years (Table 3-4). Individual-year, among-month differences were not examined due to the expected dynamic nature of zooplankton populations.

The zooplankton community of West Long Lake was largely dominated by cladocerans (i.e., *Daphnia* and *Ceriodaphnia*). Results of the two-way ANOVAs of density (Table 3-5) for the different zooplankton categories by year and season (i.e., spring, summer, and fall) indicated varying densities in every instance as well as significant year*month interactions ($P < 0.01$). *Post-hoc* one-way analyses of variance were performed for each zooplankton category within each month to examine yearly differences (Bonferroni correction, $\alpha = 0.01$). Of 18 different statistical examinations, 13 of those had significant differences in density within seasons, among years (Table 3-5).

Individual-year, among-month differences were not examined due to the expected dynamic nature of zooplankton populations.

Chlorophyll-*a* concentrations in Pony Lake were substantially higher in all seasons than those in West Long Lake, typically more than 5-fold in difference (Table 3-6). The physical and chemical characteristics of Pony and West Long Lake are summarized in Tables 4-6 – 4-8.

Fish removal

The fish community consisted of adult common carp (234 kg/ha), fathead minnow (30 kg/ha), black bullhead (28 kg/ha), green sunfish (17 kg/ha), juvenile common carp (12 kg/ha), golden shiner (2 kg/ha), grass pickerel, yellow perch, and pumpkinseed (all less than 1 kg/ha; Table 3-1). The shoreline transect method estimated a common carp population of 5,412 fish while the Peterson mark-recapture estimate was 3,969.

Thirty-two grass pickerel were collected from Pony Lake (Jolley and Willis 2008). Total length ranged from 132 to 247 mm; mean weight was 35 g. The population estimate was 198 (95% CI \pm 22) grass pickerel with a mean density of 3 (95% CI \pm 0.35) fish/ha. The population biomass estimate was 7 (95% CI \pm 0.88) kg, with a corresponding standing stock of 0.11 (95% CI \pm 0.02) kg/ha.

High biomass of common carp may have reduced submersed vegetation in Pony Lake; our biomass estimate for common carp was 245.7 kg/ha. Common carp ranged from age 1 to age 18 and from 364 to 890 mm TL (Figure 3-1). According to state and federal agency biologists who completed the Pony Lake restoration with us, grass pickerel are more abundant in Nebraska lakes with abundant submersed vegetation.

Thus, our density and biomass estimates may be toward the low end of the potential for grass pickerel. Several studies reported higher densities and biomasses of grass pickerel (Eschemeyer and Clark 1939, Carbine and Applegate 1948, Kleinert and Mraz 1966, Schneider 1993).

The grass pickerel age structure that we obtained indicated that recruitment was relatively consistent among years. However, we are not certain why no individuals from the 2007 year class were collected. During our renovation we collected many species and sizes of fishes, including small minnows. Thus, we suspect that the 2007 year class was missing.

Grass pickerel growth was difficult to interpret because of the lack of available comparison information. The mean total lengths of age-2 through age-5 fish were 166, 172, 178, and 224 mm. Others have reported substantially faster growth rates for grass pickerel (Kleinert and Mraz 1966, Carlander 1969). We did not determine gender of our sampled specimens, which might have helped to explain overlapping lengths among length groups. Grass pickerel ranged from age 1 to age 4 years and from 132 to 247 mm TL (Figure 3-2).

Exploratory fish sampling in the summer of 2008 was conducted to confirm that the rotenone treatment was effective. A variety of species were captured in experimental gill nets, mini-fyke nets, and cloverleaf traps. These included green sunfish, pumpkinseed, golden shiner, grass pickerel, and black bullhead. The fish community of Pony Lake will be fully assessed in the summer of 2009 to characterize the fish populations.

Discussion

The primary value of this study from 2004 to 2008 is the comparison of the lower-abundance invertebrate community in a lake with a planktivore-dominated fish community (i.e., Pony Lake) compared with the invertebrate community in the piscivore-dominated fish community (i.e., West Long Lake). The zooplankton community in Pony Lake was dominated by copepods in spring and a variety of sizes of cladocerans in summer and fall. In contrast, zooplankton samples from West Long Lake were dominated by large cladocerans. Both the ekman dredge and sweep-net samples exhibited substantially higher abundances of macroinvertebrates in West Long Lake when compared with Pony Lake. The chlorophyll-*a* values were much higher at Pony Lake compared with West Long Lake, indicating the higher abundance of phytoplankton in the water body with the planktivore-dominated fish community. The abundant predators in West Long Lake (Jolley et al. 2008) reduced abundance of planktivorous/insectivorous fishes, resulting in larger and more abundant aquatic invertebrates, confirming previous research (Potthoff et al. 2008; Ward et al. 2008). These findings have obvious implications for waterfowl management on the refuge.

In addition, in-depth information regarding the biomass of abundant populations of common carp in Nebraska Sandhill lakes is limited, and our results add to the body of knowledge. Age-structure analysis indicated that year classes recruited to the adult common carp population in Pony Lake during most years, but that recruitment patterns were erratic. The common carp population biomass was estimated at 234 kg/ha, indicating that this moderate carp standing stock had the potential to negatively affect

invertebrate communities, submergent vegetation, and water quality. Future research to link common carp biomass with effects on the lake characteristics would be valuable.

The grass pickerel is a small, native esocid found in the Sandhills lakes and streams. Population structure and density information for this fish species is scarce, and our study provided valuable information on age structure and growth for this rarely-studied species. In addition, we might preliminarily conclude that this small predator could not efficiently control planktivorous fishes on a refuge lake.

Finally, an additional rotenone application is being planned for Pony Lake in 2009 to fully remove the fish community. We should be able to obtain at least one year of information on invertebrate abundance estimate without any fish predation. However, the “current” long-term plan for the lake is to assess the characteristics of a bluegill-only fish population. If the bluegill overpopulate (i.e., high density, slow growth), that would provide evidence that some type of biotic control must influence bluegill recruitment in Nebraska Sandhill lakes. If the bluegill do not overpopulate in the absence of a predator (e.g., largemouth bass), the environment may be regulating bluegill recruitment in some manner (e.g., wind, waves, temperature fluctuations, or lack of appropriate habitat).

Research for this objective thus will continue in segments 11-15.

Acknowledgments

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Table 3-1. Mean density (number/m²) of primary benthic invertebrate taxa in Pony Lake, Nebraska 2004-2008. Standard errors are in parentheses.

Lake	Year	Month	Order Ephem- eroptera	Order Odonata	Order Trichoptera	Order Amphipoda	Class Hirudinea	Class Gastropoda	Family Chiron- omidae
Pony	2004	Jul	0 (0)	0 (0)	12.7 (12.7)	1.8 (1.8)	0 (0)	0 (0)	54.4 (13.7)
		Sep	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	358.7 (101.0)
	2005	Apr	0 (0)	0 (0)	0 (0)	16.3 (10.4)	0 (0)	0 (0)	97.8 (38.1)
		Jul	0 (0)	0 (0)	0 (0)	10.9 (10.9)	0 (0)	0 (0)	173.9 (72.6)
		Sep	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	103.2 (20.6)
	2006	Apr	0 (0)	0 (0)	0 (0)	5.4 (5.4)	0 (0)	0 (0)	298.9 (119.5)
		Jul	0 (0)	10.9 (6.3)	0 (0)	27.1 (20.6)	5.4 (5.4)	0 (0)	250.0 (105.9)
		Sep	0 (0)	0 (0)	0 (0)	10.9 (10.9)	0 (0)	5.4 (5.4)	190.2 (22.4)
	2007	Apr	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	521.8 (51.8)
		Jul	0 (0)	0 (0)	0 (0)	27.2 (27.2)	0 (0)	0 (0)	2,152.3 (257.5)
		Sep	0 (0)	0 (0)	5.4 (5.4)	0 (0)	0 (0)	0 (0)	81.5 (10.4)
	2008	Jan	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	32.6 (20.8)
		Apr	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	65.2 (51.0)
		Jul	0 (0)	0 (0)	0 (0)	92.4 (43.8)	0 (0)	0 (0)	16.3 (10.4)
		Sep	0 (0)	0 (0)	5.4 (5.4)	97.8 (59.2)	228.3 (146.5)	0 (0)	10.9 (6.3)

Table 3-2. Mean density (number/m²) of primary benthic invertebrate taxa in West Long Lake, Nebraska 2004-2008. Standard errors are in parentheses.

Lake	Year	Month	Order Ephem- eroptera	Order Odonata	Order Trichoptera	Order Amphipoda	Class Hirudinea	Class Gastropoda	Family Chiron- omidae
West Long	2004	Jul	105.1 (30.4)	54.5 (10.9)	391.3 (131.9)	8,612.7 (3,254.2)	217.4 (123.1)	692.1 (340.7)	11,906.3 (9,027.3)
		Sep*	-	-	-	-	-	-	-
	2005	Apr	0 (0)	152.2 (145.0)	2,163.1 (2,148.6)	804.4 (653.7)	43.5 (19.8)	32.6 (20.8)	6,005.7 (2,346.1)
		Jul	10.9 (10.9)	114.1 (27.2)	87.0 (67.0)	2,038.1 (713.1)	27.2 (20.6)	65.2 (58.2)	483.7 (184.3)
		Sep	0 (0)	10.9 (6.3)	27.2 (10.4)	179.4 (75.5)	10.9 (10.9)	0 (0)	103.3 (44.7)
	2006	Apr	76.1 (76.1)	364.1 (220.0)	119.6 (69.6)	1,494.6 (348.3)	59.8 (59.8)	141.3 (141.3)	7,668.8 (2,881.0)
		Jul	16.3 (5.4)	21.7 (12.6)	27.2 (13.7)	1,250.1 (574.5)	21.7 (15.4)	16.3 (10.4)	2,771.9 (1,556.0)
		Sep	5.4 (5.4)	429.4 (125.0)	804.4 (528.1)	17,902.9 (7,498.8)	108.7 (58.9)	32.6 (25.9)	2,239.2 (1,711.4)
	2007	Apr	5.4 (5.4)	43.5 (23.5)	2,994.7 (2,374.8)	701.1 (520.1)	152.2 (60.8)	0 (0)	1,342.5 (237.4)
		Jul	21.7 (21.7)	43.5 (8.9)	10.9 (6.3)	4,636.1 (1,584.0)	81.5 (16.3)	684.8 (437.7)	1,815.3 (321.7)
		Sep	103.3 (68.4)	337.0 (215.5)	1,945.7 (780.0)	4,858.9 (1,862.1)	380.5 (110.0)	712.0 (560.0)	820.7 (479.4)
	2008	Jan	97.8 (32.6)	10.9 (10.9)	7,652.5 (2,819.3)	4,793.7 (3,302.4)	76.1 (37.1)	141.3 (74.0)	3,293.6 (2,301.3)
		Apr	0 (0)	27.1 (27.1)	5,837.2 (2,125.1)	467.4 (257.9)	16.3 (10.4)	0 (0)	2,565.3 (959.8)
		Jul	21.7 (15.4)	0 (0)	206.5 (67.9)	2,418.6 (1,361.0)	54.5 (28.8)	0 (0)	108.7 (67.0)
			Sep	32.6 (32.6)	32.6 (32.6)	21.7 (21.7)	2,565.3 (1,891.4)	43.5 (43.5)	0(0)

*The September sample was not collected in West Long Lake due to inclement weather

Table 3-3. Mean catch per effort (number/sweep) of primary invertebrate taxa in Pony Lake and West Long Lake, Nebraska 2004-2008. Standard errors are in parentheses.

Lake	Year	Month	Order Ephem- eroptera	Order Odonata	Order Trichoptera	Order Hemiptera	Order Amphipoda	Class Gastropoda
Pony	2004*	Jul	-	-	-	-	-	-
		Sep	-	-	-	-	-	-
	2005	Apr	0.8 (0.8)	0.5 (0.5)	0 (0)	0.3 (0.3)	3.2 (1.6)	0 (0)
		Jul	0.1 (0.1)	0.5 (0.5)	0.1 (0.1)	12.8 (9.0)	2.8 (1.9)	0.1 (0.1)
		Sep	0 (0)	0.4 (0.2)	0 (0)	2.5 (0.3)	0.5 (0.4)	0 (0)
	2006	Apr	0 (0)	0.1 (0.1)	0.1 (0.1)	0.1 (0.1)	0.3 (0.3)	0 (0)
		Jul	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
		Sep	0.3 (0.3)	2.3 (1.0)	0 (0)	1 (0.7)	0.1 (0.1)	0 (0)
	2007	Apr	0.3 (0.3)	0 (0)	0 (0)	0 (0)	0.6 (0.6)	0 (0)
		Jul	0.4 (0.1)	4.4 (2.2)	0 (0)	26.1 (9.3)	4.4 (0.3)	3.1 (2.8)
		Sep	0.1 (0.1)	0.3 (0.1)	0 (0)	0.9 (0.7)	0.9 (0.4)	0 (0)
	2008	Apr	0 (0)	0 (0)	0 (0)	0.6 (0.4)	0 (0)	0 (0)
		Jul	2.5 (1.4)	0.5 (0.3)	0 (0)	9.4 (2.7)	53.1 (29.7)	23.9 (7.3)
Sep		1.1 (0.6)	1.4 (0.7)	0.3 (0.1)	40.0 (8.5)	77.6 (16.3)	2.9 (1.0)	
West Long	2004*	Jul	-	-	-	-	-	-
		Sep	-	-	-	-	-	-
	2005	Apr	1.5 (0.6)	11.8 (10.4)	1.6 (1.6)	0.4 (0.2)	81.8 (39.0)	1.6 (0.9)
		Jul	4.4 (1.9)	0.9 (0.7)	0 (0)	0.6 (0.4)	195.9 (30.8)	37.5 (16.9)
		Sep	5.0 (3.1)	7.3 (1.7)	0.9 (0.4)	1.3 (0.8)	110.9 (54.1)	15.8 (1.3)
	2006	Apr	0.8 (0.5)	2.8 (1.8)	1.8 (1.6)	0.9 (0.4)	40.5 (24.4)	0.4 (0.4)
		Jul	6.3 (1.7)	8.1 (3.0)	4.6 (3.3)	16.7 (12.8)	699.9 (224.5)	39.4 (36.4)
		Sep	2.1 (1.2)	4.9 (2.1)	19.1 (12.5)	84.3 (45.3)	501.4 (218.9)	2.6 (0.6)
	2007	Apr	0.1 (0.1)	0.3 (0.3)	0 (0)	2.0 (0.6)	26.1 (5.0)	1.9 (0.8)
		Jul	5.5 (5.0)	25.1 (22.6)	0.6 (0.3)	155.6 (127.0)	346.6 (312.4)	3.9 (1.7)
		Sep	17.1 (14.3)	18.9 (13.2)	7.3 (4.6)	26.3 (10.1)	66.9 (41.9)	98.5 (55.7)
	2008	Apr	0.4 (0.2)	0.4 (0.4)	29.0 (1.8)	0 (0)	8.3 (3.5)	0.6 (0.2)
		Jul	0.1 (0.1)	0.4 (0.1)	29.6 (10.4)	15.6 (3.9)	115.4 (54.7)	1.0 (0.1)
Sep		0.5 (0.5)	0.5 (0.5)	123.0 (83.2)	104.9 (79.0)	363.6 (264.5)	1.5 (1.3)	

*Sweep nets were not used in 2004

Table 3-4. Mean density (number/m³) of primary zooplankton taxa in Pony Lake, Nebraska 2004-2008. Standard errors are in parentheses. Mean density in columns followed by the same letter were significantly different (within seasons, among years; $P < 0.05$); those without letters were not different.

Year	Season	<i>Bosmina</i>	Copepoda	Cladocera	Copepod nauplii	Rotifera	Total
2005	Spring	315.3a (49.1)	236.1c (46.9)	5.5a (1.9)	214.1 (95.2)	85.2c (28.6)	856.9a (159.5)
2006		4.41c (1.3)	446.8a (26.7)	0.7c (0.3)	141.7 (33.9)	75.0d (28.2)	668.6a (52.5)
2007		172.6b (23.3)	351.6b (56.1)	3.5b (1.5)	174.2 (8.1)	146.5b (35.9)	848.6a (106.7)
2008		0.2c (0.2)	115.5d (21.1)	3.1c (0.8)	128.2 (20.9)	207.7a (22.9)	454.8b (53.0)
2004		46.6c (7.4)	142.7ab (28.1)	18.9c (5.1)	29.9d (5.4)	0.2b (0.2)	238.3c (35.8)
2005	Summer	1364.3a (378.9)	175.9a (23.9)	127.1a (48.9)	145.7a (24.1)	1683.5a (375.0)	3503.6a (714.6)
2006		659.1b (234.8)	149.7a (30.7)	208.0b (38.9)	78.3bc (23.4)	115.4b (53.2)	1238.1b (273.9)
2007		583.7b (132.2)	83.7bc (26.7)	573.2b (154.1)	110.5ab (18.7)	11.8b (4.4)	1363.9b (261.0)
2008		0.2c (0.2)	60.1b (10.2)	343.5c (18.0)	67.2dc (18.6)	0.4b (0.2)	481.8c (42.4)
2004		301.1b (104.2)	47.9b (12.9)	2.9cb (2.1)	26.5b (9.1)	5.2c (0.7)	383.5cb (104.2)
2005	Fall	261.6b (79.2)	54.6b (13.9)	26.0cb (8.8)	24.8b (13.0)	47.3ab (31.8)	416.1cb (131.3)
2006		57.1c (9.2)	44.8b (2.4)	23.8c (6.3)	28.0b (4.5)	6.2bc (2.2)	166.3c (16.0)
2007		412.5a (51.5)	75.4b (18.5)	62.4b (19.1)	39.5ab (8.6)	22.6a (5.8)	612.9b (92.6)
2008		45.1c (6.4)	235.9a (46.7)	1285.9a (249.1)	56.5a (13.9)	4.4c (3.1)	1630.7a (294.2)

Table 3-5. Mean density (number/m³) of primary zooplankton taxa in West Long Lake, Nebraska 2004-2008. Standard errors are in parentheses. Mean density in columns followed by the same letter were significantly different (within seasons, among years; $P < 0.05$); those without letters were not different.

Year	Season	<i>Bosmina</i>	Copepoda	Cladocera	Copepod nauplii	Rotifera	Total
2005	Spring	0.7 (0.7)	12.9b (5.3)	522.1a (73.3)	11.8b (4.6)	121.7c (53.6)	672.2b (30.8)
2006		2.3 (1.5)	139.2a (86.3)	349.9b (59.7)	157.3a (35.4)	191.4b (110.2)	845.1b (260.4)
2007		9.3 (8.1)	17.9b (5.2)	71.9c (9.8)	4.2b (2.5)	0.5c (0.3)	113.2c (13.9)
2008		0.0 (0.0)	89.4a (8.3)	4.9c (1.7)	167.6a (10.1)	870.9a (113.4)	1136.5a (120.4)
2004	Summer	41.9 (39.3)	40.4a (13.4)	176.7a (82.2)	23.5b (7.7)	0.0b (0.0)	284.2a (74.2)
2005		10.0 (10.0)	20.3b (4.6)	146.4ab (68.1)	15.1bc (3.8)	0.2b (0.2)	201.3ab (73.2)
2006		19.3 (17.4)	6.9b (3.0)	50.7b (12.2)	10.6bc (4.1)	0.9b (0.4)	110.7b (15.3)
2007		26.1 (20.8)	44.0a (10.6)	123.3ab (17.0)	42.2a (9.1)	45.9b (27.4)	282.4a (24.3)
2008		0.5 (0.3)	9.1b (3.6)	134.1ab (36.3)	7.9a (0.6)	29.4a (19.8)	185.5ab (35.4)
2004	Fall*	-	-	-	-	-	-
2005		0.9 (0.6)	14.9 (7.9)	4.6b (3.1)	3.2 (1.6)	0.2b (0.2)	24.7b (12.4)
2006		8.5 (4.7)	14.5 (14.5)	138.5a (46.5)	12.1 (7.9)	9.6b (2.7)	189.0a (49.5)
2007		3.3 (2.9)	13.6 (3.6)	58.8ab (40.7)	3.4 (2.0)	33.0a (19.4)	120.8ab (35.7)
2008		1.1 (0.8)	8.5 (3.7)	147.8a (75.1)	15.9 (5.7)	16.0b (1.5)	197.4a (85.0)

*The fall sample was not collected in West Long Lake due to inclement weather

Table 3-6. Mean total phosphorous ($\mu\text{g/L}$) and chlorophyll-a ($\mu\text{g/L}$) in Pony Lake and West Long Lake, Nebraska 2004-2008.

Lake	Season	Year	Mean	
			Mean total phosphorous ($\mu\text{g/L}$)	chlorophyll-a ($\mu\text{g/L}$)
Pony	Spring	2005	26.9 (1.6)	67.9 (13.0)
		2006	21.4 (0.4)	163.9 (20.3)
		2007	20.9 (0.3)	87.3 (5.3)
		2008	19.1 (0.3)	24.1 (1.0)
	Summer	2004	18.6 (0.2)	40.9 (17.7)
		2005	25.4 (3.4)	159.8 (5.3)
		2006	22.5 (1.8)	121.1 (8.2)
		2007	30.6 (6.2)	149.8 (12.1)
		2008	33.6 (3.5)	244.9 (115.3)
	Fall	2004	75.3 (34.9)	22.4 (1.5)
		2005	25.0 (2.6)	133.7 (10.7)
		2006	30.0 (2.1)	173.3 (12.9)
2007		-	-	
2008		56.6 (1.5)	19.2 (1.5)	
West Long	Spring	2005	60.7 (5.2)	5.6 (1.2)
		2006	23.1 (0.6)	21.6 (1.1)
		2007	89.0 (13.9)	4.3 (0.7)
		2008	37.8 (3.0)	26.2 (3.4)
	Summer	2004	21.1 (1.0)	1.9 (1.4)
		2005	50.9 (5.9)	4.9 (1.4)
		2006	68.8 (14.0)	2.8 (0.3)
		2007	33.0 (3.5)	5.0 (2.3)
		2008	143.3 (7.6)	26.0 (1.2)
	Fall	2005	19.5 (0.7)	4.3 (0.5)
		2006	102.7 (14.7)	6.5 (1.3)
		2007	-	-
		2008	69.7 (5.9)	2.7 (0.6)

Table 3-7. Physical and chemical characteristics of Pony Lake, 2004-2008.

Lake	Season	Year	Mean Secchi depth (cm)	Mean dissolved oxygen (mg/L)	Mean conductivity ($\mu\text{S}/\text{cm}$)	Mean TDS (g/L) ¹	Mean alkalinity (mg/L)	Mean pH
Pony	Spring	2005	28 (2)	8.1 (0.1)	408 (3)	570 (5)	222 (7)	8.8 (0.1)
		2006	22 (2)	7.6 (0.1)	1,420 (341)	2,580 (696)	171 (7)	9.0 (0.2)
		2007	30 (1)	9.7 (0.1)	273 (3)	280 (0)	158 (4)	9.0 (0.0)
		2008	40 (4)	10.4 (0.2)	275 (3)	290 (5)	136 (7)	6.2 (1.8)
	Summer	2004	15 (1)	4.2 (0.5)	410 (0)	420 (0)	152 (4)	8.1 (0.1)
		2005	16 (1)	8.8 (0.1)	263 (3)	275 (5)	1368 (0)	8.9 (0.1)
		2006	17 (1)	9.7 (0.1)	418 (3)	475 (5)	162 (5)	8.3 (0.1)
		2007	19 (1)	26.3 (18.2)	300 (0)	300 (0)	136 (0)	7.5 (0.0)
		2008	50 (14)	7.3 (0.2)	368 (8)	385 (15)	175 (4)	8.1 (0.1)
	Fall	2004	15 (2)	10.8 (1.0)	235 (10)	245 (12)	162 (8)	9.8 (0.1)
		2005	17 (1)	10.2 (0.1)	910 (630)	300 (0)	205 (7)	8.5 (0.2)
		2006	14 (1)	8.4 (0.5)	415 (5)	435 (5)	248 (5)	8.0 (0.0)
		2007	18 (1)	-	343 (20)	380 (40)	102 (0)	9.4 (0.1)
		2008	37 (2)	8.8 (0.5)	310 (0)	320 (0)	119 (0)	8.5 (0.0)

¹Total dissolved solids

Table 3-8. Physical and chemical characteristics of West Long Lake, 2004-2008.

Lake	Season	Year	Mean Secchi depth (cm)	Mean dissolved oxygen (mg/L)	Mean conductivity ($\mu\text{S}/\text{cm}$)	Mean TDS (g/L) ¹	Mean alkalinity (mg/L)	Mean pH
West Long	Spring	2005	76 (4)	7.3 (0.1)	588 (124)	785 (253)	248 (5)	8.5 (0.0)
		2006	60 (7)	7.3 (0.2)	905 (92)	1,505 (169)	180 (5)	9.3 (0.1)
		2007	-	9.9 (0.2)	361 (3)	372 (3)	-	8.0 (0.0)
		2008	38 (2)	9.9 (0.2)	330 (7)	340 (8)	145 (5)	7.9 (0.1)
	Summer	2004	-	8.2 (0.1)	625 (5)	630 (10)	192 (4)	7.8 (0.1)
		2005	-	8.1 (0.1)	373 (5)	380 (0)	145 (9)	8.4 (0.1)
		2006	-	9.3 (0.2)	478 (3)	530 (6)	197 (11)	8.1 (0.1)
		2007	-	7.3 (0.1)	320 (15)	330 (17)	133 (4)	9.9 (0.1)
		2008	58 (6)	9.1 (0.1)	403 (9)	365 (19)	145 (5)	8.5 (0.2)
	Fall	2005	-	10.6 (0.1)	378 (3)	395 (5)	235 (8)	8.0 (0.0)
		2006	-	-	353 (3)	360 (0)	115 (4)	8.3 (0.1)
		2007	-	14.8 (1.4)	370 (10)	400 (20)	197 (5)	8.5 (0.0)
		2008	-	9.6 (0.4)	345 (5)	355 (5)	107 (4)	8.0 (0.0)

¹T total dissolved solids

Table 3-9. Population density and biomass estimates for the fish community in Pony Lake, Nebraska revealed during rotenone renovation, October 2007.

Species	Number/m (SE)	Total abundance	Density (number/ha)	Mean weight (g)	Total biomass (kg)	Standing stock (kg/ha)
Black bullhead	7.63 (4.01)	25,179	393.42	71.33	1796.02	28.06
Adult common carp	1.64 (0.44)	5,412	84.56	2,769.79	14990.10	234.22
Grass pickerel	0.06 (0.03)	198	3.09	35.12	6.95	0.11
Green sunfish	14.80 (7.00)	48,840	763.13	22.39	1093.53	17.09
Golden shiner	2.13 (0.32)	7,029	109.83	17.26	121.32	1.90
Juvenile carp	4.85 (1.01)	16,005	250.08	46.03	736.71	11.51
Pumpkinseed	0.04 (0.03)	132	2.06	24.44	3.23	0.05
Yellow perch	0.01 (0.01)	33	.52	132.33	4.37	0.07
Fathead minnow	805.53 (21.74)	2,658,249	41,535.14	0.72	1913.94	29.91
Totals	-	2,761,077	43,141.83	-	20,666.17	322.91

Figure 3-1. Age- and length-frequency histograms for common carp collected from Pony Lake, Nebraska, 2007, after rotenone application.

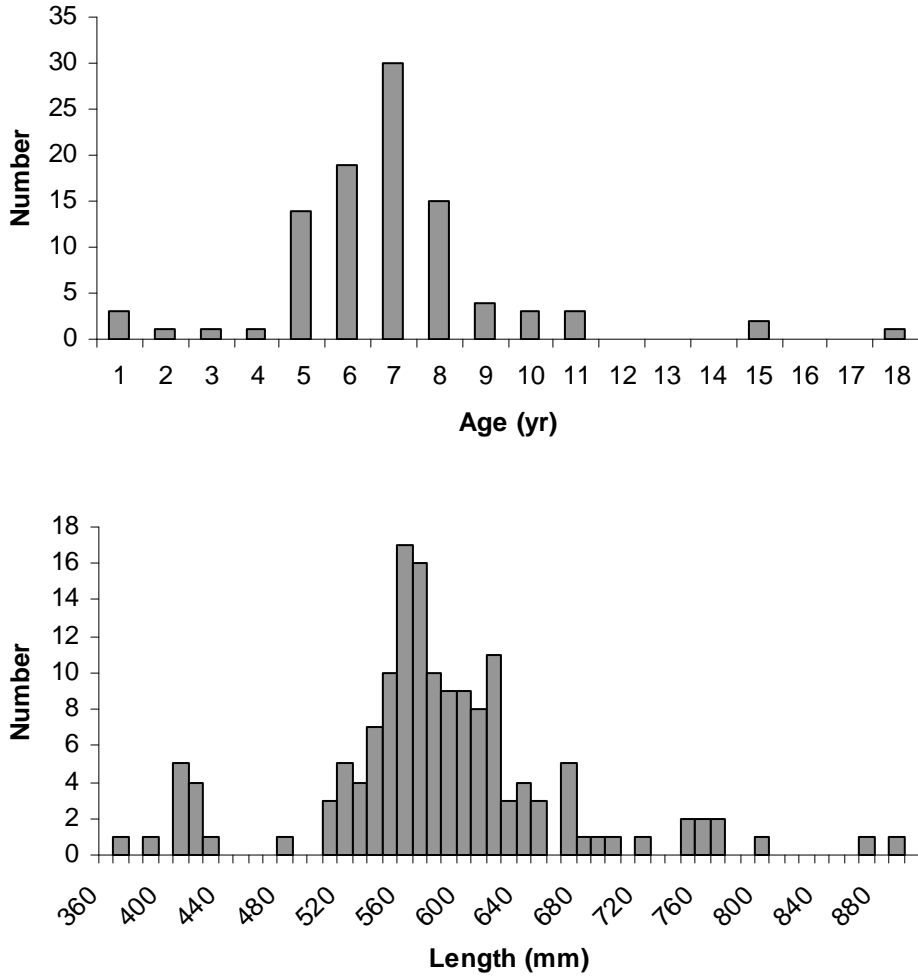
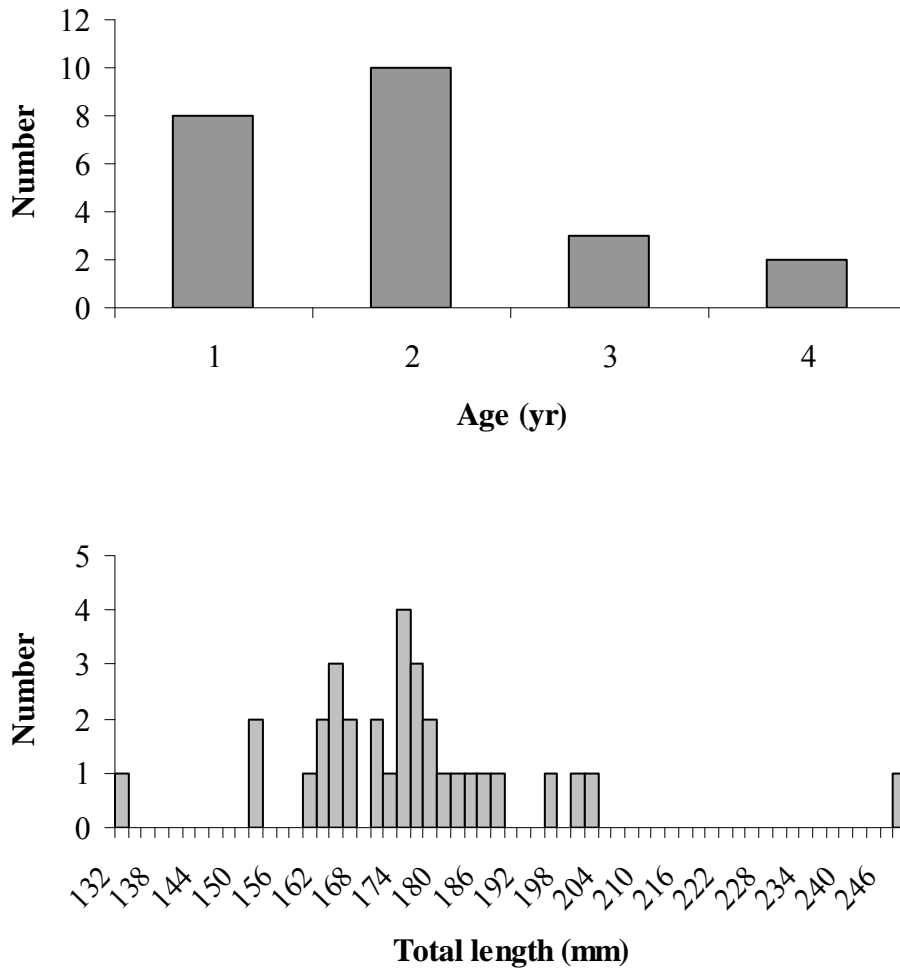


Figure 3-2. Age- and length-frequency histogram for grass pickerel collected from Pony Lake, Nebraska, 2007, after rotenone application.



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Appendix 1. Legal description and physical characteristics of lakes sampled in 2004-2009.

Lake	County	Latitude	Longitude	Surface area (ha)	Mean depth (m)	Maximum depth (m)
Cameron	Rock	42.38	99.42	39	1.8	2.9
Hackberry	Cherry	42.56	100.67	275	1.5	2.1
Marsh	Cherry	42.53	100.51	907	1.2	1.9
Pelican	Cherry	42.53	100.67	332	1.3	2.8
Pony	Cherry	42.49	100.51	64	1.0	1.8
West Long	Cherry	42.53	100.72	25	1.3	1.8

Appendix 2. Mean density of larval bluegill (number/100 m³) and number of trawls (N) from Pelican Lake, 2004-2008. Standard errors are in parentheses. Inshore locations are <100 m from emergent vegetation and offshore locations are ≥100 m from emergent vegetation.

Year	Date	Inshore		Offshore		Overall	
		Mean density (number/100 m ³)	N	Mean density (number/100 m ³)	N	Mean density (number/100 m ³)	N
2004	6-Jun	0 (0)	5	0 (0)	5	0 (0)	10
	16-Jun	0 (0)	5	0 (0)	5	0 (0)	10
	26-Jun	1 (1)	6	3 (2)	4	2 (1)	10
	7-Jul	9 (3)	5	9 (2)	5	9 (2)	10
	17-Jul	140 (59)	5	91 (21)	5	116 (31)	10
	27-Jul	52 (33)	4	94 (34)	6	77 (24)	10
	6-Aug	27 (15)	4	93 (38)	4	60 (23)	8
	16-Aug	120 (53)	5	169 (80)	5	144 (46)	10
	27-Aug	36 (9)	5	126 (26)	5	81 (20)	10
2005	5-Sep	23 (13)	4	52 (9)	4	37 (9)	8
	2-Jun	0 (0)	5	0 (0)	5	0 (0)	10
	13-Jun	0 (0)	5	0 (0)	4	0 (0)	9
	22-Jun	0 (0)	4	0 (0)	5	0 (0)	9
	30-Jun	122 (58)	5	67 (21)	5	95 (30)	10
	11-Jul	413 (135)	5	173 (60)	5	293 (80)	10
	21-Jul	551 (263)	5	1,297 (661)	5	924 (358)	10
	1-Aug	219 (65)	5	338 (92)	5	278 (57)	10
	11-Aug	95 (23)	5	89 (11)	5	92 (12)	10
2006	22-Aug	36 (14)	5	21 (8)	5	28 (8)	10
	29-May	0 (0)	4	0 (0)	5	0 (0)	9
	8-Jun	24 (12)	5	0 (0)	5	12 (7)	10
	19-Jun	60 (17)	4	81 (19)	6	72 (13)	10
	29-Jun	111 (32)	5	114 (40)	5	113 (24)	10
	10-Jul	643 (273)	5	353 (104)	5	498 (146)	10
	20-Jul	1,894 (1,224)	5	1,633 (684)	5	1,764 (662)	10
	1-Aug	1,113 (157)	5	1,687 (384)	5	1,400 (218)	10
	10-Aug	202 (69)	5	320 (133)	5	261 (73)	10
2007	4-Jun	0 (0)	5	0 (0)	5	0 (0)	10
	14-Jun	0 (0)	5	0 (0)	5	0 (0)	10
	25-Jun	113 (40)	5	256 (143)	5	185 (74)	10
	5-Jul	425 (48)	5	642 (176)	5	533 (93)	10
	16-Jul	600 (147)	5	864 (110)	5	732 (97)	10
	26-Jul	163 (40)	5	382 (108)	5	273 (65)	10
	6-Aug	11 (2)	5	7 (2)	5	9 (2)	10
	15-Aug	3 (2)	5	3 (2)	5	3 (1)	10
	19-Jun	0 (0)	5	0 (0)	5	0 (0)	10
2008	30-Jun	42 (26)	5	37 (27)	5	40 (18)	10
	8-Jul	1,472 (315)	5	661 (95)	5	1,066 (206)	10
	17-Jul	2,234 (710)	5	2,355 (571)	5	2,294 (430)	10
	28-Jul	848 (318)	5	832 (324)	5	840 (214)	10
	7-Aug	203 (73)	5	291 (66)	5	247 (49)	10
	19-Aug	356 (194)	5	131 (29)	5	243 (100)	10

Appendix 3. Mean density of larval yellow perch (number/100 m³) and number of trawls (N) from Cameron Lake, 2004-2005, and Pelican Lake, 2004-2007. Standard errors are in parentheses. Inshore locations are <100 m from emergent vegetation and offshore locations are ≥100 m from emergent vegetation.

Lake	Year	Date	Inshore		Offshore		Overall	
			Mean density (number/100 m ³)	N	Mean density (number/100 m ³)	N	Mean density (number/100 m ³)	N
Cameron	2004	29-Apr	-	-	-	-	782 (235)	11
		13-May	1,929 (899)	5	1,379 (220)	4	1,685 (492)	9
		18-May	985 (488)	4	999 (535)	4	992 (335)	8
		27-May	33 (22)	4	94 (48)	4	64 (27)	8
		7-Jun	308 (179)	4	225 (192)	4	267 (122)	8
	2005	15-May	7 (4)	4	3 (1)	4	5 (2)	8
		24-May	10 (5)	4	24 (10)	4	17 (6)	8
		2-Jun	2 (2)	4	5 (2)	4	4 (1)	8
Pelican	2004	18-Apr	-	-	-	-	0 (-)	1
		8-May	166 (23)	6	208 (66)	5	185 (31)	11
		17-May	168 (98)	5	36 (28)	2	131 (72)	7
		26-May	16 (11)	3	7 (2)	5	11 (4)	8
		6-Jun	0 (0)	5	0 (0)	5	0 (0)	10
	2005	21-Apr	-	-	0 (-)	1	0 (-)	1
		3-May	9 (5)	5	2 (1)	5	5 (3)	10
		14-May	0 (0)	5	0 (0)	5	0 (0)	10
	2006	9-May	33 (10)	5	77 (16)	5	55 (12)	10
		18-May	266 (102)	5	156 (68)	5	211 (61)	10
		29-May	7 (3)	5	2 (1)	5	5 (2)	10
		8-Jun	4 (4)	5	8 (8)	5	6 (4)	10
		19-Jun	0 (0)	4	0 (0)	6	0 (0)	10
	2007	30-Apr	77 (3)	5	103 (28)	5	90 (14)	10
		8-May	513 (213)	5	533 (210)	5	523 (141)	10
		18-May	5 (4)	5	14 (12)	5	9 (6)	10
		24-May	2 (1)	5	2 (1)	5	2 (1)	10
		4-Jun	0.45 (0.45)	5	0 (0)	5	0.22 (0.22)	10
		14-Jun	0 (0)	5	0 (0)	5	0 (0)	10
	2008	22-Apr	0 (0)	5	0 (0)		0 (0)	9
28-Apr		4 (2)	5	1 (0.6)		3 (1)	10	
8-May		223 (56)	4	174 (24)		196 (28)	9	
19-May		246 (79)	5	386 (77)		316 (57)	10	
29-May		968 (474)	5	1,298 (602)		1,133 (365)	10	
9-Jun		17 (2)	5	17 (5)		17 (2)	10	
19-Jun		1 (0.3)	5	1 (0.5)		1 (0.3)	10	
	30-Jun	0 (0)	5	0 (0)		0 (0)	10	

Appendix 4. Mean daily growth (mm/d), standard error (SE), and sample size (N) of larval yellow perch in Cameron Lake, 2004-2005, and Pelican Lake, 2004-2008 and for larval bluegill by hatch period in Pelican Lake, 2004-2008. Hatch periods for bluegill are early (i.e., June) and late (i.e., July and August).

Lake	Species	Year	Hatch period	Mean daily growth (mm/d)	SE	N
Cameron	Yellow perch	2004	Overall	0.35	0.02	75
		2005		0.27	0.02	24
Pelican	Yellow perch	2004		0.37	0.01	50
		2005		0.33	0.05	10
		2006		0.30	0.01	59
		2007		0.23	0.01	65
		2008		0.18	0.01	68
		2008		0.18	0.01	68
Pelican	Bluegill	2004	Early	0.22	0.01	43
		2005		0.37	0.01	60
		2006		0.31	0.01	82
		2007		0.20	0.01	85
		2008		0.19	0.01	84
		2004	Late	0.33	0.01	106
		2005		0.32	0.01	88
		2006		0.39	0.02	49
		2007		0.29	0.02	8
		2008		0.37	0.01	87

Appendix 5. Bivariate Pearson correlation matrix between bluegill and yellow perch life stage indices. Contrasts correspond to the same year class of fish. Correlation coefficients (r) are followed in parentheses by number of data pairs and P value. Catch per unit effort is denoted CPUE.

	Bluegill peak larval abundance	Bluegill fall CPUE	Bluegill spring (age 1) CPUE	Bluegill age 2 CPUE	Yellow perch peak larval abundance	Yellow perch fall CPUE	Yellow perch spring (age 1) CPUE	Yellow perch age 1 CPUE
Bluegill peak larval abundance		-0.11 (5, 0.87)	0.67 (4, 0.33)	0.93 (4, 0.07)	0.61 (5, 0.27)	-0.55 (5, 0.33)	0.20 (4, 0.80)	0.33 (5, 0.58)
Bluegill fall CPUE	-0.11 (5, 0.87)		0.87 (4, 0.13)	-0.17 (4, 0.83)	-0.72 (5, 0.16)	0.05 (5, 0.93)	-0.53 (4, 0.47)	0.68 (5, 0.20)
Bluegill spring (age 1) CPUE	0.67 (4, 0.33)	0.87 (4, 0.13)		0.95 (3, 0.21)	-0.10 (4, 0.90)	0.12 (4, 0.88)	-0.06 (4, 0.94)	0.90 (4, 0.11)
Bluegill age 2 CPUE	0.93 (4, 0.07)	-0.17 (4, 0.83)	0.95 (3, 0.21)		0.71 (4, 0.29)	-0.99 (4, 0.01)	-0.17 (3, 0.89)	0.02 (4, 0.98)
Yellow perch peak larval abundance	0.61 (5, 0.27)	-0.73 (5, 0.16)	-0.10 (4, 0.90)	0.71 (4, 0.29)		-0.15 (7, 0.74)	-0.04 (6, 0.95)	-0.21 (5, 0.73)
Yellow perch fall CPUE	-0.55 (5, 0.33)	0.05 (5, 0.93)	0.12 (4, 0.88)	-0.99 (4, 0.01)	-0.15 (7, 0.74)		0.59 (6, 0.22)	0.16 (5, 0.80)
Yellow perch spring (age 1) CPUE	0.20 (4, 0.80)	-0.53 (4, 0.47)	-0.06 (4, 0.95)	-0.17 (3, 0.89)	-0.04 (6, 0.95)	0.59 (6, 0.22)		0.29 (4, 0.71)
Yellow perch age 1 CPUE	0.33 (5, 0.58)	0.68 (5, 0.20)	0.89 (4, 0.11)	0.02 (4, 0.98)	0.21 (5, 0.73)	0.16 (5, 0.80)	0.29 (4, 0.71)	

Appendix 6. Mean biomass (g/m²) of major invertebrate taxa in Pelican Lake and Cameron Lake, Nebraska 2004-2008. Standard errors are in parentheses.

Lake	Year	Date	Order Ephemeroptera	Order Odonata	Order Trichoptera	Order Amphipoda	Class Hirudinea	Class Gastropoda	Family Chironomidae
Pelican	2004	18-Apr	-	0.44 (0.44)	0.08 (0.08)	0.18 (0.18)	0.03 (0.03)	-	1.01 (0.31)
		8-May	0.01 (0.01)	-	-	-	-	-	2.01 (0.37)
		17-May	0.01 (0.01)	-	-	0.08 (0.08)	0.01 (0.01)	-	1.92 (0.40)
		26-May	0.01 (0.01)	0.03 (0.02)	-	-	0.02 (0.01)	-	2.87 (0.51)
		6-Jun	-	-	-	0.01 (0.01)	0.06 (0.02)	-	3.60 (1.13)
		16-Jun	0.01 (0.01)	0.04 (0.04)	-	0.04 (0.03)	0.16 (0.04)	35 (15)	2.34 (0.75)
		26-Jun	-	-	-	-	0.05 (0.02)	-	9.37 (2.25)
		7-Jul	-	-	-	-	0.09 (0.02)	-	10.37 (2.54)
		17-Jul	-	-	-	0.03 (0.03)	0.32 (0.10)	13 (13)	3.95 (0.85)
		27-Jul	0.02 (0.01)	-	0.01 (0.01)	0.10 (0.10)	0.23 (0.12)	10 (5)	1.56 (0.47)
	6-Aug	0.02 (0.01)	-	-	-	0.08 (0.06)	28 (28)	4.61 (1.01)	
	16-Aug	-	-	-	-	0.08 (0.03)	-	0.80 (0.24)	
	27-Aug	-	0.01 (0.01)	-	-	0.08 (0.04)	20 (20)	0.02 (0.01)	
	5-Sep	-	0.80 (0.43)	-	0.29 (0.19)	0.60 (0.39)	98 (78)	1.10 (0.53)	
	2005	3-May	-	-	-	0.02 (0.01)	0.05 (0.02)	29 (29)	1.37 (0.23)
		14-May	-	-	0.01 (0.01)	0.03 (0.02)	0.02 (0.01)	4 (4)	0.55 (0.15)
		23-May	-	0.04 (0.03)	-	0.06 (0.03)	0.13 (0.06)	291 (251)	0.50 (0.11)
		2-Jun	-	-	-	-	0.03 (0.01)	11 (7)	0.18 (0.05)
		13-Jun	0.03 (0.03)	0.15 (0.08)	-	0.03 (0.02)	0.18 (0.07)	11 (11)	0.59 (0.29)
		22-Jun	-	-	-	-	0.03 (0.01)	8 (8)	1.21 (0.40)
30-Jun		0.01 (0.01)	-	-	-	0.04 (0.02)	-	2.02 (0.83)	
11-Jul		-	-	-	0.02 (0.01)	0.10 (0.03)	-	5.42 (0.86)	
21-Jul		-	-	-	-	0.01 (0.01)	-	7.16 (2.31)	
1-Aug		-	-	-	-	0.15 (0.07)	19 (19)	9.54 (1.47)	
2006	11-Aug	-	0.02 (0.02)	-	0.11 (0.10)	0.19 (0.07)	10 (10)	1.13 (0.30)	
	22-Aug	-	-	-	-	0.11 (0.04)	-	2.18 (0.87)	
	9-May	-	0.02 (0.02)	-	0.01 (0.01)	1.60 (0.66)	-	2.37 (0.46)	
	18-May	-	-	-	-	0.85 (0.20)	-	2.12 (0.48)	
	29-May	-	0.07 (0.07)	0.02 (0.01)	0.04 (0.03)	0.64 (0.24)	25 (17)	0.93 (0.28)	
	8-Jun	0.01 (0.01)	0.05 (0.03)	-	0.03 (0.01)	2.34 (1.07)	-	1.36 (0.41)	
	19-Jun	-	-	0.01 (0.01)	0.01 (0.01)	0.78 (0.51)	30 (16)	1.33 (0.67)	
	29-Jun	-	-	0.01 (0.01)	0.01 (0.01)	3.00 (1.14)	-	0.63 (0.12)	
	10-Jul	-	-	-	0.01 (0.01)	3.89 (1.18)	22 (14)	0.51 (0.12)	
	20-Jul	-	0.01 (0.01)	-	-	2.74 (2.05)	2 (2)	0.33 (0.10)	
2007	1-Aug	-	-	-	-	1.26 (0.41)	53 (53)	0.46 (0.17)	
	10-Aug	-	-	-	0.02 (0.02)	0.86 (0.36)	-	0.80 (0.24)	
	30-Apr	0.01 (0.01)	-	-	0.02 (0.01)	0.01 (0.01)	-	1.97 (0.47)	
	8-May	-	-	-	0.01 (0.01)	0.02 (0.01)	24 (18)	1.25 (0.26)	
	18-May	-	-	-	-	0.01 (0.01)	-	0.39 (0.11)	
	24-May	0.01 (0.01)	-	0.01 (0.01)	-	0.04 (0.02)	-	0.51 (0.18)	
	4-Jun	0.02 (0.01)	-	0.01 (0.01)	0.02 (0.02)	0.20 (0.05)	2 (2)	0.94 (0.30)	
	14-Jun	-	-	-	-	0.14 (0.06)	-	0.88 (0.30)	
	25-Jun	0.03 (0.03)	-	-	0.03 (0.03)	0.11 (0.07)	0.85 (0.85)	0.88 (0.31)	
	5-Jul	-	-	-	-	0.02 (0.01)	-	0.49 (0.13)	
16-Jul	-	-	-	-	0.02 (0.01)	-	0.62 (0.16)		
26-Jul	-	-	-	-	0.05 (0.02)	-	0.32 (0.05)		
6-Aug	-	-	-	-	0.01 (0.01)	-	0.65 (0.35)		
15-Aug	-	-	-	-	0.01 (0.01)	-	0.19 (0.03)		

Appendix 6 continued

Lake	Year	Date	Order Ephem- eroptera	Order Odonata	Order Trichoptera	Order Amphipoda	Class Hirudinea	Class Gastropoda	Family Chiron- omidae
Pelican	2008	22-Apr	-	-	-	-	0.34 (0.24)	-	1.05 (.13)
		28-Apr	-	-	-	-	-	-	1.60 (0.53)
		8-May	-	0.02 (0.02)	-	-	0.03 (0.03)	-	0.95 (0.31)
		19-May	0.01 (0.01)	0.01 (0.01)	0.01 (0.01)	-	0.13 (0.05)	-	0.29 (0.10)
		29-May	-	-	-	-	-	-	0.17 (0.08)
		9-Jun	0.01 (0.01)	-	-	-	-	-	0.08 (0.03)
		19-Jun	-	-	-	-	-	-	0.69 (0.22)
		30-Jun	-	-	-	-	0.02 (0.02)	-	2.58 (0.83)
		8-Jul	-	-	0.01 (0.01)	-	0.21 (0.21)	-	2.02 (0.71)
		17-Jul	-	-	-	-	0.01 (0.01)	-	3.49 (1.06)
		28-Jul	-	-	-	-	-	-	1.45 (0.44)
		7-Aug	-	-	-	-	-	-	1.02 (0.27)
		19-Aug	-	-	-	-	-	-	1.23 (0.21)
		Cameron	2004	13-May	-	-	-	-	-
18-May	-			-	0.01 (0.01)	-	-	-	2.48 (0.43)
27-May	-			-	-	-	-	-	0.82 (0.20)
2005	2-Jun		-	0.02	-	-	-	-	-
	7-Jun		-	-	-	-	-	-	0.47 (0.17)
	15-May		-	-	-	-	-	-	0.35 (0.07)
	2005	24-May	-	-	-	0.01 (0.01)	-	-	0.18 (0.04)
		2-Jun	-	-	-	-	-	-	0.90 (0.16)

Appendix 7. Physical and chemical characteristics of Pelican Lake and Cameron Lake, 2004-2008.

Lake	Year	Date	Mean Secchi depth (cm)	Mean dissolved oxygen (mg/L)	Mean conductivity ($\mu\text{S}/\text{cm}$)	Mean TDS (g/L) ¹	Mean alkalinity (mg/L)	Mean pH
Pelican	2004	8-May	-	8.4 (0.1)	417 (3)	-	239.40 (3.12)	8.0 (-)
		17-May	-	9.0 (0.1)	888 (57)	1356.7 (113.4)	245.10 (3.60)	8.2 (0.1)
		26-May	86 (11)	7.1 (0.0)	849 (247)	604.4 (60.9)	254.60 (4.46)	8.1 (0.1)
		6-Jun	-	8.7 (0.0)	384 (5)	392.0 (5.33)	239.40 (-)	8.5 (-)
		16-Jun	93 (4)	8.9 (0.2)	392 (4)	396.0 (2.67)	235.98 (2.28)	8.0 (-)
		26-Jun	94 (4)	7.8 (0.2)	390 (2)	400.0 (-)	210.33 (2.61)	8.1 (0.1)
		7-Jul	106 (3)	8.5 (0.1)	372 (6)	384.0 (6.53)	218.88 (2.28)	8.2 (0.1)
		16-Jul	102 (4)	7.1 (0.0)	372 (2)	380.0 (-)	230.85 (3.82)	8.1 (0.1)
		27-Jul	75 (6)	8.4 (0.1)	394 (3)	404.0 (2.67)	210.33 (5.13)	8.3 (0.1)
		6-Aug	53 (2)	8.8 (0.4)	388 (3)	400.0 (5.35)	226.58 (4.27)	8.5 (-)
		16-Aug	55 (3)	6.7 (0.2)	426 (16)	472.0 (30.58)	222.30 (-)	8.5 (-)
		27-Aug	40 (2)	8.3 (0.1)	658 (18)	864.0 (31.52)	213.75 (2.85)	8.9 (0.1)
	5-Sep	43 (3)	9.5 (0.0)	690 (80)	1035.0 (168.13)	190.24 (2.14)	9.1 (0.1)	
	2005	3-May	130 (6)	8.8 (0.1)	343 (4)	358.0 (2.00)	165.87 (2.61)	8.0 (-)
		14-May	54 (6)	7.1 (0.2)	-	-	159.03 (2.61)	8.1 (0.1)
		23-May	80 (16)	7.0 (0.1)	-	532.0 (16.11)	147.49 (3.13)	8.3 (0.1)
		2-Jun	-	5.1 (0.6)	-	-	169.29 (1.71)	8.1 (0.1)
		13-Jun	111 (18)	6.9 (0.4)	-	-	147.49 (3.13)	8.0 (-)
		22-Jun	152 (-)	6.2 (0.4)	-	-	167.58 (3.42)	8.0 (0.1)
		30-Jun	109 (3)	7.7 (0.1)	-	-	169.29 (3.07)	7.9 (0.1)
		11-Jul	124 (11)	8.1 (0.1)	-	-	182.97 (2.61)	8.1 (0.1)
		21-Jul	78 (2)	4.8 (0.7)	566 (71)	728.0 (137.44)	230.85 (3.82)	8.3 (0.1)
1-Aug		94 (8)	7.0 (0.1)	446 (24)	524.0 (48.52)	215.46 (3.78)	8.7 (0.1)	
11-Aug	70 (2)	8.4 (0.1)	702 (29)	1028.0 (55.87)	254.79 (3.99)	8.2 (0.1)		
22-Aug	55 (4)	7.0 (0.0)	494 (75)	632.0 (142.46)	159.03 (3.65)	8.8 (0.1)		
2006	9-May	47 (34)	5.3 (0.3)	506 (18)	620.0 (32.11)	199.50 (2.85)	8.2 (0.1)	
	18-May	-	6.0 (0.6)	866 (190)	1324.0 (369.52)	253.08 (4.27)	8.4 (0.1)	
	29-May	100 (5)	7.4 (0.0)	415 (8)	438.0 (17.24)	242.82 (3.42)	8.4 (0.1)	
	8-Jun	52 (3)	7.2 (0.1)	506 (6)	628.0 (13.40)	266.76 (4.56)	8.2 (0.1)	
	19-Jun	87 (4)	7.6 (0.1)	471 (16)	586.0 (33.21)	251.37 (3.65)	8.6 (0.1)	

Appendix 7 continued.

Lake	Year	Date	Mean Secchi depth (cm)	Mean dissolved oxygen (mg/L)	Mean conductivity (µS/cm)	Mean TDS (g/L) ¹	Mean alkalinity (mg/L)	Mean pH	
Pelican	2006	29-Jun	54 (2)	6.8 (0.3)	402 (24)	470.0 (46.88)	225.72 (4.27)	9.0 (0.1)	
		10-Jul	52 (3)	7.4 (0.0)	421 (19)	528.0 (37.97)	213.75 (2.85)	8.8 (0.1)	
		20-Jul	30 (1)	4.5 (0.2)	364 (3)	408.0 (6.80)	217.17 (3.65)	9.0 (-)	
		2-Aug	21 (1)	7.2 (0.2)	342 (2)	352.0 (3.27)	222.30 (2.55)	9.3 (0.1)	
		10-Aug	23 (1)	7.5 (0.1)	462 (22)	602.0 (41.79)	191.52 (4.27)	9.5 (-)	
	2007	30-Apr	102 (5)	7.9 (0.2)	376 (2)	384.0 (2.67)	133.38 (4.27)	8.0 (-)	
		8-May	152 (4)	7.0 (0.0)	372 (1)	380.0 (-)	159.03 (2.61)	8.3 (0.1)	
		18-May	59 (5)	7.5 (0.1)	414 (17)	428.0 (19.82)	160.74 (2.80)	8.7 (0.1)	
		24-May	82 (6)	6.9 (0.1)	371 (3)	394.0 (6.00)	147.06 (3.78)	8.9 (0.1)	
		4-Jun	70 (6)	7.5 (0.0)	625 (104)	898.0 (201.36)	155.61 (3.99)	8.8 (0.1)	
		14-Jun	66 (3)	7.7 (0.3)	746 (48)	1106.0 (92.43)	162.45 (3.82)	8.3 (0.1)	
		25-Jun	38 (2)	7.6 (0.1)	651 (66)	886.0 (114.66)	150.48 (3.42)	8.6 (0.1)	
		5-Jul	37 (1)	7.5 (0.0)	764 (41)	1204.0 (88.98)	275.31 (7.84)	9.0 (0.1)	
		16-Jul	23 (1)	7.8 (0.3)	639 (137)	944.0 (269.75)	128.25 (2.85)	9.0 (0.1)	
		26-Jul	23 (1)	6.0 (0.3)	368 (6)	414.0 (14.92)	160.74 (3.78)	9.3 (0.1)	
		6-Aug	22 (1)	7.3 (0.1)	412 (55)	504.0 (109.33)	159.03 (3.65)	9.5 (0.1)	
		15-Aug	17 (1)	6.8 (0.3)	342 (2)	368.0 (6.80)	167.58 (2.28)	9.3 (0.1)	
		2008	22-Apr	61 (3)	-	378 (1)	392.0 (3.3)	126.54 (9.26)	8.3 (0.1)
			28-Apr	82 (3)	8.2 (0.1)	402 (7)	440.0 (14.0)	143.64 (2.79)	8.3 (0.1)
			8-May	66 (5)	8.4 (0.0)	666 (54)	960.0 (100.8)	148.77 (2.61)	8.0 (0.0)
19-May	52 (1)		10.4 (0.9)	394 (9)	472.0 (40.9)	143.64 (2.79)	8.5 (0.0)		
29-May	40 (1)		15.1 (0.6)	820 (156)	1244.0 (294.2)	148.77 (2.61)	8.5 (0.1)		
9-Jun	50 (3)		13.4 (0.3)	720 (137)	1114.0 (284.4)	147.06 (2.79)	8.3 (0.1)		
19-Jun	85 (4)		13.2 (0.8)	815 (62)	1240 (120.0)	139.65 (2.85)	8.0 (0.0)		
30-Jun	112 (4)		9.7 (0.2)	476 (24)	572.0 (48.6)	148.77 (2.61)	8.0 (0.0)		
8-Jul	57 (3)		8.7 (0.4)	484 (29)	604.0 (57.7)	165.87 (4.45)	8.3 (0.1)		
17-Jul	45 (1)	8.6 (0.2)	436 (7)	480.0 (11.2)	162.45 (3.82)	8.1 (0.1)			
28-Jul	41 (1)	11.2 (0.7)	398 (5)	424.0 (7.8)	201.78 (6.14)	8.2 (0.1)			
7-Aug	35 (2)	10.3 (0.5)	388 (5)	412.0 (9.0)	215.46 (5.22)	8.5 (0.1)			
19-Aug	36 (1)	10.2 (0.1)	428 (15)	488.0 (26.2)	212.04 (5.81)	8.5 (0.1)			

Appendix 7 continued.

Lake	Year	Date	Mean Secchi depth (cm)	Mean dissolved oxygen (mg/L)	Mean conductivity (µS/cm)	Mean TDS (g/L) ¹	Mean alkalinity (mg/L)	Mean pH
Cameron	2004	13-May	21 (2)	7.9 (0.4)	600 (144)	844.0 (196.1)	129.96 (4.19)	8.5 (-)
		18-May	39 (2)	9.1 (0.0)	410 (29)	600.0 (45.7)	136.80 (-)	8.4 (0.1)
		27-May	52 (2)	7.8 (0.1)	260 (9)	290.0 (19.6)	162.45 (3.23)	7.9 (0.1)
	2005	7-Jun	24 (1)	7.7 (0.1)	225 (2)	230.0 (3.8)	153.90 (-)	8.1 (0.1)
		15-May	56 (3)	7.0 (0.1)	210 (-)	220.0 (-)	104.74 (3.87)	8.0 (-)
		24-May	54 (2)	7.9 (0.2)	223 (2)	230.0 (3.8)	111.15 (3.23)	8.4 (0.1)
		2-Jun	57 (2)	7.1 (0.1)	493 (35)	750.0 (56.2)	119.70 (-)	7.7 (0.1)

¹ Total dissolved solids

Appendix 8. Bivariate Pearson correlation matrix between bluegill and yellow perch abundance and growth indices and principal components (PC). Correlation coefficients (r) are followed in parentheses by number of data pairs and P value. Mean catch per unit effort is denoted CPUE and total length is denoted TL.

	Invertebrate		Physicochemical		Habitat		Zooplankton		Climate	
	PC1	PC2	PC1	PC2	PC1	PC2	PC1	PC2	PC1	PC2
Bluegill peak larval abundance	-0.79 (5, 0.11)	0.15 (5, 0.82)	-0.23 (5, 0.70)	-0.14 (5, 0.83)	0.71 (5, 0.18)	0.40 (5, 0.50)	0.59 (5, 0.29)	-0.22 (5, 0.72)	-0.56 (4, 0.44)	-0.50 (4, 0.50)
Bluegill fall CPUE	0.43 (5, 0.47)	-0.47 (5, 0.41)	0.04 (5, 0.95)	0.57 (5, 0.32)	-0.64 (5, 0.24)	-0.14 (5, 0.82)	-0.59 (5, 0.29)	0.61 (5, 0.28)	-0.13 (4, 0.87)	-0.99 (4, 0.01)
Bluegill spring (age 1) CPUE	-0.11 (4, 0.88)	-0.01 (4, 0.99)	-0.47 (4, 0.53)	0.57 (4, 0.43)	-0.73 (4, 0.27)	0.05 (4, 0.95)	0.21 (4, 0.79)	0.78 (4, 0.22)	-0.60 (4, 0.40)	-0.79 (4, 0.21)
Bluegill age 2 CPUE	-0.22 (3, 0.86)	0.39 (3, 0.74)	-0.99 (3, 0.06)	0.68 (3, 0.53)	-0.90 (3, 0.29)	0.10 (3, 0.93)	0.98 (3, 0.13)	0.97 (3, 0.16)	-0.94 (3, 0.23)	-0.84 (3, 0.36)
Yellow perch peak larval abundance	-0.66 (5, 0.22)	0.74 (5, 0.16)	-0.43 (5, 0.48)	-0.18 (5, 0.78)	0.84 (5, 0.08)	-0.14 (5, 0.82)	0.97 (5, 0.01)	-0.23 (5, 0.71)	-0.56 (4, 0.44)	-0.50 (4, 0.50)
Yellow perch fall CPUE	0.34 (5, 0.58)	0.34 (5, 0.57)	-0.63 (5, 0.25)	0.34 (5, 0.58)	-0.52 (5, 0.37)	-0.18 (5, 0.77)	0.15 (5, 0.81)	0.77 (5, 0.13)	-0.50 (4, 0.51)	0.33 (4, 0.67)
Yellow perch spring (age 1) CPUE	-0.76 (4, 0.24)	0.08 (4, 0.92)	-0.81 (4, 0.19)	-0.56 (4, 0.44)	0.02 (4, 0.98)	0.73 (4, 0.27)	0.86 (4, 0.14)	0.21 (4, 0.79)	-0.76 (4, 0.24)	0.63 (4, 0.37)
Yellow perch age 1 CPUE	-0.40 (4, 0.60)	-0.45 (4, 0.55)	-0.31 (4, 0.69)	0.22 (4, 0.78)	-0.34 (4, 0.66)	0.36 (4, 0.64)	0.01 (4, 0.99)	0.44 (4, 0.56)	-0.53 (4, 0.47)	-0.77 (4, 0.23)

Appendix 8 continued.

	Invertebrate		Physicochemical		Habitat		Zooplankton		Climate	
	PC1	PC2	PC1	PC2	PC1	PC2	PC1	PC2	PC1	PC2
Overall										
bluegill daily growth	0.32 (5, 0.60)	-0.39 (5, 0.51)	0.55 (5, 0.34)	0.45 (5, 0.45)	-0.11 (5, 0.87)	-0.29 (5, 0.63)	-0.54 (5, 0.35)	-0.01 (5, 0.99)	0.37 (4, 0.63)	-0.90 (4, 0.10)
Early-hatched										
bluegill daily growth	0.18 (5, 0.77)	-0.88 (5, 0.05)	0.57 (5, 0.31)	-0.11 (5, 0.86)	-0.37 (5, 0.54)	0.16 (5, 0.80)	-0.81 (5, 0.09)	-0.20 (5, 0.75)	0.42 (4, 0.58)	-0.57 (4, 0.43)
Late-hatched										
bluegill daily growth	-0.21 (5, 0.72)	0.17 (5, 0.78)	-0.19 (5, 0.77)	0.54 (5, 0.35)	0.30 (5, 0.63)	-0.08 (5, 0.89)	0.28 (5, 0.64)	0.30 (5, 0.63)	-0.36 (4, 0.64)	-0.93 (4, 0.07)
Yellow perch daily growth	0.90 (5, 0.04)	-0.41 (5, 0.49)	0.44 (5, 0.46)	0.50 (5, 0.39)	-0.78 (5, 0.12)	-0.51 (5, 0.38)	-0.87 (5, 0.06)	0.36 (5, 0.56)	0.66 (4, 0.34)	-0.61 (4, 0.39)
Fall bluegill mean TL	-0.79 (5, 0.11)	-0.20 (5, 0.74)	-0.55 (5, 0.33)	-0.21 (5, 0.74)	0.22 (5, 0.72)	0.71 (5, 0.18)	0.38 (5, 0.53)	0.17 (5, 0.78)	-0.76 (4, 0.24)	-0.35 (4, 0.65)
Fall yellow perch mean TL	-0.91 (5, 0.03)	0.02 (5, 0.98)	-0.76 (5, 0.14)	-0.41 (5, 0.49)	0.31 (5, 0.62)	0.80 (5, 0.11)	0.64 (5, 0.25)	0.13 (5, 0.83)	-0.90 (4, 0.10)	0.11 (4, 0.89)