CHAPTER 4

Population dynamics and ecological changes in an urban artificaly mixed shallow lake in Colorado, one year after restoration.

Abstract

Ecological conditions and phytoplankton succession in Sheldon Lake were analyzed from 2004 to 2005, one year after restoration of this shallow urban lake, in order to determine the trophic state and environmental variables controlling the phytoplankton, macrovertebrates and zooplankton compositions. The anthropogenic eutrophic process of the lake is characterized by increased nutrient concentrations, due to input of urban runoff. This addition triggers a chain of events starting with a massive increase in the growth of primary producers, since these are generally growth-limited by nutrients in freshwater ecosystems. Although the lake was artifically mixed, cyanobacteria were dominant as primary producer during the whole summer until they collapsed and were then replaced by diatoms after a period of high winds and rainfall. The absence of macrophytes in the main basin due to bottom sediment removal had a major effect on the juvenile blue gill sunfish, macroinvertebrates, and *Bosmina* sp. As an alternative *Bosmina* sp. used the surface blooms of cyanobacteria in summer as refuge from grazing by planktivorous fish. The greatest disturbance on the macroinvertebrate community richness and eveness was observed in the areas of incoming stormwater inlets.

*Keywords***:** cyanobacteria, artifical mixing, bottom sediment removel, macrophytes, urban runoff

4.1 Introduction

Water quality within urban water bodies is often under great pressure, owing to impacts such as storm water nutrient and contaminant loading. Contamination problems in natural receiving waters are complex, generally involving mixtures of chemicals in one or more matrices (e.g., water, sediment and biota). Especially prevalent within urban water bodies are concerns about eutrophication. Many of the same pesticides and nutrients used in large-scale agricultural practices are also used by municipal entities including urban and rural households. Runoff from fertilized lawns, household termite control, city streets, construction sites, storm sewers, household waste, organic nutrients from grass clippings may alter the soil makeup and can render a habitat unsuitable for certain organismes. US municipal discharges affect 16% of rivers and streams, 17% of lakes and reservoirs, and over 35% of estuarie (US EPA 1994).

Skinner *et al*. (1999) showed that stormwater runoff produced significant toxicity in the early life stages of medaka (*Oryzia latipes*) and inland silverside (*Menidia beryllina*). Developmental problems and toxicity are strongly correlated with the total metal content of the runoff and corresponded with exceedences of water quality criteria of Cd, Cu and Zn. The deterioration of water quality manifests itself in changes in the chemical, physical and biological characteristics of the receiving waters. These alterations include the following, reduction in water transparency, increase in pH, deoxygenation of the hypolimnion and the accompanying production of H_2S and CH_4, and release of nitrogen (N), phosphorus (P) and iron (Fe) from sediments during periods of anoxia. The increase in biological productivity which characterizes eutrophication has negative biotic impacts such as (a) an increase in

nuisance phytoplankton species and a reduction in macroalgal and higher plant species due to decrease in water transparency; (b) shifts from plankton-based to benthic or detrital-based food webs due to toxicity, poor palatability and the relative immunity to grazing of bloom phytoplankton species; (c) the toxicity of water contaminated by neuro- and hepatotoxins produced by certain bloom-forming cyanobacteria to resident invertebrates and fish; and (d) interference with recreational use due to water discoloration, and foul odours assocated with blooms (Harper 1992). The objective of this study was to describe the ecological status, as well as the limnological condition of Sheldon Lake a year after restoration based on water quality, phytoplankton, zooplankton and macroinvertebrate community structure. Phytoplankton is the major primary producers in many aquatic ecosystems and as such an essential component of the trophic structure of freshwater. Any changes in phytoplankton community structure or function due to e.g., urban runoff are likely to be reflected at higher levels of the food web.
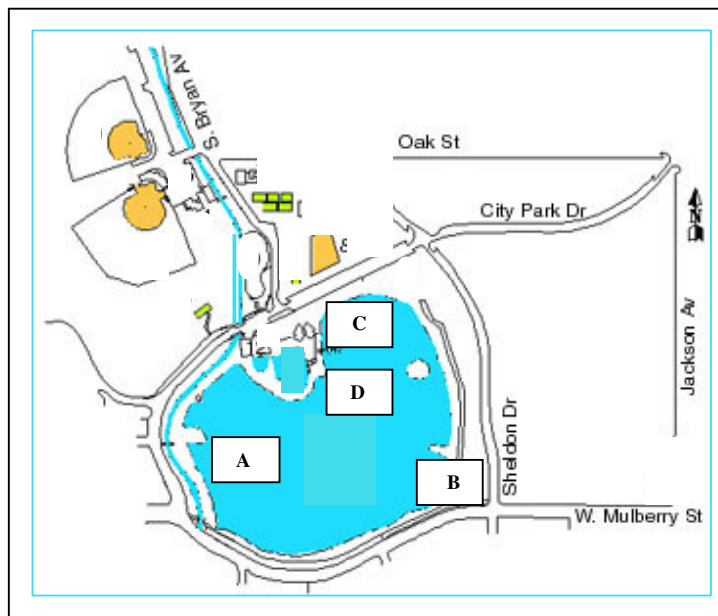
4.2 Materials and methods

4.2.1 Study area

The study was conducted in Larimer County, Colorado. The 6.07-hectare Sheldon Lake is a shallow man-made lake that was excavated in 1874, and it is a focal point for recreational activities for Fort Collins residents (Fig. 4.1). During December 2002 the Sheldon Lake Drainage Improvement Project was started with the initial purpose of removing over 250 structures from the 100-year floodplain which will provide flood protection by increasing the stormwater detention of Sheldon Lake, especially

after a major flood in 1997 causing five deaths and millions of dollars in damage
(Sheldon Lake Drainage Improvement Project 2002, 2003a, b; Endres 2004).

The effects of urban runoff after major storms on receiving-aquatic organisms or other
beneficial uses are very site-specific. Different land-development practices create
substantially different runoff-flow characteristics. Different rain patterns cause
different particulate washoff, transport, and dilution conditions (Laws 1993). Land use
in the drainage catchment area of Sheldon Lake is medium to high density residential
with arterial streets, schools, businesses, houses and apartments, serviced to about
70% by curb, gutter and storm sewers, and 30% by roadside ditches and culverts.
Soils in the study area were considered to have low infiltration and high runoff
potential. During November 2002 Sheldon Lake was drained and allowed to dry out
for the month of December while waterfowl and approximately 21 000 fish were
relocated. This period was chosen because fish then tended to aggregate making
capturing easier, oxygen stress on the fish are also minimized in winter, and
furthermore are freezing temperatures ideal for removing bottom of sediment. The
project began December 16, 2002 and was completed by June 23, 2003. During this
time 41 285 m^3 of lake bottom sediment was removed and 746 m of storm sewer pipe,
457 m of box culvert, 274.32 m of water line and nine stormwater inlets installed
(Sheldon Lake Drainage Improvement Project 2002, 2003a, b).

After completion of the project Sheldon Lake was refilled and stocked during the
summer of 2003 with largemouth bass *(Micropterus salmoides)*, bluegill sunfish
(*Lepomis macrochirus*), rainbow trout (*Oncorhynchus mykiss*), crappie (*Pomoxis
annularus*) and channel catfish (*Ictalurus punctatus*). Due to the eutrophic history of

Lake Sheldon, artificial mixing was introduced. The mixing is provided by two air compressors connected with a network of tubes and air bubble aerators in order to prevent growth of bloom-forming cyanobacteria. To regulate Sheldon Lake water depth variation in the summer and fall, adding water is supply by Pleasant Valley, Cache la Poudre River and the Colorado-Big Thompson watershed (Sheldon Lake Drainage Improvement Project 2002, 2003a, b).

Figure 4.1 Map of Sheldon Lake and City Park recreational area, Colorado (Sheldon Lake Drainage Improvement Project 2002) (Scale: 10 m = 5 mm).

4.2.2 Sampling Protocol

Sampling was performed at four sampling stations at different depths and was carried out in the morning (8.00-9.00) and afternoon (16.00-17.00)(Fig. 4.1). According to Bottrell *et al.* (1976) and Cobelas and Arauzo (1994), the most ideal sampling intervals for plankton communities should be less than their generations time, which

can be very short, and call for intensive sampling protocols. The use of daily, or at least weekly, samples appears to be a reasonable compromise, but this was not possible during this study and samples were taken every three to four weeks. Most field observations show more than 1-3 dominant species at any phase of seasonal development as predicted by the competitive exclusion theory (Hardin 1960). The reasons are found in the different responses of phytoplankton on the frequency of disturbances or changes in abiotic resource conditions at different scales (Reynolds 1984). These different scales are (1) shorter than one generation time induce physiological responses, (2) frequencies between 200 and 20 h interact with the phytoplankton growth rate, and (3) disturbances at up to 10 days intervals can initiate a successional sequence in phytoplankton development. Due to our three to four weekly sampling frequencies, we confined our analyses to successional sequence in phytoplankton development and influencing abiotic factors. Most samples were analysed within one week after collection. Species composition and community structure were assessed from 1 mL aliquots sampled from 200 mL vertical water samples, then fixed with buffered 5% (v/v) formaldehyde. Phytoplankton cells were identified and counted from transects in a Sedgewick-Rafter sedimentation chamber using an inverted microscope. Collection, handeling, identification and counting of phytoplankton were done following the procedures of Lund *et al.* (1958) and Padisak (1993). Zooplankton was sampled by means of two net hauls (mesh size, 80 µm) from bottom to surface. The two samples were pooled and preserved in formaldehyde (2% (v/v) final concentration). We measured the length of as many animals as required to ensure that the standard error was < 10% of the mean length of each species. Abundance was determined from length-weight regressions (Downing & Rigler 1984).

An Ekman grab sampler (Brower & Zar 1977) was used to collect samples of benthic macroinvertebrates on a monthly base at all four sampling sites. Organisms were futher sampled on the submerged stems of macrophytes at sampling sites A and B by scraping. These benthic samples were then preserved with 70% (v/v) ethanol, and stored at 4 °C before they were sorted. Organisms were pooled together from all four sample sites to form an aggregate sample unit. Samples were washed through a 0.25 mm mesh sieve and retained animals were carefully separated from detritus. The macroinvertebrates were identified according to Pennak (1978) and Merritt and Cummins (1996). Specimens were identified to the lowest possible taxonomic category and counted using a dissecting microscope. The classification method of Cummins (1974) was used to classify the different macroinvertebrates into functional feeding groups, since it has been suggested that the bioassessment of water quality based on functional feeding groups of macroinvertebrates may be superior to that based on community structure, because it reflects more ecologically significant attributes of freshwater systems (Rabeni *et al.* 1985).

Birds were sampled monthly from shore with a x 45 spotting scope. A running record of time, location, and behaviour was kept for all individual birds encountered to decrease the likelihood that individuals were counted more than once. For community analysis, we used only non-passerine birds that feed at or beneath the surface of the water. We chose these species because they are most strongly and unambiguously linked to lake characteristics, and were most reliably visible and identifiable by sight alone. Classification of the different bird species was done using the Colorado breeding birds' atlas (Kingery 1998).

Chl*a* was extracted from lyophilized GF filters using N, N-dimethylformamide for 2 hours at room temperature. Chl*a* was measured photospectrometrically at 647 and 664 nm, respectively and calculated according to Porra *et al*. (1989). Standard analytical techniques were used for all the chemical and physical variables. Nutrients, dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus (SRP) were analyzed using classical spectrophotometric methods (American Public Health Association, American Water Works Association, and Water Pollution Control Federation 1980). Temperature profiles and pH of the water column were measured with an YSI model 2100 thermometer and a 211 microprocessor pH meter at all four sampling stations. Data of wind speed, direction and rainfall was measured at the meteorological station of the Colorado State University, 2 km away from the lake. Fish species were collected from hook and line anglers, as well as with capturing gear that consisted of monofilament nylon nets with a 6.25 mm mesh size. Nets were set in late afternoon close to shore and macrophyte patches of sampling site A and was retrieved early the following morning. Classification of the different fish species was done using the field guide to freshwater fish (Schultz 2004).

The trophic state of Lake Sheldon was based on a combination of the classification of Carlson (1977) and Willen (2000). Freshwater lakes all over the world exists from eutrophic (rich in nutrients), to oligotrophic (low nutrient levels), while in between states also exist e.g., mesotrophic. In general, the amount of Chl*a*, cell density, and /or type of algal species present can be used as indicators of the trophic status of the water, so long as no toxic compounds that inhibit algal growth are present. Other factors have also to be taken into account, such as temperature and pH (Premazzi &

Chiaudani 1992). In our study, we used algal composition, secchi depth, total phosphorus and Chl*a* as indicators of the trophic state of Sheldon Lake a year after restoration. The dominant species shown in Table 4.1 are associated with corresponding water types (Willen 2000).

Table 4.1 Dominant species associated with water of different trophic levels (Modified from Willen 2000).

OLIGOTROPHIC

Chlorophyceae	[*Staurastrum* sp., *Gleoocystis* sp., *Spaerocystis schroeteri*]
Diatoms	[In Europe, *Cyclotella* associated with other species such as *Fragilaria* sp., *Synedra* sp., *Dynobrion* sp. and *Melosira* sp.while in North America lakes, *Asterionella* sp., *Tabellaria* sp.and *Melosira* sp.associated with *Dynobrion* sp.]
Crysophytes	Associated with very low nutrient levels [*Dynobrion* sp., *Mallomonas* sp., *Synura* sp.]
Chlorococcales	Some lakes may be dominate by *Oocystis* sp.
Dinoflagellates	[*Ceratium hirundinella, Peridinium incospicuum* and *Peridinium willei*]

MESOTROPHIC

Dinoflagellates	[In Finnish lakes, *Ceratium* sp., *Gymnodinium* sp., *Peridinium bipes* and *Peridinium cinctum*]
Chlorococcales	[*Pediastrum* sp., *Scenedesmus* sp.]
Chlorophyceae	[*Staurastrum gracile*, *Straurastrum pingue,* and *Straurastrum planctonicum*]

EU/HYPERTROPHIC

Cyanophyceae	[*Microcystis* sp*., Anabaena* sp*., Aphanizomeno*n sp. and *Lyngbya* sp. and *Oscillatoria rubescens*]
Euglenophyceae	[*Euglena* sp.]
Chlorococcales	[*Cosmarium* sp.]
Diatoms	[*Asterionella* sp*., Fragilaria* sp*., Synedra* sp*., Melosira* sp.]

We also applied Carlson's trophic state index approach to classify the trophic state of the lake that is based upon the calculation of an index with a range between 0-100 (Table 4.2). The theory of the index is based upon the statistical relationships between phosphorus loading, phosphorus concentration, chlorophyll and transparency, by using $\log_2$ of the secchi disc transparency as the starting point. Zero is set at 64 m, the integer greater than the maximum transparency ever recorded [42 m for Lake Masyuko in Japan (Hutchinson 1957)], and each halving of transparency increases the index by 10. Chlorophyll and total phosphorus concentration were related to transparency by regression equations and then added to the scale in Table 4.2. (Carlson 1977). An index category of less than 20 represents ultra-oligotrophic, 30-40 oligotrophic, 45-50 mesotrophic, 53-60 eutrophic and above 70 hypertrophic (Kratzer & Brezonik 1982).

Table 4.2 Carlson's Trophic State Index and its associated parameters (Carlson 1977).

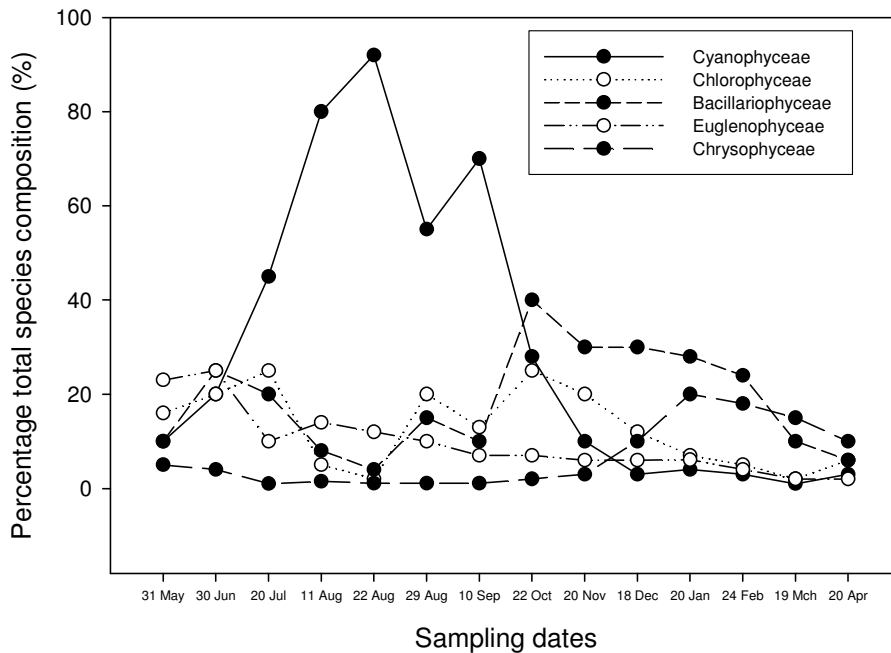| TSI	Secchi disc depth(m)	Total phosphorus(µg/L)	Chlorophyll(µg/L)
0	64	0.75	0.04
10	32	1.5	0.12
20	16	3	0.34
30	8	6	0.94
40	4	12	2.6
50	2	24	6.4
60	1	48	20
70	0.5	96	56
80	0.25	192	154
90	0.12	384	427
100	0.06	768	1183

4.2.3 Data analyses

The results were recorded on standard Excel spreadsheats for data processing, and statistical analysis was performed using SYSTAT® 7.0.1 (1997).
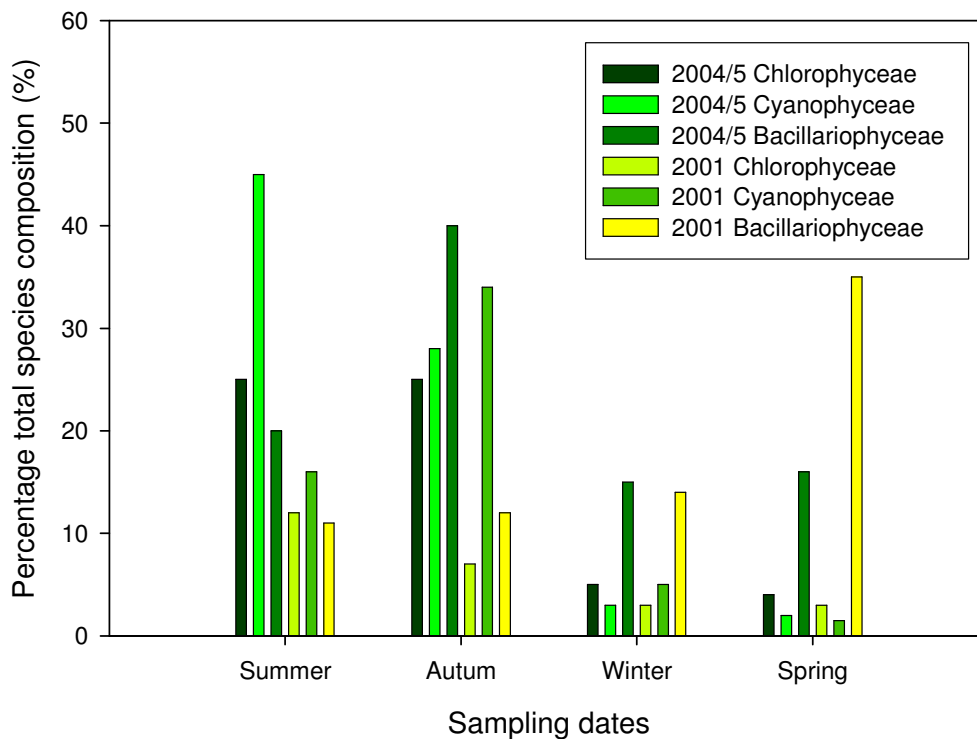
4.3 Results

4.3.1 Phytopankton

The phytoplankton succession in Sheldon Lake *per se* provides a valuable model for the changing environment in a restored, artificial mixing shallow urban lake. The mid and late summer phytoplankton community composition and structure of 2004 did not vary much. Cyanobacteria were the most abundant group of phytoplankton across all sampling sites, followed by Chlorophyceae, Bacillariophyceae and Euglenophyceae (Fig. 4.2).

Figure 4.2 Seasonal changes and contribution to phytoplankton species composition as percentage for the five major algal classes 2004-2005 from the four sampling sites.

The dominant cyanobacteria species were *Microcystis aeruginosa* and relatively less abundant was *Woronichinia naegeliana* while *Microcystis viridus* were less abundant to *Microcystis aeruginosa* in 2001 without mixing (Table 4.3, Fig. 4.3). These species disappear from the scene in early October after a period of high winds and heavy rainfall. A shift to a mostly Bacillariophyceae dominated community was observed due to high silica concentrations (silica 6.3 mg/L) and resuspension events in autumn 2004. Diatoms appeared to be more tolerant than summer species like cyanobacteria to reduced light doses and rapid fluctuations in irradiance (Reynolds 1984).

Figure 4.3 Seasonal changes and contribution to phytoplankton species composition as percentage for the Chlorophyceae, Cyanophyceae and Bacillariophyceae in 2001 and then 2004-2005 from the four fixed sampling sites.

During the winter of 2004-2005, reduce temperatures, light energy input as well as silica limitation led to lower primary production and a decline in diatom biomass which were replaced by Chrysophyceae during the clear water phase. Towards late spring Chlorophyceae became dominant after the clear water phase, when the phytoplankton biomass declines rapidly to very low amounts (Fig. 4.2). However this was not the case during the spring of 2001 without mixing when *Melosira varians* were abundant. The concentration of total phosphorous (TP) and Silica increased significantly from 2001 without mixing to 2004-2005 with mixing, while Secchi-depth decreased correspondingly (Table 4.3).

Table 4.3 Annual mean values and ranges for selected variables for the years 2001 and May 2004 to April 2005 (Data for 2002-2003 is not available due to restoration; ± indicate standard error).
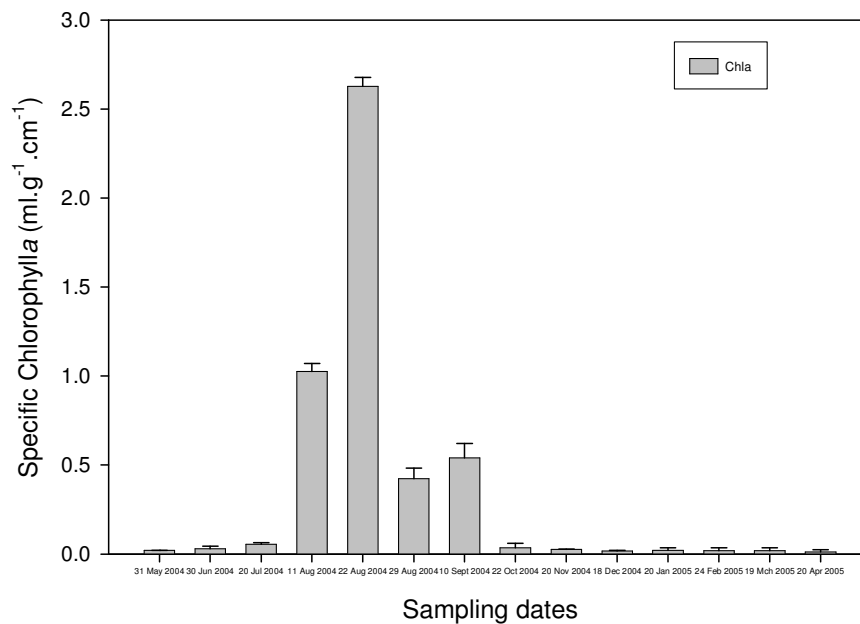
Varaible (unit)	Year 2001* Sheldon Lake before restoration (n=4)	2004-2005 Sheldon Lake one year after restoration (n=24)	2004-2005 Inflow from Urban run-off inlets and Cache la Poudre River (n=20)
Secchi depth (m)	0.51 (0.31-0.70)	0.61 (0.25-0.80)	Nd.
pH	8.5	8.4	8.1
SRP (µg/L)	Nd	4 (0-16)	3 (0-22)
Ammonia (mg/L)	Nd	0.467 (± 0.139)	0.302 (± 0.100)
Total Nitrate (mg/L)	2.280 (± 0.542)	2.87 (± 0.046)	2.47 (± 0.046)
Total Phoshorous (mg/L)	1.81 (± 0.066)	2.21 (± 0.061)	1.40 (± 0.161)
Silica (mg/L)	4.9 (± 0.052)	6.3 (± 0.031)	Nd
Conductivity (ms/cm)	Nd	1.34 (± 0.12)	1.73 (± 0.21)
DO (mg/L)	Nd	8.3 (± 0.4)	10.7 (± 0.7)
BOD (mg/L)	Nd	6.8	3.3
Chlorophyl*a* (µg/L)	72 (6-420)	58 (10-693)	Nd
Dominant phytoplankton spp.	*Microcystis aeruginosa; M. viridus* (summer) *Asterionella formosa* (winter)	*Microcystis aeruginosa; Woronichinia naegeliana* (summer) *Melosira varians, Asterionella formosa* (winter)	Bacillariophyceae (summer and winter; only for Cache la Poudre) Urban runoff (Nd)

Nd = not determined
n = sample times per year
* - Unpublished data and personal communications by B. Whirty, Park Operations Supervisor, City of Ft. Collins, and S. Seal, Fish health, Environmental Sciences, CSU

The secchi depth was quite shallow during summer 2004-2005 with an average of
0.25 m during a peak of cyanobacteria (i.e. *Woronichinia naegeliana* and *Microcystis*
aeruginosa) blooms from August to September (Table 4.3). There also was a strong
negative correlation between Chl*a* and the secchi depth during this time indicating
that the dense blooms of cyanobacteria significantly reduced light penetration through
the water column at the sampling sites (Fig. 4.4).

Figure 4.4 Specific Chl*a* measured from May 2004 to April 2005, Sheldon Lake,
Colorado. Error bars represent standard deviation from the mean value.

The classification of the lake's trophic state one year after restoration, using the
Carlson (1977) trophic state index and its associated parameters as well as the
phytoplankton indicator concept of Willen (2000), indicated that Sheldon Lake is
eutrophic to hypertrophic according to the following parameters taken during
summer: Secchi disc depth (0.25m); Total Phosphates between 0.2 mg/L to 0.3 mg/L;
Total Nitrogen between 0.2 mg/L to 0.4 mg/L; Chlorophyll 0.693 mg/L; Silica 6.3
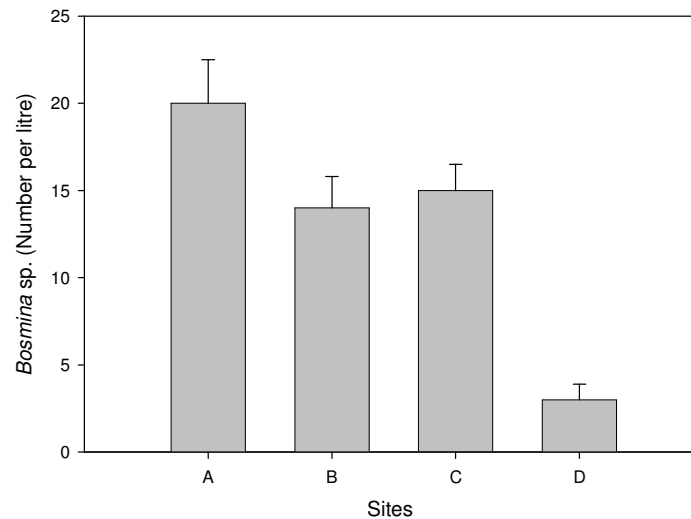
mg/L. Dominanted phytoplankton indicators: *Asterionella* sp., *Synedra* sp., *Melosira* sp., *Cosmarium* sp., *Euglena* sp. and *Microcystis* spp.

4.3.2 Zooplankton

The abundances of herbivorous zooplanton (cladocerans; omnivorous copepods and rotifers) were close to zero in Lake Sheldon from June to the middle of July, when the abundances increased somewhat, although they never exceeded 10 ind/L. During August to the end of September numbers increased above 20 ind/L (results not shown). An interesting feature of the zooplankton was the relationship between cladocerans (*Bosmina* sp.) and a cyanobacterial bloom of *Microcystis aeruginosa*. During dense surface blooms of cyanobacteria in August to the end of September, there was a 5-fold higher density of *Bosmina* sp. at sampling site C which contains high numbers of cyanobacterial colonies than at site D were cyanobacterial colonies were absent (Fig. 4.5).

At the end of September there were a decline in abundance of cyanobacteria and *Bosmina* sp. due to a temperature drop and storm weather activities. However, zooplankton community was dominated for most of the summer by the small-bodied *Bosmina* sp., which provides little potential for the removal of significant amounts of phytoplankton. This contrasts with the situation in spring during the clear water phase, when a relatively high population density of larger cladocerans (*Daphnia magna*) occurs.

Figure 4.5   Abundance in *Bosmina* sp. during summer 2004 (A) Site with cyanobacterial surface bloom and sparse macrophyte growth; (B) Site with cyanobacterial surface bloom and sparse macrophyte growth; (C) Site with cyanobacterial surface bloom but without a macrophyte growth; and (D) Site without cyanobacterial surface bloom and macrophyte growth. Error bars represent standard deviation from the mean value.
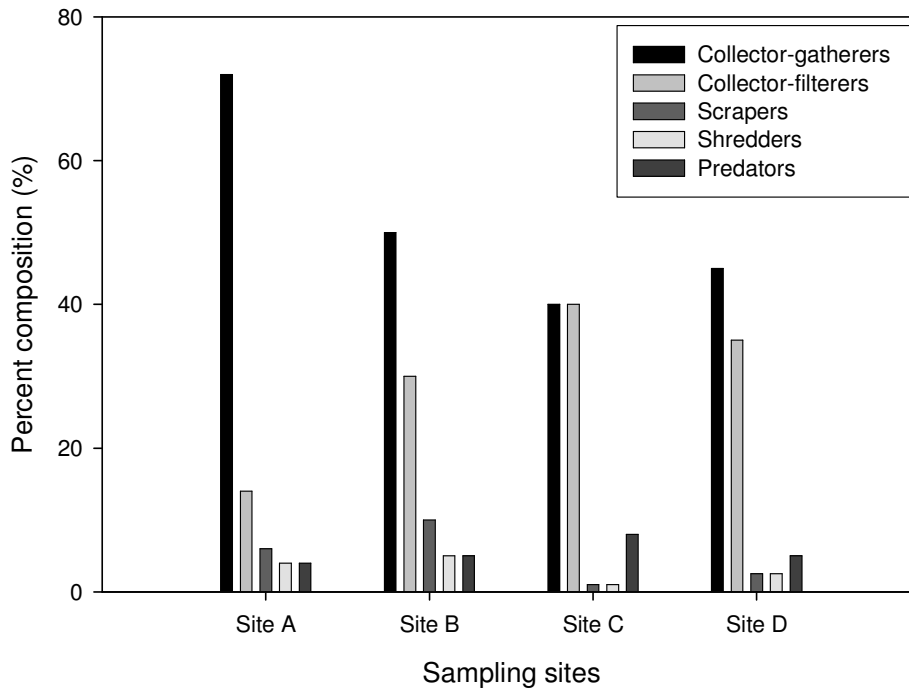
4.3.3 Benthic macroinvertebrates

Fluctuations in lake levels due to variable seasonal inflow of stormwater runoff, as well as the short time after removing of bottom sediment for restoration purposes, give rise to a very unstable and ill-defined littoral zone, with negligible development of a littoral flora.

The most abundant macrophyte recorded were *Typha angustifolia* at sampling sites A and B. Macroinvertebrate diversity in a sampling programme carried out monthly were very low and most of the lake were dominated by Nematoda*,* Oligochaete worms together with chironomid larvae of the genus *Chironomus* (Table 4.4).

Table 4.4 Dominant taxa in each functional feeding group.

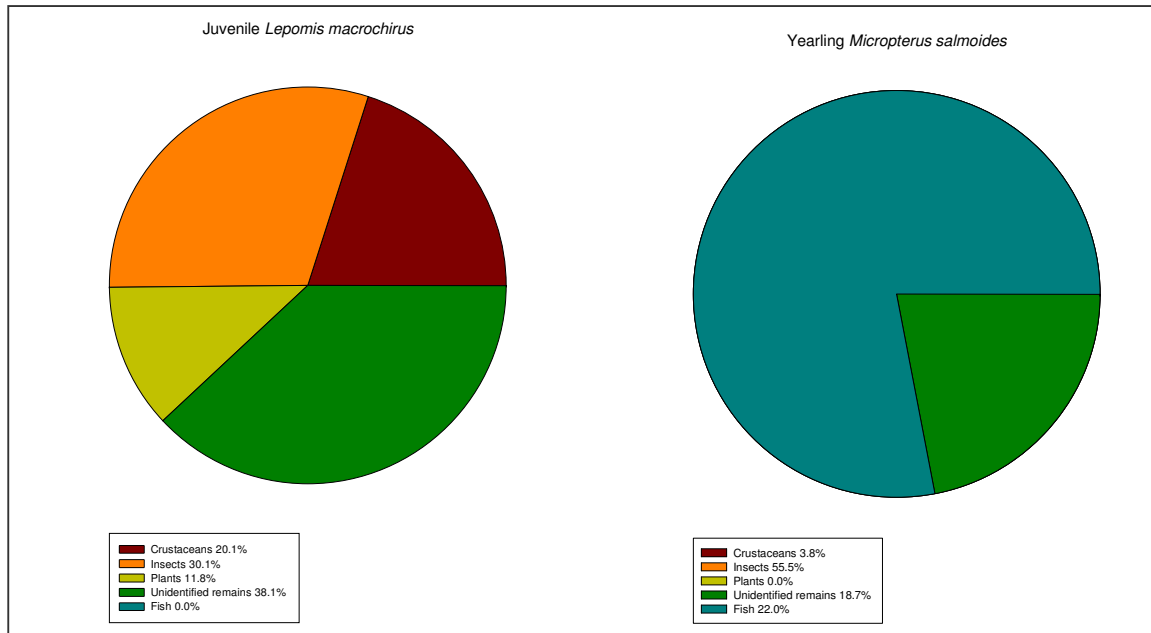| Functional feeding group	Taxa
Collector-gatherers	Chronomidae, Oligocheata, *Tricorythodes minutus*, *Acentrella insignificans*, *Baetis* spp.
Collector-filterers	*Cheumatopsyche* spp., *Simulium* spp., *Hydropsyche occidentalis*, *H. cockerelli*, *H. morose*
Scrapers	*Hydroptila pecos*, *Petrophila avernalis*
Shredders	*Caecidotea cummunis*, *Oecetis inconspicua*
Microphyte-piercers	*Hydroptilia pecos*
Predators	Nematoda, *Acari, Dugesia dorotocephala, Oecetis inconspicua*

Oligochaetes and chironomids were the most abundant at sampling site A followed by collector-filterers, scrapers and shredders which were particularly poorly represented at sampling sites C and D (Fig. 4.6).

Figure 4.6 Macroinvertebrate abundance (%) at the four sampling sites, Sheldon Lake, Colorado during summer 2004.

4.3.4 Vertebrates

With the removal of the bottom sediment and restoration of the lake, the fish fauna were restock in 2003 by the Colorado Division of Wildlife and included the following species: 3 000 largemouth bass (*Micropterus salmoides*); 3 000 bluegill sunfish (*Lepomis macrochirus*); 12 000 crappie (*Pomoxis annularus*); 3 000 channel catfish (*Ictalurus punctatus*) and rainbow trout (*Oncorhynchus mykiss*). Sufficient specimens of two of these species (largemouth bass and bluegill sunfish) were collected during the investigation to allow for a meaningful analysis of the stomach contents.

Figure 4.7 Composition by volume of the gut contents of juvenile *Lepomis macrochirus* and yearling *Micropterus salmoides* of Sheldon Lake, Colorado during summer 2004 (n = 50).

Figure 4.7 shows the percentage of the volume of the stomach contents contributed by different food items. The fish were mainly juveniles' bluegill sunfish and yearling largemouth bass, with modal lengths varying between 20-200 mm. While zooplankton

was found in the stomachs of all the species that were dissected, macroinvertebrates were the most important food type to the juvenile bluegill sunfish that was caught in the littoral zone of sampling sites A and B where macrophytes were sparce.
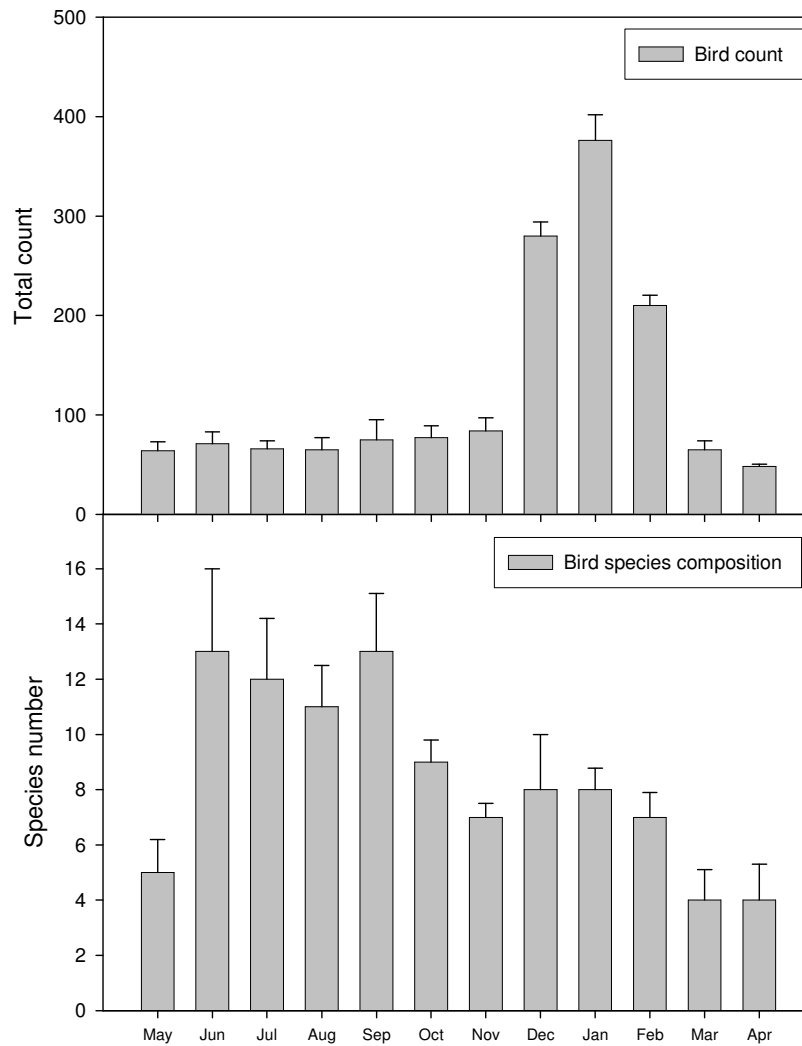
4.3.5 Waterbirds and nearshore birds

The monthly count and species number of waterbirds from May 2004 to April 2005 is reported in Figures 4.9a and b. A year's total count was 13 species and an average of 65 individuals per month. During this time the following bird species were observe: double-crested cormorant (*Phalacrocorax auritus*); back-crowned night-heron (*Nycticorax nycticorax*); great blue heron (Ardea *herodias*); spotted sandpiper (*Actitis macularia,*); black tern (*Chlidonias niger*); ring-billed gull (*Larus delawarensis*); canada goose (*Branta canadensis*); mallard (*Anas platyrhynchos*); northern shoveler (*Anas clypeata*) (Fig. 4.8); bufflehead (*Bucephala albeola*); common goldeneye (*Bucephala clangula*); common merganser (*Mergus merganser*) and the hooded merganser (*Lophodytes cucullatus*).

Figure 4.8 Northern shoveler's clustered around air bubbles that form surface 'boils' in the ice cover during artificial mixing.

The number of birds was high during the winter, but the diversity was higher during the summer months with the exception of the duck species with a higher diversity during the winter months. The most dominant birds during our survey included mallards (*Anas platyrhynchos*), Canada goose (*Branta canadensis*) and the ring-billed gull (*Larus delawarensis*). The number of sandpipers remained quite small throughout the year while the number of predator birds increased during the summer, which was probably a result of the increase in benthic biomass.

Figure 4.9 Monthly distribution of waterbirds and nearshore birds in Sheldon Lake, Colorado (a) count and (b) species. Error bars represent standard deviation from the mean value.

4.4 Discussion

4.4.1 Phytoplankton

Phytoplankton communities in the water column of temperate lakes undergo significant changes within individual years. These changes have been termed as 'seasonal succession' by plankton ecologists. Seasonal succession of phytoplankton has more similarities with the succession of terrestrial vegetation than with its seasonal aspect. Many generations are involved with several quite predictable and distinct phases. On the scale of generation times, the several months in plankton succession corresponds to tens of years in terrestrial grassland, and to centuries in forest succession. Under favourable physical conditions the intrinsically transient nature of early and mid-successional phases, and the selfsustainability of final stages can be shown. It is only the external cycle in climatic and hydrological conditions that forces phytoplankton succession to restart each year (Sommer 1991). The phytoplankton composition in 2004-2005, especially the bloom of cyanobacteria in the second half of the year is an indication of the eutrophic to hypertrophic state of Sheldon Lake.

A shift from cyanobacterial dominance to a phytoplankton community dominated by green algae and diatoms has been observed in artificially mixed lakes or reservoirs (Symons *et al*. 1970; Haynes 1973; Hawkins & Griffith 1993; Cowell *et al*. 1987). However, this was not the case in our study since cyanobacteria dominated the phytoplankton community the whole summer of 2004. The predominance of cyanobacteria can be due to the fact that the artificial mixing velocity was not

sufficient on sunny, calm days to keep *Microcystis* entrained in the turbulent flow in the entire lake. Other factors that could have determined the insufficiency of the artificial mixing were the depth of the lake where the aeration tubes were installed, and the distribution of the tubes over the lake area (Cooke *et al*. 1993). The failure to reduce dominance by colony-forming cyanobacteria can often be ascribed to insufficient mixing velocities and/or a low mean depth of the lake or reservoir (Knoppert *et al*. 1970; Lackey 1973; Osgood & Stiegler 1990). Other cyanobacteria in Sheldon lake with mixing, during the summer of 2004-2005 were *Woronichinia naegeliana* while *M. viridus* where abundant in 2001 without mixing.

Detailed analyses on the summer succession of Lake Sheldon in 2004 have shown that the meteorological (rain and wind) and chemical conditions (Silica and total Phoshorous), especially the frequency of storms and a drop in temperature, have a fundamental effect on phytoplankton succession. These periodic events together with the artificial mixing did enable cyanobacteria to maintain high population density during the late summer. This observation is supported by the variable physical and morphometric characteristics, such as those typical of many urban water bodies. These characteristics can play a role in influencing phytoplankton community composition, which include the following: mixing patterns (Harris & Trimbee 1986), high wind events (Carrick *et al*. 1993), flooding (Van den Brink *et al*. 1994), water level fluctuations (Garcia de Emiliani 1997) and changes in flushing rate (Bailey-Watts *et al*. 1990). The diatom peak (*Synedra rumpens var fusa* and *Melosira varians*) in October of 2004 that succeeded the cyanobacteria peak (*Microcystis aeruginosa*) in summer was correlated strongly with resuspension events caused by meteorological

activities (rain and wind) on the 4, 21, 25 and 27th of September 2004 when wind velocity reach a maximum of 10.2 m/s.

 The most dominant species found with mixing during the winter months of 2004-2005, were *Melosira varians* and *Asterionella formosa*, which occur, in significant abundance under the ice cover. These phenomena correlated with findings of Agbeti and Smoll (1995). *Melosira varians* were abundant only in spring of 2001 without mixing, while they where abundant in the winter of 2004-2005 with mixing (Fig. 4.3). The fact that the winter season showed a consistent positive relationship between water temperature and algal biomass is expected, because most phytoplankton species for e.g., *Microcystis aeruginosa* reached their optimal growth rate in the range of 20-25 °C (Reynolds 1984). However during spring, summer and autumn periods other factors, such as grazer-, nutrient- or light-limitation tend to play an important role, while these factors seems less important during winter periods in Sheldon Lake. Therefore, the direct effects of temperature ranging between 1-4 °C on phytoplankton growth and reproduction become relatively more significant. Although there is little experimental evidence that shows diatom species dominate in winter and spring at temperatures below 5 °C, because of their higher growth rate in colder temperatures compared to cyanobacteria or Chlorophyta (Lund 1955; Foy & Gibson 1993). The Chlorophyceae were much higher in 2004-2005 with mixing compared to 2001 without mixing (Fig. 4.3).

Foy and Gibson (1993) have demonstrated in culture experiments on three planktonic diatom species that growth rate show a progressively decreasing response to increasing temperature above 10 °C. During the clear water phase of 2005 the most

common species were *Dinobryon suecicum* and *Chrysococcus minutes*. These Chrysophytes species tended to dominate only in conditions of low productivity and low to moderate pH. They were replaced by the end of spring with chlorophytes species with a generally higher nitrogen optima and the ability to utilize bicarbonate in photosynthesis, as well as to thrive in a higher pH environment (Sandgren 1988; Reynolds 1998; Levine & Schindler 1999). The reduction of *Daphnia magna* at the end of the clear-water phase may be the result of higher transparency when the phytoplankton biomass was low, and this may have increased the predation risk from visual hunting fish. Furthermore an increase in the N:P ratio during this time is associated with waste material of free-living waterfowl, which has assembled, in great numbers at surface 'boils' produced by artificial mixing. The artificial mixing created open areas in the ice-covered lake during the winter months of December and January. Most of these waterfowl has migrated from other neigbouring ice covered lakes where food was limited.

4.4.2 Zooplankton

Results of laboratory studies (Lampert 1987) suggest that many colonial cyanobacteria are either not eaten or are a poor food source for large zooplankton, particularly *Daphnia*. Therefore, at times when these colonial forms dominate the phytoplankton, *Daphnia* populations might be expected to show decreased growth and fecundity in response to food limitation or toxicity. However, the theory was supported by our field studies in 2004-2005 when we observed that the *Daphnia* population decreased during cyanobacteria bloom periods probably due to the combined impact of fish predation and food limitation and was replaced by *Bosmina*

sp. This seasonal pattern is in agreement with other the observations (McNaught 1975; Gliwicz 1977; Beaver & Crisman 1982). The observed increase in *Bosmina* sp. in Sheldon Lake during this time may be related to its ability to avoid predators by positioning in the cyanobacterial surfaces blooms that provide temporary refuge in the absence of macrophytes.

A year after the removal of the bottom sediment of the lake for restoration purposes, macrophytes and submerged vegetation in Lake Sheldon were sparse and only a few patches of the macrophyte, *Typha anustifolia* was prominent at the littoral zone of sampling sites A and B. This correlate strongly with the observation of Moss *et al.* (1986), who found failure of submerged plant colonization of the majority of the open water in the Coockshoot Broad after bottom sediment removal in 1982. The failure of plant colonization is not yet understood but may be due to the lack of sufficient inoculums of plant material or simply to the fact that phytoplankton populations are still preventing adequate light transmission. De Bernardi and Giussano (1990) also observed that in shallow lakes where plants have been lost, circumstances leading to cyanophyte dominance also make restoration of the plant communities difficult because cyanophytes are often indigestible or nutritionally poor for invertebrate grazers. Macrophytes fill multiple roles in the ecosystem function (Carpenter & Lodge 1986) and in the mediation of predator-prey interactions involving fish and macroinvertebrates (Crowder & Cooper 1982; Savino & Stein 1982). Investigators have suggested that macrophytes provide a refuge to cladocerans from fish predation and thus contribute to biomanipulation efforts to reduce phytoplankton-standing stock (Timms & Moss 1984). Scheffer *et al.* (1993) suggested that macrophyte refuges for

Daphnia contribute significantly to the stability of the high *Daphnia*-low phytoplankton-high macrophyte state in shallow lakes.

Zooplankton in shallow, non-stratified lakes does not have the option of moving down to the metalimnion and hypolimnion to avoid predation. However, studies in shallow lakes have shown that zooplankton aggregate in nearshore areas among the structurally complex macrophyte beds during the day. For example, the density of *Daphnia magna* was 20-fold higher within the macrophyte beds during daytime than during night-time (Lauridsen & Buenk 1996), indicating that horizontal migration between the structurally complex macrophyte beds and the open water may be a way to reduce the risk of predation by planktivores in lakes where vertical migration is restricted. In parallel to vertical migration, zooplankton exposed to chemical cues 'i.e. kairomones'from fish also increases their use of the macrophyte habitat (Lauridsen & Lodge 1996). The data correlates with Schriver *et al*. (1995) that *Daphnia* tolerate a higher fish density in macrophyte-rich lakes and that the presences of submerged vegetation may further enhance grazer control. We proposed that there is a compensatory interaction between cyanobacteria and zooplankton in shallow lakes where floating leaved macrophytes and submerged vegetation is scarce. The zooplankton may prevent intense predation by zooplanktivorous fish by using cyanobacteria blooms as temporary refuge in the absence of macrophytes.

 It was also found in previous studies that macrophyte densities could affect chemical fate processes by increasing the surface area available for sorption of hydrophobic compounds. Caquet *et al*. (2000) reported the presence residues of deltamethrin and lindane in the macrophyte samples 5 weeks after treatment but never in the sediment.

Macrophytes can also affect physicochemical composition in surrounding waters, influencing the distribution and community structures of many aquatic organisms (Barko *et al*. 1988). In addition, macrophytes provide three-dimensional structure within constructed ecosystems, which affect organism distribution and interactions. Others have shown that macroinvertebrate community diversity is influenced by patchy macrophyte abundance and specific macrophyte types (Schramm *et al*. 1987; Learner *et al*. 1989). Cladoceran communities are not only associated with macrophytes for refuges but also graze on the periphytic algae on aquatic macrophytes (Campbell & Clark 1987). The impact of the low to zero densities of macrophytes in Sheldon Lake may cause indirect effects on organisms by influencing trophic linkages such as predator-prey interaction between invertebrates and vertebrates.

4.4.3 Macroinvertebrates

The differences in dominant taxa of macroinvertebrates in terms of production among sampling sites in 2004-2005, suggest that these differences might be related to the variations in physicochemical and food-related variables. The numbers of Oligochaetes and chironomids, that significantly increase in abundance in the soft bottom sediment at sampling site A, may probably be due to the exposure to organic pollution at the inlet of a urban runoff pipe situated in the sampling area (Fig. 4.6). These taxa contain many generalist feeders that benefit from increased organic loading. Other studies have also shown that the increased Oligochaetes and chironomid population densities were in association with organically polluted lotic systems (Prat & Ward 1994; Zamora-Munoz & Alba-Tercedor 1996). Wiederholm (1984) also reported that the presence of Oligochaeta, *Physella* sp., *Simulium* sp., and

a dominance of Chironomidae genera, was characteristic of aquatic systems affected by organic pollution.

The distributional patterns of the dominant taxa in each of the functional feeding groups which reflect resource distribution and use, and facilitate the understanding of organic matter processing in a freshwater ecosystem like Sheldon Lake were as followes: Collector-gatherers (Chironomidae, Nematoda, Oligochaeta); Collector-filterers (*Simulium* spp.); Scrapers (*Physella* sp.); Shredders (*Caecidotea cummunis*), Predators (Acari) (Fig 4.4). The density of the collector-filterers were the highest in November and May at the standing stock of macrophytes (*Typha angustifolia*) of sampling sites A and B, which created more surface attachment sites for collector-filterers, such as simuliids. The macrophytes at these sampling sites overwinter as rhizomes. Although the ability to absorb nutrients from the sediment and light from above the water surface make emergent macrophytes (*Typha angustifolia*) competitively superior, it also restricts them to shallow water, making emergent macrophytes a less dominant primary producer. A higher abundance of scrapers were observed in the fall and summer 2004 when higher water temperature and light intensity stimulate primary production and resulted in an increase in the food resources, especially during the increase of the diatom population in late summer. Higher densities of shredders occur in fall 2004 during a period of maximum availability of deciduous leaves. Predators showed a variation in density, with the highest densities during summer and fall. Due to the fact that sampling sites C and D did not support any macrophytes that create more diverse habitats for the attachment of benthic macroinvertebrates, the abundance and species richness were very low at these sites. On the contrary, the filamentous green alga, *Cladophora* sp. was found in

low numbers at all of the four sampling sites. Moreover, Steinman (1996) indicated that *Cladophora* sp. is not utilized as a food resource by macroinvertebrate scrapers because they are too large to graze. In addition, numerous *Caecidotea cummunis* were collected in the filamentous alga *Cladophora* during our survey. Macroinvertebrate ecology in Sheldon Lake supported the concept of Kerans and Karr (1994) which stated that the direction of change with increasing human impacts would increase proportions of individuals comprising of collector-gatherers and collector-filterers, and decrease proportions of scrapers, shredders and predators.

 In contrast with Sheldon Lake, lakes that retain submerged plant populations are more diverse and contain several mollusks species (Mason & Bryant 1975). Wortley (1974) demonstrated that this was a direct result of lack of habitat diversity and showed that when artificial plants constructed from polypropylene were introduced into the lakes without submerged macrophytes, large number of invertebrates established populations. Crayfish is one of the predators feeding on mollusks species in Sheldon Lake and account for a large proportion of the biomass (data not shown). These are not important only in terms of numbers and biomass but also because of their functional role in the ecosystem. On the other hand, crayfish may have negative effects on macrophytes, periphyton and mollusks especially if macrophytes are spares like in the case of Sheldon Lake (Lodge *et al*. 1994).

4.4.4 Vertebrates

On the basis of identifiable remains, macroinvertebrates and not zooplankton is the most important component of the diet of young non-piscivorous bluegill sunfish (*Lepomis macrochirus*) in Sheldon Lake. This phenomenon may be due to the

presence of piscivorous largemouth bass (*Micropterus salmoides*) resulting in change of habitat from the pelagic zone with no submerge macrophytes to the refuge of the littoral zone with sparse macrophytes. However, the juvenile bluegill sunfish had a strong negative effect on the density of macroinvertebrates, resulting in a smaller average size of macroinvertebrates. This observation agrees with previous reports that found that predators also affect the behaviour of their prey resulting in reduced activity or a change in habitat/ refuges (Werner *et al.* 1983; Werner & Hall 1988; Fuiman & Magurran 1994; Turner 1997). These predator-mediated changes in the habitat use of an intermediate consumer may affect its diet and thus change the direction of interactions in food chains.

4.4.5 Waterbirds and Nearshore birds

For aquatic birds, depth and size play an important role in shaping assemblages, but size appears to be the predominant factor. Differences in the composition of bird assemblages strongly paralleled differences in species richness and reflected species-area relationship reminiscent of those seen among aquatic bird communities in other geographical settings (Elmberg *et al.* 1993, 1994; Hoyer & Canfield 1994).

Small shallow lakes like Sheldon Lake (size: 6,07 hectares and max. depth: 1,4 meters) support only a core of widespread, generalist species simply because of their size, not because the water body is invariably shallow. A detectable but smaller effect of a lake's position in the landscape on its bird assemblage is also important, especially if it is situated in an urban environment like in the case of Sheldon Lake. Although birds are perhaps the best-studied taxon in human-dominated areas, our

understanding of the effects of human settlement on bird communities is in its infancy (Marzluff *et al.* 1998). Aerial piscivores which served as effective indicators of fish-assemblage type were relative scarce during our summer months survey of 2004, this may be due to the fact that the transparency of the water body was low because of cyanobacterial blooms (Secchi depth 0.25 m) and therefore not a profitable feeding site for aquatic birds that forage by surface plunging (Eriksson 1985).

During the winter months the birds may be a contributory factor to eutrophication when they assemble in great numbers at areas where artificial mixing provide openings in the ice-cover especially, the northern shoveler's which clustered around air bubbles that form surface 'boils' (Fig 4.8). This mixing method destroys or prevents thermal stratification and cause vertical circulation of phyto and zooplankton to the photic zone in the form of surface 'boils'. It is then an available food source for the northern shoveler's that filter the water through their 'Spoon-bill' ejecting the refuse through its 'sieve', and retaining whatever nutritious matter there may be. Their diet explains their preference for shallow, muddy areas that provide a bounty of free-swimming invertebrates (DuBowy 1996). A bald eagle (*Haliaeetus leucocephalus*) was frequently observed at sampling sites C and D during the winter months of December and January 2004-2005. The bald eagle is almost non-migratory, and only deserts its home during the coldest weather when the water is frozen. Artificial mixing ensure that a small percentage of Sheldon Lake at sampling sites C and D were not covered with ice during the winter months of December to January, this gave the bald eagle an opportunity for food. Occasionally the bald eagle was joined by crows and ravens when they were feeding upon carrion. On two occasions the bald eagle was spotted attacked waterfowl during January, and even killing a common goldeneye

duck at one of these occasions. Other birds frequently observed in trees near the lake during our survey included the belted kingfisher (*Ceryle alcyon*); northern flicker (*Colaptes auratus*); common raven (*Corvus corax*) and American crow (*Corvus brachyrhynchos*).

4.5 Conclusion

Colorado is typical of states in the Rocky Mountain region with a growth rate that is three times the national average and manifested in urban and suburban sprawl. These phenomena are particularly acute in counties along Colorado's Front Range, where approximately 80% of the states population lives. There, annual population increases exceeding 4-6% are not uncommon (US Bureau of the Census 1998) however this tendency will in the long run subject urban lakes increasingly more to negative environmental impacts like storm water nutrients and contaminant loading which is synonymous to urbanization. Adding water from Pleasant Valley, Cache la Poudre River and the Colorado-Big Thompson watershed to Sheldon Lake could also be ecologically damaging for it usually brings in water of different chemical nature and may introduce alien organisms. The observed large proportion of collector-gatherers and large fraction of organic pollution-facultative organisms is indicative of excessive organic loading in Sheldon Lake during our survey. Moreover, due to the loss of aquatic vegetation through bottom sediment removal, Sheldon Lake formed a turbid phytoplankton-dominated state instead of a clear macrophyte-dominated state. The submerged vegetation that was lost during the restoration process could have enhance water clarity by reducing resuspension of bottom material, providing zooplankton with a refuge from grazing by planktivorous fish, egg-laying sites and source of food for a wide variety of invertebrates, suppressing algal growth by competing for

nutrients, and releasing allelopathic substances that are toxic to algae (Scheffer 1998). However, the loss of the submerged macrophyte communities in general caused reduction in diversity of keystone macroinvertebrate and zooplankton species that acted as a key in determining much of Sheldon Lake's ecological function.

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