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Biotic interactions in temporal trends (1992–2010) of organochlorine contaminants in the aquatic food web of Lake Laberge, Yukon Territory

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HIGHLIGHTS

► Organochlorine contaminants in a sub-Arctic lake were monitored post-fishery closure.
► We observed significant temporal declines in OC from 1992 to 2010.
► Declines in OC were related to growth dilution for some species.
► Declines in OC were related to decreased lipids for some species.
► Zooplankton community changes may also have added to the declining trend in OC.

ARTICLE INFO

ABSTRACT

Declines in 6 organochlorine (OC) contaminant groups; chlordane (CHL), DDT, HCH, toxaphene (CHB), PCB and chlorinated benzenes (CBz) were measured in biota of a sub-Arctic lake (Lake Laberge, YT) following the closure of a commercial fishery in 1991. This study examined morphological (length, weight, age), biochemical (lipid content, $\delta^{13}C$, $\delta^{15}N$), population and OC data for 9 fishes and zooplankton between 1993 and 2003 (2010 for lake trout) to investigate causes for the OC declines. Growth dilution was a major factor influencing the decrease of OCs in lake trout, round whitefish and possibly zooplankton most notably in the early 2000s. A decline in lipids of most fish species also contributed to OC declines, although no such change was evident for zooplankton. It is suspected that increases in fish populations or climate variations over the 1990s, may have contributed towards a shift in plankton community composition. From 1991 to 1999, CPUE increased for 7 of the fish species and declined for 2 others. Concurrently, the zooplankton community shifted from an abundance of *C. scutifer* in 1993 to dominance by *D. pribilofensis* in 2001. Nitrogen and carbon stable isotope data suggested that food web interactions for most fish species have not changed over time. Although concentrations of OCs have declined in many fishes, the "rate" of OC transfer (using slopes of log OC vs. nitrogen isotope ratios) through the food web was greater in 2001 than in 1993. Overall, the declines in OC concentrations in the fish from Lake Laberge occurred concurrently with changes in their growth, lipid, and abundance, suggesting that ecosystem responses to the closure of the fishery were in part responsible for the lower contaminants in these fishes. As a result of this study, the Yukon government rescinded the health advisory for limiting the consumption of fish from Lake Laberge.

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1. Introduction

It is well established that a wide range of chlorinated pesticides (e.g. toxaphene (CHB), chlordanes (CHL), hexachlorocyclohexanes (HCH)) and industrial chemicals such as polychlorinated biphenyls (PCB) undergo long-range atmospheric transport and deposition in the sub- and high Arctic regions where they biomagnify up aquatic food chains (Barrie et al., 1992; Muir et al., 1992; Van Dijk and Guicherit, 1999; CACAR, 2003). Lake Laberge in the Yukon Territory was studied intensively in the early 1990s because of abnormally high levels of these organochlorine (OC) contaminants in fish, relative to surrounding lakes. Researchers showed the primary cause was not due to point source pollution (Nordin et al., 1993; Diamond et al., 2005), but rather Lake Laberge had a longer food chain (Kidd et al., 1998) and, as a result, its top predators had higher-than-average

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contaminant concentrations when compared to other regional lakes (Kidd et al., 1995b). It was hypothesized that the long food chain in this lake was an effect of the commercial fishery that existed on the lake for over 100 years.

Lake Laberge was commercially exploited since the summer of 1898 primarily for lake trout and whitefish, although incidental catches of other species such as Arctic grayling and burbot were also kept for sale or consumption (Seigel and McEwen, 1984; Thompson, 1996). After high OCs were measured in lake trout and burbot, Health Canada placed a health advisory on the consumption of fish from Lake Laberge causing a closure of the fishery in 1991–92. Since the first study of contaminants in Lake Laberge in 1991 (Connor and Sparling, 1996), concentrations of OCs have dropped dramatically in fish (Ryan et al., 2005). Several hypothesis have been put forth to explain the trends (Ryan et al., 2005), including biomass/growth dilution or shifts in trophic levels of predators and prey, or events unrelated to the cessation of the fishery, such as declines in atmospheric contaminant concentrations over the same period. We hypothesized that the closure of the 100-year commercial fishery contributed to the decline in OCs through biomass/growth dilution.

Several changes have occurred in the fish populations of Lake Laberge after the closure of the fishery. Fish abundances have increased from 1991 to 1999 for seven out of ten of the species (Foos, 2001). Lake trout also appear to have increased growth rates (Thompson, 1996) and were larger in body size (Ryan et al., 2005). An increased rate of growth, and recruitment, in fish following exploitation is an often-observed response (Hewson, 1955; Healey, 1975, 1978; Mills et al., 2000) but, to our knowledge, the effect of the closure of a fishery on contaminant levels in a sub-Arctic food web has not been studied.

Several key morphological and biochemical parameters can be used to examine the hypothesis that biomass/growth dilution contributed to the decreases in fish contaminant levels. Body size characteristics (length, weights) have been linked to contaminant concentrations in fish (Larsson et al., 1993; Kidd, 1996; Ryan et al., 2005) and it has been previously demonstrated that growth rates have a major influence on contaminant levels in biota (Thomann, 1989; Larsson et al., 1991; Sijm et al., 1992). OC concentration trends in Great Lakes lake trout were tied to fluctuations in the growth of alewives, and their varying diet, suggesting that food web interactions regarding food intake play a strong role in contaminant regulation (Borgmann and Whittle, 1991; Madenjian et al., 1999). Growth rates influence contaminant loads in organisms through biomass or growth dilution (Sijm et al., 1992). As fish increase their rate or size of prey consumption, they increase their growth rate (Matuszek et al., 1990; Pazzia et al., 2002) resulting in a larger biomass as well as potential changes in bioenergetics of assimilation, excretion or food conversion efficiencies (Persson and Greenberg, 1991; Pazzia et al., 2002). The body size/metabolism of the animal may increase in relation to contaminant intake or contaminant storage in lipids thus 'diluting' the contaminants within the body (Thomann, 1989; Clark and Mackay, 1991; Sijm et al., 1992; Hebert et al., 1997). Variability in tissue lipid content during growth may also cause changes in contaminant concentrations in aquatic organisms and there is significant evidence supporting the link between OC biomagnification and lipids (Thomann, 1989; Larsson et al., 1991, 1993).

An alternate hypothesis for the changes in OCs in Lake Laberge fishes relates to changes in food web structure. Although fishing exploitation may increase growth rates of targeted species, it also has the capacity to restructure a food web by eliminating predators (or by-catch species) thereby changing the balance of the ecosystem (Mills et al., 2000). Levels of contaminants are significantly related to the trophic position of aquatic animals (Kidd et al., 1995a, 1998; Fisk et al., 2001). Species at higher trophic levels have higher contaminant levels (of biomagnifying compounds) compared to those lower in the food web (Rasmussen et al., 1990; Kidd et al., 1995a; Zaranko et al., 1997). A change in prey or predator populations within an ecosystem may result in changes in dietary habits and subsequent OC accumulation. Since fish population abundances appear to have changed over time in Lake Laberge (Foos, 2001), it is possible that changes have also occurred within the food web structure. Such a hypothesis may be evaluated by comparing changes in carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) isotope ratios, because they are used to assess food sources and relative trophic level, respectively (Vander Zanden and Rasmussen, 1996; Das et al., 2000; Campbell et al., 2000).

Although studies of OC concentrations in biota often monitor a few key species, far fewer assess entire ecosystems for changes to abiotic inputs, productivity, dietary habits, fish abundance and OC concentrations across trophic levels. There are no other known studies of the effects of the closure of a commercial fishery on the contaminant loads in the food web of a sub-Arctic lake. This research focuses on temporal changes in six OC groups in a sub-Arctic lake food web and the relationship between biotic parameters and the concentrations of OCs over the span of more than 10 years (assessment of abiotic influences from atmospheric loading and primary productivity as determined through sediment core analysis (Ryan et al., in preparation) are reviewed in Ryan (2006). The main objective of Ryan et al. (2005) was to review differences in OC between two top fish predators (lake trout and burbot) among three lakes in the Yukon region, whereas the primary objective of this research was to determine the most probable biotic causes of the recent declines observed in OC concentrations specifically in fish from Lake Laberge using analyses of temporal trends of OC concentrations in seven species of fish and in bulk zooplankton, morphological (i.e., length, weight, age), biochemical (i.e., lipid content) and population data for fish, and food web structure using $\delta^{13}C$ and $\delta^{15}N$.

2. Materials and methods

2.1. Sample collections

Lake Laberge is approximately 40 km north of the city of Whitehorse in the Yukon Territory (61°11′N, 135° 12′W; Fig. 1). The lake is classified as oligotrophic and has had little change in this status over time as determined by limnological data collected and presented several times over the past two decades (Jack et al., 1983; Shortreed and Stockner, 1983; Kirkland and Gray, 1986; Kidd, 1996; Thompson, 1996; Foos, 1998; Ryan et al., 2005).

Nine fish species were collected from Lake Laberge including lake trout (*Salvelinus namaycush*), burbot (*Lota lota*), inconnu (*Stenodus leucichthys*), northern pike (*Esox lucius*), lake whitefish (*Coregonus clupeaformis*), least cisco (*Coregonus sardinella*), longnose sucker (*Catostomus catostomus*), round whitefish (*Prosopium cylindraceum*) and broad whitefish (*Coregonus nasus*) several times between 1992 and 2010 (Table 1). The primary collection years for all fish were 1992, 1993, 2000 and 2001 with a few species collected in 1996, 1998 and 1999. Burbot were collected in both spring and summer (March/April, July to September) while all other species were collected during the mid to late summer months (July to September) with the exception of 1992 lake trout which were collected in March and June. Additional lake trout were collected each summer from 2002 to 2010 (except for 2006). All 1992–1993 data were originally compiled as a single year and are subsequently referred to only as "1993" data in this study. Only seven of the nine species of fish captured from Lake Laberge were used for analysis. The inconnu, broad whitefish and Arctic grayling were not assessed as part of this study because of insufficient data for assessment of temporal trends.

Burbot were caught using long line angling while all other species were captured using small mesh index nets as described in Thompson (1996) and Foos (1998). These small mesh index nets included two gangs of nets each with three panels 23 m long, 2.4 m depth using mesh sizes of 3.8 cm, 6.4 cm and 7.6 cm. Net set times, mesh sizes and capture locations were kept as similar as possible over time. Small mesh nets were substituted with angling and tackle for
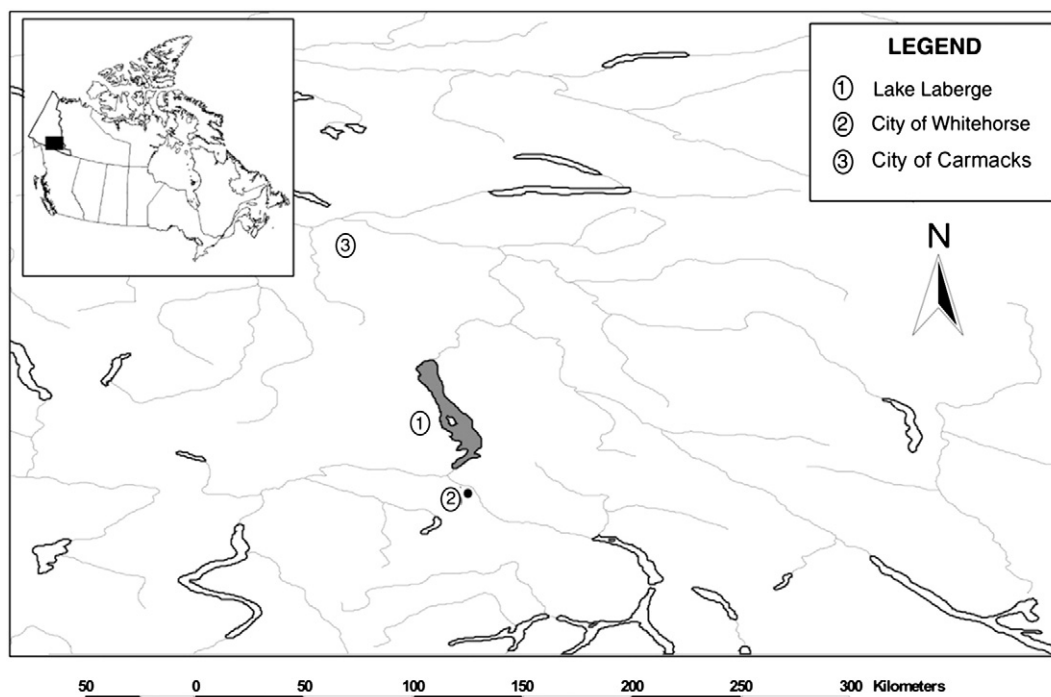
Fig. 1. Location of Lake Laberge in the Yukon Territory, Canada.

collection of lake trout from 2008 to 2010 (due to cost constraints). Yukon Territorial Government fishing quotas limited capture of lake trout to 15 or less for some years.

All fish were transported on ice to the laboratory where they were weighed and measured (total length and fork length); sex was determined and otoliths (or cleithra for pike) were removed for age analysis (Table 1). Dorsal muscle samples with skin removed (and liver for burbot) were taken, wrapped in solvent-washed aluminum foil and then shipped frozen to the Freshwater Institute in Winnipeg and stored at −20 °C.

Condition factors (K) were calculated using [mass $(g) \times 10^5 /$ length $(mm)^3$].

In addition, the Yukon Territorial Government (Renewable Resources) conducted population assessments of the fish stocks of Lake Laberge in 1991 and 1999 (Thompson, 1996; Foos, 1998, 2001). Changes in catch per unit effort (CPUE) were calculated as CPUE factors according to the following equation:

$$\text{CPUE factor} = \text{CPUE (\# in gillnet/hour)}_{1999} / \text{CPUE (\# in gillnet/hour)}_{1991}.$$

Burbot were excluded from CPUE analyses as the small mesh netting method was not designed for the capture of this species (A. Foos, Yukon Territorial Government, Whitehorse, pers. comm.).

Invertebrates were collected with small Ponar benthic dredges and shoreline sampling with kick nets during the months of June and August of 1992–1994 and June and July of 2000 and 2001. Macro-invertebrates were identified to Family, and then frozen in Whirl Paks© for storage prior to stable isotope analyses.

Zooplankton samples were also collected for community composition analyses in late July and early August of 1993 and 2002 using a 250 μm mesh, 40 cm diameter net and vertical tows through the top 15 m of the water column at different locations on the lake. The 1993 collection involved 6 vertical hauls through the upper 15 m of the water column in 2 locations for one pooled sample (K. Kidd, unpublished results) while the 2002 data involved 6 vertical hauls at 15 m (2 pooled samples) at locations in close proximity to the 1993 collection zone. These zooplankton samples were preserved in buffered formalin for species identification and enumeration (A. Salki, DFO, Winnipeg). Data were compared to samples collected in 1982 (Kirkland and Gray, 1986). However due to the low number of samples and lack of raw data for the 1982 collections, a direct statistical analysis between community compositions of 1993 and 2002 was not possible.

2.2. Organochlorine analysis

Methods for organochlorine extractions followed the same procedures reported in Ryan et al. (2005) and Kidd et al. (1998). Skinless fish muscle was homogenized, mixed with anhydrous sodium sulfate or Hydromatrix and spiked with PCB-30 and octachloronapthalene (OCN) as internal standards. The tissues were processed in a Dionex ASE300 with dichloromethane (DCM) and hexane (mixed 50:50). The extract was reduced in a warm water bath and a rotovap, filtered and a 1 mL sub-sample was measured gravimetrically in a pre-weighed aluminum dish for calculation of lipid content. The remaining extract was processed through manual gel permeation chromatography (GPC) followed by fractionation on a 1.2%-deactivated Florisil column.

Burbot livers and invertebrates (including zooplankton) were extracted with hexane using the Ball Mill method as described by Ryan et al. (2005), spiked with the internal standards (PCB-30 and OCN), diluted and sub-sampled to gravimetrically determine tissue lipid (%). A portion of the remaining sample was fractionated on a 1.2%-deactivated Florisil column as described for fish tissues. These fractions were concentrated down to 1 mL or 500 μL and aldrin was added as a volume corrector prior to GC analysis. Duplicates were run every 10 samples to verify consistency of the results. A total of 103 PCB congeners (including co-eluting congeners) and 40 OC pesticides were quantified using external standard mixtures (Ultra Scientific, North Kingstown, RI), which were run after every 5 samples.

All extracts were injected onto a Varian Star 3400 GC equipped with an electron capture detector (Ni^{63} electron source) and an 8100 auto-sampler. The column used was a J&W Scientific (Agilent Tech) DB-5, 60 m, 250 μm ID with TMP as the solvent, Hydrogen

Table 1

Morphological parameters (mean ± SE) for fishes collected from Lake Laberge; 1993–2010. Lipid content relates to liver tissue for burbot and to muscle tissue for all other fishes.

Species	Year	N[a]	Age	Length	Weight	K[b]	% lipid
Lake trout	1993	24 (8)	15 ± 2	480 ± 16	1523 ± 191	1.2	7.9 ± 0.9
	1996	13 (9)	22 ± 5	491 ± 27	1867 ± 415	1.4	9.6 ± 1.4
	2000	6	12 ± 2	590 ± 44	2753 ± 812	1.2	3.7 ± 0.8
	2001	17	14 ± 2	646 ± 25	3323 ± 388	1.2	4.9 ± 0.5
	2002	5 (2)	12 ± 4	61 ± 6 17	2874 ± 337	1.2	4.2 ± 0.9
	2003	8 (0)	n/a	593 ± 35	2485 ± 427	1.1	4.7 ± 0.8
	2004	6 (5)	13 ± 2	602 ± 28	2467 ± 389	1.1	8.7 ± 1.6
	2005	10 (9)	14 ± 2	606 ± 31	2672 ± 426	1.1	2.0 ± 0.4
	2007	9 (7)	14 ± 2	674 ± 32	3296 ± 398	1.0	1.2 ± 0.3
	2008	10 (9)	12 ± 2	580 ± 25	2216 ± 259	1.1	2.3 ± 0.4
	2009	10 (9)	10 ± 1	538 ± 18	1824 ± 164	1.2	3.0 ± 0.3
	2010	10	9 ± 1	547 ± 15	1878 ± 159	1.1	2.3 ± 0.4
Burbot	1993	29 (27)	13 ± 1	569 ± 21	1395 ± 129	0.7	41.8 ± 2.1
	1996	15 (0)	n/a	485 ± 27	922 ± 145	0.7	36.8 ± 2.3
	1999	11	17 ± 1	648 ± 22	1518 ± 153	0.5	53.8 ± 2.0
	2000	18	15 ± 1	626 ± 24	1399 ± 135	0.5	27.9 ± 2.2
	2001	30	17 ± 1	635 ± 19	1383 ± 111	0.5	23.5 ± 2.5
Northern pike	1993	8	7 ± 1	634 ± 27	1825 ± 257	0.7	0.4 ± 0.0
	2000[c]	5 (4)	7 ± 1	686 ± 75	2428 ± 843	0.7	0.3 ± 0.0
	2001[c]	6 (0)	n/a	778 ± 25	3170 ± 303	0.7	0.3 ± 0.1
Lake whitefish	1993	30 (26)	10 ± 1	318 ± 15	432 ± 46	1.1	2.8 ± 0.3
	1998	5	9 ± 2	335 ± 16	428 ± 79	1.1	1.4 ± 0.1
	2000	14 (13)	9 ± 1	319 ± 8	331 ± 16	1.0	0.5 ± 0.1
	2001	14 (0)	n/a	332 ± 10	381 ± 44	1.0	0.6 ± 0.1
Cisco	1993	7 (3)	4 ± 0	172 ± 19	58 ± 15	0.9	5.0 ± 1.1
	2000[d]	22 (20)	4 ± 1	211 ± 5	86 ± 6	0.9	0.7 ± 0.1
	2001[d]	2	5 ± 2	233 ± 18	85 ± 15	0.7	0.9 ± 0.5
Longnose sucker	1993	12 (3)	24 ± 4	433 ± 6	852 ± 32	1.0	1.0 ± 0.2
	2000[d]	12	17 ± 1	474 ± 9	979 ± 44	0.9	0.7 ± 0.2
	2001[d]	2	21 ± 2	448 ± 8	770 ± 30	0.9	0.6 ± 0.3
Round whitefish	1993	7 (1)	15	222 ± 43	168 ± 107	0.8	2.1 ± 0.2
	2000	5 (4)	9 ± 1	314 ± 20	290 ± 58	0.9	0.4 ± 0.1
	2001	38 (20)	5 ± 1	268 ± 10	170 ± 21	0.8	0.6 ± 0.0

[a] Numbers of samples in brackets applies to mean ages.
[b] K is Fulton's Condition Factor (see Section 2.1).
[c] Data combined for statistical analysis of age.
[d] Data combined for statistical analysis of all morphometric parameters.

Ultra High Purity (UHP) carrier gas (at ~2 mL/min), nitrogen UHP make-up gas (at ~50 mL/min). Injection and temperature programs conditions were as follows:

Injector temperature 80 °C, detector temperature 300 °C, initial temperature 100 °C (hold time 2 min), 1st temperature program increasing to 150 °C at 15 °C/min, 2nd temperature program increasing to 265 °C at 3 °C/min.

Recoveries of the internal standard PCB-30 and OCN were 64.6% ± 1.5 SE and 75.8% ± 1.7 SE for all fish samples combined (max. n = 373), respectively. Correction factors were not applied to the study sample results. Samples were analyzed for ΣDDT (p,p′-DDT, p,p′-DDE, p p′-DDD, o,p′-DDT, o,p′-DDE and o,p′-DDD), ΣHCH (α- β- and γ-HCH isomers), ΣCHL (sum of all technical chlordane compounds and related metabolites, including heptachlor epoxide), ΣCBz (1,2,4,5-TCB, 1,2,3,4-TCB, P5-CBz and HCBz), and ΣPCB (CB1, 3, 4/10, 7, 6, 8/5, 19, 18, 17, 24/27, 16/32, 26, 25, 31, 28, 33, 22, 45, 46, 52, 49, 47, 48, 44, 42, 41/71, 64, 40, 74, 70/76, 66, 95, 56/60, 91, 84/89, 101, 99, 83, 97, 87, 85, 136, 110, 82, 151, 144/135, 149, 118, 134, 114,131, 146, 153, 132, 105, 141, 130/176, 179, 137, 138, 158, 178/129, 175, 187, 183, 128, 185, 174, 177, 171, 156, 201/157, 172/197, 180, 193, 191, 200, 170, 190, 198, 199, 196/203, 189, 208, 195, 207, 194, 205, 206, 209).

2.3. Stable isotope analyses

Replicate 1 mg samples of oven dried and powdered tissue (skinless muscle for fish, whole bulk zooplankton and soft body parts of other aquatic invertebrates) were sent for stable isotope analysis at the University of Waterloo, Earth Sciences Department. Samples were analyzed for $\delta^{15}N$ and $\delta^{13}C$ on an Isochrom Continuous Flow Stable Isotope Mass Spectrometer (Micromass) coupled to a Carlo Erba Elemental Analyzer. Results were corrected to laboratory nitrogen standards IAEA-N1 and IAEA-N2 (both ammonium sulfate) and carbon standards IAEA-CH6 (sugar), EIL-72 (cellulose) and EIL-32 (graphite). EIL-70b is a lipid extracted/ball-milled fish material, and was used as an external monitoring standard. The precision for clean ball-milled standard material was ± 0.2‰ for $\delta^{13}C$ and ± 0.3‰ for $\delta^{15}N$. A known isotope material (Pharmamedium) was run with every 10 samples as a blind, duplicate test of the lab accuracy. Nitrogen and carbon isotopes ratios are reported in 'delta (δ)' notation in parts per thousand, or per mil (‰) according to the following equation:

$$\delta^{15}N \text{ or } \delta^{13}C = \left(\left(Ratio_{sample}/Ratio_{standard} \right) - 1 \right) \times 1000$$

where Ratio = $^{15}N/^{14}N$ or $^{13}C/^{12}C$.

Trophic levels (TL) of individual fish were calculated using the following equation:

$$TL \text{ fish}_{year} = \left(\left(\delta^{15}N \text{ fish}_{year} - \delta^{15}N \text{ snails}_{year} \right)/3.4‰ \right) + 2.$$

Lymnaeidae snails (TL = 2) were used to adjust all fish data for temporal differences in $\delta^{15}N$ at the base of the food chain (Post et al., 2000) prior to statistical analyses. Snails were selected because they were a longer-lived primary consumer present in this lake. Delta ^{13}C values were adjusted for differences in lipid content, however

significant correlations between lipid and ^{13}C were dependent on the species and year. Since δ^{13}C is known to correlate with body lipid (Post et al., 2007), and lipid content for some Lake Laberge species changed over time, the lipid adjusted ^{13}C values are presented.

2.4. Statistical analysis

Statistical analyses were conducted using Systat® with α set to 0.05 for all tests. For each fish species, temporal differences in morphological parameters (length, weight, age, condition factor), trophic level and % lipid were examined using analysis of variance (ANOVA; alpha = 0.05) followed by Tukey's post-hoc analyses to test for differences between individual years (Kruskal Wallis non-parametric test was used if assumptions for the ANOVA were not met followed by Dwass–Steel–Chritchlow–Fligner test for post hoc pairwise comparisons). Log$_{10}$ values were used if the transformation improved the distribution of the data. Due to low sample sizes for ages in some years, some year categories were removed prior to the ANOVA of mean ages over time (e.g., Northern pike 1998, Lake trout 2002 and 2003). The 2000 and 2001 datasets were also combined (denoted as 2000–2001) for statistical analyses of stable isotopes (when n<5) and for all multiple linear regressions to improve sample sizes. For three species (least cisco, longnose suckers, and round whitefish), data were combined to test for differences in OC and morphometric parameters over time only where n<5 for a specific parameter in either 2000 or 2001. Data were not pooled for tests of differences in OC or morphometric parameters over time for all remaining species sampled.

Major OC groups were tested for significant differences within fish species among years using ANOVA. OC concentrations were not adjusted for the influence of lipid using analysis of covariance (ANCOVA) nor were concentrations expressed on a lipid weight basis because lipid was not always a significant predictor of OC concentrations within a species and year, and/or the regression slopes of OC versus lipid were significantly different across years (lipid∗year interaction in ANCOVA). For burbot data only, some OC concentrations were adjusted using ANCOVA with age as a covariate, because there was a significant linear relationship between the OC and age within years and slopes across years were not significantly different. Wet weight OC concentrations were used in the statistical analyses and all OC data were log$_{10}$ transformed prior to statistical analyses to normalize the distribution of the data.

For fish that exhibited significant differences in OC concentrations and morphology among years, multiple linear regressions (MLR) were used to examine relationships between OC concentrations and biological parameters including tissue lipid (%), log$_{10}$ weight and log$_{10}$ length. This included least cisco, round whitefish, longnose sucker, lake whitefish, lake trout and burbot. Northern pike were excluded due to a large leverage in two points of data (based on Studentized residuals >4), along with one outlier and low sample numbers. Based on research from Kidd (1996) and from the results of preliminary single regressions and stepwise regressions, models for longnose sucker included log length and tissue lipid (%) while the models for round whitefish included weight and tissue lipid. Cisco and lake whitefish regression models included log weight and tissue lipid as variables. For this study, regressions were run to determine the most influential factors on OC concentrations for each fish species within each year. Results for lake trout and burbot (1993 to 2003) have already been reported in Ryan et al. (2005). However, some results for burbot (Ryan et al., 2005) were re-analyzed due to a recent addition of 16 fish from 1993 (Kidd, 1996), and additional lake trout data (2004–2010) are presented.

The slope of the log-transformed OC concentrations versus δ^{15}N, provides an estimate of contaminant biomagnification through the food web (Kidd et al., 1995a; Kiriluk et al., 1995). General linear models were used, with δ^{15}N and year as factors (and a δ^{15}N∗year interaction term) to compare the changes in slopes of OC concentrations from biota sampled between 1992–1993 and 2000–2001. The OC-δ^{15}N relationships for the 2000 and 2001 datasets were also tested for homogeneity of slopes prior to pooling the data for these two years.

3. Results

3.1. Temporal trends in OC contaminants, lipids and morphology of fish

3.1.1. Lake trout

Mean concentrations of all six OC groups in lake trout decreased significantly from 1993 to 2010 (Table 2). An analysis of data from 1993 to 2003 is described in more detail by Ryan et al. (2005), but the additional (more recent) data also showed a declining trend in all OC concentrations with evidence that the concentrations may be leveling (declining at a slower rate). Results of the Tukey's pairwise comparisons indicated no significant differences (p<0.05) in OC among years from 2003 to 2010 for ΣCBz, ΣHCH, ΣCHL, and no differences between the years 2005 and 2010 for ΣCHB, ΣDDT and ΣPCB (although mean concentrations of these 3 OCs are lowest in 2010 compared to any other year from 2005 to 2009 suggesting a continuing declining trend). The decreases in OC concentrations from 1993 to 2003 were concurrent with significant increases in lake trout lengths and weights up to 2003, and to a lesser extent, significant decreases in % lipid as noted in Ryan et al. (2005) (Table 3). The MLR analysis of more recent lake trout data indicated lipid was not a significant factor correlating with ΣCHB, ΣCHL, ΣDDT and ΣPCB in any year from 2003 to 2010 (p>0.05 for all years), although it was more consistently a factor for ΣCBz and ΣHCH (p<0.05 for 5 of 7 years sampled). Fish weight continues to be a periodic, significant factor correlating with all six OC groups in 2 or 3 of the years (depending on the OC) between 2003 and 2010. It is notable that lake trout captured in 2007 had the highest mean fish weight and length, but only the third oldest mean ages, behind the smaller sized yet older trout collected in 1993 and 1996 which were a magnitude higher in concentrations of all six OC groups (Tables 1 and 2). Lake trout weights, lengths and ages appeared to decline from 2008 to 2010, correlating with a change in the fishing method from nets to angling, although there were no significant differences in these parameters among the fish collected in the 2000s. Lipid content in trout was more variable among years with a general decline in 2005 followed by an overall leveling to 2010 compared to earlier years (Table 1).

3.1.2. Burbot

As described in Section 2.4, 16 additional samples from 1993 were included in the statistical analyses and as a result, some of the temporal trends have changed slightly from those reported by Ryan et al. (2005).

Chlordane, DDT and PCB were shown to have a positive linear relationship with age (ΣCHL (p = 0.01, r^2 = 0.09; ΣDDT p = 0.05, r^2 = 0.05; ΣPCB p<0.01, r^2 = 0.16). After adjusting for the effects of age, only the ΣDDT concentration showed a significant decrease over time (p = 0.03). Liver concentrations of ΣHCH (unadjusted for age) were also significantly lower in 2001 when compared to 1993. No temporal trends were observed for ΣCBZ and ΣCHB (Table 3).

Mean burbot ages increased significantly from 1993 to 2001 (p = 0.01; Tables 1 and 4). Mean fork lengths also increased significantly (p<0.01), while mean weights had no significant change from 1993 to 2001 (p = 0.09). Lipid contents of the livers and condition factors (K) decreased significantly (p<0.01) from 1993 to 2001.

MLR analysis for the 1993 and 2000–2001 datasets found significant predictors for each OC group within each period. Tissue lipid (%) and log age were the most consistent, significant factors correlating with OC concentrations in burbot in both the 1993 and 2000–2001 data.

Table 2
Mean (±SE) organochlorine concentrations (ng/g ww) by group, species and year for fishes (except broad whitefish and inconnu) and zooplankton collected from Lake Laberge; 1993–2010.

Species	Year	N	$\sum$ CBz	$\sum$ HCH	$\sum$ CHL	$\sum$ DDT	$\sum$ PCB	$\sum$ CHB
Lake trout	1993	24	3.9 ± 0.6	4.7 ± 0.8	47.6 ± 8.8	391.5 ± 132.7	328.3 ± 121.5	311.0 ± 62.4
	1996	13	4.9 ± 1.2	6.5 ± 1.8	53.4 ± 13.7	236.5 ± 41.4	209.3 ± 52.1	212.2 ± 28.3
	2000	6	2.3 ± 0.6	2.3 ± 1.1	22.4 ± 5.8	96.5 ± 14.2	139.0 ± 60.9	207.3 ± 49.9
	2001	16	2.1 ± 0.2	0.8 ± 0.1	26.4 ± 5.1	89.5 ± 14.0	139.7 ± 53.8	154.2 ± 60.5
	2002	5	1.2 ± 0.3	1.6 ± 0.5	7.3 ± 1.6	54.5 ± 11.6	48.6 ± 8.8	139.2 ± 16.9
	2003	8	1.2 ± 0.3	0.5 ± 0.1	7.4 ± 2.2	61.5 ± 8.6	81.0 ± 29.8	179.3 ± 42.8
	2004	6	0.5 ± 0.1	0.2 ± 0.04	7.5 ± 2.0	93.8 ± 24.7	48.7 ± 14.0	79.9 ± 21.2
	2005	10	0.4 ± 0.1	0.16 ± 0.03	2.8 ± 0.4	50.9 ± 9.6	28.9 ± 6.4	34.5 ± 6.3
	2007	9	0.3 ± 0.04	0.10 ± 0.03	5.3 ± 1.4	44.0 ± 10.0	37.4 ± 8.6	25.8 ± 4.9
	2008	10	0.8 ± 0.1	0.18 ± 0.03	4.0 ± 0.9	70.1 ± 13.1	50.2 ± 11.7	24.5 ± 5.3
	2009	9	0.5 ± 0.1	0.14 ± 0.02	2.4 ± 0.3	37.5 ± 6.9	27.4 ± 5.0	33.0 ± 4.6
	2010	10	0.3 ± 0.04	0.12 ± 0.02	1.5 ± 0.2	40.4 ± 3.8	12.1 ± 1.2	24.9 ± 4.4
Burbot[a]	1993	29	26.0 ± 2.8	32.2 ± 3.2	235.3 ± 31.7	2393.5 ± 314.8	1287.4 ± 161.1	2414.0 ± 264.7
	1996	15	40.1 ± 7.1	42.1 ± 7.5	270.2 ± 52.1	2675.4 ± 475.6	1598.3 ± 292.1	2659.2 ± 443.0
	1999	11	41.6 ± 5.3	50.3 ± 7.2	247.8 ± 36.7	2849.8 ± 361.7	1625.4 ± 219.2	3053.5 ± 364.7
	2000	17	26.3 ± 3.4	21.1 ± 3.0	219.0 ± 24.9	2069.6 ± 216.5	1172.1 ± 109.0	1281.1 ± 111.9
	2001	27	34.0 ± 3.1	8.4 ± 0.8	258.7 ± 29.0	2101.9 ± 381.0	1753.4 ± 211.2	1106.5 ± 150.1
Northern pike	1993	8	0.18 ± 0.02	0.12 ± 0.01	1.0 ± 0.1	8.4 ± 1.1	7.5 ± 1.1	9.7 ± 1.4
	2000	5	0.13 ± 0.01	0.06 ± 0.01	0.5 ± 0.1	13.7 ± 3.9	8.1 ± 2.9	1.8 ± 0.4
	2001	5	0.6 ± 0.3	0.08 ± 0.01	8.0 ± 3.1	13.8 ± 3.4	7.1 ± 1.9	4.6 ± 1.1
Lake whitefish	1993	30	1.2 ± 0.1	1.6 ± 0.2	8.9 ± 1.0	82.5 ± 11.6	66.6 ± 10.4	62.1 ± 8.5
	1998	5	0.4 ± 0.04	0.3 ± 0.1	3.0 ± 0.3	25.0 ± 2.4	5.1 ± 0.7	42.8 ± 7.7
	2000	14	0.3 ± 0.04	0.13 ± 0.01	0.6 ± 0.1	15.7 ± 1.8	9.3 ± 2.7	2.5 ± 0.6
	2001	14	0.3 ± 0.1	0.12 ± 0.02	1.0 ± 0.4	17.1 ± 3.8	20.4 ± 6.0	4.8 ± 2.1
Cisco	1993	7	1.4 ± 0.3	1.6 ± 0.4	6.4 ± 1.4	31.8 ± 6.0	21.2 ± 3.3	68.3 ± 13.7
	2000–2001[b]	23[c]	0.28 ± 0.07	0.18 ± 0.04	0.8 ± 0.3	7.1 ± 1.5	3.8 ± 0.6	3.5 ± 0.6
Longnose sucker	1993	12	0.3 ± 0.04	0.4 ± 0.1	2.2 ± 0.4	16.1 ± 2.2	13.6 ± 2.1	17.5 ± 3.5
	2000–2001[b]	14	0.19 ± 0.04	0.11 ± 0.02	0.5 ± 0.1	16.8 ± 2.8	14.1 ± 4.5	1.5 ± 0.4
Round whitefish	1993	7	0.8 ± 0.2	0.9 ± 0.2	3.3 ± 0.8	22.6 ± 4.2	17.4 ± 3.7	25.8 ± 4.5
	2000–2001[b]	24[d]	0.4 ± 0.1	0.16 ± 0.02	2.6 ± 0.9	5.7 ± 1.6	3.5 ± 0.7	1.4 ± 0.3
Zooplankton	1993	4	0.5 ± 0.1	1.3 ± 0.4	1.4 ± 0.5	4.1 ± 0.8	5.3 ± 0.4	15.0 ± 3.9
	2000–2001[b]	11	0.08 ± 0.01	0.24 ± 0.03	0.16 ± 0.03	1.1 ± 0.2	1.67 ± 0.8	2.8 ± 0.5

[a] Concentrations are from burbot liver tissue.
[b] Data were combined when samples analyzed for OC from either 2000 or 2001 had n<5 (Section 2.4).
[c] 21 samples in 2000 and 2 samples in 2001 of CIS were analyzed for OC.
[d] 4 samples in 2000 and 20 samples in 2001 of RWF were analyzed for OC.

3.1.3. Northern pike

Mean concentrations of OCs in northern pike had no consistent trends from 1993 to 2001 (Tables 2 and 3). The ΣHCH and ΣCHB levels decreased significantly over time (p<0.01) while ΣDDT and ΣPCB showed no significant change in concentrations from 1993 to 2001 (p=0.30 and 0.74, respectively). In contrast, ΣCBz and ΣCHL showed significant increases in concentrations (p=0.04 and 0.01, respectively).

Although there appears to be a trend towards higher lengths and weights of pike (Table 1), there were no significant differences in mean length, weight, age (combined 2000–2001 data), or lipid content between 1993 and 2001 (p=0.79, 0.08, 0.14, and 0.10, respectively; Tables 1 and 4).

3.1.4. Lake whitefish

OC concentrations in lake whitefish (Tables 2 and 3) decreased significantly from 1993 to 2001 (p≪0.01 for all 6 OC groups). There was a significant decrease in lipid content (p≪0.01) but no significant changes in age (p=0.55), length (p=0.87), weight (p=0.45) or condition (p=0.1) in lake whitefish over time (Tables 1 and 4).

In 1993, MLR showed that OC concentrations were significantly related to both lipid and log weight with two exceptions; neither ΣDDT nor ΣPCB were related to lipid content, and ΣHCH was not related to log weight. When compared to the MLR results for 1993, in 2000–01 lipid became a more significant predictor than log weight for 5 out of the 6 OC groups.

Table 3
Statistical results from temporal trend analysis of OC by fish over time for Lake Laberge biota from 1993 to 2001 (to 2003 for lake trout) and a list of probable causes attributing to the changes in OC concentrations.

Fish	# temporal points	ΣCBz	ΣHCH	ΣCHL	ΣDDT	ΣPCB	ΣCHB	Possible cause for OC trend
Lake trout	6	d	d	d	d	d	d	Decreased lipid, growth dilution, decrease in prey OC
Burbot	5	nc	d	nc	d	nc	d	Decreased lipid, older fish, decrease in prey OC, lower trophic level
Northern pike	3	i (m)	d	i	nc	nc	d	No consistent OC trends evident
Lake whitefish	4	d	d	d	d	d	d	Decreased lipid, decrease in prey OC
Cisco	2	d	d	d	d	d	d	Decreased lipid (related to only 2 OC), decrease in prey OC, some growth dilution
Longnose sucker	2	nc	d	d	nc	nc	d	Possible decline in prey OC, variation related to younger fish samples
Round whitefish	2	d	d	d	d	d	d	Decreased lipid, growth dilution, decrease in prey OC
Zooplankton	3	d	d	d	d	d	d	Possible growth dilution (higher densities), community species composition change

i = increase significantly (p<0.05).
d = decrease significantly (p<0.05).
nc = no change (p>0.05).
m = marginal significance (p=0.04–0.06).

Table 4
Significant temporal trends in morphological parameters and tissue lipid (%) over time in aquatic biota from Lake Laberge collected from 1993 to 2001 (to 2003 for lake trout).

Fish	# temporal points	Tissue lipid (%)	Length	Weight	Age	K
LT	6	d	i	i	nc	nc
BB	5	d	i	nc	i	d
INC	2	–	–	–	–	–
NP	3	nc	nc	nc	nc	nc
LWF	4	d	nc	nc	nc	nc
CIS	2	d	i	i	nc	nc
LNS	2	nc	i	nc	nc[a]	d
RWF	3	d	i	i	nc[a]	nc
BWF	1	–	–	–	–	–
Plankton	2	nc	–	–	–	–

i = increase significantly (p<0.05).
d = decrease significantly (p<0.05).
nc = no change (p>0.05).
[a] See Results section for caveats.

3.1.5. Least cisco

Concentrations of all 6 OC groups in least cisco decreased significantly (p<0.01) from 1993 to 2000–2001 (Tables 2 and 3).

Some morphological parameters of least cisco changed significantly over time (Tables 1 and 4). Although mean age and condition factor were not different from 1993 to 2000–2001 (p=0.71 and 0.57), lengths and weights increased (p<0.01 and 0.01, respectively) while tissue lipid content decreased during this period (p<0.01).

Results of the MLR showed a significant relationship of most OC concentrations with log weight and tissue lipid (%) for the 1993 samples. Tissue lipid was not as significant as log weight within the model except for ΣHCH. In contrast, % tissue lipid was the only significant variable for ΣCBz and ΣCHB in 2000–2001. Additionally, these models accounted for much less of the variance in 2000–2001 OC concentrations compared to those for the 1993 data.

3.1.6. Longnose sucker

No consistent temporal trends were evident across OC groups in longnose sucker from 1993 to 2000–2001 (Tables 2 and 3). There were no significant changes in concentrations of ΣCBz (p=0.14), ΣDDT (p=0.88) or ΣPCB (p=0.43). In contrast, significant decreases were noted for ΣHCH, ΣCHL and ΣCHB concentrations (all p<0.01).

Analyses of the morphological parameters showed that mean age in suckers was not significantly different among years 1993 to 2000–2001 (p=0.09). Although only three fish were aged in 1993 (Tables 1 and 4), the mean age was >3 years older than fish collected in 2000–2001. Mean lengths of longnose sucker showed a significant increase (p<0.01) from 1993 to 2000–2001, but there was no corresponding change in mean weights during this period (p=0.09). In conjunction, condition factors decreased significantly from 1993 to 2000–2001 (p<0.01). Although it appears that tissue lipid was declining (Table 1), there was no statistical difference over time (p=0.25).

Results of the MLR showed no consistent relationship between OC concentrations and lipid content. A significant relationship was found between 3 OC groups (ΣCBz, ΣHCH and ΣCHB) and lipid in 1993 but no such relationships were found for fish collected in 2000–2001. Additionally, there were no significant relationships between OCs and log lengths for longnose sucker in 2000–2001.

3.1.7. Round whitefish

Results for round whitefish demonstrated several significant decreases in contaminants from 1993 to 2000–2001 (Tables 2 and 3). The ΣCBz and ΣCHL concentrations had marginally significant declines (p=0.04 and 0.03, respectively) while declining trends in OC concentrations were more significant (all p<0.01) for the remaining 4 OC groups, ΣHCH, ΣDDT, ΣPCB and ΣCHB.

A temporal comparison of round whitefish ages was not done due to the low number of samples from 1993 (Tables 1 and 4). Both mean

length and weight of this species increased significantly from 1993 to 2001 (both p<0.01). In contrast, tissue lipid displayed a significant decrease (p>0.01) while condition factors showed no significant difference from 1993 to 2001 (p=0.31).

The MLR modeling did not find any significant predictors of OC contaminant groups for the 1993 data. In contrast, OC concentrations, for all groups but ΣHCH and ΣCHB, were significantly predicted by both lipid and log weight for the 2000–2001 samples.

3.1.8. Zooplankton

Concentrations of all 6 OC groups in zooplankton decreased significantly (p<0.01) between 1993 and 2000–2001 (Tables 2 and 3). In 2000–2001, mean concentrations of ΣCBz, ΣHCH, ΣCHL, ΣDDT, ΣPCB and ΣCHB were 16%, 18%, 11%, 27%, 32% and 19% of 1993 values, respectively.

Tissue lipid (%) content for zooplankton showed no significant change (p=0.62) between 1993 (1.5%±0.5% SE) to 2000–2001 (1.3%±0.2% SE). Multiple linear regressions showed no correlation between any of the 6 OC groups and tissue lipid in plankton (p>0.05 for all groups).

3.2. Zooplankton abundance and community composition

An increase in zooplankton densities, as well as a change in species composition, was found when comparing 1982, 1993 and 2002 vertical hauls (Table 5). In 2002, the calanoid copepod *Cyclops scutifer* had decreased abundance, accounting for less than half of the populations represented in 1982 and 1993 samples. Conversely, the cyclopoid copepod *Diaptomus pribilofensis* had increased in abundance and became the dominant zooplankton in Lake Laberge. Zooplankton densities were also higher in 2002 by 42% when compared to 1993.

3.3. Food web structure

Raw δ15N data for food web organisms from Lake Laberge increased slightly across all taxa from 1993 to 2001. Within the fishes, northern pike and lake whitefish showed the biggest increases in mean δ15N of 1.36 and 1.10‰, respectively (Table 6). With the exception of Trichoptera collected in 2001 (+3.05‰), most invertebrate taxa had similar δ15N when comparing 1993 to 2000–2001 data, although sample sizes were limited for some biota; δ15N increased by 0.5‰ for snails and by 0.4‰ for zooplankton from 1993 to 2000–2001.

Table 5
Total densities and community composition (as a percent of total counts) of zooplankton collected in Lake Laberge in 1982, 1993 and 2002.

Species	Body length (mm)	% of community 1982[b]	1993[c] (n=6×1)[d]	2002 (n=6×2)[d]
Cyclops scutifer	1.1–1.9[a] (0.9–1.2)[b]	84.6	80.0	35.3
Cyclopoid nauplii		0.0	0.4	0.1
Diaptomus pribilofensis	1.0–1.8[a] (0.9)[b]	9.6	0.9	46.6
Senecella calanoides	2.3[b]	0.1	0.0	0.5
Daphnia middendorffiana (also *D. pulex*)	1.3–3.0[a]	<0.1	0.0	7.4
Daphnia longiremis	0.8–1.2	4.0	18.0	6.9
Daphnia spp. (no ephippia)			0.0	2.8
Daphnia rosea		1.5		
Eubosmina longispina (also *E. coregoni*)		0.2	0.0	0.6
Total crustaceans per liter		n/a	9.73	13.89

[a] From Ward et al. (1959) and Pennak (1989).
[b] From Lake Laberge samples (Kirkland and Gray, 1986).
[c] From K. Kidd (pers. comm.).
[d] See Materials and methods section.

Table 6
Stable nitrogen isotope ratios (δ^{15}N, ‰) by species or invertebrate family over the sampling period.

Species	1993[a]			2000			2001		
	n	Mean	SD	n	Mean	SD	n	Mean	SD
Lake trout	34	12.38	0.84	6	13.49	0.33	17	13.18	0.39
Burbot	32	12.06	0.59	18	12.34	0.41	30	12.41	0.41
Northern pike	10	9.62	0.75	5	10.83	0.52	5	11.09	0.21
Lake whitefish	36	8.22	1.15	15	9.27	0.45	13	9.38	0.71
Cisco	14	7.87	0.47	22	8.68	0.39	2	8.99	0.61
Arctic grayling							1	8.97	
Longnose sucker	12	7.57	0.92	14	8.15	0.56	6	8.62	0.75
Round whitefish	8	7.16	0.76	6	8.05	0.87	20	7.65	0.85
Trichoptera	9	2.13	0.91				3	5.18	1.02
Chironomids	8	4.02	1.14				1	4.20	
Snails (Lymnaeidae)	6	2.05	1.01	17	1.96	1.36	18	3.21	0.56
Snails (Planorbidae and Valvatidae)	4	1.26	1.39						
Clams (Sphaeridae)	2	3.60	1.99	1	3.11		1	2.48	
Dipterans (other than chironomids)							3	2.18	0.38
Ephemeroptera							1	1.65	
Zooplankton	6	4.94	1.22	9	4.95	0.69	5	5.97	0.11

[a] Some data included from Kidd (1996).

In both 1993 and 2000–2001, lake trout and burbot were the top predators in Lake Laberge as indicated by their trophic levels (TL) of 4.8 to 5.2 (Table 7). During both time periods, northern pike occupied a TL (4.3 to 4.5) that was intermediate between the top predators and the other fish species collected. The remaining fish species (whitefish, sucker and cisco) had mean TLs (3.5 to 3.9) that were lower than those of the pike and higher than the littoral invertebrates or zooplankton collected. Calculated TL showed a significant decrease for burbot while all other species remained unchanged from 1993 to 2000–2001 (pooled; Table 7).

Fishes from Lake Laberge varied in their source of carbon but these among-species differences did not change from 1993 to 2001. Lake trout and cisco had mean δ^{13}C values that were between those of zooplankton and snails, indicating that these two species relied on more pelagic than benthic carbon. In contrast, longnose sucker and northern pike had δ^{13}C values that were similar to littoral, benthic invertebrates (Lymnaeidae), suggesting heavier reliance on littoral carbon. Other fishes had δ^{13}C that fell between lake trout and longnose sucker, suggesting that these species rely more on benthic than pelagic carbon. Delta^{13}C varied over time but only two species had significant differences from 1993 to 2000–2001 (Table 8). Burbot were about 0.5‰ lighter in δ^{13}C in 2001 when compared to 1993 ($p \leq 0.01$). In contrast, lake whitefish were significantly ($p <= 0.01$) heavier in δ^{13}C (by 1.6‰) in 2001 when compared to 1993. Zooplankton δ^{13}C was similar across all three sampling dates. In contrast, benthic invertebrate taxa were more variable but no consistent trends over time were found across all taxa. However, small sample sizes for all littoral taxa (with the exception of Lymnaeidae and Trichoptera) precluded

Table 7
Calculated trophic levels for fish from Lake Laberge collected in 1993 and 2000–2001 and ANOVA results for within species comparisons.

Species	1993			2000–2001			
	n	Mean	SE	n	Mean	SE	p
Lake trout	44	5.20	0.04	23	5.13	0.02	0.32
Burbot	32	5.07	0.03	48	4.88	0.02	<0.05
Northern pike	10	4.32	0.07	10	4.47	0.04	0.08
Lake whitefish	36	3.91	0.06	28	3.98	0.03	0.35
Cisco	13	3.81	0.04	24	3.80	0.02	0.76
Longnose sucker	12	3.72	0.08	20	3.64	0.04	0.61
Round whitefish	8	3.60	0.08	26	3.51	0.05	0.40

any statistical analyses. Mean δ^{13}C between 1993 and 2001 did not change for Lymnaeidae ($p > 0.05$) but decreased for Trichoptera ($p = 0.03$).

3.4. Fish populations

Results of fish population surveys showed increases from 1991 to 1999 in the catch per unit effort (CPUE) for 6 out of 9 species including lake trout, broad whitefish, northern pike, inconnu, longnose sucker and least cisco (Fig. 2). Two notable changes include increased CPUE factors for lake trout (top predator) and least cisco (primary prey of lake trout) of 2.1 and 3.8 fold, respectively. Both of the CPUE values (# and kg caught per gillnet hour; Ryan, 2006) increased for each of the previously mentioned species. Catch data for lake trout has more recently (2004 and 2009) indicated that CPUE values are similar to that observed in 1991.

In contrast to lake trout and cisco, decreases in CPUE values occurred for both round whitefish and lake whitefish. However, the kg caught per gillnet hour increased for lake whitefish. This indicated that fewer lake whitefish were captured but their total mass in 1999 was greater when compared to 1991.

Morphological data obtained by the Yukon Territorial Government (YTG) indicates that mean weights of lake whitefish may have increased from 371 g to 436 g (n = 255 and 446 for 1991 and 1999, respectively; Foos, 2001) and although YTG had not analyzed this data statistically, these averages fall well within the ranges presented in Table 1.

3.5. Trophic biomagnification of OCs

Log-transformed concentrations of OCs were significantly related to the δ^{15}N of biota from Lake Laberge in both 1993 and 2000–2001 (Table 9). Within both time periods, the slopes were lowest for the least lipophilic contaminant group ΣHCH and highest for the more lipophilic OCs (e.g. ΣCHB), while δ^{15}N accounted for 31 to 64% of the variability in the OC data. When these relationships were compared across years and within OC groups, significantly higher slopes were observed for the 2000–2001 data when compared to the results from 1993 (Table 9).

4. Discussion

Concentrations of CHL, DDT, HCH, CHB, PCB and CBz, decreased significantly in zooplankton and six species of Lake Laberge fish from 1993 to 2003, with declines continuing in lake trout to 2010 (the only species monitored since 2004). This contrasts research for Great Lakes fish which, shows that OC concentrations had stabilized and were no longer undergoing rapid declines as seen in the 1970s and 80s (Borgmann and Whittle, 1991; Stow et al., 1995). The purpose of our research was to investigate the causes behind the large temporal decline of contaminants in the Lake Laberge ecosystem that was not observed in other regional lakes (Ryan et al., 2005). This research has focused on specific morphological, biochemical and population variables as well as food web structure which are known to affect contaminant concentrations in fish.

4.1. Growth dilution and OC contaminants

Models and field data have shown that the growth rate of fish has a greater effect on OC bioaccumulation than lipid content, indicating that the level of contaminants in a fish is affected by more than just a lipid-partitioning mechanism (Thomann, 1989; Borgmann and Whittle, 1991; Larsson et al., 1991; Hammar et al., 1993; Stow et al., 1995). Contaminant burdens were related to the bioenergetics of the organisms (food conversion efficiency; Sijm et al., 1992) as well as the assimilation efficiency and excretion rate of organochlorine

Table 8
Stable carbon isotope ratios (δ^{13}C, ‰) by species or invertebrate family from 1993 to 2001. Values for fish δ^{13}C are lipid adjusted (except burbot).

Species	1993[a]			2000			2001		
	n	Mean	SD	n	Mean	SD	n	Mean	SD
Lake trout	32	−28.56	1.75	6	−29.29	2.36	17	−27.92	1.39
Burbot[b]	32	−26.89	0.73	18	−27.45	0.8	30	−27.41	0.47
Northern pike	10	−24.62	0.45	5	−24.69	0.71	5	−25.32	0.73
Lake whitefish	36	−26.25	1.56	15	−25.01	1.37	13	−24.65	1.41
Cisco	12	−28.99	3.02	22	−29.61	0.37	2	−29.90	0.29
Longnose sucker	12	−21.97	1.08	14	−21.35	0.83	6	−21.14	0.54
Round whitefish	9	−23.68	2.03	6	−24.34	2.64	20	−25.21	3.17
Trichoptera	9	−24.37	0.93				3	−32.61	3.29
Chironomids	8	−22.85	0.63				1	−22.54	
Snails (Lymnaeidae)	7	−23.63	0.67	17	−20.38	2.78	18	−24.27	1.69
Snails (Planorbidae and Valvatidae)	4	−26.30	2.99						
Clams (Sphaeridae)	3	−22.61	4.53	1	−27.69		1	−28.32	
Dipterans (other than chironomids)							3	−25.20	2.94
Ephemeroptera							1	−29.58	
Zooplankton	6	−33.31	0.93	9	−33.03	1.26	5	−33.17	0.22

[a] Some data included from Kidd (1996).
[b] Burbot values were not lipid adjusted as fat content was determined for liver tissue and not muscle.

chemicals in fish (Olsson et al., 2000; Ruus et al., 2002). Clark and Mackay (1991) noted that an increase in the concentration of chlorinated hydrocarbons in the food created a proportional increase in the contaminant concentrations in the test fish, while an increase in feeding rates did not create a similar proportional effect. The hypothesis is that the consumed food moves faster through the body resulting in decreased uptake of OC (Sijm et al., 1992). Alternatively, increases in nutrient assimilation (possibly due to higher food intake), may provide a greater capacity for biotransformation and hence elimination of contaminants (Sijm et al., 1992).

The declines in organochlorine concentrations in fish from Lake Laberge from 1993 to 2001 (up to 2010 for lake trout) have coincided with major changes in the morphology of some species. Significant relationships exist between body weight and OC concentrations for lake trout and forage fishes (cisco, lake whitefish and round

whitefish; this study; Ryan et al., 2005). Growth dilution is a likely explanation for the declines in OCs for lake trout from Lake Laberge, and research has shown that an increase in fish growth rates is elicited by an increase in prey size, although not by increases in prey numbers (Kerr, 1971b; Matuszek et al., 1990; Trippel and Beamish, 1993; Pazzia et al., 2002). Lake Laberge lake trout feed primarily on least cisco (Kidd, 1996; Tables 6 and 8), and this species increased in body size (and numbers) while declining in OC concentrations from 1993 to 2001. Additionally, OC concentrations (from 1993 to 2003) were inversely related with lake trout mass (Ryan et al., 2005), a relationship that was also observed but more intermittently from 2004 to 2010. Notably, this significant relationship between the 6 OC groups and mass occurred far less often in lake trout sampled from another regional Yukon lake during the same period (M. Ryan, unpublished results). Hence growth dilution of OCs may be occurring
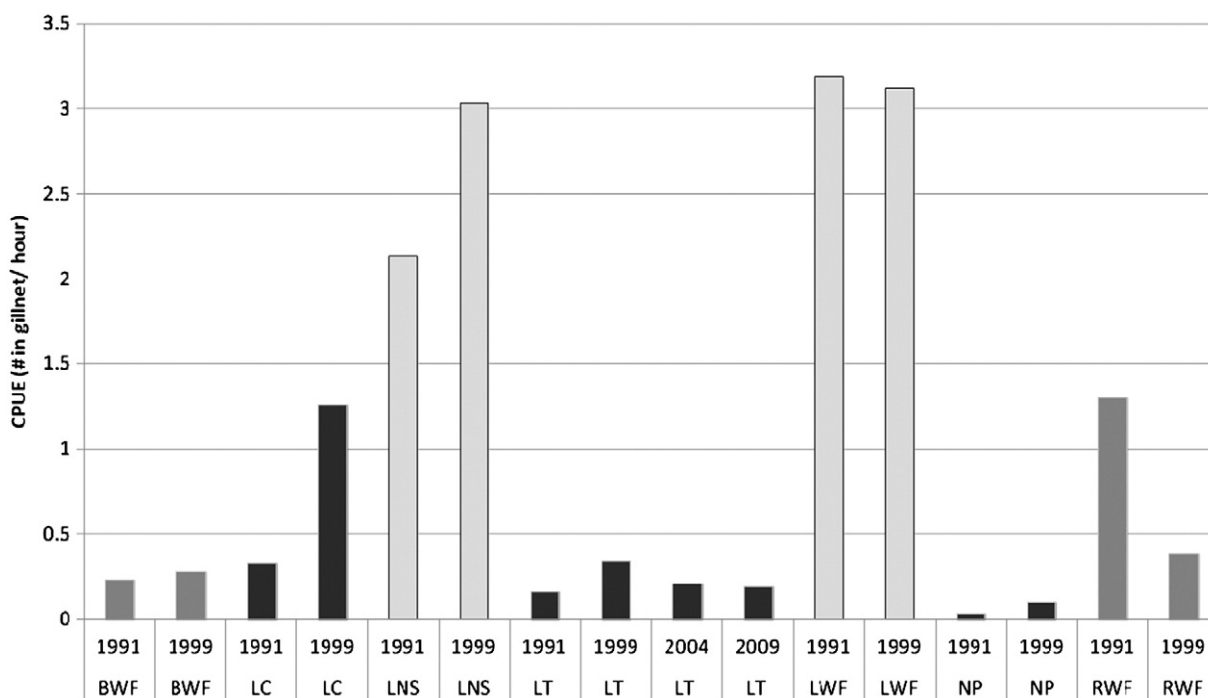

Fig. 2. Catch per unit effort (# in gillnet per hour) for selected fishes from Lake Laberge netted in 1991 and 1999 (additionally 2004 and 2009 for lake trout) using small mesh net surveys (data courtesy of Yukon Territorial Government). BWF—broad whitefish, LC—least cisco, LNS—longnose sucker, LT—lake trout, LWF—lake whitefish, NP—northern pike, RWF—round whitefish.

Table 9

Slopes and intercepts ($\pm$ SE) from $\log_{10}$ OC concentrations (ng/g ww) versus δ^{15}N regression models for 1993 (N = 100) and 2000–2001 (pooled; N = 137) biotic samples from Lake Laberge.

	1993			2000–2001			
	Slope (m)	Intercept (b)	r^2	Slope (m)	Intercept (b)	r^2	p
$\sum$ CBz	0.17 (0.02)	−1.52 (0.22)	0.40	0.30 (0.02)	−3.07 (0.20)	0.56	<0.01
$\sum$ HCH	0.16 (0.02)	−1.31 (0.24)	0.31	0.25 (0.02)	−2.81 (0.21)	0.44	0.01
$\sum$ CHL	0.22 (0.02)	−1.11 (0.24)	0.47	0.43 (0.02)	−3.80 (0.23)	0.67	<0.01
$\sum$ DDT	0.23 (0.03)	−0.31 (0.26)	0.45	0.42 (0.02)	−2.73 (0.22)	0.68	<0.01
$\sum$ PCB	0.21 (0.02)	−0.23 (0.24)	0.44	0.41 (0.02)	−2.77 (0.21)	0.68	<0.01
$\sum$ CHB	0.23 (0.02)	−0.27 (0.24)	0.48	0.44 (0.02)	−3.32 (0.25)	0.64	<0.01

in the Lake Laberge lake trout if they are growing at faster rates in more recent years (2000 to present) compared to the fish sampled in the 1990s. Lake trout otoliths from Lake Laberge had no abnormal growth patterns in 1993 versus 2001 (K. Mills, pers. comm.). There were no indications of "growth stanzas", which are characterized by periods of slow growth followed by accelerated growth and indicate that food consumption has changed. However, otoliths from 2000 and 2001 showed a faster first year of growth when compared to those from 1993, suggesting that increased growth rates of lake trout started at the young-of-the-year (YOY), planktivorous stage.

Although the evidence is not as strong, it is possible that growth dilution also explains some of the declines in OC concentrations in Lake Laberge for other fishes. Round whitefish were larger and body weight was inversely related to some OC concentrations in 2000–2001 when compared to 1993. Least cisco increased in length and weight from 1993 to 2000–2001 and weight was a significant predictor of OC concentrations in 1993 but not in 2000–2001, showing a shift in its importance. Cisco OC concentrations have decreased more likely due to the decrease in contaminant concentrations of their prey, zooplankton. There is no evidence to support the hypothesis of growth dilution for burbot, lake whitefish or longnose sucker.

Another factor that likely contributes to the changes in OC concentrations in Lake Laberge fishes is age as higher concentrations of OCs are typically found in older individuals (Kidd, 1996; Sijm et al., 1992). The mean age of burbot has increased while concentrations of ΣDDT, ΣCHB and ΣHCH have declined; the increase in age may be offsetting some of the decline in contaminants over this time period. Longnose sucker sampled from Lake Laberge in 2000–2001 were younger on average compared to fish sampled in 1993 and their ΣCHB, ΣHCH and ΣCHL concentrations declined concurrently.

Concentrations of OCs in fishes are closely related to levels in their prey; and for this reason the declines in OCs in Lake Laberge fish may be linked to reductions in prey contaminants. OC concentrations in zooplankton collected in 2001 from Lake Laberge declined to less than 30% of the 1993 levels. Lower OC concentrations in zooplankton were likely due to decreased contaminant levels in primary producers (phytoplankton), but this decrease may also be caused by growth dilution. Research has shown that OC concentrations and bioconcentration factors in plankton have an inverse relationship with plankton biomass and cell densities (Lederman and Rhee, 1982; Taylor et al., 1991). Although zooplankton OC levels do correlate with water OC concentrations, the process of equilibration is not particularly fast, and seasonal variables such as light and water temperatures, are highly influential on growth (Swackhamer and Skoglund, 1993; Smith, 1995). The rate of zooplankton biomass growth and death may exceed the rates of contaminant equilibration with the environment such that plankton communities maintain diluted OC concentrations per weight of mass (Smith, 1995) and OC concentrations in water become a weak predictor of levels in plankton (Swackhamer and Skoglund, 1993). This dilution effect provides planktivorous predators with a food base that is less contaminated (Smith, 1995). Zooplankton densities were higher in 2001 than 1993, although these data are limited, and lend support to biomass dilution of OCs at the base of the food web. In this study,

we did not specifically examine changes in phytoplankton, although it is possible to review this through the analysis of sediment cores (Gajewski et al., 1997; Stern et al., 2005). Analyses of sediment cores indicated increases in most OC fluxes in Lake Laberge (Ryan, 2006; Ryan et al., in preparation) indicating that a reduction of OC influx from abiotic sources was not likely responsible for the declines observed in fish. Additionally, increases in phytoplankton abundance observed over the 1990s (Ryan, 2006) provide further evidence that growth dilution, and not changes in atmospheric inputs (Hung et al., 2010), were the leading factors for OC declines in fish.

4.2. Lipids and biomagnification

Lipid content is known to be a major predictor of OC in fish (Kidd et al., 1998) as OC concentrations are positively related to lipid content in the tissues (Rasmussen et al., 1990; Larsson et al., 1991; Rowan and Rasmussen, 1992; Kucklick et al., 1996). In the present study, decreases in lipid body contents of lake trout, burbot, round whitefish, and lake whitefish were correlated to declines in OC concentrations (Ryan, 2006). However, analysis of the most recent lake trout data indicates that lipids were not correlated with 4 of the 6 OC groups studied from 2003 to 2010 indicating that changes to the OC concentrations are related to factors more influential than fish lipid content further supporting the hypothesis of growth dilution in this species.

The decline in tissue lipids for fishes in Lake Laberge may be due to changes in foraging behavior (e.g. intensity), prey lipid and increased reproduction. Because fish populations and zooplankton densities have increased in this lake in the early 2000s, foraging behaviors may have also changed over time. For example, with more cisco in Lake Laberge, lake trout may have spent less energy foraging for this primary prey, which would have increased their growth rates (Kerr, 1971a; Hebert et al., 1997; Pazzia et al., 2002). Since the ciscos in Lake Laberge were also larger, fewer would be needed to maintain a high rate of growth in lake trout (Matuszek et al., 1990). There was no evidence of a shift in the dietary habits of lake trout away from their main prey cisco based on δ^{15}N and δ^{13}C. Lastly, lipids and OC levels may also be affected from an increase in intraspecific competition for spawning and the effects of reproduction on OC concentrations in reproducing fish (Larsson et al., 1993). Since OC can be passed along to young via reproductive routes (Larsson et al., 1993; Fahraeus-Van Ree and Payne, 1997; Delorme et al., 1999), it is possible that increased spawning in Lake Laberge contributed to lower lipids and OCs for some species.

The lipid contents measured in zooplankton from 1993 to 2001 remained unchanged, yet the OC concentrations declined by more than 70% for all 6 OC groups in this period. Since the decrease in OC levels were not linked to the lipid content of zooplankton, other factors must be responsible such as a community composition change, biomass dilution, or a reduction in prey OC concentrations. Tissue lipid content may not account for the entire decrease in OC concentrations in some Lake Laberge biota, but it is a more likely factor

for fish that are not exhibiting patterns typical of growth dilution (e.g. lake whitefish).

In Lake Laberge, the OC-δ^{15}N regressions have increased significantly between 1993 and 2000–2001 for all OC groups, indicating that the "rates" of food web biomagnification were higher in the most recent sampling period. The biomagnification increases may be caused by factors specific to the physiology or behavior of each species, or because of the lower contaminant levels found in zooplankton or forage fish. Feeding behavior and diet can have a significant effect on bioaccumulation rates. For example, two groups of lake trout from Lake Michigan had distinctly different bioaccumulation rates of PCB (Miller et al., 1992). The lower PCB bioaccumulation rate in deep water trout was attributed to their consumption of larger, higher caloric content fish (Miller et al., 1992). As prey size increases, predators may also receive the offsetting benefit of growth dilution. Secondly, as fish feeding rates increase with higher prey populations, their biomagnification factors are expected to increase linearly; however this only goes to a point where OC uptake efficiency declines with increased food consumption (Sijm et al., 1992). In Lake Laberge, the higher slopes of the regressions for samples from 2000–2001 is surprising given that concentrations of OCs have declined in most components of the food web. Although there may be many explanations for this shift, the increased slope suggests that the greater magnitude of OC declines in organisms lower in the food chain are the likely factors influencing this trend.

4.3. Populations and food web dynamics

Exploitation, addition or removal of a fish species affects the growth rates (Hewson, 1955; Healey, 1975, 1978; Mills et al., 2000), body size (Healey, 1975), fecundity (Munkittrick and Dixon, 1989), and maturation ages (Healey, 1975, 1978; Munkittrick and Dixon, 1989) as well as affecting the structure of the prey community (Mills and Forney, 1983; Trippel and Beamish, 1993; McNaught et al., 1999). It was expected that changes in such population characteristics would occur following the closure of the century long commercial fishery on Lake Laberge. Changes to the status of lake trout are known to have significant impacts on the food web community structure due to changes in fish predation (Northcote, 1988; Merrick et al., 1992; Trippel and Beamish, 1993; Wetzel, 2001). Trout stocking of an alpine lake did result in a change in the plankton community towards a heavier dominance by a smaller *Diaptomus* and smaller phytoplankton species over larger crustaceans (McNaught et al., 1999). Lake trout have also influenced the size characteristics of slimy sculpin as an increase in sculpin size coincided with the decline in lake trout predation (Mcdonald and Hershey, 1992). Both lake trout and round whitefish have been shown to affect the distribution, size and density patterns of molluscs in an Alaskan lake through size selective predation (Merrick et al., 1992). Evidence of fluctuation in fish populations in Lake Laberge towards the late 1990s likely influenced the body size characteristics of some prey fish and zooplankton thus shifting the community composition or species characteristics observed in Lake Laberge.

The evidence indicating the zooplankton structure in Lake Laberge changed over time may also be due to a shift in community composition, production, or in the timing of seasonal succession. Densities of zooplankton increased and there was a shift from the dominance of *C. scutifer* in 1982 (85%) and 1993 (80%) towards dominance by *D. pribilofensis* in 2002 (47%). Aside from potential changes in pressure from fish predation, this shift could be indicative of a long-term change caused by several dry and above average temperature summers (1981–1984, 1994–1997; Ryan, 2006) in the Yukon region (Environment Canada, 2000). Climate change effects on plankton were studied in the Experimental Lakes Area in Ontario (Canada) where a rise in temperatures (2 °C) in the 1980s resulted in both increased phytoplankton biomass and number of species, along with

a shift in the community species composition, during drought years (Findlay et al., 2001). Abnormal variations in climate can easily offset a pattern in succession as shown by Edmundson et al. (2003), who concluded that the depth of the euphotic zone in an Alaskan lake decreased due to an increase in glacial runoff and silt loadings during dry, hot years, resulting in a significant decrease in planktivorous copepod biomass and subsequent increases in numbers of primary producers. However, temporal cycles of *Cyclops* populations as those observed in Lake Laberge have been reported for Alaskan lakes, while *Diaptomus* did not exhibit similar patterns over the same period (Edmundson et al., 2003). Therefore it is possible that the observations of community shifts in Lake Laberge may just be part of the natural life cycle for these zooplankton species and are unrelated to climate changes.

Plankton succession occurs due to shifts in water nutrients, temperatures and light periods, and as such, is largely moderated by climate (Wetzel, 2001). Changes in a seasonal pattern over extended periods of time may result in shifts of the plankton succession times and ultimately a shift in the habits or growth potential of planktivorous predators. Fish production is known to closely correlate with primary production (Downing et al., 1990) and even non-planktivorous species such as lake trout have been related to zooplankton densities (O'Brien et al., 2004). For example, an increase in the availability of *D. pribilofensis* was shown to account for growth differential in YOY lake trout from inshore and offshore habitats, between two Arctic lakes (Mcdonald et al., 1992) and it was suggested that *Diaptomus* might be an important supplemental food resource for salmonids in years when cyclopoid copepods such as *C. scutifer* are in lesser abundance (Edmundson et al., 2003). The addition of *Mysis relicta* to a Montana lake resulted in dramatic increases in the lake trout populations through improved survival of YOY (Stafford et al., 2002). Other research concluded that copepod biomass in spring, along with the abundance of fish spawners, is directly related to the fall fry salmonid production and strongly influences recruitment success in the next generation (Edmundson et al., 2003). Both fry weight and abundance were related to the biomass populated by *Cyclops* and *Diaptomus* species in that research. With the increasing shift towards *D. pribilofensis* dominance in the zooplankton community in Lake Laberge, it is speculated that an increased growth rate, and ultimately growth dilution of contaminants, is beginning at the YOY stage. Changes in the zooplankton community within Lake Laberge as previously noted might directly account for the increased growth in some fish, including lake trout, although no direct correlations were studied in this project. Factors determining the size spectrum and species community composition of plankton are important determinants of the structure of the pelagic food web and ultimately the partitioning of contaminants.

Although there are several viable reasons for a change in the zooplankton community of Lake Laberge, there are fewer hypotheses regarding the decline in zooplankton OC concentrations (over 70%) between 1993 and 2001. These OC declines may be due to the shift in the composition of the plankton from *C. scutifer*, to a smaller and more herbivorous *D. pribilofensis* (noted as having a smaller maximum body size by length) (Table 5; Kirkland and Gray, 1986). Contaminant concentrations are affected by body size (Fisk et al., 2001) and hence the surface area that is in contact with the water, the primary source of OC contaminants for plankton (Swackhamer and Skoglund, 1993). Other species differences include varying elimination rates (Swackhamer and Skoglund, 1993) or different body composition (e.g. lipids, proteins; (Parsons et al., 1961)), potentially affecting contaminant partitioning rates (Lederman and Rhee, 1982). Such a change in the community composition will eventually relate to changes in contaminants higher in the food chain (Smith, 1995; Stewart et al., 2003). Stewart et al. (2003) concluded that major changes in fish OC levels following the Red River flood of 1997 were not linked to the transport of new chemicals into the

ecosystem, but were due to species shifts within the plankton community. These studies provide information on the importance of plankton community composition interactions to upper trophic levels, which may be influencing OC contaminant levels in Lake Laberge.

Previous studies demonstrated that Lake Laberge had an atypically long food chain compared to other regional lakes (Kidd et al., 1995b, 1998). After the commercial fishery was closed on the lake, it was hypothesized that the food chain may shorten, due to increased intraspecific competition (Kidd, 1996) and consequently reduces concentrations of OCs in fish because fish that feed at a lower trophic position are known to have lower contaminant burdens (Rasmussen et al., 1990). Although small increases in $\delta^{15}N$ were found for all biota between 1993 and 2000–2001, the relative trophic levels for fish and invertebrates were still similar 10 years after the fishery closure (Table 7) and did not support the hypothesis of a change in food chain length in Lake Laberge.

Changes in the source of carbon for fish may also affect its OC concentrations as a few studies have found higher PCBs and other OCs in fish that rely more on pelagic than benthic carbon (Campbell et al., 2000; Guildford et al., 2008). In Lake Laberge there was no evidence of changes in the mean $\delta^{13}C$ of zooplankton over time, based on a definition of enrichment between predator and prey ranging from 0.8 to $1.3‰ \pm 1.1‰$ relative to the diet; (DeNiro and Epstein, 1978; McCutchan et al., 2003), but some obvious increases were found for benthic invertebrates (e.g. Lymnaeidae increased by 1.25‰ while a larger shift $>3.0‰$ was noted for Trichoptera; Table 8). Fishes with high lipid tend to have more negative $\delta^{13}C$ ratios, and this can confound interpretations of carbon source (Post et al., 2007). Lipid adjusted mean $\delta^{13}C$ (using the method in Post et al. (2007)) became heavier from 1993 to 2001 for lake trout (-28.56 and $-27.92‰$, respectively), longnose sucker (-21.97 and $-21.35‰$) and lake whitefish (-26.25 and $-24.65‰$). In contrast, lipid adjusted mean $\delta^{13}C$ was lighter for round whitefish (-23.68 and $-25.21‰$) and cisco (-28.99 and $-29.61‰$) in 2001–01 compared to 1993. Absolute differences between a fish species and zooplankton (mean $\delta^{13}C_{fish}$ – mean $\delta^{13}C_{zoop}$) were similar in both 1993 and 2000–2001 for lake trout (4.7 and 4.5, respectively). Pike (8.7 and 8.1; 1993 and 2000–2001, respectively), burbot (6.4 and 5.7), round whitefish (9.5 and 8), and cisco (4.0 and 3.4) appeared to feed more on pelagic carbon in 2000–2001 than 1993, whereas the opposite trends were found for longnose sucker (11.3 and 11.8) and lake whitefish (6.9 and 8.2). Although the trends support subtle shifts in the food sources for some fish, the changes in lipid adjusted $\delta^{13}C$ were very small ($<0.7‰$ for the majority of species). Since the trends towards more pelagic food sources did not coincide with increases in OCs (e.g. round whitefish and cisco), the evidence suggests that any subtle changes in the food sources of Lake Laberge fish was not the primary factor affecting the large decline in OC concentrations in fish tissue.

Although the fish populations of Lake Laberge have shown evidence of a change since the closure of the commercial fishery in 1991, any resulting shifts in the food web through foraging competition or food abundance has not dramatically altered the structure or hierarchy of the food chain while OC concentrations in fish have significantly declined over this period. The reduction in OCs was related to differences in lipid content and body mass over time providing evidence for the hypothesis that growth dilution is a primary factor responsible for the declines in contaminant concentrations. A general observation can made that the more pelagic fish species (cisco, lake whitefish, lake trout) and zooplankton have demonstrated the most consistent changes in OC concentrations compared to predominantly benthic organisms (burbot, longnose suckers) which has likely been caused by differences in the food web interactions beginning at the plankton trophic level.

Considering the slow recovery of exploited lake trout, possible changes in fish populations and subsequent species interactions within the food web, OC concentrations in the Lake Laberge ecosystem may continue to fluctuate. However, OCs are expected to continue declining likely at a slower rate over time.

Acknowledgments

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