Long-term Cladocera dynamics and inferred fish zooplanktivory in Kootenay Lake, British Columbia

NSERC Alliance grant "New approaches to assess changes in nutrients and aquatic production in large lakes impacted by dams in Western Canada"

Report to B.C. Hydro, July 2025

Isaac Armstrong[1], Kathleen Laird[1], Brian Cumming[1]

[1]Paleoecological Environment Assessment and Research Laboratory, Queen's University Department of Biology, Kingston, Ontario, Canada

Executive Summary

- Kootenay Lake, BC, is a long, deep, glacial lake valued for recreational and subsistence fishing. Numerous anthropogenic impacts (e.g., invasive species, cultural eutrophication, watershed impoundment, land-use changes) have affected fisheries, leading to management interventions.

- In 1992, an artificial fertilization project was implemented to restore ecosystem productivity in response to oligotrophication attributed to the retention of nutrients behind upstream dams and the closure of an upstream fertilizer plant.

- To understand how anthropogenic impacts altered secondary production, we assessed cladoceran remains in four dated sediment cores (two from Kootenay Lake, one from Duncan Reservoir, and one from Trout Lake, a reference site). We quantified assemblage composition and concentration. We also used cladoceran size structure as a proxy for zooplanktivory to reconstruct past food-web dynamics.

- We find evidence that invasive *Bosmina (Eubosmina) coregoni* was present in Kootenay Lake in substantial abundances ~1965, a remarkably early invasion, and has since been introduced to both Duncan Reservoir and Trout Lake.

- While Trout Lake was a suitable reference core for primary production, it was not as effective in isolating the effects of climate on secondary production. Connectivity of fish populations between Trout and Kootenay Lakes likely influenced cladoceran trends.

- Duncan Reservoir showed a typical trophic upsurge response to inundation, but changes to assemblage composition in the 1980s suggest enhanced zooplanktivory, potentially because of changing reservoir characteristics.

- Both *Mysis* and kokanee likely regulated cladoceran concentrations and size in Kootenay Lake. Duncan and Libby Dam construction was associated with short-term pulses of cladoceran production, but 1980s declines in cladoceran concentration followed lake recovery from cultural eutrophication with the closure of upstream Cominco Fertilizer Plant on the Kootenay River.

- Artificial fertilization increased cladoceran abundances in Kootenay Lake, though the large initial increases were quickly mediated. Cladoceran concentrations and size metrics show cyclical trends following kokanee abundances, indicating top-down regulation.

- Inferred planktivory was much higher in the artificial fertilization period pre-kokanee collapse than the historic or reference baseline period. Following the kokanee spawner collapse, the size and concentration of subfossil *Daphnia* increased.

- Kootenay Lake is supporting larger *Daphnia* than seen in Duncan or Trout lakes, likely the result of artificial fertilization increasing productivity above reference levels.

- Overall, artificial fertilization has contributed to novel food web dynamics in Kootenay Lake which have persisted after the collapse of kokanee spawners c. 2013.

Table of Contents

1. Project Overview

Kootenay Lake, British Columbia, is a long, deep lake that is prized for cultural, aesthetic, and recreational services. The system hosts important fisheries, including trophy Gerrard rainbow trout (*Onchorhynchus mykiss*) and bull trout (*Salvelinus confluentus*), which are largely dependent on kokanee (lacustrine sockeye salmon; *Onchorhynchus nerka*) as prey. However, the system has experienced many anthropogenic impacts: mining and forestry beginning in the mid-late 1800s, successive watershed impoundments, the introduction of invasive *Mysis*, and major anthropogenic phosphorus (P) loading from an upstream fertilizer plant. The subsequent impacts to Kootenay Lake limnology and fish populations have been well-described (see Northcote, 1972; Daley et al., 1981; Ashley et al., 1997; Northcote et al., 2005; Binsted & Ashley, 2006; Arndt, 2009; Schindler et al. 2020).

As a result of these impacts, and largely due to the collapse of kokanee stocks in the 1980s, management efforts have focused on fisheries restoration alongside protecting wildlife and riparian habitat. Artificial fertilization was undertaken in 1992 to restore kokanee stocks by increasing the biomass of high-quality plankton (Schindler et al., 2020). In 2007, a Water Use Plan (WUP) was enacted to enhance wildlife habitat and reduce erosion downstream of Duncan Dam (BC Hydro, 2007). However, collapse of kokanee stocks c. 2013 and the continued depression of kokanee populations by piscivores (Warnock et al., 2022) suggests the food web has become imbalanced.

Here, our goal is to provide long-term records on the response of Cladocera (crustacean zooplankton) to successive anthropogenic impacts and management efforts in the Kootenay Lake system. Cladocera (specifically *Daphnia*) are the preferred prey item of kokanee (Schindler et al., 2020; Wilson et al., 2011) but are sensitive to both bottom-up (nutrient) and top-down (predation) controls (Korhola & Rautio, 2001). As *Daphnia* have higher nutritional requirements and a longer reproductive time than many other Cladocera (e.g., *Bosmina*), they succeed in systems with enough nutrients to ensure sufficient availability and quality of food (Persson et al., 2007). However due to their large size, *Daphnia* are vulnerable to visual predation by fish, and can be suppressed in lakes with high salmonid densities despite increased nutrients (Chen et al., 2011). Thus, understanding the long-term controls on cladoceran production and composition is important to contextualizing management outcomes. However, zooplankton data prior to the 1970s were sparse and sampling occurred sporadically (Daley et al. 1981).

We employ a paleolimnological approach to assess cladoceran dynamics over time in dated sediment cores encompassing pre-1900s conditions to 2023. Using subfossil cladoceran remains, we reconstruct temporal trends in cladoceran assemblage metrics (species composition, overall abundance, and size structure). Assemblage metrics of subfossil cladocera are responsive to trophic change, and the classical responses to nutrient loading and fish predation regimes are well-described (Leavitt et al.,1994; Korhola & Rautio, 2001; Bos & Cumming, 2003; Nevalainen & Luoto, 2016; Armstrong et al., 2025). Taxa have differing food and habitat requirements and also differ in their response to predation (Nevalainen & Luoto, 2016). Cladocera alter their morphology in response to predation pressure, and these changes can be measured to reconstruct past changes to the abundance of planktivorous fish (reviewed in Korosi et al. 2013). Overall, Cladocera are well-suited to reconstructing the structure and function of aquatic food webs.

In brief, we investigate the impacts of watershed impoundment, reservoir operations, and artificial fertilization on cladoceran dynamics and on cladoceran size-structure (as a proxy for fish planktivory) in Kootenay Lake, Duncan Reservoir, and the reference system of Trout Lake. This research is part of a larger project prepared for BC Hydro and for the linked NSERC Alliance Grant “New Approaches to Assess Changes in Nutrients and Aquatic

Production in Large Lakes Impacted by Dams in Western Canada." A companion report analyzed diatom remains, diatom-inferred total phosphorus (DI-TP) and sediment-inferred lake-water TP (SI-TP; Laird et al., 2025).

2. Study Location

2.1 Regional geography and climate

The Kootenay Lake system is a branch of the upper Columbia River drainage basin, located in the Purcell Trench, between the Selkirk and Purcell mountains (southeastern British Columbia; **Figure 2.1**). The north-south mountain ranges are a dominant influence on climate, as most precipitation is delivered to the mountains, leaving the valley comparatively drier (Daley et al., 1981; Rodenhuis et al., 2009). The region's hydrologic regime is highly influenced by melting snowpack and glaciers, which can provide important cold-water discharges during the summer (Stahl & Moore, 2006; Brahney et al., 2017). At the Kaslo station (49.9142° N, 116.9155° W), annual precipitation ranges 700-1000 mm, average temperatures are ~8°C, and 17-19 ºC in the summer months (**Figure 2.2**).

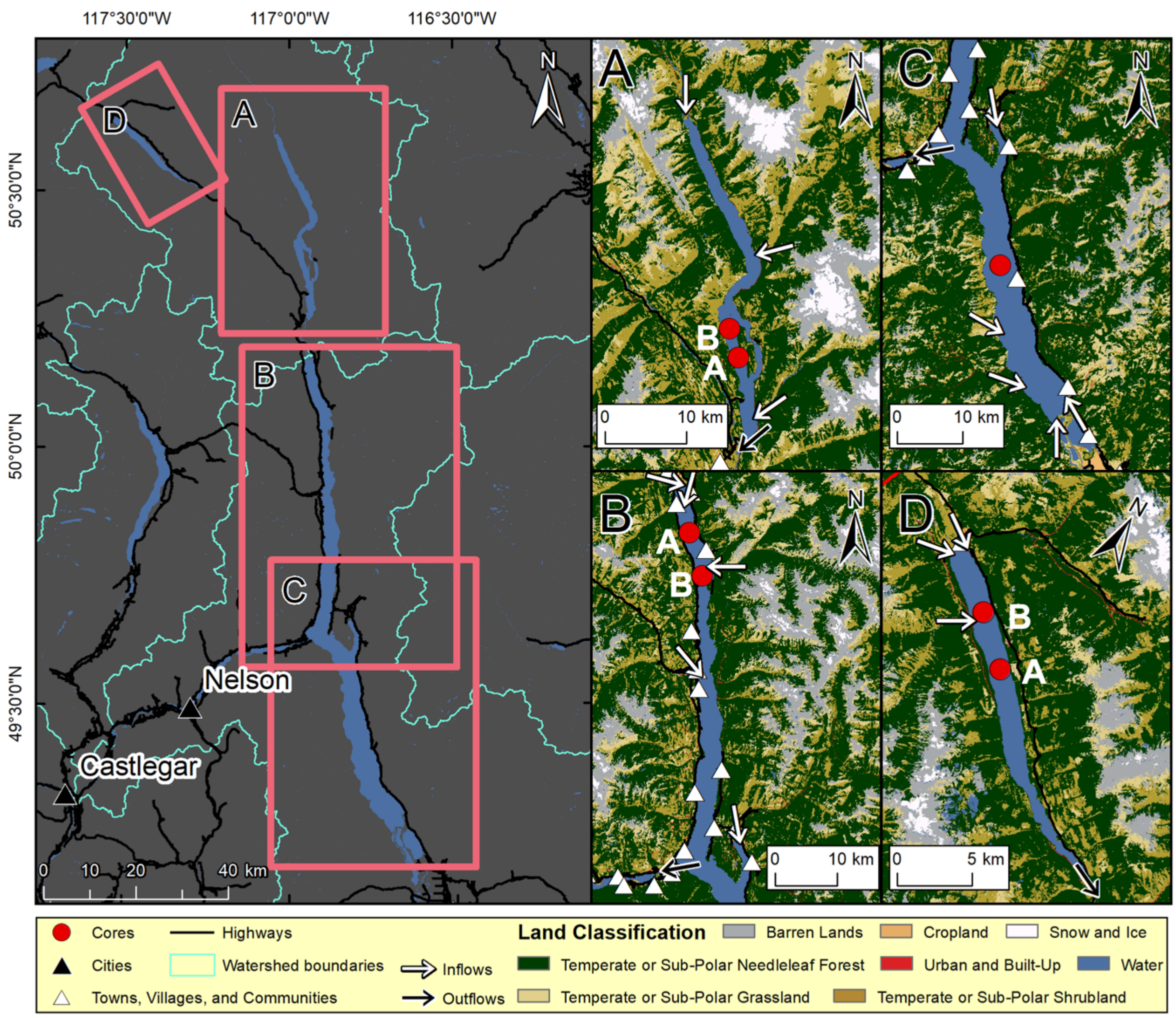

Figure 2.1. *Kootenay Lake project basins: A) Duncan Reservoir, B) Kootenay Lake North Arm, C) Kootenay Lake South Arm, and D) Trout Lake. Coring locations are indicated by solid red circles. Basins where two cores were taken have locations labelled with A and B.*

The Kootenay Lake drainage basin has been subject to land-use changes such as agriculture, forestry, and mining (Daley et al., 1981). Mining activity boomed from approximately 1890 to 1920, though some major mines persisted, such as the Bluebell mine in Riondel which discharged waste rock, tailings, and mine-water drainage into Kootenay Lake before the mine closed in 1971 (Daley et al., 1981). Forest harvesting was widespread in the region and log booms were common on Kootenay Lake as they transported logs from northern sites in the Duncan-Lardeau basin to a mill in Nelson (Daley et al., 1981). Forestry activities lessened with the closure of Kootenay Forest Products in 1984 (Nelson Museum & Archive, 2022; https://nelsonmuseum.ca). The spatial extent and magnitude of this watershed disturbance is unclear, as are the potential limnological impacts, however as this activity occurred throughout our study region, we would not expect it to impact our ability to isolate the effects of local stressors (e.g., dam construction) within our cores.

In general, air temperatures in the Canadian portion of the Columbia Basin have risen 1.2-2ºC over the last century (Rodenhuis et al., 2009). Climate data from the Kaslo, BC, station show that temperature has increased consistently over the past century. The onset of this increase varied seasonally, but generally between 1970 and 1990. Annual precipitation has also increased, most notably between 1955 and 1960, with spring precipitation showing a consistent increase beginning ~1980. Duncan Dam climate station data show a similar trend, with warmer springs since ~1985 (**Figure 2.3**) and increasing summer and winter temperatures beginning in the 1990s. Fall and spring precipitation have increased over time, while summer and winter precipitation have decreased. Total snow has also decreased over time at both stations since the 1980s (**Figure A1**).

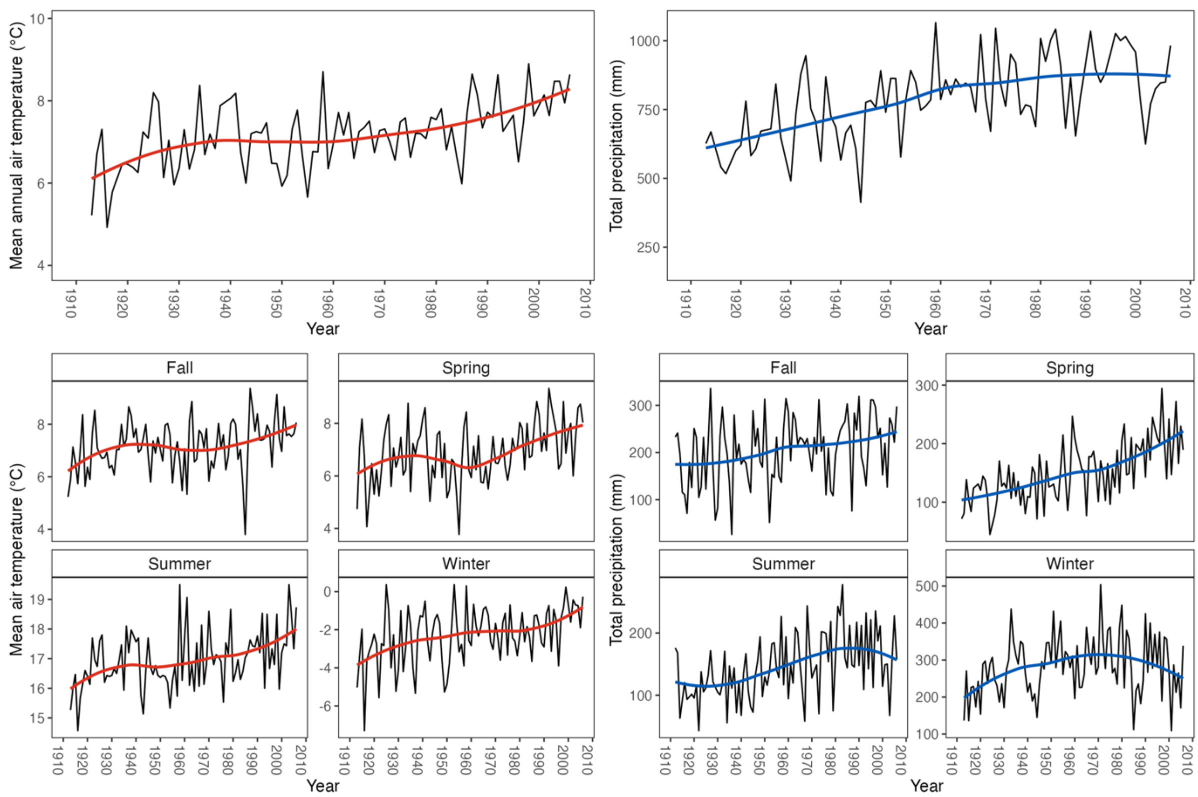

Figure 2.2. *Annual and seasonal trends for air temperature (ºC) and total precipitation (mm) from the climate station at Kaslo, British Columbia, climate station (ID #1124). Trends are smoothed with a basic* loess *function. Data were downloaded from Environment and Climate*

Change Canada using the function `weather_dl` from the R package weathercan (LaZerte and Albers, 2018).

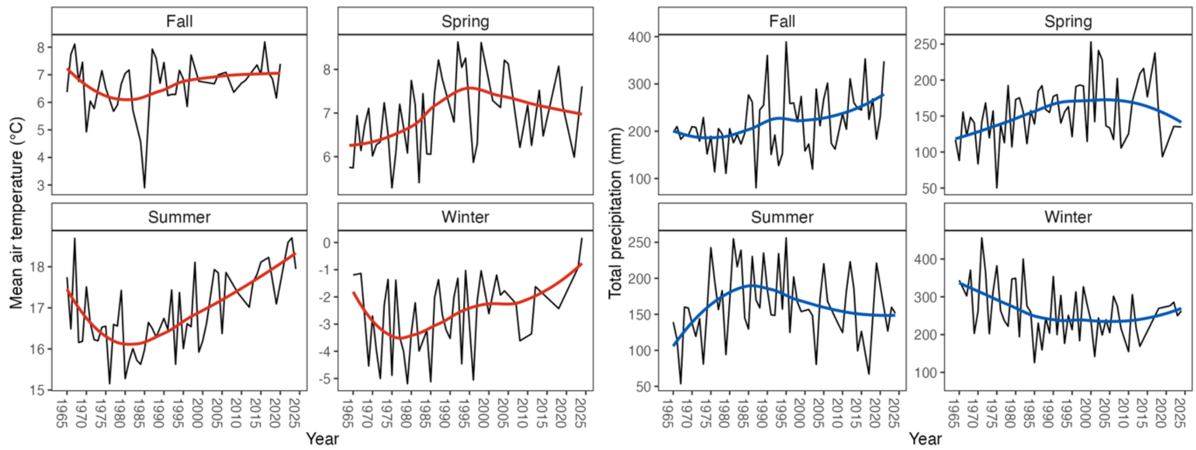

Figure 2.3. *Seasonal trends for air temperature (ºC) and total precipitation (mm) from the climate station at Duncan Dam, British Columbia (ID #1115). Trends are smoothed with a basic* loess *function. Data were downloaded from Environment and Climate Change Canada using the function* `weather_dl` *from the R package* weathercan *(LaZerte and Albers, 2018).*

2.2 Trout Lake

Trout Lake is a headwater lake whose outlet is the Lardeau River which drains into Kootenay Lake. The Lardeau River is an important spawning habitat for Gerrard Rainbow trout and kokanee. While biomonitoring data for Trout Lake are limited, rainbow trout, bull trout, burbot (*Lota lota*), dolly varden (*Salvelinus malma*), and kokanee are all regularly caught (https://www.anglersatlas.com/, 2025; https://www.troutlakebc.com, n.d.). Rainbow trout in particular move freely between Trout and Kootenay Lakes, and a low number of rainbow trout fry may also migrate upstream into Trout Lake after spawning (Acara, 1969).

Trout Lake was selected as a reference site for the effects of climate on production, as it was not directly impacted by dams or major anthropogenic P loading, though the watershed experienced similar mining and forestry activity characteristic of this region. The ice phenology for Trout Lake is unclear; an older report remarks that it did not freeze in 1967 (suggesting that not freezing was unusual; Acara, 1969), while a 1908 newspaper article suggests that Trout Lake was "icebound" for 2-3 months a year (Slocan Mining Review 1908, UBC Library Open Collections https://open.library.ubc.ca/).

2.3 Duncan Reservoir

Duncan Reservoir is a long (45 km), large (7290 ha) reservoir which undergoes a ~30 m annual drawdown that can dewater ~50 km^2 (Cope, 2008). It is isothermal from November to May, and in the summers of 2010-2012, it stratified with a thermocline at 10-12 m (AMEC & Poisson, 2012). Water is discharged through low-level outlets (541.63 m), with high-elevation spillways used sporadically from mid-July to September (AMEC & Poisson, 2012).

Duncan Reservoir was created c.1965-67 with the construction of Duncan Dam, which extended the pre-existing Duncan Lake from 25 to 45 km (Arndt, 2009). The resulting inundation increased lentic area by 2.5x and flooded a diverse area of riparian floodplain, alluvial fans, creek inflows, wetland, woodland, shallow ponds, and braided streams (Arndt, 2009; Utzig & Schmit, 2011). At full pool, the reservoir area is 73.3 km^2, and at low pool it is reduced to 26.4 km^2 (the size of pre-dam Duncan Lake; Weir & Irvine, 2023). Construction of the reservoir presented concerns for fish populations by flooding low-gradient fish habitat and exposing spawning areas during winter drawdown (Arndt, 2009). However, data are largely absent for fish populations within Duncan Reservoir, though kokanee, bull trout, and burbot are present, and rainbow trout stocking occurred within the past century (Neufeld & Burrows, 2017).

The maximum elevation of Duncan Reservoir is 576-577 m asl, which is met most years (**Figure 2.4**). Mean elevations were generally higher prior to 1990, decreased to 2000, but have been stable since 2004. Minimum elevations were higher through 1980s-early 1990s, then have been stable since 2004. The change in minimum elevation is driven by changes in spring (April) when reservoir level is lowest before refilling begins in May (**Figure A2**). Discharge patterns have been variable over time, with the highest discharges generally in the winter and the lowest in June (**Figure A3**).

Dam discharge patterns affect the Duncan River inflow to Kootenay Lake, affecting downstream habitat quality (Binsted & Ashley, 2006). The spawning patterns of fish are impacted by water velocity and temperature, both of which are impacted by discharge (lower discharge results in warmer temperatures; Acara, 1969; Morbey & Ydenberg 2003, AMEC & Poisson, 2012). Dam discharge rates also affect the erosion of downstream areas and high discharge (> 400 m^3/s) could cause backflow into Meadow Creek (DDWUPCC, 2005). As such, a Water Use Plan was officially implemented in 2007, to better protect riparian areas, reduce erosion, and facilitate passage of bull trout (BC Hydro, 2007).

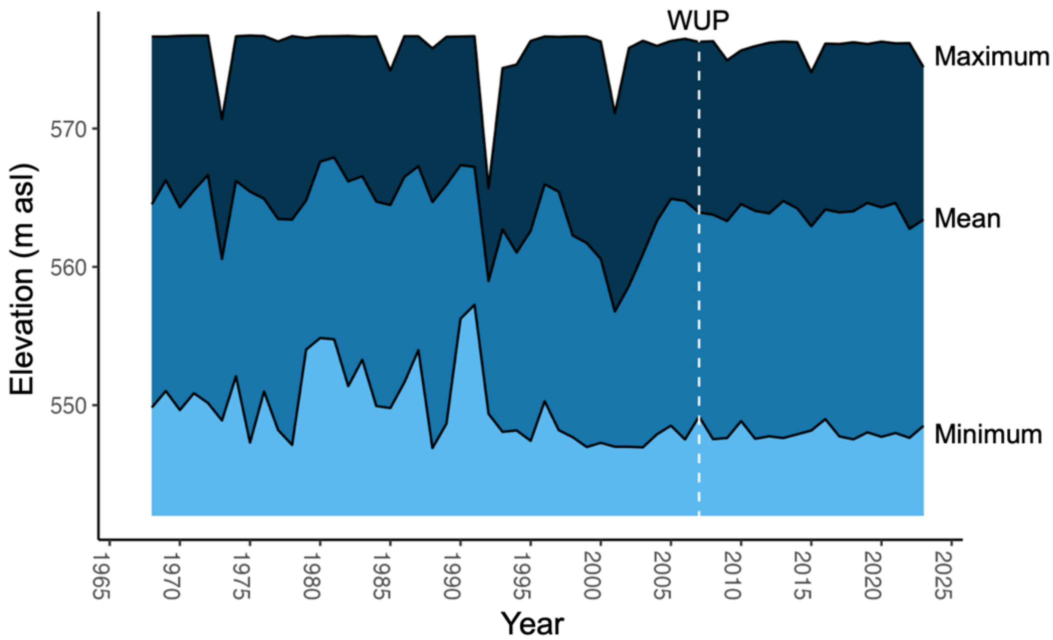

Figure 2.4. *Annual maximum, mean, and minimum elevation (m asl) for Duncan Reservoir from 1968 to 2023. The Water Use Plan (WUP) is shown as a dashed white line. Historical hydrometric data from Environment and Natural Resources Canada (https://wateroffice.ec.gc.ca/).*

2.4 Kootenay Lake

Kootenay Lake is long and narrow, characteristic of a fjord-type lake, with steep sides that rise to summit elevations of 2100-2700 m asl (Pharo & Chamberlain, 1979), though the lake itself has an elevation of 530 m asl. Historically, water levels varied as much as ~4 m seasonally (Daley et al., 1981) and currently vary 2-3 m (Fortis BC, 2025; https://www.fortisbc.com/). It has three sub-basins (North, South, and West Arms) which are morphologically distinct but experience mixing (Binsted & Ashley, 2006). The North Arm is primarily fed by the Duncan and Lardeau rivers (~25% of the annual inflow), while the South Arm is fed by the Kootenay River (~50% of annual inflow; Daley et al., 1981). The West Arm is the outlet to the Columbia River.

Kootenay Lake has a water residence time of 1.5-1.8 years (Daley et al., 1981; Carmack et al., 1986). Kootenay Lake has a drainage basin of almost 46,000 km^2, 80% of which is largely drained by the Kootenay River into the South Arm (Northcote, 1972; Daley et al., 1981). Inflow from the Duncan and Kootenay rivers controls circulation patterns (Daley et al., 1981) and seasonal nutrient availability (Carmack & Gray, 1982). As river discharge differs between the North and South Arms, so does timing and amount of productivity (Jasper et al., 1983). Inflow also affects temperature (AMEC & Poisson, 2002). Kootenay Lake does not freeze, at least since the 1940s (Zyblut, 1967).

Alongside regional land development in the 1900s, Kootenay Lake was impacted by several direct stressors. In 1939, the Corra Linn Dam, built on the lake outlet, became operational, raising water levels by ~2.4 m (Arndt, 2009). In 1949, the invasive amphipod *Mysis* was introduced under the belief it would act as food for kokanee, though instead it acted as a competitor (Thompson, 2000). The Cominco phosphorus fertilizer plant on the St. Mary's River opened in 1953 and discharged phosphorus to the lake, with inputs tripling by the late 1960s (Daley et al., 1981). Phosphorus loadings led to cultural eutrophication, blue-green algal blooms were common (Ennis et al. 1983), and kokanee were abundant (Schindler et al., 2020).

Subsequent watershed impoundments altered flow regimes and nutrient availability and seasonality. Duncan Dam was constructed in 1967 on the Duncan River, approximately 8 km north of Kootenay Lake (BC Hydro, 2007). Libby Dam was constructed upstream on the Kootenay River in 1973. As a result of both dams, springtime freshet has been greatly reduced and light penetration has increased (Daley et al., 1981), while winter flows have increased (Binsted & Ashley, 2006). Reservoirs generally retain nutrients behind dams, reducing downstream productivity and impacting food webs (Maavara et al., 2015), though Daley et al. (1981) also suggest that dams can increase productivity by reducing turbidity, decreasing flow-related "washout" of algal cells, and mobilizing nutrients released during reservoir inundation.

Kokanee stocks collapsed in the 1980s, which was attributed to cultural oligotrophication from upstream impoundments (Ashley et al., 1997). To enhance food-web productivity, artificial fertilization began in 1992 using a blend of liquid nitrogen (28-0-0, N-P_2O_5-K_2O; % by weight) and phosphorus (10-34-0, N-P_2O_5-K_2O ; % by weight) fertilizer additions to the North Arm (Binsted & Ashley, 2006). Fertilizer addition was timed as to approximate spring freshet and then continued into the summer (total of 20 weeks). The baseline data used to guide this restoration was from the 1940s, as eutrophication was considered to begin in the 1950s and intensify in the 1960s (Binsted & Ashley, 2006; Schindler et al., 2020). Earlier limnological data are not available, so the historic productivity of the lake before the Corra Linn dam raised water level is unknown.

Artificial fertilization initially succeeded in raising plankton biomass, and fertilization of the South Arm was undertaken in 2004 (Binsted & Ashley, 2006). This contributed to historic highs of kokanee biomass by 2007 (Warnock et al., 2022), but the resultant increase in piscivore biomass likely contributed to the collapse of kokanee stocks c. 2013 (Warnock et al., 2022). Since 2015, kokanee have been trapped in a "predator pit" where biomass and juvenile survival are depressed (Warnock et al., 2022).

3. Methods

3.1 Field and lab methods

Seven sediment cores were collected from the Kootenay Lake system in June 2023. Cores were retrieved in June 2023 from Duncan Reservoir (two cores), Trout Lake (two cores, Kootenay Lake North Arm (two cores), and Kootenay Lake South Arm (one core). A Glew gravity corer (Glew et al. 2001; 7.6 cm inner diameter) was used to collect the cores, which were then sectioned into 0.25-cm intervals and kept cold until they could be transferred to cold storage (4ºC) at Queen's University. In this study, we used one core from each of the lake basins for cladoceran analysis (**Table 3.1**).

Table 3.1 *Coring information for the four cores used in this study. Adapted from Laird et al. (2025).*

Core	Date taken	Location	Depth collected (m)	Core length (cm)
Trout A	June 8, 2023	N50° 35' 36.3" W117° 26' 51.6"	265	37.25
Duncan A	June 5, 2023	N50° 20' 54.5" W 116° 58' 06.6"	111	51.5
Kootenay North Arm A	June 2, 2023	N50° 06' 31.0" W116° 53' 34.8"	110	64.0
Kootenay South Arm	June 11, 2023	N49° 29' 36.3" W116° 48' 10.1"	165	75.5

Gamma spectroscopy measured radioisotope activity in freeze-dried sediments (22-27 subsamples per core). Core chronologies were constructed using ^{210}Pb activities, specifically by measuring the decay of unsupported ^{210}Pb (half-life of 22.3 years). Unsupported ^{210}Pb is calculated as the difference between total ^{210}Pb and background ^{210}Pb (with ^{214}Pb used as a proxy). Chronologies used a Constant Rate of Supply (CRS) model tied to the ^{137}Cs peak as an independent marker of 1963 (Appleby, 2001). Ages between CRS-dated samples were derived using linear interpolation. More detailed methodological information is given in Laird et al. (2025). Chronological information for all sediment cores is presented in **Appendix B**.

3.2 Cladoceran processing

Sediment was selected for cladoceran analysis based on the core dating models, where a 0.5-cm resolution was used for post-1990 samples and 1 to 2-cm was used for pre-1990 analyses, based on the temporal resolution of the core for periods of interest. As more recently deposited intervals in the sediment core inherently have a higher temporal resolution, intervals post-1990 generally had the highest temporal resolution (e.g., lowest average years/cm; **Table 3.2**).

Table 3.2 *Number of intervals used in cladoceran analysis and the average temporal resolution (years between each 0.5-cm interval)*

Period	Trout		Duncan		Kootenay North Arm		Kootenay South Arm	
	n	Average yrs/0.5 cm	*n*	Average yrs/0.5 cm	*n*	Average yrs/0.5 cm	*n*	Average yrs/0.5 cm
Post-1990	15	2.1	18	1.9	22	1.5	18	1.9
1963-1990	9	1.7	9	1.8	10	1.6	9	2.3
Pre-1963	5	1.9	6	3.1	12	4.6	16	2.1

Sediment was processed for cladoceran following standard methods (Korhola & Rautio, 2001); briefly, ~1.5 g of wet sediment was deflocculated in a heated 10% KOH solution for ~30 minutes then rinsed through a 35-μm sieve (an appropriate size to avoid underrepresentation of taxa with small remains that can be lost during sieving; Alric and Perga, 2011). Samples were then stored in 20 mL vials and aliquots of known volume were plated onto cover slips that were allowed to dry before being permanently mounted to slides using Entallan®. For intervals where sediment was unavailable (from use in other analyses), an adjacent interval was processed instead when possible. Additional samples were digested around historic events (e.g., Duncan Dam construction) to better characterize impacts, or around very anomalous samples to better contextualize the observations.

Cladoceran identification and measurement was performed at 200-400x magnification with a compound microscope and brightfield illumination. Identification followed Korosi and Smol (2012a,b) and Szeroczyńska and Sarmaja-Korjonen (2007). A minimum of 70 individuals was counted in each interval (Kurek et al., 2010) and the count of the most abundant remain of each taxon in each interval was used to determine the number of individuals within that sample.

Bosminid taxa were distinguished as either *Bosmina longirostris/Bosmina (Eubosmina) longispina* (hereafter *Bosmina* spp.) or *Bosmina (Eubosmina) coregoni* by the precence or absence of a mucro on their carapace valves. *Bosmina (Eubosmina) coregoni* is an invasive species within the Great Lakes and to our knowledge not previously detected in the Kootenay region, but it is recognizable by characteristic lack of a mucro and strong carapace reticulation (de Melo & Herbert, 1994; **Figure 3.3**). Differentiation of *Bosmina longirostris* and *B. (E.) longispina* was difficult and relies on rostrum pore locations which can often be obscured or fragmented in subfossil remains (Korosi & Smol, 2012a), but based on remains with clear pore locations, both taxa were present in our samples. For intervals where the total number of bosminid antennules counted exceeded total bosminid carapaces, the ratio of *Bosmina* spp. and *B. (E.) coregoni* carapaces was applied to antennule counts to determine the relative abundance of each taxa.

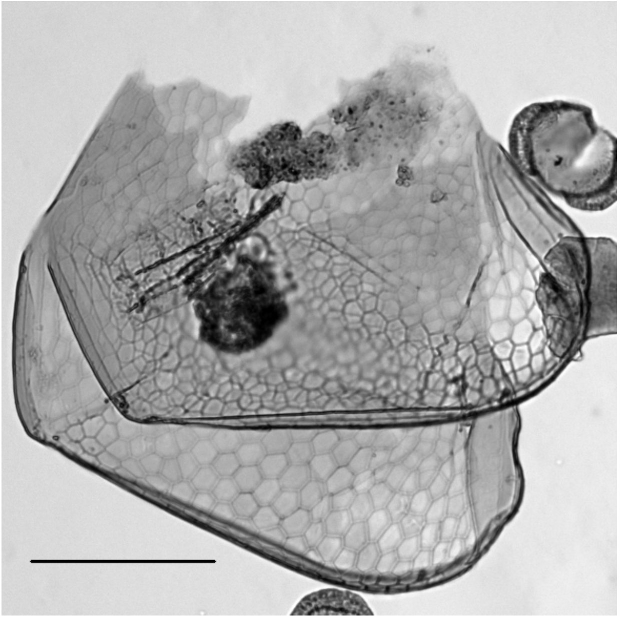

Figure 3.1. *Carapace of* Bosmina (Eubosmina) coregoni *(scale bar 100 μm). Photomicrograph from the Kootenay North Arm Core A 18.5 – 18.75 cm interval (~1965).*

Daphnia were grouped into either *Daphnia pulex* species complex or *Daphnia longispina* species complex based on the presence or absence of stout pecten on the post-abdominal claw (Szeroczyńska & Sarmaja-Korjonen, 2007; Korosi & Smol, 2012a). Given the most dominant daphnids in Kootenay Lake, our *D. longispina* complex is likely representing *D. mendotae*, *D. longispina*, and *Ceriodaphnia reticulata*, while *D. pulex* complex is representing *D. pulex* (Bassett et al., 2018).

Littoral taxa were identified to the highest possible resolution, often to species or species group, though many species could not be reliably distinguished by carapace or headshield. *Alona* cf. *guttata/circumfibriata* included *A. guttata, A. circumfibriata*, and remains of a very similar taxa whose headshield did not quite match the descriptions for either taxon but was previously described in British Columbia lakes as *Alona* cf. *circumfibriata* (Bos et al., 1999). *Chydorus sphaericus* is a complex which includes individuals from the closely related *C. brevilabris*; the former nomenclature is more prevalent in water monitoring data while the latter nomenclature is more typical to paleolimnological studies (e.g., Bos & Cumming, 2003; Korosi & Smol, 2012b).

Cladoceran measurements were taken as they can indicate changes in predation pressure over time or shifts between species that cannot be reliably identified using subfossil remains (Korosi et al., 2013). While measured sizes in sediment remains tend to be smaller on average than water column, they faithfully capture the temporal change within water column samples (Alric & Perga, 2011). *Bosmina* and *Daphnia* measurements were taken for both Kootenay Lake cores for all processed intervals, but due to time limitations, only *Daphnia* post-abdominal claw lengths were measured for Trout Lake (and only for the post-1990 time period), while no measurements were taken for Duncan Reservoir.

Measurements were performed using the photomicrography software *Eclipse* using the free-drawing tool (for *Bosmina* mucro and *Daphnia* post-abdominal caw measurements) and the line-measure tool for (for *Bosmina* carapace measurements). *Bosmina* carapace and mucro measurements were paired on the same individual with carapaces in good condition (not overly fragmented or folded) and mucro attached, and *Daphnia* claws were measured separately for *D. longispina* complex and *D. pulex* complex (**Figure 3.4**). When possible, 20 individuals were measured per interval. While 30-50 individuals is conventional (Brahney et al., 2011), the statistical method we employed (described in 2.4) better accounts for lower sample sizes in calculations of error than conventional methods (Armstrong et al., 2025). Using a lower minimum sample size allowed more intervals to be measured.

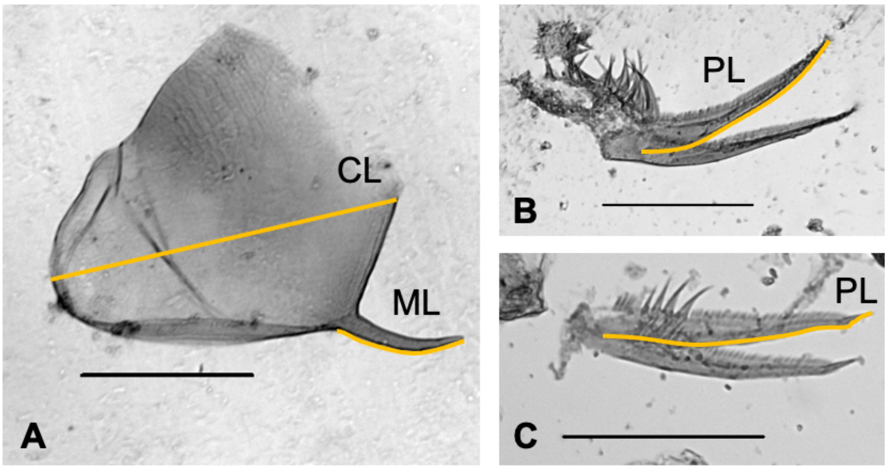

Figure 3.2. *Measurement locations for cladoceran size metrics. A)* Bosmina *carapace length (CL) and mucro length (ML). B)* Daphnia longispina *species complex post-abdominal claw length (PL). C)* Daphnia pulex *species complex post-abdominal claw length (PL). All scale bars 100 µm.*

3.3 Data analysis

Statistical analyses and data visualization were performed in RStudio v. 2024.9.0.375 (Posit team 2024; R Core Team, 2024) with general use of the packages *dplyr* (Wickham et al., 2023), *riojaPlot* (Juggins, 2023), and *ggplot2* (Wickham, 2016).

We calculated relative abundance for each taxon for each interval by dividing the number of individuals of that taxon by the total number of individuals of that interval, then multiplying by 100. Within each core, if the number of indistinguishable remains between two similar taxa was consistently higher than the number of distinguishable individuals, then their relative abundances were grouped prior to visualization: e.g., *A. affinis* and *A. quadrangularis* are identifiable by headshield, but not by carapace, while *Acroperus harpae* and *Camptocercus* can be distinguished by carapace but not by always by headshields, which are often folded down the midline (keeled). For relative abundance visualizations for each core, we grouped littoral taxa with very low relative abundances but similar morphological attributes were grouped. As a result, the groupings of littoral taxa for visualizations differ slightly by core. However, these groupings did not affect the results of any statistical analyses.

Overall cladoceran concentrations were calculated for each interval by using the number of individuals counted for each cover slip (a known volume) to back-calculate the total number of individuals in the sample. Using total number of individuals per wet weight of sample, and the percent water of each sample (either calculated from freeze-drying or by linear interpolation), we calculated the number of exoskeletons per mg dry sediment (exos / mg dw). Concentrations of individual taxa were calculated by dividing their relative abundance by 100 then multiplying it by the total concentration for each interval. Concentrations of littoral taxa were grouped, excepting *C. sphaericus*, which was calculated and presented separately. We did this as *C. sphaericus* was generally the most abundant littoral taxon in our samples and as it is a strong indicator of trophic state (Bos & Cumming, 2003; Currie et al., 2023) as it can utilize planktonic habitat during periods of (cyano)filamentous algae (Fryer, 1968). While cladoceran concentration is not a direct measure of biomass (it includes both corpses and exuviae; Kerfoot, 1981), concentration values in sediment cores have been significantly related to standing crop of an ecosystem (Hann et al. 1994), thus allowing us to infer changes in productivity.

Bosmina mucro lengths were corrected to carapace length to better reconstruct mucro-length change independent of body-size change (Armstrong et al., 2025). Essentially, the natural log (ln) of the mucro length was modelled as a function of ln-carapace length and each sediment interval (as a discrete sample), then the predicted value of mucro length (when carapace length was held constant) was calculated for each interval alongside a 95% confidence interval. As *Bosmina* carapace and *Daphnia* claw lengths do not have the same ability to be corrected based on allometry, we modelled them just as a function of each sample interval (one-way ANOVA) to determine mean length and 95% confidence intervals. To compare how overall *Daphnia* size changed, we also calculated the weighted average *Daphnia* claw length, weighting the modelled mean *D. longispina* complex and *D. pulex* complex claw lengths according to their relative abundances for each interval.

We assessed changes in cladoceran size between time periods of interest for kokanee populations to compare inferred food-web structure and function. We separated samples within four time periods: historic (pre-1930s), restoration baseline (1940-1965), artificial fertilization pre-kokanee collapse (1992-2013) and artificial fertilization post-kokanee collapse (2015-2023). A minimum of one year was left between time periods to reduce the chance that samples on the “edge” would incorporate data from the adjacent period. Even though kokanee stocks did not decline until the 1980s, the 1940-1965 period was chosen as it represents the ecosystem productivity considered ideal and guiding the restoration process before Duncan Dam became operational (Binsted & Ashley, 2005; Schindler et al., 2020). The pre-1930s period was chosen to display general properties of the site before acceleration of anthropogenic impacts beginning with the Corra Linn dam c. 1939, and likely as its broader temporal resolution also incorporates a degree of natural variability. All measurements within each period (i.e., using all raw data, not modelled means) were compared using ANOVA, with Tukey’s Honest Significant Difference to determine which periods had significantly different means.

In the North Arm samples, modelled *Bosmina* mucro and *D. longispina* post-abdominal claw measurements from 1990-2023 were tested for cyclicity using sinusoidal functions (using *sin* and *cos* functions within the standard ‘*lm’* function in R). *Daphnia pulex* measurements were not tested as *D. pulex* were either absent or in low abundances for many of these intervals. Models were created by using *sin* and *cos* functions within a linear model. The period for each model was calculated by dividing the total number of years by the number of observed maxima, and model terms were assessed with manual best fitting. These models were not made to be predictive of cladoceran lengths, but to assess if these size measurements significantly showed cyclicity, as might be expected of predator-prey cycles.

4. Results

4.1 Trout Lake

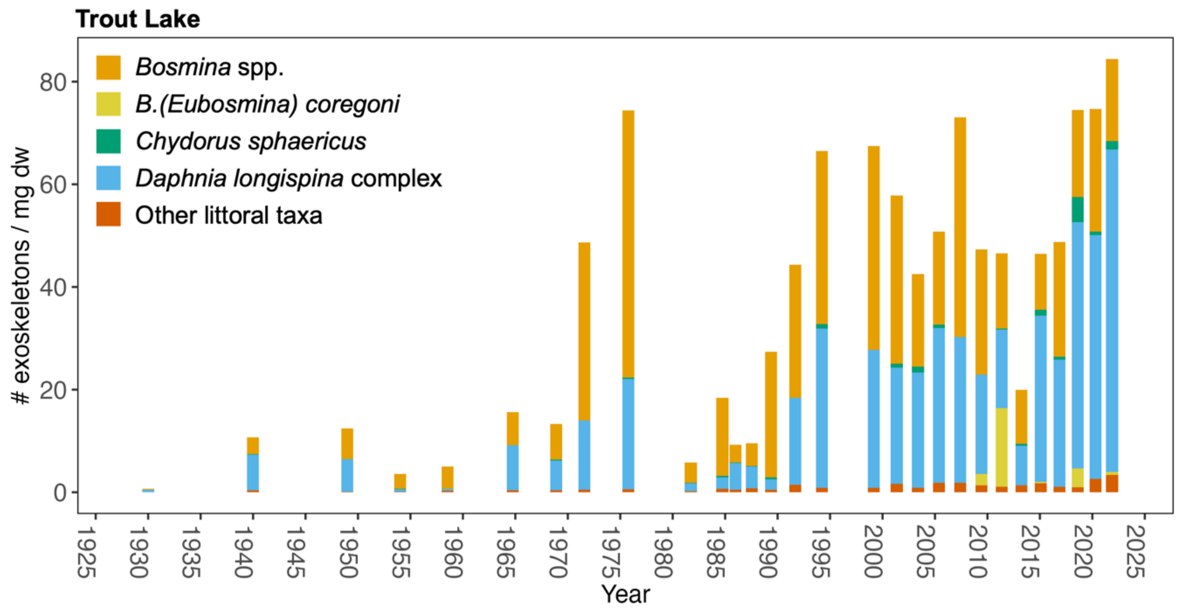

Figure 4.1. *Cladoceran concentrations (exoskeletons / mg dry weight) for Trout Lake core A by CRS date of each sediment interval.*

The age-depth chronology of the Trout Lake core was complex and may contextualize some of the concentration trends. There were periods of very high sedimentation which resulted in several, short-lived periods of 'plateaus in age' around 19-20 cm, 15-16 cm, and 8.5-10 cm, (**Figure B3**). We noticed more organic material and debris in the prepared cladoceran slurries and on plated slides for the 8.5 cm, 19.5 cm, and 21.5 cm intervals than was characteristic for other intervals in the core (**Figure B4**). Further, some of these correspond to intervals with very low concentrations (e.g., 19.5 cm, or ~1958) and/or low relative abundance of *Daphnia* compared to *Bosmina* (e.g., 8.5 cm or 1989).

Cladoceran concentrations were very low ~1930 (< 1 exos mg/dw) but increased in 1940 and 1950, with the assemblage dominated by *D. longispina* complex (**Figure 4.1, Figure C1**). Concentrations of *Daphnia* then decreased sharply in 1955-1960 before increasing again in the 1960s. Concentrations of both *Daphnia* and *Bosmina* increased in the 1970s, though *Bosmina* had a larger proportional increase, and then declined post-1980. After 1990, overall concentrations increase, and *Daphnia* return to approximately 40-50% of the assemblage. The relative abundance and concentration of littoral taxa increased, and in ~2011 there was a spike of *B.(E.) coregoni* at 30% relative abundance. In 2009-2013, concentrations of *Daphnia* decreased, but post ~2014, *Daphnia* dominated the assemblage (60-70% relative abundance) and reached their highest concentrations within the sediment core (up to 62 exos mg/dw). *Chydorus sphaericus* concentrations increased in 1995, with their highest values seen 2018/2019.

Daphnia longispina complex claw lengths were ~146 µm at 1992 (**Figure 4.2**). They increased to 157 µm by 1999, then decreased to a low of 136 µm around 2003 and remained low, with minima of 131 µm in 2007 and 136 µm in 2013. Post-2014, claw lengths increased, with a maximum of 160 µm ~2016 and most recent values of 157 µm ~2022.

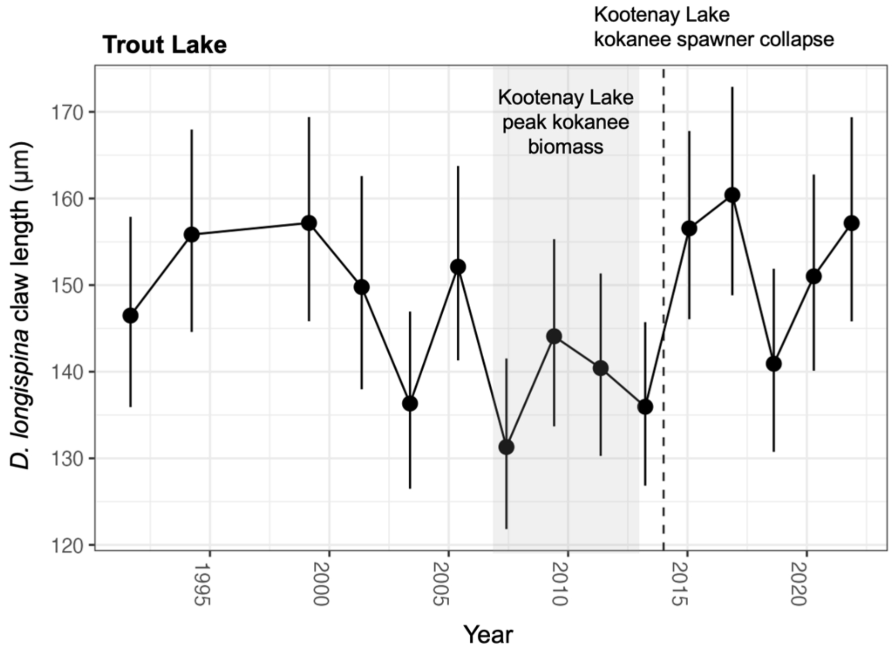

Figure 4.2. Daphnia longispina *complex post-abdominal claw lengths for Trout Lake from ~1991 to ~2021. The shaded grey period is the time of peak kokanee biomass in Kootenay Lake, and the dashed line is the Kootenay Lake kokanee spawner collapse.*

4.2 Duncan Reservoir

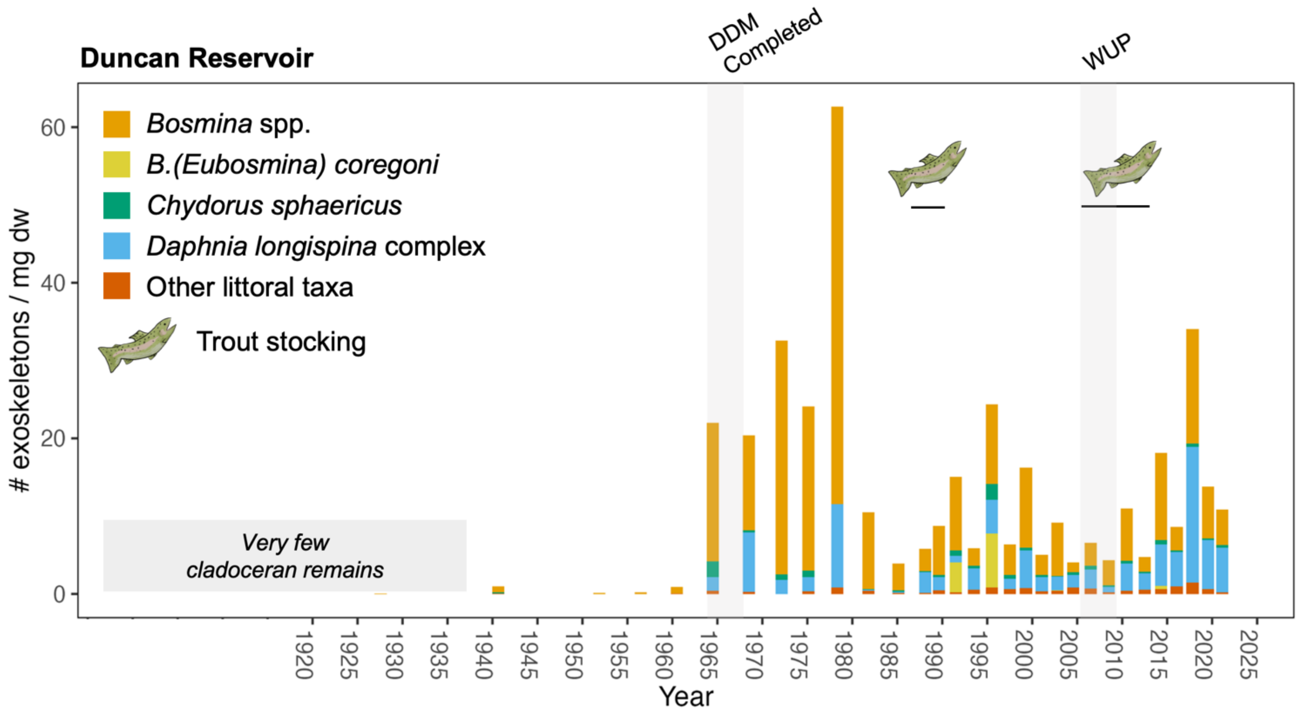

Figure 4.3. *Sedimentary cladoceran concentrations (# of exoskeletons / mg dry weight) by CRS-modelled year of Duncan Reservoir sediment core A. The approximate timing of Duncan Dam (DDM) completion and the Water Use Plan (WUP) are shown by grey bars, and periods of annual rainbow trout stocking are indicated by the horizontal black line. Rainbow Trout illustration by IA.*

Sedimentary concentrations of cladoceran remains are low prior to the creation of Duncan Reservoir (**Figure 4.3**). Relative abundance pre-dam could not be characterized due to the scarcity of remains, but the most common taxa found were *Bosmina* spp. and littoral *Alona,* while *D. longispina* complex remains were not observed in sediment samples prior to reservoir creation. Concentrations increase c. 1965 (going from < 1 exoskeletons/ mg dw to > 20), coincident with the creation of Duncan Dam. Following reservoir creation, the cladoceran assemblage is dominated by *Bosmina* spp. (74% relative abundance), though *D. longispina* is also present in notable relative abundances (initially 6%, increasing to 36.5% ~1969, before decreasing < 10%; **Figure C2**). There was a sharp decline in cladoceran concentrations c. 1980, from 64 to 10.5 exoskeletons /mg dw, in which *D. longispina* complex decrease proportionately more than *Bosmina.*

Daphnia return just prior to 1990, and overall concentrations increase, with pulses of *Chydorus sphaericus* and *B. (E.) coregoni* in 1991 and 1996. There has also been an increase in the concentration of other littoral Cladocera post-1990, with *Alonella excisa, Alona affinis/quadrangularis,* and *Camptocercus/Eurycercus*-type as the most prominent littoral taxa (aside from *C. sphaericus*). Since ~2000, the cladoceran assemblages show a gradually decreasing relative abundance of *Bosmina* spp. and an increasing relative abundance of *D. longispina* complex, which is most prominent post-2015.

The year-to-year concentrations post-1990 are variable. Some of this variation may be an artifact of subsampling: approximately half of the post-1990 samples had very little sediment left after prior subsampling, and this could have biased concentration values. When looking at a more general trend, we do see concentrations decreasing from 2000 to 2015, then increasing again (with a notable increase in *Daphnia)* post-2015. Monitoring data from Duncan Reservoir found a huge biomass of *Daphnia* in 2018 due to a longer growing season (temporally, this aligns with our data – it is the c. 2018 interval when *Daphnia* concentrations are highest), which shows that sediment concentrations are (to an extent) reflecting water column samples.

4.3 Kootenay Lake North Arm

Cladoceran concentrations were relatively low prior to the 1930s (< 15 exos/mg dw), after which they increased until ~1960 (~20 exos/mg dw), when they declined by 90% (~2 exos/mg) and remained very low until ~1990 (**Figure 4.4**). There is an exception c. 1965 (around the time of Duncan Dam completion, given error in our dating model), when there was a sudden spike in *Bosmina* and *B. (E.) coregoni* concentrations. In this interval, *B.(E.) coregoni* reached a relative abundance of 30%, whereas it only appeared in a few other intervals in low relative abundances (**Figure C3**). The assemblage structure was evenly dominated between *Bosmina* and *Daphnia* until ~1970, when relative abundances of both taxa declined in favour of littoral taxa (reflecting a decrease in the concentrations of *Bosmina* and *Daphnia*, not an increase in the concentrations of littoral taxa). Post-1980, *Bosmina* dominated (relative abundance > 80%) until the kokanee collapse.

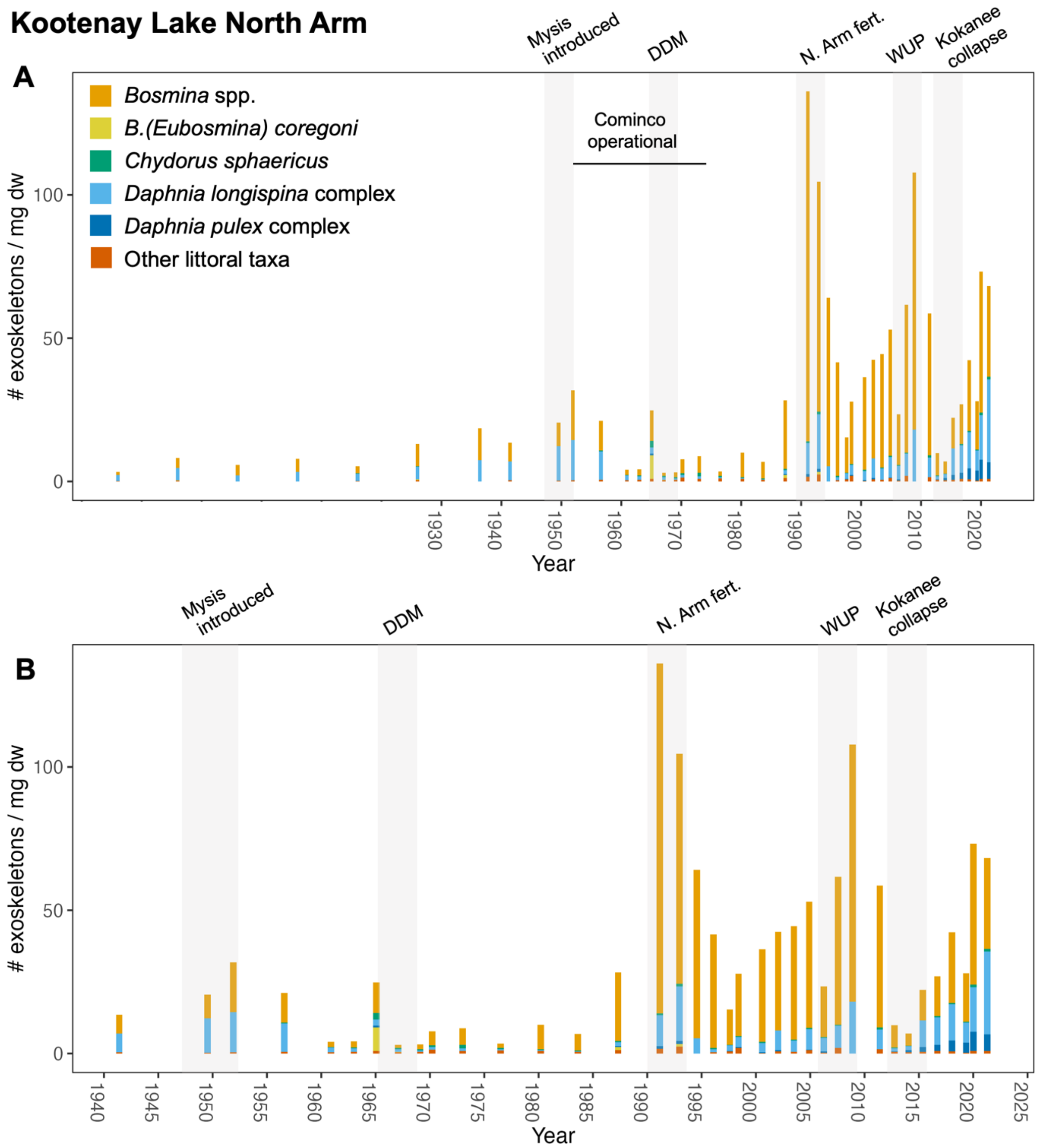

__Figure 4.4.__ Sedimentary cladoceran concentrations (# of exoskeletons / mg dry weight) by CRS-modelled year for (A) all measured intervals and (B) post-1940 intervals for the Kootenay Lake North Arm sediment core A. The approximate timing of historic events including Duncan Dam (DDM) completion and the Water Use Plan (WUP) are shown by grey bars. Undated intervals are spaced evenly (10 years per cm core depth).

Concurrent with artificial fertilization, the concentrations of *Bosmina* and *D. longispina* complex increased dramatically post-1990, with an initial increase of 480% for *Bosmina* (23 to 133 exos/mg dw) and 670% for *Daphnia longispina* comlex (1.4 to 10.8 exos/mg dw). Subsequently, concentrations declined, reaching lows in 1997 and in 2006, before peaking again ~2009 (90 and 18 exos/mg dw for *Bosmina* and *D. longispina* complex respectively). *Daphnia* were very low again in 2012-2014 (< 1.5 exos/mg dw) but increased in the post-kokanee-collapse state and reached their highest concentrations in the core in

2021 at 29 exos/mg dw. Post-2014, *D. pulex* complex increased dramatically in relative abundance (18-20%) and concentration (up to 6 exoskeletons mg / g dw). Post-2014, *Bosmina* decreased to ~50% of the assemblage, with *D. longispina* complex at ~28% and *D. pulex* complex at ~17%.

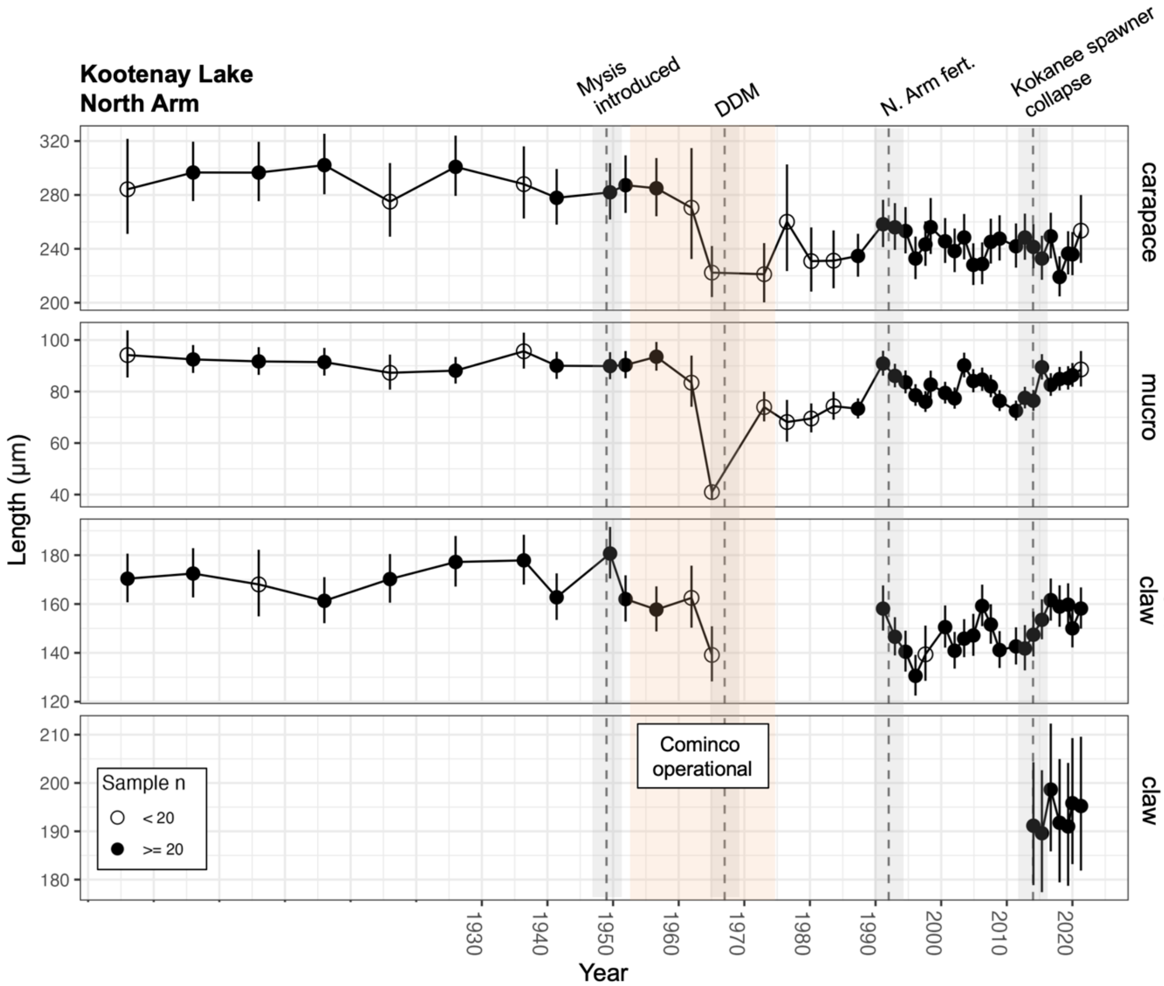

Figure 4.5. *Size metrics for the North Arm core. The approximate timing of historic events is given by dashed lines, ± 2.5 years to account for error in our dating model and to fit the resolution of the figure. The approximate operation time of intense Cominco fertilizer discharge is given in the pink overlay. Undated intervals are spaced evenly.*

Bosmina carapace lengths were high (mean = 287 µm) until 1965 (approximate Duncan Dam completion), when they declined to 222 µm and remained low until ~1991, excepting a peak of 260 µm ~1977 (**Figure 4.5, 4.6**). Mucro lengths showed a similar trend: they were initially high (mean = 91 µm) and then declined to 41 µm ~1965, though they then increased to ~72 µm until ~1991. Following artificial fertilization in 1992, mucro lengths increased to 91 µm and carapace lengths to 258 µm. After this initial increase, both mucro and carapace lengths decline and show a cyclical trend. Mucro lengths had another maximum ~2004 and were lower (< 80 µm) during the time of peak kokanee biomass. They increased post-kokanee collapse and averaged 86 µm post-2014.

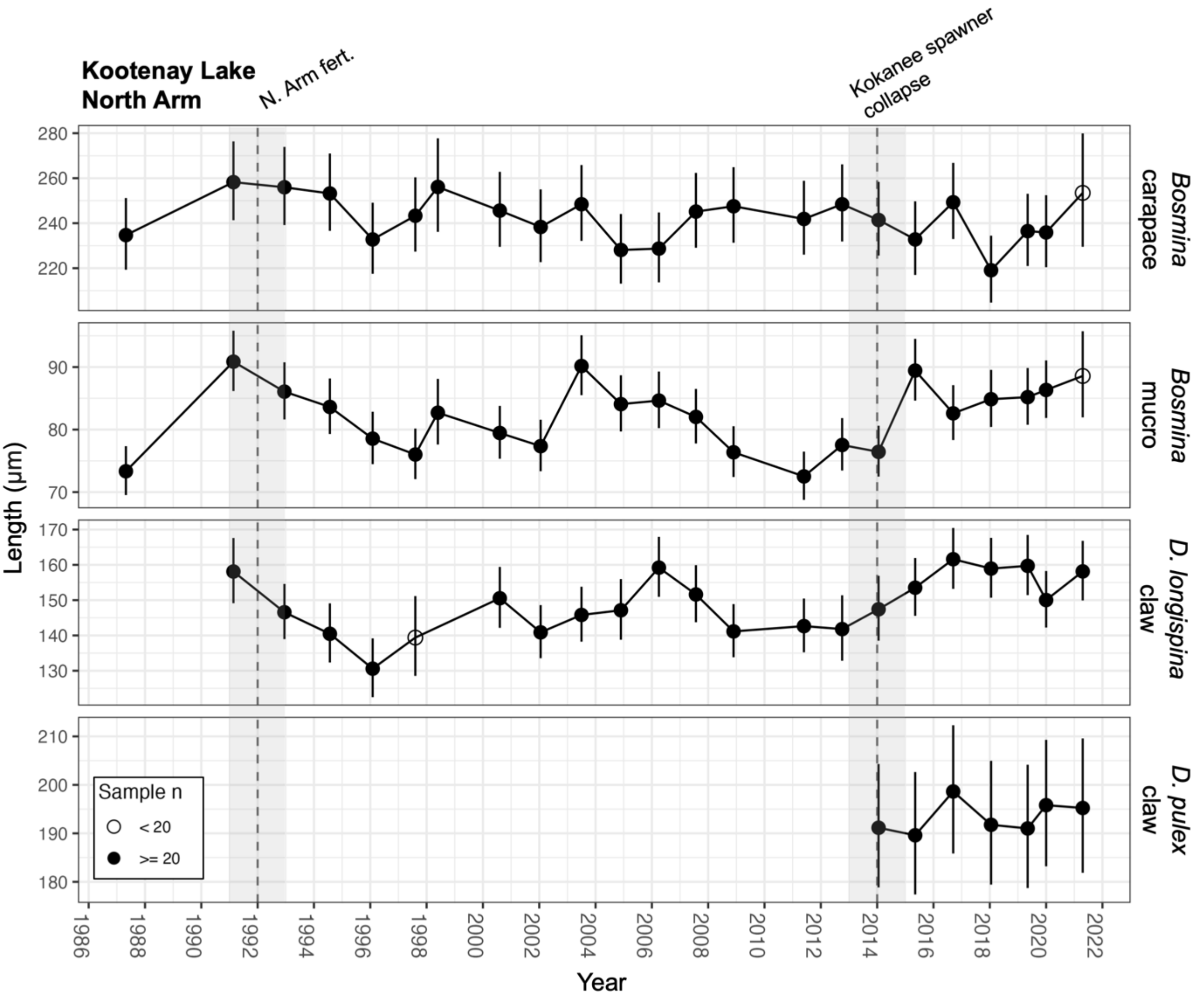

Figure 4.6 *Size metrics for the North Arm core. The approximate timing of historic events is given by dashed lines, ±1 years to account for error in our dating model and to fit the resolution of the figure.*

Daphnia longispina complex claw lengths were high (mean = 171 µm) until 1950, after which they declined to 139 µm around 1965. Concentrations of *Daphnia* were too few to measure claw length until ~1992 (158 µm), and then claw lengths decreased to 130.5 µm by 1996. Claw lengths remained low (average = 146 µm) until the kokanee spawner collapse, after which they increased > 150 µm. The *D. pulex* complex measured in our samples were larger than the *D. longispina* complex, with a claw length of 194 µm post-2014. While *D. longispina* complex claw lengths do not return to the pre-1930s levels, the weighted average claw length of *Daphnia* post-2014 was 172 µm, similar to the pre-1950s values.

In the North Arm, both *Daphnia* claw length and *Bosmina* mucro length can be effectively modelled using sinusoidal curves ($p < 0.001^{***}$; **Figure C4**; **Equation C1, C2**) showing that these sizes are behaving in a cyclical way from the early 1990s to the kokanee spawner collapse, with an approximately 8-year period (double the 4-year life span of kokanee). When these metrics are compared to kokanee population data (Warnock et al., 2022), they show an inverse relationship to population, suggesting these curves may reflect top-down predation. Evidently, these models do not quite fit the data as they assume increase and decrease to be symmetrical, whereas the *Bosmina* data suggest abrupt increase and gradual decreases.

3.4 Kootenay Lake South Arm

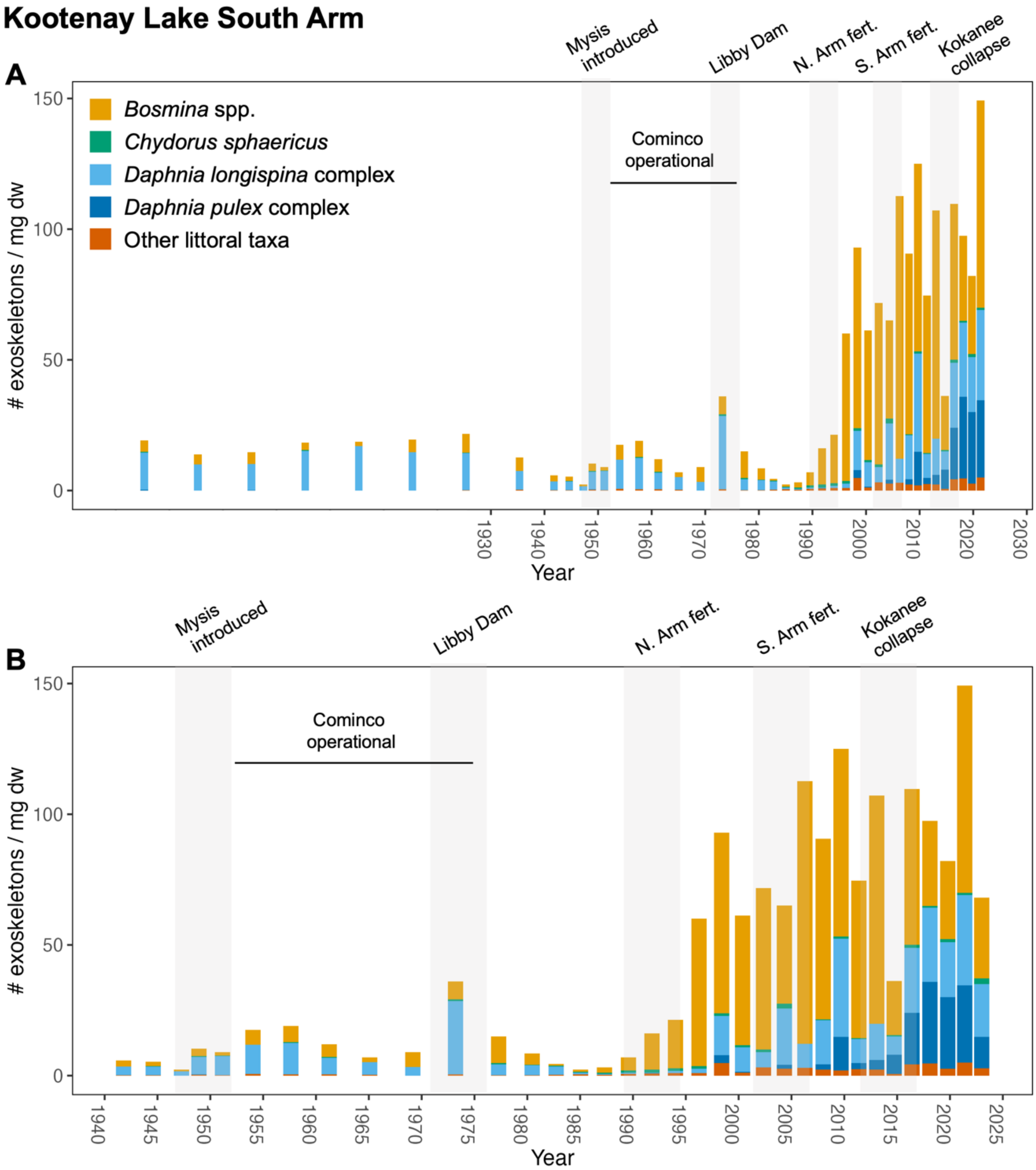

Figure 4.7. *Sedimentary cladoceran concentrations (# of exoskeletons / mg dry weight) by CRS-modelled year for (A) all measured intervals and (B) post-1940 intervals for the Kootenay Lake South Arm sediment core A. The approximate timing of historic events are shown by grey bars and the time that the Cominco fertilizer discharge was most intense is shown by a black bar. Undated intervals are spaced evenly (10 years per cm core depth).*

Pre-1930, cladoceran concentrations in the South Arm averaged 4 exos/mg dw for *Bosmina* and 14 exos/mg dw for *D. longispina complex* (**Figure 4.7**). Both *Bosmina* and *D. longispina* complex decreased in concentration post-1930, with their lowest values by 1947 (0.7 exos/mg dw for *Bosmina* and 1.6 exos/mg dw for *D. longispina* complex). Concentrations of both taxa increased in the 1950s coinciding with the onset of Cominco operation, though concentrations remained lower than the pre-1930s values. *Daphnia longispina* complex dominated the assemblages prior to 1969 when they dropped to ~37% of the assemblage (**Figure C5**). However, there was a dramatic and short-lived increase in *D. longispina* concentrations c. 1973, at the time of Libby Dam construction: concentrations were over 6x higher than adjacent intervals, with no corresponding increase in *Bosmina*. Following this, *Daphnia* and *Bosmina* declined, coinciding with both dam construction and the closure of the Cominco fertilizer plant, and were in low abundances through the 1980s and early 1990s (both taxa with concentrations < 1 exos mg/dw).

At the timing of onset of artificial fertilization in the North Arm, *Bosmina* increased in concentration, from 5 exos/mg dw pre-1990 to 62 exos/mg dw by 2002, while *D. longispina* complex were low until ~1998, when they increase from 1.5 exos/mg dw to 15 exos mg/dw. *D. pulex* complex becomes notable in the record following 1998. Aligning with South Arm fertilization, *D. longispina* concentrations increase again ~2004 (to 21.5 exos/mg dw) and *Bosmina* have a major increase in the following interval, ~2006, to 100 exos mg/dw. *Daphnia* concentrations peak in 2009 and combined *Daphnia* reach 40% relative abundance. Concentrations then declined, with low values (< 10 exos mg/dw each for *D. longispina* and *D. pulex*) in 2014. Post-kokanee collapse, concentrations increased, with combined *Daphnia* concentrations exceeding > 50 exos/mg dw by 2018. This was driven by a large increase in *Daphnia pulex*, reaching up to 30% relative abundance. *Bosmina (Eubosmina) coregoni* also appears in the South Arm record post-2014, though in low relative abundances.

Bosmina carapaces were large (279 µm) until ~1949 (the time of *Mysis* introduction), at which point they decreased to 226 µm by ~1951 (**Figure 4.8, 4.9**). Carapace lengths remained low until 1970 (up to 250 µm), then decreased again to a minimum of 213 µm ~1992 (onset of North Arm fertilization) before showing an upward trend that continues to modern-day samples. *Bosmina* mucro lengths were high (mean = 89 µm) before 1949, and then decreased to 67 µm following *Mysis* introduction before increasing to a post-1960 average of 76 µm. Around 1992, they decreased to 60 µm, then increased gradually and from 2000 to 2023 they averaged 78 µm.

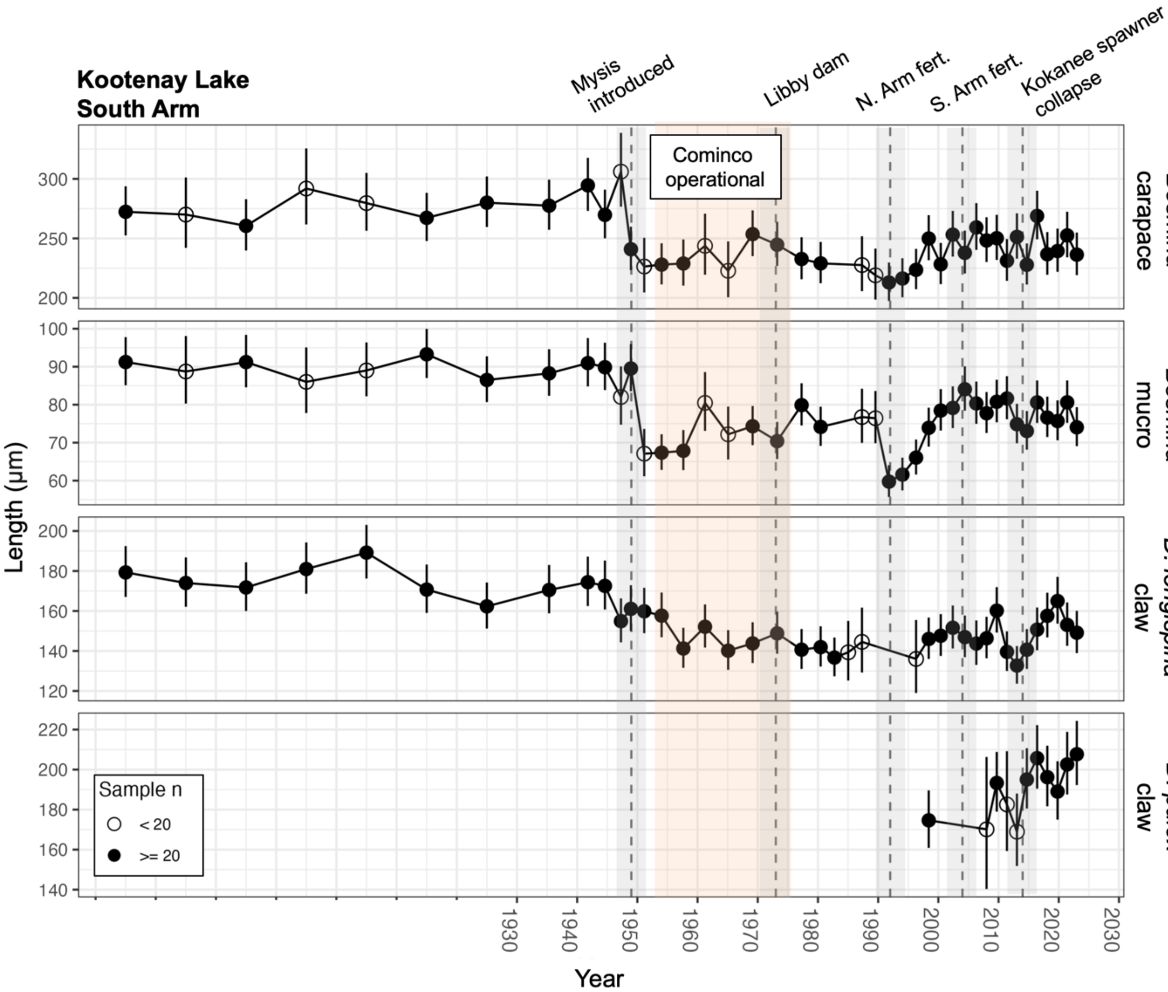

Figure 4.8. *Size metrics for the South Arm core. The approximate timing of historic events is given by dashed lines, ± 2.5 years to account for error in our dating model and to fit the resolution of the figure. The approximate operation time of intense Cominco fertilizer discharge is given in the pink overlay. Undated intervals are spaced evenly.*

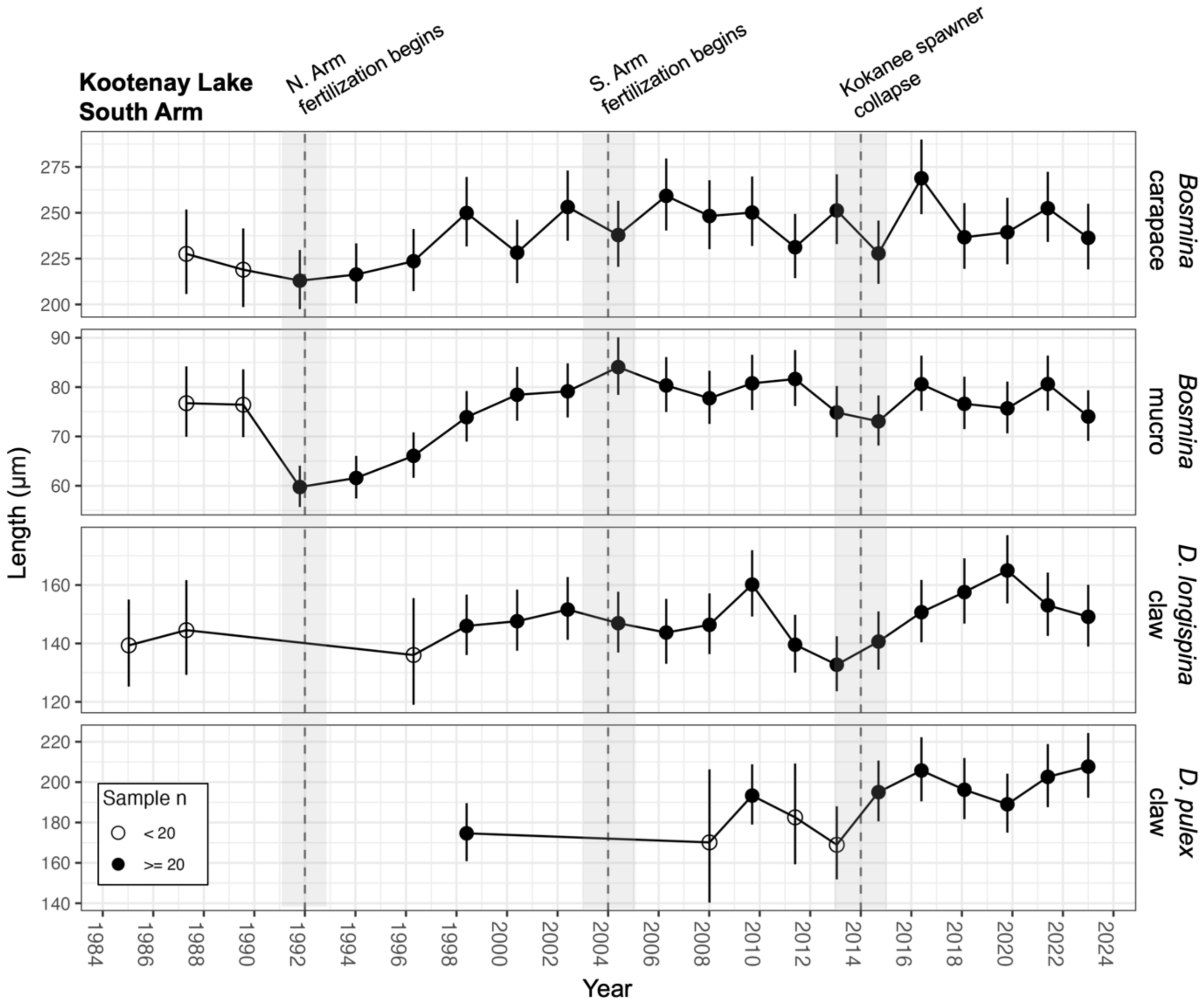

Figure 4.9. *Size metrics for the South Arm core post-1984. The approximate timing of historic events is given by dashed lines, ± 1 years to account for error in our dating model and to fit the resolution of the figure.*

D. longispina claw lengths were largest at the bottom of the core (179 µm) and declined to 162 µm prior to 1930. They declined again in 1949 and reached minima of 141 µm ~1958 and 137 µm ~1982. Claw lengths increased starting in 1998, up to 160 µm in 2009, and then fell to 133 µm in 2013. Post-2015, they averaged 155 µm (lower than pre-1930s values). *Daphnia pulex* claw lengths averaged 182 µm prior to 2014, then increased to 200 µm post-kokanee collapse. The weighted average *Daphnia* claw size is 176 µm; e.g., when the size and relative abundance of both *Daphnia* species are considered, the overall *Daphnia* size structure is similar to pre-1930.

3.5 Between core comparisons

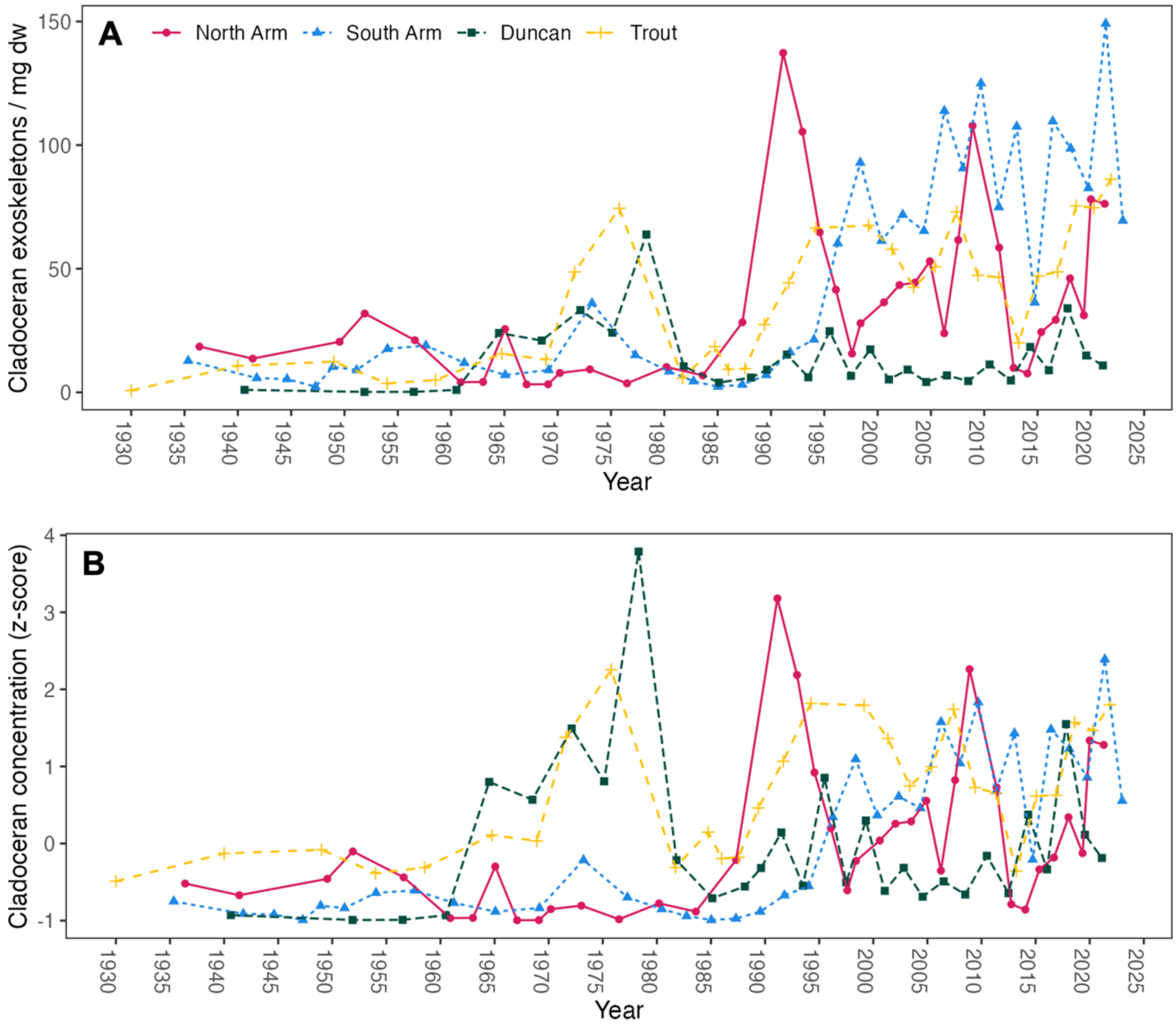

Figure 4.10 *Cladoceran concentrations by exoskeletons / mg dw (A) and by Z-score for each core (B) for each of the four sediment cores by CRS-modelled date from 1930 to 2023.*

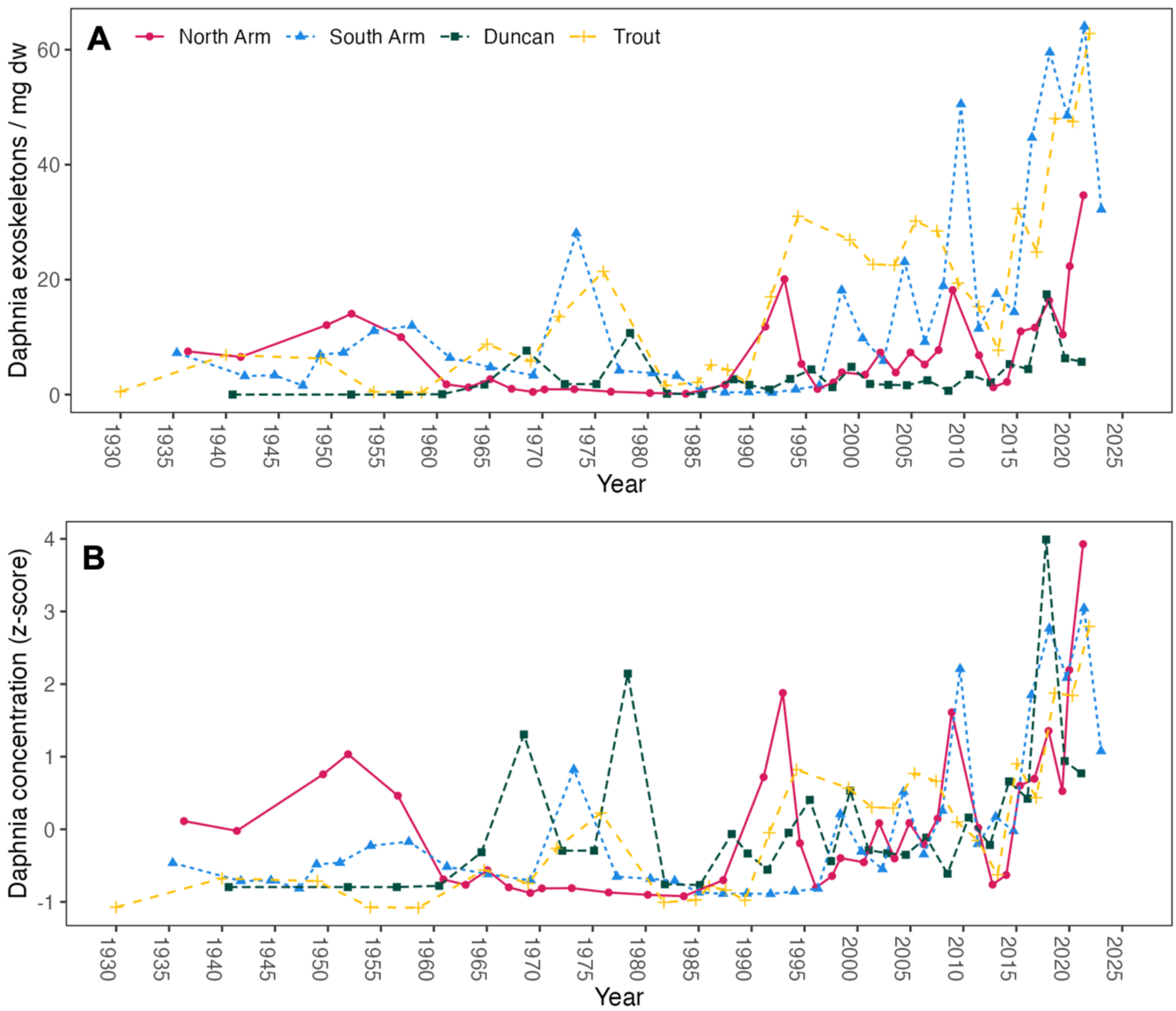

Figure 4.11 Daphnia *concentrations (including both* D. longispina *complex and* D. pulex *complex) by exoskeletons / mg dw (A) and by Z-score for each core (B) for each of the four sediment cores by CRS-modelled date from 1930 to 2023.*

Cladoceran concentrations in all four cores have increased over time (**Figure 4.10**). Duncan Reservoir concentrations are highest during the initial inundation period, while Kootenay Lake concentrations are lowest during cultural eutrophication and highest during artificial fertilization. Trout Lake concentrations do increase post-1990 but are at similar values to the 1970-1980 intervals. Both Kootenay Lake cores have *Daphnia* at or over 50% relative abundance but show a shift to a *Bosmina*-dominated assemblage in the 1970s until ~2014. All four cores show an increase in *Daphnia* relative abundance and concentrations since 2015 (**Figure 4.11**), though *Daphnia pulex* complex is only consistently present in the Kootenay Lake cores. Littoral taxa are rare in all four cores, making it difficult to interpret shifts in relative abundance, especially as periods of higher relative abundance co-occur with low pelagic concentrations. However, all four cores show an increasing concentration of littoral taxa beginning in the 1990-2000 period.

The measured size values for the North and South Arms (for each metric) are quite similar and are generally consistent with each other through much of the sediment core (**Figure 4.13**). For example, *Bosmina* carapace length in both cores declines to ~220 µm in 1965 and claw length in both cores is also at ~139 µm. Further, both cores show very similar carapace values post-2000, and average *Daphnia* claw lengths reach similar minima in both 1996 and in 2014. The weighted average *Daphnia* claw lengths increase in 2015 in both cores to very similar (~170 µm) lengths. These congruences suggests they are an accurate characterization of temporal trends in the mean *Bosmina* and *Daphnia* population size structure. This also suggests that core-specific differences are reflective of spatial patterns in Kootenay Lake between cores; e.g., *Bosmina* carapace and mucro lengths decrease in the South Arm c. 1949 but do not change in the North Arm core. *Bosmina* measurements also show opposing trends between cores ~1992. In Trout Lake, the *D. longispina* complex claw lengths are very similar to those measured in Kootenay Lake (**Figure 4.13, Figure C6**). Post-2014, they do not increase to the same weighted *Daphnia* value (due to a lack of *D. pulex* in Trout Lake; **Figure 4.14**).

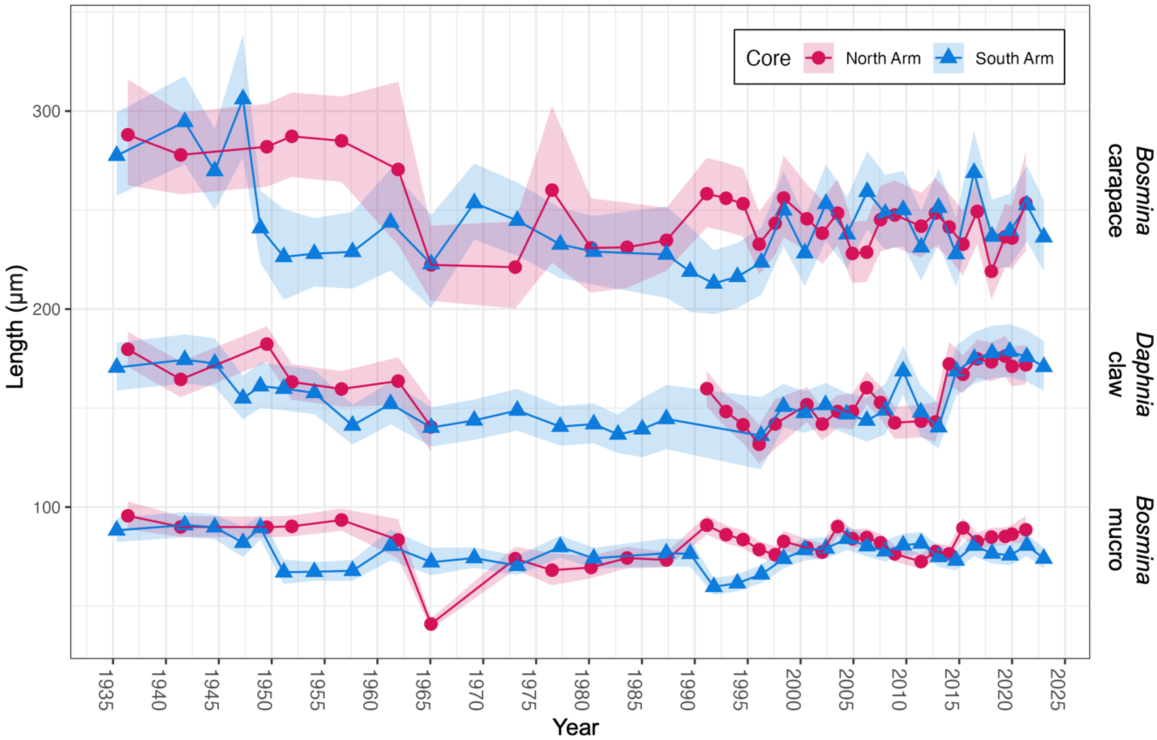

Figure 4.12. *Summary of cladoceran size metrics between the North and South Arms of Kootenay Lake in CRS-dated intervals.*

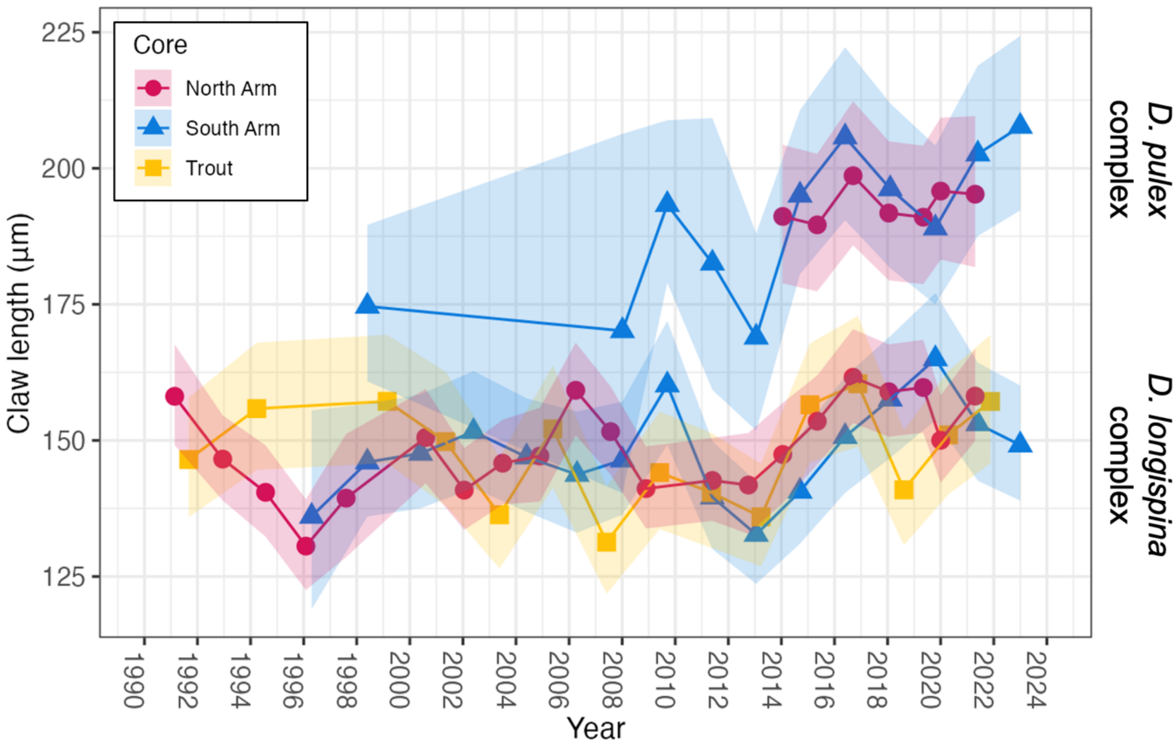

Figure 4.13. Daphnia pulex *complex (top) and* Daphnia longispina *complex (bottom) post-abdominal claw lengths for North and South Arms of Kootenay Lake and for Trout Lake for 1990-onward, with error shading showing the 95% confidence intervals.*

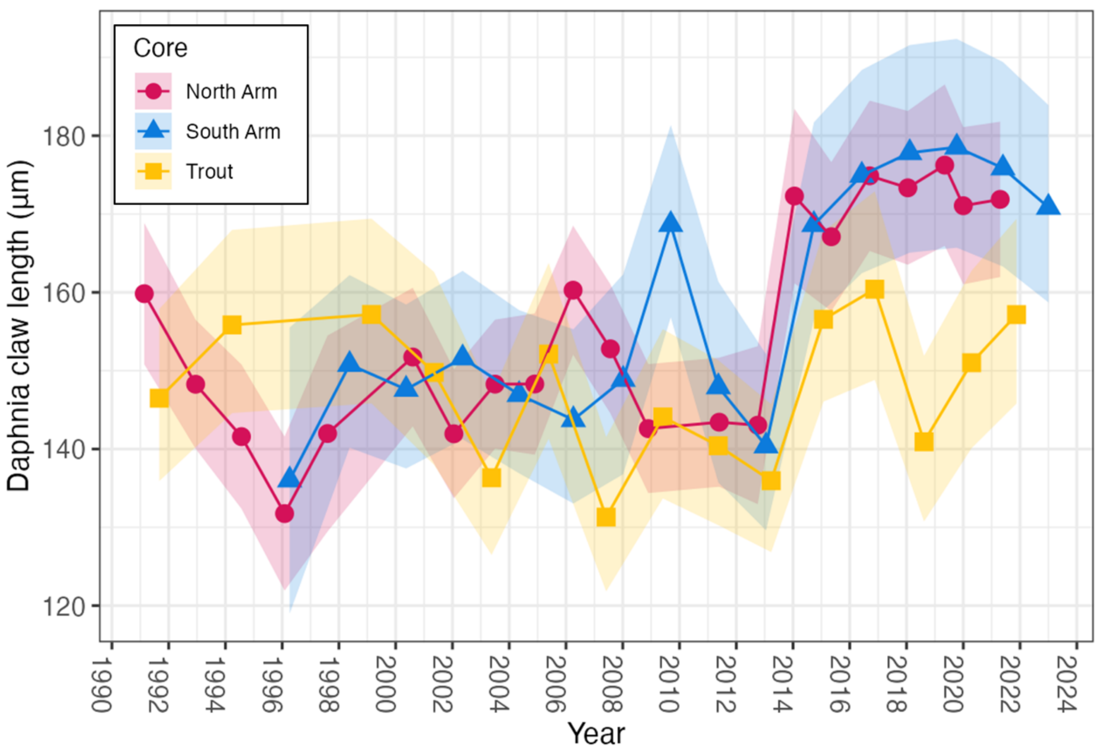

Figure 4.14. *Weighted average* Daphnia *post-abdominal claw lengths for North and South Arms of Kootenay Lake and for Trout Lake for 1990-onward, with error shading showing the 95% confidence intervals.*

4 .6 Size structure: time-period comparisons

In the North Arm, *Bosmina* size structure was not significantly different between the historic and restoration baseline periods (**Figure 4.15**). However, carapace and mucro lengths in the historic and baseline periods were significantly higher than the artificial fertilization period. Mucro lengths in the post-collapse period were significantly larger than the artificial fertilization period, and on par with the baseline period, while carapace lengths in the post-collapse period did not differ significantly from the artificial fertilization period. In the South Arm, *Bosmina* carapace and mucro lengths are significantly larger pre-1930s than all other periods, and are smallest in the artificial fertilization period, while size-structure in the post-collapse period occupies an intermediate position between artificial fertilization and restoration baseline. Overall, this suggests that predation on Kootenay Lake *Bosmina* from invertebrate predators may have increased post-collapse, but *Bosmina* body size remains smaller than the historic and restoration periods.

Daphnia longispina complex claw lengths in the North Arm did not differ significantly between historic and restoration baseline periods but were significantly smaller in the artificial fertilization period (**Figure 4.14**). In the post-collapse period, claw lengths are an intermediate length, and the mean is significantly different from the other periods. In the South Arm, *Daphnia longispina* complex claw lengths are significantly longer in the historic period than in all other periods and are significantly smaller during the artificial fertilization period. Claw lengths in the restoration baseline period are similar to the post-collapse period. Inferred zooplanktivory was lowest pre-1930s and highest in the artificial fertilization period, with some lessening of pressure post-kokanee collapse, though *Daphnia longispina* body size remains smaller. More detailed output for the Tukey’s Honest Significant Difference testing is given in **Table C1.**

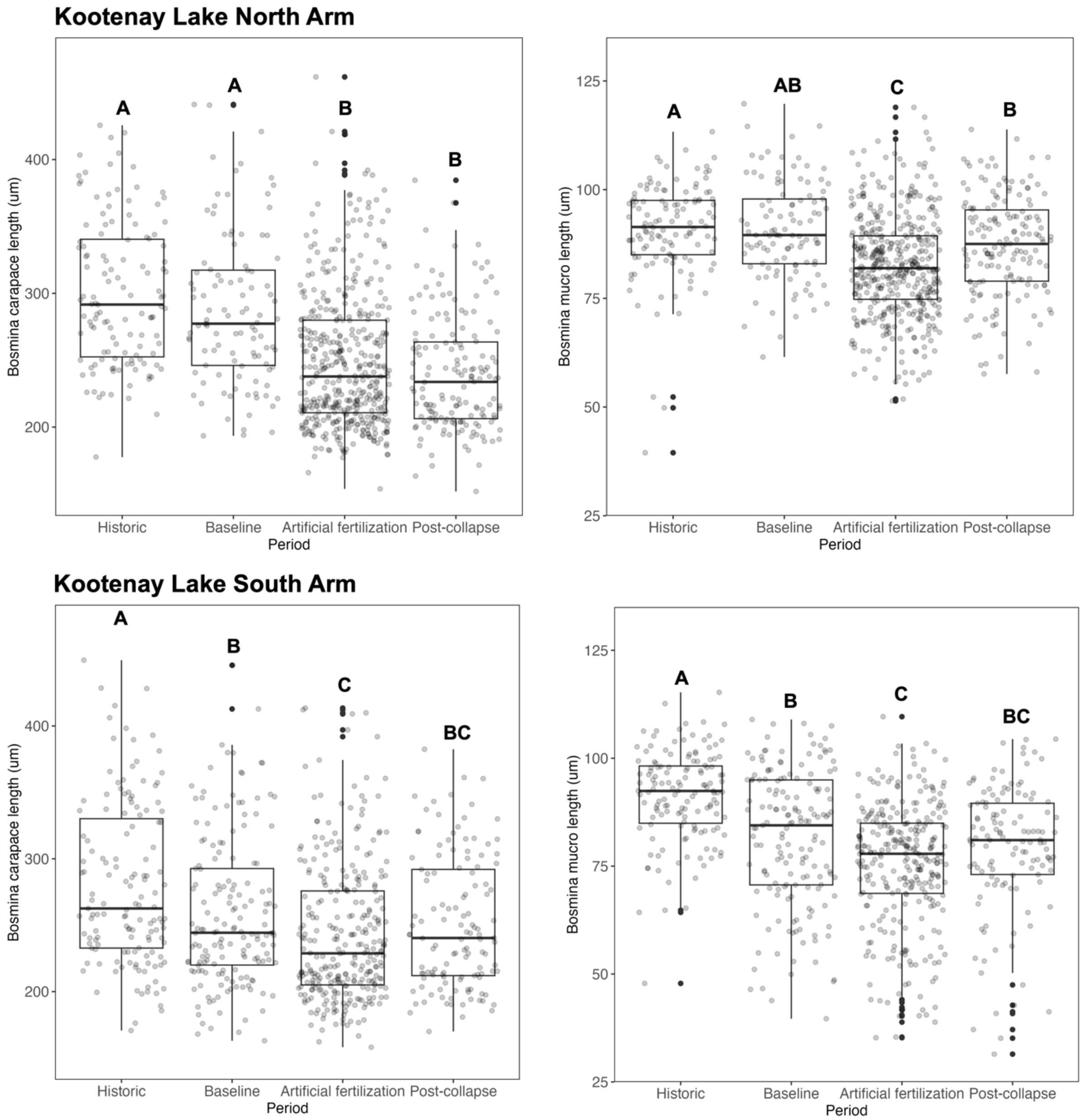

Figure 4.15. *Bosmina spp. carapace and mucro lengths for the North and South Arms for the different periods: historic (pre-1930s), restoration baseline (1940-1965), artificial fertilization, pre-kokanee collapse (1992-2013), and post-kokanee collapse (post-2015). Bolded letters show periods within each sediment core where means are not significantly different (e.g., AB would have the same mean as both A and B, while A and B themselves are significantly different) as determined by Tukey's Honest Significant Difference.*

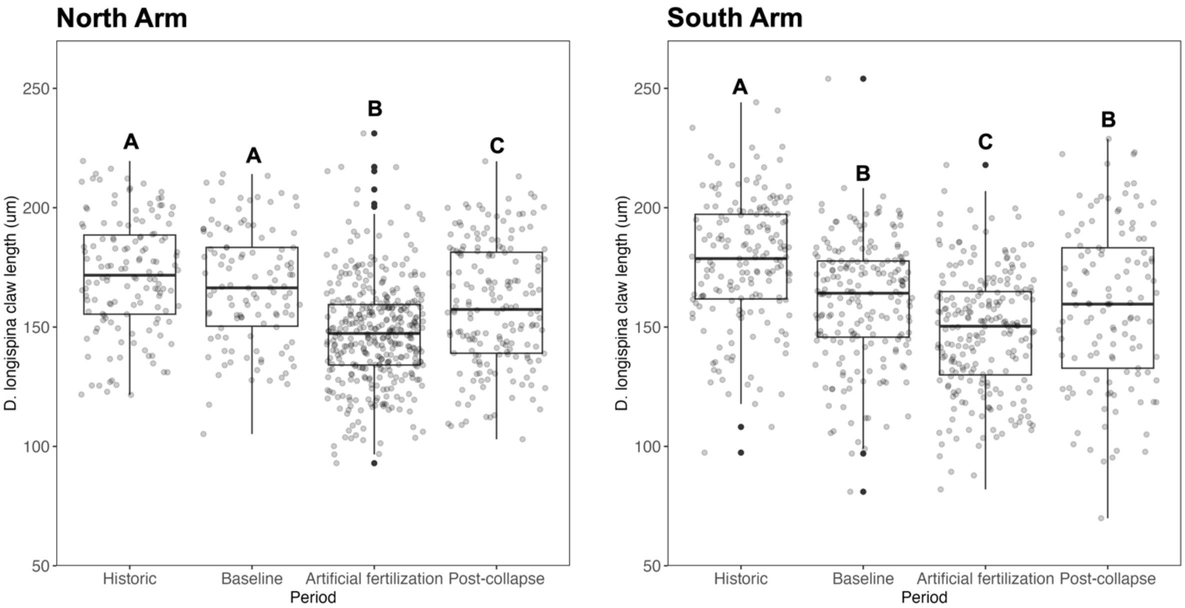

Figure 4.16. Daphnia longispina-*complex post-abdominal claw lengths for the North (left) and South (right) Arms for the different periods: historic (pre-1930s), restoration baseline (1940-1965), artificial fertilization, pre-kokanee collapse (1992-2013), and post-kokanee collapse (post-2015). Bolded letters show periods within each sediment core where means are not significantly different (e.g., AB would have the same mean as both A and B, while A and B themselves are significantly different) as determined by Tukey's Honest Significant Difference.*

5. Discussion

5.1 Early introduction of invasive Bosmina (Eubosmina) coregoni?

An invasive species, *B. (E.) coregoni* was first detected in the Great Lakes in 1965. It was found in Lake Winnipeg, Manitoba, in the late 1980s (Suchy et al., 2010), and its first detection in the Pacific Coast was in 2008 (Smits et al., 2013). While a taxon referred to as *Bosmina coregoni* has been documented in BC lakes in the early 1900s (Zyblut, 1967; Patalas and Salki, 1973), this is likely referring to either *Bosmina (Eubosmina) longispina* (as it has been previously known as *Bosmina coregoni longispina*) or *Bosmina longirostris* (e.g., Zyblut refers to *Bosmina coregoni* as the dominant bosminid in Kootenay Lake and makes no distinction between mucronate forms). The taxonomy of *Bosmina* is notoriously complex, and many species have been renamed/redesignated (Kotov et al., 2009). However, to our knowledge, *B.(E.) coregoni* is the only taxa where the mucro is absent (De Melo & Herbert, 1994).

In the North Arm core, our data suggest that *B. (E.) coregoni* was present in Kootenay Lake at a similar timing to when it invaded the Great Lakes (**Figure 4.4**). The dominance of *B. (E.) coregoni* c. 1965-7, at 30% relative abundance, is unlikely to be artificial. It may be possible that the introduction of *M. relicta* to the lake inadvertently introduced *B.* (*E.) coregoni. Mysis* prey on *Bosmina* resting eggs in the sediment of the Baltic Sea (Viitasalo and Viitasalo, 2004). Consumption and excretion of resting eggs could be a vector for introduction: many invasive aquatic invertebrates have been accidentally introduced during stocking events of other invertebrates, and resting eggs can survive unfavourable conditions (Hänfling et al., 2011)

The timing of *B. (E.) coregoni* in the Trout, Duncan, and South Arm cores is in line with a westward invasion trajectory (Suchy et al., 2010). In Trout and Duncan cores, however, *B. (E.) coregoni* displays an odd pattern of being present in only a few intervals, but in high relative abundances (>20%) – similar to its appearance in North Kootenay. It is possible that *B. (E.) coregoni* has been continually present in low abundances since introduction but can only bloom in favourable conditions. In the Great Lakes, it is associated with meso-eutrophic conditions and filamentous algae or cyanobacterial blooms (Currie et al., 2023). It is also vulnerable to both invertebrate and vertebrate predation, as it occupies a middle size class and has no mucro. Thus, in the sediment cores it might indicate times when zooplanktivory is low and food resources are abundant. In the North Arm, the timing of *B. (E.) coregoni* appearance coincides with the period of high anthropogenic P loading from Cominco and with the initial inundation of Duncan Reservoir, which could have released a high load of nutrients into Kootenay Lake for an anomalously productive growing season.

5.2 Trout Lake: a suitable reference lake?

The complex sedimentation rate in the Trout Lake core suggests the occurrence of short-lived disturbances, which could alter the concentration of cladoceran remains by sediment dilution, or the relative abundance of taxa (e.g., favouring disturbance-tolerant taxa such as *Bosmina* over *Daphnia*; Goulden et al., 1982). It is unclear to what extent cladoceran assemblages are reflecting the oligotrophic state of the lake and changes to climate patterns (as shown by diatom assemblages; Laird et al., 2025) or potential watershed disturbances from historic forestry, mining, erosion activities, or log rafting. However, we lack the information to disentangle the magnitude or effect of these potential disturbances.

Cladoceran concentrations are elevated in the 1970s relative to the 1960s and 1980s (**Figure 4.1**). While logging and other forestry impacts can destabilize soils and increase nutrient loading, the DI-TP in Trout Lake showed no increase at this time (**Figure F1**), and cladoceran assemblages in other BC lakes have shown very minimal assemblage response to forestry practices (Bredersen et al., 2002). Thus, sustained changes to production from forestry impacts are unlikely. The increased cladoceran concentrations could reflect a climatic change. Our climate record suggests increased warming and precipitation at this time, and in 1967, Trout Lake did not freeze, which was recorded as unusual (Acara, 1969). A longer growing season can increase concentrations, and this is consistent with changes seen in the Trout Lake diatoms (Laird et al., 2025). The increase in concentrations post-1990 is also consistent with climate warming and with changes to diatom assemblages which indicate a lengthened growing season (Laird et al., 2025).

It is also possible that Cladocera in Trout Lake were affected by the changes to fish populations in Kootenay Lake and even by the construction of Duncan Dam through its effect on fish spawning. The creation of Duncan Dam altered river discharge and temperature and altered spawner migration timing to the Trout Lake outlet, which could have limited reproduction (Acara, 1969). The correlation between *Daphnia* concentrations and claw length in Trout Lake and the Kootenay Lake kokanee data suggests that some connection, or synchrony, currently exists, and likely did so in the past. Rainbow Trout do move between Kootenay Lake and Trout Lake (Irvine, 1978; Arndt, 2009). Kokanee are also present in Trout Lake, though we are unsure if they move freely between Trout and Kootenay Lake. While overall, trying to interpret these fish dynamics would be too complex for our data, our results do suggest zooplankton in Trout Lake may be affected by changes in connected systems.

This is in opposition to the diatom data which suggest that Trout Lake is a suitable reference site for primary production (Laird et al., 2025).

5.3 Duncan Reservoir: ontogeny and food-web changes

The increase in cladoceran concentrations following Duncan Dam completion is consistent with the trophic surge following reservoir creation (**Figure 4.2**). The inundation of riparian area to form Duncan Reservoir almost certainly resulted in a pulse of nutrients to the system (typical for reservoir creation; Grimard & Jones, 1982), and the 2.5x increase in lentic area (Arndt, 2009) would have created more open-water habitat to support zooplankton production. This upsurge in nutrient concentrations and pelagic production is also documented in diatom concentrations, diatom-inferred total phosphorus, and the P retention model (Laird et al., 2025). The domination of *Bosmina* in newly created Duncan Reservoir is also consistent with inundation. Due to their high growth rate, *Bosmina* are also more disturbance-tolerant than *Daphnia* and can adapt more quickly to increased nutrient loading (Adamczuk, 2016). As such, they may be favoured in conditions of rapid change, such as a reservoir experiencing nutrient pulses following inundation or experiencing annual water level fluctuations from drawdown. Generally, taxa with shorter lifespans and higher reproductive rates are favoured in reservoir environments (Furey et al., 2006).

The cladoceran assemblage changes c. 1980 co-occur with a decrease in DI-TP, though SI-TP and bioavailable P remained elevated until ~1990 (Laird et al., 2025**, Figure D1, D2**). The overall decline in cladoceran concentrations agrees with a decline in nutrient concentrations and subsequently production. Hydrologic changes to Duncan Reservoir c. 1980 might have led to reduced nutrient availability within the system, as minimum reservoir elevation increases by 7 m in 1979 and is higher than any year prior, and minimum elevations continue to be higher through the 1980s (**Figure A2**). Considering that a 30-m drawdown decreases lake area by ~50 km^2 (Cope, 2008), the change in drawdown area between these years (and between decades) could have been quite notable; less area exposed by drawdown would reduce remobilization of nutrients and sediments (Kenney & Walker, 1990). Fall discharge was also much lower through the 1980s than the previous decade (**Figure A3**), which can alter mixing regimes in the fall and subsequently nutrient retention and availability, though unfortunately, the resolution of our data cannot assess seasonality.

The sharp decline in *Daphnia* concentration (relative to *Bosmina*) would be unexpected if the changes to cladoceran assemblages c.1980 were only related to nutrient declines. *Daphnia* are a key component of cladoceran assemblages in low-nutrient lakes (Straile, 2015). While *Daphnia* have higher nutritional requirements than *Bosmina* (Schulz & Sterner, 1999), and could decline if the quantity/quality of food changed, the *Daphnia* concentrations and relative abundances post-1990 in Duncan Reservoir are higher than during the period of nutrient upsurge, despite lower diatom-inferred TP. *Daphnia longispina* are also in high relative abundances in oligotrophic Trout Lake sediments and in Kootenay Lake samples with low DI-TP. Therefore, a reduction in nutrients would not be expected to reduce *Daphnia* abundance.

As a key prey item for fish, *Daphnia* dynamics are often regulated by top-down forcings in oligotrophic lakes (Parker and Schindler, 2006). Their decline could signal a change in higher trophic levels occurring c. 1980. A higher water level may have been favourable for planktivorous fish in Duncan Reservoir (e.g., kokanee) as it could have reduced silt reintroduced during drawdown (Spence & Neufeld, 2000), favouring visual predation by fish. A higher fall water level could reduce dewatering of kokanee eggs, which are often deposited on alluvial fans in Duncan Reservoir and suffer a high mortality rate during winter drawdown (Penfold, 2012), while a higher spring water level might reduce predation pressure on kokanee fry, expanding the area of lentic habitat and consequently reducing the density of predators (Weir & Irvine, 2023). Alternatively, the changes seen c. 1980 could be the result of *Mysis relicta* invasion, as *Mysis* is present in Duncan Reservoir, but the date of its introduction is unknown. *Mysis* are omnivorous and their introductions have been found to alter zooplankton communities by shifting them to a smaller size (Ellis et al., 2011). The recovery of *Daphnia* following trout stocking in the late 1980s (which might have suppressed zooplanktivores) supports the idea that they are responding to top-down pressures.

Changes to the timing of nutrient availability or to the growing season (with a warming climate) could also lead to bottom-up control. Changes to growing season timing can lead to phenological mismatches between grazers and phytoplankton (Winder & Schindler, 2004). Changes to light penetration (from an altered water-level regime) can also cause phenological mismatches, as light penetration can alter the hatching time of overwintering *Daphnia* eggs, or decrease the quality of algal food (Hessen, 2008; Pérez-Martínez et al., 2013).

It is difficult to determine the relative influence of co-occurring stressors in driving the major ecological change seen c. 1980. Further investigation of cladoceran size structure using the processed Duncan Reservoir samples could be helpful in disentangling changes in food web structure. Shifts in the post-abdominal claw length of *Daphnia* can be used to assess changes between different species within the *D. longispina* complex, which cannot be reliably identified by subfossil remains, but may occupy different size ranges. These species may have different temperature preferences and feeding habits which affect their response to environmental changes (Infante & Litt, 1985; Ellis et al., 2011).

5.4 Kootenay Lake history pre-1990

Diatom assemblages show that DI-TP increased beginning with the Corra Linn dam operation c. 1939, which raised water levels and increased organic inputs into the system by flooding shoreline (**Figure D1**; Laird et al., 2025). The dam would have also increased pelagic habitat. Cladoceran concentrations in the North Arm increase at this time (**Figure 4.4**). However, cladoceran concentrations decreased in the South Arm, though the assemblage structure was consistent (**Figure 4.7**). The South Arm was more light-limited (Northcote et al., 2005), which could have reduced secondary production. Alternatively, lower concentrations might reflect a higher sedimentation rate; the post-1930 samples likely incorporate less time than prior intervals, though dating error are higher in samples with lower ^{210}Pb activities.

The North Arm did not show a response to initial *Mysis* invasion, but concentrations declined, especially post-1960 as *Mysis* became more established and as kokanee reached high abundances (Daley et al., 1981). This agrees with a study that found that in 1963, Cladocera were less abundant in the North Arm than the South Arm, and less abundant than they had been in 1949, with *Daphnia* being particularly rare in the North Arm (Zyblut, 1967). This was attributed to high predation pressure from kokanee, which were in higher abundances in the North Arm (Zyblut, 1967). While the North Arm diatoms do also show impacts of Cominco fertilizer loading (Laird et al., 2025), this is not reflected in cladoceran concentrations. Intense predation pressure from kokanee may have depressed Cladocera in the North Arm (Zyblut, 1967), while kokanee would have had the ability to forage in other parts of Kootenay Lake with higher prey availability: Daley et al. (1981) suggest that in the 1970s, kokanee were "getting fat" in the West Arm from mysids washed into shallower water. Declines in *Daphnia* from eutrophication also occur as eutrophication may diminish the nutritional value of algae (Taipale et al., 2019).

In the South Arm, there is a sharp decline in cladoceran size structure with the introduction of *Mysis* in 1949 (**Figure 4.8**). Cladocera likely responded to this increased predation pressure by decreasing their size at reproduction, which relieves planktivory pressure and allows for earlier reproduction (Amundsen et al., 2009), while the pressure imposed by *Mysis* on predatory invertebrates would have released *Bosmina* from some forms of predation, letting them decrease their mucro length. While *Mysis* is also an invertebrate predator, it can eat prey with a large range in size (Viherluoto et al., 2000), so it was likely not efficient for *Bosmina* to increase mucro lengths as a predation defense. Further, *Mysis* was more commonly found on the lake bottom in the North Arm, while in the South Arm it was observed near the surface (as light penetration was much lower; Zyblut, 1967). This could explain our observed size-structure differences (**Figure 4.12**); in the South Arm, *Mysis* could prey directly on Cladocera, while in the North Arm predation may have been limited, or *Mysis* may have preyed on cladoceran resting eggs on the lake bottom (Viitasalo & Viitasalo 2004)

The shift from larger to smaller *Bosmina* in the South Arm could also represent a shift in the relative abundances of larger-bodied *B. (E.) longispina* to smaller-bodied *B. longirostris* (Alric & Perga, 2011). The replacement of *B. (E.) longispina* by *B. longirostris* is considered a typical eutrophication response, largely as eutrophication increases size-selective predation (Stenson, 1976). No decline in cladoceran concentrations occurred at this time in the South Arm, indicating that the size trade-off may have adequately relieved predation pressure, or that the nutrients supplied from the Cominco fertilizer plant in the 1950s helped support Cladocera despite increased predation.

Both cores show a pulse in production coincident with their respective dam constructions. In the North Arm, the sharp decrease in *Bosmina* mucro length in this interval suggests that either invertebrate predators were absent, or food resources were sufficient to relieve predation pressure. While single intervals are admittedly difficult to interpret, the timing of these anomalous intervals in both cores corresponds well to the construction of the Duncan Dam and the Libby Dam respectively. This suggests that the initial inundation of reservoirs could have mobilized nutrients downstream, temporarily increasing productivity. High P loading from the Cominco plant could also increase production, though other core intervals from the 1960s and 1970s (during the time of high P loading) do not show the same elevated cladoceran concentrations. Interestingly, this “pulse” is not recorded in the diatom assemblages (Laird et al., 2025). TP and SRP inputs from the Kootenay River are also not higher in the early 1970s than prior (Binsted & Ashley, 2006). It is possible that if dam construction led to top-down effects, Cladocera could have been released from predation and thereby able to take advantage of high nutrient inputs without any corresponding change to primary production.

The low cladoceran abundances (and specifically *Daphnia*) in the 1980s combined with low measured sizes, suggests that planktivory was high at this time, and higher than pre-1930s conditions (e.g., the historic food-web structure). The decline in *Daphnia* is offset between the North and South Arms, beginning ~1975 in the North Arm and following declines in DI-TP and biovolume of planktonic diatoms (**Figure D3**). In the South Arm, *Daphnia* concentrations decline ~1985. This decrease precedes the decline in DI-TP but aligns with the decline in planktonic biovolume (**Figure D4**). Likely, a decrease in primary production as the Kootenay Lake recovered from cultural eutrophication inhibited the ability of *Daphnia* to persist under artificially heightened predation pressure.

5.5 Artificial fertilization and food-web dynamics

Artificial fertilization began in 1992 to replace what was considered the natural contribution of phosphorus that was retained in upstream reservoirs and to add enough nitrogen to avoid the growth of undesirable algae (Binsted & Ashley, 2006), though the low TP state in the 1980s was largely a result of the closure of the Cominco fertilizer plant. Laird et al. (2025) found that artificial fertilization increased diatom concentrations, initially favouring larger, pennate diatoms which are less edible and tend to occur in meso-eutrophic conditions. In the North Arm, both *Bosmina* and *Daphnia* increased, but the increase in *Bosmina* was proportionately much larger (**Figure 4.4**). In the South Arm, there was an increase in *Bosmina* following the 1992 fertilization of the North Arm (**Figure 4.7**). Binsted and Ashley (2006) found that fertilizer loads to the North Arm impacted the South Arm, and diatoms in the South Arm also indicate nutrient enrichment in 1992 (Laird et al., 2025). The artificial fertilization thus did increase the growth of *Daphnia* (considered desirable), but also benefited other Cladocera.

However, our data suggest that when nutrients were added to the North Arm, the dramatic increases in cladoceran production were quickly regulated, as concentrations and size structure declined in subsequent years, inverse to changes in kokanee populations (**Figure 4.6, 4.9**; **Figure D5**; Warnock et al., 2022). Artificial fertilization has been found to intensify fish planktivory (Hann et al., 1994). In a fertilization experiment of an oligotrophic Norwegian lake, herbivore biomass (mainly Cladocera) increased by 111% with fertilization, but then quickly declined in concentration and shifted toward smaller species (Reinertsen & Langeland, 1982). It was suggested that eutrophication "sped up" the ecosystem processes, leading to excess fish predation that exerts pressure on large zooplankton grazers (Reinertsen & Langeland, 1982). Thus, our data aligns with previous studies suggesting that artificial fertilization of secondary production will be quickly regulated by zooplanktivory.

The South Arm also shows a predation signal in 1992. *Bosmina* mucro length decreases sharply and then shows a gradual increase, which is opposite to the North Arm, where mucro lengths increase sharply and then decline (**Figure 4.12**). In the North Arm, it is likely that artificial fertilization favoured both *Bosmina* and larger zooplankton (*Daphnia*, copepods), which increased predation pressure on *Bosmina* until larger zooplankton were regulated by increases in the kokanee population. The South Arm data suggest that predation pressure on *Bosmina* decreased, perhaps as more food availability helped offset pressure or encouraged more rapid growth.

Through the artificial fertilization period, prior to 2014, *Daphnia* claw sizes are very small, showing us that planktivory was much higher than in pre-1930s or the restoration baseline conditions (**Figure 4.16**). Despite the enhanced predation pressure, *Daphnia* were able to persist, likely from the positive effects of nutrient enrichment. This could have contributed to the all-time high kokanee biomass in 2007-2013 (Warnock et al., 2022), which therefore would have been larger than what the system had naturally supported. Similarly, inferred DI-TP for the artificial fertilization period is higher than pre-1930s data (**Figure D1**), suggesting lower-food-web production in Kootenay Lake was (and continues to be) enriched relative to historic conditions.

Even as weighted *Daphnia* size structure returns to pre-1930s values post-kokanee collapse (**Figure 4.14**), potentially indicating a return to historic predation pressure, the assemblage composition in Kootenay Lake is different. *Bosmina* remain smaller (**Figure 4.15**), perhaps reflecting climate warming (Beaver et al., 2020) or continued pressure by *Mysis*, and *B. (E.) coregoni* is present in the system and capable of blooming. *Daphnia pulex* is also present in the Kootenay Lake cores post-1990, but absent or very rare in the pre-disturbance intervals and in the Trout and Duncan cores. *Daphnia pulex* is a larger daphnid more likely to be selected by planktivorous fish (Leavitt et al., 1994), which is seen here by its increase in numbers following the kokanee collapse c. 2013-2014. *Daphnia pulex* also has a higher food requirement than smaller daphnids (Asaeda & Acharya, 2000). This suggests that artificial fertilization in Kootenay Lake has increased productivity beyond a) historic conditions, and b) more than what would have been expected to occur with climatic warming (using Trout Lake and Duncan Reservoir as references).

5.6 The Water Use Plan

The WUP was officially implemented in 2007 (BC Hydro, 2007) and co-occurred with a decrease in DI-TP in both Duncan Reservoir and Kootenay Lake, though this decrease could also have been from a decline in reservoir P storage and/or a changing climate (Laird et al., 2025). Given the importance of reservoir operations in altering thermal regimes (Ma et al., 2008; Carr et al., 2020) and in discharge regime affecting inflow temperatures and timing of nutrient availability to Kootenay Lake (AMEC & Poisson, 2012), we would expect to see an effect of the WUP on secondary production. The cladoceran assemblages did not show an obvious effect of the WUP in either the Duncan or Kootenay lake cores, potentially as the top-down influence of kokanee predation (in Kootenay) or trout stocking (in Duncan) was a stronger regulator of cladoceran dynamics. However, there was an increase in cladoceran concentrations in Kootenay Lake post-2007, with both Arms showing near-maximum values in 2009. This could be related to kokanee dynamics and warming temperatures, which potentially could interact with the WUP, but not in a way our data can distinguish.

5.7 Recent climate influences

All four cores show an increase in *Daphnia* concentrations post-2015 (**Figure 4.11**). Increased concentrations do not always translate to increased biomass, as they can reflect smaller individuals with higher turnovers, though *Daphnia* populations in Kootenay Lake had a larger size structure (by weighted average claw size) than in preceding years and as large as the pre-1930s samples. In Trout Lake, *D. longispina* claw size is lower than the pre-1950s values for Kootenay Lake (**Figure C6**), so it is conceivable that the increase of *D. longispina* complex in Trout Lake might not represent a large biomass increase relative to what we infer in Kootenay Lake, even if *Daphnia* increase in concentrations in both lakes.

The timing of the *Daphnia* increase aligns with the kokanee spawner collapse and with increases in air temperatures, particularly in the summer and winter (**Figure 2.3**), two factors which are very likely to favour *Daphnia*. A decline in kokanee would reduce planktivory pressure on *Daphnia*. Even in Duncan Reservoir, fish populations may play a role in the *Daphnia* release, as rainbow trout were stocked between 2007 and 2013 (Neufeld & Burrows, 2017), and some bull trout pass from Kootenay Lake through Duncan Reservoir to spawn (Warnock et al., 2022). Monitoring data for the system have linked *Daphnia* abundance to temperature, and *Daphnia* in Kootenay Lake tend to bloom in August and September (Bassett et al., 2024). These two factors are difficult to isolate, however they are likely not independent.

Warmer water temperatures are generally expected to favour smaller cladocerans with lower metabolic requirements (Forster et al., 2012; Beaver et al., 2020). Widespread replacement of *Daphnia* by *Bosmina* has been seen in temperate lakes across Canada (Armstrong & Kurek, 2019), and a lab experiment found that *Daphnia* have lower population sizes when reared at higher temperatures (Tseng et al., 2021). Warming temperatures are also forecasted to reduce the quality of algal food (Arts et al., 2015; Tseng et al., 2011), and may increase predation risk on Cladocera by altering predator behaviour (Riessen, 2015). However, this may be less relevant in systems like Kootenay Lake where a) there is enough high-quality food to support larger zooplankton, and b) predation pressure on large zooplankton is low. The diatom assemblages in the three lakes show a decline in colonial and chain-forming inedible algae and an increase in concentrations of smaller, edible algae (Laird et al., 2025). Warmer water temperatures are likely to reduce the impact of *Mysis* predation on *Daphnia*, since *Mysis* prefer colder, deeper waters and cannot feed as effectively on *Daphnia* occupying the warm epilimnion (Ellis et al., 2011).

In short, the combination of a longer growing season, increased primary production, a decline in fish populations (and possibly in *Mysis* predation) likely support the increase in *Daphnia* in all three lakes. The increase in larger *Daphnia pulex* in Kootenay Lake is likely additionally supported by the maintenance of high-quality algal food by the manipulation of nutrient ratios, despite low overall DI-TP. However, the suppression of *Daphnia* concentration and size in the peak kokanee biomass era indicates it is possible for the top-down effects of predation to override the positive effects of increased primary production from climate and artificial fertilization.

6. Conclusions

Aquatic food webs are complex. Here we use cladoceran subfossils in dated sediment cores to show that Cladocera (including key prey item *Daphnia*) are responsive to nutrient enrichment in the Kootenay Lake system but are heavily regulated by top-down factors (predation). Our results show spatial differences in cladoceran production and predation intensity in Kootenay Lake, consistent with previous data. We also provide evidence that artificial fertilization increased planktivory pressure beyond historic levels, which aligns with other paleolimnological investigations of aquatic fertilization programs. While predation pressure has decreased since the kokanee spawner collapse, cladoceran species composition in Kootenay Lake remains different from both historic and reference lake assemblages, suggesting that both artificial fertilization and climate warming are contributing to novel food-web dynamics in Kootenay Lake.

7. References

Acara, A. H. (1969). Kootenay Lake Rainbow Trout-fry production, fry and spawner survival. B.C. Fish and Wildlife Branch, Misc. Unpublished Manuscript, 35p.

Adamczuk, M. (2016) Past, present, and future roles of small cladoceran *B. longirostris* (O.F. Müller, 1785) in aquatic ecosystems. *Hydrobiologia*, 767, 1-11.

Alric, B., & Perga, M. (2011) Effects of production, sedimentation and taphonomic processes on the composition and size structure of sedimenting cladoceran remains in a large deep subalpine lake: paleo-ecological implications. *Hydrobiologia*, 676, 101-116.

AMEC and Poisson 2012. Lower Duncan River Water Quality Monitoring (DDMMON-7). Year 2 Report. Report Prepared for: BC Hydro, Castlegar. Prepared by: AMEC Environment & Infrastructure Ltd. and Poisson Consulting Ltd. 51 pp. + 4 App.

Amundsen, P., Siwertsson, A., Primicerio, R., & Bøhn, T. (2009) Long-term responses of zooplankton to invasion by a planktivorous fish in a subarctic watercourse. *Freshwater Biology*, 54, 24-34.

Appleby, P. G. (2002) Chronostratigraphic techniques in recent sediments. In Last, W.M., & Smol, J.P. (Eds) Tracking Environmental Change Using Lake Sediments, Volume 1: Basin Analysis, Coring, and Chronological Techniques. Kluwer Academic Publishers, pp. 171-203.

Armstrong, I., & Kurek, J. (2019) Sensitivity and response of low-nutrient lakes to post-twentieth century environmental change in New Brunswick, Canada. *Journal of Paleolimnology*, 61, 85-98.

Armstrong, I., Jones, A., Would, J.A., Graves, E.L., Dysart, A.J., Kurek, J., & Cumming, B.F. (2025) Allometry improves water flea (Crustacea: Cladocera) size-structure interpretations in lake sediments. *Journal of Paleolimnology*.

Arndt, S. (2009) Footprint impacts of BC Hydro dams on kokanee populations in the Columbia River Basin, British Columbia.

Arts, M.T., Hixson, S.M., & Mercieca, S. (2015) Climate warming reduces essential fatty acid production in algae. *Journal of Environmental Indicators*, 9, 7-8.

Asaeda, T., & Acharya, K. (2000) Application of individual growth and population models of *Daphnia pulex* to *Daphnia magna*, *Daphnia galeata* and *Bosmina longirostris*. *Hydrobiologia*, 421, 141-155.

Ashley, K., Thompson, L.C., Lasenby, D.C., McEarchern, L., Smokorowski, K.E., & Sebastian, D. (1997) Restoration of an interior lake ecosystem: the Kootenay Lake fertilization experiment. *Water Quality Research Journal*, 32, 295-324.

Bassett, M., Schindler, E.U., Johner, D., Weir, T., Vidmanic, L., and Ashley, K.I. 2018. Kootenay Lake Nutrient Restoration Program North Arm and South Arm 2014 and 2015 Report. Ministry of Forests, Lands and Natural Resource Operations, Province of British Columbia, Nelson, B.C.

Bassett, M., Weir, T., & Johner, D. (2024) Kootenay Lake North Arm Nutrient Restoration Program 2023-34 (F24) Activity Report. COL-F24-F-3797-DCA. Prepared for Fish & Wildlife Compensation Program (FWCP).

BC Hydro (2007) Duncan Dam Project Water Use Plan. Revised for Acceptance by the Comptroller of Water Rights.

Beaver, J.R., Tausz, C.E., Black, K.M., Bolam, B.A. (2020) Cladoceran body size distributions along temperature and trophic gradients in the conterminous USA. *Journal of Plankton Research*, 42, 1-17.

Binsted, G.A., & Ashley, K.I. (2006) Phosphorus Loading to Kootenay Lake from the Kootenay and Duncan rivers and Experimental Fertilization Program. Report prepared for the British Columbia Conservation Foundation.

Bos, D.G., & Cumming, B.F. (2003) Sedimentary Cladoceran remains and their relationship to nutrients and other limnological variables in 53 lakes from British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences*, 60(10), 1177-1189.

Bos, D.G., Cumming. B.F., & Smol, J.P. (1999) Cladocera and Anostraca from the Interior Plateau of British Columbia, Canada, as paleolimnological indicators of salinity and lake level. *Hydrobiologia*, 392, 129-141.

Brahney, J., Routledge, R., Bos, D.G., & Pellat, M.G. (2011). Changes to the productivity and trophic structure of a sockeye salmon rearing lake in British Columbia. *North American Journal of Fisheries Management*, 30, 433-444.

Brahney, J., Weber, F., Foord, V., Janmaat, J., & Curtis, P.J. (2017) Evidence for a climate-driven hydrologic regime shift in the Canadian Columbia Basin. *Canadian Water Resources Journal*, 42, 179-192.

Bredersen, E.L., Bos, D.G., Laird, K.R., & Cumming, B.F. (2002) A cladoceran-based paleolimnological assessment of the impact of forest harvesting on four lakes from the central interior of British Columbia, Canada. *Journal of Paleolimnology*, 28, 389-402.

Carmack, E.C., & Gray, C.B.J. (1982) Patterns of circulation and nutrient supply in a medium residence-time reservoir Kootenay Lake, British Columbia. *Canadian Water Resources Journal*, 7, 51-70.

Carmack, E.C., Wiegand, R.C., Daley, R.J., Gray, C.B.J., Jasper, S., & Pharo, C.H. (1986) Mechanisms influencing the circulation and distribution of water mass in a medium residence-time lake. *Limnology and Oceanography*, 31, 249-265.

Carr, M.K., Sadeghian, A., Lindenschmidt, K., Rinke, K., & Morales-Marin, L. (2020) Impacts of varying dam outflow elevations on water temperature, dissolved oxygen, and nutrient distributions in a large prairie reservoir. *Journal of Environmental Engingeering Science*, 37, 78-97.

Chen, G., Selbie, D.T., Finney, B.P., Schindler, D.E., Bunting, L., Leavitt, P.R., & Gregory-Eaves, I. (2011) Long-term zooplankton responses to nutrient and consumer subsidies arising from migratory sockeye salmon *Oncorhynchus nerka*, 120, 1317-1326.

Cope, S. (2008). Footprint impacts of BC Hydro dams on burbot populations in the Columbia River Basin, British Columbia. Prepared for Fish and Wildlife Compensation Program Columbia Basin. Prepared by Westslope Fisheries Ltd.

Currie, W.J.S., Bowen, K.L., Niblock, H.A., Fitzpatrick, M.A.J., Rozon, R., Munawar, M., Cuddington, K.M.D., & Ward, C. (2023). Final Status Assessment of BUI 13 Phytoplankton and Zooplankton Populations for the Bay of Quinte Area of Concern. *Canadian Manuscript Report of Fisheries and Aquatic Sciences*, 3261, x + 114 p.

Daley, R.J., Carmack, E.C., Gray, C.B.J., Pharo, C.H., Jasper, S., & Weigand, R.C. (1981). The effects of upstream impoundments on Kootenay Lake, B.C. Canada Inland Waters Directorate, Research Institute, Scientific Series, West Vancouver, British Columbia.

De Melo, R., & Herbert, P.D.N. (1994) A taxonomic reevaluation of North American Bosminidae. *Canadian Journal of Zoology*, 72(10), 1808-1825.

DeWeese, N., Favot, E.J., Branstrator, D.K., Reavie, E.D., Smol, J.P., Engstrom, D.R., et al. (2021) Early presence of *Bythotrephes cederströmii* (Cladocera: Cercopagidae) in lake sediments in North America: evidence or artifact? *Journal of Paleolimnology*, 66, 389-405.

Duncan Dam Water Use Plan Consultative Committee (DDWUPCC) (2005) Consultative committee report: Duncan Dam water use plan. BC Hydro. ISBN 0-7726-5439-5

Ellis, B.K., Stanford, J.A., Goodman, D., Stafford, C.P., Gustafson, D.L., Beauchamp, D.A., et al. (2011) Long-term effects of a trophic cascade in a large lake ecosystem. *Proceedings of the National Academy of Sciences*, 108, 1070-1075.

Ennis, G.L., Northcote, T.G., & Stockner, J.G. (1983) Recent trophic change in Kootenay Lake, British Columbia, as recorded by fossil diatoms. *Canadian Journal of Botany*, 61, 1983-1992.

Forster, K., Hirst, A.G., & Atkinson, D. (2012) Warming-induced reductions in body size are greater in aquatic than terrestrial species. *Ecology*, 109(47), 19310-19314.

Fryer, G. (1968) Evolution and adaptive radiation in the Chydoridae (Crustacea: Cladocera): a study in comparative functional morphology and ecology. *Philosophical Transactions of the Royal Society B*, 254(795), 221-384.

Furey, P.C., Nordin, R.N., & Mazumder, A. (2006) Littoral benthic macroinvertebrates under contrasting drawdown in a reservoir and a natural lake. *Freshwater Science*, 25, 19-31.

Gaugaush, R.F. (1984) Mixing events in Eau Galle Lake. Lake and Reservoir Management 1: 286-291.

Glew, J. R., Smol, J. P., & Last, W. M. (2001). Sediment core collection and extrusion. In W. M. Last, & J. P. Smol (Eds.), Tracking environmental change using lake sediments. Volume 1: Basin analysis, coring, and chronological techniques. Kluwer Academic Publishers.

Goulden, C.E., Henry, L.L., & Tessier, A.J. (1982) Body size, energy reerves, and competitive ability in three species of Cladocera. *Ecology*, 63, 1780-1789.

Grimard, Y., & Jones, H.G. (1982) Trophic upsurge in new reservoirs: a model for total phosphorus concentrations. *Canadian Journal of Fisheries and Aquatic Sciences*, 39(11), 1473-1483.

Hänfling, B., Edwards, F., & Gherardi, F. (2011) Invasive alien Crustacea: dispersal, establishment, impact and control. *BioControl*, 56, 573-595.

Hann, B.J., Leavitt, P.R., & Chang, P.S.S. (1994) Cladocera community response to experimental eutrophication in Lake 227 as recorded in laminated sediments. *Canadian Journal of Fisheries and Aquatic Sciences*, 2312-2321.

Hessen, D.O. (2008) Efficiency, energy and stoichiometry in pelagic food webs; reciprocal roles of food quality and food quantity. *Freshwater Reviews,* 1, 43–57.

Infante, A., & Litt, A.H, (1985) Differences between two species of *Daphnia* in the use of 10 species of algae in Lake Washington. *Limnology and Oceanography*, 30, 1053-1059.

Irvine, J.R. (1978) The Gerrard rainbow trout of Kootenay Lake, British Columbia – a discussion of their life history with management, research and enhancement recommendations. Province of British Columbia Fisheries Management Report No. 72

Jasper, S., Carmack, E.C., Daley, R.J., Gray, C.B.J., Pharo, C.H., & Wiegand, R.C. (1983) Primary productivity in a large, temperate lake with river interflow: Kootenay Lake, British Columbia. *Canadian Journal of Fisheries and Aquatic Science*, 40, 319-327.

Juggins, S. (2023) riojaPlot: stratigraphic diagrams in R, package version (0.1-20). https://github.com/nsj3/riojaPlot

Kennedy, R.H., & Walker, W. W. (1990) Reservoir Nutrient Dynamics. In: Thornton, K.W., Kimmel, B.L., & Forrest, E.P. (Eds.) Reservoir Limnology: Ecological Perspectives. John Wiley & Sons, 256pp.

Korhola, A., & Rautio, M. (2001). Cladocera and other branchiopod crustaceans. In J. P. Smol, H. J. B. Birks, & W. M. Last (Eds.), Tracking environmental change using lake sediments. Volume 4: Zoological indicators. Kluwer Academic Publishers.

Korosi, J.B., & Smol, J.P. (2012a). An illustrated guide to the identification of Cladoceran subfossils from lake sediments in northeastern North America: part 1 – the Daphniidae, Leptodoridae, Bosminidae, Polyphemidae, Holopedidae, Sididae, and Macrothicidae. *Journal of Paleolimnology,* 48, 571-586.

Korosi, J.B., & Smol, J.P. (2012b). An illustrated guide to the identification of Cladoceran subfossils from lake sediments in northeastern North America: part 2 – the Chydoridae. *Journal of Paleolimnology,* 48, 587-622.

Korosi, J.B., Kurek, J., & Smol, J.P. (2013). A review on utilizing Bosmina size structure archived in lake sediments to infer historic shifts in predation regimes. *Journal of Plankton Research,* 35, 444-460.

Kotov, A.A., Ishida, S., Taylor, D.J. (2009) Revision of the genus *Bosmina* Baird, 1845 (Cladocera: Bosminidae), based on evidence from male morphological characters and molecular phylogenies. *Zoological Journal of the Linnean Society*, 156, 1-51.

Kurek, J., Korosi, J.B., Jeziorski, A., & Smol, J.P. (2010) Establishing reliable minimum count sizes for cladoceran subfossils sampled from lake sediments. *Journal of Paleolimnology*, 44, 603-612.

Laird, K.R., Boyle, J., Moyle, M., & Cumming, B.F. (2025) Diatom and sedimentary phosphorus analyses for the Kootenay Lake Nutrient Loading Palaeolimnological Assessment (and the linked NSERC Alliance Grant “New Approaches to Assess Changes in Nutrients and Aquatix Production in Large Lakes Impacted by Dams in Western Canada”). Report prepared for BC Hydro. 77pp + 12 appendices.

LaZerte, S.E., & Albers, S. (2018) weathercan: Download and format weather data from Environment and Climate Change Canada. *The Journal of Open Source Software*, 3(22), 571.

Leavitt P.R., Sanford, P.R., Carpenter, S.R., & Kitchell, J.F. (1994). An annual fossil record of production, planktivory and piscivory during whole-lake manipulations. *Journal of Paleolimnology,* 11, 133-149.

Ma, S., Kassinos, S.C., Kassinos, D.F., & Akylas, E. (2008) Effects of selective water withdrawal schemes on thermal stratification in Kouris Dam in Cyprus. Lakes & Reservoirs 13: 51-61.

Maavara, T., Parsons, C.T., Ridenour, C., Stojanovic, S., Dürr, H.H., Powley, H.R., & Van Cappellen, P.V. (2015) Global phosphorus retention by river damming. *Proceedings of the National Academy of Science*, 112(51), 15603-15608.

Morbey, Y.E., & Ydenberg, R.C. (2003) Timing games in the reproductive phenology of female pacific salmon (*Onchorhynchus* spp.) American Naturalist 161: 284-298.

Mushet, G.R., Laird, K.R., Leavitt, P.R., Maricle, S., Klassen, A., & Cumming, B.F. (2020) Bottom-up forces drive increases in the abundance of large daphnids in four small lakes stocked with Rainbow Trout (*Oncorhynchus mykiss*), Interior British Columbia, Canada. *Ecosystems,* 23, 873-890.

Neufeld, M., & Burrows, J. (2017) Evaluation of the potential impact of Duncan Reservoir Rainbow Trout stocking on Kootenay Lake piscivore abundance. Ministry of Forests, Lands and Natural Resource Operations, Nelson BC.

Nevalainen, L., & Luoto, T.P. (2016) Relationship between cladoceran (Crustacea) functional diversity and lake trophic gradients. *Functional Ecology*, 31, 488-498.

Northcote, T.G. (1972) Kootenay Lake: man’s effects on the salmonid community. *Journal of the Fisheries Research Board of Canada*, 29, 861-865.

Northcote, T.G., Pick, F.R., Fillion, D.B., & Salter, S.P. (2005) Interaction of nutrients and turbidity in the control of phytoplankton in a large western Canadian lake prior to major watershed impoundments. *Lake and Reservoir Management*, 21, 261-276.

Parker, B.R., & Schindler, D.W. (2006) Cascading trophic interactions in an oligotrophic species-poor alpine lake. *Ecosystems*, 9, 157-166.

Patalas, K., & Salki, A. (1973) Crustacean plankton and the eutrophication of lakes in the Okanagan Valley, British Columbia. *Journal of the Fisheries Board of Canada*, 30, 519-542.

Penfold, G.E. (2012) A review of the range of impacts and benefits of the Columbia River treaty on basin communities, the region, and the province. Prepared for: Ministry of Energy, Mines and Natural Gas, Columbia River Treaty Review

Pérez-Martínez, C., Jiménez, L., Moreno, E., & Conde-Porcuna, J.M. (2013) Emergence pattern and hatching cues of *Daphnia pulicaria* (Crustacea, Cladocera) in an alpine lake. *Hydrobiologia*, 707, 47-57.

Persson, J., Tobias, V., & Ravet, J.L. (2007) Food quantity and quality regulation of trophic transfer between primary producers and a keystone grazer (*Daphnia*) in pelagic freshwater food webs. *Oikos*, 116, 1152-1162.

Pharo, C.H., & Chamberlain, V.A. (1979) Kootenay Lake, British Columbia: II, geographic and geological setting. Monograph prepared for Environment Canada.

Posit team (2024). RStudio: Integrated Development Environment for R. Posit Software, PBC, Boston, MA. http://www,posit.co/

R Core Team (2024) R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Reinertsen, H, & Langeland, A. (1982) The effect of a lake fertilization on the stability and material utilization of a limnetic ecosystem. *Ecography*, 5, 1982.

Riessen, H.P. (2015) Water temperature alters predation risk and the adaptive landscape of induced defenses in plankton communities. *Limnology and Oceanography*, 60, 2037-2047.

Rodenhuis, D.R., Bennett, K.E., Werner, A.T., Murdock, T.Q., Bronaugh, D. (2009) Hydro-climatology and future climate impacts in British Columbia. Pacific Climate Impacts Consortium, University of Victoria, Victoria BC, 132pp.

Schindler, E.U., Shafii, B., Anders, P.J., Price, W.J., Holderman, C., Ashley, K.I., & Bassett, M. (2020) Characterizing the phytoplankton and zooplankton communities in Kootenay Lake: a time series analysis of 24 years of nutrient addition. *Canadian Journal of Fish and Aquatic Sciences*, 77, 904-916.

Schulz, K.L., & Sterner, R.W. (1999) Phytoplankton phosphorus limitation and food quality for *Bosmina*. *Limnology and Oceanography*, 44, 1549-1556

Smits, A.P., Litt, A., Cordell, J.R., Kalata, O., & Bollens, S.M. (2013) Non-native freshwater cladoceran *Bosmina coregoni* (Baird, 1857) established on the Pacific coast of North America. *BioInvasions Records*, 2, 281-286.

Spence, C., & Neufeld, M. (2002) Radio telemetry studies of Duncan Reservoir Burbot. Report prepared by the Ministry of Water, Land, and Air Protection, Nelson, B.C., for the Habitat Conservation Trust Fund and the Bonneville Power Administration.

Stahl, K., & Moore, R.D. (2006) Influence of watershed glacier coverage on summer streamflow in British Columbia, Canada. *Water Resources Research*, 42, W06201.

Stenson, J.A.E. (1976) Significance of predator influence on composition of *Bosmina* spp. populations. *Limnology and Oceanography*, 21, 814-822.

Straile, D. (2015) Zooplankton biomass dynamics in oligotrophic versus eutrophic conditions: a test of the PEG model. *Freshwater Biology*, 60, 174-183.

Suchy, K.D., Salki, A., & Hann, B.J. (2010) Investigating the invasion of the nonindigeneous zooplankter, *Eubosmina coregoni*, in Lake Winnipeg, Manitoba, Canada. *Journal of Great Lakes Research*, 36, 159-166.

Szeroczyńska, K., & Sarmaja-Korjonen, K. (2007). Atlas of Subfossil Cladocera from Central and Northern Europe. Friends of the Lower Vistula Society.

Taipale, S.J., Vuorio, K., Aalto, S.L., Peltomaa, E., & Tiirola, M. (2019) Eutrophication reduces the nutritional value of phytoplankton in boreal lakes. *Environmental Research*, 179, 108836.

Thompson, L.C. (2000) Relationship of kokanee salmon (*Oncorhynchus nerka*) abundance to total phosphorus loading, and *Mysis relicta* density in Kootenay Lake, British Columbia. *SIL Proceedings, 1922-2010*, 27, 1545-1550.

Tseng, M., Di Fillipo, C.M., Fung, M., Kim, J.O., Forster, I.P., & Zhou, Y. (2021) Cascading effects of algal warming in a freshwater community. *Functional Ecology*, 25, 920-929.

Utzig, G., & Schmit, D. (2011) Dam footprint impact summary: BC Hydro dams in the Columbia basin. Prepared for: Fish and Wildlife Compensation Program: Columbia Basin, Nelson, BC.

Viherluoto, M., Kuosa, H., Flinkamn, J., & Viitasalo, M. (2000) Food utilization of pelagic mysids, *Mysis mixta* and *M. relicta*, during their growing season in the northern Baltic Sea. *Marine Biology*, 136, 553-559.

Viitasalo, S., & Viitasalo, M. (2004) Predation by the mysid shrimps *Mysis mixta* and *M. relicta* on benthic eggs of *Bosmina longispina maritima* (Cladocera) in the northern Baltic Sea. *Marine Ecology Progress Series*, 281, 155-163.

Warnock, W.G., Thorley, J.L., Arndt, S.K., Weir, T.J., Neufeld, M.D., Burrows, J.A., & Andrusak, G.F. (2022) Kootenay Lake kokanee (*Oncorhynchus nerka*) collapse into a predator pit. *Canadian Journal of Fisheries and Aquatic Sciences*, 79, 234-348.

Weir T., & Irvine, A. (2023) Duncan Reservoir Kokanee Stock Assessment (DDMON-17) – Implementation Year 3 (2018) and 2016 to 2018 Synthesis Report. Prepared for: BC Hydro. 80pp. + 13 App.

Wickham, H. (2016) ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag, New York.

Wickham, H., François, R., Henry, L., Müller, K., & Vaughan, D. (2023) dplyr: A grammar of data manipulation. R package version 1.1.4.

Wilson, S.M., Corsi, M.P., Brandt, D.H., & Stark, E.J. (2021) The response of *Daphnia* to nutrient additions and kokanee abundance in Dworshak Reservoir, Idaho. *Canadian Journal of Fisheries and Aquatic Science*, 78, 1677-1688.

Winder, M., & Schindler, D.E. (2004) Climate change uncouples trophic interactions in an aquatic ecosystem. *Ecology*, 85, 2100-2106.

Zyblut, E.R. (1967) Temporal and spatial changes in distribution and abundance of macro-zooplankton in a large British Columbia lake. MSc thesis, University of British Columbia. 165pp.

Appendix A: Environmental data

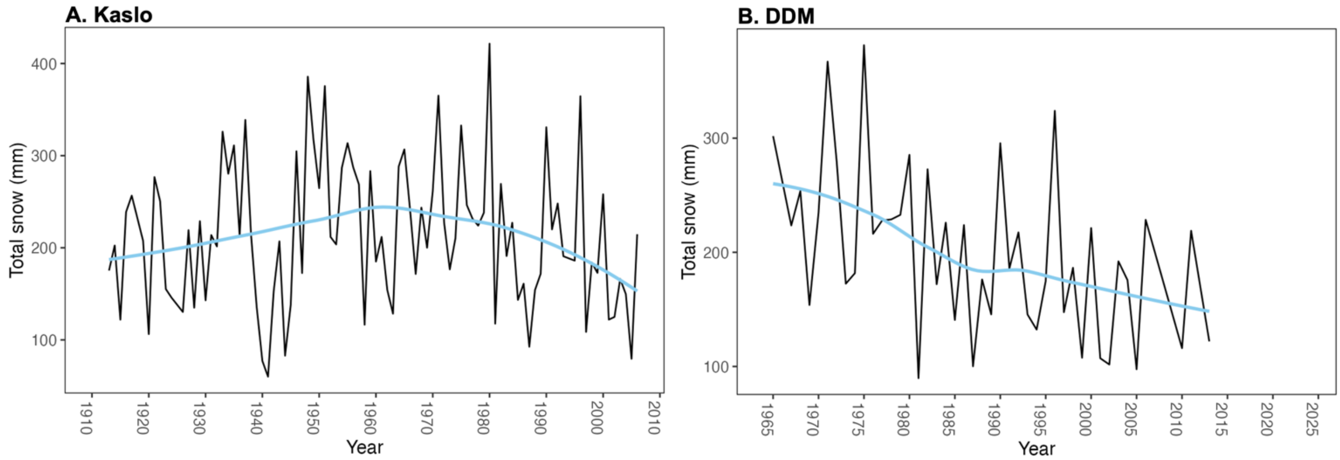

Figure A1. Total annual snow (mm) from the climate station at A) Kaslo (#1124) and B) Duncan Dam (ID #1115). Trends are smoothed with a basic loess function. Data were downloaded from Environment and Climate Change Canada using the function `weather_dl` from the R package weathercan (LaZerte and Albers, 2018).

Figure A2. Monthly mean elevation (m asl) from Duncan Dam for 1968 to 2023. Historical hydrometric data from Environment and Natural Resources Canada.

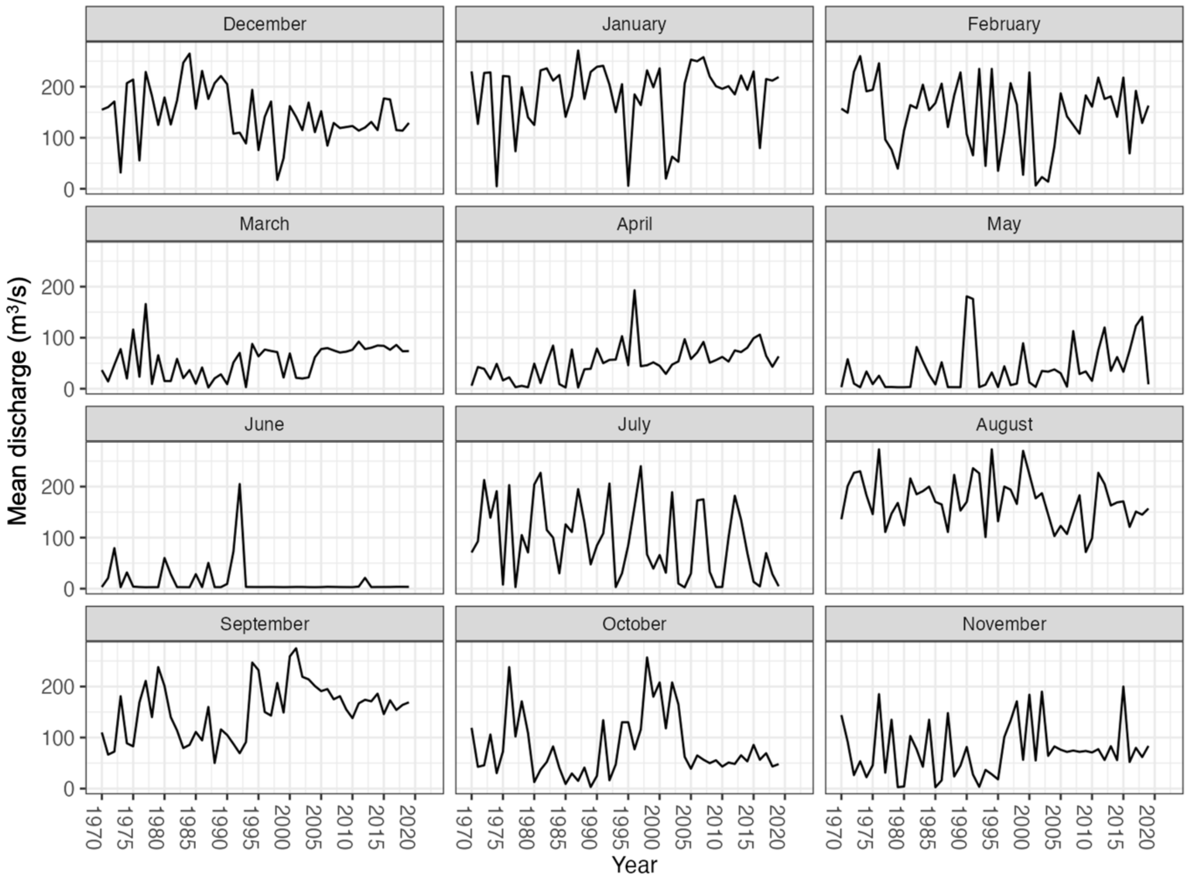

Figure A3. Monthly mean discharge (m^3/s) from Duncan Dam for 1970 to 2019. Historical hydrometric data from Environment and Natural Resources Canada.

Appendix B: Core chronologies

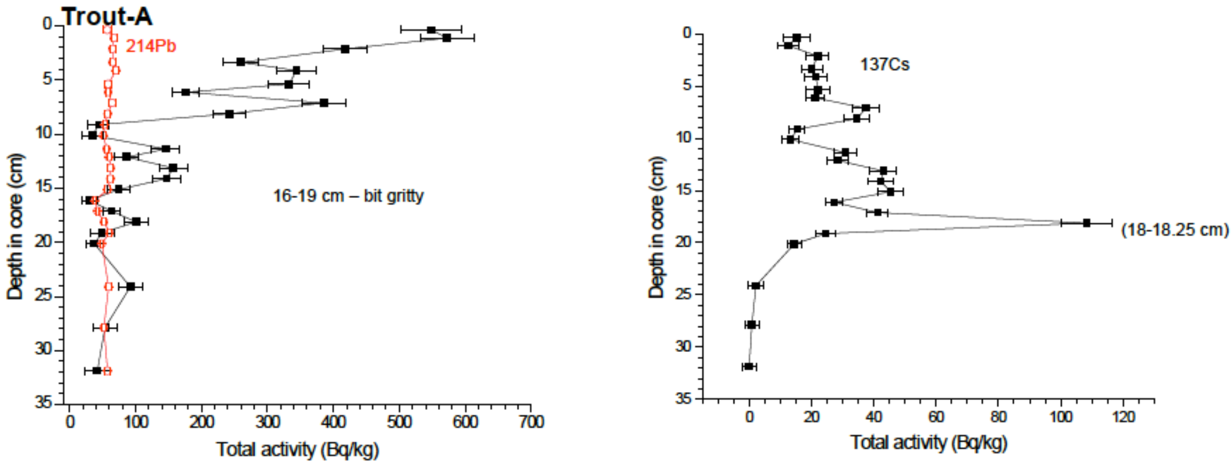

Figure B1. Measured radioisotope activity (Bq/kg) by midpoint depth (cm) for Trout Lake Core A. From Laird et al. (2025).

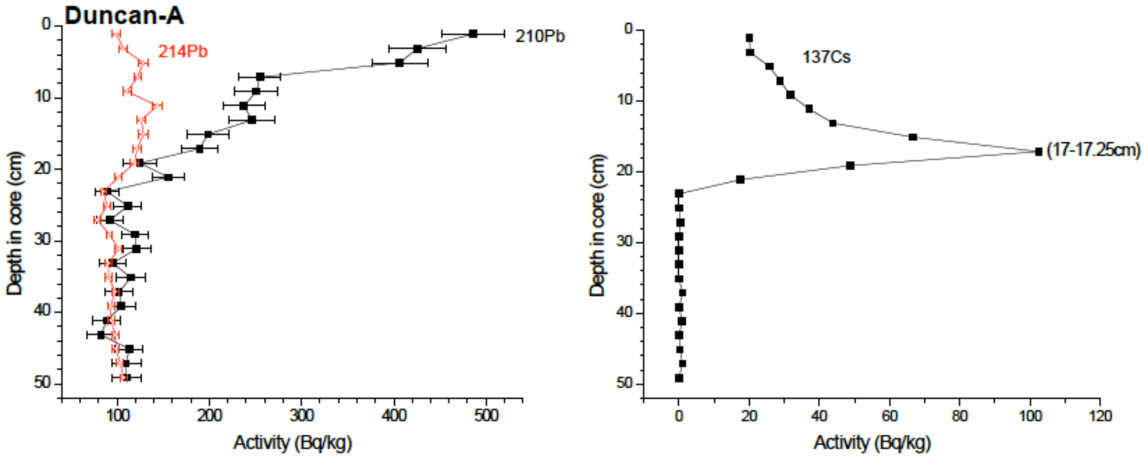

Figure B2. Measured radioisotope activity (Bq/kg) by midpoint depth (cm) for Duncan Reservoir Core A. Taken from Laird et al. (2025).

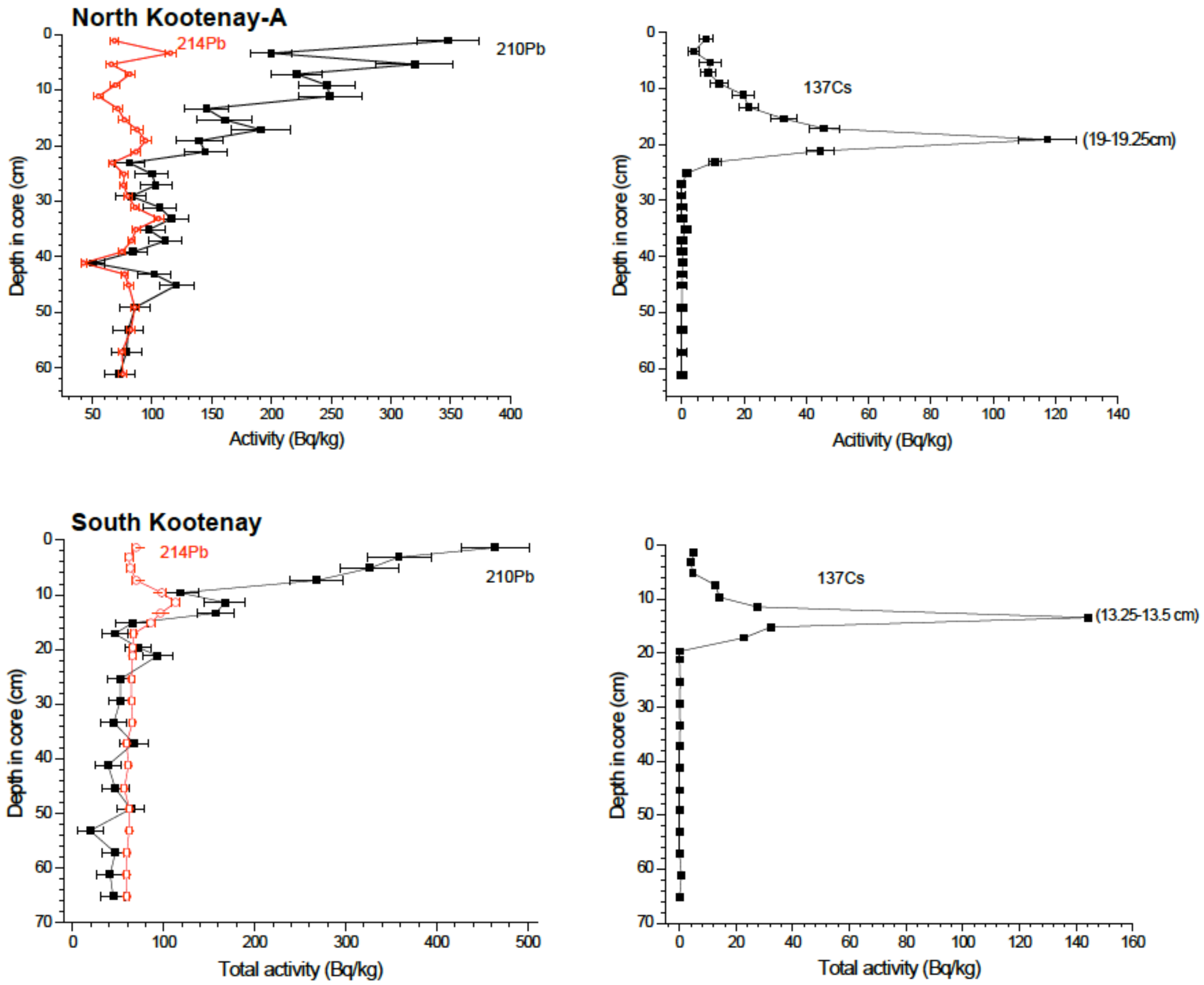

Figure B3. Measured radioisotope activity (Bq/kg) by midpoint depth (cm) for the Kootenay Lake cores (North Arm Core A, South Arm). Taken from Laird et al. (2025).

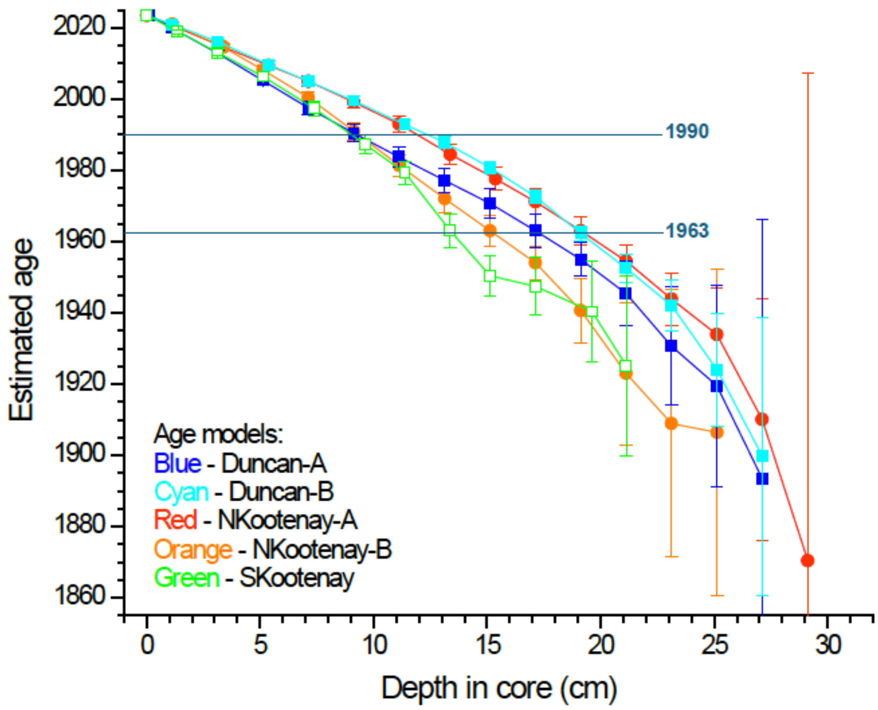

Figure B4. Age-depth models with modelled year (CRS date) by midpoint (cm) for the Duncan and Kootenay sediment cores, with bars showing estimated error. Taken from Laird et al. (2025).

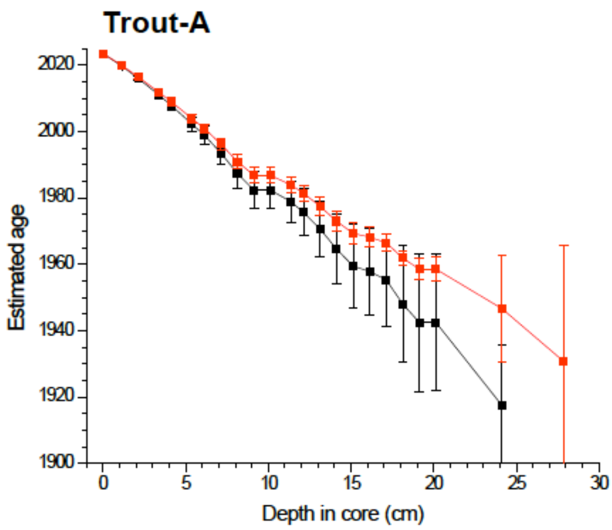

Figure B5. The CRS dating model for Trout Lake Core A showing the model based only on ^{210}Pb decay (black line) and the model fixed to the ^{137}Cs peak (red line, used in study). Taken from Laird et al., 2025).

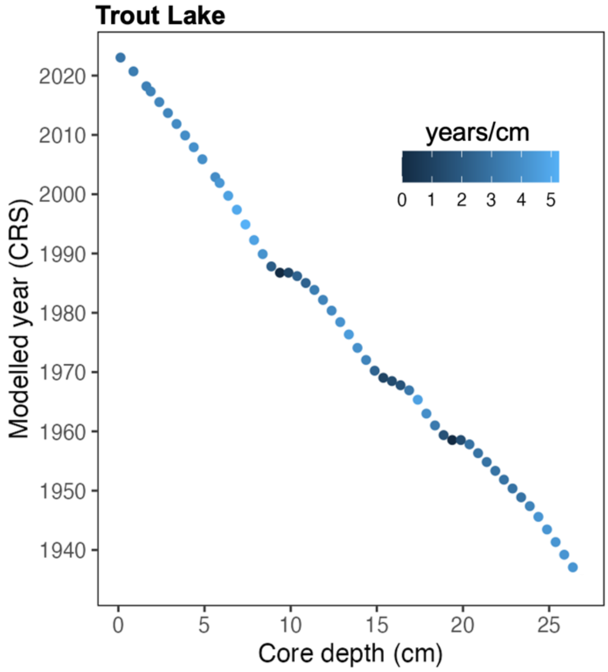

Figure B6. The age-depth relationship for Trout Lake core A based on interpolated ages from CRS-dated sediments. The colour of each point shows the years/cm for the sectioned sediment interval associated with each depth.

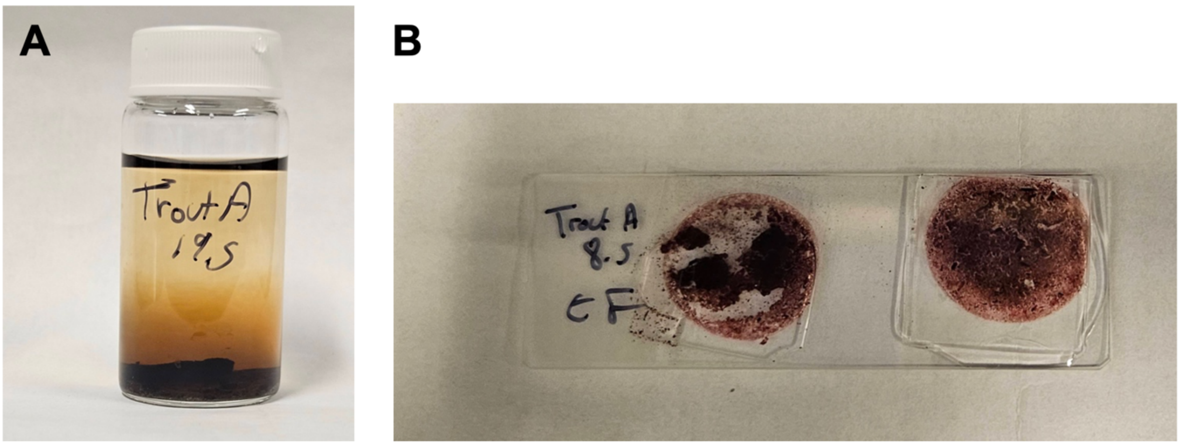

Figure B7. Discoloured cladoceran slurry (A) and failed plated slide (B) for intervals with anomalous amounts of organic debris from Trout Lake core A.

Appendix C: Cladoceran results

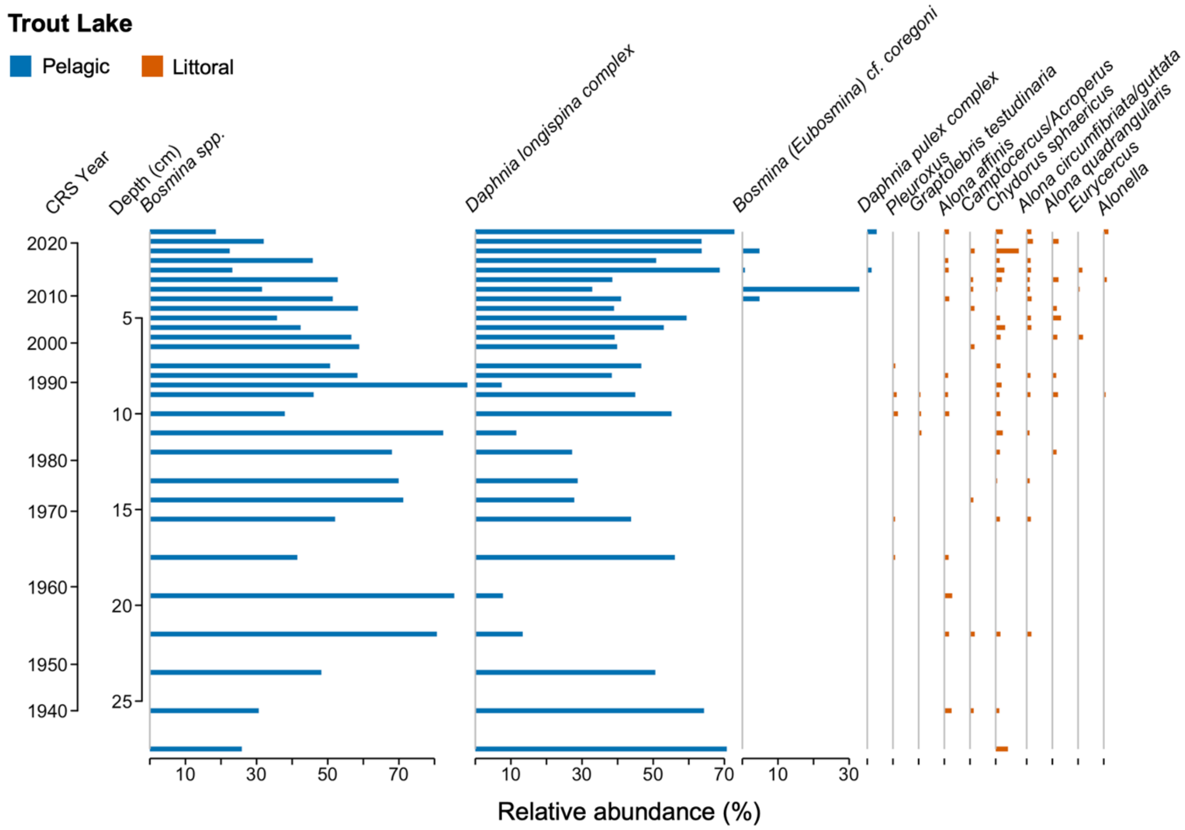

Figure C1. Relative abundance (%) of cladoceran taxa in the Trout Lake sediment core by depth (cm) and CRS year. Pelagic taxa are depicted in blue and littoral taxa in orange. Within each habitat, taxa are sorted left-to-right by direction of change within the core.

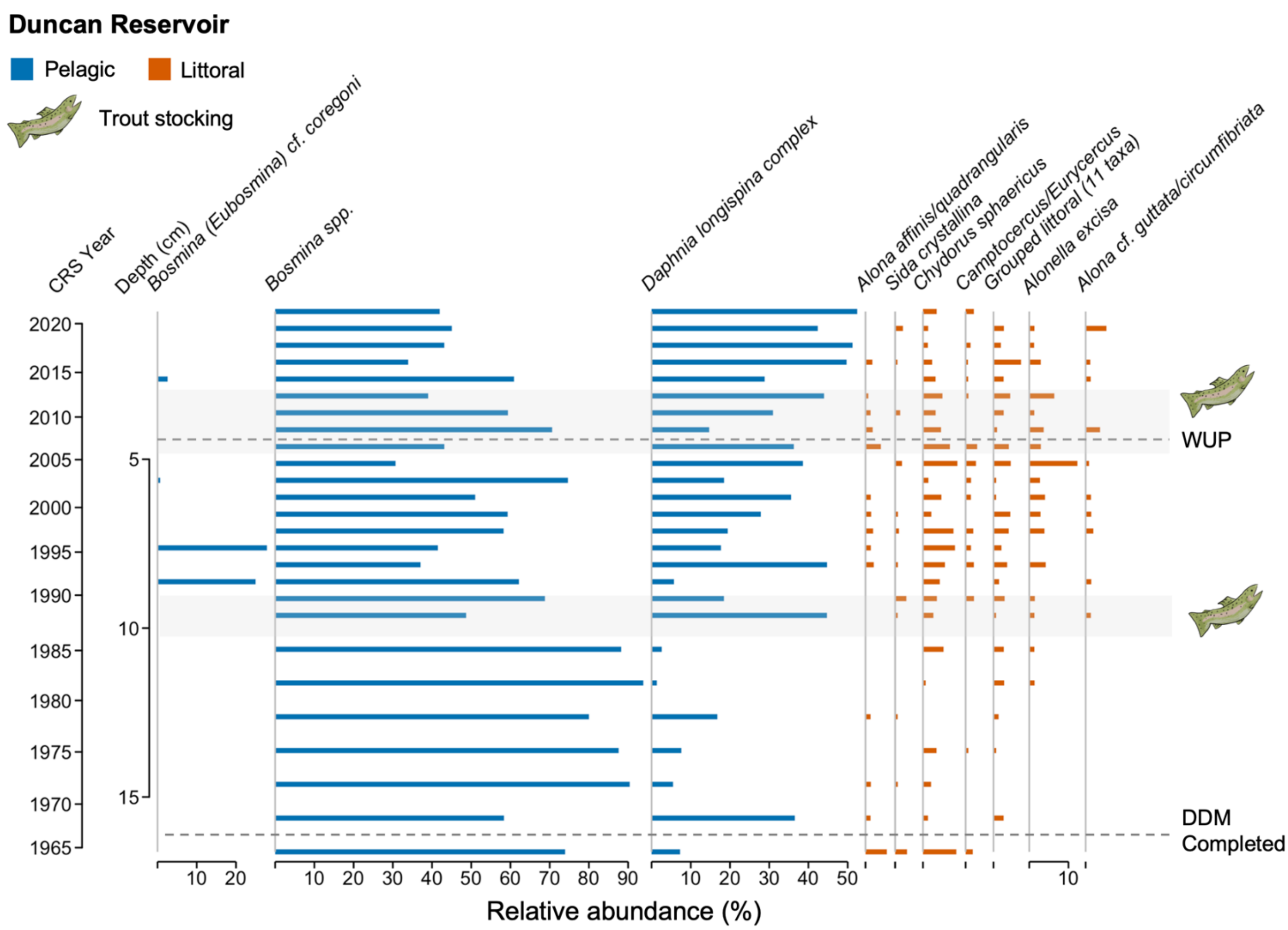

Figure C2. Relative abundance (%) of cladoceran taxa in the Duncan Reservoir sediment core A by CRS-modelled year. Starting values on the x-axis are not shown to reduce clutter, but all axes start at 0. Duncan Dam (DDM) completion and the Water Use Plan (WUP) are shown by dashed lines, and periods of annual rainbow trout stocking are shown by the grey overlay. Taxa are coloured according to primary habitat (pelagic = blue, littoral = orange). Within each habitat, taxa are ordered left-to-right by direction of change within the core.

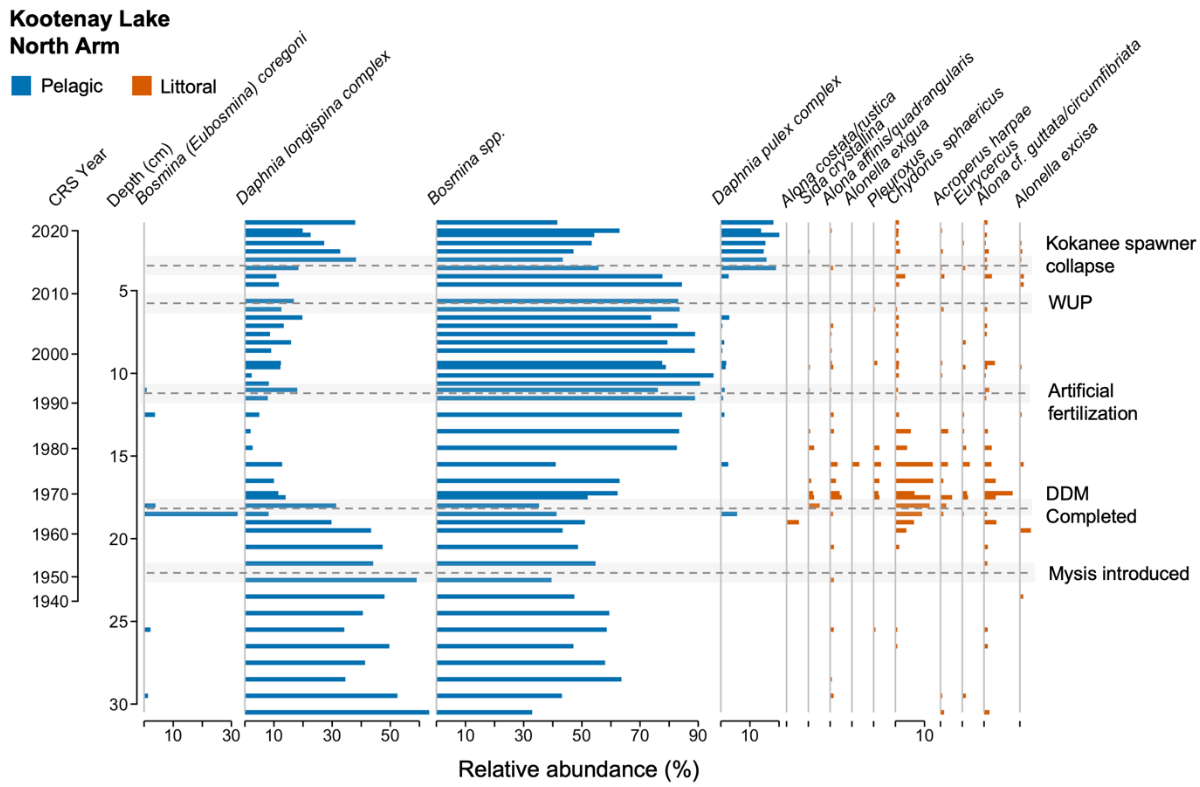

Figure C3. Relative abundance (%) of cladoceran taxa in the Kootenay Lake North Arm sediment core A by CRS-modelled year. Starting values on the x-axis are not shown to reduce clutter, but all axes start at 0. Approximate timing of historic events including Duncan Dam (DDM) completion and the Water Use Plan (WUP) are shown by dashed lines and with grey overlay. Taxa are coloured according to primary habitat (pelagic = blue, littoral = orange). Within each habitat, taxa are ordered left-to-right by direction of change within the core.

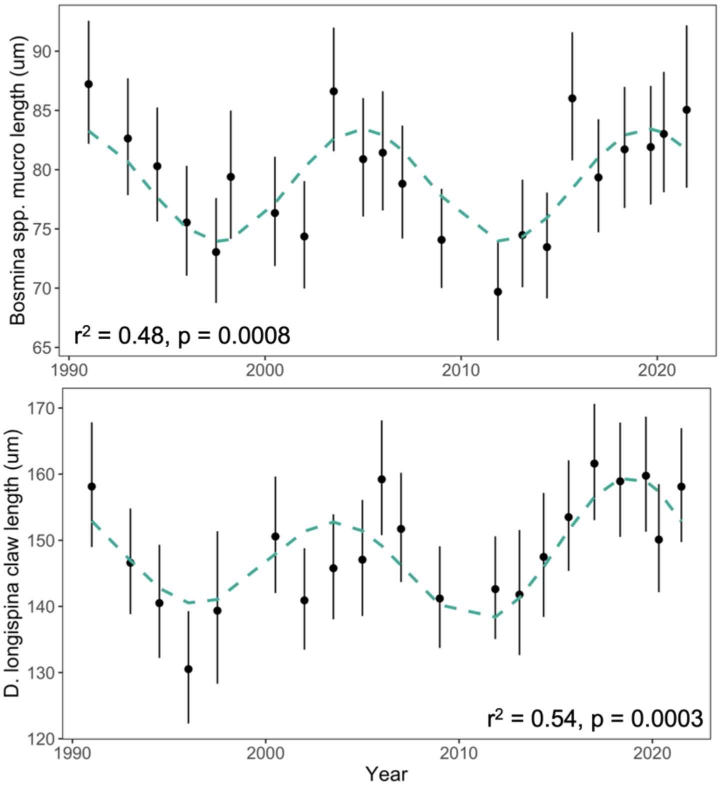

Figure C4. Sinusoidal curves for cladoceran size metrics from Kootenay Lake North Arm core A in the 1990-2023 time period.

Equation C1. Best-fit sinusoidal equation for modelled *Bosmina* mucro length from 1990-2023 for North Arm Core A, where t = year.

$$y \sim \sin\left(\frac{1.1\text{pi}}{8} * t\right) + \cos\left(\frac{1.1\text{pi}}{8} * t\right)$$

Equation C2. Best-fit sinusoidal equation for modelled *Daphnia longispina* claw length from 1990-2023 for North Arm Core A, where t = year.

$$y \sim \sin\left(\frac{0.9\text{pi}}{8} * t\right) + \cos\left(\frac{0.95\text{pi}}{8} * t\right)$$

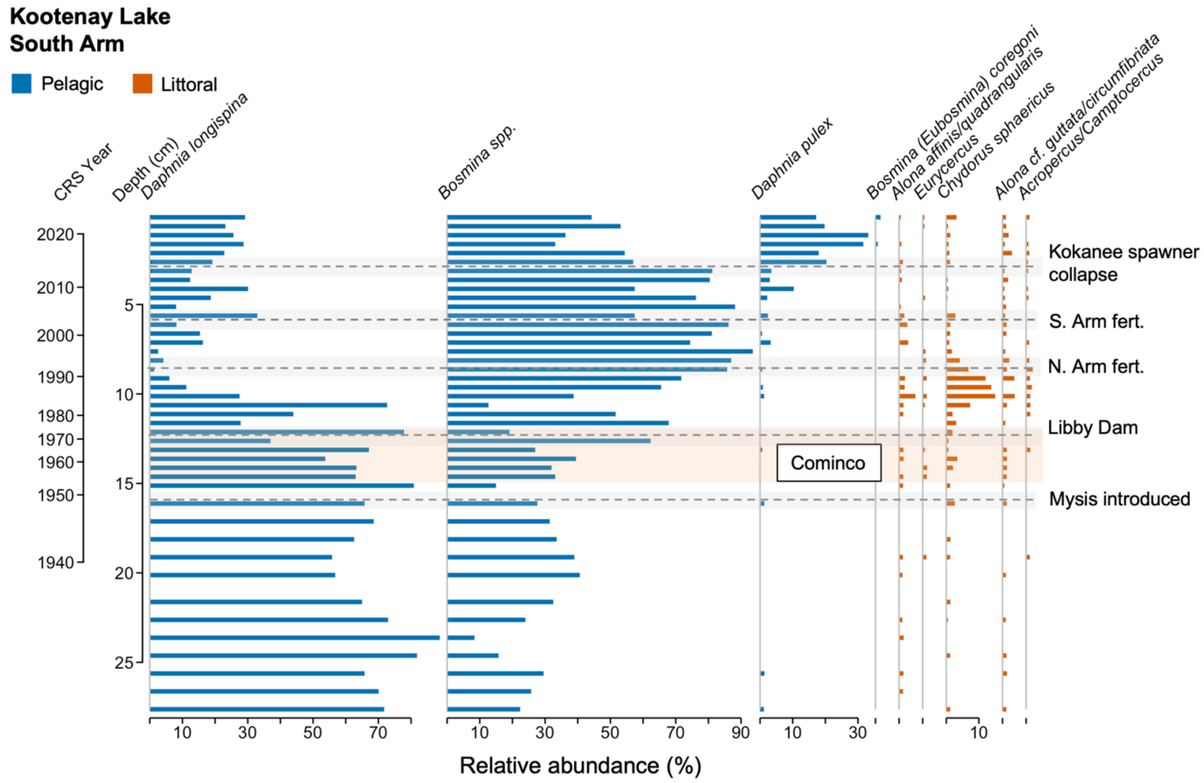

Figure C5. Relative abundance (%) of cladoceran taxa in the Kootenay Lake South Arm sediment core A by CRS-modelled year. Starting values on the x-axis are not shown to reduce clutter, but all axes start at 0. Approximate timing of historic events are shown by dashed lines and with grey overlay, with the time of Cominco operation shown by pink overlay. Taxa are coloured according to primary habitat (pelagic = blue, littoral = orange). Within each habitat, taxa are ordered left-to-right by direction of change within the core.

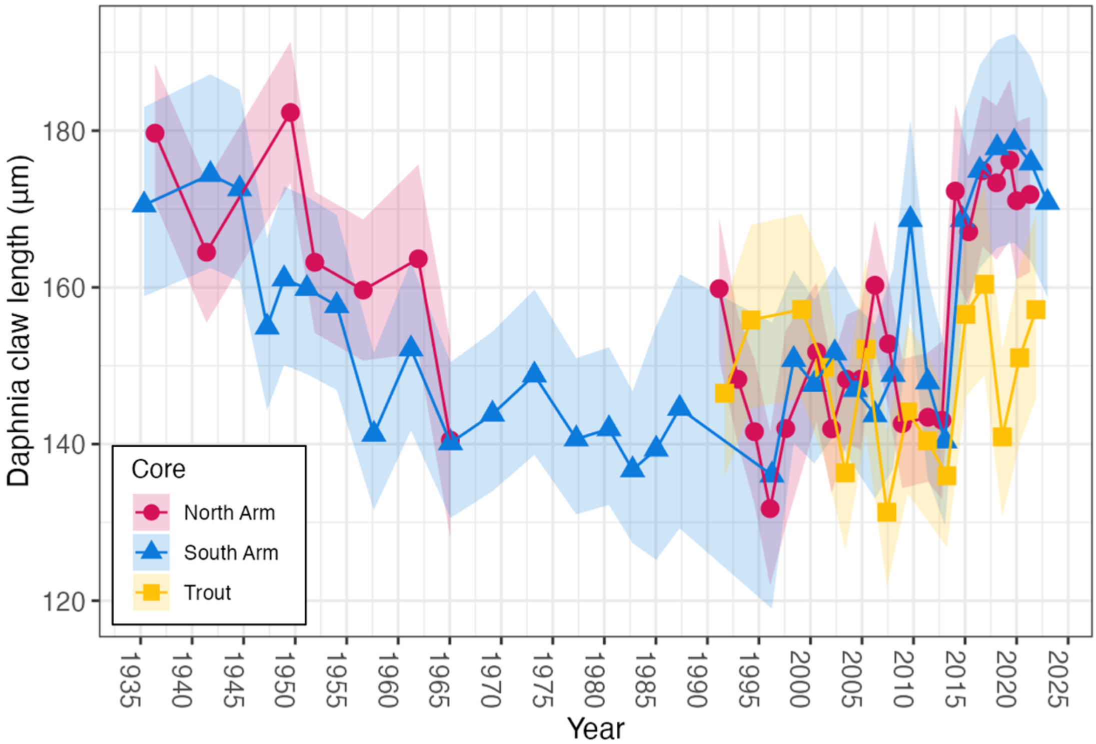

Figure C6. Weighted average Daphnia spp. post-abdominal claw lengths for North and South Arms of Kootenay Lake and for Trout Lake with error shading showing the 95% confidence intervals.

Table C1 Tukey's Honest Significant Difference results showing the difference in means for cladoceran size metrics between time periods. Bolded values indicate when means are significantly different, with asterisks denoting degree of statistical significance (* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$).

	Kootenay Lake North Arm			**Kootenay Lake South Arm**		
Period comparison	*Bosmina* carapace	*Bosmina* mucro	*D. longispina* complex claw	*Bosmina* carapace	*Bosmina* mucro	*D. longispina* complex claw
Baseline vs. historic	-12.4	-0.13	-4.8	**-20.6****	**-9.9*****	**-16.4*****
Artificial fertilization vs. historic	**-50.8*****	**-8.4*****	**-24.3*****	**-36.7*****	**-15.8*****	**-29.6*****
Post-collapse vs. historic	**-59.6*****	**-3.6***	**-12.6*****	**-28.6*****	**-12.3*****	**-18.7*****
Artificial fertilization vs. baseline	**-38.4*****	**-8.3*****	**-19.6*****	**-16.0***	**-5.9****	**-13.2*****
Post-collapse vs. baseline	**-47.2*****	-3.5	**-7.9***	-7.9	-2.4	-2.3
Post-collapse vs. artificial fertilization	-8.8	**4.8*****	**11.7*****	8.12	3.5	**10.9****

Appendix D: Other proxy and biological data

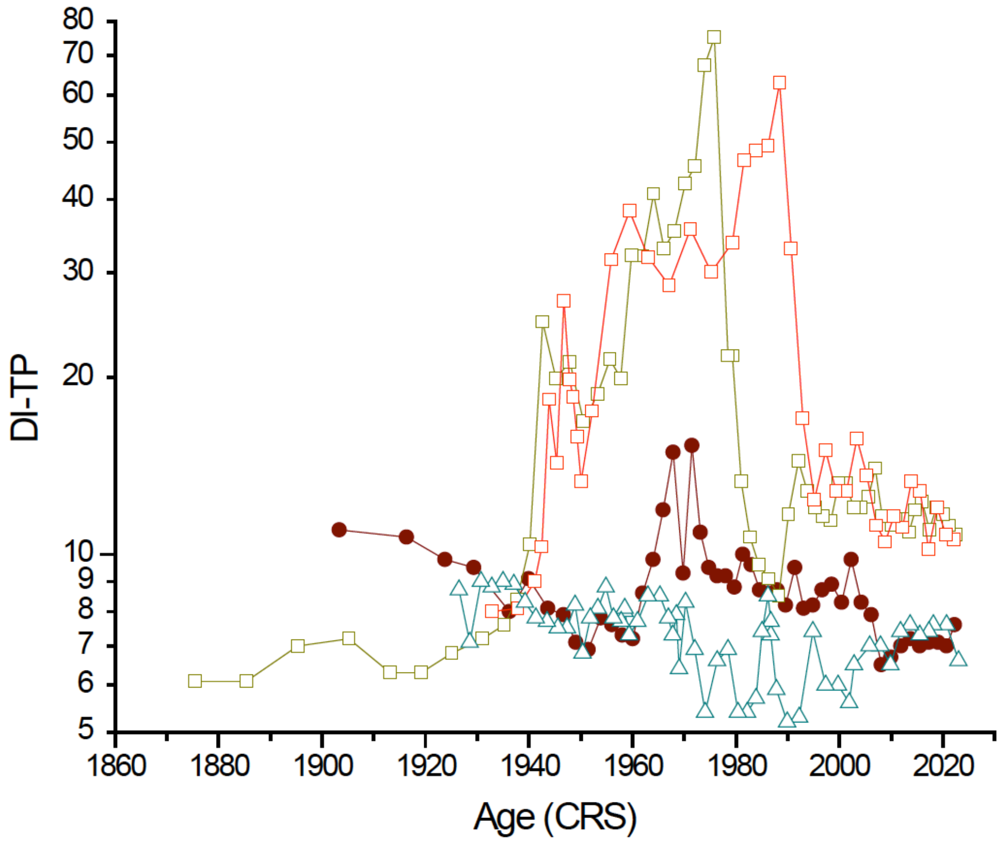

Figure D1. Diatom-inferred total phosphorus (DI-TP, ug/L) by CRS-modelled year for Trout Lake (core A, cyan open triangle), Duncan Reservoir (core A, maroon circle), Kootenay Lake North Arm (core A, dark yellow open square), and Kootenay Lake South Arm (red open square). Taken from Laird et al. (2025).

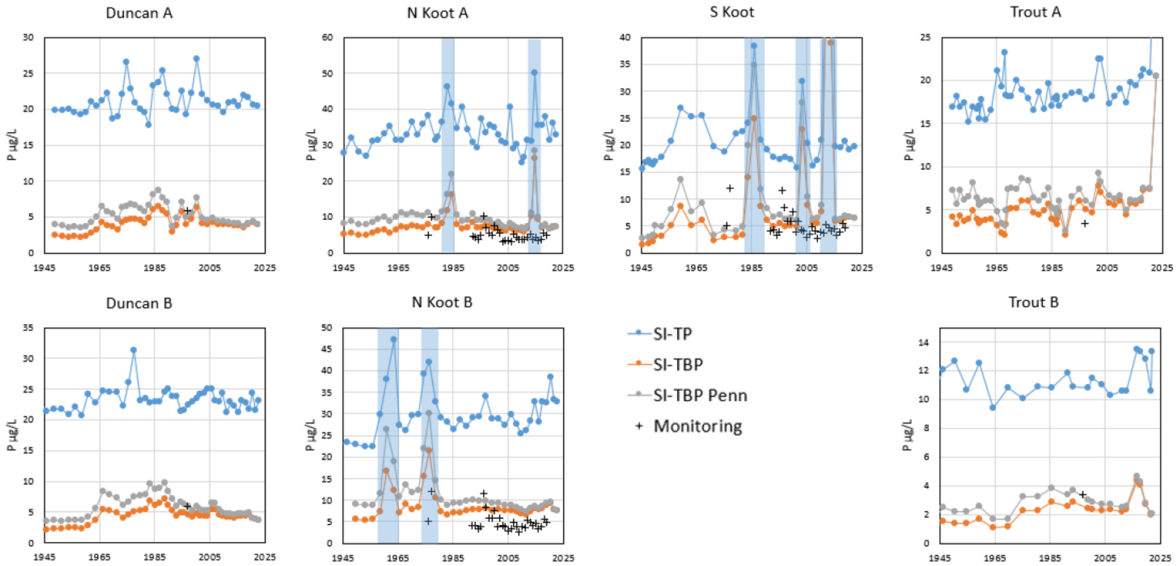

Figure D2. Sediment-inferred lake-water TP (SI-TP), TP inference based on bioavailable forms (SI-TP_{bio} or SI-TBP) and Penn corrected inferred lake-water bioavailable TP (SI-TBP Penn) for all study cores by CRS-modelled year. Horizontal blue bars indicate samples with vivianite present and black crosses show values from monitoring data. Taken from Laird et al. (2025).

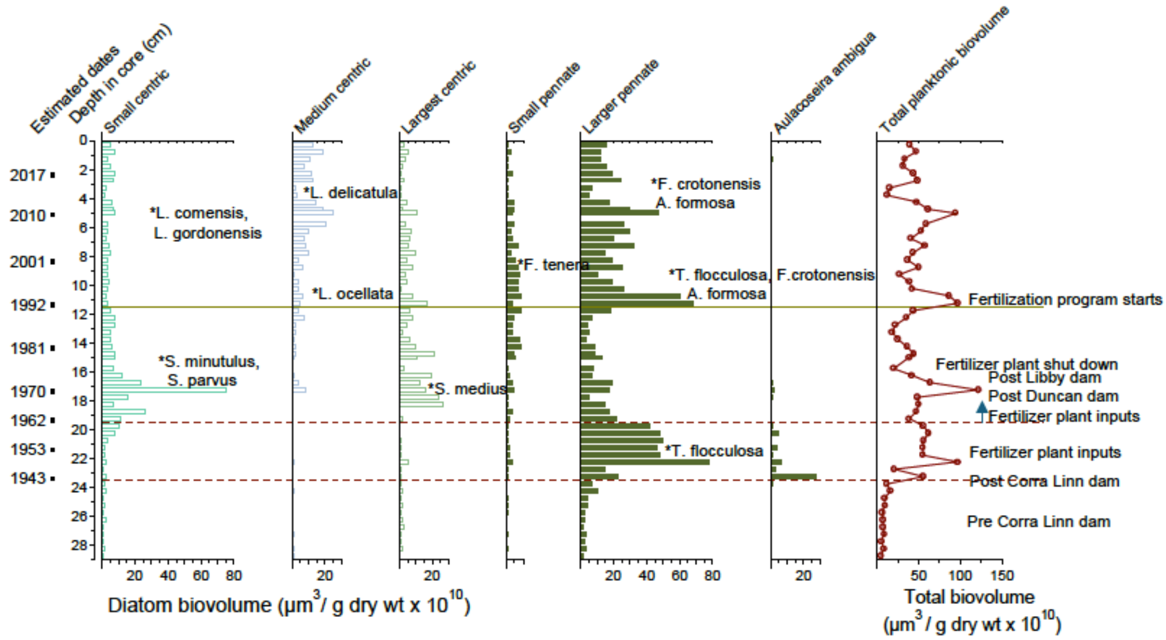

Figure D3. Biovolume of planktonic diatom groups for the Kootenay Lake North Arm (Core A) with labels for the dominant taxa comprising each group in each zone. Historic events are labelled on the right-hand side. Size groups: small centric (10-80 µm^3/cell), medium centric: (100-200 µm^3/cell), large centric (up to 500-600 µm^3/cell), small pennate (65-140 µm^3/cell), large pennate (160-210 µm^3/cell to upwards of 550-660 µm^3/cell). Taken from Laird et al. (2025).

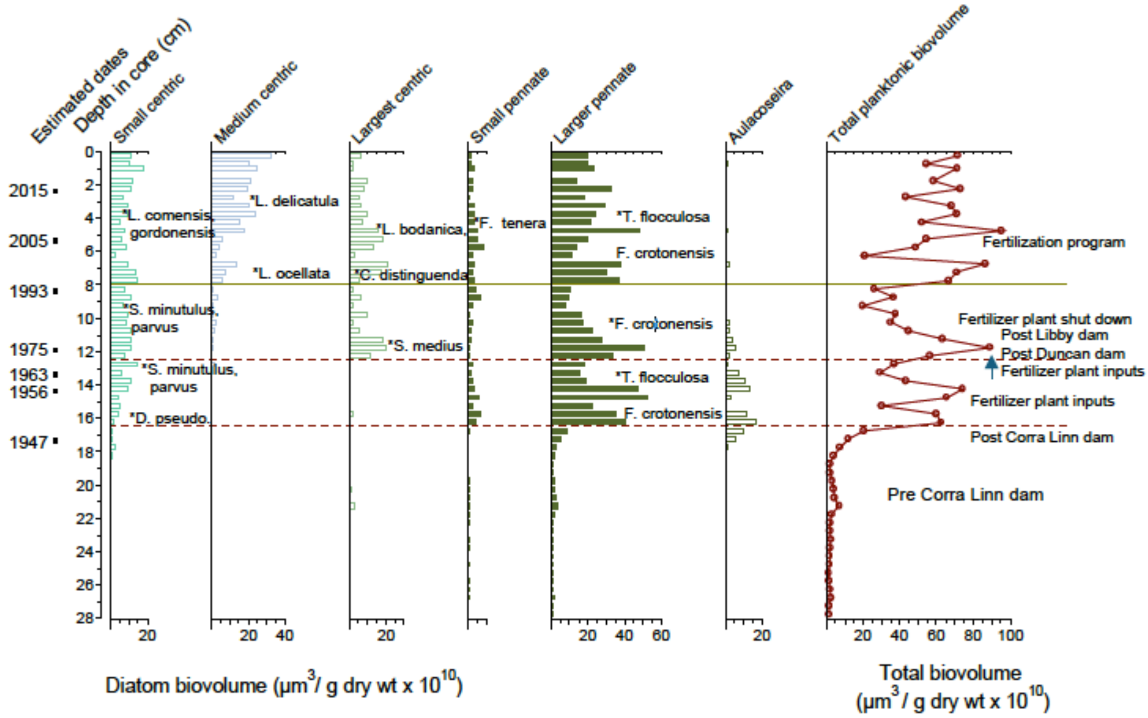

Figure D4. Biovolume of planktonic diatom groups for the Kootenay Lake South Arm with labels for the dominant taxa comprising each group in each zone. Historic events are labelled on the right-hand side. Size groups: small centric (10-80 μm^3/cell), medium centric: (100-200 μm^3/cell), large centric (up to 500-600 μm^3/cell), small pennate (65-140 μm^3/cell), large pennate (160-210 μm^3/cell to upwards of 550-660 μm^3/cell). Taken from Laird et al. (2025).

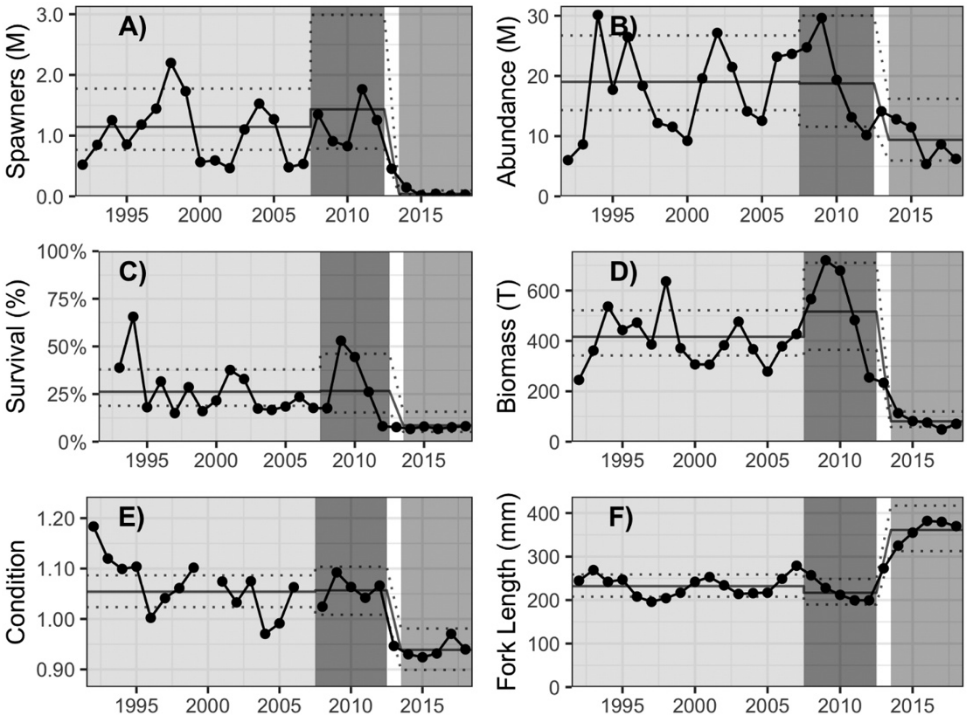

Figure D5. *Figure and figure caption taken directly with permission from Warnock et al. 2022:* Kokanee population metrics for (A) spawner abundance, (B) age 0 abundance, (C) juvenile survival, (D) total biomass, (E) age 1 body condition (Fulton's *K*) and (F) spawner fork length. Shading indicates the historical (light grey), peak (dark grey) and post-collapse (grey) periods. The black dots are the measured annual values, connected by black lines; the grey solid line and grey dotted lines indicate the estimated mean value for the time period with 95% percentile credible intervals, respectively.