University of Alberta

Selenium dynamics in Canadian Rocky Mountain lakes

by

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Abstract

I investigated, water, invertebrates and fishes from lakes in Banff National Park and Kananaskis Country, Alberta for selenium (Se), an element known to be toxic to vertebrates. At some depths, Se concentrations in sediment exceeded recognized thresholds for bird and fish reproductive impairment. Se concentrations in water were over USEPA guidelines after spring melt runoff. In aquatic invertebrates, Se concentrations exceeded values known to cause reproductive impairment in fish and bird predators. Se concentrations in all fish species exceeded known thresholds for reproductive impairment in avian consumers and the majority surpassed concentrations that would negatively affect wildlife and human consumers. Se concentrations in some fish species have significantly increased over the past 6-16 years. The strongest predictors of fish Se concentrations were growth rate, condition factor, age, weight, trophic position (within lakes) and vegetation type (among lakes). These results suggest that consumption advisories are desirable for several lakes in the Banff and Kananaskis area, and that Se concentrations in fish from other area lakes should be investigated.

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1.0 Introduction[1]

Selenium (Se) is an essential trace element at low concentrations which becomes toxic to organisms at higher concentrations (Frost and Lish 1975, Hodson et al. 1980, Lemly 1993a, Wang and Gao 2001). Fishes require 0.1-0.5 mg Se/kg dw (dry weight) from their diet (Lemly 1997). Se becomes poisonous to fishes if their dietary intake exceeds 3 mg Se/kg dw (Lemly 1993b). Se bioaccumulates in organisms and biomagnifies within aquatic food webs, causing increased risk to wildlife (Sappington 2002, Lemly 1999b, Lemly 2004) and human (Barceloux 1999, CCME 2007) consumers.

Elevated concentrations of Se in fishes from three lakes (Lake Minnewanka, Johnson Lake and Bighorn Lake) in southeastern Banff National Park were identified during a 2001 study that surveyed metal concentrations in fishes from Canadian Rocky Mountain lakes (Kelly 2007a, Kelly 2007b). The mean concentrations of Se in trout muscle tissue from Lake Minnewanka, Johnson Lake and Bighorn Lake were 5.78 mg Se/kg dw (1.74 mg Se/kg ww (wet weight)), 8.10 mg Se/kg dw (1.88 mg Se/kg ww) and 14.87 mg Se/kg dw (3.09 mg Se/kg ww), respectively (Kelly 2007a). These concentrations are above those known to affect wildlife and human health.

Sublethal effects for fishes are observed at 7-9 mg Se/kg dw in fish muscle and teratogenesis occurs at concentrations <10 mg Se/kg dw (United States Department of the Interior 1998). Toxic effects of Se have also been observed in aquatic birds. Avian reproductive failure can occur if average prey Se concentrations exceed 2.9 mg Se/kg dw, and mortality of adult birds can occur when concentrations of Se in bird muscle tissue are over 4-8 mg

[1] A review of relevant literature is presented in Chapter 2.

Se/kg dw (United States Department of the Interior 1998). Se concentrations in some fishes from Lake Minnewanka, Johnson Lake and Bighorn Lake are greater than Se concentration in bird muscle that can be lethal. This indicates that birds are likely at great risk because they are top aquatic predators that feed on fish and aquatic invertebrates. Tissue guidelines have been established by the Government of British Columbia to protect aquatic life (invertebrates and fishes) at 1.0 mg Se/kg ww (Government of British Columbia, Ministry of the Environment 2001). Thus, mean Se concentrations in fishes from Lake Minnewanka, Johnson Lake and Bighorn Lake likely put the health of their wildlife consumers at risk.

USEPA guidelines indicate that human health advisories for limited fish consumption by healthy adults and no consumption by children and pregnant women should be issued at 2 mg Se/kg ww (United States Department of the Interior 1998). Mean Se concentrations in fishes from Lake Minnewanka and Johnson Lake are very close to this guideline and Se concentrations in fishes from Bighorn Lake exceed the guideline, indicating that a human health risk exists. Lake Minnewanka and Johnson Lake are very accessible for anglers. Lake Minnewanka is the most fished lake in Banff National Park (Charlie Pacas, personal communication). Although Kelly (2007a, 2007b) identified elevated Se concentrations in fishes from 3 lakes in southeastern Banff, Se dynamics were not the focus of that study, which provided motivation for the research documented in this thesis.

In 2007, I studied Se dynamics in lakes within or near Banff National Park (Chapter 3). I collected a sediment core from Johnson Lake in February 2007 to examine inputs of Se to the lake over time. During the ice-free season (May to October 2007), I collected water and invertebrates monthly from Johnson Lake and Lake Minnewanka, in Banff National Park. These samples were analysed for Se to examine seasonal patterns. In August 2007, I sampled fishes from Lake Minnewanka, Johnson Lake and invertebrates and fishes from 10 other lakes in or near Banff National Park. All invertebrate (including those collected monthly from Johnson Lake and Lake Minnewanka) and fish samples were analysed for Se. This allowed me to determine whether elevated Se concentrations in fishes were more geographically widespread than only Johnson Lake, Lake Minnewanka and Bighorn Lake.

Fishes were also analyzed for stable carbon and nitrogen isotopes. As with other trace metals (Cabana and Rasmussen 1994, Kidd et al. 1995, Camusso et al. 1998), I was able to characterize relationships between Se concentrations in biota and food web structure within lakes. I compared Se concentrations to established guidelines to determine if Se concentrations posed a health risk to fish and their wildlife and human consumers. I then determined which factors (*i.e.,* water chemistry parameters, fish biology and fish metal concentration variables and GIS-generated data for lake and catchment area, geology and climate, etc.) affect Se concentrations in water, invertebrates and fishes within Johnson Lake and Lake Minnewanka and among all study lakes. Finally, I compared current Se concentrations in fishes to Se concentrations in fish specimens collected previously (Kelly 2007a) from Johnson Lake (2001 Brook Trout, Kelly, University of Alberta), Lake Minnewanka (1991 Lake Trout, Donald, Environment Canada) and Spray Lakes (1992 Lake Trout, Donald, Environment Canada) to ascertain whether changes occurred over time. Se concentrations from prior invertebrate collections at Johnson Lake (2000/01 Amphipoda, Ephemeroptera, Trichoptera, Odonata and zooplankton) and Lake Minnewanka (2000/01 Ephemeroptera and zooplankton) were also available for comparison (Kelly, unpublished data).

Significance

Kelly (2007a and 2007b) identified three lakes in southeastern Banff National Park that contained fishes with elevated Se concentrations. My study determined whether other lakes in or near southeastern Banff contained fishes with Se concentrations that could affect the health of wildlife or human consumers. I also detailed Se dynamics in two lakes seasonally, determined which factors influence Se concentrations within and among lakes, and determined whether Se concentrations in biota changed over time. My results will aid Parks Canada, Alberta Environment and Sustainable Resource Development in their effort to protect the vitality of lakes in Canadian Rocky Mountain Parks by providing them with baseline data and information on Se dynamics that can be used for management purposes. These data will be useful to government agencies because many lakes in southeastern Banff National Park and its surrounding area are very accessible to residents and visitors who frequently use the lakes for leisure activities, such as fishing. High Se concentrations in fishes could put the health of human and wildlife consumers at risk. My study was the first detailed assessment of Se dynamics in Rocky Mountain lakes. Previous research on Se in the Canadian Rocky Mountains has focussed solely on rivers and creeks. Thus, my results will complement other research and help to provide a broader understanding of Se dynamics in aquatic ecosystems of the Canadian Rocky Mountains.

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2.0 Review of Relevant Literature

Sources of Se

Se in the environment can be of natural or anthropogenic origin (Table 2.1) Natural sources include rock weathering and biogenic emissions (Table 2.1). Natural concentrations of Se in the earth's crust are estimated to be 0.2 mg/kg (Lemly 1985). Coal in some soils and mineral layers are enriched by a factor of ~65 (Ensminger 1981). This enrichment factor increases to 1250 when the coal is burned to produce electricity (Lemly 1985, Lemly 2004). Many industrial activities contribute to anthropogenic sources of Se (Table 2.1). Atmospheric deposition is an important source of Se contamination (Haygarth et al. 1991, Bennett 1995, Beavington et al. 2004, Cutter and Cutter 2004, Lemly 2004). High Se concentrations, far from anthropogenic emission sources, have been documented in air (Beavington et al. 2004), water (Cutter and Cutter 2004), soil (Beavington et al. 2004), lichens (Bennett 1995) and grasslands (Haygarth et al. 1991).

Se cycling

Gaseous, elemental Se (Se^0) is emitted into the atmosphere from several sources (*i.e.,* combustion of coal) (Anden and Klein 1975) (Table 2.1 and Figure 2.1). Once in the air, Se^0 is chemically transformed into inorganic selenite (Se (IV)) and selenate (Se (VI)) (Bronikowksi et al. 2000, Wen and Carignan 2007) and can be wet or dry deposited to surface waters and soil (Barceloux 1999, Wen and Carignan 2007). Se can also enter aquatic systems by surface (*i.e.,* snowmelt or runoff) or subsurface drainage (Barceloux 1999).

In water, Se can exist in several forms: as selenious (H_2Se0_3) or selenic (H_2Se0_4) acid (Barceloux 1999, Wen and Carignan 2007), soluble Se (IV) and

Se (VI) and elemental Se^0 (due to reduction of Se (IV) with $S0_2$) (Barceloux 1999). Once in an aquatic ecosystem, Se can be absorbed or ingested by animals, bind with particulate matter in surficial sediment, or it can remain free in solution (Lemly 1998, Simmons and Wallschlager 2005). Selenium can be sequestered to the sediment from solution through chemical and microbial reduction of Se (IV) to Se (VI) whereby it is adsorbed onto clay and particulates, reacts with minerals and settles (Lemly 1998). With the deposition of biologically incorporated Se and sedimentation, most Se is located in the top layer of sediment and detritus. However, sediments are only an interim storage compartment as Se can be cycled from sediment into biota (Lemly 1998).

Once in the sediment, several processes occur by which Se becomes available for uptake by aquatic organisms. Plant roots and microorganisms can convert elemental Se (Se^0) and inorganic Se (Se (IV) and Se (VI)) to organic compounds such as selenocysteine, selenomethionine, dimethyl selenide (DMSe) and dimethyl diselenide (DMDSe) (Barceloux 1999). In its organic form of selenomethionine or selenocysteine, Se is available for uptake by aquatic organisms, such as molluscs (Barceloux 1999), benthic invertebrates and detritus-feeding fish and wildlife (Lemly 1998). Inorganic forms of Se in water and food are also taken up by organisms (Bowie et al. 1996), although the organic forms are more bioaccumulative and toxic (Simmons and Wallschlager 2005). In its methylated forms (DMSe and DMDSe), organic Se can bioaccumulate (Jasonsmith et al. 2008), but is mostly released to the atmosphere because of its volatility (Frankenberger and Karlson 1994). The residence time in the atmosphere as DMSe is short (~ 6 hours) (Wen and Carignan 2007).

Se can be readily exchanged between the water column and the sediment because of high sedimentary biological activity and redox potential. Interactions between the sediment and the benthic food web can result in long-term Se contamination of aquatic systems (Simmons and Wallschlager 2005).

Health Effects of Se

Although symptoms of exposure to toxic Se exhibited by humans, wildlife and fishes are variable, exposure pathways are similar. Se exposure begins as excess Se replaces sulphur (S) in the ionic disulfide bonds (S-S) of tertiary protein formation (Lemly 2002). This replacement produces triselenium linkages (Se-Se-Se) or selenotrisulfide linkages (S-Se-S), both of which prevent the formation of disulfide bonds (Lemly 2002). The end result is a malformation of enzymes and proteins (Lemly 2002).

Symptoms of Se poisoning in humans include brittle hair and nails, skin lesions, dental cavities, fatigue, irritability, muscle tenderness, tremor, depression, nervousness, nausea and vomiting (Wilber 1980, Barceloux 1999). There is no conclusive evidence that Se is a carcinogen (Barceloux 1999, CCME 2007), nor a teratogen to humans (Agency for Toxic Substances and Disease Control 1997, CCME 2007). Several studies support an inverse relationship between cancers and elemental Se exposure in humans (Coates et al. 1988, Van't Veer et al. 1990, Hunter et al. 1990).

Wildlife, especially birds, exhibit more severe symptoms than humans when exposed to high concentrations of Se. Maternal exposure to Se results in transfer to eggs and can result in reproductive impairment (Ohlendorf et al. 1986, Skorupa and Ohlendorf 1991), including embryo malformation and mortality (Hoffman et al. 1988). Se can also cause mortality of chicks and adult birds (O'Toole and Raisbeck 1998, Skorupa 1998). Teratogenic effects may occur in birds exposed to elevated Se, including limb reductions, craniofacial and axial skeletal malformations (Franke and Tully 1935, Hoffman et al. 1988, Barceloux 1999), brain defects (*i.e.* hydrocephaly) and organ damage (Hoffman et al. 1988).

Fishes also exhibit serious symptoms when exposed to high concentrations of Se. These involve swelling of the gill lamellae, elevated lymphocytes, anemia, corneal cataracts, bulging of the eye, and pathological alterations of the liver, kidney, heart and ovary (Lemly 1997, Lemly 2002). Teratogenic deformities can occur in the spine, head, mouth and fins (Lemly 1997, Lemly 2002, Holm et al. 2005). Reproductive failure can be caused by embryo mortality (Lemly 1997, Holm et al. 2005), reduced production of viable eggs due to ovarian deformities and post-hatch mortality due to the bioaccumulation of Se in eggs (Lemly 2002).

Factors that affect Se accumulation in fishes

Several factors can influence Se accumulation in fishes. These include: sources of Se to a catchment or lake, factors that affect transport of Se within a catchment to a lake and factors that affect bioaccumulation and biomagnification of Se within a lake.

Factors that affect sources of Se to a catchment or lake

Catchment geology can be an important source of Se to lakes, through the weathering of rocks (Presser et al. 1994). Cretaceous marine sedimentary rock (Presser et al. 1994), black shales (Wang and Gao 2001, Wen and Qiu 2002), carbonate sandstone and slate (Wang and Gao 2001) have all been identified as Se-rich rock. In parent bedrock, Se exists in the form of selenites and selenides in association with sulphide minerals (Barceloux 1999). Erosion of Se-bearing rock formations into river sediments is a major geologic source of Se in the lower Colorado River (Presser et al. 1994). In the Western United States, agricultural drainage through Se-bearing rock is an important source of Se to wetlands (Presser et al. 1994). The Cascade Thrust formation has been identified as a potential natural geologic source of Se to the Bow River south of Lake Louise, Alberta (Glozier et al. 2004).

Se is subject to atmospheric transport (Haygarth et al. 1991, Bennett 1995, Beavington et al. 2004, Cutter and Cutter 2004), and atmospheric deposition can be a significant source of Se (Barceloux 1999, Wen and Carignan 2007). Therefore, the deposition of Se to an aquatic system is likely affected by climatic variables such as wind direction and speed, and seasonal variations in temperature and precipitation (Wen and Carignan 2007).

Factors that affect transport of Se within a catchment to a lake

Contaminant transport in a catchment is affected by factors such as aspect, slope, soil type, vegetation type, litter, leaf or needle inputs, lake area, presence of wetlands, catchment area, catchment volume and catchment area to lake area ratio (as reviewed by Kelly 2007b). Intuitively, transport of Se, would likely be influenced by these same factors, although little is documented in the literature. Sappington (2002) has shown that hydrological events (*i.e.* rainfall and snowmelt) can increase the quantity of Se entering a lake from its catchment. Increases in catchment disturbance also affect the transport of contaminants to lakes and streams (Foster and Charlesworth 1996).

Factors that affect bioaccumulation and biomagnification

The biogeochemical cycling and accumulation of Se in water is affected by rates of biotic and abiotic Se transformations, redox conditions in sediment, food web structure (Sappington 2002) and species-specific differences in Se uptake and excretion (Sappington 2002, Rainbow 2002, Wang 2002). Also important are the sediments, not only because of the bioaccumulation of Se through the benthic food web, but also due to microbial reduction, which transforms Se to a form (*i.e.* selenate, selenite, organic selenides) that is biologically available to aquatic organisms (Bowie et al. 1996). Inundation of terrestrial soils with water to create reservoirs can decrease the availability of Se, through reduction (under anoxic conditions) to DMSe which is released to the atmosphere (Mailman et al. 2006). Many reservoirs are located in and near southeastern Banff National Park, and several are represented in this study.

The presence of other elements can increase or decrease Se bioaccumulation in aquatic organisms (Hamilton and Palace 2001). Selenium interacts synergistically, additively or antagonistically with other trace metals (*i.e.,* antimony (Sb), arsenic (As), cadmium (Cd), cobalt (Co), copper (Cu), germanium (Ge), lead (Pb), silver (Ag), tellurium (Te) and tungsten (W)) in fishes, birds and mammals (reviewed by Hamilton 2004).

The antagonistic relationship between Se and mercury (Hg) has been widely studied as a potential means to mitigate both atmospheric and point-source Hg contamination in aquatic systems (Turner and Rudd 1983, Klaverkamp et al. 1983, Mailman et al. 2006). The addition of Se to a food web contaminated with Hg resulted in fishes bioaccumulating less Hg (which is more toxic than Se) (Turner and Rudd, 1983, Turner and Swick 1983). Se has also been shown to reduce the methylation rate of Hg to toxic methylmercury (MeHg) in sediments (Jin et al. 1997).

Parks Canada, in consultation with Health Canada, issued a fish consumption advisory and precautionary consumption advice for Hg in fishes from all Canadian Rocky Mountain Park waterbodies (Parks Canada 2005) based on data from Kelly (2007b). The antagonistic relationship that exists between Se and Hg may influence the concentrations of these metals in fishes from Canadian Rocky Mountain lakes (Kelly 2007b).

Se in mountain lakes

Trace metals can exhibit similar patterns of spatial distribution in mountain ecosystems (Šoltés 1992 and 1998, Kock et al. 1995, Zechmeister 1995, reviewed in Kelly 2007b). Fishes from high altitude lakes contain elevated concentrations of trace metals, including Cd and Pb (Kock et al. 1995). Trace metal concentrations (Pb, Cd, Cu, Zn, Cr, Fe, Mn, Mo and S) in moss from European mountains are known to increase with elevation (Šoltés 1992 and 1998, Zechmeister 1995, reviewed by Kelly 2007b). Cadmium concentrations in amphipods from Canadian Rocky Mountain lakes also increase with altitude (Varty, unpublished data). It seems likely that Se concentrations in a variety of biota would be elevated in mountainous areas and increase with elevation.

Geographic and topographic characteristics can result in enhanced atmospheric deposition of contaminants to mountainous areas. Increased wind speed, cloud immersion time and precipitation volume in mountainous areas can enhance metal deposition rates (Petty and Lindberg, Battarbee et al. 2009). These factors would likely impact my study area which is located in the Canadian Rocky Mountains.

The type of precipitation can also affect the quantity of contaminants deposited to mountainous regions. The majority of precipitation to mountains falls as snow (Yang et al. 2007), which scavenges more contaminants from the air than rain (Moldovan et al. 2007). Therefore, snow-bound metals can be transferred to aquatic systems in snowmelt runoff (Mast et al. 2005, Moldovan et al. 2007)

Many factors other than rates of atmospheric deposition can influence contaminant concentrations in biota from mountain aquatic systems. Catchment characteristics, such as vegetation and soil type, also change with altitude which can influence water chemistry parameters (*e.g.,* pH, alkalinity, conductivity, total phosphorus and total nitrogen) (Larson et al. 1999). Lake productivity is lower at higher elevations (Schindler 1990), which favours higher concentrations of contaminants in organisms (Kidd et al. 1999, Essington and Houser 2003, Kamman et al. 2005, reviewed by Kelly 2007b).

Several studies have identified aquatic biota with high Se concentrations in mountainous areas. In two lakes in the National Parks of the Patagonian Andean Range in Argentina, Se concentrations in fish muscle ranged from 0.8 to 1.5 mg/kg dw (Arribere et al. 2008). In the San Joaquin Valley, (located between the Sierra Nevada and Coast Range Mountains in California, USA) Se concentrations in water, sediment, zooplankton, invertebrates and mosquitofish from reference sites did not exceed published toxicity guidelines (Saiki and Lowe 1987). However, the Se concentration of some fishes from mountain lakes in western US national parks were elevated (WACAP 2009).

In Canada, Se concentrations of invertebrates and salmon fry from rivers and streams in the Cascade Mountains of British Columbia caused the calculated tolerable daily intake (TDI) of the predatory American dipper (*Cinclus mexicanus*) to be surpassed (Morrissey et al. 2005). In the Rocky Mountains of Alberta, Se concentrations in aquatic macroinvertebrates (Wayland and Crosley 2006) and bull trout (*Salvelinus confluentus*) (Palace et al. 2004) collected from impacted and reference waterbodies exceeded established toxicity thresholds. Previous Se research in the Canadian Rockies focussed on the environmental impacts of coal mining (Wayland and Crosley 2006, Palace et al. 2004). Results indicated that mining activities can exacerbate already elevated concentrations of Se in water, sediment and biota (Wayland and Crosley 2006, Palace et al. 2004).

Climate Change and Se

Climate change has caused alterations to the physical characteristics of Rocky Mountain lakes (Hauer et al. 1997). Increased air temperatures result in increases in water temperature (Schindler 1997), which affect seasonal heat budgets and the timing and onset of lake stratification and ice-breakup (Schindler 1997, Hauer et al. 1997). Climate warming may potentially result in alterations to river flow pattern, hydrology, biogeochemical cycles and fluxes (Hauer et al. 1997), water chemistry (*i.e.* phosphorus, dissolved organic carbon (DOC), sulphates, etc.) (Schindler 1997), and increased lake productivity (Murdoch et al. 2000). These changes could lead to modifications in the community composition of aquatic organisms (Schindler 1997, Parker et al. 2008). Decreases in lake flushing rates and higher evaporation rates could cause increases in concentrations of atmospherically transported contaminants within lakes (Moore et al. 1997). These climate change-induced alterations to mountain lake ecosystems will likely affect factors that influence Se concentrations in water and biota from Rocky Mountain lakes.

Reservoirs and Se

Both lakes and reservoirs can act as sinks for catchment derived and atmospherically deposited particulates (Shotbolt et al. 2005). However, some characteristics of reservoirs differ from those of lakes. Water levels in reservoirs regularly fluctuate, resulting in sediment disturbance (Shotbolt et al. 2006). Reservoirs also have higher sedimentation rates than lakes and these rates can be better understood with records of the drawdown and filling of the reservoir (Shotbolt et al. 2006). Seasonal drawdown and refilling cycles in reservoirs can create conditions that are favorable for the methylation of metals such as mercury (Hg) (Ambers and Hygelund 2001). The seasonal flooding (*i.e.,* reservoir filling) of Se-rich soils could also result in the precipitation of Se salts that could be transported in surface runoff to the reservoir from the catchment (Rainbow and Luoma 2008).

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Table 2.1 Sources of selenium to the environment

Sources	Reference
Natural	
Crustal weathering-cretaceous marine shales	Mosher and Duce 1987, Nriagu and Pacyna 1988, Nriagu 1989, Presser et al. 1994, Barceloux 1999
Biogenic emissions (*i.e.* continental particulates, continental volatiles, marine)	Mosher and Duce 1987, Nriagu 1989
Anthropogenic	
Fossil fuel acquisition, combustion and processing	Lemly 1985, Lemly 2004,
Agricultural irrigation	Presser and Ohlendorf, 1987, Presser 1994, Presser et al. 1994, Seiler et al. 1999.
Metal smelting	Germani et al. 1981, Small et al. 1981, Nriagu and Wong 1983, Lemly 1994
Landfills	Lemly 2004
Oil use	Hawkins et al. 1997, Ohlendorf and Gala 2000
Mining	Lemly 1994, Lemly 1999a, Desborough et al. 1999, Herring et al. 1999, Lemly 2004
Constructed wetlands	Ohlendorf et al. 1986, Lemly 1999a, Skorupa 1998, Hansen et al. 1998, Lemly and Ohlendorf 2002
Fly ash disposal	Lemly 1999a, Lemly 2004
Livestock waste	Oldfield 1998, Lemly 1999b, Lemly 2004
Electrolytically produced metals	Lemly 2004

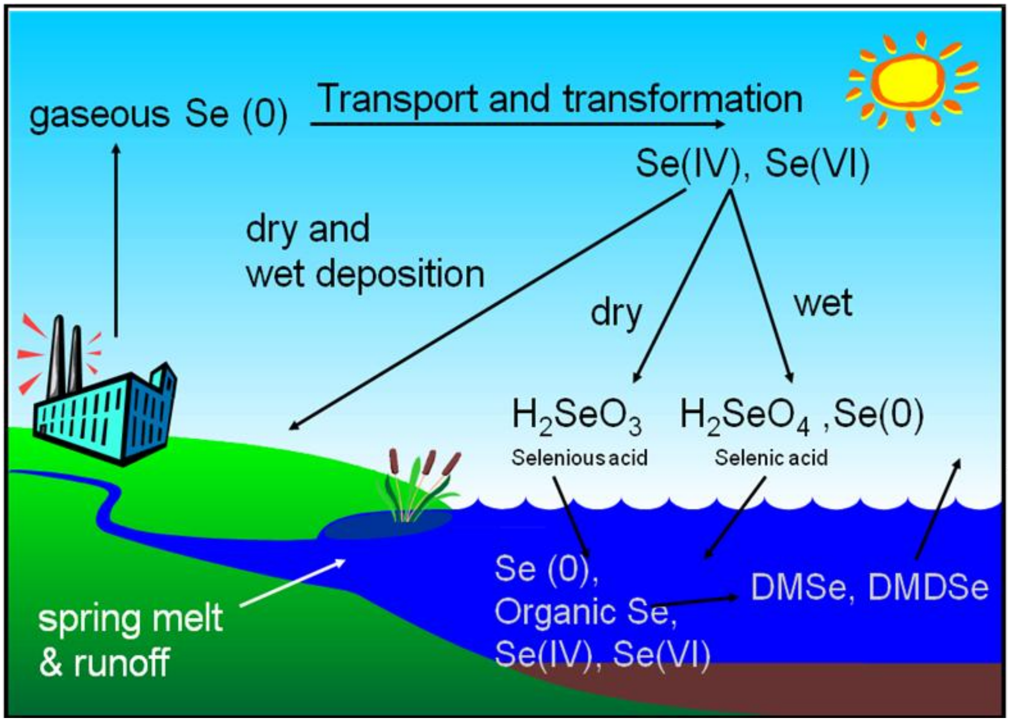

Figure 2.1 Selenium cycling in an aquatic environment (modified from Kelly 2007).

3.0 Selenium concentrations in sediment, water and biota from lakes in the Canadian Rocky Mountains: temporal and spatial trends and comparisons with guidelines for human and wildlife health

Introduction

Elevated concentrations of selenium (Se) in fishes and invertebrates from three lakes (Lake Minnewanka, Johnson Lake and Bighorn Lake) in southeastern Banff National Park were identified during a 2001 study of metal concentrations in fishes from Canadian Rocky Mountain lakes (Kelly 2007). In 2007, I studied temporal and spatial patterns of Se accumulation in lakes within or near Banff National Park. To examine temporal trends, I explored: 1) Se inputs to Johnson Lake over time, 2) seasonal changes in Se concentrations of water and invertebrates from Johnson Lake and Lake Minnewanka and 3) changes in Se concentrations of fishes and invertebrates from Johnson Lake, Lake Minnewanka and Spray Lakes over time, by comparing 2007 samples to those collected in the 1990s-2001 (Kelly 2007). I hypothesized that: 1) Se concentrations would be greater in newly deposited versus older sediments, if Se inputs to Johnson Lake increased over time, 2) seasonal changes to Se concentrations in water and invertebrates would be evident with peaks during snow melt runoff, 3) that Se concentrations in invertebrates and fishes would increase over time, if Se inputs concurrently increased.

To determine if elevated Se concentrations in biota were more widespread in Rocky Mountain lakes, I measured Se concentrations in invertebrates and fishes from 10 additional lakes over a larger geographic area. I hypothesized that elevated concentrations of Se in biota would be more widespread than Johnson Lake, Lake Minnewanka and Bighorn Lake if regional factors, such as geology, were related to increased Se concentrations in biota.

Se concentrations in sediment, water and biota were compared to established guidelines to determine if there was a health risk to fishes and their human and wildlife consumers. I hypothesized exceedences of published Se thresholds for sediment, water and biota in all lakes if elevated Se concentrations were widespread. Finally, I determined which factors within lakes (*e.g.,* fish biology variables such as age, fork length, total length, weight, condition factor, growth rate, $\delta^{13}C$, $\delta^{15}N$, %C, %N and concentration of other trace metals) and among lakes (*e.g.*, lake area, catchment area, watershed area to lake area ratio, aspect, slope, elevation, catchment geology, catchment vegetation composition, climate, fish biology variables including baseline-adjusted $\delta^{15}N$ and concentration of other trace metals) affected Se concentrations in water, invertebrates and (age-corrected) fishes. I hypothesized that Se concentrations in fishes, both within and among lakes, would be related to factors that were shown to affect Se concentrations of aquatic biota in other published studies.

Selenium (Se) is an essential trace element that can be toxic to organisms when present in high concentrations (Frost and Lish 1975, Hodson et al. 1980, Lemly 1993a, Wang and Gao 2001). Se contamination in the aquatic environment can originate from natural (*i.e.* Se-rich rock weathering (Nriagu and Pacyna 1988, Nriagu 1999, Presser et al. 1994, Barceloux 1999) and biogenic emissions (Mosher and Duce 1987, Nriagu 1989)) or anthropogenic sources (*i.e.* combustion of coal (Lemly 1985, Lemly 1993b, Lemly 2004), mining activities (Lemly 1994, Lemly 1999a, Skorupa 1998) and agricultural irrigation (Presser and Ohlendorf 1987, Presser 1994)). Se is subject to atmospheric transport and can be deposited to the landscape through precipitation (Haygarth et al. 1991, Bennett 1995, Beavington et al. 2004, Cutter and Cutter 2004, Lemly 2004) and dry deposition (Haygarth et al.

1991). Se has been shown to bioaccumulate in plankton, benthos and fish and biomagnify through aquatic food webs (Turner and Swick 1983, Lemly 1999b, Sappington 2002, Lemly 2004).

Elevated Se concentrations in aquatic biota can put the health of human and wildlife consumers at risk. Human exposure to elevated concentrations of Se produces symptoms ranging from brittle hair and nails to nausea and vomiting (Wilber 1980, Barceloux 1999). In birds, Se concentrations exceeding toxicity thresholds can cause teratogenesis and reproductive failure (Ohlendorf et al. 1986, Skorupa and Ohlendorf 1991). Symptoms of Se poisoning in fish can range in severity from swelling of the gill lamellae and bulging of the eye (Ohlendorf et al. 1986, Skorupa and Ohlendorf 1991) to deformities of the spine, head, mouth and fins (Lemly 1997, Lemly 2002, Holm et al. 2005) and reproductive failure (Lemly 1997, Holm et al. 2005).

Selenium concentrations in fishes can be influenced by several factors that affect sources, transport, and bioaccumulation of Se. Catchment geology (Presser et al. 1994) and climatic variation (Wen and Carignan 2007) can influence the sources of Se to a catchment or lake. Factors that affect the transport of Se in a catchment to a lake (*i.e.,* slope, aspect, lake area, catchment area) (as reviewed by Kelly 2007) also influence Se concentrations in aquatic biota. Factors that enhance bioaccumulation and biomagnification include redox conditions in sediment (Sappington 2002), flooding of reservoirs (Mailman et al. 2006), the presence of other elements that interact with Se (Hamilton and Palace 2001), uptake and excretion rates of Se by aquatic organisms (Rainbow 2002, Wang 2002, Sappington 2002) and food web structure (Sappington 2002, Lemly 2002).

Research on Se concentration in the Canadian Rocky Mountains has previously been conducted on rivers (Wayland and Crosley 2006) and creeks (Palace et al. 2004, Holm et al. 2005), but not on lakes (with the exception of Kelly 2007). My research is the first detailed assessment of Se dynamics in

Canadian Rocky Mountain lakes, and will generate baseline data and information that can be used by Parks Canada and Alberta Environment and Sustainable Resource Development for management purposes. These data will also be useful for assessing the effects of climate change on Se cycling in Canadian Rocky Mountain lakes. Climate change is responsible for several modifications to lake characteristics in the Rocky Mountains (Palace et al. 2004, Holm et al. 2005), Climate warming may potentially cause changes in biogeochemical cycling (Hauer et al. 1997), community structure of aquatic organisms (Schindler 1997) and higher evaporation rates (Moore et al. 1997) and may lead to increased deposition rates of atmospherically transported contaminants into lakes (Moore et al. 1997), which will likely impact Se dynamics in Rocky Mountain lakes.

Methods

Study Area

I sampled twelve lakes in the Canadian Rocky Mountains, Alberta, Canada (Figure 3.1, Appendix 1). In Banff National Park, Lake Minnewanka, Johnson Lake, Vermilion Lakes, Two Jack Lake and Cascade Pond were sampled. I also studied 7 lakes in Alberta Provincial Parks near Banff (Spray Lakes (Spray Valley Provincial Park), Mud Lake, Lower Kananaskis Lake, Upper Kananaskis Lake (Peter Lougheed Provincial Park), Gap Lake, Barrier Lake and Whiteman's Pond (Bow Valley Wildland Provincial Park)).

Nine of my twelve study lakes are reservoirs (Cascade Pond, Two Jack Lake, Johnson Lake, Lake Minnewanka, Whiteman's Pond, Spray Lakes, Upper Kananaskis Lake, Lower Kananaskis Lake and Barrier Lake). Some characteristics of reservoirs (e.g., higher sedimentation rates and sediment disturbance, seasonal drawdown and refilling, etc.) could be related to elevated Se concentrations in biota.

Field Sampling

In February 2007, a sediment core from Johnson Lake was obtained using a Glew gravity corer and 1.0 m acid-washed core tube. A 27 cm core was collected from the deepest area of the lake and extruded on site. Sediment was sliced in 0.5 cm sections from 0 to 20 cm and 1.0 cm sections from 20-26.5 cm. Sections were stored in Whirlpaks™ and subsequently frozen.

Lake Minnewanka and Johnson Lake were sampled monthly for water, zooplankton and benthic macroinvertebrates from May to October 2007. Surficial water samples were collected from a boat. Water to be analysed for Se and other trace metals and water chemistry was gathered in acid-washed high density polyethylene (HDPE) bottles. Unfiltered water for alkalinity, pH, conductivity, ammonia (NH_4^+), nitrite + nitrate (NO_2^{2-} + NO_3^-), total phosphorus (TP), turbidity, silica (SO_2), Se and other trace metals was kept cool until analysis. Water was filtered prior to analysis for anions (sulphate (SO_4^{2-}), chloride (Cl^-)), cations (sodium (Na^+), potassium (K^+), calcium (Ca^{2+}), magnesium (Mg^{2+}), dissolved organic carbon (DOC), colour (abs 350), total dissolved nitrogen (TDN), and total dissolved phosphorus (TDP). Chlorophyll *a* and particulates (for CHN) were collected on Whatman (Maidstone, England) GF/F filters and frozen.

Zooplankton samples were amassed by hauling a conical 140 µm mesh net vertically through the water column from a boat. Bulk samples were collected into Whirlpaks™ and frozen. For density determination and identification purposes, another sample was kept in an HDPE bottle and preserved in formalin. Benthic macroinvertebrates were gathered with sweep nets and by turning over rocks at the littoral zone. Samples were sorted by order into Whirlpaks™ and frozen.

In August 2007, small fishes were caught in minnow traps baited with dog food that were set overnight. I set multi-mesh "test mesh" gillnets (1, 1.5, 2, 3, 4") to capture larger fish. I collected up to 30 fish of each species over the size range present in Lake Minnewanka and Johnson Lake and a maximum of 10 fish of each species present in the 10 other study lakes. Individual fish were identified (Nelson and Paetz 1992) to species, weighed, measured (fork length and total length) and frozen. When available, benthic macroinvertebrates were collected from the study lakes (8 of 10 lakes; Two Jack Lake, Cascade Pond, Vermilion Lakes, Gap Lake, Barrier Lake, Lower Kananaskis Lake, Spray Lakes, and Mud Lake).

Laboratory Preparation

Sediment

Sediment samples were freeze-dried for 72 hours and ground with an acid-washed agate mortar and pestle until homogeneous.

Invertebrates

Zooplankton were identified to Order (copepods) or to Family (cladocerans) (Edmondson 1959), and enumerated to determine community structure. Benthic macroinvertebrates were identified to the lowest possible taxonomic level using Clifford (1991) and Thorp and Covich (1991). Bulk zooplankton and benthic macroinvertebrate samples were freeze-dried for 48 hours and ground with an acid-washed glass mortar and pestle until homogeneous.

Fishes

Fishes were dissected to determine sex, age and maturity. Sagittal otoliths were removed, preserved in 50% ethanol and aged by counting otolith rings under a dissecting microscope. Growth rates were determined two ways:

$$\text{Growth rate 1} = \frac{\text{age}}{\text{FL}}$$

$$\text{Growth rate 2} = \frac{W}{(\text{age-1})}$$

Where FL=fork length (mm) and W=weight (g). The calculation for growth rate 2 was used in addition to growth rate 1 because it accounts for lower weight gain in younger fishes (Kidd et al. 1995). Condition factor (K) was calculated as:

$$K = \frac{W \times 10^5}{L^3}$$

where W=weight (g), L=length (mm) and 10^5 is a scaling factor applied to bring K close to 1 (Barnham and Baxter 1998, Nash et al. 2006). Stomach contents were removed, preserved in 4% formalin and archived. Dorsal fish muscle tissue was prepared for trace metal and stable isotope analysis as described for zooplankton and macroinvertebrates.

Se and other trace metal analysis

Water, sediment, invertebrates and fishes were all analysed for Se and other trace metals at the University of Alberta's Radiogenic Isotope Facility (http://research.eas.ualberta.ca/rif/) using a Perkin Elmer Elan 6000 inductively coupled plasma mass spectrometer (ICP-MS). Water samples were diluted five times and analysed.

Sediment samples were weighed, extracted overnight using 10 mL of 1N HNO_3, decanted, diluted and run 10 times for labile metal concentration (including Se). The sample was then dried, the residue weighed, 5 mL of 1N

HNO_3 added and heated at 125°C until dry. After addition of 10 mL of 8N nitric acid, the sample was heated again for a few hours. When the entire sample had reached solution, it was diluted and analyzed for metal concentrations (including Se).

Prior to analysis, invertebrate and fish samples were acid digested with HNO_3. The acid digested samples were prepared for analysis by adding 0.1 ml each of trace metal grade HNO_3 and an internal standard solution (containing 100 ppb Bi, Sc, and In) and 8.8 ml of distilled deionized water to 1 ml of the digested sample solution. Samples were then analysed.

Stable isotope analysis

Freeze-dried and ground invertebrates were treated with HCl to dissolve surficial carbonates, oven dried at 65°C and reground. Invertebrate, zooplankton and fish samples were then weighed into tin cups to be analysed for stable ^{15}N and ^{13}C isotopes at the University of California at Davis Stable Isotope Facility (http://stableisotopefacility.ucdavis.edu/) using a PDZ Europa ANCA-GSL elemental analyzer interfaced to a PDZ Europa 20-20 isotope ratio mass spectrometer (IRMS) (Sercon Ltd., Cheshire, UK). Samples were combusted, oxides removed in a reduction reactor. Nitrogen and CO_2 were separated on a Carbosieve GC column (65°C, 65 mL/min) before entering the IRMS. Reference gas peaks were used to calculate provisional isotope ratios of the sample peaks. Final $\delta^{15}N$ and $\delta^{13}C$ values were calculated by adjusting the provisional values such that correct values for laboratory standards were obtained. Two laboratory standards were analysed every 12 samples. Glutamic acid was used for the concentration and ^{13}C regressions and ammonium sulfate and sucrose were used for drift correction. All laboratory standards were calibrated against NIST Standard Reference Materials (IAEA-N1, IAEA-N2, IAEA-N3, IAEA-CH7, and NBS-22). All stable isotope results are expressed in delta notation (δ):

$$\delta R‰ = [(R_{sample}/R_{standard}) - 1] \times 1{,}000$$

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

Fishes from Johnson Lake were analysed for ^{15}N and ^{13}C isotopes at the University of Alberta Biogeochemical Analytical Laboratory using a comparable method to that described above.

Geographical Information Systems (GIS)

I used ArcGIS 9.2 (Environmental Systems Research Institute 2006) ArcMap to complete GIS analyses for statistical modelling. I merged National Topographic Data Base (NTDB) data (Natural Resources Canada 2002) by entity type (*e.g.*, all watercourses, all waterbodies, all toponymy etc.) to create one data layer for each entity within my study area and projected the data to NAD 1983 UTM Zone 11.

Waterbodies were dissolved (to remove boundary edges) and attributes were updated with lake names and toponym. Waterbody polygons were converted to lines and merged with watercourses for use with the digital elevation model (DEM) reconditioning tool. Study lake lines were converted to raster grids for use in the watershed calculation. I converted Canadian Digital Elevation Data (CDED) (Natural Resources Canada 2003) to raster grids which were mosaicked to create one data layer. If required, I defined and projected the CDED to NAD 1983 UTM Zone 11 using a 20 meter cell size. I used ArcHydro (Environmental Systems Research Institute 2007) DEM Reconditioning, Fill sinks, and Flow Direction Tools to create a flow direction grid from the CDED and calculated the value 'fdr'. I calculated watersheds using map algebra in Spatial Analyst's Raster Calculator (*e.g.*, 1.shp = GridShape(Watershed([fdr], Select([lakel], 'value = 1')), NOWEED) and merged all outputs (*i.e.*, 1.shp...12.shp).

I projected NM11gelp.shp (Wheeler and McFeely 1991) in NAD 1983 UTM Zone 11 and clipped by study area boundary. I then intersected the merged watersheds with geology, and dissolved the watershed name and geology attributes of interest. I calculated the AREA field and exported the table to dBase.

For landcover, I mosaicked all Earth Observation for Sustainable Development (EOSD) data (Canadian Forest Service 2007) to one raster grid. I put my watersheds in Hawth's Tools (Beyer 2007) and used the "Clip Raster by Polygons" tool on the landcover raster. I then exported the attributes table to dBase, merged tables, joined attributes with watersheds, calculated new fields (*i.e.*, landcover class names) and created a Pivot Table.

I converted climate data (McKenney et al. 2008) to raster grids, projected them into NAD 1983 UTM Zone 11 and applied ZonalStats using individual watersheds converted to rasters (*i.e.,* w1...w12) on the climate grids (*e.g.*, biop01_1 = ZonalStats (w1, bio_out_p01, ALL) which I repeated for the other climate variables (biop, maxt, mint and p). I then merged the tables for each climate variable.

Statistical analysis

All statistical analyses were conducted using SYSTAT 12.00.08 for Windows (Systat Software, Inc. 2007) and Prism 5 for Windows Version 5.02 (Trial) (GraphPad Software, Inc. 1992-2009). Data were tested for normality using the Wilks-Shapiro test and by visual inspection of histograms. Linear regression, analysis of variance (ANOVA) and analysis of covariance (ANCOVA) are robust for slight violations of some assumptions (Zar 1999). Some of the data varied slightly from the normal distribution, but untransformed data were used and residuals checked for homoscedasicity and normality. I used these parametric analyses, rather than the non-parametric equivalents, to avoid loss of statistical power.

Temporal trends

Historical trends of Se in sediment

Selenium concentrations were plotted against sediment core depth to examine historical Se inputs to Johnson Lake. The sediment core was not dated, so conclusive trends of Se deposition over time could not be established. However, I examined temporal trends on the basis that older sediments would be deeper, and newer sediments would be closer to the sediment-water interface. Labile and total Se concentrations were visually inspected and compared to published toxicity thresholds.

Seasonal trends

I examined graphs of Se concentrations in water, invertebrates and zooplankton collected monthly throughout the ice free season (May to October) from Johnson Lake and Lake Minnewanka to detect seasonal variability. The Se concentrations were then compared to toxicity thresholds.

Long-term trends in Selenium concentrations of aquatic organisms

Se concentrations in invertebrates were graphed and compared between 2000/01 and 2007. Monthly data throughout the ice-free season were not available for 2000/01; therefore comparisons were made using 2007 data from the same month sampled in 2000/01. To compare Se concentrations in brook trout from Johnson Lake (2001 to 2007), lake trout from Lake Minnewanka (1991 and 2007) and lake trout from Spray Lakes (1992 to 2007) over time, I analyzed archived fish tissue (Donald: 1991, 1992 and Kelly: 2001). I then performed an ANCOVA with age as the covariate, Se concentration as the dependent variable and year as the factor. Age was chosen as the covariate rather than length or weight because fish growth rates decrease with elevation in Canadian Rocky Mountain lakes (Donald and Alger 1988), so fish length may not be a reliable measure of Se exposure in fishes from mountain lakes. I verified homogeneity of slopes by determining that the interaction factor was not significant before performing all ANCOVAs. If the F value was significant, I always used Tukey's Honestly Significant Differences (HSD) post-hoc test to determine the significance of pairwise comparisons between the factor (*i.e.,* year). I also calculated the percent change in Se concentration over time.

Within lake trends

Within lakes, I used ANCOVA to detect differences in Se concentrations, among fish species. The ANCOVA was performed using age as the covariate, Se concentration as the dependent variable and fish species as the factor. Cascade Pond, Mud Lake and Upper Kananaskis Lake were excluded from this analysis because there was only one species of fish present in those lakes. Gap Lake, Johnson Lake and Whiteman's Pond were omitted from the ANCOVA as there was a significant interaction term found between fish species and age. Linear regression was used to define within-lake relationships between age-corrected Se concentrations and baseline-adjusted $\delta^{15}N$ for organisms at the base of the food web (*i.e.,* snails) and all fish species. Please see the "among lakes" section below for age and baseline adjustment methods. The slope of each linear regression was an indicator of species-specific bioaccumulation rates. I evaluated slopes within lakes to assess differences in the rate of Se bioaccumulation among fish species. I then performed a one-way ANOVA and Tukey's HSD to determine if there were significant differences in slopes between fish species in the same lake.

Spatial trends

Among lakes

Se concentrations of invertebrates collected during August 2007 from all lakes were graphed and visually inspected for trends. Concentrations of

Se in the same species of fish were compared among lakes using ANCOVA. Age was the covariate, Se concentration the dependent variable, and lake was the factor. Three fish species (cisco (*Coregonus artedii*), cutthroat trout (*Salmo clarki*) and rainbow trout (*Salmo gairdneri*)) were not compared in this manner as these fishes were only present in one lake. All trout (combined) was also omitted as there was a significant interaction term in the ANCOVA between lake and age. The linear regression intercept and slope from the previously discussed linear regression of age-corrected Se concentrations and baseline-adjusted $\delta^{15}N$ were compared among all lakes that contained a particular species. Comparison of intercepts allowed me to assess differences in background Se concentrations among lakes. I then performed a one-way ANOVA and Tukey's HSD to determine if there were significant differences in slopes and intercepts among lakes for each fish species.

Factors that affect Se concentrations

Within lakes

Pearson product-moment correlation was used to characterize seasonal relationships at Johnson Lake and Lake Minnewanka between: 1) water Se concentrations and invertebrate Se concentrations, 2) Se concentrations in lake water and zooplankton Se concentrations, 3) water Se concentrations and the concentrations of other metals in water, 4) Se concentrations in water and water chemistry parameters, 5) zooplankton Se concentration and the concentration of other metals in zooplankton, 6) zooplankton Se concentration and zooplankton community composition (% taxa) and 7) Se concentrations in invertebrates and the concentration of other metals in invertebrates. Linear regression was used to determine if seasonal relationships existed between zooplankton and invertebrate Se concentrations and their respective values for $\delta^{13}C$ and $\delta^{15}N$.

I used ANCOVA to determine if fish biology variables (*i.e.*, condition factor, $\delta^{15}N$ and $\delta^{13}C$) were different among fish species within a lake. Age was the covariate, the fish biology variable the dependent variable, and the fish species was the factor. If a significant interaction was found between age and the fish species, an ANCOVA was not performed for that lake.

Pearson product-moment correlations were used to detect relationships between the concentrations of other trace metals and Se concentrations in fishes within lakes. I then used linear regressions to ascertain whether relationships existed between Se concentrations in each fish species and several explanatory variables (*i.e.*, age, fork length, total length, weight, condition factor, growth rate 1 and 2, $\delta^{15}N$ and $\delta^{13}C$. Within linear regression, the p value for the t-statistic indicates the significance of the slope. This corresponds to testing the significance of the correlation between the dependent and independent variables and is what is presented. I then applied multiple linear regression including only significant independent variables (those identified by previous Pearson product-moment correlations and linear regressions) to determine which of the factors were important for predicting Se concentrations in fish species within lakes. Forward stepwise multiple regressions (with alpha-to-enter and alpha-to-remove set to 0.15) were completed, and included variables that significantly affect fish Se concentration based on previous analyses. Collinearity of variables within each model was eliminated by creating correlation matrices and removing variables that correlated with a correlation coefficient greater than 0.6.

Among lakes

Age correction

To compare fish biology variables (*i.e.,* age, fork length, total length, weight, condition factor, growth rate 1 and 2, $\delta^{13}C$, $\delta^{15}N$ and concentrations of other trace metals) with Se concentrations for each species among lakes, all variables must first be age-corrected to a species-specific common age (*i.e.*, cutthroat trout (*Salmo clarki*) (age 4), rainbow trout (*Salmo gairdneri*) (age 4), brook trout (*Salvelinus fontinalis*) (age 4), brown trout (*Salmo trutta*) (age 5), lake trout (*Salvelinus namaycush*) (age 6), bull trout (*Salvelinus confluentus*) (age 6), mountain whitefish (*Prosopium williamsoni*) (age 6), cisco (*Coregonus artedii*) (age 6), white sucker (*Catostomus commersoni*) (age 8) and longnose sucker (*Catostomus catostomus*) (age 13)). When significant relationships existed between age and fish biology variables, linear regression was used for age correction. When no significant relationship existed, the mean value of the fish biology variable or Se concentration was used.

δ^{15}N baseline adjustment

Baseline δ^{15}N variability must be removed by baseline-adjustment before organism δ^{15}N values can be compared among lakes (VanderZanden and Rasmussen 1999, Post 2002). Snails (*i.e.*, *Stagnicola* spp., *Physa* spp., and *Helisoma* spp.) were used as the baseline organism. Within each lake, the δ^{15}N of snails was subtracted from invertebrate and fish δ^{15}N.

I used ANCOVA to determine if fish biology variables (i.e., condition factor, δ^{15}N and δ^{13}C) were different among lakes for each species of fish. No baseline organisms (i.e. gastropods) were collected in Upper Kananaskis Lake or Whiteman's Pond. Therefore, among lake comparisons of baseline adjusted δ^{15}N did not include Whiteman's Pond and Upper Kananaskis Lake. The covariate was age, the fish biology variable was the dependent variable, and lake was the factor. If a significant interaction was found between age and lake, an ANCOVA was not completed for that fish species.

Pearson product-moment correlations were used to establish relationships between the Se concentration of each fish species and the factors that affect Se concentrations in fishes among lakes (*i.e.*, lake area, catchment area, catchment area to lake area ratio, aspect, elevation, slope, geology, vegetation and climate within the catchment of the lake, and other trace metal concentrations in fish muscle). I also used correlation to establish any relationships between Se concentrations in fishes among lakes and Se concentrations in invertebrates. Linear regression was applied to characterize the among-lake relationship between Se concentrations in each fish species and growth rate, fork length, weight, condition factor and the concentrations of other trace metals. I then used multiple linear regression to determine which independent factors (that were significant in previous Pearson product-moment correlations and linear regressions) were important for predicting Se concentrations in fish species among lakes. Forward stepwise multiple regressions (with alpha-to-enter and alpha-to-remove set to 0.15) included variables that significantly affected fish Se concentration based on previous analyses. Collinearity of variables within each model was eliminated by creating correlation matrices and removing variables that correlated to the variable in question with a correlation coefficient greater than 0.6.

Results

Temporal trends

Historical trends of Se in sediment

Although the sediment core was not dated, I inferred temporal trends because sediment age should increase with sediment depth. Labile Se concentrations (n=47, $\bar{x}$=0.37, SE=0.20, range=0.10-0.93 mg Se/kg dw) remained constant throughout the sediment profile and did not increase in more recently deposited sediment (Figure 3.2). Total Se concentrations were more variable, ranging from 0.91-5.76 mg Se/kg dw (n=47, $\bar{x}$=3.98, SE=0.14).

Recently deposited sediments (0-7.5 cm) were lower in total Se than older sediment.

Seasonal trends

In May at Johnson Lake, Se concentrations in water (2.53 µg Se/L) were much greater than in any other month (Figure 3.3). This spring pulse, however, was not detected in Lake Minnewanka. In Lake Minnewanka, Se concentrations in water remained consistent over the entire ice-free season (range=0.42–0. 61 µg Se/L) (Figure 3.3). With the exception of the month of May, Se concentrations in water were greater in Lake Minnewanka than in Johnson Lake throughout the six-month period.

Concentrations of Se in zooplankton from Johnson Lake (range=1.16–2.55 mg Se/kg dw) and Lake Minnewanka (range=0.75–2.08 mg Se/kg dw) were variable throughout the ice-free season. No seasonal trends were observed in either lake (Figure 3.4).

Among macroinvertebrates, the lowest Se concentrations were in Gastropoda and Amphipoda from both Johnson Lake and Lake Minnewanka (Figure 3.5). Ephemeroptera and Trichoptera had the greatest Se concentrations among all taxa found in these lakes.

Se concentrations in amphipods were similar in both Johnson Lake (*Gammarus lacustris*, *Hyalella azteca*) and Lake Minnewanka (*G. lacustris*). Se concentrations in amphipods from Johnson Lake ranged from 1.66 to 3.06 mg Se/kg dw in *H. azteca* and 2.64 to 3.72 mg Se/kg dw in *G. lacustris* (Table 3.1, Figure 3.5a). In Lake Minnewanka, Se concentrations in *G. lacustris* ranged from 2.01 to 3.19 mg Se/kg dw (Table 3.1, Figure 3.5a).

Se concentrations in Ephemeroptera increased from May to June and then decreased in August in both Johnson Lake (*Callibaetis*) and Lake Minnewanka (*Leptophlebia* and *Siphlonurus*). Ephemeropteran Se concentrations in Johnson Lake ranged from 4.75 to 18.83 mg Se/kg dw in

Callibaetis (Table 3.1, Figure 3.5b). In Lake Minnewanka, Se concentrations in *Leptophlebia* ranged from 5.43 to 8.64 mg Se/kg dw (Table 3.1, Figure 3.5b).

Three genera of Trichoptera (*Anabolia*, *Agrypnia* and *Limnephilus*) were collected from Johnson Lake; however, only one genus was sampled from Lake Minnewanka (*Hesperophylax*). In Johnson Lake, Se concentrations increased in *Agrypnia* and *Limnephilus* from May to June. *Anabolia* Se concentrations ranged from 2.76 to 3.61 mg Se/kg dw, *Limnephilus* Se concentrations ranged from 5.33 to 7.85 mg Se/kg dw and *Agrypnia* had the greatest Se concentration at 8.58 mg Se/kg dw (Table 3.1, Figure 3.5c). Trichopterans were only collected in May from Lake Minnewanka, therefore, no seasonal trends were identified. However, *Hesperophylax* pupae and larvae had similar Se concentrations (13.44 and 13.49 mg Se/kg dw respectively) (Table 3.1, Figure 3.5c).

Odonates (Aeshnidae) were collected each month only from Johnson Lake. Se concentrations in Aeshnidae gradually decreased from May to October. Se concentrations of Aeshnidae ranged from 5.01 (in May) to 8.36 (in October) mg Se/kg dw (Table 3.1, Figure 3.5d).

Se concentrations of Gastropoda remained similar throughout the ice-free season in both Johnson Lake and Lake Minnewanka. In Johnson Lake, Se concentrations in *Stagnicola* ranged from 0.85 to 1.01 mg Se/kg dw (Table 3.1, Figure 3.5e). In Lake Minnewanka, Se concentrations of *Physa* ranged from 0.77 to 1.46 mg Se/kg dw (Table 3.1, Figure 3.5e).

Long-term trends in Selenium concentrations of aquatic organisms

Within taxa, Se concentrations of invertebrates were similar when compared between 2000/01 and 2007, with the exception of Ephemeroptera and possibly Trichoptera from Johnson Lake (Figure 3.6) which exhibited variable changes between 2000/01 and 2007. Concentrations of Se in Amphipoda and Odonata from Johnson Lake and Ephemeroptera from Lake Minnewanka were similar when compared between 2000/01 and 2007. The concentrations of Se in zooplankton from Johnson Lake and Lake Minnewanka were consistently greater in 2007 than in 2000/01.

In Johnson Lake, age-adjusted Se concentrations of brook trout significantly increased by 31.35% from 2001 to 2007 (ANCOVA: F=4.80, p=0.04). Similarly, age-adjusted Se concentrations in lake trout from Lake Minnewanka increased 30.98% from 1991 to 2007 (ANCOVA: F=7.54, p<0.01). In Spray Lakes, lake trout exhibited a 37.78% increase in age-adjusted Se concentrations from 1992 to 2007 (ANCOVA: F=10.06, p<0.01) (Figure 3.7).

Within lake trends

Within lakes, concentrations of Se in fishes were variable (Table 3.2, Figure 8). Within Lake Minnewanka, Se concentrations were significantly different among fish species (ANCOVA: F=15.63, p<0.01). Concentrations of Se in mountain whitefish (*Prosopium williamsoni)* were significantly greater than Se concentrations in both cisco (*Coregonus artedii*) (Tukey's HSD: p<0.01) and lake trout (*Salvelinus namaycush*) (Tukey's HSD: p<0.01). There was also a significant difference in Se concentrations among fish species from Two Jack Lake (ANCOVA: F=7.91, p<0.01). Post hoc tests indicated that Se concentrations in lake trout were significantly lower than concentrations in mountain whitefish (Tukey's HSD: p<0.01) and longnose sucker (*Catostomus catostomus*) (Tukey's HSD: p=0.02). Se concentrations were not significantly different when compared between lake trout and mountain whitefish from Spray Lakes (ANCOVA: F=3.60, p=0.08), brown trout (*Salmo trutta*), mountain whitefish or longnose sucker from Barrier Lake (ANCOVA: F=0.36, p=0.70), and bull trout (*Salvelinus confluentus*) and longnose sucker from Lower Kananaskis Lake (ANCOVA: F=0.25, p=0.63). The Se concentrations of brook trout (*Salvelinus fontinalis*) and white sucker (*Catostomus commersoni*) from Vermilion Lakes were also not significantly different (ANCOVA: F=0.01, p=0.91).

The slope and intercept of the relationship between Se concentration and baseline-adjusted $\delta^{15}N$ for each lake by fish species is presented in Table 3.3. No significant difference was identified among the slopes of the Se versus $\delta^{15}N$ relationship for fish species from Johnson Lake (one-way ANOVA: F=1.93, p=0.15). The slopes of the Se versus $\delta^{15}N$ relationship for three species of fish from Lake Minnewanka were significantly different (one-way ANOVA: F=27.76, p<0.01). The Se bioaccumulation rates of mountain whitefish were greater than both cisco (Tukey's HSD: p<0.05) and lake trout (Tukey's HSD: p<0.05). A significant difference in the rate of Se bioaccumulation among fish species from Gap Lake was also identified (one-way ANOVA: F=11.75, p<0.01). The slope of the Se versus $\delta^{15}N$ relationship was greater for white sucker than mountain whitefish (Tukey's HSD: p<0.05) and lake trout (Tukey's HSD: p<0.05). In Two Jack Lake, a significant difference in the bioaccumulation rate of Se among fish species was also detected (one-way ANOVA: F=6.55, p<0.01). Lake trout from Two Jack Lake had lower rates of Se bioaccumulation than both mountain whitefish (Tukey's HSD: p<0.05) and longnose sucker (Tukey's HSD: p<0.05). No significant difference was found when the slopes of the Se versus $\delta^{15}N$ relationship were compared among fish species from Vermilion Lakes (one-way ANOVA: F=2.68, p=0.24) and Barrier Lake (one-way ANOVA: F=0.06, p=0.94).

As expected, there were no significant difference in the y-intercept (baseline Se concentration) among fish species within lakes (Johnson Lake: one-way ANOVA: F=0.27, p=0.78, Lake Minnewanka: one-way ANOVA: F=0.25, p=0.79, Barrier Lake: one-way ANOVA: F=0.0002, p=0.99, Gap Lake: one-way ANOVA: F=0.05, p=0.95, Two Jack Lake: one-way ANOVA: F=0.07, p=0.93.).

Spatial trends

Among lakes

The Se concentrations of five orders of benthic invertebrates collected from study lakes other than Johnson Lake and Lake Minnewanka in August 2007 are presented in Figure 3.9. Benthic macroinvertebrates collected from Vermilion Lakes, a wetland complex, had the greatest Se concentration among all lakes for three of four orders (Trichoptera, Odonata and Gastropoda) and one species of Amphipoda (*Gammarus lacustris)*. Concentrations of Se in Amphipoda exhibited little variability among lakes. Se concentrations in *Gammarus lacustris* ranged from 2.19 mg/kg dw in Barrier Lake to 3.92 mg Se/kg dw in Vermilion Lakes (Table 3.1, Figure 3.9a), while Se concentrations of *Hyalella azteca* ranged from 3.79 mg Se/kg dw in Vermilion Lakes to 3.93 mg/kg dw in Two Jack Lake (Table 3.1, Figure 3.9a).

Se concentrations in *Siphlonurus* (Ephemeroptera) decreased from west to east among lakes (Figure 9b). *Siphlonurus* Se concentrations ranged from 2.93 mg Se/kg dw from Barrier Lake to 6.93 mg Se/kg in Cascade Pond. The Se concentrations of *Leptophlebia* and *Callibaetis* from Cascade Pond were within the range of Se concentrations observed in *Siphlonurus* (Table 3.1, Figure 3.9b).

Psychoglypha (Trichoptera) Se concentrations also decrease from west to east among lakes (Figure 3.9c). *Psychoglypha* Se concentrations ranged from 3.99 mg Se/kg dw in Gap Lake to 7.91 mg Se/kg dw in Two Jack Lake. Se concentrations of all other Trichopteran taxa were within the range of *Psychoglypha* with exception of *Anabolia* from Vermilion Lakes (14.65 mg Se/kg dw).

Odonates were only collected from two lakes (Figure 3.9d). Their Se concentrations ranged from 3.90 mg/kg dw in Corduliidae from Mud Lake to 14.56 mg/kg dw in Coenagrionidae from Vermilion Lakes (Table 3.1, Figure 3.9d).

Se concentrations in 3 genera of Gastropoda (*Helisoma*: 2.08 mg/kg dw, *Stagnicola*: 3.25 mg/kg dw and *Physa*: 4.13 mg/kg dw) from Vermilion Lakes were greatest when compared to gastropods collected from other lakes (Table 3.1, Figure 3.9e). In lakes where both *Physa* and *Stagnicola* were collected, the Se concentration of *Physa* was always greater than that of *Stagnicola*. *Stagnicola* Se concentrations ranged from 0.54 mg/kg dw in Cascade Pond to 3.25 mg/kg dw in Vermilion Lakes (Figure 3.9e). Se concentrations of *Physa* increased from east to west and ranged from 1.12 mg Se/kg dw in Barrier Lake to 4.13 mg Se/kg dw in Vermilion Lakes (Figure 3.9e). *Helisoma* was only collected from Vermilion Lakes and thus no comparisons can be made.

Se concentrations in the same species of fish that were collected from 3 or more lakes were variable (Table 3.2, Figure 3.10). Se concentrations of fish species collected from fewer than three lakes are presented in Figure 3.11. Brook trout Se concentrations were found to be significantly different among lakes (ANCOVA: F=11.42, $p<0.01$) (Figure 3.10a). Se concentrations of brook trout from Vermilion Lakes were significantly lower than brook trout from Johnson Lake (Tukey's HSD: $p<0.01$) and Cascade Pond ($p=0.050$) (Figure 3.10a). Lake trout Se concentrations were significantly different among lakes (ANCOVA: F=21.80, $p=0.04$) (Figure 3.10b). Lake trout from Lake Minnewanka had significantly greater Se concentrations than those from Spray Lakes (Tukey's HSD: $p=0.03$) (Figure 3.10b). A significant difference in Se concentrations of brown trout was also detected among lakes (ANCOVA: F=10.44, $p<0.01$). Brown trout from Gap Lake contained greater Se concentrations than those from Barrier Lake (Tukey's HSD: $p<0.01$) and Whiteman's Pond (Tukey's HSD: $p=0.02$) (Figure 3.10c). Se concentrations of mountain whitefish were also significantly different among lakes (ANCOVA: F=8.71, $p<0.01$) (Figure 3.10d). Mountain whitefish from Barrier Lake had significantly lower Se concentrations than mountain whitefish from Gap Lake (Tukey's HSD: $p<0.01$), Lake Minnewanka (Tukey's HSD: $p<0.01$) and Two

Jack Lake (Tukey's HSD: $p<0.01$) (Figure 3.10d). Se concentrations in longnose sucker were also significantly different among lakes (ANCOVA: F=17.68, $p<0.01$) (Figure 3.10e). Longnose sucker from Two Jack Lake had greater Se concentrations than those from both Barrier Lake (Tukey's HSD: $p<0.01$) and Whiteman's Pond (Tukey's HSD: $p<0.01$). Longnose sucker from Lower Kananaskis Lake also had higher Se concentrations than longnose sucker from Barrier Lake (Tukey's HSD: $p<0.01$) (Figure 3.10e). Se concentrations in white sucker also differed significantly among lakes (ANCOVA: F=4.66, $p<0.01$) (Figure 3.10f). White sucker from Johnson Lake contained significantly greater Se concentrations than white sucker from Gap Lake (Tukey's HSD: $p=0.02$) and Vermilion Lakes (Tukey's HSD: $p=0.04$) (Figure 3.10f). There were no significant differences in the Se concentrations of bull trout from Lower and Upper Kananaskis Lakes (ANCOVA: F=3.93, $p=0.06$) (Figure 3.11).

The slope and intercept of the relationship between Se concentration and baseline-adjusted $\delta^{15}N$ for each lake by fish species is presented in Table 3.3. When all trout species were combined, there was a significant difference in the slope of Se versus baseline-adjusted $\delta^{15}N$ (*e.g.*, bioaccumulation rate of Se) among lakes (one-way ANOVA: F=5.16, $p<0.01$). The bioaccumulation rate of brook trout from Cascade Pond was significantly greater than that of brown trout from Barrier Lake (Tukey's HSD: $p<0.05$), bull trout from Lower Kananaskis Lake (Tukey's HSD: $p<0.05$), lake trout from Lake Minnewanka (Tukey's HSD: $p<0.05$) cutthroat trout (*Salmo clarki*) from Mud Lake (Tukey's HSD: $p<0.05$), lake trout from Spray Lakes (Tukey's HSD: $p<0.05$), lake trout from Two Jack Lake (Tukey's HSD: $p<0.05$) and brook trout from Vermilion Lakes (Tukey's HSD: $p<0.05$). Se bioaccumulation rates in brook trout from Johnson Lake were significantly greater than brook trout from Vermilion Lakes (Tukey's HSD: $p<0.05$), bull trout from Lower Kananaskis Lake (Tukey's HSD: $p<0.05$) and lake trout from Two Jack Lake (Tukey's HSD:

$p<0.05$). Among lakes containing brook trout, there were significant differences detected in the slope of the Se versus baseline-adjusted $\delta^{15}N$ relationship (one-way ANOVA: $F=6.57$, $p<0.01$). Post-hoc testing revealed that brook trout from Vermilion Lakes had lower Se bioaccumulation rates than those from Cascade Pond (Tukey's HSD: $p<0.05$) and Johnson Lake (Tukey's HSD: $p<0.05$). The bioaccumulation rate of Se was also significantly different among lakes for brown trout (one-way ANOVA: $F=13.13$, $p<0.05$). Brown trout from Gap Lake had a greater rate of Se bioaccumulation than brown trout from Barrier Lake (Tukey's HSD: $p<0.05$). There were significant differences in the slope of Se versus baseline-adjusted $\delta^{15}N$ for mountain whitefish among lakes (one-way ANOVA: $F=4.57$, $p<0.01$). Mountain whitefish from Lake Minnewanka had greater rates of Se bioaccumulation than those from Barrier Lake (Tukey's HSD: $p<0.05$) and Gap Lake (Tukey's HSD: $p<0.05$). Se bioaccumulation rates were also significantly greater in mountain whitefish from Two Jack Lake than those from Gap Lake (Tukey's HSD: $p<0.05$). Among-lake differences in the slope of the Se versus baseline-adjusted $\delta^{15}N$ relationship were also detected for white sucker (one-way ANOVA: $F=6.46$, $p<0.01$). White suckers from Gap Lake had greater rates of Se bioaccumulation than white suckers from Johnson Lake (Tukey's HSD: $p<0.05$). There were no significant among-lake differences in slope (*i.e.*, Se bioaccumulation rates) for lake trout (one-way ANOVA: $F=0.68$, $p=0.51$) or longnose sucker (one-way ANOVA: $F=9.12$, $p=0.09$).

Differences in y-intercept (*e.g.*, the baseline concentration of Se in the lake) were not statistically significant when compared among lakes by species of fish (Brook trout: one-way ANOVA: $F=2.73$, $p=0.08$, Brown trout: one-way ANOVA: $F=2.46$, $p=0.21$, Lake trout: one-way ANOVA: $F=1.17$, $p=0.32$, All trout: one-way ANOVA: $F=1.28$, $p=0.25$, Mountain whitefish: one-way

ANOVA: F=1.82, p=0.14, White sucker: one-way ANOVA: F=1.59, p=0.21, Longnose sucker: one-way ANOVA: F=1.75, p=0.28).

Factors that affect Se concentrations

Within lakes

In Johnson Lake and Lake Minnewanka, Pearson product-moment correlation were used to investigate relationships between Se concentrations in water and both the concentration of other trace metals in water and water chemistry parameters (Appendix 2). In Johnson Lake, Se concentrations in water were positively correlated with the concentrations of many other metals in water (Table 3.4). In Lake Minnewanka, the concentrations of Phosphorus (P) (r=-0.99, p<0.01), Zinc (Zn) (r=-0.84, p=0.04), Niobium (Nb) (r=-0.99, p<0.01) and Tantalum (Ta) (r=-0.99, p<0.01) in water were negatively correlated while Magnesium (Mg) (r=0.86, p=0.03) was positively correlated with the concentration of Se in water (Table 3.4). Three water chemistry parameters (*i.e.,* chloride, ammonium and silica) were correlated with water Se concentrations in Johnson Lake (Table 3.5), but there were no similar significant relationships in Lake Minnewanka.

Relationships between Se concentrations in water and Se concentrations in zooplankton and invertebrates were investigated in Johnson Lake and Lake Minnewanka using Pearson product-moment correlations. Se concentrations in water were not significantly correlated with zooplankton (p>0.05) or benthic invertebrate (p>0.05) Se concentrations in either lake.

Using Pearson product moment correlations, Se concentrations in zooplankton from Johnson Lake were negatively related to the concentration of some metals (aluminum (Al), zirconium (Zr) and lead (Pb), (p<0.05), see Table 3.6) and positively related with cyclopoid density (r=0.85, p=0.03). In Lake Minnewanka, Se concentrations in zooplankton were negatively correlated with the concentrations of 34 other metals ((p<0.05), see Table 3.6) and positively correlated with Zinc (Zn) concentration ($r=0.89$, $p=0.02$ see Table 3.6) in zooplankton. The concentration of Se in zooplankton Lake Minnewanka was not correlated with zooplankton density or community composition ($p>0.05$).

Pearson product-moment correlations were used to examine relationships between Se concentrations of invertebrate taxa and the concentrations of other metals in invertebrates from both Johnson Lake (Table 3.7) and Lake Minnewanka (Table 3.8). Significant relationships were identified between Se and other metals in Trichoptera, Amphipoda, Odonata and Ephemeroptera in Johnson Lake (Table 3.7) and Amphipoda and Gastropoda in Lake Minnewanka (Table 3.8), but with a different suite of metals for each taxa.

In Johnson Lake, Se concentrations in benthic invertebrates and zooplankton were not significantly related to their respective $\delta^{13}C$ and $\delta^{15}N$ based on the results of linear regression analyses ($p>0.05$). The same trend was evident within Lake Minnewanka, with one exception. Trophic position (*i.e.*, $\delta^{15}N$) was positively related to Se concentration in *Gammarus lacustris* ($R^2=0.74$, $p=0.03$).

Individual linear regression and ANCOVA analyses were performed to examine relationships between Se concentrations in fishes and fish variables within lakes (Appendix 3). These factors included: age, fork length, weight, condition factor, growth rate 1 (weight/ (age-1)), growth rate 2 (fork length/age), other trace metals, ^{13}C and $\delta^{15}N$. Significant linear regression results are reported in Table 3.9.

Despite Se concentrations differing only between fish species within Lake Minnewanka and Two Jack Lake, I found that $\delta^{15}N$, $\delta^{13}C$ and condition factor were significantly different among fish species within each lake. In Lake Minnewanka, $\delta^{15}N$ values were significantly different among fish species (ANCOVA: $F=35.87$, $p<0.01$). The $\delta^{15}N$ of lake trout was greater than both cisco (Tukey's HSD: $p<0.01$) and mountain whitefish (Tukey's HSD:

p<0.01). The $\delta^{15}N$ of cisco was also significantly greater than that of mountain whitefish (Tukey's HSD: p=0.02). The $\delta^{15}N$ of fish species from Two Jack Lake was significantly different (ANCOVA: F=59.63, p<0.01). The $\delta^{15}N$ of lake trout was greater than both mountain whitefish (Tukey's HSD: p<0.01) and longnose sucker (p<0.01). Mountain whitefish had a significantly greater $\delta^{15}N$ than longnose sucker (Tukey's HSD: p=0.01). Interestingly, $\delta^{15}N$ was significantly different between fish species in all lakes except Lower Kananaskis Lake (ANCOVA: p<0.05), however, there was no significant difference in Se concentrations between species in the majority of the study lakes.

The $\delta^{13}C$ of fish species from Lake Minnewanka was significantly different (ANCOVA: F=14.03, p<0.01). The $\delta^{13}C$ of cisco was significantly more negative than both lake trout (Tukey's HSD: p<0.01) and mountain whitefish (Tukey's HSD: p<0.01). In Two Jack Lake, there was also a significant difference in $\delta^{13}C$ among fish species (ANCOVA: F=4.41, p=0.03). Longnose sucker were significantly greater in $\delta^{13}C$ than mountain whitefish (Tukey's HSD: p=0.042). The only other lake where $\delta^{13}C$ was different among species was in Whiteman's Pond (ANCOVA: p<0.05).

Condition factor was significantly different among fish species from Lake Minnewanka (ANCOVA: F=6.94, p<0.01). Mountain whitefish had a significantly greater condition factor than both cisco (Tukey's HSD: p=0.020) and lake trout (Tukey's HSD: p=0.02). Within Two Jack Lake, longnose suckers had a greater and significantly different condition factor when compared to lake trout (ANCOVA: F=5.98, p=0.01, Tukey's HSD: p<0.01). The only other lake where condition factor was significantly different among species was Lower Kananaskis Lake (ANCOVA: p<0.05).

Within lakes, many significant yet variable correlations between the concentration of Se and the concentrations of other metals in fish species were identified for all lakes, except cutthroat trout from Mud Lake (Tables 3.10-

3.20). Zirconium (Zr) was the only metal that was significantly correlated with Se concentrations in cutthroat trout from Mud Lake ($r=-0.72$, $p=0.03$).

All significant but uncorrelated fish biology variables were subjected to multiple linear regression for each species within each lake. Only seven significant multiple regression models were calculated for fish species within lakes (*i.e.*, white sucker from Johnson Lake, bull trout from Lower Kananaskis Lake, mountain whitefish from Gap Lake, brook trout from Cascade Pond, lake trout from Spray Lakes and brown trout and longnose sucker from Whiteman's Pond). The significant variables included were different in each lake. A significant multiple regression model for white sucker from Johnson Lake ($F=9.87$, $p<0.01$) included growth rate 1 and weight and explained 42.22% of variation in mean Se concentration:

$$\text{Mean [Se] Johnson Lake white sucker} = -0.40 * \text{growth rate 1} + 0.0096 * \text{weight} + 16.25$$

The multiple regression model for bull trout from Lower Kananaskis Lake was significant ($F=27.35$, $p<0.01$), included condition factor and trophic position and explained 88.66% of variation in mean Se concentration:

$$\text{Mean [Se] Lower Kananaskis Lake bull trout} = -3.82 * \text{condition factor} + 3.52 * \delta^{15}N - 22.19$$

A multiple regression model explained 73.49% of variation in mean Se concentrations of mountain whitefish from Gap Lake, and was statistically significant ($F=9.70$, $p<0.01$). It included growth rate 2 and trophic position:

$$\text{Mean [Se] Gap Lake mountain whitefish} = 0.16 * \text{growth rate 2} - 6.33 * \delta^{15}N + 48.42$$

The multiple regression model for brook trout from Cascade Pond was significant ($F=15.83$, $p<0.01$) included fork length and explained 66.44% of variation in mean Se concentration:

$$\text{Mean [Se] Cascade Pond brook trout} = 0.16 * \text{fork length} - 2.18$$

A significant multiple regression model ($F=7.17$, $p<0.01$) that included weight and explained 66.44% of variation in mean Se concentration was calculated for Spray Lakes:

$$\text{Mean [Se] Spray Lakes lake trout} = 0.00036 * \text{weight} + 4.51$$

A significant multiple regression model ($F=19.84$, $p<0.01$) that included condition factor, explained 71.26% of variation in mean Se concentration of brown trout from Whiteman's Pond:

$$\text{Mean [Se] Whiteman's Pond brown trout} = -6.44 * \text{condition factor} + 11.80$$

The multiple regression model for longnose sucker from Whiteman's Pond was significant ($F=21.12$, $p<0.01$) included age and growth rate 2 and explained 89.42% of variation in mean Se concentration:

$$\text{Mean [Se] Whiteman's Pond longnose sucker} = 1.48 * \text{age} + 1.47 * \text{growth rate 2} - 61.27$$

Among lakes

When Se concentrations of all fish species were correlated to the Se concentration of invertebrates among lakes, only all trout species grouped was positively correlated to Se concentration of Ephemeroptera ($r=0.64$, $p=0.046$).

ANCOVA was performed to examine relationships between Se concentrations in fishes and $\delta^{15}N$, $\delta^{13}C$ and conditions factor.

The baseline-adjusted $\delta^{15}N$ of brook trout differed significantly among lakes (ANCOVA: $F=235.30$, $p<0.01$) (Appendix 4). Baseline-adjusted $\delta^{15}N$ was greater in brook trout from Johnson Lake than those from Vermilion Lakes (Tukey's HSD: $p<0.01$) and Cascade Pond (Tukey's HSD: $p<0.01$). The $\delta^{13}C$ of brook trout was also significantly different among lakes (ANCOVA: $F=72.03$, $p<0.01$). The $\delta^{13}C$ of brook trout from Johnson Lake was significantly lower than those from Vermilion Lakes (Tukey's HSD: $p<0.01$) and Cascade Pond ($p<0.01$).

Among lakes, $\delta^{13}C$ of brown trout was significantly different (ANCOVA: F=98.22, $p<0.01$). Brown trout from Gap Lake had greater $\delta^{13}C$ than brown trout from both Barrier Lake (Tukey's HSD: $p<0.01$) and Whiteman's Pond (Tukey's HSD: $p<0.01$).

Bull trout from Lower Kananaskis Lake had greater baseline-adjusted $\delta^{15}N$ (ANCOVA: F= 18.04, $p<0.01$) (Appendix 4) than bull trout from Upper Kananaskis Lake. No baseline organisms (*e.g.*, gastropods) were obtained from Upper Kananaskis Lake. Since Upper and Lower Kananaskis Lakes share the same water, I assumed a similar baseline in both lakes and compared unadjusted $\delta^{15}N$ values for bull trout from both lakes. Bull trout from Lower Kananaskis Lake had significantly lower $\delta^{13}C$ than those from Upper Kananaskis Lake (ANCOVA: F=5.58, p=0.03).

Significant difference in the baseline-adjusted $\delta^{15}N$ of lake trout were detected among lakes (ANCOVA: F=18.11, $p<0.01$) (Appendix 4). Lake trout from Two Jack Lake were greater in $\delta^{15}N$ than those from Lake Minnewanka (Tukey's HSD: p=0.01) and Spray Lakes (Tukey's HSD: $p<0.01$). However, lake trout from Lake Minnewanka had greater $\delta^{15}N$ than those from Spray Lakes (Tukey's HSD: $p<0.01$). When compared among lakes, the $\delta^{13}C$ of lake trout was significantly different (ANCOVA: F=56.49, $p<0.01$). The $\delta^{13}C$ of lake trout from Spray Lakes was significantly lower than that of lake trout from both Lake Minnewanka (Tukey's HSD: $p<0.01$) and Two Jack Lake (Tukey's HSD: $p<0.01$). Lake trout from Lake Minnewanka also had lower $\delta^{13}C$ than lake trout from Two Jack Lake (Tukey's HSD: $p<0.01$).

The baseline-adjusted $\delta^{15}N$ of mountain whitefish was significantly different among lakes (ANCOVA: F=26.04, $p<0.01$) (Appendix 4). Mountain whitefish from Barrier Lake had greater $\delta^{15}N$ than mountain whitefish from Gap Lake (Tukey's HSD: $p<0.01$). Mountain whitefish from Lake Minnewanka had higher $\delta^{15}N$ than those from both Gap Lake (Tukey's HSD: $p<0.01$) and Spray Lakes (Tukey's HSD: $p<0.01$), while mountain whitefish from Two Jack Lake had greater $\delta^{15}N$ than those from Spray Lakes (Tukey's

HSD: $p<0.01$), Gap Lake (Tukey's HSD: $p<0.01$) and Barrier Lake (Tukey's HSD: $p=0.02$).

The $\delta^{13}C$ of mountain whitefish also varied significantly among lakes (ANCOVA: F=112.60, $p<0.01$). Mountain whitefish from Gap Lake were greater in $\delta^{13}C$ than those from Whiteman's Pond (Tukey's HSD: $p<0.01$), Lake Minnewanka (Tukey's HSD: $p<0.01$), Two Jack Lake (Tukey's HSD: $p<0.01$), Barrier Lake (Tukey's HSD: $p<0.01$) and Spray Lakes (Tukey's HSD: $p<0.01$). Mountain whitefish from Lake Minnewanka were greater in $\delta^{13}C$ than those from Spray Lakes (Tukey's HSD: $p<0.01$), Barrier Lake (Tukey's HSD: $p<0.01$) and Whiteman's Pond (Tukey's HSD: $p<0.01$). The $\delta^{13}C$ of mountain whitefish from Two Jack Lake was greater than mountain whitefish from Spray Lakes (Tukey's HSD: $p<0.01$), Barrier Lake (Tukey's HSD: $p<0.01$) and Whiteman's Pond (Tukey's HSD: $p<0.01$).

The condition factor of mountain whitefish also varied significantly among lakes (ANCOVA: F=8.57, $p<0.01$). Mountain whitefish from Lake Minnewanka had greater condition factor than those from Spray Lakes (Tukey's HSD: $p<0.01$), while mountain whitefish from Two Jack Lake had greater condition factor than those from Spray Lakes (Tukey's HSD: $p<0.01$), Whiteman's Pond (Tukey's HSD: $p=0.02$) and Barrier Lake (Tukey's HSD: $p<0.01$).

The baseline-adjusted $\delta^{15}N$ of longnose sucker was significantly different among lakes (ANCOVA: F=24.36, $p<0.01$) (Appendix 4). Longnose sucker from Barrier Lake had lower $\delta^{15}N$ than those from Two Jack Lake (Tukey's HSD: $p<0.01$) and Lower Kananaskis Lake (Tukey's HSD: $p<0.01$). The $\delta^{13}C$ of longnose sucker also varied significantly among lakes (ANCOVA: F=36.09, $p<0.01$). Longnose sucker from Two Jack Lake had significantly greater $\delta^{13}C$ than those from Whiteman's Pond (Tukey's HSD: $p<0.01$), Barrier Lake (Tukey's HSD: $p<0.01$) and Lower Kananaskis Lake (Tukey's HSD: $p<0.01$). The condition factor of longnose sucker was also significantly different among lakes (ANCOVA: F=16.23, $p<0.01$). The trend apparent for

$\delta^{13}C$ was the same as condition factor. Longnose sucker from Two Jack Lake had greater condition factor than those from Whiteman's Pond (Tukey's HSD: $p<0.01$), Barrier Lake (Tukey's HSD: $p<0.01$) and Lower Kananaskis Lake (Tukey's HSD: $p<0.01$).

The baseline-adjusted $\delta^{15}N$ of white sucker also varied significantly among lakes (ANCOVA: F=356.88, $p<0.01$) (Appendix 4). White sucker from Johnson Lake had greater $\delta^{15}N$ than those from Gap Lake (Tukey's HSD: $p<0.01$) and Vermilion Lakes (Tukey's HSD: $p<0.01$). White sucker from Vermilion Lake were lower in $\delta^{15}N$ than those from Gap Lake (Tukey's HSD: $p<0.01$). The $\delta^{13}C$ of white sucker were also significantly different among lakes (ANCOVA: F=449.88, $p<0.01$). White sucker from Johnson Lake were significantly lower in $\delta^{13}C$ than those from Vermilion Lakes (Tukey's HSD: $p<0.01$), Whiteman's Pond (Tukey's HSD: $p<0.01$) and Gap Lake (Tukey's HSD: $p<0.01$). The $\delta^{13}C$ of white sucker from Gap Lake was significantly greater than those from Whiteman's Pond (Tukey's HSD: $p<0.01$) and Vermilion Lakes (Tukey's HSD: $p<0.01$).

I used Pearson product-moment correlations to assess variables that affect Se concentrations in fishes among lakes. Variables examined were: lake area, catchment area, catchment area: lake area ratio, aspect, elevation, slope, % geology, % vegetation cover, concentrations of other trace elements, climate variables (temperature and precipitation), growth rate 1, growth rate 2, fork length, weight, condition factor, $\delta^{15}N$ and $\delta^{13}C$. (Appendix 1 and 3).

When compared among lakes, Se concentration of brook trout, brown trout, all trout, mountain whitefish and longnose sucker were significantly correlated with a variety of different explanatory variables (Table 3.21), including metals (Table 3.22).

I used multiple regression to assess uncorrelated variables that were significantly related to Se concentrations in fishes among lakes in other analyses. Significant multiple regression models were generated for mountain whitefish and all trout species combined. The multiple regression model was significant (F= 413.45, p<0.01), included weight of fish and vegetation characteristics and explained 99.84% of variation in the mean Se concentrations of mountain whitefish:

$$\text{Mean [Se] in all mountain whitefish} = -26.52 * \%\ \text{open conifer} + 18.09 * \%\ \text{dense broadleaf} + 0.01 * \text{weight} + 8.09$$

For all trout species combined, the multiple regression model was also significant (F=5.99, p=0.034), included a vegetation characteristic and explained 37.46% of variation in mean Se concentration:

$$\text{Mean [Se] in all trout} = -1.40 * \%\ \text{open conifer} + 8.36$$

Discussion

Toxicity thresholds and spatial trends

Se concentrations in sediment, water, invertebrates and fishes from my study lakes exceeded published toxicity thresholds and were generally greater than those identified in other aquatic systems. This likely has implications for the health of aquatic biota and their wildlife and human consumers in my study area.

Sediment

Of the 47 sediment sections analyzed from Johnson Lake, 44 exceeded the USEPA thresholds for impairment of bird reproduction (1-3 mg Se/kg dw) and 7 exceeded the threshold for fish reproduction (3-5 mg Se/kg dw) (United States Department of the Interior 1998). At every depth analysed, the total Se concentration in sediment from Johnson Lake exceeded typical background Se concentrations in freshwater environments of <1 mg Se/kg dw (United States Department of the Interior 1998). I use the term "background" to refer to Se concentrations in various media (*e.g.* sediment, water, invertebrates, fishes and birds) that are indicative of aquatic systems unimpacted by natural or anthropogenic sources of Se. "Background" concentrations are provided by the United States Department of the Interior (1998) with the published toxicity thresholds applied within this thesis.

Previous studies of Se-contaminated lakes have reported adverse effects in fishes at sediment concentrations from 4-12 mg Se/kg dw (Lemly 1997) or 100 mg Se/kg dw (Presser and Barnes 1984, Saiki and Lowe 1987). At coal-mining sites in Alberta, Se concentrations of creek sediment were within the range of Johnson Lake and were between 1.7 and 3.0 mg Se/kg dw whereas reference site concentrations were 0.2 mg Se/kg (Casey 2005). Therefore, the Se concentration of sediment in Johnson Lake is more similar to the concentrations in a mine-affected creek than those of a reference site.

Water

Throughout the ice free season, Se concentrations in epilimnetic water (Figure 3.3) from Johnson Lake and Lake Minnewanka remained below the threshold that can cause fish reproductive effects (2.00 μg Se/L), with the exception of Johnson Lake in May (2.53 μg Se/L). Background Se concentrations in water are typically below 0.2 μg Se/L (United States Department of the Interior 1998). Background concentrations were exceeded in all months in Lake Minnewanka and in Johnson Lake with the exception of October in Johnson Lake (Se concentration=0.1 μg/L).

Nine of the twelve lakes (including Johnson Lake and Lake Minnewanka) I sampled are reservoirs (Parks Canada 2003). Background levels of total Se in water from lakes in hydroelectric reservoirs in northwestern Ontario and northern Quebec were much lower and range from 0.01-0.18 μg Se/L (Mailman et al. 2006).

Invertebrates

Invertebrate-related toxicity guidelines (United States Department of the Interior 1998) were compared to the Se concentration of invertebrates collected for this study in Table 3.1. The background Se concentration for aquatic invertebrates is 0.4-4.5 mg Se/kg dw (United States Department of the Interior 1998). Reproductive impairment of birds and fish occurs when their diets exceed 2.9 and 4-8 mg Se/kg dw respectively (United States Department of the Interior 1998).

Se concentrations in zooplankton from Johnson Lake and Lake Minnewanka were variable throughout the ice free season and ranged from 1.16-2.55 and 0.75-2.08 mg Se/kg dw respectively. These Se concentrations were within range of background concentrations for aquatic invertebrates. Concentrations of Se in zooplankton from Johnson Lake and Lake Minnewanka were lower than those in Lake Michigan (6.00 mg Se/kg dw) (Copeland and Ayers (1972) as cited by Saiki and Lowe 1987).

The maximum Se concentration for most invertebrate orders collected seasonally from Johnson Lake and Lake Minnewanka exceeded background concentrations for aquatic invertebrates and concentrations that would cause reproductive impairment in birds that consume aquatic invertebrates. Amphipoda Se concentrations were within the range of background concentrations, although the maximum values of *Gammarus lacustris* and *Hyalella azteca* exceed the threshold to protect avian consumers. Se concentrations in Ephemeroptera in Johnson Lake (*Callibaetis*) and Lake Minnewanka (*Leptophlebia* and *Siphlonurus*) were all within the range known to cause adverse effects in their bird and fish consumers. The Se concentrations of *Agrypnia* in Johnson Lake and *Hesperophylax* in Lake Minnewanka (Trichoptera) far exceeded concentrations of Se that would affect the reproduction of their fish and bird consumers. *Anabolia* from Johnson Lake remained within the background concentration, but could have adverse effects on avian consumers. In both Johnson Lake and Lake Minnewanka, all genera of Odonata (Aeshnidae, Coenagrionidae and Libellulidae) exceed background levels and the threshold that causes sublethal effects in fish and bird consumers. Gastropod Se concentrations in Johnson Lake (*Stagnicola)* and Lake Minnewanka (*Stagnicola* and *Physa*) were within background levels throughout the ice-free season.

Among lakes, Amphipoda Se concentrations remained within range of background, but in all lakes could potentially cause bird reproductive impairment (Table 3.1). Mean Se concentrations of Ephemeroptera in all lakes (with the exception of Barrier Lake) surpassed background concentrations and far exceeded those concentrations which cause fish and avian reproductive effects (Table 3.1). Mean Trichoptera Se concentrations were above those concentrations which cause avian reproductive effects and fish sublethal effects in all lakes, with the exception of Cascade Pond (Table 3.1). Odonata mean Se concentrations exceeded the established toxicity thresholds for fish and bird diets in all lakes. Mud Lake would be the only exception, where the mean Se concentration of Odonates was very close to, but did not exceed the threshold for fish reproductive impairment (Table 3.1). Gastropoda Se concentrations remain within background for all lakes with the exception of Vermilion Lakes, where the maximum Se concentrations exceeded those which cause reproductive impairment in fish and bird consumers (Table 3.1). Hemiptera (collected in Johnson Lake only) and Oligochaeta (collected in Lake Minnewanka only) both had Se concentrations which surpassed those which cause reproductive effects in fish and avian consumers (Table 3.1).

Se concentrations of mixed invertebrates in non-impacted streams in British Columbia, Canada range from 5.14-5.55 mg Se/kg dw (Morrissey et al. 2005), which are lower than those in my study. However, Ephemeroptera (7.11-9.29 mg Se/kg dw) and Trichoptera (4.83-8.67 mg Se/kg dw) from creek reference sites in the McLeod River Basin Alberta, Canada (Wayland and Crosley 2006) are comparable to those in my study.

Fishes

The range, mean and standard error of Se concentrations in fish species from all lakes are presented in Table 3.2 and compared to published toxicity thresholds for fish and their wildlife and human consumers. The background concentration for fish muscle is between 1 and 4 mg Se/kg dw. Fish sublethal effects and teratogenic effects are observed when muscle concentrations range from 7 to 9 and 10 to 20 mg Se/kg dw respectively. Bird reproductive impairment occurs when they consume fish muscle with between 3-8 mg Se/kg dw. Human health advisories for children and pregnant women are often implemented when edible tissues exceed 8 mg Se/kg dw and a complete ban on human consumption usually occurs when concentrations surpass 20 mg Se/kg dw (United States Department of the Interior 1998, Table 3.2).

Among lakes, all species of fish had mean and maximum Se concentrations that exceeded background. Values also exceeded concentrations that cause reproductive impairment in their avian consumers (Table 3.2). Rainbow trout (collected only in Johnson Lake) mean and maximum Se concentrations exceeded thresholds for fish sublethal effects, fish teratogenesis and for human consumption by children or pregnant women (Table 3.2). In all lakes where brook trout were collected (with the exception of Vermilion Lakes) mean and maximum Se concentrations also surpassed thresholds for fish sublethal effects, fish teratogenesis and for human consumption by children or pregnant women (Table 3.2). Fish sublethal effects, fish teratogenesis and consumption by children or pregnant women thresholds were all exceeded by the maximum Se concentrations of lake trout from all lakes except Spray Lakes. Maximum Se concentrations in cutthroat trout from Mud Lake and bull trout from Upper Kananaskis Lake exceeded the threshold of fish sublethal effects (Table 3.2). In brown trout, only those from Gap Lake had mean and maximum Se concentrations which were greater than the thresholds for fish sublethal effects and consumption by children or pregnant women. Maximum values of brown trout from Gap Lake also exceeded fish teratogenic thresholds (Table 3.2). Cisco were only collected in Lake Minnewanka where maximum Se concentrations exceeded thresholds for fish sublethal and teratogenic effects and for consumption by children or pregnant women (Table 3.2). Mean and maximum Se concentrations for mountain whitefish in all lakes exceeded thresholds for fish sublethal and teratogenic effects and for consumption by children or pregnant women. In addition, maximum Se concentrations of mountain whitefish in Lake Minnewanka and Gap Lake were greater than those which would warrant a complete ban on human consumption by the US Department of the Interior (1998) (Table 3.2). Longnose sucker maximum Se concentrations from all lakes (with the exception of Barrier Lake) surpassed thresholds for fish sublethal effects. Those from Two Jack Lake and Whiteman's Pond also exceeded thresholds for fish teratogenic effects and for consumption by children or pregnant women (Table 3.2). White sucker maximum Se concentrations in all lakes (with the exception of Gap Lake) were greater than thresholds established for fish sublethal effects and for the consumption by children or pregnant women. Maximum Se concentrations in white sucker from Johnson Lake also exceeded the threshold to cause fish teratogenic effects (Table 3.2).

Se concentration in bull trout from my study (4.65-6.03 mg Se/kg dw) were within range of bull trout from creeks in an area of active coal mining in Alberta (1.2-9.4 mg Se/kg dw) and greater than those from reference sites (0.6-5.5 mg Se/kg dw) (Palace et al. 2004). Brown trout Se concentrations from my study were elevated (4.48-8.75 mg Se/kg dw) when compared to brown trout from Argentinian mountain National Park lakes (0.56-1.7 Se/kg dw) (Arribére et al. 2006). Rainbow trout Se concentrations from my study ranged from 5.15-12.07 mg Se/kg dw, which were similar to those from mining-impacted creeks in Alberta at 6.0 mg Se/kg dw (Holm et al. 2005) (converted from 1.5 mg Se/kg ww using Lemly's (1993b) 4x conversion factor from ww to dw assuming 75% moisture content; which was similar to calculated conversion factors from this study (4.66x)) and considerably greater than concentrations from reference sites (2.0 mg Se/kg dw) and from a lake in Rocky Mountain National Park in the United States (0.68-1.94 mg

Se/kg dw) (WACAP 2009 unpublished data). Rainbow trout in Argentinian mountain national park lakes were also much lower in Se (0.57-1.7 mg Se/kg dw) (Arribére et al. 2006) than those from out study. Brook trout Se concentrations in my study ranged from 4.90-10.15 mg Se/kg dw, which was less than those from mining-impacted sites (15.2 mg Se/kg dw), but greater than those from reference sites (2.4 mg Se/kg dw) (Holm et al. 2005) and from lakes in National Parks in the Pacific northwest United States (0.08-6.17 mg Se/kg dw) (WACAP 2009 unpublished data). Brook trout Se concentrations in my study were also greater than those from Argentinian mountain national park lakes (0.80-3.1 mg Se/kg dw). Cutthroat trout Se concentrations in my study ranged from 4.98-7.8 mg Se/kg dw, and were considerably greater than. Cutthroat trout from lakes in Rocky Mountain National Park in the United States (0.97-2.81 mg Se/kg dw) (WACAP 2009 unpublished data). Lake trout Se concentrations in my study ranged from 5.06-8.38 mg Se/kg dw. This range of values was similar to lake trout from Alaskan National Park lakes (0.44-11.90 mg Se/kg dw) (WACAP 2009 unpublished data). However, despite a high maximum Se concentration for lake trout from an Alaskan National Park (11.90 mg Se/kg dw), the mean values are much lower than those from my study (my study: $\bar{x}$ =7.12 mg Se/kg dw, SE=0.72; Alaskan lakes: $\bar{x}$ =3.54 mg Se/kg dw, SE=0.90).

Temporal trends

In other aquatic systems, high concentrations of Se in sediment, water, invertebrates and fishes have been related to natural (*e.g*. geologic (Presser et al. 1994)) and anthropogenic (*e.g*. combustion of fossil fuels (Lemly 1999a)) sources. Sediment cores are often used to distinguish between these two types of inputs on a broad temporal scale. Inputs of Se from the surrounding catchment to a waterbody can vary seasonally (Hillwalker et al. 2006) and impact sediment and water concentrations (Borgmann and Norwood 1999, Hillwalker et al 2006). This variability can often influence Se concentrations in biota (Wen and Carignan 2007). I have shown that Se concentrations in biota are elevated, thus, determining whether concentrations change temporally has important implications for the health of fish and their wildlife and human consumers.

Historical trends of Se in sediment

The labile Se concentration in sediment from Johnson Lake did not substantially increase from deeper to surface sediments. This suggests that anthropogenic Se concentrations to Johnson Lake have not increased substantially over time because the labile, or leachable metal concentration, corresponds to anthropogenic inputs (Katz and Kaplan 1981). Labile metals are bound to the exterior of the sediment particle and are not part of the interior lattice structure (Katz and Kaplan 1981). High Se concentrations identified in Kelly (2007) in biota are likely due to natural sources instead of anthropogenic sources as the labile Se concentration is below toxicity thresholds and does not increase with time. Although the labile Se concentration did not increase with time, total Se decreased temporally, which causes the labile concentrations to be proportionally more important in more recent sediments. This indicates that natural concentrations appear to be declining which could be linked to climate variability. Higher temperatures can decrease spring runoff (Cayan et al. 2001) and therefore result in a decrease in natural Se concentrations to a lake (Sappington 2002).

The proportion of total Se that was labile gradually increased from sections ~20-24 cm in depth. At 20-22 cm, the total Se concentration was almost entirely labile Se. Mudslides or turbidite deposits may be responsible for the substantial decrease in natural Se deposits to Johnson Lake. Trace metal concentrations in sediment can be extremely variable and toxicity is more dependent on the bioavailable, or labile metal concentration than the total metal concentration, which also includes the highly bound metals in the interior of the sediment particle (Borgmann and Norwood 1999).

Most metals have a high affinity for binding to sediment, thus even the ingestion of a small amount of sediment by aquatic invertebrates can result in high body concentrations (Borgmann and Norwood 1999). Se-rich sediment can also be a good locale for Se methylation (Jasonsmith et al. 2008) which is further encouraged by an alkaline pH and reducing conditions (Masscheleyn and Patrick 1993, Jasonsmith et al. 2008). Methylated forms of Se have been hypothesized as another source of bioavailable Se to aquatic organisms (Jasonsmith et al. 2008). In Johnson Lake, the water pH was slightly alkaline during the ice-free season ranging from 8.1 to 8.2. Therefore, Johnson Lake sediment conditions may enhance Se methylation.

Interpretation of sediment core data would have been enhanced if the sections were dated and sedimentation rates were calculated. This would have allowed for comparison with climate data over time. Speciation of the Se would also have been useful for determining sediment toxicity. Elemental (Se^0) and organoselenium are the most dominant forms of Se in sediment (Sappington 2002). Se^0 is relatively stable and insoluble, therefore not very toxic (Zhang et al. 1999), but organoselenium can accumulate in the sediment or mineralize to the most toxic form of Se, the inorganic forms (Se (IV) and Se (VI)) (Zhang et al. 1999).

Seasonal trends

The peak Se concentration observed in water collected from Johnson Lake in May is likely the result of runoff from spring snowmelt, which is a known source of increased contaminant concentrations in lake water (Barceloux 1999, Sappington 2002). The opposite seasonal trend was identified in pond water by Hillwalker et al. (2006) and creek water (Casey 2005) where Se concentrations were lower in spring than in the fall, possibly due to dilution by precipitation. I did not observe a spring pulse in the Se concentration of water from Lake Minnewanka. Possible reasons could be the differences in hydrology between the two lakes (Lake Minnewanka has regulated flow), or that I missed spring runoff in Lake Minnewanka.

Seasonal events can affect the transport of substances to a lake, which can influence the in-lake distribution of contaminants by altering pH, salinity and dissolved oxygen (Ekpo and Ibok 1998). These changes can affect water Se concentrations over longer periods of time (Hillwalker et al 2006). In addition, the uptake of Se by bacteria, algae and aquatic invertebrates can cause decreased concentrations of Se in water (Hamilton and Lemly 1999), which can lead to the false assumption that there will be no adverse effects of Se within the food web (Hamilton and Lemly 1999).

It was expected that zooplankton Se concentrations would follow the same pattern as water Se concentrations, because many zooplankton are filter feeders (Clifford 1991). However, zooplankton Se concentrations fluctuated in both Johnson Lake and Lake Minnewanka throughout the ice-free season, and only followed trends of Se in water for July through October in Johnson Lake. Se concentrations in water and zooplankton appeared to decline after fall turnover in September. Uptake of most trace metals by aquatic invertebrates is through their diet (Debruyn and Chapman 2007, Rainbow 2007), thus, low Se concentrations in water may not necessarily translate into low Se concentrations in aquatic invertebrates.

Se concentrations of amphipods and gastropods from both Johnson Lake and Lake Minnewanka were low and fluctuated very little throughout the ice-free season, similar to Se concentrations in water. Se concentrations in water samples can be variable in aquatic systems because water is sensitive to changes in transport and distribution. Therefore, water Se concentrations may not be an accurate means of predicting benthic invertebrate Se concentrations (Hillwalker et al. 2006). Amphipods are mainly detritivores and gastropods are classified as herbivorous scraper/grazers (Clifford 1991), so functional feeding group or trophic position could be responsible for low concentrations of Se in these taxa. Odonates, trichopterans and ephemeropterans contained greater Se concentration than amphipods and gastropods, which may also be related to feeding behaviour or trophic position. Odonates are generally predaceous and often consume other invertebrates (Clifford 1991). Trichopterans are primarily classified as herbivorous and omnivorous scrapers, but can be predaceous in later instars (Clifford 1991). Ephemeropterans are mainly detritivores, but can also be carnivorous (Clifford 1991). Increased Se concentrations in predaceous invertebrates are likely due to the biomagnification of Se in aquatic food webs (Saiki and Lowe 1987). Casey (2005) reported no seasonal trends in Se concentrations from June to August in invertebrates from Alberta creeks, nor any strong patterns between Se concentrations and trophic position or functional feeding group.

The sediment in Johnson Lake contained elevated concentrations of total Se, and thus may be responsible for increased Se concentrations in the invertebrate taxa that interact with the sediment. Aquatic organisms that reside at the sediment-water interface can be exposed to elevated metals concentrations, because this area contains greater concentrations of metals than the overlying water column (Borgmann and Norwood 1999). Se concentrations in sediment could also be high in other lakes, which would suggest Se inputs from a regional source. Seasonal differences in benthic invertebrate Se concentrations could also be attributed to the shedding of metallothioneins during molting (Engel 1993), reproductive status, (Hare and Campbell 1992) and prey availability (Bronmark and Hansson 2005).

Long-term trends in Selenium concentrations in organisms

Se concentrations of zooplankton from Johnson Lake and Lake Minnewanka decreased between 2000/01 and 2007. Changes in the Se concentration of benthic invertebrates during the same time period were variable, and no overwhelming trends were evident. Amphipoda and Odonata samples were nearly identical between the two sampling periods.

Although Se concentrations in Trichoptera and Odonata varied, the changes were not consistent. Lead and cadmium concentrations in aquatic invertebrates have been shown to decrease in streams in the USA over the period of one year. The decrease was attributed to an increase in stream discharge at the end of the year (Besser et al. 2007).

The Se concentrations of lake trout from Lake Minnewanka and Spray Lakes and brook trout from Johnson Lake increased significantly over time. In all three lakes, the percentage of increase was similar (~30%). Aquatic systems that have experienced high increases in Se over time in sediment, water or biota are typically related to point sources of contamination. Past studies have examined increases in Se concentrations in lakes and rivers due to mining (Hillwalker et al. 2006, Orr et al. 2006, Wayland et al. 2007), coal combustion (Lemly 2004), irrigation of seleniferous soils and the subsequent disposal of contaminated water (Presser et al. 1994, Hoffman et al. 1988) and metal smelting (Nriagu and Wong 1983). Labile Se concentrations in a sediment core from Johnson Lake did not change with depth. This could indicate that increases in Se concentrations of fishes over time are not related to increased inputs from an anthropogenic source. However, increased erosion could lead to enhanced Se deposition without altering the concentration. The increase in Se concentrations of fishes is regional, and is likely related to factors that affect Se concentrations within and among lakes.

Factors that affect Se concentrations

Differences in Se concentrations have been identified in invertebrate taxa within Lake Minnewanka and Johnson Lake. Se concentrations were variable for some invertebrate taxa but others showed little variability among lakes. Se concentrations were consistently greatest in invertebrates from Vermilion Lakes. Differences in Se concentrations by invertebrate taxa could be a result of variations in uptake and excretion, prey choice, habitat occupancy, life history and longevity (Rainbow 2002, Hillwalker et al. 2006).

Benthic invertebrates are exposed to Se from both the sediment compartment and from the water column (Hare et al. 2003).

Within lakes, differences in Se concentrations among fish species were only identified in two lakes, Lake Minnewanka and Two Jack Lake. Among lakes, ANCOVAs showed significant differences in Se concentrations for all fish species except bull trout, however Tukey's tests indicated little variability in Se concentration among lakes for the majority of fish species (*i.e.*, Se concentrations in fishes from only one or two lakes were significantly different from the other lakes). Bull trout were sampled from only two lakes, Upper and Lower Kananaskis Lakes. These two lakes are connected and therefore their watersheds and waters would share many of the same characteristics. In addition, bull trout would likely be able to swim between the two lakes.

Bioaccumulation rates

Se bioaccumulation rates (slope of the regression relationship between Se concentration and baseline-adjusted $\delta^{15}N$) differed within lakes between species in only Gap Lake, Lake Minnewanka and Two Jack Lake. Species that are often piscivorous (*i.e.,* lake trout) would be expected to have greater bioaccumulation rates than benthivorous or zooplanktivorous species (*e.g.,* white sucker, longnose sucker and mountain whitefish). In this study, I found the opposite, which could be explained by differences in diet. Differences in the bioaccumulation rate of Se in fish species among lakes were detected in mountain whitefish, brook trout, brown trout, white sucker and all trout species combined among lakes. Species-specific differences in bioaccumulation rates can be attributed to differences in the absorption and elimination rates of a metal (Veltman et al. 2008). Increased weight can be a factor that decreases the elimination rate of metals resulting in higher metals concentrations (Hendriks and Heikens 2001, Veltman et al. 2008). Water chemistry parameters can also affect the bioaccumulation of trace metals.

Metal absorption by aquatic organisms is lower at low salinity and higher in lakes that are more saline (Veltman 2008).

The y-intercept of the regression relationship between Se concentration and baseline-adjusted $\delta^{15}N$ corresponded to the baseline Se concentration in each lake. Within lakes, no significant differences were identified in the y-intercept. This was to be expected because the invertebrate baseline Se concentration would be the same for all fishes from that lake. Among lake differences in the y-intercept can be attributed to variability in baseline Se concentrations (*e.g.,* differences in Se inputs from the catchment) or differences in absorption and elimination rates of the organisms (Veltman et al. 2008). Luoma and Rainbow (2005) found that there is substantial variability in invertebrate metals bioaccumulation based on both exposure route (*i.e.*, diet vs. solution) and geochemical effects on bioavailability. Bioaccumulation of metals in aquatic organisms can also depend on the ecosystem, metal-specific chemistry and abundance, speciation of the metal and species-specific effects (*i.e.*, excretion rates) of the organism (Luoma and Rainbow 2005, Veltman et al. 2008). Baines et al. (2006) determined that the absorption rate of metals in molluscs is a function of filtration rate, which is both species-specific and temperature dependent.

$\delta^{15}N$

Trophic position was anticipated to be positively related to the Se concentration of fishes, because Se can biomagnify in aquatic food webs (Sappington 2002, Lemly 2004). Trophic position is often related to concentrations of contaminants in fishes because of biomagnification in food webs (Cabana and Rasmussen 1994). $\delta^{15}N$ was significantly different among fish species within all lakes, except for the fishes in Lower Kananaskis Lake. Lake trout from Two Jack Lake exhibited the highest baseline-adjusted $\delta^{15}N$ (trophic position) of all fish species in the lake, but the lowest Se concentration. In Lake Minnewanka, higher trophic position in fish also did not translate to a higher Se concentration. This was unexpected because Se usually increases with trophic position (Sappington 2002, Lemly 2004) (*e.g.* it is unusual for lake trout, a top predator, to have lower Se than mountain whitefish). I expected that fishes with higher $\delta^{15}N$ values would have higher Se concentrations, as this occurs with other metals, such as Hg (Cabana and Rasmussen 1994, Camusso et al. 1998), but this was not always the case in my study. Among lakes, all species except brown trout had significant differences in $\delta^{15}N$. Hendriks and Heikens (2001) found that metal concentrations in aquatic organisms did not increase with trophic position but instead by the rates or metal absorption and elimination. Lake trout from Two Jack Lake had lower concentrations of Se than both mountain whitefish and longnose sucker as well as the lowest Se bioaccumulation rates. Omnivory in food webs has been found to interfere with the relationship between of trophic position and contaminant concentrations in aquatic organisms (Kling et al. 1992, Beaudoin et al. 2001). Since omnivory in lakes often causes overlap in $\delta^{15}N$ values (Beaudoin et al. 2001), it can alter the predicted relationship between $\delta^{15}N$ and the bioaccumulation of metals.

Over time, increases in Se concentrations in fishes can also be related to increases in the concentrations of their prey items because of Se biomagnification (Besser et al. 1993, Sappington 2002, Lemly 2004). In Lake Minnewanka and Spray Lakes there was a small but significant (p<0.01, p=0.02, respectively) decrease in $\delta^{15}N$ of lake trout (Lake Minnwanka $\delta^{15}N$: 1991: $\bar{x}$ =12.40 ± 0.34, 2007: $\bar{x}$ =11.29 ± 0.11, Spray Lakes $\delta^{15}N$: 1992: $\bar{x}$ =11.09 ± 0.13; 2007: $\bar{x}$ =10.06 ± 0.35). This may indicate that over time, lake trout in these lakes began consuming food items (*i.e.* invertebrates) with a lower $\delta^{15}N$ but a higher Se concentration. The Se increase in brook trout from Johnson Lake was only over a period of six years, compared to fifteen and sixteen years in lake trout from Spray Lakes and Lake Minnewanka respectively. There was no significant (p=0.20) difference in $\delta^{15}N$ in brook trout from Johnson Lake

(2001: $\bar{x}$ =8.13 ± 0.19, 2007: $\bar{x}$ =8.37 ± 0.09) which may be the shorter time period between sampling periods or that no change in feeding behaviour occurred.

δ¹³C

Within lakes, $\delta^{13}C$ was significantly different in fish species in Lake Minnewanka, Two Jack Lake, Barrier Lake and Whiteman's Pond. Among lakes, significant differences in $\delta^{13}C$ were identified for all fish species. $\delta^{13}C$ indicates whether the carbon source of the organism originates from terrestrial or atmospheric inputs. Lower (more negative) $\delta^{13}C$ values indicate carbon sources from atmospheric inputs which are usually reflected in pelagic invertebrates, whereas higher (less negative) $\delta^{13}C$ indicate terrestrial carbon inputs reflected in littoral benthos (Post 2002). This was reflected in the differences in $\delta^{13}C$ of fish species from Lake Minnewanka. Cisco, which are often zooplanktivorous (Scott and Crossman 1973) had $\delta^{13}C$ values that were significantly more negative than omnivorous (in mountain lakes) lake trout and mountain whitefish (Scott and Crossman 1973). In Two Jack Lake, benthivorous longnose suckers (Scott and Crossman 1973) were significantly greater in $\delta^{13}C$ than omnivorous mountain whitefish (Scott and Crossman 1973).

Growth rate

It was expected that growth rate would have a negative effect on Se concentrations in fishes, because growth dilution results in incorporation of contaminants into a larger amount of tissue, reducing the overall concentration in the organism (Mailman et al. 2006). Within lakes, growth rate was not a reliable predictor of Se concentrations because growth rate was negatively related to Se concentrations in some fish species and positively related in others. Fish growth rate was positively correlated to Se concentrations in mountain whitefish among lakes. In other studies, fish growth rates were negatively related to MeHg (methyl mercury) in fishes (as reviewed by Kelly 2007), but Kelly (2007) also found a positive relationship between MeHg and growth rate of trout from Rocky Mountain lakes.

Age, length and weight

Similar to growth rate, fish age, fork length and weight were not reliable predictors of Se concentration in fishes within lakes. Some studies have observed that fish age is not related to Se concentration (Frøslie et al. 1985, Mason et al. 2000). Burger et al. (2001) determined that fish weight was positively correlated to Se concentration in several fish species. However, there was no relationship between fish length and Se concentration in fishes from Greenland (Riget and Dietz 2000), bull trout from Alberta (Palace et al. 2004) or Arctic char in Nunavut (Gantner et al. 2009). Fish fork length was perfectly negatively correlated ($r=-1.000$) with Se concentration of longnose sucker among all lakes.

Condition factor

Within lakes, condition factor was significantly different in fish species from Lake Minnewanka, Two Jack Lake and Lower Kananaskis Lake. Among lakes, mountain whitefish and longnose sucker condition factors were significantly different among lakes. Condition factor is a measure of “fish girth” and reflects the energy stores and overall condition of a fish (Eastwood and Couture 2001). Condition factor was negatively related to Se concentrations in sunfish from Belews Lake (Sorensen and Bauer 1984). In lakes in Ontario, Canada, low fish growth rates translated into lower condition factors. Fishes with lower condition factors had higher concentrations of metals when compared to their “healthier” counterparts (Eastwood and Couture 2001).

Within lakes

Se in water

There was no correlation between Se concentrations in water and Se concentrations in zooplankton or benthic invertebrates in either Johnson Lake or Lake Minnewanka. This implies that benthic invertebrates, some of which have elevated Se concentrations, are acquiring their Se from a different source, such as sediment or diet or that they have mechanisms of retention, adsorption and excretion that prevent such correlations. Detritivores have been found to have higher concentrations of Se than pelagic feeders based on their close contact with the sediment (Jasonsmith et al. 2008).

Other Metals

In Johnson Lake, Se concentrations in water were significantly and positively correlated with 20 metals, including arsenic (As), zinc (Zn), antimony (Sb), and tungsten (W). These metals in particular have been identified as having a negative interaction with Se in fish (Jasonsmith et al. 2008). In Lake Minnewanka, five of six correlated metals were related negatively to Se concentrations in water. This included a negative relationship between Se and Zn concentrations in water, which may translate to the same relationship in fishes. In other lakes, there were some significant correlations of Se concentrations in fishes to concentrations of other metals. Excess dietary carbohydrate (Hamilton 2004) and dietary differences (Jasonsmith et al. 2008) have been suggested to increase the toxicity and absorption of Se to fishes. In addition, the interactions of other metals (*e.g.*, Hg, As, bismuth (Bi), Sb, copper (Cu), cadmium (Cd), germanium (Ge), silver (Ag), Zn and W) can decrease the toxicity and bioavailability of Se in fishes (Jasonsmith et al., 2008). In my study, longnose sucker in some lakes had Se concentrations that were significantly correlated with Cd concentrations, but as a strongly positive relationship. In addition, Se concentrations were significantly positively correlated with Ag concentrations in white sucker in some lakes. Within lakes, some of these metals were correlated with Se concentrations, but results were highly variable and some relationships were positive, while others were negative.

Water Chemistry

Water chemistry variables can influence Se concentrations in water. Sulphate concentration in water has been shown to have an antagonistic effect on the uptake and toxicity of selenate (Se (VI)) in aquatic organisms (Sappington 2002). Only total Se was measured for my study, and it was not significantly correlated with sulphate concentrations in water from Johnson Lake or Lake Minnewanka. Highly saline waters can increase the uptake of Se by organisms (Veltman et al 2008). In my study, there were no significant correlations between water chemistry parameters and water Se concentrations in Lake Minnewanka. In Johnson Lake, sulphate content was not related to Se concentrations in water, but chloride anions were negatively correlated. Salinity (measured as conductivity) was also uncorrelated to Se concentrations in water.

Zooplankton Density

Zooplankton (cyclopoid copepod) density was positively correlated with the Se concentration of zooplankton in Johnson Lake. The dominant form of zooplankton changed seasonally from cladocerans to cyclopoid copepods in Johnson Lake, as indicated by monthly measurements of zooplankton density. Despite this seasonal shift, Se concentrations of bulk zooplankton samples from Johnson Lake were only related to the density of cyclopoid copepods.

Se in Invertebrates

In Johnson Lake and Lake Minnewanka, zooplankton and invertebrate Se concentrations were not related to $\delta^{13}C$ or $\delta^{15}N$ with the exception of

Gammarus lacustris in Lake Minnewanka. These results are supported by Jasonsmith et al. (2008), who found that Se concentrations in insects are a reflection of internally absorbed and externally adsorbed Se and not trophic position.

Multiple regression model

Multiple regression models that included significant predictors of Se concentrations were completed, including only independent variables that were not correlated. Growth rate, trophic position, weight, age and condition factor were significant predictors of Se concentrations in fishes within lakes. Growth rate was both positively (longnose sucker from Whiteman's Pond and mountain whitefish from Gap Lake) and negatively (white sucker from Johnson Lake) related to Se concentration in fishes. $\delta^{15}N$ was associated both positively (bull trout from Lower Kananaskis Lake) and negatively (mountain whitefish from Gap Lake) to the Se concentration of fishes within lakes. Fish weight was a significant predictor of Se concentrations in white sucker from Johnson Lake and lake trout from Spray Lakes. Age was positively related to Se concentrations in longnose sucker from Whiteman's Pond. I found that condition factor was related negatively to Se concentrations of bull trout from Lower Kananaskis Lake and brown trout from Whiteman's Pond brown trout.

Among lakes

Geology

Some geologic formations were negatively correlated to Se concentration in both brown trout and mountain whitefish, however, none were related positively. It was expected that catchment geology that was comprised of Se-rich shales (Wang and Gao 2001, Wen and Qiu 2002) and sandstones (Wang and Gao 2001) would cause greater amounts of Se to be transported to lakes and subsequently to fishes. Past studies have found that catchment geology was responsible for high Se in some aquatic systems (Presser et al. 1994, Glozier et al. 2001), and high Se concentrations in my study area have been related previously to geologic formations (Glozier et al. 2001). Perhaps the geology data I used were too coarse to identify this relationship.

Vegetation

The percentages of open conifer and dense broadleaf vegetation were negatively correlated to Se concentrations of all trout species combined and mountain whitefish. The percentage of dense conifer within a catchment was not related to Se concentrations in fishes. Trees are known to enhance capture of atmospheric pollutants (Tyler 1984, McGee et al. 2007), so a decrease in the density of conifer stands in the watershed would result in more Se reaching open soils of the watershed and subsequently the lake through atmospheric deposition. Conifers collect airborne pollutants and these contaminants can adhere to the needles of the tree (Tyler 1984, McGee et al. 2007). Conifers are better scavengers of atmospheric contaminants than deciduous trees because their needles are retained all year and they are more efficient in capturing particles than leaves (Lovett 1994, Weathers et al. 2000). Fewer contaminant particles would adhere to the needles of an open conifer stand than if the stand were dense. Increased percentages of dense broadleaf stands in lake watersheds could have a negative effect on Se concentrations in fishes because of the increased interception of atmospherically deposited contaminants by the leaves (Weathers et al. 2000).

Climate

Climate variables were both positively and negatively correlated to fish Se concentrations among lakes. Mean winter precipitation (likely as snow) from November to February was negatively correlated to Se concentrations in lakes. Snow is a better scavenger of metals from the air than rain (Moldovan et al. 2007). This negative correlation is likely because ice cover would prevent precipitation from entering lakes in winter. Thus, metals may only reach lakes during spring melt runoff. Spring precipitation (likely as rain) is positively correlated to Se concentrations. This was expected as Se would be entering lakes in the spring rather than the winter through direct deposition in addition to catchment deposition where runoff can access the ice-free lake (Veysseyre et al. 2001).

Temperature was both positively and negatively correlated with Se concentrations in fish species among lakes. Seasonal variations in temperature can be related to changes in Se deposition to a catchment or lake (Wen and Carignan 2007), thus possibly resulting in both positive and negative relationships with fish Se concentrations.

Watershed area

Watershed area was negatively correlated to Se concentration in brown trout among lakes. This implies that brown trout in lakes with smaller watersheds have higher Se concentrations. Hg concentrations in Swedish lakes also showed a negative relationship between watershed area and fish Hg concentrations (Håkanson et al. 1988).

Wetlands

The Vermilion Lakes are wetlands (Parks Canada 2003), and are the largest wetland area in the Bow Valley (Fedje et al. 1995). Wetlands have a high retention potential for trace metals (Gambrell 1994). Plants and animals strongly bioaccumulate Se in wetlands (Lemly and Ohlendorf 2002). Therefore, it follows that the Vermilion Lakes are subject to higher Se concentrations in biota than other study sites based on its wetland characteristics, as identified for invertebrates.

Reservoirs

Mercury (Hg), like Se, also bioaccumulates in aquatic organisms and biomagnifies in food webs (as reviewed by Kelly 2007). When terrestrial ecosystems are flooded (*i.e.* for hydroelectric dams), toxic MeHg is produced and bioaccumulates in fishes (Kelly et al. 1997, St. Louis et al. 2004, Mailman et al. 2006). Se addition has been hypothesized as a tool to lower Hg concentrations in reservoirs (Mailman et al. 2006). Se can be methylated and volatilized to the atmosphere (Mailman et al. 2006) during reservoir creation, resulting in low Se concentrations in some reservoirs, but other studies have found that methylation of Se creates an additional form of Se that is bioavailable to organisms in the water column (Jasonsmith 2008). Nine of the twelve lakes in this study are reservoirs (Cascade Pond, Two Jack Lake, Johnson Lake, Lake Minnewanka, Whiteman's Pond, Spray Lakes, Upper Kananaskis Lake, Lower Kananaskis Lake and Barrier Lake) and fishes from Johnson Lake, Lake Minnewanka and Spray Lakes have low concentrations of Hg (Kelly 2007), but high concentrations of Se. In other areas of the Canadian Rocky Mountain Parks, including some lakes in Banff National Park, Hg concentrations in fishes are high enough that consumption advisories have been issued (Kelly 2007), but this is not the case in my study area.

Multiple regression model

I developed significant multiple regression models (including only independent variables that were significantly related to Se concentrations in fishes) for Se concentrations of all trout species combined and mountain whitefish among lakes. Vegetation type and fish weight were the only variables that were significant predictors of Se concentrations among lakes. The percentage of the watershed covered in open conifer was negatively associated with Se concentrations for both all trout species combined and mountain whitefish. In addition, the percentage of dense broadleaf trees within the watershed and fish weight both had a positive relationship with Se concentrations in mountain whitefish. Thus results were variable when considered for all species among lakes.

In summary, concentrations of Se in water, sediment, invertebrates and fishes exceeded published toxicity thresholds. Se concentrations in these media were generally greater than those found in other studies, and often similar to concentrations at anthropogenically impacted sites. The health of fishes and their human and wildlife consumers are likely at risk. Elevated concentrations of Se in sediments, and therefore the benthic food web, likely play an important role in the high Se concentrations of several invertebrate taxa and fish species identified by Kelly (2007) and during this study. The relationships between factors that affect Se concentrations in fishes were variable within and among lakes, indicating that importance of species and lake specific factors. Within and among lake predictive multiple regression models identified several variables that were related to Se concentrations in fishes. However, significant models could not be created for many fish species within and among lakes. Within lakes, significant predictors included growth rate, weight, condition factor, trophic position and age. These predictors were inconsistent within lakes and between species, indicating that Se concentrations of fishes were influenced by species and lake specific factors. Among lakes, significant multiple regression models could only be attained for all trout species combined and mountain whitefish. This was another indicator that Se concentrations in fishes were related to lake specific factors. In both significant models, % open conifer in the catchment was related negatively to the Se concentrations of all trout species combined and mountain whitefish. Weight and % dense broadleaf were also predictors of Se concentrations in fishes. Omnivory, and other factors such as the lack of fine scale geology data or not measuring and including important predictors in the analysis may have confounded within and among lake modelling attempts. The similar temporal increases in Se concentrations of brook trout and lake trout from three different lakes are indicative of a regional effect. Analysis of climate data over time may be a useful next step. Continued temporal monitoring should be implemented because of the implications for wildlife and human health.

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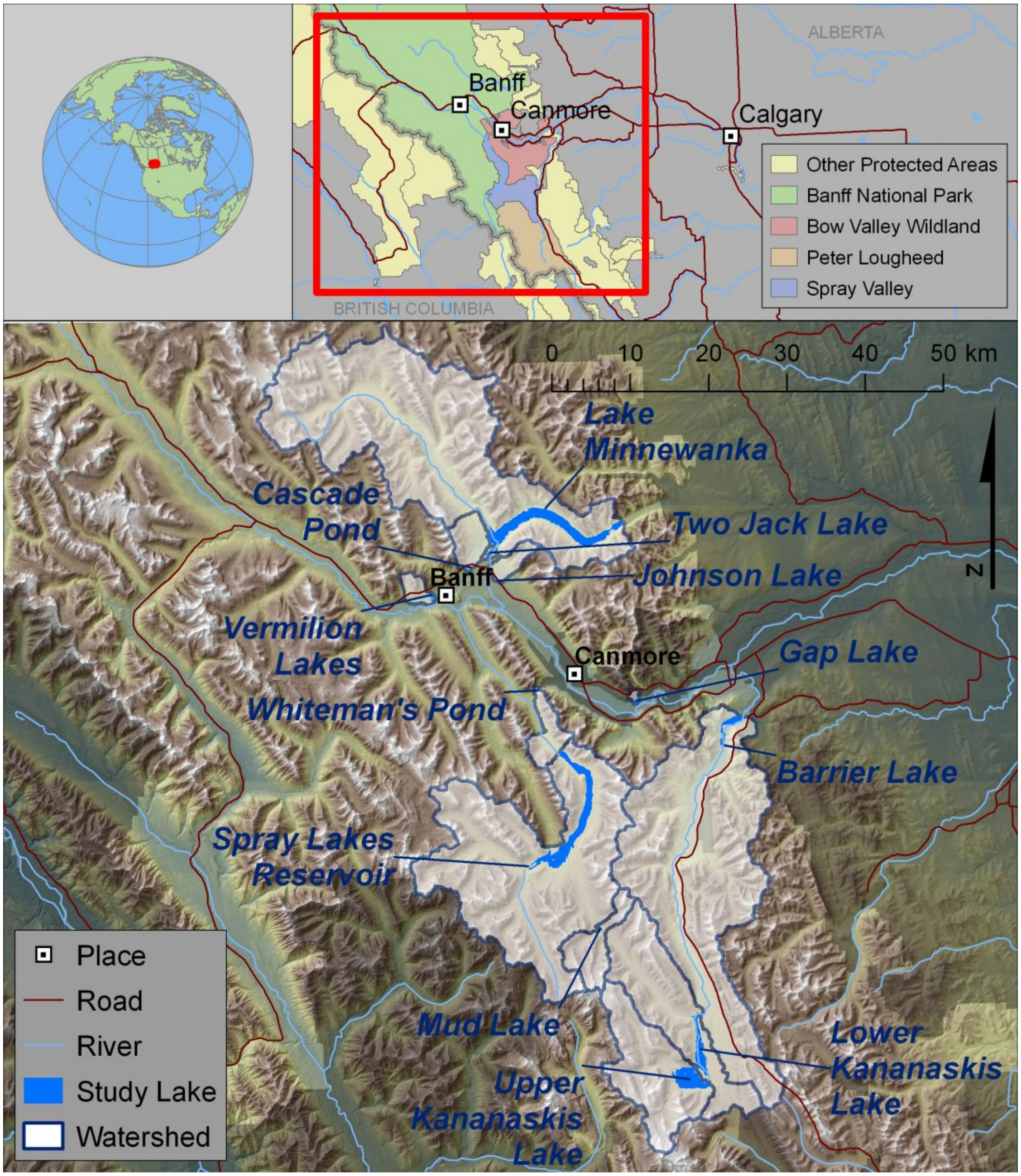


Figure 3.1 Map of the study area. Data were collected from lakes in Banff National Park, Bow Valley Wildland Provincial Park, Peter Lougheed Provincial Park and Spray Valley Provincial Park.

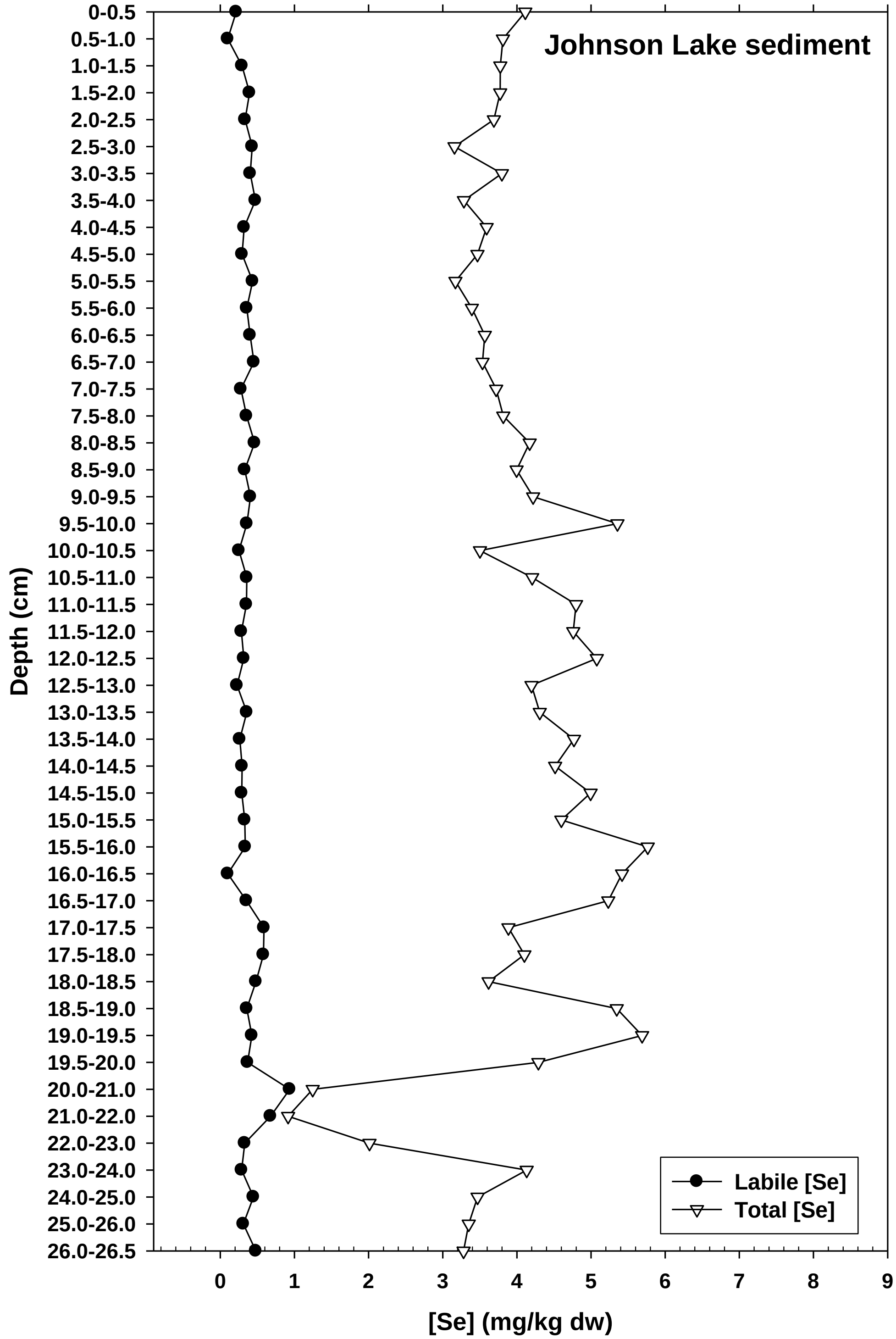


Figure 3.2 Total and labile selenium (Se) concentrations in sediment from a core collected at Johnson Lake in February 2007

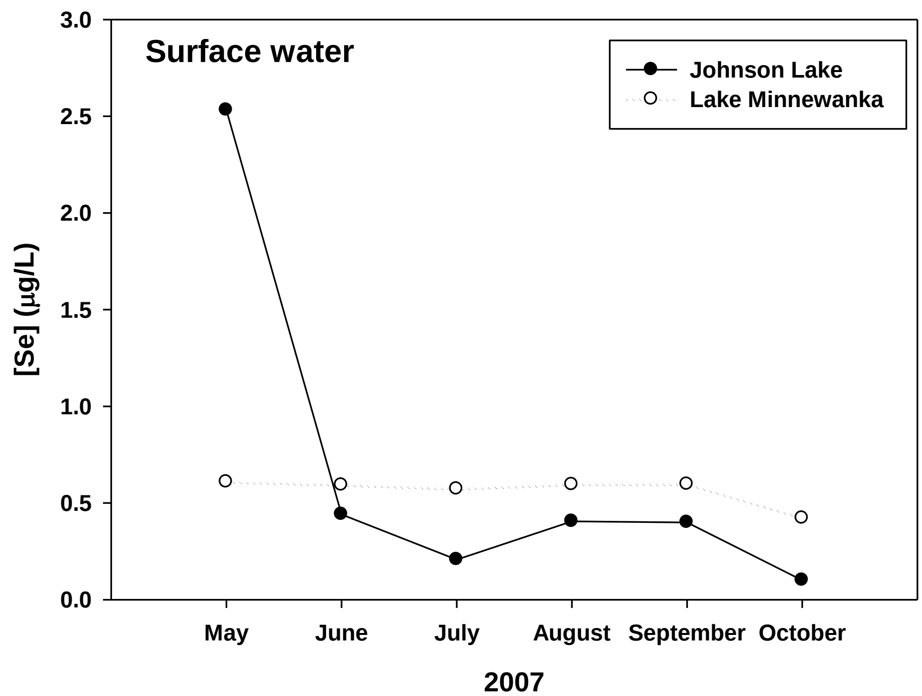


Figure 3.3 Selenium (Se) concentrations in surface water collected monthly from Johnson Lake and Lake Minnewanka throughout the ice-free season (May-October 2007)

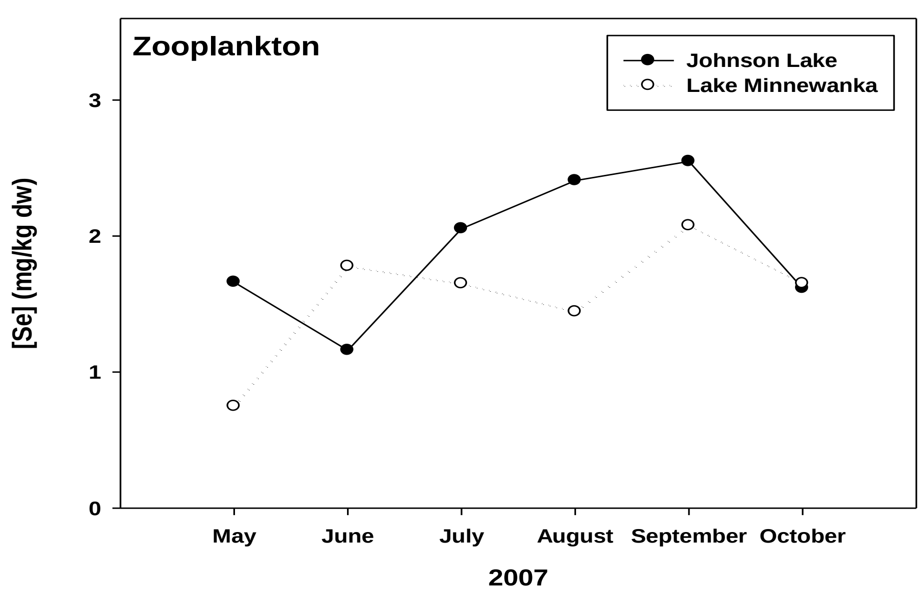


Figure 3.4 Selenium (Se) concentrations in bulk zooplankton collected monthly from Johnson Lake and Lake Minnewanka throughout the ice-free season (May-October2007)

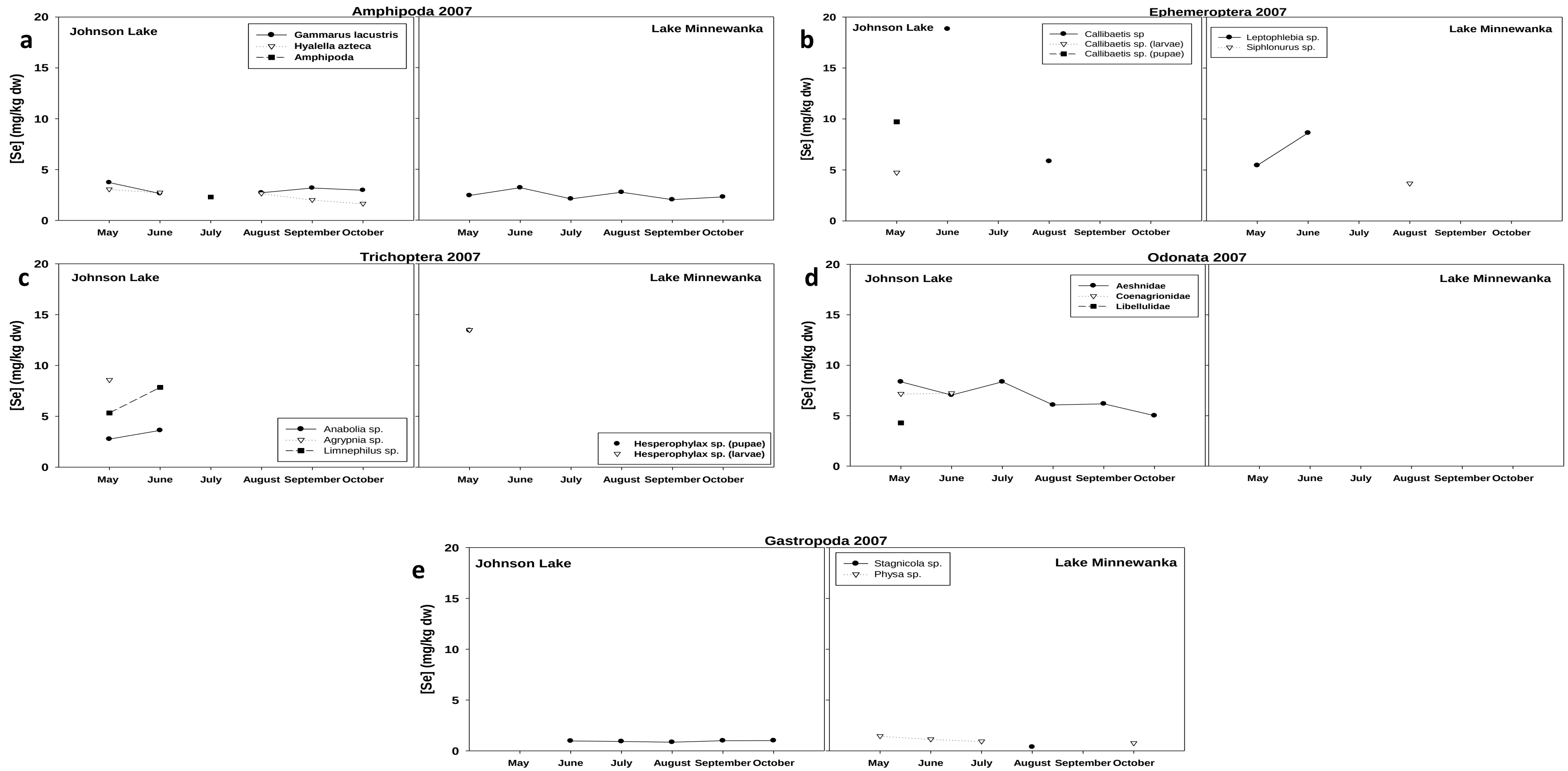


Figure 3.5 Selenium (Se) concentrations of benthic macroinvertebrates including: (a) Amphipoda, (b) Ephemeroptera, (c) Trichoptera, (d) Odonata and (e) Gastropoda collected monthly from Johnson Lake (left panel) and Lake Minnewanka (right panel) throughout the ice-free season (May-October 2007). Note: Macroinvertebrates from each Order were sometimes not collected at all sampling periods, and no odonates were available from Lake Minnewanka.

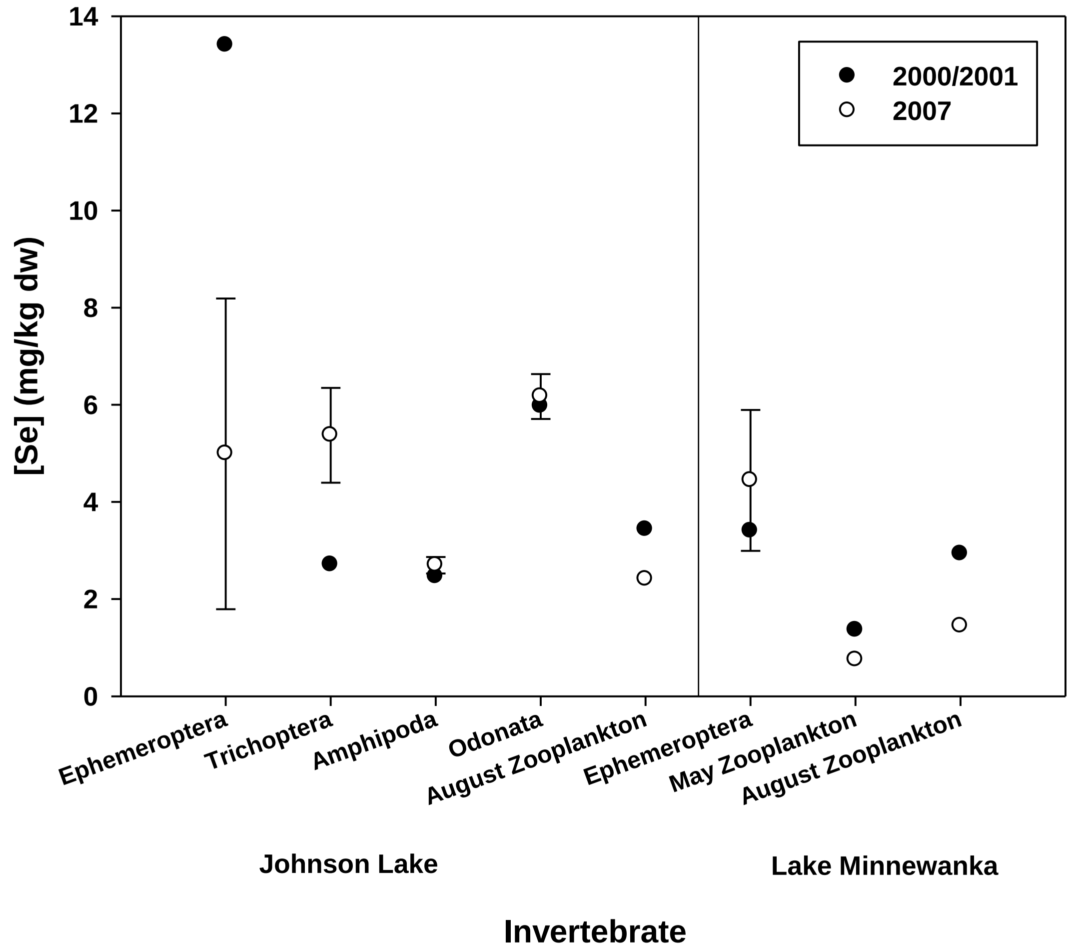


Figure 3.6 Selenium (Se) concentrations in invertebrates collected from Johnson Lake and Lake Minnewanka during the same month in 2000/01 (Kelly 2007) and 2007. Error bars represent standard error. Sample sizes in 2007= Johnson Lake: Ephemeroptera (n=4), Trichoptera (n=3), Amphipoda (n=11), Odonata (n=9); Lake Minnewanka: Ephemeroptera (n=3).

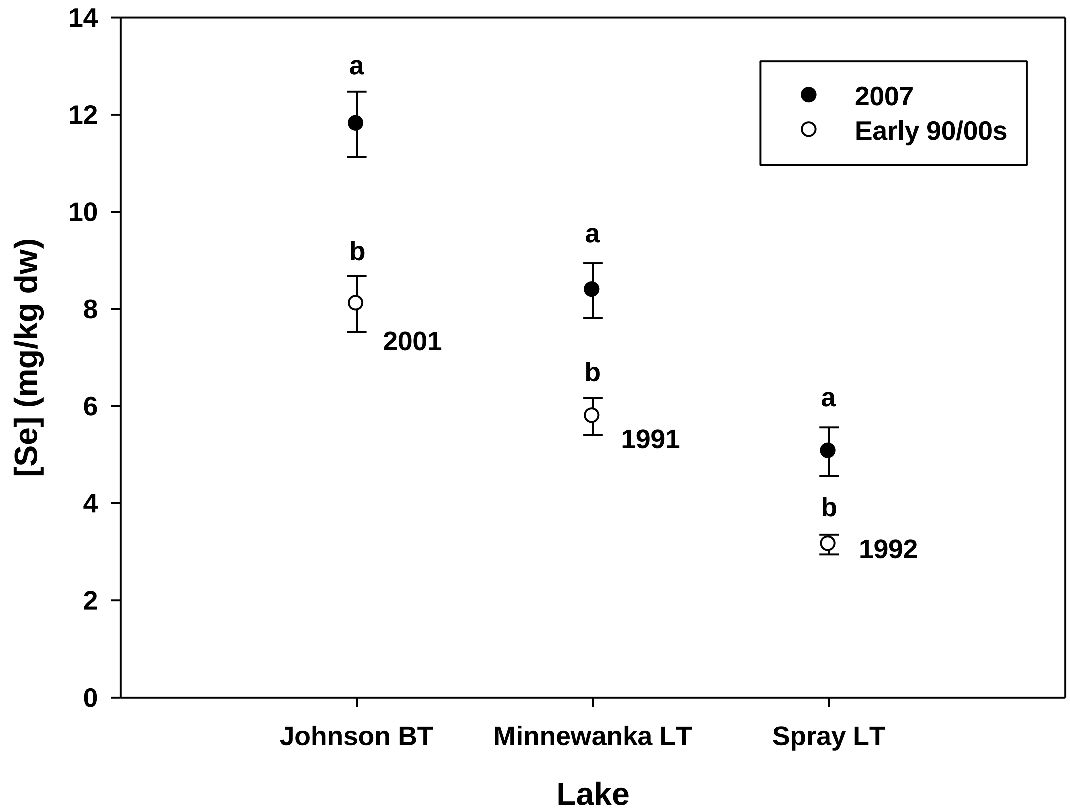


Figure 3.7 Selenium (Se) concentrations in fishes from Johnson Lake (BT=brook trout), Lake Minnewanka (LT=lake trout) and Spray Lakes (LT=lake trout) in the early 1990/00s (Kelly 2007) and 2007. Within lakes, all pairwise comparisons between years were significantly different (Tukey's Test, $p<0.05$). Letters indicate significant pairwise differences within lakes between years. Error bars represent standard error. Sample sizes = Johnson Lake BT: 2001 (n=11), 2007 (n=30), Lake Minnewanka LT: 1991 (n=10), 2007 (n=29), Spray Lakes LT: 1992 (n=7) 2007 (n=7).

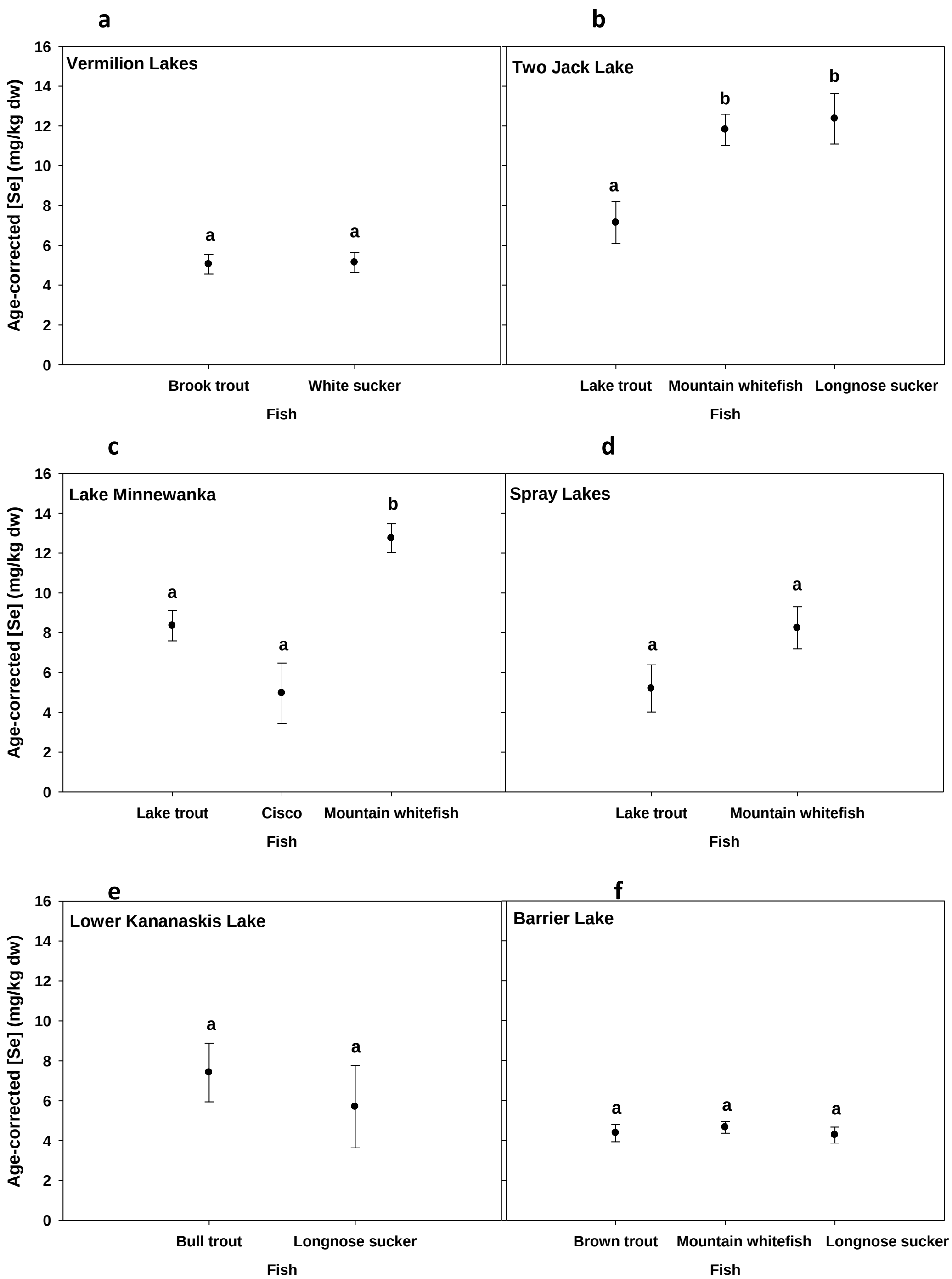


Figure 3.8 Within lake comparison of age-corrected-least squares mean selenium (Se) concentrations among fish species for: a) Vermilion Lakes, b) Two Jack Lake, c) Lake Minnewanka, d) Spray Lakes, e) Lower Kananaskis Lake and f) Barrier Lake. Only lakes in which ANCOVAs and subsequent Tukey's Tests could be performed without violation of assumptions are presented. Letters indicate significant pairwise differences based on the results of Tukey's Tests ($p<0.05$). See methods for species-specific standard age.

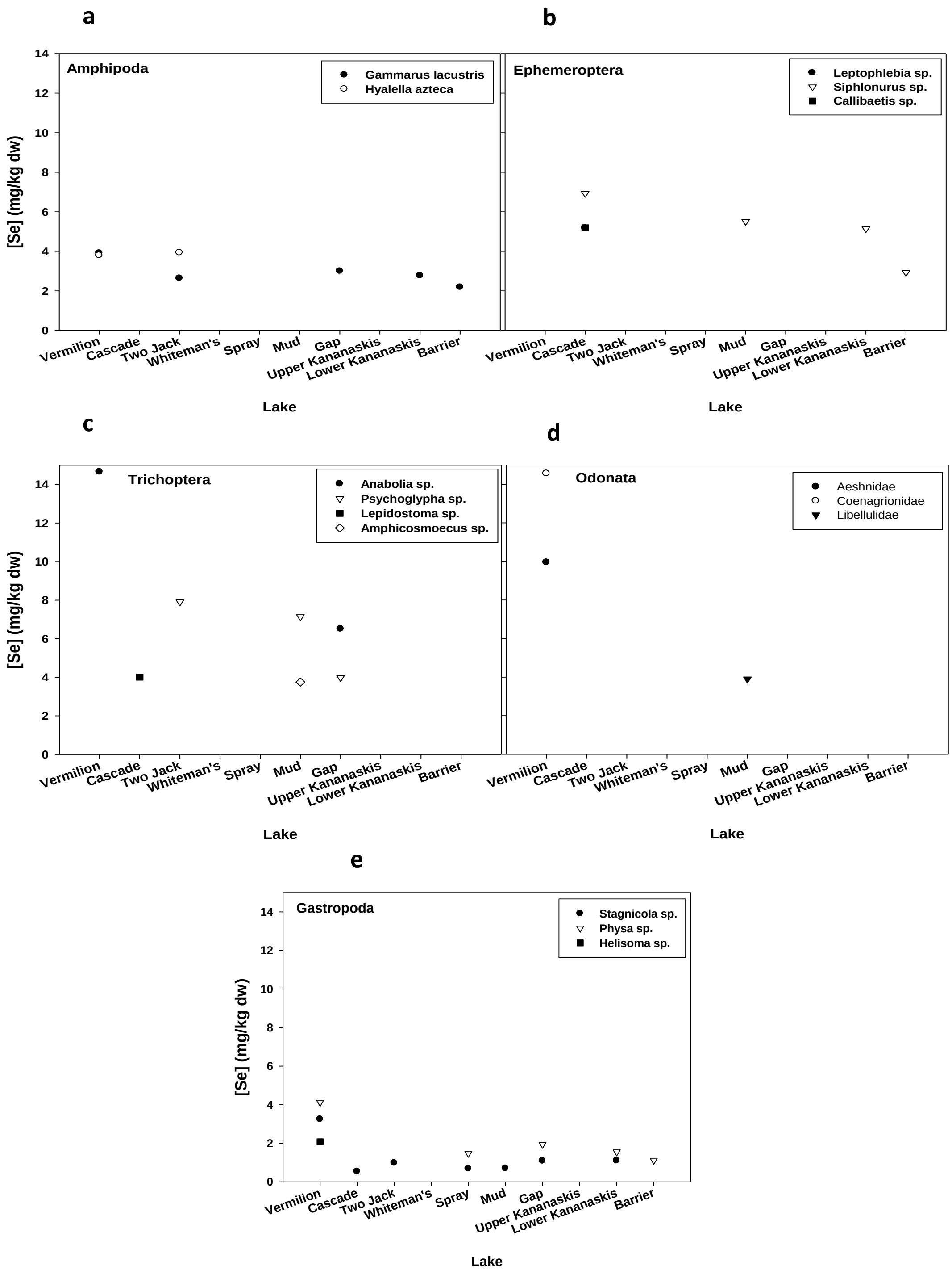


Figure 3.9 Selenium (Se) concentrations in benthic macroinvertebrates (a) Amphipoda, (b) Ephemeroptera, (c) Trichoptera, (d) Odonata and (e) Gastropoda collected from other study lakes in August 2007. Lakes are arranged on the x-axis in order from east to west. Where no data point exists, invertebrates of that order were not present at the site at the time of sampling. Note: Cascade Pond has data overlap of *Leptophlebia* and *Callibaetis*.

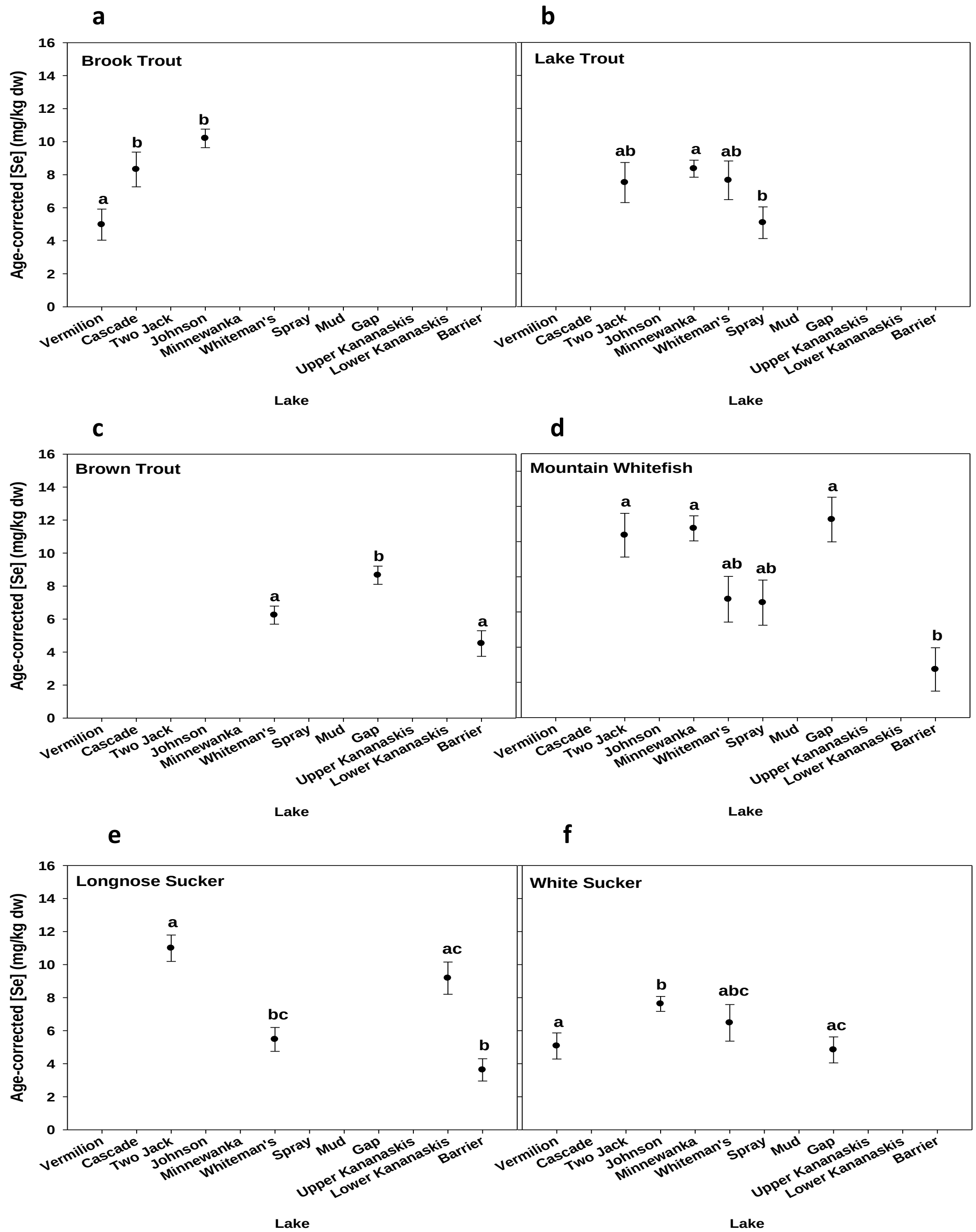


Figure 3.10 Among lake comparison of age-corrected least-squares mean selenium (Se) concentrations of: a) brook trout, b) lake trout, c) brown trout, d) mountain whitefish, e) longnose sucker and f) white sucker. Letters indicate significant pairwise differences based on the results of Tukey's Tests ($p<0.05$).

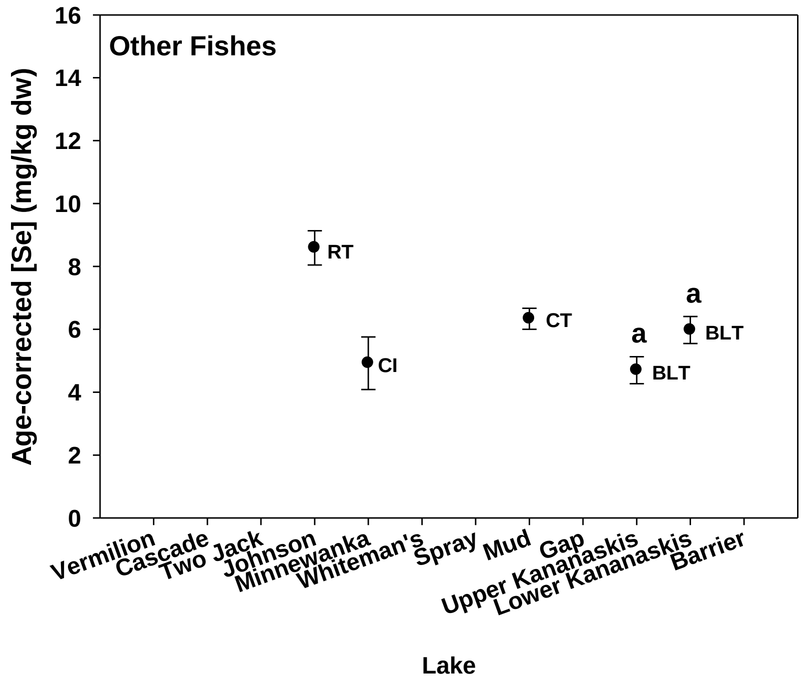


Figure 3.11 Age-corrected least-squares mean Selenium concentrations of all other fish species among lakes. Among lakes, the Se concentration in bull trout (BLT) was significantly different (Tukey's Test, $p<0.05$). Letters indicate significant pairwise differences. Since rainbow trout (RT), cisco (CI) and cutthroat trout (CT) were each collected from only one lake, therefore, among lake pairwise comparisons were not possible.

Table 3.1 Selenium (Se) concentrations (mean ± standard error) of invertebrates compared to toxicity thresholds[1]. If the mean Se concentration exceeds the threshold, "$\bar{x}$ "appears in the threshold column that has been exceeded. If the maximum range value exceeds a threshold, "max" appears in the column of the threshold.

Order	Lake	n	Se (mg/kg dw) range	Se (mg/kg dw) mean ± SE	Background for aquatic invertebrates (0.4-4.5 mg/kg dw)	Bird diet causing avian reproductive impairment (2.9 mg/kg dw)	Fish diet causing fish sublethal effects (4-8 mg/kg dw)
Amphipoda	All study lakes	7	2.19 – 3.93	2.99 ± 0.25		$\bar{x}$, max	
	Johnson	11	1.66 – 3.72	2.70 ± 0.17		$\bar{x}$,max	
	Minnewanka	6	2.01 – 3.19	2.46 ± 0.18		$\bar{x}$,max	
	Vermilion	1		3.79		$\bar{x}$,max	
	Two Jack	1		3.93		$\bar{x}$,max	
	Gap	1		3.00		$\bar{x}$,max	
	Lower Kananaskis	1		2.78			
	Barrier	1		2.19			
Trichoptera	All study lakes	7	4.00 – 14.65	8.04 ± 1.62	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max
	Johnson	3	2.76 – 8.58	5.56 ± 1.69	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max
	Minnewanka	2	13.44 – 13.49	13.47 ± 0.02	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max
	Vermilion	1		14.65	$\bar{x}$,max	max	$\bar{x}$,max
	Cascade	1		4.01		max	max
	Two Jack	1		7.91	max	max	max
	Mud	2	3.75 – 7.14	5.45 ± 1.70	$\bar{x}$,max	$\bar{x}$, max	$\bar{x}$, max
	Gap	2	3.99 ± 6.50	5.25 ± 1.26	$\bar{x}$,max	$\bar{x}$, max	$\bar{x}$, max
Coleoptera	All study lakes	2	2.17 – 9.31	5.74 ± 3.57	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max
	Johnson	4	0.90 – 3.62	2.27 ± 0.66		max	
	Lower Kananaskis	1		9.31	max	max	max
Odonata	All study lakes	4	3.90 – 12.26	8.03 ± 1.79	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max
	Johnson	9	4.28 – 8.36	6.63 ± 0.46	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max

	Vermilion	2	9.95 – 14.56	12.26 ± 2.31	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max
	Mud	1		3.90		$\bar{x}$,max	
	Lower Kananaskis	1		9.31	max	max	max
Ephemeroptera	All study lakes	6	2.93 – 11.47	6.13 ± 1.16	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max
	Johnson	4	4.75 – 18.83	9.79 ± 3.20	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max
	Minnewanka	3	3.68 – 8.37	5.92 ± 1.45	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max
	Cascade	3	5.18 – 6.93	5.77 ± 0.58	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max
	Mud	1		5.52	$\bar{x}$,max	$\bar{x}$,max	$\bar{x}$,max
	Lower Kananaskis	1		5.14	max	max	max
	Barrier	1		2.93		max	
Gastropoda	All study lakes	10	0.70 – 3.15	1.30 ± 0.22		max	
	Johnson	5	0.85 – 1.01	0.95 ± 0.03			
	Minnewanka	5	0.37 – 1.46	0.94 ± 0.18			
	Vermilion	3	2.08 – 4.13	3.15 ± 0.59		max	max
	Cascade	1		1.28			
	Two Jack	1		0.98			
	Spray	2	0.69 – 1.48	1.08 ± 0.40			
	Mud	1		0.70			
	Gap	2	1.09 – 1.95	1.52 ± 0.43			
	Lower Kananaskis	2	1.10 – 1.55	1.33 ± 0.23			
	Barrier	1		1.17			
Hemiptera	Johnson	2	0.42 – 7.62	4.02 ± 3.60	max	$\bar{x}$, max	$\bar{x}$, max
Oligochaeta	Minnewanka	2	10.16 – 10.29	10.23 ± 0.06	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max
Zooplankton	All study lakes	2	1.56 – 1.91	1.73 ± 0.18			
	Johnson	6	1.16 – 2.55	1.91 ± 0.22	max		
	Minnewanka	6	0.75 – 2.08	1.56 ± 0.18	max		

[1]Thresholds established by the United States Department of the Interior (1998).

Table 3.2 Selenium (Se) concentrations (mean ± standard error) of fishes compared to toxicity thresholds[1]. If the mean Se concentration exceeds the threshold, "$\bar{x}$ "appears in the threshold column that has been exceeded. If the maximum range value exceeds a threshold, "max" appears in the column of the threshold.

Species	Lake	n	Se (mg/kg dw) range	Se (mg/kg dw) mean ± SE	Background for fishes (<1 – 4 mg/kg dw)	Fish sublethal effects (7-9 mg/kg dw)	Fish teratogenic effects (10-20 mg/kg dw)	Bird consumer reproductive impairment (diet of 3-8 mg/kg dw)	Edible tissue: limited consumption advisories (no children or pregnant women) (8 mg/kg dw)[2]	Edible tissue: ban on human consumption (20 mg/kg dw)[2]
Rainbow trout	Johnson	20	5.15 - 12.07	8.59 ± 0.55	$\bar{x}$, max	$\bar{x}$, max	max	$\bar{x}$, max	$\bar{x}$, max	
Brook trout	All study lakes	3	4.90 – 10.15	8.22 ± 1.67	$\bar{x}$, max	$\bar{x}$, max	max	$\bar{x}$, max	$\bar{x}$, max	
	Johnson	30	5.24 – 16.28	9.62 ± 0.67	$\bar{x}$, max	$\bar{x}$, max	max	$\bar{x}$, max	$\bar{x}$, max	
	Vermilion	10	3.92 – 6.32	4.90 ± 0.28	$\bar{x}$, max			$\bar{x}$, max		
	Cascade	10	5.14 – 17.04	10.15 ± 1.26	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	
Lake trout	All study lakes	4	5.06 – 8.38	7.12 ± 0.72	$\bar{x}$, max	$\bar{x}$, max	max	$\bar{x}$, max	max	
	Minnewanka	29	4.21 – 14.26	8.38 ± 0.56	$\bar{x}$, max	$\bar{x}$, max	max	$\bar{x}$, max	$\bar{x}$, max	
	Two Jack	5	5.50 – 10.99	7.50 ± 0.97	$\bar{x}$, max	$\bar{x}$, max	max	$\bar{x}$, max	max	
	Whiteman's	6	4.60 – 10.55	7.56 ± 0.95	$\bar{x}$, max	$\bar{x}$, max	max	$\bar{x}$, max	max	
	Spray	8	3.04 – 7.79	5.06 ± 0.50	$\bar{x}$, max	max		$\bar{x}$, max		
Cutthroat trout	Mud	9	4.98 – 7.83	6.33 ± 0.33	$\bar{x}$, max	max		$\bar{x}$, max		
Bull trout	All study lakes	2	4.65 – 6.03	5.34 ± 0.69	$\bar{x}$, max			$\bar{x}$, max		
	Upper Kananaskis	10	4.19 – 9.27	6.03 ± 0.50	$\bar{x}$, max	max		$\bar{x}$, max		
	Lower Kananaskis	10	3.55 – 5.71	4.65 ± 0.26	$\bar{x}$, max			$\bar{x}$, max		
Brown trout	All study lakes	3	4.48 – 8.75	6.47 ± 1.24	$\bar{x}$, max	max		$\bar{x}$, max	max	
	Whiteman's	10	4.56 – 9.43	6.17 ± 0.49	$\bar{x}$, max	max		$\bar{x}$, max	max	
	Gap	10	6.11 – 12.78	8.75 ± 0.73	$\bar{x}$, max	$\bar{x}$, max	max	$\bar{x}$, max	$\bar{x}$, max	
	Barrier	5	3.45 – 5.33	4.48 ± 0.36	$\bar{x}$, max			$\bar{x}$, max		
Cisco	Minnewanka	7	3.33 – 8.40	4.92 ± 0.84	$\bar{x}$, max	max		$\bar{x}$, max	max	
Mountain whitefish	All study lakes	6	4.71 – 13.40	10.07 ± 1.37	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	
	Minnewanka	30	5.13 – 25.02	12.73 ± 0.87	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	max

	Two Jack	10	6.93 – 15.66	12.29 ± 0.79	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	
	Whiteman's	10	5.95 – 13.03	8.92 ± 0.68	$\bar{x}$, max	$\bar{x}$, max	max	$\bar{x}$, max	$\bar{x}$, max	
	Spray	10	4.19 – 16.50	8.36 ± 1.35	$\bar{x}$, max	$\bar{x}$, max	max	$\bar{x}$, max	$\bar{x}$, max	
	Gap	10	9.14 – 20.63	13.40 ±1.43	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	max
	Barrier	10	3.44 – 5.84	4.71 ± 0.25	$\bar{x}$, max			$\bar{x}$, max		
Longnose sucker	All study lakes	4	4.14 – 11.07	7.28 ± 1.46	$\bar{x}$, max	$\bar{x}$, max	max	$\bar{x}$, max	max	
	Two Jack	5	6.83 – 13.84	11.07 ± 1.15	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	$\bar{x}$, max	
	Whiteman's	8	4.15 – 11.68	6.22 ± 0.85	$\bar{x}$, max	max	max	$\bar{x}$, max	max	
	Lower Kananaskis	7	5.72 – 9.72	7.68 ± 0.58	$\bar{x}$, max	$\bar{x}$, max		$\bar{x}$, max	max	
	Barrier	8	3.17 – 5.84	4.14 ± 0.35	$\bar{x}$, max			$\bar{x}$, max		
White sucker	All study lakes	4	4.66 – 7.59	6.03 ± 0.65	$\bar{x}$, max	max		$\bar{x}$, max		
	Johnson	30	3.56 – 16.26	7.59 ± 0.54	$\bar{x}$, max	$\bar{x}$, max	max	$\bar{x}$, max	max	
	Vermilion	10	3.00 – 8.78	5.30 ± 0.580	$\bar{x}$, max	max		$\bar{x}$, max	max	
	Whiteman's	5	4.12 – 9.24	6.55 ± 1.07	$\bar{x}$, max	max		$\bar{x}$, max	max	
	Gap	10	2.11 – 7.16	4.66 ± 0.49	$\bar{x}$, max	max		$\bar{x}$, max		

[1]Thresholds established by the United States Department of the Interior (1998).

[2]Converted from wet weight to dry weight based on 4x conversion factor assuming 75% moisture content of fish tissue (Lemly 1993b)

Table 3.3 Slope and y-intercept of the linear regression between Se concentrations and $\delta^{15}N$ in invertebrates and fishes by species and lake. The slope is an indicator of the rate of Se bioaccumulation and the y-intercept is a measure of baseline Se concentrations. (BRT=brown trout, LNS=longnose sucker, MW=mountain whitefish, BT=brook trout, WS=white sucker, RT=rainbow trout, BLT=bull trout, CI=cisco, LT=lake trout)

Lake	Fish	Slope ± SE	y-intercept ± SE
Barrier	BRT	0.57 ± 0.16	1.59 ± 0.49
	LNS	0.65 ± 0.21	1.58 ± 0.48
	MW	0.66 ± 0.17	1.58 ± 0.48
Cascade	BT	2.23 ± 0.49	0.78 ± 0.18
Gap	BRT	1.48 ± 0.21	2.03 ± 0.55
	MW	0.79 ± 0.35	2.12 ± 0.66
	WS	2.54 ± 0.19	1.88± 0.40
Johnson	BT	1.75 ± 0.22	0.19 ± 0.83
	RT	1.35 ± 0.20	0.74 ± 0.69
	WS	1.20 ± 0.21	0.94 ± 0.74
Lower Kananaskis	BLT	0.65 ± 0.12	1.73 ± 0.47
Minnewanka	CI	0.61 ± 0.19	1.49 ± 0.69
	LT	0.96 ± 0.12	1.40 ± 0.51
	MW	2.24 ± 0.04	1.05 ± 0.13
Mud	CT	0.97 ± 0.06	0.80 ± 0.24
Spray	LT	0.80 ± 0.17	0.40 ± 0.21
	MW	1.20 ± 0.88	0.36 ± 0.10
Two Jack	LNS	1.76 ± 0.27	1.70 ± 0.85
	LT	0.68 ± 0.27	2.13 ± 1.27
	MW	1.91 ± 0.24	1.65 ± 0.78
Vermilion	BT	0.46 ± 0.99	3.31 ± 0.21
	WS	1.95 ± 1.18	3.04 ± 0.80

Table 3.4 Significant Pearson correlations between selenium (Se) concentrations and concentrations of other trace metals in surface water from Johnson Lake and Lake Minnewanka, collected monthly from May to October 2007.

Trace metal	r =	*p* =
Johnson Lake		
Lithium (Li)	0.94	<0.01
Sodium (Na)	0.96	<0.01
Magnesium (Mg)	0.99	<0.01
Potassium (K)	0.96	<0.01
Calcium (Ca)	0.96	<0.01
Zinc (Zn)	0.96	<0.01
Arsenic (As)	0.99	<0.01
Rubidium (Rb)	0.99	<0.01
Strontium (Sr)	0.99	<0.01
Niobium (Nb)	0.99	<0.01
Molybdenum (Mo)	0.99	<0.01
Ruthenium (Ru)	0.92	<0.01
Palladium (Pd)	0.99	<0.01
Antimony (Sb)	0.86	0.04
Barium (Ba)	0.99	<0.01
Europium (Eu)	0.99	<0.01
Tantalum (Ta)	0.83	0.04
Tungsten (W)	0.99	<0.01
Rhenium (Re)	0.92	<0.01
Uranium (U)	0.98	<0.01
Lake Minnewanka		
Magnesium (Mg)	0.856	0.03
Phosphorus (P)	-0.99	<0.01
Zinc (Zn)	-0.84	0.04
Niobium (Nb)	-0.99	<0.01
Tantalum (Ta)	-0.99	<0.01

Table 3.5 Significant Pearson correlations between selenium (Se) concentration and water chemistry parameters in surface water from Johnson Lake, collected monthly from May to October 2007. There were no significant correlations for these parameters from Lake Minnewanka.

Water chemistry parameter	r =	*p* =
Johnson Lake		
Ammonium (NH_4^+)	0.99	<0.01
Chloride (Cl^-)	-0.86	0.03
Silica (Si^+)	-0.89	0.02

Table 3.6 Significant Pearson correlations between bulk zooplankton selenium (Se) concentrations and the concentrations of other trace metals in zooplankton from Johnson Lake and Lake Minnewanka.

Trace metal	r =	*p* =
Johnson Lake		
Aluminum (Al)	-0.84	0.04
Zirconium (Zr)	-0.88	0.02
Lead (Pb)	-0.82	0.04
Lake Minnewanka		
Lithium (Li)	-0.85	0.03
Boron (B)	-0.88	0.02
Aluminum (Al)	-0.85	0.03
Titanium (Ti)	-0.91	0.01
Vanadium (V)	-0.85	0.03
Iron (Fe)	-0.83	0.04
Nickel (Ni)	-0.90	0.01
Zinc (Zn)	0.89	0.02
Germanium (Ge)	-0.87	0.02
Rubidium (Rb)	-0.86	0.03
Yttrium (Y)	-0.84	0.04
Zirconium (Zr)	-0.90	0.02
Palladium (Pd)	-0.91	0.01
Silver (Ag)	-0.82	0.04
Tin (Sn)	-0.82	0.04
Antimony (Sb)	-0.87	0.02
Cesium (Cs)	-0.85	0.03
Lanthanum (La)	-0.85	0.03
Cerium (Ce)	-0.85	0.03
Praseodymium (Pr)	-0.85	0.03
Neodymium (Nd)	-0.85	0.03
Samarium (Sm)	-0.85	0.03
Europium (Eu)	-0.82	0.04
Gadolinium (Gd)	-0.85	0.03
Terbium (Tb)	-0.85	0.03
Dysprosium (Dy)	-0.85	0.03
Holmium (Ho)	-0.84	0.04
Erbium (Er)	-0.84	0.04
Thulium (Tm)	-0.85	0.03
Ytterbium (Yb)	-0.84	0.04
Lutetium (Lu)	-0.84	0.04
Tungsten (W)	-0.90	0.01
Platinum (Pt)	-0.84	0.04
Lead (Pb)	-0.81	0.04
Thorium (Th)	-0.88	0.02

Table 3.7 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in invertebrates (*Limnephilus, Gammarus, Hyalella, Aeshna* and *Callibaetis*) from Johnson Lake.

Trace metal	r =	*p* =
***Limnephilus* (Trichoptera)**		
Vanadium (V)	0.99	0.04
Calcium (Ca)	0.99	0.04
Ruthenium (Ru)	-0.99	0.04
Thorium(Th)	1.000	0.04
***Gammarus* (Amphipoda)**		
Tin (Sn)	-0.86	0.03
Iridium (Ir)	0.86	0.03
***Hyalella* (Amphipoda)**		
Copper (Cu)	-0.95	<0.01
Zinc (Zn)	0.93	<0.01
Arsenic (As)	0.83	0.04
Cadmium (Cd)	0.90	0.02
Tin (Sn)	-0.93	<0.01
***Aeshna* (Odonata)**		
Titanium (Ti)	0.82	0.04
Cobalt (Co)	0.93	<0.01
Cadmium (Cd)	0.98	<0.01
Europium (Eu)	0.82	0.04
***Callibaetis* (Ephemeroptera)**		
Sodium (Na)	0.99	<0.01

Table 3.8 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in invertebrates (*Gammarus* and Gastropoda) in Lake Minnewanka.

Trace metal	r =	*p* =
***Gammarus* (Amphipoda)**		
Chromium (Cr)	-0.84	0.04
Arsenic (As)	0.82	0.04
Molybdenum (Mo)	-0.87	0.03
Tin (Sn)	-0.83	0.04
Gastropoda		
Arsenic (As)	0.97	<0.01

Table 3.9 Significant results of simple linear regressions between selenium (Se) concentrations in fish species and factors that affect Se concentrations within lakes. (RT=rainbow trout, BT=brook trout, WS=white sucker, BLT=bull trout, MW=mountain whitefish, LT=lake trout, BRT=brown trout, LNS=longnose sucker).

Fish species by lake	**Explanatory variable[1]**	**+ or -**	**F**	***p*=**	**% variation in [Se] explained**
Johnson Lake RT	Growth rate 2	-	4.34	0.05	19.4
Johnson Lake BT	Age	+	5.67	0.02	16.9
	FL	+	7.88	<0.01	22.0
	TL	+	6.45	0.02	18.7
	W	+	5.33	0.03	16.0
	Growth rate 2	-	4.93	0.03	15.0
Johnson Lake WS	Age	-	6.16	0.02	18.0
	FL	-	6.54	0.02	18.9
	TL	-	7.00	0.013	20.0
	W	-	8.29	<0.01	22.8
	Growth rate 1	-	14.86	<0.01	34.7
Lower Kananaskis Lake BLT	K	-	6.66	0.03	45.4
	^{15}N	+	30.57	<0.01	79.3
Vermilion Lakes WS	^{13}C	-	14.85	<0.01	65.0
Gap Lake WS	^{15}N	-	6.81	0.03	46.0
Gap Lake MW	Age	-	7.41	0.03	48.1
	FL	-	6.19	0.04	43.6
	TL	-	6.31	0.04	44.1
	Growth rate 2	+	10.38	0.01	56.4
	^{13}C	-	5.31	0.05	39.9
	^{15}N	-	13.95	<0.01	63.6
Cascade Pond BT	Age	+	12.71	<0.01	61.4
	FL	+	15.8	<0.01	66.4
	TL	+	14.06	<0.01	63.7
	W	+	14.17	<0.01	63.9
	Growth rate 2	-	10.74	0.01	57.3
Two Jack Lake MW	^{15}N	-	6.13	0.04	43.4
Two Jack Lake LNS	^{13}C	+	13.85	0.03	82.2
Barrier Lake MW	Growth rate 1	-	5.83	0.04	42.1
Spray Lakes LT	W	+	7.17	0.04	54.4
	Growth rate 1	+	6.39	0.04	51.6
Whiteman's Pond BRT	Age	+	10.72	0.01	57.2
	FL	+	12.16	<0.01	60.3
	TL	+	12.09	<0.01	60.2
	W	+	8.98	0.02	52.9
	K	-	19.84	<0.01	71.3
	^{15}N	+	6.00	0.04	42.8
Whiteman's Pond LNS	Age	-	11.05	0.02	64.8
	FL	-	15.43	<0.01	72.0
	TL	-	13.77	0.01	69.7
	W	-	10.17	0.02	62.9
	Growth rate 1	-	6.95	0.04	53.7
	Growth rate 2	+	18.53	<0.01	75.5
	^{13}C	+	8.76	0.03	59.4

[1]FL=fork length, TL=total length, W=weight, K=condition factor, Growth rate 1=age/FL, Growth rate 2=W/age-1

Table 3.10 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in muscle from rainbow trout, brook trout and white sucker collected from Johnson Lake in August 2007.

Trace metal	**r =**	***p* =**
Rainbow trout		
Vanadium (V)	0.77	<0.01
Chromium (Cr)	0.73	<0.01
Gallium (Ga)	0.61	<0.01
Brook trout		
Calcium (Ca)	-0.42	0.02
Vanadium (V)	0.54	<0.01
Chromium (Cr)	0.74	<0.01
Manganese (Mn)	-0.46	<0.01
Gallium (Ga)	0.74	<0.01
Strontium (Sr)	-0.36	0.05
Niobium (Nb)	0.40	0.03
Palladium (Pd)	0.48	<0.01
Silver (Ag)	0.41	0.03
Tin (Sn)	-0.50	<0.01
Antimony (Sb)	0.45	0.01
Tantalum (Ta)	0.39	0.03
Platinum (Pt)	0.42	0.02
Gold (Au)	0.50	<0.01
Thallium (Tl)	0.40	0.03
Thorium (Th)	0.39	0.04
White sucker		
Aluminum (Al)	-0.37	0.04
Cobalt (Co)	0.46	<0.01
Copper (Cu)	-0.38	0.04
Zinc (Zn)	0.48	<0.01
Gallium (Ga)	0.50	<0.01
Arsenic (As)	0.40	0.03
Cadmium (Cd)	0.47	<0.01
Barium (Ba)	0.37	0.04
Europium (Eu)	0.43	0.02

Table 3.11 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in muscle from lake trout, mountain whitefish and cisco collected from Lake Minnewanka in August 2007.

Trace metal	r =	*p* =
Lake trout		
Chromium (Cr)	0.74	<0.01
Gallium (Ga)	0.71	<0.01
Germanium (Ge)	-0.45	0.01
Mountain whitefish		
Lithium (Li)	0.43	0.02
Boron (B)	0.43	0.02
Chromium (Cr)	0.52	<0.01
Gallium (Ga)	0.57	<0.01
Ruthenium (Ru)	0.49	<0.01
Rhenium (Re)	0.39	0.03
Cisco		
Magnesium (Mg)	0.78	0.04
Chromium (Cr)	0.91	<0.01
Iron (Fe)	0.80	0.03
Gallium (Ga)	0.98	<0.01
Niobium (Nb)	0.82	0.03
Molybdenum (Mo)	0.88	<0.01
Palladium (Pd)	0.88	<0.01
Antimony (Sb)	0.97	<0.01
Tungsten (W)	0.92	<0.01
Platinum (Pt)	0.88	<0.01
Gold (Au)	0.95	<0.01
Thorium (Th)	0.87	0.02

Table 3.12 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in muscle from bull trout and longnose sucker collected from Lower Kananaskis Lake in August 2007.

Trace metal	**r =**	***p* =**
Bull trout		
Boron (B)	0.84	<0.01
Sodium (Na)	0.81	<0.01
Potassium (K)	-0.67	0.03
Calcium (Ca)	0.85	<0.01
Vanadium (V)	0.73	0.02
Manganese (Mn)	0.67	0.03
Gallium (Ga)	0.74	0.01
Arsenic (As)	-0.77	<0.01
Strontium (Sr)	0.88	<0.01
Ruthenium (Ru)	0.80	<0.01
Thallium (Tl)	0.73	0.02
Longnose sucker		
Titanium (Ti)	0.87	0.01
Copper (Cu)	0.89	<0.01
Zinc (Zn)	0.92	<0.01
Gallium (Ga)	0.77	0.04
Cadmium (Cd)	0.77	0.04
Uranium (U)	0.89	<0.01

Table 3.13 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in muscle from brook trout and white sucker collected from Vermilion Lakes in August 2007.

Trace metal	**r =**	***p* =**
Brook trout		
Iron (Fe)	0.66	0.04
Gallium (Ga)	0.79	<0.01
Ruthenium (Ru)	0.66	0.04
White sucker		
Arsenic (As)	-0.67	0.03
Lithium (Li)	-0.79	<0.01
Cadmium (Cd)	-0.67	0.03

Table 3.14 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in muscle from brown trout, mountain whitefish and white sucker collected from Gap Lake in August 2007.

Trace metal	**r =**	***p* =**
Brown trout		
Manganese (Mn)	0.83	<0.01
Gallium (Ga)	0.65	0.04
Mountain whitefish		
Phosphorus (P)	0.72	0.02
Titanium (Ti)	0.80	<0.01
Nickel (Ni)	0.72	0.02
Rubidium (Rb)	0.80	<0.01
Cesium (Cs)	0.74	0.02
Sodium (Na)	0.79	<0.01
White sucker		
Vanadium (V)	0.69	0.03
Chromium (Cr)	0.64	0.04
Gallium (Ga)	0.76	0.01
Rubidium (Rb)	0.80	<0.01
Yttrium (Y)	0.74	0.02
Zirconium (Zr)	0.64	0.04
Palladium (Pd)	0.64	0.04
Silver (Ag)	0.65	0.04
Cesium (Cs)	0.78	<0.01
Lanthanum (La)	0.70	0.03
Praseodymium (Pr)	0.74	0.01
Neodymium (Nd)	0.69	0.03
Tungsten (W)	0.67	0.04
Gold (Au)	0.65	0.04
Thorium (Th)	0.64	0.04

Table 3.15 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in muscle from brook trout from Cascade Pond in August 2007.

Trace metal	**r =**	***p* =**
Sodium (Na)	-0.80	<0.01
Aluminum (Al)	-0.73	0.02
Calcium (Ca)	-0.71	0.02
Copper (Cu)	-0.68	0.03
Zinc (Zn)	-0.70	0.02
Strontium (Sr)	-0.78	<0.01
Tin (Sn)	-0.78	<0.01
Barium (Ba)	-0.78	<0.01
Lanthanum (La)	-0.66	0.04
Cerium (Ce)	-0.65	0.04
Praseodymium (Pr)	-0.66	0.04
Lead (Pb)	-0.65	0.04
Uranium (U)	-0.64	0.04

Table 3.16 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in muscle from lake trout, mountain whitefish and longnose sucker collected from Two Jack Lake in August 2007.

Trace metal	**r =**	***p* =**
Lake trout		
Strontium (Sr)	0.97	<0.01
Cadmium (Cd)	0.90	0.04
Barium (Ba)	0.92	0.03
Mountain whitefish		
Magnesium (Mg)	0.65	0.04
Zirconium (Zr)	-0.87	<0.01
Niobium (Nb)	-0.86	<0.01
Palladium (Pd)	-0.90	<0.01
Silver (Ag)	-0.80	<0.01
Cesium (Cs)	-0.65	0.04
Hafnium (Hf)	-0.83	<0.01
Tantalum (Ta)	-0.85	<0.01
Tungsten (W)	-0.67	0.03
Gold (Au)	-0.88	<0.01
Thorium (Th)	-0.83	<0.01
Longnose sucker		
Copper (Cu)	-0.93	0.02

Table 3.17 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in muscle from brown trout, mountain whitefish and longnose sucker collected from Barrier Lake in August 2007.

Trace metal	**r =**	***p* =**
Brown trout		
Lanthanum (La)	0.93	0.02
Cerium (Ce)	0.90	0.04
Mountain whitefish		
Ruthenium (Ru)	-0.64	0.04
Longnose sucker		
Titanium (Ti)	0.71	0.04
Yttrium (Y)	0.79	0.02
Praseodymium (Pr)	0.88	<0.01
Neodymium (Nd)	0.86	<0.01

Table 3.18 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in muscle from bull trout collected from Upper Kananaskis Lake in August 2007.

Trace metal	**r =**	***p* =**
Chromium (Cr)	0.91	<0.01
Gallium (Ga)	0.95	<0.01
Rubidium (Rb)	0.65	0.04
Barium (Ba)	-0.71	0.02

Table 3.19 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in muscle from lake trout and mountain whitefish collected from Spray Lakes in August 2007.

Trace metal	**r =**	***p* =**
Lake trout		
Magnesium (Mg)	-0.86	<0.01
Phosphorus (P)	-0.78	0.02
Chromium (Cr)	0.74	0.04
Silver (Ag)	0.79	0.02
Platinum (Pt)	0.87	<0.01
Mountain whitefish		
Copper (Cu)	0.76	0.01
Arsenic (As)	-0.67	0.04
Rubidium (Rb)	-0.81	<0.01
Cesium (Cs)	-0.68	0.03
Thallium (Tl)	-0.73	0.02

Table 3.20 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in muscle from lake trout, brown trout, mountain whitefish, longnose sucker and white sucker collected from Whiteman's Pond in August 2007.

Trace metal	r =	*p* =
Lake trout		
Zirconium (Zr)	-0.86	0.03
Thorium (Th)	-0.88	0.02
Brown trout		
Potassium (K)	-0.84	<0.01
Gallium (Ga)	0.80	<0.01
Mountain whitefish		
Titanium (Ti)	-0.64	0.04
Barium (Ba)	-0.66	0.04
Lanthanum (La)	-0.66	0.04
Longnose sucker		
Sodium (Na)	0.88	<0.01
Magnesium (Mg)	0.89	<0.01
Vanadium (V)	0.09	<0.01
Cadmium (Cd)	0.92	<0.01
White sucker		
Vanadium (V)	0.97	<0.01
Chromium (Cr)	0.99	<0.01
Gallium (Ga)	0.95	0.01
Zirconium (Zr)	0.99	<0.01
Silver (Ag)	0.91	0.03

Table 3.21 Significant Pearson correlations between concentrations of selenium (Se) and explanatory variables that affect fish Se concentrations in muscle from brook trout, brown trout, all trout, mountain whitefish and longnose sucker collected from all study lakes in August 2007.

Explanatory variable	**r =**	***p* =**
Brook trout		
%Exposed land in watershed	-0.99	0.04
Maximum temperature of seasonality (coefficient of variation)	-0.99	0.01
Maximum mean temperature of wettest quarter	-0.99	<0.01
Minimum mean temperature of the driest quarter	-0.99	<0.01
Mean precipitation of wettest period	1.00	<0.01
Mean precipitation of warmest quarter	0.99	<0.01
Minimum January maximum temperature	-0.62	<0.01
Mean January maximum temperature	0.99	<0.01
Mean March maximum temperature	-0.99	0.04
Mean October maximum temperature	0.99	0.04
Mean November maximum temperature	0.99	0.01
Maximum December maximum temperature	0.99	0.04
Mean December maximum temperature	0.99	0.04
Mean January precipitation	-0.99	0.03
Mean February precipitation	-0.99	0.03
Mean April precipitation	0.99	<0.01
Mean May precipitation	0.99	<0.01
Mean November precipitation	-0.99	<0.01
Mean December precipitation	-0.99	0.02
Brown trout		
Watershed area	-0.99	0.03
%Spray River formation (shale/sandstone/cherty-limestone/dolostone/gypsum)	-0.99	0.02
%Water in watershed	0.99	0.02
All trout		
%Open conifer in watershed	-0.61	0.03
%Broadleaf-dense in watershed	-0.61	0.03
Mean Isothermality	0.60	0.04
Mountain whitefish		
%Ishbel formation (siltstone/sandstone/chert/phosphate/mudstone/orthoquartzite)	-0.81	0.04
%Snow/ice in watershed	-0.87	0.03
%Open conifer in watershed	-0.91	0.01
%Broadleaf-dense in watershed	-0.90	0.02
Fish growth rate 1	0.91	0.01
Fish weight	0.90	0.01
Longnose sucker		
Maximum temperature of seasonality (coefficient of variation)	0.99	0.02
Fish fork length	-1.000	<0.01

Table 3.22 Significant Pearson correlations between concentrations of selenium (Se) and other trace metals in muscle from brook trout, all trout species, mountain whitefish and longnose sucker collected from all study lakes in August 2007.

Trace metal	r =	*p* =
Brook trout		
Gold (Au)	-0.99	<0.01
All trout		
Sodium (Na)	0.63	0.03
Magnesium (Mg)	0.58	0.04
Europium (Eu)	0.66	0.02
Iridium (Ir)	0.60	0.04
Thallium (Tl)	-0.64	0.02
Mountain whitefish		
Potassium (K)	0.84	0.04
Longnose sucker		
Magnesium (Mg)	0.99	0.04
Phosphorus (P)	0.99	0.04
Lanthanum (La)	0.99	0.04

4.0 Conclusion

In Chapter 3 I examined temporal and spatial trends of Se accumulation in Rocky Mountain Lakes and compared Se concentrations of sediment, water and biota to published toxicity thresholds. Se concentrations in sediment from Johnson Lake exceeded thresholds though to protect fish and bird reproduction. However, Se concentrations exceeded these thresholds throughout the sediment core, suggesting that the Se inputs to the lake have always been elevated and were not anthropogenic in origin. Se concentrations in water and zooplankton from Johnson Lake and Lake Minnewanka were mostly below thresholds that cause health effects in aquatic biota. Water exceeded these thresholds in Johnson Lake during spring melt runoff. Most benthic invertebrate taxa surpassed the thresholds that protect their fish and bird consumers. Se concentrations of nearly all fish species exceeded thresholds that cause health effects to fishes and their avian and human consumers. Se concentrations in two different fishes have increased temporally in all three of the lakes where archived fish were available.

I did not identify any historical trends in the Se concentration of sediments from Johnson Lake, which suggested that anthropogenic activities were likely not contributing to elevated Se concentrations in biota. However, the possibility that Se inputs via erosion have increased without affecting Se concentrations cannot be ruled out. Seasonal trends were identified in water from Johnson Lake (in a spring melt pulse) and some invertebrate Se concentrations in Johnson Lake and Lake Minnewanka. Long-term increasing temporal trends were identified in fishes but not in invertebrates.

I identified differences in fish Se concentration within lakes and in fishes and invertebrates among lakes. Within lakes, I could only formulate significant multiple regression models (which included only independent variables) for seven fish species in six lakes which explained the variation in Se concentrations. Factors that were significantly correlated to Se concentrations within lakes were growth rate, trophic position, fish weight, fish age and fish condition factor.

Among lakes I examined factors that affected age-corrected Se concentrations of fishes. I developed significant multiple regression models using independent variables that were not correlated for all trout species and mountain whitefish. The predictors of Se concentrations in fishes among lakes were % open conifer and dense broadleaf vegetation in the catchment and fish weight.

My results will provide baseline data on Se concentrations in sediment, water and biota and information on Se dynamics in Rocky Mountain lakes. This will be useful to government agencies such as Parks Canada, Alberta Environment and Alberta Sustainable Resources Development. The results of my study should be provided to Health Canada by Parks Canada, so a health risk assessment can be performed to determine if fish consumption advisories need to be implemented for any of the fish species in my study lakes. A temporal monitoring program should be developed based on my finding that already elevated Se concentrations in fishes are increasing over time. This monitoring program should include links to current and archived climate data. Parks Canada, Alberta Environment and Alberta Sustainable Resources Development may also want to expand sampling to determine if elevated Se concentrations in lakes extends beyond my study area. An assessment of Se concentrations in river ecosystems may also be warranted. Se concentrations in fishes were high enough to cause health effects in fishes and their wildlife consumers. If available, archived fish and bird population data could be assessed for declining trends. If declines are not evident, calculation of mountain park specific Se thresholds may be needed. Perhaps Se thresholds developed for other aquatic systems in different geographic areas are too sensitive for lakes in my study area. Fishes and their wildlife predators may be more tolerant to naturally occurring high Se concentrations. However, spinal deformities consistent with Se exposure were identified in three trout (of 342 fishes collected for this study) from Johnson Lake, Lake Minnewanka and Lower Kananaskis Lake. Such a low incidence of these deformities could be the result of factors other than exposure to high Se concentrations. For this reason, mention of these deformities was not included in Chapter 3. Further studies could also include assessment of a variety of fish health parameters.

Current and future research

Currently, Sarah Lord, a Ph.D. candidate at the University of Alberta is investigating Hg, Se and other contaminants in loons from Alberta lakes. She has several study lakes in Banff National Park and the surrounding area and she will determine if Se is elevated in loons, a top avian predator. In addition, my research will be expanded to include the dynamics of other trace metals in Rocky Mountain lakes. Sediment, water, invertebrate and fish data have been analysed for other trace metals (*i.e.,* lead, arsenic, copper) and will be examined for trends.

Future experimental study could examine the teratogenic effects of elevated Se concentrations that I identified within lakes of southeastern Banff and Kananaskis Country. The effects of high Se concentrations in fishes are usually "invisible" because often, affected embryos do not survive. An experimental study could be conducted, involving fishes with high Se concentrations from my study lakes. Eggs would be fertilized and embryos would be examined for teratogenic deformities.

Appendix 1. Selected characteristics of Rocky Mountain Park study lakes

Lake Name	**Easting**	**Northing**	**Elevation (m.a.s.l.)**	**Lake Area (m^2)**	**Catchment Area (m^2)**	**Aspect**	**Mean Slope (°)**
Vermilion	597656.52	5670794.82	1400	299917	12429200	Flat	21.20
Cascade	602467.11	5674524.38	1400	38751	27204000	E	16.69
Two Jack	605115.64	5676470.24	1485	357856	2224400	W	7.53
Johnson	605872.36	5672848.79	1413	153879	15036400	SW	17.17
Minnewanka	611038.72	5681466.42	1505	21919899	607156000	SW	22.99
Whiteman's	611171.50	5658498.00	1676	134347	549157200	W	23.21
Spray	616758.10	5641285.00	1692	18801334	520474800	W	21.97
Mud	618664.75	5627082.00	1623	176544	45364000	W	26.21
Gap	623818.06	5657333.00	1292	286275	2308800	SE	24.89
Upper Kananaskis	630578.25	5608947.50	1701	8454157	145625200	E	24.42
Lower Kananaskis	631709.80	5613210.00	1670	5915781	313956000	SW	20.37
Barrier	635584.99	5655261.46	1378	2811155	851023600	W	23.20

Appendix 2. Selected water quality parameters (mean ± SE) of Lake Minnewanka and Johnson Lake

Lake Name	Water Temperature (°C)	Secchi Depth (m)	pH	TDN (µg/L)	TP (µg/L)	TDP (µg/L)	DOC (mg/L)	Cl (mg/L)	SO_4 (mg/L)	Na (mg/L)	K (mg/L)	Ca (mg/L)	Mg (mg/L)	Cond (µS/cm)
Minnewanka	10.17 ± 1.40	8.89 ±1.76	8.21 ± 0.03	64.67 ± 15.87	3.08 ± 0.55	2.17 ± 0.59	1.43 ± 0.21	0.48 ± 0.07	54.02 ±	1.11	0.53	41.40	15.38	336.00
Johnson	11.83 ± 1.85	2.58 ±0.22	8.15 ±	91.50	4.83	2.08	1.60	0.82	44.11	1.50	0.57	42.16	17.48	345.17

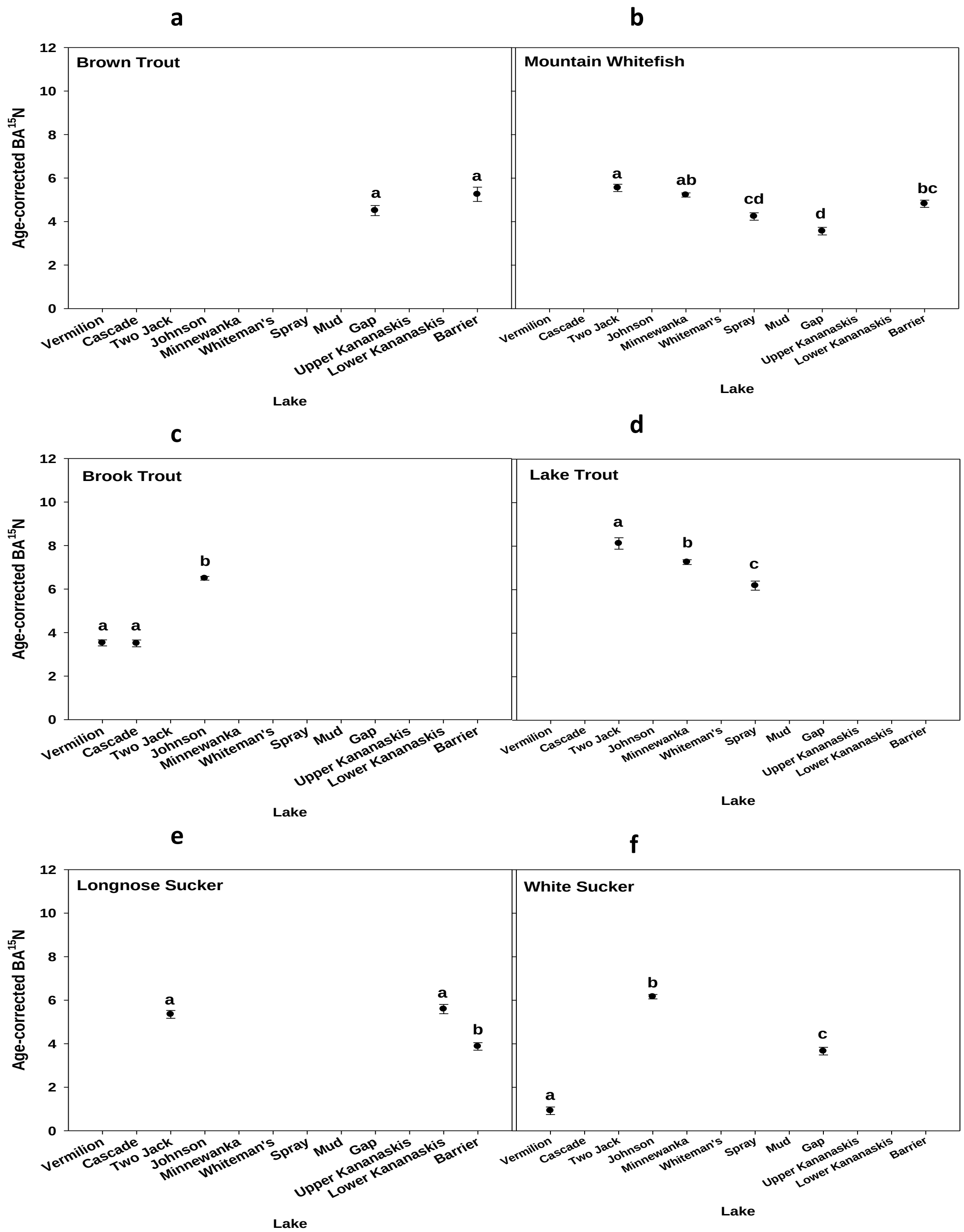
Note: Johnson Lake: secchi depth=bottom depth, TDN=total dissolved nitrogen, TP=total phosphorus, DOC=dissolved organic carbon, Cl=chloride, SO_4=sulphate, Na=sodium, K=potassium, Ca=calcium, Mg=magnesium, Cond=conductivity

Appendix 3. Fish biology variables (mean ± SE) measured in fishes from Rocky Mountain Park study lakes

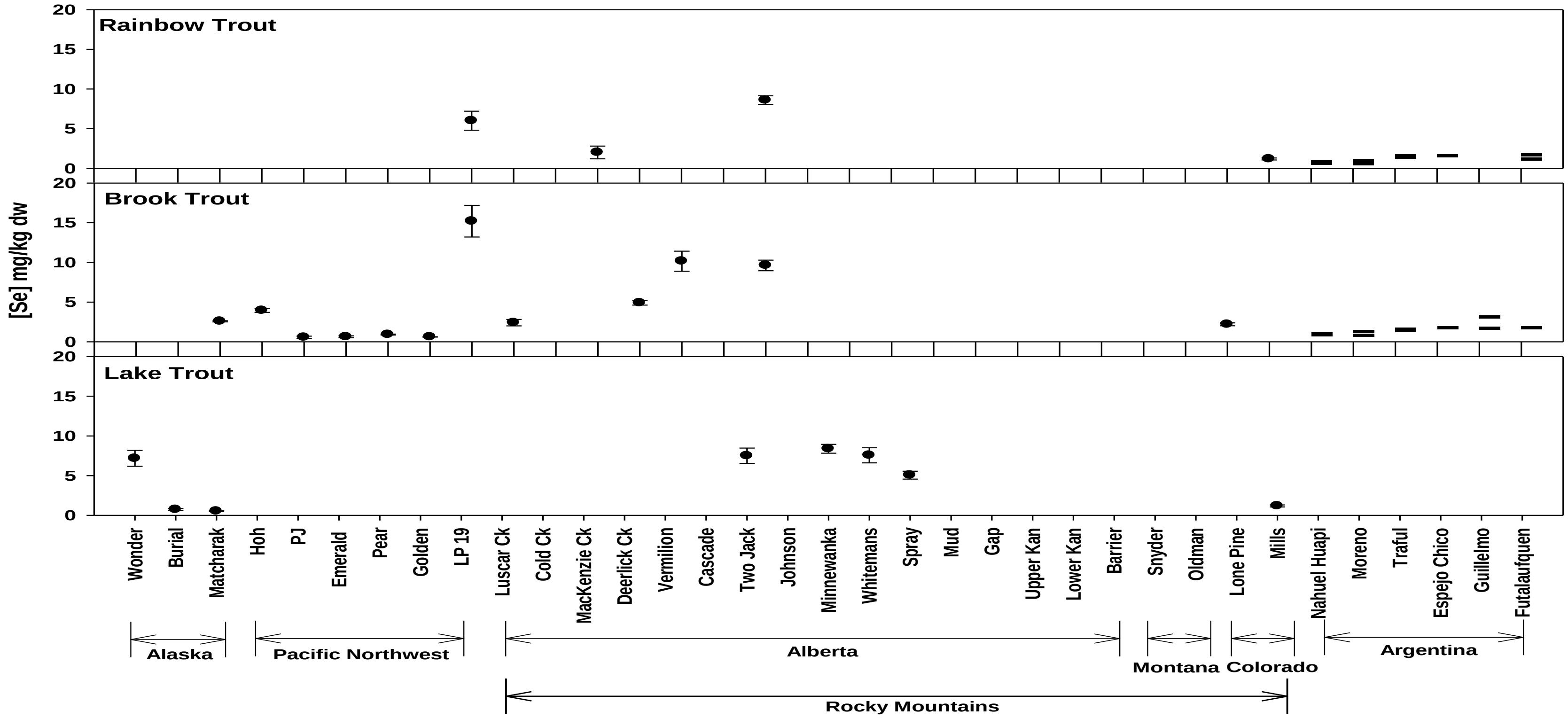
Species	Lake	n	Age (yrs)	Fork length (mm)	Weight (g)	Condition Factor	Growth Rate 1	Growth Rate 2	$\delta^{13}C$	$\delta^{15}N$		%C	%N
										UA$\delta^{15}N$	BA$\delta^{15}N$		
Rainbow trout	Johnson	20	2.70 ± 0.25	221.60 ± 18.25	179.90 ± 38.27	1.25 ± 0.04	90.05 ± 8.04	84.74 ± 2.52	-29.23 ± 0.14	7.85 ± 0.12	5.91 ± 0.12	48.22 ± 0.33	14.28 ± 0.09
Brook trout	All study lakes	3	2.98 ± 0.21	201.18 ± 7.30	113.76 ± 11.19	1.19 ± 0.02	61.73 ± 4.56	76.76 ± 3.28	-28.31 ± 0.16	8.36 ± 0.07	5.30 ± 0.21	46.71 ± 0.14	14.16 ± 0.08
	Johnson	30	2.53 ± 0.20	202.73 ± 9.83	118.27 ± 15.23	1.23 ± 0.04	75.93 ± 3.33	86.47 ± 3.41	-29.12 ± 0.11	8.37 ± 0.09	6.43 ± 0.09	46.93 ± 0.17	14.32 ± 0.06
	Vermilion	10	2.90 ± 0.28	224.30 ± 12.71	148.40 ± 23.70	1.20 ± 0.02	77.48 ± 5.17	80.28 ± 3.89	-27.29 ± 0.15	8.40 ± 0.15	3.52 ± 0.15	46.63 ± 0.15	14.02 ± 0.06
	Cascade	10	4.40 ± 0.67	173.40 ± 14.49	65.60 ± 14.37	1.09 ± 0.02	18.99 ± 0.85	44.10 ± 3.67	-26.89 ± 0.19	8.25 ± 0.11	3.70 ± 0.11	46.12 ± 0.44	13.82 ± 0.30
Lake trout	All study lakes	4	9.50 ± 0.57	413.94 ± 19.44	1177.72 ± 225.28	1.14 ± 0.03	112.20 ± 11.63	47.45 ± 2.27	-27.80 ± 0.21	10.67 ± 0.15	7.16 ± 0.13	47.24 ± 0.28	13.92 ± 0.09
	Minnewanka	29	10.52 ± 0.57	419.90 ± 22.56	1132.13 ± 240.61	1.15 ± 0.04	99.26 ± 13.10	39.99 ± 13.10	-27.50 ±0.10	11.29 ± 0.11	7.31 ± 0.11	47.24 ± 0.33	14.00 ± 0.11
	Two Jack	5	9.00 ± 0.95	464.80 ± 59.68	1405.90 ± 436.10	1.14 ± 0.05	158.43 ± 37.46	50.99 ± 2.90	-25.78 ± 0.50	9.77 ± 0.19	8.03 ± 0.19	47.33 ± 0.59	14.20 ± 0.15
	Whiteman’s	6	6.00 ± 1.06	381.00 ± 35.00	754.75 ± 290.16	1.10 ± 0.13	137.31 ± 25.98	71.45 ± 9.85	-27.52 ± 0.12	9.26 ± 0.42	-	46.59 ± 0.65	13.95 ± 0.13
	Spray	8	8.75 ± 2.35	385.25 ± 73.72	1517.63 ± 1025.05	1.13 ± 0.06	114.53 ± 40.70	54.25 ± 7.35	-30.37 ± 0.33	10.06 ± 0.35	6.07 ± 0.35	47.65 ± 1.08	13.44 ± 0.28
Cutthroat trout	Mud	9	3.22 ± 0.15	192.56 ± 5.51	76.89 ± 4.98	1.08 ± 0.04	34.85 ± 1.62	60.18 ± 1.54	-25.95 ± 0.19	6.61 ± 0.08	5.82 ± 0.08	44.75 ± 0.64	13.97 ± 0.20
Bull trout	All study lakes	2	6.20 ± 0.50	483.30 ± 21.86	1418.04 ± 159.37	1.18 ± 0.03	268.50 ± 11.16	83.33 ± 3.97	-29.60 ± 0.21	8.83 ± 0.12	6.78 ± 0.10	48.63 ± 0.37	13.06 ± 0.15
	Upper Kananaskis	10	5.20 ± 0.53	438.90 ± 11.56	1046.30 ± 88.84	1.22 ± 0.04	262.85 ± 14.73	90.18 ± 6.57	-28.08 ± 0.22	8.41 ± 0.12	-	48.06 ± 0.59	13.14 ± 0.14
	Lower Kananaskis	10	7.20 ± 0.74	527.70 ± 38.02	1789.78 ± 261.98	1.13 ± 0.04	274.15 ± 17.36	76.48 ± 3.59	-30.12 ± 0.27	9.25 ± 0.10	6.78 ± 0.10	49.21 ± 0.39	12.97 ± 0.26
Brown trout	All study lakes	3	6.16 ± 0.43	373.32 ± 22.73	564.40 ± 91.05	0.97 ± 0.04	104.59 ± 9.70	64.20 ± 3.63	-25.56 ± 0.47	8.78 ± 0.21	4.75 ± 0.22	46.45 ± 0.28	14.24 ± 0.06
	Whiteman’s	10	5.90 ± 0.99	428.60 ± 45.51	737.80 ± 160.29	0.87 ± 0.06	145.65 ± 4.86	80.52 ± 5.91	-27.38 ± 0.13	9.01 ± 0.26	-	47.23 ± 0.14	14.31 ± 0.06
	Gap	10	6.50 ± 0.37	338.20 ± 25.95	477.50 ± 148.82	1.06 ± 0.08	78.06 ± 16.76	51.60 ± 1.06	-22.89 ± 0.31	8.02 ± 0.28	4.56 ± 0.28	46.16 ± 0.26	14.32 ± 0.05

	Barrier	5	6.00 ± 0.71	333.00 ± 24.34	391.40 ± 84.41	1.01 ± 0.06	75.54 ± 6.67	56.77 ± 2.79	-27.29 ± 0.36	9.81 ± 0.30	5.14 ± 0.30	45.47 ± 1.15	13.93 ± 0.25
Cisco	Minnewanka	7	6.00 ± 0.31	227.14 ± 3.35	117.29 ± 6.19	1.00 ± 0.03	23.56 ± 0.56	38.30 ± 1.46	-28.15 ± 0.14	9.73 ± 0.10	5.75 ± 0.10	46.61 ± 0.21	14.11 ± 0.12
Mountain whitefish	All study lakes	6	7.23 ± 0.46	267.06 ± 10.02	315.39 ± 36.45	1.15 ± 0.02	42.55 ± 2.07	48.17 ± 3.11	-27.43 ± 0.21	8.43 ± 0.13	4.84 ± 0.11	47.12 ± 0.19	13.72 ± 0.07
	Minnewanka	30	7.67 ± 0.87	283.57 ± 19.56	417.20 ± 73.68	1.20 ± 0.02	51.84 ± 3.92	52.32 ± 6.30	-27.12 ± 0.09	9.20 ± 0.14	5.22 ± 0.14	47.26 ± 0.26	13.98 ± 0.08
	Two Jack	10	8.50 ± 1.63	296.10 ± 35.72	485.90 ± 137.93	1.28 ± 0.06	52.68 ± 6.48	42.77 ± 5.10	-26.87 ± 0.30	7.36 ± 0.24	5.62 ± 0.24	47.46 ± 1.20	13.03 ± 0.24
	Whiteman's	10	3.80 ± 0.59	205.80 ± 13.46	101.60 ± 16.67	1.03 ± 0.02	36.74 ± 2.85	69.73 ± 11.82	-28.50 ± 0.13	7.76 ± 0.12	-	46.19 ± 0.14	13.96 ± 0.07
	Spray	10	10.20 ± 0.74	318.20 ± 16.15	374.70 ± 43.57	1.09 ± 0.03	39.17 ± 2.75	31.83 ± 1.00	-29.73 ± 0.28	8.46 ± 0.15	4.47 ± 0.15	47.44 ± 0.19	13.72 ± 0.16
	Gap	10	4.70 ± 0.58	216.30 ± 14.88	130.90 ± 30.23	1.09 ± 0.03	32.87 ± 2.77	49.71 ± 3.77	-23.70 ± 0.11	6.75 ± 0.12	3.29 ± 0.12	47.14 ± 0.13	13.74 ± 0.08
	Barrier	10	7.60 ± 0.75	249.40 ± 14.14	178.40 ± 26.34	1.07 ± 0.03	25.98 ± 1.31	34.35 ± 1.81	-29.30 ± 0.26	9.48 ± 0.11	4.81 ± 0.11	46.93 ± 0.50	13.35 ± 0.24
Longnose sucker	All study lakes	4	16.14 ± 1.03	435.50 ± 15.22	1165.11 ± 83.29	1.30 ± 0.04	75.10 ± 3.40	29.31 ± 1.42	-27.85 ± 0.30	7.96 ± 0.12	4.84 ± 0.19	47.99 ± 0.42	13.22 ± 0.19
	Two Jack	5	15.80 ± 0.86	430.40 ± 13.39	1348.40 ± 92.15	1.70 ± 0.09	91.05 ± 2.97	27.39 ± 0.72	-24.95 ± 0.33	9.81 ± 0.30	5.14 ± 0.30	45.47 ± 1.15	13.93 ± 0.25
	Whiteman's	8	12.63 ± 1.45	398.50 ± 27.05	830.13 ± 131.14	1.16 ± 0.03	68.22 ± 5.19	33.14 ± 1.95	-28.03 ± 0.16	7.83 ± 0.13	-	46.29 ± 0.42	13.68 ± 0.13
	Lower Kananaskis	7	23.14 ± 0.34	488.57 ± 8.43	1437.29 ± 88.60	1.26 ± 0.05	66.40 ± 3.37	21.11 ± 0.13	-29.27 ± 0.29	7.94 ± 0.10	5.47 ± 1.00	49.90 ± 0.98	12.63 ± 0.48
	Barrier	8	13.75 ± 1.78	429.25 ± 41.49	1115.88 ± 190.49	1.21 ± 0.04	79.63 ± 8.72	33.87 ± 3.06	-28.22 ± 0.30	8.62 ± 0.08	3.95 ± 0.08	48.39 ± 0.70	12.85 ± 0.38
White sucker	All study lakes	4	10.16 ± 0.71	305.42 ± 15.09	515.51 ± 54.32	1.33 ± 0.03	51.05 ± 4.40	36.65 ± 2.42	-27.90 ± 0.34	7.46 ± 0.14	4.61 ± 0.31	46.02 ± 0.19	14.10 ± 0.07
	Johnson	30	10.37 ±1.05	262.20 ± 17.11	334.53 ± 51.48	1.37 ± 0.04	29.84 ± 1.99	30.40 ± 1.90	-29.51 ± 0.08	8.11 ± 0.11	6.16 ± 0.11	45.92 ± 0.34	14.32 ± 0.11
	Vermilion	10	8.50 ± 2.17	271.70 ± 42.22	413.50 ± 148.65	1.21 ± 0.03	45.16 ± 6.13	51.72 ± 10.70	-28.09 ± 0.23	5.79 ± 0.14	0.90 ± 0.14	46.04 ± 0.14	14.02 ± 0.05
	Whiteman's	5	9.60 ± 0.75	379.80 ± 13.64	728.60 ± 62.29	1.32 ± 0.06	84.88 ± 1.73	40.18 ± 1.99	-28.00 ± 0.10	7.55 ± 0.15	-	46.38 ± 0.42	14.00 ± 0.13
	Gap	10	11.50 ± 0.69	431.60 ± 5.85	1053.90 ± 44.80	1.31 ± 0.04	102.49 ± 4.51	38.55 ± 1.92	-22.82 ± 0.21	7.14 ± 0.15	3.67 ± 0.15	46.14 ± 0.22	13.55 ± 0.07

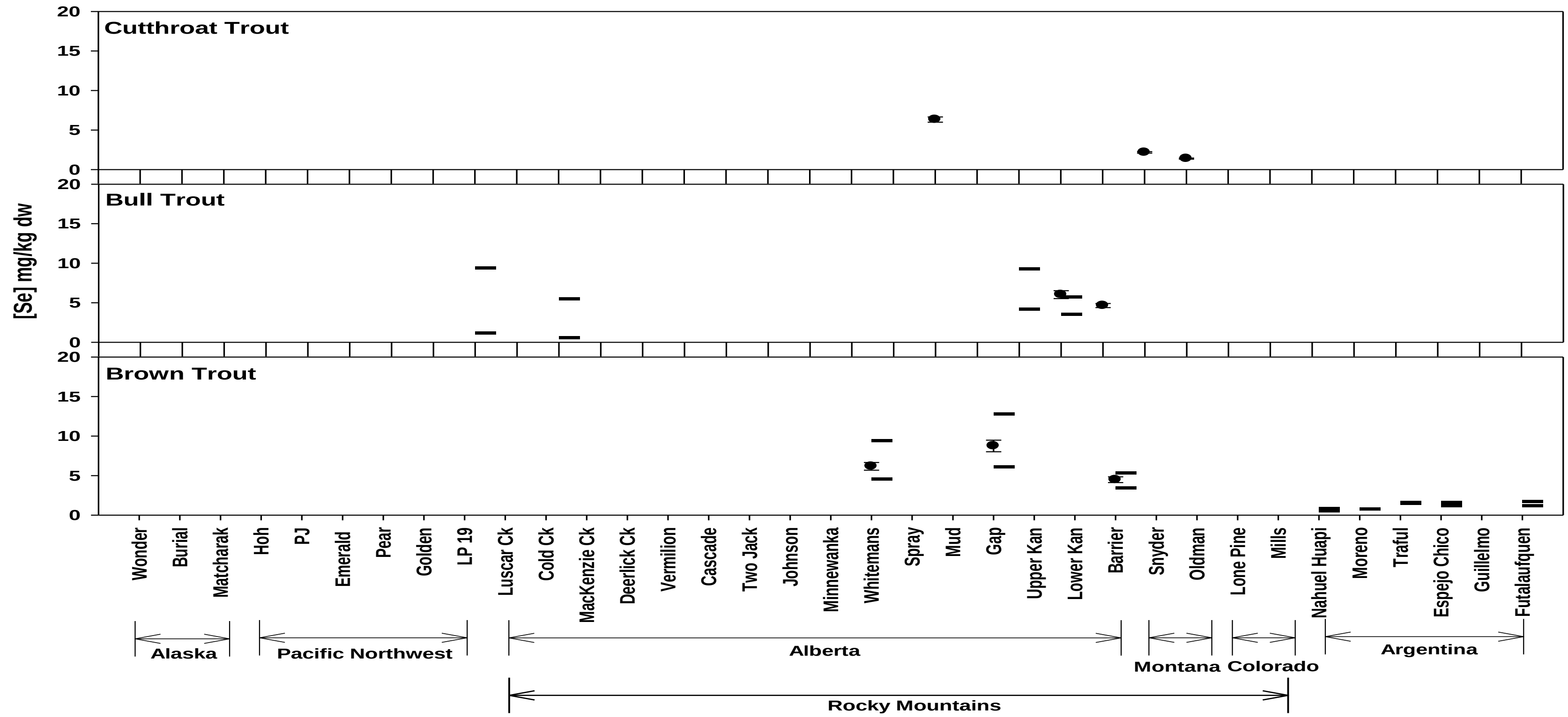
*UA=unadjusted, BA=baseline adjusted



Appendix 4. Among lake comparison of age-corrected least-squares mean baseline-adjusted $\delta^{15}N$ ($BA\delta^{15}N$) of: a) brook trout, b) lake trout, c) brown trout, d) mountain whitefish, e) longnose sucker and f) white sucker. Letters indicate significant pairwise differences based on the results of Tukey's Tests (p<0.05).



Appendix 5. A west-to-east comparison of Se concentrations (mg/kg dw) in rainbow trout, brook trout and lake trout. Data were obtained from WACAP 2009 (Alaska, Pacific Northwest and Colorado), Holm et al. 2005 (Alberta: Luscar Creek, Cold Creek and Deerlick Creek) and Arribére et al. 2006 (Argentina) and were compared to values obtained from my study. Circles (•) and error bars (I) correspond to the mean and standard error. Horizontal bars are the minimum (_) and maximum (¯) which represents the range.



Appendix 6. A west-to-east comparison of Se concentrations (mg/kg dw) in cutthroat trout, bull trout and brown trout. Data were obtained from WACAP 2009 (Montana), Palace et al. 2004 (Alberta: Luscar Creek and MacKenzie Creek) and Arribére et al. 2006 (Argentina) and were compared to values obtained from my study. Circles (•) and error bars (I) correspond to the mean and standard error. Horizontal bars are the minimum (_) and maximum (¯) which represents the range.