VARIABILITY OF FISH PRODUCTION: NUTRIENTS AS CHEMICAL DRIVERS ACROSS A DIVERSE GEOGRAPHIC RANGE

Caitlin Good
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Department of Biological Sciences
University of Lethbridge
LETHBRIDGE, ALBERTA, CANADA

Abstract

Total phosphorus (TP) is an essential nutrient established as a driver of primary productivity that we predict plays a large role in the fish biomass in rivers, lakes and reservoirs. Watershed catchment geology and anthropogenic modification to river flow including hydropower influence nutrient variability in rivers. The relationship between nutrient variation and fish biomass was compared on a local-scale between two geological distinct mountain watersheds in southeastern BC, providing an opportunity to examine smaller-scale ecosystem response to nutrient availability. Nutrient regimes were also characterised in rivers regulated by hydropower and reference rivers across Canada to assess water quality trends and expected variability in fish biomass across a broad geographic scale. Local scale aquatic assessment is useful for identifying trends in ecosystem response within a watershed, whereas regional assessments are at a scale that is more applicable for management. Using TP and fish biomass relationships from a meta-analysis of literature, we developed regionally specific nutrient-based fish models. Establishing baseline nutrient regimes and developing models to estimate expected fish biomass specific to regional fish diversity, provides a useful predictive tool for initiating mitigation and compensation for rivers affected by hydropower.

Preface

This study is part of NSERC HydroNet, a research collaboration with hydropower companies, federal agencies and universities across Canada. Our role within the multidisciplinary research group is to compare productive capacity of fish habitats, related to regional nutrient regimes. In addition to a local-scale project investigating geological influences on nutrient variability, we are comparing nutrient regimes in regulated and unregulated rivers at sites Canada-wide (BC, AB, QC, ON, NB and NL).

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List of Symbols, Abbreviations and Nomenclature

Symbol	Definition
Ag	silver
Au	gold
Ca_2^+	calcium ion
$CaCO_3^-$	calcium carbonate
CO_2	carbon dioxide
CO_2^{-3}	carbonate
DIC	dissolved inorganic carbon
DO	dissolved oxygen
Fe	iron
H^+	hydrogen ion
ma	million years ago
N	nitrogen
N:P	nitrogen:phosphorus
N_2	nitrogen gas
NH_4^+	ammonium
NO_2	nitrite
NO_3^-	nitrate
O_2	oxygen
P	phosphorus
P:B	production:biomass
Pb	lead
PO_4^{-3}	phosphate
PP	particulate phosphorus
SRP	soluble reactive phosphorus
TDS	total dissolved solids
Zn	zinc

1 Understanding the influence of nutrient source and variability in aquatic communities: a literature review

1.1 Nutrient variation

Defined as ecological drivers (Poff and Ward 1990, Wohl et al. 2007), nutrient availability and flux are established as major influences on the fitness and productivity of populations (Kwak and Waters 1997). Nutrient variation in the environment is driven by watershed characteristics defined by geology, land cover, climate and precipitation (McGroddy 2008, Nolin 2012). Therefore, these factors control the hydrological patterns and biological makeup of the ecosystem including aquatic community, population and instream biotic processes. Phosphorus (P) and nitrogen (N) are essential components of ecosystem biochemistry (Ensign and Doyle 2006), and the transport, transformation and limitation of different species of N and P directly affects the potential aquatic production in a system (Vadeboncoeur et al. 2008).

1.1.1 Nutrient sources across landscapes

In a natural watershed, there are many biotic and abiotic interactions involved in nutrient cycling and storage. Rivers were originally defined as a continuum of ecosystem processes along a longitudinal gradient from headwaters to higher stream order mainstems found lower in the watershed (Vanotte et al. 1980, Newbold et al. 1982). Further investigation into nutrient transport has focused on the importance of multi-directional interactions within the watershed, where parent lithology controls the flow interaction along a longitudinal pathway as well as lateral and vertical contact time of water throughout the basin (Triska et al. 1989b, Valett et al. 1996, Pinay et al. 2002, Ensign and Doyle 2006). Therefore, differences in geology and climatic variables control the nutrient uptake processes at a watershed scale, including local precipitation influences on the flow frequency, duration and rate of flux (Bunn and Arthington 2002, Wohl et al. 2007).

Land use

In combination with geologic and flow regime influences on nutrient concentrations, modified landscapes managed for urban development, agriculture, rangeland, forestry, water diversion and power development influence nutrient sources and biological functioning of the system. Increased anthropogenic pressure on the environment has a strong effect on nutrient variation, and land use has become a more prominent contributor to aquatic nutrient regimes (Johnson et al. 1997). Land cover modification can disrupt soil stability, increasing erosional rates and injecting concentrated inputs of sediment-bound and dissolved nutrients into aquatic systems (Triska et al. 1993, Schomberg et al. 2005). Modified nutrient sources are defined as point (direct) sources from industry outfalls, sewage, and urban effluent, or non-point sources including diffuse runoff from intensive agriculture chemical fertilizers, mine rock-drain, forest harvest, and the inundation of land through reservoir creation. These anthropogenic inputs to rivers, lakes and reservoirs have disturbed the natural nutrient regimes of watersheds (Triska et al. 1993, Stockner and Ashley 2003, Camargo et al. 2005, Lehrter 2006, Schindler et al. 2006).

Trophic state

Nutrient modification will often result in a shift in the trophic status of the waterbody. The lowest nutrient state is considered oligotrophic, and progressively higher nutrient concentrations result in mesotrophic and eutrophic conditions (Wetzel 2001). Trophic state defines the potential productivity of the waterbody, where ecosystem response to increased nutrient loads can result in high primary production including macrophytes and algal material, and depleted oxygen through biological oxidation processes (eutrophy). (Wetzel 2001, Camargo et al. 2005, Lehrter 2006). Trophic status assignment has been described extensively in lakes using indicators of primary productivity including P concentrations for nutrient levels and potential light attenuation (OECD 1982, Maassen et al. 2005), whereas river classification of trophic state is commonly identified through studies investigating abundance and community composition of primary producers (Camargo et al. 2005, Leira et al. 2009). Although different indicators of productivity are used to determine trophic state in both lake and river studies, these indicators are commonly in addition to nutrient concentration measures of both P and N (Dodds et al. 1998).

For management purposes accurate definitions of trophic status is important to compare baseline nutrient conditions across a broad range. At the national level (CCME 2004 (Canadian Council of Ministers of Environment)), trophic state is defined using ranges of nutrient values. These trophic state ranges are more established in lakes than rivers (Dodds 2007), therefore values established as CCME aquatic health trigger concentrations are often applied to both lakes and rivers. From studies throughout North America, Dodds et al. (1998) developed trophic state ranges for rivers using cumulative frequency distribution of TP concentrations from over 1300 temperate rivers (Table 1-1). The study assessed pristine river systems within a range of geologic catchments worldwide where concentration of bioavailable N nitrate (NO_3^-) ranged from 50 to 200 µg L^{-1} and bioavailable P phosphate (PO_4^{3-}) ranged from 2 to 25 µg L^{-1} (Meybeck and Helmer 1989). The trophic state of P in these rivers according to Dodds et al. (1998) and CCME (2004) would be oligotrophic to mesotrophic (Table 1-1). Many systems exhibit naturally low, oligotrophic nutrient regimes due to catchment characteristics and lack of exposure to highly urbanized or agricultural watersheds. Yet extensive research has focussed on the eutrophication of systems from increased N and P export, showing the extreme of nutrient loading. An appreciation for the individuality (lithology, climate and flow regime) of each aquatic system (river, lake, or reservoir) will help determine the natural trophic status of the waterbody (Dodds 2007).

1.1.2 Characteristics of nitrogen (N) and phosphorus (P) species

Essential nutrients, P and N, support the productivity of aquatic systems (Biggs and Close 1989, Mulholland and Webster 2010). Nitrogen is prevalent in the atmosphere and landscape, making it readily available in aquatic systems and very soluble in its inorganic form. Yet N can still be limiting because transformation into a useable dissolved form requires complex communities of N-fixing organisms (Moss 2010) that require light and stable environments to be most effective (Wetzel 2001). Phosphorus is available naturally in small quantities and its introduction is via predominantly sedimentary weathering of phosphate-bearing minerals called apatite (Moss 2010). Metabolic demand often outweighs the supply of P (Newbold et al. 1982), for phosphorus has the lowest ratio of supply to need of any essential element derived from the earth (Moss 2010). Therefore, nutrient *recycling* is very important in sustaining P concentrations in aquatic systems (Stockner and Ashley 2003), and characterisation of the baseline nutrient regime is important for understanding speciation and availability of phosphorus in different aquatic systems.

Phosphorus

Total Phosphorus (TP) consists of organic phosphorus, dissolved or soluble reactive phosphorus and total particulate phosphorus. Organic phosphorus often makes up a large component of TP in the form of cellular constituents bound in organisms (Worsfold et al. 2008), yet in order for most aquatic plants and organisms to assimilate phosphorus, it must be in an easily utilized form, dissolved inorganic P (predominantly phosphate, PO_4^{3-}) (Bostrom et al. 1988). Particulate phosphorus (PP) contains any relatively large (>0.45um) organic or inorganic material from bacteria to clays. PP is responsible for the majority of sediment-bound P transported during storms (Newbold et al. 1982). Regions with high PP include glacial fed systems where the phosphorus sorbs to the glacial flour, making it temporarily unavailable for use by primary producers. Researchers on the Flathead River in Southeastern BC, through algal assays, found that suspended loads during turbid high flow were composed of less than 6% of usable P (Ellis and Stanford 1988), indicating that environmental factors will influence the mobility and biological availability of P throughout the system (Bostrom et al. 1988). Bioavailable or useable P is defined as the sum of P that is dissolved in the water column and immediately available for uptake. Bioavailable P consists of dissolved or soluble reactive P (SRP) (predominantly PO_4^{3-}) plus the P that is reversibly bound to inorganic substrate or suspended autochthonous organic matter (Bostrom et al. 1988).

Bioavailability varies temporally among systems depending on environmental conditions. Thus, there are conflicting opinions on bioavailable estimates related to residence time of reversible P bound to inorganic and organic compounds (Bostrom et al. 1988, Ellis and Stanford 1988, Chambers et al. 1992). Some studies prefer SRP as the most direct way to measure active P, but this excludes P from slower assimilation pathways that would be included in a TP measure (Nurnberg and Peters 1984). Opposing views perceive SRP concentrations as the minimum potential amount of phosphorus available, serving as an underestimate without consideration of the potential reversibility of P into dissolved form (Bostrom et al. 1988). In systems with high dissolved nutrient loads, high production will increase available phosphorus turnover (Correll 1998), yet in systems with low inorganic nutrient values, small increases in available nutrients may make a large difference in uptake (Bothwell 1989). Therefore, regardless of trophic state, available phosphorus levels may be quickly consumed by the demand of primary producers and thus TP (including both particulate, dissolved inorganic and organic phosphorus) is considered an accurate measure of actual availability (Correll 1998, Dodds 2003).

Nitrogen

Total Nitrogen (TN) is a combination of nitrogen found in amino organic compounds, nitrogen gas (N_2), and dissolved inorganic nitrogen including oxidized nitrite (NO_2), nitrate (NO_3^-) and the reduced form as ammonium (NH_4^+). Nitrogen enters the ecosystem through nitrogen fixation, precipitation, surface water sources and ground water filtration (Wetzel 2001). Changes in nitrogen speciation occur at the sediment-water and water-air interface through oxidative and reductive processes including nitrification, denitrification, and ammonification (Triska et al. 1989b). Nitrification converts NH_4^+ into NO_2^- and finally to NO_3^-, the largest and most soluble component of dissolved nitrogen in aquatic systems (Lefebvre et al. 2005). This process decreases O_2 and acidifies the water column. Denitrification is the reduction of organic matter (carbon) which occurs where there are areas of low O_2 (i.e. areas of decomposition and sedimentation). Anoxic conditions enable heterotrophic bacteria to rob NO_3^- of O_2, releasing N_2 gas from the aquatic system into the atmosphere (Wetzel 2001, Lefebvre et al. 2005, Moss 2010). Ammonification is conversion of organic biomass or waste by bacterial decomposition to NH_4^+ (Moss 2010). Although local shortage of N can occur, generally N is found in high amounts due to increasing inputs of fertilizer and industrial emissions (Schindler et al. 2006) in the environment which can increase nitrogen deposition and acidify and eutrophy aquatic ecosystems.

1.1.3 Abiotic factors and nutrient dynamics

Water quality dynamics are important controls of nutrient speciation (Mueller et al. 2006, Frost et al. 2009). Each watershed is characterised by its unique lithology, flow pathways, climate, precipitation (either snow or rain-melt), and vegetation cover. Certain abiotic conditions respond to these watershed characteristics (including pH, dissolved oxygen (DO), conductivity, alkalinity, temperature and light attenuation), and play a large role in the continuous shift of seasonal or diel nutrient fluctuations. These factors may accelerate or impede nutrient speciation or uptake, creating a cascade of physiochemical processes.

Dissolved oxygen

Dissolved oxygen (DO) concentrations control species composition and distribution on both a spatial and temporal scale in rivers, lakes and reservoirs. As a product of photosynthesis and a requirement for respiration, dissolved oxygen is vital to the growth and distribution of bacteria, aquatic plants (including algae and macrophytes) in a freshwater system. Spatial disparities in DO are more common in reservoirs and lakes due to a high water surface area to atmosphere exposure ratio (Giller and Malmqvist 2008). Although a river is turbulent in nature, deep pools and slow water areas in rivers may experience allochthonous or litter accumulation. Although this depositional zone is a beneficiary to nutrients released as a process of decomposition, enrichment becomes a trade-off for oxygen deprivation (Giller and Malmqvist 2008). Anoxic and oxic conditions also control the release and retention of N and P at the sediment water interface. In anaerobic conditions, nitrification does not occur, limiting soluble nitrogen forms, or conversely, in aerobic conditions, P permanently binds to inorganic and organic particles. Directly influencing bioavailability, shifts in DO trigger the uptake and release of essential nutrients.

pH and dissolved inorganic carbon

To sustain health, aquatic biota depend on stable pH of their environment as a limitation for distribution, within a range of pH of 6-9 (Environment Canada 2004, Moss 2010). The measure of H^+ ion concentration in the water, pH is an important water quality variable when characterising systems with different parent geology. For acidic systems with high [H^+], the rate at which elements are dissolved from rocks increases, creating an ecosystem response to weathering products. For example, in carbonate rich streams, pH values will be slightly alkaline, yet streams tend to be naturally acidic in non-sedimentary, soft water systems. Because pH is a logarithmic scale, small shifts in values can be important in nutrient availability and cycling. Gomez et al. (1999) reported that after a pH change from 7.0 to 8.0, particulate phosphorus that was originally adsorbed to Fe, reabsorbed to available organic material. This is relevant for aquatic ecosystems, where shifts in pH can control nutrient uptake rates and availability of essential nutrients.

Dissolved inorganic carbon (DIC) is present in aquatic systems as dissolved CO_2 and bicarbonate, and carbonate. Inorganic carbon concentrations are replenished through atmospheric and mineral additions to the system from catchment weathering processes and are strongly influenced by pH (Wetzel 2001). DIC can be consumed in the form of CO_2 by algae for photosynthesis, a by-product of microbial respiration, or precipitated out as calcium carbonate ($CaCO_3$) in rivers with high calcium content (Finlay 2003).Therefore, DIC would be altered by unique geology, discharge, forest structure, and stream productivity (Finlay 2003).

Conductivity and alkalinity

Alkalinity measures the carbonate concentration of the water, determining the system’s ability to neutralize acid. Low alkalinity is present in boreal catchments (Precambrian or igneous) with low Ca^{2+} content, where the lower the alkalinity the more sensitive the system is to acidification. Conductivity is the total ionic content, related to both the major ions and total dissolved solids (TDS) of the waterbody. Conductivity can be an indicator of catchment characteristics including erosional resistance and ion uptake of parent lithology, and for differentiating between ground water and precipitation dominated watersheds (Sinokrot et al. 1995). Both conductivity and alkalinity, similar to N and P, have been used as surrogates for bottom-up studies of productive capacity (Mazzoni and Lobon-Cervia 2000, Almodovar et al. 2006, Benke and Huryn 2010) because of their connection to water ion chemistry and carbonate concentration.

Temperature and light attenuation

Directly related to the metabolic response of aquatic organisms (Giller and Malmqvist 2008), temperature indirectly influences dissolution of oxygen, organic matter decomposition, and solubility of nutrients. As water temperatures increase, nutrient and oxygen availability decrease, exemplifying the importance of thermal regime on productivity and aquatic community in freshwater systems (Caissie 2006). Aquatic communities, including fish, have thermal preferences and limitations that influence their metabolic rates and distribution (Boyer et al. 2008).

Light limits the biotic processes in aquatic systems for it is essential in photosynthetic processes in reservoirs and lakes. In the upper reaches of streams that experience high canopy cover, baseline production is influenced primarily by heterotrophic processes initiated by inputs of organic material. Because light limits photosynthesis, streams become autotrophic in the un-shaded lower reaches (Vanotte et al. 1980, Dodds 2006). Light availability in aquatic systems will shift depending on forest and canopy cover (Dodds 2006), suspended sediment and turbidity (Northcote and Larkin

1963) and algal abundance (Vadeboncoeur et al. 2008), resulting in an ecosystem response in productivity.

Nutrient speciation can respond to multiple abiotic factors in the environment including DO, pH, DIC, conductivity, alkalinity, temperature and light attenuation. Seasonal and daily variability or anthropogenic modification of flow will alter the interaction of these abiotic factors, thus shifting nutrient transport and storage.

1.1.4 Nutrient transport and storage

Research on nutrient transport and storage dynamics in freshwaters has focussed mainly on the processes in lakes. River flow pathways were understood as advective until advancement in the understanding of the attenuation of the transport process began in the late 1970’s (Morrice et al. 1997). Nutrient spiralling, a concept introduced by Webster et al. (1975), demonstrates how ecosystem structure biologically controls nutrient uptake in rivers, where nutrient pulses from the headwaters are redistributed downstream through water, particulates and consumers (Mulholland et al. 1985). Continuing this work, researchers used P-radiotracers to model river length of nutrient uptake and storage to further understand stream P dynamics (Newbold et al. 1981, Newbold et al. 1983). Rivers are considered the greatest transport systems of materials of all aquatic systems, exhibiting complex recycling processes to store nutrients that sustain productive communities (Triska et al. 1989a). Hydrologic retention and nutrient storage increase with increased channel complexity. Debris dams (Ehrman and Lamberti 1992), increased hydraulic connectivity and permeability of the lithology (Morrice et al. 1997) will influence the entrainment of nutrients exposed to slower stream velocities.

Nutrient cycling alters the bioavailability of essential nutrients in freshwater environments. Trophic state, mixing regime, abiotic conditions and the presence of reducing metals control the storage and bioavailability of P in bottom sediments (Nurnberg and Peters 1984, Mueller et al. 2006). In reservoirs, lakes, and lentic habitats of rivers, the sediment-water interface can act as a source or sink of nutrients (Valett et al. 1996). Nutrients in solution adsorb to organic or inorganic fines, settling out in slow depositional habitats. At the water-sediment interface, with stable oxygen levels and pH, P can remain permanently bound and become buried, in the only other permanent N and P removal process from aquatic systems besides denitrification (Ensign and Doyle 2006). On the other hand, internal loading, the resuspension of bound nutrients, occurs during reduced oxygen conditions (Rzepecki, 2012 and Baldwin 2007). Internal loading is common in stratified reservoirs with an anoxic hypolimnion, and in polymictic (shallow) lakes exhibiting algal blooms and anoxia in macrophyte beds. Lentic habitats in rivers can also exhibit similar conditions due to the decomposition of organic matter in sediment pore space, resulting in decreased O_2, reintroducing a flux of soluble P into the system. With a strong affinity to adsorb to particulate matter, P available in solution would decrease during times of high suspended solid levels. In multiple northern glacial rivers where a majority of P is in particulate form rather than dissolved, Hodson et al. (2004) reported that P would adsorb to fine glacial sediment throughout the watershed, and under optimal conditions, would create a bank of ‘reversible’ phosphorus. The extent of sediment nutrient retention is largely dependent on the watershed’s morphological features that would control flows and seasonal mixing (climate, thermal stratification and depth) and abiotic conditions as described above (Vadeboncoeur et al. 2008).

1.1.5 Geomorphic setting: nutrient regime baseline

Geological setting

Nutrient variability on a regional scale is controlled and altered by the parent lithology of the watershed (Valett et al. 1996, Lefebvre et al. 2005, Schomberg et al. 2005, Wohl et al. 2007). Geologic characteristics affect the physical, chemical and biological components of an ecosystem (Neff and Jackson 2011) and are vital in defining hydrologic connectivity and dissolution of minerals throughout the watershed. Permeability of the substrate will decrease in systems with fine sediments including clays, silts and till, as compared to crystalline rock structure with low solubility that mechanically erodes into coarser substrate (i.e. granite)(Morrice et al. 1997). With a reduction in fine sediments, there is increased connectivity between the river and lateral recharge (Dahm et al. 1998). Nutrient interaction and retention with hillslope flow, ground water and aquifers will vary depending on catchment lithology (Valett et al. 1996). Nutrient uptake depends on contact time of water in the watershed (Wohl et al. 2007), where you would expect to see a difference in nutrient concentration in steep flashy mountain rivers compared to meandering low elevation prairie streams. Due to different erosional resistances and chemical makeup, unique geology will determine the presence and accessibility of organic elements in aquatic systems.

Many studies examine landform differences for the purpose of nutrient regime characterisation on the basis of sedimentary vs. non sedimentary geology (Valett et al. 1996, Morrice et al. 1997, Neff and Jackson 2011). Although simplified, this approach allows a clear distinction between the differences that occur at a broad scale across North America between contrasting Precambrian Shield and sedimentary landforms.

Sedimentary drainage basins have high weathering potential, erosion rates, and availability of soil minerals, and therefore river systems have high nutrient concentrations. Limestone is a common phosphate bearing carbonate rock, and streams with this parent lithology are considered hard water systems due to their ionic content. Granite catchments have low weathering rates, stream-bed solubility, and are considered soft water systems with low ion content (Camargo et al. 2005, Brown et al. 2011). In addition, a lower buffering capacity, common in waters with minimal ion concentration will affect the streams ability to adapt to shifting anthropogenic influences including acidification (Schindler 2001). Igneous rock is a major component of the continental crust forming the Precambrian Shield found extensively throughout the boreal region of Canada, and is also found as granite batholiths intruding sedimentary or metamorphic layers in mountain building regions (Willett 1999). A watershed's baseline nutrient regime is distinct due to differences in solute uptake and source minerals present in geologically unique catchments.

Climate

Local climate variables play a role in modifying the nutrient regime within the catchment, and is often characterised using latitude (Cooke et al. 1993), elevation (Lewis and McCutchan 2010), temperature (Lessard and Hayes 2003), precipitation (Hauer et al. 1997) and the resulting flow regime (Green and Finlay 2010). As latitude increases, snowline elevation will decrease from at or above 5000 m at the equator, to sea level in the arctic (Ward 1994), influencing the quantity and rate of material loading in the system (Cooke et al. 1993). In Montane systems, elevation gradients influence state of precipitation (rain or snow) and water temperature, which in turn influence dissolution rates (Lewis and McCutchan 2010), and fish growth rates (Parra et al. 2009). Flow regime of the watershed, whether glacial, snowmelt or rain dominant, will also influence the solute concentration of the waterbody (Sickman et al. 2003). In the summer, glacial-fed systems typically have a diel influx of nutrients due to daytime melt, including high amounts of suspended sediment, influencing light attenuation and nutrient dynamics (Ward 1994). Snowmelt dominated systems will experience high spring runoff flows during melt, and are similar to glacial melt, which are often low in dissolved and high in particulate nutrients (Hauer et al. 1997). With greater inter-annual variability, rain dominated regions will often have low concentrations of nutrients due to the dilution effect from high annual rainfall, especially in regions with reduced mineral content such as coastal British Columbia (Giller and Malmqvist 2008).

1.2 Influence of hydropower on nutrient regimes

1.2.1 Introduction

Harnessing the flows of most major rivers in the world, dams provide numerous benefits for flood control, irrigation, water diversion, recreation, fisheries and hydropower (Bednarek 2001, Miranda 2001). Dams disrupt the natural flow of rivers, and instream fluctuations are no longer governed chiefly by seasonal fluxes but through operational requirements and resource management decisions. Nutrient dynamics are inherently dependent on flow patterns, and therefore the impoundment of rivers has a great effect on the natural transport and storage of a river’s nutrient regime.

1.2.2 River to reservoir

River impoundment can change the timing, magnitude and frequency of the natural flow regime (Ward and Stanford 1983). Similar to river dynamics, reservoir nutrient regimes respond to their upstream catchment, with the exception of greater depositional habitat for sedimentation of nutrients. Transitioning from natural riverine flows where turbulent mixing generally sustains nutrients and particulates in suspension, dams alter stream habitat by creating slower, lentic conditions. Lentic regions of the reservoir experience greater accumulation of particulates, decreasing transport downstream (Newbold et al. 1982).

Although directly influenced by the chemistry of its tributaries, reservoir shape (Northcote and Larkin 1963), residence time (De Oliveira Naliato et al. 2009), age (Grimard and Jones 1982), and presence of littoral zones (Matzinger et al. 2007) are important in determining and sustaining trophic state. Depth is negatively correlated with productivity in lakes (Wetzel 2001), therefore deeper reservoirs with poor mixing would likely sustain oligotrophic conditions. Shallow, well mixed systems with a greater surface area to depth ratio would maintain higher nutrient conditions (Cooke et al. 1993). Residence time is generally the average length of time water spends within the waterbody (Thornton 1990). Reservoirs in Brazil with different retention times were found to exhibit significantly different limnological conditions from one another. Reservoirs with longer residence times stratified all year long compared to ones with shorter residence time of 10-15 days where stratification was highly variable, and only during the summer (De Oliveira Naliato et al. 2009). Short residence times should not be overlooked, as any retention upstream of a dam is still longer than the natural river flow (Stanley and Doyle

2002). A longer retention time enhances P cycling and that may result in a more productive planktonic community, highly influencing ecosystem function (Ward and Stanford 1983, Kennedy 2001). Researchers have also found reservoir age to be an important determinant in nutrient regimes (Grimard and Jones 1982). Dams cause flooding of terrestrial landscape inundating vegetation and initiating decomposition within the reservoir (Stanford et al. 1996). The receiving reservoir and stream experiences an increase in nutrients related to the flux in organic matter that can last as long as a decade, followed by reduced nutrient concentrations as the reservoir ages (Grimard and Jones 1982).

Parallels drawn between reservoirs and lakes are often rejected due to a reservoir's water level fluctuations which create unstable littoral zones that lower primary productivity (Thornton 1990, Kennedy 2001, Stockner et al. 2005). Littoral zones, described as the shoreline region of a water body exposed to light (Cooke et al. 1993, Vadeboncoeur et al. 2008), host epilimnetic plants and algae that play a large role in P turnover (Rigler 1956). A disproportionate amount of production is found in the littoral zone of a lake (Matzinger et al. 2007), a habitat that is often deficient in reservoirs (Stockner et al. 2005). Water level fluctuations destroy the opportunity for community establishment, and thereby sustained production throughout the reservoir (Bunn and Arthington 2002, Matzinger et al. 2007).

Along with geomorphic attributes, dam design and operation will influence downstream ecosystem functioning, whether a large reservoir with a large retention of water, a storage headpond or a smaller run of the river dam, with shorter water retention times (Kennedy 2001, De Oliveira Naliato et al. 2009). Energy demand results in increased releases during periods of high demand (winter), or in some cases, even daily ramping rate fluctuations in hydro-peaking dams (Arthington et al. 2006). Minimal flow requirements for aquatic life are highly variable. Attempts by water managers to incorporate environmental flows that mimic a natural system, sustain ecosystem cues and aquatic habitats below dams need to be system specific (Arthington et al. 2006). Therefore, the extent of limnological processes of reservoirs are controlled by the unique operational regime of the dam, and nutrient response may be limited by reservoir depth, residence time, or stratification, either enhancing or disrupting the trophic status of the system (Northcote and Larkin 1963, Friedl and Wuest 2002, Ahearn et al. 2005).

1.2.3 Reservoir to river

Downstream of dams, rivers experience disruption in their flow patterns, stream temperature, substrate composition, sediment loads, and nutrient regimes (Ward and Stanford 1983). The most noticeable indicator of a dams influence on a system is the biological downstream effects. Greatly studied, thermal stratification is common in reservoirs, where water warms on the surface (epilimnion) and cold dense water remains on the bottom (hypolimnion) (Edwards 1978, Kennedy 2001, Nurnberg 2007). Similar to processes in rivers and lakes, modified temperature regimes will instigate dissolved oxygen swings, influencing transport and storage of nutrients. Either periodic or in some cases long term, lack of exposure to surface mixing and light attenuation, the hypolimnion becomes anoxic.

The structure and function of the dam will determine outflow dynamics, and is especially important in reservoirs that experience stratification (Nurnberg 1998). The vertical positioning of spill-water release at the head of the dam can determine the water quality downstream, whether it releases from the epilimnion or more commonly, hypolimnetic release (Kennedy 2001, De Oliveira Naliato et al. 2009). Pulses of cold, nutrient poor water from the hypolimnion during summer months will influence nutrient availability and thermal refugia for downstream organisms, where thermal shifts can be detected tens of kilometers downstream (Sinokrot et al. 1995).

Dams disrupt the longitudinal transport of materials downstream, as well as they increase abundance of fine sediments below dams from the lack of scouring flows (De Oliveira Naliato et al. 2009). Therefore, the alteration of sediment transport directly influences the transport of particulate nutrients a great distance from the dam (Bednarek 2001). Reservoirs can act as either a source or a sink of P (and a lesser extent N) through sedimentation and internal loading (Friedl and Wuest 2002). Due to temperature density gradients when rivers enter reservoirs, upstream river flows may plunge below the photic zone of the reservoir, becoming biologically unavailable, either depositing nutrients in bottom sediments, or bypassing influence on reservoir production altogether by flowing directly out through the hypolimnetic outflow (Pieters and Lawrence 2012).

1.3 Influence of nutrient regime on aquatic productivity: bottom up processes

1.3.1 Primary productivity

The relationship between nutrient concentrations and primary productivity is well studied (Elwood et al. 1981, Biggs and Close 1989, Rosemond et al. 1993), where bottom-up processes indicate that algae and aquatic plants depend on growth limiting nutrients for metabolism and production (Rosemond et al. 1993, Stockner and Ashley

2003). N:P ratios, commonly used as an indicator of aquatic nutrient deficiency (Dodds 2003) and interactions (Green and Finlay 2010), are calculated from measured total or dissolved nutrient values as proportions. Introduced by Redfield (1958), N:P ratios of 16:1 indicated a proportion limited in neither nitrogen nor phosphorus (Redfield 1958). N:P stoichiometry in freshwater and marine science has been widely studied, and its variability is believed to be driven by different processes in unique aquatic systems including decomposition of seston in oceans (Redfield 1958), anthropogenic influence in lakes (Downing and McCauley 1992, Arbuckle and Downing 2001), and hydrologic transport (Green and Finlay 2010) and geology (Valett et al. 1996, McGroddy 2008) in streams.

1.3.2 Nutrient enrichment projects

Nutrient enrichment projects have experimentally demonstrated ecosystem responses to shifts in bioavailable nutrients in a range of aquatic systems including lakes, reservoirs and rivers (Ashley and Stockner 2003, Stockner and Ashley 2003). The response of fertilization programs in large lakes and reservoirs (Pieters et al. 2003, Hyatt et al. 2004), small coastal watersheds (Perrin et al. 1987, Slaney et al. 2003), inland rivers (Bothwell 1989), or high latitudes (Deegan and Peterson 1992), all support the influence of N and P on freshwater productivity across diverse waterbodies. The initial goal of enrichment projects on rivers was to increase food supply to re-establish historically prominent fisheries in oligotrophic rivers (Perrin et al. 1987, Pieters et al. 2003, Hyatt et al. 2004). Damming has increased retention of terrestrial derived nutrients and disconnected upstream migration of anadromous salmonids (Wipfli and Baxter 2010).

Fertilization experiments are designed as short-term habitat enhancement. A flux of nutrients into the watershed supplements low nutrient budgets and instigates a shift in trophic state by stimulating productivity as a mitigation effort for river disruption (Hyatt et al. 2004).

1.3.3 Secondary productivity

Secondary productivity is the biomass produced by the consumers of the population from ascending trophic levels (Wetzel 2001). Consumers are the initial step in trophic transfer, and production is the rate of biomass (weight per area) accumulated by the aquatic population over a period of time (Allen 1971, Benke 1993, Wetzel 2001). Higher nutrient concentrations will increase algal growth in streams (Newbold et al. 1982, Huryn and Wallace 2000), translating to more food availability for fish and invertebrates. Production estimates include community structure and growth rates; measures that are used to quantify ecosystem conditions for fitness and survival of aquatic organisms (Poff and Huryn 1998). Production to biomass (P:B) ratios are commonly used as an index for biomass turnover, where high ratios would be expected in fast growing, small organisms, providing an indicator of life histories within the population (Robertson 1979).

1.4 General organization of thesis

Establishing baseline characteristics of a system allows for application of nutrient management criteria (Dodds 2007). This project was designed to characterise nutrient regimes across a diverse geographic range, and identify trends in bottom up controls of energy transfer to fish production (biomass). The introduction of this thesis outlined the background overview from the literature on the processes that influence nutrient cycling and variability. The focus is specifically on the natural variation of natural systems due to geologic characteristics of the watershed, and nutrient variability due to flow management from Hydropower development. Chapter 2 presents a local study that is a comparison of nutrient variability between regions, in two geologically distinct mountain ranges in Southeastern British Columbia. These mountain systems provide an opportunity to investigate trends in ecosystem response to nutrient availability on the lowest end of the productivity spectrum, where a paucity of nutrients may influence fish biomass and productive capacity of the system. The third chapter is a study across a larger geographic range as part of a research network (NSERC HydroNet) in collaboration with hydropower companies, federal agencies and universities across Canada. Our role within the multidisciplinary research group is to investigate a relationship between nutrient regimes and fish productivity in regulated and reference rivers.

These two studies complement one another as they provide a connection between testable local scale trends and larger scale management challenges. The more intensive local study of fish population response to nutrient concentrations indicates the complexity of systems within regions, whereas the HydroNet study provides a regional approach, adapting to challenges with applying local trends across larger scales with unique fish communities, and anthropogenic influences. From a management standpoint, this thesis is an important contribution in the development of a nutrient database, and providing an indication of regional similarities and differences in nutrient regimes. In addition, this study provides a tool for estimating regionally specific expected fish biomass through the development of nutrient based fish models derived from the literature. TP and TN were chosen for analyses in both studies due to their robustness (Nurnberg 2007), reliability and comparability across large data sets. Although TN is discussed throughout the thesis, TP has been established as driver of primary productivity and we predict it will play a larger role in defining fish productivity in aquatic systems.

Table 1-1 Trophic state for TP ranges in aquatic systems (lakes and rivers) according to CCME guidelines (Environment Canada 2004) and TP values from a cumulative frequency study of rivers (Dodds et al. 1998).

Trophic State	CCME Range (μg L^{-1})	Rivers only (μg L^{-1}) (Dodds et al. 1998)
Ultra-Oligotrophic	<4	-
Oligotrophic	4-10	<25
Mesotrophic	10-20	25-75
Meso-eutrophic	20-35	-
Eutrophic	35-100	>75
Hyper-eutrophic	>100	-

2 Nutrients as drivers of fish production: geologic variability in mountain streams

2.1 Abstract

Nutrient concentrations in streams are dependent on catchment geomorphology, climate and land use. Without anthropogenic sources, stream parent lithology controls the natural variability of nutrients in aquatic systems. Total phosphorus has been established as a bottom-up driver of productivity; therefore, characterising the water quality attributes of the underlying geology will help identify how variation of nutrient regimes in stream environments affects productivity of fish communities. To establish baseline relationships between nutrients and fish biomass at a local scale, I compared streams with metamorphic and granite intrusion lithology to sedimentary streams dominated by limestones and shales in the Purcell and Southern Canadian Rocky Mountains. Catchment geology, water quality, nutrient regime, invertebrate sampling and two-pass depletion electrofishing in a closed site were used to identify the relationships between nutrient limitations and fish production. Nutrient and chemical parameters including Total phosphorus, alkalinity, conductivity, DIC and pH were greater at limestone sites than granite sites and fish population differed in biomass, production, and abundance. Using Total phosphorus as our aquatic productivity metric, evaluation of the physical properties in aquatic systems will help to identify geographic variability of nutrient regimes, and their influence on fish communities.

2.2 Introduction

Freshwater productivity studies have focused predominantly on lake ecosystems (Northcote and Larkin 1963, Downing and Plante 1993), using a variety of variables to predict whole-system productivity. Productivity calculation is labour intensive requiring multiple time consuming field and lab measurements, therefore river studies have used surrogates including total suspended sediment (Northcote et al. 2005), alkalinity (Kwak and Waters 1997), conductivity (Mazzoni and Lobon-Cervia 2000), and phosphorus concentration (Peterson et al. 1993, Rosemond et al. 1993, Castillo et al. 2003). All of these surrogate variables are related to the ionic content of the water, and are dependent on regional scale parameters such as climate, flow regime, and geology (Lefebvre et al. 2005, Wohl et al. 2007). Whether considered a surrogate or a direct driver of bottom-up processes, total phosphorus (TP) has been correlated with increased primary production (Rosemond et al. 1993), influencing biological community biomass and ecosystem function of the watershed (Hoyer and Canfield 1991, Freeman et al. 2007, Lewis and McCutchan 2010).

From a regional scale, landscape differences across much of Canada can be broadly identified as shield or off shield environments (granite vs. sedimentary landforms), a common approach used to identify chemical and biological trends between watersheds (Horton et al. 1999, Wohl et al. 2007). Granite catchments are considered soft water systems with low ion content, due to low weathering rates and solubility of the watershed material (Camargo et al. 2005, Brown et al. 2011). Alternatively, sedimentary drainage basins have high erosion rates and weathering potential, and therefore a greater availability of soil minerals results in high nutrient concentrations. High nutrient variation is often attributed to differences in land use, and potential point and nonpoint source nutrient inputs (Vitousek et al. 1997). Rivers in Norway exhibit high variability in water chemistry on a local scale due to their diverse parent lithology, and can therefore make diagnosing anthropogenic impacts difficult (Reimann et al. 2009). Mountainous regions can be spatially heterogeneous in productivity; therefore, baseline nutrient regimes characterized by their catchment lithology will help to detect these spatial anomalies. As compared to lower elevation watersheds, mountain systems higher in the watershed provide an opportunity to assess nutrient variability with minimal interaction with urban and agriculture inputs.

Nutrient concentrations in water are comparable field measures that can be used as a baseline productivity metrics. TP has been developed as an appropriate index for measuring phosphorus availability in aquatic systems for it includes bioavailable dissolved organic and inorganic phosphorus, and internally loaded particulate phosphorus which serves as future phosphorus reserves (Sickman et al. 2003). Available P is dependent on chemical content of watershed surface geology with small quantities naturally available through rock weathering, many aquatic systems worldwide are P limited (Moss 2010). Alternatively, in eutrophic systems, hyper-production of primary producers has been the focus of many studies in both lakes and rivers. Anthropogenic fluxes including point and non-point sources, in addition to experimental nutrient enrichment studies, have been a strong defense for the central role P plays in bottom-up processes of community biomass and production (Stockner and Shortreed 1978, Benke and Huryn 2010). In oligotrophic coastal streams, even a very small amount of P addition to a stream increased the productivity twofold (Bothwell 1989), indicating that even on the lowest end of nutrient availability, significant differences in biomass can be found. This is significant from a management perspective for salmonid systems that historically depended on nutrient subsidies from migratory fish spawning migrations (Sanderson et al. 2009, Wipfli and Baxter 2010). Subsidies to systems with low production and nutrient concentrations will depend on the watershed's connectivity to downstream sources (migratory fish) or to upstream landscape sources of nutrients. Increased human disturbance and fragmentation of aquatic systems has necessitated a greater understanding of the driving influences of the productivity sustaining aquatic communities (Woodward and Hildrew 2002).

2.2.1 Objectives

Across British Columbia, a geographically diverse province, expected values for productivity in rivers will reflect their regional geology, land use, and climate characteristics. At a local scale, a comparison of nutrient variation between two geologically different watersheds will provide a high enough resolution to determine if there is a response in fish productivity of the community. Our objectives are:

1) To examine natural variability of nutrient concentrations in 14 rivers across two geologically distinct mountain regions (sedimentary verses granite intrusion lithology).
2) To identify reach-scale relationships between nutrient concentrations (Total phosphorus) and fish productivity between regions.

We expect that nutrient concentrations (predominantly total phosphorus) will be greater at sedimentary sites than granite sites and that total phosphorus will be a strong predictor of aquatic productivity.

2.3 Site description

2.3.1 Geologic setting

At the headwaters of the Kootenay-Columbia River system, the Purcells are the southernmost range in the Columbia Mountains extending north to south, confined by the Rocky Mountain trench to the east and Kootenay Lake to the west. The southern Canadian Rocky Mountains (Continental Ranges) rise to the east of the trench and extend along the BC and Alberta border. Mountain building of the Canadian Cordillera developed in a southwest to northeastern direction, making the Purcell Mountains older (185ma) than the Southern Canadian Rocky Mountains (100ma). Due to differential erosion rates and time of exposure, there is a continuum throughout the cordillera where ranges in the southwest are predominantly lower in elevation than mountain ranges in the northeast (Rockies). Relative to the Rocky Mountains that consist of less weathered, rugged peaks, the southern Purcell Mountains are more rounded with older exposed rock.

The Purcell Mountain Range is within the Belt-Purcell Super-group, with exposed layers composed of quartzite, argillite and mafic sills, a geologic sequence that extends within the Purcell region of the East Kootenays south through Montana and into Idaho (McFarlane and Pattison 2000, Joncas and Beaudoin 2002). Multiple metamorphic events led to the shifting of faults and intrusion of basement Precambrian granite throughout the meta-sedimentary layers (Paiement et al. 2012). Precipitation of ore materials during the transformation of metamorphic rock deposited mineral veins (Fe, Pb, Au, Ag, Zn) that weave along these lateral sills and fault lines of the Purcell Super-group (Joncas and Beaudoin 2002). Historic Sullivan and Moyie deposits (Hoy et al. 2003), and numerous small placer mines and claims have been the focus of geologic exploration and extraction for over a century.

The Rocky Mountains are comprised of predominantly Paleozoic sedimentary rock (calcareous shale from historic inland marine deposits) with little exposed basement geology (Bally et al. 1966). Although close in geographic range, these two mountain systems exhibit unique sedimentary and granite intrusion surface geology resulting in different erosional resistance properties which modifies flow and potential nutrients in solution (Valett et al. 1996, Bouchard and Jolicoeur 2000).

2.3.2 Study Site

The study was carried out in 14 forested mountain streams, seven in each of the two regions, at equivalent elevations (approximately 1500m) upstream from the Elk and St. Mary River valley alluvial deposits (Hoy et al. 2003) (Figure 2-1). Both regions are resource rich, with resource extraction occurring on small and large scales. In both regions, all study sites were above active logging, coal and placer mining, and upstream of urban point and non-point sources of nutrients. Because of this, these systems are referred to as ‘pristine’ relative to downstream reaches. The two mainstem rivers draining these regions include the St. Mary River (Purcells) and the Elk River (Rockies) which converge into the Kootenay/ Columbia Pacific drainage system. Because specific location in mountain watersheds can be an indicator of climate influences on the flow regime

(Wohl et al. 2007), elevation was used as a consistent physiographic measure to alleviate bias in site selection. With stream size standardized, elevation would also provide comparability in location within the watershed as an indicator of water temperature which is a known influence of fish growth and distribution (Jensen 1990, Parra et al. 2009).

Both drainages exhibit relatively high relief (Brown et al. 2011) with peaks above elevations as high as 3000 m. The Southern Purcells within the study area watersheds are smoothed and tree covered, 300-600 m lower than the Rockies region. Both regions have similar hydrographs, dominated by spring snowmelt and rainfall events during the summer. Peak discharges occur in late May due to snowmelt dominated hydrographs (Water Survey of Canada Accessed March 2013), with declining flows until Autumn rain events; lowest stream flows occurr in the winter from November to March. Rivers were chosen for perennial flow, including habitat access for all life stages of local fish fauna. Both regions are known to have relatively low species richness compared to other regions across Canada (Chu et al. 2003), but comparable to other salmonid dominated systems (Cote et al. 2011). Local fish species sampled at higher elevations in the watershed include native westslope cutthroat trout (*Oncorhynchus clarkii lewisi*), bull trout (*Salvelinus confluentus*), mountain whitefish (*Prosopium williamsoni*), sculpin species (*Cottus species*), and long nose dace (*Rhinichthys cataractae*). Non-native brook trout (*Salvelinus fontinalis*) and rainbow trout (*Oncorhynchus mykiss*) are invasive due to historic and current stocking programs (DeMarco 1993).

2.4 Materials and methods

2.4.1 Sampling

Following initial investigation using mapping software, topographic maps and air photos, stream attributes were analyzed to ensure that stream characteristics including watershed size, wetted width and discharge were comparable. Contributing watershed areas were calculated using Arc GIS 10.0, with a 1:50000 DEM. GIS mapping software was also used to determine catchment characteristics including surficial geology, regional elevation ranges, and identify comparability of stream order. In July 2011, after freshet, 20 sites were visited and a coarse site survey was completed to allow for exclusion of sites that exhibited differences in stream size, obvious anthropogenic inputs of nutrients including treatment facilities, agriculture, rangeland, and mining activity. Field methodology was developed in the winter of 2011, while the field sampling season occurred May-September 2011.

2.4.2 Habitat characterization

Stream habitat variation was quantified using a modified fish habitat assessment procedure, including 20 measured field habitat parameters (FHAP) (Johnston and Slaney 1996). Reaches of approximately 150 m were broken down into habitat unit type, including riffle, glide, and pool, and cross sectional and longitudinal measurements of each habitat unit were taken to establish metrics weighted to area. Riffles and glides habitat units were combined and described as fast water, including riffle and non-turbulent depending on stage height. Pools, or slow water, were defined using a minimum residual pool depth (difference between pool crest depth and maximum pool depth) to ensure that they act as an adequate cover element related to the wetted width of the reach (Johnston and Slaney 1996). Reach scale habitat variables measured at each site included riparian vegetation and forest type, and percent cover of instream cover elements (visually estimated total % of habitat unit area with undercut banks, deep pools, overhanging vegetation, woody debris or large boulders). Stratified by habitat type, cross-sectional depths, wetted and bank full widths, and bankfull height were recorded. Quantification of substrate distribution was completed using a Wollman pebble count, including a zigzag sampling pattern of 100 rocks through habitat units proportional to their occurrence in the total reach. D_{50} was calculated from the width of the intermediate axis following the substratum classes of the Wentworth scale (fines<2 mm, gravel 2-64 mm, cobble 56-256 mm, boulder 256 + mm) (Gordon et al. 1992). Streambed slope was measured using a hand held clinometer for each habitat unit, and a weighted average for habitat unit area was used. Discharge was calculated as the product of the rivers cross sectional area and velocity. Flow was measured using a Swoffer velocity meter across a representative transect at 15-20 intervals along the tape depending on wetted width, at 60% of depth. A weighted mean of all habitat variables by proportion of habitat units was calculated within each reach, and the mean values of all seven sites were compared between regions.

2.4.3 Water quality

The twelve water quality parameters measured at each site included TP, TPP, SRP, TN, NO_2+NO_3, DIC, temperature (^oC), dissolved oxygen (mg L^{-1}), pH, and conductivity (µS cm^{-1}). Temperature, DO, pH, and conductivity were measured using a multi-parameter YSI and alkalinity (mg L^{-1} $CaCO_3$) was determined using a Hanna Instruments chemical titration kit. Water sampling (2-3 replicates per site) was completed in the summer of 2011. The study was designed to compare between regions, therefore individual chemical profiles and long term temporal trends of each stream were not examined. Grab samples during low flow served as an indicator of baseline water quality within a site. Samples were taken in 500mL acid washed polyethylene nalgene bottles and nutrient concentrations were measured as Total Phosphorus (TP) and Total Nitrogen (TN). TP was determined using persulfate oxidation, and TN was analyzed with the chemical digestion flow injection method (Ng 2010). Additional water quality analyses included dissolved inorganic carbon (DIC), combination of nitrite and nitrate (NO_2+NO_3) and total particulate phosphorus (TPP). Sub-samples were filtered into a 15mL centrifuge tube in the field using a 0.7um GF/F syringe filter for both DIC and for NO_2+NO_3. DIC sub-sample was left with no headspace in the centrifuge and 12mL of filtrate for NO_2+NO_3 analysis was acidified with 20uL Sulfuric acid (H_2SO_4) (Ng 2010). TPP was total weight of particulate remaining on a 0.45µm filter. Water duplicate samples were taken for every 10 field samples to ensure that contamination and field treatment variability could be quantified. Water was cooled at 4^{o}C and shipped to the University of Alberta Biogeochemical lab for analysis.

2.4.4 Aquatic invertebrate sampling

Variation in the presence or absence of benthic insects will help to identify trends in fish food found in the streams, as a useful measure of lower level trophic response to primary productivity (Huryn et al. 1995, Lewis and McCutchan 2010). At each sampling site, benthic aquatic invertebrates were collected in riffles (velocity and depth compared between sites) using a 343 micron mesh surber sampler (0.09 m^2 sample area). All invertebrate samples were collected within a two week period in August, to limit variation in invertebrate biomass due to seasonality. Three samples were collected along a cross-section from margins to mid-channel within each of the electrofishing sites, controlling for sampling area, time and effort. Samples were field sorted, frozen, counted and identified to Order in the laboratory (Clifford 1991). Samples were dried at 60^oC drying oven for 48 hours to reduce variation in species moisture composition, and dry weight biomass, density, and % EPT (Ephemeroptera, Plecoptera, and Trichoptera) composition metrics were reported.

2.4.5 Fish sampling

River reaches (mean site area = 860 m^2) were sampled by a two person wading crew using a Smith Root LR24 backpack electrofisher. In small wadeable streams, two pass depletion removal can provide robust estimates of population size and biomass (Randall 1995, Lockwood and Schneider 2000). Sites were isolated with mesh block nets at the upstream and downstream end of the reach to maximize the capture efficiency and minimize the movement of fish either in or out of the closed fish biomass estimate (Peterson et al. 2005). Electrofisher power settings were determined according to conductivity and temperature measures taken before sampling, and a consistent two hour cool down period was taken between passes. Two-pass depletion estimate calculation depends on the calculation of the rate at which catch decreases between subsequent passes. With the assumption that catchability would not differ between passes or among rivers, we calculated the population estimate using an average catch probability estimate between passes among all sites. Using this rate of decline between successive passes, I calculated a cumulative estimate of the fish population for each site.

For each fish caught, non-lethal measurements were recorded including species, sex, maturity, fork length (mm), and weight (0.1 g). Scale samples were removed from behind the dorsal fin, anterior to the lateral line, and dried in coin envelopes (Mackay et al. 1997). Scales were sorted and at least 4 scale impressions per fish were made on acetate slides and viewed on either a 4X or 10X dissecting microscope and digitally photographed and catalogued using Amscope software ©. Radial distances from scale focus to annuli were measured using ImageJ ©. Measures of scale increments were replicated and mean values were used for back-calculation of fish length (Pierce et al. 1996). Fish length-at-age was determined using Fraser Lee scale length at age back-calculation method (Devries 1996):

$$L_i = [(L_c - a)/S_c] \times S_i + a$$

Where L_i=back calculated length of fish *i*th increment, L_c=fork length of fish at capture, S_c=total scale radius, S_i=radius at *i*th annulus, and a=intercept parameter from linear regression between scale length and fork length at capture, as a correction factor (Klumb et al. 2001). Fish that did not have scale age data were assigned ages based on length increments calculated by species from aged fish for each region. Weight at age was determined using back calculated lengths and species specific regional length weight regressions generated from region specific and collected data. Specific growth rates were calculated using:

$$\text{SGR } (g_t) = [\log W_2 - \log W_1)/(t_2 - t_1)]$$

Where t= time (years) between successive age intervals, W_2-W_1= difference in weight of fish between this interval, and therefore SGR (g_t) = specific growth rate over time t. Due to limitation in external gender determination, and diverse spawning habits of fish in the system, a GSI (gonadal somatic index) correction was not applied to account for the proportion of fish body weight associated with gonads (Strange 1996). Biomass was calculated as the mean mass (g) of the individual fish at age per meter square of habitat. For each species of fish, individual cohort production was calculated as:

$$P_t = g_t B_t.$$

Cohort production (P_t) was calculated according to Ricker (1975) over an annual time interval (t), as the product of the specific growth rate (g_t) and biomass (B_t) per year (t_1 to t_2). Total site production was calculated as a sum of all cohorts, for all species. Biomass is a simple field measure that directly relates to the richness of the system, but does not account for age and size structure (Cote 2007). Production calculation can incorporate community structure and therefore energy shifts within the ecosystem (Cyr and Peters 1996). Production to biomass ratios were calculated for each site as rate indicating population turnover (Robertson 1979). Differences in fish communities were assessed between regions using Shannon's and Simpson's diversity indices (1-D), calculating the evenness of species abundance within each site, and the diversity index calculated for the number of species relative to the total abundance of the site.

2.4.6 Statistical analyses

Principle component analysis (PCA) was used to recognize patterns in summarizing the habitat and chemical explanatory parameters between regions. Habitat parameters included habitat type (%fast and slow water), discharge (m^3), D_{50}, substrate composition, wetted width, bankfull height, water depth, large woody debris (LWD), and percent cover (adjusted to site area). Chemical parameters included TP, TN, NO_2+NO_3, DIC, TPP, alkalinity ($CaCO_3$), conductivity, pH, dissolved oxygen (DO) and temperature (^{o}C). PCAs were completed using arithmetic mean values for each region, weighted to site area for a more accurate proportion comparison where appropriate. Ultimately, the lack of interaction between the habitat variables would provide confidence in our site comparability whereas chemical parameter interactions would help direct the statistical approach between regions. Data were log-transformed where it did not satisfy the assumptions of a parametric test (assessed by the Shapiro-Wilk test for normality), stabilizing the variability in the data. All water quality variables violated the Shapiro-Wilk test for normality and therefore data sets were log transformed before analysis. Once assumptions were verified, a two sample independent t-test was used to compare between the two region's mean chemical, habitat, and fish population variables. A significance value of p=0.05 was established for all tests.

2.5 Results

2.5.1 Habitat comparison

Using PCA analysis, all chemical water quality parameters and all habitat variables were compared on two biplots along with individual sites (Figure 2-2). Parameters that were controlled for (site length, elevation, site area) were not used in the analysis. The first two components account for 83% of the variance in data of the chemical biplot, and accounted for 66% in the habitat biplot. The habitat variables did not exhibit the differentiation that the chemical biplot showed between regions.

The 14 sites varied in whole watershed area with a mean of 49.94 square kilometers (Table 2-1), with the smallest area for West Alexander Creek and the largest for Meachen Creek.

The Rockies and Purcell habitat variables were averaged among habitat units within each reach, and the mean values of all seven sites were compared between regions (Table 2-2). Using t-tests, no significant difference was detected between regions for discharge ($t(_{12})=0.70$, $p=0.50$), Wollman substrate analysis D50 ($t(_{12})=0.63$, $p=0.54$), % cover ($t(_{12})=1.73$, $p=0.11$), slope ($t(_{12})=0.57$, $p=0.58$), abundance of LWD ($t(_{12})=0.72$, $p=0.49$), mean depth ($t(_{12})=0.08$, $p=.94$), mean wetted width ($t(_{12})=1.69$, $p=0.13$), and bankfull height ($t(_{12})=1.12$, $p=0.29$). Habitat units were grouped together for analysis as fast water (riffle and glide) and slow water (pools) and no significant difference was detected between regions for % slow and % fast water ($t(_{12})=0.43$, $p=0.67$). Elevation, site length and area were controlled for in the sampling design and therefore no difference was identified between regions for these variables. In combination with the exploratory PCA, t-test results identified no difference between the Rockies and Purcells based on fish habitat variables.

2.5.2 Water quality

Water quality parameters demonstrated grouping behaviour for sites by region in the PCA analysis (Figure 2-2). T-tests were used to compare between the two regions (Table 2-3), using a mean for all measurement values collected during the two site visits in July and August for the collection of fish, invertebrate and habitat data. A significant difference was found between regions for TP concentrations (t($_{12}$)=2.66, p=0.02), DIC(t($_{12}$)=3.56, p<0.01), conductivity (t($_{12}$)=3.54, p<0.01), alkalinity (t($_{12}$)=3.56, p<0.01), and pH (t($_{12}$)=6.17, p<0.0001). No significant difference was found between regions for TN (t($_{12}$)=1.48, p=0.16), NO_2+NO_3 (t($_{12}$)=0.84, p=0.42), or for the particulate and dissolved forms of Phosphorus, TPP (t($_{12}$)=0.11, p=0.91), and TDP (t($_{12}$)=1.54, p=0.15). No difference was found between regions for site DO (t($_{12}$)=1.88, p=0.09) or surface water temperature (t($_{12}$)=0.42, p=0.69).

When compared to long term monitoring TP values (median) of mainstem Elk, Kootenay and St.Mary Rivers, our trophic state and values were consistent with their range of 5-14 µg L^{-1} (Pommen 2001). The only TP value that was outside of this range was Lizard Creek, in the Rockies (22µg L^{-1}), which is consistent with Elk River Watershed summer monitoring values for this site (ERA 2013, personal communication).

2.5.3 Aquatic invertebrate biomass

Total invertebrate biomass was significantly higher at the Rockies sites compared to Purcell sites (t($_{12}$)=4.2, p<0.001), with a mean (dw) biomass of 1.48 ±SE 0.23 g m^{-2} and 0.53 ±SE 0.08 g m^{-2} respectively (Figure 2-3). Identification to the scale of Order did not allow for an extensive analysis of invertebrate diversity composition as an aquatic health indicator. No significant difference was found between regions for their % composition of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) (EPT %), a common biomonitoring tool.

2.5.4 Fish biomass, growth, and production

Arithmetic mean biomass differed between regions ($t_{(12)}$=4.08, p=0.0015) with lower minimum values occurring in Purcell streams (mean of 0.516±0.10 g m^{-2}) compared to Rocky streams (mean of 1.82±0.30 g m^{-2}) (Table 2-4). There was no difference between the regions in the proportion of fish caught on the first to second pass ($t_{(12)}$=0.585, p=0.28). Similar to Reid et al. (2009), capture probability was highly variable between passes among all sites from 0.23-0.93. Interestingly, when we calculated biomass caught between passes, the variability of the catch proportion was the same across all sites (0.2-1.09). Because there was no relationship between first and second passes and the large range of values, we calculated the capture probability across all sites in both regions (0.65), rather than by individual site for the population estimate. Calculated population density for each region ranged from 10-40 m^{-2} and 21-55 fish m^{-2} for the Rockies and Purcells, respectively, as compared to the actual abundance (sum of two passes) range of 6-21 m^{-2} and 14-29 m^{-2} (Table 2-4).

Relatively low species richness was identified at all sites, but our values were comparable with low species abundance found in higher altitude, low order mountain streams (Santoul et al. 2004) (Table 2-4). Simpsons index of diversity (1-D) indicates the greater the value, the greater the diversity. Shannons index of evenness is a metric for how equal the abundance of each species is within the community, and will approach zero as one species dominates. There were no differences between regions for species richness ($t_{(12)}$=0.99, p=0.17) or evenness ($t_{(12)}$=0.52, p=0.31).

Fish ages were assigned with confidence for 60 fish that had readable scales, 26% of the total fish caught. Scale ageing can be subjective and therefore a secondary scale reader reviewed 25% of the scales, randomly selected across age class and species. Comparison showed no significant difference between ages of two reviewers, and discrepancies between ages were settled by a third reviewer. Westslope cutthroat trout (WCT) comprised of 57% of the fish abundance caught, and were present at 13 of the 14 sites. Although specific growth rates were calculated for all species for production estimates, WCT growth was compared between regions (Figure 2-4). The growth rates of the Rockies streams (adjusted to age by length increments) were noticeably higher for the smaller fish, before levelling off consistently between regions for fish over 4+. The same trend was observed when growth rates for all salmonid species were combined, where growth was higher in the first three years as compared to the Purcells.

In our sites both biomass and production were higher in the Rockies (Table 2-4). The Purcells had lower growth rates, biomass and production. However, P:B ratio, the population turnover rate, was higher than the Rockies (Figure 2-5) along with fish abundance and density. To put these mountain stream sites in context on the productivity scale, sites graphically compared with a more wide scale biomass/ Total Phosphorus (TP) relationship from measured values in salmonid dominated systems in Newfoundland (Figure 2-6). TP data encompassed a broader nutrient concentration spectrum, from nutrient poor to enriched sites influenced by urban inputs. This strengthened the comparison, providing a continuum for the linear TP/biomass relationship (R^2=0.728).

2.6 Discussion

In close proximity to one another, the Purcells and Southern Rocky Mountains are cold interior salmonid streams with comparable climate, flow regime and elevations.

Relative to lower elevation high order rivers, both systems fall on the lower end of the spectrum for productivity and nutrient enrichment. Even so, through comparison of rivers across these two regions, unique geology of each catchment played a role in the significant differences found in water quality and fish populations.

Rocky Mountain streams, with sedimentary catchments, had a higher ionic content than granite intruded systems in the Purcells. Conductivity, alkalinity and DIC, which are surrogate measures of total dissolved solids, Ca^{2+}content and high CO_2^{-3} concentration, were all higher in sedimentary streams. Granite intruded systems were less basic, with a difference in mean pH of 0.5, with no overlap in range (Table 2-3). The logarithmic scale of pH units indicates an exponential increase between regions, where a higher pH in the Rockies may increase nutrient solubility and potentially expedite the dissolution of the sedimentary minerals. Total and inorganic nitrogen values did not vary between systems, and fell within a natural range of nitrogen found in aquatic systems worldwide (50-150 µg L^{-1}, (Moss 2010)). TN values in the mainstem Elk, Kootenay and St.Mary Rivers were highly inflated compared to our lower order, upper watershed sites. This shift in TN downstream is potentially due to local anthropogenic influences including urban effluent or explosives used in overburden mining (Nagpal 1982).

P values were expected to be higher in the Rockies region due to apatite in sedimentary rocks. Total P concentrations and a greater proportion of dissolved P was bioavailable in Rockies systems. A minimal shift in bioavailable P can drive dynamic biologic processes influencing fish production in the system; an indication of the limiting nature of oligotrophic streams (Hill et al. 2010).

Higher elevation low order streams carrying capacity for fish can be limited by temperature (Parra et al. 2009), habitat (Poff and Huryn 1998), and food availability (Wipfli and Baxter 2010). Temperature variability can be a driver of shifts in productivity in invertebrates (Benke 1993) and fish production (Dineen et al. 2007), where warmer waters initiate higher metabolic rates and faster growth to an optimal point. Both temperature and oxygen, abiotic factors known to limit fish distribution (Armstrong et al. 2003), were comparable across all sites. Elevation of study sites was controlled to fall between 1100-1550m for all sites, within a range consistent with supporting all life stages of trout distribution (Kruse et al. 1997). Habitat availability and complexity, including shifts in depth and velocity, are important to accommodate multiple species and life stages in the stream (Fausch et al. 1988). There were no significant differences between the 20 measured habitat variables between regions, therefore fish biomass and production differences between regions were not confounded by the quantity of high quality habitat (Bosek and Hubert 1992).

Variation in nutrient concentration and food (invertebrate) availability in these two watersheds would influence their potential to support aquatic populations. Benthic invertebrate biomass was higher in the Rockies (high nutrient sites), consistent with characteristics of limestone dominated systems in other studies with higher dissolved nutrients and stable temperatures (Wohl et al. 2007). An increase in invertebrate biomass provided an initial indication in this study of an ecosystem response to nutrient content in steams.

Closed two-pass electrofishing increased confidence in the comparability of fish biomass estimates across different watersheds. The assumptions of a two pass depletion are that the probability of capture between passes must be equal (Lockwood and Schneider 2000), no changes in environmental conditions and effort occur, and there is no emigration in and out (Schnute 1983). All other assumptions met, capture probability could be influenced by size and life histories of species present (Reid et al. 2009). Larger fish may inflate the data for a population's size structure with only a single pass, as smaller species and sizes are more cryptic (Otis et al. 1978). For the purpose of this study, with lower expected abundance and biomass in upper watershed mountain systems, two passes in a closed system would provide a stronger estimate of biomass and age classes present.

Fish biomass was significantly higher in Rockies systems, corresponding with higher P concentrations and invertebrate biomass between regions. Although biomass is more straight forward to estimate, fish production values are important indicators of ecosystem status. Calculation of production is rigorous and requires detailed information about fish community age and growth on a temporal scale (Jonsson et al. 2011). Production values calculated (Table 2-4) were comparable to lower range of other studies in low fertility northern regions including Norway (0.1-6.1 g m^{-2} (Power 1973)), and two regions in Newfoundland including Terra Nova Park (0.32-10.98 g m^{-2} (Cote 2007)) and Copper Lake headwaters (0.99-6.51 g m^{-2} (Clarke and Scruton 1999)). Mean biomass values of our sites were also comparable to granitic coastal streams of British Columbia (1.81 g m^{-2} (BC Hydro 2004 unpublished)), well below the estimate of 10-30 g m^{-2} developed as a range for highly productive trout streams (Kwak and Waters 1997).

Fish growth rates can be influenced by abiotic and biotic factors providing an indication of environmental condition of the watershed. Therefore, fish growth can be highly variable regionally within the same species (Carlson et al. 2004). Cutthroat trout growth rates were higher in nutrient rich sites as compared to the nutrient poor Purcell region, moreso in the youngest fish. Higher growth rates can be partially attributed to optimal temperatures and resource availability (Dmitriew 2011), therefore a higher cost is associated with feeding in systems with limited resources. A study of nutrient addition in an oligotrophic river in Alaska by Peterson et al. (1993), found both growth and biomass were nutrient limited. Enrichment projects experimentally demonstrate how small increases in the nutrient regime will influence ecosystem function in nutrient poor systems. Therefore, higher trout growth rates in nutrient rich rivers of the Rockies supports the bottom-up relationship with nutrient concentrations and productivity found in enrichment projects. The Rockies streams in our study exhibit higher phosphorus concentrations, invertebrate biomass as potential food availability, fish growth rates and fish biomass as compared to the more nutrient poor Purcell streams.

Differences in P:B ratio (production: biomass) will be present between or within species with faster growth rate : smaller size. P:B therefore is an indicator of the rate of turnover of the community, and is controlled by the lifespan and size of the population (Robertson 1979). Production calculations revealed that the Purcell region’s fish population was composed of a greater proportion of 1-3 year old fish than the nutrient rich systems. A greater abundance of small fish increased the P:B ratio as a measure of an ecosystem with a quick turnover rate. Productivity is known to have a diminishing returns curve where, as nutrients increase, production will increase until an optimum level is reached (Ryder 1978). In the case of eutrophic systems, high nutrient levels would begin to decrease production of a population due to stress and anoxic conditions (Jonsson et al.

2011). In rivers with low nutrients, the carrying capacity of a system may be limited by its nutrient regime, where, similar to Cote (2007), larger fish are moving downstream into areas with higher resource abundance. Spatial distribution of fish is a balance of energy gain and risk (Heggenes et al. 1999), where enriched habitats will sustain larger individuals in the population. Areas of high food abundance due to bottom up processes have less competition than those with low resource availability (Chapman 1966).

Fish abundance was significantly higher in nutrient poor systems, with a greater proportion of small, juvenile trout. Production calculations revealed that older, larger fish did not make up a large proportion of the community in nutrient poor sites as compared to the rich sites of the Rockies where larger individuals were sampled within consistent habitat types. Smaller individuals may make up a proportion of a resident population, or could represent migratory populations using upper reaches as rearing habitat for juveniles. The ability of salmonids to move throughout the system to potential population sources, including lakes or higher order rivers, would also increase the likelihood of multi-life histories (Watry and Scarnecchia 2008).

Fish inherently move to optimal habitats daily and seasonally. Fish move to avoid unfavourable environmental conditions, elude competition and predation, and to maximize their efficiency in growth and reproduction (Chapman 1966, Watry and Scarnecchia 2008). Whereas habitat is often the limiting factor in the winter (Cunjak 1996), salmonids require a higher nutrient budget during the summer (Heggenes et al. 1999). Although nutrient concentrations may not be productive enough to support an adult population, low nutrient streams were able to support a cohort of the fish population with a lower metabolic requirement. This difference in metabolic traits has been found between species with native cutthroat trout and invasion patterns of rainbow trout (Rasmussen et al. 2012). Rainbow trout have higher metabolic requirements and are predominantly found in lower elevations of the watershed (Paul and Post 2001). Salmonid systems are often below historic levels and ecosystem saturation, where density dependence factors would play a role in community structure (Regetz 2003). In low order mountain streams in oligotrophic regions, low fertility upper reaches create a resource limited habitat with less competition from larger individuals, able to support and rear fry and juveniles.

Although my study streams were on the lower scale of productivity, these mountain systems still showed a two-fold difference in biomass between regions. Nutrients in natural systems are influenced by catchment characteristics, where unique solute concentrations play an important role in the regional patterns of fish biomass, growth and production. The initial response of an ecosystem is limited by its nutrient regime, where bottom up enrichment triggers an increase in ecosystem production. Nutrient deprived systems could potentially limit the carrying capacity for all life stages of fish in the stream, regardless of optimal habitat requirements.

Baseline nutrient regimes at the lowest end of the spectrum of productivity can help to identify trends in phosphorus and fish in natural systems. Many rivers and streams in Canada are inherently oligotrophic, especially in higher latitudes and elevations, the Boreal region, and habitats of the east and west Coasts that historically depended on marine derived subsidies for seasonal nutrient fluxes. With increasing human alteration of aquatic systems, the threat of continually pressuring these ecosystems is apparent.

This study examined the headwaters upstream of the Elk River, a watershed that has received increasing attention due to implications of coal mining on water quality (Hauer and Sexton 2013). In addition to this, both the Rockies and Purcell regions of southeastern BC drain into the Columbia River system, a river known internationally as one of the most fragmented rivers from hydropower development (Ferguson et al. 2011). With both of these environmental stressors in mind, understanding baseline nutrient characteristics at multiple scales is important for watershed management. Rivers are not homogeneous spatially or temporally, and it is necessary to acknowledge the challenges in quantifying all of the complex chemical, physical, and biological characteristics contributing to ecosystem function.

Table 2-1 Site locations of water quality grab samples, invertebrate surber samples, habitat mapping and electrofishing for both the Rockies (n=7) and Purcells (n=7).

Site	Region	Latitude	Longitude	Watershed Area km^2	Fish Species Sampled
Aldridge	Rockies	50.33530	-114.91035	31.29	WCT, BT, EB
Forsythe	Rockies	50.23279	-114.96854	76.86	WCT, BT, MW, EB
Lizard	Rockies	49.49109	-115.11492	35.50	WCT, BT, EB
Lodgepole	Rockies	49.28188	-114.81516	49.21	WCT
Quarrie	Rockies	50.25983	-114.97463	42.65	WCT, BT, MW, EB
Ram (Big Horn)	Rockies	49.16746	-114.78270	60.44	BT
West Alexander	Rockies	49.76981	-114.71776	16.10	WCT, BT
Buhl	Purcells	49.88765	-116.08793	71.31	WCT
Gold	Purcells	49.31303	-115.68449	26.60	WCT, CC
Mark	Purcells	49.73530	-116.04261	71.37	WCT
Meachen	Purcells	49.54818	-116.46374	79.08	WCT
Perry	Purcells	49.52523	-116.05212	58.46	WCT, EB, RN
Redding	Purcells	49.54603	-116.59387	39.07	WCT, BT
Teepee	Purcells	49.27840	-115.52148	41.28	WCT, LNC, CC

Fish species include WCT= westslope cutthroat trout, BT=bull trout, EB=eastern brook trout, MW=mountain whitefish, RN=rainbow trout, LNC=long-nose dace, CC=sculpin species.

Table 2-2 Summary of relevant fish habitat parameters for all sites in the Rockies and Purcells. Mean values were weighted by area for all habitat units in the reach.

Rockies	Elevation (m)	Site Area (m^2)	Discharge (m^3)	% Fast Water Area	% Pool Area	D50 (mm)	Mean depth (m)	Bankfull width (m)	% Cover	Slope (degrees)	LWD (#)
Aldridge	1508	764.04	0.33	0.87	0.12	75	0.24	7.74	0.09	1.6	2
Forsythe	1436	856.54	1.1	0.62	0.38	52.5	0.37	12.98	0.3	2	29
Lizard	1046	837.34	0.29	0.62	0.38	45	0.33	9.53	0.19	2.4	18
Lodgepole	1279	657.60	0.43	0.74	0.26	89.5	0.36	11.43	0.14	2.3	16
Quarrie	1486	1060.69	0.72	0.73	0.27	130	0.44	9.13	0.21	2.7	30
Ram	1513	752.10	0.23	0.71	0.29	90	0.28	10.14	0.19	1.9	7
W. Alex	1494	736.48	0.18	0.49	0.52	76.3	0.35	8.24	0.11	1.6	9
Purcells	Elevation (m)	Site Area (m^2)	Discharge (m^3)	% Fast Water Area	% Pool Area	D50 (mm)	Mean depth (m)	Bankfull width (m)	% Cover	Slope (degrees)	LWD (#)
Buhl	1490	1120.98	1.53	0.79	0.21	100	0.43	13.73	0.25	2.8	17
Gold	1480	445.88	0.12	0.5	0.5	30	0.3	5.74	0.27	1.2	47
Mark	1424	680.63	0.44	0.72	0.28	120	0.28	9.38	0.28	3.2	4
Meachen	1496	886.60	1.01	0.82	0.18	100	0.38	12.5	0.28	2.7	29
Perry	1230	1144.50	0.34	0.69	0.31	95	0.3	13.6	0.12	2	7
Redding	1478	1080.75	0.98	0.77	0.21	125	0.28	9.78	0.21	2.5	8
Teepee	1106	1020.30	0.35	0.66	0.34	70	0.39	12.8	0.24	1.4	37

Table 2-3. Water quality parameters for regions including mean value (±SE), range, t statistic for Two-sample t-test, and significance value ($p<0.05$) for Rockies ($n=7$) and Purcells ($n=7$).

Parameter		Rockies		Purcells		Test value	
Metric	Units	Mean	Range	Mean	Range	|t|	p
Total Phosphorus	P µg L^{-1}	9.3± 2.3	5-22	4.6± 0.5	3-7	2.6622	0.0207*
TPP[a]	P µg L^{-1}	1.86± 0.45	0.50-4.00	2.00± 0.65	0.50-5.00	0.1097	0.9145
TDP[a]	P µg L^{-1}	7.3± 1.9	3.8-18	2.6±0.61	0.0001-4	1.5433	0.1487
Total Nitrogen	N µg L^{-1}	54.4± 13.0	21-106	32.9±2.5	24-44	1.4810	0.1644
Nitrate +Nitrite	N µg L-1	12.5± 6.2	0.5-46	6.7±6.7	0.5-21	0.8441	0.4151
DIC[a]	C mg L^{-1}	29.6± 0.72	26.7-32.7	13.5±3.7	4.1-27	3.5588	0.0039*
Conductivity	µS cm^{-1}	146.3± 19.5	134-272.9	93.5±20.8	38.5-183	3.5367	0.0041*
pH[b]	[H+]	8.0± 0.06	7.7-8.25	7.5±0.04	7.5-7.7	6.1724	0.0001*
Alkalinity	mg L^{-1} $CaCO_3$	246.9± 6.0	222.5-272.5	112.6±30.6	34.2-225	3.5593	0.0039*
DO[a]	mg/L	9.4± 0.3	8.0-11.0	8.5± 0.3	8.0-10.0	2.1593	0.0518
Temperature	^{o}C	9.6± 1.2	5.7-13.9	9.1± 0.36	7.7-10.5	0.4223	0.6803

* Denotes significant value $p<0.05$

[a] DIC=Dissolved inorganic carbon, TPP= Total particulate phosphorus, TDP= Total dissolved phosphorus, DO= Dissolved Oxygen

[b] pH reported as median values

Table 2-4. Summary of sample reach fish data including biomass, production, density, abundance and production to biomass ratio (P:B).

Parameter		Rockies		Purcells		Test value	
Metric	Units	Mean	Range	Mean	Range	|t|	p
Biomass	g m^{-2}	1.82±0.30	0.95-3.01	0.516±0.10	0.12-0.90	4.0849	0.0015*
Production	g $m^{-2}yr^{-1}$	1.067±0.14	0.57-1.63	0.309±0.05	0.12-0.54	5.6059	0.0014*
P:B	yr^{-1}	0.648±0.05	0.49-0.91	0.790±0.09	0.52-1.25	2.2537	0.0651
Density	# m^{-2}	0.016±0.002	0.008-0.025	0.0234±0.004	0.016-0.043	1.6387	0.1272
Species Richness	#	2.7±0.5	1-4	1.7±0.3	1-3	1.6814	0.1185
Abundance	#	13.28±2.2	6-21	19.57±1.81	14-29	2.2055	0.0477*

*Denotes significant value p<0.05

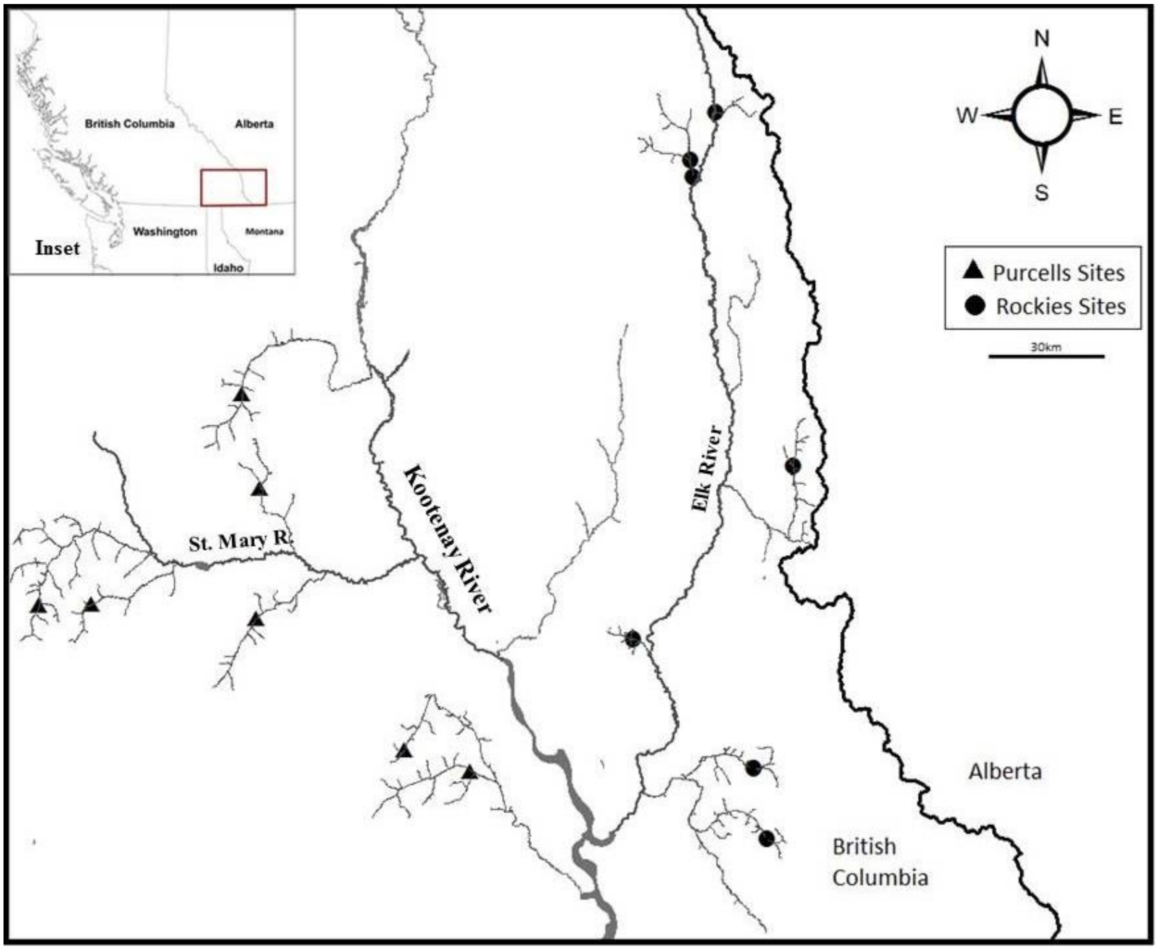

Figure 2-1 Site map (with larger scale inset) for Purcell and Rockies sampling locations

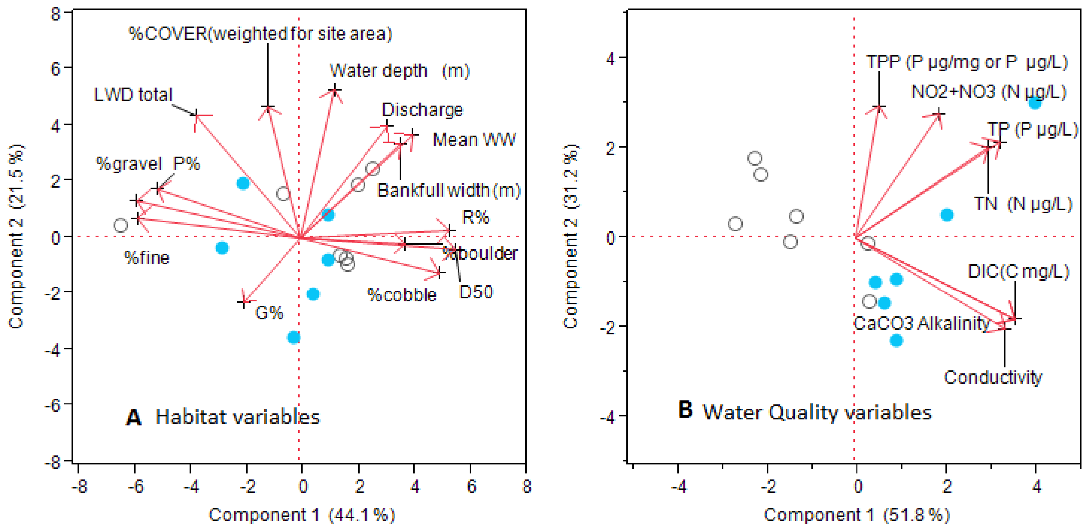

Figure 2-2 Principle component analyses (PCA) investigating variability in (A) habitat variables and (B) qater quality variables for each site (solid circles are Rockies sites, hollow circles are Purcell sites.

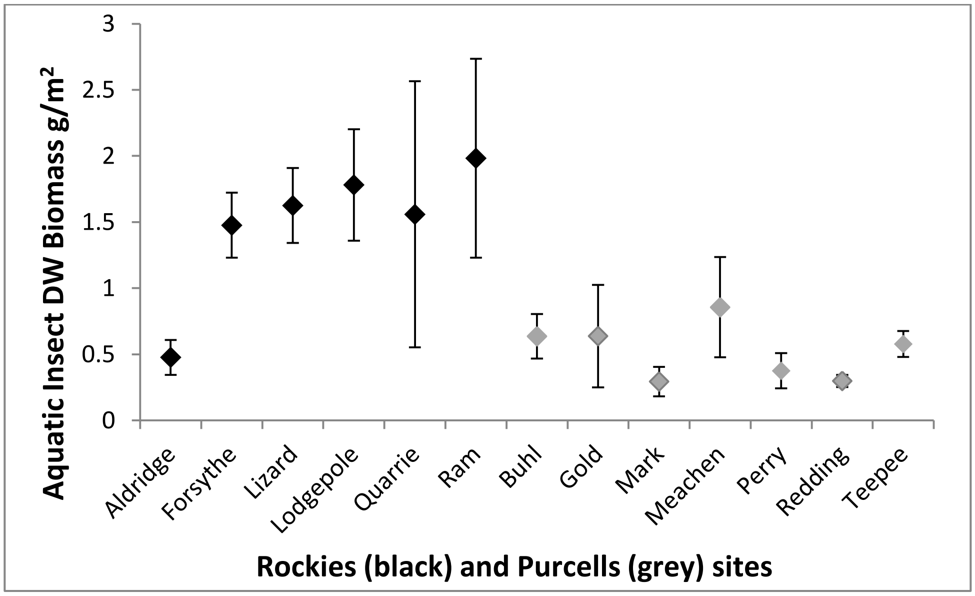

Figure 2-3 Aquatic invertebrate biomass (g of dry weight; SEM bars) from 3 replicates for Rockies (black) and Purcell sites (grey).

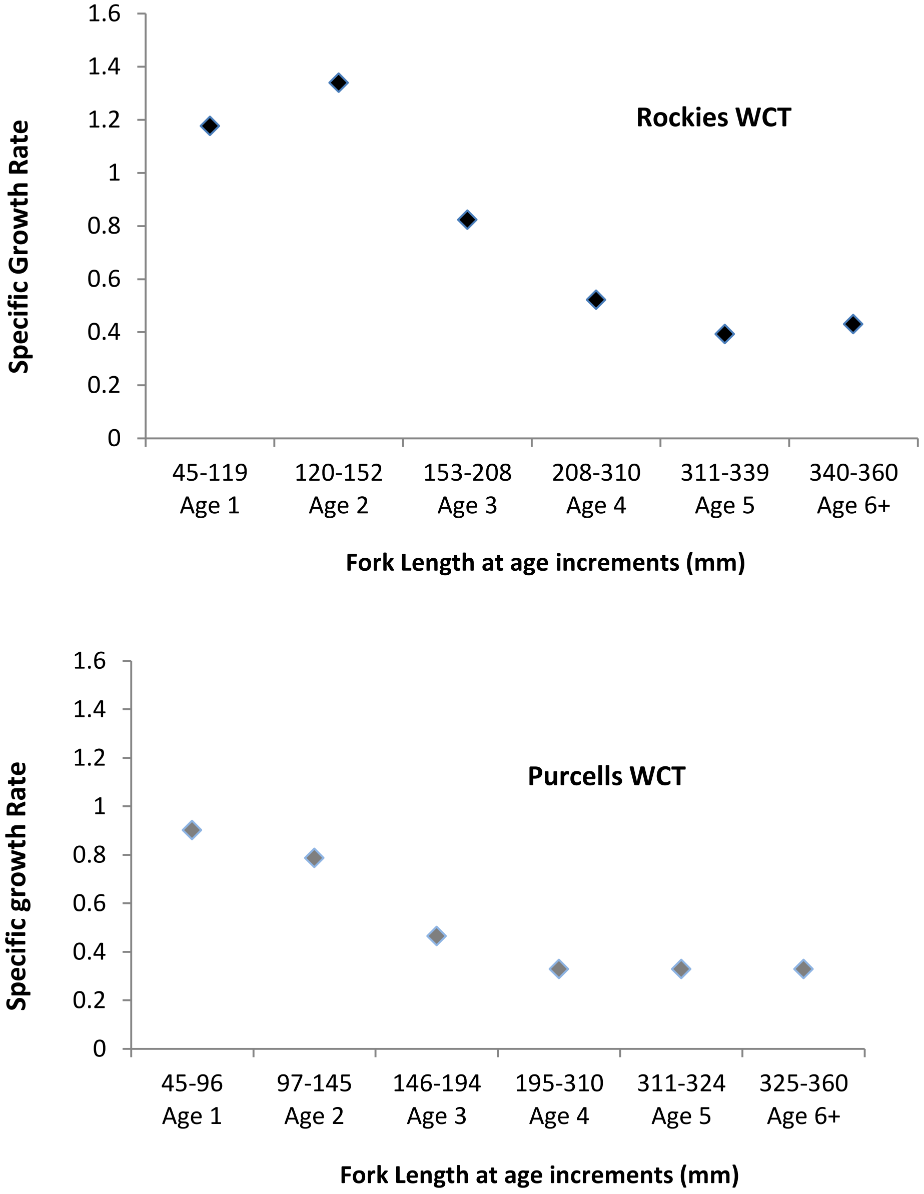

Figure 2-4 Specific growth rate for length adjusted age 1-6 westslope cutthroat trout *(O. clarki lewisi)*, the most abundant species present in the Rockies and Purcells. Each point represents a SGR calculated for fork length increment/age relationships for each region.

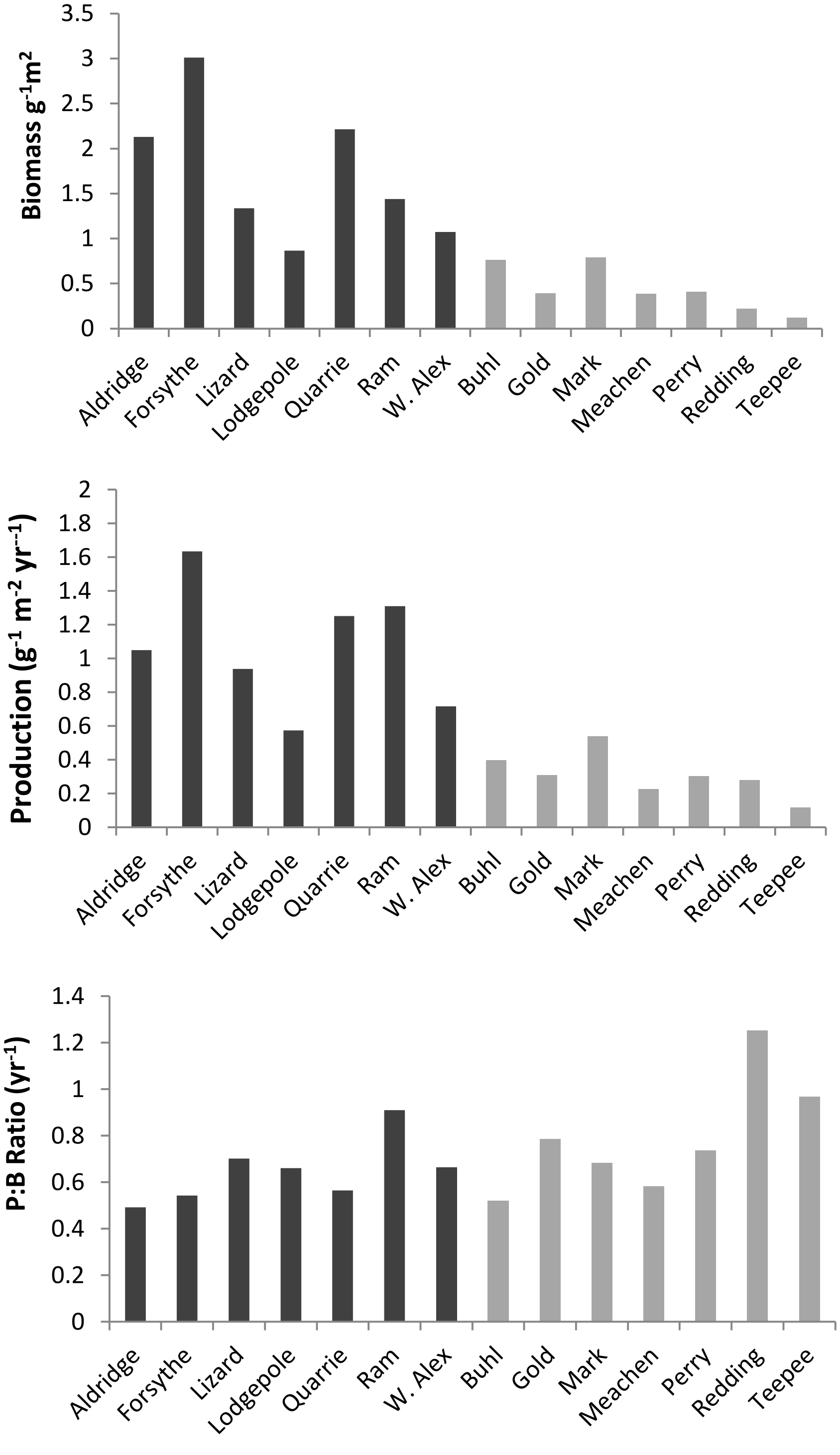

Figure 2-5. Fish biomass, production and P:B ratio for all sites in Rockies (black) and Purcells (grey).

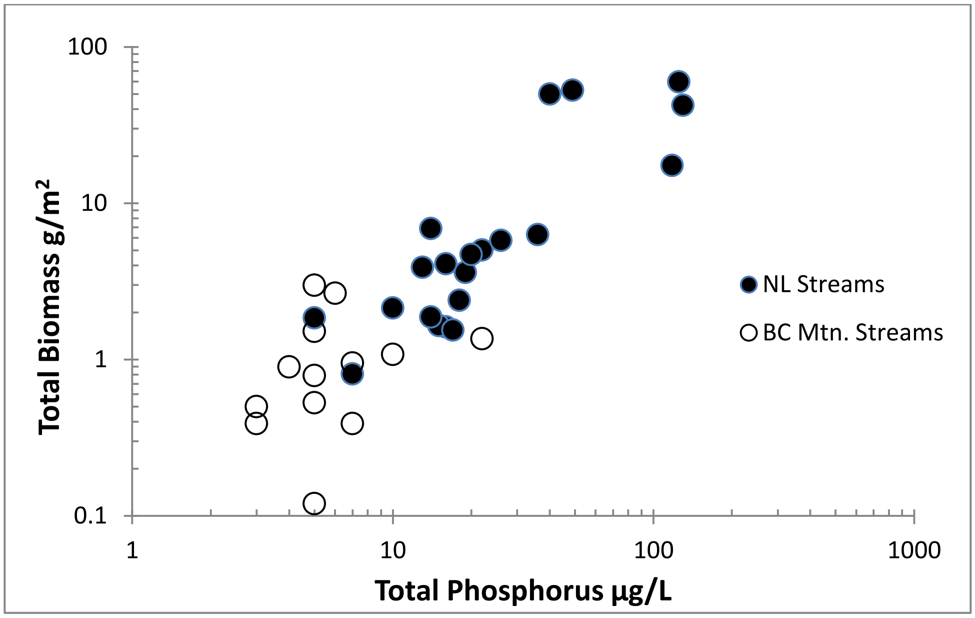

Figure 2-6 TP and total biomass of Newfoundland streams, including highly enriched sites close to anthropogenic influence. Purcell and Rocky Mountain streams fall in line at the lower end of the TP/biomass relationship

3 Characterizing nutrient regimes across Canada: water management implications for fish productivity

3.1 Abstract

Variability in nutrient cycling and accumulation within a watershed is influenced by flow dynamics and catchment characteristics including geology, land use, and climate. Nutrients have been widely established as drivers for primary productivity in aquatic habitats, yet these processes may differ among diverse freshwater ecosystems. In rivers modified by hydropower, impoundments create a depositional zone, disrupting nutrient pulses and transport of materials downstream. Reservoirs can act as a sink or source of nutrients through sediment accumulation or internally loading and releasing nutrients downstream. Evaluating nutrient variation in regulated and reference systems will help to identify how shifts in nutrient regimes influence fish biomass. This study compared Total Phosphorus and Total Nitrogen in regionally diverse reference rivers and rivers modified by hydropower (regulated) in BC, AB, QC, ON, NB and NL in 2011 and 2012. To support this study, local regulated systems in BC and AB were sampled up and downstream of reservoirs to monitor nutrient loading trends on the same river. Nutrient dataset developed for HydroNet river sites using comparable robust sampling parameters. With the expectation that species diversity would influence community biomass production estimates, species specific nutrient-based fish biomass models developed from the literature were used to estimate expected fish biomass in salmonid dominated and multi-species systems across Canada. Expected fish biomass based on TP values provides a useful tool for hydropower management across diverse aquatic waterbodies.

3.2 Introduction

Nutrient regimes have been modified by the development of dams on river ways (Power et al. 1996, Friedl and Wuest 2002). Modification of flow due to hydropower development changes the multi-dimensional nutrient transfer throughout the river, disrupting natural flood pulses, trapping and releasing nutrients at different rates (Power et al. 1996, Freeman et al. 2007). As compared to natural flow conditions, regulated rivers dampen seasonal variability of nutrient distribution (Bunn and Arthington 2002). Impounded systems can have a headpond or storage reservoir, which reserves spring floods for many purposes including irrigation, recreation or hydroelectric power production for late fall and winter energy demand. Alternatively, a run of the river facility may have very limited upstream ponding, and operational requirements may include hydro-peaking flows, with seasonal and even diel water level fluctuation (Renofalt et al. 2010). Hydrologic variability is ecologically important, influencing the temporal and spatial distribution of stream materials and seasonal nutrient spiralling processes (Mulholland et al. 1985). Tributary catchment characteristics directly influence the initial chemistry of the reservoir, however, in regulated systems, impounded and intermittent flows modify nutrient distribution and retention processes, directly influencing bioavailable resources for aquatic community production (Poff and Ward 1989).

3.2.1 Water quality up and downstream of reservoirs

Downstream of the reservoir, regulated receiving streams are highly influenced by the release of reservoir outflow (Sinokrot et al. 1995, Nurnberg 2007) which is affected by the trophic state and morphometrics of the impoundment. Modified temperature regimes in a stratified storage reservoir can cause oxygen depletion in the hypolimnion, and can initiate nutrient loading which can increase nutrients in the receiving stream (Power et al. 1996, Friedl and Wuest 2002, Nurnberg 2007). However, reservoirs where inflows plunge below the photic zone and deposit particulates in the reservoir will retain nutrients and decrease material transport downstream (Stanley and Doyle 2002). These changes can cause ecosystem shifts in the downstream including fluctuations in nutrient concentrations, algal production, invertebrate biomass, and ultimately the productive capacity of fish.

Mass balance (nutrient input/output) models have successfully been used to identify nutrient loading rates in lakes, reservoirs and even rivers (Bosch et al. 2009, Baldwin et al. 2010). However, they can be challenging for they depend on accurate long-term hydrological and chemical modelling (Butturini 2005) and require an understanding of sedimentation processes in the system (Nurnberg 1998). Alternatively, water quality monitored in the regulated river system upstream of the reservoir (inflow) and the outflow into the receiving waters downstream, would be a simpler approach to identifying nutrient regimes and ecological condition when considering multiple systems across a broad geographic range. In addition to this, developing reliable estimates of nutrient regimes in diverse aquatic systems requires strong regional reference sites for making informed regional scale management decisions (Peterson et al. 1999).

3.2.2 Regional characterization

A regional approach to aquatic ecosystem characterisation has been developed in Provincial monitoring programs (Perrin 1998, Brown et al. 2011), as well as terrestrial ecozones across Canada (Ecological Stratification Working Group 1995, Environment Canada 2000). Both terrestrial and aquatic characteristics have been used to define ecoregions, that can be used for establishing reference conditions relative to regionally specific catchment and watershed characteristics (Verdonschot 2000). Although ecoregions have been considered to cover too large of a range to be applicable at a watershed scale (Verdonschot 2000), they are useful for establishing broad scale grouping of homogenous physical and environmental variables for application of management criteria. Four primary terrestrial ecozones represent the HydroNet sites throughout Central and Eastern Canada. Generally, unique geology distinguishes between the Boreal ecozone (granite), Atlantic Maritimes (Appalachian volcanics and granites) and sedimentary systems throughout the Montane Cordillera, and Prairies (Ecological Stratification Working Group 1995) (Figure 3-2). In aquatic systems, geologic features influence natural variability, for watershed lithology affects mineral availability, weathering rates and flow dynamics (Valett et al. 1996, Morrice et al. 1997). Regional characteristics will also be an indicator of the vulnerability and resilience of the aquatic system to ecosystem stress or change (Peterson et al. 1999). An example of this is in the Boreal shield, a region identified by igneous (granite) bedrock geology comprised of 22% of Canada's surface freshwater (Ecological Stratification Working Group 1995). Characterized by low buffering capacity, and therefore an intolerance for acidic inputs, Boreal systems are prone to acidification as compared to sedimentary counterparts (Schindler 2012). Not only is the natural trophic state important for developing regional trigger concentrations of N and P, but also for recognizing the system's response to other contributing factors including modification of flow and anthropogenic point and non-point sources of enrichment. Ecozone classification provides a framework based on geographic and climatic variables to be used for comparability of ecosystem function and change. Characterization of diverse watersheds, although broadly defined within ecozones, can assist in interpreting expected regional water quality and aquatic community including fish.

3.2.3 Fish community and biomass distribution

On a regional scale, native fish species distribution varies according to geographic barriers and postglacial dispersal from glacial refugia (Rempel and Smith 1998, Hewitt 2000, McPhail 2007). From this regional species pool, including invasions and introduced species, fish presence depends on abiotic and biotic factors at a local scale (Jackson et al. 2001). In order to accurately link fish biomass with water quality, the regional expected fish community needs to be accounted for. The trend of decreasing species richness at higher latitudes and altitudes has been widely studied (Bryant et al. 2004, Parra et al. 2009, Griffiths 2010). Freshwater fish diversity in Canada is also quite low in mountainous systems where distribution is restricted by geographic characteristics (McPhail 2007) which is common in salmonid dominated rivers of the east and west coast, northern regions and Montane Cordillera of BC and Alberta. Newfoundland exhibits a mean species richness in its temperate lakes of 3.6 (Scruton et al. 2005), comparable with northern and mountainous regions, the lowest freshwater species richness in Canada (Chu et al. 2003). There is high fish diversity in interior Canada at lower latitudes, where close proximity to species rich refugia influenced the regional species pool (Mandrak and Crossman 1992), and this richness is sustained due to optimal climatic variables (Chu and Jones 2011). Canada is geographically diverse, and species richness reflects the intricacies of its topographic barriers and colonization routes. Therefore, for an estimate of biomass, two communities are considered in this study, either species-poor salmonid systems or multi-species dominated systems.

The premise of applying two community specific fish models to the nutrient data is based on the relationship between expected biomass and species richness. A higher diversity of fish species will result in more specific niche partitioning of resources (Hooper et al. 2005), maximizing energy transfer and ecosystem processes in species rich regions, resulting in higher community production (Carey and Wahl 2011).

3.2.4 Objectives

Our role within NSERC HydroNet is to characterise nutrient regimes, specifically total phosphorus (TP), and the relationship with fish productivity in regulated systems. The main objectives for this study are:

1) Compare nutrient regimes in regulated and reference rivers across regionally diverse sites, developing a multi-year comparable nutrient dataset.
2) Monitor nutrient trends in regulated rivers up and downstream of reservoirs, providing a comparison for nutrient behaviour in reservoirs acting as sources or sinks in steady state conditions.
3) Develop nutrient based fish models to estimate expected fish biomass from measured nutrient values for regulated rivers across a large geographical range, considering the influence of fish community diversity on biomass.

We expect that nutrient regimes will differ due to large-scale regional catchment characteristic and land use. Using regional biomass data, the development of nutrient based fish models will provide a useful predictive tool for hydropower management.

3.3 Materials and Methods

3.3.1 Site Description

A multi-institutional sampling program was developed across Canada for paired regulated (hydropower) and reference rivers in BC, AB, QC, ON, NB and NL in 2011 and 2012, to develop a consistent nutrient database for HydroNet river sites (Figure 3-1). Reference rivers are defined as the regional representative of the regulated systems (Verdonschot 2000), with comparable climate and geomorphology parameters set out during HydroNet development (Smokorowski et al. 2011). Sites ranged from BC to NL in the Montane Cordillera, Prairies, Boreal Shield, and Atlantic Maritime ecozones (Figure 3-2). Rivers up and downstream of 15 reservoirs were also sampled in the Kootenay region of southeastern British Columbia (BC) and east of the Rocky Mountains in southern Alberta (AB); in the Montane Cordillera and Prairie ecozones (Figure 3-5).

3.3.2 Sampling Protocol

In 2011 and 2012, nutrient data was collected for 16 regulated and 16 reference sites, with 175 grab samples collected across six provinces (Figure 3-1). Partner universities provided sampling assistance across Canada, therefore annual site visits were determined by existing field sampling plans.

The local reservoir sampling design focussed on inflow and outflow to investigate trends in water quality within the same system, on run-of-the-river and peaking systems. Nutrient concentrations for systems with multiple inflow points (major tributaries) into the reservoir were proportionally adjusted by average discharge input for sampling month. BC and AB reservoirs sampled were facilities managed for hydropower, recreation, drinking water supply, irrigation and flood control (storage)(Table 3-2, Figure 3-5).Variation between systems is inherent, and therefore, sampling riverine inflow and outflow below the dam on the same river was a simple measure to characterize nutrients during baseline flows from a watershed scale (Bosch 2008). To characterize longitudinal differences in nutrient concentration, we sampled within one control stream, Belly River (upstream of diversion structures).

A water sampling protocol was developed to ensure sampling consistency, cleanliness and simplicity for study participants to follow, including water duplicate samples taken for every 10 field samples. Samples were collected in 500 mL acid washed polyethylene Nalgene bottles supplied to partners across Canada prior to field sampling. All water samples were kept cool at 4^{o}C and shipped to the University of Alberta Biogeochemical lab for TN and TP analysis. TP was determined using persulfate oxidation, and TN was analyzed with the chemical digestion flow injection method (Ng 2010).

Literature meta-analysis provided fish biomass estimates from rivers that spanned a range of nutrient concentrations (oligotrophic to eutrophic), for rivers with salmonid and multi-species fish communities. These estimates, along with findings in Chapter 2 were used to characterise trends in the variability across geographically diverse aquatic systems.

3.3.3 Statistical Analysis

Reference and regulated sites were compared using mean water quality values obtained from sampling in June- August 2011 and 2012. Descriptive statistics were calculated and t-tests were used to compare between reference and regulated sites across Canada. Molar N: P ratios were calculated from mean site values, for Total P and Total N concentrations. Regression plots were used to indicate the most influential component of the N:P ratio among regulated and reference rivers, as a function of TN and TP concentration (Green and Finlay 2010). Paired t-tests were applied to compare the mean difference between upstream and downstream TP and TN concentrations for local reservoir sites in BC and AB. Proportional difference between up and downstream TP concentrations was calculated to indicate loading behaviour, whether or not the reservoir was trapping or releasing P (acting as a source or sink) to the receiving stream. A weighted 20% difference between up and downstream values was used as the cut-off point for depositional or internal loading dynamics.

Regression models used for fish community comparisons were constructed from fish biomass and TP data in published scientific journals, government publications and consulting reports. Data was sorted for regional comparison with two fish community identifiers, either salmonid dominated or multi-species systems. Salmonid data was sourced primarily from insular Newfoundland, whereas multispecies data was combined for species rich systems from Ontario, Quebec and northern U.S. states (Rasmussen et al. in Prep).

3.4 Results

3.4.1 HydroNet regulated and reference rivers

No significant difference was found between nutrient regimes in regulated and reference rivers, using two sample t-tests for TP ($t_{(30)}$, t=0.55, p=0.58) and TN ($t_{(30)}$, t=0.29, p=0.77). Data was also compared across years (2011 and 2012) and no significant difference was found. With a combination of all TP HydroNet data from 32 sites, over two years, the range of mean values were consistently low (2-12 $\mu g\ L^{-1}$) predominantly within the oligotrophic trigger range developed by the CCME guidelines (Environment Canada 2004)(Figure 3-4). One outlier mean value (over two years) of 22 $\mu g\ L^{-1}$ was found for St. Mary River in southern Alberta that may be the direct influence of sampling coinciding with seasonal upstream agricultural land use. N:P ratios are used to identify nutrient deficiency in aquatic systems, as an indication of limitation for algal growth. N:P ratios calculated for regulated and reference streams ranged from 12-200, with only three of the 33 sites (9%) falling below the Redfield ratio of 16 (Redfield 1958)(Figure 3-3), indicating phosphorus limitation. Although TP values fell within a small oligotrophic range, TN values were highly variable across regions with a greater range across systems from 64-1303 $\mu g\ L^{-1}$. Variation in N:P ratio for HydroNet regulated and reference rivers sampled during the summer months was influenced most by the variability of TN (R^2=0.91), rather than TP.

3.4.2 Up and downstream BC and Alberta reservoirs

Using paired t- tests, no significant difference was found between up and downstream sites for select reservoirs in BC and Alberta for TP ($t_{(15)}$, t=1.74, p= 0.10) or TN ($t_{(15)}$, t=0.82, p= 0.43). Percent difference of TP values between inflow and outflow was calculated for all 15 regulated systems, as well as the control system (Belly R.). When looking at proportional differences between up and downstream sites three distinct nutrient retention processes were observed: trapping, releasing or no change. Proportional difference between upstream TP and downstream TP varied between sites, resulting in some reservoirs acting as a sink and others as a source of TP to the receiving stream. Proportional difference between inflow and outflow (Figure 3-6) indicates that the majority of sites varied less than 20% between the inflow and outflow TP concentrations. Of the reservoirs that fell outside of this range, the greatest trappers in 2012 included Spray and Glenmore. Whereas, reservoirs that were releasing TP downstream either due to internal loading or organic nutrient flux included Twin Valley and Bearspaw.

3.4.3 Expected fish biomass models

Nutrient based fish regression models (Rasmussen et al. in prep) were applied using 2011 and 2012 nutrient data, as an example of how similarities in nutrient regimes would reflect different fish biomass results (Figure 3-7). Based on regional species composition, HydroNet sites in Montane Cordillera, New Brunswick and Newfoundland would reflect the salmonid dominated model whereas the multi-species model would be applied to species rich rivers in interior Quebec and Ontario, with biomasses an order of magnitude higher.

3.5 Discussion

3.5.1 Regulated and reference HydroNet rivers

Across a broad geographical range, TP and TN concentrations did not significantly differ between regulated and reference HydroNet Rivers across two years. Modification of flow dynamics influences longitudinal (Bunn and Arthington 2002) and lateral (Ward and Stanford 1995) transport of nutrients, therefore, the expectation was for highly variable nutrient regimes between reference and regulated rivers across a broad geographic range. Based on regional characterization approach, I also expected systems to exhibit similarities due to comparable geologic and climactic habitats found across Canada. A large amount of Hydropower development occurs in upper elevation Montane rivers (Finger et al. 2007), and in the Boreal region (Schindler 1998), which coincided with over half of the river sites characterised for HydroNet across Canada which were within these regions (Table 3-1). Therefore, low P concentrations would be expected due to the nutrient limiting nature of Boreal and Montane rivers (Environment Canada 2000, Ensign and Doyle 2006). In addition to these nutrient poor regions, the results of two years of sampling identified relatively low P values for rivers in interior Quebec, Ontario, and the Prairie region, not corresponding with regional nutrient characteristics. These sedimentary-based rivers would be expected to exhibit a higher trophic state for they occur lower in the watershed with closer proximity to a higher anthropogenic density. Consistently low ranges of TP concentrations were measured in all HydroNet rivers across Canada from Montane British Columbia to Newfoundland. The range of values fell below 25 µg L^{-1}, the upper oligotrophic range defined specifically for rivers (Dodds et al. 1998).

Greater site-specific differences were measured in TN between systems across Canada. Land-use was considered when pairing regulated and reference sites but this high range of variability of N is likely linked to non-point source anthropogenic inputs. Aquatic systems' natural nitrogen concentrations (NO_3^-) range from subarctic systems at <25 µg L^{-1} to Amazon rivers with 200 µg L^{-1} (Wetzel 2001), whereas nitrogen range for our regulated and reference rivers in some cases exceeds these extremes six fold, an indication of anthropogenic inputs. Total nitrogen fluctuations are most influenced by trends in inorganic nitrogen (NO_3^-) (Wetzel 2001). High fluxes in nitrogen can be a direct response of anthropogenic inputs from agriculture fertilizers (Schindler et al. 2006) or mining processes (explosives)(Nagpal 1982), both of which are highly soluble in aquatic systems as NO_3^-. HydroNet rivers with the highest N outlier levels included the Elk River, and Etchemin River, with potential non-point sources from mining and agriculture, respectively. Saturation of N in rivers can elevate nutrient imbalances (Vitousek et al. 1997), for high concentrations of TN would instigate hyper-algal production, requiring all available P in the system. Therefore, higher levels of TN could influence these rivers to be persistently P limited.

N:P ratios are used to determine overall limiting nutrients in aquatic ecosystems. Accepted in freshwater science is that a N:P ratio greater than 20 would be P limited and less than 10 would be N limited (Schanz and Juon 1983, Ashley and Stockner 2003). Using this range, N:P ratios calculated from two years of data indicated that both reference and regulated HydroNet Rivers across Canada were P limited.

3.5.2 Upstream and downstream reservoir water quality

The process of reservoir trapping and nutrient deprivation of downstream systems has been well established (Power et al. 1996, Friedl and Wuest 2002, Cavaliere and Homann 2012), however, impoundments can also enrich downstream waters through internal loading and hypolimnetic outflow release of nutrients (Camargo et al. 2005, Nurnberg 2007). A shift from turbulent riverine habitats to slow back-watered velocities, creates depositional environments act as nutrient sinks or sources to downstream receiving waters (Friedl and Wuest 2002, Nurnberg 2009). Nutrient trapping and releasing trends are generally identified using mass balance modelling to predict net fluxes in external nutrient loading, identifying seasonal and annual shifts (Butturini 2005, Bosch and Allan 2008). Yet the basis of loading trends is not easily identified, having been influenced by many contributing factors including residence time (De Oliveira Naliato et al. 2009), age (Grimard and Jones 1982), presence of littoral zones (Matzinger et al. 2007) and morphometrics of the reservoir (Northcote and Larkin 1963). Therefore, for a broad approach, determining baseline trophic state of a system can roughly define the potential productivity of the ecosystem. Monitoring of systems during one season can limit the understanding of annual nutrient flux but precision of nutrient load estimates is exchanged for generality, by increasing the geographic range and potential variation and distribution of the data (Cote 2007). Therefore, baseline monitoring of nutrient concentrations during steady state conditions (July, August) indicates a rough estimate of nutrient limitation during the growing season.

Nutrient concentrations and storage depend on different catchment characteristics, climate, and flow pathways (Newbold et al. 1981, Lefebvre et al. 2005).

Reservoir loading trends can also vary from year to year depending on climate and hydrologic patterns (Friedl and Wuest 2002). An overview of local reservoirs' retentive processes, measured as inflow and outflow along the river continuum, indicated variable trends in trapping and releasing. Interesting, most reservoirs showed little to no difference between up and downstream site, and nutrient levels were consistently low. The small variation between up and downstream sites on rivers across a diverse range could be a consequence of water sampling in the summer when natural inflows were minimal, and therefore tributary nutrient fluxes would be limited. Only a few sites indicated signs of trapping, where TP is settling out or releasing, potentially due to site-specific factors (Baldwin et al. 2010). Twin Valley is a Prairie irrigation reservoir, which does not stratify due to wind exposure. Higher outflow levels at Twin Valley could reflect the well mixed optimal summer growing conditions of the reservoir, with outflow TP concentrations dominated by organic P in the form of cellular biomass accrued from summer algae production. Spray reservoir, a high altitude mountain reservoir in the Rocky Mountains, had significantly higher TP values upstream of the reservoir in Smith Dorien River. This is likely attributed to particulates from glacial meltwaters (rock flour), where turbid tributaries elevated upstream total values from particulate inflow (Camargo et al. 2005). In BC and Alberta, the reservoirs that showed the greatest change between up and downstream nutrient regimes were associated with reservoirs on rivers with multiple dams. This was not an exclusive trend, but greater accumulated residence time would influence nutrient transport processes. Interestingly, most sites exhibited very little change between inflow and outflow

Influence of hydropower

Some watersheds are chronically fragmented by multiple dams between the headwaters and marine environment, compounding the influence of regulation on nutrient cycling. A study in historic salmonid populations in the Columbia River system found that carrying capacity of rivers was limited by the decline of marine derived nutrients and not habitat (Achord et al. 2003). Systems impounded by hydropower dams shift nutrient regimes by trapping sediments in reservoirs, and fragmenting upstream tributaries from marine derived and other migratory fish movements (Wipfli and Baxter 2010). The Columbia River system has 14 mainstem dams and over 130 impoundments on its tributaries (Ferguson et al. 2011). On the Columbia River upstream of the Canada/US border, the Arrow Lakes (impounded lakes upstream of Keenleyside dam) are nutrient deprived from sediment trapping of upstream dams (Pieters et al. 2003). In mountain fed systems that are naturally low in nutrients and depend on nutrient subsidies for increased production, impoundments will further exacerbate nutrient limitation. Sedimentation processes can compound nutrient deprivation consistently with multiple dams downstream from one another, resulting in systems that were naturally nutrient poor now exhibiting ultra-oligotrophic conditions (Friedl and Wuest 2002). The oligotrophic trophic state will become more extreme, and this process will be elevated in rivers with multiple impoundments. The current trophic state of a system influences nutrient behaviour with a positive feedback loop increasing the disparity between trophic states. Although cycling rates do not vary with shifting concentration (Hudson et al. 2000), solubility of P will increase in eutrophic systems (Hudson et al. 2000, Dodds 2003). Therefore, there is a trend for solubility of P to increase in systems with high P concentrations and decrease in systems with low P concentrations. Although nutrient retention may assist with management of eutrophic reservoirs, decreased nutrients in oligotrophic reservoirs would limit ecosystem functioning downstream (Friedl and Wuest 2002). As described in Chapter 2, even on the lowest end of nutrient concentration spectrum, small shifts in available nutrients can influence fish production.

Application of fish models

For a cross Canada comparison, regions were identified and paired with reference systems based on broad scale physical and chemical parameters. Although an appreciation for the unique characteristics of each site is important when trying to understand ecosystem function at a local scale, this can be challenging for implementing regional management objectives. Thus, when environmental conditions are approached at a lower resolution, large-scale patterns can be observed.

The structure of the fish community is responsible for energy flow and function within the ecosystem (Arend and Bain 2008). Therefore, regional species diversity and potential biomass need to be considered. Distribution of fish species is assessed at varying scales, and how large a role certain local characteristics play is determined by the study question. For a regional approach to identifying fish communities, distribution depends on historic zoogeographic colonization and landscape constraints developing a regional fish pool. Whereas, on a local scale, abiotic and biotic factors play a role in determining fish presence from within the regional pool (Jackson et al. 2001). Fish biomass differed between regions on a local scale in the study in Chapter 2, with variation in nutrient concentrations influencing productivity. Nutrient based fish models applied to the dataset for across Canada indicated an order of magnitude difference in biomass between salmonid and multi-species systems with the same nutrient regimes. This relationship between TP and productive capacity for fish provides an estimate of expected fish productivity relative to measured nutrient values.

Conclusions

The assessment of nutrient regimes at an ecologically relevant scale is challenging. Streams generally have higher nutrient concentrations than lakes (Randall 1995), therefore you would assume that reservoirs would fall within the continuum between the two, with contrasting high flushing rates and lentic depositional zones. When comparing regulated and free flowing rivers across HydroNet sites of Canada, paired sites may exhibit comparable trophic state due to regional similarities of concentrated hydropower development. A large percentage of the large hydropower facilities in Canada occur in the northern, boreal region (Schindler 1998), although not void of anthropogenic influences, have limited intensive agriculture due to bedrock dependant coniferous forests (Environment Canada 2000). Nutrient regime of regulated systems will therefore be influenced by regional catchment characteristics and land use, where the current trophic state will indicate its retentive and cycling processes. Natural availability of N and P has been altered by anthropogenic activities (Meybeck and Helmer 1989, Moss 2010) and each nutrient becomes limiting in scenarios where the other is found in excess (Wetzel 2001). Although both N and P are essential to aquatic systems, P has been more closely linked with fish production or biomass as a measure of water fertility in diverse aquatic systems (Hanson and Leggett 1982, Hoyer and Canfield 1991). Therefore, P limitation would directly reflect systems potential productivity as compared to Nitrogen. Understanding the baseline TP concentration of the system and the expected fish biomass based on fish community will provide a useful predictive tool for hydropower management.

Table 3-1 Regulated and Reference site location across Canada

Province	River	Latitude	Longitude	**R**egulated/ **N**on**R**egulated	Ecoprovince
NFL	West Salmon R.[1]	48.1728	-56.2309	**R**	Boreal Shield
Ontario	Magpie R.[2]	48.0217	-84.8122	**R**	Boreal Shield
Ontario	Mississagi R.[3]	46.8747	-83.3193	**R**	Boreal Shield
Quebec	Saint Jean R.[4]	48.1978	-70.2533	**R**	Boreal Shield
Quebec	Etchemin R.[5]	46.6674	-71.0856	**R**	Atlantic Maritime
Quebec	Fourchue R.[6]	47.6506	-69.5123	**R**	Atlantic Maritime
NB	Dee R.[7]	-67.0062	47.1218	**R**	Atlantic Maritime
NB	Serpentine R.[7]	-66.9924	47.1960	**R**	Atlantic Maritime
BC/Montana	Kootenay (below Libby)[8]	48.3669	-115.3198	**R**	Montane Cordillera
BC	Elk R. (below Elko)[9]	49.2797	-115.0998	**R**	Montane Cordillera
BC	Bull R. (below Aberfeldie)[10]	49.4927	-115.3618	**R**	Montane Cordillera
Alberta	Kananaskis R. (below Pocaterra)[11]	50.7023	-115.1183	**R**	Montane Cordillera
Alberta	Kananaskis R. (below Barrier)[14]	51.0442	-115.0298	**R**	Montane Cordillera
Alberta	Elbow R.(below Glenmore)[13]	51.0089	-114.0862	**R**	Prairies
Alberta	Old Man R.[12]	49.5591	-113.8785	**R**	Prairies
Alberta	Waterton R.[15]	49.3331	-113.6664	**R**	Prairies
Alberta	St. Mary's R.[16]	49.3613	-113.0624	**R**	Prairies
NFL	Twillick Br.[1]	48.1155	-55.5766	NR	Boreal Shield
Ontario	Batchawena R.[2]	46.9980	-84.5230	NR	Boreal Shield
Ontario	Aubinadong R.[3]	46.8944	-83.4129	NR	Boreal Shield
Quebec	Petite Saguenay R.[4]	48.1977	-70.0579	NR	Boreal Shield

Table 3.1 Regulated and Reference sites across Canada continued.

Province	Regulated River	Latitude	Longitude	**R**egulated/ **N**on**R**egulated	Ecoprovince
Quebec	Aux Saumons R.[5]	45.5838	-71.3877	NR	Atlantic Maritime
Quebec	Becancour R.[5]	46.3168	-71.4463	NR	Atlantic Maritime
Quebec	Beaurivage R.[5]	46.6891	-71.2677	NR	Atlantic Maritime
Quebec	Du Loop R.[6]	47.5828	-69.6688	NR	Atlantic Maritime
NB	Gulquac R.[7]	-67.2676	46.9860	NR	Atlantic Maritime
BC	Bull R.[9,a]	49.5331	-115.3274	NR	Montane Cordillera
BC	Elk R.[10,a]	49.3070	-115.0828	NR	Montane Cordillera
BC	Kootenay R.[11,13,a]	49.4346	-115.4233	NR	Montane Cordillera
Alberta	Waterton R.[12]	49.2591	-113.6889	NR	Prairies
Alberta	Elbow R.[8,14,a]	50.9929	-114.1481	NR	Prairies
Alberta	Castle R.[15]	49.5001	-114.1999	NR	Prairies
Alberta	Belly R.[16]	49.2610	-113.5737	NR	Prairies

Super script number corresponds to reference and regulated pairings

NR sites will be either an unregulated site or upstream of dam by 10km

[a] Duplicate river names refer to rivers that were sampled upstream of an impoundment

Table 3-2 Local reservoirs in BC and Alberta, nutrient concentrations sampled at riverine inflow and downstream outflow

Dam	Reservoir	Year built	Purpose	Type
Glenmore	Glenmore	1932	WS	Run-of-river
Carsland	Carsland	1973	I/R	Weir
Bearspaw	Bearspaw	1954	H/C	Peaking
Ghost	Ghost	1929	H/R	Run-of-river
Spray	Spray Lakes	1951	H/R	Peaking
Pocaterra	Lower Kananaskis	1955	H/R	Peaking
St. Mary	St.Mary	1951	I/H	Run-of-river
Twin Valley	Twin Valley	2004	I/R/C	Irrigation storage
Keho	Keho	1923	I/R	Irrigation storage
Waterton	Waterton	1963	I/H/R	Run-of-river
Barrier	Barrier	1947	H/R	Peaking
Oldman	Oldman	1990	I/H/R/C	Run-of-river
Aberfeldie	Aberfeldie	1922	H	Run-of-river
Elko	Elko	1924	H	Run-of-river
Libby	Koocanusa	1973	H/R/C	Run-of-river
Belly (control)	Belly (control)	n/a	Weir d/s	Control section

I=irrigation, H=hydropower, R=recreation, C= flood control, WS=water supply

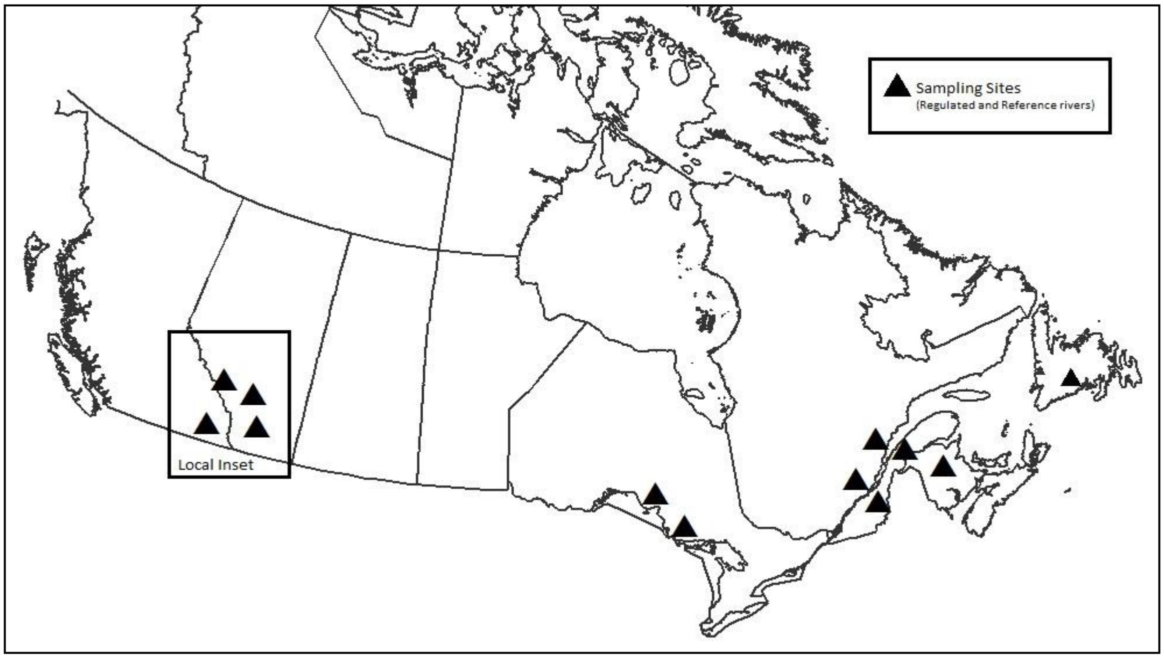

Figure 3-1 Map of regional distribution of Canadian sampling, triangle refers to location of multiple reference and regulated sites in each region, n=33.

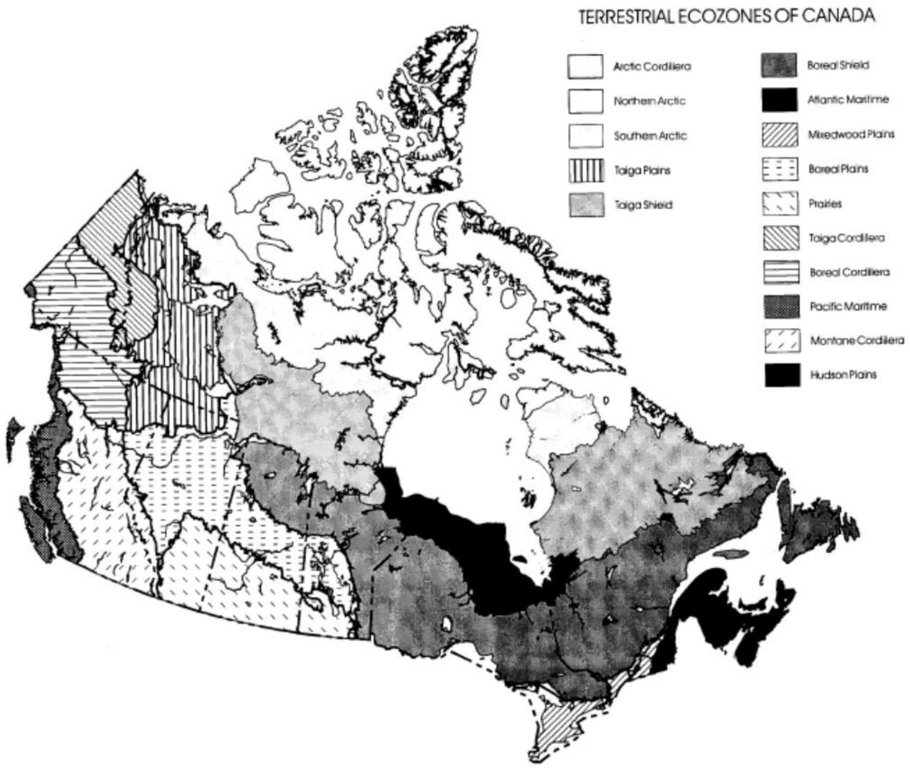

Figure 3-2 Adapted from Terrestrial Ecozones of Canada, (Ecological Stratification Working Group 1995).

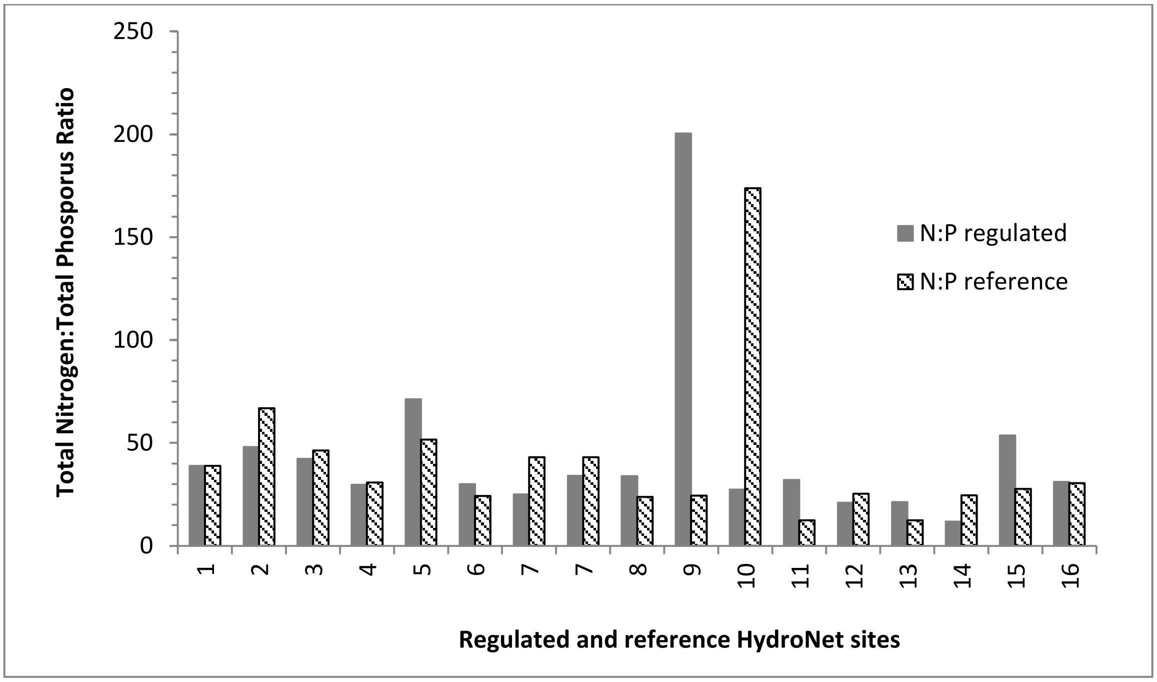

Figure 3-3 N:P ratios for regulated and reference HydroNet rivers, site numbers corresponding to paired sites in Table 3-1.

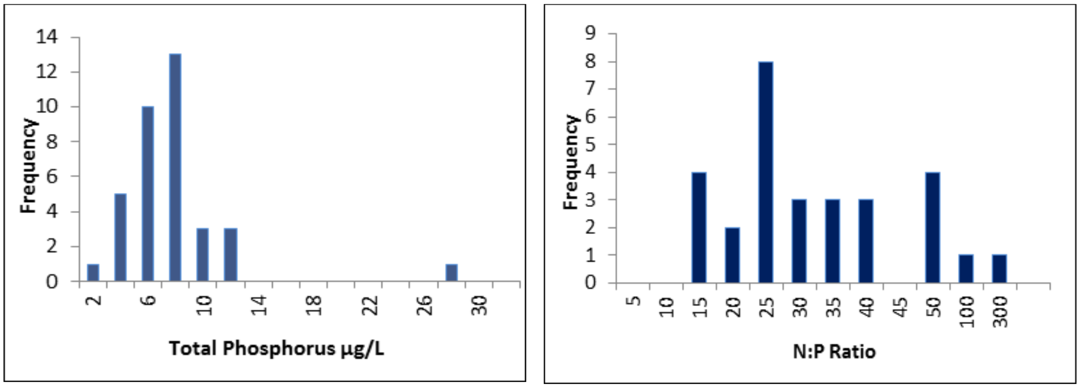

Figure 3-4 TP distribution for HydroNet rivers across Canada (left) and N:P ratios (right) indicating no sites were nitrogen limited (less than 10), and most were phosphorus limited (greater than 20).

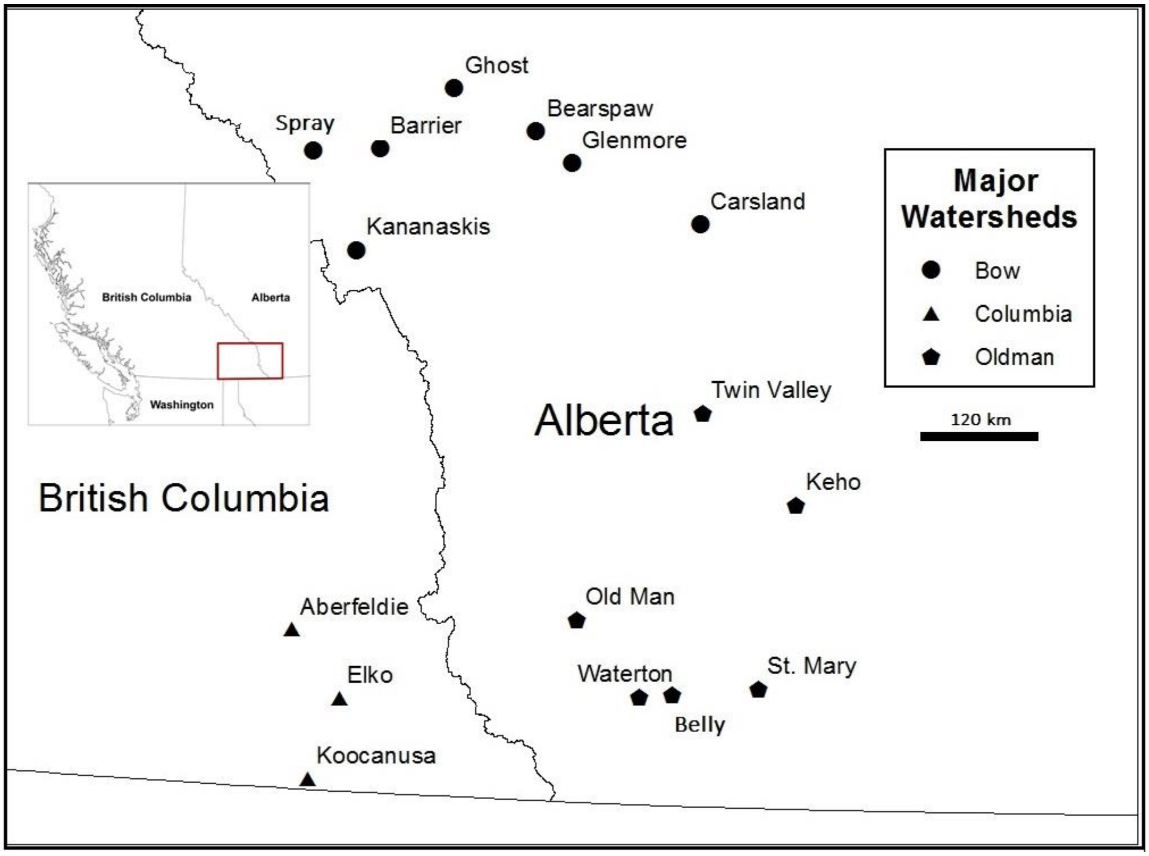

Figure 3-5 Local reservoirs in BC and AB sampled in riverine habitat upstream and downstream of impoundment.

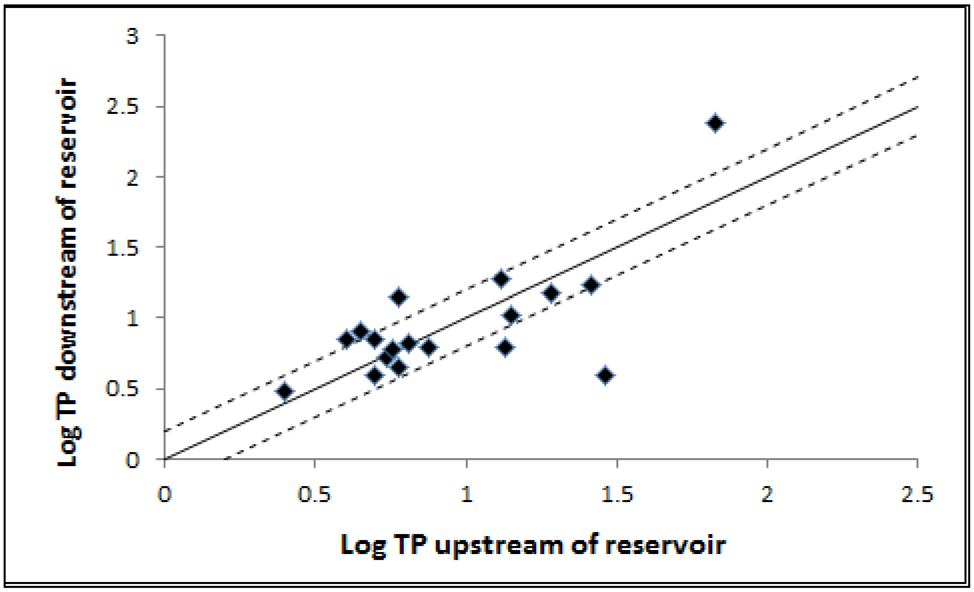

Figure 3-6 TP sampling up and downstream of reservoirs in BC and Alberta. Solid line indicates no change in TP up and downstream of reservoirs. Dashed lines indicate a 20% difference between up and downstream values, where the reservoir was acting as a sink (trapping) or source (releasing) nutrients downstream.

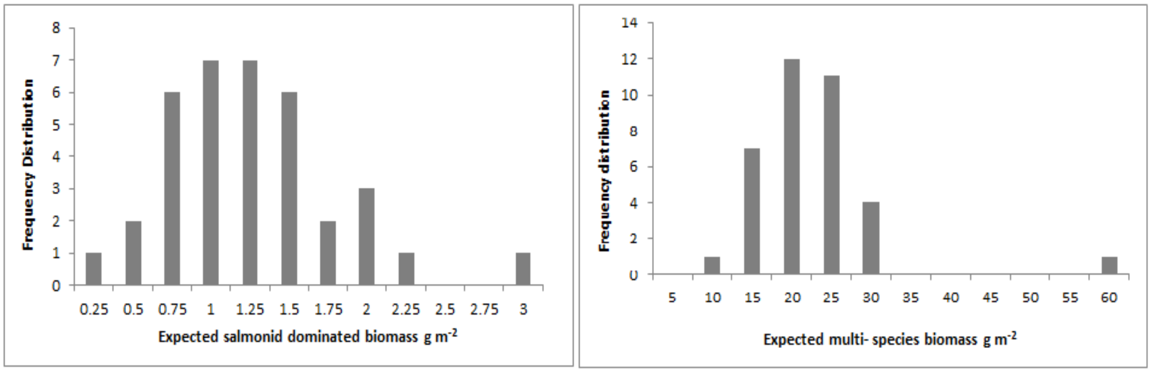

Figure 3-7 Nutrient based fish models, with the HydroNet measured nutrient values applied for systems across Canada. An order of magnitude difference between salmonid and multi-species systems biomass confirms the importance of regionally specific models for management application. Note that x-axis scale (biomass) is different.

4 Conclusions and Further Research

4.1 Nutrient and fish relationships

Rivers, lakes and reservoirs’ nutrient concentrations are altered by upstream catchment characteristics, therefore, a clear understanding of natural regional variability is required to develop target water quality ranges. In hydrologically altered watersheds, the current state of water quality may be different from expected regional values. Thus, an understanding of how flow modification such as hydropower development influences nutrient concentrations and resultant fish communities is required to effectively manage the aquatic health of modified watersheds.

Hydropower facilities affect fish communities in many ways, by shifting available habitats, fragmenting fish movement patterns, and shifting available food resources. Dam and reservoir development and operation threatens aquatic health by disrupting nutrient transport downstream, and fragmenting upstream reaches from seasonal nutrient subsidies (including migratory fish). For mitigation and compensation of environmental impact resulting from the development and operation of dams, hydropower companies require an accurate aquatic baseline assessment to use as a benchmark of expected values for the application of accurate environmental management criteria. For effective assessment of fisheries in areas modified by hydropower, predictive tools can aid aquatic management decisions by providing comparative assessment from straightforward field measurements. Predicting expected fish biomass from nutrient concentrations requires:

1) A baseline Total Phosphorus concentration measure
2) An understanding of the regional fish species pool, for there is a trend of increased total fish biomass with greater fish species diversity.

Total phosphorus (TP) is the limiting nutrient in a range of aquatic systems, supporting its role as a driver of aquatic productivity at lower trophic levels, and is a simple water quality metric. For applicable fish production assessment, nutrient data should be collected during base flows, when primary productivity and fish growth are at their peak. The accumulation of stressors on the environment has shifted the natural nutrient regimes in most aquatic systems; therefore, regionally scaled nutrient-based fish models provide a management tool applicable to both natural aquatic systems and those highly modified by anthropogenic pressure (hydropower).

Hydropower development and operation disrupts natural aquatic processes, and therefore, through environmental monitoring and assessment, appropriate mitigation or compensation efforts can be applied (Trussart et al. 2002). Mitigation and compensation efforts are used to achieve regional expectations. In order to properly mitigate or minimize the effect of the development or operation of the facility, a benchmark of regional productivity is required, especially if no pre-dam environmental data is available. Nutrient-based fish models provide a framework that can be continuously strengthened, and used to predict expected fish biomass values from baseflow nutrient concentrations. An overall approach to mitigating the effects of hydropower facilities on water quality is challenging, for each site location will resemble the unique regional characteristics, and be influenced by facility construction and operation (Kennedy 2001, Trussart et al. 2002). Therefore, watersheds influenced by hydropower must be assessed to apply suitable mitigation and compensation efforts. Examples of mitigation and compensation efforts that are specific to promoting optimal water quality and fish community targets include: controlling potential upstream point and non-point sources of nutrient loading, modifying reservoir outflow location (surface or hypolimnetic withdrawal) to meet dissolved oxygen and temperature requirements for aquatic species present, meeting instream-flow requirements for all life stages of fish species present, and improving fish passage facilities to promote transfer of nutrient subsidies to upstream reaches. The resources required for long-term environmental monitoring and assessment should be allocated during the planning and development stage of hydropower projects to ensure the success and longevity of environmental mitigation and compensation. In order to apply environmental mitigation and compensation to hydrologically altered watersheds, regional ecological benchmarks need to be defined. Predictive tools based on nutrient and fish biomass relationships can be continuously strengthened with regional reference data, providing an environmental goal or expectation that hydropower managers can use to aid in management, mitigation and compensation decisions.

4.2 Research collaboration and application

Research collaboration and data sharing are instrumental in addressing large-scale research questions. As the scale of stressors on the environment increase, aquatic assessment also has to be adjusted to incorporate larger scale influences to the ecosystem (Verdonschot 2000). There is a trade off when investigating ecosystem function relationships; they need to be measured at an ecologically relevant scale, and in a way that systems can be effectively monitored and managed. Local studies provide insight into community interactions and ecosystem response at a reach scale, however, the high resolution of these findings can be challenging to apply within management criteria. The local and regional studies of Chapter 2 and 3 complement one another as they provide a connection between testable local-scale trends and larger-scale management challenges.

Data sharing and development of comparable datasets developed from robust, repeatable sampling programs is crucial to strengthening current models. The benefit of a multidisciplinary research group that includes industry, academic and government agencies is the development of applicable studies, where research findings are translated smoothly through to management and policy applications.

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Appendix A

Table A1: Chapter 2 fish sampling raw data 2011

Region: Rockies
Site: Quarrie Creek (LAT LONG)
Closed site area: 1032 m^2
Date: August 5, 2011
Temperature: 11 ^{o}C
Conductivity: 167 µS
EF seconds (first, second haul): 658, 612

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	MW	270	254	2	BT	150	35
1	MW	220	144.5	2	MW	245	143
1	MW	230	151	2	BT	147	35.7
1	MW	272	265	2	WCT	170	68.2
1	EB	140	36.5	2	WCT	235	180
1	BT	140	30.2	2	BT	134	31
1	WCT	170	66	2	WCT	164	62.1
1	MW	275	274				
1	MW	330	425				
1	MW	220	147.5				

Region: Rockies
Site: West Alexander Creek (LAT LONG)
Closed site area: 692 m^2
Date: August 4, 2011
Temperature: 5.7 ^{o}C
Conductivity: 167 µS
EF seconds (first, second haul): 953, 881

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	WCT	274	240.5	2	WCT	208	122.5
1	WCT	245	185.4	2	BT	158	40
1	WCT	248	161.5				
1	BT	160	43				

Region: Rockies

Site: Lizard Creek (LAT LONG)

Closed site area: 837 m^2

Date: August 6, 2011

Temperature: 13 ^oC

Conductivity: 226 µS

EF seconds (first, second haul): 729,750

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	WCT	142	31	2	WCT	130	32
1	WCT	120	21	2	WCT	60	3
1	WCT	180	86	2	WCT	60	4
1	WCT	185	95	2	WCT	60	4
1	WCT	124	27	2	WCT	165	64
1	WCT	74	4	2	WCT	170	69
1	EB	170	56	2	WCT	60	4
1	WCT	110	13	2	WCT	60	4
1	BT	130	19	2	WCT	60	4
1	WCT	60	20	2	WCT	60	4
1	WCT	360	572				

Region: Rockies

Site: Aldridge Creek (LAT LONG)

Closed site area: 764 m^2

Date: August 8, 2011

Temperature: 10 ^oC

Conductivity: 176 µS

EF seconds (first, second haul): 943,882

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	WCT	310	406	2	WCT	310	305
1	EB	100	8	2	WCT	155	57
1	BT	87	7	2	BT	155	40
1	MW	264	185				
1	WCT	360	645				

Region: Rockies

Site: Ram Creek (LAT LONG)

Closed site area: 752 m^2

Date: August 14, 2011

Temperature: 10 °C

Conductivity: 273 µS

EF seconds (first, second haul): 1169, 952

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	BT	254	198	2	BT	202	109
1	BT	214	105	2	BT	203	113
1	BT	195	87	2	BT	183	77
1	BT	173	69				
1	BT	81	7				
1	BT	116	23				
1	BT	134	29				
1	BT	126	90				
1	BT	155	45				
1	BT	110	17				
1	BT	138	31				
1	BT	140	33				
1	BT	202	109				

Region: Rockies

Site: Lodgepole Creek (LAT LONG)

Closed site area: 658 m^2

Date: August 18, 2011

Temperature: 7.5 °C

Conductivity: 237 µS

EF seconds (first, second haul): 1000, 892

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	WCT	163	48	2	WCT	215	124
1	WCT	149	42	2	WCT	137	28
1	WCT	163	66	2	WCT	155	54
1	WCT	232	154				
1	WCT	137	54				

Region: Rockies

Site: Forsythe Creek (LAT LONG)

Closed site area: 856 m^2

Date: August 19, 2011

Temperature: 7^{o}C

Conductivity: 220 µS

EF seconds (first, second haul): 982, 752

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	MW	235	154	2	WCT	368	650
1	EB	200	104	2	MW	308	300
1	BT	105	16	2	MW	244	165
1	EB	150	47	2	WCT	120	23
1	WCT	123	24	2	BT	92	9
1	BT	94	11	2	WCT	117	117
1	WCT	300	389	2	BT	130	130
1	BT	102	11				
1	WCT	336	503				
1	WCT	120	23				

Region: Purcells

Site: Gold Creek (LAT LONG)

Closed site area: 446 m^2

Date: August 9, 2011

Temperature: 8 ^{o}C

Conductivity: 148 µS

EF seconds (first, second haul): 860, 881

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	WCT	120	17	2	WCT	129	21
1	WCT	125	24	2	WCT	130	19
1	WCT	135	25	2	WCT	127	20
1	WCT	100	8	2	CCG	88	6
1	WCT	80	7				
1	WCT	84	3				
1	WCT	84	3				
1	CCG	47	1				
1	CCG	47	1				
1	WCT	137	20				

Region: Purcells

Site: Teepee Creek (LAT LONG)

Closed site area: 1020 m^2

Date: August 11, 2011

Temperature: 10.5 ^oC

Conductivity: 221 µS

EF seconds (first, second haul): 1358, 1222

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	CCG	75	8	2	CCG	41	1
1	CCG	72	6	2	CCG	82	10
1	CCG	75	7	2	LNC	104	18
1	WCT	103	12	2	LNC	104	11
1	WCT	84	9	2	WCT	87	8
1	LNC	92	7	2	CCG	41	1
1	CCG	41	1	2	CCG	41	1
1	CCG	41	1	2	LNC	104	11
1	CCG	82	10				

Region: Purcells

Site: Perry Creek (LAT LONG)

Closed site area: 1144 m^2

Date: August 10, 2011

Temperature: 9 ^oC

Conductivity: 42 µS

EF seconds (first, second haul): 1600, 1334

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	RB	200	80	2	EB	130	16
1	RB	150	37	2	EB	140	20
1	EB	140	21	2	RB	145	36
1	EB	138	23				
1	EB	140	20				
1	EB	140	20				
1	RB	170	48				
1	EB	150	27				
1	EB	120	16				
1	EB	92	7				
1	EB	92	7				
1	EB	130	16				
1	EB	140	24				
1	EB	110	11				
1	RB	150	40				

Region: Purcells

Site: Redding Creek (LAT LONG)

Closed site area: 1081 m^2

Date: August 27, 2011

Temperature: 9.5 ^{o}C

Conductivity: 57.5 µS

EF seconds (first, second haul): 1465, 1338

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	BT	73	5	2	BT	132	23
1	BT	93	11	2	BT	60	3
1	BT	122	21	2	BT	70	3
1	WCT	128	30	2	BT	125	24
1	WCT	110	14	2	BT	105	10
1	BT	68	4	2	BT	65	3
1	BT	72	4	2	BT	89	8
1	BT	70	3	2	BT	107	14
1	BT	95	10	2	BT	70	3
1	BT	70	3	2	BT	70	3
1	BT	59	3	2	BT	122	30
1	BT	59	3	2	BT	122	30
1	BT	122	30	2	BT	122	30
1	BT	122	30	2	BT	122	30
1	BT	122	30	2	BT	122	30
1	BT	122	30				
1	BT	122	30				
1	BT	122	30				

Region: Purcells

Site: Mark Creek (LAT LONG)

Closed site area: 680 m^2

Date: August 15, 2011

Temperature: 9.7 °C

Conductivity: 35 µS

EF seconds (first, second haul): 900, 852

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	WCT	155	38	2	WCT	80	4
1	WCT	130	24	2	WCT	152	32
1	WCT	123	18	2	WCT	124	16
1	WCT	112	16	2	WCT	146	36
1	WCT	154	37	2	WCT	108	15
1	WCT	93	7	2	WCT	140	29
1	WCT	81	7	2	WCT	123	18
1	WCT	82	5	2	WCT	139	29
1	WCT	158	39	2	WCT	113	12
1	WCT	100	11	2	WCT	96	9
1	WCT	77	4	2	WCT	81	5
1	WCT	142	22	2	WCT	124	21
1	WCT	98	11	2	WCT	125	21
1	WCT	155	38	2	WCT	92	7
1	WCT	79	5				

Region: Purcells

Site: Meachen Creek (LAT LONG)

Closed site area: 886 m^2

Date: August 16, 2011

Temperature: 9 °C

Conductivity: 83 µS

EF seconds (first, second haul): 1448, 1427

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	WCT	47	2	2	WCT	113	17
1	WCT	195	81	2	WCT	129	20
1	WCT	165	54	2	WCT	50	2
1	WCT	149	34	2	WCT	59	2
1	WCT	149	34	2	WCT	59	3
1	WCT	149	34	2	WCT	93	12
1	WCT	73	7	2	WCT	90	8
1	WCT	73	7	2	WCT	96	10
1	WCT	73	7	2	WCT	97	11

Region: Purcells

Site: Buhl Creek (LAT LONG)

Closed site area: 1120 m^2

Date: August 24, 2011

Temperature: 7.7 ^oC

Conductivity: 95.5 µS

EF seconds (first, second haul): 1384, 1185

Haul Pass	Species	Length	Weight	Haul Pass	Species	Length	Weight
1	WCT	50	2	2	WCT	130	28
1	WCT	163	52	2	WCT	180	73
1	WCT	139	32	2	WCT	132	25
1	WCT	168	54	2	WCT	195	82
1	WCT	157	43	2	WCT	149	37
1	WCT	185	68	2	WCT	117	16
1	WCT	199	77	2	WCT	127	20
1	WCT	140	31	2	WCT	138	26
1	WCT	34	7	2	WCT	135	25
1	WCT	148	42	2	WCT	165	56
				2	WCT	153	46
				2	WCT	114	12

Fish species codes: WCT= westslope cutthroat trout, BT=bull trout, EB=eastern brook trout, MW=mountain whitefish, RN=rainbow trout, LNC= long-nose dace, CC=sculpin species.

Haul pass: refers to two pass closed electrofishing using stop nets

Length= fork length, other than CC with was recorded as standard length

Weight= to nearest gram

Appendix B

Table B1. Chapter 2 surber sampling invertebrate data summary by site

Region: Rockies
Site: Lizard Creek
Sampling date: August 9, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site Biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	102	0.0895	0.994	4.874	0.20	0.84
Plecoptera	106	0.1846	2.051		0.42	
Trichoptera	42	0.1397	1.552		0.32	
Diptera	26	0.0157	0.174		0.04	
Coleoptera	23	0.0085	0.094		0.02	
Nematoda	5	0.0007	0.008		0.00	

Region: Rockies
Site: Forsythe Creek
Sampling date: August 19, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site Biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	361	0.1571	1.746	2.822	0.62	0.94
Plecoptera	74	0.0314	0.349		0.12	
Trichoptera	20	0.0465	0.517		0.18	
Diptera	29	0.019	0.211		0.07	

Region: Rockies
Site: Quarrie Creek
Sampling date: August 5, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site Biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	132	0.1796	1.996	4.674	0.43	0.92
Plecoptera	82	0.0353	0.392		0.08	
Trichoptera	25	0.1732	1.924		0.41	
Diptera	20	0.0325	0.361		0.08	

Region: Rockies
Site: Aldridge Creek
Sampling date: August 15, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	55	0.0524	0.582	0.943	0.62	1.00
Plecoptera	68	0.0239	0.266		0.28	
Trichoptera	29	0.0086	0.096		0.10	

Region: Rockies
Site: Ram Creek
Sampling date: August 13, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site Biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	134	0.2153	2.392	5.948	0.40	0.58
Plecoptera	121	0.052	0.578		0.10	
Trichoptera	34	0.1821	2.023		0.34	
Diptera	196	0.0691	0.768		0.13	
Coleoptera	10	0.0022	0.024		0.00	
Nematoda	1	0.0146	0.162		0.03	

Region: Rockies
Site: Lodgepole Creek
Sampling date: August 18, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site Biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	99	0.0898	0.998	5.343	0.19	0.86
Plecoptera	56	0.0303	0.337		0.06	
Trichoptera	36	0.2828	3.142		0.59	
Diptera	31	0.078	0.867		0.16	

Region: Purcells
Site: Mark Creek
Sampling date: August 26, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site Biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	71	0.0439	0.488	0.882	0.55	1.00
Plecoptera	29	0.0273	0.303		0.34	
Trichoptera	7	0.0082	0.091		0.10	

Region: Purcells
Site: Buhl Creek
Sampling date: August 23, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site Biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	72	0.0996	1.107	1.909	0.58	0.98
Plecoptera	40	0.0121	0.134		0.07	
Trichoptera	17	0.0587	0.652		0.34	
Diptera	3	0.0014	0.016		0.01	

Region: Purcells
Site: Gold Creek
Sampling date: August 9, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site Biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	100	0.0405	0.450	1.913	0.24	0.97
Plecoptera	60	0.017	0.189		0.10	
Trichoptera	18	0.1109	1.232		0.64	
Diptera	2	0.0033	0.037		0.02	
Coleoptera	4	0.0005	0.006		0.00	

Region: Purcells
Site: Teepee Creek
Sampling date: August 11, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site Biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	110	0.0543	0.603	1.739	0.35	0.93
Plecoptera	64	0.0336	0.373		0.21	
Trichoptera	33	0.0322	0.358		0.21	
Diptera	16	0.0364	0.404		0.23	

Region: Purcells
Site: Perry Creek
Sampling date: August 14, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site Biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	76	0.0249	0.277	1.126	0.25	0.91
Plecoptera	35	0.0085	0.094		0.08	
Trichoptera	23	0.048	0.533		0.47	
Diptera	4	0.0186	0.207		0.18	
Coleoptera	10	0.0013	0.014		0.01	

Region: Purcells
Site: Meachen Creek
Sampling date: August 26, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site Biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	86	0.0283	0.314	2.649	0.12	0.95
Plecoptera	37	0.1503	1.670		0.63	
Trichoptera	17	0.0574	0.638		0.24	
Diptera	7	0.0024	0.027		0.01	

Region: Purcells
Site: Redding Creek
Sampling date: August 25, 2011

Order	Count	Dry Weight (g)	Biomass* (g/m^2)	Total site Biomass (g/m^2)	Proportion of total biomass by order	Proportion EPT** composition
Ephemeroptera	63	0.0579	0.643	0.896	0.72	0.95
Plecoptera	26	0.0069	0.077		0.09	
Trichoptera	11	0.0133	0.148		0.17	
Diptera	3	0.0022	0.024		0.03	
Coleoptera	2	0.0003	0.003		0.00	

*Invertebrate biomass calculation based on surber sampler area of 0.09 m^2

**EPT= Ephemeroptera, Plecoptera, Trichoptera