# Effect of Permafrost Thaw Slumps on Benthic Invertebrates and on Concentrations of Persistent Organic Pollutants in lakes of the Mackenzie Delta Uplands, NT

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# Contents

## Abstract

Permafrost thaw slumping along lakeshores in lakes of the Mackenzie Delta Uplands, NT is known to alter water chemistry significantly. Its impact on benthic communities and persistent organic pollutant (POP) behaviour in lakes is not known. Benthic invertebrate communities responded to slumps through changes to community composition and size spectra. Larger taxa tended to dominate in lakes with slumps. Variability in biomass size spectra was related to total dissolved nitrogen concentration and slump size. Concentrations of POPs in *Gammarus* were negatively correlated with total phosphorus and positively correlated with the percentage of the catchment slumped. Lakes with slumps generally had higher mean concentrations of POPs in *Gammarus* (ex. $\Sigma PCBs_{Disturbed}$ = 27.54 ng/g lipid, $\Sigma PCBs_{Undisturbed}$ = 16.97 ng/g lipid; $\Sigma DDT_{Disturbed}$ =18.47 ng/g lipid and $\Sigma DDT_{Undisturbed}$ =10.86 ng/g lipid). Benthic invertebrate biomass was also negatively correlated with concentrations of contaminants in *Gammarus*, supporting the biomass dilution hypothesis. Thaw slumps have large enough impacts on the physico-chemical characteristics of lakes that they alter benthic invertebrate community composition and size-structure, and contaminant concentrations in *Gammarus*.

## Résumé

Les glissements de fonte rétrogressifs au bords des lacs dans la région du Delta du Mackenzie, NT, ont un effet connu sur la chimie de l'eau. Cepedant, l'effet de glissements de fonte rétrogressifs sur les communautés de macroinvertébrés benthiques et sur le comportement de polluants organiques persistents (POPs) dans l'eau n'est pas connu. Les communautés de macroinvertébrés benthiques ont répondus aux glissements de fonte avec des changements dans la composition de communautés et des changements dans la spectre de taille des lacs. Les taxons plus grands ont dominés dans les lacs pertubés. La variabilité des spectres de taille était lié à la concentration d'azote total dissous et à la taille des glissements. Les concentrations de polluants dans l'amphipode *Gammarus* étaient négativement correlés avec la concentration de phosphore total et positivement correlés avec le pourcentage du bassin versant affecté par un glissement de fonte. Les lacs avec les glissements avaient des concentrations de POP plus hautes que les lacs sans glissements (ex. $\Sigma BPC_{Perturbés}$ = 27.54 ng/g de lipide, $\Sigma BPCs_{Non\text{-}perturbés}$ = 16.97 ng/g de lipide; $\Sigma DDT_{Perturbé}$ =18.47 ng/g de lipide and $\Sigma DDT_{Non\text{-}perturbé}$ =10.86 ng/g de lipide). La biomasse de macroinvertbré benthique était aussi négativement correlé avec

la concentration de POPs dans *Gammarus*, ce qui soutient l'hypothèse de la dilution par biomasse. Les glissements de fonte rétrogressifs ont un impact assez fort sur les charactéristiques physico-chimiques des lacs qu'ils ont un effet sur les communautés de macroinvertébrés benthiques (composition des communautés et spectre de taille), et sur les concentrations de POPs dans *Gammarus.*

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# General Introduction

## Permafrost thaw and the Mackenzie Delta Uplands

While climate change is a global phenomenon, the increase in annual average temperature since 1980 has been twice as high over the Arctic as it has been over the rest of the world (AMAP 2011). The Intergovernmental Panel on Climate Change suggests that the global mean temperature will probably further increase by 1.4-5.8°C over the next 100 years (IPCC 1995) with warming that will be more pronounced around the poles (Lemke et al. 2007). The temperatures between the years 2005-2010 have been the warmest period ever recorded in the Arctic (AMAP 2011). In the Mackenzie Delta uplands east of the Mackenzie Delta, NWT, the mean annual air temperature has increased by more than 2.5°C since the 1970s (Burn and Kokelj 2009). Higher surface temperatures are driving changes in the ground temperatures and in the cryosphere (AMAP 2011). Permafrost underlies approximately 24% of the exposed land area of the Northern hemisphere (Taylor et al. 1999), a large portion of which is in Canada.

Thawing permafrost can lead to different terrain features. If thawing occurs in areas with large amounts of ground ice, the terrain will have a very pronounced physical response to thawing. Permafrost thaw slumps are a common type of thaw formation in permafrost terrain. They are large crater-like depressions initiated by the exposure of ice-rich permafrost following a disturbance or erosion (Burn and Lewkowicz 1990). Increased rates of permafrost thaw slump formation along lake banks have recently been observed in the Mackenzie Delta region (Lantz and Kokelj 2008). Once ice-rich permafrost thaws, the material and meltwater flow to the foot of the thawed face (Burn and Lewkowicz 1990). Slump expansion may span several years, with the head-wall retreating as more ice-rich substrate becomes exposed. A slump head-wall may retreat several metres in a single season. Slump expansion may slow when the ground ice becomes exhausted, or when the angle of the footslope becomes such that mud does not flow off it. As the slump stabilizes, it may become revegetated as plants begin to colonize the footslope. Stable slumps may also be re-initiated after several years. This may occur as a result of talik (the unfrozen zone beneath a lake) expansion beneath the lake into ice-rich materials, causing lake-bottom subsidence and rejuvenation of shoreline slumping (Kokelj et al. 2009a).

In this way, permafrost slumps can persist and expand for many years (Lantz and Kokelj 2008; Kokelj et al. 2009a).

The tundra terrain of the Mackenzie Delta uplands is underlain with ice-rich permafrost that can be several hundred meters thick (Mackay 1971). The near-surface permafrost is also ion rich because of soluble materials that become entrapped in the permafrost by a rising permafrost table, in conjunction with downward migration of ions along thermally induced moisture gradients (Kokelj and Burn 2003). The Mackenzie Delta uplands are also interspersed with thousands of small lakes (Burn and Kokelj 2009), almost one in ten of which are directly impacted by permafrost thaw slumps (Kokelj et al. 2009b). Because of the ion-rich nature of the permafrost, slump materials in the region tend to have very high concentrations of major ions and can have significant effects on lake physico-chemical and biological parameters (Kokelj et al. 2005; Thompson et al. 2010). In lakes with slumps, specific conductivity and the concentration of major ions (Ca, Mg, Na, K) increased significantly (Kokelj et al. 2005). pH also correlated positively with the % of the catchment slumped. Slump size negatively correlated with water colour and dissolved organic carbon (DOC) in the water column. The decrease in DOC and colour is likely due to the higher concentrations of base cations in lakes with slumps, which can increase the adsorption and flocculation of coloured dissolved organic matter (CDOM) from the water column (Thompson et al. 2008). Decreased concentrations of chlorophyll *a*, total dissolved nitrogen (TDN) and total phosphorus (TP) in the water column of lakes with slumps have also been found (Thompson et al. 2010). Mesquita (et al. 2010) also linked increased macrophyte biomass to permafrost slump presence, potentially due to the higher degree of light penetration to the sediments. Because of the rapid rate of climate warming in the Canadian Arctic, understanding how thawing permafrost affects lake ecosystems is of great importance.

This thesis will examine how thawing permafrost is currently altering benthic invertebrate assemblages and persistent organic pollutant (POP) behavior in lakes of the Mackenzie delta region. Studying a subset of lakes may shed light on how the Arctic landscape will change in the coming years.

## The biomass size spectrum

The first chapter of my thesis describes the response of benthic invertebrate assemblages to permafrost thaw slumps. Benthic macroinvertebrates are an important component of secondary production in lake ecosystems, and are a food source for fish. Changes to benthic macroinvertebrate assemblages will also likely impact larger lake biota. Benthos may respond to the water chemistry changes caused by permafrost thaw slumping, and to the changes in pelagic and littoral primary productivity related to decreased planktonic chlorophyll, increased light penetration, and increased macrophyte biomass in slumped lakes.

I will also combine a taxonomic description with size spectra analysis to describe how benthic invertebrate assemblages vary along a spatial gradient of increasing permafrost thaw slumping. The biomass size spectrum describes the distribution of living biomass across the entire range of organism size. The concept was first developed by Sheldon et al. (1972, 1973) to describe the planktonic communities of marine systems. The distribution of biomass among individuals in a community is a useful indicator of ecosystem structure, and describes how energy and nutrients are partitioned in an ecosystem(Boudreau et al. 1991), and how energy flows up an ecosystem through predator-prey interactions (Kerr and Dickie 2001). Size spectra are quantified as a log-density versus log-body size linear relationship. They are remarkably consistent across ecosystem types, though the parameters of the spectra such as slope, intercept or number of size classes will vary slightly with local environmental conditions (Boudreau et al. 1991; Rasmussen 1993; Bourassa and Morin 1995; Petchey and Belgrano 2010). Because size-spectra are independent of taxonomic composition, they allow for the comparison of ecosystem structure between different communities and therefore allow comparisons among assemblages formed by different species or in different habitat types. They can be used to describe, compare, and make predictions about biological assemblages and are therefore a complementary technique to taxonomic description of assemblages. Biomass size distributions have been shown to vary with environmental factors and with perturbations to the ecosystem (Sprules and Munawar 1986; Hanson 1990; Rasmussen 1993; Bourassa and Morin 1995). They have also been used to assess human impacts on systems, for example the impact of fisheries, agricultural practices, or warming temperatures (Boudreau and Dickie 1989; Gamble et al. 2006; Yvon-Durocher et al. 2011). I propose to use biomass size spectra to assess the impact of permafrost thaw slumps on benthic invertebrate communities. The changes to the system caused by thawed permafrost material should be reflected by changes to the biomass size spectrum. If larger taxa are favoured by the changes, the angle of the slope should decrease, while the angle of the slope would increase if smaller taxa are favoured. Alternatively, if all size classes are impacted to the same degree, an over-all decrease in biomass in each size class is expected.

## Persistent Organic Pollutants and Permafrost thaw

Changes to water chemistry caused by thawed permafrost material will also necessarily impact other parts of the lake systems. Among the water chemistry variables altered by thaw slumps, many will also impact the behavior of persistent organic pollutants in water. The second chapter of my thesis will focus on persistent organic pollutant (POP) concentrations in lake biota. The Arctic is largely free from direct inputs of many chemicals and contaminants, and there are now strict restrictions on production and use of persistent contaminants and pesticides throughout North America and Europe. However, contaminants remain present in air, water, animal and human populations in the Arctic. The presence of these chemicals in the Arctic has been known for many years, but the current changing Arctic climate raises new questions with regard to contaminant behaviour and presence in the Arctic.

Two POP groups are examined in this study, polychlorinated biphenyls (PCBs) and organochlorine pesticides (OCs). PCBs and OCs were widely used throughout the world beginning in the 1940s (Macdonald et al. 2000). Contaminant concentrations in the Arctic abiotic environment have been measured since the late 1980s, and in some cases even earlier (Harner et al. 2003). The Stockholm convention was the major international, legally binding instrument to manage POPs on a global scale, and came into force in 2004. It has since been ratified by 153 countries(AMAP 2009).

PCBs are a group of related chlorinated hydrocarbons formed by the chlorination of a biphenyl base molecule with molecular chlorine. A PCB molecule can have from one to ten chlorine atoms substituted around the biphenyl ring structure, making 209 different congeners. PCBs are synthetic compounds that are not easily degraded in the environment, have high molecular weights and low vapour pressures. They are highly stable, and have been used as lubricants, heat conductors and plasticizers (Newman and Unger 2003). Their high stability, coupled with their lipophilicity, allows them to remain in the environment for long periods and to accumulate in biota through the food chain. OCs are also synthetic, chlorinated compounds. These compounds have chlorine substituted for hydrogen on a hydrocarbon molecule. Like PCBs, they are very stable in the environment, degrade slowly, and tend to accumulate in lipids. These POPs have various negative impacts on human and animal health, and can affect the nervous system, immune system, reproductive system and endocrine system (Newman and Unger 2003). These chemicals are of particular interest because they accumulate through the food chain and are highly persistent in the environment.

Contaminants are transported to the arctic via long range transport (Wania and Mackay 1996). Most POPs are volatile enough to evaporate and deposit among air, water and soil at ordinary temperatures. Warm temperatures favour evaporation whereas cooler temperatures at higher latitudes favour deposition. Furthermore, contaminants of different volatilities will migrate at different velocities, and deposit at different temperatures. Highly volatile POPs tend to remain in the atmosphere for longer periods of time and at cooler temperatures, and less volatile POPs will deposit sooner. In this way, different compounds may reach the same latitudes at different rates. Once a compound reaches the latitude at which it will deposit, it may partition onto water, snow, ice, soil or vegetation. By this mechanism, contaminants tend to migrate and accumulate at the poles of the Earth.

Once the contaminant is deposited on lakes, the water chemistry of the lake will affect how it will partition within the lake system, Contaminant uptake to biota from water is largely governed by the chemical properties of the contaminants, and the physico-chemical properties of the water and sediments (Newman and Unger 2003). Because permafrost thaw slumps affect water chemistry, they will also likely impact contaminant uptake to biota. DOC, one parameter altered significantly by thaw material, has a well-established effect on contaminant behavior in lake systems (Kukkonen 1991; Haitzer et al. 1998; Haitzer 1999). Because contaminants are hydrophobic, freely dissolved POPs tend to bind to DOC in the water column when it is present. This renders them less bioavailable for uptake by organisms.

Total phosphorus and total nitrogen also tend to decrease in lakes with thaw slumps (Kokelj et al. 2005), which in turn affects planktonic biomass. Biomass of algae and other aquatic organisms can in turn impact contaminant concentration through biodilution, since biota in more productive lakes also tend to have lower concentrations of contaminants (Larsson et al. 1992; Holmqvist et al. 2005). Algae settling can also draw contaminants down to the sediments leading to their burial.

While concentrations of legacy POPs in Arctic air have been decreasing since the 1990s (Hung et al. 2009), conflicting trends for contaminant concentrations in Arctic freshwater biota have been reported. Some studies indicate that concentrations of POPs in biota from arctic freshwater systems may be rising through time, while others seem to indicate the opposite. Carrie et al. (2010) found that concentrations of PCBs and mercury in burbot of the Mackenzie River have increased in the last 25 years, despite declining or stable atmospheric concentrations. This change correlates with warming temperatures and reduced ice cover in the area. This also correlates with increased permafrost thaw in the Mackenzie Valley and along the Mackenzie River (Aylsworth and Duk-Rodkin 1997). Conversely, lower concentrations of POPs have been observed in Yukon Lakes between the period of 1992 to 2003 and 1992 and 2010 (Ryan et al. 2005; Ryan et al. 2013). These changes were mainly attributed to changes in biotic factors (decreased lipid content and increased body mass) over the study period, and possibly to the increase of lake plankton productivity. This study will shed light on how thawing permafrost impacts contaminant loads in biota in Arctic lakes.

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# Chapter 1- Impact of Permafrost Thaw Slumps on Benthic Macroinvertebrate Assemblages and on the Biomass Size Spectrum of Small Lakes of the Mackenzie Delta Uplands

## Abstract

Permafrost slumping along lakeshores has been shown to alter water chemistry significantly, but its impact on benthic communities has not been determined. Here, benthic invertebrates were sampled in lakes of the Mackenzie Delta Uplands, NT, that were affected to varying degrees by permafrost thaw slumps. Ostracoda, Diptera and Arachnida (mites) were the dominant taxa in undisturbed lakes. In contrast, disturbed lakes were dominated by Bivalvia, Ostracoda and Nematoda. In general, larger taxa had higher densities in disturbed lakes (Mollusca, Nematoda and Trichoptera) whereas smaller taxa (Ostracoda and Arachnida) had higher densities in undisturbed lakes. Variability in size spectra was most significantly related to variation in total dissolved nitrogen (TDN) concentration and permafrost thaw slump size. Overall abundance per size class increased with increased TDN and decreased with increased disturbance size, however responses to TDN were largest for the smallest invertebrates whereas the larger invertebrates seemed to benefit from increased thaw slump size. The positive effect of disturbance is possibly due to the increased macrophyte biomass in lakes with slumps, which increases the habitat for certain large invertebrate taxa. This study indicates that thawing permafrost is altering not only composition, but also the size-structure of benthic invertebrate communities in small lakes.

# Introduction

The uplands east of the Mackenzie Delta, NT, are covered in ice-rich permafrost that is several hundred meters thick (Mackay 1971) and is interspersed with thousands of small lakes (Burn and Kokelj 2009). Almost one in ten lakes greater than 1ha in area are directly impacted by retrogressive thaw slumps, one of the most dramatic forms of thawing permafrost (Kokelj et al. 2009b). These thaw slumps are common formations in permafrost terrain, and are initiated by the exposure of ice-rich permafrost following a disturbance or erosion of the active layer (Burn and Lewkowicz 1990). Permafrost slumps can persist and expand for many years , and may be re-initiated by talik (unfrozen ground) expansion beneath the lake, causing lake-bottom subsidence and rejuvenation of shoreline slumps (Lantz and Kokelj 2008; Kokelj et al. 2009b). Thawing of ice-rich sediment creates a mud-slurry which, if formed on the banks of rivers or lakes, can flow into open waters. Due to climate warming, permafrost thaw slump activity in the Mackenzie Delta uplands has been increasing since the 1950s and is expected to continue increasing as temperatures warm (Lantz and Kokelj 2008). The clays of the Mackenzie Delta uplands are ion-rich, and can have significant impacts on water chemistry. Changes to water chemistry include decreased dissolved organic carbon (DOC) and water colour in lakes, and increased concentration of certain major ions (Ca, Mg, Na, K, $SO_4^{2-}$, $CO_3^{2-}$, and $HCO_3^{-}$) (Kokelj et al. 2005). The increased water clarity associated with slumping, along with increased nutrients in the sediments of lakes with slumps (calcium, magnesium and strontium) are also linked to increased macrophyte biomass and the development of a more complex benthic habitat (Mesquita et al. 2010). Slumps are also linked to decreases in the concentrations of chlorophyll-*a*, total dissolved nitrogen (TDN) and total phosphorus (TP) in the water column. These parameters are sometimes used as indicators of lake productivity, and changes to them may affect higher trophic levels within the lake systems.

Benthic macroinvertebrates are important contributors to secondary production in lake ecosystems. The biomass size spectrum, describing the distribution of living biomass across the range of organism sizes, is a useful indicator of ecosystem structure and describes how energy and nutrients are partitioned in an ecosystem. Because spectra are independent of taxonomic composition, they facilitate the comparison of ecosystem structure among assemblages formed by different species or in different habitat types. They can be used to describe, compare, and make predictions about biological communities, and are therefore a complementary technique to taxonomic description of communities.

Biomass size distributions have been shown to vary with environmental factors and with perturbations to the ecosystem (Sprules and Munawar 1986; Hanson 1990; Rasmussen 1993; Bourassa and Morin 1995). They have also been used to assess human impacts on systems, for example the impact of fisheries, agricultural practices, or warming temperatures (Boudreau and Dickie 1989; Gamble et al. 2006; Yvon-Durocher et al. 2011).

In the uplands east of the Mackenzie Delta, where dissolved ions, nutrients, and organic carbon concentrations in lakes are dramatically altered following permafrost thaw slump formation, it is likely that macroinvertebrates will also respond in some way to thaw slumping. However, the response is difficult to predict. The increased macrophyte biomass of lakes with slumps would provide greater habitat complexity for benthos and allow a wider variety of species, greater densities and greater biomass. Alternatively, decreased chlorophyll-*a,* dissolved nutrients, and DOC, as found in slump affected lakes, may coincide with lower inverterbrate biomass and density (Jorgenson 1992). If invertebrates are resource-limited, it is expected that benthic invertebrate biomass and density will be lower in slump affected lakes, due to their lower chlorophyll-*a* concentrations. On the other hand, if invertebrate abundance is limited by habitat complexity in these systems, the negative impact of reduced pelagic nutrients on abundance of invertebrates could be compensated by the increased habitat complexity related to higher macrophyte biomass in slumped lakes.

## Study Site

Sixteen small lakes were sampled in the tundra uplands to the east of the Mackenzie Delta between June and July 2010 (figure 1). The density of lakes in the uplands area is very high (Kokelj et al. 2009b), many of which have taliks beneath them (Burn and Kokelj 2009). The permafrost in this area contains large amounts of ground ice that, if exposed, is susceptible to thaw. The ice and surrounding sediments are ion-rich, due to entrapment of soluble materials by a rising permafrost table, and downward migration of ions along thermally-induced suction gradients (Kokelj and Burn 2005). Thawing of this iron-rich permafrost and ground ice due to the formation of permafrost thaw slumps along the banks of lakes has been shown to significantly alter water chemistry (Kokelj et al. 2005). Lakes with slumps have increased concentrations of major ions (Ca, Mg, Na, K, $SO_4^{2-}$, $CO_3^{2-}$, and $HCO_3^-$) and increased specific conductivity (Kokelj et al. 2005). The ion-rich clays are also linked to the flocculation of organic material in the water and its precipitation to the sediments, which in turn leads to lower dissolved organic carbon (DOC) and water colour (Thompson et al. 2008). Permafrost thaw slumps are also linked to decreases of planktonic chlorophyll-*a* concentrations, total dissolved nitrogen (TDN) and total phosphorus (TP) in the water column. Changes to the macrophyte communities have also been observed (Mesquita et al. 2010), with higher macrophyte abundances being found in lakes affected by slumping. This was linked to increased nutrients in sediments of slump lakes and increased water clarity.

All lakes selected for this study were taken from a list of lakes studied in the area, provided by Dr. Steven Kokelj (Aboriginal and Northern Affairs Canada, Yellowknife, NT). In order to quantify the effect of the permafrost thaw slumping on the lake, the % of the catchment that is thawed was calculated using the area of the permafrost thaw slump and the area of the catchment (Kokelj et al. 2005). This proportion has been shown to correlate with changes to water chemistry that are associated with permafrost thawing, such as dissolved organic carbon and specific conductivity (Kokelj et al. 2005).

The percentage of the lake's total lake catchment area occupied by retrogressive thaw slump ranged between 0-34% (Table 1). Five reference lakes that had no thaw slumps in their catchments were also included. To minimize the effect of latitude, distances between lakes were kept to a minimum: all lakes were within 70km of each other.

## Materials and Methods

All 17 lakes were accessed by helicopter between June 12$^{th}$ 2010 and July 6$^{th}$ 2010. Water was sampled twice during this period, once at the beginning and once at the end. Benthic invertebrates were sampled with an Ekman dredge once between these dates. Three Ekman samples were taken in the littoral zone of each lake, at similar depths (between 50 cm and 90 cm) and in similar soft substrate types – macrophytes were avoided (due to problems with the Ekman dredge jaws closing around macrophytes), as were very sandy or gravelly substrate. Samples were stored in sealed plastic containers until the return to the laboratory, at which point they were sieved to 500 µm and stored in 95% ethanol. Samples were split using a Folsom splitter in order to achieve a more manageable sample size, and identified, generally to Family, using keys from McCafferty (1981) and Thorp & Covich (2010). One Ekman dredge was also taken at each lake to obtain a surficial sediment sample for isotopic analysis (analysed for %C and %N by the G.G. Hatch Isotope Lab, University of Ottawa).

All identified invertebrates were measured in order to estimate biomass. Digital images were captured with a scanner at 600DPI, and each individuals' total length was measured using Image-Pro software (2008). In order to account for small invertebrates that were lost through the 500µm sieve, correction factors were calculated using a sieve retention model (Morin et al. 2004). The model was of the form:

$$\ln(p/[1-p]) = a + b \log_{10}(RL) + c \log_{10}(RL)\log_{10}(M)$$

Where *p* is the probability that an organism is retained in a sieve, the relative length (*RL)* is the ratio of body length to sieve mesh size, *M* is the sieve mesh size, and *a, b* and *c* are fitted coefficients that varied according to the shape of the invertebrate (from perfectly round to long and thin). Dry-mass for all invertebrates was then determined using length-dry mass regression models (Benke et al. 1999). Density (ind/m $^2$) and biomass (µg/m$^2$) were then calculated by dividing the total number of individuals and the dry-mass of individuals by the area of the Ekman dredge (15.2 cm x 15.2cm).

Lakeshores were seined in order to obtain amphipod and fish biomass for a separate analysis (Chapter 3). Fish were identified to species level and their length was measured.

Water samples were sent to the National Centre for Environmental Testing (NLET, Burlington, ON) for nutrient analysis and to the Taiga Environmental Laboratory (Yellowknife, NT) for analysis of ions. Average values of the two sample periods were used for statistical analyses.

## Statistical Analysis

Linear regressions of nutrients (chlorophyll-*a*, total dissolved nitrogen and total phosphorus) were first constructed as a function of % Catchment Disturbed to establish that the trends observed in previous studies on nearby lakes were consistent with trends in this selection of lakes (see Figure 3). A principal component analysis was then run on a correlation matrix of environmental variables to determine if other strong trends existed between environmental variables and different matrices of catchment disturbance. The first two principal components are plotted in Figure 4. A second PCA was run on a correlation matrix of environmental variables and invertebrate biomass per taxon. Taxa that accounted for over 95% of biomass were included in the ordination to remove noise caused by rare taxa.

Size spectra were constructed for each lake. Organisms were grouped into $\log_2$ size classes based on dry-mass (µg), and plotted as $\log_{10}$ density (ind/m$^2$) against the upper limit of $\log_{10}$ mass (µg) for each size class following previously established methods (Ahrens and Peters 1991). A polynomial regression model was fitted to the constructed size spectra to best represent the shape and variability of the data. Several water chemistry parameters and slumping parameters were included in the initial model (including specific conductivity, chlorophyll-*a*, % catchment disturbed, disturbance size, catchment area, lake area, % $O_2$, TDN and TP). Stepwise regression analysis was then used to select the terms that best described the variation in density per size class.

All statistical analyses were done using R 2.15.2 (R Core Team 2013). Data were log-transformed prior to analysis to meet the assumptions of parametric analyses.

## Results

The taxa collected from the litoral zone with the Ekman dredge were Ostracoda (Order Podocopida, Families Candonidae, Ilyocyprididae, Limnocytheridae, Cytherididae, Cyprididae), Arachnida (Suborder Hydracarina, Halacaridae, Oribatei), Bivalvia (Family Sphaeriidae), Copepoda (Order Cyclopoida), Gastropoda (families Valvatidae and Lymnaeidae), Clitellata (Subclasses Oligochaeta and Hirudinae), Nematoda, Malacostra (Order Amphipoda, Family Gammaridae (Genera *Gammarus*)), Insecta (Order Trichoptera, Families Limnephilidae, Molannidae, Phryganeidae, Lepidostomatidae, Leptoceridae and Order Diptera, Family Chironomidae), Coleoptera (Families Chrysomelidae, Haliplidae) and the phylum Tardigrada. These taxa were similar to those found by Mesquita (2008) in nearby lakes of the Mackenzie Delta.

To compare densities between lakes, raw data were used that had not been put through the sieve retention model (Fig. 2). This was done in order to make values more comparable with other studies that had not used this model. Ostracoda had the highest mean abundance when comparing mean abundance per taxa in all lakes (44 903 ind/$m^2$). This group was followed by Diptera (29 823 ind/$m^2$) and Arachnida (19 820 ind/$m^2$). In undisturbed lakes, the three most abundant taxa were also Ostracoda(63 123 ind/$m^2$), Diptera(42 373 ind/$m^2$) and Arachnida (34 097 ind/$m^2$). In disturbed lakes, the three most abundant taxa were Bivalvia (31 235 ind/$m^2$), Ostracoda (26 683 ind/$m^2$) and Nematoda (24 021 ind/$m^2$). Dominant taxa are those presented in the PCA analysis (Fig. 4). T-tests revealed no significant differences between the mean densities of all disturbed and undisturbed lakes for any taxa aside from mites and diptera (which had higher densities in Undisturbed lakes). Mean abundances of Ostracods, Copepods, Coleoptera, Hirudinae and Oligochaetes were also higher in undisturbed lakes, but not significantly. Disturbed lakes had higher mean densities of Bivalva, Mollusca, Nematoda and Tardigrada, though no differences were significant.

Total dissolved nitrogen, total phosphorous and chlorophyll-*a* in lake water decreased significantly with increasing % catchment disturbed (see Figure 3). Total dissolved nitrogen varied from 0.69mg/L to 1.56mg/L, total phosphorus varied from 0.32 μg/L to 20 μg/L, with one very high concentration of 61μg/L , and chlorophyll-a varied from 0.7μg/L to 4.55μg/L, though one lake had very high concentrations (14.6μg/L).

Average biomass and density per lake were significantly negatively correlated with disturbance size and positively correlated with total dissolved nitrogen (Table 2). Biomass alone was positively correlated with TP. There were no significant correlations with % catchment disturbed or catchment size.

Total nitrogen, total phosphorous and chlorophyll-*a* in lakes did not vary between lakes with and without slumps (One-way ANOVA, $p>0.05$), although total dissolved nitrogen was close to significance, with a $p = 0.052$. When lakes were further divided by slump activity (active slump, stable slump or no slump), no significant differences were detected.

The percent organic Carbon and Nitrogen in the sediments were very strongly correlated ($r=0.98$). Both %C and %N appeared negatively correlated with % catchment slumped, though neither was a significant correlation ($r = -0.44$ , $p=0.074$ and $r = -0.45$, $p=0.071$ respectively). This is likely because of the limited data set of 17 lakes.

**PCAs**

A principal components analysis was run on lake characteristics and water chemistry variables (% Catchment disturbed, specific conductivity, latitude, pH, lake surface area, temperature, dissolved oxygen, catchment size, chlorophyll-*a* in the water column, dissolved organic carbon, total phosphorous, total dissolved nitrogen and water colour) (see Figure 4a). Axis 1 of the principal components analysis explained 46% of the variance among sites and reflected mostly the gradients in % catchment disturbed and water colour, which were negatively correlated. Sites on the right side of the ordination had low values for water colour but relatively high proportions of slumping in their catchments, whereas sites on the left had a low proportion of slumping in their catchments and high values for water colour. Axis 2 explained an extra 16% of the variance among sites. This axis reflected mostly the size of the catchment and the temperature of the water. Total phosphorous, total nitrogen and chlorophyll-*a* contributed slightly more to the first axis, but contributed to both axes. Lakes falling on the left of the figure are mainly reference lakes, whereas lakes on the right have thaw slumps. Lakes on the left of the figure tend to have higher concentrations of chlorophyll-*a*, total nitrogen and total phosphorous. Lakes in the bottom of the figure tend to have larger catchments and higher chlorophyll-*a* in their water columns.

A PCA with a sub-set of water chemistry variables and biomass of the most abundant taxa was run (Fig. 4b) to determine which taxa correlated with these parameters. Axis 1 explained 33% of the variance between sites, and the second explained an extra 17% (50% in total). The taxon whose biomass varied most along the first axis was the Acariformes. Along the second axis, Chironomidae, Ostracoda and Sphaeriidae varied most. Acariforme biomass seemed to correlate strongly with chlorophyll-*a* and total phosphorous, whereas Chironomidae, Sphaeridae and Ostracoda seemed to correlate most strongly with total nitrogen. The taxa that seemed positively correlated with % catchment disturbed were Gastropoda, Amphipoda and Oligochaeta (though only slightly).

## Size Spectra

Stepwise regression led to the selection of the following final model. A polynomial regression, including three mass terms, was the best fit for the data. Total dissolved nitrogen was retained as a main term, and the interaction between total dissolved nitrogen and the three mass terms. Disturbance size was also retained as a main effect, and the interaction term between log mass$^2$ and Disturbance size. The coefficients of all the retained terms were significant. An ANOVA comparing the model with and without interactions found that the model with interactions was significantly better than the model without.

$$\begin{aligned}\text{logDensity} = {} & 5.46 + 1.06(\text{logmass}) - 1.13*(\text{logmass})^2 + 0.17(\text{logmass})^3 \\ & + 1.36(\text{logTDN}) - 0.56(\text{logDisturbance}) + 1.13(\text{logmass})(\text{logTDN}) - 1.16(\text{logmass})^2(\text{logTDN}) \\ & + 0.21(\text{logmass})^3(\text{logTDN}) + 0.061(\text{logmass})^2(\text{logDisturbance})\end{aligned}$$

Figure 5 is a graphical representation of the size spectrum model ($R^2$= 0.71, p<$2.2e^{-16}$). Invertebrate density increased with increasing nitrogen concentrations, and peaked in lakes with no disturbances. The size-spectrum model explained 70% of the total variability in density, with body mass accounting for the largest portion (67%). Total dissolved nitrogen and catchment size both explained around 1% of the total variability, though both were statistically significant in the model. The effect of nitrogen is a positive effect on the density of invertebrates in the smaller size classes. However, the positive effect of nitrogen on density per size classes is much more pronounced near the mode of the size spectra.

Figure 5 describes the effect of disturbance on density, statistically controlled for different dissolved nitrogen concentrations (low, mid and high concentrations). The largest effect of disturbance on densities occurs in the small size classes, and this trend is clear at all concentrations of nitrogen. The densities of large invertebrates are much less variable in these lakes, and changes to small size classes drive the differences in total biomass and density. At high nitrogen concentration, the effect of disturbance on size spectra is minimal. At very low concentrations of nitrogen, the effect of disturbance is much more pronounced, where the peak density varies from 8 326 ind/$m^2$ with a large disturbance to 24 672 ind/$m^2$ in lakes with no slump.

When comparing between high total dissolved nitrogen and low total dissolved nitrogen lakes, the most obvious difference is the peak density (120 000 ind/$m^2$ for average TDN, 7 000 000 ind/$m^2$ for high TDN and 25 000 ind/$m^2$ for low TDN). For larger size classes, there are no large differences in densities between lakes, though the positive effect of slumping on this part of the figure is more pronounced for lakes with low total nitrogen.

## Fish

Fish were caught in 6 of the 17 lakes studied. All fish were caught in disturbed lakes, and species caught were Ninespine Stickleback, Pond Smelt, and Northern Pike (see table 3). No effect of fish was visible on size-spectra (no difference in shape of the spectra was obvious between lakes with and without fish), though no statistical analyses were conducted. Including fish in the spectra would require an estimate of biomass, which was not measured (lakes were seined until enough biomass was reached for a separate analysis, without taking into account the area seined or the number of passes made).

## Discussion

As witnessed in previous studies (Kokelj et al. 2005), permafrost slumping on the banks of lakes caused significant differences in water chemistry ( Fig 4b). Specific conductivity and pH correlate positively and significantly with % catchment slumped whereas water colour and DOC correlated negatively and significantly with % catchment slumped. Total dissolved nitrogen, total phosphorous and chlorophyll-*a* were all negatively correlated with % catchment slumped. It appears that permafrost slumping is removing these TDN and TP from the water column, potentially precipitating them to the sediment or even removing them from surface run-off before they reach the lake by adsorption onto exposed clays . Thompson *et al.* (2008) showed that the experimental addition of slump materials into lake water likely causes the precipitation of organic matter from the water column, due to flocculation of organic matter brought about by increased cations in lakes with slumps. Carbon and Nitrogen content in the sediments were very strongly correlated. This might indicate that carbon and nitrogen are precipitating simultaneously from the water column as organic matter. Phosphorus has been shown to adsorb onto organic and inorganic particles in the water column, and onto sedimenting clays or organic matter. Major divalent metal ions in water, such as calcium and magnesium (which are higher in slumped lakes) tend to enhance the aggregation and sedimentation rates of these particles (Wetzel, 2001). Because slump material is composed of ion-rich clay, its addition to the water column increases specific conductivity of the water, resulting in organic matter (carbon, nitrogen) and phosphorus precipitation from the water column. As phosphorus and nitrogen are removed, algal productivity decreases and chlorophyll-*a* in the water column drops.

The slight decreases in phosphorus and nitrogen in slump lakes, and subsequent decreases in chlorophyll-*a* concentrations, are likely partially responsible for the observed decreases in invertebrate biomass and densities in slump lakes, suggesting these small arctic lakes are nutrient limited. This relationship is similar to how invertebrate biomass increased following experimental additions of phosphorus and nitrogen to arctic lakes (Jorgenson 1992). The regression analysis also revealed that the abundance of small animals declines with increasing slump size, independent of nitrogen. The additional decrease of small invertebrates in slumped lakes (beyond what would be expected based on nitrogen alone) is possibly a result of the changes to habitat brought about by shoreline slumps. The increased macrophyte abundance may favour larger invertebrates, such as bivalves, mollusks and amphipods.

No significant correlation between biomass or density of invertebrates and %catchment area slumped was discernible, though both invertebrate biomass and density are negatively correlated with disturbance size (ha). In lakes with slumps, water is clearer (Kokelj et al 2005) and others have indicated that benthic macrophyte communities are better developed (Mesquita et al. 2010). Gilinsky (1984) showed that when macrophytes were experimentally added to caged portions in a lake, density of invertebrates in surrounding sediments tended to decrease as certain taxa migrated onto the new habitat. Since this study sampled sediments only (macrophytes were avoided), the lower densities on sediments in lakes with more macrophytes (lakes with permafrost thaw slumps, as described by Mesquita et al. 2010) might be caused by a similar phenomenon of migration of invertebrates into macrophytes.

Invertebrate taxa identified here were similar to those identified by Mesquita (2008) in other lakes of the Mackenzie Delta Uplands. However, densities calculated here were an order of magnitude higher than densities from Mesquita. This is likely caused by the differences between the sampled areas in both studies, since only littoral sediments were sampled here as opposed to transects through the entire lake irrespective of sediment type and depth in Mesquita (2008). The most abundant taxa were not significantly different between disturbed and undisturbed lakes (when comparing densities with t-tests). However, in general, the smallest taxa had higher densities in undisturbed lakes (Ostracoda, Arachnida and Copepoda). Bivalvia and Mollusca both had higher mean densities in undisturbed lakes, taxa often associated with macrophytes. The size-spectra revealed that size-structure of benthic invertebrate communities was also changing in response to slumping and to nitrogen concentrations.

## Size Spectra

Variations in density of invertebrates in each size class depended most on body mass. Density was also significantly related to total dissolved nitrogen and disturbance size, though these factors explained only a small portion of the variability in density in each size class compared to body mass alone. The main effect of nitrogen was an increase in invertebrate density per size class, whereas the main effect of permafrost slump size was a decreased density per size class across spectra. For the beginning portion of the figure (smallest size classes), total nitrogen had a positive effect on density per size class. Smallest invertebrates respond most to total nitrogen. The PCA analysis of biomass and water chemistry variables shows that Acariformes (mites), Ostracoda and Chironomidae are the taxa that vary most closely with TDN. These taxa accounted for most of the density of small and medium size classes. The distribution of benthic ostracods at the sediment-water interface is a function of the availability of food, the substratum surface and the particle size distribution (Smith and Delorme 2010). No particle-size analysis was done on the sediments sampled, so a comparison of these is difficult in the context of the present study. However, as similar habitats were targeted while sampling, I will assume that substratum surface and particle size distribution were similar among lakes. Food availability is likely the dominant driver of Ostracoda abundance. Ostracods are detrivores and herbivores, and require particulate organic matter that can easily be picked up and "swept" into their mouths. Ostracods can also feed off algae or plants, if present. Here, chlorophyll-*a* concentration (which is an expression of algal biomass) was significantly related to total dissolved nitrogen, which is higher in lakes without slumps. Because aquatic mites were not identified past Order, little can be said about their relationship with nitrogen and chlorophyll. Different species can inhabit different habitats, and there are several functional feeding groups including predacious, parasitic and algivorous (Smith et al. 2010). In these lakes, their density and biomass is related to nitrogen concentrations, and a large portion of species present might therefore be algivorous.

The negative size-nitrogen interaction on density for medium to larger size classes (2.99 µg to 696µg) is unexpected, and also corresponds to a positive influence of disturbance size in this same part of the figure. Lakes with low nitrogen and permafrost thaw slumps would also be lakes with higher macrophyte biomass, which would alter the benthos species composition in sediments. Certain taxa (such as chironomids) might migrate onto macrophytes when they are present, leading to lower densities in the sediments (Gilinsky 1984). The increased densities of very large invertebrates (above 1000 μg) related to the % of the catchment slumped is probably also related to increased macrophyte biomass in slump lakes. The PCA shows amphipod and gastropod biomass are the most closely correlated with % catchment slumped. These two taxa composed the majority of the large size classes. Amphipods are mobile, epi-benthic invertebrates whose abundance is generally higher in macrophytes, where refugia from predators exist. Gastropods in freshwater habitats are generally detrivores or herbivores, which are often associated with macrophytes or cobbles where they scrape periphyton for food (Brown and Lydeard 2010). They are also mobile and can migrate between habitats. The increased abundance of macrophytes in lakes with disturbances, in spite of decreasing nitrogen in these lakes, is the most likely explanation for increased density of large taxa.

Fish were found in a small sub-set of lakes (6 of the 17 lakes), and were only present in lakes with permafrost slumps (though fish have been caught in lakes without permafrost slumps in similar nearby lakes). No clear effect of fish presence could be seen on size spectra. This might perhaps be caused by low fish densities in lakes. Lakes with slumps do appear to present better habitat for fish. Macrophyte biomass is higher (Mesquita et al. 2010), which acts as refugia and shelter. Certain invertebrate taxa that are normally found in and around macrophytes, such as amphipods and chironomids, are also prey for many minnow-size fish. Periphyton growing on macrophytes, and the macrophytes themselves, are also a food source for small minnow-size fish. Ninespine stickleback in particular require vegetation for nesting, and adults are often found in association with dense vegetation (Richardson et al. 2001).

## Conclusion

Benthic invertebrate density, biomass, and the size-structure of the benthic community appear to be changing following thaw slump formation. Density and biomass tended to decrease with disturbance size, likely due to the decreased trophy of disturbed lakes (decreased chlorophyll-*a*, total dissolved nitrogen and total phosphorus). This is therefore an indirect effect of slumping on benthic communities.

Size spectrum analysis suggests that nitrogen concentration in the water column and the size of the permafrost thaw slump affect the benthic invertebrate size-structure of lakes. Densities of small size classes are positively correlated with total dissolved nitrogen. Small invertebrate density is also reduced in lakes with large disturbances but this reduction is less important or reversed for large invertebrates. This differential response to slumping and nitrogen by small and large invertebrates could be due to the increased macrophyte biomass in slump lakes. In slump lakes, density of certain invertebrates in sediments might decrease due to migration onto macrophytes, whereas other invertebrate species which tend to prefer macrophyte habitats such as amphipods and gastropods would become more abundant, resulting in spill-over onto sediments from macrophytes.

This research focused only on one substrate type, though studies have shown that different substrates can significantly alter species composition and size-structure of communities (Gilinsky 1984; Bourassa and Morin 1995). Because macrophyte abundance is increasing in slump lakes, sampling invertebrate communities from macrophytes would give a clearer picture of how the entire lake system is likely to change. Obtaining data for fish would also expand the size-spectra, potentially revealing how fish communities are affected by slump formations.

As temperatures in the Arctic continue to rise, increased permafrost thaw in the Northern hemisphere is inevitable. This study indicates that freshwater lakes in ice rich permafrost terrain will likely be changing in the coming years, with possible decreases in biomass and density of benthic invertebrates, and shifts in the size-structure of benthic communities. If the entire Canadian Arctic responds in the same manner to permafrost thaw, the potential impacts on Northern communities may be vast.

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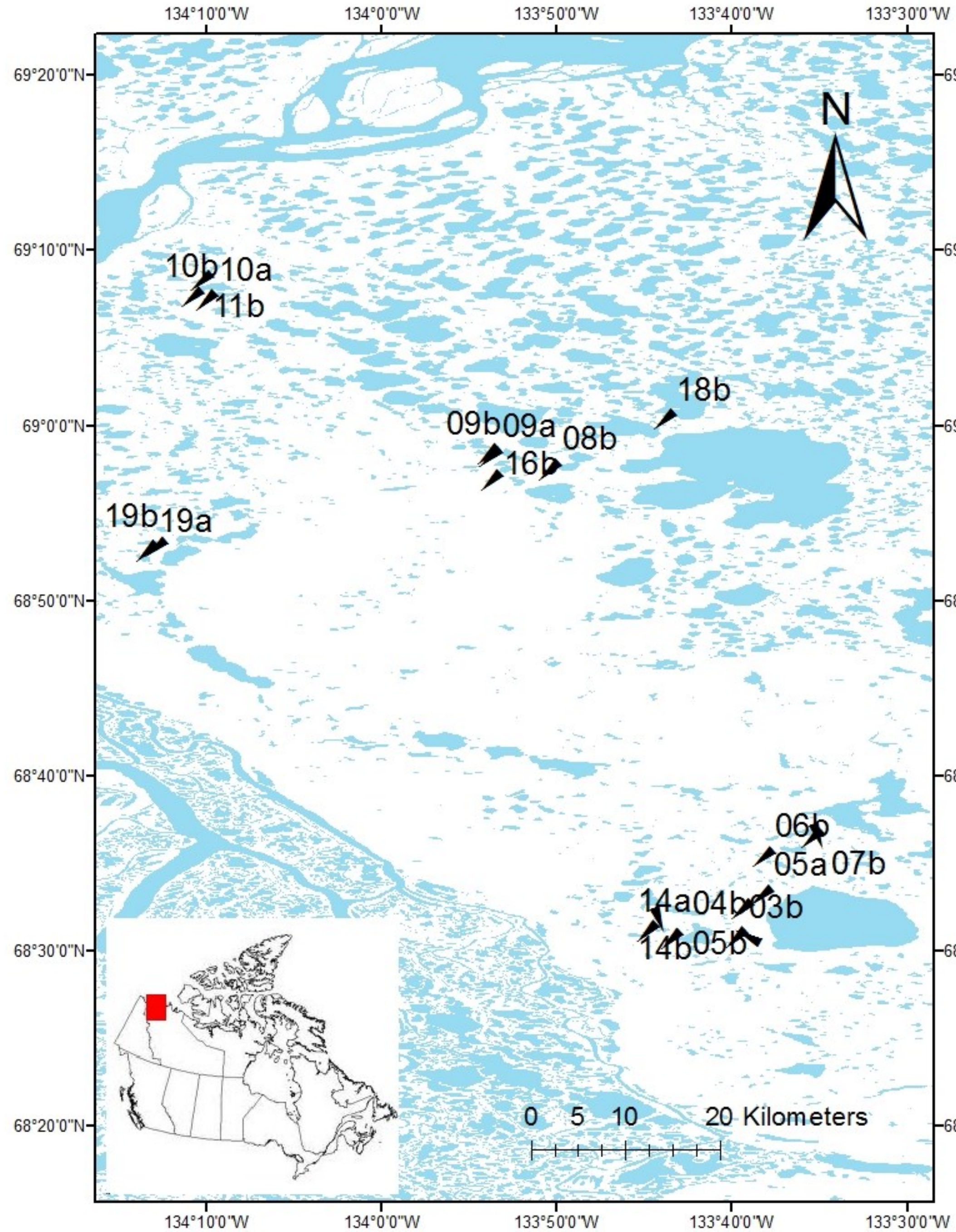



Figure 1: Map of the studied lakes in the upland tundra study region east of the Mackenzie River Delta. Lakes with permafrost slumps are labeled “b” and control lakes are labeled “a”

Table 1: Physical characteristics of sampled lakes, Mackenzie Delta Uplands, NT. Lakes sampled between June and July 2010.

| Lake | % Catchment disturbed by slump | Lake Region | Catchment Area (ha) | Lake Surface Area (ha) |
|---|---|---|---|---|
| 03b | 23.6 | South | 16.3 | 4 |
| 04b | 13.8 | South | 17.8 | 5 |
| 05a | 0 | South | 20.9 | 2.9 |
| 05b | 7.3 | South | 27.7 | 2.8 |
| 07a | 0 | South | 18.1 | 1.4 |
| 07b | 3 | South | 34.7 | 3.1 |
| 08b | 12.1 | Central | 32.7 | 6.5 |
| 09a | 0 | Central | 29.3 | 3.1 |
| 09b | 34.3 | Central | 7.2 | 3.6 |
| 10b | 30.9 | Central | 23.3 | 11.4 |
| 11b | 6.4 | Central | 39.4 | 10.5 |
| 14a | 0 | South | 33.5 | 3.4 |
| 16b | 6 | Central | 63 | 14.1 |
| 18b | 12.1 | Central | 4.7 | 2.7 |
| 19a | 0 | Central | 24.6 | 2.8 |
| 19b | 5.9 | Central | 28.1 | 6.4 |
| 36b | 19.9 | South | 24.4 | 3.9 |

*source: Kokelj et al. 2005, and Kokelj (unpublished)

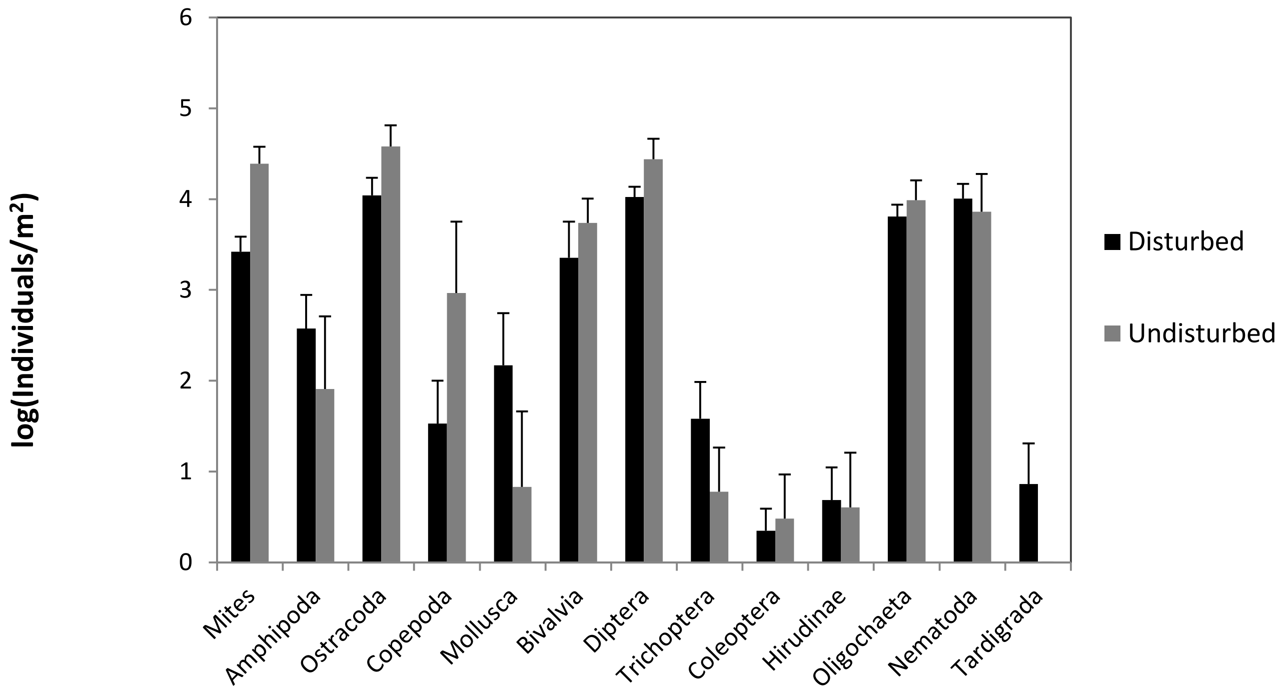



Figure 2: Average benthic invertebrate density (+ standard error) of different taxonomic groups in lakes of the Mackenzie Delta Uplands.

Table 2: Coefficients of determination of simple linear regressions between benthic invertebrate biomass, density and different lake parameters. P-values in parentheses. For significant coefficients, (+) or (-) is included to indicate if the correlation is positive or negative

| | Catchment size (ha) | Disturbance size (ha) | % Catchment disturbed | TDN | TP |
|---|---|---|---|---|---|
| Biomass ($\mu g/m^2$) | 0.06 (0.35) | 0.27 (0.03) (-) | 0.11 (0.20) | 0.48 (0.002) (+) | 0.26 (0.045) (+) |
| Density ($ind/m^2$) | 0.04 (0.43) | 0.36 (0.01) (-) | 0.19 (0.08) | 0.31 (0.02) (+) | 0.24 (0.057) |

Table 3: Sampled lake and fish species caught

| Lake | Fish species |
|---|---|
| 7b | Ninespine stickleback (*Pungitius pungitius)* |
| 9b | Northern Pike (*Esox lucius)* |
| | Ninespine stickleback (*Pungitius pungitius)* |
| 8b | Northern Pike(*Esox lucius)* |
| | Pond Smelt (*Hypomesus olidus)* |
| 19b | Northern Pike(*Esox lucius)* |
| | Pond Smelt (*Hypomesus olidus)* |
| 16b | Ninespine stickleback (*Pungitius pungitius)* |
| 11b | Ninespine stickleback (*Pungitius pungitius)* |
| | Pond smelt (*Hypomesus olidus)* |

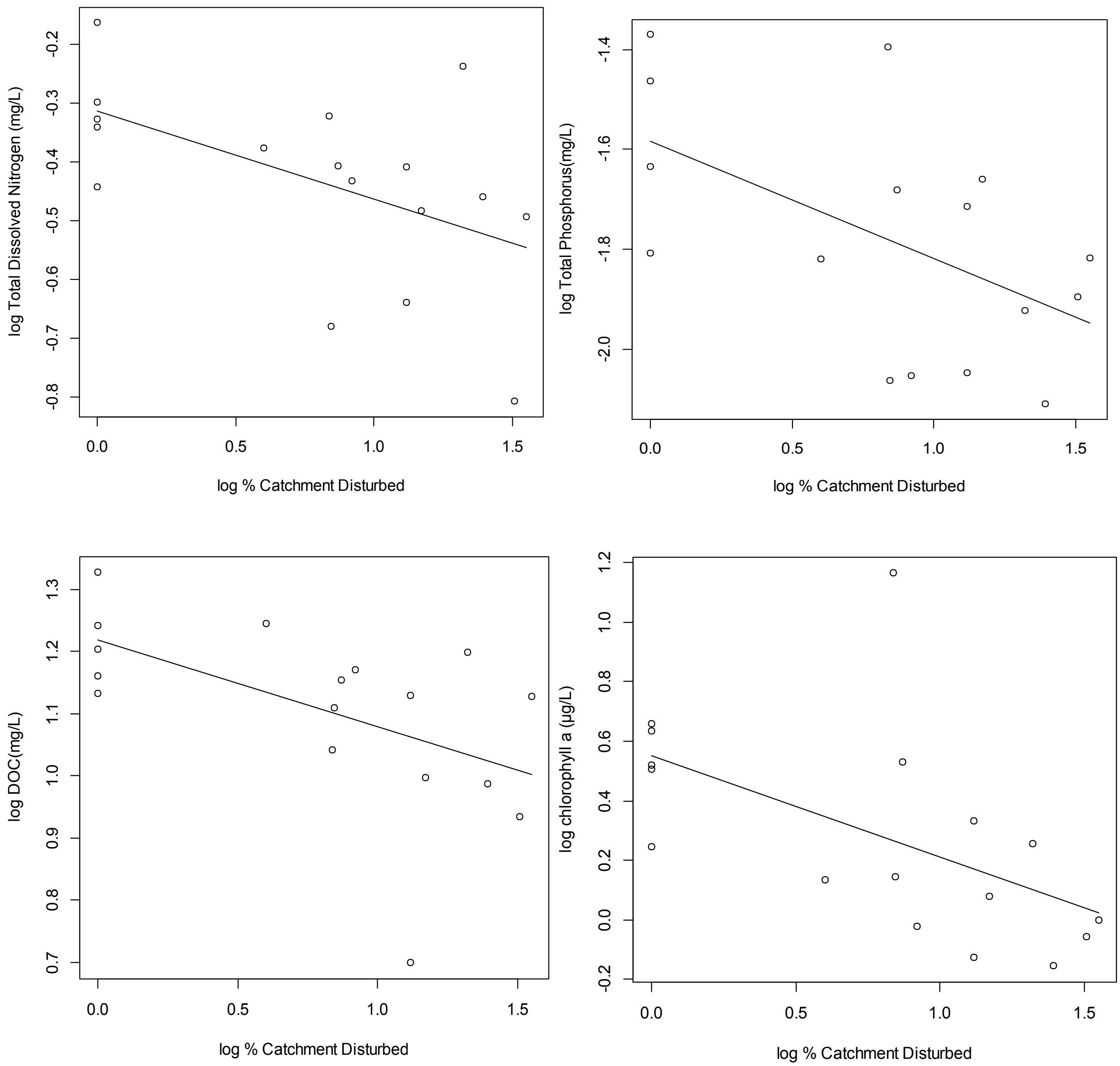



Figure 3: Correlations between nutrients and chlorophyll-a in the water column and % catchment disturbed by slumps. Nutrients presented here are total dissolved nitrogen ($r2=0.28$ $p= 0.027$), total phosphorus ($r=0.56$ $p=0.026$), dissolved organic carbon ($r=0.54$ $p=0.023$) and chlorophyll-a ($r= 0.55$ $p=0.019$)

a)

b)

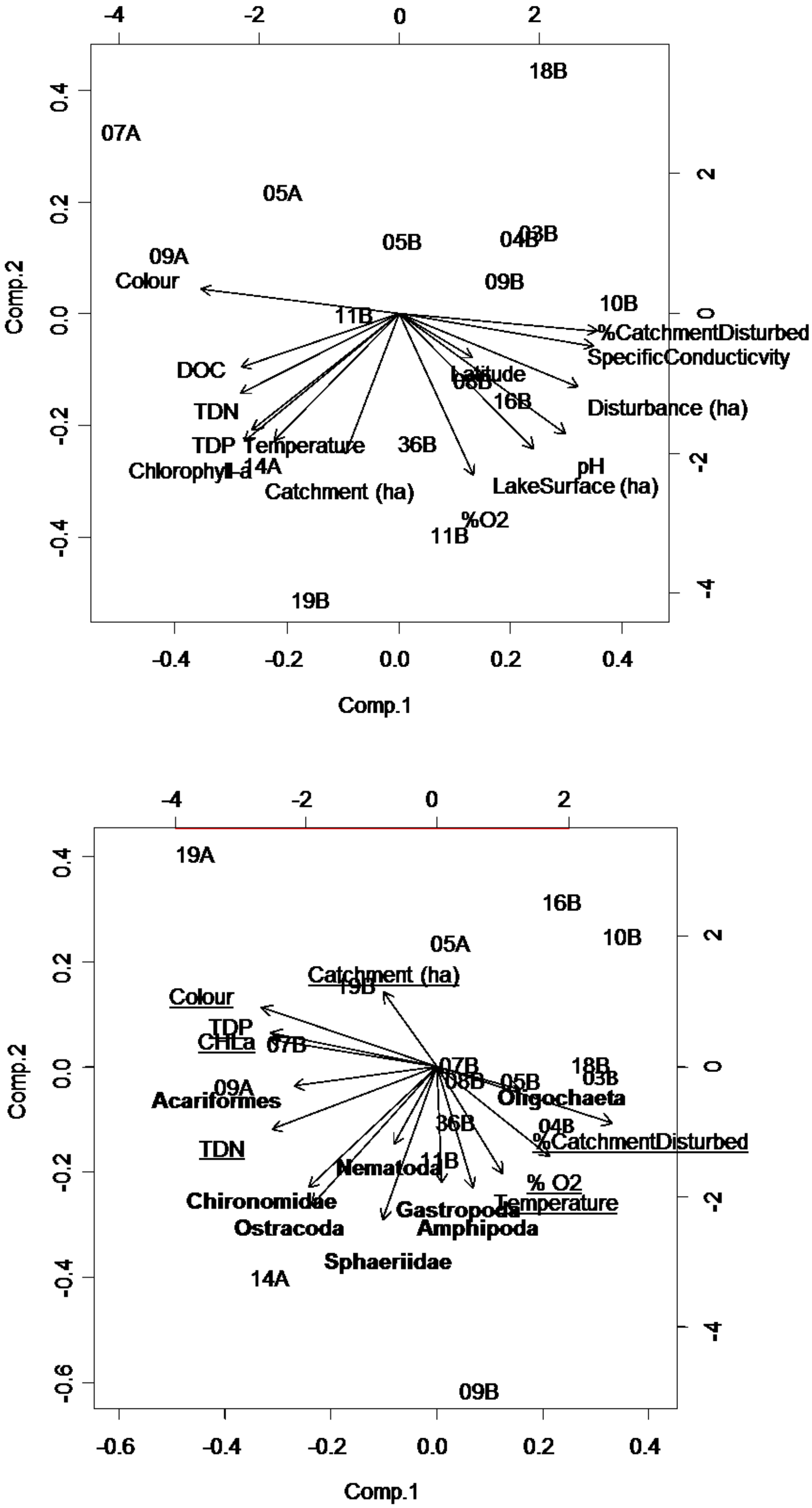
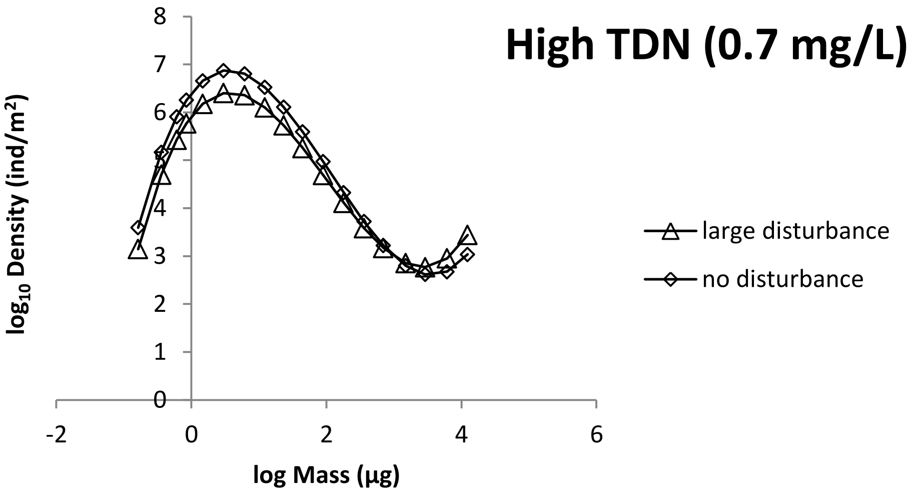
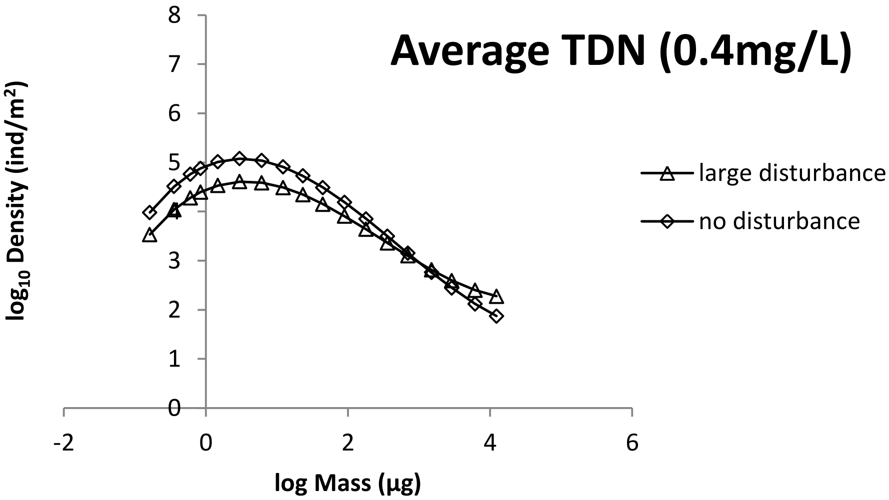



Figure 4: a) Principal Components Analysis of water chemistry and physical variables. b) Principal Components Analysis of water chemistry and physical variables variables with taxa that accounted for over 95% of Total biomass. Taxa are in bold, water chemistry and physical variables are underlined.



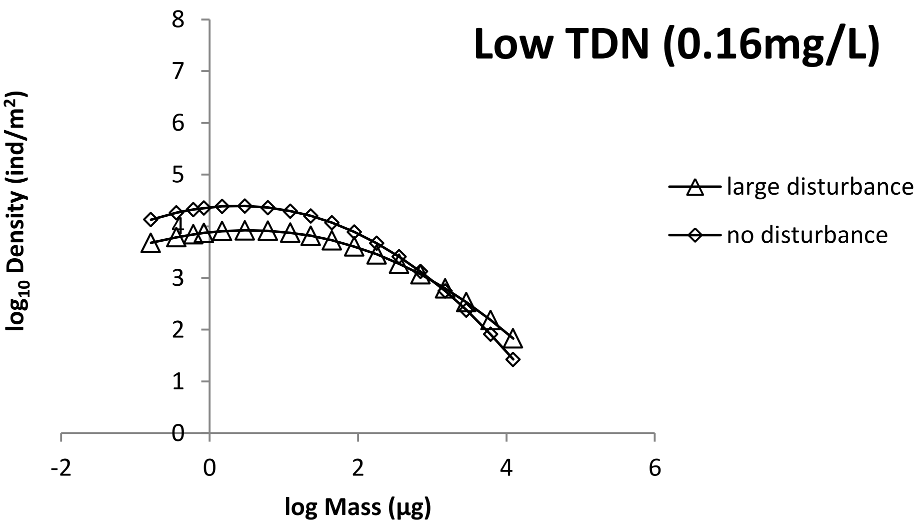




Figure 5: Size Spectra for all lakes with varying concentrations of total dissolved nitrogen and with different size permafrost thaw slumps (average-size disturbance= 2.01 ha, large disturbance=7.19 ha)

# Chapter 2 – Impact of Permafrost Thaw Slumps on Contaminant Concentrations in Amphipods and Fish of the Mackenzie Delta uplands, NT

# Abstract

Permafrost thaw slumps are large formations on lake banks that are a source of thawed material to lakes. When present, they alter water chemistry parameters that may affect the way persistent organic pollutants (POPs) behave in the environment. Polychlorinated biphenyls (PCBs) and organochlorine pesticides (OCs) were measured in macroinvertebrates, fish, and sediments from a remote system of lakes in the Northwest Territories to examine the effect of permafrost thaw slumps on contaminant concentrations in biota. POPs in fish, the amphipod *Gammarus* and sediments were found in comparable concentrations to other Arctic lakes. The concentrations were negatively correlated with lake trophic status (total phosphorus) and positively correlated with the percentage of the catchment affected by slumping. Benthic invertebrate biomass was also negatively correlated with concentrations of contaminants in amphipods, supporting the biomass dilution hypothesis. Lakes with slumps generally had 1.1 -1.7x higher concentrations of POPs in amphipods than lakes without slumps (ex. $\Sigma PCBs_{Disturbed}$ = 27.54 ng/g lipid, $\Sigma PCBs_{Undisturbed}$ = 16.97 ng/g lipid; $\Sigma DDT_{Disturbed}$ =18.47 ng/g lipid and $\Sigma DDT_{Undisturbed}$ =10.86 ng/g lipid; $\Sigma Hexachlorohexanes_{Disturbed}$ =7.16 ng/g lipids, $\Sigma Hexachlorohexanes_{Undisturbed}$ =6.69 ng/g lipids). These results suggest that permafrost thaw slumps have a large enough impact on the physico-chemical characteristics of lakes that they alter contaminant concentrations in amphipods.

## Introduction

Despite the fact that there are strict restrictions on production and use of persistent contaminants and pesticides throughout North America and Europe, they are displacement via long range transport of these contaminants allow their presence in air, water, animal and human populations in the Arctic (Wania and Mackay 1996). While the presence of these chemicals has been known for many years, the current changing Arctic climate and the expected widespread thawing of permafrost raise new questions about contaminant dynamics in the Arctic. It is currently unclear how these changes will affect the distribution and behaviour of persistent organic pollutants (POPs). Conflicting trends for contaminant concentrations in Arctic freshwater biota have been reported, with some studies indicating concentrations of some legacy contaminants may be rising (e.g. Carrie et al. 2010), while others seem to indicate the opposite (Ryan et al. 2013).

The uplands east of the Mackenzie Delta, NWT, are underlain by ice-rich permafrost, interspersed with thousands of small lakes (Burn and Kokelj 2009) that may be affected by warming. In this region, the thawing of permafrost creates thermokarst topography and various thermokarst formations. One example of these formations are permafrost thaw slumps, which are crater-like formations created when permafrost with large quantities of ground ice thaws. Studies have shown that climate warming is increasing the rate of permafrost thaw slump activity in the Mackenzie Delta Uplands (Lantz and Kokelj 2008). Currently, almost one in ten of the lakes are directly impacted by permafrost thaw slumps (Kokelj et al. 2009b). When a slump forms on a lake bank, the mud slurry enters the water, leaving a large depression on the bank. Thaw slumping on lake banks has been shown to alter water chemistry significantly (Kokelj et al. 2005; Thompson et al. 2010), as well as sediment chemistry and lake productivity (Thompson et al. 2010; Mesquita et al. 2010; Deison et al. 2012).

Many of the water chemistry parameters altered by permafrost slumps have documented effects on the uptake of POPs to lake biota (Larsson et al. 1992; Haitzer 1999; Berglund et al. 2001; Holmqvist et al. 2005; Roessink et al. 2010). Dissolved organic carbon (DOC) flocculates out of the water column when slump material is added to water (Thompson et al, 2008), and DOC has a negative impact on the uptake of hydrophobic contaminants from water (Black and McCarthy 1988; Kukkonen

1991; Haitzer et al. 1998) due to contaminant sorption (absorbtion or adsorption) onto dissolved or particulate organic matter in the water column. Slumps have also been shown to lower the nutrient concentration of lakes and hence decrease algal productivity (Thompson et al. 2010), two parameters which can alter contaminant concentration in biota due to reduced biomass dilution (Taylor et al. 1991; Larsson et al. 1992; Berglund et al. 2001; Holmqvist et al. 2005; Carrie et al. 2010).

Because thaw slump activity alters the concentration of DOC in the water column so drastically, and because the intensity of thaw slump activity varies spatially, there is a very wide range of DOC in slump-affected lakes in the Mackenzie Delta Uplands. This location therefore presents the opportunity to study the effect of dissolved organic carbon on the behavior of contaminants in lake-water. Likewise, the significant change in lake productivity over the range of catchment slumping is of equal interest. Because these changes reflect the effect of thawing permafrost on lakes, this study may also be indicative of how wide-spread permafrost thaw will impact the arctic landscape over the coming years.

The objective of this study was to quantify the effect of permafrost thaw slumps on POPs in amphipods and fish. Amphipods were chosen as study species because of their presence in most lakes of the study area. Samples were taken from a set of small lakes that had varying degrees of thaw slump activity on their banks. Because of the known impacts of thermokarst slumps on water chemistry (especially the shifts in DOC), it is likely that there will be a variation in contaminant concentrations in lakes depending on the degree of thaw slump activity. This study aims to relate contaminant levels measured in biota to the magnitude of permafrost slumping and to physico-chemical parameters of the water-column altered by slumps.

## Methods

### Study area

Fourteen small lakes were sampled in the tundra uplands east of the Mackenzie Delta between June and July 2010. The area is underlain by continuous permafrost that is up to several hundred metres thick (Mackay 1971). The permafrost in the area contains large amounts of ground ice that, if exposed, is susceptible to thaw. The ice and surrounding sediments are also ion-rich (Mackay 1979; Kokelj et al. 2002), with the near-surface permafrost becoming ion-rich due to entrapment of soluble materials by a rising permafrost table and the downward migration of ions along thermally-induced suction gradients (Kokelj and Burn 2005). Thawing of the permafrost along the banks of lakes leads to the release of this material, significantly altering water chemistry once it enters the water column (Kokelj et al. 2005).

Lakes were selected along a range of catchment slumped (up to 34% catchment slumped) that included 4 reference lakes that were unaffected by slumps. To minimize the effect of latitude, distances between lakes were kept to a minimum: all lakes were within 70km of each other.

Water was sampled twice during the sampling season, on June 12$^{th}$ 2010 and on July 6$^{th}$ 2010. These values were averaged for all statistical analyses. All biota were sampled between these dates. Water samples were sent to the National Centre for Environmental Testing (NLET, Burlington, ON) for nutrient analysis and to the Taiga Environmental Laboratory (Yellowknife, NWT) for analysis of metals and ions.

Sampling was conducted from one random location in the littoral zone along lake banks, in areas free of macrophytes. Sediment samples were taken in each lake by scraping the top 1cm of sediment from an Ekman dredge sample. For duplicates, a separate Ekman dredge was sampled. Sediment samples were stored in Whirlpack bags and frozen until analysis. After sediment sampling, seining for *Gammarus* and fish was conducted in the same area that was sampled for sediments until around 8 grams of biomass was collected. If fish were found, they were euthanized by a blow to the head and pithing of the brain. Samples were stored in 50mL centrifuge tubes over ice until return to the laboratory. Fish samples were then measured and weighed, rinsed with clean water and frozen until analysis. *Gammarus* samples were allowed to sit in clean water in the fridge over-night to clear their guts. After 24 hours, they were rinsed, blotted to remove excess water, and frozen until analysis. Only amphipods over 2 mm in length were kept for analysis.

## Sampling methodology

Biota and sediment samples were analysed for PCB and OC contaminant concentrations. Sub-samples of sediments were analysed for inorganic, organic and total carbon by the GG Hatch Isotope lab at the University of Ottawa. $\delta^{15}N$ and $\delta^{13}C$ were measured in biota by the GG Hatch Isotope Laboratory at the University of Ottawa (measured by combustion on a VarioEL III Elemental Analyser followed by "trap and purge" separation and on-line analysis by continuous-flow with a DeltaPlus Advantage isotope ratio mass spectrometer coupled with a ConFlo II). These measurements were taken to approximate the trophic level of biota.

Average benthic invertebrate biomass for 17 lakes was quantified (see chapter 2). 11 of these 17 also coincided with lakes that were sampled for POPs. Biomass was calculated from Ekman dredge samples of the sediments. Three Ekman samples per lake, at similar depths (between 50cm and 90cm) and in similar substrate types were chosen (sediments free of macrophytes and without clay substrates). Invertebrates were identified using keys from McCafferty (1981) and Thorp & Covich (2010). Inverterabrate biomass was estimated and a sieve retention model was applied (see Chapter 2 for methods).

## Analsysis of Persistent Organic Pollutants

### *Quality control*

All glassware used for the extraction of organic pollutants was cleaned using established protocols. Glass Pasteur pipettes were heated for 12 hours at 200°C. Any metallic tools used for sample processing (such as blender blades and metal spatulas) were cleaned using the same procedures.

Method blanks were run with each round of sample extractions. To blank correct before analysis, mean blank concentration was subtracted from the extract concentration. Concentrations of biological samples were also corrected for the loss of analyte in the 10% aliquot used in lipid determination (see below).  Analysis of certified reference materials (SRM2974a organics in freeze-dried mussel tissue, SRM1944 New York/New Jersey waterway sediment) were conducted with every 10-15 samples for quality assurance.

Method detection limits were calculated according to Loconto (2006).  Detection limits for individual congeners and metabolites can be found in Table 5 and 6.

### Contaminant Extraction and Lipid Analysis

#### *Sample Preparation*

Biological samples were homogenized by freezing in liquid nitrogen and grinding to a fine powder in a glass blender with metal blades.   Six-eight grams of material were used per extraction for both fish and *Gammarus*.  For fish, whole bodies were used and several individuals were combined to form one sample.  Pond smelt (*Hypomesus olidus)* size averaged 5.26 cm and varied between 2.7 cm – 9 cm, while nine-spine stickleback (*Pungitius pungitius)*size averaged 3.78 cm and varied between 2.4 cm - 6.8 cm.

Sediment samples were homogenized by mixing thoroughly by hand.  Any pebbles or macrophytes were removed prior to analysis.  Sediments were then centrifuged to remove excess water, and weighed out to 12-15 grams for extractions.

#### *Extraction*

Samples were ground with pre-cleaned Hydromatrix.  This mixture was packed into cells for accelerated solvent extraction (ASE).  Samples were spiked with an internal standard (composed of PCB 30 and 205, as well as 1,2,4,5 tetrabromobenzene, δ-hexachlorohexane and endrin ketone) to monitor recovery.  The internal standards are chosen to monitor the recovery over the entire chromatogram. The samples were extracted with dichloromethane, hexane and acetone (Omnisolv, high-purity grade).

Methods for PCB and DDT analysis followed the National Center for environmental Testing (NLET) protocol. Liquid-liquid extraction was then performed to remove excess water using DCM, a 3 % sodium chloride solution and a saturated sodium sulphate solution, before being passed through sodium sulphate.

Percent lipids for biota samples was determined by drying a small aliquot (10%) of the extracted sample in pre-weighed aluminum weighing dishes for 24 hours. The dishes were allowed to dry and were then re-weighed to determine the portion of the aliquot composed of lipid. The samples were then filtered through a 0.2 µm PTFE filter before being cleaned further using Preparative Liquid Chromatography (Prep-LC) through 19x150 mm and 19x300 mm Envirogel GPC clean-up columns connected in series, using dichloromethane. Final cleanup and fractionation of samples was performed by column chromatography. The final extract was passed through a column packed with activated silica gel, evaporated to 500 µl and spiked with the internal standard octachloronaphthalene (OCN). Samples were analyzed on an HP6890 Series gas chromatography system with electron-capture detector, using pulsed splitless injection. Chromatograms were analysed using Agilent ChemStation software.

## Statistical Analysis

Statistical analyses were done using R 2.15.2 (R Core Team 2013) and Microsoft Excel. All data were log-transformed prior to analysis to meet the assumptions of parametric testing.

BSAFs (biota-to-sediment accumulation factors) were calculated using the following formula:

$$\mathrm{BSAF} = \frac{\mathrm{C_B}}{\mathrm{C}_{sed}}$$

Where $C_B$ is equal to the lipid-normalized concentration of the contaminant in biota, and $C_{sed}$ is the organic carbon-normalized concentration of contaminant in sediments. BSAFs are used to evaluate the bioavailability and absorption of chemicals from sediments (Gewurtz et al. 2000). BSAF > 1 indicate that bioaccumulation may be playing a role in the transfer of organic contaminants in the benthic food web.

Principal Component Analysis (PCA) was used to summarize variation and correlation structure among observations of physical variables and contaminant concentrations in biota and sediments. Linear regressions were then used to test the relationship between variables. Independent variables included were % catchment slumped, disturbance size (ha), disturbance size(ha)/lake size(ha), specific conductivity, DOC (mg/L), chlorophyll-*a* (μg/L) , average biomass of invertebrates per lake and total nitrogen (mg/L).

In certain cases, similar but insignificant trends were present for regressions of different contaminants or congeners. Fisher's method for combining probabilities was used in these cases to combine p-values from several tests and determine an aggregate p-value (Sokal and Rohlf 1995).

## Results

All lakes in this study except one are oligotrophic (4-10 μg/L of phosphorus; Wetzel 2001) but nevertheless cover a gradient of water chemistry correlated with catchment slumping (Figure 2) similar to what has been observed by others in this area (Kokelj et al. 2005; Thompson et al. 2008; Kokelj et al. 2009b). Total dissolved nitrogen varied from 0.687mg/L to 1.56mg/L and was negatively correlated with % catchment slumped. Total phosphorus varied from 3.2 μg/L to 20 μg/L, with one high outlying value of 61μg/L, and was also negatively correlated with % catchment slumped. Chlorophyll-*a* concentrations varied from 0.7μg/L to 4.55μg/L in all lakes except one, which had a significantly higher concentration of 14.6μg/L. Chlorophyll-*a* was negatively correlated with % catchment slumped ($p=0.01$), and positively correlated with total nitrogen and total phosphorus. Slumping also correlated negatively with DOC in the water column ($p=0.0056$), which varied tenfold from 2.6 mg/L to 26mg/L. Specific conductivity varied from 42.4μS/cm to 925.5μS/cm, and correlated positively with % catchment disturbed ($p=0.0012$).

Based on the comparison of $\delta^{15}N$ values observed here with those from other studies, the sampled biota likely occupied two trophic levels (Kidd et al. 1998b). Pond smelt (mean of 10.62 ‰, standard error= 0.21) and ninespine stickleback (mean = 10.20 ‰, standard error= 0.47) are likely secondary consumers and *Gammarus* (mean of 5.82‰) is likely a primary consumer. $\delta^{13}C$ values for amphipods and fish were very similar (averages of -32.63‰ and -32.16‰ respectively).

## Results of POP extractions

Different contaminant groups were quantified by summing concentrations of isomers and related compounds. ΣHCH was quantified as the sum of αHCH, βHCH and γHCH, with a log $K_{ow}$ s of 3.7-3.8. ΣCHL was quantified as the sum of chlordane-related compounds (heptachlor, heptachlor epoxide, cis-chlordane, trans-nonachlor, oxychlordane, γ-chlordane), with a log $K_{ow}$ of 6.0. ΣCBz was calculated as the sum of pentachlorobenzene and hexachlorobenzene, with log $K_{ow}$s of 4.5-5.5. ΣDDT is the sum of *o-p'*-DDT, *o-p'*-DDT, *p-p'*-DDD, *o-p'*-DDD, *p-p'*-DDE, *o-p'*-DDE, with log $K_{ow}$ s of 6.2-6.9. ΣAldrins was calculated as the sum of Aldrins and Dieldrin, with log $K_{ow}$s ranging from 5.48-5.66. ΣPCB was the sum of PCB congeners 18, 29, 31-28, 52, 49, 44, 99, 87, 110, 149, 118, 146, 153, 132, 105, 138, 163, 187, 183,128, 156, 201, 157, 180, 170, 195,194, 206 and 209, with a range of log $K_{ow}$s of 6.2-7.0. In general, the lipophilicity of the contaminant groups in this study are as follows: ∑DDT ≈ ∑PCB >∑CHL >∑CBz ≈ Aldrins >∑HCH.

Among the PCB congener groups, the highest concentrations were detected in hexachlorbyphenyls, pentachlorobiphenyls, tetrachlorobiphenyls and trichlorobiphenyls. Heptachlorobiphenyls were also detected, but in much lower concentrations. Nonachlorobiphenyls and decachlorobiphenyls, congeners with the highest degree of chlorination, were not detected.

Concentrations of ΣPCBs and of different OC pesticides in biota are listed in tables 2 and 3. Table 2 shows lipid corrected concentrations (ng/g lipid), whereas table 3 shows concentrations in ng/g wet weight. Concentrations in *Gammarus* were lower than in fish for all contaminants, which is expected for bioaccumulating contaminants due to their lower trophic position. Pond smelt generally had higher concentrations than ninespine stickleback. PCB congeners found in highest concentrations in *Gammarus* were 153, 49 and 52. In fish, highest concentrations were found for PCB congeners 49, 44 and 153. ΣPCBs were the contaminant found in highest concentrations, followed by ΣDDTs, ΣCBz and ΣAldrins. Duplicate samples of *Gammarus* were collected for lakes 7B (for sum PCBs, coefficient of variation =6.2%) and 19B (for sum PCBs, coefficient of variation=9.4%).

For *Gammarus*, the concentrations of contaminants generally followed the pattern of contaminant lipophilicity, though chlordanes were found in the lowest concentrations. In ninespine stickleback, the concentrations of the different contaminant groups followed the same trends as for *Gammarus* except for ΣDDT, which was the 3$^{rd}$ lowest concentration. In pond smelt, ΣPCBs, ΣAldrins and ΣCHLs were in highest concentrations.

Sediment concentrations of POPs were comparable to other arctic lakes (Allen-Gil et al. 1997), though were lower than values reported by Kidd (1998) from the Nunavut region (table 4). ΣPCB were found in the highest concentrations followed by ΣHCH. ΣDDT and ΣCHL had very similar concentrations. ΣAldrins and ΣCBz were detected, though in quite low concentrations.

BSAFs could only be calculated for lakes where concentrations in both *Gammarus* and surficial sediments were known (10 of the 14 lakes). For ΣPCBs, the mean BSAFs value was 8.9 and ranged from 1.47 to 12.8 (with one very high value of 45.17). This range was similar for Hexachlorohexanes (mean= 2.4, range of 0.56 to 9.17), DDT (mean=11.75, range of 2.22 to 29.1) and Chlordanes (mean= 13.9, range of 1.88 to 37, with one high value of 53.7). Other OC pesticide groups had higher BSAFs: Aldrins ranged from 1.95 to 114.3 (mean of 36.9) and Chlorobenzenes from 3.25 to 127.7 (mean of 30.29).

## Results of statistical analyses

In order to decide which metric (% catchment disturbed, disturbance size or disturbance size/lake size) of permafrost slumping to use, linear regressions were run between different water chemistry variables and these metrics. They were all found to correlate significantly with water chemistry parameters. Percent catchment slumped was therefore chosen to remain consistent with the literature (Kokelj et al. 2005; Deison et al. 2012).

The compounds in this study represent a range of lipophilicities, and generally more lipophilic, higher chlorinated compounds bioaccumulate to a greater degree and have longer half-lives in biota than the less chlorinated compounds. The log $K_{ow}$s, which represent the tendency of the chemical to partition between the organic phase and the aqueous phase, were taken from Kidd (Kidd et al. 1998a), Mackay (Mackay 1982) and Mackay (Mackay et al. 2006).

Concentrations of ΣPCBs in *Gammarus* , fish and sediments were comparable to levels measured in other Canadian and American Arctic Lakes (Kidd et al. 1998; Kidd 1998, Schindler and Kidd from AMAP 2009). These values are similar to what has been observed by others. For example, Brunson (Brunson et al. 1998) calculated a range of 2.5 to 26.6 for field collected oligocheates. Tracey and Hansen (1996) found similar BSAFs for PCBs (mean= 43.65) and OC pesticides (mean=91.2) in benthic invertebrates of several functional feeding groups, and Viganò et al. (2007) found a mean BSAF of 50.11 for PCBs.

The PCA conducted on water chemistry variables revealed a clear gradient in water chemistry associated with increased % catchment slumped (Figure 3). Lakes with permafrost thaw slumps on their banks tended to have higher specific conductivity, but low DOC, chlorophyll-*a*, invertebrate biomass and total nitrogen. The first principal component explained 59% of the variance and the second principal component explained an additional 18%. Over-all, parameters differed between disturbed and undisturbed lakes with undisturbed lakes clustering mostly on the left of the figure and lakes with slumps clustering on the right side of the figure (Figure 3, panel B). The first principal component mainly reflected the gradients in % catchment disturbed, disturbance size, disturbance size/lake size and total nitrogen. The second principal component mainly reflected the gradients in catchment and lake surface area, and benthic invertebrate biomass.

A second PCA analysis run on environmental variables, concentrations of contaminants in *Gammarus*, concentrations of contaminants in sediments, and BSAFs (Figure 4). The 1$^{st}$ principal component explained 31% of the variance, and the 2$^{nd}$ principal component explained an additional 15%. The first principal component reflected mostly the gradients of BSAFs, concentrations in sediments, % catchment slumping, invertebrate biomass and certain water chemistry parameters (DOC and total nitrogen). The second principal component reflected mostly variations in total phosphorus, chlorophyll-*a,* specific conductivity, and concentrations of POPs in Gammarus. Lakes falling on the right of the figure tended to have higher concentrations of most contaminants in their sediments, whereas lakes on the left had higher BSAF values. Lakes falling in the bottom portion of the PCA had higher concentrations of POPs in *Gammarus*, and had higher portions of their catchments affected by slumps. Lakes with high chlorphyll-*a* and total phosphorus tended to fall towards the top of the figure.

Following the PCA, linear regressions and t-tests were used to determine which parameters were significantly correlated to POP concentrations in *Gammarus*. T-tests were performed, though the dataset was highly uneven and contained only 3 lakes unaffected by thaw slumps. Parameters chosen for these analyses were benthic invertebrate biomass, % catchment slumped, chlorophyll-*a* in the water columns and DOC in the water column. Latitude and lake size were also parameters that were tested, and neither had a significant correlation.

Total phosphorus and total nitrogen correlated negatively with concentrations of POPs in *Gammarus*. The correlation between Total Phosphorus with ΣPCB ($p=0.041$), ΣDDT ($p=0.0047$) and Aldrins ($p=0.016$) was significantly negative (Figure 6). Combining all p-values for POPs with Fisher's method was also significant ($p=0.0257$) over-all. Certain individual PCB congeners (PCB 49, 163 and 156) were significantly negatively correlated with TP in the water column. Chlorophyll-*a* concentration in the water column also had an over-all negative correlation with concentration of contaminants in biota, though no significant trends were detected.

POP concentrations in *Gammarus* were negatively correlated with invertebrate biomass for only certain contaminant groups (Figure 7). This correlation was significant for ΣAldrins ($p=0.035$) and ΣChlordanes ($p=0.039$). The combined p-value with the Fisher method was not significant for the main POP groups.

POP concentrations in *Gammarus* were inversely related to overall benthic biomass, suggesting dilution of a relatively constant pool of contaminants in variable amount of biomass. If contaminants are diluted in biomass, the concentration of contaminants in the biota should be proportional to the reciprocal of biomass (Taylor et al. 1991), and therefore that the log of concentration declines with the log of biomass with a slope of -1. Model 2 regressions (Ranged Major Axis) were used to calculate slopes with biomass as the independent variable. RMA regressions were used because the variables on both the independent and the dependent axes had error associated with them, and therefore Ordinary Least Squares regressions were not appropriate. For regressions with invertebrate biomass as the independent variable, the slopes Σ PCBs (-0.59), ΣDDT ( -0.89), ΣAldrins ( -1.06) and ΣChlordanes (-1.18) were not significantly different than -1 (Table 7). The biomass dilution hypothesis has therefore not been refuted.

Concentrations of POPs in surficial sediments (ng/g organic carbon) of shallow zones showed no significant correlations with physical or chemical lake parameters (Table 8). All contaminant groups tended to be negatively correlated with total nitrogen in the water column. Though this trend was only significant for DDT ($r=-0.54$, $p=0.04$), the combined p-value using the Fisher's method was significant ($p=0.023$). There was also a general negative correlation between the concentrations of POPs in surficial sediments and DOC in the water column, and POPs in surficial sediments and the % of the catchment that was slumped, though none were significant. Concentrations of POPs in surficial sediments (ng/g dry weight) of shallow zones showed no significant correlations with physical or chemical lake parameters (Table 9).

Correlations were also computed for BSAFs and water chemistry parameters (Table 10). All contaminant groups tended to be positively correlated with Total Nitrogen. This trend was significant over-all when combining p-values with the Fisher's method.

## Discussion

Concentrations of POPs in biota in this study are similar but slightly lower than concentrations in Arctic lakes from the late 1990s. Concentrations in invertebrates and fish were similar to those measured in small Arctic lakes in the Yukon (Kidd et al. 1998a). Compared to a lake in the Keewan region of Nunavut (Kidd et al. 1998b), concentrations were in the same order of magnitude but were lower for all contaminants measured. This is consistent with long-term PCB and OC trends measured from air in the Canadian arctic, which shows a slow decline over the same time period (Hung et al. 2005). Differences between concentrations in fish and *Gammarus* likely reflected the effect of trophic level, such as has been observed in other lakes (Kidd et al. 1998b). Within the study region, organochlorine pollutant levels were uncorrelated with latitude, supporting the assumption that lakes received similar atmospheric fallout of contaminants.

The pattern of PCB congeners detected in amphipods is what would be expected given the log-$K_{ow}$s of the different congener groups. PCBs with mid-range log-Kows were found in highest concentrations. Log-$K_{ow}$ represents the partition coefficient of a compound between n-octanol and water, and is used to reflect the hydrophobicity of a compound and to imply relative partitioning between aqueous phases of the environment and lipids in an organism. Generally, congeners with a higher degree of chlorination have higher log-$K_{OW}$s (Newman and Unger 2003). For compounds with very low log-$K_{OW}$s, a larger portion stays dissolved in water and is not easily taken up by organisms or bound to organic carbon in the water column. For compounds with log-$K_{OW}$ values of approximately 3-6, uptake to organisms increases with $K_{ow}$ (Newman and Unger 2003). Above log-$K_{ow}$ values of 6-7, the rate of uptake slows and begins to decrease (Kelly et al. 2004). Dietary uptake efficiency is therefore generally highest for contaminant groups with mid log-$K_{ow}$s (Gobas et al. 1993). This pattern is consistent with concentrations found here: none of the most highly chlorinated congeners (nonachlorobiphenyl or decachlorobiphenyl) were detected in amphipods, nor the least chlorinated congeners (monochlorobiphenyl or dichlorobiphenyl).

The mean concentrations of contaminants in amphipods were generally higher in lakes with permafrost thaw slumps. This was true for PCBs, Endosulfans, Aldrins, Chlordanes, hexachlorocyclohexanes and DDTs, and t-tests showed that the difference was significant for PCBs, Endosulfans, Chlordanes and Hexachlorocyclohexanes. Linear regressions confirmed that amphipods in lakes with a larger portion of their catchments affected by slumps tended to have higher concentrations of contaminants. *Gammarus* in lakes with slumps generally had 1.07 -1.7x higher concentrations of POPs than lakes without slumps. Mean ΣPCB concentrations in *Gammarus* for undisturbed lakes was 16.97ng/g (±1.80), while the mean in disturbed lakes was 27.54 ng/g (±8.08). For aldrins, the mean for disturbed lakes was 12.15 ng/g (±4.84), and 9.59 ng/g (±3.03) in undisturbed lakes. For ΣDDT, the mean in disturbed lakes was 18.47 ng/g and 10.86 ng/g lipid in undisturbed lakes. For ΣHexachlorohexanes, the mean in disturbed lakes was 7.16 ng/g lipids, and 6.69 ng/g lipids in undisturbed lakes. The presence of thaw slumps therefore seems to coincide with higher concentrations of contaminants in *Gammarus.* Slumps therefore appear to be altering the accumulation of contaminants in *Gammarus*.

The accumulation of contaminants by organisms is dependent on several factors related to uptake and elimination. For aquatic organisms, the major routes of chemical uptake are dietary uptake and gill uptake. The major routes of elimination are gill elimination, metabolic transformation and fecal egestion and growth dilution (Arnot and Gobas 2004). If the changes to water chemistry brought about by permafrost slumps alter these uptake and elimination routes, there should be measurable changes of contaminant burdens in amphipods. Water chemistry variables altering these routes of uptake and elimination should also be linked to how concentrations of *Gammarus* changed over the studied lakes.

The effect of decreasing contaminant burdens in biota due to biodilution in a system has been observed in plankton (Taylor et al. 1991; Berglund et al. 2001), invertebrates (Holmqvist et al. 2005) and fish (Taylor et al. 1991; Larsson et al. 1992; Sijm et al. 1992; Kidd et al. 1999; Holmqvist et al. 2005; Evans et al. 2005; Ryan et al. 2013). The dilution hypothesis predicts that concentration of contaminants should be proportional to the reciprocal of biomass (Taylor et al. 1991). The biomass dilution hypothesis was not refuted here (the slopes of regressions between concentrations of POPs in amphipods as dependent variables and benthic invertebrate biomass as independent variables were not significantly different than -1).

Biodilution can occur when growth rates of biota exceed the uptake kinetics of POPs, or when a finite amount of contaminant is diluted over a larger total biomass resulting in less contaminant per unit biomass. If lake productivity shifts and results in changes to biomass, it may therefore alter contaminants per unit biomass. Here, statistical analysis indicated that concentrations of POPs in biota were negatively correlated with lake trophy (total phosphorus). *Gammarus* in more oligotrophic lakes tended to have higher concentrations of POPs. This pattern was also visible for lakes with higher benthic invertebrate biomass and for lakes with higher algal biomass (estimated by the concentration of chlorophyll-*a*). Lakes with higher phosphorus, chlorophyll-*a* and invertebrate biomass also corresponded to those without slumps. In this way, slumps may indirectly alter contaminant concentrations in biota.

A major route of uptake of POPs to aquatic organisms, uptake through gills, is dependent on the freely dissolved portion of contaminants in the water column. Hydrophobic contaminants dissolved in the water become sorbed (absorbed or adsorbed) to dissolved or particulate organic carbon. DOC is normally found to have a negative effect on the uptake of hydrophobic contaminants from water to biota (Black and McCarthy 1988; Kukkonen 1991; Taylor et al. 1991; Haitzer et al. 1998). When large quantities of DOC are present, the fraction of freely dissolved contaminant in the water column, the fraction that can be taken up by the gills, is decreased. DOC is drawn out of the water column in lakes with slumps, likely increasing the portion of freely dissolved contaminant in the water column. Lakes with slumps also had higher concentrations of contaminants in *Gammarus*.

Another major uptake route is uptake through food. It may be altered by the presence of slumps if *Gammarus* food sources are impacted. *Gammarus* are epi-benthic generalist feeders, and may be shredders, grazers and predators (Macneil et al. 1997). Because they are epi-benthic, they spend time in the water column and in sediments and may feed in either area. If concentrations of POPs in plankton are biodiluted because of high plankton biomass, *Gammarus* may be accumulating smaller concentrations of POPs from a planktonic diet.

Gammarus may also feed within the sediments. For this reason, concentrations of POPs in sediments and biota-sediment accumulation factors were calculated to determine if accumulation from the sediments was altered by the presence of thaw slumps. No significant correlations were apparent for surficial sediment concentrations that were not carbon-corrected. For carbon-corrected concentrations there was a very weak negative correlation between total nitrogen in the water column and all contaminants in sediments (this trend was significant for ΣDDT only). Though this trend was very weak over-all, it reoccurred for all contaminant groups. There was also a negative correlation between DOC in the water column and carbon-corrected contaminants in sediments (significant when combining p-values with the Fisher's method). This indicates that, for each gram of surficial sediments, the total amount of contaminants does not change as a function of lake nutrients. However, in lakes with higher total nitrogen, there seems to be slightly less contaminant in the sediments per gram of carbon. Lakes with higher total nitrogen were generally more productive (there was a significant correlation between nitrogen in the water column and chlorophyll-*a*), with higher chlorophyll-*a* and likely higher algal biomass. These also correspond to lakes with higher organic carbon in their sediments and with higher algae settling rates (Deison et al. 2012). Since *Gammarus* may feed on organic particles in the sediments, and if these particles have smaller concentrations of POPs, it's possible that *Gammarus* are accumulating less contaminant from a sedimentary diet in lakes with higher total nitrogen (more productive lakes, without slumps) than in lakes with slumps. This would support the biodilution theory since it would indicate that in more productive lakes with higher algal settling fluxes, the amount of contaminant actually being drawn down into the sediments by algae is lower.

The lack of strong correlations between concentrations of POPs in sediments and any lake physico-chemical parameters may be due to the fact that only shallow sediments were sampled. Shallow sediments were selected to be near amphipod sampling sites, and therefore represent the conditions amphipods were exposed to. However, shallow sediments are also unstable with higher resuspension rates, impacting the transfer of organic pollutants to the benthic food chain (Charles et al. 2005). Depth, lake morphometry and macrophyte biomass will also affect sediment deposition, stability and resuspension (Kemp et al. 1974; Blom et al. 1992; Evans 1994; M.O.E. 1999; Madsen et al. 2001). Because of these factors, it is possible that shallow sediments are simply too unstable to show significant correlations, considering the relatively low POP concentrations detected in all sediments.

### Biota-Sediment Accumulation Factors and Sediments

BSAFs are used to calculate the partitioning of a contaminant between an organism's lipids and the organic carbon fraction of the sediments (Kukkonen et al. 2004). The range found here indicates that the majority of contaminants are being accumulated from sediment. BSAFs were not correlated with slumping in the catchment, though showed slight positive trends with DOC and total nitrogen (significant when combining p-values with the Fisher's method). It therefore appears as though more contaminants are being accumulated from lakes without slumps, even though these were also lakes with lower concentrations of contaminants per gram organic carbon. It's possible that in these lakes, sediments make up a larger portion of the amphipod diet.

*Gammarus* may be exposed to varying amounts of contaminants from the overlying water which is not accounted for in the BSAF calculation. Feeding behaviour will affect uptake of contaminants (Leppänen 1995) but *Gammarus* feeding strategies are highly variable (Macneil et al. 1997; Kelly et al. 2002),and the amount of sediment being ingested as food may vary between communities in different lakes depending on availability of different food sources (Morrison et al. 1996). If *Gammarus* feed mostly from algae in the water column, they would not necessarily reflect concentrations in sediments. Lastly, shallow sediments, such as those sampled here, are generally unstable and are easily resuspended by wind and wave action.

## Conclusion

As permafrost thaw rates continue to rise in the Canadian Arctic, predicting how these changes will affect wildlife is increasingly important. Persistent organic pollutants are of particular interest because they bioaccumulate through the food chain.  This study has shown that a particular form of permafrost thaw, permafrost thaw slumps on lake banks, has impacts on contaminant burdens in biota from Arctic lakes.

Carrie et al. (2010) showed that concentrations of PCBs and mercury in burbot of the Mackenzie River have increased in the last 25 years, despite declining or stable atmospheric concentrations.  This change correlates with warming temperatures and reduced ice cover in the area.  This also correlates with increased permafrost thaw in the Mackenzie Valley and along the Mackenzie River (Aylsworth and Duk-Rodkin 1997).  This study found that for lakes with higher degrees of slumping *Gammarus* tended to have higher concentrations of POPs.

Changes to water chemistry that alter the main routes of uptake and elimination of contaminants will necessarily alter contaminant concentrations in *Gammarus*.   This study found that concentrations of POPs in amphipods were negatively correlated with % catchment thawed.    Contaminant concentration in amphipods was also negatively correlated with benthic invertebrate biomass and with lake trophy, which suggests that biomass dilution may be the main mechanism for the lower contaminant levels in biota of slump lakes.  Carbon-corrected sediment concentrations showed negative trends with Total Nitrogen in the water columns, with BSAFs correlating positively and sediment concentrations (carbon corrected) correlating negatively.  Though weak, these trends may indicate that the changes brought about by slumping are enough to alter sediments, and affect the way contaminants are being taken up.

There are many other types of thaw formations that occur in the Arctic, and they may not all have the same impacts on lakes as those seen here.  However, because the rate and occurrence of permafrost thaw slumps on lake banks on the open tundra is increasing, the effects seen here may become more widespread as the arctic continues to warm.  The decreased lake productivity of lakes as a result of slumping may increase contaminant loads in biota.  However, warming temperatures in the Arctic over this time period are also expected, potentially counter-acting this trend.  Warmer temperatures will lead to longer ice-free seasons, lengthening the growing season for aquatic invertebrates and phytoplankton. This will potentially create more productive lakes, which in turn may decrease concentrations in biota through biodilution. However, this study indicates that lake productivity may play a key role in determining how arctic lakes respond to permafrost thaw and changing climate.

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Table 1: Physical and chemical characteristics of studied lakes, Mackenzie Delta Uplands, NT

| Lake | Catchment Area (ha) | Surface Area (ha) | % Catchment disturbed by slump | Temperature ($^{o}$C) | pH | Specific Conductivityty (uS/cm @ 25 oC) | DOC (mg/L, Unfiltered Water) | TN Dissolved (mg/L) | TP Dissolved (mg/L) | Chl a uncorrected (ug/L, Water) |
|---|---|---|---|---|---|---|---|---|---|---|
| 03b | 15.3 | 4 | 23.6 | 16.9 | 8.1 | 668.5 | 7.9 | 0.347 | 0.0039 | 0.70 |
| 04b | 17.8 | 5 | 13.8 | 9.9 | 7.8 | 925.5 | 9.9 | 0.329 | 0.0032 | 1.20 |
| 05a | 20.9 | 2.9 | 0.0 | 14.5 | 7.4 | 55.3 | 13.0 | 0.361 | 0.0059 | 1.77 |
| 05b | 27.7 | 2.8 | 7.3 | 15.7 | 7.6 | 436.9 | 13.2 | 0.370 | 0.0051 | 0.95 |
| 07a | 18.1 | 1.4 | 0.0 | 13.0 | 6.7 | 46.5 | 21.2 | 0.457 | 0.014 | 4.30 |
| 07b | 34.7 | 3.1 | 3.0 | 15.5 | 7.6 | 266.2 | 14.9 | 0.420 | 0.0069 | 1.37 |
| 08b | 32.7 | 6.5 | 12.1 | 10.8 | 7.8 | 228.6 | 6.45 | 0.392 | 0.0057 | 2.15 |
| 09a | 29.3 | 3.1 | 0.0 | 13.7 | 6.9 | 42.4 | 17.4 | 0.471 | 0.014 | 4.55 |
| 09b | 7.2 | 3.6 | 34.3 | 12.8 | 8.0 | 439.7 | 5.5 | 0.322 | 0.0057 | 1.00 |
| 10b | 23.3 | 11.4 | 30.9 | 10.5 | 7.8 | 812.4 | 2.7 | 0.156 | 0.0036 | 0.88 |
| 11b | 39.4 | 10.5 | 6.4 | 13.3 | 8.2 | 449.8 | 14.2 | 0.393 | 0.008 | 3.40 |
| 14a | 33.5 | 3.4 | 0.0 | 16.2 | 8.2 | 192.9 | 14.7 | 0.687 | 0.012 | 3.30 |
| 16b | 63 | 14.1 | 6.0 | 11.6 | 8.1 | 333.7 | 12.9 | 0.210 | 0.0045 | 1.40 |
| 18b | 4.7 | 2.7 | 12.1 | 10.3 | 8.1 | 300.4 | 5 | 0.230 | 0.0043 | 0.750 |
| 19a | 24.6 | 2.8 | 0.0 | NA | NA | NA | 16.3 | 0.503 | 0.020 | 3.20 |
| 19b | 28.1 | 6.4 | 5.9 | 14.2 | 8.1 | 119 | 11.2 | 0.476 | 0.061 | 14.6 |
| 36b | 24.4 | 3.9 | 19.9 | 16.8 | 8.1 | 787.3 | 12.9 | 0.579 | 0.0071 | 1.80 |

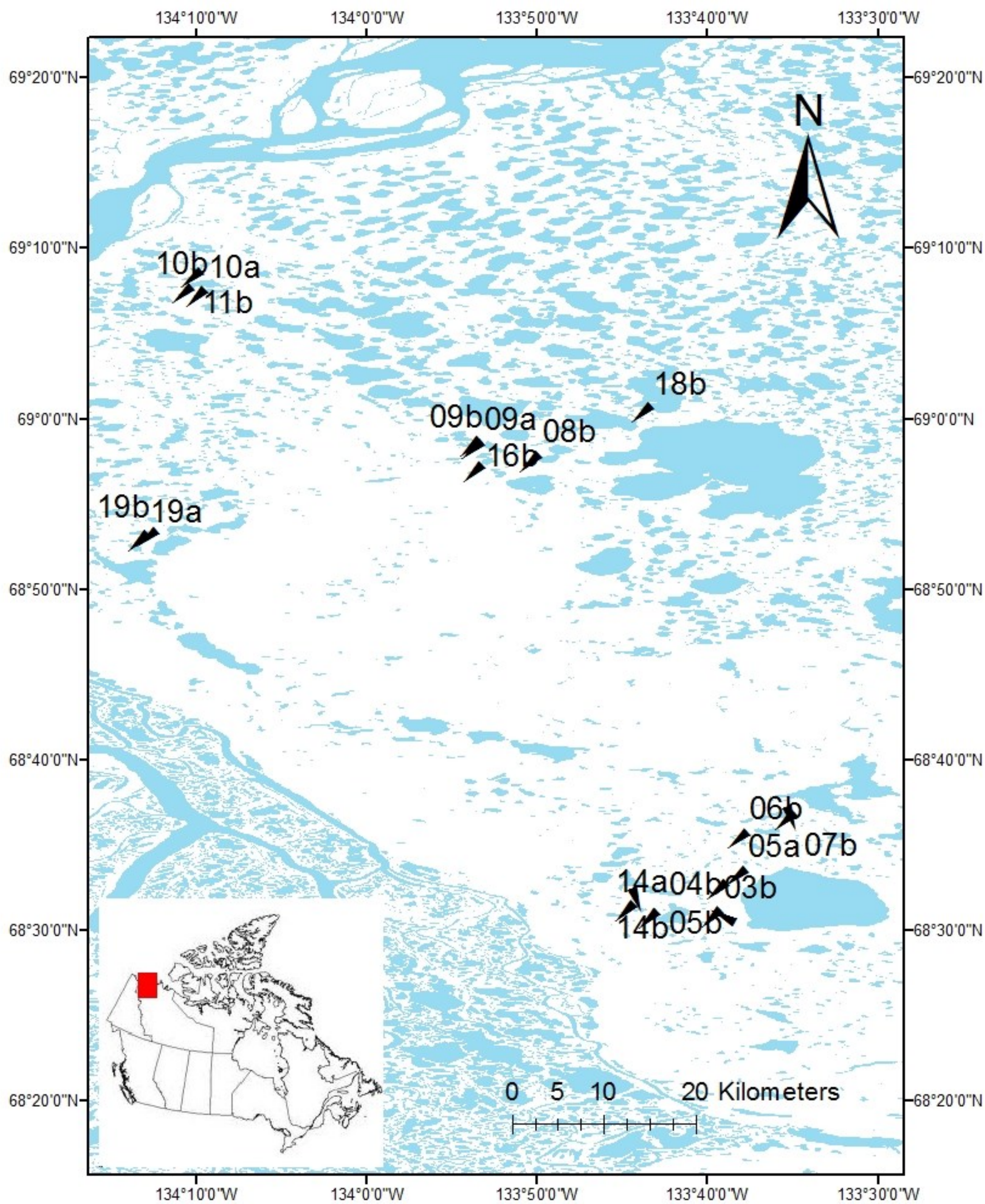


Figure 1: Map of the studied lakes in the upland tundra study region east of the Mackenzie River Delta. Lakes with permafrost slumps are labeled "b" and control lakes are labeled "a"

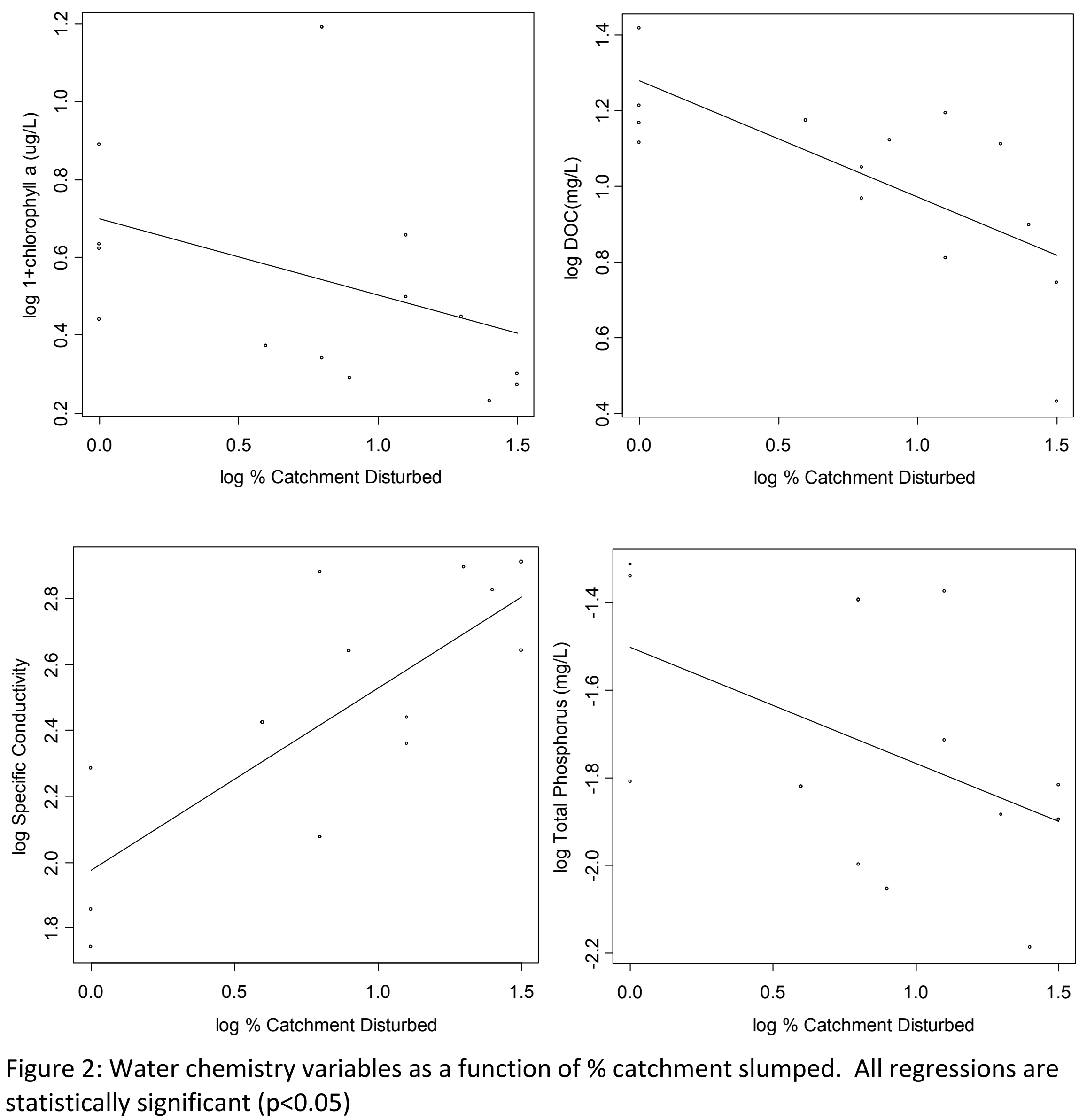


Figure 2: Water chemistry variables as a function of % catchment slumped. All regressions are statistically significant ($p<0.05$)

Table 2: Mean (±SD) concentrations of Σpolychlorinated biphenyls (ΣPCB), Σ hexachlorocyclohexanes (ΣHCH), Σ chordanes (ΣCHL), and ΣAldrin-Dieldrin (ΣAldrins) (ng/g lipid) in biota across all studied lakes.

| | | n | ΣHCH[1] | ΣCHL[2] | ΣDDT[3] | ΣAldrins | ΣCBz[4] | ΣPCB[5] |
|---|---|---|---|---|---|---|---|---|
| *Gammarus* | *Gammarus* | 14 | 6.51 ±1.72 | 4.59 ±3.59 | 16.57 ±7.90 | 11.09 ±5.32 | 9.81 ±3.66 | 24.90 ±8.42 |
| Ninespine stickleback | *Pungitius pungitius* | 4 | 22.01 ±6.15 | 10.88 ±10.37 | 22.74 ±17.69 | 32.816 ±23.00 | 39.54 ±4.36 | 50.04 ± 8.29 |
| Pond smelt | *Hypomesus olidus* | 3 | 33.39 ±24.72 | 75.87 ±51.62 | 31.78 ±26.38 | 84.84 ±57.76 | 41.39 ±24.46 | 88.85 ± 10.32 |

[1] ΣHCH= sum of αHCH, βHCH and γHCH

[2] ΣCHL= sum of CHL-related compounds – heptachlor, heptachlor epoxide, cis-chlordane, trans-nonachlor, oxychlordane, γ-chlordane

[3] ΣDDT=sum of *o-p'*-DDT, *o-p'*-DDT, *p-p'*-DDD, *o-p'*-DDD, *p-p'*-DDE, *o-p'*-DDE

[4] ΣCBz=sum of pentachlorobenzene and hexachlorobenzene

[5] ΣPCB= sum of 18, 29, 31-28, 52, 49, 44, 99, 87, 110, 149, 118, 146, 153, 105, 138, 163, 187, 183,128, 156, 201, 157, 180 and 170

Table 3: Mean (±SD) concentrations of Σpolychlorinated biphenyls (ΣPCB), Σ hexachlorocyclohexanes (ΣHCH), Σ chlordanes (ΣCHL), and ΣAldrin-Dieldrin (ΣAldrins) (ng/g wet weight) in biota across all studied lakes.

| | | n | ΣHCH | ΣCHL | ΣDDT | ΣAldrins | ΣCBz | ΣPCB |
|---|---|---|---|---|---|---|---|---|
| *Gammarus* | *Gammarus* | 14 | 0.10 ±0.02 | 0.09 ±0.08 | 0.23 ±0.12 | 0.12 ±0.06 | 0.11 ±0.09 | 0.45 ±0.34 |
| Ninespine stickleback | *Pungitius pungitius* | 4 | 0.41 ±0.13 | 0.19 ±0.16 | 0.42 ±0.34 | 0.59 ±0.41 | 0.73 ±0.12 | 0.97 ± 0.21 |
| Pond smelt | *Hypomesus olidus* | 3 | 0.46 ±0.27 | 1.11 ±0.40 | 0.50 ±0.28 | 1.81 ±1.96 | 0.60 ±0.19 | 1.25 ± 0.28 |

See table 2 for definition of contaminant abbreviations

Table 4: Mean (±SD) concentrations of Σpolychlorinated biphenyls (ΣPCB), Σ hexachlorocyclohexanes (ΣHCH), Σ chordanes (ΣCHL), and ΣAldrin-Dieldrin (ΣAldrins) in sediment samples across all studied lakes. Concentrations are presented in ng/g dry weight and ng/g carbon

| | n | ΣHCH | ΣCHL | ΣDDT | ΣAldrins | ΣCBz | ΣPCB |
|---|---|---|---|---|---|---|---|
| Sediments (ng/g dry weight) | 16 | 0.28 ±0.182 | 0.13 ±0.116 | 0.12 ±0.0955 | 0.04 ±0.0272 | 0.05 ±0.0338 | 0.44 ±0.32 |
| Sediments (ng/g organic carbon) | 16 | 3.93 ±3.055 | 2.41 ±3.054 | 2.30 ±2.630 | 0.59 ±0.527 | 0.96 ±1.142 | 11.26 ± 9.64 |

See table 2 for definition of contaminant abbreviations

Table 5: Mean (±SD) concentrations of Σpolychlorinated biphenyls (ΣPCB), Σ hexachlorocyclohexanes (ΣHCH), Σ chordanes (ΣCHL), and ΣAldrin-Dieldrin (ΣAldrins) (ng/g wet weight) in Gammarus in control lakes and lakes with permafrost thaw slumps

| | n | ΣHCH | ΣCHL | ΣDDT | ΣAldrins | ΣCBz | ΣPCB |
|---|---|---|---|---|---|---|---|
| Control lakes | 4 | 5.07 ± 0.77 | 2.37 ±1.24 | 13.16 ±7.56 | 6.59 ±4.86 | 9.70 ±4.92 | 16.67 ±1.70 |
| Lakes with slumps | 10 | 6.99 ±1.70 | 5.34 ±3.85 | 17.68 ±5.23 | 12.56 ±4.84 | 9.84 ±3.42 | 26.95 ± 7.93 |

See table 2 for definition of contaminant abbreviations

Table 6: Method detection limits of PCB congeners for biota (data is in ng/g)

| PCB | Amount | PCB | Amount |
|---|---|---|---|
| 18 | 0.36 | 146 | 0.16 |
| 29 | 0.50 | 153 | 0.64 |
| 31-28 | 0.24 | 105 | 0.45 |
| 52 | 0.56 | 138 | 0.26 |
| 49 | 0.21 | 163 | 0.15 |
| 44 | 0.41 | 187 | 1.68 |
| 66 | 0.14 | 183 | 1.55 |
| 101 | 0.78 | 128 | 0.077 |
| 99 | 0.26 | 156 | 0.082 |
| 87 | 0.38 | 201 | 0.061 |
| 110 | 0.17 | 157 | 0.17 |
| 149 | 0.68 | 180 | 0.10 |
| 118 | 0.43 | 170 | 0.077 |

Table 7: Method detection limits of OC Pesticides for biota (data is in ng/g dry weight)

| Pesticides by Groups | Amount |
|---|---|
| Pentachlorobenzen | 0.30 |
| a-HCH | 0.026 |
| Hexachlorobezene | 0.00 |
| b-HCH | 0.00 |
| g-HCH | 0.096 |
| Heptachlor | 6.57 |
| Aldrin | 0.00 |
| Heptachlor Epoxide | 0.00 |
| Oxychlordane | 0.00 |
| g-Chlordane | 0.00 |
| 2,4'-DDE | 0.00 |
| Cis-Chlordane + Endosulfan I | 0.94 |
| Trans-Nonachlor | 0.14 |
| 4,4'-DDE | 0.19 |
| Dieldrin + 2,4'-DDD | 0.38 |
| Endrin | 5.47 |
| Endosulfan II | 0.00 |
| 4,4'-DDD + 4,4'-DDT | 0.28 |
| 2,4'-DDT | 2.33 |
| Mirex | 0.84 |

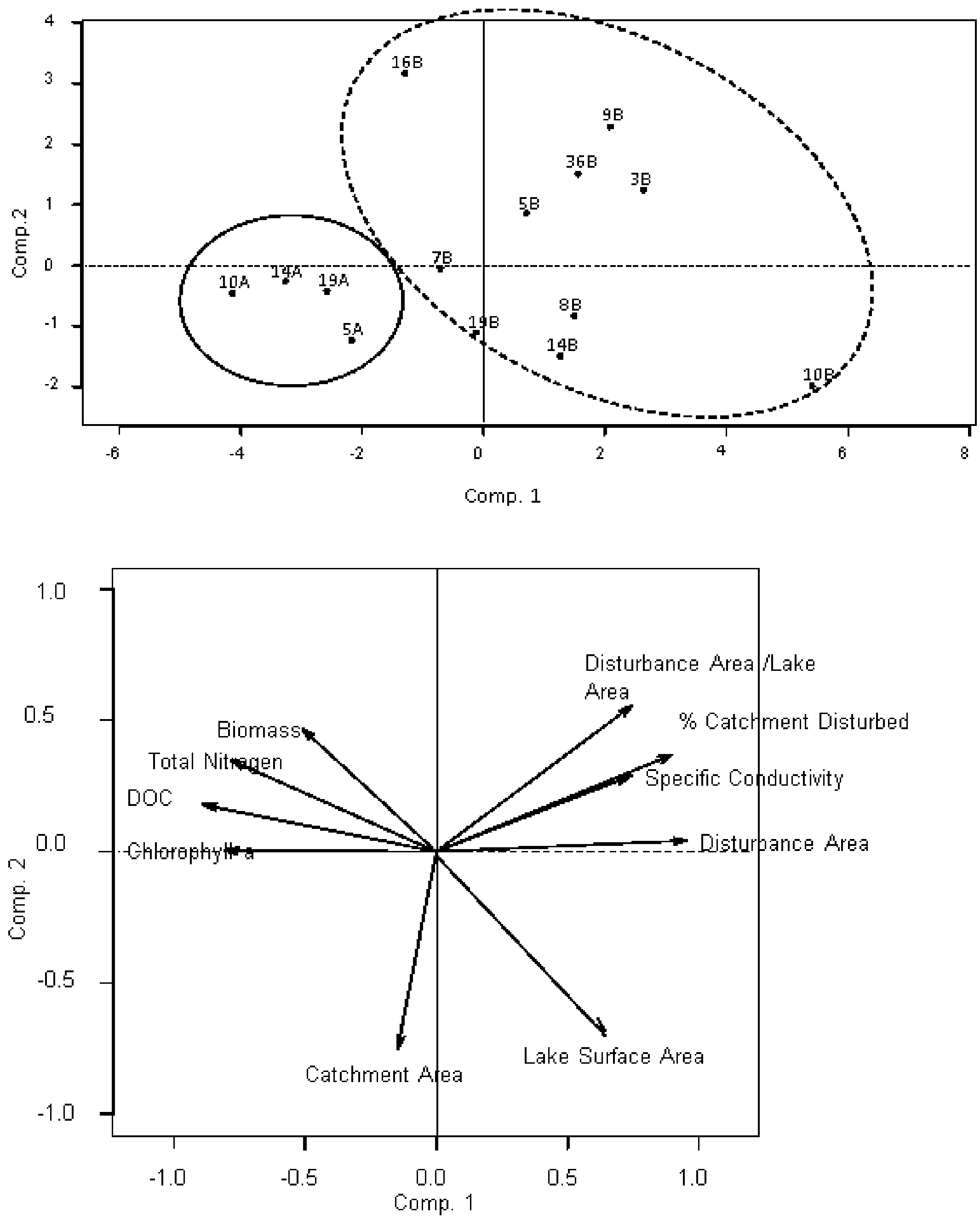


Figure 3: PCA biplot of environmental variables. Variables are log transformed or log(x+1) transformed in the case of disturbace area, % Catchment slumped and disturbance area/lake area. Panel A shows the factor loading plot of environmental variables, panel B shows the lake clusters (the dashed line encompasses lakes with permafrost thaw slumps, and the solid line encompasses lakes without permafrost thaw slumps)

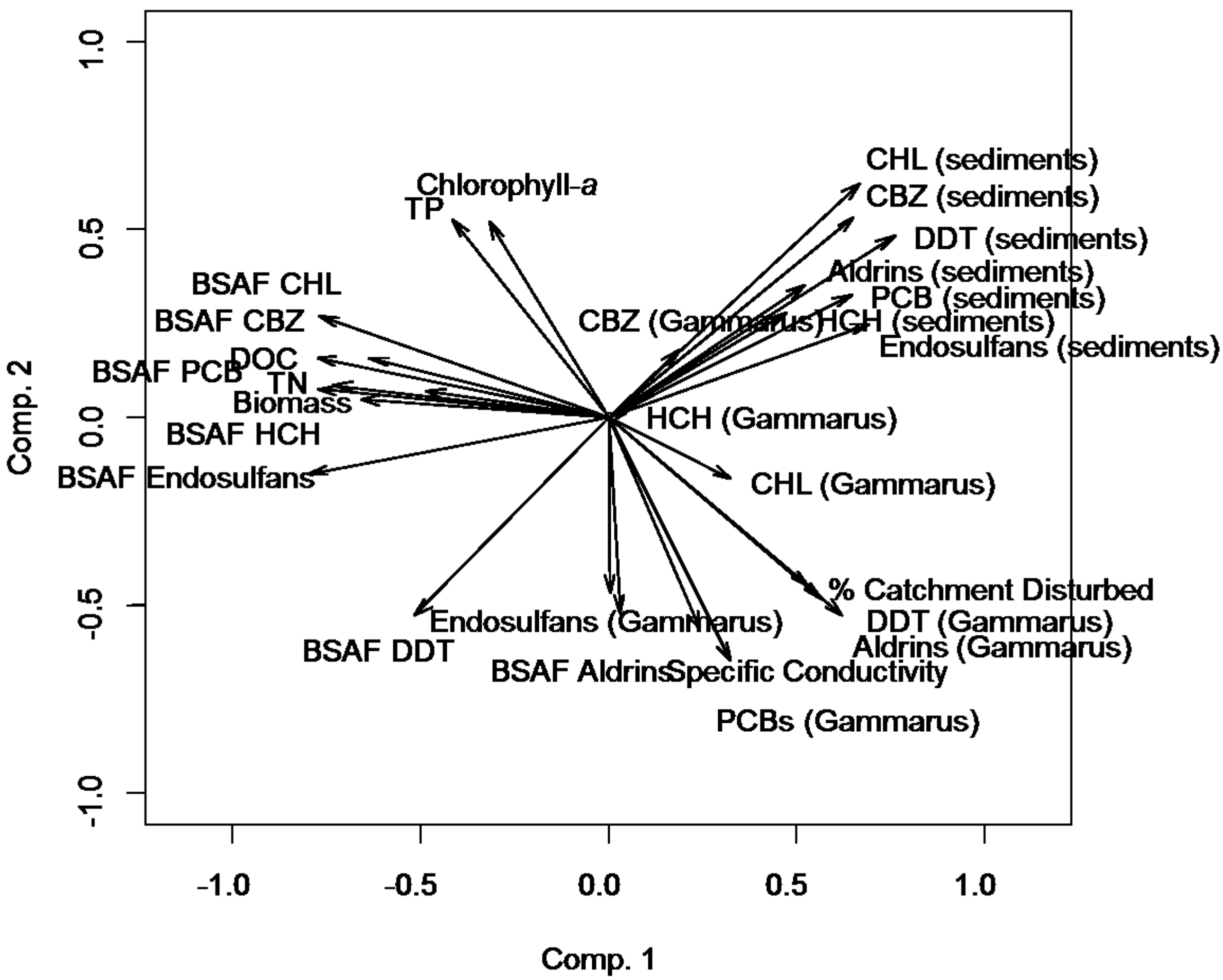


Figure 4: PCA biplot of environmental variables, concentrations of contaminant groups in amphipods (ng/g lipid), concentrations of contaminant groups in sediments (ng/g organic carbon), and of BSAFs. All data was log-transformed prior to analysis. (CHL = chlordanes, CBZ=chlorobenzenes, HCH=Hexachlorocyclohexanes)

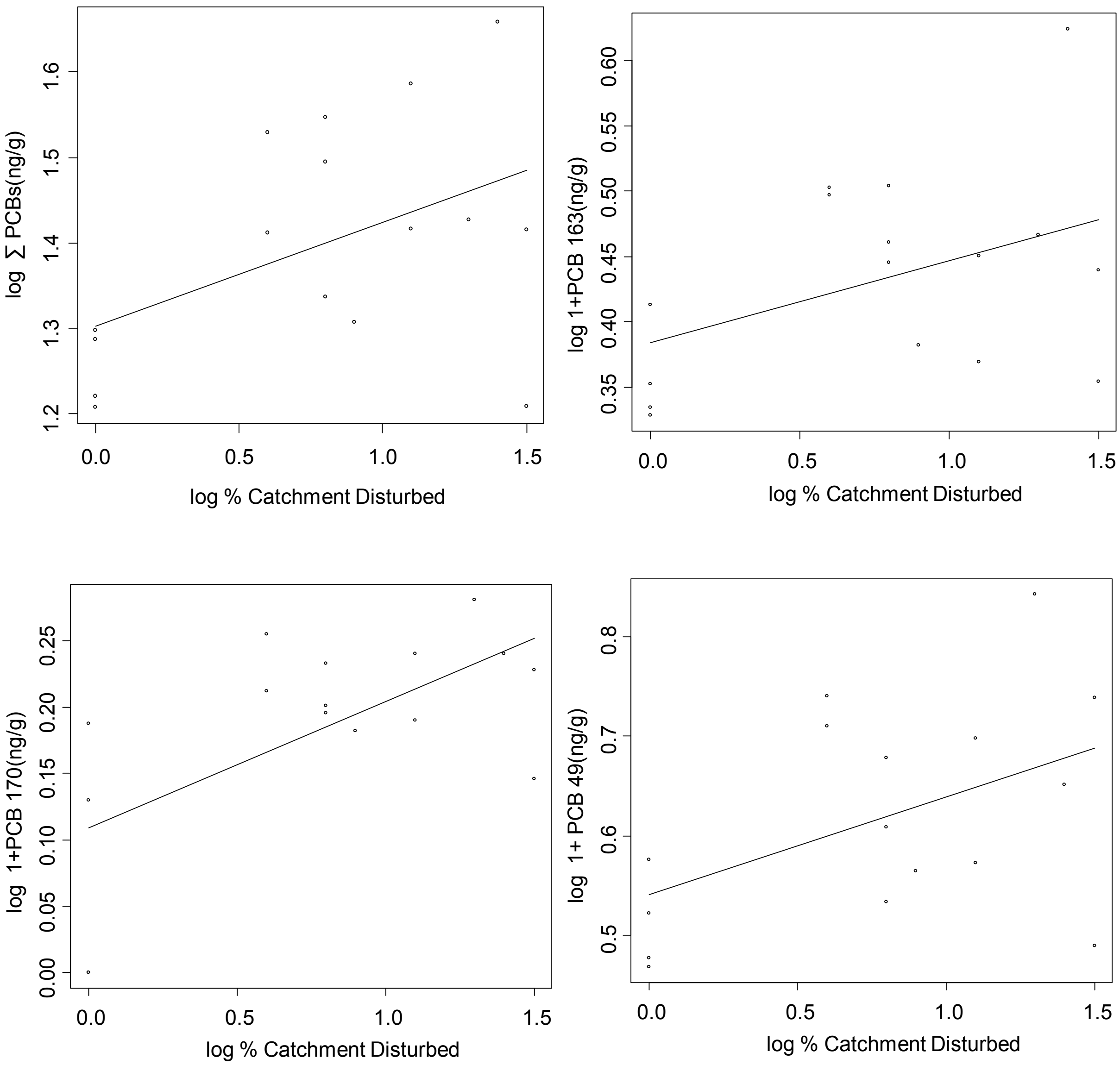


Figure 5: Linear regression analysis relating concentration of ΣPCBs ($p=0.06$) and PCB congeners 163 ($p=0.09$), 170 ($r^2=0.36$, $p=0.008$) and 49 ($p=0.05$) to % of the catchment affected by thaw slumps.

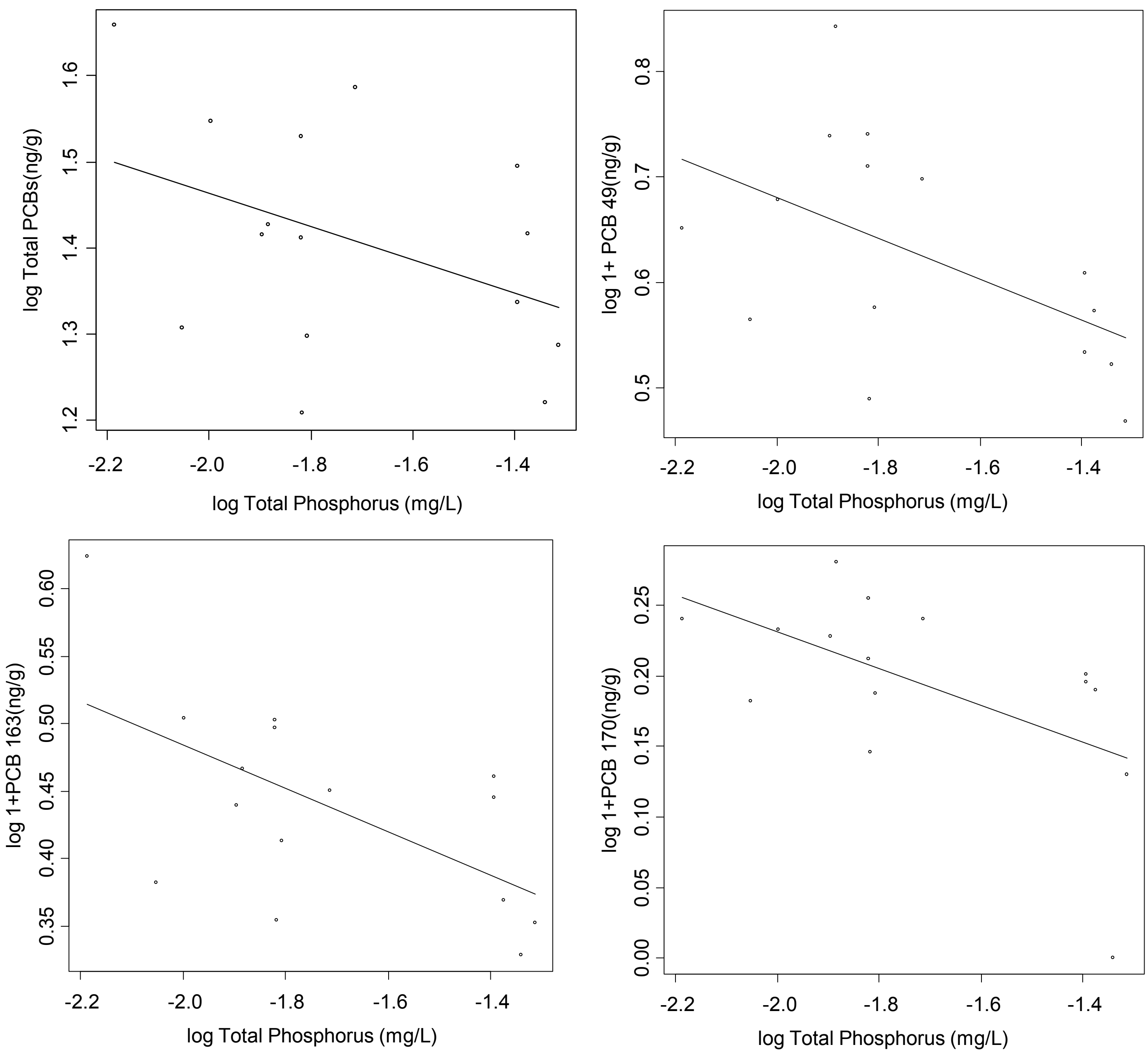


Figure 6: Linear regression analysis relating concentration of ΣPCBs ($p=0.04$), and PCB congeners 163 ($p=0.019$ ), 170 ($r^2=0.36$, $p=0.03$ ), 49 ($p=0.049$) to the concentration of Total Phosphorus (mg/L).

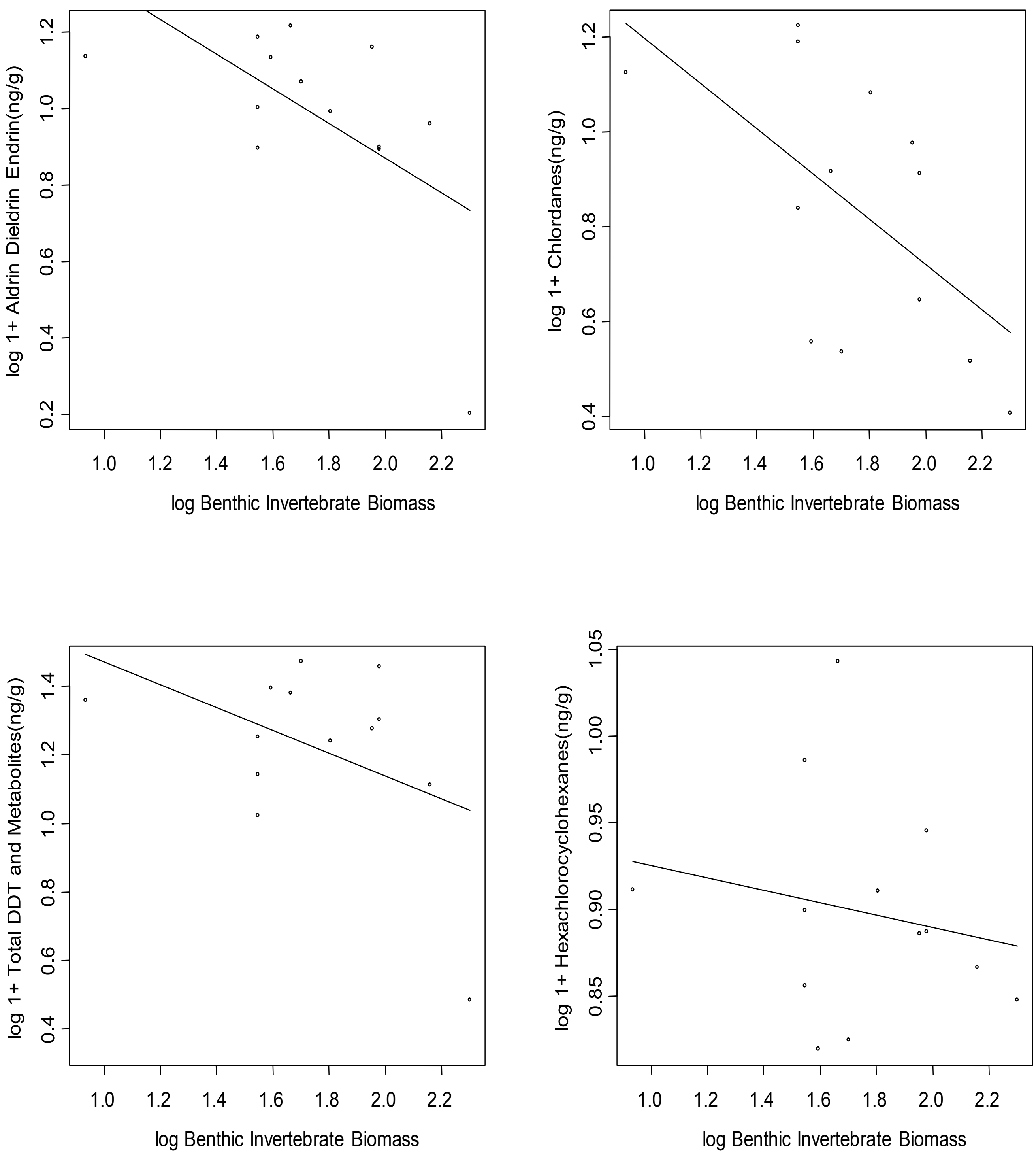


Figure 7: Linear regression analysis relating concentration of ΣAldrins (0.035), ΣChlordanes (p=0.2 ), Σ DDT (p=0.24 ) and Σ HCH (p=0.5) to benthic invertebrate biomass (g/m$^2$).

Table 7: Slopes, Confidence intervals and p-values of RMA regressions with invertebrate biomass as an independent variable and different contaminant groups as dependent variables.

| Contaminant | Slope | 95% Confidence intervals | p-value |
|---|---|---|---|
| PCB | -0.59 | 1.20, -0.082 | 0.10 |
| DDT | -0.89 | -16.94, -0.092 | 0.047 |
| Aldrins | -1.06 | 1.20, -0.082 | 0.0061 |
| Chlordanes | -1.18 | -5.19, -0.50 | 0.013 |
| Chlorobenzenes | 1.37 | -0.0052, -0.15 | 0.31 |
| Hexachlorocyclohexanes | -0.037 | -0.16, 0.087 | 0.24 |
| Endosulfans | -2.20 | 0.12, -0.071 | 0.40 |

Table 8: Correlations and p-values of concentrations of contaminants in sediments (ng/g organic carbon)and different physical and chemical parameters of lakes. All data was log-transformed prior to analysis.

| | chlorophyll-*a* | | % Catchment disturbed | | DOC | | Specific Conductivity | | Total Nitrogen | | Total Phosphorus | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | R | p-value | R | p-value | R | p-value | R | p-value | R | p-value | R | p-value |
| PCB | -0.02 | 0.95 | 0.35 | 0.20 | -0.31 | 0.26 | 0.13 | 0.64 | -0.41 | 0.13 | -0.23 | 0.41 |
| DDT | 0.08 | 0.78 | 0.17 | 0.55 | -0.33 | 0.24 | 0.01 | 0.98 | -0.54 | 0.04 | -0.03 | 0.90 |
| CHL | 0.37 | 0.17 | 0.01 | 0.96 | -0.41 | 0.13 | -0.24 | 0.40 | -0.38 | 0.16 | 0.04 | 0.88 |
| CBZ | 0.02 | 0.93 | 0.24 | 0.38 | -0.49 | 0.07 | -0.03 | 0.91 | -0.49 | 0.07 | -0.03 | 0.91 |
| Endosulfans | -0.34 | 0.21 | -0.06 | 0.83 | -0.41 | 0.13 | -0.05 | 0.00 | -0.69 | 0.33 | -0.27 | 0.33 |
| Aldrin | 0.35 | 0.20 | 0.13 | 0.65 | -0.30 | 0.28 | 0.04 | 0.89 | -0.34 | 0.21 | 0.00 | 0.99 |
| HCH | -0.10 | 0.71 | 0.36 | 0.17 | -0.31 | 0.25 | 0.09 | 0.73 | -0.42 | 0.11 | -0.10 | 0.70 |

Table 9: Correlations and p-values of concentrations of contaminants in sediments (ng/g dry weight)and different physical and chemical parameters of lakes. All data was log-transformed prior to analysis.

| | chlorophyll-*a* | | % Catchment disturbed | | DOC | | Specific Conductivity | | Total Nitrogen | | Total Phosphorus | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | R | p-value | R | p-value | R | p-value | R | p-value | R | p-value | R | p-value |
| PCB | 0.27 | 0.32 | 0.09 | 0.74 | 0.26 | 0.33 | 0.1 | 0.73 | 0.26 | 0.33 | -0.07 | 0.80 |
| DDT | 0.29 | 0.28 | 0.02 | 0.93 | -0.04 | 0.87 | -0.07 | 0.80 | -0.07 | 0.80 | 0.01 | 0.97 |
| CHL | -0.22 | 0.41 | 0.19 | 0.49 | -0.42 | 0.10 | 0 | 0.99 | -0.53 | 0.034 | -0.3 | 0.26 |
| CBZ | 0.64 | 0.0082 | -0.07 | 0.78 | -0.11 | 0.68 | -0.3 | 0.27 | -0.03 | 0.92 | 0.19 | 0.48 |
| Endosulfans | 0.52 | 0.041 | -0.08 | 0.77 | -0.25 | 0.35 | -0.3 | 0.26 | -0.18 | 0.51 | 0.12 | 0.65 |
| Aldrin | 0.22 | 0.41 | 0 | 0.99 | -0.07 | 0.79 | 0.15 | 0.59 | 0.01 | 0.98 | -0.09 | 0.75 |
| HCH | -0.18 | 0.51 | 0.29 | 0.27 | -0.05 | 0.87 | 0.18 | 0.75 | 0.09 | 0.35 | -0.25 | 0.35 |

Table 10: Correlations and p-values of concentrations of Biota-Sediment Accumulation Factors (BSAFs) and different physical and chemical parameters of lakes. All data was log-transformed prior to analysis.

| | chlorophyll-*a* | | % Catchment disturbed | | DOC | | Total Nitrogen | | Total Phosphorus | |
|---|---|---|---|---|---|---|---|---|---|---|
| BSAFs | R | p-value | R | p-value | R | p-value | R | p-value | R | p-value |
| PCB | 0.52 | 0.23 | -0.5 | 0.43 | 0.36 | 0.43 | 0.54 | 0.21 | 0.54 | 0.21 |
| DDT | -0.25 | 0.59 | 0.26 | 0.38 | 0.39 | 0.38 | 0.43 | 0.34 | -0.48 | 0.27 |
| CHL | 0.71 | 0.076 | -0.46 | 0.30 | 0.58 | 0.18 | 0.73 | 0.060 | 0.69 | 0.086 |
| CBZ | 0.66 | 0.13 | -0.39 | 0.39 | 0.61 | 0.15 | 0.75 | 0.051 | 0.61 | 0.15 |
| Endosulfans | 0.4 | 0.37 | -0.11 | 0.82 | 0.63 | 0.13 | 0.82 | 0.023 | 0.24 | 0.60 |
| Aldrin | -0.25 | 0.59 | 0.31 | 0.50 | 0.18 | 0.69 | 0.09 | 0.84 | -0.53 | 0.22 |
| HCH | 0.73 | 0.061 | 0.36 | 0.34 | 0.08 | 0.87 | 0.36 | 0.43 | 0.75 | 0.050 |

## General Conclusions

There is evidence of permafrost warming and degradation throughout high latitudes of North America (Lemke et al. 2007). The results of such wide-spread permafrost degradation are likely to affect the flora and fauna in all arctic habitats, along with human infrastructure and communities. The study of the effect of permafrost thaw slumps on aquatic communities allows researchers to examine how thawed permafrost material affects lakes, while simultaneously allowing for comparison with lakes that are not affected by thaw. Results presented in chapters one and two allow to better quantify and understand ecological impacts associated with permafrost thaw slumping.

Retrogressive permafrost thaw slumps, which are increasing in frequency in the Arctic in a warmer climate, have an indirect effect on benthic invertebrate assemblages in small lakes, and on the concentrations of contaminants in the amphipod *Gammarus*. Most changes are related to the impact of permafrost thaw slumps on nutrients in the water column, which in turn lead to altered productivity. Water chemistry concetration presented here corroborates past research. Total nitrogen, total phosphorus and chlorophyll-*a* were lowered by thaw slump material entering lake water. In contrast, specific conductivity and certain major ions (Ca, Mg, Na, K, $SO_4^{2-}$, $CO_3^{2-}$, and $HCO_3^-$) increased in lakes with permafrost thaw slumps influenced by the input of ion-rich clays from slump material. DOC and humic substances in the water column are lower in lakes with slumps, which was likely a result of increased specific conductivity and ionic content of water, causing these substances to flocculate out to the sediments (Thompson et al. 2008). These changes lead do decreased algal productivity and lower chlorophyll-*a* in lake water. The effects of thaw slumps on lakes were also found to impact the benthic communities of lakes.

I showed how benthic invertebrate density and biomasses, and the size-structure of benthic communities, are changed following thaw slump formation (Chapter 1). Average density and average biomass per lake tended to decrease with decreased nutrient level of lakes, and lakes which had higher levels of chlorophyll-*a*, total dissolved nitrogen and total phosphorus generally had higher average benthic invertebrate density and biomass. Average density and average biomass per lake were not directly correlated with % catchment disturbed, but were significantly related to the size of the permafrost thaw slump. Analysis of the constructed size spectrum suggests that the concentration of nitrogen in the water column has an over-all positive effect on the biomass of invertebrates in most size classes, though the increase was less significant for the biomass of larger size classes.

The size-spectrum also suggests that slumping has an over-all negative effect on the invertebrate community size structure, though different size classes responded differently to the presence of slumps: the density of small invertebrates is reduced in lakes with large disturbances, but this reduction is less important for large invertebrates. This differential response by small and large invertebrates to slumping and nitrogen concentrations could be due to increased macrophyte biomass in slumped lakes. Nitrogen tends to increase invertebrate density and biomass of invertebrates in smaller size classes. For medium –large size classes, nitrogen had a negative effect on invertebrate density, whereas slump size had a positive effect on invertebrate density. The positive effect due to slumping on larger size classes is possibly linked to the higher biomass of macrophytes in lakes with thaw slumps (Mesquita et al. 2010). The largest invertebrates collected (snails and amphipods) are also those that tend to prefer macrophyte habitats. The spill-over of these larger invertebrates from macrophytes onto open sediments might therefore account for the higher abundance of large size classes in lakes with slumps and low total nitrogen. There might also be migration from the open sediments to macrophytes of some of the smaller invertebrates, potentially explaining the negative impact of thaw slumps on benthic invertebrate biomass of small invertebrates.

In Chapter 2, I showed that the presence of permafrost thaw slumps on lake banks is linked to increased contaminant burdens in *Gammarus*. Lakes with slumps tended to have higher chlorophyll a and invertebrate biomass (as established in Chapter 2). Here, I found that lakes without slumps (those with higher productivity and invertebrate biomass) tended to have lower concentrations of contaminants in the amphipod *Gammarus* (1.07 – 1.7x higher concentrations in lakes without thaw slumps). This is likely due to biomass dilution, as contaminant burdens per unit biomass are decreased when higher biomass is present. There were no such trends for fish, though very few lakes contained fish and no definitive conclusions could be drawn.

Concentrations of carbon corrected contaminants in surficial sediments along lake banks were negatively correlated to Nitrogen in lake water when combining p-values with Fisher's method. Though it was already known that sediments in lakes affected by thaw slumps are physically altered by slump material (Deison et al. 2012), this study may serve as confirmation that slumps alter lakes sufficiently to impact contaminant uptake and elimination in amphipods. Likewise, Biota-sediment accumulation factors showed positive trends with slumping, possibly indicating that the composition of sediments is being altered by thaw slumps, and decreasing the uptake of contaminants from sediments.

Further research is also needed to assess the impact of permafrost thaw slumps on biota. Benthic invertebrates should be sampled from different habitat types in lakes. Because macrophytes may be more abundant in lakes with slumps, sampling benthic invertebrates from macrophytes may give a more complete picture of how benthic communities are changing in the entire lake. If macrophytes are increasing in abundance in lakes with permafrost thaw slumps, it would also be interesting to measure contaminant loads in macrophytes. This would allow us to assess if the increasing macrophyte biomass in lakes with slumps is taking a significant amount of contaminant from the water column. If possible, it would also be interesting to widen the data set and sample a larger selection of lakes. While 16 lakes can indicate some trends, having more would increase the power of the analysis and likely allow us to detect stronger correlations.

Results of this thesis used in conjunction with other work being done currently on lakes affected by permafrost thaw slumping will no doubt be a useful tool for assessing how aquatic communities will change with a warming Arctic, and how contaminant uptake and concentrations in biota are likely to change. Other research currently being conducted on these lakes will help give a clearer picture of how permafrost thaw slumps are affecting different components of the system. Sediment cores from these lakes are currently being analyzed and are indicating that algal scavenging is not increasing contaminant loads in lakes because over-all fluxes of contaminants into slumped and control lakes were not significantly different even though algal productivity was higher in lakes without slumps (Eickmeyer, unpublished). This work also indicates that over-all, concentrations of TOC normalized contaminants were higher in sediments of lakes with slumps, though this is likely because there is simply less organic carbon in sediments of lakes with slumps. Studies are also being conducted on concentrations of contaminants in lake water (Houben, unpublished). Once we have data for all different components of the system, further analysis and modelling of these lakes will be possible.

Though this thesis studied only a specific kind of permafrost thaw, the results of this thesis demonstrate how changing arctic climate has the potential to significantly alter different aspects of freshwater systems in the Arctic. Not all permafrost thaw will occur in the same type of permafrost terrain, and therefore may have different impacts on lake chemistry and biota. However, these results shed light on the potential wide-spread impact of permafrost thaw on freshwater systems.

## Appendix A – Benthic invertebrate identification (raw data)

| Lake | Family | Code | Individuals | Density(Indp/m2) | BugDM(ug) |
|---|---|---|---|---|---|
| 03B | Amphipoda | AMPHI | 112 | 4848.484848 | 37516.514 |
| 03B | Bivalvia | PELY | 96 | 4155.844156 | 1632.61133 |
| 03B | Candonidae | O_Can | 40 | 1731.601732 | 44.5570437 |
| 03B | Ceratopogonidae | CERAT | 24 | 1038.961039 | 580.925176 |
| 03B | Chironominae | CHIR | 232 | 10043.29004 | 5505.48934 |
| 03B | Cyprididae | O_Cypr | 8 | 346.3203463 | 13.9997144 |
| 03B | Diptera | D_pupa | 8 | 346.3203463 | 174.218336 |
| 03B | Hirudinea | HIRU | 8 | 346.3203463 | 120.799948 |
| 03B | Ilyocyprididae | O_Ilyo | 24 | 1038.961039 | 62.9360965 |
| 03B | Limnocytheridae | O_Limn | 8 | 346.3203463 | 24.0575233 |
| 03B | Nematoda | NEMA | 304 | 13160.17316 | 1698.66267 |
| 03B | Odontoceridae | T_Odon | 8 | 346.3203463 | 1534.62409 |
| 03B | Oligochaeta | OLIG | 128 | 5541.125541 | 67233.7358 |
| 03B | Oribatei | ORIB | 8 | 346.3203463 | 36.2896724 |
| 04B | Acari | ACARI | 8 | 346.3203463 | 18.0432115 |
| 04B | Amphipoda | AMPHI | 24 | 1038.961039 | 36569.7232 |
| 04B | Bivalvia | PELY | 264 | 11428.57143 | 2421.03621 |
| 04B | Candonidae | O_Can | 432 | 18701.2987 | 1001.02035 |
| 04B | Chironominae | CHIR | 528 | 22857.14286 | 3986.43078 |
| 04B | Copepod | C_Cant | 32 | 1385.281385 | 56.4201278 |
| 04B | Cyprididae | O_Cypr | 88 | 3809.52381 | 111.646884 |
| 04B | Cytheridae | O_Cyth | 8 | 346.3203463 | 20.5915585 |
| 04B | Gastropoda | GAST | 144 | 6233.766234 | 7895.69048 |
| 04B | Invertebrate | HYGR | 8 | 346.3203463 | 0.12913306 |
| 04B | Molanna | T_Mol | 16 | 692.6406926 | 3057.5324 |
| 04B | Nematoda | NEMA | 192 | 8311.688312 | 558.488723 |
| 04B | Oligochaeta | OLIG | 200 | 8658.008658 | 23886.629 |

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|---|---|---|---|---|---|
| 04B | Oribatei | ORIB | 40 | 1731.601732 | 123.625583 |
| 05A | Amphipoda | AMPHI | 18 | 779.2207792 | 1634.69146 |
| 05A | Bivalvia | PELY | 24 | 1038.961039 | 1637.41016 |
| 05A | Candonidae | O_Can | 153 | 6623.376623 | 721.351768 |
| 05A | Chironominae | CHIR | 100 | 4329.004329 | 3407.31398 |
| 05A | Chrysomelidae | C_Chry | 6 | 259.7402597 | 341672.57 |
| 05A | Cyprididae | O_Cypr | 47 | 2034.632035 | 322.923183 |
| 05A | Diptera | D_pupa | 9 | 389.6103896 | 1051.65288 |
| 05A | Invertebrate | HYDA | 9 | 389.6103896 | 0.60392894 |
| 05A | Invertebrate | HYGR | 2 | 86.58008658 | 0.33523324 |
| 05A | Lepidostomatidae | T_Lepi | 1 | 43.29004329 | 5432.46838 |
| 05A | Nematoda | NEMA | 39 | 1688.311688 | 6034.82592 |
| 05A | Oligochaeta | OLIG | 195 | 8441.558442 | 42342.2005 |
| 05A | Oribatei | ORIB | 124 | 5367.965368 | 1420.70912 |
| 05B | Amphipoda | AMPHI | 16 | 692.6406926 | 4308.69107 |
| 05B | Bivalvia | PELY | 696 | 30129.87013 | 10051.9724 |
| 05B | Candonidae | O_Can | 400 | 17316.01732 | 510.307452 |
| 05B | Ceratopogonidae | CERAT | 32 | 1385.281385 | 574.801387 |
| 05B | Chironominae | CHIR | 176 | 7619.047619 | 1396.38482 |
| 05B | Cyprididae | O_Cypr | 128 | 5541.125541 | 122.323672 |
| 05B | Gastropoda | GAST | 120 | 5194.805195 | 1260.78597 |
| 05B | Nematoda | NEMA | 80 | 3463.203463 | 1030.04808 |
| 05B | Oligochaeta | OLIG | 960 | 41558.44156 | 49853.2989 |
| 05B | Oribatei | ORIB | 80 | 3463.203463 | 94.3227137 |
| 07A | Bivalvia | PELY | 536 | 23203.4632 | 24820.3148 |
| 07A | Candonidae | O_Can | 56 | 2424.242424 | 88.3872793 |
| 07A | Chironominae | CHIR | 440 | 19047.61905 | 3187.23878 |
| 07A | Copepod | C_Cant | 48 | 2077.922078 | 97.9449086 |
| 07A | Cyprididae | O_Cypr | 656 | 28398.2684 | 536.846947 |
| 07A | Halacaridae | HALA | 64 | 2770.562771 | 53.8327076 |

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|---|---|---|---|---|---|
| 07A | Nematoda | NEMA | 8 | 346.3203463 | 4.8531443 |
| 07A | Oligochaeta | OLIG | 248 | 10735.93074 | 1206.85508 |
| 07A | Oribatei | ORIB | 416 | 18008.65801 | 365.740271 |
| 07B | Amphipoda | AMPHI | 72 | 3116.883117 | 140905.247 |
| 07B | Bivalvia | PELY | 44 | 1904.761905 | 11047.2264 |
| 07B | Chironominae | CHIR | 147 | 6363.636364 | 2633.73565 |
| 07B | Copepod | C_Cant | 10 | 432.9004329 | 1.37785936 |
| 07B | Cyprididae | O_Cypr | 66 | 2857.142857 | 30.7243103 |
| 07B | Diptera | D_pupa | 59 | 2554.112554 | 1326.96949 |
| 07B | Gastropoda | GAST | 65 | 2813.852814 | 17066.6093 |
| 07B | Haliplidae | C_Hali | 8 | 346.3203463 | 920.316221 |
| 07B | Invertebrate | HYGR | 25 | 1082.251082 | 7.44859327 |
| 07B | Nematoda | NEMA | 31 | 1341.991342 | 784.544812 |
| 07B | Oligochaeta | OLIG | 138 | 5974.025974 | 15320.3875 |
| 07B | Oribatei | ORIB | 410 | 17748.91775 | 320.133439 |
| 07B | Phryganeidae | T_Phry | 16 | 692.6406926 | 2904.14044 |
| 07B | Tardigrada | TARDI | 64 | 2770.562771 | 11.4399165 |
| 08B | Amphipoda | AMPHI | 64 | 2770.562771 | 8917.4932 |
| 08B | Bivalvia | PELY | 48 | 2077.922078 | 194.308713 |
| 08B | Candonidae | O_Can | 912 | 39480.51948 | 964.657086 |
| 08B | Chironominae | CHIR | 416 | 18008.65801 | 7213.10201 |
| 08B | Copepod | C_Cant | 32 | 1385.281385 | 2.31967453 |
| 08B | Cyprididae | O_Cypr | 240 | 10389.61039 | 184.828164 |
| 08B | Gastropoda | GAST | 112 | 4848.484848 | 958.312368 |
| 08B | Invertebrate | HYGR | 32 | 1385.281385 | 0.81623511 |
| 08B | Nematoda | NEMA | 232 | 10043.29004 | 789.914981 |
| 08B | Oligochaeta | OLIG | 168 | 7272.727273 | 507.590335 |
| 08B | Oribatei | ORIB | 96 | 4155.844156 | 87.4933052 |
| 09A | Bivalvia | PELY | 110 | 4761.904762 | 27038.1813 |
| 09A | Candonidae | O_Can | 16 | 692.6406926 | 18.1958101 |

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|---|---|---|---|---|---|
| 09A | Chironominae | CHIR | 758 | 32813.85281 | 53547.644 |
| 09A | Copepod | C_Cant | 128 | 5541.125541 | 26.7718112 |
| 09A | Cyprididae | O_Cypr | 2234 | 96709.95671 | 946.49809 |
| 09A | Diptera | D_pupa | 14 | 606.0606061 | 5489.46585 |
| 09A | Limniphilidae | T_Limn | 4 | 173.1601732 | 11081.1104 |
| 09A | Nematoda | NEMA | 988 | 42770.56277 | 726.11919 |
| 09A | Oligochaeta | OLIG | 40 | 1731.601732 | 420.058855 |
| 09A | Oribatei | ORIB | 1924 | 83290.04329 | 1061.91315 |
| 09B | Amphipoda | AMPHI | 32 | 1385.281385 | 15171.1764 |
| 09B | Bivalvia | PELY | 6680 | 289177.4892 | 44783.3191 |
| 09B | Candonidae | O_Can | 1056 | 45714.28571 | 1872.18268 |
| 09B | Ceratopogonidae | CERAT | 104 | 4502.164502 | 1546.98148 |
| 09B | Chironominae | CHIR | 2112 | 91428.57143 | 44990.5828 |
| 09B | Copepod | C_Cant | 80 | 3463.203463 | 17.9664713 |
| 09B | Cyprididae | O_Cypr | 296 | 12813.85281 | 301.708305 |
| 09B | Diptera | D_pupa | 24 | 1038.961039 | 78.6800241 |
| 09B | Gastropoda | GAST | 792 | 34285.71429 | 11476.0068 |
| 09B | Invertebrate | HYGR | 16 | 692.6406926 | 0.11135698 |
| 09B | Leptoceridae | T_Lepto | 16 | 692.6406926 | 176.553862 |
| 09B | Limnocytheridae | O_Limn | 80 | 3463.203463 | 89.1950302 |
| 09B | Nematoda | NEMA | 2440 | 105627.7056 | 3212.39033 |
| 09B | Oligochaeta | OLIG | 120 | 5194.805195 | 11057.9797 |
| 09B | Oribatei | ORIB | 24 | 1038.961039 | 24.1600061 |
| 09B | Tardigrada | TARDI | 128 | 5541.125541 | 8.53502536 |
| 10B | Bivalvia | PELY | 2 | 86.58008658 | 240.153818 |
| 10B | Candonidae | O_Can | 48 | 2077.922078 | 95.1583334 |
| 10B | Chironominae | CHIR | 72 | 3116.883117 | 849.319212 |
| 10B | Invertebrate | HYDA | 8 | 346.3203463 | 0.06387768 |
| 10B | Nematoda | NEMA | 184 | 7965.367965 | 1123.7385 |
| 10B | Oligochaeta | OLIG | 104 | 4502.164502 | 6190.44598 |

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|---|---|---|---|---|---|
| 10B | Oribatei | ORIB | 24 | 1038.961039 | 120.080458 |
| 11B | Acari | ACARI | 16 | 692.6406926 | 163.633831 |
| 11B | Amphipoda | AMPHI | 2 | 86.58008658 | 14.2063454 |
| 11B | Bivalvia | PELY | 691 | 29913.41991 | 46325.2731 |
| 11B | Candonidae | O_Can | 1439 | 62294.37229 | 23446.9542 |
| 11B | Ceratopogonidae | CERAT | 2 | 86.58008658 | 470.78057 |
| 11B | Chironominae | CHIR | 124 | 5367.965368 | 20208.1175 |
| 11B | Copepod | C_Cant | 3 | 129.8701299 | 12.3069996 |
| 11B | Cyprididae | O_Cypr | 400 | 17316.01732 | 6062.12046 |
| 11B | Cytheridae | O_Cyth | 11 | 476.1904762 | 215.994431 |
| 11B | Diptera | D_pupa | 6 | 259.7402597 | 4962.23709 |
| 11B | Gastropoda | GAST | 751 | 32510.82251 | 1115328.66 |
| 11B | Haliplidae | C_Hali | 1 | 43.29004329 | 8180.8091 |
| 11B | Ilyocyprididae | O_Ilyo | 558 | 24155.84416 | 6613.94622 |
| 11B | Invertebrate | HYGR | 2 | 86.58008658 | 0.07996261 |
| 11B | Limnocytheridae | O_Limn | 330 | 14285.71429 | 3237.69986 |
| 11B | Nematoda | NEMA | 84 | 3636.363636 | 5724.32553 |
| 11B | Oligochaeta | OLIG | 151 | 6536.796537 | 46226.655 |
| 11B | Oribatei | ORIB | 9 | 389.6103896 | 92.9529499 |
| 14A | Amphipoda | AMPHI | 152 | 6580.08658 | 26582.6099 |
| 14A | Bivalvia | PELY | 480 | 20779.22078 | 9086.22433 |
| 14A | Candonidae | O_Can | 3248 | 140606.0606 | 1440.71745 |
| 14A | Ceratopogonidae | CERAT | 48 | 2077.922078 | 462.108116 |
| 14A | Chironominae | CHIR | 2264 | 98008.65801 | 8101.88981 |
| 14A | Copepod | C_Cant | 984 | 42597.4026 | 282.139971 |
| 14A | Cyprididae | O_Cypr | 432 | 18701.2987 | 108.691235 |
| 14A | Diptera | D_pupa | 152 | 6580.08658 | 164.761742 |
| 14A | Gastropoda | GAST | 328 | 14199.1342 | 4444.77414 |
| 14A | Hirudinea | HIRU | 24 | 1038.961039 | 18728.5086 |
| 14A | Invertebrate | HYDA | 96 | 4155.844156 | 3.07925644 |

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|---|---|---|---|---|---|
| 14A | Invertebrate | HYGR | 272 | 11774.89177 | 7.72271853 |
| 14A | Nematoda | NEMA | 656 | 28398.2684 | 1122.86185 |
| 14A | Oligochaeta | OLIG | 864 | 37402.5974 | 1719.66838 |
| 14A | Oribatei | ORIB | 472 | 20432.90043 | 440.814252 |
| 16B | Candonidae | O_Can | 16 | 692.6406926 | 37.7327717 |
| 16B | Ceratopogonidae | CERAT | 16 | 692.6406926 | 31.6850377 |
| 16B | Chironominae | CHIR | 56 | 2424.242424 | 777.923014 |
| 16B | Diptera | D_pupa | 24 | 1038.961039 | 222.064027 |
| 16B | Nematoda | NEMA | 104 | 4502.164502 | 431.265317 |
| 16B | Oligochaeta | OLIG | 16 | 692.6406926 | 57.9465007 |
| 16B | Oribatei | ORIB | 8 | 346.3203463 | 15.8918411 |
| 18B | Amphipoda | AMPHI | 17 | 735.9307359 | 3969.64656 |
| 18B | Bivalvia | PELY | 35 | 1515.151515 | 638.176213 |
| 18B | Candonidae | O_Can | 210 | 9090.909091 | 301.709393 |
| 18B | Ceratopogonidae | CERAT | 33 | 1428.571429 | 1139.34543 |
| 18B | Chironominae | CHIR | 65 | 2813.852814 | 661.920378 |
| 18B | Copepod | C_Cant | 128 | 5541.125541 | 73.6688828 |
| 18B | Cyprididae | O_Cypr | 240 | 10389.61039 | 204.399132 |
| 18B | Cytheridae | O_Cyth | 2 | 86.58008658 | 26.6085552 |
| 18B | Diptera | D_pupa | 72 | 3116.883117 | 708.022155 |
| 18B | Halacaridae | HALA | 16 | 692.6406926 | 11.9786572 |
| 18B | Hirudinea | HIRU | 32 | 1385.281385 | 3114.7138 |
| 18B | Invertebrate | HYGR | 8 | 346.3203463 | 1.4698163 |
| 18B | Nematoda | NEMA | 349 | 15108.22511 | 3763.59436 |
| 18B | Oligochaeta | OLIG | 586 | 25367.96537 | 48681.5317 |
| 18B | Oribatei | ORIB | 456 | 19740.25974 | 319.58011 |
| 18B | Plecoptera | PLEC | 1 | 43.29004329 | 414.539435 |
| 18B | Trichoptera | T_pupa | 16 | 692.6406926 | 73.0008128 |
| 19A | Amphipoda | AMPHI | 16 | 692.6406926 | 13.4596858 |
| 19A | Bivalvia | PELY | 48 | 2077.922078 | 354.354309 |

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|---|---|---|---|---|---|
| 19A | Candonidae | O_Can | 240 | 10389.61039 | 235.020711 |
| 19A | Ceratopogonidae | CERAT | 48 | 2077.922078 | 117.601422 |
| 19A | Chironominae | CHIR | 1004 | 43463.20346 | 12297.3854 |
| 19A | Copepod | C_Cant | 32 | 1385.281385 | 18.8877718 |
| 19A | Cyprididae | O_Cypr | 210 | 9090.909091 | 212.530697 |
| 19A | Diptera | D_pupa | 58 | 2510.822511 | 752.95722 |
| 19A | Nematoda | NEMA | 670 | 29004.329 | 2606.74546 |
| 19A | Oligochaeta | OLIG | 344 | 14891.77489 | 4214.90423 |
| 19A | Oribatei | ORIB | 560 | 24242.42424 | 549.322005 |
| 19B | Amphipoda | AMPHI | 48 | 2077.922078 | 3093.56955 |
| 19B | Bivalvia | PELY | 16 | 692.6406926 | 122.735507 |
| 19B | Candonidae | O_Can | 96 | 4155.844156 | 80.2858546 |
| 19B | Chironominae | CHIR | 216 | 9350.649351 | 4309.53173 |
| 19B | Cyprididae | O_Cypr | 72 | 3116.883117 | 32.129518 |
| 19B | Diptera | D_pupa | 32 | 1385.281385 | 1925.9704 |
| 19B | Hirudinea | HIRU | 8 | 346.3203463 | 5785.63609 |
| 19B | Limniphilidae | T_Limn | 16 | 692.6406926 | 13705.2533 |
| 19B | Nematoda | NEMA | 256 | 11082.25108 | 721.629397 |
| 19B | Oligochaeta | OLIG | 224 | 9696.969697 | 7732.85839 |
| 19B | Oribatei | ORIB | 48 | 2077.922078 | 72.2344024 |
| 19B | Tardigrada | TARDI | 32 | 1385.281385 | 40.4048491 |
| 36B | Amphipoda | AMPHI | 33 | 1428.571429 | 47406.8741 |
| 36B | Bivalvia | PELY | 88 | 3809.52381 | 7358.53409 |
| 36B | Candonidae | O_Can | 159 | 6883.116883 | 1123.19737 |
| 36B | Ceratopogonidae | CERAT | 37 | 1601.731602 | 3564.96196 |
| 36B | Chironominae | CHIR | 172 | 7445.887446 | 14673.4828 |
| 36B | Cyprididae | O_Cypr | 31 | 1341.991342 | 196.632626 |
| 36B | Gastropoda | GAST | 5 | 216.4502165 | 1853.90485 |
| 36B | Invertebrate | HYDA | 7 | 303.030303 | 3.34018342 |
| 36B | Invertebrate | HYGR | 18 | 779.2207792 | 2.95258358 |

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|---|---|---|---|---|---|
| 36B | Leptoceridae | T_Lept | 8 | 346.3203463 | 2891.44464 |
| 36B | Nematoda | NEMA | 2404 | 104069.2641 | 52413.8463 |
| 36B | Oligochaeta | OLIG | 50 | 2164.502165 | 3788.76318 |
| 36B | Oribatei | ORIB | 170 | 7359.307359 | 778.929846 |

## Appendix B – Benthic invertebrate PCB concentration (ng/g lipid)

| Congener | 9B | 5B | 5A | 10B | 7B-D | 14A | 19B | 6B | 10A | 19B-D | 7B | 14B | 8B | 3B | 19A | 36B |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 18 | 1.35 | 1.75 | 2.29 | 3.08 | 2.12 | 3.26 | 1.76 | 1.62 | 1.60 | 1.84 | 3.89 | 2.93 | 4.05 | 2.56 | 1.57 | 2.87 |
| 29 | 0.73 | 0 | 2.38 | 2.66 | 0 | 1.79 | 0 | 0 | 0.91 | 0 | 0 | 0 | 1.81 | 7.64 | 1.06 | 0 |
| 31-28 | 0.92 | 1.39 | 0 | 1.66 | 1.65 | 1.04 | 0 | 3.99 | 1.16 | 1.51 | 2.98 | 0 | 3.01 | 0 | 1.26 | 2.54 |
| 52 | 0 | 2.59 | 0 | 0 | 0 | 0 | 2.09 | 3.09 | 2.04 | 3.16 | 0 | 3.76 | 5.11 | 2.76 | 2.34 | 0 |
| 49 | 2.09 | 2.67 | 2.77 | 4.48 | 4.13 | 2.33 | 2.44 | 2.74 | 1.94 | 2.59 | 4.30 | 3.77 | 3.99 | 3.48 | 2.00 | 5.96 |
| 101 | 4.80 | 5.47 | 2.46 | 2.34 | 0 | 4.23 | 4.07 | 6.92 | 0 | 4.28 | 9.20 | 0 | 17.23 | 0.56 | 1.36 | 0.60 |
| 99 | 1.40 | 1.63 | 0 | 1.98 | 1.95 | 2.09 | 1.57 | 1.22 | 1.20 | 1.74 | 2.31 | 1.89 | 2.40 | 2.78 | 0 | 0 |
| 87 | 0.46 | 0 | 0 | 0 | 0 | 0 | 1.01 | 0 | 0.75 | 1.20 | 1.79 | 1.97 | 1.54 | 1.29 | 0 | 1.24 |
| 110 | 0.65 | 0.88 | 1.71 | 1.42 | 1.62 | 0.92 | 1.44 | 0.78 | 0.70 | 2.17 | 2.99 | 2.95 | 2.30 | 2.24 | 0.63 | 1.07 |
| 149 | 1.17 | 1.44 | 1.88 | 1.93 | 2.46 | 1.28 | 1.85 | 1.35 | 1.29 | 2.12 | 2.64 | 2.67 | 2.20 | 2.35 | 1.00 | 1.33 |
| 118 | 1.09 | 0.92 | 1.63 | 1.63 | 1.80 | 1.40 | 1.26 | 1.27 | 1.25 | 1.62 | 2.61 | 2.86 | 1.82 | 2.31 | 0.96 | 1.33 |
| 146 | 0 | 0 | 0 | 0 | 0.92 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 153 | 2.04 | 4.09 | 2.62 | 3.09 | 4.21 | | 3.54 | 3.38 | 1.94 | 2.42 | 2.80 | 2.75 | 3.13 | 4.40 | 1.99 | 2.76 |
| 132 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 105 | 0.42 | 0 | 0.57 | 0 | 0 | 0 | 0 | 2.51 | 0.38 | 0.53 | 0.83 | 0.86 | 0 | 0.73 | 0 | 0.71 |
| 138 | 0.69 | 1.02 | 0 | 0 | 0 | 0 | 1.07 | 0 | 0.80 | 0.94 | 1.33 | 1.38 | 1.49 | 0 | 0.81 | 1.79 |
| 163 | 1.26 | 1.41 | 1.59 | 1.75 | 2.18 | 1.13 | 1.81 | 1.34 | 1.25 | 1.60 | 2.04 | 2.19 | 1.82 | 3.21 | 1.16 | 1.93 |
| 187 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.92 | 0.81 | 0.76 | 0 | 1.02 | 0.96 | 1.78 | 0 | 1.00 |
| 183 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.47 | 0 | 0.69 | 0.72 | 0 | 0 | 0 |
| 128 | 0.30 | 0.27 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.65 | 0 | 0 | 0.29 | 0 |
| 156 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.72 | 0 | 1.23 | 0 | 0 |
| 201 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.86 | 0 | 0 |
| 157 | 0 | 0 | 0.52 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 180 | 0.19 | 0.08 | 0.33 | 0.67 | 1.00 | 0.38 | 0.48 | 0.28 | 0 | 0.40 | 0.18 | 0.49 | 0.49 | 1.26 | 0.03 | 0.27 |
| 170 | 0.40 | 0.52 | 0.54 | 0.69 | 0.80 | 0 | 0.60 | 0.55 | 0.35 | 0.49 | 0.60 | 0.71 | 0.74 | 0.74 | 0 | 0.91 |

## Appendix C – Fish PCB concentrations (ng/g lipid)

| congeners | 11B-9SS-D | 10A-9SS | 11B-9SS | 19B-PS | 19B-PS-D | 16B-9SS | 11B-PS | 7B-9SS |
|---|---|---|---|---|---|---|---|---|
| 18 | 4.72 | 3.62 | 2.18 | 6.83 | 0.00 | 2.95 | 6.57 | 5.49 |
| 29 | 0.00 | 0.00 | 0.00 | 6.66 | 4.96 | 0.00 | 0.00 | 3.95 |
| 31-28 | 2.50 | | 1.99 | 4.67 | 3.56 | 1.47 | 4.74 | 1.77 |
| 52 | 0.00 | 4.44 | 0.00 | 0.00 | 0.00 | 3.48 | 0.00 | 2.79 |
| 49 | 5.25 | 4.30 | 5.94 | 8.61 | 7.80 | 3.25 | 10.71 | 5.85 |
| 44 | 4.46 | 3.98 | 4.06 | 4.72 | 9.33 | 2.36 | 9.26 | 3.95 |
| 66 | 0.00 | 3.47 | 0.00 | 5.07 | 0.00 | 0.00 | 0.00 | 4.79 |
| 99 | 3.60 | 3.75 | 3.40 | 5.17 | 7.44 | 2.68 | 4.78 | 2.44 |
| 87 | 1.99 | 2.17 | 2.08 | 3.79 | 5.28 | 1.63 | 3.65 | 1.47 |
| 110 | 3.34 | 3.17 | 3.15 | 5.11 | 10.38 | 2.32 | 5.88 | 3.40 |
| 149 | 2.93 | 2.76 | 3.13 | 5.83 | 7.58 | 2.58 | 4.58 | 3.87 |
| 118 | 2.21 | 1.77 | 2.39 | 3.70 | 5.68 | 1.44 | 3.18 | 2.66 |
| 146 | 0.81 | 0.00 | 1.12 | 2.56 | 0.00 | 0.91 | 0.00 | 0.00 |
| 153 | 3.94 | 3.09 | 3.11 | 6.61 | 9.48 | 2.63 | 5.30 | 4.78 |
| 132 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| 105 | 1.01 | 0.51 | 0.82 | 1.31 | 2.11 | 0.56 | 1.90 | 0.94 |
| 138 | 2.18 | 1.43 | 1.89 | 2.85 | 4.52 | 1.53 | 4.74 | 2.60 |
| 163 | 2.58 | 1.69 | 2.33 | 3.92 | 6.15 | 1.80 | 3.58 | 3.20 |
| 187 | 1.05 | 1.09 | 1.18 | 1.86 | 2.96 | 1.17 | 2.40 | 2.81 |
| 183 | 0.66 | 0.89 | 1.27 | 1.29 | 1.63 | 0.83 | 0.00 | 1.77 |
| 128 | 0.00 | 0.70 | 0.00 | 0.00 | 0.79 | 0.47 | 0.63 | 0.00 |
| 156 | 0.00 | 0.50 | 0.99 | 1.52 | 1.90 | 0.00 | 0.78 | 0.00 |
| 201 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| 157 | 0.00 | 0.00 | 1.06 | 1.44 | 1.58 | 0.00 | 1.07 | 0.00 |
| 180 | 1.83 | 1.67 | 2.56 | 3.40 | 4.80 | 1.11 | 3.31 | 2.89 |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| 170 | 0.72 | 0.62 | 0.86 | 1.19 | 1.62 | 0.41 | 1.08 | 1.05 |
| 195 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| 194 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| 206 | 0.79 | 0.00 | 0.00 | 0.00 | 0.00 | 0.20 | 0.80 | 0.00 |
| 209 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |

## Appendix D – Concentrations of OC Pesticides in *Gammarus* (ng/g lipid)

| Pesticide Classes (ng/g organic carbon) | 5A | 14A | 10A | 19A | 6B | 7B | 7B0D |
|---|---|---|---|---|---|---|---|
| Hexachlorocyclohexanes | 5.59 | 6.04 | 7.99 | 7.13 | 7.62 | 6.71 | 7.81 |
| DDT | 23.87 | 2.03 | 1.18 | 16.36 | 18.04 | 19.14 | 27.62 |
| Aldrins | 12.54 | 0.59 | 6.05 | 8.78 | 9.81 | 6.83 | 6.89 |
| Chlordanes | 2.60 | 1.54 | 2.57 | 11.08 | 2.92 | 7.16 | 3.42 |
| Chlorobenzenes | 17.02 | 13.87 | 0.94 | 15.48 | 11.69 | 12.71 | 17.91 |
| Endosulfans | 2.30 | 1.95 | 2.57 | 2.53 | 3.41 | 4.40 | 3.42 |

| Pesticide Classes (ng/g organic carbon) | 9B | 5B | 10B | 19B0D | 19B | 14B | 8B | 3B | 36B |
|---|---|---|---|---|---|---|---|---|---|
| Hexachlorocyclohexanes | 6.35 | 6.69 | 7.15 | 6.17 | 8.67 | 6.07 | 6.93 | 5.67 | 10.03 |
| DDT(dichlorodiphenyltrichloroethane) & Metabolites | 11.96 | 17.91 | 21.82 | 9.55 | 16.85 | 14.26 | 12.89 | 28.62 | 22.95 |
| Aldrin-Dieldrin-Endrin | 8.11 | 13.38 | 12.63 | 6.85 | 9.07 | 13.44 | 14.29 | 10.72 | 15.36 |
| Chlordanes | 2.27 | 8.48 | 12.34 | 14.44 | 5.88 | 2.54 | 15.71 | 2.42 | 7.26 |
| Chlorobenzenes | 13.26 | 8.36 | 14.35 | 7.23 | 14.20 | 2.68 | 5.84 | 8.00 | 6.99 |
| Endosulfans | 1.76 | 2.77 | 2.25 | 3.04 | 2.11 | 2.54 | 4.28 | 1.96 | 4.40 |

## Appendix E – Concentrations of OC Pesticides in Fish (ng/g lipid)

| Pesticide Classes (ng/g organic carbon) | 10A-9SS | 11B-9SS | 11B-9SS-D | 16B-9SS | 7B-9SS | 11B-PS | 19B-PS | 19B-PS-D |
|---|---|---|---|---|---|---|---|---|
| Hexachlorocyclohexanes | 24.35 | 22.31 | 27.92 | 40.49 | 13.45 | 4.89 | 48.85 | 46.46 |
| DDT(dichlorodiphenyltrichloroethane) & Metabolites | 4.63 | 36.84 | 39.48 | 60.05 | 10.05 | 18.69 | 62.08 | 13.89 |
| Aldrin-Dieldrin-Endrin | 34.34 | 51.48 | 0.00 | 4.61 | 45.44 | 141.38 | 85.91 | 26.54 |
| Chlordanes | 4.68 | 5.89 | 6.53 | 104.45 | 26.40 | 29.48 | 131.48 | 66.66 |
| Chlorobenzenes | 38.29 | 40.12 | 45.87 | 69.95 | 35.54 | 13.47 | 59.00 | 51.73 |
| Endosulfans | 4.68 | 5.59 | 18.88 | 5.12 | 4.95 | 7.66 | 6.53 | 4.73 |

## Appendix F – Concentrations of PCB homologues in sediments (ng/g organic carbon)

| Homologues (ng/g organic carbon) | 9A-1 | 9B-1 | 14A-1 | 6B-1 | 14B-1 | 7B-1 | 4B-1 | 5A-1 | 3B-1 | 5B-1 |
|---|---|---|---|---|---|---|---|---|---|---|
| monochlorobiphenyl (1-3) | 0.0571 | 0.08 | 0.08 | 0.12 | 0.05 | 0.14 | 0.05 | 0.05 | 0.11 | 0.00 |
| dichlorobiphenyl (4-15) | 0.04 | 0.03 | 0.03 | 0.04 | 0.02 | 0.19 | 0.02 | 0.00 | 0.03 | 0.02 |
| trichlorobiphenyl (16-39) | 0.03 | 0.03 | 0.02 | 0.07 | 0.01 | 0.04 | 0.02 | 0.07 | 0.02 | 0.03 |
| tetrachlorobiphenyl (40-81) | 0.03 | 0.04 | 0.04 | 0.04 | 0.03 | 0.06 | 0.01 | 0.02 | 0.03 | 0.07 |
| pentachlorobiphenyl (82-127) | 0.03 | 0.04 | 0.07 | 0.07 | 0.08 | 0.05 | 0.03 | 0.06 | 0.06 | 0.09 |
| hexachlorobiphenyl (128-169) | 0.01 | 0.01 | 0.01 | 0.03 | 0.01 | 0.01 | 0.01 | 0.00 | 0.02 | 0.01 |
| heptachlorobiphenyl (170-193) | 0.00 | 0.00 | 0.04 | 0.00 | 0.03 | 0.00 | 0.01 | 0.01 | 0.01 | 0.00 |
| octachlorobiphenyl (194-205) | 0.00 | 0.00 | 0.00 | 0.01 | 0.00 | 0.01 | 0.00 | 0.00 | 0.00 | 0.01 |
| nonachlorobiphenyl (206-208) | 0.00 | 0.00 | 0.00 | 0.01 | 0.00 | 0.01 | 0.01 | 0.01 | 0.00 | 0.00 |
| decachlorobiphenyl (209) | 0.20 | 0.23 | 0.30 | 0.39 | 0.25 | 0.49 | 0.17 | 0.22 | 0.28 | 0.23 |

| Homologues (ng/g organic carbon) | 16B-3 | 19A-1 | 16B-1 | 36B-1 | 19B-1 | 10B-1 | 11B-1 | 36B-3 | 7B-4 | 7A-1 |
|---|---|---|---|---|---|---|---|---|---|---|
| monochlorobiphenyl (1-3) | 0.14 | 0.00 | 0.11 | 0.13 | 0.25 | 0.12 | 0.22 | 0.15 | 0.13 | 0.12 |
| dichlorobiphenyl (4-15) | 0.09 | 0.02 | 0.07 | 0.06 | 0.04 | 0.01 | 0.14 | 0.13 | 0.03 | 0.04 |
| trichlorobiphenyl (16-39) | 0.02 | 0.05 | 0.00 | 0.47 | 0.02 | 0.05 | 0.54 | 0.63 | 0.10 | 0.27 |
| tetrachlorobiphenyl (40-81) | 0.02 | 0.02 | 0.02 | 0.05 | 0.05 | 0.07 | 0.14 | 0.11 | 0.03 | 0.07 |
| pentachlorobiphenyl (82-127) | 0.05 | 0.03 | 0.01 | 0.10 | 0.07 | 0.05 | 0.15 | 0.19 | 0.12 | 0.09 |
| hexachlorobiphenyl (128-169) | 0.02 | 0.00 | 0.01 | 0.02 | 0.03 | 0.04 | 0.02 | 0.04 | 0.09 | 0.08 |
| heptachlorobiphenyl (170-193) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.02 | 0.04 | 0.01 |
| octachlorobiphenyl (194-205) | 0.00 | 0.01 | 0.01 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| nonachlorobiphenyl (206-208) | 0.00 | 0.01 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.01 |
| decachlorobiphenyl (209) | 0.35 | 0.14 | 0.23 | 0.83 | 0.47 | 0.35 | 1.21 | 1.26 | 0.54 | 0.69 |

## Appendix G – Concentrations of OC Pesticides in sediments (ng/g organic carbon)

| | 9A-1 | 14A-1 | 5A-1 | 19A-1 | 7A-1 | 6B-1 | 14B-1 | 9B-1 |
|---|---|---|---|---|---|---|---|---|
| Hexachlorocyclohexanes | 2.97 | 1.58 | 4.43 | 3.03 | 2.62 | 0.83 | 0.92 | 0.00 |
| DDT(dichlorodiphenyltrichloroethane) & Metabolites | 0.14 | 0.35 | 6.76 | 0.56 | 3.14 | 1.16 | 0.57 | 1.71 |
| Aldrin-Dieldrin-Endrin | 0.06 | 0.30 | 0.55 | 0.34 | 0.75 | 0.09 | 0.61 | 0.00 |
| Chlordanes | 0.09 | 0.03 | 7.63 | 0.29 | 3.87 | 0.00 | 0.63 | 0.00 |
| Chlorobenzenes | 0.16 | 0.16 | 3.00 | 0.12 | 1.20 | 0.36 | 0.23 | 0.00 |
| Endosulfans | 0.09 | 0.03 | 2.18 | 0.05 | 0.54 | 0.00 | 0.17 | 0.00 |

| | 7B-1 | 4B-1 | 3B-1 | 5B-1 | 10B-1 | 11B-1 | 14B-4 | 36B-3 |
|---|---|---|---|---|---|---|---|---|
| Hexachlorocyclohexanes | 4.79 | 4.90 | 8.58 | 11.99 | 4.77 | 1.38 | 2.26 | 5.92 |
| DDT(dichlorodiphenyltrichloroethane) & Metabolites | 3.98 | 1.48 | 1.25 | 1.83 | 9.81 | 2.53 | 0.83 | 2.00 |
| Aldrin-Dieldrin-Endrin | 0.00 | 0.68 | 0.82 | 1.18 | 2.04 | 0.79 | 0.33 | 0.16 |
| Chlordanes | 0.00 | 1.06 | 0.76 | 1.22 | 10.26 | 1.68 | 2.14 | 1.48 |
| Chlorobenzenes | 1.32 | 0.37 | 0.79 | 0.39 | 4.06 | 0.60 | 0.82 | 0.41 |
| Endosulfans | 0.00 | 0.14 | 0.11 | 0.17 | 3.11 | 0.24 | 0.26 | 0.18 |