

Establishment of Ecological Baseline Metrics for Integrated Ecomonitoring and Assessment of Cumulative Impacts on Great Slave Lake Fisheries Ecosystems

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by

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ABSTRACT

An integrated monitoring program was conducted in the main basin of Great Slave Lake (GSL) in the summers (June–August) of 2011–2016. The overarching objectives of this study were to 1) establish standardized sampling protocols guiding community-based monitoring of water quality, phytoplankton, zooplankton, benthos and fish; 2) quantify spatial and temporal variation in the relative abundance and biomass of major biological components, and 3) explore indicator relationships between fish community diversity and environmental variables to characterize the structure of the GSL ecosystem. Collaborating with the indigenous communities and decision makers, we adopted a pathways approach, outlined by cumulative impact monitoring program (CIMP), and applied depth-stratified random sampling protocols to collect quantitative information covering the limnology, biological production and fisheries.

A 245 grid system, separated by 5 minutes of latitude and 10 minutes of longitude, was developed to cover the six management areas of the main basin of GSL. During the implementation, the management area based surveys have been conducted in a rotating format over Resolution Bay, Moraine Bay, Yellowknife Bay and Simpsons' Island, whereas the western basin has been surveyed every summer to account for inter-annual variation.

Over the six management areas (IW-IV) and eight depth strata (20 m intervals), there were significant differences in environmental variables, such as temperature, dissolved oxygen and turbidity, by management area and depth strata. Over 2011-2016, temperature showed a consistent pattern, but turbidity differed by years. Zooplankton was dominated by copepods, comprising 80% of all samples. Approximately 78% of the total zooplankton density from all samples occurred in shallow sites (<20 m) and was negatively correlated to depth. Significant temporal variability was observed in the western basin, approximately four and three times denser in 2013, respectively, than the remaining years. Benthos was dominated by ostracods (mean \pm SE; 599 ± 74 individuals/m²) and amphipods (551 ± 47 individuals/m²), which accounted for over 69% of the total density of benthic invertebrates, followed by oligochaetes, bivalves and chironomids. No evident temporal variation in benthos in the western basin was found, but the greatest invertebrate density was found in areas IV ($2\,792 \pm 348$ individuals/m²) and V ($3\,312 \pm 1\,051$ individuals/m²).

A total of 387 effective gillnet settings were deployed recording 24 fish species. The grand average catches were 108 individuals and 33.65 kg per set. Combined with percentages of number, weight and frequency of occurrence, three coregonids, Lake Whitefish (*Coregonus clupeaformis*), Least Cisco (*C. sardinella*) and Lake Herring (*C. artedii*), were determined as “dominants” of the fish community in the lake. Standardized by soak duration, type of experimental gillnets and area of mesh-sized panel, the grand averages of relative abundance and biomass, equivalent to number or biomass per unit effort (NPUE or BPUE), were estimated to 0.28 ± 0.02 individuals/m² and 65.62 ± 4.04 g/m², respectively. Spatial distribution of NPUE differed among management areas, whilst significant difference in BPUE was found through depth stratified settings. Multivariate analyses, including cluster analysis and canonical correspondence analysis, revealed that depth and temperature were important environmental variables dictating the spatiotemporal dynamics and predator-prey relationships of the multispecies biological production in the lake.

RÉSUMÉ

Un programme multidisciplinaire de suivi écologique a été conduit dans le bassin majeur du Great Slave Lake (GSL) durant les étés 2011 à 2016 (Juin-Aout). Les objectifs principaux de cette étude étaient 1) d'établir les conditions environnementales de bases et leur variabilité dans ce grand lac arctique et 2) d'explorer les indicateurs environnementaux et les changements cumulés pour caractériser la productivité aquatique, la biodiversité, et les associations environnementales. En collaboration avec les communautés affectées et les décideurs, nous avons adopté une CIMP Pathways Approach pour prendre en compte les préoccupations des communautés et appliquer des protocoles d'échantillonnages aléatoires stratifiés en fonction de la profondeur pour collecter des informations quantitatives traitant de la limnologie, les niveaux trophiques inférieurs chez les invertébrés et les pêcheries. Une grille de 245 points séparés par 5 min de latitude et 10 minutes de longitude a été développée pour couvrir les six secteurs de gestion du bassin principal de GSL. Durant la mise en place, des relevés de bases ont été conduits sur les secteurs de gestion en alternant entre Resolution Bay, Moraine Bay, Yellowknife Bay et Simpson Island, alors que l'est du bassin a été étudié chaque été pour prendre en compte les variations interannuelles. Pour les six secteurs de gestion et les huit profondeurs (20 m d'intervalle) étudiés, des différences significatives ont été observées pour les variables environnementales (température, l'oxygène dissous, et la turbidité), les secteurs de gestion et la profondeur.

De 2011 à 2016, la température a montré une tendance constante alors que des changements de turbidité ont été observés en fonction des décharges de Slave River, celles-ci dépendantes de la régulation de l'eau et des fluctuations naturelles d'évaporation et de précipitations. Le zooplancton était dominé par les copépodes, comprenant 80% de l'ensemble des échantillons. Approximativement 78% des densités totales de zooplancton ont été observées dans les sites peu profond (<20m) et étaient négativement corrélées avec la profondeur. Une variabilité temporelle significative a été observée dans la partie ouest du bassin, approximativement trois à quatre fois plus importante en 2013 que les autres années. Le benthos était dominé par les ostracodes (moyenne \pm écart-type, $599 \pm 1\,051$ individus/m²) et amphipodes (551 ± 47 individus/m²) qui représentent plus de 69% de la densité totale des invertébrés benthiques, suivit par les oligochètes, bivalves et chironomidés. Aucune variation temporelle n'a été observée pour le benthos dans le bassin ouest du lac mais des densités supérieures ont été relevées dans les secteurs IV ($2\,792 \pm 348$ individus/m²) et V ($3\,312 \pm 1\,051$ individus/m²). Ce résultat est probablement dû en partie aux substrats meubles qui favorisent les échantillonnages par la drague benthique comparé aux substrats durs ou sableux.

Au total, 387 filets maillants ont été déployés et 24 espèces de poisson ont été collectées. La moyenne générale était de 108 individus et 33,65 kg par filet. Le nombre, le poids et la fréquence d'occurrence combinés ont déterminé que trois espèces de Corégonidés « dominant » la communauté de poissons de ce lac : le grand corégone, le cisco sardinelle et le cisco de lac. La moyenne générale d'abondance et de biomasse relative ont été standardisées par la durée d'immersion, le type de filet maillant et l'aire du panneau de maillage, ce qui équivaut au

nombre par unité d'effort ou à la biomasse par unité d'effort (NPUE or BPUE), et ont été estimés à 0.28 ± 0.02 individus/m² et 65.62 ± 4.04 g/m², respectivement. La distribution spatiale du NPUE diffère selon les secteurs de gestion alors que des différences significatives ont été observées pour la BPUE en fonction des intervalles de profondeur étudiées. Les analyses multivariées incluant une analyse par regroupement et une analyse canonique de correspondance, ont relevé que la profondeur et la température étaient des variables environnementales importantes qui influencent les dynamiques spatio-temporelles et les relations proies-prédateurs dans la production de pêche multi espèces de ce lac.

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INTRODUCTION

Great Slave Lake (GSL) is a deep, cold, oligotrophic freshwater lake in the sub-arctic region of the Northwest Territories (Rawson 1950; Evans 2000). It is the second largest arctic great lake in Canada, the deepest lake (max depth = 614 m) in North America, and the 9th largest lake in the world (Munawar 1987; MRBB 2004). Often considered a relatively pristine arctic great lake, the vulnerability to cumulative impacts is currently understudied despite increasing anthropogenic activities such as contamination (Moore et al. 1979; Mudroch et al. 1992; Evans 2000; Wayland et al. 2000), mining (Cott et al. 2016; Evans and Muir 2016), hydroclimate changes (Gibson et al. 2006) and fishery exploitation (Rawson 1949; Keleher 1972; Healey 1975; Zhu et al. 2016). For example, the discovery of gold during the 1930s on the northern shore of GSL subsequently caused arsenic contaminants to pass through the aquatic environment into human consumption of fish products from the watershed (Moore et al. 1979; Cott et al. 2016; Evans and Muir 2016). Large-scale temporal variation in discharges from major input and output tributaries has resulted in anomalous transports of sediments and nutrients (Rawlins et al. 2009). In terms of data collected prior to 1995, up to 75% of Indigenous Peoples that completed the survey from Fort Smith and Fort Fitzgerald noted more algal growth and 44% noted higher turbidity. All those changes of water quality are believed to be the result of a series of recent hydroclimate regime shifts (MRBB 2004; IPCC 2014).

Most importantly, GSL has been renowned to offer the largest commercial, recreational and Aboriginal (CRA) freshwater fisheries in the Northwest Territories, Canada, benefitting the indigenous communities with abundant and valuable Lake Whitefish, *Coregonus clupeaformis* (Mitchill) and Lake Trout, *Salvelinus namaycush* (Walbaum). In the early 1950s, a historical peak in commercial harvest, over 4000 tonnes of Lake Whitefish and Lake Trout combined, appeared to be the largest commercial fishery in the Northwest Territories (DFO 2015). During the 1960s, the GSL commercial fishery had stabilized at about 1700 tonnes of whitefish annually, whereas trout started a steady decline to 333 tonnes. In 1972, the commercial Lake Whitefish harvest also declined to a level of 1000 tonnes, compared with a total of 2604 tonnes in 1950 (Bond and Turnbull 1973). At the same period, commercial Lake Trout harvest varied from 1823 tonnes in 1949 to as low as 86 tonnes in 1972 (Healey 1972). A downsizing commercial market and soaring fuel prices caused the annual harvest levels of GSL commercial fisheries to remain at the level of approximately 1500 tonnes until the early 1990s and steadily dropped to 620 tonnes for all fish species in 2015-2016 (Zhu et al. 2014).

As interests in exploring natural resources and warning signals of cumulative impacts in the GSL ecosystem increase, scientific investigation has become an essential opportunity to help understand the structure, function and response of the ecosystem to the cumulative impacts. In 1944, the pioneer commercial fishery feasibility survey was undertaken by the Fishery Research Board of Canada to study the relationship between fisheries production and the environment (Rawson 1950; Kennedy 1953). It recommended commencing the GSL commercial fisheries at the beginning of the 1946 fishing season. Subsequent research has addressed fundamental

questions relating to the age, growth, maturity, mortality and gillnet-based capture efficiency for species of commercial interest, such as Lake Whitefish (Kennedy 1951; Kennedy 1953; Healey 1975), Lake Trout (Kennedy 1954) and Inconnu (Fuller 1955). Based on the outcomes of these studies, in 1972 the Department of Fisheries and Oceans Canada (DFO) established fisheries management areas with annual fisheries quotas and regulations. Additional studies on commercial catches (Moshenko and Low 1978; Read and Taptuna 2003), fish plant samples (Bond and Turnbull 1973), annual harvests (Yaremchuk et al. 1989), multi-panel gillnet selectivity (Bond 1975; Moshenko and Low 1978; Roberge et al. 1985b; Stewart et al. 1999; Day 2002), annual reproductive behaviours (Roberge et al. 1985a) and fishing effort (Falk et al. 1975; Thompson et al. 1988) have contributed to the current fishing regulations. These include 1) no commercial fishing of Lake Trout in the East Arm, 2) a minimum stretched mesh size of 140 mm (5") for commercial gillnets, 3) Inconnu fishing closures in southern shallow waters during the summer and fall seasons, and 4) establishment of a long-term stock assessment monitoring program. Over the 70+ year history of the GSL commercial fisheries, substantial changes in the relative importance of composing species, productivity and ecosystem resilience to multiple cumulative impacts have been observed. However, the mechanisms and possible consequences of these changes remain unknown due to a lack of continuous time-series monitoring data. Our recent analyses indicated that during 1960–2010, riverine discharge from the Slave River tended to increase in winter (February) and gradually decreased in early summer (June) largely as a result of water regulations in the Peace-Athabasca-Slave system caused by the operation of the W.A.C. Bennett Dam (Zhu et al. 2016). As a result, significant positive correlations between riverine inflow and fish length were identified in southern GSL, indicating that reduced discharge from the Slave River may depress the growth of Lake Whitefish. In addition to hydroclimate variables, local meteorological conditions, including air temperature, precipitation and wind, that prevail in a particular region may influence the regimes of water temperature and the flow of watercourses (Jonsson and Jonsson 2009). In GSL, positive correlations were detected between air temperature, condition factors and asymptotic size of Lake Whitefish in southern GSL before May, and negative relationships were found during June through November. This suggests that warmer winters may benefit growing conditions for overwintering schools while warmer summers may retard the growth efficiency of this cold-water fish (Zhu et al. 2016). Because of lack of relevant scientific research and long-term monitoring activities, many information gaps exist to be able to interpret internal mechanisms governing interaction among several connecting components of the GSL ecosystem. For instance, at the fish population level, there are no definitive studies conducting quantitative fisheries stock assessments to generate biological reference points such as maximum sustainable yield (MSY), total allowable catches (TACs) and harvest control rule (HCR; Walters and Martell 2004). At the multispecies level, there is a lack of detailed information describing the fish community, such as species richness, relative importance and associations with environmental variables (Alexander et al. 2011). At the lake ecosystem level, there is no baseline information on the GSL fisheries ecosystem, such as aquatic productivity, trophic linkages, and carrying capacity or key indicators

reflecting the nature of GSL in response to natural and external stressors (McCarthy et al. 1997; del Monte-Luna et al. 2004; Link 2005; Pinnegar and Engelhard 2008).

In 2012, the Fisheries Protection Provisions of Canada's Fisheries Act, amended via Bill C-38, called for 1) the development of adaptive management strategies addressing the extent of cumulative impacts with a measurable approach, 2) the evaluation of the sustainability of fishable populations and 3) the integration of uncertainties and environmental risks in fisheries management. GSL provides an ideal ecological platform for researchers, indigenous peoples and decision makers to work together to address these scientific strategies in the subarctic aquatic ecosystem. Incorporated with traditional ecological knowledge, this study establishes a current baseline for the ecosystem, indicators of ecosystem change and assessing the cumulative impacts on the sustainability and ongoing productivity of the lake. The overarching objectives of this six-year basin-wide baseline study were designed to address the community and decision makers' concerns of cumulative impacts on the sustainability and ongoing productivity of the GSL ecosystem. This study strategically focused to 1) establish standardized sampling protocols guiding community-based monitoring activities for monitoring water quality, phytoplankton, zooplankton, benthos and fish; 2) quantify the relative abundance and biomass of major biological components at the species and population level and to index the temporal changes and 3) explore relationships between fish community diversity and environmental variables to characterize the structure of the GSL ecosystem.

MATERIALS AND METHODS

Study area

Geographically, GSL is a major sub-arctic component of the 4200 km long Mackenzie River drainage system, overlain at the boundary of the Interior Platform and the Canadian Shield. In the southwest area of the lake, Paleozoic sediments lie on Precambrian sedimentary and crystalline rocks overlain by thin Lower Cretaceous sediments (MRBB 2004). In the southeast area of the lake, the bedrock is of mixed sedimentary igneous origin and of Proterozoic age. The East Arm, a distinctive feature of GSL, is a Proterozoic fold belt of low-grade metamorphic rocks, intruded by gabbro sills and diabase dykes and overlain by a series of sediments. The Canadian Shield, extending across the northeast area of GSL, is an eroded Archean peneplain of granites and sedimentary-volcanic greenstone belts. The Slave River inputs 87% of the total annual discharge into GSL, carrying waters originating from northern British Columbia and Alberta. Mean residence times are 12 years for the whole lake and 7 years for the western basin (Rawson 1950). The huge volumes of river water discharge entering GSL greatly affect water clarity. The transparency varies from 0.2 m in the southern shoreline of the Slave River Delta to 15 m in the east Arm, and is strongly influenced by the catchment characteristics associated with the proximity to the treeline (Rühland and Smol 1998).

Survey Design

A 245 grid system, separated by 5 minutes of latitude and 10 minutes of longitude, was developed to cover the six management areas of the main basin of GSL (Fig. 1). During 2012 through 2016, the western basin (areas IW and IE) was sampled annually, while one of the remaining management areas (II, III, IV or V), was surveyed to cover the entire lake. Sampling occurred during summer each year, between June and August. According to the depth-stratified random sampling protocol (Zhu et al. 2011), a total of 50 grids were designed, positioned at the center of each grid with navigation using a GPS (Garmin GPSMAP 60 CSX and 62, <https://www.garmin.com>). Two field research crews were organized consisting of DFO research biologists and boat captains, technicians and helpers contracted from community members of K'atl'odeeche First Nation (KFN), West Point First Nation (WPFN), Deninu Kué First Nation (DKFN), North Slave First Nation (NSFN), the Hay River Métis Council (HRMC) and the Fort Resolution Métis Council (FRMC). One fishing boat was contracted for each field crew.

Environmental variables

The field sampling procedures were completed sequentially for limnology, zooplankton and benthos before deploying the standardized and commercial gillnets. When arriving at a grid, the actual grid depth was determined by a depth sounder from the fishing boat. A YSI 6600 v2 water quality Sonde (www.ysi.com) with a multiparameter display system (MDS 650) was used to measure six limnological variables: depth (m), temperature (°C), dissolved oxygen (mg/l), pH, turbidity (NTU) and conductivity ($\mu\text{S}/\text{cm}$). Vertical profiles of the lake were performed by recording measurements every meter between 0–10 m, every 2 m between 10–20 m, and every 5 m interval after 20 m to reach the lake bottom.

In addition to the YSI sonde, secchi disk depth (m) was determined with a 30 cm diameter, black and white disk lowered over the shady side of the boat. Meanwhile, weather conditions, including wind direction, wind speed (km/h), cloud cover (%), precipitation (mm), visibility (km) and wave height (m) were determined visually at each grid at the time of sampling.

Low-trophic biological sampling

Low trophic biological components included chlorophyll a, zooplankton and benthic invertebrate samples. Chlorophyll a was measured by the YSI chlorophyll a sensor. Zooplankton were collected with standard plankton nets (1.5 m long with a 0.5 m diameter opening, $A = 0.1963 \text{ m}^2$; www.aquasample.com) with a 118 μm mesh size. Two integrated replicates of zooplankton samples were vertically pulled from 2 m above the lake bottom to the surface at a rate of 0.8–1.0 m/s. All zooplankton samples were washed into 250 ml jars and preserved with sugared ethanol reagent (1 l of 95% ethanol mixed with 20 g sugar) for shipment to the DFO Freshwater Institute in Winnipeg, MB.

For benthic invertebrates, sediment and mud samples were collected using a petite (6" \times 6", $A = 0.0232 \text{ m}^2$) or standard (9" \times 9", $A = 0.0523 \text{ m}^2$) PONAR dredge. Three replicates were taken at each grid and transferred to steel buckets for transport. To be considered a valid replicate, more than 50% chamber space had to be filled. When on land, the substrate sediment samples were

first washed by using an 83 µm washing bucket. Two washing sieves (250 µm and 1000 µm) were stacked to remove coarse gravel, sand, mud and fine clays from the samples by gently rinsing under freshwater. The remaining was transferred into a 250-ml jar with 10% buffered formalin solution for transport to Winnipeg.

Low-trophic sample processing

Zooplankton samples were washed under lightly running distilled water using a 90 µm sieve. Mysids were counted and removed from the entire sample and placed in a vial containing sugared-ethanol solution. The remaining zooplankton were transferred to a vial and filled with a sugared-ethanol solution to a final volume of 40 ml. The vial was lightly shaken to evenly distribute the zooplankton and a 1 ml Hensen-Stempel pipette was immediately submerged into the vial to collect a subsample. The subsample was transferred to a Bogorov tray for counting using a dissecting microscope at 40x magnification. If less than 400 individuals were counted then a subsequent subsample was taken from the vial and processed, until the total abundance in the subsamples represented a minimum of 400 individuals or 20% (8 ml) of the initial 40 ml volume had been processed. Taxa were identified as Mysidae, Calanoida, Cyclopoida, Bosminidae, Daphniidae or other Cladocera. Final counts for each replicate were estimated by dividing the counts from the processed subsample(s) by the proportion of volume subsampled from the 40 ml sample.

For benthic invertebrates, two replicates from each grid were washed through a 250 µm sieve with freshwater and then preserved in a 70% ethanol solution. Sample processing was completed by transferring samples to gridded Petri dishes and manually picking invertebrates under a dissecting microscope at 40x magnification. Samples were identified to the following taxa: Amphipoda, Bivalvia, Ceratopogonidae, Chironomidae, Coleoptera, Collembola, Diptera (unknown), Ephemeroptera, Gastropoda, Hemiptera, Hirudinea, Hydridae, Lepidoptera, Megaloptera, Mysidae, Nemata, Odonata, Oligochaeta, Ostracoda, Phoridae, Plecoptera, Scathophagidae, Tardigrada, Trichoptera and Turbellaria. Nemata were the dominant taxa found in all samples (48 % of total abundance) but were excluded from the analysis due to their low biomass. Counts of each taxon were determined for each replicate, and then averaged to provide a final abundance for each grid.

Fish sampling

Fish were collected with up to three different sets of gillnets, deployed at pre-assigned depths to facilitate analysis of the strata-based abundance and biomass data. Two kinds of experimental gillnets, called pelagic and benthic nets, were designed, with the same numbers of mesh-sized panels but different panel heights of 3.66 m and 1.83 m, respectively. Ten different mesh size panels, ranging from 13 mm (½") to 140 mm (5 ½"), were randomly strung together into one gang of nets. An additional panel with a mesh size of 133 mm (5 ¼"), 183 m long and 9.1 m depth, was attached to the end of the experimental gillnet set to mimic commercial catch methods.

The type and number of gillnets used depended on the depth of the sampling grid (Table 1). Benthic nets were deployed on the lake bottom at grids < 10 m deep and > 20 m deep. A set of pelagic nets was placed at a depth of 5 m in sites >10 m deep. An additional midwater pelagic net set was set at depths of 20 m and 30 m for grids 40–60 m and 60+ m deep, respectively. According to sampling specifics by Ontario Ministry of Natural Resources (OMNR, Morgan and Snucins 2005), all gillnets kept in the water overnight for 18–30 h were considered as an effective set. Any gillnet sets outside this time range were treated as invalid and excluded from further analysis.

Fish processing

All fish samples were sorted into species level, classifying by common names (Scott and Crossman 1998; Nelson et al. 2004), and separated by the mesh-size panel and net set they were collected in. Species-specific enumeration and round weight were measured to account for the variation of species richness, abundance and biomass quantities. Sub-sampling from panels of the nets varied with the underlying research interests and importance of the species studied. A minimum requirement for the fishery-independent study is that all recreational fish are completely sampled; all common fish species are sampled by a quantity of at least 200 individuals and all uncommon or rare species are sampled completely (Zhu et al. 2011). The target species, Lake Whitefish, Lake Trout and Inconnu, are sampled either in numbers proportional to the overall catch for each net set and mesh size or at least 1000 samples over the entire sampling season. If ten or fewer fish of a given species are captured by a panel, the detailed sampling procedure is required for the entire sample. If more than 10 fish are captured, only 10 randomly selected individuals that represent the size distribution of the species captured in the net and mesh size are sampled.

Detailed sampling of important fish species (Table 2) involves measurement of i) length (total, fork length; mm), ii) weight (round, dressed and gonad weight; g), iii) sex (male, female, unidentified), iv) maturity and v) feeding habits (stomach fullness and contents). Total length was measured from the tip of the snout to the tip of the caudal fin (caudal fin is compressed slightly vertically for maximum measurement). Fork length was measured from the tip of the snout to the fork in the caudal fin. Round weight was the weight of the entire fish and dressed weight was the weight of the gutted fish including the head. The gonad development stages refer to standardized terminology for describing reproduction of fish (Murua et al. 2003; Brown-Peterson et al. 2011; Lowerre-Barbieri et al. 2011). The code 0 was used when the sex was unknown. The codes of 1 to 5 were assigned to females and codes 6 to 10 for males for the respective developmental stages: immature, mature, running ripe, spent and resting. Their biological characters by visual examination are summarized in Table 3.

Statistical analysis

The one-way analysis of variance (ANOVA) is used to determine whether there are any statistically significant differences between the means of two or more independent (unrelated) groups. The depth strata were divided in 20 m groupings. It can be applied to examine if density

of observations of zooplankton, benthos and fish is different from stratified depth, levels of area, year, temperature or turbidity. If a significant difference has been identified, a post-hoc multiple comparison such as a Turkey multiple comparison or Bonferroni-corrected critical value, at a significance level of $p=0.05$, was used. Prior to ANOVA, significant outliers of bivariate variables were detected by use of a dotplot. Normality of dependent variables was diagnosed using skewness and kurtosis tests. If the dependent variables were not normally distributed, a logarithmic transformation was used. All statistical analyses were conducted using Stata 10 (www.stata.com).

For zooplankton ($n=167$) and benthic invertebrates ($n=151$), replicates collected from the same location were averaged to generate an estimate of the taxa density from each sample. The total density for each taxon was calculated to compare the overall composition of taxa. The density of each sample, and this data was used to calculate the mean (\pm SE) density in each depth category and area. Data from the western basin, areas IW and IE, was used to examine temporal variability over the study for zooplankton ($n=70$) and benthic invertebrates ($n=61$).

Population biology

Abundance or biomass indices varied by individual biological components under study. For zooplankton, density (individuals/m³) was calculated by dividing the total abundance of taxa in each sampling grid by the maximum depth (m) of the sample. For benthic invertebrates, the density (individuals/m²) was calculated by dividing the final count of each taxon by the area of the PONAR dredge used. Replicates collected from the same location were averaged to estimate the sample taxa density for zooplankton ($n=167$) and benthic invertebrates ($n=151$). The total density for each taxon was calculated to compare the overall composition of taxa. The sample taxa density was used to calculate the mean (\pm SE) density in each depth category and area. Data from the western basin, areas IW and IE, was used to examine temporal variability over the study for zooplankton ($n=70$) and benthic invertebrates ($n=61$).

For fish, all gillnet-based catch per unit effort (CPUE) were calculated as total individuals and weight caught per 24 h set, divided by total mesh-sized panel area, and reported as species-specific relative number (NPUE, individuals/m²) and biomass (BPUE, g/m²).

Fish community

Compared with other lacustrine habitats, information on fish species diversity in GSL is still rather limited. However, the importance of species can be evaluated by uses of the composition of each species to the total abundance or biomass. It was determined by use of an index of relative importance (IRI, Pianka 1975) to depict the position of the species composition,

$$\%IRI = \frac{(\%N_i + \%W_i) \times \%F_i}{\sum_{i=1}^n (\%N_i + \%W_i) \times \%F_i}$$

Where, $\%N_i$, $\%W_i$ and $\%F_i$ are the relative abundance, biomass and occurrence frequency used for each species (i) out of the total species richness (n), respectively. Three ranks of fish species,

dominant, regular and rare, were graded in term of %IRI as >10.0%, 1.0-10.0% and <1.0%, respectively (Zhu et al. 2008).

Before performing the gradient analyses, we made a few decisions about the inclusion of species and samples. To remove any undue effects of rare species on the ordination analyses (ter Braak 1995), species occurring as % F <5%, % N or % W <0.10% within the entire sampling system were excluded. Three ordination techniques, hierarchical cluster analysis (HCA), detrended correspondence analysis (DCA) and canonical correspondence analysis (CCA), were explored to examine the data, with the intent to highlight different aspects of fish community associations with pelagic and demersal (benthic) environmental conditions. Differing from k-mean clustering, the HCA builds a hierarchy from the bottom-up, and doesn't require specifying the number of clusters beforehand. HCA is a numerical, rather than a statistical procedure and no assumption of normality is required. Given a set of observations, we employed the HCA to partition site-based observations of six environmental variables into a dendrogram. The environmental matrix (site \times environmental variables) was standardized by use of generic function scale, which determines how column scaling is performed after median-based centering. We used the 'Ward' clustering method to classify spatial variation in environmental variables. The hierarchical tree obtained from the cluster analysis was split stepwise into an increasing number of cut-off levels, each time expanding the spatial organization patterns of the environmental variables by one. The procedure was stopped when there was no further increase in the 'indicator values' suggesting that no additional information was obtained by further subdividing the data set. Phylogenetic tree plot was used to express the spatial division patterns of environmental variation in terms of results of within-cluster sum-of-squares.

We used the DCA to investigate spatial variation of fish community structure. It is an indirect eigenvector ordination technique based on reciprocal averaging that corrects for the "arch effect" observed in correspondence analysis (Legendre and Legendre 1998). Primary gradients within fish communities are effectively displayed by DCA, and species turnover rates can be inferred by scaling the axes by using standard deviation units of sample scores, with 50% turnover in species composition occurring over approximately one standard deviation (Gausch 1982). We limited the DCA to data collected from late June through mid-August, the only months sampled over the years. The DCA determines hypothetical linear gradients that best explain the species composition in each sample, irrespective to any environmental variables. Therefore, the analysis deals explicitly with all biological components to separate the error variance from the explanatory variance. Sites were then ordinated via DCA axes whose length may be related to species turnover via units of standard deviation in compositional turnover (Hill and Gausch 1980).

Significance tests for CCA models were based on Monte Carlo permutation tests (2500 permutations) for the sum of all eigenvalues. The significance of relationships between the synthetic gradients and individual environmental variables were evaluated by t-tests for the inner-set correlations and canonical coefficients (ter Braak 1986, 1994). The weighted average

species scores were used in all CCA ordination plots and only the environmental variables that the inter-set correlation coefficient and canonical coefficient were significant at critical level $p < 0.05$ were included in the plots.

Three sets of data, including limnology, lower-trophic sampling and fish catch, were organized for integration of fishery-independent survey information. Separated by water columns (top, mid-water and bottom), two site-based matrixes were produced as environmental variable (site \times environment) and abundance (site \times species-specific abundance). The HCA and phylogenetic tree plot were conducted with packages “cluster” (<https://cran.r-project.org/web/packages/cluster/index.html>) and “ape” (<https://cran.r-project.org/web/packages/ape/index.html>) under R environment version 3.4 (www.r-project.org). DCA and CCA were carried out in CANOCO 4.5 (ter Braak and Verdonschot 1995; ter Braak and Smilauer, 1997).

RESULTS

Environmental variables

Over 2011–2016, 228 grids have been surveyed in GSL, including repeated sampling in the western basin areas IW and IE (Table 4). The annual number of grids sampled varied from 54 grids in 2016 to 17 grids in 2011, the first year of the survey when only one crew was conducting surveys (Fig. 2).

Water temperature

Water temperature varied between 3.05 °C in the bottom and 22.3 °C on the surface, with an overall average (\pm SE) of 10.03 ± 4.91 °C during the summer months. Combined with all years’ (2011–2016) observation, a two-way ANOVA demonstrated that water temperature differed significantly with management areas ($df=5$, $F=25.23$, $p < 0.0001$) and depth strata ($df=7$, $F=82.85$, $p < 0.0001$) without significant interaction between area and depth strata ($df=10$, $F=0.57$, $p=0.84$). Among management areas, average depth (\pm SE) was the shallowest in IW (9.96 ± 0.37 m) whilst the deepest in area IV (63.53 ± 1.61 m). Bonferroni’s multiple comparison resulted that the mean grid depths in areas IE (26.36 ± 0.55 m), III (24.22 ± 0.91 m) and V (35.57 ± 3.77 m) did not significantly differ ($p > 0.15$).

Correspondingly, the area-based mean temperature varied negatively with changes of grid depth ($n=1859$, $r=-0.41$, $p < 0.0001$). The average temperature at each depth was profiled by management areas (Fig. 3), demonstrating the presence of an evident thermocline structure at the depth of 10–15 m across all areas. Within the thermocline (change of temperature is $>2^\circ\text{C}$ per meter), the temperature reduces with depth more quickly than in the warm layers above (epilimnion) or cold layers below (hypolimnion). This stable thermal system continues in the water column throughout the summer months, and the position and thickness change with grid depth. As seen at area IW, the epilimnion-thermocline-hypolimnion boundaries were 9 and 12 m, with a 3 m thermocline layer. In area IE, the thickness of the thermocline was up to 4 m, with

temperature changes from 11 °C at 11 m deep to 6.7 °C at 13 m. In other off-shore deep-water areas, the stable hypolimnion layer <5 °C occurred at depths >20 m.

Constraining analysis to the surface environment only (the 5 m deep water column below the surface), a two-way ANOVA showed significant differences of temperature by management area ($df=5$, $F=6.84$, $p<0.0001$) and depth ($df=4$, $F=13.14$, $p<0.0001$) without a significant area and depth interaction ($df=19$, $F=1.21$, $p=0.24$). In the upper water column (0–5 m below the surface), mean temperature in area IW is the warmest (16.60 ± 0.23 °C) with a >3 °C difference among the six management areas by use of Bonferroni's multiple comparison test ($p<0.05$). There is no significant difference in temperature in the upper layer of the deep-water areas ($p>0.90$).

Using all field measurements at 1–5 m depths, spatial distribution of the surface temperature showed uniform patterns over most of the main basin of GSL (Fig. 4a). This reflects the well mixture of heat energy in this interface during the summer months. Two warm water masses appeared as a result of quick heating in the shallow waters of the western basin and the warm discharge from the Slave River at the mouth in the main basin. In the lake bottom, there was a mostly uniform thermal structure, uniquely characterized by combinations of depth and substrate (Fig. 4b). In the shallow areas IW and V east of the mouth of the Slave River, observed warm temperatures serve as thermal requirement for many important fishes in the lake.

Measurement of water temperature has been repeatedly conducted in the western basin over 2011-2016, displaying no significant variation by year ($df=5$, $F=0.75$, $p=0.58$), year and depth strata interaction ($df=7$, $F=0.67$, $p=0.70$), but showing significant difference with depth strata ($df=2$, $F=91.15$, $p<0.0001$). Bonferroni's multiple comparison revealed no significant difference in surface temperature was identified in area IW ($p>0.05$), except in 2012 versus 2015 ($p<0.05$). In area IE, a significant difference in surface temperature ($p>0.05$) was found over the years, as seen by warmer years in 2011, 2012 and 2016 (Fig. 5a). Bottom temperature (Fig. 5b), did not significantly differ by year ($df=5$, $F=1.01$, $p=0.41$) and interaction of year and area ($df=5$, $F=1.45$, $p=0.21$), but was different between the two areas ($df=1$, $F=54.85$, $p<0.001$), as area IW is shallow and bottom waters can occur within the warm epilimnion.

Dissolved oxygen

The dissolved oxygen (DO) concentrations varied in the range of 6.19–12.99 mg/l with a grand average (\pm SE) of 10.43 ± 1.00 mg/l ($n=1839$). Concentrations differed by area ($df=5$, $F=35.91$, $p<0.0001$), depth strata ($df=7$, $F=3.59$, $p<0.001$) and the interaction of area and strata ($df=10$, $F=1.93$, $p<0.05$). The spatial distribution of DO in the surface water tended to be a somewhat uniform pattern (Fig. 6a). Cooler, offshore waters (Fig 5a) had greater oxygen concentrations than the western basin and the Slave River output areas (Fig. 6b). The strata-based average DO showed the dissolved oxygen concentration seemed to be uniformly distributed with depth (Fig. 7).

Significant variations in dissolved oxygen by year ($df=5$, $F=26.67$, $p<0.0001$), depth strata

($df=2$, $F=4.30$, $p<0.05$) and year and depth interaction ($df=7$, $F=6.04$, $p<0.0001$) were identified by a two-way ANOVA (Fig. 8). In the surface water column (0–5 m), there were significant effects of year ($df=5$, $F=90.81$, $p<0.0001$), area ($df=1$, $F=13.29$, $p<0.0005$) and marginal interaction between year and area ($df=5$, $F=2.30$, $p<0.05$). In the waters deeper than 5 m, effects of year ($df=5$, $F=23.60$, $p<0.0001$) and year and area interaction ($df=5$, $F=4.34$, $p<0.001$), but no effect of area ($df=1$, $F=3.49$, $p=0.06$).

Turbidity

Turbidity, a measure of clarity of the water, varied between 0.1– 80 NTU with a grand average (\pm SE) of 7.89 ± 0.28 NTU. Two-way ANOVA showed no consistent trend across depth strata ($df=7$, $F=2.60$, $p<0.05$), management areas ($df=5$, $F=12.06$, $p<0.0001$) and strong depth strata and area interaction ($df=10$, $F=4.10$, $p<0.0001$). In the upper water column (0–5m), higher turbidity was observed near discharges from major tributaries, particularly the Slave River, while large clear water masses can be seen in the off-shore and deep waters of areas II and IV (Fig. 9a). In the bottom waters, turbid water mass can penetrate into the east part of area III and south part of area IV, indicating the considerable effects of discharge from the Slave River (Fig. 9b).

Relatively, western basin (area IW: 4.49 ± 0.49 NTU and area IE: 5.39 ± 0.23 NTU) has less turbid water than the other areas (II: 8.76 ± 1.04 NTU, IV: 6.14 ± 0.48 NTU, V: 11.16 ± 1.57 NTU). In Area III, it is relatively shallow but turbidity is the greatest (18.10 ± 1.12 NTU) because it is within the plume of the Slave River discharge (Fig. 10). Over 2011-2016, there was no significant difference in turbidity in the depth strata ($df=2$, $F=0.38$, $p=0.38$), but remarkable inter-annual variation in turbidity ($df=5$, $F=19.05$, $p<0.0001$) and strong year and depth strata interaction ($df=7$, $F=3.3$, $p<0.005$). Conspicuous higher turbidity can be seen in the western basin in 2011 and 2013 (Fig. 11), compared to 2012, 2014 and 2016.

Lower trophic components

Zooplankton

Zooplankton density significantly differed between taxa ($df=5,996$; $F=375$; $p<0.0001$) with copepods comprising 80% of the total density from all samples. The mean (\pm SE) density of calanoid copepods ($2\,572 \pm 210$ individuals/ m^3) was similar to cyclopoid copepods ($1\,927 \pm 231$ individuals/ m^3 ; Fig. 12a). Bosminids (632 ± 95 individuals/ m^3) were the most abundant cladoceran followed by Daphniidae (446 ± 73 individuals/ m^3). Calanoid and cyclopoid copepods were collected in every grid, while Bosminids and Daphniidae were found in 90% and 92%, respectively. Mysidae (0.5 ± 0.1 individuals/ m^3) were the least collected taxa (31% of grids).

Significant temporal variability was observed in the western basin (areas IW and IE) among years ($df=4$, 65; $F=20.77$; $p<0.0001$). Densities in 2013 ($12\,095 \pm 1\,826$ individuals/ m^3) and 2015 ($7\,818 \pm 1\,502$ individuals/ m^3) were significantly greater than in other years (Tukey HSD, $p<0.05$), and were approximately four and three times more dense, respectively, than the remaining years (Fig. 12d).

Zooplankton density significantly differed between areas ($df= 5,144$; $F= 2.39$; $p= 0.04$) and depth categories ($df= 3,144$; $F=17.96$; $p< 0.0001$) and had a non-significant interaction ($df= 14,144$; $F=1.73$; $p= 0.06$). Mean density was negatively correlated to depth ($df= 1000$, $r= -0.1901$, $p < 0.0001$) and shallow sites less than 20 m contained 78% of the total density from all samples (Fig. 12c). Samples collected in 0-10 m ($9\,150 \pm 1\,018$ individuals/m³) and 10-20 m ($6\,567 \pm 924$ individuals/m³) contained, on average, densities three and two times greater than samples collected from sites deeper than 20 m, respectively. Area V ($9\,229 \pm 1\,952$ individuals/m³), surveyed in 2015, had the greatest mean density of zooplankton but was only significantly different from area II (Tukey HSD $p<0.05$; Fig. 13b). In general, grids located near shorelines had greater densities of zooplankton compared to offshore survey locations (Fig. 12b). Grids with the greatest densities of zooplankton were found in all areas except area II.

Benthos

Densities were significantly different among taxa from all samples (Kruskal-Wallis rank sum, $df= 4$, $\chi^2=11.716$, $p= 0.0196$). The mean densities (\pm SE) of ostracods (599 ± 74 individuals/m²) and amphipods (551 ± 47 individuals/m²) were both larger than all the remaining taxa combined, and accounted for over 69% of the total density of benthic invertebrates (Fig. 13a). Oligochaetes (235 ± 30 individuals/m²), bivalves (119 ± 35 individuals/m²) and chironomids (82 ± 21 individuals/m²) were also commonly collected benthic fauna.

In the western basin, no significant difference in benthic invertebrate densities were found among years (ANOVA, $df= 4,56$, $F=3.30$, $p=0.15$). Densities ranged from the lowest in 2016 (742 ± 242 individuals/m²) to the greatest in 2013 ($2\,113 \pm 435$ individuals/m²).

The greatest variation in benthic invertebrate density was found among areas ($df=5,131$, $F=7.65$, $p<0.0001$). Areas IV ($2\,792 \pm 348$ individuals/m²) and V ($3\,312 \pm 1\,051$ individuals/m²) had significantly greater (Tukey HSD $p<0.05$) densities than areas IE ($1\,122 \pm 129$ individuals/m²), II (893 ± 124 individuals/m²) and III ($1\,082 \pm 191$ individual/m²). More invertebrates were collected from sites <10 m deep ($2\,589 \pm 553$ individuals/m²) than the deeper sites, however, depth was not a significant factor in density ($df= 3,131$, $F=2.52$, $p= 0.06$), but did have an interaction effect with area ($df=11,131$, $F=1.86$, $p= 0.05$). The greater densities of benthic invertebrates in areas IV and V may be partially due to the soft substrates favorable for sampling by benthic dredges compared to hard or sandy substrates. PONAR devices penetrated deeper into the sediments, collecting a large volume of substrates to process for invertebrates.

Combined with area-based analyses above, spatial distributions of the density of low trophic components (zooplankton and benthos) showed no consistent patterns with depth strata (Fig. 14). However, the higher density of zooplankton appeared in the nearshore or 20 m shallow areas, whereas relatively lower densities were found in the deep waters (Fig. 14a). Spatial distribution of benthic invertebrates showed higher density in the deep water especially in the Yellowknife Bay (area IV), dominated by Ostracoda and Amphipoda (Fig. 14b).

Fish community diversity

Two types of experimental gillnets were used for the quantitative sampling studies: pelagic and benthic nets. During June–August, 2011–2016, numbers of overall ($n=392$) and effective ($n=387$) net sets are summarized in Table 5. Less than 2% of the total settings were excluded from the following analysis because of longer deployment times caused by severe weather conditions. Among all effective sampling sets, a total of 17–46 sets were deployed annually in the western basin, whereas the numbers of sets varied by areas in terms of the depth-stratified random sampling protocol (Zhu et al. 2011).

Species richness

A total of 24 fish species were recorded from the depth-stratified random sampling in the main basin of the GSL (Fig 15). All species except for Lake Chub were collected in benthic gillnets (Table 6). In pelagic gillnets, 17 species (71%) were collected in the surface nets (5 or 12 m deep), compared with 8 (33%) and 7 (29%) species collected in mid-water nets 20 m and 30 m deep, respectively. Six fish species (25%), Arctic Grayling, Arctic Lamprey, Common Dace, Slimy Sculpin, Sauger and Walleye, were found only in the benthic gillnets and six fish species (25%), Burbot, Least Cisco, Lake Herring, Lake Trout, Lake Whitefish and Shortjaw Cisco, appeared in the whole water column. So, most fish species can disperse in the upper or bottom GSL environments during summers.

Catch by area and setting

A total of 42,312 individual fish with a combined weight of 13,024 kg were collected. More than 47% of individuals were from area IE, followed by III (20%), II, IW and IV (~10%), whereas individuals from area V were less than 4% (Fig. 16a). In terms of weight, the catch percentage in area IE was down to 41%, followed by III (20%), IW and IV (13%), II (9%) and V (4%). This may convey information on size differences by fish species over spatial areas (Fig. 16b).

In addition to the effects of spatial areas, total catches differed by management areas and types of settings (gillnet type and depth; Fig. 16c). Individuals comprising 52% of the total catch abundance came from pelagic sets whereas catch from bottom and mid-water sets took up 42% and 8%, respectively. Area IE had the greatest total catch compared to the other areas, and catch from pelagic and benthic nets comprised 28% and 18%, respectively. According to total weight, greater than 58% came from bottom sets whereas 39% were from pelagic sets. Eliminating confounding effects of the numbers of different sets, the overall average catch in number (CNPS) and weight (CWPS) per set were 109 individuals and 34.08 kg, respectively (Fig. 16). The highest area-based mean CNPS of 224 individuals/set was seen in mid-water of 20 m in area III. In the upper water, greater catch rate appeared in areas IE (164 individuals/set) and III (189 individuals/set). In the bottom, CNPS seemed to decrease along a west to east direction (IW: 131 individuals/set, IE: 113 individuals/set, III: 90 individuals/set, V: 72 individuals/set, Fig. 16c). Through water columns sampled, the greatest mean CWPS was found in the bottom (40.52 kg/set), while decreased from 30.75 kg/set in the surface, 16.32 kg/set in 20-m mid-water and 5.15 kg/set in 30-m mid-water settings (Fig. 16d). Horizontally, among areas in the

southern GSL, CWPS in the surface increased from 22.79 kg/set in area IW, 32.77 kg/set in area IE to 50.27 kg/set in area III. Among the deep-water areas, a similar spatial trend can be seen, along a southwest (14.89 kg/set in area II) to northeast gradient (22.23 kg/set in IV and 32.10 kg/set in area V). In the bottom environment, a steady decrease of CWPS along southwest to east was found both in the shallow-water areas, from 54.04 kg/set in area IW, 42.86 kg/set in area IE, 41.12 kg/set in area III, as well as in deep-water areas of 33.68 kg/set in area II, 31.57 kg/set in area IV and 24.39 kg/set in area V. There was no consistent trend of CWPS in the mid-water sets.

Density, abundance and biomass

We applied the index of relative importance (IRI) to rank species-specific contributions to the quantitative survey (Table 7). The ranks of 24 fishes differed by water columns. In the upper water environment, three coregonids, Least Cisco (28%), Lake Herring (27%) and Lake Whitefish (26%), are considered as dominants. Inconnu (6%), Longnose Sucker (5%), Burbot (4%) and Lake Trout (3%) belong to regular components, and the remaining 17 species rarely appeared in the upper waters. In middle waters, three coregonids were dominant, Least Cisco (38%), Lake Whitefish (33%) and Lake Herring (20%), followed by three regular species, Lake Trout (4%), Burbot (2%) and Shortjaw Cisco (1%). The bottom environment was predominantly Lake Whitefish (52%), followed by Burbot (15%) and Lake Herring (15%), as a dominant component. Five fishes, Least Cisco (9%), Longnose Sucker (4%), Northern Pike (2%), Troutperch (1%) and Shortjaw Cisco (1%), are regulars. Combining all three sampled depth strata, three coregonids, Lake Whitefish (38%), Lake Herring (20%) and Least Cisco (19%) comprised the dominants, in association with five regulars, Burbot (10%), Longnose Sucker (5%), Inconnu (3%), Lake Trout (2%) and Northern Pike (1%). Thus, we can suggest that ecological characteristics of the fish community in GSL are controlled by moderate- (whitefish) and small-sized (Cisco) coregonids, which interact with large-sized predacious fishes (Burbot, Inconnu, Lake Trout, Northern Pike) as regulars.

To remove effects from the number of sets, soak duration of nets, and catching area of individual gears, we evaluated the relative importance of all species by use of two relative indices, NPUE (individuals/m²) and BPUE (g/m²). NPUE ranged from 0.001 to 2.38 individuals/m² with a grand average of 0.28 ± 0.02 individuals/m² ($n=254$). Using a two-way ANOVA, we found a significant difference in NPUE over six management areas ($df=5$, $F=3.64$, $p<0.005$), but no difference through three types of setting (top, mid-water and bottom; $df=2$, $F=2.34$, $p=0.10$) and area and setting interaction ($df=9$, $F=1.29$, $p=0.24$). Among the six management areas, the greatest average NPUE appeared in the shallow areas of the southern GSL (IW: 0.34 ± 0.05 , IE: 0.40 ± 0.04 and III: 0.37 ± 0.05 individuals/m²) than those in deep-water areas (II: 0.22 ± 0.02 , IV: 0.16 ± 0.02 and V: 0.20 ± 0.06 individuals/m²). In particular, NPUE in area IV was considerably lower than those in the southern shallow areas ($p<0.05$) revealed by use of Bonferroni's multiple comparison. BPUE varied in a range of 0.05 g/m² to 441.90 g/m² with a grand average of 65.62 ± 4.04 g/m² ($n=254$). A two-way ANOVA revealed the significant

difference in BPUE by types of settings ($df=2$, $F=27.79$, $p<0.001$), but no significant difference among areas ($df=5$, $F=1.07$, $p=0.38$) or area and water column interactions ($df=9$, $F=0.91$, $p=0.52$). Bonferroni's multiple comparison indicated that remarkably higher BPUE emerged in the bottom ($p<0.0001$) than those in upper ($p=0.11$) and middle ($p=0.26$) water columns. No outstanding BPUE difference was identified from upper ($p=0.11$) and mid-water settings ($p=0.26$). Our analyses that no significant spatial biomass differences were observed further confirm that fishes utilize different habitats for foraging resources and rapid growth during their growing season (Kennedy 1953, 1954; Healey 1975).

Compared with upper (top) and bottom (bot) settings, the spatial distributions of the NPUE and BPUE for all fish species combined are shown in Fig. 17. Higher values of both indices appeared in the southern nearshore grids, compared to relatively lower values from grids in deeper water. Area IW seemed to support higher BPUE than the other areas. One-way ANOVAs indicated a significant difference for NPUE among areas in the bottom ($df=5$, $F=3.42$, $p<0.01$) but not the top ($df=5$, $F=1.7$, $p=0.14$). In the upper water column, the area-based average NPUE varied between 0.10 ± 0.05 individuals/m² in IW to 0.38 ± 0.09 individuals/m² in III, with grand average of 0.24 ± 0.03 individuals/m² (Fig. 18). In the bottom, lowest and highest NPUE were 0.20 ± 0.03 individuals/m² in IV and 0.48 ± 0.09 individuals/m² in IW with grand average of 0.32 ± 0.03 individuals/m², significantly varying with depth ($df=4$, $F=3.38$, $p<0.05$). The ratio of mean NPUE values between top and bottom layers was 0.75, showing a marginal difference through the entire water columns ($df=1$, $F=4.66$, $p<0.05$). With regard to BPUE, no area-based difference was found both in the top ($df=5$, $F=1.09$, $p=0.37$), ranging between $0.16\text{--}222.21$ g/m² with a grand average of 39.50 ± 3.90 g/m², and in the bottom ($df=5$, $F=0.83$, $p=0.53$) varying in the range of $0.32\text{--}441.90$ g/m² with a grand average of 100.03 ± 6.51 g/m². The ratio of mean BPUE between top and bottom layers was 0.39. BPUE in the bottom was more than 2.5-times greater than the top. One-way ANOVA showed significant effects of grid depth on the distribution of BPUE ($df=4$, $F=4.17$, $p<0.005$). As revealed by use of Bonferroni's multiple comparison, the highest BPUE appeared in the shallow waters (<30 m) with the average of 90.83 ± 6.85 g/m², followed by 49.02 ± 4.81 g/m², 39.45 ± 5.34 g/m², 11.78 ± 5.37 g/m² and 9.13 ± 4.25 g/m² in the depth groups of 30-60 m, 60-90 m, 90-120 m and >120 m, respectively. Thus, BPUE for all fishes tended to gradually decrease with lake depth ($n=119$, $r=-0.45$, $p<0.0001$) to the bottom of GSL.

Over a period of 2011-2016, higher NPUE and BPUE occurred in 2012, showing a tendency of inter-annual variation (Fig. 18). In area IW, the lowest and highest NPUE were 0.13 individuals/m² and 0.53 individuals/m², leading to more than a 4-times change over the years, but was statistically not significant ($df=5$, $F=1.00$, $p=0.44$). In area IE, NPUE varied in a range of 0.21 individuals/m² and 0.96 individuals/m² with more than a 4-times change over the years, which was statistically different ($df=5$, $F=11.54$, $p<0.0001$). Remarkable inter-annual variation of BPUE in area IW was detected with changes from 60.36 g/m² in 2014 to 193.95 g/m² in 2012, but was not statistically significant ($df=5$, $F=1.32$, $p=0.28$). In area IE, minimum and maximum BPUE were 50.81 g/m² in 2015 and 164.91 g/m² in 2012, which a significant

difference in annual BPUE was seen ($df=5$, $F=6.78$, $p<0.0001$).

Lake Whitefish

Lake Whitefish is the most important fish in GSL, sustaining the largest highly-valued commercial and subsistence uses in the Northwest Territories. With regards to NPUE, no considerable difference with area ($df=5$, $F=1.13$, $p=0.35$) or depth strata ($df=7$, $F=0.31$, $p=0.95$) was identified, showing uniform distribution of the fish abundance (Fig. 19). In the top, NPUE varied in the range of 0–0.34 individuals/m² with a grand average of 0.02 ± 0.01 individuals/m², whereas in the bottom it ranged between 0–0.91 individuals/m² with an average of 0.08 ± 0.01 individuals/m² (Fig. 20a and c). Compared Lake Whitefish NPUE by three types of settings, a significant difference was found between pelagic and benthic settings ($df=1$, $F=31.77$, $p<0.0001$). Lake Whitefish BPUE distribution seemed to be rather uniform without significant area-based differences in upper ($df=5$, $F=1.2$, $p=0.32$), mid ($df=4$, $F=0.51$, $p=0.73$) and bottom sets ($df=5$, $F=1.4$, $p=0.23$). The mean values of BPUE were 4.5-times greater in the top (9.64 ± 1.19 g/m²) than the bottom (43.60 ± 2.66 g/m²; Fig. 20 b). Moreover, one-way ANOVA revealed that BPUE differed in depth strata ($df=7$, $F=2.87$, $p<0.01$) with the greatest BPUE (39.99 ± 6.53 g/m²) in depths of 20–40 m. Bonferroni's multiple comparison test showed significant difference ($p<0.005$) in depth-dependent BPUE of fish collected in depth strata one, 19.98 ± 2.09 g/m² (<20 m) versus two, 39.99 ± 6.53 g/m², (20–40 m). At depth strata three (40–60m) and four (60–80m), the average of the BPUE was 33.12 ± 4.66 g/m² and 31.54 ± 8.10 g/m², without significant difference ($p>0.05$). The BPUE briskly dropped to 5.70 ± 1.76 g/m² ($n=6$) after 80 m. During 2011–2016, Lake Whitefish NPUE in area IW differed by year ($df=5$, $F=3.49$, $p<0.05$). It reached the highest level of 0.17 ± 0.05 individuals/m² in 2012 (Fig. 20e), compared with the rest of the years (NPUE<0.07 individuals/m²). There was a similar year effect to Lake Whitefish NPUE in area IE ($df=5$, $F=8.17$, $p<0.0001$), with the highest NPUE in 2012 (0.16 ± 0.04 individuals/m²). BPUE of Lake Whitefish differed significantly by year ($df=5$, $F=3.84$, $p<0.005$), but no difference was detected in area ($df=1$, $F=0.33$, $p=0.57$). The greatest average value of Lake Whitefish BPUE was found in 2012, which was significantly different from 2013–2016 values (difference>30g/m², $p<0.05$; Fig. 20f).

Lake Trout and Inconnu

Lake Trout used to be a target species for GSL commercial fisheries until 1972 when the fish stock could not tolerate a higher intensity of commercial gillnet fishing. Inconnu has long been a by-catch species and has recently become concerned with the stock status as a result of higher international demands. In the present research, we considered Lake Trout and Inconnu as regulars among GSL fish community members (Table 7). For Lake Trout, the spatial distribution for the BPUE indicated that the presence was constrained to areas of clear water, away from high turbidity areas near the mouth of the Slave River (Fig. 21). The BPUE in the upper water ranged from 0–39.57 g/m² with an average of 3.92 ± 0.75 g/m² ($n=107$), which was similar to the range (0–78.57 g/m²) and average (3.91 ± 1.11 g/m²; $n=119$) in the bottom (Fig. 22 a, b). No significant difference of BPUE was detected by depth strata ($df=7$, $F=0.23$, $p=0.98$).

Inconnu were collected over the south and east part of the main basin of GSL, and more than doubled BPUE (7.39 ± 1.52 g/m²; $n=107$) and appeared more in the top sets than bottom sets (3.42 ± 1.43 g/m², $n=119$; Fig 22 d and e). There was a marginal difference in setting-based BPUE ($df=1$, $F=3.64$, $p=0.06$). Among sites ($n=58$) Inconnu was detected (NPUE or BPUE > 0), at the depth ranged of 4-39 m with an average of 9.01 ± 1.02 m, but the BPUE seemed significantly positively related to setting depth ($r=0.33$, $F=6.89$, $p<0.05$). This mentioned that Inconnu can expand into as deep as 40 m during the summer season.

Temporal variation was detected by examining BPUE over the western basin of GSL (Fig. 22 c and f). One-way ANOVAs showed no considerable BPUE difference by year ($df=5$, $F=1.83$, $p=0.11$) and settings ($df=2$, $F=1.33$, $p=0.27$) for Lake Trout, as well as by year ($df=5$, $F=0.63$, $p=0.67$) and settings ($df=2$, $F=1.87$, $p=0.16$) for Inconnu. These results suggest that both fishes have stable populations.

Fish community and environmental associations

Six environmental variables, depth, temperature, dissolved oxygen, pH, turbidity and conductivity, have been included for multivariate cluster analysis (CA) and canonical correspondence analysis (CCA). CA for environment variables showed the presence of three clusters for pelagic environment and four clusters for demersal environment (Fig. 23).

The GSL sites varied widely in water quality and habitat characteristics (Table 8). The pelagic environment ($n=108$) can be separated into three spatial clusters (Table 7). Under unique thermal conditions, the majority of sites ($89/108=82\%$) belonged to offshore site Cluster A, characterized by the greatest depth (38.32 ± 2.86 m), coldest temperature (12.16 ± 0.34 °C) and lowest turbidity (2.25 ± 0.28 NTU). Within Cluster B, it covered nearshore sites with shallow average depth (13.93 ± 2.03 m) and an intermediate level of turbidity (19.87 ± 1.57 NTU). Cluster C included 11 sites in offshore (36.15 ± 7.52 m), warm (14.51 ± 0.54 °C), low pH (8.08 ± 0.19) and high turbidity (42.36 ± 3.11 NTU) waters. With regards to the benthic environment, Cluster A contained 35 nearshore sites (9.96 ± 0.94 m) with warmer (13.75 ± 0.61 °C), lower dissolved oxygen concentrations (9.27 ± 0.22 mg/l) and less turbid waters (11.89 ± 2.62 NTU). Compared with nearshore clusters, there were two site clusters, B and C, located in deeper and colder waters. Cluster B encompassed deep water sites in relatively clearer waters (7.26 ± 0.65 NTU), compared with the rather turbid water mass ($40.41 \pm$ NTU) in Cluster C.

Using forward selection, the single variable giving the highest eigenvalue (λ) CCA for intra-set correlations of the six environmental variables is selected by the class variable depth in the pelagic environment and temperature in the benthic environment (Table 9). Hereafter, all remaining environmental variables are ranked on the basis of the fit that each separate variable gives in conjunction with the variables of conditional effects. The statistical significance of the effect of each variable is tested by a Monte Carlo permutation test. Thus, in the pelagic environment all variables except pH showed significant conditional effects with extra fit of comparable magnitude. In the benthic environment, depth and turbidity have somewhat higher eigenvalues with statistically significant additional effects ($p<0.001$). As judged by the Monte

Carlo test, dissolved oxygen and conductivity contribute non-significant additional effects ($p < 0.05$).

The first two canonical axes of ordination analysis accounted for 12.30% of the variance in pelagic fish species composition and 72% of the environment-species variance. Sample site groupings, determined using cluster analysis, were superimposed to reflect the spatial pattern corresponding to the association of fish components on both CCA axes (Fig. 24a). Of 17 fish found in the pelagic environments, three fish, Troutperch, Spottail Shiner and Lake Chub, were most commonly collected in shallow, warm waters, while Ninespine Stickleback preferred deep, high turbidity areas. The environmental associations of the majority of fish species in the pelagic habitat were divided into shallow or deep habitats.

Of the 22 fish collected in the benthic environment, the first two canonical axes of the ordination analysis accounted for 24.10% of the variance in species composition and 84% of the environment-BPUE variance (Table 8). When sample site clusters overlapped, three habitat clusters described the spatial distributions of individual fish assemblages (Fig. 24b). The shallow warm waters contained most prey species (Spottail Shiner, Common Dace, Northern Pearl Dace, Ninespine Stickleback) as well as Arctic Grayling, Goldeye, White Sucker, Walleye and Northern Pike. In deep and clear waters, Arctic Sculpin and Shorthorn Sculpin were found in the deepest sites and Shortjaw Cisco, Burbot, Lake Trout, Lake Whitefish and Lake Herring were found in the mid to deep waters. Cold and shallow waters contained Troutperch, Least Cisco, Inconnu, Longnose Sucker, Round Whitefish and Sauger (Fig. 24b). Overall, fish BPUE composition was significantly different between spatial divisions (clusters) of environmental variables and several species were identified as diagnostic species of each cluster.

DISCUSSION

The results of this six-year project improve our understanding of the cumulative impacts on the GSL ecosystem, particularly in 1) lake-river interaction, 2) fish communities and environmental associations, 3) correspondence of fish communities to changing aquatic environment conditions and 4) implications for resource management.

Limnological environment and river-lake interaction

In this study, our field survey covered a bathymetric range from 2.7 to 150 m across six management areas in the main basin of GSL, from 2011–2016. During the survey period, significant differences in environmental variables, such as depth, temperature and turbidity, were identified in surface waters (0–5 m) across the lake by multivariate statistical analyses, but little to no variation was seen at depths 20 m or more. Spatial variations during summer time were presumably related to multiple hydrological events and dynamic processes, such as seasonal water level regulations (Gibson et al. 2006a, b; Prowse et al. 2006), river-lake catchment interactions (Anderson et al. 2002) and seasonal patterns in the localized physical environment (Rawson 1950). Among environmental variables, turbidity, an indicator of trophic condition, had large spatial variation in horizontal expansion of the turbid water mass. This might correspond to

the period from mid-July to early August when high volumes of riverine discharge enter GSL, predominately via the Slave River, which is responsible for 77% of the total inputs to the lake (Gibson et al. 2006). The resulting discharge may be caught and diluted in the Slave River Delta and adjacent waters. This localized hydrodynamic event has been recognized to profoundly impact the spatiotemporal dynamics of the physical limnology, low-trophic biological productivity, and fish and fisheries (Rawson 1950; Gibson et al. 2006a; Lehodey et al. 2006; Zhu et al. 2016).

Fish community diversity and environmental associations

Combined with the information on limnology and fish community descriptors, we applied multivariate statistics to examine what structures the fish community and how environmental conditions impact the stability of the fish community in GSL. Grid depth was selected as the most important physical parameter, and separates the fish community into shallow and deep habitats, associated with negative relationship with water temperature. Fish preferences for cool (deep) or warm (nearshore, surface) waters were apparent. Turbidity seemed to impose the success of prey-predator match-mismatch pairs of typically visual predators, like Lake Trout, Walleye and Northern Pike (predators) versus Lake Herring and Least Cisco (preys).

In the shallow, turbid, nearshore waters, catchment and influx of nutrients and quick lower trophic biological production are closely linked to discharges from the Slave, Little Buffalo, Buffalo and Hay rivers. Turbidity is an important abiotic indicator to probe the trophic status, especially in the pelagic environment. The direct effect of turbidity-driven shallow environments can be linked to the food web structured by a vector of plankton → cisco → piscivore, such as Lake Trout and Inconnu. Similarly, the effect of turbidity and light intensity on prey-predator interaction has been documented for planktivorous Smelt (*Osmerus eperlanus*) and Phantom Midge (*Chaoborus flavicans*), indicating smelt-mediated mortality of midge larvae was the highest at intermediate turbidity (20 NTU), and turbidity exceeding 30 NTU combined with lower light intensity may provide an efficient daytime refuge for midge even in the presence of predacious Smelt (Horppila et al. 2004). A group of small-sized fish, Goldeye, Lake Chub, Troutperch and Spottail Shiner, living in turbid shallow water, seem to be an example of refuge-shelter selection.

Lake Whitefish was identified as the most important dominant fish species, especially based on biomass per unit effort (BPUE) composition. CCA found grid depth and turbidity seemed to jointly impact the spatial distribution of Lake Whitefish catch per unit effort (NPUE). Size-dependent distribution was evident in Lake Whitefish, with small whitefish being most abundant in shallow turbid habitats and large whitefish collected in deep, cold habitats (Qadri 1961).

Response of the fish community to changing environments

Within particular bathymetric environments during the summer season, changes in depth-dependent temperature seemed to be an important driver of predator-prey relationships between fish in the hypolimnion. The direct impacts of temperature can be ascribed to production of prey

resources as a bottom-up control and the physiological requirements of predators as top-down regulation, trading off trophic cascade effects along food web pathways. In a laboratory experiment on the effect of temperature on community structure of a three trophic level food chain, Kishi et al. (2005) suggested those trophic cascading effects varied with temperature even in the presence of abundant predators. In the Arctic deep-sea environment of Davis Strait and Baffin Island region, the majority of fish species were found to distribute along environmental gradients from the warmer water temperatures in the lower latitudes to cooler temperatures in higher latitudes (Chamber and Dick 2007). Consequently, changing thermal regimes will alter the productivity of the aquatic ecosystem (ACIA 2005) and contribute to species distribution shifts in the fish community (Perry et al. 2005). At present, we have insufficient data to determine the ecological consequences of changing temperature regimes on the structure and functions of the GSL fish community. Continued field observations, long-term research, and model-based simulations will be necessary to examine the possible responses of fisheries and fish communities to changing climate and land-lake interactions.

Resource management implications

This study will substantially improve scientific advice and options available to decision-makers of co-management governance. Since 2006, DFO has incorporated a precautionary approach (PA) decision making framework when taking fisheries population dynamics and uncertainties into consideration. As for the existence of information, knowledge and research gaps, this six-year study, after Dr. Donald Rawson's pioneer study in the 1940s, is the first comprehensive study of Great Slave Lake. Incorporated with national and international freshwater fisheries monitoring protocols, this research project was strategically designed to be a standardized monitoring framework applied in the northern great lake. As the fishery-independent activities continue over years, the accumulation of relative abundance and biomass of multispecies, age structure matrix and total harvest for commercial, recreational and aboriginal uses, can be integrated into a quantitative fisheries stock assessment. Aligning with DFO mandates and the PA framework, the overarching objectives of this project can be particularly empowered to help formulate a set of biological reference points (BRPs) including maximum sustainable yield (MSY), total allowable catches (TAC) and overfishing limits (OLs). Therefore, it will greatly enhance and improve decision-making processes for monitoring and managing Great Slave Lake fish and fisheries. Applying the protocols and frameworks developed in this study for long-term monitoring activities will benefit future decision-making processes addressing concerns of 1) relationship between the sustainability of fisheries and ongoing biological productivity, which will be estimated in terms of quantitative surveys of multi-trophic species-specific abundance and biomass, as well as fishing efforts reflecting the exploitation intensity; 2) influencing formulation and modification of area-based fishing quotas and numbers of fishing licenses for managing commercial fisheries; and 3) determination of conservation and protection of habitat uses by sensitive fish species, like closure zones for Inconnu in southern GSL or Short-jaw Cisco (*Coregonus zenithicus*).

It can contribute to future decision-making processes for the establishment of fisheries management frameworks, such as an integrated fisheries management plan (IFMP) and an ecosystem-based fisheries management (EBFM). In order to implement these ambitious and promising frameworks, it is necessary to combine commercial, recreational and aboriginal (CRA) fisheries data with data-intensive ecosystem research to ensure the sustainability of fisheries and healthy aquatic ecosystems.

CONCLUSION AND RECOMMENDATIONS

In the main basin of Great Slave Lake, a total of 245 grids, evenly spaced in terms of longitude and latitude, were configured for the depth-stratified multidisciplinary study. Between 2011–2016, summer time field research was conducted collecting the ecosystem information on limnology, zooplankton, benthic invertebrates and fish in 226 grids, which covers all management areas and repeated annual sampling in the western basin. Throughout the summer months, temperature showed consistent patterns in the horizontal scales but remarkable variation was identified in the depth and thickness of the thermocline. There was significant variation in turbidity as a result of strong river-lake interactions, impacting bottom-up biological production.

Zooplankton density, dominated by copepods, significantly differed between years, areas and depth categories. Mean density was negatively correlated to depth and shallow sites less than 20 m contained 78% of the total density from all samples. The greatest variation in benthic invertebrate density, dominated by ostracods and amphipods, was found among areas, but no significant temporal variation among years.

A total of 24 fish species were identified, dominated by three coregonids (Lake Whitefish, Least Cisco and Lake Herring). Multivariate statistical analysis indicated that the stability and functionality of the fish community are more or less associated with changes of depth, temperature and turbidity.

Through this multi-year field survey and data analyses, we recommended,

- ***Consideration of the spatial distribution patterns of target fish populations***

Despite wide spread abundance, Lake Whitefish are a typical benthivore that largely reside in the bottom environment and primarily consume benthic resources. During the summer months, abundant immature Lake Whitefish approach nearshore areas for feeding and potential predator avoidance, while adult fish reside in deeper water below the epilimnion. This is helpful information for resource management staff to monitor the activities of commercial harvests within fishing quota limits and the recruitment of juveniles into the adult population.

- ***Incorporation of temporal variation of fish population production dynamics***

Our results showed no significant temporal variation of Lake Whitefish abundance and biomass in the western basin of GSL. Analysis of the GSL fish plant sampling program dating

back to the 1970s found that a strong cohort of Lake Whitefish recruits can persist for more than 10 years in the population and therefore might not be captured in the 6 years of our study. Other reason regarding unclear patterns in inter-annual variations in Lake Whitefish abundance is resulted from low commercial fisheries harvest since 2000, compared to historical levels of GSL Lake Whitefish fisheries. Information on temporal variation of target fish populations is critically important to be incorporated into adaptive fisheries management plans by timely adjusting commercial catch quotas in response to adult abundance and cohort recruitment. Combined with those, long-term monitoring of multispecies population dynamics should be supported to inform sustainable fisheries management.

- ***Management strategy should combine with environmental regime shifts and interaction of multispecies***

In terms of the six years studied, we found that GSL environmental conditions are experiencing dynamic variations driven by cumulative impacts including water regulation, climate change, habitat loss from mining, and exploitation. Variation in the concentrations of dissolved oxygen in the hypolimnion might reflect dynamic changes of supplies-demands relationships for aquatic organisms, especially predacious fishes, in the deep-water. Our analysis has also identified turbidity as an important environmental variable impacting the distribution of biological components including zooplankton, benthos and fish. Despite plumes of turbid water in the local areas, rivers play an ecological role in suppling river-oriented nutrients, transport warm water masses and function to be detrimental to typical visual predators. As a result of changing GSL environments, the overall effects are augmented and exerted on quantities and quality of supporting fish habitats in GSL during summers. Subsequently, not only does it modify low-trophic biological production, but it drives changes of growth, reproduction and mortality rates of animal populations through physiological regulations. Therefore, we recommend future fisheries management strategies should be combine with 1) multidisciplinary information collected from our 245-grid geographic system, 2) interaction of multiple species structuring and maintaining the stability of the fish community by sharing trophic linkages and habitat connections and 3) food-web information delineating predator-prey relationships and systematic responses to changing hydroclimate and trophic regimes as well as harvest strategies.

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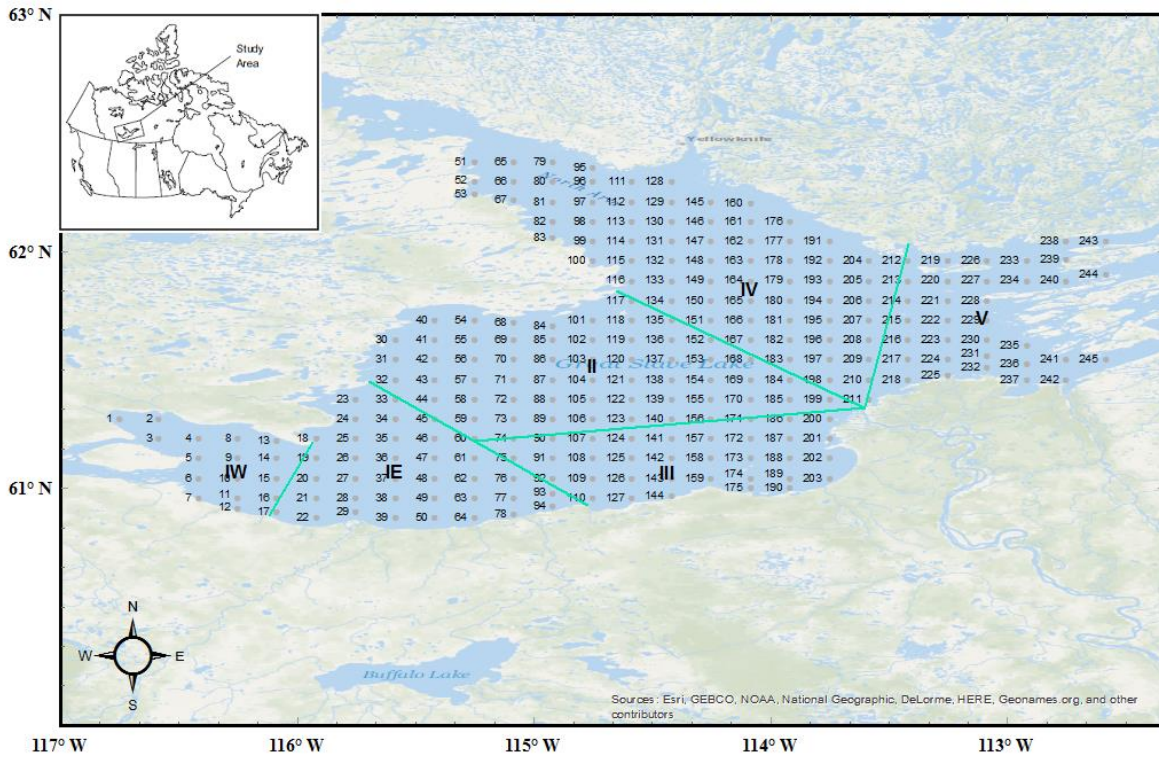


Fig. 1 Map of Great Slave Lake showing location (inset), management areas and sampling grids used for summer research between 2011-2016.

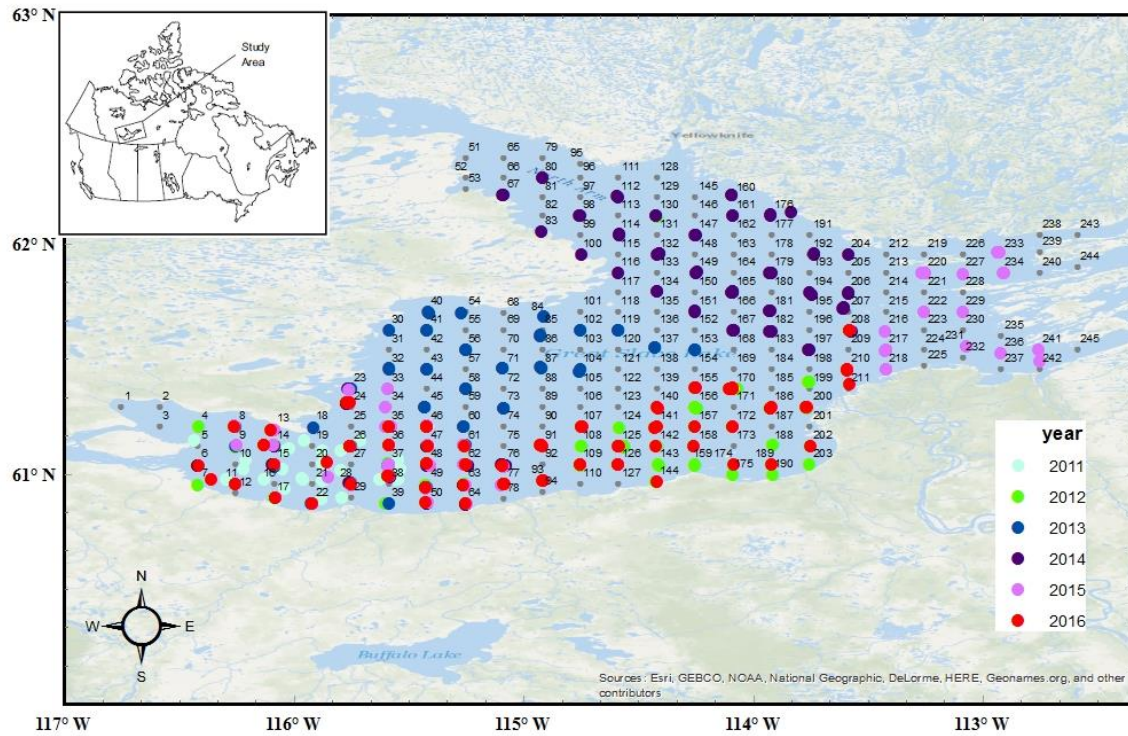


Fig. 2 Annual sampling grids for integrated fisheries ecosystem surveys in the main basin of Great Slave Lake, summers of 2011-2016

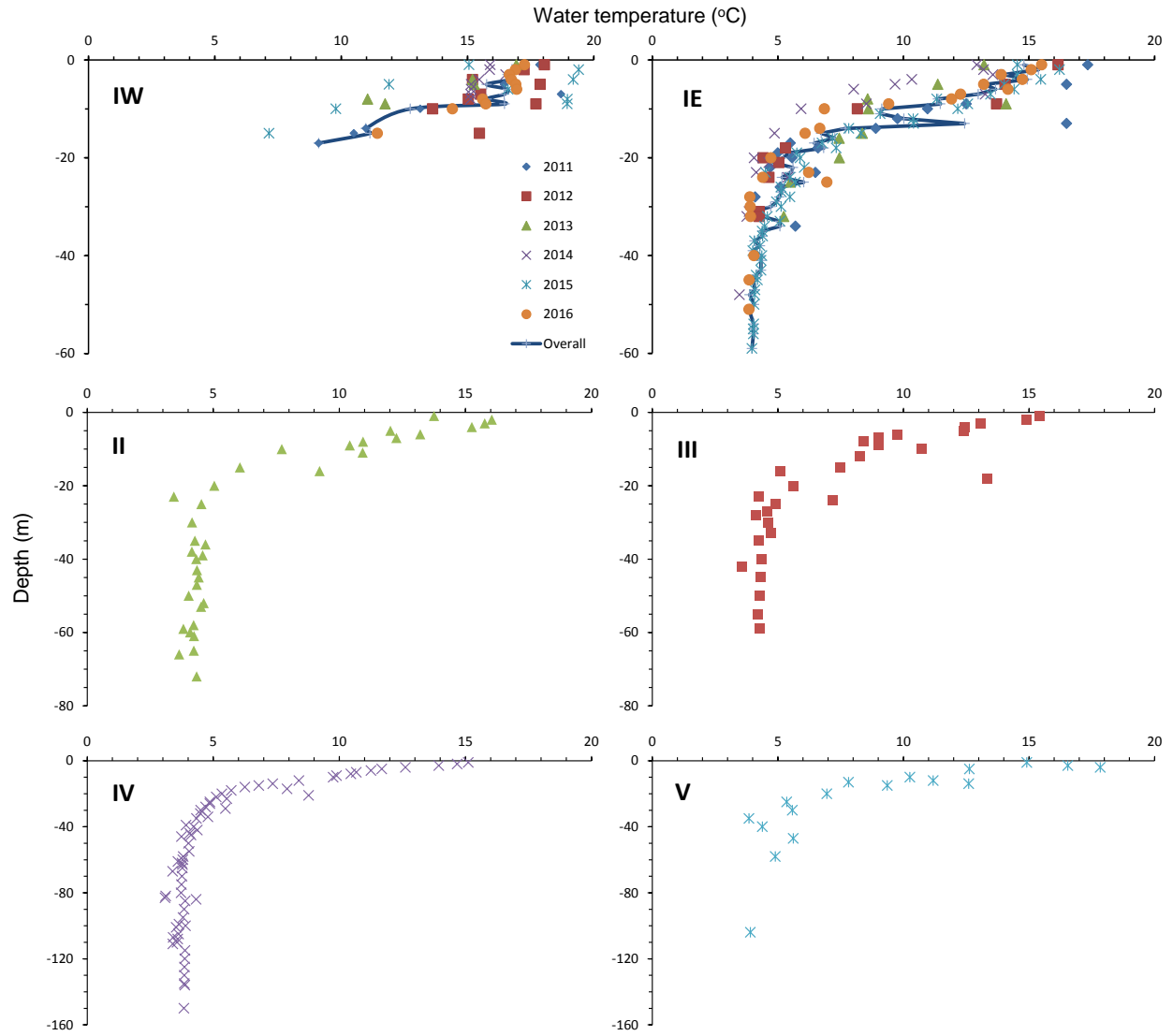


Fig. 3 Profile of average water temperature in management areas IW to V, 2011-2016.

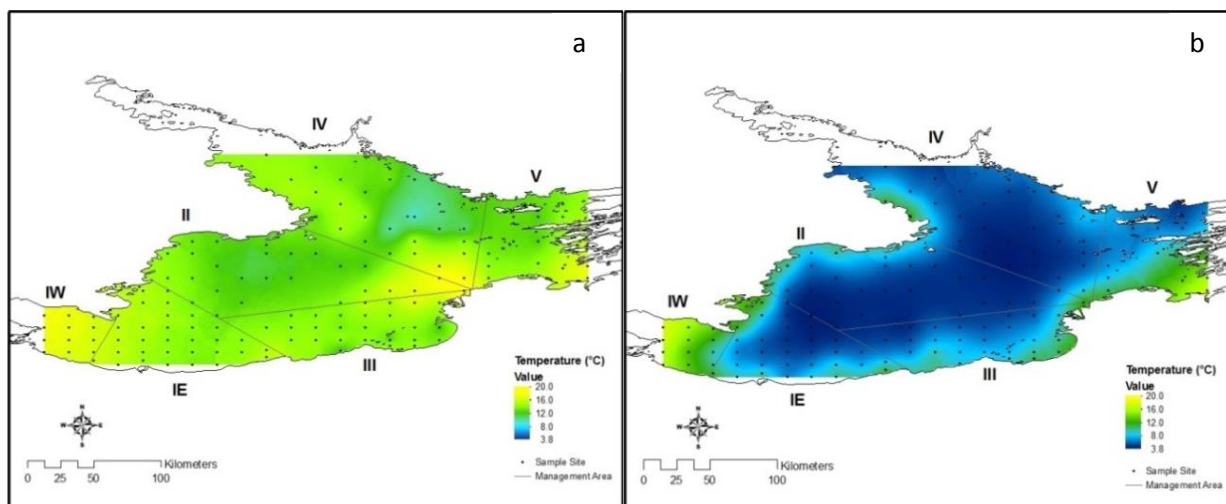


Fig. 4 Spatial distribution of water temperature ($^{\circ}\text{C}$) in the upper water column (a: 0–5 m) and in the bottom measurement (b) from the main basin of GSL, 2011–2016.

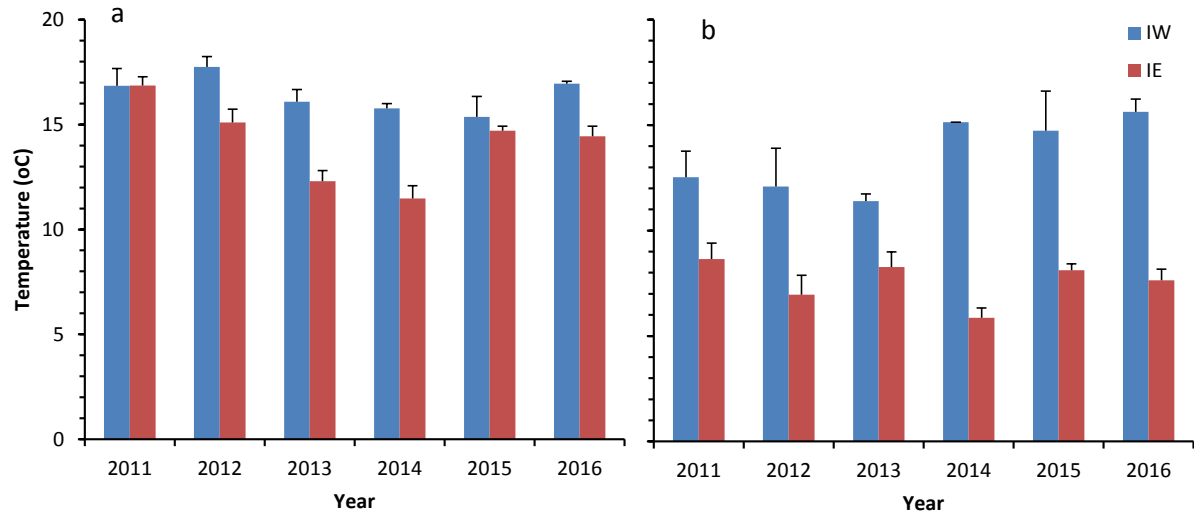


Fig. 5 Comparison of mean (\pm SE) temperature ($^{\circ}$ C) in the surface (a: 0–5 m) and bottom (b) in the western basin of GSL, 2011–2016.

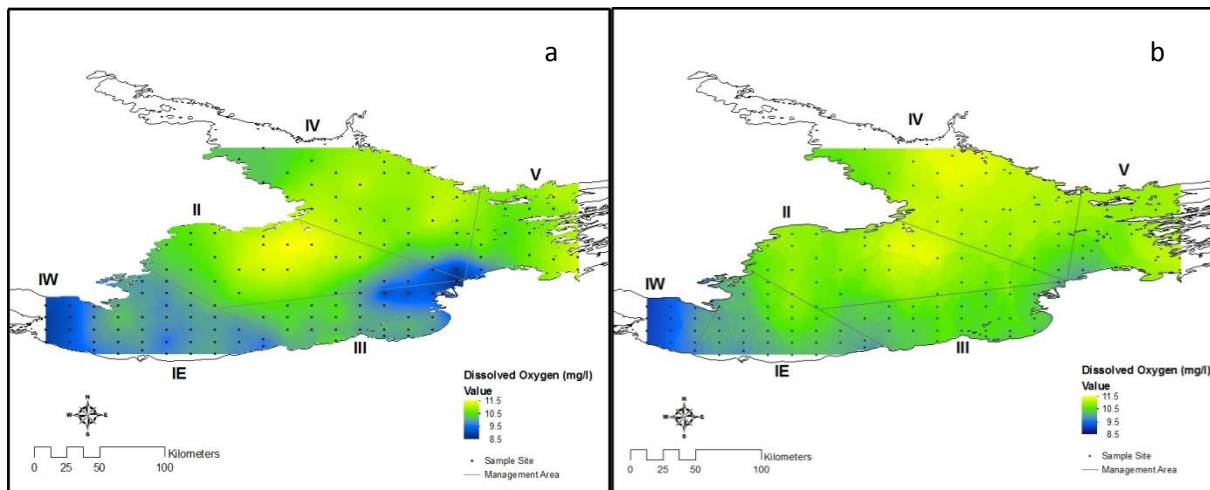


Fig. 6 Spatial distribution of dissolved oxygen concentration (mg/l) in the surface waters (a: 0–5 m) and bottom environments (b) of Great Slave Lake, observed from June through August, 2011–2016.

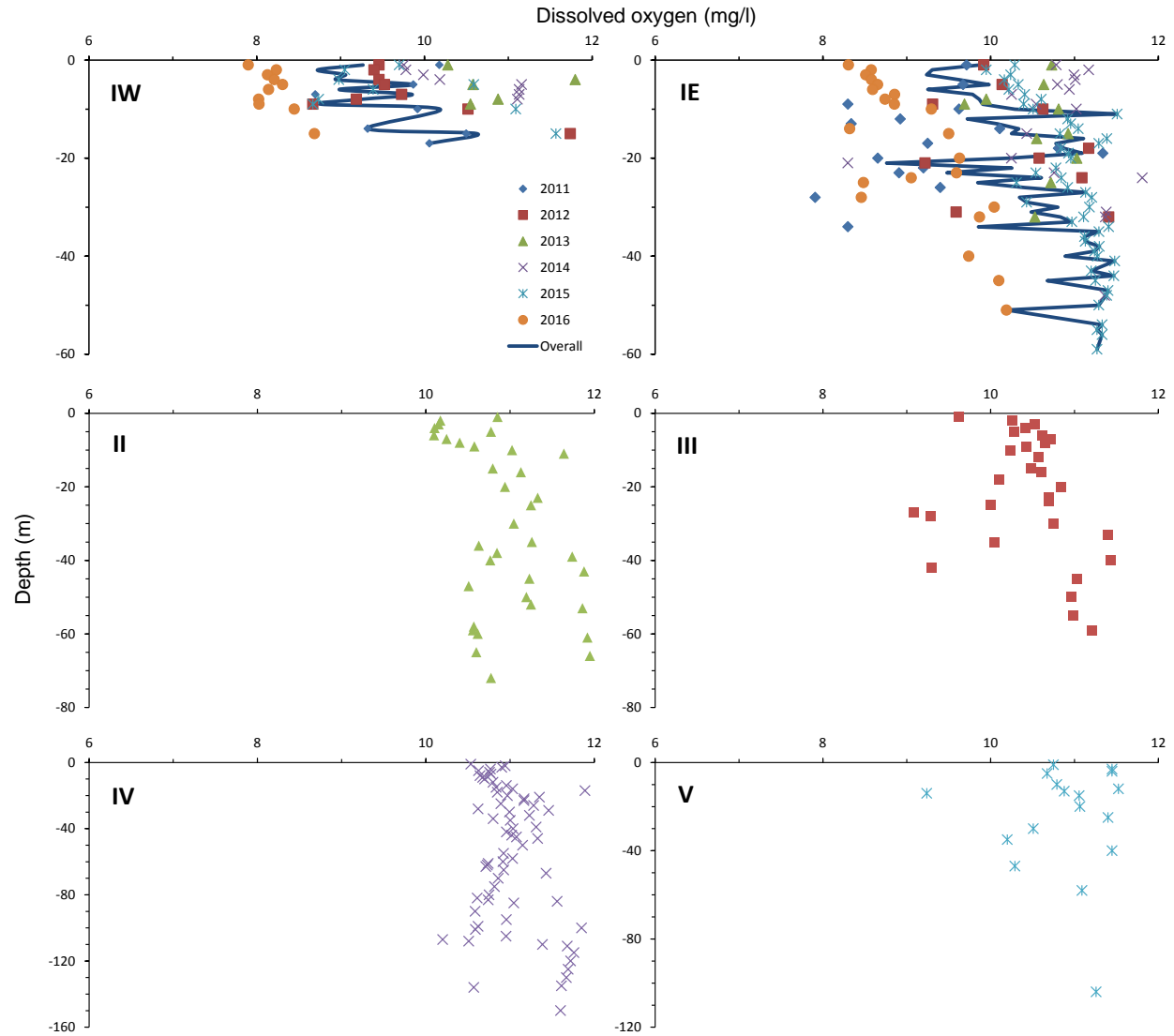


Fig. 7 Annual changes of strata-based average of dissolved oxygen (mg/l) measured in the main basin of Great Slave Lake, June through August, 2011-2016.

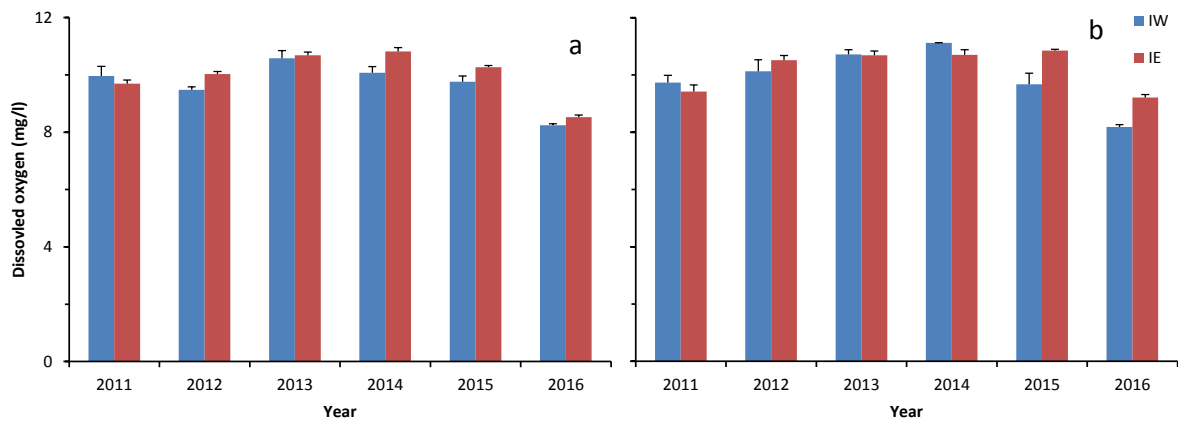


Fig. 8 Comparison of mean (\pm SE) dissolved oxygen (mg/l) in the surface (a: 0–5 m) and bottom (b: deeper than 5 m) in the western basin of GSL, 2011–2016.

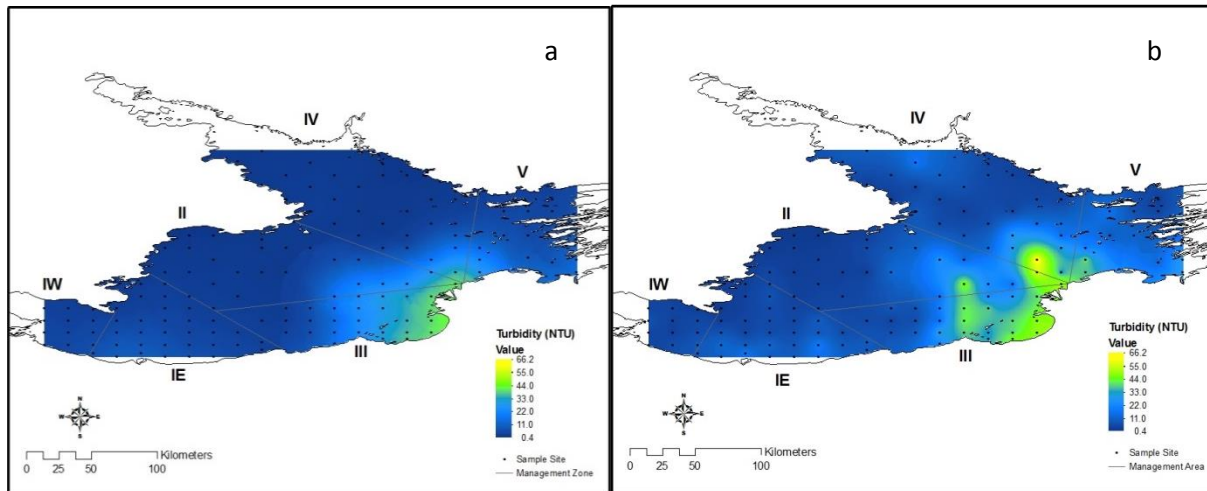


Fig. 9 Spatial distribution of turbidity across the surface (a: 0–5 m) and bottom waters (b) of GSL, 2011–2016.

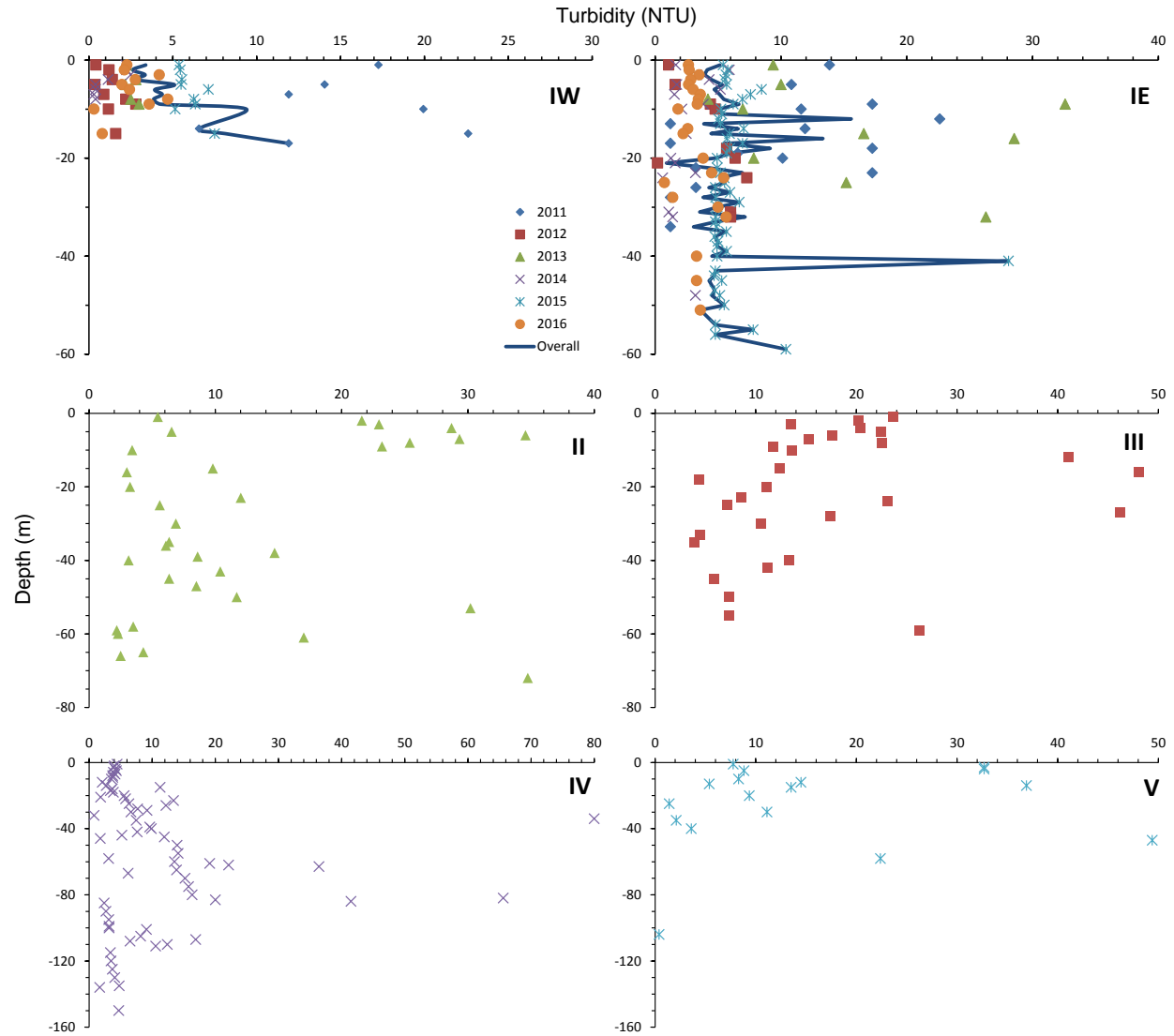


Fig. 10 Changes of strata-based average of turbidity (NTU) in the main basin of Great Slave Lake. No clear patterns through depth strata can be seen.

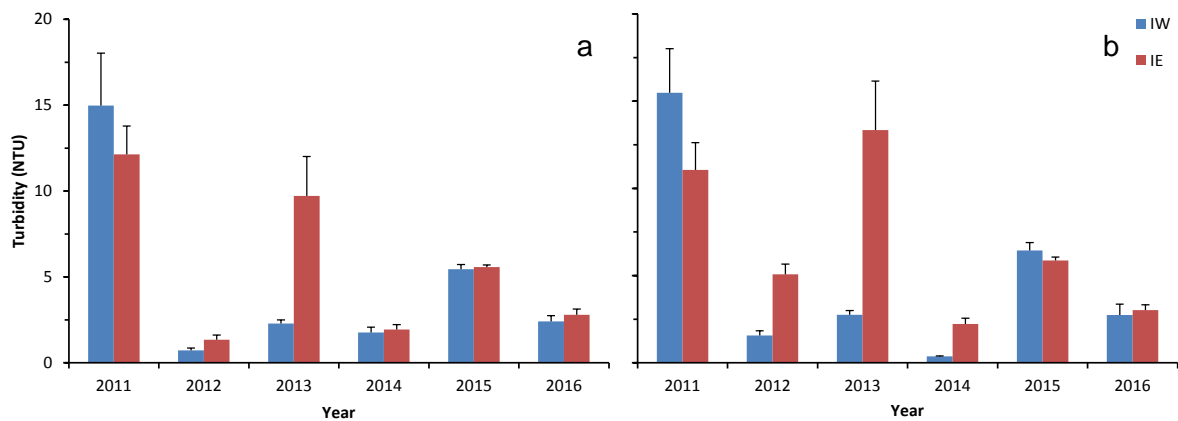


Fig. 11 Comparison of mean (\pm SE) turbidity (NTU) in the surface (a: 5 m shallow) and bottom (b) in the western basin of GSL, 2011-2016.

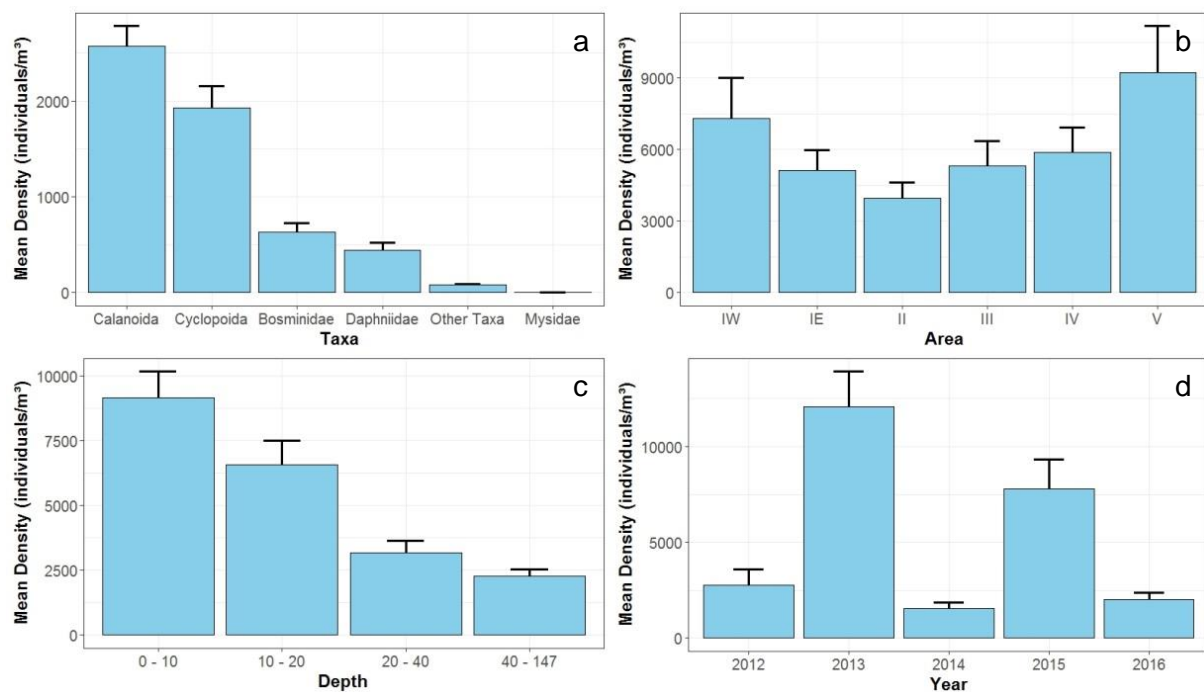


Fig. 12 Mean (\pm SE) density of zooplankton (individuals/m³) collected in vertical net tows by taxa (a), area (b), depth (m; c) and yearly from 2012–2016 (d). In a) “Other Taxa” include the Cladocerans *Diaphanosoma* sp., *Holopedium* sp., and *Leptodora* sp. Data for a-c) represent all collections from the lake ($n=167$) and for d) all collections from areas IW and IE where annual sampling occurred ($n=70$).

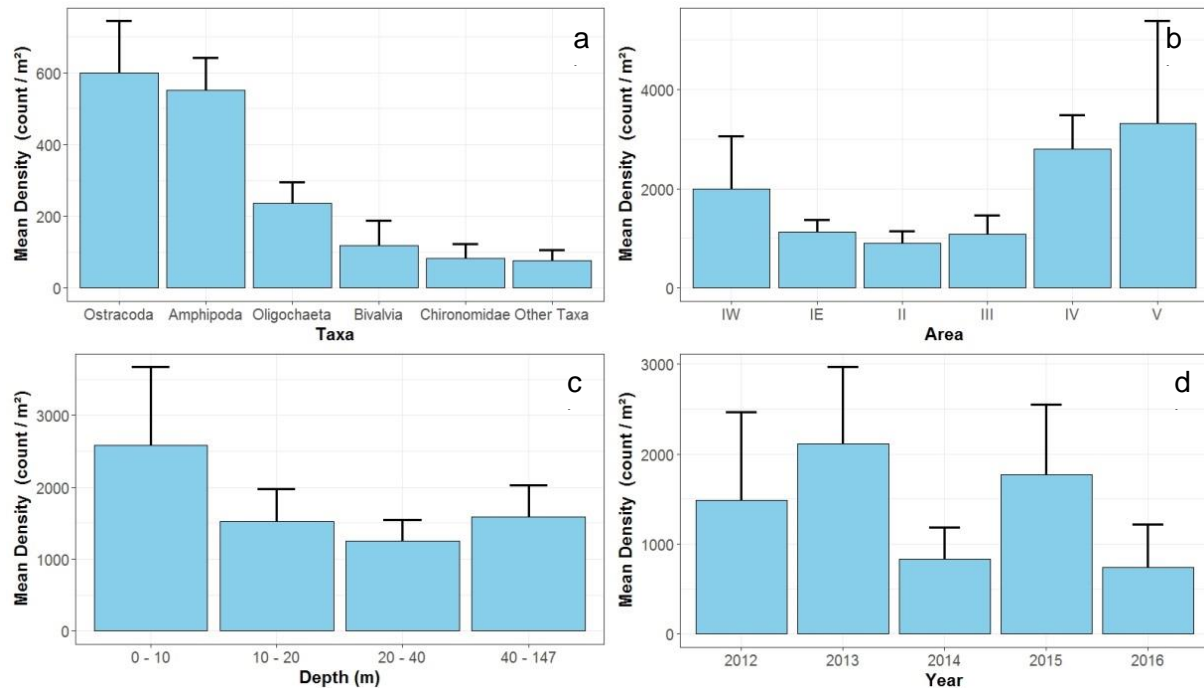


Fig. 13 The mean (\pm SE) density of benthic invertebrates (individuals/m²) collected by PONAR dredges, by taxa (a), area (b), depth category (m; c) and year (d). In a, “Other Taxa” comprises 20 taxa that each had a mean density <25 individuals/m². Data for a-c) represent all collections from the lake (n=151) and for d) collections from areas IW and IE where annual sampling occurred (n=61).

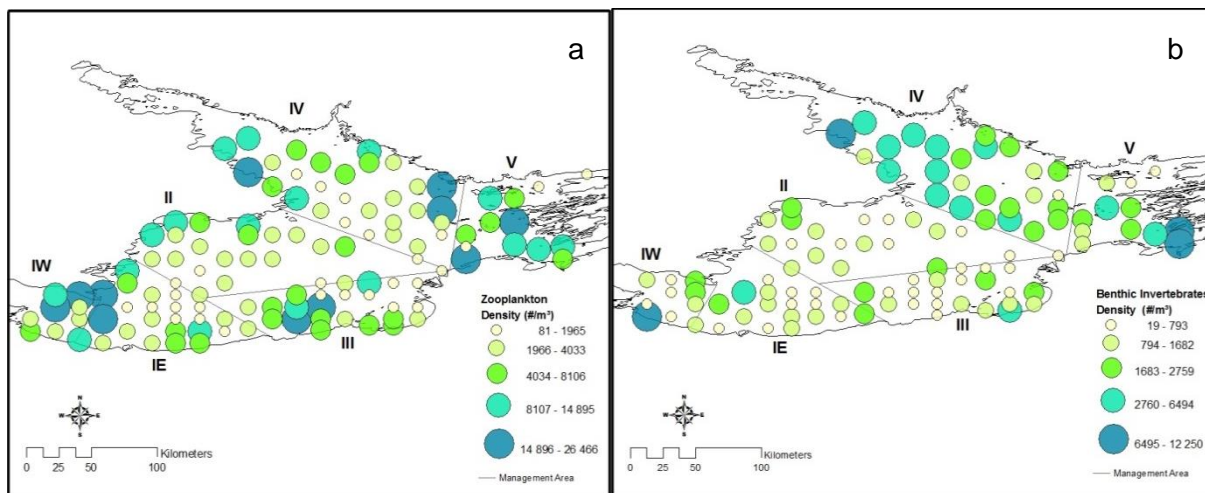


Fig. 14 Distribution of grid-based density of zooplankton (a: individuals/m³; n=120 grids) and benthic invertebrates (b: individuals/m²; n=108 grids), collected in the main basin of Great Slave Lake, June-August, 2012-2016.

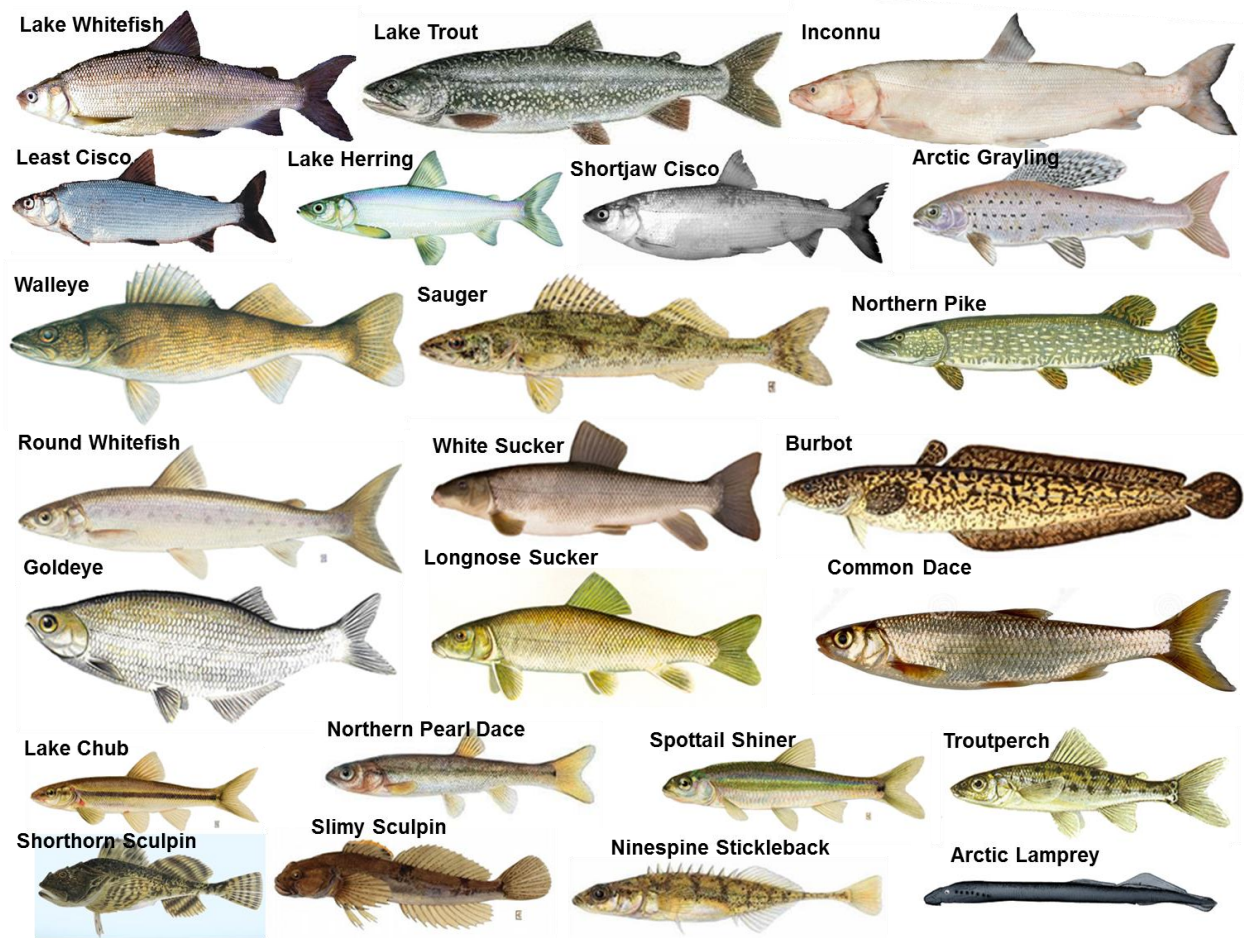


Fig. 15 Fish species collected in the main basin of Great Slave Lake, June through August, 2011-2016, and images are from website (<https://en.wikipedia.org>).

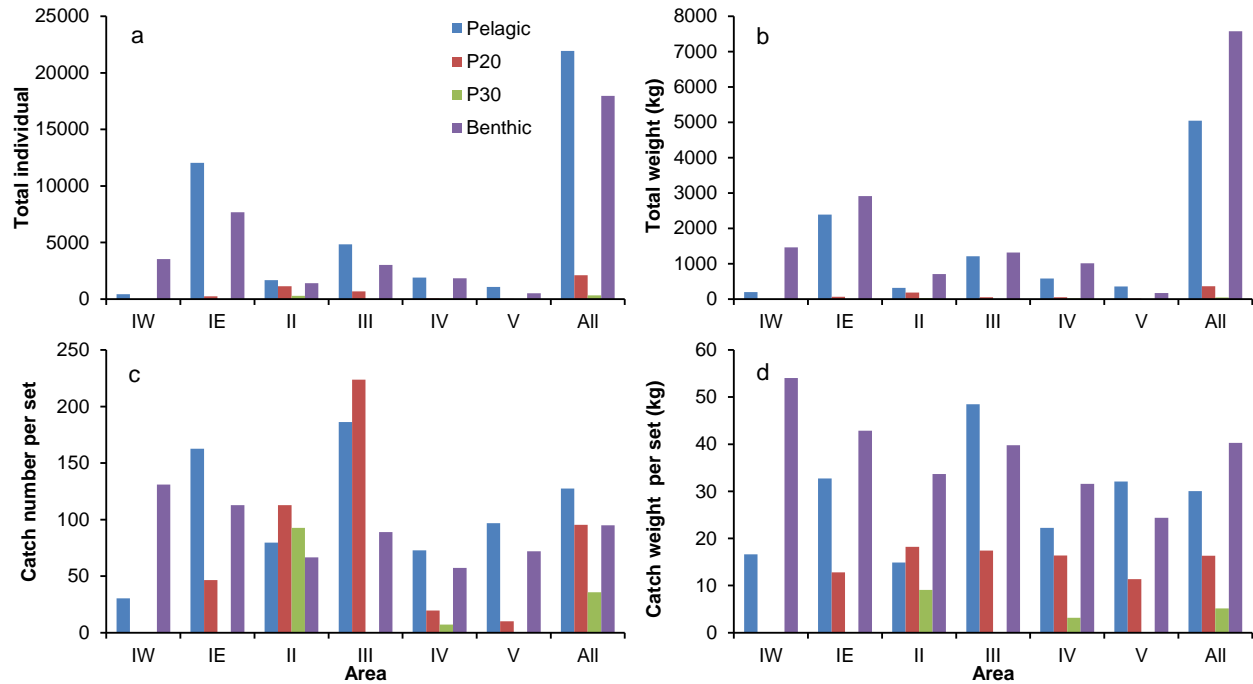


Fig. 16 Total number (a) and weight (b) of all fish species caught by experimental gillnets deployed in the upper water column (pelagic), 20 m (P20) and 30 m (P30) mid-water, and the lake bottom (benthic). Catch number per set (c: CNPS) and catch weight per set (d: CWPS) were calculated by dividing total catch by total sets.

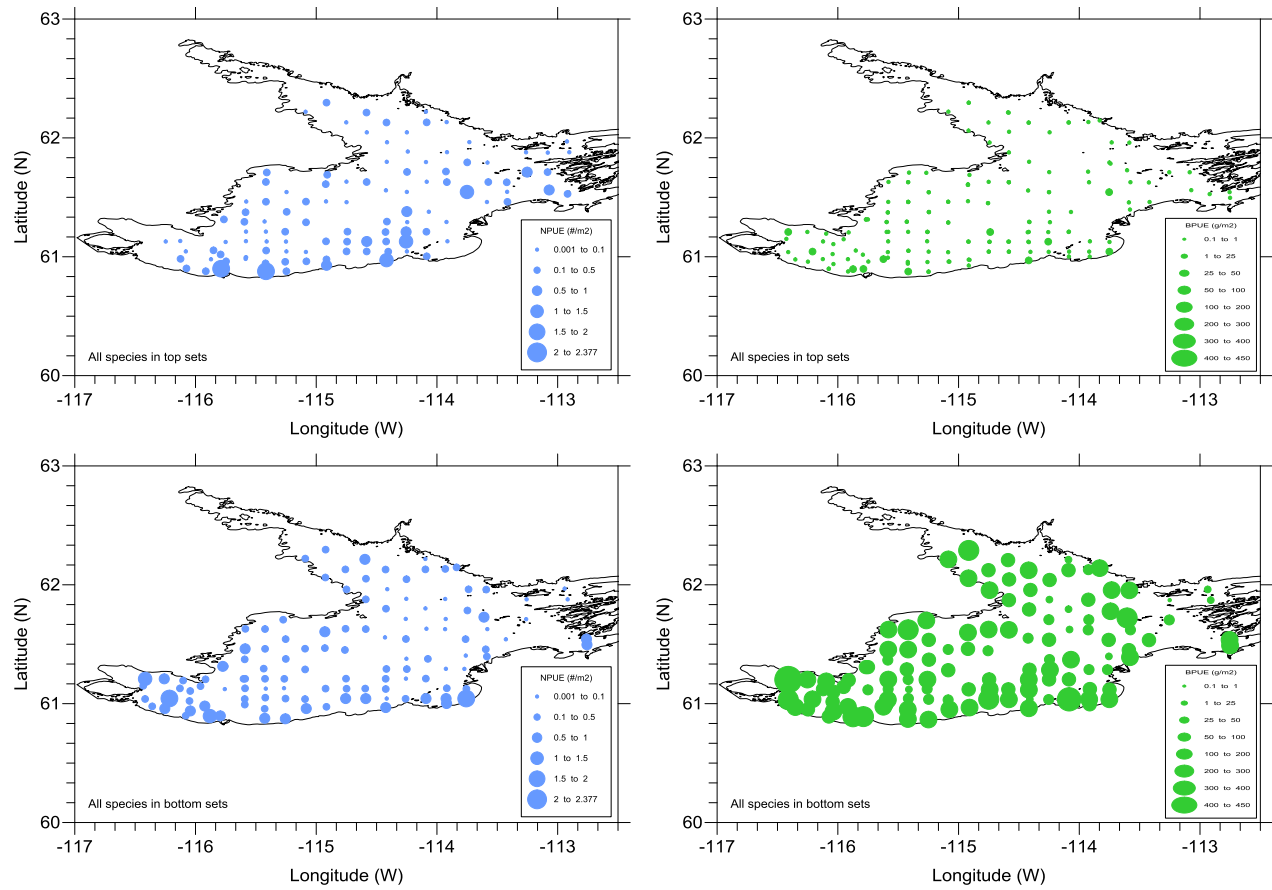


Fig. 17 Spatial distribution of NPUE (left panel, individual/m²) and BPUE (right panel: g/m²) of all fish in the surface (upper) and bottom environment (lower) of Great Slave Lake.

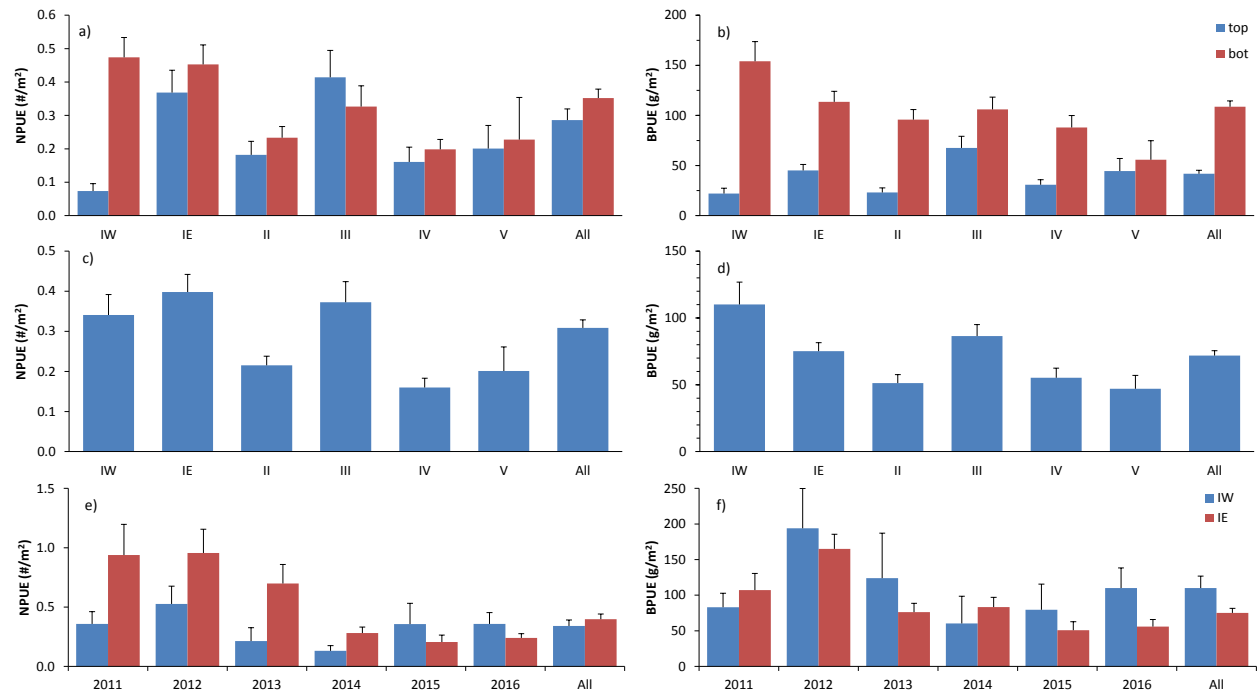


Fig. 18 Comparison of average (mean±SE) NPUE (left, #/m²) and BPUE (right, g/m²) for all fish species in the top versus bottom (a,b), area (c,d) and inter-annual variation in IW and IE (e,f) of Great Slave Lake.

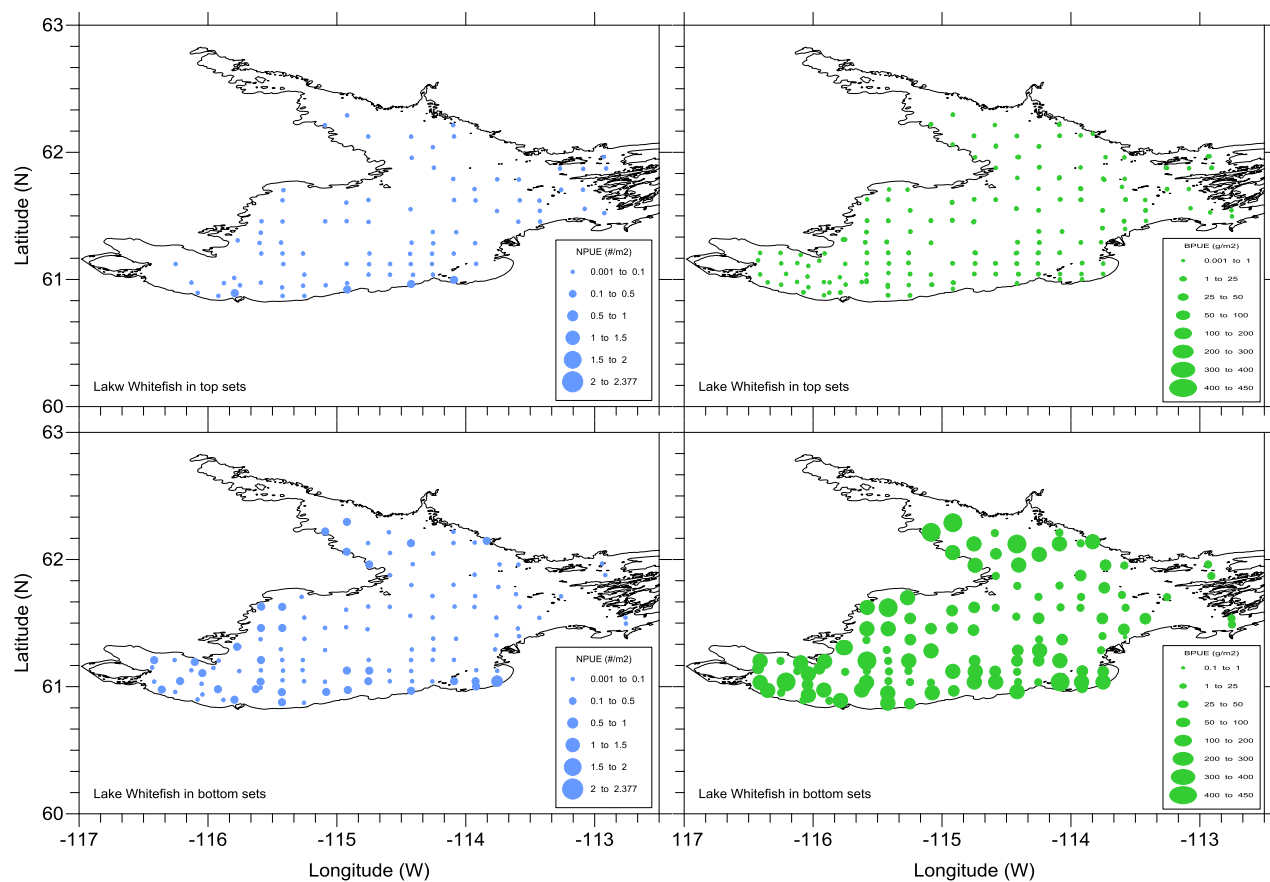


Fig. 19 Spatial distribution of NPUE (left panel, individuals/m²) and BPUE (right panel: g/m²) of Lake Whitefish in the surface (upper) and bottom environment (lower) of Great Slave Lake.

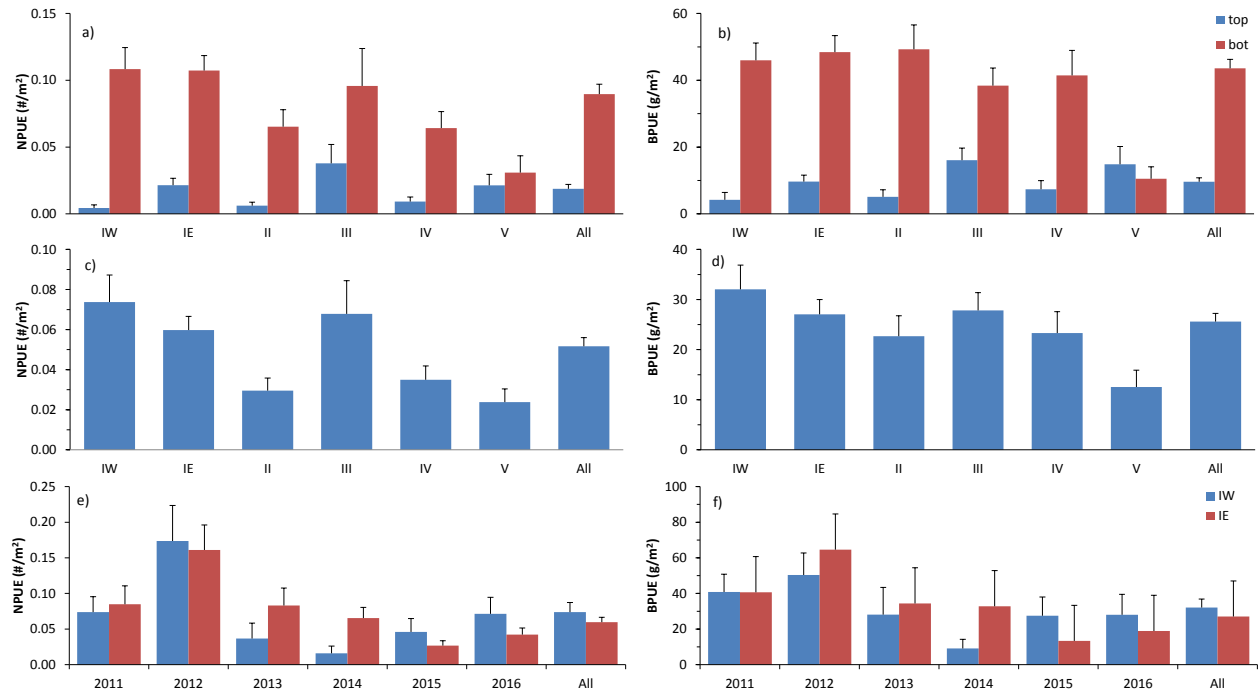


Fig. 20 Comparison of average (mean \pm SE) NPUE (left, individuals/m²) and BPUE (right, g/m²) for Lake Whitefish in the top versus bottom (a,b), area (c,d) and inter-annual variation in IW and IE (e,f) of Great Slave Lake.

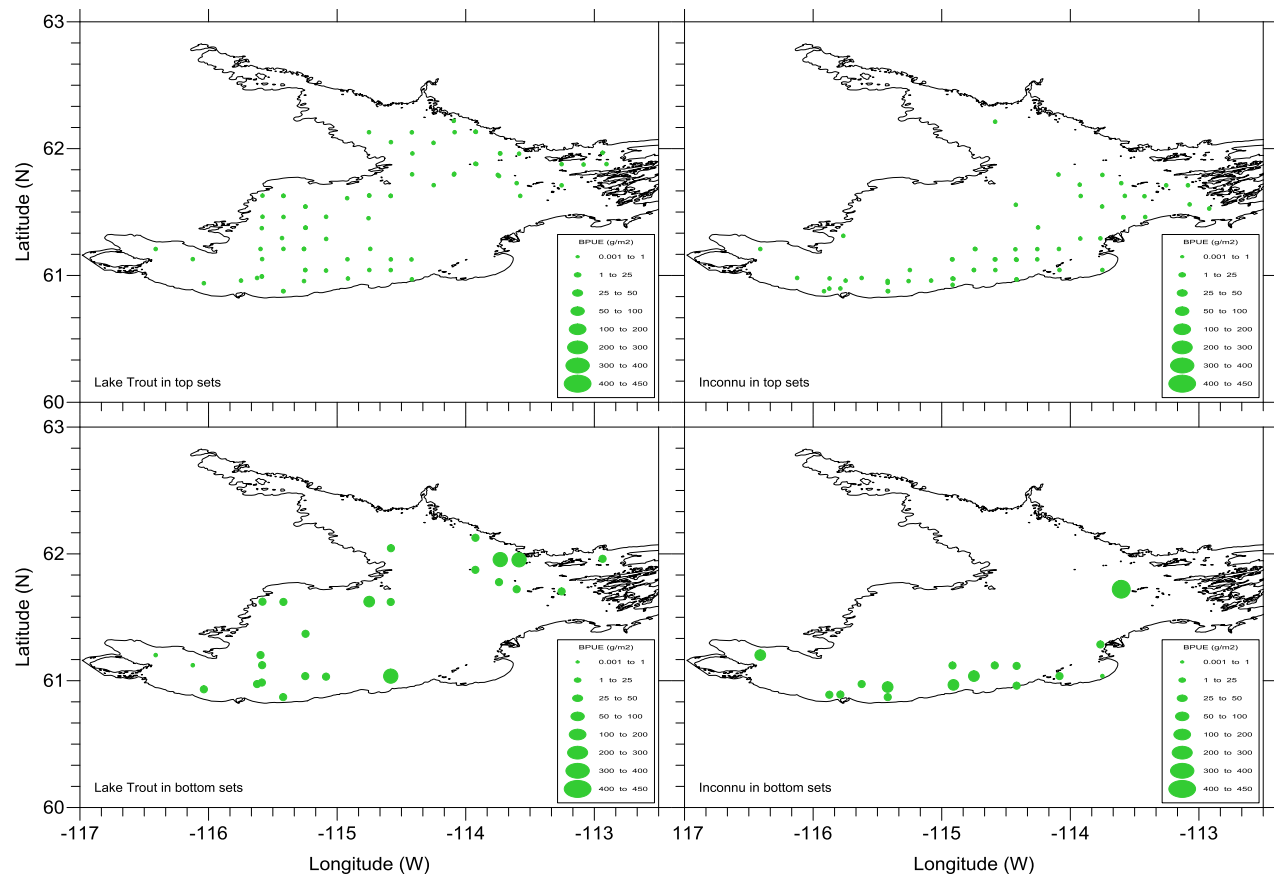


Fig. 21 Spatial distribution of BPUE (g/m²) in the upper (upper panels) and bottom (lower panels) sets in Great Slave Lake for Lake Trout (left) and Inconnu (right).

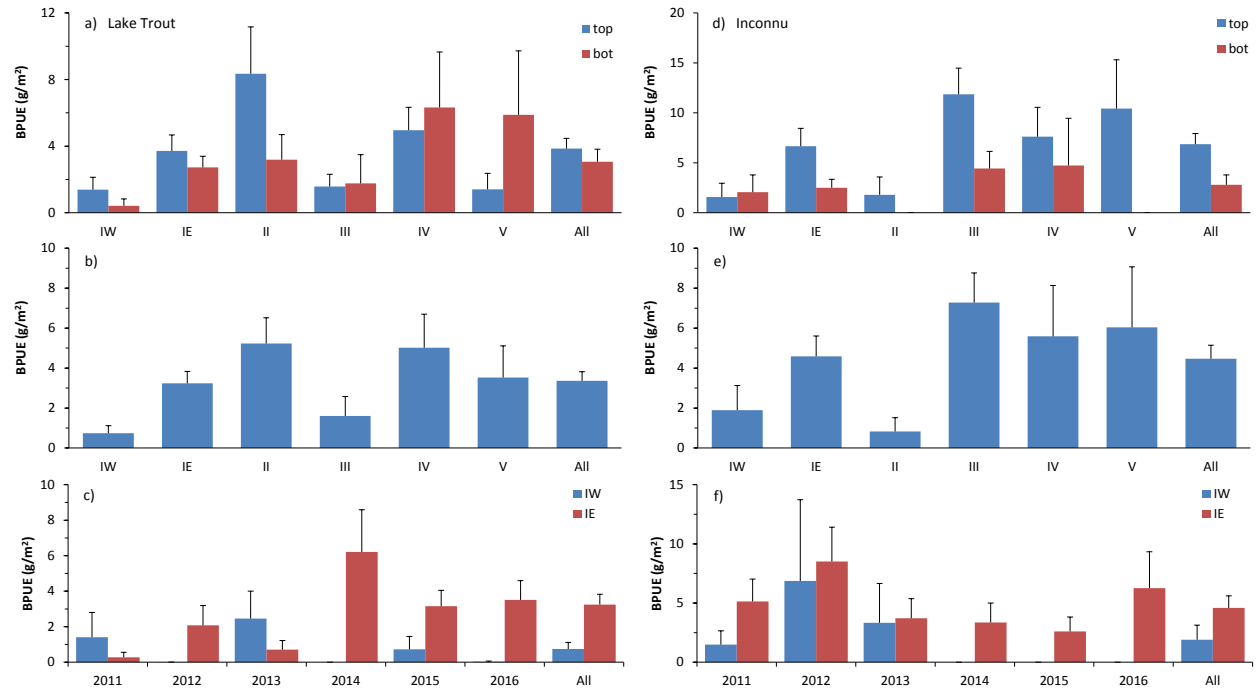


Fig. 22 Comparison of average BPUE for Lake Trout (left panel) and Inconnu (right panel), compared to the top versus bottom (a, d), area (b, e) and inter-annual variation, 2011-2016, in IW and IE (e) of Great Slave Lake.

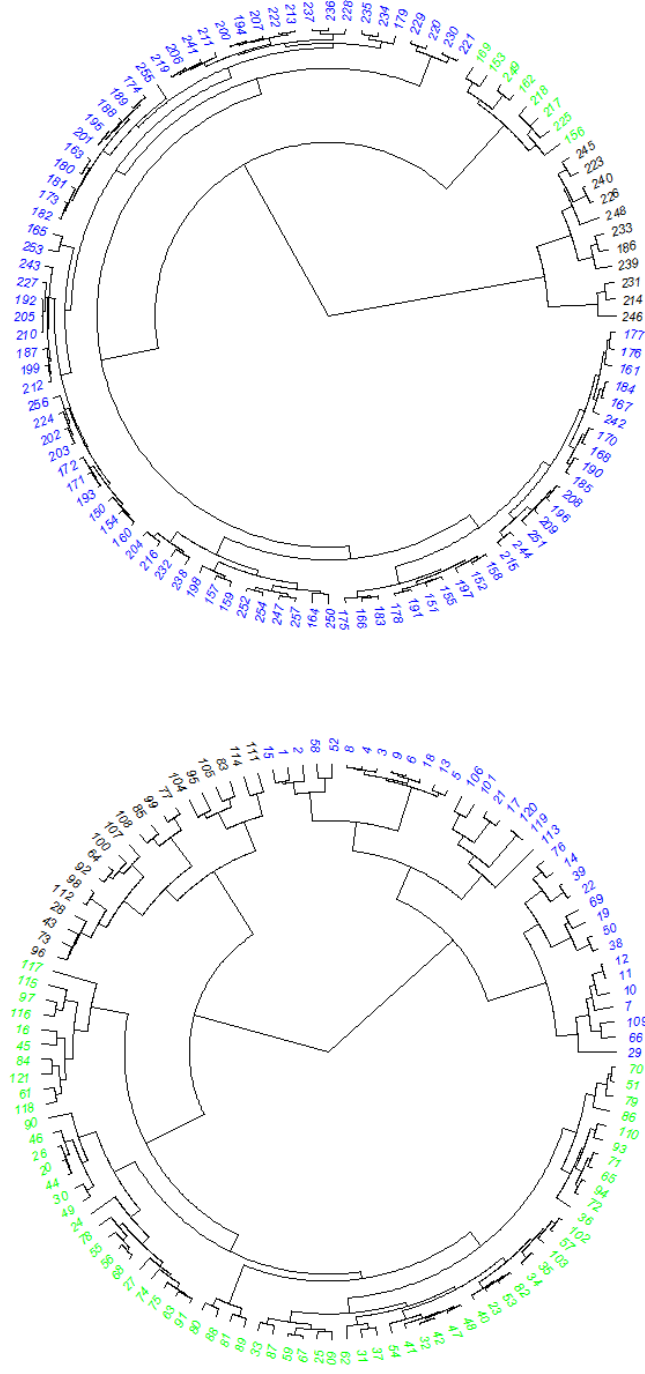


Fig. 23 Cluster analysis for grouping environmental variables in the pelagic (upper) and benthic (lower) waters of Great Slave Lake, using within group sum squares. Clusters were expressed by colors.

Table 1. Summary of gillnet deployment depths and net types based on the grid depth (m) at the sample site. Experimental gillnets included benthic (B) and pelagic (P) nets, and had a commercial (C) net attached at the end.

Net Depth (m)	Grid Depth (m)				
	0–10	10–20	20–40	40–60	60+
5	-	P + C	P	P	P
20	-	-	-	P	-
30	-	-	-	-	P
Lake bottom	B + C	-	B + C	B + C	B + C

Table 2. Biological sampling information collected from fishery-independent survey by use of quantitative gillnets during 2012-2016.

	Inconnu	Lake Trout	Lake Whitefish	Shortjaw Cisco	Least Cisco	Northern Pike	Walleye	Longnose Sucker	Burbot	all others
Sample size	all	all	all ¹⁾	all	all ²⁾	all	all	all	all	any dead fish
Total length (mm)						√	√	√	√	
Fork length (mm)	√	√	√	√	√	√				√
Round weight (g)	√	√	√	√	√	√	√	√	√	√
Dressed weight (g)	√	√	√	√	√	√	√	√	√	
Sex	√	√	√	√	√	√	√	√	√	
Maturity	√	√	√	√	√	√	√	√	√	
Gonad weight (g)	√	√	√	√	√	√	√	√	√	
Otolith	√	√	√	√	√	√	√	√	√	
Scale	√		√ (left)	√ (left)	√	√	√	√	√	
Pectoral fin	√	√	√ (left)	√ (left)	√					
DNA	√	√	√	√	√					
Stomach frozen if has food	√	√	√	√	√	√	√	√	√	
Photo (image)	√	√	√	√	√					

- 1) Sample of first 20 fish and 20% sub-sample to be used. For example in one panel 25 Lake Whitefish are captured and a total of 21 fish are sampled
- 2) If less than 10 Cisco fish are captured in a single panel, all are sampled. If more than 20 ciscoes are caught, then a 20% subsample is taken.

Table 3. Definition and codes for development stage of the gonads during life histories of fish collected from GSL multispecies survey, 2012-2016.

Maturity	Code	Female (F)	Code	Male (M)
Immature (virgin)	1	<ul style="list-style-type: none"> - never spawned - gonad bumpy in texture - hard and shaped like a long triangle - up to full length of body cavity - gonad skin firm - eggs visible, but tiny 	6	<ul style="list-style-type: none"> - never spawned - gonads long and thin - tube-like shape - up to full body length - putty-like firmness
Mature (spawner)	2	<ul style="list-style-type: none"> - current year spawner - gonad fills body cavity - small blood vessel visible - eggs growing but not loose - not expelled by pressure 	7	<ul style="list-style-type: none"> - current year spawner - gonads growing and more firm - milt not expelled by pressure - centers may feel juicy
Running Ripe	3	<ul style="list-style-type: none"> - current year spawner - gonads fill body cavity - eggs full size and almost see-through - eggs released from by pressing stomach 	8	<ul style="list-style-type: none"> - current year spawner - gonads full size - usually white - milt expelled by slight pressure
Spent	4	<ul style="list-style-type: none"> - spawning complete - gonad skin burst open and loose - small eggs visible - some loose full sized eggs found 	9	<ul style="list-style-type: none"> - spawning complete - loose with some milt - blood vessels obvious - gonads darker in color
Resting	5	<ul style="list-style-type: none"> - not spawning this year, but did in past - gonads about half the size of the body cavity - gonad skin is thin, loose, and almost see-through - healed from spawning - tiny eggs visible - some full-size eggs may be found - gonad loose, flappy 	10	<ul style="list-style-type: none"> - not spawning this year, but did in past - gonads tube-shaped, less bulky - healed from spawning - no fluid in center (no juicy feel) - usually full length of body cavity - usually dark and blotchy in color

Table 4. Summary of sampling grids by management areas over 2011-2016.

Area	2011	2012	2013	2014	2015	2016	Total
IW	5	8	4	2	6	8	11
IE	12	11	14	18	24	24	35
II		2	20			3	25
III		22				17	27
IV				30		2	31
V					13		13
Sum	17	43	38	50	43	54	228

Table 5. Summary of effective sets of experimental gillnets used for monitoring fish in the main basin of Great Slave Lake, during June through August, 2011-2016.

Setting	Area	2011	2012	2013	2014	2015	2016	Sum
Pelagic (5,12 m)	IW	2	1	3	2	2	2	12
	IE	8	6	7	13	19	20	73
	II		1	18			2	21
	III		14				11	25
	IV				24		2	26
	V					11		11
	Total	10	22	28	39	32	37	168
Upper Mid (20 m)	IW							
	IE					3	2	5
	II			10				10
	III		1				2	3
	IV				2		1	3
	V					1		1
	Total		1	10	2	4	5	22
Lower Mid (30 m)	IW							
	IE							
	II			3				3
	III							
	IV		1		4		1	6
	V							
	Total		1	3	4		1	9
Benthic	IW	6	5	2	4	4	6	27
	IE	6	7	5	15	18	17	68
	II		1	18			2	21
	III		18				15	33
	IV		1		29		2	32
	V					7		7
	Total	12	32	25	48	29	42	188
Overall		22	56	66	93	65	85	387

Table 6. Fish species collected from depth-stratified surveys in the main basin of Great Slave Lake, 2011-2016. The experimental gillnets were deployed at 5 m below the surface (pelagic), at 20 m (P20) and 30 m (P30) as well as benthic gillnet (benthic), in terms of grid depth.

Acronym	Common name	Scientific name	Pelagic	P20	P30	Benthic	Overall
AG	Arctic Grayling	<i>Thymallus arcticus</i>				+	+
ALP	Arctic Lamprey	<i>Lethenteron camtschaticum</i>				+	+
BBT	Burbot	<i>Lota lota</i>	+	+	+	+	+
CHUB	Lake Chub	<i>Couesius plumbeus</i>	+				+
GDI	Goldeye	<i>Hiodon alosoides</i>	+			+	+
INCO	Inconnu	<i>Stenodus nelma</i>	+	+		+	+
LCK	Least Cisco	<i>Coregonus sardinella</i>	+	+	+	+	+
LKDS	Common Dace	<i>Leuciscus leuciscus</i>				+	+
LKH	Lake Herring	<i>Coregonus artedii</i>	+	+	+	+	+
LKT	Lake Trout	<i>Salvelinus namaycush</i>	+	+	+	+	+
LKWF	Lake Whitefish	<i>Coregonus clupeaformis</i>	+	+	+	+	+
LNSK	Longnose Sucker	<i>Catostomus catostomus</i>	+			+	+
NPD	Northern Pearl Dace	<i>Margariscus nachtriebi</i>	+			+	+
NPK	Northern Pike	<i>Esox lucius</i>	+			+	+
NSSBK	Ninespine Stickleback	<i>Pungitius pungitius</i>	+			+	+
RDWF	Round Whitefish	<i>Prosopium cylindraceum</i>	+	+		+	+
SAUGER	Sauger	<i>Sander canadensis</i>				+	+
SHSP	Shorthorn Sculpin	<i>Myoxocephalus scorpius</i>			+	+	+
SLSP	Slimy Sculpin	<i>Cottus cognatus</i>				+	+
SJCK	Shortjaw Cisco	<i>Coregonus zenithicus</i>	+	+	+	+	+
SPT	Spottail Shiner	<i>Notropis hudsonius</i>	+			+	+
TP	Troutperch	<i>Percopsis omiscomaycus</i>	+			+	+
WSK	White Sucker	<i>Catostomus commersonii</i>				+	+
WY	Walleye	<i>Sander vitreus</i>	+			+	+
Sum			17	8	7	23	24

Table 7. Relative importance of fishes in upper (top), mid-water (mid; combination of 20 m and 30 m nets), bottom (bot) and entire water columns of the main basin of Great Slave Lake, June through August, 2011-2016. Common and scientific names for fish codes are shown in Table 5.

Fish	Number (%)				Weight (%)				Frequency				%IRI			
	Top	Mid	Bot	All	Top	Mid	Bot	All	Top	Mid	Bot	All	Top	Mid	Bot	All
AG	0.00	0.00	0.02	0.01	0.00	0.00	0.03	0.02	0.00	0.00	1.65	0.79	0.00	0.00	0.00	0.00
ALP	0.00	0.00	0.08	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.83	0.39	0.00	0.00	0.00	0.00
BBT	2.00	1.30	7.16	4.22	8.22	6.85	17.86	13.65	48.15	39.29	82.64	43.70	3.91	2.44	15.05	9.75
CHUB	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.93	0.00	0.00	0.39	0.00	0.00	0.00	0.00
GDI	0.12	0.00	0.63	0.34	0.14	0.00	0.51	0.35	3.70	0.00	4.96	3.94	0.01	0.00	0.04	0.03
INCO	1.27	0.22	0.33	0.76	20.45	7.61	3.75	10.35	36.11	10.71	13.22	18.50	6.23	0.64	0.39	2.56
LCK	40.45	51.12	17.73	31.35	5.10	16.08	1.68	3.66	77.78	75.00	60.33	44.49	28.13	38.41	8.53	19.44
LKDS	0.00	0.00	0.04	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.83	0.39	0.00	0.00	0.00	0.00
LKH	32.99	36.59	20.06	27.58	8.50	12.55	6.05	7.29	80.56	53.57	77.69	45.67	26.54	20.06	14.76	19.88
LKT	0.77	0.52	0.45	0.61	9.59	14.88	3.99	6.64	36.11	32.14	20.66	23.23	2.97	3.77	0.67	2.10
LKWF	12.55	7.22	30.27	19.92	28.25	39.67	43.17	37.27	78.70	92.86	96.69	53.15	25.50	33.18	51.68	37.95
LNSK	4.39	0.00	5.10	4.30	13.52	0.00	8.65	10.13	37.96	0.00	38.02	25.98	5.40	0.00	3.81	4.68
NPD	0.01	0.00	0.12	0.06	0.00	0.00	0.00	0.00	0.93	0.00	3.31	1.97	0.00	0.00	0.00	0.00
NPK	0.42	0.00	1.83	1.00	3.18	0.00	8.17	5.88	12.04	0.00	22.31	13.78	0.34	0.00	1.62	1.18
NSSBK	0.09	0.00	0.29	0.17	0.01	0.00	0.00	0.00	2.78	0.00	8.26	5.12	0.00	0.00	0.02	0.01
RDWF	0.09	0.13	0.04	0.07	0.05	0.15	0.01	0.03	0.93	3.57	2.48	1.97	0.00	0.01	0.00	0.00
SAUGER	0.00	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.83	0.39	0.00	0.00	0.00	0.00
SHSP	0.00	0.13	0.12	0.06	0.00	0.01	0.00	0.00	0.00	7.14	5.79	2.76	0.00	0.01	0.01	0.00
SLSP	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
SJCK	2.22	2.77	3.42	2.80	0.49	2.19	0.65	0.65	19.44	39.29	34.71	19.29	0.42	1.48	1.03	0.83
SPT	0.02	0.00	2.51	1.12	0.00	0.00	0.03	0.02	1.85	0.00	9.92	5.12	0.00	0.00	0.18	0.07
TP	1.63	0.00	6.72	3.74	0.05	0.00	0.13	0.09	7.41	0.00	26.45	14.17	0.10	0.00	1.32	0.68
WSK	0.00	0.00	0.45	0.20	0.00	0.00	2.10	1.19	0.00	0.00	6.61	3.15	0.00	0.00	0.12	0.05
WY	1.00	0.00	2.63	1.63	2.43	0.00	3.22	2.77	16.67	0.00	18.18	13.78	0.45	0.00	0.77	0.76

Table 8. Summary of cluster analysis for pelagic and benthic environmental variables of Great Slave Lake.

	Mean	SE	95% CI	Mean	SE	95% CI	Mean	SE	95% CI
Top	Cluster A (n=89)			Cluster B (n=8)			Cluster C (n=11)		
Depth (m)	38.32	2.86	32.64–43.99	13.93	2.03	9.12–18.73	36.15	7.52	19.40–52.90
Temperature (°C)	12.16	0.34	11.48–12.84	13.79	1.66	9.87–17.71	14.51	0.54	13.31–15.72
Dissolved oxygen (mg/l)	10.23	0.12	10.00–10.46	9.81	0.35	8.99–10.64	9.50	0.26	8.91–10.09
pH	8.43	0.04	8.34–8.51	8.57	0.36	7.72–9.41	8.08	0.19	7.65–8.50
Turbidity (NTU)	2.25	0.28	1.69–2.81	19.87	1.57	16.16–23.59	42.36	3.11	35.44–49.29
Conductivity (µs/cm)	0.25	0.01	0.24–0.27	0.24	0.01	0.21–0.27	0.31	0.03	0.25–0.37
Bottom	Cluster A (n=35)			Cluster B (n=66)			Cluster C (n=20)		
Depth (m)	9.96	0.94	8.05–11.87	44.26	3.38	37.51–51.02	37.77	5.55	26.16–49.38
Temperature (°C)	13.75	0.61	12.50–14.99	4.49	0.11	4.27–4.71	5.77	0.46	4.81–6.73
Dissolved oxygen (mg/l)	9.27	0.22	8.81–9.72	10.63	0.11	10.41–10.85	10.96	0.12	10.70–11.21
pH	8.38	0.09	8.21–8.56	8.11	0.05	8.01–8.22	8.11	0.11	7.88–8.33
Turbidity (NTU)	11.89	2.62	6.57–17.22	7.26	0.65	5.97–8.55	40.41	3.62	32.82–47.99
Conductivity (µs/cm)	0.27	0.01	0.24–0.29	0.21	0.01	0.20–0.23	0.21	0.01	0.19–0.23

Table 9. Conditional effects of environmental variables with the first two axis of CCA for BPUE of all species in pelagic and benthic environments of Great Slave Lake.

Variable	Pelagic			Benthic		
	λ	F	p	λ	F	p
Depth	0.12	7.04	0.00	0.08	5.78	0.00
Temperature	0.06	4.39	0.00	0.40	28.97	0.00
Conductivity	0.05	2.94	0.00	0.02	1.85	0.04
Turbidity	0.04	2.88	0.01	0.05	3.46	0.00
Dissolved oxygen	0.04	2.24	0.02	0.02	1.74	0.06
pH	0.01	0.46	0.89	0.02	1.44	0.15
	Axis 1	Axis 2		Axis 1	Axis 2	
Eigenvalue	0.17	0.06		0.41	0.08	
Correlation	0.66	0.52		0.88	0.66	
Cumulative percentage variance						
Species	9.10	12.30		20.00	24.10	
Species-environment	53.20	72.00		69.80	84.00	

APPENDICES

Here, a series of reports, conference presentations and posters are listed as appendices. If interested, please log in to the NWT discovery portal

(<http://nwtdiscoveryportal.enr.gov.nt.ca/geoportal/catalog/main/home.page>) to find them.

- **Non peer-reviewed report**

- 1) NWT CIMP 2012-13 Annual Project Status Report. Available at <http://nwtdiscoveryportal.enr.gov.nt.ca/geoportal/catalog/main/home.page>
- 2) NWT CIMP 2013-14 Annual Project Status Report Available at <http://nwtdiscoveryportal.enr.gov.nt.ca/geoportal/catalog/main/home.page>
- 3) NWT CIMP 2014-15 Annual Project Status Report Available at <http://nwtdiscoveryportal.enr.gov.nt.ca/geoportal/catalog/main/home.page>
- 4) NWT CIMP 2015-16 Annual Project Status Report Available at <http://nwtdiscoveryportal.enr.gov.nt.ca/geoportal/catalog/main/home.page>
- 5) NWT CIMP 2016-17 Annual Project Status Report Available at <http://nwtdiscoveryportal.enr.gov.nt.ca/geoportal/catalog/main/home.page>

- **Peer-reviewed report**

- 1) Zhu, X., Wastle, R. J., Howland, K. L., Leonard, D. J., Mann, S., Carmichael, T. J., and Tallman, R. F. 2015. A comparison of three anatomical structures for estimating age in a slow-growing subarctic population of Lake Whitefish. *North American Journal of Fisheries Management* 35: 262-270. Available at <http://dx.doi.org/10.1080/02755947.2014.996683>
- 2) Zhu, X., Tallman, R. F., Howland, K. L., and Carmichael, T. J. 2016. Modeling spatiotemporal variabilities of length-at-age growth characteristics for slow-growing subarctic populations of Lake Whitefish, using hierarchical Bayesian statistics. *Journal of Great Lakes Research* 42: 308-318. Available at <http://dx.doi.org/10.1016/j.jglr.2015.08.013>
- 3) Zhu, X., Wastle, R., Leonard, D., Howland, K., Carmichael, T.J., and Tallman, R.F., 2017. Comparison of Scales, Pectoral Fin Rays, and Otoliths for Estimating Age, Growth, and Mortality of Lake Whitefish, *Coregonus clupeaformis*, in Great Slave Lake. DFO CSAS Res. Doc. 2016/115. v + 28 p. Available at http://www.dfo-mpo.gc.ca/csas-sccs/Publications/ResDocs-DocRech/2016/2016_115-eng.pdf.
- 4) Zhu, X., Day, A.C., Taptuna, W.E.F., Carmichael, T.J., and Tallman, R.F. 2015. Hierarchical modeling of spatiotemporal dynamics of biological characteristics of Lake Whitefish, *Coregonus clupeaformis* (Mitchill), in Great Slave Lake, Northwest Territories, 1972-2004. DFO Can. Sci. Advis. Sec. Res. Doc. 2015/038. v + 56 p. http://www.dfo-mpo.gc.ca/csas-sccs/Publications/ResDocs-DocRech/2015/2015_038-eng.html
- 5) DFO. 2015. Assessment of Lake Whitefish Status in Great Slave Lake, Northwest Territories, Canada, 1972-2004. DFO CSAS SAR 2015/042. 10 p. <http://www.dfo->

mpo.gc.ca/csas-sccs/publications/sar-as/2015/2015_042-eng.html

- 6) Zhu, X., and et al. 2013. Fishery-independent gillnet study (FIGS) sampling protocol used for multi-species study in Great Slave Lake, Northwest Territories, Canada. CSAS Research Document. To be posted on <http://www.dfo-mpo.gc.ca/csas-sccs/index-eng.htm>.
- 7) Zhu, X., and et al. 2013. Evaluation of capture efficiency and mesh-sized gillnet selectivity for important fishes in Great Slave Lake, Northwest Territories, Canada. DFO CSAS Research Document. To be posted on <http://www.dfo-mpo.gc.ca/csas-sccs/index-eng.htm>.
- 8) Zhu, X., and et al. 2013. Proceedings for the Regional Review of the Stock Assessment Approach for Sustainable Fisheries Development for Great Slave Lake. DFO Science Advisory Report (SAR): Integrated fishery stock assessment plan for sustainable harvest of Lake Whitefish in Great Slave Lake, Northwest Territories. June 18-24, 2013. Winnipeg, MB. To be posted on <http://www.dfo-mpo.gc.ca/csas-sccs/index-eng.htm>.
- 9) Zhu, X., and et al. 2013. Monitoring and Assessing Environmental and Cumulative Impacts on Great Slave Lake Fishery Population Productivity and Fish Community Association, 2012/13. Canadian Manuscript of Fisheries and Aquatic Sciences No. ####: VIII+41 p. Available at <http://nwtDiscoveryportal.enr.gov.nt.ca/geoportal/catalog/main/home.page>

- **Presentation**

- 1) Zhu, X. 2012. Monitoring and assessing environmental and cumulative impacts on Great Slave Lake fishery population productivity and fish community association. Presented at The Ecological & Evolutionary Ethology of Fishes (EEEEF) in Windsor, Ontario, June 18-22, 2012.
- 2) Zhu, X. 2012. Integrative ecomonitoring of fisheries production and fish community in Great Slave Lake. Presented at 2013 NWT CIMP Workshop in Yellowknife, NWT, December 14-18, 2013.
- 3) Zhu, X. and et al. 2013. Integrated Ecomonitoring of Cumulative Impacts on Great Slave Lake Fisheries Ecosystems. Presented at 2013 GSLAC Fall meeting in Hay River, NWT, November 5-8, 2013.
- 4) Zhu, X. et al. 2013. Fish Community Dynamics and Environmental Association: Implications for Decision Making for Sustainable Fisheries in the Great Lakes of the NWT. Presented at the GeoScience Forum in Yellowknife, NWT, November 16-20, 2013.
- 5) Zhu, X., et al. 2014. Fish Community Dynamics and Environmental Association: Implications for Decision Making for Sustainable Fisheries in GSL, Northwest Territories. Presented at the Canadian Conference for Fisheries Research (CCFFR), Yellowknife, NWT, January 3-5, 2014.
- 6) Zhu, X., et al. 2014. Ecotrophic modeling of anthropogenic cumulative impacts on the sustainability of fisheries productions: comparison of Lake Erie and GSL ecosystems.

Presented at the Ecopath with Ecosim (EwE) 30 Years' International Conference, Barcelona, Spain, November 10-12, 2014.

- 7) Zhu, X., et al. 2015. Monitoring Great Slave Lake Fisheries Ecosystem. Presented at the GSLAC Spring meeting in Hay River, NWT, May 23-14, 2015.
- 8) Zhu, X., et al. 2015. Integrated Ecomonitoring of Cumulative Impacts on Great Slave Lake Fisheries Ecosystems. Presented at the GSLAC Fall meeting in Hay River, NWT, November 3-4, 2015.
- 9) Zhu, X., et al. 2015. Great Slave Lake Fisheries Ecosystem Studies, 2012-2015. Presented at the 2015 Fall GSLAC meeting, Hay River, NWT, November 3-6, 2015.
- 10) Zhu, X., et al. 2016. Integrated Ecomonitoring of Great Slave Lake (GSL) Fisheries Ecosystems. Presented at the GSLAC spring meeting in Hay River, NWT, May 13-14, 2016.
- 11) Zhu, X., et al. 2016. Fish Community Dynamics and Its Association with Great Slave Lake Environmental Changes. Presented at the GSLAC fall meeting in Hay River, NWT, November 8-9, 2016.
- 12) Zhu, X., et al. 2017. Integrated Ecomonitoring and Assessment of Great Slave Lake Fisheries Ecosystem — Project Progress and Summaries. Presented to the NWT-CIMP office, January 5, 2017.

- **Poster**

- 1) Zhu, X., and et al. 2013. Age estimate comparison of Lake Whitefish in Great Slave Lake. Presented at 2013 NWT CIMP workshop in Yellowknife, NWT, December 14-18, 2013.
- 2) Zhu, X., and et al. 2013. Evaluation of capture efficiency and mesh-sized gillnet selectivity for dominant fishes in Great Slave Lake. Presented at 2013 NWT CIMP workshop in Yellowknife, NWT, December 14-18, 2013.
- 3) Zhu, X., and et al. 2013. Spatial distribution of Inconnu, *Stenodus leucichthys nelma* (Pallas, 1773), in the southern Great Slave Lake. Presented at 2013 NWT CIMP workshop in Yellowknife, NWT, December 14-18, 2013.