Conservation of Arctic Flora and Fauna

State of the Arctic Freshwater Biodiversity Report

CIRCUMPOLAR BIODIVERSITY MONITORING PROGRAM

ARCTIC COUNCIL

Acknowledgements

For more information please contact:
CAFF International Secretariat
Borgir, Nordurslod
600 Akureyri, Iceland
Phone: +354 462-3350
Fax: +354 462-3390
Email: caff@caff.is
Website: www.caff.is

CAFF Designated Agencies:

- Environment and Climate Change Canada
- Faroese Museum of Natural History
- Finnish Ministry of the Environment
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Editing: Tom Barry, Tom Christensen, Joseph Culp, Willem Goedkoop, and Jennifer Lento

CBMP Freshwater Steering Group: Willem Goedkoop (co-lead), Joseph Culp (co-lead), Jennifer Lento, Petri Liljaniemi, Kirsten S.Christoffersen, Elena Fefilova, Jón S. Ólafsson, Steinar Sandøy, Chris Zimmerman, Jan Rene Larsen, Tom Christensen, Sara Longan, and Kári Fannar Lárusson

CBMP Leads: Tom Christensen, Sara Longan

CAFF Secretariat: Tom Barry, Courtney Price, Kári Fannar Lárusson, Hólmgrímur Helgason, and Olga Pálsdóttir

Authors of report: Jennifer Lento, Willem Goedkoop, Joseph Culp, Kirsten Christoffersen, Elena Fefilova, Guðni Guðbergsson, Kári Fannar Lárusson, Petri Liljaniemi, Jón S. Ólafsson, Steinar Sandøy, Christian Zimmerman, Tom Christensen, Patricia Chambers, Jani Heino, Seppo Hellsten, Maria Kahlert, Francois Keck, Sarah Laske, Danny Chun Pong Lau, Isabelle Lavoie, Brianna Levenstein, Heather Mariash, Kathleen Rühland, Emilie Saulnier-Talbot, Ann Kristin Schartau, Martin Svenning.

Contributors to report: Ekaterina Abramova, Per-Arne Amundsen, Jukka Aroviita, Tom Barry, Maria Baturina, Carolina Behe, Daniel Bogan, John Brittain, Bob Brua, Stéphane Campeau, Krista Chin, Catherine Docherty, Olga Dubovskaya, Frauke Ecke, Arni Einarsson, Jaakko Erkinaro, M. Falkegård, Frank Farsø Riget, Laura Forsstrom, Nikolai Friberg, Larisa Frolova, Ole Geertz-Hansen, Brian Hayden, Kerstin Holmgren, Þóra Katrín Hrafnsdóttir, Brian Huser, Haraldur R. Ingvarsson, Elena Ivanova, Dean Jacobsen, Ben Jones, I-R. Jonsson, Kimmo Kahilainen, Satu-Maaria Karjalainen, Jan Karlsson, Jennie Knopp, Elena Kochanova, Olga Kononova, Elena Kravchuk, Ånund Kvambekk, Torben Lauridsen, Olga Loskutova, Annette M Watson, Kjetil Melvold, Alexander Milner, Marit Mjelde, Heikki Mykrä, Gulnara Nigamatzyanova, Anna Novichkova, Panu Orell, Johan Östergren, Michael Power, Courtney Price, Ruslan Rafikov, Milla Rautio, Atso Romakkaniemi, Jukka Ruuhijärvi, Svein Jakob Saltveit, Susanne Schneider, Rebecca Shaftel, John Smol, Jan-Idar Solbakken, Heidi Swanson, Tobias Vrede, Matt Whitman, Jukka Ylikörkkö, Ivan Zuev.

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CIRCUMPOLAR BIODIVERSITY MONITORING PROGRAM

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Layout/Design: Maria Dýrfjörð, www.mariacreativestudio.com

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1. Executive Summary

Introduction

Arctic freshwater ecosystems (lakes, rivers, and associated wetlands) are threatened by climate change and human development that can affect freshwater biodiversity. Such effects will change not only the distributions and abundances of aquatic species, but also the lives of Arctic Peoples that are dependent on the ecosystem services supplied by lakes and rivers. Thus, the freshwater biodiversity program of the Circumpolar Biodiversity Monitoring Program (Freshwater-CBMP) focuses on lake and river ecosystems and has established a long-term monitoring framework for these Arctic freshwaters. Developed for the Conservation of Arctic Flora and Fauna (CAFF; the biodiversity Working Group of the Arctic Council), this framework facilitates more accurate and rapid detection, communication and response to significant trends in Arctic water quality and biodiversity. Freshwater-CBMP goals are addressed in the *"Arctic Freshwater Biodiversity Monitoring Plan",* which describes an integrated, ecosystem-based approach for monitoring Arctic freshwaters (Culp et al. 2012a).

This State of Arctic Freshwater Biodiversity Report (SAFBR) is the first circumpolar assessment of key biotic elements, or ecosystem components, in Arctic freshwaters. The overall aim was to assess the current status and trends of freshwater biodiversity by geographical regions across the circumpolar Arctic. Specific objectives were to use existing monitoring data to: 1) assess alpha and beta biodiversity; 2) identify geographical locations with high biodiversity (i.e., biodiversity hotspots); 3) where possible, determine the primary environmental and human stressors associated with the observed changes in biodiversity; and 4) identify key monitoring locations for inclusion in future circumpolar assessments of ecological change in freshwaters.

The primary biotic elements examined were Focal Ecosystem Components (FECs), which are biotic assemblages that are ecologically pivotal and/or sensitive to changes in biodiversity and/or environmental conditions, and that are routinely monitored in the Arctic (e.g., fish, benthic macroinvertebrates, zooplankton, planktonic algae, algae from benthic samples, and macrophytes). Ecosystem changes that would affect biodiversity of FECs were placed in the context of testable impact hypotheses (or predictions). These impact hypotheses outline a cause-effect framework that describes how FECs are expected to respond to anticipated change in environmental and anthropogenic stressors. For example, permafrost degradation is expected to result in harsher physical disturbance regimes that increase sediment loads and turbidity of rivers. A full set of these impact hypotheses is listed in the Freshwater Biodiversity Monitoring Plan (Culp et al. 2012a).

Biodiversity was assessed using existing data for FECs gathered from all available sources (i.e., academia, government, industry, and documented Traditional Knowledge gathered from systematic literature searches) for the contemporary period (1950 to present), and where possible, for the post-industrial period (1900 to 1950) and historical (pre-1900) periods. Centralized data sources were available in national monitoring databases for some countries, but even in these cases (e.g., Sweden, Norway), considerable data formatting and harmonization were required before the data could be compiled for the circumpolar region. Consolidated databases were very limited in other countries, which induced extensive data searches and recovery of government reports, published literature and industry registries and digitization/ harmonization before data could be added to the CBMP-Freshwater database. The extensive circumpolar freshwater database is a primary deliverable of the CBMP-Freshwater to CAFF, as it documents the underlying SAFBR data and will facilitate future assessments of change in Arctic freshwaters.

Abiotic Variables

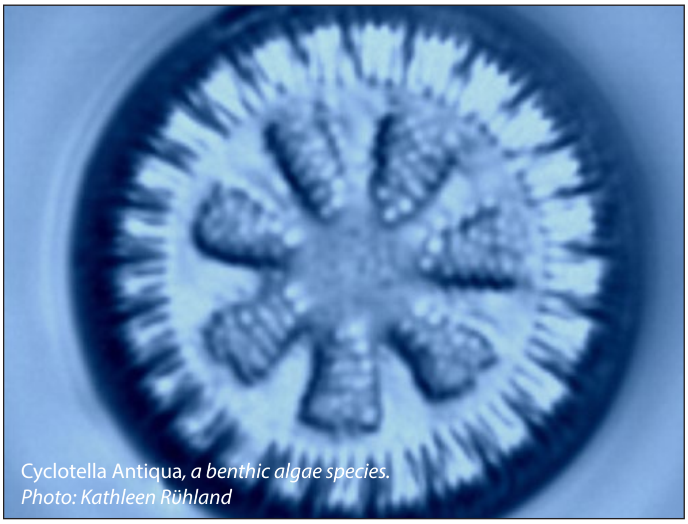
Lakes and rivers are closely interlinked with the surrounding landscape and reflect climate- and human-induced changes in land-use and development, with shifts in abiotic drivers of biodiversity being early warning indicators of ecological change. The Freshwater Monitoring Plan identifies nine major environmental and anthropogenic stressors to freshwater ecosystems that can be summarized as (1) permafrost thaw and changes in the hydrological regime resulting in higher loads of nutrients, solids, and organic matter; (2) long-range transboundary air pollutants and point source pollution originating from industrial development and urbanization; (3) fisheries over-harvesting; (4) climate-driven changes to riparian vegetation from grasses to shrub-dominated flora, i.e., greening of the Arctic; and (5) flow alterations and regulation due to hydropower dams and other forms of development that can lead to substantial habitat fragmentation and destruction. The Abiotic chapter of the SAFBR provides examples of long-term declines in ice-cover duration and increases in water temperature that have been observed in the Arctic. Long-term declines in total phosphorus concentrations are presented for major rivers in northern Sweden that illustrate the ongoing decline in freshwater nutrient concentrations (oligotrophication) of the Arctic/alpine regions of the Scandinavian Peninsula. In contrast to these slowly progressing changes are the rapid alterations of water turbidity and chemistry following the formation of permafrost thaw slumps, i.e., the collapse of landscape structures due to permafrost thawing. These examples highlight some of the various abiotic changes that are ongoing in Arctic landscapes and that affect water quality and biodiversity in lakes and rivers.

Scenarios of biodiversity change in Arctic freshwaters predict a net increase in biodiversity with warming temperatures, assuming dispersal routes exist for southern species to colonize northern regions. However, as water quality and habitat conditions shift to more closely resemble southern latitudes, this shift is expected to come with a reduction in the habitat range of cold-tolerant species endemic to the Arctic. In other words, along with an overall predicted increase in the number of species, there will be a net loss of unique Arctic-specific biodiversity. Alterations of habitat conditions originating from changes in air and water temperatures, permafrost extent, nutrient availability, and terrestrial vegetation will change the zonation of the Arctic region by globally decreasing the size of the sub-, low, and high Arctic regions, and by reducing habitats critical to cold-tolerant Arctic species. These alterations to aquatic biodiversity and food webs will ultimately induce changes to freshwater fisheries around the Arctic and to the ecosystem services that they supply to Arctic residents.

Biodiversity Assessment

Spatial patterns in diversity were assessed for each FEC for the circumpolar region by using a regionalization approach. Stations were grouped into climate-based terrestrial ecoregions, and patterns of alpha and beta diversity were evaluated within and among ecoregions. Alpha diversity (the number of taxa – species-level or higher, depending on the FEC) was assessed by using rarefaction curves to estimate taxonomic richness at a set number of stations within each ecoregion, in order to correct for variation in sampling effort across the Arctic. Comparisons of rarefied alpha diversity across ecoregions were used to assess broad spatial patterns. Beta diversity (change in species composition across stations) was assessed within ecoregions by grouping stations at a smaller spatial scale (hydrobasins, which are standardly-derived catchments) and estimating the relative contributions of turnover (replacement of taxa with new/different taxa across stations) and nestedness (with some stations containing a subset of the same taxa found at the richest stations) to beta diversity. Circumpolar and regional analyses were conducted on data with harmonized taxonomic names.

Algae from Benthic Samples


Cyclotella Antiqua, *a benthic algae species.*
Photo: Kathleen Rühland

Algae are key primary producers in Arctic freshwaters, and benthic samples include diatoms and a number of classes of other algal groups. This assessment focused on diatoms, as this is a major group in Arctic freshwaters and data availability was high. Lake diatom stations were the most evenly distributed across the circumpolar region of all the FECs, although coverage was patchy in Russia and lacking in the High Arctic of Greenland or Svalbard. The highest alpha diversity for lake sediments was found at low- to mid-level latitudes and in coastal ecoregions, including coastal Alaska, the Arctic archipelago and southern coast of Hudson's Bay in Canada, Iceland, and Norway. Beta diversity indicated that there was generally moderate to high dissimilarity in community structure among lake stations. Lake beta diversity was dominated by the turnover component in all ecoregions indicating that there was a high degree of species replacement across stations. The highest alpha diversity of river diatoms was in coastal Alaska and western Canada, and high diversity was also evident in Fennoscandian ecoregions. The lowest alpha diversity was found in eastern and southern Canadian ecoregions, which had on average half as many diatom taxa as in the most diverse ecoregions. Beta diversity within an ecoregion was highly variable for river diatoms, but turnover was the predominant component of beta diversity for river diatoms.

Samples with the highest diatom richness for both lakes and rivers were generally between 60-75°N latitude. However, the decline in richness outside this latitudinal range was small, and partly due to the fact that fewer samples were collected at the highest latitudes (above 75°N), particularly in rivers. Diversity was lower in the high Arctic than in the sub- or low Arctic, particularly for lakes, and analysis identified groups of taxa in both lakes and rivers that were characteristic of high latitude samples. Diatom taxa that were dominant across the circumpolar region are generally also common to other regions of the world. This is consistent with the observation that although temperature may affect diatom diversity, the distribution of species is also driven by local geology and water chemistry conditions. Many of the taxa found across the Arctic are typical of waters with low nutrient levels and neutral pH, although indicators for nutrient-rich conditions were also found. Assessment of paleolimnological data indicated that temporal change in diatom assemblage composition was lowest in the eastern Canadian Arctic, which has historically been subjected to less warming than other areas of the Arctic. Shifts in dominant taxa over time were indicative of strong community changes, likely due to changes in the thermal stratification regimes of lakes since circa 1800.

Lake diatoms are so far not generally included as part of routine monitoring programs, and thus assessment must rely on academic data. Although time series for these data are largely absent, the advantage of diatom samples in lakes is that long-term changes can be inferred from diatoms stored in sediment cores. However, the collection of cores should be expanded to a broader spatial area across the Arctic to facilitate broad-scale assessment of long-term trends for the circumpolar region. River samples were more sparse than lake samples, and were lacking from Russia, Iceland, Greenland, Svalbard, and central and western Canada. Although river algae monitoring is done routinely in some Arctic countries (e.g., Norway, Sweden, Finland), it is limited elsewhere in the circumpolar region. Furthermore, even in countries where monitoring occurs, the samples may not always be comparable if they focus on soft algae (non-diatoms, e.g., in Norway) or do not follow comparable sample processing procedures. Thus, there is a clear need to increase the spatial scope of river diatom monitoring in order to better capture biodiversity of this important group across the circumpolar region.

Phytoplankton

Achnanthes minutissima.
Photo: Chris Carter

Phytoplankton are microscopic algae that are suspended in the water column, and include diatoms and a number of non-diatom algal taxa. Assessment of rarefied alpha diversity within ecoregions indicated that phytoplankton diversity was highest in Fennoscandia and lowest in Russia and the Canadian High Arctic. Beta diversity was high in a number of ecoregions in Alaska, Russia, Fennoscandia, and southern Canada. Ecoregions in these areas showed the highest differentiation in phytoplankton assemblages and large among-lake differences in water body types (e.g., size/depth and water quality). Low and high Arctic lakes generally had higher beta diversity than sub-Arctic lakes. Turnover was the predominant component of beta diversity in all ecoregions, which is indicative of the introduction of new species across stations. This result suggests that spatially extensive monitoring of lake phytoplankton is required to provide reliable estimates of species turnover and biodiversity.

Cyanobacteria, which often include toxin-producing species, did not show long-term unidirectional trends in biovolume. However, there were similar peaks in Cyanobacteria biovolume across a number of lakes during years with high temperatures, with two-thirds of the Cyanobacteria peaks happening during one of the 10 hottest years on record. Since rising temperature and decreased ice cover potentially enhance cyanobacterial dominance (Paerl and Huisman 2008), continued monitoring of cyanobacteria in all Arctic regions may be useful in tracking associated climate and nutrient changes in Arctic water bodies. Long-term monitoring data for the full phytoplankton assemblage indicated a decrease in total biovolume in a highly productive lake in Greenland, while conversely, biovolume in a number of low productivity lakes in Finland and Sweden increased. If these trends continue into the future, phytoplankton biovolume will be expected to be more similar across these Arctic lakes.

Phytoplankton are not regularly monitored in all Arctic countries, therefore, data are patchy both in spatial and temporal coverage. The most extensive monitoring occurs in Fennoscandia and Greenland. In contrast, very little sampling occurs in the high Arctic and there is a need for increased monitoring across North America, Russia, and other northern areas of the Arctic. Future monitoring efforts for lake phytoplankton must improve consistency in sample processing methods, particularly with respect to the estimation of biovolume, and improve taxonomic resolution to the species-level where possible.

Polygonal lake, Khibiny mountains, Russia
Photo: Gregory A. Pozhvanov/Shutterstock.com

Macrophytes

Water milfoil (Myriophyllum alterniflorum*).*
Photo: Mps197/Shutterstock.com

Macrophytes (macroscopic water plants) are primary producers that act as a food resource and supply habitat structure for other aquatic organisms. The highest alpha diversity of macrophytes was in Fennoscandian lakes. Alpha diversity was lowest at high latitudes and remote locations such as the Canadian High Arctic, Greenland, Iceland, and the Kola Peninsula. Three of the ecoregions with the lowest species richness had an average latitude > 70°N, suggesting that alpha diversity of macrophytes declines in high-latitude Arctic regions. The most common taxa across all stations were *Myriophyllum alterniflorum, Potamogeton gramineus,* and *Ranunculus reptans.* Aquatic moss species comprised a higher percentage of total species richness with increasing latitude.

For most ecoregions, turnover was the dominant component of beta diversity as it accounted for more than 70% of the total beta diversity. This indicates that variation in diversity within an ecoregion was due to finding different species across stations, and emphasizes the importance of increasing sample coverage. Beta diversity of macrophyte assemblages ranged between 0 (no inter-station differences in species composition) and 1 (no inter-station overlap in species) within the ecoregions. Macrophyte beta-diversity was largely driven by ecoregion connectivity, with remote ecoregions generally having lower beta diversity.

Extensive macrophyte data were available for some areas of the Arctic (e.g., Fennoscandia), but data were sparse for large areas of Canada, Alaska, and Russia. Macrophyte monitoring is not part of regular assessments in Canada, Alaska, and Russia, thus limiting the spatial scope of available data. Across the entire circumpolar region, there are very few lakes that are monitored regularly. As a result, time series data are generally not available, and many lake observations are outdated (e.g., 1970s or earlier) with no repeated visits to the same lakes. Such data do not allow for the detection of shifts in macrophyte distribution and may not provide an accurate view of contemporary patterns in diversity. Moreover, monitoring may not include the identification or enumeration of aquatic mosses, helophytes, or bryophytes, which may be of particular concern if these groups are dominant in a region, as often occurs in the sub- and high Arctic. Improvements to the monitoring of macrophytes are necessary across the circumpolar region, and should focus on regular and repeated monitoring of representative lakes with standardized monitoring protocols.

Zooplankton

Daphnia longispina.
Photo: Deiter Ebert

Zooplankton are microscopic invertebrates that live suspended in the water column and provide an important food source for fish in lakes. Zooplankton include crustacean taxa and rotifers, the latter of which are often not identified in samples. Crustacean zooplankton showed the highest alpha diversity for lakes in northern Russia, Fennoscandia, and Alaska. A limited set of stations with rotifer information indicated that rotifers added a small to moderate number of taxa to regional zooplankton diversity. Assessment of the full zooplankton assemblage provided evidence of high alpha diversity in coastal regions, particularly in Fennoscandia, Russia, and Alaska. This pattern is consistent with predictions that high richness would be found in areas that were unaffected by recent glaciation (e.g., Alaska) and in coastal areas (Rautio et al. 2008, Samchyshyna et al. 2008).

Beta diversity of zooplankton (crustaceans and rotifers) varied, with some ecoregions in Alaska, Russia, and Fennoscandia indicating high assemblage differences among lakes, and other ecoregions in the high Arctic or where few lakes were sampled indicating low differences in species composition among lakes. These findings highlight the importance of monitoring zooplankton in a wide variety of lakes within an ecoregion, to ensure the full diversity in an ecoregion is captured. Diversity was generally dominated by species turnover in ecoregions where more lakes were sampled over a wider spatial extent. Consequently, widespread sampling would be necessary to accurately summarize the full diversity of species in an area and ensure differences among lakes were captured.

The most diverse groups in the zooplankton dataset were the calanoid copepods, cyclopoid copepods, cladocerans, and rotifers. Common species of rotifers and crustaceans are also common and abundant outside the Arctic. Cladocerans were numerically dominant in sub-Arctic lakes (approximately 50% of all specimens), however, this group decreased in the presence of cyclopoid copepods in the low Arctic and high Arctic. The relative abundance of calanoid copepods was similar between the sub-Arctic and low Arctic, and declined in the high Arctic zone. Ongoing climate change may provide opportunities for the spread of Eurasian species, such as Bythotrephes longimanus and Limnosida frontosa, to the North American continent and lead to potential shifts in biodiversity and food web structure.

Greenland and Norway are the primary regions with routine monitoring at established stations for zooplankton, whereas data from other regions often come from environmental impact studies (e.g., Canada) rather than long-term programs intended to evaluate natural variation or monitor for effects of climate change. The lack of data in some European countries may be due to the fact that zooplankton are not considered an *"ecological quality element"* according to the European Water Framework Directive and thus have lower priority in monitoring. The necessary reliance on data from academia, industry, or other non-governmental organizations means that there are few time series, and in some areas, limited sampling of the full zooplankton assemblage (e.g., areas with research focused on Crustacea or just on cladocerans or copepods). Future monitoring efforts should be based on a set of permanent monitoring sites covering all climatic regions in each country, with an aim to standardize collection methods and the habitats sampled.

Benthic Macroinvertebrates

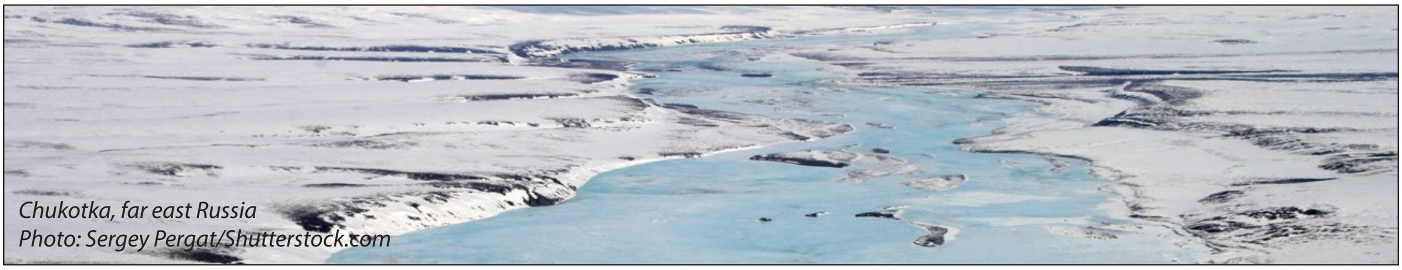

Baetidae (top) and Heptageniidae (bottom)
Photo: Jan Hamrsky

Benthic macroinvertebrates are macroscopic invertebrates (predominantly insects) that live on the bottom of lakes and rivers and provide an important food source for fish. Alpha diversity of lake littoral (near-shore) habitats showed strong regional differences, with the lowest alpha diversity in remote areas and islands (e.g., Greenland, Iceland, Faroe Islands, Wrangel Island) and the highest taxonomic richness in Fennoscandia and the coastal regions of Alaska. Similarity in diversity estimates for the most taxonomically-poor ecoregions suggests that barriers to dispersal, such as proximity to mainland and presence of mountains, limit biodiversity in these northern lakes. Beta diversity within ecoregions was variable, with a higher importance of species loss evident in remote island ecoregions. Macroinvertebrate diversity in the lake profundal (deep water) zone habitat was lower and less variable than littoral zone observations; nevertheless, circumpolar trends showed a similar pattern.

Alpha diversity of river macroinvertebrates was lowest at the highest latitudes and on remote islands (e.g., Canadian high Arctic, Svalbard, Greenland, Iceland, Wrangel Island). Diversity also appeared to be lower in mountainous ecoregions. Conversely, the highest alpha diversity was observed at the lowest latitudes on the mainland where connectivity does not affect dispersal of taxa from southern regions and thermal regimes are the warmest. Beta diversity for rivers was high within all ecoregions, and taxonomic nestedness (loss of species) contributed more to beta diversity in high latitude, high altitude, and remote island ecoregions.

Further analysis of alpha diversity in lakes in rivers in relation to latitude indicated a strong latitudinal decline in both rivers and lake littoral zones above 68°N. Declines were likely a result of high-Arctic environments exceeding the thermal tolerances of taxa. In rivers, variability in this pattern at the mid-latitudes was associated with a west-east temperature gradient that exists in North America and colder thermal regimes in the eastern Canadian Arctic relative to similar latitudes in Fennoscandia. Lower diversity was also evident where dispersal was limited. This was particularly evident in lakes located on islands, where diversity was consistently lower than mainland stations, even at similar latitudes.

Monitoring gaps for benthic invertebrates of lakes and rivers are largely related to the need for harmonized sampling design and method. River benthic macroinvertebrate data were among the most extensive of all FECs with good spatial coverage across the circumpolar region, and with a relatively standardized sampling method. However, single-event sampling of riverine macroinvertebrates was common, and with the exception of Sweden, time series data were scarce. In lakes, there were large gaps in the spatial coverage of benthic invertebrate data due to a lack of routine monitoring in many areas, and because the sampled habitats (e.g., near-shore vs. deep-water zones, which have different assemblages of benthic macroinvertebrates) and sampling methods varied by country. To support future macroinvertebrate assessment in lakes, countries need to standardize the sampling approach, ideally including sampling of the taxonomically-rich littoral habitat. An additional limitation to the strength and scope of diversity assessment for both rivers and lakes is the current inconsistency in the taxonomic resolution, particularly for midges (chironomids), which are predominant in the Arctic. Future assessments should continue to make use of the strong spatial coverage of data and accessibility of data from national databases, but monitoring activities must include higher taxonomic resolution of the Chironomidae (i.e., to sub-family using microscopic techniques or to species-level using genetic barcoding) and schedule regular re-sampling of areas to establish the time-series data required to assess the impacts of climate change and development.


Chukotka, far east Russia
Photo: Sergey Pergat/Shutterstock.com

Sarek National Park, Jokkmok, Sweden"
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Fish

Arctic Char
Photo: Dan Bach Kristensen/Shutterstock.com

Freshwater fish are ecologically, socially, and economically important in the Arctic, and more information is known about the distribution and diversity of fish species in Arctic lakes and rivers than is known about other FECs. Within the ecoregions included in this assessment, 100 fish species are known to occur. Large-scale alpha diversity varied among ecoregions, ranging from a single species in the high Arctic to as many as 47 species in Fennoscandia. Fish alpha diversity varied across continents with northern and mountainous ecoregions having lower diversity. Islands (e.g., Iceland, Greenland) had fewer fish species due to biogeographic constraints.

Fourteen species of fish had a distributional range across continents - including salmonids, smelts, sticklebacks, freshwater cod, pike, and lamprey. Three additional species (all Salmonids) have been introduced to Fennoscandia and Russia from North America. Longitudinal distribution patterns of fish species showed a marked decline in the Atlantic zone, from generally more than 50 species in North America to many stations with less than 50 species in Fennoscandia. Our analysis also showed that alpha diversity at latitudes above 72°N declined to a single species, Arctic charr, although more species are known to occur.

Beta diversity differed across ecoregions, with higher values in Alaska and inland Fennoscandia. The turnover component of beta diversity was dominant in ecoregions in these areas. This indicates that the replacement of species across spatial or environmental gradients drives diversity patterns across a range of ecoregion types in North America and Fennoscandia, including alpine and taiga habitats. The nestedness component of beta diversity was greater only in Iceland, where only three species were represented in the data, and changes in species composition across the region would result from sub-setting the richest fish community.

While fish are key species in aquatic ecosystems and are important to communities of the North, it is evident that there are significant gaps in monitoring effort and data coverage across the circumpolar region. Although in some cases the spatial extent is limited because existing datasets were not accessible, there remain significant gaps in monitoring effort and coordination of routine monitoring in some areas. Across Canada, for example, a large number of historical studies focused on monitoring commercial or subsistence fisheries, and thus quantified a selection of fish species rather than assessing the diversity of the full assemblage. Furthermore, many sites across North America have only been sampled one time, thus precluding temporal analyses of trends. Similarly, there are large areas that have not been sampled sufficiently to allow for analyses of spatial patterns or temporal trends. Until broader spatial and temporal data coverage is available, the ability to assess changes in biodiversity, especially at large spatial scales, will be limited.

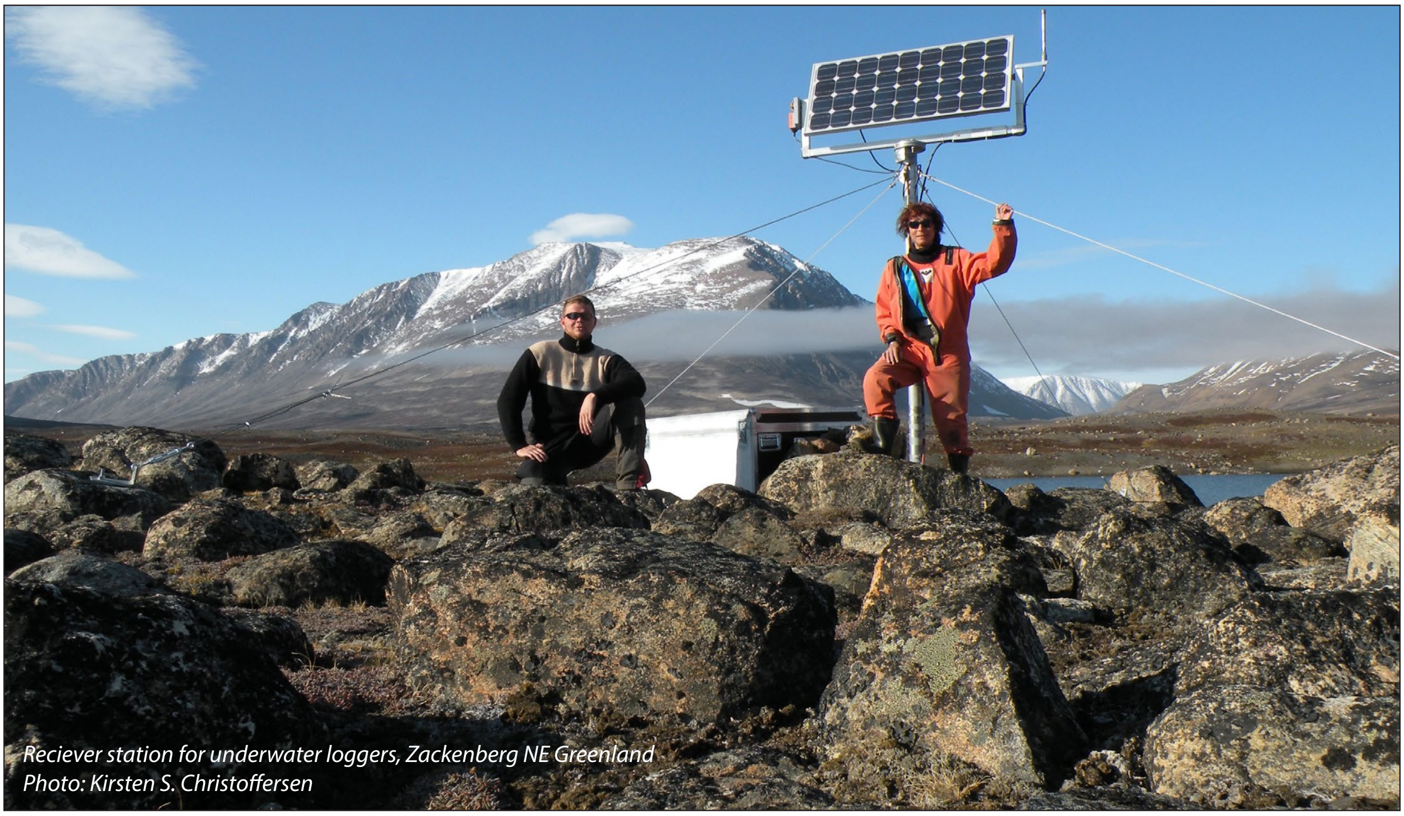
Freshwater Biodiversity Synthesis

Warming temperatures in Arctic rivers and lakes will likely lead to an increase in biodiversity as southern species expand their range northwards, and cold stenotherms are extirpated from waters that exceed their thermal tolerance threshold. Where cold-water endemic species are limited to the Arctic region, this will result in global losses of these species, e.g., for fish such as Arctic charr. A warmer and wetter climate will also increase rates of mineral weathering, decomposition of soil organic matter, erosion and sedimentation. This likely will lead to higher concentrations of organic matter, minerals, and nutrients. Such change in key drivers of the freshwater environment can affect large-scale processes (e.g., brownification, nutrient enrichment, sedimentation) of lake and river ecosystems leading to changes in alpha and beta diversity and ecosystem productivity.

We compared spatial diversity patterns among FECs to identify areas of the Arctic with consistently high or low diversity. Fennoscandian lakes represented a diversity hotspot for macrophytes, zooplankton, benthic macroinvertebrates, and fish. The warmer climate in Fennoscandia and strong connectivity to the mainland may play a role in the overall high diversity of the area. The coastal ecoregion in Alaska and western Canada ranked as the most diverse for lake diatoms and phytoplankton, and one of the most diverse ecoregions for lake fish. Connectivity of the Alaskan coastal region and lack of recent glaciation in that area may have contributed to high diversity of lake diatoms, phytoplankton, and fish. Ecoregions in Canada, Greenland, and Iceland were generally less diverse for many of the lake FECs.

Similar results were obtained when the diversity of river FECs were compared across ecoregions. Fennoscandia was overall the most diverse region across diatom, benthic macroinvertebrate, and fish FECs, though the coastal ecoregion in Alaska and western Canada showed the highest diversity of diatoms and fish. As observed for lakes, river diversity in the mountainous ecoregions of Alaska and western Canada was low, suggesting an impact of harsh environmental conditions associated with higher elevations. Alaskan ecoregions south of the Brooks-British Range ranked low for fish diversity, possibly reflecting the effect of dispersal barriers to anadromous species immigrating from the diverse Arctic Coastal Tundra. Eastern and northern Canada, which have colder long-term average temperatures than western North America or Fennoscandia, had the lowest diversity of river diatoms and benthic macroinvertebrates.

Regional evaluations of the relationships between FECs and environmental drivers revealed the importance of temperature as an overriding driver for multiple FECs in both lakes and rivers. For example, latitudinal and longitudinal patterns in river benthic macroinvertebrates reflect temperature gradients across the North American Arctic. Other factors related to dispersal, glaciation history, and bedrock geology were also identified as important drivers of diversity in North American river FECs. In Fennoscandia, FECs in lakes were strongly influenced by climatic drivers (e.g., latitude, temperature, precipitation) and vegetation cover. The drivers in both regions include both large-scale, slowly progressing landscape-level processes that will have long-lasting effects, as well as rapid modifications which have more local and short-term effects. The concerted action of these environmental drivers, and their subsequent effects on biological assemblages, will depend on regional conditions. Slow response times will make some of these processes progress for decades to come, while others may induce sudden biological shifts with strong repercussions on aquatic ecosystems when critical threshold levels are exceeded. These analyses form the baseline against which future assessment can be compared, and begin to address some of the impact hypotheses in the freshwater biodiversity monitoring plan (Culp et al. 2012a).



Reciever station for underwater loggers, Zackenberg NE Greenland
Photo: Kirsten S. Christoffersen

State of Monitoring and Advice

Chapter 6 of the SAFBR provides an overview of ongoing freshwater monitoring activities in the Arctic countries and summarizes the various parameters measured in the Arctic countries. This overview illustrates the large differences in the organization of monitoring by the each country, the FECs monitored, and the spatial coverage of monitoring in the Arctic. We demonstrate that the availability and coverage of data varied among the Focal Ecosystem Components. Lake ecosystems are not routinely monitored for many FECs in large countries such as Russia, Canada and the US because monitoring is dependent on irregular or insecure funding. However, Canada, Greenland, and Iceland have a monitoring focus on fish monitoring. In contrast, the Fennoscandian countries have well-established monitoring programs for lake FECs based on secure funding (e.g., Water Framework Directive) although the spatial coverage is poor for some FECs. Monitoring of river FECS shows a similar trend except that Canada routinely monitors the benthic macroinvertebrate and fish FECs. In general, the Arctic countries monitor abiotic parameters in rivers to a much greater extent than in lake ecosystems.

Freshwater biomonitoring has traditionally focused on the assessment of ecosystem health and pollution-effects, and has used standardized sampling effort and sample processing to reduce observation variability and increase ability to detect ecological change. While this type of monitoring can be used to estimate biodiversity, these techniques are not designed to measure the full biodiversity of a site because they can underestimate the presence of rare species. Future monitoring must focus on harmonized methods, with sampling in a sufficient number of stations across representative ecoregions to support the detection of trends related to testing impact hypotheses. Chapter 6 suggests a number of improvements for future monitoring in the Arctic that build on the long tradition of bioassessment in freshwaters and that include community engagement. More specifically, we provide the following key recommendations for consideration in future biodiversity monitoring of freshwater ecosystems in the Arctic:

Emerging Approaches

- Incorporate Traditional Knowledge as an integral part of future circumpolar monitoring networks.
- Engage local communities in monitoring efforts through Citizen Science efforts.
- Include an increased focus and use of remote sensing approaches.
- Make use of recent advances in environmental DNA (eDNA) methods and genetic barcoding.

Future Monitoring Methods

- Further harmonize sampling approaches among countries, and select appropriate sampling methods and equipment to balance between maintaining consistency and comparability with historical data and alignment with common methods used across the circumpolar region.
- Develop supplementary monitoring methods that provide better standardized estimates of biodiversity to maximize the likelihood of detecting new and/or invasive species.
- Use a regionalization approach based on ecoregions to guide the spatial distribution of sample stations and, ultimately, to provide better assessments.
- Ensure that spatial coverage of sampled ecoregions is sufficient to address the overarching monitoring questions of the CBMP across the circumpolar region, maintain time series in key locations, and fill gaps where monitoring data are sparse.
- Ensure the number of monitoring stations provides sufficient replication within ecoregions and covers common water body types.

Future Monitoring Design and Assessment

- Arctic countries should establish a circumpolar monitoring network based on a hub-and-spoke (intensive-extensive) principle in remote areas.
- Experimental design for the hub-and-spoke network should largely focus on addressing the Impact Hypotheses developed in the CBMP freshwater plan to increase focus on assessing biotic-abiotic relationships in Arctic freshwater systems.
- The Freshwater Steering Group of the CBMP should continue to serve as the focal point for the development and implementation of pan-Arctic, freshwater biodiversity monitoring.
- There should be a focus on continuing monitoring efforts at stations with existing time series, as these stations form key sites for future evaluations of temporal changes.
- Resources must be provided to maintain and build the freshwater database for future assessments in order to maximize the benefits of this database
- Arctic countries should make better efforts to document and preserve data from short-term research projects and research expeditions, as well as from industrial, university and government programs.
- Due to the patchy nature of sampling, future assessments require the continued use of rarefaction curves for scientifically-sound comparisons of alpha diversity across ecoregions.

Considering the rapid changes occurring in Arctic ecosystems, there is an urgent need for the CBMP-Freshwater of CAFF to continue building baseline databases to aid the assessment of future biodiversity change. In addition, harmonization of monitoring efforts among Arctic countries and a greater focus on Arctic lakes and rivers should be a strategic goal. Lastly, we stress that status assessments of Arctic lakes and rivers must explore the close association of biodiversity with spatial patterns of physico-chemical quality of aquatic habitats that can drive biological systems.

Content

Landmannalaugar, Iceland
Photo: Corepics/Shutterstock

2. Introduction

2.1. Monitoring Freshwater Biodiversity in a Changing Arctic

The State of Arctic Freshwater Biodiversity Report (SAFBR) is the first Circumpolar Biodiversity Monitoring Program (CBMP) assessment to summarize the status and trends of key biotic elements, or Focal Ecosystem Components (FECs), in the Arctic freshwater environment. The assessment used existing data for FECs gathered from all available sources (i.e., academia, government, industry, and documented Traditional Knowledge collected through systematic literature searches) to improve the detection and understanding of changes in circumpolar freshwater biodiversity. The CBMP-Freshwater effort represents the first international initiative to develop an integrated, ecosystem-based approach for monitoring Arctic freshwater biodiversity.

Although Arctic freshwater ecosystems have been defined by the CBMP as rivers, streams, lakes, ponds, and their associated wetlands (Culp et al. 2012a), this assessment focuses only on rivers, streams, lakes, and ponds due to a lack of monitoring data for wetlands. These environments are threatened by climate change and human development that can affect freshwater biodiversity (Wrona et al. 2013). Climate-related increases in air temperatures can thaw permafrost, change ice cover regimes, increase growth and spatial coverage of terrestrial vegetation (e.g., shrubification), and modify hydrological processes including water balance. In glacially-fed systems, climate change is expected to lead to significant changes in community structure and function along down-stream longitudinal gradients as the loss of glaciers affects hydrological and thermal regimes of receiving waters (Milner et al. 2017). Biodiversity shifts in Arctic regions may cause more significant changes to ecosystem function than in lower latitudes because of the low functional redundancy in these remote locations (Post et al. 2009), as warming and glacial retreat are expected to lead to increases in species richness in concert with increased functional diversity (Brown and Milner 2012, Brown et al. 2018). Overall, the distributions and abundances of freshwater species in Arctic freshwaters, as well as the lives of Arctic Peoples, are expected to be altered in response to such environmental regime shifts (see section 3.3).

2.2. The Circumpolar Biodiversity Monitoring Program (CBMP)

The CBMP is the cornerstone program of the Conservation of Arctic Flora and Fauna (CAFF; the biodiversity Working Group of the Arctic Council), and is organized into Marine, Freshwater, Terrestrial, and Coastal ecosystem groups that develop CBMP monitoring plans and authoritative assessments. The program was developed to improve long-term monitoring of Arctic biodiversity to facilitate more rapid detection, communication and response to significant trends in biodiversity, and to identify the factors driving those trends (Barry et al. 2013). It is an international network of scientists, governments, indigenous organizations, and conservation groups working to harmonize and integrate efforts to monitor the Arctic's living resources. This adaptive monitoring program incorporates management questions, conceptual ecological models, experimental monitoring design, data collection and reporting. In addition, the CBMP aims to gather data from both Traditional Knowledge (TK) and science, and make this information more readily available to policy-makers and the public in order to improve conservation and management of the Arctic's natural resources. The design adopted by the CBMP follows the steps required for an effective and adaptive scientific and ecosystem-based monitoring program (Lindenmayer and Likens 2009), and includes a consideration of what future priority questions and user needs should be addressed by the program (Figure 2-1). Future questions will be guided by the CAFF Board and other Arctic biodiversity data users (Barry et al. 2013, Christensen et al. 2018).

Disko field work
Photo: Kirsten S. Christoffersen

2.3. The Arctic Freshwater Biodiversity Monitoring Plan

CBMP goals for freshwater ecosystems were addressed by the Freshwater Expert Monitoring Group (CBMP-Freshwater) who developed an integrated, ecosystem-based approach for monitoring Arctic freshwater biodiversity (Culp et al. 2012a). The monitoring principles followed the current practice of assessing the distribution and abundance of biota (i.e., biodiversity) in relation to the physical and chemical environmental conditions of freshwater ecosystems. The plan details the rationale and framework for monitoring circumpolar Arctic freshwaters with the aim of harmonizing freshwater biodiversity monitoring activities across the Arctic countries. It is designed to increase the spatial and temporal extent of monitoring data and improve monitoring in Arctic freshwaters where representative biological data sets and long time series are mostly lacking. Data from these programs can be used over the long-term to produce status and trend assessments for Arctic freshwaters. An intent of this approach is to stimulate future research initiatives to improve predictions for environmental change in Arctic freshwaters, facilitate implementation of long-term monitoring strategies, and improve reporting on the state of Arctic freshwater ecosystems. The use of TK should be increased in the future because, to date, TK has received limited attention as a result of limited funding support and human capacity.

2.3.1. Arctic freshwater ecosystems

Freshwater ecosystems are an integral part of Arctic landscapes. For example, some 121,000 lakes are found within the land mass defined by the CAFF-boundary (Figure 2-2), of which 68% are situated in Canada and 21% in Russia. Among the Arctic countries, lakes on average cover 3.7% of the land area (range 0.5 on Greenland to 8.8 in Canada), stressing their importance in Arctic landscapes. Lake and river ecosystems reflect changes and activities in their catchments, thus these systems and their biota can be used to detect ecological shifts at large spatial scales. Freshwater biodiversity for this assessment was evaluated within the Arctic Biodiversity Assessment (ABA) and CAFF spatial boundaries, with the Arctic divided into sub-regions (high Arctic, low Arctic, sub-Arctic; Figure 2-2). These Arctic sub-regions cover a wide range of biomes from glaciers and permafrost areas to northern forests, have relatively uniform biogeographical characteristics and are typically characterised by low biodiversity and relatively simple food webs. Assessments of individual Focal Ecosystem Components (FECs; see section 2.3.2) were conducted at a circumpolar as well as at the country or regional scale. Several countries were grouped within regional subdivisions based on geographic proximity (i.e., North America, Fennoscandia including Svalbard).

2.3.2. Focal Ecosystem Components and Impact Hypotheses

Focal Ecosystem Components (FECs) are biotic taxa that are ecologically pivotal, charismatic and/or sensitive to changes in biodiversity and/or environmental conditions. Arctic freshwater experts chose the most representative FECs to be used as practical indicators of Arctic freshwater ecosystem health. Expert consensus identified these FECs as central to the functioning of an ecosystem, sensitive to potential stressors, and most likely to be commonly represented in existing databases for the circumpolar Arctic (Table 2-1). For example, although microbial assemblages are important for biogeochemistry of freshwaters, they are not routinely monitored in the Arctic and thus cannot be assessed across the circumpolar region at this time. FECs are placed in the context of expected ecosystem change through the development of testable impact hypotheses (or predictions) that outline a cause-effect framework regarding how change in environmental and anthropogenic stressors is expected to affect FECs. A full set of impact hypotheses has been described in the Freshwater Biodiversity Monitoring Plan (Culp et al. 2012a). For example, permafrost degradation is expected to result in increased sediment loads and turbidity of rivers (i.e., Sediment Regime Change), thus negatively affecting the light and physical disturbance regimes of rivers. More examples of impact hypotheses that specifically act on the various FECs are given in section 3. Testing of these hypotheses requires targeted assessments that are designed to detect impacts of the stressors of interest, or long time-series that can indicate temporal stressor-response patterns. As such, it may not be possible to test all of the impact hypotheses with the data that have been collected for this report (e.g., hydrologic regime changes due to flow regulation or glacial retreat). However, assessment of the current status of biodiversity across the Arctic provides a baseline with which future data can be compared, and the impact hypotheses provide both guidelines for future scientific data collection and a focus for management decision-making.

Table 2-1 Biotic Focal Ecosystem Components (FEC) selected for inclusion in Arctic freshwater monitoring and assessment.

Focal ecosystem Component	Ecosystem
Fish	Lakes and rivers
Benthic Invertebrates	Lakes and rivers
Benthic agae	Lakes and rivers
Zooplankton	Lakes
Phytoplankton	Lakes
Macrophytes	Lakes

A list of indicators previously used in freshwater monitoring was considered for the assessment of FECs, however, taxa presence/absence and abundance (numerical and biomass) were chosen because they provided the maximum coverage across the Arctic (Table 2-2). The indicators used built on those employed by the Marine Expert Monitoring Group (Gill et al. 2011) with key criteria being that they were: 1) sensitive to environmental change and anthropogenic stressors; 2) scientifically valid and relevant; and 3) likely to be monitored into the future. These parameters allowed estimation of several indicators of ecological structure, including alpha diversity and beta diversity.

Figure 2-1 The CBMP takes an adaptive Integrated Ecosystem based Approach to monitoring and data creation. This figure illustrates how management questions, conceptual ecosystem models based on science and Traditional Knowledge (TK), and existing monitoring networks are designed to guide the four CBMP Steering Groups (marine, freshwater, terrestrial, and coastal) in their development. Monitoring outputs (data) are designed to feed into the assessment and decision-making processes (data, communication and reporting). The findings are then intended to feed back into the monitoring program.

Figure 2-2 Arctic freshwater boundaries from the Arctic Council's Arctic Biodiversity Assessment developed by CAFF, showing the three sub-regions of the Arctic, namely the high (dark purple), low (purple) and sub-Arctic (light purple), and the CAFF boundary (red line).

Table 2-2 List of monitored parameters for Focal Ecosystem Components (FECs) and the list of potential indicators and indices for each FEC in lake and river ecosystems. For the State of Arctic Freshwater Biodiversity Report, the most widely available data for Arctic freshwaters included information on taxa presence/absence and taxa abundance (numerical and/or biomass). The ecosystem to which each FEC applies can be found in Table 2-1.

FECs	Monitored	Indicators/Indices
Benthic algae and phytoplankton	Number of individuals or biomass of each taxon	Community indices (e.g., abundance and density, taxonomic richness, diversity and dominance, biomass and numbers of keystone taxa)
		Numbers of red-listed (endangered) and rare taxa
		Distribution and range (e.g., latitudinal and altitudinal)
	Biomass (including chlorophyll a and biovolume)	Bulk algal biomass
		Size structure of entire population or of keystone taxton
Fish, benthic macro-invertebrates and zooplankton	Number of individuals or biomass of each taxon	Community indices (e.g., abundance and density, taxonomic richness, diversity and dominance, biomass and numbers of keystone taxa, ecological traits)
		Numbers of red-listed (endangered) and rare taxa
		Distribution and range (e.g.,latitudinal and altitudinal, residency/anadromy for fish)
	Genotypes and alleles (fish)	Genetic diversity
	Biomass (including biovolume, length, and weight)	Size structure of an entire population or of keystone taxton
	Age of individuals	Age structure of entire population or of a keystone taxon; growth rates (size at age or age at length (fish), or life cycle stage at length (BMI) and age at maturity (age combined with biomass)
	Timing of key life history events	Migratory phenology
		Emergence timing
	Body burden of key contaminants in fish	Concentrations of contaminants in fish tissues above consumtion guidelines or above environmental thresholds for sub-lethal or lethal effects
Macrophytes	Areal cover of each taxon	Community indices (e.g., abundance and density, taxonomic richness, diversity and dominance, biomass and numbers of keystone taxa, ecological traits)

2.3.3. Global Linkages of Freshwater CBMP

The size and nature of Arctic ecosystems make them critically important to the biological, chemical and physical balance on a global scale (Meltofte 2013). Therefore, CAFF makes significant efforts to develop strategic partnerships and ensure that Arctic biodiversity information provides added value to other Arctic Council and related global activities and forums. This approach helps CAFF contribute to the attainment of global biodiversity goals, targets and commitments of biodiversity-related Multilateral Environmental Agreements (MEAs) and other relevant international biodiversity forums. For example, CAFF has a framework of agreements with the biodiversity MEAs that are relevant to the Arctic, e.g., Ramsar and the Convention on Biological Diversity (CBD) (Meltofte 2013). Furthermore, CAFF is undertaking work focused on enhancing engagement in relation to the roles and functions of Arctic wetlands as a resource for humans and biodiversity to support sustainable development and resilience in the Arctic. CAFF also has the Arctic Migratory Bird Initiative (AMBI), which aims to improve sustainability of populations of migratory birds that breed and make use of freshwater ecosystems in the Arctic.

The CBMP is endorsed by the Arctic Council and the United Nations Convention on Biological Diversity (CBD) (Convention on Biological Diversity 2010, Barry et al. 2013), and contributes to the Sustaining Arctic Observing Networks (SAON). In relation to the Arctic Council, the CBMP is an important tool for CAFF to implement several of the 17 recommendations from the *Arctic Biodiversity Assessment Report for Policy Makers* (CAFF 2013). In addition, the CBMP is the official Arctic Biodiversity Observation Network (Arctic BON) of the Group on Earth Observations – Biodiversity Observation Network (GEO BON) and a partner to the Global Biodiversity Indicators Partnership (BIP). In relation to GEO BON, the Freshwater CBMP aligns very closely with the approach taken by GEO BON's global Freshwater Biodiversity Observation Network (FW BON) that working to implement common standards and methodologies around the world for in-situ and remotely-sensed observation of freshwater biodiversity. As well, the Focal Ecosystem Component approach used by the CBMP can be relatively easily translated into the Essential Biodiversity Variable concept of GEO BON (Pereira et al. 2013). Countries that are subject to the Water Framework Directive (WFD) of the European Union (EU) (i.e., Finland, Iceland, Norway, Sweden) can also promote synergies between their CBMP activities and WFD outputs to improve the protection of Arctic inland surface waters.

SAFBR outputs will contribute to the above mentioned partnerships and/or national obligations —for example, by helping to measure progress towards the CBD Aichi Biodiversity Targets and measuring the Arctic's progress towards the soon to be released post-2020 Biodiversity Targets —and will gather relevant and reliable information that can inform regional and global processes that affect Arctic biodiversity.

2.4. The State of Arctic Freshwater Biodiversity Report (SAFBR)

2.4.1. Objectives and Overview of Report

The overall goal of the SAFBR was to assess the current status and trends of freshwater biodiversity of FECs both within geographical regions and across the circumpolar Arctic. Specific objectives were to:

- Assess alpha and beta biodiversity and evaluate species distributions and community composition across the Arctic;
- Appraise whether alpha and beta diversity are stable, increasing or decreasing, and if the distribution of particular species is changing;
- Identify geographical locations with high biodiversity (i.e., biodiversity hotspots);
- Determine the primary environmental and human stressors associated with the observed patterns in biodiversity; and
- Identify key monitoring locations for inclusion in future circumpolar assessments of freshwater biodiversity.

2.4.2. Collection and harmonization of data

Metadata collection was undertaken to identify high-quality sources for data collection in the Arctic region. Well-established national monitoring networks and databases in some countries facilitated the collection of contemporary data from a small number of sources, but a lack of coordinated monitoring in other countries required more extensive searches. To ensure broad spatial coverage for the assessment in those countries that lacked national databases (e.g., USA, Canada, Greenland), data were acquired by identifying potential sources, including government-funded monitoring programs, industry-funded monitoring programs, peer-reviewed published literature, and the grey literature, and extracting published data or requesting data access where necessary. Data collection was not exhaustive (for example, it was not possible to obtain data from some sources in Canada), but was as extensive as possible given time constraints. Additionally, Russian involvement in the data collection process was delayed until 2016, significantly limiting the amount of data that could be collected for this effort. Data for lakes and rivers of all sizes were collected for a variety of FECs for the contemporary period (1950 to present), and where possible, for the post-industrial period (1900 to 1950) and pre-1900 (paleo data). Spatial and temporal coverage of data was patchy for many regions, and historical data were lacking for many biotic FECs; however, the initial data collection and assessment can serve to inform the expansion of monitoring to fill identified gaps.

Data were compiled by each country using a standardized format, then reviewed and revised as needed to ensure compliance with data formatting requirements for a single circumpolar database on freshwater biodiversity and supporting variables. Once data from all countries were compiled, reviewed for quality assurance, and entered into the database, harmonization procedures were completed to ensure comparability of data from a variety of data sources. In particular, a nomenclature table was created for each FEC (fish, benthic invertebrates, zooplankton, macrophytes, diatoms, and phytoplankton) to correct and update the taxonomic identifications from the raw data and ensure standardization of taxonomic nomenclature across data originating from a variety of sources. Nomenclature tables updated outdated naming conventions, corrected misspellings or regionally inconsistent spellings, and in some cases were used to create taxonomic complexes where it was necessary to group ambiguous or higher-order identification of individuals.

For each sample in the database, sampling method and equipment details were recorded and compared prior to selection of data for analysis. To reduce variability in the data due to differing methodologies, subsets of data were selected for analysis based on compatibility of equipment (e.g., selecting samples with similar mesh size), collection methods (e.g., grouping methods that would be expected to sample similar portions of the assemblage), sampled habitats (e.g., separation of littoral and profundal samples), and approaches (e.g., samples targeting individual species versus those that collect the full assemblage). Data included a number of measurement types (e.g., presence/absence, counts, densities, biomass). All data were converted to presence/absence to allow broad-scale comparison of data that was inclusive of the greatest number of samples (e.g., including both quantitative and qualitative data), but subsets of quantitative data were also retained for analysis where appropriate. See section 4.1.1.3 for more details on FEC-specific data processing.

Available remote sensing and geospatial data were also collected for the circumpolar region, but largely represented supporting variables (e.g., abiotic variables). Though the use of remote sensing data can support assessment of biodiversity at global scales in terrestrial systems, for example (Turner et al. 2003, Pettorelli et al. 2014), the application of such data in freshwaters is generally limited to description of environmental drivers or Chlorophyll a biomass. Current technology does not support the evaluation of biodiversity of freshwater benthos, plankton, or fish through remote sensing, and data collection was necessarily limited to variables that could describe potential environmental drivers of biodiversity patterns.

The final circumpolar freshwater database is one of the main deliverables of CBMP-Freshwater to CAFF as it documents the data underlying the 2018 SAFBR. This database includes original data, nomenclature tables, and final harmonized data, and will be incorporated into the Arctic Biodiversity Data Service (ABDS). The ABDS is an online, interoperable data management system that serves as a focal point and common platform for all CAFF programs and projects as well as a dynamic source for up-to-date circumpolar Arctic biodiversity information and emerging trends. Incorporation in the ABDS will secure the freshwater database for future assessments and allow the database to be expanded over time as more data are collected or located. This will facilitate future assessments of change in freshwater biodiversity across the circumpolar region.

Permafrost slump in Northwest Territories, Canada.
Photo: Jennifer Lento

3. Drivers of Change in Arctic Freshwaters

3.1. Introduction

Freshwater ecosystems are highly abundant and diverse throughout the circumpolar region and include lakes, ponds, rivers, streams, and associated wetlands, all interconnected in the hydrological cycle of the Arctic (Wrona et al. 2005, Prowse et al. 2006b, Wrona et al. 2006b, Vincent and Laybourn-Parry 2008). Arctic freshwaters are generally nutrient-poor and ice-covered during a large part of the year. Key environmental and anthropogenic drivers in the Arctic, operating singly or in concert, affect the distribution and abundance of freshwater ecosystems, their water chemistry and related habitats, and structural (i.e., community composition) and functional (i.e., drinking water and food supply) ecological properties. In this chapter we summarize the major drivers of change that act on freshwater ecosystems in the Arctic, and provide examples of the effects of several key drivers of change.

3.2. Major Environmental and Human Impacts on Arctic Freshwaters

The Freshwater Monitoring Plan identifies nine major environmental and anthropogenic stressors to freshwater ecosystems (Culp et al. 2012a, Table 2) that can be summarized as (1) permafrost thaw and changes in the hydrological regime resulting in higher loads of nutrients, solids and organic matter (e.g., Kokelj et al. 2013), (2) long-range transboundary air pollutants and point source pollution originating from industrial development and urbanisation, (3) fisheries over-harvesting, (4) climate-driven changes to riparian vegetation from grasses to shrub-dominated flora (e.g., Elmendorf et al. 2012), i.e., greening of the Arctic (Jia et al. 2003, 2009), and (5) flow alterations and regulation due to hydropower dams and other forms of development that leads to substantial habitat fragmentation and destruction. As the water quality and biota of lakes and rivers reflect local- and landscape-scale processes in their catchment, these freshwater ecosystems are highly suitable for the monitoring and detection of both diffusive and point-source pollution. For example, increased nutrient loads from agricultural land use or point-source pollution will result in higher primary production and higher abundances of grazing benthic invertebrates in lakes and rivers. Conversely, increased loads of suspended solids or dissolved organic matter can decrease light penetration and cause a decline in primary production (Karlsson et al. 2009). Permafrost thaw will result in increased turbidity and a leakage of soil organic carbon (Kokelj et al. 2009). Large-scale thawing of permafrost layers can dramatically alter Arctic landscapes through the drainage of lakes on permafrost, resulting in the disappearance of these water bodies and large-scale landscape transformations.

3.2.1. Key Examples of Environmental Drivers Affected by Climate

Many of the environmental factors that affect the physical-chemical environment of lakes and rivers are primarily driven by climate. This includes changes in the duration and thickness of ice cover as well as snow pack conditions (Borgstrøm 2001, Schindler and Smol 2006, Christoffersen et al. 2008, AMAP 2011, Prowse et al. 2011b). For example, long-term shifts in ice cover duration have been observed in lakes and rivers in Fennoscandia. In the Torne River located in Northern Sweden, ice-on is 10 days later and ice-out 10 days earlier than in the early 1900s, meaning that on average the present-day duration of river ice cover is some 20 days shorter than a century ago (Figure 3-1). A similar long-term trend in seasonal ice duration is evident for Lake Torneträsk, Sweden (Figure 3-1). Such changes in ice cover affect the thermal budget of freshwater ecosystems. For example, long-term temperature data from Utsjoki Nuorgam in the Tana River (69°N, in Finland) show that there has been a gradual increase in the number of days exceeding a daily mean temperature of 5°C between 1970 and 2017 (Figure 3-2). The trend was most evident from 1995 to 2017, when the number of days above 5°C increased significantly, at a rate of 0.56 days per year (Sen's slope of trend; Mann-Kendall trend test for 1995- 2017 significant at $p = 0.01$). Overall, this has led to an increase to 21 more days of temperatures over 5°C since 1970 (Figure 3-2). Warmer water and subsequently shorter ice cover will allow more sunlight and heat to enter freshwaters, thus resulting in more degree days (i.e., the cumulative heat that organisms experience), and drive photosynthesis,

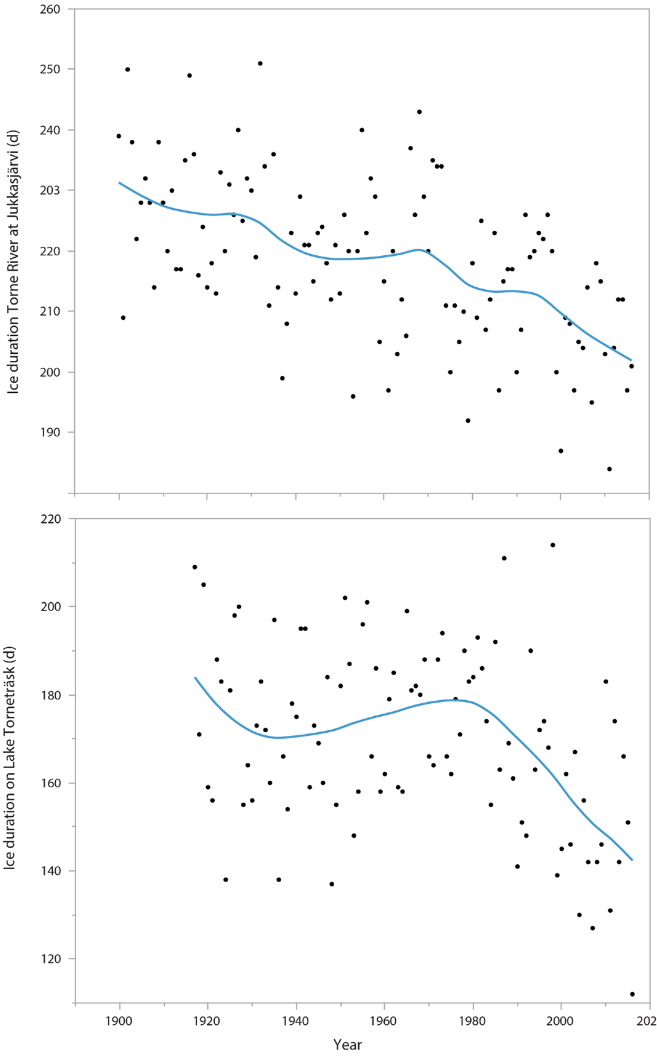


Figure 3-1 Long-term trends in ice duration (as days) in the River Torne (upper plot) and Lake Torneträsk (lower plot) at 68° north on the Scandinavian peninsula. Lines show smooth fit. Data source: Swedish Meteorological and Hydrological Institute.

leading to higher primary production and subsequent effects on production at higher trophic levels. Higher water temperatures may also allow the northward movement of species with a more southerly distribution, thus increasing the biodiversity of lakes and rivers (Culp et al. 2012b).

The climate-change-driven impact on the Torne River ice regimen has been accompanied by a gradual decrease in concentrations of total phosphorus, a key nutrient that limits the photosynthesis rates of primary producers in aquatic ecosystems. Phosphorus concentrations in the Torne River have declined on average by some 7 µg/L per year and, although small, these declines are highly significant. Similar, but slightly more pronounced declines in total phosphorus concentrations, i.e., 0.14–0.26 µg/L per year, have been found for other major unregulated rivers that drain the Boreal Highlands of Sweden (Figure 3-3). Declines in total phosphorus concentrations have also been found for lakes at northern latitudes in Finland (Arvola et al. 2011), Canada (Eimers et al. 2009, Stammler et al. 2017), and Sweden (Huser et al. 2018). Large-scale catchment processes that contribute to reductions in nutrient run-off to lakes and rivers are (i) the observed changes in tundra vegetation cover, a.k.a. the "Greening of the Arctic" (Pouliot et al. 2009, Elmendorf et al. 2012) mediated by an increased nutrient uptake by and storage in rooted plants (Aerts et al. 2006), and (ii) the more efficient trapping of P in soils that originates from soil pH increases induced by declines in acid precipitation (Geelhoed et al. 1997, Gérard 2016). The concerted action of these large-scale changes contributes to the gradual transformation of northern lakes toward more nutrient-poor conditions and is expected to increase in the predominance of N2-fixing cyanobacteria. Long-term declines in total phosphorus concentrations have repercussions on the primary production in these rivers and lakes and may push them towards more ultra-oligotrophic conditions. These declines in nutrient concentrations partly counteract the positive effects of a longer growing season due to changes in the ice regimen.

Another important environmental driver of change in Arctic freshwaters is the thawing of ground ice across landscapes as climates warm and precipitation increases (Kokelj et al. 2015). For example, permafrost degradation via retrogressive thaw slumps can increase transport of solutes, including nutrients, and sediments into Arctic lakes and rivers (Kokelj et al. 2009, Chin et al. 2016). Shoreline slumps on lake ecosystems appear to reduce DOC and increase water clarity as a result of the adsorption of nutrients onto settling sediment particles (Thompson et al. 2012). Such environmental change is associated with reduced phytoplankton productivity and increases in rooted macrophyte biomass (Mesquita et al. 2010). In contrast, shoreline slumps in rivers increase turbidity and suspended sediments by multiple orders of magnitude (Figure 3-4) and can lead to overwhelming sediment effects including reduced benthic algal biomass (Levenstein et al. 2018) and macroinvertebrate abundance (Chin et al. 2016). Furthermore, climate models indicate that increasing permafrost degradation will lead to a loss of wetlands in the Arctic as meltwater from thawing ground ice drains to deeper soil levels rather than contributing to surface soil moisture (Avis et al. 2011). It is clear that biological communities in Arctic freshwaters are at risk from environmental changes that can affect food web dynamics, biological production, and biodiversity, thereby having potential effects on the ecosystem services valued by northerners.

The retreat of glaciers and ice sheets provides a unique example of how climate change can have both positive and negative effects on freshwaters and freshwater biodiversity. Glacier retreat and accompanying shifts in glacial outflow can lead to river piracy when flowing waters are dependent on the glacier for source water. River piracy is the re-routing of headwater streams into different river systems, significantly altering flow regimes and even causing rivers to dry up, and a recent example from the Yukon in the Canadian Arctic showed such events happening on an accelerated time scale in response to glacier retreat (see Shugar et

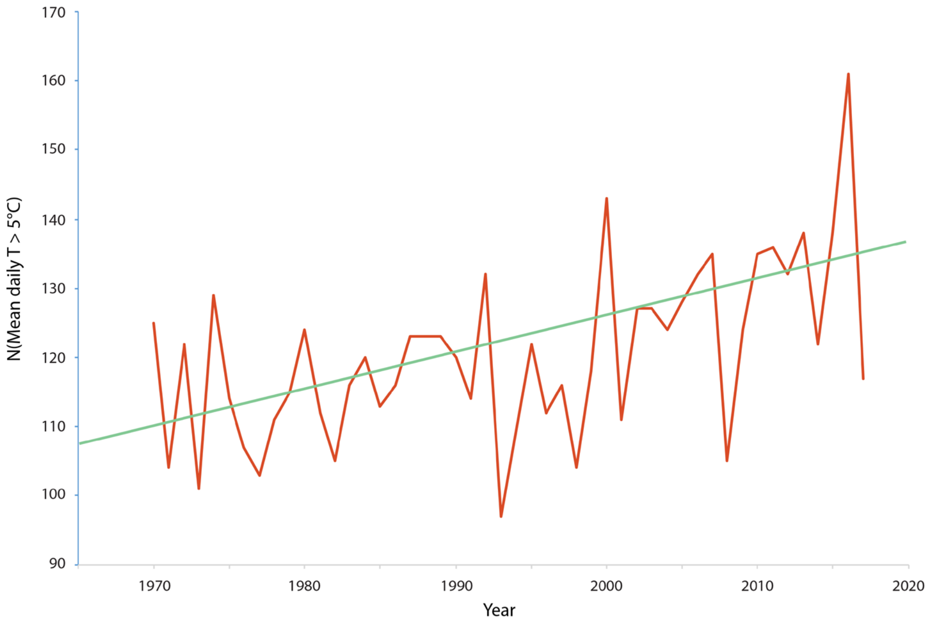


Figure 3-2 Long-term water temperature trends (1970–2017) for the Utsjoki Nuorgam station in the River Tana (69°N in Finland). The diagram shows the number of days per year with a mean temperature exceeding 5°C. The data show that from 1995 to 2017, this indicator increased by over 0.5 days per year. Data source: Finnish Meteorological Institute.

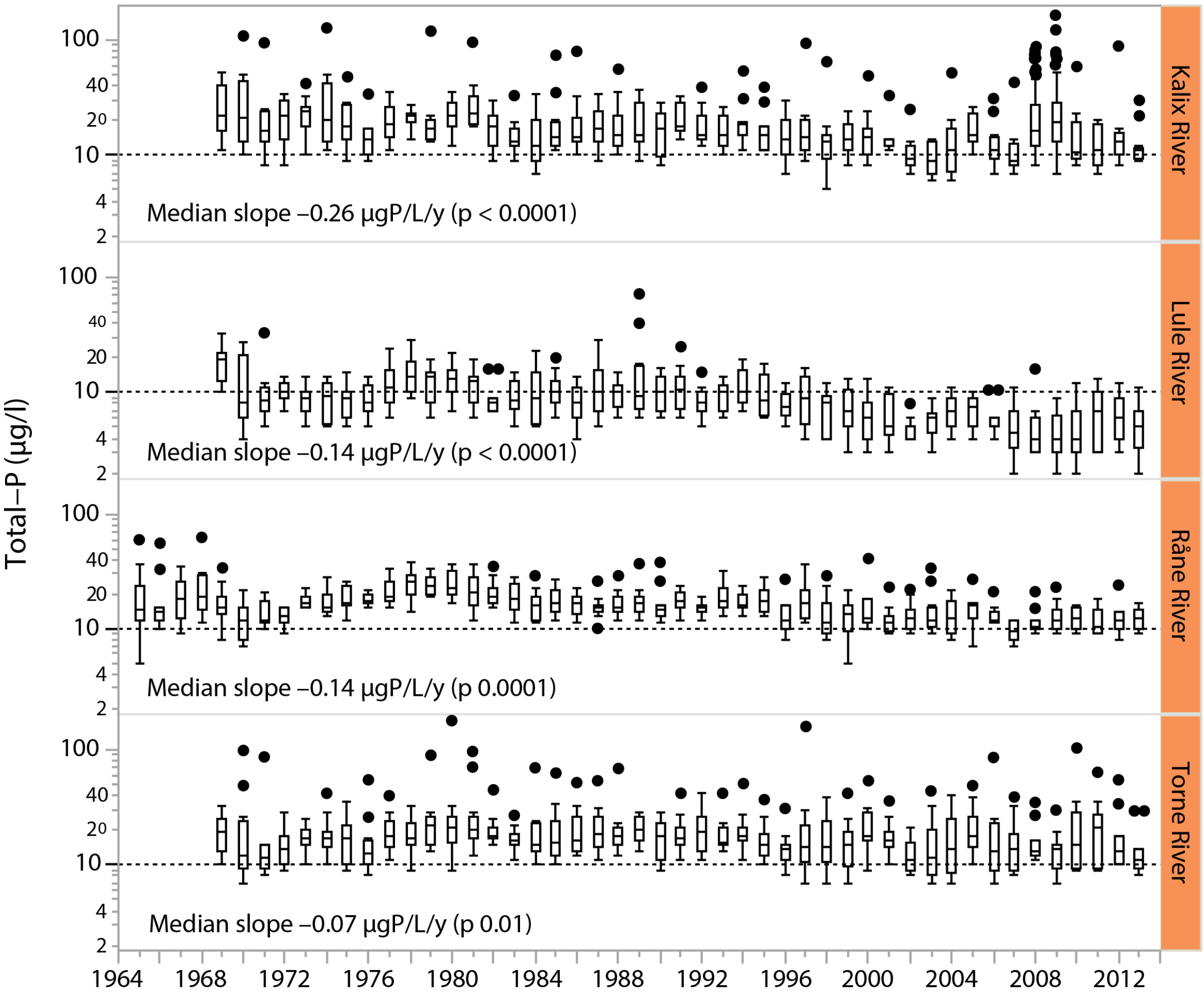


Figure 3-3 Long-term trends in total phosphorus water concentrations (µg/L) in four major, unregulated rivers that drain the subarctic Arctic/alpine ecoregion of the Scandinavian peninsula, the Kalix river, The Lule river, the Råne river, and the Torne river. Slopes and p-values are given in the different panels. Boxes indicate medians and 25th and 75th percentiles, while whiskers give the 10th and 90th percentiles.


Photo credit: Jennifer Lento.

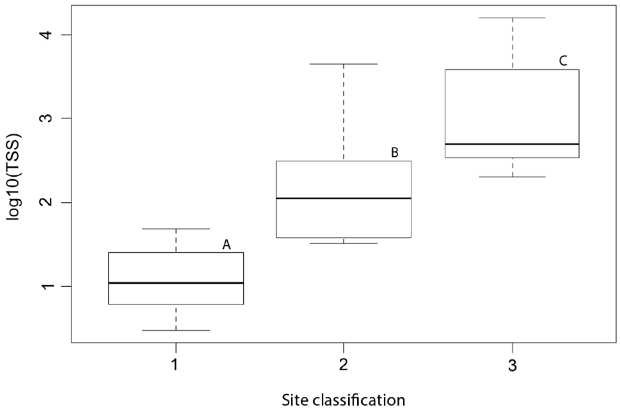


Figure 3-4 Effects of permafrost thaw slumping on Arctic rivers, including (upper) a photo of thaw slump outflow entering a stream on the Peel Plateau, Northwest Territories, Canada, and (lower) log10-transformed total suspended solids (TSS) in (1) undisturbed, (2) 1-2 disturbance, and (3) > 2 disturbance stream sites, with letters indicating significant differences in mean TSS among disturbance classifications Plot reproduced from Chin et al. (2016).

al. 2017). In contrast, meltwater from ice sheets in west Greenland has led to an increase in the number and size of lakes on the landscape, increasing available freshwater habitat (Carrivick and Quincey 2014). Climate-induced glacier loss alters hydrological regimes, sediment transport, and biogeochemical and contaminant fluxes from rivers to oceans (Brown et al. 2018). Declining glacial cover will initially cause a decline in alpha diversity (number of species) in receiving waters as glacial meltwater inputs increase and water temperature drops (Figure 3-5), leading to stronger differences in assemblage composition (beta diversity). A further reduction of glacial influence will alter cold river biodiversity, leading to increased alpha diversity and functional diversity (Figure 3-5), and will completely reshape many river systems (Brown and Milner 2012, Milner et al. 2017, Brown et al. 2018). With the loss of glaciers, hydrologic regimes of rivers and lakes will be more reliant on inputs from other water sources, such as snowmelt, groundwater, and rain events (Milner et al. 2017). This will profoundly influence the natural environment, including many facets of biodiversity, and the ecosystem services that glacier-fed rivers provide to humans, particularly provision of water for agriculture, hydropower, and consumption.

3.3. Predicted Scenarios of Species Richness Response to Climate Warming

Scenarios of species richness response to increased temperatures in Arctic lakes and rivers were proposed by Culp et al. (2012b). These biodiversity predictions specifically address changes in the relative share of eurythermic species (i.e., those that can function at a wide range of temperatures) and stenothermic species (i.e., those that are adapted to a narrow range of temperatures). As temperature regimes in Arctic freshwaters warm, the northward movement of eurythermic species will affect biodiversity at all scales from species composition within rivers, lakes and ponds (alpha biodiversity) to changes in regional assemblages (gamma biodiversity), with the overall state change depending on the relative rates of gains and losses in eurythermic and stenothermic species (Vincent et al. 2011). These changes in species richness can also be expected to modify functional diversity in Arctic freshwaters (Brown et al. 2018). A rapid increase in the abundances of eurythermic species and a slow loss of stenotherms will produce a pulsed increase in gamma biodiversity that likely will settle at a new equilibrium dominated by eurythermal species (Figure 3-6a). In contrast, a more moderate dispersal rate by eurythermal species (assuming that barriers to dispersal are limited) coupled with the rapid loss of stenotherms will produce a pulsed decrease in gamma biodiversity that will also eventually settle at a new equilibrium dominated by eurythermal species (Figure 3-6b). An equilibrium dominated by eurythermal species is reached more rapidly through a rapid increase in eurytherms coupled with a rapid decrease in stenotherms (Figure 3-6c). In contrast, a slow increase in eurytherms coupled with a slow decrease in stenotherms will lead to a slow increase in gamma biodiversity that eventually will settle at a new equilibrium dominated by eurytherms (Figure 3-6d). The actual changes in species diversity will, therefore, depend critically on the relative rates of change in eurythermal and stenothermic species, with the panels in Figure 3-6 representing possible response scenarios. Declines in eurytherms are expected as temperatures increase beyond thermal tolerance levels (not shown in Figure 3-6) (Woodward et al. 2010).

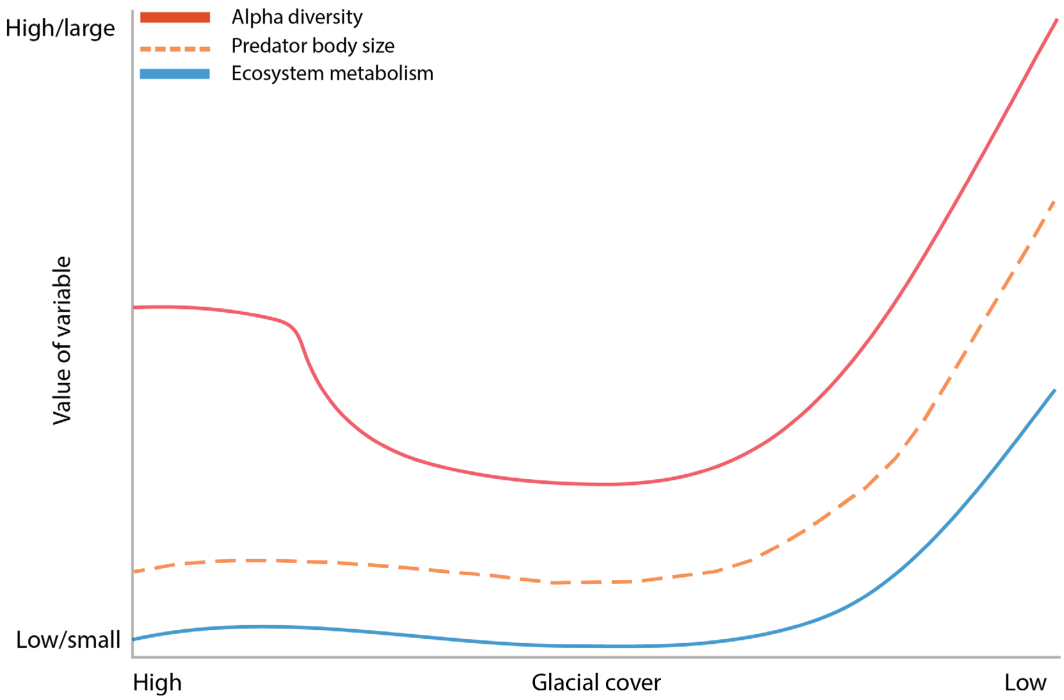


Figure 3-5 Changes in alpha diversity (red line), predator body size (blue dashed line), and ecosystem metabolism (blue solid line) with a shift in glacial cover from high (left) to low (right). Redrawn from Milner et al. (2017).

Where dispersal routes do not exist (e.g., isolated high Arctic or high-altitude water bodies), the climate-driven loss of stenotherms may not be compensated by eurythermic species invasion and an overall decline in gamma biodiversity is expected. The effect is expected to predominate more among vertebrates whose dispersal patterns rely on habitat connectivity, although the response of invertebrate composition and functional diversity to climate change may also be affected by dispersal limitations (Brown et al. 2018). Avian range expansion associated with climate warming, however, may lead to increased invertebrate diversity at local (alpha) and regional (gamma) scales via dispersal facilitation (Santamaría and Klaassen 2002).

Scenarios of biodiversity change in Arctic freshwaters (Figure 3-6), including glacially-fed systems (Figure 3-5), predict a net increase in biodiversity with warming temperatures, assuming dispersal routes exist for southern species to colonize northern regions. However, as water quality and habitat conditions shift to more closely resemble southern latitudes, this shift is expected to come with a reduction in the habitat range of cold-tolerant species endemic to the Arctic. Thus, along with an overall predicted increase in the number of species, there will be a net loss of unique Arctic-specific biodiversity. Alterations of habitat conditions originating from changes in air and water temperatures, permafrost extent, nutrient availability, and terrestrial vegetation will change the zonation of the Arctic region by globally decreasing the size of the sub-, low, and high Arctic regions and by reducing habitat critical to cold-tolerant Arctic species.

In summary, climatic regime change is likely to produce substantial effects on the physical and chemical habitat template of Arctic freshwater ecosystems. This change in the abiotic environment is expected to cause transformations in biological production and biodiversity as some existing resident taxa are selected against while others are favored such as the northerly dispersal of taxa previously unable to tolerate Arctic conditions (Wrona et al. 2006a, Vincent et al. 2011). Resulting alterations to aquatic biodiversity, therefore, have the potential to produce changes to freshwater fisheries around the Arctic, and to modify the distributions of aquatic invertebrates, vertebrates and plants. These changes to aquatic biodiversity and food webs will affect not only Arctic freshwater ecosystems, but also the ecosystem services that they supply to Arctic residents.

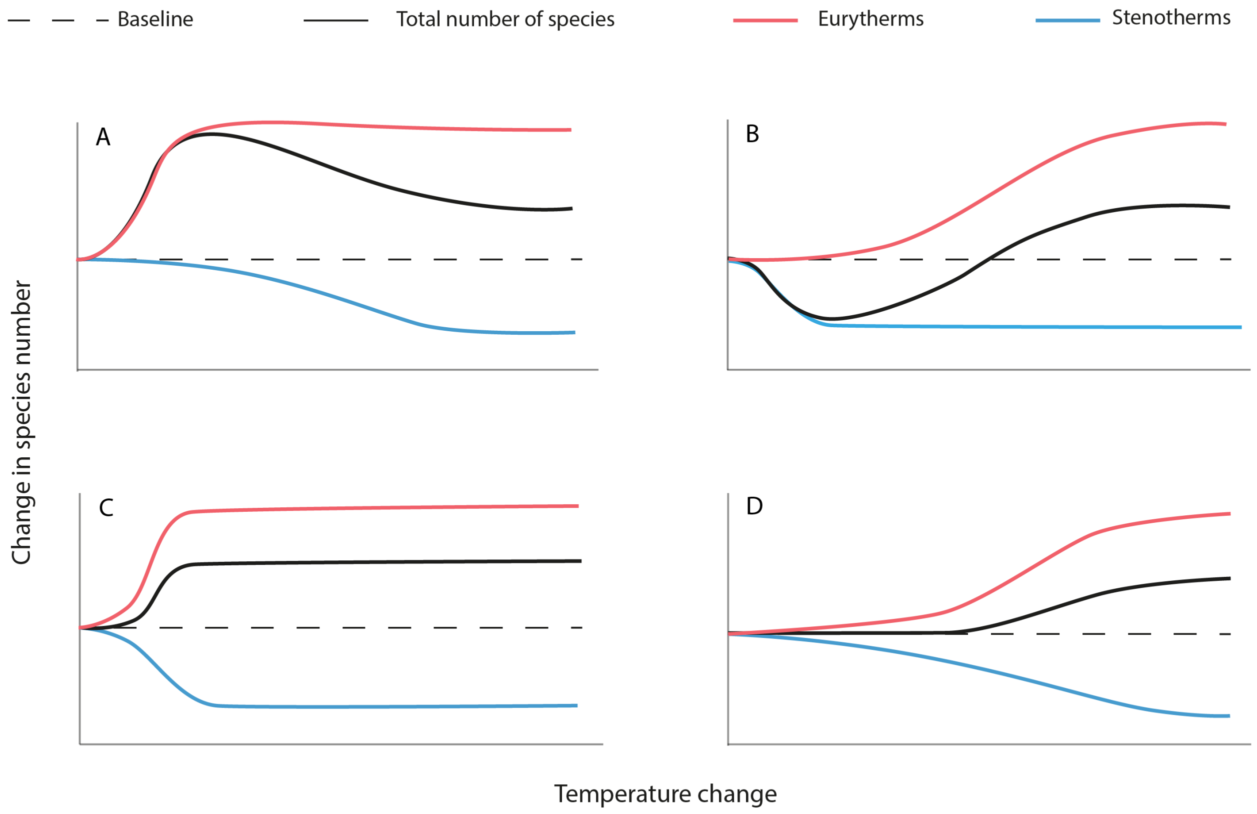


Figure 3-6. The hypothesized effects of rising mean water temperature on biodiversity (as total species number) of Arctic freshwater ecosystems. A pulsed increase in gamma biodiversity (a) results from the combination of high eurythermal invasion and establishment and low stenothermic loss with increasing water temperature. A pulsed decrease in gamma biodiversity (b) results from the combination of low eurythermal invasion and establishment and high stenothermic loss. Rapid increases (c) and slow increases (d) in species diversity occur, respectively, with high eurythermal invasion and establishment coupled with high stenothermic loss or low eurythermal invasion and establishment and low stenothermic loss as temperatures increase. For simplification, barriers to dispersal have been assumed to be limited in these models.

Darner dragonfly (Aeshna cyanea) *nymph extending its labium*
Photo: Jan Hamrsky

4. Status and Trends in Arctic Freshwater Biodiversity

4.1. Introduction

This chapter provides assessments of spatial and temporal trends in freshwater biodiversity of lakes and rivers for each biological FEC, including algae from benthic samples, phytoplankton, water plants (macrophytes), zooplankton, benthic macroinvertebrates, and fish. Primary producers are represented by both microscopic (algae) and macroscopic (water plants) organisms, while animals include primary (zooplankton, benthic macroinvertebrates) and secondary consumers (invertebrate predators, fish). Aquatic food webs are driven by the photosynthesis of primary producers and by the microbial decomposition of organic matter entering lakes and rivers from their terrestrial environment. The sources represent the autotrophic and heterotrophic pathways, respectively, in aquatic food webs (Figure 4-1). Primary production (photosynthesis) and decomposing leaf litter (detritus) form the base of food webs that supplies food for primary consumers (herbivores and detritivores) and predators at higher trophic levels.

Primary production in lakes and rivers is dependent on access to sunlight and is thus limited to the shallow zones of lakes. Allochthonous inputs (e.g., terrestrial vegetation) contribute food for primary consumers (Figure 4-2), but may be limited at higher latitudes. Benthic algae can grow attached to stones, water plants or sediments in the shallow areas of lakes, while planktonic algae (or phytoplankton) are free-floating in the open water, i.e., the pelagial zone (Figure 4-2). In nutrient-poor, clear-water Arctic lakes, photosynthesis by benthic algae is the main source of food for higher trophic levels and can occur at deeper depths due to increased light penetration. Pelagic phytoplankton production can become relatively more important in more nutrient-rich lakes, where light penetration is decreased. In the deep and dark profundal zone of large lakes, no photosynthesis occurs, and biological production is entirely dependent on organic matter settling to the bottom. In rivers, food webs are more simplified with low biomass of plankton, and benthic algae and allochthonous materials form the base of the food web (Figure 4-3). The illustrations in Figure 4-2, Figure 4-3, and Figure 4-4 highlight reference conditions for ecosystem processes and biodiversity in lakes, rivers and glacial-fed rivers and the potential impact to these freshwater ecosystems as a result of climate change.

This chapter focuses on the dominant FECs within lakes and rivers, assessing status and trends in lake algae from benthic samples, phytoplankton, macrophytes, zooplankton, benthic macroinvertebrates, and fish, as well as river algae from benthic samples, benthic macroinvertebrates, and fish. In line with freshwater monitoring tradition, our assessments primarily focus on changes in the assemblages of these FECs, rather than on single taxa. Key aims for the assessments were to provide an overview of the spatial and temporal trends in current biodiversity of FECs for rivers and lakes in the Arctic and sub-Arctic regions, including assessment of alpha and beta diversity, and evaluation of historical changes that have occurred; to determine any data gaps in spatial coverage across these regions; and to establish a baseline for future monitoring for these geographical regions. Additionally, we provide the steps required to optimize future global bioassessments. By collecting and assessing data from the circumpolar region, we have attempted to establish the state of knowledge for these FEC groups, while allowing for future re-assessments of status and trends as more data become available.

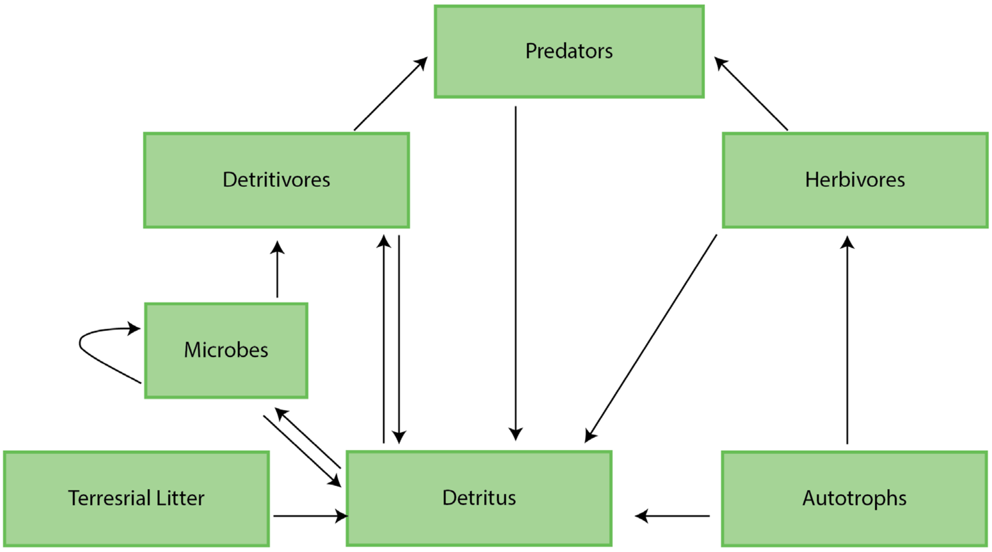


Figure 4-1 A generic food web diagram for a lake or river, indicating the basic trophic levels (boxes) and energy flow (arrows) between those levels. Reproduced from Culp et al. (2012a).

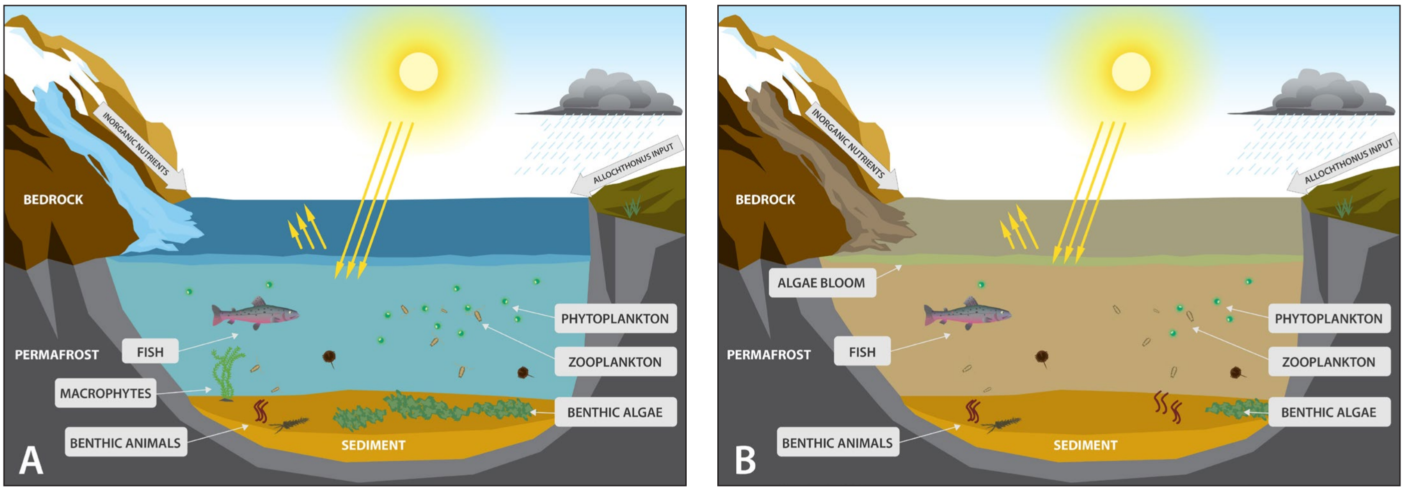


Figure 4-2 (a) Typical Arctic lake food web, including primary and secondary consumers, and indicating vertical placement of food web components in the water column, and (b) Arctic lake food web following climate-change induced inputs of sediments and nutrients, indicating resulting food web shifts with declines in water transparency.

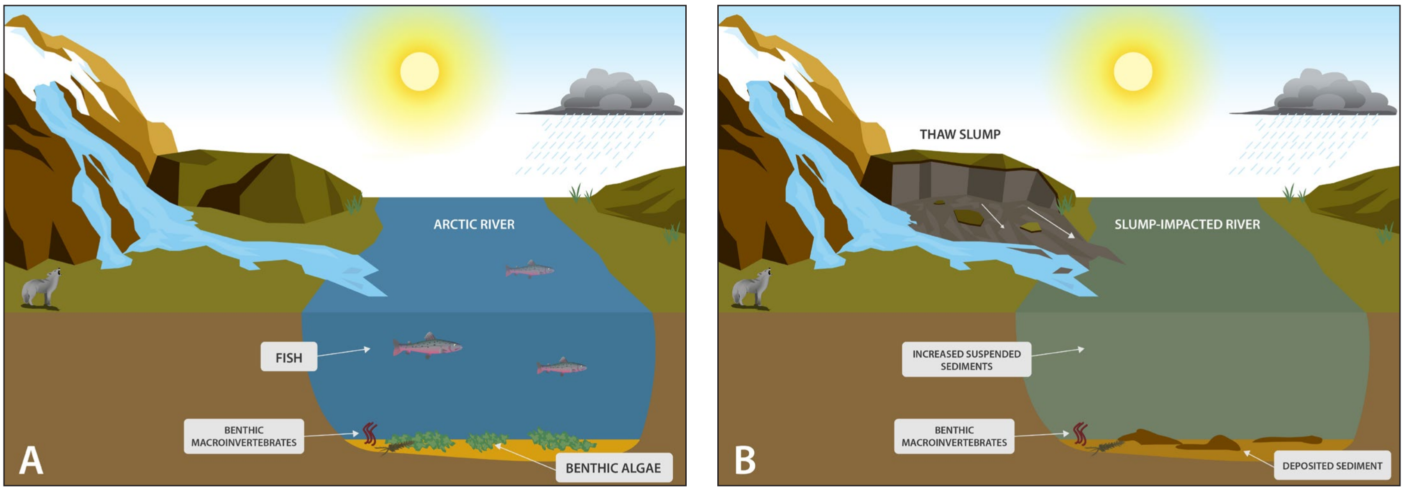


Figure 4-3 (a) Typical Arctic river food web, including primary and secondary consumers, and (b) Arctic river food web impacted by inputs from permafrost thaw slump, with increased suspended solids and increased deposited sediment causing burial of benthic producers and consumers.

Photo: Dan Bach Kristensen/Shutterstock.com

4.1.1. Analytical Approach

Our data assessment emphasizes the importance of alpha and beta diversity indicators. The species diversity of a region (gamma diversity) is comprised of both local species diversity (alpha diversity) and the variation in species diversity among sites (beta diversity). These measures of diversity are scale-dependent, and their pattern of change should be considered along multiple spatial and temporal scales. However, the feasibility of such a multi-scale assessment is limited by data availability, and in particular, the spatial and temporal coverage of available data. Although spatial coverage of stations is extensive in some regions with established routine monitoring (e.g., Fennoscandia), sparse coverage in other areas limits the scope of such assessments.

To standardize the spatial scale of the circumpolar analysis of diversity, stations were classified on the basis of the terrestrial ecoregion in which they were found (Terrestrial Ecoregions of the World, TEOW; Olson et al. 2001). These geographic and climatic regions allowed stations to be grouped based on regional conditions, which would be expected to affect habitat conditions within freshwater ecosystems (see Figure 4-5 for a complete map of ecoregions included in the assessment). Although ecoregions based on global freshwater basins have been derived (e.g., Freshwater Ecoregions of the World, FEOW; Abell et al. 2008), this classification groups together several terrestrial ecoregions that would be expected to differ ecologically due to differences in biogeography and climate. For example, Alaska is largely included in one freshwater ecoregion despite the contrast between coastal areas and the mountainous Brooks and British Ranges. Svalbard, which is classified as Arctic Desert in TEOW, is grouped with northern Norway, Finland, the Kola Peninsula, and northwestern Russia as a single freshwater ecoregion in FEOW. Furthermore, Greenland was not assigned freshwater ecoregions, whereas it has been split into northern and southern ecoregions in TEOW. It was determined that the freshwater ecoregions were at too great a spatial scale for assessment, and climate-based terrestrial ecoregions were instead chosen to better allow assessment of the freshwater response to climate change.

Stations were further grouped into hydrobasins within the ecoregions for some analyses (Lehner and Grill 2013). Hydrobasins are standardly-derived basins that reflect natural flow patterns over the landscape, are at a smaller scale than ecoregions, and allow for the grouping of hydrologically-related systems instead of deriving catchment areas for all stations in the database. Because fish stations were more spatially concentrated, analysis of fish data used level 7 hydrobasins, whereas analysis of other FECs used the larger level 5 hydrobasins (to ensure a sufficient number of stations within each hydrobasin).

Because of differences in the number of stations sampled in each ecoregion, rarefaction curves were used to estimate alpha diversity within each ecoregion at a chosen number of stations (see detailed explanation in section 4.1.1.1), thus controlling for variability in sampling effort. This approach allowed for comparison of standardized estimates of alpha diversity across the circumpolar region that were less affected by local sample frequency. Beta diversity was estimated at the hydrobasin level to determine average beta diversity within each ecoregion and assess dominant components of beta diversity (see detailed explanation in section 4.1.1.2).

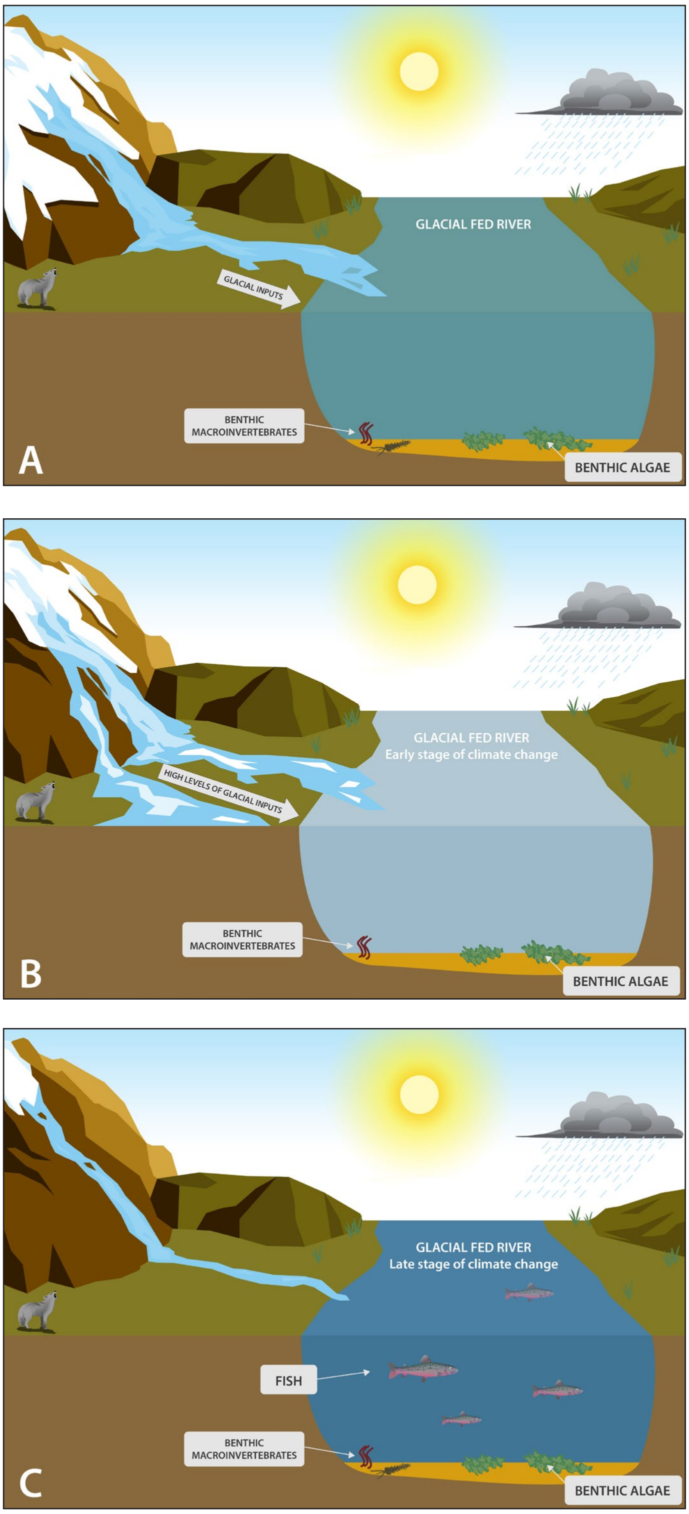


Figure 4-4 (a) Typical glacier-fed river food web, including primary producers and consumers and low-moderate glacial inputs, (b) glacier-fed river food web in the early stages of climate change, with increased glacial inputs, and (c) glacier-fed river in the late stages of climate change, when glacial inputs have ceased due to glacier retreat.

Figure 4-5 Terrestrial ecoregions that are included within the circumpolar region within the CAFF boundary and/or the ABA boundaries. Source: Terrestrial Ecoregions of the World (TEOW; Olson et al. 2001).

4.1.1.1. Alpha Diversity

It is well established that the number of species encountered at a station correlates proportionally to the sampling effort, i.e., the total area sampled (Rosenberg and Resh 1993, Gotelli and Colwell 2001). Thus, the number of species observed increases as the sampling effort (number of samples) within a water body is increased or as more water bodies are sampled, reaching a plateau above a certain threshold that may depend on the sampling method. Because sampling effort varied among countries and within ecoregions, we applied rarefaction procedures to allow for sound comparisons of alpha diversity (or taxonomic richness) among our data. In this procedure, species accumulation curves were generated for each ecoregion (using the program EstimateS; Colwell 2013, Colwell and Elsensohn 2014), and curves were extrapolated to a higher number of sampled stations when necessary (e.g., see Colwell et al. 2004, Colwell et al. 2012). These curves can be used to estimate the accumulation of new species with added sampling effort, but they can also be used as rarefaction curves, to compare among different regions at a standard number of samples (e.g., if region A included 50 sampled stations and region B included 100 stations, diversity could be rarefied along the curve for region B to 50 stations for a more standardized comparison). Rarefaction curves were randomized 100 times, and the average taxonomic diversity (with upper and lower 95% confidence intervals) was rarefied for every station along the curve (to the maximum number of stations in an ecoregion or to a predetermined extrapolation point). Rarefied alpha diversity was then extracted from the curve at a set number of stations for all ecoregions to allow comparison of taxonomic richness at a standardized sampling level. A sub-analysis was completed for each FEC using ecoregions with high levels of sampling to rarefy diversity to a greater number of stations, thus assessing patterns where spatial coverage of monitoring was high. All ecoregions with two or more stations were then rarefied (with extrapolation of the species accumulation curve, as needed) to a standard level of 10 stations for broad-scale assessment. Besides providing better comparisons of alpha diversity among water bodies, rarefaction procedures also alleviate the effects of sampling artifacts related to spatial scale or sampling effort that can vary across localities or regions.

Alpha diversity can be estimated as the number of species encountered, i.e., species richness. However, where identification to species level is not possible, or where taxa have been combined at a higher level to avoid mixed-level taxonomy, alpha diversity can describe family-level richness or simply taxonomic richness, which is the number of taxonomic units at a station. The taxonomic level at which alpha diversity was assessed differed depending on the FEC (see section 4.1.1.3)

4.1.1.2. Beta Diversity and Its Components

Beta diversity describes the level of dissimilarity in assemblage structure among stations within a defined region (Whittaker 1972). If beta diversity is high, stations likely have very large differences in taxonomic composition, whereas low beta diversity indicates that similar taxa are found at all stations. Beta diversity is complementary to alpha diversity, which simply assesses the number of taxa, and is defined as the diversity among ecosystems or the degree of assemblage differentiation in a region. Hence, beta diversity not only accounts for the relationship between local and regional diversity, but also informs about the degree of differentiation among biological assemblages. For example, it is possible for two stations to have a similar number of species (similar levels of alpha diversity), but to have different sets of species found at each station (high beta diversity across these stations).

There are many different ways to estimate beta diversity (Tuomisto 2010a, b), but one that is both simple and effective is to use Sørensen's dissimilarity coefficient, β_{SOR} (Jost 2007, Baselga 2010). β_{SOR} ranges from 0 to 1, with 0 indicating that the same taxa are found at all stations, and 1 indicating that stations have completely different sets of taxa. This measure of beta diversity has the added benefit that it can be partitioned into the portion of beta diversity that is due to spatial turnover and the portion that is due to nestedness (Baselga 2010, Baselga and Orme 2012, Baselga et al. 2012). These components describe the way in which taxonomic composition differs across stations. For example, spatial turnover refers to replacement of taxa from one station to another, indicating that stations contain unique species (Baselga 2010, Baselga et al. 2012). In contrast, nestedness describes species loss from one station to another, and indicates that taxonomic composition at some stations is a subset of what is found at the richest station (Baselga 2010, Baselga et al. 2012). Figure 4- 6 provides visual examples of differences between these components of beta diversity, as presented in Baselga (2010). In Figure 4-6a, there is nestedness across sites, as sites A2 and A3 contain a subset of the species found at A1. Figure 4-6b indicates an example of strong spatial turnover with equal richness across sites, as compositional differences among sites are due to unique species at each site. Figure 4-6c indicates a situation with both turnover and nestedness, as turnover is evident between sites C2 and C3 (unique species at each site, not found at the other site), but the species at both C2 and C3 are nested within the species found at site C1. Northward migration of species into the Arctic might be expected to increase spatial turnover within an area, as more new species begin to contribute to station differences. Estimating the relative contribution of spatial turnover and nestedness to beta diversity can provide important information for monitoring (Socolar et al. 2016). In a region with high spatial turnover, it would be necessary to sample a large number of stations to ensure accurate assessment of species richness (as new species would be introduced at each additional station). However, where nestedness dominates, it may be sufficient to monitor the stations with the highest richness to make accurate regional diversity estimates, though this depends in part on spatial scale of assessment.

4.1.1.3. Data Considerations for FECs

The database compiled by CBMP-Freshwater contains extensive information from over 9000 stations across the circumpolar Arctic (with stations defined as sampling locations with unique geographic coordinates) from which samples have been collected one or more times. For substantially fewer sites, there exists time series data that exceed 10 years. Paleolimnological data were also collected to allow for an assessment of historical trends beyond the contemporary time period. Data were collected and harmonized to standardize units of measurement and nomenclature as outlined in section 2.4.2.

An important consideration in the assessment of data collected by the different Arctic countries is that sampling methods must be comparable. For some biological FECs, the methods used were very similar across the circumpolar Arctic. However, differences were evident with respect to sampled habitats, sampling equipment, mesh size, and sample processing. Due to these differences it was necessary to simplify data (e.g., as presence/absence) or select subsets of data collected with more similar methods to account for these differences. Below, we outline the primary considerations for each of the FECs:

- *Algae from benthic samples*: Data on soft algae (non-diatoms) were only available for Norway, Greenland, and parts of Canada, and therefore the circumpolar analysis focused on diatoms, which were sampled across all regions. Data included lake rocky shoreline scrapes, top sediments of lakes, lake sediment cores (full cores and top/bottom cores), and river scrapes. Analysis generally focused on each group separately, though data from tops of cores were included in the analysis of lake top sediments. Counts or relative abundance data were available for most regions, with the exception of Greenland that only had presence/absence data, and therefore was excluded from some analyses. An adjusted taxonomic nomenclature was used that combined ambiguous or easily misidentified species at the genus level or into species complexes. Where information on sampling dates and depths of sediment core samples was not available or chronology was deemed to be problematic or not reliable, these data were excluded from the paleolimnological analysis.

- *Phytoplankton*: Data included a mix of presence/absence, counts, biovolume, and density, but

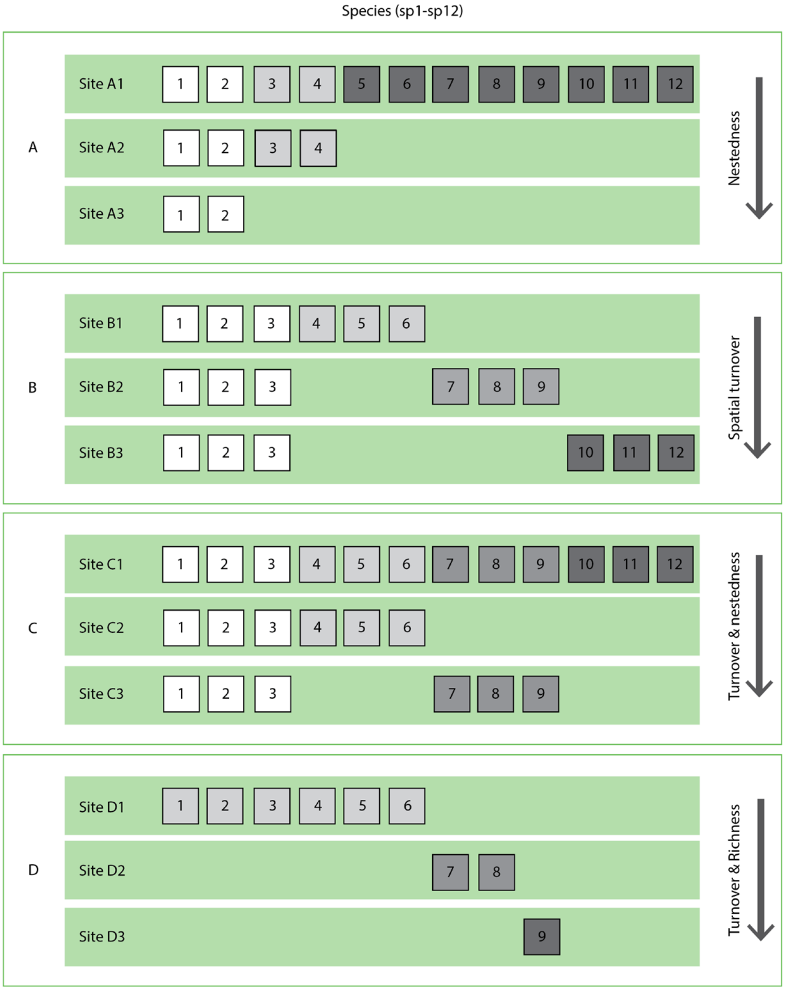


Figure 4-6 Examples of the spatial turnover and nestedness components of beta diversity, with four hypothetical locations (A-D), each with three sites. Location A is completely nested, as sites A2 and A3 each contain a subset of the richest site (A1). Location B is dominated by spatial turnover, as compositional differences are due to the introduction of new species across each site. Location C includes a combination of turnover and nestedness, as C2 and C3 have unique species relative to each other, but both sites contain a subset of what is found at site C1. Location D shows differences in richness across sites that are complete due to spatial turnover (unique species at each site). Figure reproduced from (Baselga 2010).

most stations had biovolume data. Biovolume of phytoplankton taxa can be calculated in multiple ways, and it was necessary to identify the procedure used for each dataset and apply a correction for some data to make them all comparable. There was some mixed-level taxonomy (genus only or genus and species), and corrections were made to avoid taxonomic redundancy.

- *Macrophytes*: Sampling methods for macrophytes varied substantially, ranging from simple observational notes to experimental designs with quadrats and measurements of biomass. To allow for broad-scale comparisons, all data were converted to presence/absence. Some regions had less emphasis on identification of mosses, and this was accounted for with subset analyses of moss data. Analysis was at the species level.

- *Zooplankton*: Zooplankton data included vertical hauls (composite samples through the water column) and depth-specific samples. The latter were summed across the entire water column to make them comparable with composite samples. Mesh size of sample nets varied across studies, and an effort was made to select samples with comparable mesh sizes. Not all datasets included identification and enumeration of rotifers (i.e., studies were focused only on crustacean taxa), and therefore analysis was completed once with the subset of samples that included data on the full zooplankton assemblage (including rotifers), and once using all samples but selecting only crustacean taxa for analysis. Datasets that included identification only at a coarse level (e.g., order) were generally excluded from analysis and corrections were made to avoid mixed-level taxonomy. Pelagic samples were selected for analysis, and littoral crustacean taxa were removed from the data prior to analysis to ensure a focus on the pelagic habitat and assemblages.

- *Benthic macroinvertebrates*: Data for lake benthic macroinvertebrates were obtained from the littoral, sub-littoral, and profundal zones where samples were collected using a variety of samplers (e.g., kick nets, stone scrubs, Surber samplers, dredges, Ekman grab, Ponar, corers). When top sediments were collected using corers, only Chironomidae (midges) were identified and enumerated to support paleolimnological work. Analysis therefore focused on subsets of data based on habitat, sampling method, and whether the full assemblage was assessed. Littoral data collected with kick nets or stone scrubs were combined for analysis, and profundal data collected with dredges or grab samplers were combined for a second analysis. Chironomidae were only identified to family level for many of these samples, so analysis was conducted at the taxonomic level of family or higher. A subset of profundal stations (including the paleo top cores) with Chironomidae to genus level were also analyzed. River samples had greater similarity in sampling methods and habitats than lakes, and analysis focused on samples from rocky substrates collected by kick nets. Mesh size was generally around 400-500 µm, though in some rare cases samples with a mesh size of 200 µm were used if no other samples were available for the region (generally only in Norway). Taxonomic level varied for river samples, and some samples were excluded if taxonomic level was too high (e.g., order level for USA samples) or excluded important groups (e.g., Finland samples that did not identify or enumerate chironomids and/or oligochaete worms). Because samples from Finland, USA, and some areas of Canada only identified chironomids to the family level, analysis was conducted on family-level data for the circumpolar region.

- *Fish*: Sampling methods varied widely across the circumpolar region, with many different types of sampling equipment (e.g., gill nets, hoop nets, seine nets, electrofishing, minnow traps, angling) and different mesh sizes. In addition, a large number of datasets (particularly for Canada) targeted individual species or groups of species (e.g., those important for commercial fishing) and thus did not identify or enumerate species in the full fish assemblage. To account for this and improve comparability of data, analysis focused on data from samples that enumerated the full fish assemblage, and data were converted to presence/absence to focus on composition trends rather than patterns in abundance. Analysis was at the species level.

Black fly swarms by Lake Mývatn, Iceland
Photo: Árni Einarsson

Cyclotella Antiqua*, a benthic algae species.*
Photo: Kathleen Rühland

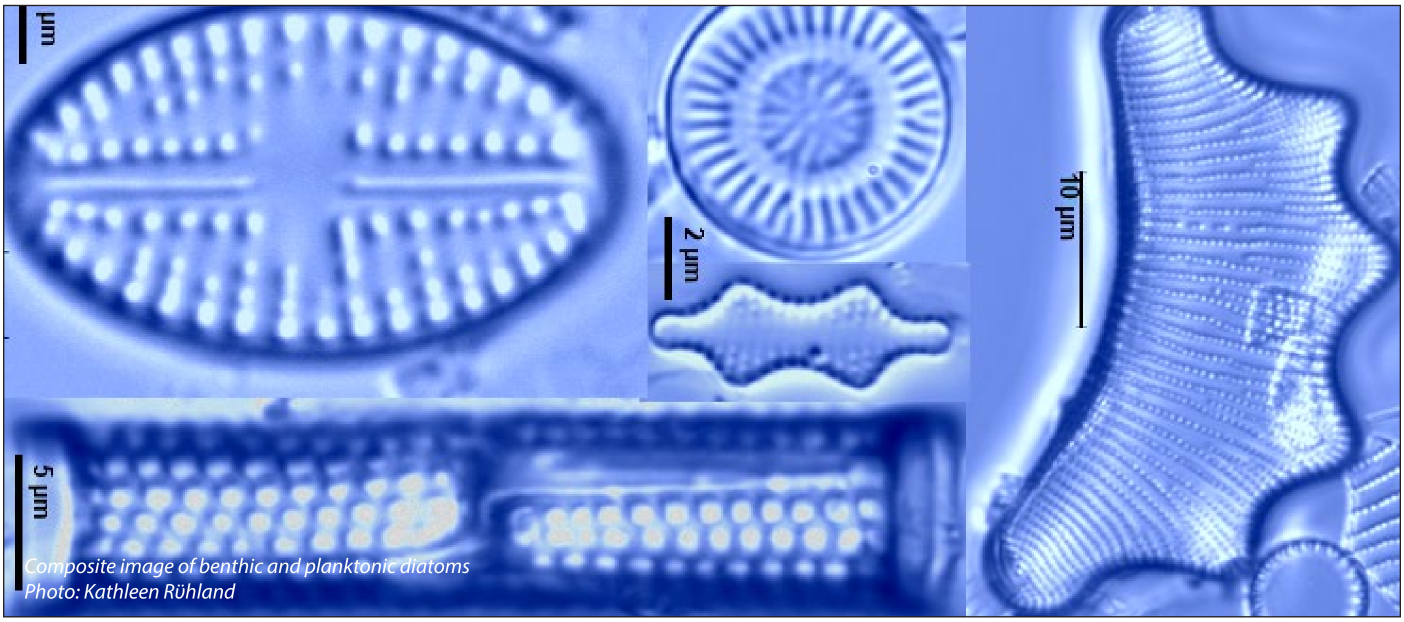
4.2. Algae from Benthic Samples

4.2.1. Introduction

Benthic algae are here defined in terms of the habitat in which samples were collected (e.g., lake sediments or streambeds), and thus include taxa collected from benthic habitats, regardless of whether they are otherwise defined as planktonic or benthic species. This definition allows for the comparison of all monitoring data collected by a particular method, without a need to classify species as benthic or planktonic. The Benthic Algae FEC describes primary producers that mainly occur in aquatic habitats, and includes both diatoms (class Bacillariophyceae) and soft algae (non-diatoms such as cyanobacteria, green algae, etc.). Algae from benthic samples are a heterogeneous group, ranging in size from micrometer-long small diatoms, cyanobacteria, or green algae to meter-long filamentous green algae. In benthic habitats, algae generally form a slippery, millimeter thick biofilm or thicker algal mats of around a centimeter. Benthic algae are important in food webs, and are often the main primary producers in open and shallow river stretches or lakes in the Arctic (refer to, e.g., Stevenson and Pan 1999, Wetzel 2001).

Studies of Arctic fresh waters have revealed that lake and stream benthic algae mats are commonly dominated by cyanobacteria, diatoms, and green algae (Conjugatophyceae and Chlorophyceae) (Wrona et al. 2013). Periphytic algal assemblages in Arctic streams are commonly composed of similar classes of algae, though assemblages are dominated by diatoms and cyanobacteria in the Canadian high Arctic (Lento et al. unpublished). In general, these algae provide important nutritional benefits to higher trophic levels, with diatoms in particular having substantial amounts of important fatty acids (Torres-Ruiz et al. 2007). Grazing of algae by herbivores can be affected by algal traits (e.g., morphology), nutritional value that can be modified by algal composition and environmental factors (Müller-Navarra et al. 2000, Ravet et al. 2003), as well as by invertebrate and fish predation on herbivores.

Most algae species in Arctic freshwaters are not restricted to polar regions, and their distributions reflect local geology and water chemistry conditions rather than thermal preferences (Wrona et al. 2013). Biomass of algae from benthic samples in lakes and rivers declines with increasing latitude, which may reflect the shorter growing season at high latitudes (Wrona et al. 2013). Arctic diatom diversity in both lakes and rivers is predicted to be affected by pH, nutrients, and metals, as found in temperate regions (Smol and Stoermer 2010). Additionally, lake diatom assemblage composition is expected to be driven by differences in the thickness and seasonal duration of ice cover, as these can affect light levels, length of the stratification period, and change across the 10°C July isotherm (Pienitz et al. 2004). In high Arctic regions, shallow, small water bodies might be most sensitive to the effects of climate change, and might display the most pronounced community shifts (Pienitz et al. 2004). Large-scale assessments of algal distributions from benthic samples in Arctic lakes and streams have not been previously done, and thus the broad-scale effect of climate change and human development on this FEC is unknown. In Arctic regions, diatom sampling is preferred over sampling the full assemblage (diatoms and non-diatoms) because of the greater ease of processing and identification of diatoms. Moreover, research on algae from benthic samples in Arctic lakes and rivers generally considers only diatoms or estimates biomass of the entire biofilm through bulk chlorophyll assessment. Thus, a broad-scale assessment of algal biodiversity from benthic samples using existing data must focus on the diatom component of these samples. Diatoms are advantageous for monitoring because they are sessile and thus cannot escape impacts, they grow quickly, react readily to environmental changes, and integrate impacts over a period of weeks. They are often the dominant algal group in benthic samples, are ubiquitous, and are diverse (> 100 taxa possible per cm2). Sample collection is simple, and the glass diatom frustule is preserved well over time (and in sediments), facilitating species-level identification. Diatom ecology and taxonomy is well-studied, and environmental preferences of species are well known, including the response to stressors such as eutrophication, acidification, pollution, salinisation and toxicants (Pienitz et al. 2004, Smol and Stoermer 2010, Villeneuve et al. 2013). These characteristics make diatoms valuable indicators of environmental change in freshwater ecosystems.



Composite image of benthic and planktonic diatoms
Photo: Kathleen Rühland

4.2.2. Objectives and Approach

Circumpolar assessment of contemporary diatom assemblages was completed for lake surface-sediment samples (i.e., top sediments or tops of cores) and river benthic scrapes. For this analysis, stations were grouped by ecoregion, which are terrestrial regions with similar geography and climate (Olson et al. 2001). Alpha diversity was assessed for each ecoregion by rarefying to a standard set of stations, while beta diversity was estimated within ecoregions and broken down into its components, i.e., turnover and nestedness. Diatom assemblages in lake surface sediments and shoreline scrapes, as well as in river benthic scrapes were also evaluated spatially to study contemporary biodiversity and compositional patterns. This was done both on a circumpolar scale and across a latitudinal (temperature) gradient, and for comparisons between North America and Europe. In addition, paleolimnological data (full lake sediment cores and tops/bottoms of cores) were used to assess long-term changes in biodiversity of lake diatom assemblages. The top-bottom approach is commonly applied in paleolimnological studies in order to assess change between two periods of time in a study set including a large number of lakes (e.g., Rühland et al. 2003). The results provide a before-and-after snapshot of lake conditions, often between the pre-industrial period and the late 20th century, when most lakes have experienced some form and degree of anthropogenic impact.

Diatom data were collected from both national monitoring programs and academic research studies, as routine monitoring of diatoms is limited to only a few countries (e.g., Sweden, Finland). Samples included those for which diatom assemblage structure was analyzed from prepared slides and where at least 300 diatom valves were enumerated under the microscope. Diatom nomenclature was harmonized at the genus or species level across the circumpolar region, and taxonomic differences due to variability in laboratory analysis methods were minimized by grouping some taxa into species complexes.

4.2.3. Overall Patterns and Trends

4.2.3.1. Circumpolar Diversity

4.2.3.1.1. Lakes

Lake diatom stations were arguably the most evenly distributed across the circumpolar region of all the FECs, although coverage was patchy in Russia and stations were lacking in the High Arctic of Greenland or Svalbard (Figure 4-7a). Ten ecoregions across the Arctic had sufficient numbers of samples to allow for comparison of rarefied richness estimates at 40 stations. Among these ecoregions, the highest alpha diversity was found in the Arctic Coastal Tundra in Alaska, with an average of 239 taxa (harmonized to include species complexes) in 40 stations, which was significantly higher than all other ecoregions (Figure 4-7b). High alpha diversity was also evident in northern Canada (Middle Arctic Tundra and High Arctic Tundra) and Iceland (Iceland Boreal Birch Forests and Alpine Tundra), where on average over 205-215 taxa were found in 40 stations (Figure 4-7b). Russia and Fennoscandia had slightly lower, but not significantly different alpha diversity levels (in the Taimyr-Central Siberian Tundra and Scandinavian Montane Birch Forest and Grasslands). The lowest alpha diversity was found in southern Greenland, in the Kalaallit Nunaat Low Arctic Tundra, which both had only 28 taxa on average in 40 stations (Figure 4-7b), whereas the southern ecoregions in Canada had low to moderate levels of alpha diversity (126 to 178 taxa in 40 stations).


Photo: Elena Shchipkova/Shutterstock.com

Similar patterns were evident when rarefied taxonomic richness was compared across all 19 ecoregions at a level of 10 stations. The Arctic Coastal Tundra, Middle Arctic Tundra, and Iceland Boreal Birch Forests and Alpine Tundra remained high alpha diversity ecoregions, but the Scandinavian Montane Birch Forest and Grasslands had similar richness to these ecoregions when only 10 stations were considered (Figure 4-7c). The highest diversity was in the Southern Hudson Bay Taiga in Canada, with an average of 155 taxa in 10 stations; however, this ecoregion did not differ significantly from those in the range of 135-150 taxa. The High Arctic Tundra was in a lower alpha richness class when only 10 stations were considered, and was similar to the Scandinavian and Russian Taiga and the other Russian ecoregions (East Siberian Taiga, Taimyr-Central Siberian Tundra, and Northwest Russian-Novaya Zemlya tundra), where 120-130 taxa were found on average in 10 stations. The Kalaallit Nunaat Low Arctic Tundra continued to have the lowest alpha diversity across all regions, but low richness was also found in a mountainous Alaskan ecoregion (Brooks-British Range Tundra) and in eastern and southeastern Canadian ecoregions (Baffin Coastal Tundra, Eastern Canadian Forests, and Eastern Canadian Shield Taiga), where average alpha diversity ranged from 65 to 98 taxa in 10 stations (Figure 4-7c).

Beta diversity β_{SOR} ranged from 0.50 to 0.88 when averaged across hydrobasins in an ecoregion, which indicated that there was generally moderate to high dissimilarity in community structure among stations within hydrobasins. The highest dissimilarity among stations ($\beta_{SOR} = 0.88$) was found in the Kalaallit Nunaat Low Arctic Tundra, which had the lowest alpha diversity. Thus, despite the low number of taxa, stations in the ecoregion were highly dissimilar. Furthermore, lake beta diversity was dominated by the turnover component in all ecoregions, which indicated that there was a high degree of species replacement across stations (Figure 4-7d), and a low contribution of nestedness to beta diversity (ranging from 3% to 23%, but at 15% or lower for 16 ecoregions). This suggests that spatially-extensive sampling is necessary to capture the majority of species across these ecoregions.

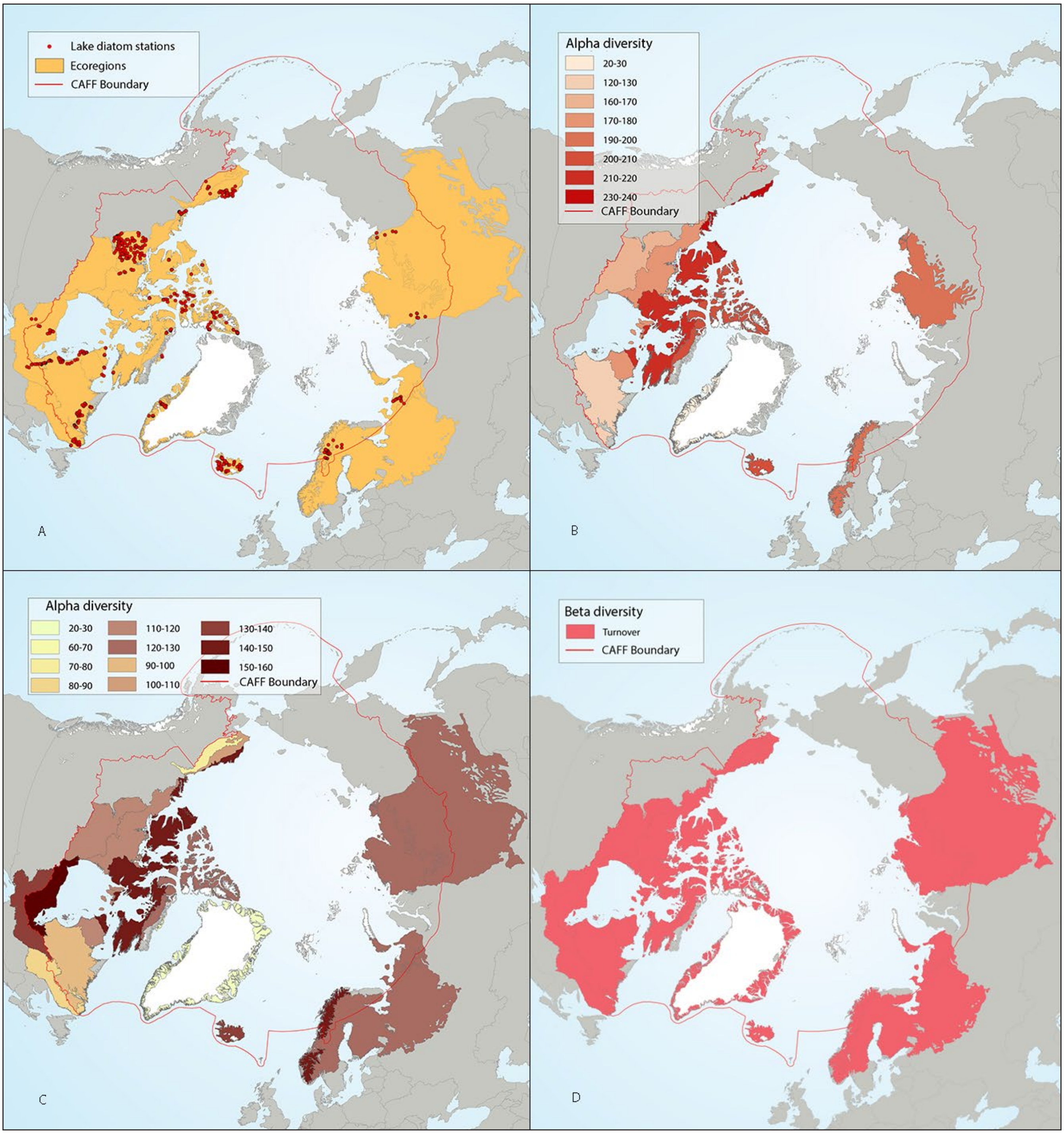


Figure 4-7 Circumpolar assessment of lake diatoms, indicating (a) the location of lake diatom stations, underlain by circumpolar ecoregions; (b) ecoregions with many lake diatom stations, colored on the basis of alpha diversity rarefied to 40 stations; (c) all ecoregions with lake diatom stations, colored on the basis of alpha diversity rarefied to 10 stations; (d) ecoregions with at least two stations in a hydrobasin, colored on the basis of the dominant component of beta diversity (i.e. species turnover, nestedness, approximately equal contribution, or no diversity) when averaged across hydrobasins in each ecoregio.

Photo: Nick Pecker/ Shutterstock.com

4.2.3.1.2 Rivers

River diatom data included large gaps in Russia, Iceland, and in the western Canadian low Arctic and sub-Arctic regions (Figure 4-8a). Eight ecoregions had a sufficient number of stations to allow for comparison of richness rarefied at the level of 40 stations. Among these, there were distinct and significantly different groupings of ecoregions with similar alpha diversity. The highest alpha diversity was in the Arctic Coastal Tundra in Alaska, which on average had 252 taxa in 40 stations (Figure 4-8b). The Fennoscandian ecoregions (Scandinavian and Russian Taiga, and Scandinavian Montane Birch Forest and Grasslands) were similar in alpha diversity, which ranged from 180 to 197 taxa. The northern Canadian ecoregions of the Middle Arctic Tundra and High Arctic Tundra had similar alpha diversity at 115-116 taxa on average in 40 stations (Figure 4-8b). Finally, the lowest richness was found in the eastern and southern Canadian ecoregions (Torngat Mountain Tundra, Eastern Canadian Shield Taiga, and Low Arctic Tundra), which ranged from 91 to 95 taxa on average in 40 stations.

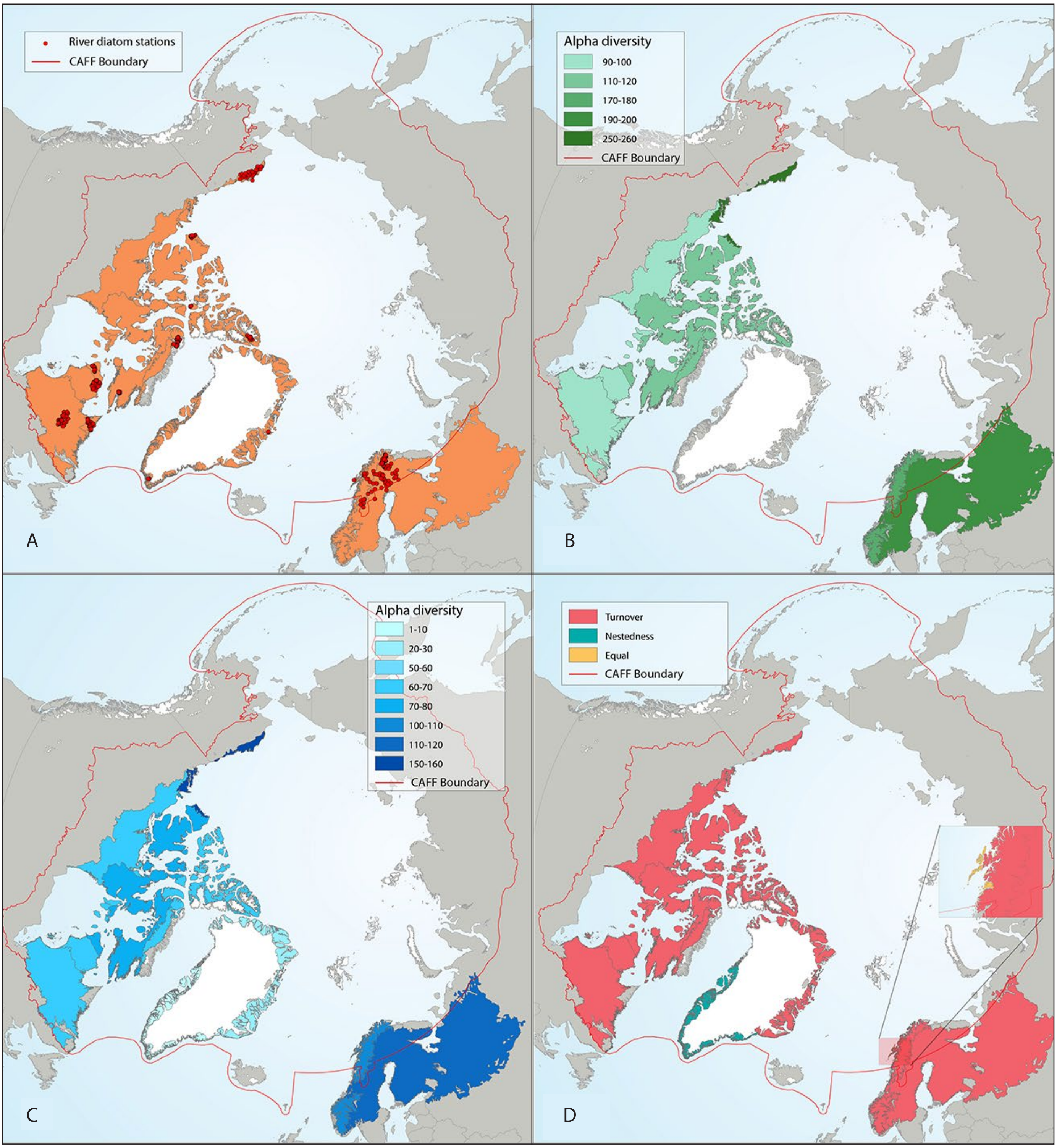


Figure 4-8 Results of circumpolar assessment of river diatoms, indicating (a) the location of river diatom stations, underlain by circumpolar ecoregions; (b) ecoregions with many river diatom stations, colored on the basis of alpha diversity rarefied to 40 stations; (c) all ecoregions with river diatom stations, colored on the basis of alpha diversity rarefied to 10 stations; (d) ecoregions with at least two stations in a hydrobasin, colored on the basis of the dominant component of beta diversity (species turnover, nestedness, approximately equal contribution, or no diversity) when averaged across hydrobasins in each ecoregion.

Comparison of richness values rarefied to 10 stations allowed the inclusion of two additional ecoregions in Greenland and one in Fennoscandia. Both of the Greenland ecoregions had the lowest alpha diversity estimates across the study area (Figure 4-8c). The Kalaallit Nunaat Low Arctic Tundra had an average of 3 taxa in 10 stations, whereas that for the Kalaallit Nunaat High Arctic Tundra averaged 28 taxa in 10 stations, with both values being significantly lower than those for the remaining ecoregions (Figure 4-8c). The Scandinavian Coastal Conifer Forests had low alpha diversity, similar to the Torngat Mountain Tundra, which is also a coastal ecoregion. Comparing the remaining ecoregions at only 10 stations (rather than at 40) resulted in less clear groupings of ecoregions in the low to moderate classes of alpha diversity (e.g., those ranging 52–72 taxa in 10 stations), with some shifts in the similarity of ecoregions. For example, High Arctic Tundra became more similar to the Low Arctic Tundra than to the Middle Arctic Tundra (Figure 4-8c). The observation that groupings of ecoregions were less evident when only 10 stations were sampled emphasizes the importance of sampling a sufficient number of stations in order to capture the range of species present and, consequently, accurately characterize alpha diversity within ecoregions.

Beta diversity within an ecoregion was highly variable for river diatoms, as β_{SOR} ranged from 0.2 (indicating strong similarity among stations in an ecoregion) to 0.81 (indicating strong dissimilarity among stations). Seven of the ecoregions had moderate dissimilarity among stations, with β_{SOR} between 0.5 and 0.65. The lowest beta diversity was in the Kalaallit Nunaat Low Arctic Tundra ($\beta_{SOR} = 0.2$) and the Scandinavian Coastal Conifer Forests ($\beta_{SOR} = 0.37$). In contrast, the highest beta diversity was in the Torngat Mountain Tundra ($\beta_{SOR} = 0.81$) and the Scandinavian Montane Birch Forest and Grassland ($\beta_{SOR} = 0.75$). Interestingly, one of the lowest beta (Scandinavian Coastal Conifer Forest) and the highest beta (Torngat Mountain Tundra) ecoregions had similar alpha diversity estimates at 10 stations (56 and 52 taxa, respectively). These ecoregions also differed with respect to the dominant component of beta diversity, as the Torngat Mountain Tundra was dominated by taxonomic turnover (90% of beta), whereas the Scandinavian Coastal Conifer Forests had more similar contributions of turnover and nestedness (64% and 36% of beta, respectively; Figure 4-8d). Across all ecoregions, turnover generally remained the dominant component of beta diversity. However, in the Kalaallit Nunaat Low Arctic Tundra, beta diversity was completely due to nestedness (Figure 4-8d), which reflected the low richness and low number of stations in this ecoregion.

4.2.3.2. Regional Diversity

Circumpolar analysis of ecoregions showed high biodiversity in both lakes and rivers along the northern Alaskan coast, in the Canadian Arctic Archipelago, and in Fennoscandia. At the site scale, lake stations with high alpha diversity were found throughout the circumpolar region (Figure 4-9), whereas clusters of high diversity river stations were somewhat more evident in Alaska and Fennoscandia (Figure 4-10). However, data were more spatially limited for rivers, which may have affected the distribution of taxonomic hotspots. Differences in diversity across the circumpolar regions may be related to a number of factors, including time since glaciation, underlying geology type, and site-specific substrate and nutrient conditions. For example, a number of samples from Alaska were collected in marshy habitats with sandy substrates and high levels of organic material, which may have led to different taxonomic composition than would be expected in rocky habitats. Furthermore, northern Alaska and the Arctic Archipelago are primarily underlain by sedimentary and sedimentary/volcanic bedrock, in contrast to the non-sedimentary bedrock underlying the southern Canadian Arctic regions. These geological differences may have contributed to contrasting diversity across these regions, as diatom species composition is strongly affected by underlying geology through its effect on water chemistry (Grenier et al. 2006).

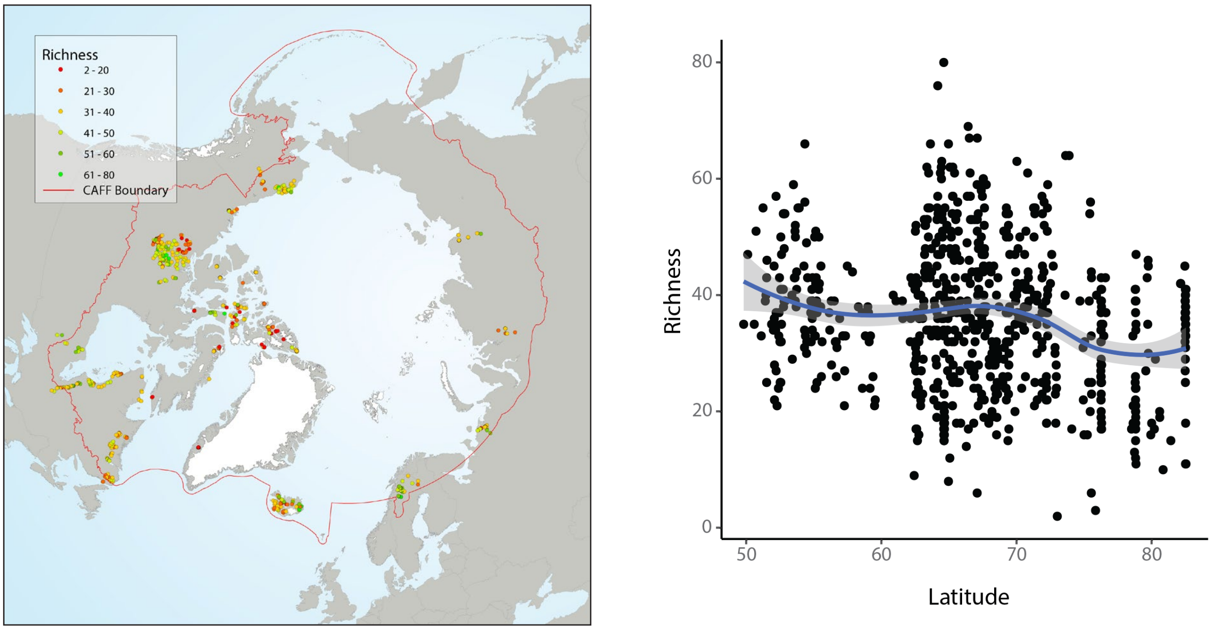


Figure 4-9 (left) Global species richness of diatom communities in Arctic lakes, with stations over the global mean taxonomic richness in red, and (right) local species richness of diatom communities in Arctic lakes across a latitudinal gradient.

Samples with the highest diatom richness, for both lakes and rivers, were generally in the latitude range of 60-75°N (Figure 49, Figure 4-10). However, the decline in richness outside this latitudinal range was small, and partly due to the fact that fewer samples were collected at the highest latitudes (above 75°N), particularly in rivers. Taxa accumulation curves indicated that an asymptote was not reached for either lake or river samples in any Arctic zone (sub-, low, or high Arctic; Figure 4-11), which suggests that raw richness estimates were affected by sampling effort. Taxa accumulation curves for lakes were similar until approximately 100 samples, above which high Arctic lakes had significantly lower taxonomic richness than sub- or low Arctic samples (Figure 4-11).

Similar results were observed for lakes, though the species accumulation curve for the low Arctic appeared to reach a richness value that was significantly higher than the estimate for the sub-Arctic (with no overlap of 95% confidence intervals for the two regions) at around 225 samples. Differences among Arctic zones were more evident for river data. The high Arctic had significantly lower taxonomic richness than the other two zones at 50 samples, whereas the low Arctic reached significantly higher richness than the sub-Arctic at approximately 75 samples. Patterns in lakes and rivers across Arctic zones were thus similar, but differences were stronger and more evident for river samples.

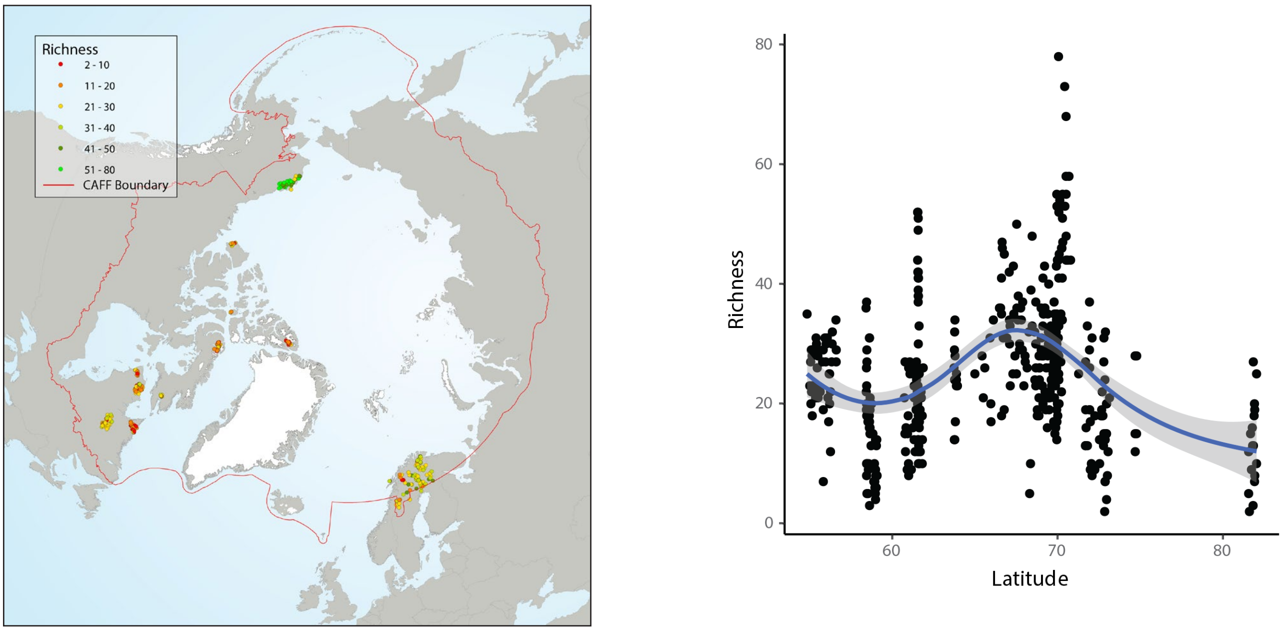


Figure 4-10 (left) Global species richness of diatom communities in Arctic streams, with stations over the global mean taxonomic richness in red, and (right) local species richness of diatom communities in Arctic streams across a latitudinal gradient.

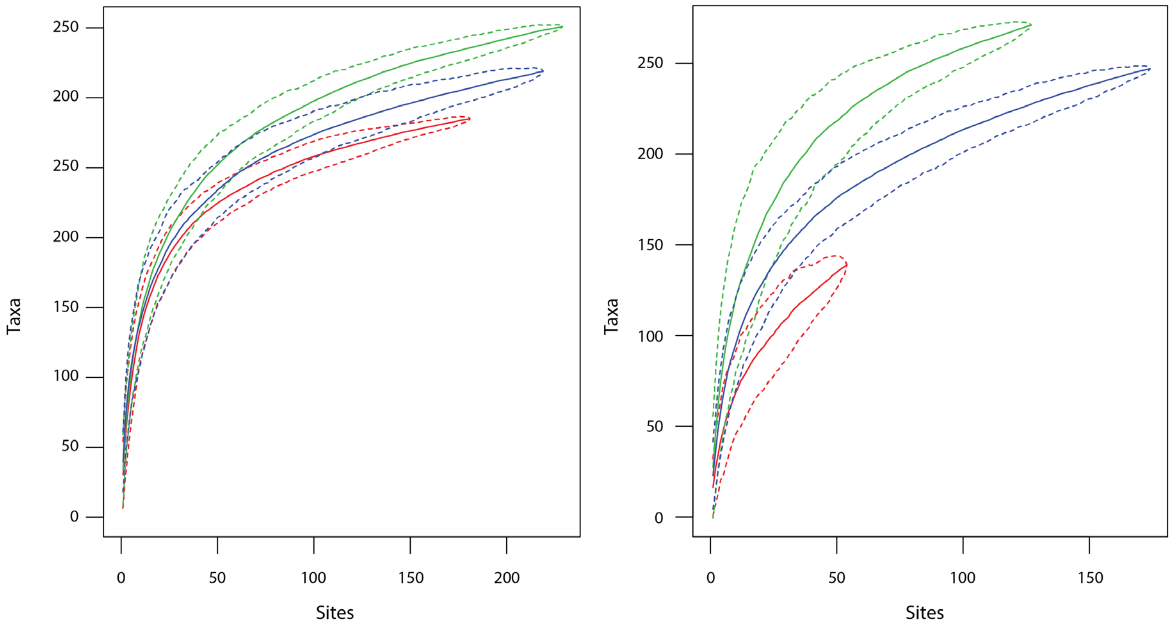


Figure 4-11 Taxa accumulation curves for (left) lakes and (right) rivers in the high Arctic (red), low Arctic (green), and sub-Arctic (blue) zones.

4.2.3.3. Compositional Patterns

Diatom taxa that were found to dominate the samples from across the circumpolar region are generally also common in other regions of the world, consistent with the suggestion that most algal species are not limited to the polar regions (Wrona et al. 2013). Within the studied regions, among-site differences in diatom assemblages were small, resulting in gradual shifts in composition across regions rather than clear thresholds of change. Many of the taxa that were present across the Arctic have an ecological preference for low nutrient levels and neutral pH, though a number of the identified taxa display a wide range of tolerances to environmental conditions and can be found in nutrient-rich conditions.

Diatom samples were grouped into biotypes with self-organizing maps (SOMs) to determine natural groupings of sites based on assemblage structure similarity and to evaluate spatial distributions of taxa. There were gradual changes between groups of sites rather than sharp differences, which indicated that there were no profound shifts in species taxa composition among regions. Clusters of samples did not differ significantly between North America and Europe/Russia, which indicated that taxa composition was not linked to longitudinal position (Figure 4-12). Instead, some of the similarity among sites was related to latitude, with apparent clusters of taxa associated with high latitude and low temperatures, particularly for the most spatially extensive dataset of lake top sediments (Figure 4-12).

Lake top sediment samples were characterized by six SOM clusters (biotypes), one of which represented a diatom assemblage that only occurred at very high latitudes (>70°N) and cold temperatures (< -15°C degrees annual average). This cluster was also characterized by a high number of unique species, contributing to the distinction of these northern samples. There were four SOM biotypes found for rivers, two of which were associated with high latitudes (>70°N). Of the high latitude biotype in rivers, only one was associated with cold temperatures (< -7°C annual average degrees). Lake shoreline samples were collected only at lower latitudes and thus did not allow for assessment of compositional differences between Arctic zones. Shoreline samples were characterized by three SOM biotypes, one of which was associated with higher latitudes than the others (>60°N) and colder temperatures (approximately < -7°C annual average degrees).

Figure 4-12 Diatom groups from Self Organizing Maps (SOMs) in lake top sediments, showing the geographical distribution of each group (with colors representing different SOM groups).

4.2.3.4. Temporal Trends

4.2.3.4.1. Lake sediment core top-bottom analysis

Analysis of differences between the tops and bottoms of lake cores was completed for 116 lakes, only 5 of which were located outside of North America. We divided the lakes into four categories based on geographical location, climate and vegetation (Boreal forest = BF, transition zone = T, Low Arctic = LA, High Arctic = HA). For certain analyses, sites were also categorized into 3 depth categories: ponds (< 2m), shallow lakes (2.1–6 m) and deep lakes (> 6.1m). Figure 4-13 illustrates the number of lakes per geographic zone for each of the depth categories.

The dataset for the top-bottom analysis included 452 diatom taxa, many of which were rare (e.g., found in low numbers at a single station). To eliminate rare taxa and reduce noise, only taxa that represented at least 2% in at least one sample were retained for the analysis, which reduced the number of species in the data set from 452 to 174. We then used the Bray-Curtis (B-C) dissimilarity coefficient to evaluate the difference in assemblage composition between ~1800 CE and contemporary sediments within each study lake. High values of this coefficient (which ranges from 0 to 100%) indicate strong dissimilarity between samples, whereas low values indicate that samples are similar. Mean values of the B-C dissimilarity coefficient were compared to assess if depth (pond, shallow, deep) and geographical zone (BF, T, LA, HA) influenced similarity between the bottoms and tops of cores.

B-C dissimilarity coefficient analysis showed a very wide range of similarity between tops and bottoms of cores among the stations (Bray-Curtis ranging between 14% and 91%; geographical distribution presented in Figure 4-14). Although the number of lakes outside North America was limited, the results suggested that systems in northern Siberia have changed more over time than those in northern Finland. In North America, western and high Arctic clusters (Tuktuyaktuk, Alaska, Ellesmere Island and other high Arctic Islands) displayed the largest change between now and c. 200 years ago. These were followed by the central Canadian sites (NWT and Hudson Bay lowlands). Finally, lakes in the central and eastern Canadian Arctic (lower islands of the Arctic Archipelago, Baffin Island and Northernmost Ungava) showed the least change in diatom assemblages in the past c. 200 years.

In North America, continent-wide long-term climatic changes, such as deglaciation, are known to follow a west-east spatio-temporal gradient (Dyke 2004). This is also true of recent climatic warming, evidenced by the degree of change observed in biotic assemblages over the past c. 200 years (Smol et al. 2005). The low degree of change in many eastern Canadian lakes may reflect the temperature stability that has been noted in this area, as the easternmost regions of the Canadian Arctic have not warmed as much as the west and the far north (Prowse et al. 2006a). At sites on central high Arctic islands including Victoria, Prince of Wales and Devon, in eastern and southern Baffin Island, as well as in northernmost Ungava, across the Hudson Strait, changes in diatom assemblages over the past 200 years have remained minor. This is likely to change in the near future, as climate warming and long-distance atmospheric deposition of pollutants increasingly encroach into the region (e.g., Saulnier-Talbot et al. 2015).

The highest latitudes in the Canadian Arctic region have some of the largest temporal variability among stations. This region includes some of the lakes with the largest change over the last c. 200 years, but also includes stations that have recorded very low levels of change. The duration of ice cover is the main environmental variable that has been driving the changes in diatom assemblage composition in this region (Griffiths et al. 2017), with the ice-free season changing from almost none to several weeks per summer. However, not all lakes in the Canadian Arctic region are subject to homogeneous decreases in ice-cover (Keatley et al. 2008), which can explain the large variation in temporal patterns among lakes at high latitudes. Temporal shifts in diatom communities may be less evident where there has not be a change to ice cover duration due to lake morphometry (e.g., deeper lakes, such as Elison Lake; Smol and Douglas 2007, Keatley et al. 2008), localized conditions (e.g., Skeleton Lake, where shading and shelter from wind have sustained ice

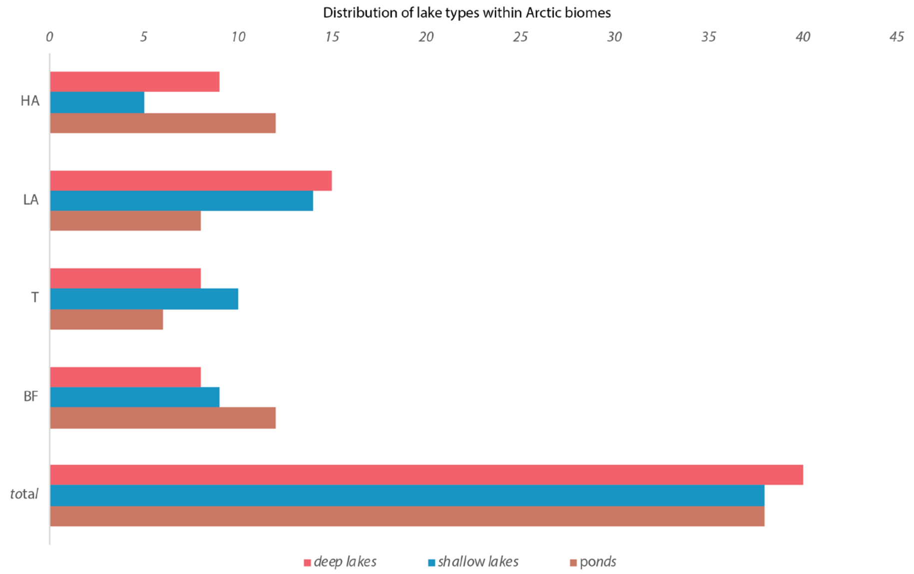


Figure 4-13 Number of deep lakes (red), shallow lakes (blue), and ponds (brown) in each geographical zone (BF, T, LA, HA). BF = Boreal Forest, T = Transition Zone, LA = Low Arctic, HA = High Arctic.

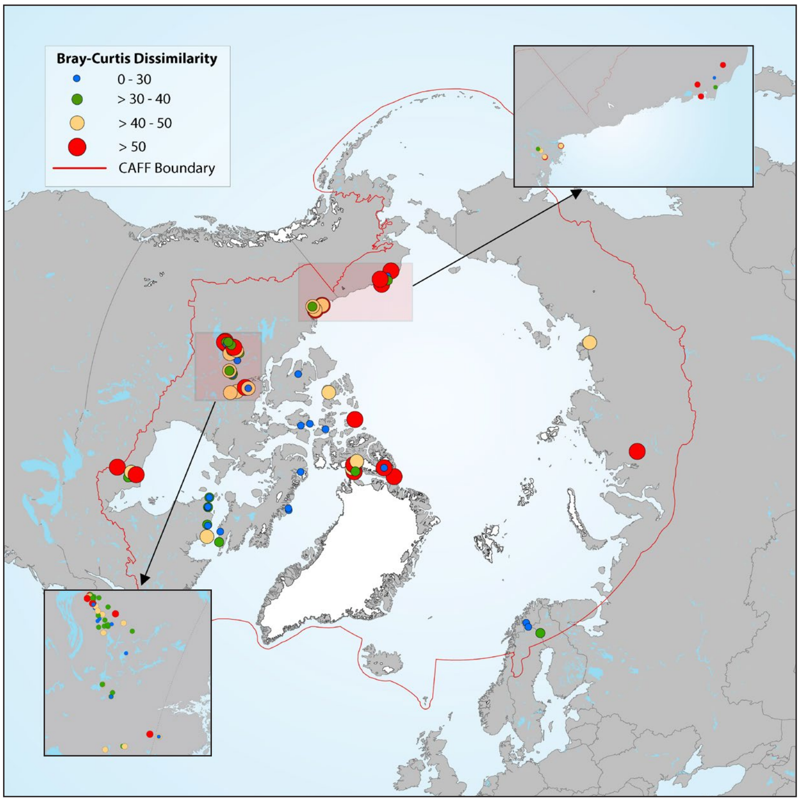


Figure 4-14 *Map showing the magnitude of change in diatom assemblages between bottom (pre-industrial) and top (modern) section of the cores, estimated by Bray-Curtis (B-C) dissimilarity. Boundaries for the B-C dissimilarity categories are based on distribution quartiles (0-30%, 30-40%, 40-50% and >50%), where the lowest values (blue dots) represent the lowest degree of change in diatom assemblage composition between top and bottom sediment core samples in each lake.*

cover; Keatley et al. 2008)(e.g., Skeleton Lake, where shading and shelter from wind have sustained ice cover; Keatley et al. 2008), or where the climate signal is dampened by other impacts (e.g., Eider Pond, which is dominated by the effects of bird colonies; Michelutti et al. 2010).

In some areas of the Arctic, there were clear shifts in taxa that contributed strongly to dissimilarity between bottom and top core samples. The large change recorded in the Russian sites reflected a transition in the assemblages from small *Fragilarias sensu lato* to epiphytic taxa (especially *Achnanthidium minutissimum*) in one lake, and from a strong dominance of *Aulacoseira humilis* (over 50% of the assemblage) to a more diverse assemblage dominated by small *Fragilarias sensu lato* in the other lake. Changes in the diatom assemblages of lakes located in the Central Canadian Arctic (continental NWT and Hudson Bay Lowlands) were characterised by a switch from a predominance of small, benthic *Fragilaria sensu lato* species to planktonic taxa such as *Discostella stelligera* (Rühland et al. 2003), reflecting changes in the length of the ice-free season and in stratification of the water column (Saros et al. 2016). Temporal change within lakes showed a high degree of similarity across this region. In contrast, in the subarctic Hudson Bay lowlands, temporal change was highly variable between lakes. In this region, diatom assemblages also responded to climatic warming and decreased ice-cover with an increase in planktonic species, relative to the pre-1850 period (Rühland et al. 2014).

Geographic zone and depth did not have a significant effect on B-C dissimilarity (two-way ANOVA; $p > 0.05$). However, mean within-zone and -lake type B-C dissimilarities suggested that ponds have changed more than shallow and deep lakes and that changes in boreal forest and transition zones are more pronounced than in the low Arctic and high Arctic. Ponds in the boreal forest and in the high Arctic changed more than those in the transition and low Arctic zones; shallow lakes in the transition and low Arctic zones changed more than in the boreal forest and high Arctic; and deep lakes in the boreal forest and transition zones changed more than those in the low Arctic and high Arctic zones (Table 4-1). This strong spatial heterogeneity suggests that different lake types respond differently to change in different biomes.

Table 4-1 Lake depth categories that have changed the most in each geographic zone.

Zone	Ponds	Shallow lakes	Deep lakes
Boreal Forest	X		X
Transition Zone		X	X
Low Arctic		X	
High Arctic	X		

4.2.3.4.2. Identification of Sentinel Species

A SIMPER analysis (similarity percentage) was conducted using the Top-Bottom data to identify individual taxonomic groups that contributed most to the observed differences between pre-industrial and present-day assemblages. This multivariate analysis quantifies the contribution of each variable in a group and can help to identify the variables responsible for the differences between groups (here, fossil versus modern samples). For selected indicator taxa, we plotted the bottom vs. top relative abundances to evaluate the concordance between past and modern data (1:1 plot). The distribution of relative abundances along the 1:1 line indicates the shifts in abundance of these taxa between past and present (below the 1:1 line indicates higher abundances in fossil samples while above the 1:1 line indicates higher abundances in modern samples).

A number of diatom taxa showed large changes in relative abundances between bottom and top samples of the cores. In the boreal forest, the most striking assemblage change was from a predominance of small *Fragilaria sensu-lato* to the planktonic species *Discostella stelligera* in all the deep lakes and in most shallow lakes. This was also the case in many low Arctic lakes, and to a lesser degree was also visible in the transition lakes. In the high Arctic, there was a change from small *Fragilaria sensu lato* to a more diverse assemblage dominated by other small benthic taxa such as *Achnanthes minutissima*, which were found at higher relative abundances in modern samples than in fossil samples for ponds. Ponds in the boreal forest showed a shift to lower relative abundances of *Nitzschia fonticola* in modern compared with fossil sediments. *Pseudostaurosira brevistriata* similarly declined in modern samples relative to fossil samples in lakes of all sizes in the transition zone.

When samples were pooled across all Arctic zones and lake sizes, there was evidence that the Discostella complex showed a higher relative abundances in modern sediments than in fossil sediments (Figure 4-15), indicating a major temporal shift in the prevalence of this group. Increased relative abundance in modern sediments was also evident for *Psammothidium marginulatum/scoticum/levenderi* (Figure 4-15). Whereas both of these groups increased in dominance in modern sediments, there were other taxonomic groups that became less prevalent in modern samples, including *Aulacoseira perglabra* complex and *Staurosirella pinnata* complex (Figure 4-15). Such a shift in taxa could indicate changes in lake stratification over time in response to to climatic changes.

4.2.3.4.3. Downcore Analysis

Diatom composition was analyzed along the length of the core (downcore) to evaluate gradual temporal changes in assemblage structure and biodiversity since the early 19th century. This analysis provided a representation of ecological reorganization that was similar, albeit more in-depth, to the analysis of top and bottom core samples. Only cores that had robust chronologies were retained for this analysis (a total of 52). Samples along the length of the core were compared in multivariate analysis (Detrended Correspondence Analysis) to obtain a measure of beta diversity, which indicated the degree of dissimilarity across the c. 200 years of sediments analyzed from the core. The years in which the sediments were sampled varied considerably. As such, some cores may have been retrieved before the beginning of significant warming in certain areas of the study region, affecting the degree of change between recent and past.

Sediment core analysis results showed high variability in beta diversity across downcore samples, which indicated

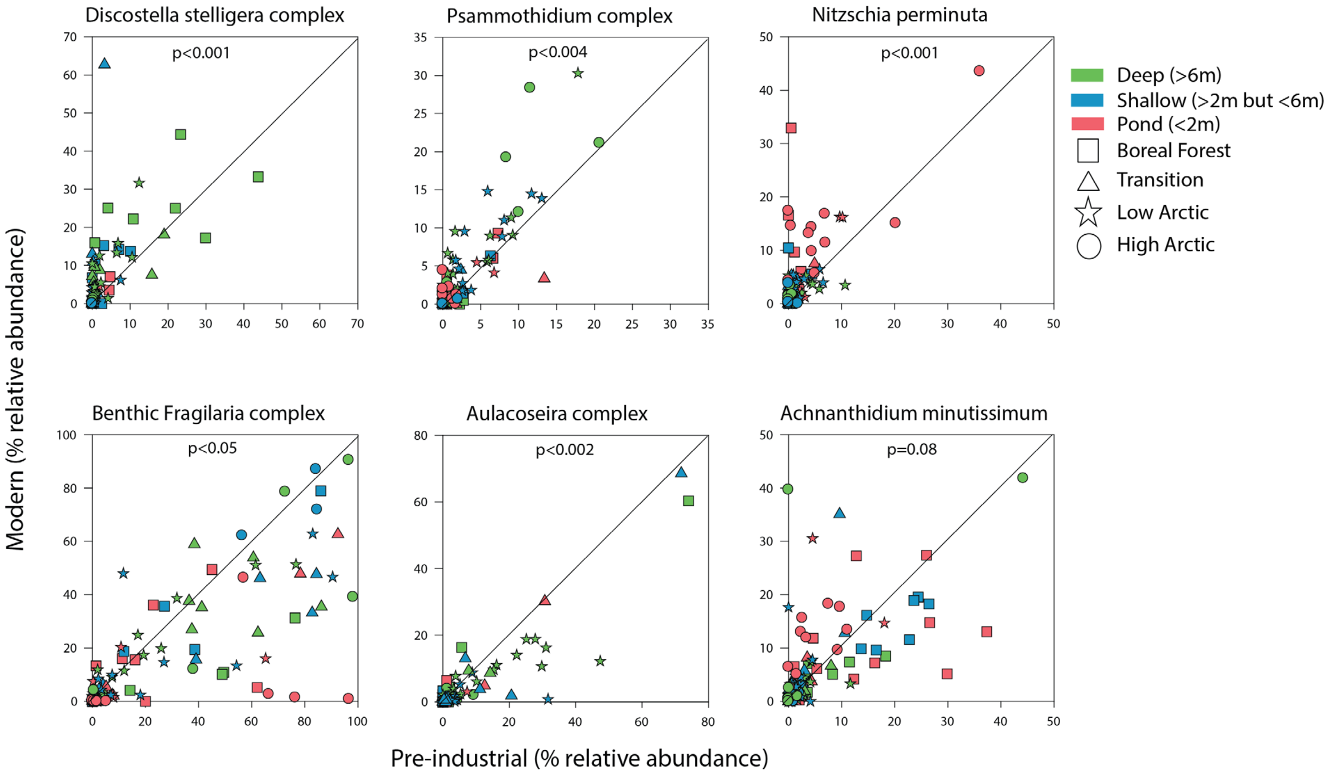


Figure 4-15 Comparison of the relative abundance of select diatom taxonomic groups between core bottoms (pre-industrial sediments; x-axis) and core tops (modern sediments; y-axis) with a 1:1 line to indicate whether there were higher abundances in fossil samples (below line) or modern samples (above line).

that rates of change in diatom assemblage composition were not the same across the sampled lakes. Nevertheless, some regional trends were observed. The areas that showed the least change since c. 1800 (lowest beta diversity) included northernmost Québec and eastern Baffin Island in Canada, the Central Canadian Archipelago and northern Finland (Figure 4-16). Sites in the Canadian high Arctic and northeastern Northwest Territories, Canada showed moderate overall change, whereas the largest changes were observed in the westernmost cluster of sites, near the Beaufort Sea in Canada, and southernmost sites (southern Hudson Bay; Figure 4-16). Variability in relative change among lakes was high at these sites. However, these results are similar to those obtained by Smol et al. (2005).

4.2.4. Gaps in Knowledge and Monitoring

Lake top sediment samples covered a large extent of the circumpolar region, but this coverage was primarily due to academic research, and thus does not represent repeated monitoring. Additional data from academia do exist (for example, additional samples are available from paleolimnological training datasets for Europe, including samples on Svalbard and on the Kola Peninsula in Russia), and these could be integrated into the CBMP database for future assessments. Any further assessment of diatoms from lake sediments must rely primarily on academic data, as these samples are not generally part of monitoring programs, with the exception of monitoring by industry and local- or regional-scale government monitoring (e.g., the province of Québec in Canada). However, the advantage of diatom samples in lakes is that long-term changes can be inferred from collecting sediment cores, thus compensating for the lack of routine monitoring activities.

River samples were more sparse, and were lacking from Russia, Iceland, Greenland, Svalbard, and central and western Canada. Although river algae monitoring takes place in some Arctic countries (e.g., Norway, Sweden, Finland), it is limited elsewhere in the circumpolar region (though some local- or regional-scale government monitoring occurs, e.g., the province of Québec in Canada). Furthermore, even in countries where monitoring occurs, the samples may not always be comparable if they focus on non-diatoms (e.g., in Norway) or do not follow comparable procedures (e.g., if they do not digest samples, thus potentially obscuring diatom taxa and making species-level identification impossible). Obviously, there is a clear need to increase the spatial scope of river diatom monitoring in order to capture the full variability in biodiversity across the circumpolar region.

Paleolimnological data for lakes similarly had gaps, in particular across most countries outside North America, resulting from the need to acquire downcore and top-bottom core data from academic sources, as this work is not conducted as part of routine monitoring. In some cases, core data were obtained from published research, but critical information on dates and/or depths may have been missing, or data were obtainable as presence/absence only, when analysis focused on relative abundances. Additional paleolimnological data repositories exist and could be accessed to improve data coverage, particularly for Europe where there was little collection of data of this type. Furthermore, it would be beneficial to make a targeted effort to approach paleolimnological researchers who have not been involved in the CBMP process, to request access to full core data that could fill the gaps in spatial coverage and allow for a broader assessment of long-term trends.

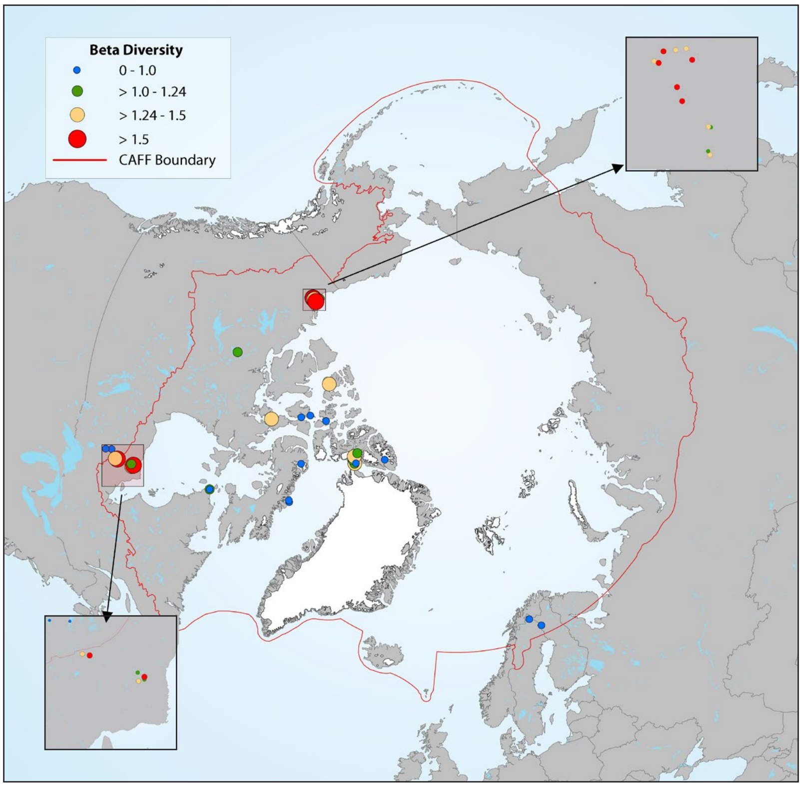


Figure 4-16 Map showing the magnitude of change in diatom assemblages for downcore samples, with beta diversity used as a measure of the compositional differences between samples at different depths along the core. Boundaries for the beta diversity categories are based on distribution quartiles (0-0.1, 0.1-1.24, 1.24-1.5, >1.5), where the lowest values (blue dots) represent the lowest degree of change in diatom assemblage composition along the length of the core in each lake.

Achnanthes minutissima.
Photo: Chris Carter

4.3. Phytoplankton

4.3.1. Introduction

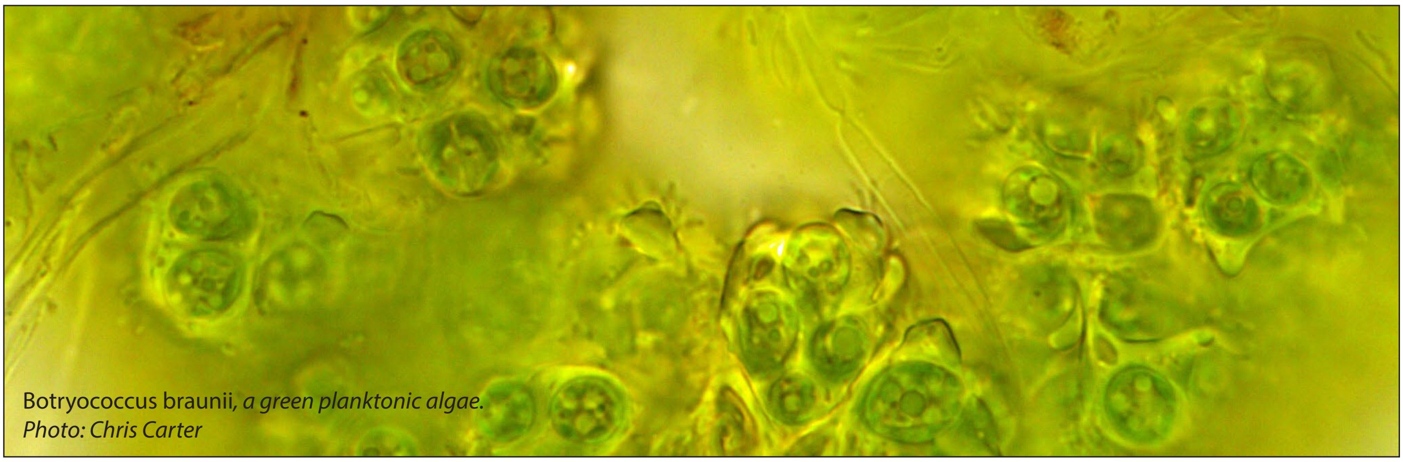
Primary producers living in the water column (pelagic zone) of lakes are microscopic phytoplankton that include single cells and small-cell colonies that rely on dissolved nutrients and light for growth. Although phytoplankton cells account for < 1% of photosynthetic biomass on earth, they are responsible for about 50% of global net primary production and are the primary source of energy in lake ecosystems (Field et al. 1998). Phytoplankton communities of Arctic lakes typically include diatoms, dinoflagellates, chrysophytes, and benthic cyanobacteria, while chlorophytes and pelagic cyanobacteria are less common (Sheath 1986, Forsström et al. 2005). Few, if any, species of phytoplankton are exclusively found in the Arctic; diatoms species are numerous but chrysophytes (e.g., *Dinobryon*) and dinoflagellates (e.g., *Gymnodinium*) are represented by fewer species. Species such as *Dinobryon* are adapted to utilise bacteria as an energy source, thus they can grow during winter with little or no light and then switch to photosynthesis as light becomes available. In addition, the dynamics of phytoplankton community composition have important implications for biogeochemical cycling processes (Winder and Sommer 2012).

Species richness and biomass can range greatly across Arctic lakes depending on environmental conditions that regulate their need for resources, including factors such as temperature, precipitation, UV radiation levels, spread of viruses and fungi, and predation by zooplankton (Prowse et al. 2006b, Reynolds 2006). Species numbers can total up to several hundred per lake and be correlated negatively with latitude and negatively with altitude (and thus water temperature; Stomp et al. 2011). Phytoplankton biomass and community composition is commonly regulated by the availability of light and the key nutrients, nitrogen (N) and phosphorus (P) (Sterner and Elser 2002, Mette et al. 2011). Under N-deficient conditions, Cyanobacteria can become dominant because this group is capable of converting atmospheric N2 (i.e., nitrogen fixation) to biologically-available N forms (Pick and Lean 1987). In clear, nutrient-poor Arctic lakes, phytoplankton production is low and dominated by small species. Phytoplankton species are often the preferred food source for filter feeders; however, in nutrient-poor lakes some pelagic feeders can obtain a higher proportion of their diet from benthic primary production (Vadeboncoeur et al. 2003, Mariash et al. 2014). Small-bodied taxa like the Chrysophyte, *Cryptomomas,* are often the most dominant pelagic algal groups; however, diatoms and dinoflagellates can also be numerically dominant (Christoffersen et al. 2008).

Phytoplankton species distribution and composition has been well studied in some regions of the Arctic but is not standardly included in monitoring programs. There has been no large-scale description of the biogeographical distribution of phytoplankton species in the Arctic regions (Wrona et al. 2013); however, studies from lower latitudinal gradients (e.g., Stomp et al. 2011) point to a decrease in diversity towards the North that may reflect responses of environmental drivers to geographical gradients (latitude, longitude and altitude). Since important abiotic factors driving phytoplankton growth, such as nutrients and temperature, decrease towards higher latitudes, species richness is expected to be lower in the Arctic compared to temperate regions. Nevertheless, phytoplankton richness can be relatively high in oligotrophic boreal lakes (Forsström et al. 2005), and this is especially the case for shallow lakes with many semi-planktonic species of desmids and diatoms. Mass blooms of harmful, toxin-producing cyanobacteria or other nuisance species are rarely reported in the water column, although they often dominate the microbial mat communities in Arctic lakes (Bonilla et al. 2005).

4.3.2. Objectives and Approach

The aim of this assessment is to provide a summary of phytoplankton biodiversity in lakes from the high Arctic to the sub-Arctic regions to provide a baseline for future monitoring programs, detect any changes that have occurred over time and identify gaps in spatial coverage across the Arctic. The assessment also includes summaries of spatial and temporal patterns of biodiversity and biomass in Arctic lakes. Sample locations were grouped into ecoregions of similar geography and climate (Olson et al. 2001) at a circumpolar scale, and alpha diversity, beta diversity, and its component parts (turnover and nestedness) were evaluated within ecoregions. Spatial and temporal patterns in phytoplankton diversity and species composition were also assessed at a regional scale (by country and by Arctic region) for the circumpolar Arctic. This assessment of phytoplankton diversity, which is based on contemporary data, provides a baseline with which future monitoring results can be compared and identifies gaps in the current distribution of phytoplankton monitoring.



Botryococcus braunii, *a green planktonic algae.*
Photo: Chris Carter

4.3.3. Overall Patterns and Trends

4.3.3.1. Circumpolar Diversity

For the among-ecoregion comparisons, alpha diversity (taxon richness) was assessed for 6 ecoregions that had 30–69 lakes/stations (Figure 4-17a). When data were rarefied to assess taxonomic richness at 35 stations for each ecoregion, the highest alpha diversity was found for the Arctic Coastal Tundra in Alaska (268 taxa), followed by the Low Arctic Tundra in Canada and the Kalaallit Nunaat High Arctic Tundra in Greenland (with 225 and 219 taxa, respectively; Figure 4-17b). Alpha diversity was lower in southern Greenland, where lakes on the Kalaallit Nunaat Low Arctic Tundra had 186 taxa, and in the ecoregions sampled in Russia (lakes on the East Siberian Taiga had 112 taxa and those on the Taimyr-Central Siberian Tundra had 81 taxa; Figure 4-17b). Differences between ecoregions were significant for all but the Low Arctic Tundra and the Kalaallit Nunaat High Arctic Tundra.

Another eight ecoregions had data for 19 or fewer stations and were not included in among-ecoregion comparisons above, as extrapolation to 35 stations was far outside the range of stations per ecoregion. To include ecoregions with

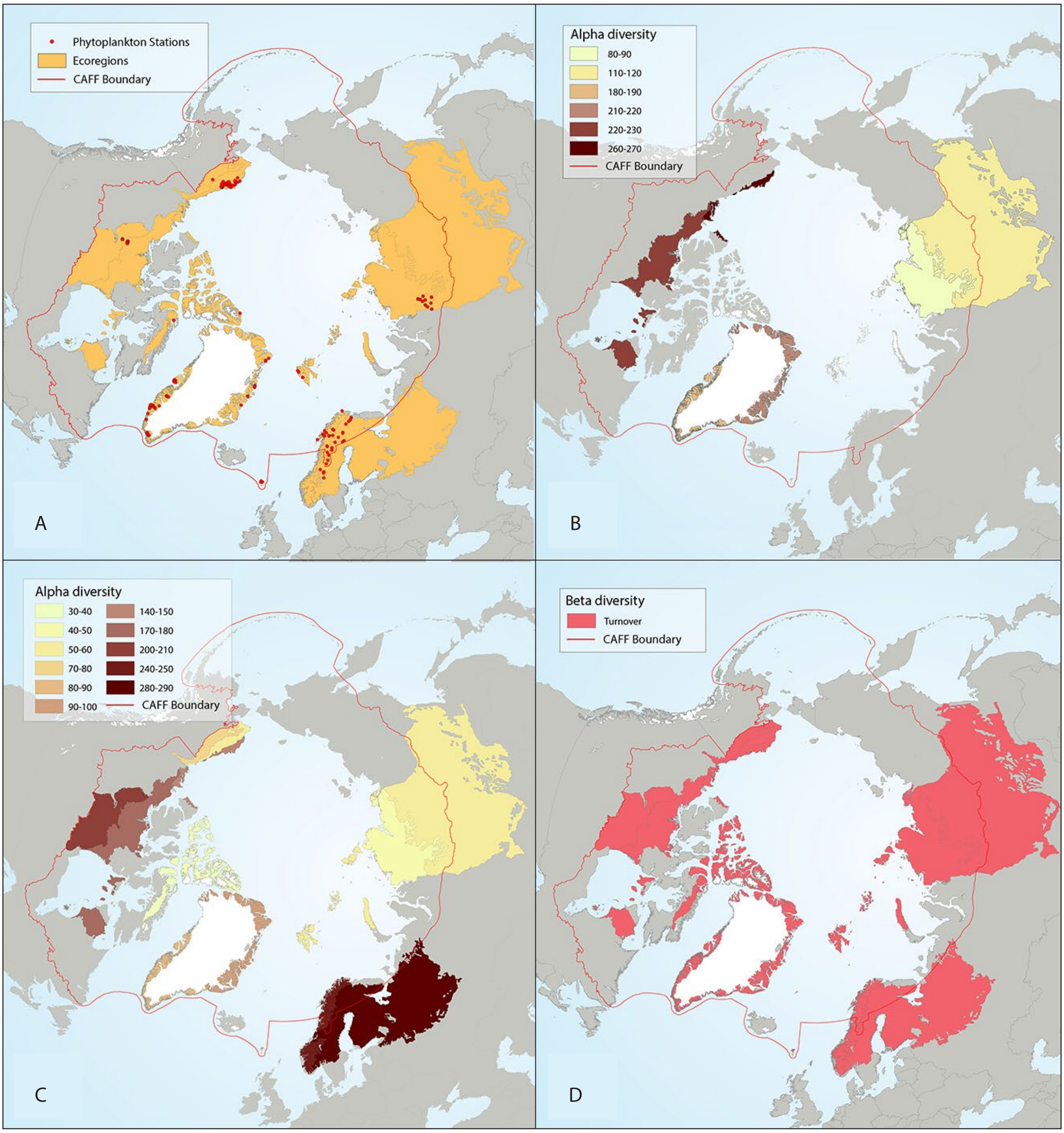


Figure 4-17 Results of circumpolar assessment of lake phytoplankton, indicating (a) the location of phytoplankton stations, underlain by circumpolar ecoregions; (b) ecoregions with many phytoplankton stations, colored on the basis of alpha diversity rarefied to 35 stations; (c) all ecoregions with phytoplankton stations, colored on the basis of alpha diversity rarefied to 10 stations; (d) ecoregions with at least two stations in a hydrobasin, colored on the basis of the dominant component of beta diversity (species turnover, nestedness, approximately equal contribution, or no diversity) when averaged across hydrobasins in each ecoregion.

fewer sampled stations, the analysis was conducted on all ecoregions with rarefaction and extrapolation used to assess taxonomic richness at a sampling effort of 10 stations in a region. The Scandinavian and Russian Taiga, the Scandinavian Montane Birch Forest and Grasslands, and the Northern Canadian Shield Taiga had the highest alpha diversity at 280, 247, and 201 taxa, respectively (Figure 4-17c). These estimates of taxonomic richness were all significantly different (no overlap among 95% confidence intervals). In this assessment at 10 stations, there was significantly lower taxonomic richness in the Arctic Coastal Tundra (144 taxa), the Low Arctic Tundra (170 taxa) and the Kalaallit Nunaat High Arctic Tundra (100 taxa), which were the ecoregions with the highest alpha diversity when more stations were considered in the analysis (Figure 4-17c). Alpha diversity estimates for these three ecoregions were also significantly different from one another. The other eight regions had 40–87 taxa, but for some of these only between 2 and 6 stations were included, suggesting a potential for large error in the estimate of alpha diversity at 10 stations. This assessment highlighted the importance of increasing sampling effort within ecoregions to include more stations, as patterns of diversity may differ greatly depending on how many stations are sampled and how much variability exists among those stations.

Without extrapolation to a larger number of stations, there remained evidence that the Scandinavian ecoregions, in particular, had higher alpha diversity than other regions. For example, the Scandinavian and Russian Taiga had a total of 351 taxa found across the full 19 stations that were sampled in this ecoregion, and the Scandinavian Montane Birch Forest and Grasslands had 333 taxa across its 19 stations. In contrast, the Arctic Coastal Tundra (in Alaska) had an estimate of 202 taxa at 19 stations and only reached a total of 343 taxa across the 64 stations that were sampled in that ecoregion, whereas the Kalaallit Nunaat Low Arctic Tundra (in Greenland) had an estimate of 132 taxa at 19 stations and only reached a total of 256 taxa across its 69 stations. Presumably, differences in alpha diversity between the Scandinavian ecoregions and the ecoregions in Greenland and in North America would have been even greater had a comparable number of stations been sampled.

Beta diversity among ecoregions ranged between 0.31 and 0.90. Beta diversity exceeded 0.80 for the Arctic Coastal Tundra, the Arctic Foothill Tundra, the East Siberian Taiga, the Low Arctic Tundra, the North Canadian Shield Taiga, Scandinavian Montane Birch Forest and Grasslands, and the Taimyr-Central Siberian Tundra. This means that the lakes in these regions showed the highest among-station diversity, i.e., showed a high differentiation in phytoplankton assemblages. Beta diversity was low in the Brooks-British Range Tundra, the High Arctic Tundra, the Kalaallit Nunaat High Arctic Tundra, and the Scandinavian and Russian Taiga. Homogeneous conditions in the catchment, resulting in rather similar water body types and water quality can contribute to this. Turnover was the dominant component of beta diversity in all ecoregions (Figure 4-17d), accounting for at least 70% of the total beta diversity. This reflected the important contribution of the introduction of new species across stations, and indicates the importance of spatially extensive monitoring of phytoplankton in lakes, to ensure the full variability due to species turnover is captured.

4.3.3.2. Regional Diversity

Beta diversity was assessed for each Arctic region (sub-Arctic, low Arctic, and high Arctic) by comparing the mean statistical distance of lakes to the centroid for each Arctic region in multivariate space, where larger distances are indicative of greater differences among assemblages. The low and high Arctic lakes on average had higher beta diversity than the sub-Arctic lakes when the distance to the centroid was used as an estimate of beta diversity (Figure 4-18). However, the average distance to the centroid was not significantly different among the three regions, mainly due to the large variability among sub-Arctic lakes. Sub-Arctic lakes are more heterogeneous due to a higher variability in catchment characteristics (e.g., vegetation cover, permafrost, nutrient concentrations) than lakes at higher latitudes. The fact that there were more samples from the sub-Arctic region (and more samples per lake) as well as wider geographic sample coverage likely also contributed to this pattern.

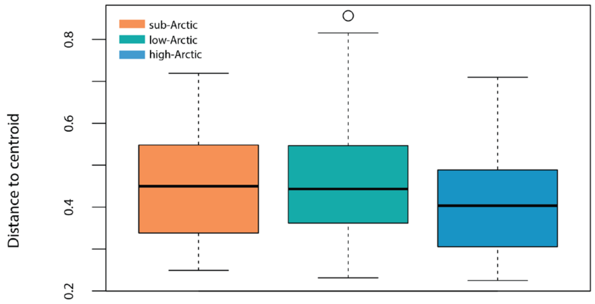


Figure 4-18 Box plot represents the homogeneity of assemblages in high Arctic (n=190), low Arctic (n=370) and sub-Arctic lakes (n=1151), i.e., the distance of individual lake phytoplankton assemblages to the group centroid in multivariate space. The mean distance to the centroid for each of the regions can be seen as an estimated of beta diversity, with increasing distance equating to greater differences among assemblages.

Macrophytes in lake, Rybachy peninsula near Murmansk, Russia
Photo: svic/Shutterstock.com

4.3.3.3. Compositional Patterns

There were 8-10 phytoplankton classes present within each Arctic region. Chrysophytes and Chlorophyceae (green algae) were the most dominant phytoplankton groups across all geographical regions (Figure 4-19), with the exception of Russia where cyanobacteria was most abundant across lake sites. The most common phytoplankton classes in the sub-Arctic regions were Chrysophyceae and Chlorophyceae. The next most common were Cyanophyceae, Bacillariophyceae (diatoms), Synurophyceae, and Dinophyceae with more than 1000 occurrences in the sub-Arctic (Figure 4-19). In the low Arctic, the assemblage composition was more balanced, with nearly 71% of the community equally represented by congugatophytes, chrysophytes, diatoms, and cyanobacteria (Figure 4-19). In the high Arctic, chrysophytes clearly dominated, making up 34% of the community, whereas Dinoflagellates and green algae together contributed to 27% of the community on average (Figure 4-19).

4.3.3.4. Temporal Trends

Phytoplankton alpha diversity was compared based on the time period of collection to evaluate whether general patterns in species richness among Arctic regions has varied over time. The oldest records, from 1940-1980, showed high taxa richness, but these data covered very few years and sites (n< 34), making it difficult for direct comparison with the later contemporary years (n> 100) (1980-2000 and 2000-2015; Figure 4-20). Across both of the later time periods, there were significant differences in taxa numbers between Arctic regions (p = 0.03), with lakes in the high Arctic and low Arctic generally both having < 20 taxa, compared to the sub-Arctic sites that had approximately 30 taxa per site. Average richness was slightly higher from 2000-2015 than it was from 1980-2000 in the high Arctic and sub-Arctic, whereas the low Arctic showed a larger increase in richness in the later time period, but neither trend was significant. However, these patterns may have been affected by increased frequency and geographic coverage of sampling stations in later years.

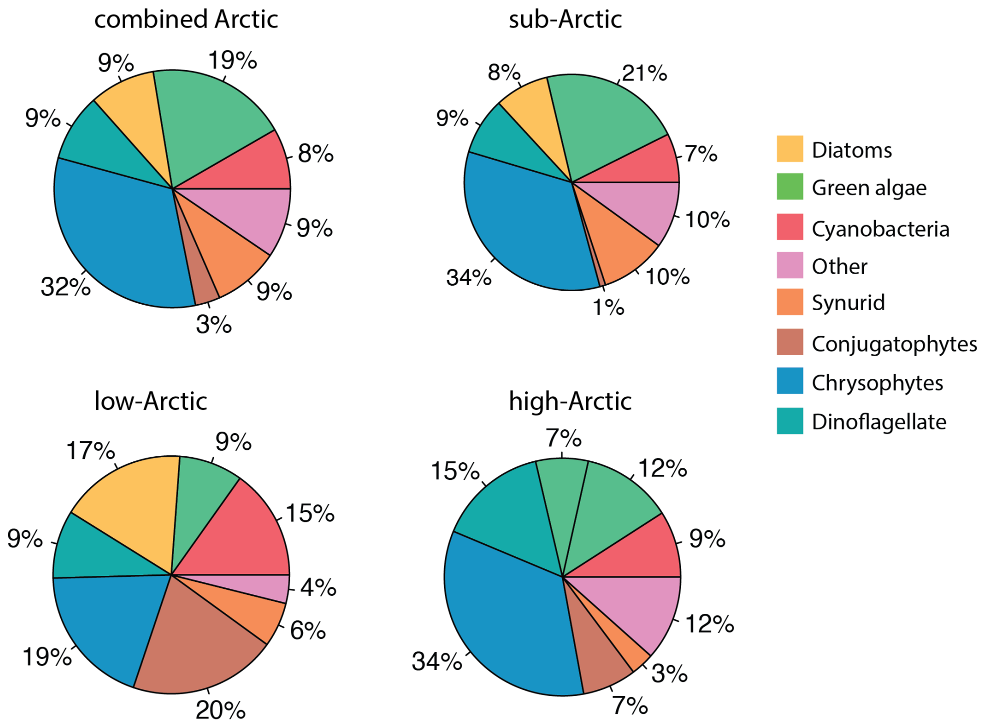
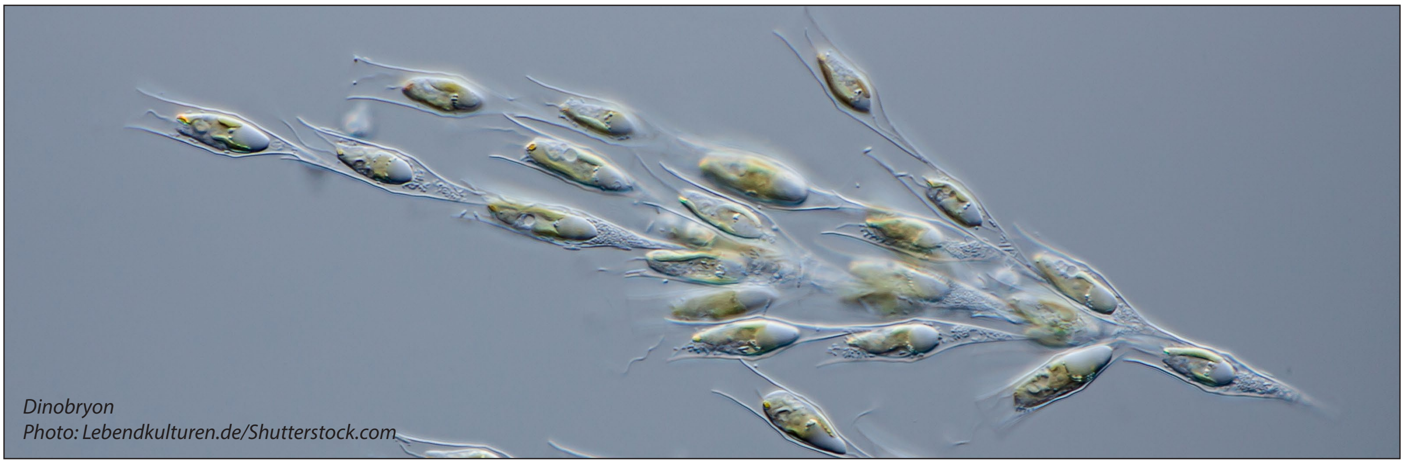


Figure 4-19 Phytoplankton percent composition by dominant classes across the three Arctic regions, using relative presence across stations calculated from from presence –absence data.



Dinobryon
Photo: Lebendkulturen.de/Shutterstock.com

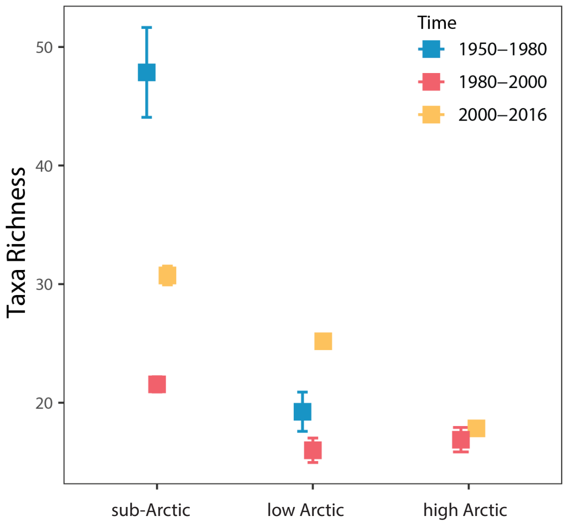


Figure 4-20 Phytoplankton species richness averaged by time periods ±SE in each Arctic region.

4.3.3.4.1. Temporal Trends in Total Biovolume and Composition

Temporal trends in community composition were more closely evaluated for lakes with greater than 10 years of phytoplankton biovolume data. Lakes with long time series were identified in Finland, Greenland, and Sweden. Whereas Finland and Greenland each had two lakes with greater than 10 years of data, Sweden had 12 lakes with more than 10 years of phytoplankton monitoring data. Productivity differed among lakes, with generally high productivity (>250 $mm^3 L^{-1}$) for the two Greenland lakes and for Pallasjärvi in Finland. The Swedish lakes and Inarijärvi in Finland had a lower productivity of on average less than 100 mm3L-1. Shifts in biovolume were compared among lakes to identify monotonic (i.e., single-directional) temporal trends.

From the late 1980s to present, total biovolume of phytoplankton increased in Inarijärvi in Finland and in several Swedish lakes, while biovolume decreased in Langemandssø in Greenland. Temporal trends in Greenland and Finland were variable in general, with some apparent outliers often masking trends. For example, when an extremely high value in 2014 (biovolume = 854 $mm^3 L^{-1}$) was excluded, there was a significant decrease in total biovolume in Greenland lake Langemandssø (Mann-Kendall trend test (M-K) $p = 0.024$; Sen's slope of trend = -11.59). Finland's Inarijärvi had extremely variable total biovolume of phytoplankton prior to 1995, but from 1995 to 2014 there was evidence of a significant increasing trend (MK $p = 0.001$; slope = 1.19) despite an extremely high value in 2001. For Sweden, the trends were more clear across lakes, which were either sampled from circa 1988 to present (e.g., Abiskojaure, Jutsajaure, Stor-Tjulträsket), or sampled from circa 2000 to present. Clear significant increasing trends (all with $p < 0.05$) in total biovolume were evident in the Swedish lakes Abiskojaure (slope = 0.64), Båtkåjaure (slope = 1.63), Övre Fjätsjön (slope = 2.41), Jutsajaure (slope = 3.92), and Stor-Tjulträsket (slope = 2.30). Remaining Swedish lakes either displayed no trend over time, or had trends driven by a single outlier year. Overall, these trends indicate that total biovolume is decreasing in some of the highest productivity lakes and increasing in many low productivity lakes, leading to a more similar level of phytoplankton biovolume across these systems.

Though there was evidence of significant temporal trends in overall biovolume, it was often less clear which groups of phytoplankton contributed to those trends. In Greenland and Finland, trends in Chrysophyceae appeared to most closely reflect overall biovolume trends, with increasing biovolume of chrysophytes in Finland lakes and decreasing chrysophyte biovolume in Greenland lakes (though this decrease was only significant for Sommerfuglesø, which did not display a significant trend in overall biovolume). However, Chrysophyceae only showed a clear trend in one Swedish lake (Abiskojaure), and there was little evidence of broad trends in other phytoplankton classes in Swedish lakes, which indicated that shifts in biovolume over time could not be attributed to a single group.

4.3.3.4.2. Cyanobacteria response to a Changing Climate

Four main climate drivers of phytoplankton can be summarized as i) water temperature, ii) water column irradiance and clarity, iii) stratification regime and residence time, which are also influenced by local precipitation patterns, and iv) availability of nutrients (Paerl and Huisman 2008). More specifically in the Arctic, the seasonality of phytoplankton communities is directly affected by changes in ice coverage (Vincent 2007, Prowse et al. 2011c). Not only will earlier ice-off dates and later ice-on dates increase the length of the growing season, but they will shift the peak spring phytoplankton bloom earlier (Prowse et al. 2011c). Furthermore, a decrease in the period of ice cover and increased nutrient inputs could contribute to increased prevalence of cyanobacteria blooms (Prowse et al. 2011c). Current climate trends are showing rising temperatures in the Arctic, lengthening of the ice-off period, and changes in precipitation patterns (IPCC 2007).

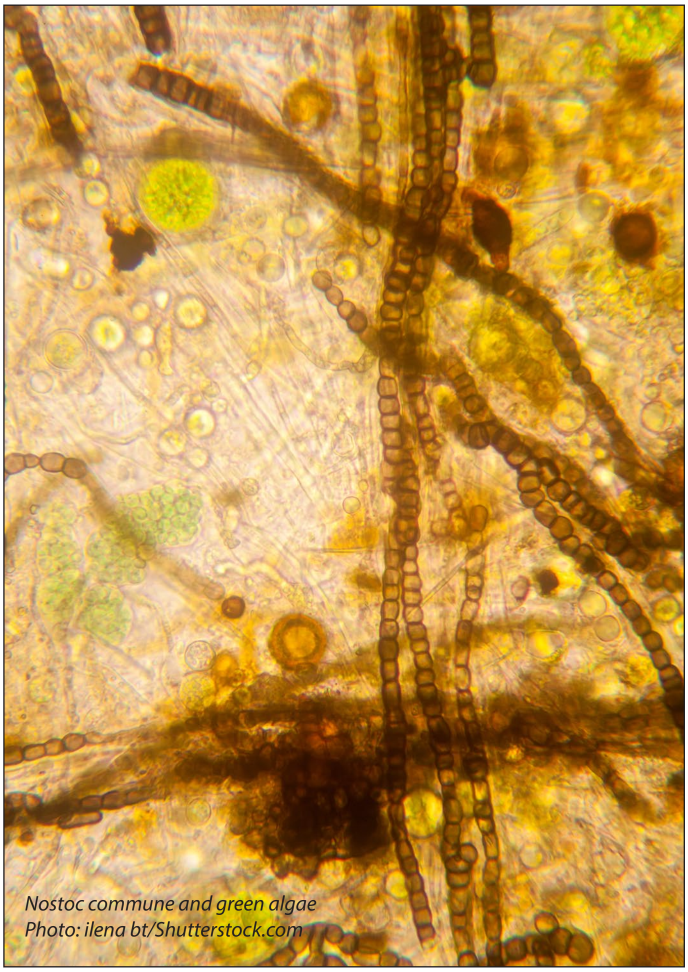

Nostoc commune and green algae
Photo: ilena bt/Shutterstock.com

Figure 4-21 Circumpolar Arctic distribution of Cyanophyceae using presence- absence data from all sites sampled between 1980-2015.

Cyanobacteria are often considered to be a nuisance or even toxic phytoplankton group, as they are capable of creating thick surface blooms and outcompeting other phytoplankton. Although mostly thought to inhabit warm-temperate climates, they are commonly found in Polar regions (Vincent 2007). The circumpolar database confirmed these patterns, as cyanobacteria were found in low Arctic and sub-Arctic lakes, including about half the Russian lakes, some northern Fennoscandian lakes, as well as a few lakes in western Greenland and Canada (Figure 4-21). Blooms of toxic algae do not generallyoccur in Arctic lakes (Wrona et al. 2013), and toxin presence has only been confirmed in one case (Trout-Haney et al. 2016), though such occurrences might become more prevalent if climate change leads to warmer temperatures and higher nutrient inputs to lake and river systems. Cyanobacteria often dominate the benthic mats and algae in the littoral areas in cold habitats (Vincent 2007), but there was evidence of pelagic cyanobacteria in some of the high Arctic lakes in our dataset.

Temporal patterns in cyanobacteria biovolume were used as an indicator of how climate change has affected lake phytoplankton assemblages in the Arctic. With Cyanobacteria favouring warmer waters and abundant nutrients, we predicted an expanded geographical range and increased dominance of cyanobacteria since 1990, concordant with a period of increased warming. However, our results showed no overall increase in the regional distribution of cyanobacteria from 1980-2000 compared with 2001-2015. There was limited evidence of unidirectional trends in cyanobacteria biovolume across long-term data records from Greenland, Finland, or Sweden.

Despite the lack of long-term trends, cyanobacteria biovolume showed similar peaks across a number of lakes that may have corresponded with shifts in temperature. Long-term records were examined for Finland and Sweden (which had lakes with more consistent time series, with fewer gaps in recent years) to identify the years in which cyanobacteria biovolume appeared to peak (that is, was notably increased relative to other years on record), and 66% of these peaks (37 of 56 high values of cyanobacteria biovolume) occurred during one of the hottest years on record (Table 4-2). High biovolume of cyanobacteria was most prominent in 2014, the hottest year on record since 1880, when 64% of lakes had a peak value (and in some cases these values were the highest recorded for cyanobacteria across the lake's time series; e.g., Pahajärvi had a peak of 2944 mm^3/L in 2014, compared with the next highest value of 192 mm^3/L in 2006). More than 20% of the lakes showed peak cyanobacteria biovolume in 2013 and 2003 (ranked the 5th hottest years), 2006, 2007, and 2010 (ranked the second hottest year; Table 4-2). An additional 32% of the peaks in cyanobacteria biovolume across lake times series (19 of 56 high values) were found to follow a hot year (e.g., a peak was noted in 2004, 2008, or 2011), and there were also five lakes sampled in Greenland that had cyanobacteria blooms of > 2000 mm^3/L in 2004. These peaks in years following record hot years could reflect a delayed reaction to temperature increases, particularly as there appeared to be certain lakes that regularly showed these off-pattern peaks. Peaks in these years may also have occurred as a delayed response to consecutive warm years (e.g., 2004 followed record hot years in 2003 and 2004; 2008 followed hot years in 2005 through 2007; and 2011 followed record hot years in 2009 and 2010). Notably, these off-pattern peaks in cyanobacteria were generally not followed by another high biovolume value in the next year when temperatures were once again elevated. Since rising temperature and decreased ice potentially enhance cyanobacterial dominance (Paerl and Huisman 2008), continued monitoring of cyanobacteria in all Arctic regions may be useful in tracking associated climate and nutrient changes in Arctic water bodies.

4.3.4. Gaps in Knowledge and Monitoring

Monitoring of phytoplankton is not completed regularly in all Arctic countries, and data are therefore patchy both in spatial coverage and temporal coverage. The best coverage of phytoplankton monitoring data exists in Fennoscandia and Greenland, though most data are located in low Arctic or sub-Arctic regions, and high Arctic coverage is sparse (particularly in Svalbard). Monitoring designs vary among these countries, with a different focus on maximizing spatial or temporal data coverage in different regions. For example, monitoring in Finland, Sweden, and Greenland takes place at a small number of sites, but focuses on preserving long time series. In contrast, phytoplankton monitoring in Norway includes repeated sampling within a year in many stations (in some cases including monthly sampling), but the suite of sites differs from year to year. Thus stations may have only 6 years of biovolume data over a 15-year period, with many gaps in the time series.

There is virtually no consistent phytoplankton monitoring in North America. Data for Canada, Russia, and USA were largely sourced from academic research or from monitoring data collected by industry, which tends to include a large number of stations from a small number of lakes, though sampling is repeated annually and often monthly for those stations. Academic data can provide spatial coverage for limited areas, but rarely includes repeated sampling over a long time period, thus limiting the number of time series that can be examined. The result is that there is insufficient data, particularly for Canada, to accurately describe biodiversity across this region.

The need for more monitoring sites across North America, Russia, and other northern areas of the Arctic is clear. Norway began monitoring approximately 15 lakes in northern regions in 2017, which will begin to fill some gaps. But to allow for comparison and assessment across phytoplankton monitoring data, future monitoring efforts must endeavor to improve consistency in sample processing methods (particularly with respect to the estimation of biovolume) and taxonomic resolution. Data collected for this assessment included a mixture of biovolume (estimated by multiple methods), biomass, density, abundance, and presence/absence. Such a range of measurements are difficult to combine in a way that retains maximum information (i.e., retaining more information than presence/absence).

Furthermore, phytoplankton data included a mix of species-level and genus-level data, which can complicate assessment of taxonomic structure. Where possible, potential taxonomic redundancy from multi-level nomenclature was removed from the data, but future efforts should focus on obtaining species-level data where possible.

Table 4-2 The percent of lakes with a peak in cyanobacteria biovolume in each of the 10 hottest years on record from 1880 to 2014 (temperature rankings from NOAA National Centers for Environmental Information 2015). Percents were calculated using only lakes sampled in a particular year, and a peak was defined as a notable increase in biovolume relative to other years on record, with magnitude of peaks varying by lake and by year.

Rank (1 = Warmest)	Year	Percent of lakes with cyanobacteria peak
1	2014	64%
2 (tie)	2010	21%
2 (tie)	2005	15%
4	1998	18%
5 (tie)	2013	36%
5 (tie	2003	36%
7	2002	18%
8	2006	29%
9 (tie)	2009	14%
9 (tie)	2007	29%

Water milfoil (*Myriophyllum alterniflorum*).
Photo: Mps197/Shutterstock.com

4.4. Macrophytes

4.4.1. Introduction

Macrophytes are a diverse group of aquatic plants large enough to see with the naked eye. There are 644 described species of vascular macrophytes in the Nearctic region and 497 species in the Palearctic region (Chambers et al. 2008), though Arctic zones of these biogeographical regions are expected to be less diverse. Macrophytes are taxonomically and phenologically wide ranging, from macroalgae (such as macroscopic species of green algae or Chlorophyta), to mosses and liverworts (Bryophyta), ferns (Pteridophyta) and seed-bearing plants (Spermatophyta) (Chambers et al. 2008). Macroscopic forms of Cyanobacteria, Xanthophyta (yellow-green algae) and Rhodophyta (red algae) can also be classified as aquatic macrophytes. Morphological forms of aquatic macrophytes include emergent (rooted plants with foliage extending into the air), floating-leaved (plants rooted to the lake or stream bottom with leaves that float on the water surface), submersed (plants growing completely submerged under the water and attached to, or closely associated with the substrate), and free-floating macrophytes (plants that typically float on or under the water surface) (Chambers et al. 2008). In addition, the depth distribution of macrophytes in lakes and rivers is often determined by the light penetration through water.

Macrophytes are an important functional component of lake ecosystems. They remove nutrients (e.g., nitrogen and phosphorus) from the water column (e.g., Gumbricht 1993, Jeppesen et al. 1998) and decrease wave energy and water currents, which leads to increased sedimentation and stabilization of sediment within macrophyte beds (e.g., Carpenter and Lodge 1986, Sand-Jensen 1997). Moreover, these beds provide habitat for fish, invertebrates, and epiphytes, and are an important food source for some invertebrates (e.g., insects) and vertebrates (e.g., fish, birds, moose) (Lodge 1991, Newman 1991). Wrona et al. (2013) indicate there are several major environmental factors that affect macrophyte distribution including nutrient levels, water clarity and water temperature (including ice regimes). Because macrophyte presence and abundance is closely associated with these environmental factors as well as substrate type, the composition of macrophyte communities can provide diagnostic information on water quality and is part of many countries' assessment criteria (Jeppesen et al. 1998, Søndergaard et al. 2010).

4.4.2. Objectives and Approach

This circumpolar assessment provides a summary of broad spatial patterns of aquatic macrophyte biodiversity in the Arctic. To accomplish this we examined presence/absence data for macrophyte species-level data compiled for 440 lakes in all Arctic countries except Russia (Figure 4-22a). We examined spatial distribution patterns of macrophyte species composition, alpha diversity (i.e., species richness), and beta diversity and its component parts (i.e., turnover and nestedness) for regions with numerous data records. Using this approach, we produced a baseline for current macrophyte species distribution and composition to which future monitoring results can be compared. Knowledge gaps related to macrophyte monitoring in lakes and rivers were also identified.

4.4.3.Overall Patterns and Trends

4.4.3.1. Circumpolar Diversity

Among the three ecoregions with the largest number of sampling stations, there was significantly lower alpha diversity in the Iceland Boreal Birch Forests and Alpine Tundra (estimate of 35 species at 70 stations) than in either the Scandinavian and Russian Taiga or the Scandinavian Montane Birch Forest and Grasslands (estimate of 111 and 112 species at 70 stations, respectively; Figure 4-22b). Macrophyte distribution ranges are thought to be largely determined by seed dispersal via migratory birds and human activity, though continental drift and geographic proximity may have influenced dispersal patterns (Les et al. 2003, Chambers et al. 2008). These processes may have contributed to lower observed alpha diversity in Iceland compared to the continental Scandinavian ecoregions. When compared to one another using 100 stations, alpha diversity estimates were similar for the two Scandinavian ecoregions (120 species in the Scandinavian and Russian Taiga and 130 species in the Scandinavian Montane Birch Forest and Grasslands), indicating generally high diversity within these two ecoregions.

Rarefaction of species across all ecoregions, using 10 stations as the assessment threshold, showed alpha diversity estimates were lowest (< 15 species) for the tundra ecoregions of the Brooks-British Range, Kalaallit Nunaat High Arctic, Kola Peninsula and High Arctic (Figure 4-22c). Three of the ecoregions with the lowest species richness were located at the highest latitudes (average latitude > 70°N for the stations in the ecoregion), suggesting that alpha diversity of macrophytes declines in high-latitude Arctic regions. This is consistent with past research, which has suggested that there are latitudinal and altitudinal gradients in alpha diversity (Chambers et al. 2008), with aquatic vascular macrophytes showing a decline in species richness with latitude (Wrona et al. 2013). The highest alpha diversity (> 45 species) was in lakes of the Arctic Coastal Tundra, Northwest Territories Taiga, Scandinavian and Russian Taiga and Scandinavian Montane Birch Forest and Grasslands (Figure 4-22c). Interestingly, the Scandinavian and Russian taiga ecoregion had the highest estimated alpha diversity when only 10 samples were used (60 species), whereas it generally had lower alpha diversity than the Scandinavian Montane Birch Forest and Grasslands when a more representative sample size (e.g., over 70 stations) was considered. This result highlights the importance of sampling a sufficient number of stations across these regions.

Beta diversity of macrophyte assemblages ranged between 0 (no inter-station differences in species composition) and 1 (no inter-station overlap in species) within the ecoregions. Ecoregions with the highest inter-station differences ($\beta_{SOR} > 0.80$) included the Arctic Coastal Tundra, Brooks-British Range Tundra, Kalaallit Nunaat High Arctic Tundra, Kalaallit Nunaat Low Arctic Tundra, Northwest TerritoriesTaiga, Scandinavian Coastal Conifer Forests, and Scandinavian Montane Birch Forest and Grasslands. Beta diversity was lowest (high inter-station composition overlap) in the remote ecoregions with low connectivity, such as the Faroe Islands Boreal Grasslands, High Arctic Tundra, and Kola Peninsula Tundra. For most ecoregions, turnover was the dominant component of beta

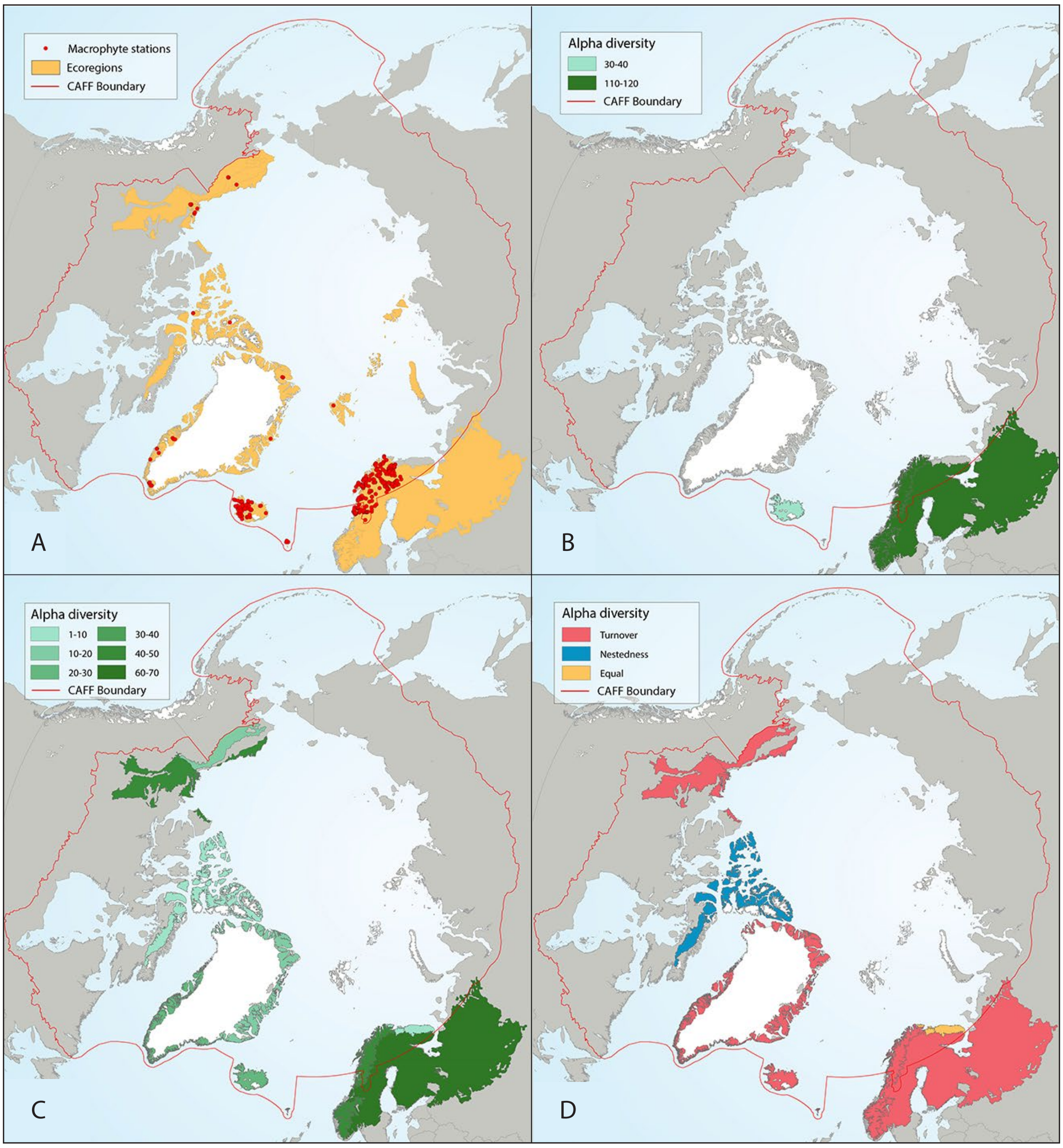


Figure 4-22 Results of circumpolar assessment of lake macrophytes, indicating (a) the location of macrophyte stations, underlain by circumpolar ecoregions; (b) ecoregions with many macrophyte stations, colored on the basis of alpha diversity rarefied to 70 stations; (c) all ecoregions with macrophyte stations, colored on the basis of alpha diversity rarefied to 10 stations; (d) ecoregions with at least two stations in a hydrobasin, colored on the basis of the dominant component of beta diversity (species turnover, nestedness, approximately equal contribution, or no diversity) when averaged across hydrobasins in each ecoregion.

diversity as it accounted for more than 70% of the total beta diversity (Figure 4-22d). This indicates that variation in diversity within an ecoregion is due species replacement across stations, rather than finding a subset of the species found at the richest station. The High Arctic Tundra ecoregion had no beta diversity as species composition was the same among stations, and beta diversity of the Kola Peninsula Tundra was a result of both turnover and nestedness.

4.4.3.2. Regional Diversity

Species richness of circumpolar macrophytes varied widely among lakes in the sub-Arctic region, ranging from 0 to a maximum of 29 species when mosses and algae were excluded (Figure 4-23). The highest alpha diversity was observed in Fennoscandia and the Faroe Islands, and alpha diversity was significantly lower in Greenland (Figure 4-23). Species richness was highly variable in Fennoscandia, owing in part to the wide variety of stations and ecoregions sampled in that area (Figure 4-23). Beta diversity in these regions was primarily driven by species turnover, indicating that differences among stations were due to the replacement of species.

4.4.3.3. Compositional Patterns

All major taxonomic groups were included in the circumpolar dataset, although there were several lakes without macrophytes or with only aquatic mosses. The most common taxa were *Myriophyllum alterniflorum*, *Potamogeton gramineus*, and *Ranunculus reptans*. Aquatic moss species comprised a higher percentage of total species richness with increasing latitude. Bryophytes (or charophytes) commonly dominate the macrophyte assemblages in high latitude lakes (e.g., Welch and Kalff 1974, Vincent and Hobbie 2000) where macrophyte growth rate is extremely low (e.g., Sand-Jensen et al. 1999). Multivariate analysis of macrophyte assemblages for highly-sampled regions indicated some separation among countries based on species composition (Figure 4-24). In particular, macrophyte species composition in Greenland and Norway differed from stations in Sweden and Finland, which were highly similar (Figure 4-24). Species composition in a number of Greenland stations was distinct from all other countries included in the analysis.

In Arctic lakes, aquatic macrophyte abundance and composition is largely driven by physicochemical conditions including climate, which imposes latitudinal and altitude zonation patterns; local weather that modifies the regional patterns; water clarity, which is largely determined by natural or anthropogenic erosional activities; and nutrients, which are inherently in low supply and increase with human activity (Chambers et al. 2008). Harsh climate and ice conditions restrict the distribution of helophytes (i.e., perennial marsh plants with overwintering buds underwater) in the littoral zone, and the number of submerged vascular plants decreases as they are successively replaced by mosses at northern latitudes. This compositional change is likely due to the superior competitive ability of mosses under low light and temperature conditions (Sand-Jensen et al. 1999). Most of the lakes included in the analysis were in pristine condition, but there was some evidence of nutrient enrichment as indicated by the presence of freely-floating lemnids (*Lemna trisulca)* and ceratophyllids *(Ceratophyllum demersum)*.

4.4.3.4. Temporal Trends

Paleoecological analysis has identified shifts in macrophyte taxonomic composition in response to changing environmental conditions. For example, pollen records from a number of lakes in Greenland show a loss of aquatic angiosperms and their replacement by aquatic mosses (8000 - 1000 BP), associated with lake oligotrophication (i.e., the reduction in salt concentrations, ions and nutrients) (Fredskild 1983, 1992). Similarly, analysis of the depth distribution and abundance of aquatic pollen taxa from six Alaskan lakes indicated increases in macrophyte abundance 14,000-12,000 and 8000 BP, likely due to temperate-linked changes in productivity and/or changes in water depth (Edwards et al. 2000).

4.4.4. Gaps in Knowledge and Monitoring

Extensive data were available for some areas of the Arctic (e.g., Fennoscandia), but data were sparse elsewhere, particularly for Canada, Alaska, and Russia. No data were obtained from Russian lakes, and data for Canada and Alaska were extracted from a small number of published papers. In the case of Canada, this resulted in a moderate number of samples covering a small geographic area. Data from Alaska included a single species list that summarized observations from over 100 lakes that covered a wide geographic area, with no details about the lakes in which macrophyte species were found. Macrophyte monitoring is not part of regular assessments in these countries, thus limiting the spatial scope of available data.

Across the entire circumpolar region, there are very few lakes that are monitored regularly. As a result, time series data are generally not available, and many lake observations are outdated (e.g., 1970s or earlier) with no repeated visits to the same lakes. Such data do not allow for the detection of shifts in macrophyte distribution and may not provide an accurate view of contemporary patterns in diversity.

There may also be inconsistencies in sampling methods and taxonomic identification; this can introduce variability that constrains data comparisons. For example, identification of aquatic mosses and Charophytes is sometimes difficult and may result in errors. Moreover, monitoring may not include the identification or enumeration of aquatic mosses, helophytes, or bryophytes, which may be of particular concern if these groups are dominant in a region. Improvements to the monitoring of macrophytes are necessary across the circumpolar region, and should focus on regular and repeated monitoring of representative lakes with standardized monitoring protocols.

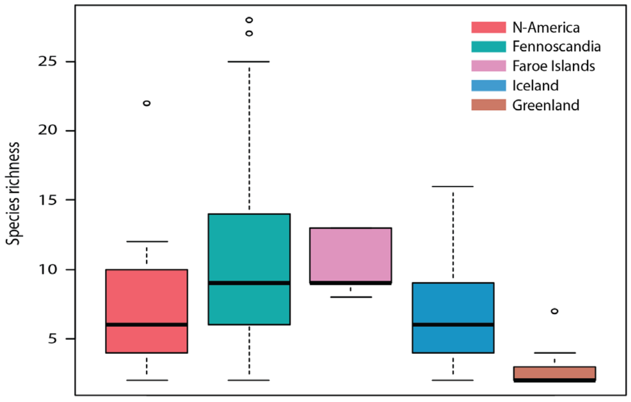


Figure 4-23 Species richness of aquatic macrophytes excluding mosses and algae in five geographic regions of the Arctic.

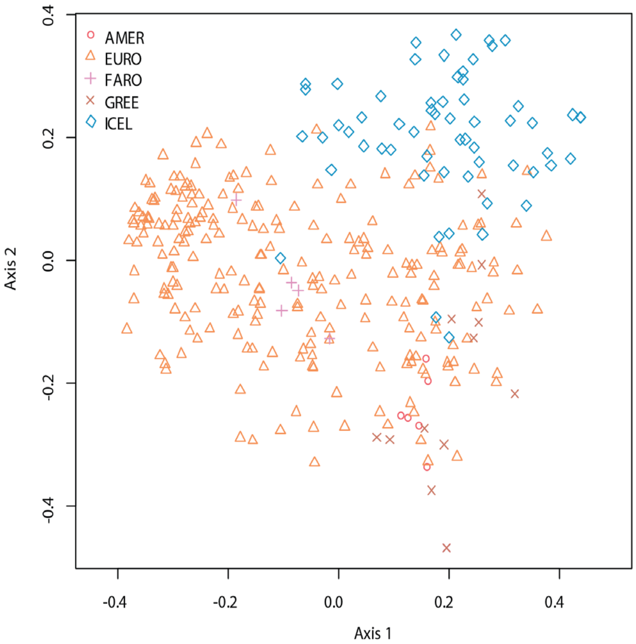


Figure 4-24 Principal coordinates analysis of aquatic macrophytes presence-absence data for North America (AMER), Fennoscandia (EURO), Faroes (FARO), Iceland (ICEL) and Greenland (GREE).

Daphnia longispina.
Photo: Deiter Ebert / Flickr/CC 2.0

4.5. Zooplankton

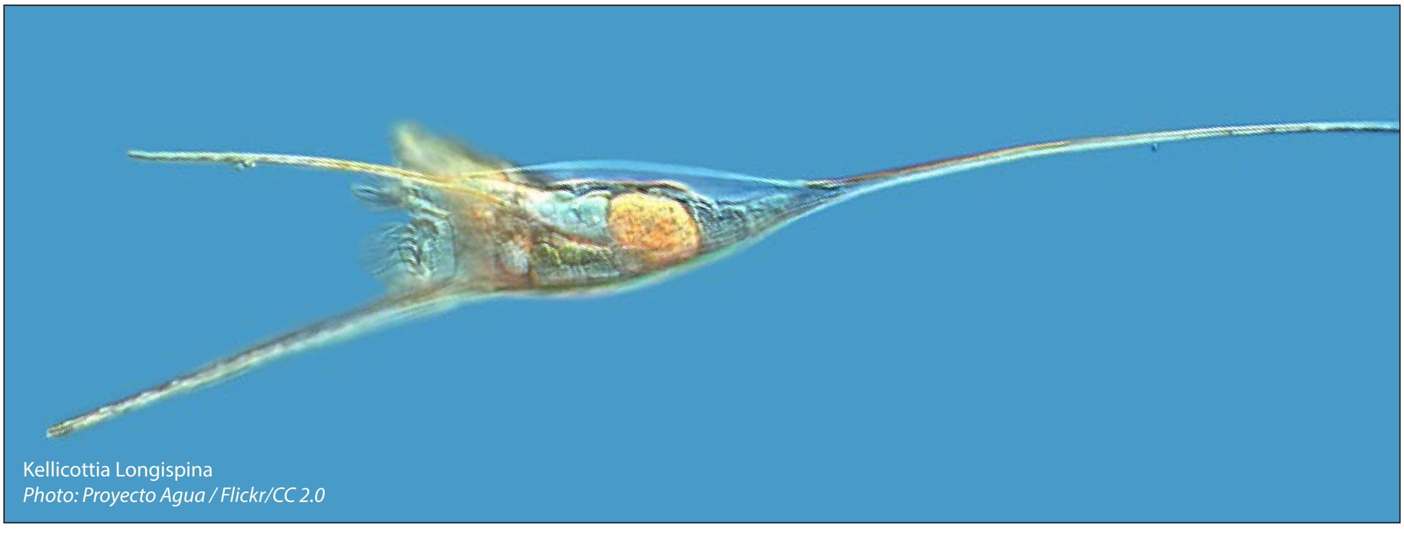
4.5.1. Introduction

Zooplankton are small, heterotrophic invertebrates that live in the water column (pelagic zone) of standing water bodies (i.e., lakes, ponds, pools), although some species have juvenile and resting stages that may occur in benthic habitats. The zooplankton of freshwaters (including the Arctic) are represented by three major groups: rotifers (Rotifera), cladocerans (Cladocera) and copepods (Copepoda). Arctic lakes are dominated by rotifers of the Monogononta class, cladocerans of all families including many genera of Ctenopoda, Anomopoda, Haplopoda and Onychopoda, and copepods of the orders Calanoida and Cyclopoida. In Arctic ponds and small water bodies, cladocerans of the families Chydoridae and Macrothricidae, and copepods from the order Harpacticoida come into account. Zooplankton assemblages are composed of true pelagic taxa, but can also contain benthic species, particularly in small water bodies. In order to obtain a complete picture of biodiversity in these ecosystems and compare them with biodiversity in shallow water bodies, it is necessary to analyze the composition of both aquatic assemblages. However, data for littoral or benthic taxa and stages of zooplankton are generally less available than pelagic data, which may preclude assessment of this component of the zooplankton assemblage.

The distribution of zooplankton species in Arctic lakes differs along a longitudinal gradient, with the greatest species richness corresponding with proximity to areas without recent glaciation (e.g., Alaska, northern Greenland; Wrona et al. 2013). Recolonization of lake zooplankton following the Pleistocene period was from these areas, and thus species richness declines with distance from locations such as Alaska and northern Greenland (Samchyshyna et al. 2008, Rautio et al. 2011). Zooplankton diversity is also increased by proximity to coastal regions, where species richness is increased by freshwater species that originate from marine ecosystems (Rautio et al. 2008).

Zooplankton density, biomass, and community composition are regulated by the abundance of food resources (i.e., planktonic and/or benthic algae and bacteria), predation, and indirectly by the nutrient status of lakes. Despite the low nutrient conditions that are typical of the Arctic, zooplankton density and biomass can be relatively high in shallow Arctic lakes because of the presence of benthic algal mats that are important feeding habitats (e.g., Rautio and Vincent 2006, Mariash et al. 2014). Eutrophication primarily has an indirect effect on zooplankton through increased primary production and changes in food quantity/quality. In low-productivity waters, even small inputs of nutrients may lead to increased production and biomass of zooplankton as well as richness. In contrast, in lakes with higher natural nutrient levels and in lakes with heavy nutrient loads, the main response may be a shift in zooplankton composition. Many zooplankton species are sensitive to predation from macroinvertebrates and fish. In fish-free lakes, zooplankton assemblages include large-bodied species, but the presence of fish will drive the size distribution of zooplankton towards smaller species (O'Brien et al. 2004). Increased fish predation, as a consequence of invasive species or changes in lake trophy, often have strong effects on zooplankton composition, with a shift from larger to smaller cladocerans and copepods and changes in the size distribution of individual species.

Industrial pollution and climate change both have the potential to affect the biodiversity of zooplankton in Arctic lakes. For example, metal and acid contamination in the areas along the boundary between Russia (Kola Peninsula) and Norway (South-Varanger) affects the zooplankton community composition in lakes, leading to reduced species richness, changes in species composition, and reduced complexity of the zooplankton assemblage (Vandysh 2002, Shustova et al. 2009). Lakes on the Kola Peninsula have very simple zooplankton communities with low species numbers (Vandysh 2002), similar to patterns that have been observed in temperate regions of Canada (Yan and Strus 1980). Changes in water temperature influence zooplankton through shifts in the composition of dominant species, changes to phenology of certain species, and introduction of alien species to northern regions (Rautio et al. 2011). Increased water temperature may also affect stratification of deep lakes and cause shifts in the vertical distribution of species in the water column. Other drivers related to climate change, including increased UV, shifts in precipitation, and increased turbidity, have the potential to impact the diversity, biomass, and feeding efficiency of zooplankton.



Kellicottia Longispina
Photo: Proyecto Agua / Flickr/CC 2.0

4.5.2. Objectives and Approach

This assessment provides an evaluation of zooplankton biodiversity and composition across the entire Arctic, which allows for analyses of latitudinal differences and trends across all Arctic regions. At a circumpolar scale, stations were grouped into ecoregions of similar geography and climate (Olson et al. 2001) and alpha diversity, beta diversity, and its component parts (turnover and nestedness) were evaluated within ecoregions. Zooplankton data from 3082 samples representing 482 stations and 421 lakes were used in this report. Of these, 416 lakes included data on crustaceans (Figure 4-25a), and 172 lakes included data on both crustacean zooplankton and rotifers (Figure 4-26a). Thus, diversity was assessed separately for the whole zooplankton assemblage (using stations in which Crustacea and Rotifera were sampled and enumerated) and also for just crustacean zooplankton. Spatial patterns in zooplankton diversity were also assessed at a regional scale (by country and by Arctic region) for the circumpolar Arctic. In the regional assessment, beta diversity was assessed in terms of the dispersion of stations around a group centroid in multivariate space (based on assemblage composition). Using contemporary zooplankton data, we produce a baseline to which future monitoring results can be compared and identify gaps in the current distribution of monitoring.

4.5.3. Overall Patterns and Trends

4.5.3.1. Circumpolar Diversity

For a subset of ecoregions with moderate sample frequency (range 25-72 stations; Figure 4-25a), alpha diversity estimates of crustacean zooplankton were rarefied to 25 stations for comparison. Among these ecoregions, alpha diversity of crustacean zooplankton was highest for lakes in the Northwest Russian-Novaya Zemlya Tundra ecoregion, where on average 29 taxa were found, and the Scandinavian and Russian Taiga ecoregion, where 23 taxa were found (Figure 4-25b). Alpha diversity of crustacean zooplankton was lowest in the Brooks-British Range Tundra in Alaska, where seven taxa were found. For the other four ecoregions, alpha diversity ranged from nine to 16 crustacean taxa, with higher diversity found in coastal Alaska and in Canada than was found in Iceland or Greenland. When all ecoregions were compared at a rarefied alpha diversity level of 10 stations, ecoregions in Russia and Fennoscandia remained the most diverse, with 22 crustacean taxa in the Northwest Russian-Novaya Tundra ecoregion, 19 taxa in the Scandinavian

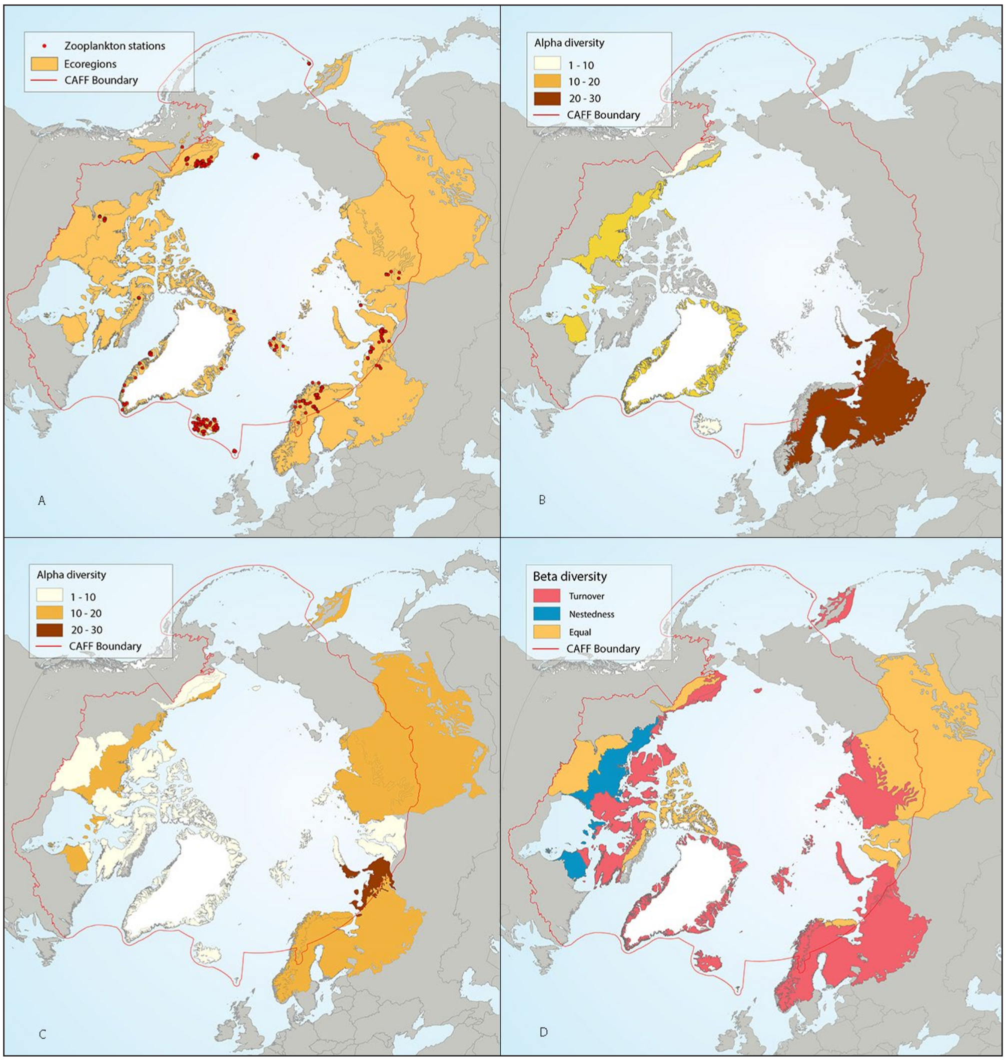


Figure 4-25 Results of circumpolar assessment of lake zooplankton, focused just on crustaceans, and indicating (a) the location of crustacean zooplankton stations, underlain by circumpolar ecoregions; (b) ecoregions with many crustacean zooplankton stations, colored on the basis of alpha diversity rarefied to 25 stations; (c) all ecoregions with crustacean zooplankton stations, colored on the basis of alpha diversity rarefied to 10 stations; (d) ecoregions with at least two stations in a hydrobasin, colored on the basis of the dominant component of beta diversity (species turnover, nestedness, approximately equal contribution, or no diversity) when averaged across hydrobasins in each ecoregion.

Montane Birch Forest and Grasslands, and 17 taxa in the Scandinavian and Russian Taiga (Figure 4-25c). The majority of ecoregions (18 ecoregions from across the circumpolar region) had similar alpha diversity estimates, ranging from six to 13 taxa (Figure 4-25c). The lowest diversity was found in the Canadian high Arctic (High Arctic Tundra), Svalbard (Arctic Desert) and mountainous regions of Alaska (Brooks-British Range Tundra), where fewer than seven crustacean taxa were estimated to be found at 10 stations.

A limited set of stations also had data for rotifers (Figure 4-26a), allowing for a more inclusive assessment of alpha diversity patterns. Four ecoregions had moderate levels of sampling of both Crustacea and rotifers, and alpha diversity estimates were rarefied to 25 stations for comparison. Similar to the crustacean analysis, the Northwest Russian-Novaya Zemlya Tundra ecoregion was found to be most diverse, having on average 102 taxa (95% confidence interval 92-111; Figure 4-26b). The Arctic Coastal Tundra was also significantly more diverse than the other ecoregions, and had an average of 62 taxa (95% confidence interval 55-70). The remaining two ecoregions had similar alpha diversity estimates (23-26 taxa). All ecoregions with data for both crustaceans and rotifers were compared at a rarefied alpha diversity level of 10 stations. Ecoregions covered a wider range of alpha diversity (which spanned four to 86 taxa on average per ecoregion) than when only Crustacea were considered. Russian and Fennoscandian ecoregions remained the most diverse ecoregions, though the Scandinavian Montane Birch Forest and Grasslands had the highest richness, with an estimated 86 crustacean and rotifer taxa at 10 stations (though this was not significantly different from the Northwest Russian-Novaya Zemlya Tundra, which had an estimated 67 taxa; Figure 4-26c). In this analysis, the Arctic Desert had the lowest diversity with an estimated four taxa (Figure 4-26c).

Assessment of the full zooplankton assemblage and of crustacean zooplankton provided some evidence of high alpha diversity in coastal regions, particularly in coastal ecoregions of Fennoscandia and Russia. This pattern is consistent with predictions that high richness would be found in coastal areas where there would be more influence from the marine habitat (Rautio et al. 2008). Richness was also high in Alaska (Arctic Coastal Tundra), which supports the prediction of high diversity in areas that were unaffected by recent glaciation (Samchyshyna et al. 2008). The high alpha diversity of rotifers observed for some Russian ecoregions (particularly the Northwest Russian-Novaya Zemlya Tundra)

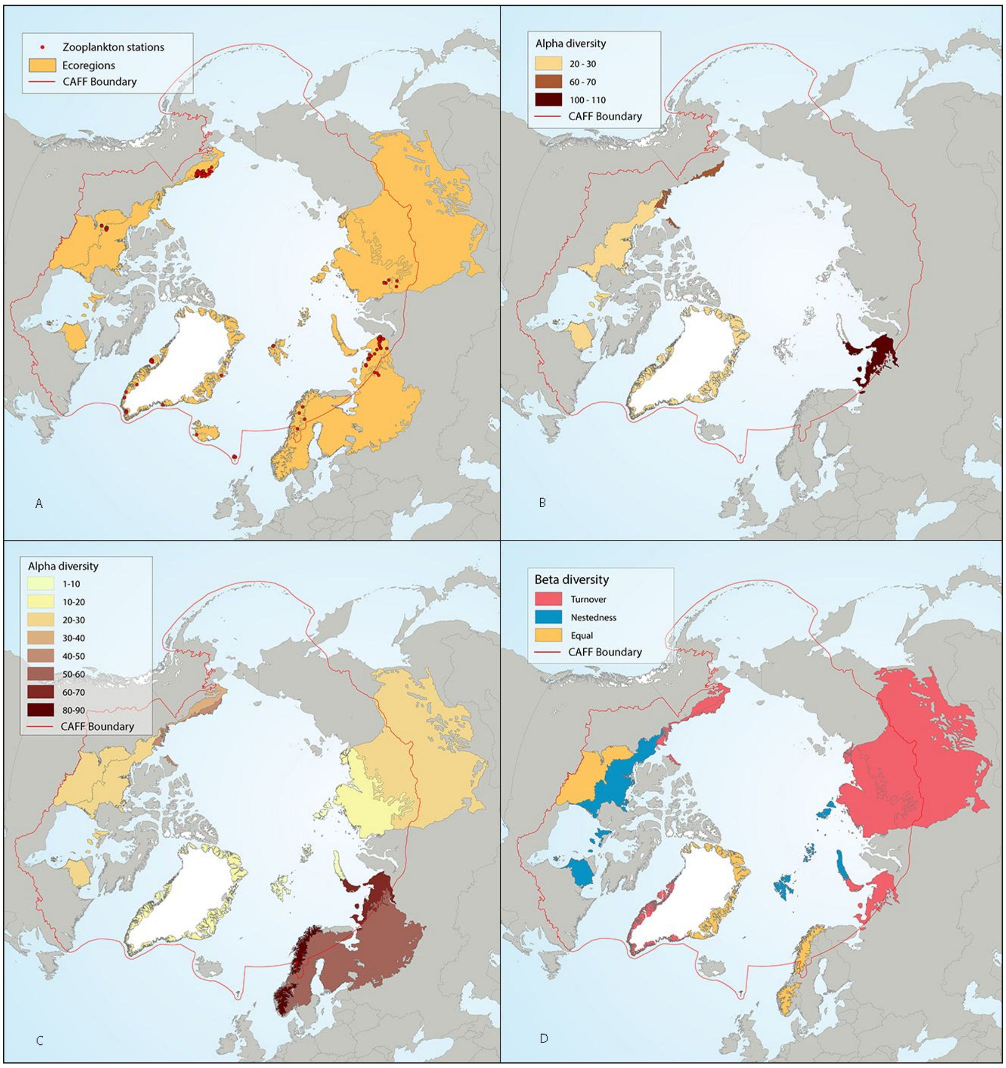
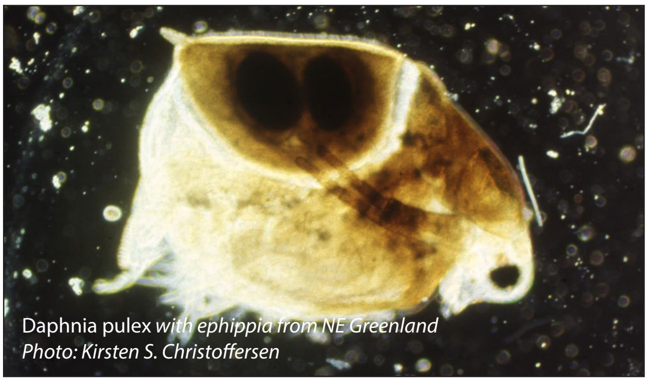


Figure 4-26 Results of circumpolar assessment of lake zooplankton, including crustaceans and rotifers, and indicating (a) the location of zooplankton stations, underlain by circumpolar ecoregions; (b) ecoregions with many zooplankton stations, colored on the basis of alpha diversity rarefied to 25 stations; (c) all ecoregions with zooplankton stations, colored on the basis of alpha diversity rarefied to 10 stations; (d) ecoregions with at least two stations in a hydrobasin, colored on the basis of the dominant component of beta diversity (species turnover, nestedness, approximately equal contribution, or no diversity) when averaged across hydrobasins in each ecoregion.

may partly reflect differences in the taxonomic identification approaches used by researchers in different regions (e.g., where taxa may be split into multiple species in some regions but grouped in other regions). Further consultation with taxonomic experts from different regions may be required to confirm taxonomic groupings and further develop nomenclature harmonization.



Daphnia pulex *with ephippia from NE Greenland*
Photo: Kirsten S. Christoffersen

Beta diversity for crustacean zooplankton, calculated for 21 ecoregions, exceeded 0.80 in three of the ecoregions: the Arctic Coastal Tundra, Arctic Desert, and Kamchatka-Kurile Meadows and Sparse Forests. The high beta diversity values indicated that these ecoregions had the highest differences in crustacean zooplankton assemblage composition among lakes. Eight of the 21 ecoregions had low beta diversity ($\beta_{SOR} \leq 0.50$), suggesting that the lakes in these regions were more similar in their crustacean zooplankton assemblages. The lowest beta diversity was in the High Arctic Tundra ($\beta_{SOR} = 0.20$), whereas seven other ecoregions had beta diversity ranging from 0.43 to 0.5; however, these ecoregions all had data from between four and seven lakes, and low beta diversity values may have reflected the fact that assemblage composition was compared among few lakes. Beta diversity for the remaining ecoregions ranged from 0.51 to 0.77, indicating intermediate similarities among assemblages. Species turnover was the most important component of beta diversity in 13 of the 21 ecoregions analyzed, accounting for 70–100% of beta diversity in these ecoregions (Figure 425d). Of the remaining ecoregions, one was dominated by nestedness (Low Arctic Tundra in Canada; 80% of beta diversity) whereas all other ecoregions had approximately equal contribution of turnover and nestedness, with the turnover component accounting for 38-62% of beta diversity. Where nestedness played a larger role, there were generally fewer lakes sampled in an ecoregion, and differences among lakes may not have been captured. These results highlight the importance of monitoring zooplankton in a wide variety of lakes within an ecoregion, as widespread sampling may be necessary to accurately summarize the full diversity of species in an area.

Beta diversity for samples with both crustacean and rotifer data ranged from 0.28 to 0.87 across 12 ecoregions. For some ecoregions, the addition of rotifers resulted in only minor deviations from the results of the crustacean beta diversity analysis. However, there was a strong increase in beta diversity in the Arctic Foothills Tundra (0.5 to 0.8) and the Northern Canadian Shield Taiga (0.51 to 0.73), which indicated that the inclusion of rotifers led to stronger dissimilarity among stations. In contrast, there was a sharp decline in beta diversity estimates for the Arctic Desert (0.81 to 0.29) and Kalaallit Nunaat High Arctic Tundra (0.71 to 0.43), which indicated that consideration of rotifer taxa led to stronger similarity among stations in these high Arctic ecoregions. Nestedness became more important in some ecoregions and contributed approximately equally to beta diversity (e.g., Kalaallit Nunaat High Arctic Tundra and Scandinavian Montane Birch Forest and Grasslands), or even became the dominant component of beta diversity (e.g., Arctic Desert; Figure 426d). Species turnover remained an important component of beta diversity in ecoregions in Alaska, Russia, and southern Greenland.

4.5.3.2. Regional Diversity

Average taxon richness per site decreased with increasing latitude for crustaceans and rotifers. For rotifers, taxa richness was significantly lower in the high Arctic than in the low Arctic ($p < 0.005$), but there was substantial overlap in richness between the high Arctic and the sub-Arctic/alpine zone. When crustacean richness was assessed, there was evidence of significantly lower alpha diversity in the high Arctic than in either low-Arctic or sub-Arctic/alpine lakes ($p < 0.001$; Figure 4-27). Though these patterns in diversity may have been driven in part by lower sample size in the high Arctic, they are also indicative of lower diversity of both crustaceans and rotifers at higher latitudes.

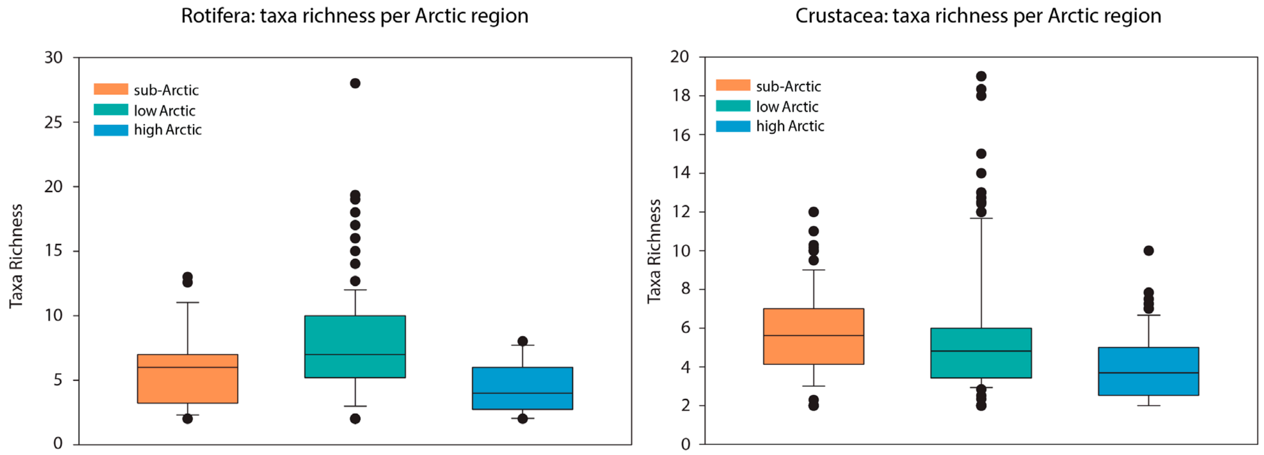


Figure 4-27 Box-plots of taxa richness (average per lake) by Arctic regions for rotifers (left) and crustaceans (right). Crustacean taxa are restricted to taxa within Calanoida, Cyclopoida and Cladocera. Samples with only a single taxon have been excluded. Boxes represent median and interquartile range.

4.5.3.3. Compositional Patterns

The zooplankton dataset included 357 taxa, of which 282 were at the species level. The main groups of zooplankton were calanoid copepods (30 spp.), cyclopoid copepods (37 spp.), cladocerans (66 spp.) and rotifers (121 spp.). Other groups like Harpacticoida, Ostracoda and others were represented with <20 species each. Approximately 40% of all taxa were found in only one or two samples, whereas the most common species groups (*Daphnia longispina* gr., which likely included several species), and *Cyclops scutifer* were found in 44% and 37% of the samples, respectively. Among the rotifers, *Kellicottia longispina* was most widespread, found in 69% of the samples with rotifer data.

Distributional patterns differed among species groups. Many of the common and highly abundant species, i.e., crustaceans (*Bosmina longirostris, Bosmina longispina/ coregoni gr., Daphnia longispina gr.*) and rotifers (*Asplanchna priodonta, Conochilus unicornis, Kelicottia longispina*) are common throughout the Holarctic area. Some species were common, but were not found in all regions. For example, the cladoceran *Daphnia pulex* gr., commonly associated with fishless lakes, was observed in 31% of the stations, but was lacking in Scandinavia. The reason for this may be that Scandinavian lakes are relatively large and deep and contain fish. Additionally, *Holopedium gibberum* was observed in 26% of the circumpolar lakes, but was absent in the eastern part of Russia. Other common species with a more restricted spatial distribution were *Bythotrephes longimanus*, *Limnosida frontosa*, and *Heterocope appendiculata*, which were found in Scandinavia and Eastern and Western Siberia, but not in Canada or parts of Beringia (Alaska), in agreement with the described Eurasian distribution of these species. *Leptodora kindtii* and *Cyclops abyssorum* had a similar distribution in the data, but have been found in North America in the past.

Comparison of the relative abundance of each of the main crustacean groups (Calanoida, Cyclopoida and Cladocera) across Arctic zones indicated that the dominance of cladocerans in sub-Arctic lakes(approximately 50% of all specimens) was diminished by an increased presence of cyclopoid copepods in the low Arctic and high Arctic (Figure 4-28). Conversely, the relative abundance of calanoid copepods was similar between the sub-Arctic and low Arctic, and declined in the high Arctic zone (Figure 4-28). It is known that copepods are more cold-adapted than cladocerans, and are thus potentially able to tolerate high Arctic conditions. Cladocerans have advantages in colonization of arctic areas thanks to a cyclic parthenogenesis that includes lentic stages (epiphia), but they are more sensitive to the ratio of P:N in waterbodies (Novichkova and Azovsky 2017), and may be limited in the nutrient-poor systems of the high Arctic.

4.5.3.4. Gaps in Knowledge and Monitoring

Data for this assessment were lacking particularly from Canada and Russia, but there were several regions of the Arctic where spatial coverage was less extensive than for other FECs. Zooplankton are not generally included in routine monitoring in North America, with the exception of monitoring by industry. The lack of data in some European countries may be due to the fact that zooplankton is not a so-called “ecological quality element” according to the European Water Framework Directive. Greenland and Norway are the primary regions with routine monitoring at established stations, whereas monitoring data from other regions are often from impact studies rather than long-term programs intended to evaluate natural variation or monitor for effects of climate change. Throughout the circumpolar region, therefore, there is a need to rely in part or in whole on data from academia, industry, or other non-government research. As a result, there is limited availability of time series, and in some areas, limited assessment of the full zooplankton assemblage (e.g., areas with research focused on crustaceans or just on cladocerans or copepods).

There were some historical data from Greenland (records from the early 1900s from published papers), but data were generally from within the last 30 years for most countries. Time series were rare, and only nine stations in the database (from Greenland, Norway, and USA) had ≥ 10 years of sampling data. Some paleolimnological cladoceran data were obtained for the database, but these data covered a small spatial scale and additional data would be required to improve assessment of long-term historical changes.

Current assessments are limited by a lack of routine monitoring, which would be necessary to detect changes in response to climate change and anthropogenic stressors. To facilitate the future status assessment for freshwater zooplankton across the circumpolar region, there is need to standardize sampling methods and habitats. For example, data included a variety of samplers (with different mesh sizes) and sampling approaches (depth-specific or depth-integrated sampling), and the depth/region of the lake where samples were collected was not always available. For many of the samples, >50% of the taxa are generally associated with littoral habitats (very few truly pelagic species). We assume that the vast majority of these sites are very shallow, and therefore that most samples represent a mixed habitat (littoral + pelagic). However, future monitoring would preferably include the collection of quantitative samples with vertical net-hauls from open waters and horizontal net-hauls from the littoral zone. Furthermore, identification of the complete sample (crustaceans and rotifers) should be completed using the most recent nomenclature.

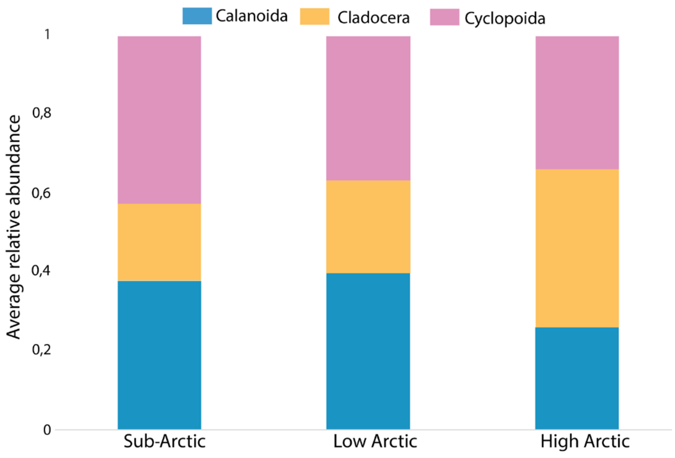


Figure 4-28 Average relative abundance of the main zooplankton groups (calanoid copepods, cyclopoid copepods, cladocerans) for the sub-Arctic (n=150), low-Arctic (n=154), and high-Arctic (n=55) regions. Samples with a single taxon have been excluded.

*The larval stage of non-biting midges (*Chironomidae*) occurs in aquatic environments. Chironomidae are cold-tolerant and are therefore the dominant benthic macroinvertebrate group in many Arctic freshwater systems. Because they are so abundant, they are an important component of aquatic and terrestrial food webs, and provide food sources for fish and other organisms.*
Photo: Jan Hamrsky

4.6. Benthic Macroinvertebrates

4.6.1. Introduction

Benthic macroinvertebrates are a diverse group of animals including insect larvae, crustaceans, worms, molluscs, and mites that are generally visible to the naked eye, and that are typically collected using nets with a mesh size of 0.25-0.5 mm. They live on or close to the bottom substrates of rivers and lakes, i.e., in sediments, on stony substrates and/or rocky shores. Benthic macroinvertebrate monitoring in lakes is generally either of the littoral species that inhabit the shallow waters along the shoreline or the profundal species that live in the soft sediments of the deep, dark parts of lakes. Macroinvertebrates in rivers are commonly collected in stony riffle habitats or in the sandy/clay sediments of pool habitats as well as in association with water plants. Benthic macroinvertebrates feed on water plants, algae, detritus, and other macroinvertebrates, and form the trophic link between the base of the food web (i.e., the primary producers) and predators higher up in the food chain such as invertebrate predators, fish and waterfowl. Many benthic macroinvertebrate taxa have a wide distribution and well-defined ecological niches, which make them good indicators of ecological condition.

The diversity of benthic macroinvertebrate assemblages in the Arctic has not been studied on a circumpolar scale, but regional assessments have suggested that species-specific physiological tolerance to cold temperatures may play a role in determining assemblage structure (Milner et al. 2001, Wrona et al. 2013, Culp et al. In Press). Riverine studies from higher northern latitudes (> 40o N) indicate that alpha diversity of benthic macroinvertebrates decreases with increasing latitude, shifting from high abundances of mayflies (Ephemeroptera), caddisflies (Trichoptera) and stoneflies (Plecoptera) to communities dominated by true flies (Diptera) (e.g., Oswood 1997, Castella et al. 2001, Scott et al. 2011). At lower latitudes, the benthic macroinvertebrate community in rivers typically includes the mayfly families Baetidae and Heptageniidae, stoneflies belonging to the Nemouridae and Chloroperlidae, oligochaete worms, and the water mites or Hydracarina. Further North, cold water temperatures and scarce food resources limit benthic macroinvertebrate survival and growth (Wrona et al. 2013), and adaptations such as freeze-avoidance or freeze-tolerance (Irons III et al. 1993) are necessary for survival (Danks 1992, Danks et al. 1994). The predominant taxa at the highest latitudes are the non-biting midge (family Chironomidae) subfamily Diamesinae, which dominates in glacial streams where maximum temperatures are 0–2°C, while worms (Oligochaeta), crane flies (Tipulidae), and the midge subfamily Orthocladiinae are found in streams with maximum temperatures between 2°C and 4°C (Milner et al. 2001).

Important environmental drivers of benthic macroinvertebrate assemblages in northern rivers include substrate composition, water velocity, nutrients, temperature, catchment geology and catchment vegetation (Lento et al. 2013). Arctic streams and rivers are also highly dynamic systems, with long periods of ice cover followed by peaks in flow during the spring freshet, when melt-water from snow and ice enters the system, leading to high water levels, fast-flowing water, and the potential for flooding (Prowse and Culp 2003, Prowse et al. 2006b, Prowse et al. 2011a). These ecosystems have recurrent high turbidity due to unstable streambeds and high sediment loads. Such physical disturbances are important drivers of macroinvertebrate biodiversity and are especially pronounced during periods of high discharge, such as during ice break-up or in glacial-fed systems, or during periods of anchor ice formation (Milner and Petts 1994, Power and Power 1995). The dynamics of Arctic streams and rivers impose constraints on macroinvertebrate biodiversity and population size, as invertebrates need specific traits to grow and reproduce in these environments (e.g., life history strategies relating to the length of the life cycle and number of generations and life stages, cold tolerance, and feeding habits; Brown et al. 2018).

Arctic lakes have lower invertebrate taxon richness than temperate lakes, but still maintain functionally and taxonomically diverse benthic macroinvertebrate communities (Johnson and Goedkoop 2002, Wrona et al. 2013). Primary production in Arctic lakes is to a large extent from benthic algae, mainly diatoms and cyanobacteria. Due to the high water clarity of Arctic lakes (except glacier-fed lakes), light penetrates and supports benthic primary production even at large depths, thus supplying food for benthic invertebrates. Common taxa in the shallow, highly-productive littoral zone of Arctic lakes are midges (Chironomidae), crane flies (Tipulidae), case-building caddisflies in the family Limnephilidae, mayflies, especially in the family Ameletidae, and stoneflies belonging to the genus Capnia in the family Capniidae. In contrast, the profundal zone of lakes largely consists of homogeneous sediments and is characterized by relatively low diversity. Chironomids, amphipod crustaceans, and oligochaete worms are common taxa in the profundal zone. Fishless lakes frequently contain predation-sensitive macroinvertebrate fauna, such as the crustacean *Lepidurus arcticus* (Branchiopoda, Notostraca) and the beetle *Colymetes*. These organisms may be highly abundant in fishless lakes, but are very rare on larger spatial scales.


Rhantus Suturalis
Photo: Jan Hamrsky

Freshwater biomonitoring of benthic invertebrates in lakes and rivers has a long tradition (Rosenberg and Resh 1993), starting with the early work by Kolkwitz and Marsson (1909) and progressively developing into multiple, pollution-specific indicator taxa that are used to construct biological metrics. These metrics quantify effects on ecosystems by both point-source pollution and large-scale diffusive pollution on macroinvertebrate communities (e.g., Wiederholm 1980, Armitage et al. 1983, Raddum and Fjellheim 1984) by summarizing knowledge of the tolerance range of multiple macroinvertebrate species along environmental stress gradients into a single value. Such metrics (e.g., EPT and tolerance metrics) have been developed for ecological integrity, acidification, and eutrophication and form the core of assessment tools in many countries, but are not adapted to Arctic water bodies. Similarly, as lakes and rivers are closely linked to landscape modifications, macroinvertebrate communities will react to climate-induced changes in landscape-level processes such as changes in temperature, ice-regimes, and vegetation. As biomonitoring using benthic macroinvertebrates is a well-established approach, standard methods exist for the sampling, processing, and analysis of samples that can facilitate large-scale assessments of their diversity.

4.6.2. Objectives and Approach

Benthic macroinvertebrate assemblages were assessed for rivers and for lake littoral and profundal zones. Analysis focused on samples collected using similar approaches (e.g., kick net in lake littorals and rivers, grab samples in profundal zone of lakes). Analysis was conducted at the lowest taxonomic level that allowed maximum inclusion of stations, which was generally at family level or higher as a large number of stations did not identify chironomids to lower taxonomic units (though limited analysis was done at the genus level for chironomids in lake profundal zones). The circumpolar assessment of benthic macroinvertebrates provided a summary of broad spatial patterns of biodiversity in Arctic streams and lakes. Stations were grouped into ecoregions with similar geography and climate (Olson et al. 2001) and rarefaction curves were used to compare alpha diversity across ecoregions at a standardized number of stations (to control for differences in sampling effort). We assessed beta diversity within ecoregions (averaging beta diversity across hydrobasins in each ecoregion), as well as the proportion of beta diversity that was due to turnover or nestedness.


Ephemeroptera (top) and Heptageniidae (bottom)
Photo: Jan Hamrsky

Alpha diversity within hydrobasins was compared with the average latitude of stations within each hydrobasin to evaluate latitudinal trends in diversity. Assemblage structure was also assessed to describe regional and latitudinal shifts in taxonomic composition, contrasting different areas of the circumpolar Arctic. Using this approach and contemporary data, we produce a baseline to which future monitoring results can be compared and identify gaps in the current distribution of benthic macroinvertebrate monitoring.


Baetidae
Photo: Jan Hamrsky

4.6.3 Overall Patterns and Trends

4.6.3.1 Circumpolar Diversity

4.6.3.1.1. Lake Littoral Zones

Lake littoral invertebrate samples were not collected across the entire circumpolar region, but primarily came from Fennoscandia, Iceland, and USA, with a small number of stations in southern Greenland, Faroe Islands, and northern Russia (Kola Peninsula and Wrangel Island; Figure 4-29a). There were four ecoregions in the highly sampled regions with sufficient sampling to allow the assessment of littoral zone alpha diversity rarefied to 80 stations. Among these ecoregions, the lowest alpha diversity was found in the Iceland Boreal Birch Forests and Alpine Tundra, which had an average of 16 taxa in 80 stations (Figure 4-29b). This may have been due in part to the sampling method used in this country, as invertebrate samples were collected using rock scrapes rather than kick nets. Although these methods are broadly comparable, rock scrapes have been shown to collect fewer taxa than kick nets (Lento and Morin 2014). The Arctic Coastal Tundra in Alaska had significantly higher alpha diversity, with an average of 37 taxa in 80 stations (Figure 4-29b). The Fennoscandian ecoregions had the highest taxonomic richness, with an average of 56 taxa in the mountainous Scandinavian Montane Forests and Grasslands and an average of 70 taxa in the Scandinavian and Russian Taiga. Alpha diversity estimates for Fennoscandia were significantly higher than for Iceland or Alaska, suggesting strong regional differences in taxonomic richness across the sampled area.

Alpha diversity was rarefied to 10 stations to allow comparison of all ecoregions in which invertebrate samples were collected. In this analysis, low alpha diversity (9-13 taxa on average in 10 stations) was found for four ecoregions. These ecoregions were all found on remote islands, and included the Faroe Islands Boreal Grasslands, Wrangel Island Arctic Desert in Russia, Kalaallit Nunaat Low Arctic Tundra in Greenland, and the Iceland Boreal Birch Forests and Alpine Tundra (Figure 4-29c). The low diversity in these island ecoregions is indicative of a dispersal effect on taxonomic richness, with barriers to dispersal limiting the number of taxa that can colonize a region. Alpha diversity estimates were higher for the Arctic Coastal Tundra in Alaska (average of 24 taxa), the Kola Peninsula Tundra (average of 24 taxa), the Scandinavian Montane Forests and Grasslands (average of 34 taxa) and the Scandinavian and Russian Taiga (average of 42 taxa), with the latter two ecoregions having significantly higher alpha diversity than the low diversity ecoregions (Figure 4-29c). The similarity in diversity estimates for the most taxonomically-poor ecoregions suggests that barriers to dispersal, such as proximity to mainland and presence of mountains, limits biodiversity in these northern lakes. Even in areas of high biodiversity, such as Fennoscandia and northern Alaska, there was evidence of lower diversity where the presence of mountainous ecoregions likely limited dispersal. Beta diversity within ecoregions ranged from 0.19 to 0.77, indicating a relatively large range in the level of similarity between lakes. The lowest beta diversity (βSOR = 0.19) was in the Faroe Islands Boreal Grasslands, where only four lakes were sampled over a relatively small area, alpha diversity was low, and composition among lakes was extremely similar. Other ecoregions with low to moderate beta diversity included the Kola Peninsula Tundra (βSOR = 0.43) and the Iceland Boreal Birch Forests and Alpine Tundra (βSOR = 0.56). In all three of these ecoregions, nestedness contributed to beta diversity, either as the predominant component (in the case of the Kola Peninsula) or approximately equally to taxonomic turnover (Figure 4-29d). Thus, among-lake differences in composition in these ecoregions were due primarily or in part to a loss of species. In the remaining ecoregions, within which βSOR ranged from 0.65 to 0.78, turnover was generally the dominant component of beta diversity (with the exception of the Kalaallit Nunaat Low Arctic Tundra, where there were equal contributions of turnover and nestedness), indicating that the replacement of taxa across lakes drove differences in composition.

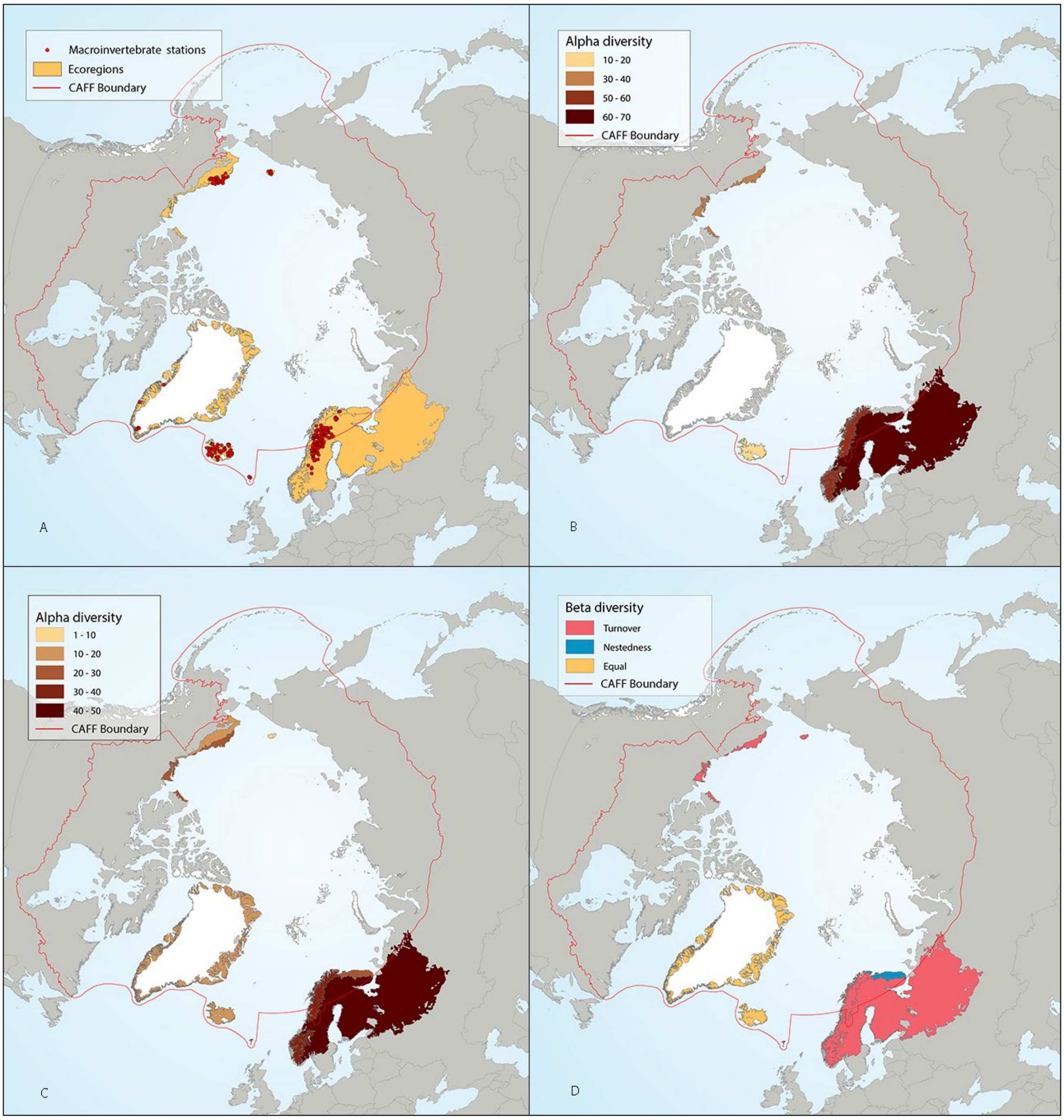


Figure 4-29 Results of circumpolar assessment of lake littoral benthic macroinvertebrates, indicating (a) the location of littoral benthic macroinvertebrate stations, underlain by circumpolar ecoregions; (b) ecoregions with many littoral benthic macroinvertebrate stations, colored on the basis of alpha diversity rarefied to 80 stations; (c) all ecoregions with littoral benthic macroinvertebrate stations, colored on the basis of alpha diversity rarefied to 10 stations; (d) ecoregions with at least two stations in a hydrobasin, colored on the basis of the dominant component of beta diversity (species turnover, nestedness, approximately equal contribution, or no diversity) when averaged across hydrobasins in each ecoregion.

4.6.3.1.2. Lake Profundal Zones

Lake profundal zone samples were obtained for sub-Arctic and low Arctic regions of Canada and Fennoscandia. Profundal samples had predictably lower taxonomic richness for benthic invertebrates than littoral zone samples. Sample richness was also less variable than was found in the littoral zone samples, resulting in smaller confidence intervals around richness estimates. Alpha diversity was rarefied to 20 stations for comparison among ecoregions. The lowest diversity was found in the Low Arctic Tundra (average of 8 taxa in 20 stations) and the Northern Canadian Shield Taiga (average of 9 taxa, significantly higher than the Low Arctic Tundra estimate), both in central Canada. The remaining ecoregions had similar alpha diversity, ranging from 17 to 26 taxa on average in 20 stations. These included the Central Canadian Shield Forests and Southern Hudson Bay Taiga in Central and southern Canada, and the Scandinavian and Russian Taiga and Scandinavian Montane Birch Forest and Grasslands in Fennoscandia.


Simuliidae
Photo: Jan Hamrsky

A number of stations in Canada (sub-, low, and high Arctic), Greenland, and Russia had genus-level data for Chironomidae (midges) in top surface sediments of lake profundal zones (collected using corers or grab samplers). Surface sediment samples were analyzed to compare Chironomidae diversity across ecoregions, with genus-level alpha diversity rarefied to 10 stations in each ecoregion. The lowest alpha diversity was at the highest latitudes, in the High Arctic Tundra in Canada and the Kalaallit Nunaat High Arctic Tundra in Greenland (average of 13 and 14 genera, respectively). Lower latitudes in Canada had significantly higher alpha diversity, with 20 genera in the Low Arctic Tundra, 25 genera in the Middle Arctic Tundra, and 28 genera on average in the Northern Canadian Shield Taiga. The Northwest Russian-Novaya Zemlya Tundra had significantly higher alpha diversity than all other ecoregions, at an average of 64 genera in 10 stations, but this highly elevated estimate may have reflected a different taxonomical approach, with additional splitting of genera relative to the samples from North America and Greenland.

4.6.3.1.3. Rivers

River benthic macroinvertebrate stations had better spatial coverage across the circumpolar region than lake stations, and were particularly prevalent in Canada (Figure 4-30a). There were six ecoregions in Canada and Fennoscandia with a sufficient number of stations to allow for comparison of family richness rarefied to 100 stations. The lowest alpha diversity was in two mountainous ecoregions: the Ogilvie-MacKenzie Alpine Tundra in Canada (average of 46 taxa in 100 stations) and the Scandinavian Montane Birch Forest and Grasslands (average of 56 taxa; Figure 4-30b). In contrast, the highest alpha diversity was in the low-latitude Southern Hudson Bay ecoregion in Canada (average of 76 taxa in 100 stations; Figure 4-30b) and the Scandinavian and Russian Taiga in Fennoscandia (average of 69 taxa in 100 stations; Figure 4-30b); both ecoregions had significantly higher alpha diversity than the two least diverse ecoregions. The Northwest Territories Taiga and Muskwa-Slave Lake Forests ecoregions in Canada had intermediate taxonomic richness, at 60 and 68 taxa, respectively (Figure 4-30b).

A total of 24 ecoregions had river benthic macroinvertebrate stations, and were comparable at a rarefied alpha diversity level of 10 stations. The lowest diversity was in the Arctic Desert ecoregion on Svalbard, with an average of 2 taxa in 10 stations (Figure 4-30c). Low diversity (ranging from 6 to 11 taxa in 10 stations) was also evident in other high Arctic and low Arctic island ecoregions, including the Kalaallit Nunaat High Arctic Tundra and Kalaallit Nunaat Low Arctic Tundra in Greenland, Iceland Boreal Birch Forests and Alpine Tundra, Wrangel Island Arctic Desert in Russia, and High Arctic Tundra in Canada (Figure 4-30c). The highest alpha diversity was evident in sub-Arctic mainland ecoregions, including the Scandinavian and Russian Taiga (average of 47 taxa in 10 stations), and the Muskwa-Slave Lake Forests (44 taxa), Southern Hudson Bay (53 taxa), and Central Canadian Shield Forests (56 taxa) ecoregions in Canada. Across the sampled region, alpha diversity generally was lower at the highest latitudes, on remote islands, and in mountainous ecoregions. In contrast, the highest alpha diversity was evident at the lowest latitudes on the mainland where connectivity allows for greater dispersal of taxa from southern regions. Thus, alpha diversity may reflect a combination of dispersal constraints and thermal tolerances.

Beta diversity for rivers was variable across ecoregions, with average β_{SOR} ranging from 0.21, implying strong similarity among stations, to 0.95, which indicated a large among-site variability in assemblage structure. Beta diversity was highest in the Southern Hudson Bay Taiga ecoregion, which indicated that the largest differences among stations were evident within one of the most diverse ecoregions. Turnover was the predominant component of beta diversity (>70%) in most ecoregions (Figure 4-30d), but there was an increased contribution of nestedness in colder ecoregions and ecoregions potentially affected by dispersal limitations. In particular, both turnover and nestedness contributed approximately equally to beta diversity in Brooks-British Range Tundra and Arctic Foothills Tundra in Alaska, the High Arctic Tundra in Canada, the Kalaallit Nunaat High Arctic Tundra and Low Arctic Tundra in Greenland, and the Wrangel Island Arctic Desert in Russia (Figure 4-30d). Furthermore, beta diversity was completely attributed to nestedness in the Arctic Desert in Svalbard, which was not surprising, as only two taxa were found in this ecoregion. For the remaining ecoregions, the dominance of the turnover component of beta shows that taxon replacement was the main driver of among-river compositional differences.

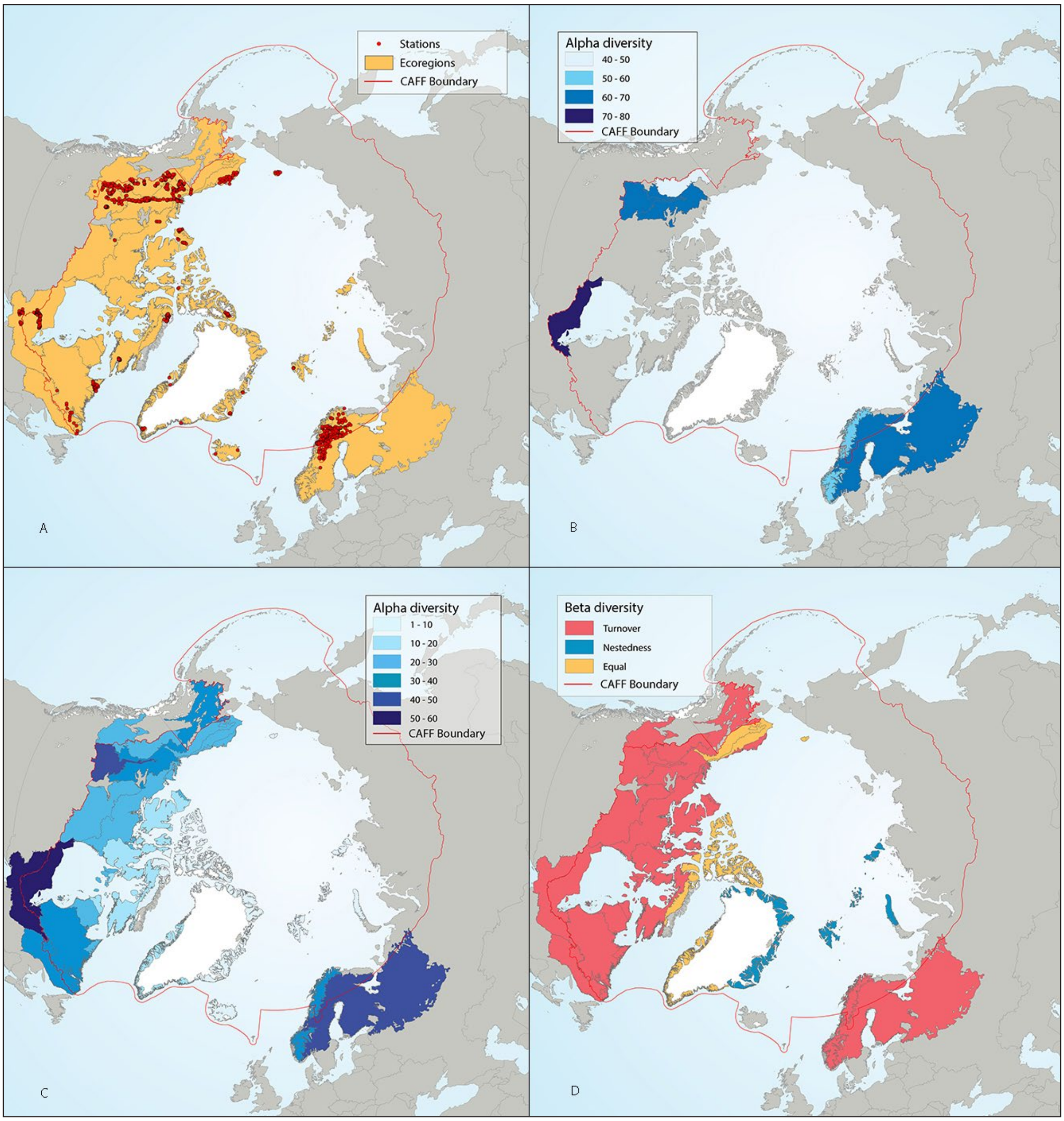


Figure 4-30 Results of circumpolar assessment of river benthic macroinvertebrates, indicating (a) the location of river benthic macroinvertebrate stations, underlain by circumpolar ecoregions; (b) ecoregions with many river benthic macroinvertebrate stations, colored on the basis of alpha diversity rarefied to 100 stations; (c) all ecoregions with river benthic macroinvertebrate stations, colored on the basis of alpha diversity rarefied to 10 stations; (d) ecoregions with at least two stations in a hydrobasin, colored on the basis of the dominant component of beta diversity (species turnover, nestedness, approximately equal contribution, or no diversity) when averaged across hydrobasins in each ecoregion.

Lepidurus Arcticus
Photo: Per Harald Olsen

4.6.3.2. Regional Diversity

The relationship between alpha diversity and latitude was explored to evaluate whether there was evidence of a decline in richness with increasing latitude, as suggested in previous studies (e.g., Scott and Crossman 1973, Castella et al. 2001). Because many ecoregions covered a wide range of latitudes, stations were grouped at a smaller spatial scale into level 5 hydrobasins, and analysis focused only on hydrobasins with at least 4 stations. To ensure comparability of richness estimates across hydrobasins with different levels of sampling, rarefied alpha diversity was compared at the level of 10 stations.


Sphaerium
Photo: Jan Hamrsky

Rarefied taxonomic richness for lake littoral macroinvertebrates showed evidence of a declining trend in alpha diversity above 68°N for samples in Fennoscandia and Alaska (Figure 4-31). Other hydrobasins were located on remote islands (e.g., Iceland, Wrangel Island, Greenland, Faroe Islands) and rarefied alpha diversity in these hydrobasins was lower than those in Fennoscandia and Alaska by approximately 10 or more taxa, regardless of latitude. The low diversity of island ecoregions across all latitudes provided strong evidence for an island biogeography effect on BMI diversity in lakes. For example, Iceland has limited EPT taxa due to dispersal constraints for these taxa. In island hydrobasins, the effect of dispersal constraints on BMI diversity appeared to be stronger than latitudinal constraints, as diversity was similar across all latitudes for these hydrobasins. In contrast, in mainland (e.g., higher connectivity) hydrobasins where dispersal was less limited, a decline in diversity with increasing latitude was the predominant trend, likely related to thermal tolerances.

The river data assessment showed stronger evidence of a latitudinal decline in alpha diversity of benthic invertebrates (Figure 4-32). River data covered a wider range of latitudes (from 49°N to 83°N) and revealed clear evidence of higher taxonomic richness at the lowest latitudes and a strong decline in taxonomic richness above 68°N (Figure 43-2). However, these data also covered a wider range of longitudes, and there was evidence that the strength of the latitudinal decline in diversity differed by longitude/region, related in part to longitudinal temperature gradients across the Arctic. For example, a west-east temperature gradient exists in North America, with more historical warming in the west than along the eastern Canadian Arctic coast, and colder temperatures in the east at similar latitudes. Eastern Canadian hydrobasins clearly showed a stronger decline in diversity that began at lower latitudes than in other regions of the Arctic, and generally had lower diversity than western Canada or USA/western Canada hydrobasins at similar latitudes (Figure 4-32). Furthermore, the eastern Canadian Arctic is colder than Fennoscandia at similar latitudes. Within the mid-latitudes, western North American stations and Fennoscandia stations had higher average alpha diversity than eastern Canadian stations, consistent with patterns expected to occur with warmer temperatures. The lowest alpha diversity values in the mid-latitudes were attributed to the Kalaallit Nunaat Low Arctic Tundra in Greenland (average richness of 12 taxa at 10 stations and average latitude 61°N) and a hydrobasin in the Middle Arctic ecoregion on southern Baffin Island in eastern Canada (average richness of 16 taxa at 10 stations and average latitude of 63.8°N). Both areas (southern Greenland and southern Baffin Island) have experienced less warming since 1990 than other areas of the Arctic (NASA GISS).

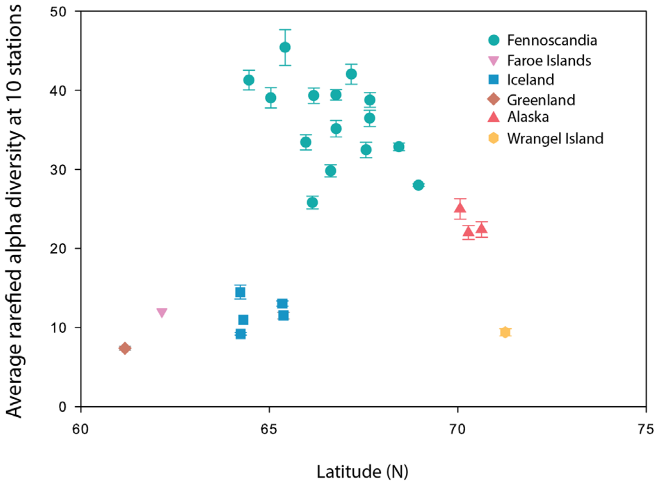


Figure 4-31 Alpha diversity (rarefied to 10 stations, with error bars indicating standard error) of littoral lake benthic macroinvertebrates plotted as a function of the average latitude of stations in each hydrobasin. Hydrobasins are coloured by country/region.

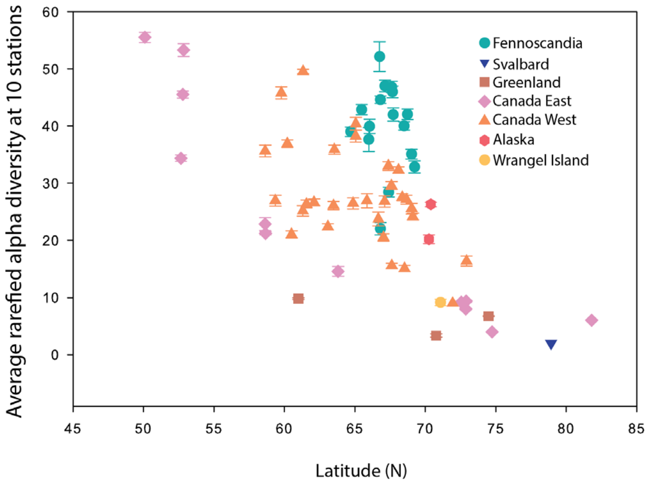


Figure 4-32 Alpha diversity (rarefied to 10 stations, with error bars indicating standard error) of river benthic macroinvertebrates plotted as a function of the average latitude of stations in each hydrobasin. Hydrobasins are coloured based on country/region

4.6.3.3. Compositional Patterns

The most abundant taxa were compared spatially across highly-sampled areas of the Arctic to identify similarities and differences in composition. Geographic areas for comparison were selected by broadly grouping stations in highly-sampled areas by locale (see Figure 4-33 and Figure 4-34 for locations chosen for lake and river BMI, respectively). Data were summarized by selecting the most abundant taxonomic families in each area, comprising a total of 85% of the organisms found in the area. To account for regional differences at the family level, data were summarized by order level or higher, providing a broad picture of composition across geographic areas. Lake littoral samples were generally numerically dominated by Dipteran taxa (true flies, primarily chironomids) and oligochaete worms in all Arctic areas (Figure 4-33). The numerical abundance of Diptera and Oligochaeta was strong enough in Greenland that these were the only two groups that contributed to the dominant portion of the assemblage. Ephemeroptera (mayflies) were not generally abundant in littoral samples, but Trichoptera (caddisflies) were among the important taxa in Alaska and Plecoptera (stoneflies) were abundant in Fennoscandia. Alaska and Fennoscandia also differed with respect to non-insects, as nematode worms were important in Alaska whereas isopods were among the abundant taxa in Fennoscandia. However, differences with respect to

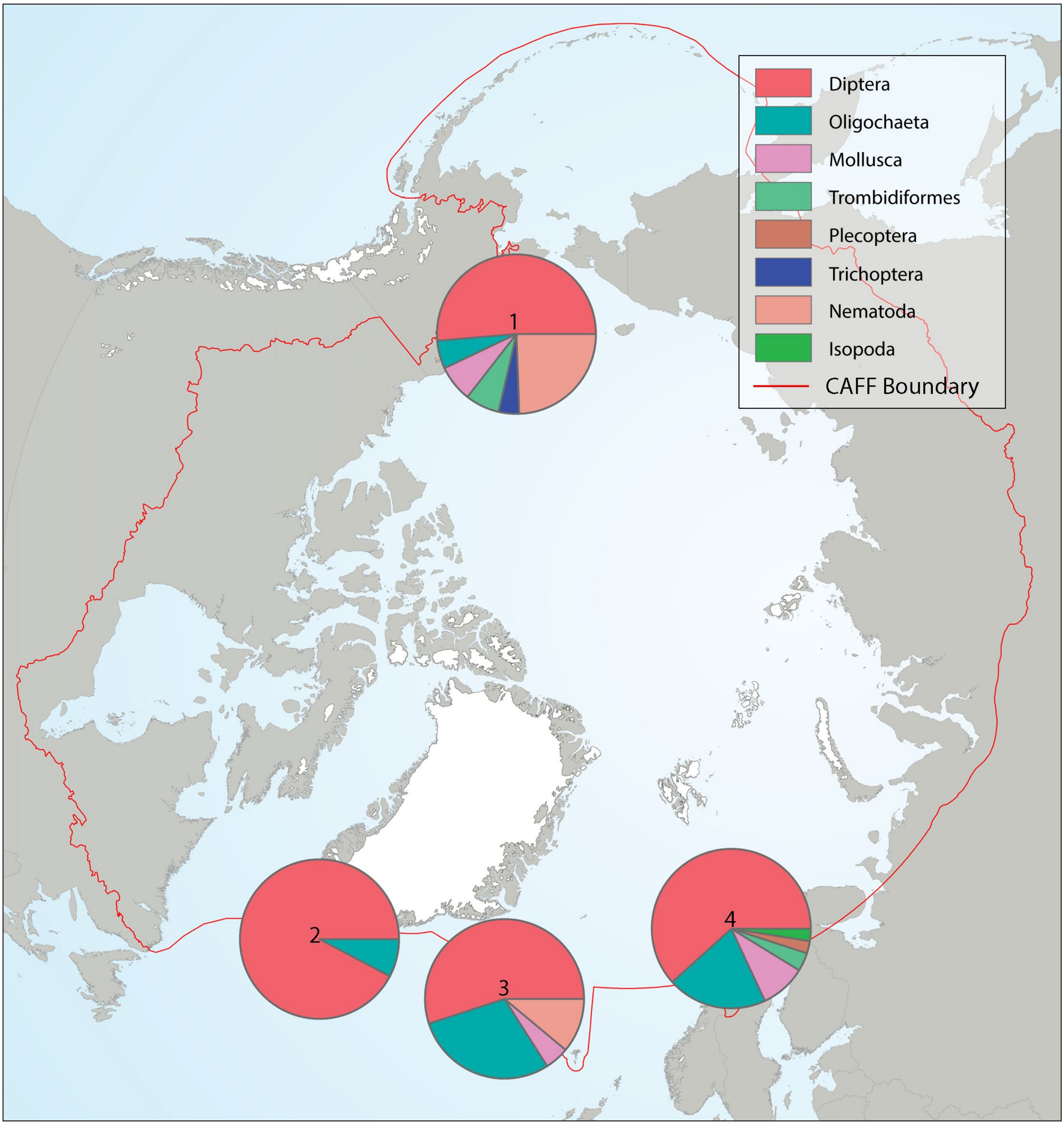


Figure 4-33 Summary of the taxa accounting for 85% of the lake littoral benthic macroinvertebrates collected in each of several highly-sampled geographic areas, with taxa grouped by order level or higher in pie charts placed spatially to indicate sampling area. Pie charts correspond to (1) Alaska, (2) Greenland low Arctic, (3) Iceland, and (4) Fennoscandia.

nematode abundance may have reflected differences in sample sorting, as nematodes are often not counted in lake littoral samples of Fennoscandia.

River benthic invertebrate assemblages were compared across more areas of the Arctic, and showed large differences in composition and dominance, both latitudinally and longitudinally. Diptera alone made up 85% of the assemblage in the high Arctic islands (Ellesmere Island in Canada, and Svalbard), as well as in Iceland (Figure 4-34). The high Arctic in Greenland was dominated by Diptera, but oligochaetes were also abundant in these systems, and composition of the most abundant groups was extremely similar between the low and high Arctic regions of Greenland. At lower latitudes, other groups contributed more to assemblage composition. Alaska and northern Baffin island in eastern Canada had similar composition of oligochaetes and nematode worms, but Alaska also had high abundance of mollusks whereas Ephemeroptera were more common on Baffin Island. Ephemeroptera were highly abundant in several areas of eastern and southern Canada (Baffin Island, northern Labrador, and south of Hudson Bay), but were not abundant in other areas of the Arctic. In contrast, Plecoptera and Trichoptera were far more abundant in Fennoscandia, western Canada, and south of Hudson Bay than they were in eastern Canada. Overall, Fennoscandia had the largest contribution from non-Dipteran organisms.

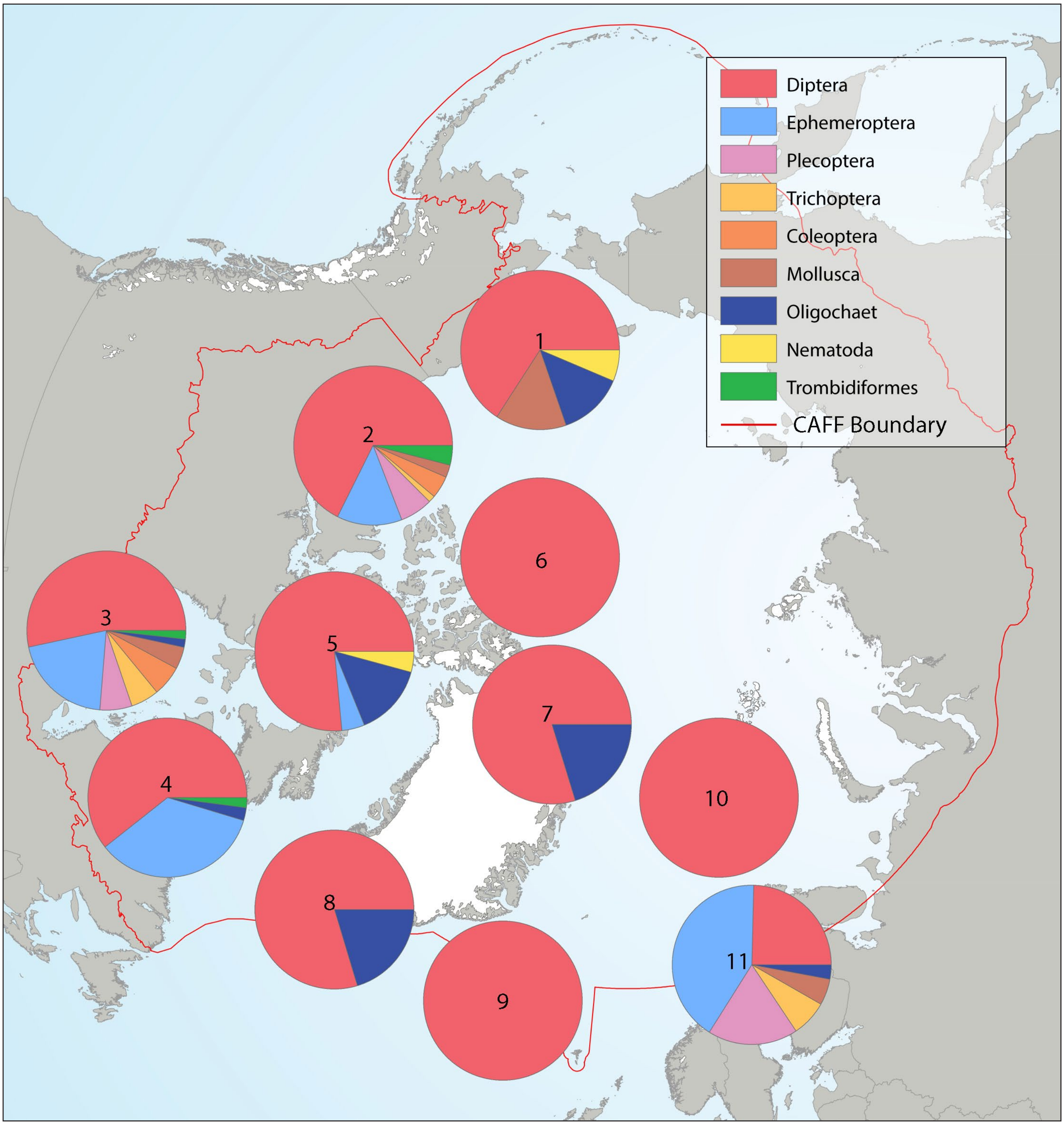


Figure 4-34 Summary of the taxa accounting for 85% of the river benthic macroinvertebrates collected in each of several highly-sampled geographic areas, with taxa grouped by order level or higher in pie charts placed spatially to indicate sampling area. Pie charts correspond to (1) Alaska, (2) western Canada, (3) southern Canada, south of Hudson Bay, (4) northern Labrador, (5) Baffin Island, (6) Ellesmere Island, (7) Greenland high Arctic, (8) Greenland low Arctic, (9) Iceland, (10) Svalbard, and (11) Fennoscandia.

4.6.3.4. Temporal Trends in Lakes

Few long-term records of benthic macroinvertebrates exist from biological monitoring in Arctic lakes. In Lake Abiskojaure (68°N) and Lake Stor-Tjulträsk (66°N) in Sweden, the stony littoral zones (1 m depth) have been monitored annually since 1988 (Figure 4-35). Taxonomic richness (alpha diversity) of littoral macroinvertebrates shows a high inter-annual variability for both lakes. This is mainly due to the low densities of many taxa, i.e., many taxa occur only with a single or few individuals in a sample. While there is no obvious trend in Lake Abiskojaure, richness in the more southern Lake Stor-Tjulträsk has been increasing significantly (Mann-Kendall trend test, $p < 0.001$) since the 1990s. Taxonomic richness has been calculated using a standardized list of taxa. Note, however, that two major groups of benthic macroinvertebrates, the Chironomidae (midges) and Oligochaeta (worms) have not been identified to species. The plots in Figure 4-35 therefore show underestimates of true alpha diversity.

4.6.4. Gaps in Knowledge and Monitoring

There were large gaps in the spatial coverage of lake benthic invertebrate data, due in part to a lack of routine monitoring in many areas. However, a larger issue with regards to lake monitoring is that different countries and researchers focus on a different lake habitats, leading to a heterogeneous dataset that does not allow for broad comparisons. For example, Fennoscandia, Iceland, Alaska, and a few stations in Greenland have data from the littoral zone of lakes, with data collected using kick nets or rock scrapes (broadly comparable with respect to taxonomic composition, though rock scrapes have been shown to collect fewer taxa; Lento and Morin 2014). In contrast, Canada, Russia, and most of Greenland have data from the profundal zone, with grab samplers or dredges used to collect benthic macroinvertebrates. Data from the two quite different habitats and sampler types are not comparable, as profundal samples collected by grabs or dredges are dominated by organisms that prefer to burrow in soft sediments, such as midges and worms. Lake littoral samples, collected using a kick net (i.e., D-net) or rock and sediment scrapes, include more taxa that live on rocky substrates, such as mayflies, stoneflies, and caddisflies. Profundal samples are naturally less diverse than littoral samples, precluding broad comparisons across the circumpolar region. Moreover, in many areas (such as the Canadian high Arctic), only top sediment samples or sediment cores are collected for paleolimnological samples (or training datasets) using chironomids. These samples, although spatially extensive, offer a further limitation on circumpolar assessments, particularly when lake littoral and profundal samples do not identify past the family level for Chironomidae. In order to support future assessment efforts, there should be a standard sampling approach for lake benthic macroinvertebrate assemblages, ideally focused on the more taxonomically-rich littoral zone.

Spatial coverage of river benthic macroinvertebrate data was strong across the circumpolar region, though data were generally lacking for Russia. Despite this gap, river benthic macroinvertebrates had one of the best spatial extents of all the FECs, due in large part to the prevalence of this group in monitoring programs, the relatively standardized sampling approaches used across the circumpolar region (generally kick nets or similar, with comparable mesh size in most regions), and the fact that it is the only FEC to have a national database in Canada, with data contributed to the database from academia and government. As a result, most ecoregions in Canada were represented by river benthic macroinvertebrate samples, allowing for strong spatial assessments. However, despite the strong spatial coverage, much of the sampling that has occurred (particularly in Canada) has been single-event sampling, and thus time series are scarce. An additional limitation to the strength and scope of diversity assessment is inconsistency in the taxonomic level of identification across the Arctic. In Alaska, Finland, and some stations in Canada, Chironomidae were not identified past family level (or were not counted at all for some samples in Finland). Chironomidae are ubiquitous in the Arctic and make up a large proportion of the benthic macroinvertebrate assemblage, and identification at even the subfamily level can help identify more subtle shifts in composition across broad regions (e.g., Culp et al. In Press). Future assessments can continue to make use of the strong spatial coverage of data and accessibility of data from national databases, but monitoring activities should endeavour to include a focus on Chironomidae and include re-sampling of stations to establish time series if possible.

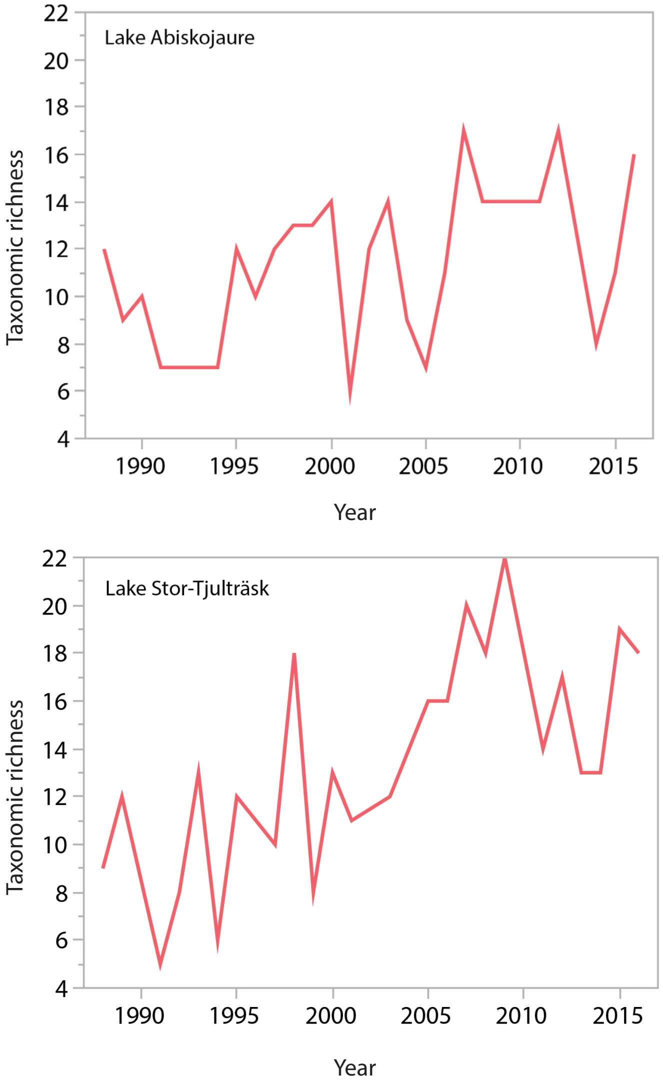


Figure 4-35 30-year trends in alpha diversity of benthic macroinvertebrates in the stony littoral zones (1 m depth) of two Scandinavian Arctic/alpine lakes: Lake Abiskojaure (upper panel) and Lake Stor-Tjulträsk (lower panel).

Arctic Charr
Photo: Dan Bach Kristensen /Shutterstock.com

4.7. Fish

4.7.1. Introduction

Fishes are often the top consumer in freshwater habitats, feeding on an array of prey including zooplankton, macroinvertebrates, and other fish. Over 100 species of fish, from 18 families, are reported from freshwaters throughout the circum-Arctic region (Wrona et al. 2013). Five families account for most of the freshwater taxonomic diversity and include Salmonidae (salmon, trout, and whitefish), Cottidae (sculpins), Cyprinidae (carps and minnows), Percidae (perch), and Petromyzontidae (lamprey) (Wrona et al. 2013). Much of the diversity in Arctic fish species occurs within one family, the Salmonidae. Within the Salmonidae family, there is a high degree of phenotypic and ecological diversity throughout the circumpolar Arctic (Klemetsen 2010), and those species that are closely related are often considered as a species complex that includes phenotypic, systematic, and taxonomic variation. Furthermore, fish of the same species may occur sympatrically (e.g., in the same lake), but occupy different niches -- and the use of different resources is often accompanied by differentiation of physical or morphometric characteristics (Knudsen et al. 2007, Siwertsson et al. 2010). For example, the Arctic charr (*Salvelinus alpinus*) species complex, including the North American Dolly Varden (*S. malma*) and at least 10 other Siberian charrs (Wrona et al. 2013), is widely spread across the Arctic and exhibits a broad range of behavioral and morphological variation (Reist et al. 2013). Within a species, charr exhibit a range of biodiversity characteristics including variation in life history (see Box 4-1), trophic status, and size structure, resulting in a highly adaptable taxonomic group (Snorrason et al. 1994, Wrona et al. 2013). In fact, diadromous Arctic charr is the only fish species present through much of the extreme high Arctic (~75° N latitude; Wrona et al. 2013).

At large spatial scales (e.g., continents or regions), historical and modern features of climate, geology, and hydrology influence species distributions. Dispersal barriers, landscape differences, and climate events such as glaciations, sea level rise and fall, and flooding regimes have influenced present day patterns of species richness and distribution (Tonn 1990, Matthews 1998, Hugueny et al. 2010). For example, the North Atlantic region is species-poor as it was colonized only by anadromous species, whereas Beringia and northwest Canada are relatively specious because of high variation in space and time of glaciation events that allowed for multiple refugia from glaciation and recolonization pathways (Mims et al. 2010, Wrona et al. 2013). Within northern regions, high variability in environmental conditions across the landscape can support equally high variation in the richness of species at smaller spatial scales (Tonn 1990). Climate change will likely alter the geographic range of species – affecting both local and regional biodiversity – through the northward expansion of southern "non-native" species and the contraction or loss of "native" Arctic species (Wrona et al. 2006b, Culp et al. 2012b, Hayden et al. 2017).

At local scales, a number of environmental drivers affect fish diversity through direct impacts on distribution, growth, and phenology (Culp et al. 2012a, Culp et al. 2012b). Water temperature can have significant impacts on the timing of ontogeny and incubation, on growth and energy requirements of all life stages, and may affect the timing of migrations in diadromous fishes. Connectivity is a strong driver of fish distributions and community composition and is affected by water availability and human development (Schindler and Smol 2006, Dias et al. 2014, Laske et al. 2016). Nutrient concentrations can also affect body size and production of fish (Hayden et al. 2017). While these examples illustrate local-scale influences of environmental shifts on fish diversity, interacting effects across local and regional spatial scales complicate the prediction of species shifts with changing environmental drivers (Box 4-2).

Box 4-1. Fish Life History

At high latitudes, fish life history strategies predominantly promote survival and reproduction in environments that are suboptimal or resource-limited (North America, Mims et al. 2010). Many Arctic fish species are large-bodied, long-lived, late-maturing, highly fecund, and often migratory (e.g., Atlantic salmon; Niemelä et al. 2006, Erkinaro et al. 2018). Migration is common among many species of Arctic fishes. Migrations can either be fully within freshwater habitats (e.g., Arctic Grayling; West et al. 1992, Heim et al. 2016), or between fresh- and saltwater habitats (i.e., diadromy; Gross et al. 1988). Strictly, anadromous fish migrate from sea to freshwater to reproduce, while catadromous fish migrate from freshwater to reproduce in the sea (Mecklenburg et al. 2002). Among the Arctic fishes that migrate between fresh- and saltwater, 39 species (families: Acipenseridae [sturgeons], Gasterosteidae [sticklebacks], Osmeridae [smelts], Petromyzontidae [lampreys], and Salmonidae [salmon, trout, and whitefish]) are anadromous, and two are catadromous (family Anguillidae [eels]; Wrona et al. 2013).

In northern climates, anadromy is the dominant migratory strategy because it allows fishes to exploit the highly productive marine environment, thereby increasing growth rates and reproductive advantages (Gross et al. 1988, Wrona et al. 2013). Anadromous migratory behavior may be reduced at the lower latitudes where feeding and rearing resources are more available to fish in freshwater (Reist et al. 2006). At higher latitudes, anadromy and its benefits to Arctic charr (the only fish species in the extreme high Arctic) decrease because of limited access to marine habitats (Svenning and Gullestad 2002).

4.7.2. Objectives and Approach

The circumpolar fish assessment provides a summary of broad spatial patterns of species diversity in Arctic rivers/ streams and lakes, including those in sub, low, and high Arctic regions. Fish diversity was evaluated across continental (104 – 107 km2) and regional scales (103 - 105 km2). Specifically, we assessed fish diversity by examining spatial distribution patterns of species composition, alpha diversity, and beta diversity and its component parts (turnover and nestedness). Data were assessed for the circumpolar region by evaluating alpha diversity (species richness) at the ecoregion scale. Beta diversity and its components were calculated within ecoregions for this circumpolar assessment. Regional-scale assessments were completed for highly-sampled ecoregions in Alaska, Iceland, and Fennoscandia, where data were aggregated at the hydrobasin scale to compare diversity patterns at more standardized spatial scales. Diversity measures for this regional assessment included ecoregion-scale gamma diversity (species richness), basin-scale alpha diversity (species richness), and basin-scale beta diversity (species compositional differences). At the regional scale, hydrobasins were divided across ecoregions of similar geography and climate (Olson et al. 2001). Only stations with data for the entire fish assemblage were included in analyses of biodiversity (Figure 4-36a). We identified gaps in the current distribution of fish monitoring, providing a baseline of information for comparison with future monitoring.

4.7.3. Overall Patterns and Trends

Freshwater fish biodiversity was assessed using fish presence data from 3148 stations spanning c. 240° longitude and 23° latitude and 25 ecoregions (Figure 4-36a). Two-thirds (n = 2116) of the stations occurred in lotic systems, with the remainder in lentic systems (n = 1058). In several ecoregions and countries the provided data were robust - having high numbers of both lotic and lentic sample stations. For example, in Fennoscandia (Norway, Sweden, and Finland) 1807 stations were available for analysis, including 1585 lotic stations and 237 lentic stations. Overall, only six ecoregions (24 %) were represented by more than 100 stations, and 12 ecoregions (48 %) were represented by more than 10 stations. The remaining 13 ecoregions were represented by fewer than 10 stations, and often stations of only one type - lentic or lotic. Poor data coverage among the circumpolar ecoregions results from lack of sampling, sampling focused only on particular species, or lack of data contribution to the CBMP freshwater database. Within the 25 ecoregions included in this assessment, 100 fish species of 52 genera and 15 families are known to occur (Figure 4-36b). Sixty-five of the species are in the two most specious families: the Salmonidae, with 34 species from 8 genera, and the Cyprinidae, with 31 species from 21 different genera (Appendix A). The remaining 35 species are from 13 families of 23 genera.

4.7.3.1. Circumpolar Diversity

Large-scale alpha diversity (i.e., species richness within ecoregions) varied among 25 ecoregions, ranging from a single fish species in the Arctic Desert (Arctic charr) to as many as 47 species in the Scandinavian and Russian taiga (Figure 4-36b). Fish alpha diversity varied across continents; northern ecoregions or mountainous ecoregions had lower

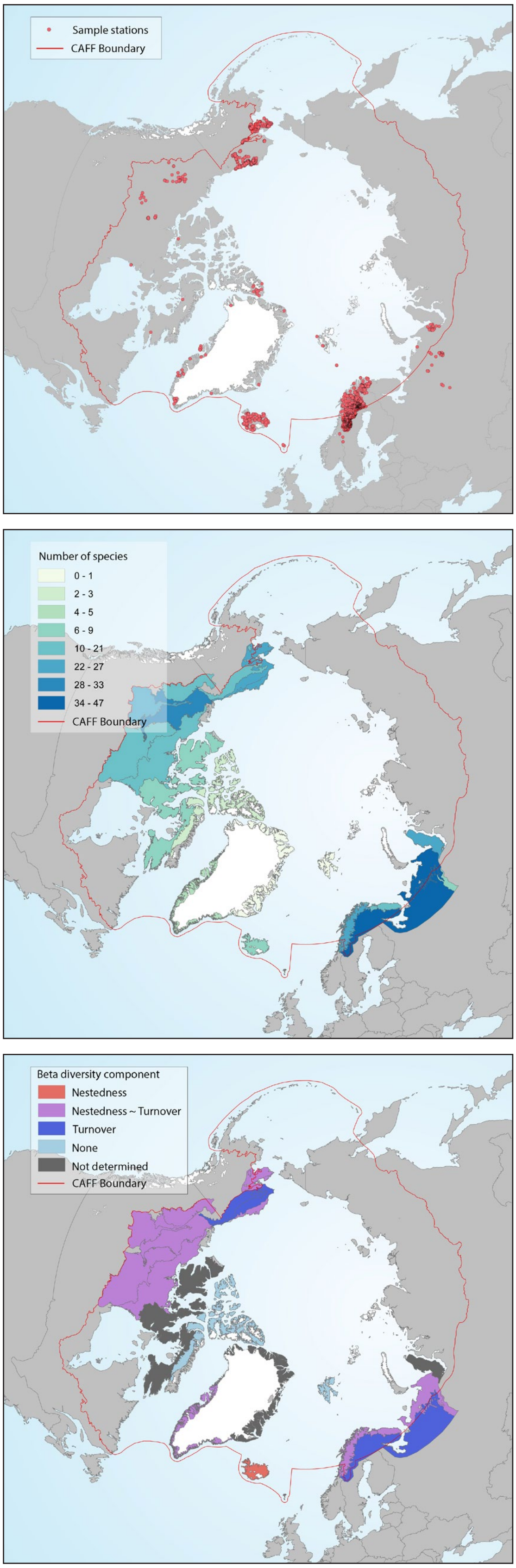


Figure 4 -36 Freshwater fish sampling stations (A), ecoregion alpha diversity in each of the sampled ecoregions, as quantified by estimates of species richness from reference texts (Muus and Dahlstrøm 1971, Scott and Crossman 1973, Mecklenburg et al. 2002) and expert knowledge (academic and government scientists and traditional knowledge) (B), and ecoregion beta diversity (C) characterized according to components of beta diversity as either nestedness, turnover, no diversity (none, beta = 0), or similar nestedness and turnover (nestedness ~ turnover) in the circumpolar Arctic. Ecoregions are shown only where sampling stations occur. Fish sampling stations included in this study assessed complete fish assemblages at each location.

numbers of fish species. As expected, Islands (e.g., Iceland - 9 species, Greenland - 5 species) also had fewer fish species (Figure 4-36b), representing only one-fourth of all freshwater families when compared to continental ecoregions. Based on a standard sample size (n = 10 stations), alpha diversity of fish varied across ecoregions, ranging from a single species in the high Arctic zones (e.g., Arctic Desert and High Arctic Tundra) to between 2 and 13 species in the low Arctic zones (e.g., Interior Yukon-Alaska Alpine Tundra) and subarctic (e.g., Northwest Territories Taiga) ecoregions of North America. In the high Arctic (above 75 °N) ecoregions with low richness are both isolated from continents by salt water (e.g., islands such as Svalbard and Ellesmere) and are extremely cold, making it difficult for freshwater species to access and persist in these areas. The relatively high alpha diversity, at 7 to 15 species, in Fennoscandia and western Russia (e.g., Scandinavian and Russian Taiga, Figure 4-36c) is likely due to the combination of history (e.g., glaciations), fish access via streams that run north, and large spatial extent (Reist et al. 2006, Wrona et al. 2013, Stein et al. 2014). Estimates of richness in all ecoregions were generally reduced from those known from academic and government researchers, Traditional Knowledge, and literature (Figure 4-36b, Figure 4-37). For example, alpha diversity at latitudes above 72°N declined to a single species, Arctic charr, based on observations at 36 stations in 17 hydrobasins across four ecoregions. However, eight additional species are known to occur in the Middle Arctic Tundra and one additional species in the High Arctic Tundra of the Archipelago (Figure 4-36B; Scott and Crossman 1973). In addition, TK records can provide additional information about observations of fish species diversity outside of the ecoregions for which monitoring and research data were obtained, such as northern Quebec (Nunavik) and Labrador in Canada, as well as Russia (Figure 4-37).

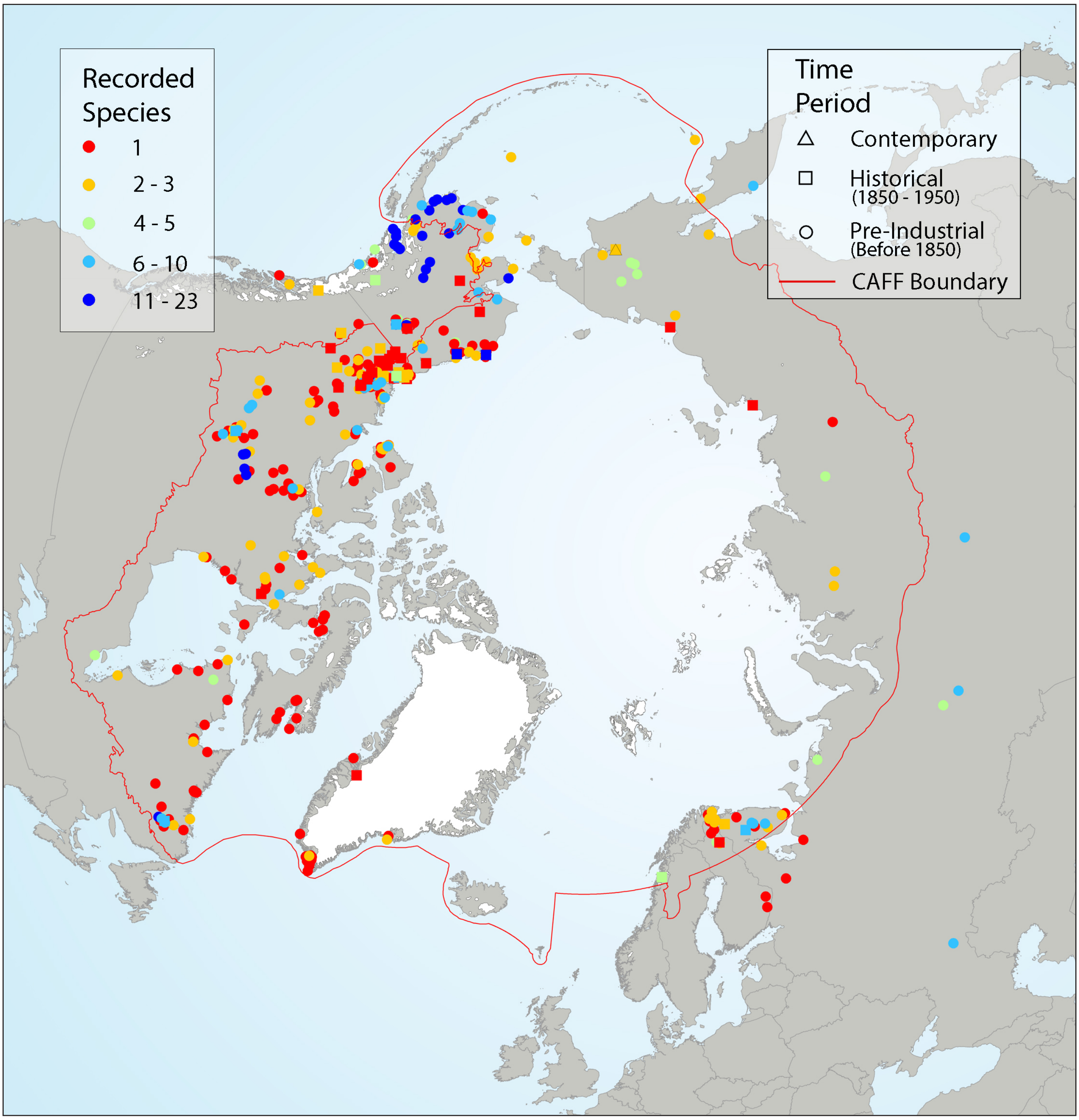


Figure 4-37 Fish species observations from Traditional Knowledge (TK) literature, plotted in the approximate geographic location of observed record, with symbol colour indicating the number of fish species recorded and shape indicating the approximate time period of observation. Results are from a systematic literature search of TK sources from Alaska, Canada, Greenland, Fennoscandia, and Russia.

Beta diversity assessment across 25 ecoregions was focused on determining the dominant component of beta diversity (i.e., nestedness or turnover) within an ecoregion. Three ecoregions had insufficient data for calculating beta diversity: Kalaallit Nunaat High Arctic Tundra, Middle Arctic Tundra, and Yamal Gydan Tundra. In the Arctic Foothills Tundra, Brooks-British Range Tundra (North America) and in the Scandinavian and Russian Taiga, the turnover component of beta diversity was greater than the nestedness component (confidence intervals did not overlap; Figure 4-36c). This indicates that the replacement of species across spatial or environmental gradients appeared to drive diversity patterns across a range of ecoregion types in North America and Fennoscandia, including alpine and taiga habitats (See Box 4-2). Generally, a heterogeneous mix of habitats or a broad range of locations (including both lakes and streams) would capture higher biodiversity in these ecoregions, because species and communities are more dissimilar over greater distances (Socolar et al. 2016). The nestedness component of beta diversity was greater only in the Iceland Boreal Birch Forests and Alpine Tundra where only three species were represented in the data, and changes in species composition across the region would result from subsetting the richest fish community. In this instance, monitoring or conserving biodiversity in high richness locations (e.g., sites, lakes, river reaches) may provide the best option of maintaining current biodiversity (Socolar et al. 2016). Beta diversity in all other ecoregions showed no significant differences in turnover and nestedness components, indicating that compositional differences within these ecoregions are due to a combination of stations containing subsets of the species found in richer communities and stations containing additional species not found elsewhere.

Beta diversity in two of the most northern ecoregions (Arctic Desert and High Arctic Tundra) equaled zero, as only a single species (Arctic charr) was captured and there was no change in freshwater fish composition among stations. These locations are species-poor and less accessible to freshwater species, presently and in the past. Reduced colonization potential in these regions prevents the addition of more species, while a hierarchy of species-specific traits may dictate distribution within those regions (Henriques-Silva et al. 2013). In these low richness regions, within-species biodiversity (e.g., polymorphisms) may be of most interest or importance for future monitoring of species.

4.7.3.2. Regional Diversity

Regional analysis was completed for five highly-sampled ecoregions, which included the Arctic Coastal Tundra and Brooks-British Range Tundra in Alaska, the Iceland Boreal Birch Forests and Alpine Tundra in Iceland, and the Scandinavian Montane Birch Forest and Grasslands and the Scandinavian and Russian Taiga in Fennoscandia. The regional species pool (based on literature and expert knowledge) in the Iceland Boreal Birch Forests and Alpine Tundra was 8 species, the lowest number among the five ecoregions. In the mountainous ecoregions, the Brooks-British Range Tundra and the Scandinavian Montane Birch Forest and Grasslands, there were 19 and 25 species, respectively. The Arctic Coastal Tundra had a species richness estimate of 26 species, and in the largest ecoregion, Scandinavian and Russian Taiga, 47 species occurred (Figure 4-36b). When compared across a standard sample size (n = 200 stations), the lowest species richness was found in the Iceland Boreal Birch Forests and Alpine Tundra (3 species, significantly lower than all other ecoregions), which is isolated from other ecoregions by the North Atlantic Ocean. Rarefied alpha diversity was highest in the Scandinavian and Russian Taiga (average of 20 species; Fennoscandian stations only) and the Arctic Coastal Tundra (average of 19 species; Figure 4-38). Species richness estimates were similar for these two ecoregions and did not differ significantly even when rarefaction curves were compared at a sampling frequency of 1500 stations. Rarefied alpha diversity (at 200 stations) in the two mountainous ecoregions was reduced compared to lower elevation Taiga and Coastal Tundra, though only the Brooks-British Range Tundra (average of 9 species) had a significantly lower species richness estimate, whereas confidence intervals for the Scandinavian Montane Birch Forest and Grasslands (average of 15 species) overlapped with those of the lower-elevation ecoregions.

In a subset of 7th level hydrobasins that contained at least 10 sampling stations, the pool of available species ranged from 3 species in Iceland Birch Forest and Alpine Tundra to 21 species in Arctic Coastal Tundra (Figure 4-39a). The average species richness of hydrobasins was typically reduced from the available species pool. Mean basin richness was 9 ± 2.5 species in the Arctic Coastal Tundra, 4.5 ± 1 species in the Brooks-British Range Tundra, 9.5 ± 1.2 species in the Scandinavian and Russian Taiga, and 5.9 ± 1.2 species in the Scandinavian Montane Birch Forest and Grassland (Figure 4-39b). Only in Iceland did the basin richness of 3 ± 0 species mirror the available species pool (Figure 4-39a-b). In Alaska and Fennoscandia, the richness of mountain region basins was consistently lower than the richness of adjacent lowland (tundra or taiga) basins. Mountain regions often have fewer speciesdue to the challenges of accessing habitats (e.g., steep stream gradients) or because of harsher climate conditions (e.g., earlier freeze-up dates).

Beta diversity differed across ecoregions, with higher values ($\beta_{SOR} > 0.70$) in the Arctic Coastal Tundra, Brooks-British Range Tundra, and Scandinavian and Russian Taiga. The Scandinavian Montane Birch Forest and Grasslands and the Iceland Boreal Birch Forests and Alpine Tundra showed moderate beta diversity (β_{SOR} values between 0.56 and 0.66). The value of β_{SOR} in Iceland Boreal Birch Forests and Alpine Tundra was likely reduced due to its low species richness and isolation. Spatial isolation may have also contributed to differences in the importance of nestedness relative to species replacement. Among the five ecoregions, only the Iceland Boreal Birch Forests and Alpine Tundra showed greater nestedness-resultant similarity compared to turnover (Figure 4-39c). Turnover, the replacement of species in space, was more important relative to nestedness in the remaining four ecoregions (Figure 4-39c), indicating that assemblages would vary across landscapes with either distance between sites or along another environmental gradient (e.g., elevation or temperature).

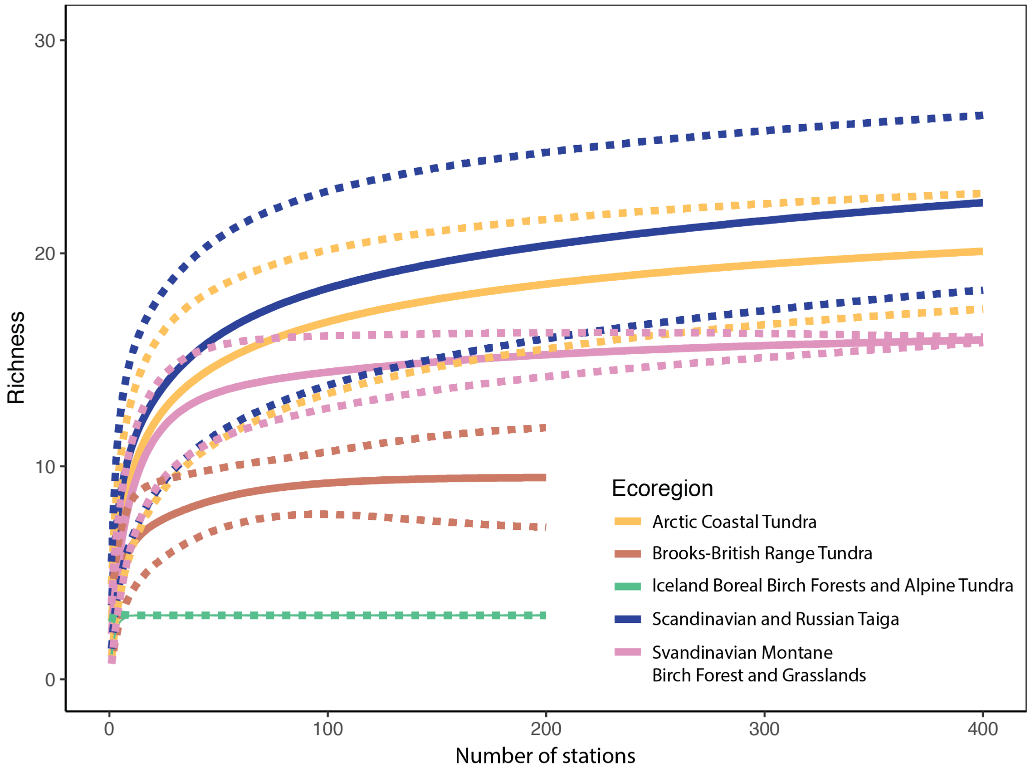


Figure 4-38 Rarefaction curves of fish species richness in the five ecoregions with robust sampling data. Dashed lines are the 95% confidence intervals. Curves for the Brooks-British Range Tundra and Iceland Boreal Birch Forests and Alpine Tundra were extrapolated to 200 stations (from 63 and 73 stations, respectively), Scandinavian Montane Birch Forest and Grasslands, Scandinavian and Russian Taiga, and Arctic Coastal Tundra were truncated at 400 stations.

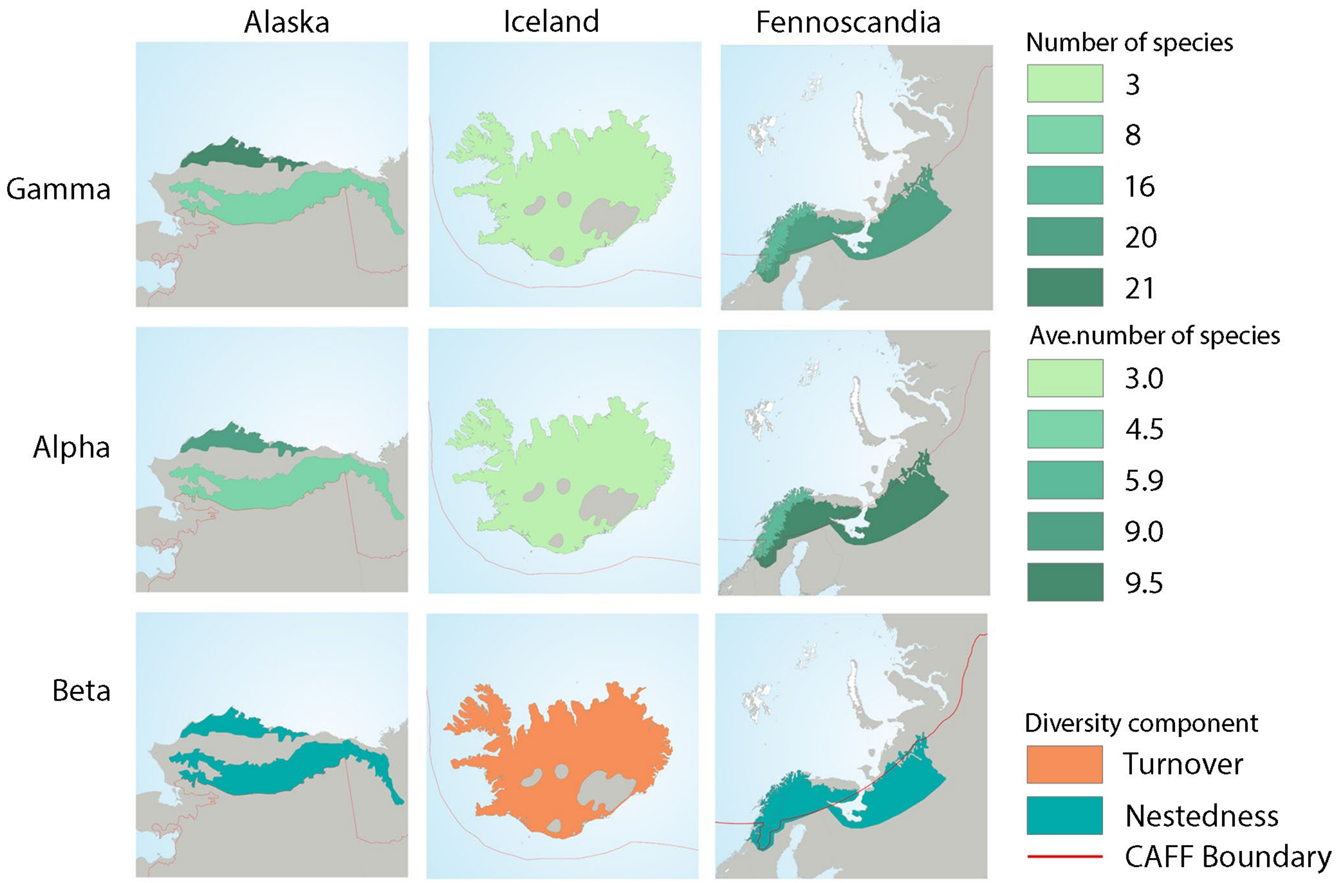


Figure 4-39 Fish diversity characteristics in three geographical regions: Alaska, Iceland, and Fennoscandia. Gamma diversity is based the total number of species sampled in hydrobasins of each ecoregion. Alpha diversity shows the mean basin species richness (95% confidence interval) and beta diversity shows the component of beta diversity, nestedness or turnover, that dominated within each of the ecoregions; gamma, alpha, and beta diversity estimates were based on a subset of basins where a minimum of 10 stations were sampled. All maps are drawn to the same scale.

4.7.3.3. Compositional Patterns

Across the total area with available fish presence data, there were discernible differences in the distribution of species (Figure 4-40), including the presence of certain families (e.g., Catostomidae in North America) or exchange in genera (e.g., *Salmo* in Fennoscandia and *Oncorhynchus* in Alaska). Fourteen species of fish had a distributional range across continents - including salmonids (7 spp.), smelts (2 spp.), sticklebacks (2 spp.), burbot (1 spp), pike (1 spp), and lamprey (1 spp.). Three additional species (all salmonids) have been introduced to Fennoscandia and Russia from North America. Generally, ecoregions that spanned greater spatial extents (e.g., Scandinavian and Russian Taiga) had higher numbers of species, and ecoregions that reached lower latitudes often contained minnows (Cyprinidae) and perch (Percidae). The most northern ecoregions contained few fish, sometimes only Arctic charr. Latitude limited the species richness, and therefore, the beta diversity (change in species composition) across space. Furthermore, in isolated locations like Iceland, the depauperate fish fauna and their distributional patterns - as subsets of the richest community - resulted in lower overall beta diversity, and a higher index of nestedness compared to turnover. Mountain regions may be similarly isolated, with fish species access reduced due to stream gradients or climate. In the regional analysis, species richness was reduced in the Brooks-British Range Tundra and in the Scandinavian Montane Birch Forest and Grasslands when compared to adjacent, low-elevation ecoregions (e.g., Brooks-British Rar Tundra elevation range 0-150 m; https://www.worldwildlife.org/biome-categories/terrestrial-ecoregions). Interestingly, the within-ecoregion beta diversity was comparable, and mountain and low elevation ecoregion beta diversity was primarily supported through species turnover.

Biodiversity analyses were influenced by the availability of data across and within ecoregions. For some areas, limitations based on sample size (the number of stations) hindered our ability to fully examine species richness from the data gathered for the CBMP database. For example, in our regional subset, which contained the most robust data, we could not discern differences in species richness between the Scandinavian Montane Birch Forest and Grasslands and the adjacent Scandinavian and Russian Taiga until nearly 300 stations were sampled. In all other ecoregions but one, we had far fewer than 300 sample stations, and therefore, an inability to compare richness at the hydrobasin level based on collected data. Fortunately, fish distributions are well known, especially compared to other aquatic organisms, and species richness of ecoregions could be determined based on literature, expert knowledge, and indigenous knowledge. While we were able to determine whether beta diversity within ecoregions was due to either replacement or loss of species, this often relied on small sample sizes, with one or two hydrobasins representing large spatial extents. Increasing spatial and temporal coverage, through additional monitoring or improved access to existing data, would improve our ability to determine the status of freshwater fishes.

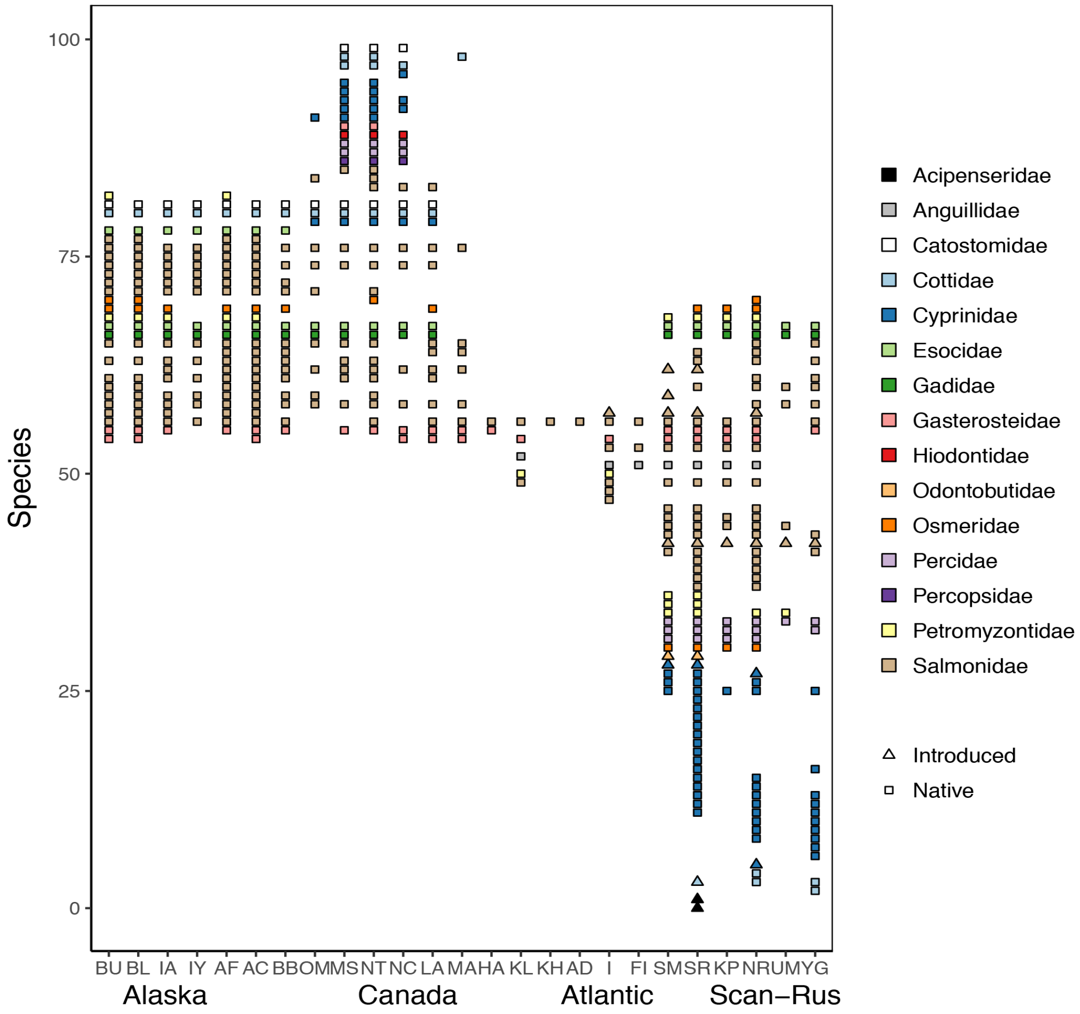


Figure 4-40 Longitudinal distribution pattern of fish species from Alaska to western Russia. Each number (y-axis) represents a single species, colored by taxonomic family. Species numbers are referenced in Appendix A. Introduced species are represented by circles. See Figure 4-36 for ecoregion abbreviations.

4.7.3.4. Temporal Trends

Changes to thermal and hydrological regimes of freshwaters due to climate change are predicted to affect the distributions and prevalence of salmonids including Atlantic salmon (*Salmo salar*), brown trout (*Salmo trutta*), and Arctic charr (*Salvelinus alpinus*) (Elliott and Elliott 2010, Finstad and Hein 2012). Northern Norway (65–71°N) and Iceland (64–66°N) are among the only regions in the world where distributions of these species overlap. Long-term catch records for these areas provide an opportunity to assess recent changes in the abundance of these fish species and evaluate whether similar trends are evident in both countries.

A 24-year record of fish relative abundance (percent of total abundance) from Iceland shows that Atlantic salmon were most abundant in the west by a margin of about 50-70% (Figure 4-41a), while trout were most abundant in the south by about 10-30% (Figure 4-41b). Communities in the north and east exhibited the strongest changes in relative abundance over time (Figure 4-41c,d). In these regions, previously similar abundances of Atlantic salmon and

Box 4-2. Case Study. Impact of climate, land-use, and human population development on fish biodiversity

Both climate and land-use affect Arctic freshwaters and their fish communities. For example, Hayden et al. (2017) examined fish communities along a gradient of altitude, human population density, and land-use intensification in the subarctic, Tornio-Muoniojoki catchment (Figure 4-43) over the period of 2009 to 2013. Levels of nutrients (phosphorus, nitrogen, carbon) in lakes increased along the gradient leading to higher ecosystem productivity. This productivity gradient was associated with a change in fish community composition with salmonids (European whitefish, *Coregonus lavaretus*) dominant in headwater lakes. Fish composition then progressively shifted downstream towards percid (perch, *Perca fluviatilis*, and ruffe (*Gymnocephalus cernua*) and finally cyprinid (roach, *Rutilus rutilus*) dominance (Figure 4-43). This progressive change was accompanied by a near 50-fold increase in relative biomass of fish, and a 50% decrease in mean body size. This massive increase in fish abundance was correlated with a reduction in the size of invertebrate prey, a shift towards smaller invertebrate species, and decreased invertebrate diversity, particularly in the most productive lakes. They also observed distribution limits and continuous range expansions over the period of record for cool and warm water species such as percids (ruffe, perch), and cyprinids (ide [*Leuciscus idus*], roach, bleak [*Alburnus alburnus*]). In contrast, range retractions were evident for the cold water species Arctic charr (*Salvelinus alpinus*), grayling (*Thymallus thymallus*), brown trout (*Salmo trutta*), and burbot (*Lota lota*). The study concludes that effects of range expansion cannot be predicted by bioclimatic envelope models alone, but that lake-specific abiotic and biotic data must be integrated to realistically assess future fish community diversity. Hence, long-term data *fr*om Arctic systems are required to optimally assess the relative roles of different abiotic and biotic factors in determining fish diversity and ecosystem functioning. However, if such long-term data are not available, space-for-time substitution studies have the potential to provide an alternative approach to predict future change in fish diversity.

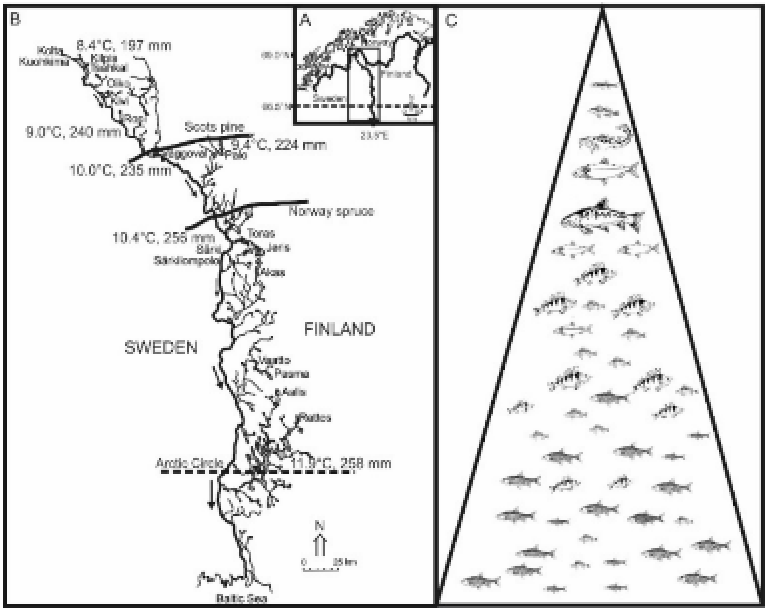


Figure 4-43 The map of northern Fennoscandia (A) and subarctic Tornio-Muoniojoki catchment showing the location of 18 tributary lakes. Open water season air temperature and precipitation (June-September 1981-2010) at six weather stations and locations of coniferous treelines are shown (B). Change in fish communities, body size, and abundance along the climate and productivity gradient are illustrated (C). (Modified from Hayden et al. 2017)

anadromous Arctic charr (~45% each) have been diverging since 2005 due to declines in the relative proportion of Arctic charr, resulting in a dominance of Atlantic salmon in these systems. At the same time, in the north and east, brown trout have steadily increased (10-15%) since 1992 (at the start of record). Potential temporal shifts in the relative abundance of fish species in Iceland's river communities will change current patterns of species diversity - lessening the evenness among species in some regions (e.g., diverging percent abundance of Arctic salmon and anadromous Arctic charr in northern rivers) while increasing the evenness of species in others (e.g., brown trout and anadromous Arctic charr in western rivers).

Long-term records from northern Norway indicate that Atlantic salmon has dominated in river-based systems for the entire period of record (1993-2016), and has been increasing in relative abundance over the last several years (Figure 4-42a). The amount of brown trout in the catches has been relatively stable throughout the period, while Arctic charr have shown a decline in relative abundance over the last 10-15 years. In lake-based systems, however, brown trout seems to be the dominant species and has shown a steady increase from 1995 until approximately 2011, while relative abundances of both Atlantic salmon and anadromous Arctic charr declined over the same period (Figure 4-42b). Thus, the relative abundance of anadromous Arctic charr has generally declined in rivers of northern Norway, both in river-based and lake-based systems (Figure 4-42a,b). However, whereas there was an early period of relative stability followed by a decline after 2002 in Norwegian river-dominated systems, similar to the patterns seen in Iceland, there was a more steady decline in anadromous Arctic charr abundance in lake-based systems in northern Norway from 1995 to 2009 (Figure 4-42b).

Coherent changes in two countries that are located on each side of the Norwegian Sea indicate that a common factor such as climate change may be causing these declines in Arctic charr. However, the mechanisms for the changes are not fully understood. In Iceland, water temperature has shown an increase in spring and autumn while the average temperature for the summer months (June – August) has not shown an increase. The effects of increased water temperatures in spring and autumn might affect and possibly cause mismatch in spawning and hatching time of Arctic charr while salmon and trout remain unaffected. The strong contrast in the dominance of brown trout and Atlantic salmon in northern Norway between lake- and river-based systems speaks to the important influence of lakes on fish assemblage composition.

4.7.4 Gaps in Knowledge and Monitoring

While fish are key species in aquatic ecosystems and are important to communities of the North, it is evident that there are significant gaps in monitoring effort and data coverage across the circumpolar region. Although in some cases the spatial extent is limited because existing datasets were not accessible (e.g., some academic sources that were not open access or government-funded programs that are no longer in operation), there remain significant gaps in monitoring effort and coordination of routine monitoring in some areas. Across Canada, for example, a large number of historical studies focused on monitoring commercial or subsistence fisheries, and thus quantified a selection of fish species rather than assessing the diversity of the full assemblage. Furthermore, many sites across North America have only been sampled once, thus precluding temporal analyses of trends. Similarly, there are large areas that have not been sampled sufficiently to allow for analyses of spatial patterns or temporal trends.

Sparse long-term data on fish assemblages exist for the Arctic. Long-term commercial fishing data are common in North America and Europe (e.g., see section 4.7.3.4), but these records generally focus on targeted fish species rather than assessing the full assemblage. There were large spatial gaps in the distribution data that could be obtained on fish assemblages that hinder assessments of fish distribution or biodiversity patterns across large spatial or circumpolar scales. Furthermore, the number of fish species included in the database represents only 42% of all Arctic species (Wrona et al. 2013). In northeastern North America, only four of 38 species (or 10.5%) were included, and only two species (Arctic charr and threespine stickleback) that were surveyed outside of commercial fishing data were used in our analyses. In two regions, which were considered relatively data rich, Beringia (i.e., Alaska) & northwestern Canada and Russia & Scandinavia, only 50% and 57% of the species, respectively, were found in the dataset. (See Appendix A for species lists). Until broader spatial and temporal data coverage is available, the ability to assess changes in biodiversity, especially at large spatial scales, will be limited.

Arctic grayling
Photo: Rostislav Stefanek/Shutterstock.com

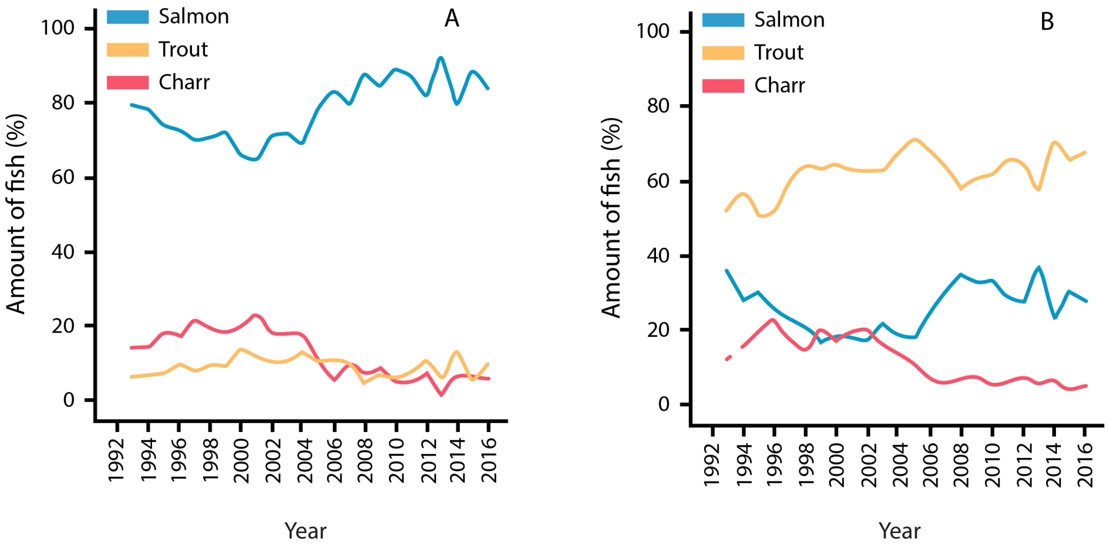
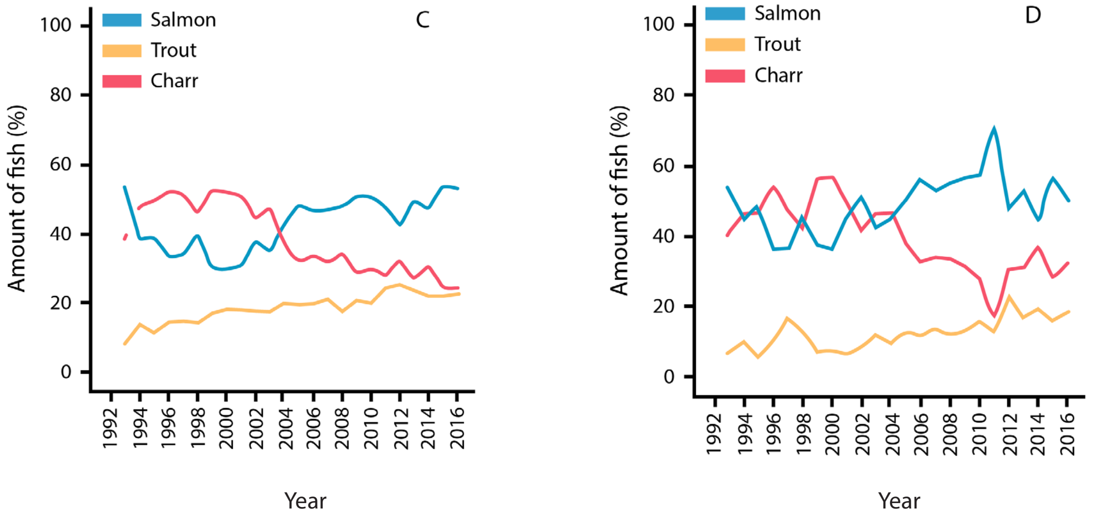



Figure 4-41 Temporal patterns in % abundance of Atlantic salmon, brown trout, and anadromous Arctic charr from catch statistics in Iceland rivers monitored from 1992 to 2016, showing results from (a) west, (b) south, (c) north, and (d) east Iceland.

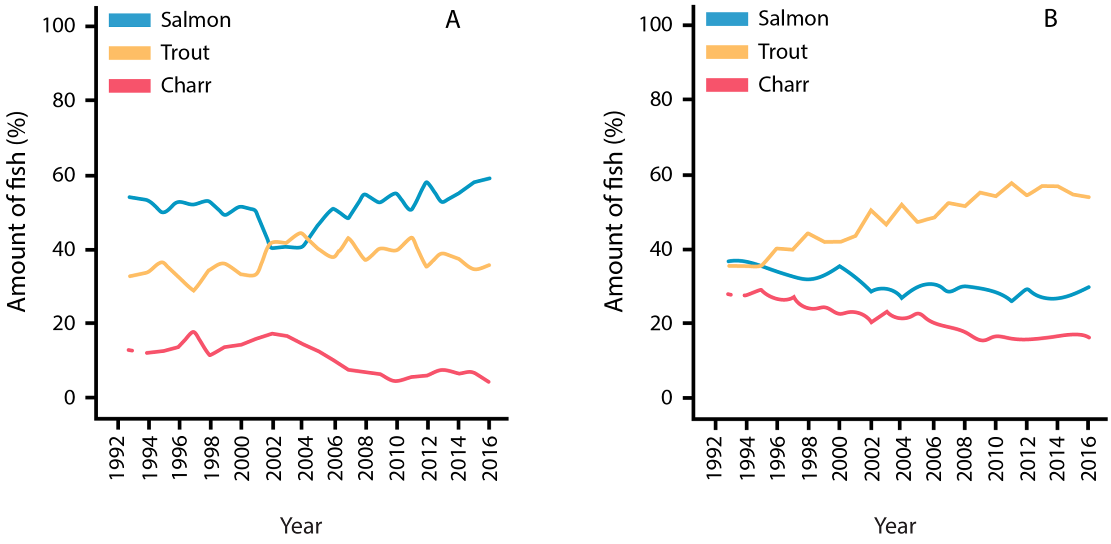


Figure 4-42 Temporal patterns in % abundance of Atlantic salmon, brown trout, and anadromous Arctic charr from catch statistics in northern Norway rivers monitored from 1993 to 2016, including basins dominated by (a) rivers and (b) lakes.

Researchers collect aquatic invertebrates from small ponds on the Arctic Coastal Plain, Alaska. Photo: Christian Zimmerman, USGS

5. Freshwater Biodiversity Synthesis

5.1. Circumpolar Comparisons Among FECs

General patterns of diversity were compared across FECs to identify areas of the Arctic with consistently high or low diversity. Such patterns can help identify diversity hotspots, where diversity is high among multiple FECs, as well as areas where diversity is low and might be expected to change if warming temperatures lead to the northward movement of eurythermal species. Furthermore, patterns in diversity across FECs can help identify relevant drivers of diversity, e.g., where regional conditions in an area of low diversity differ from those in areas of high diversity. Comparisons were made among ecoregions that had moderate to high sample coverage (e.g., those ecoregions that had sufficient numbers of stations to examine alpha diversity rarefied to 25 or more stations). There were no ecoregions with moderate to high numbers of stations for all FECs for lakes, and only three ecoregions for rivers, and therefore comparisons were made among ecoregions with sufficient coverage of at least two FECs.

Fennoscandia (in particular, the Scandinavian and Russian Taiga, which is the inland ecoregion) represented a diversity hotspot for lake macrophytes, zooplankton, benthic macroinvertebrates, and fish (there was insufficient sampling of diatoms and phytoplankton to assess the regional patterns in these FECs; Table 5-1). The Northwest Russian-Novaya Zemlya Tundra in Russia was a hotspot for zooplankton and fish, ranking first and second, respectively, for diversity of these FECs. The Arctic Coastal Tundra in Alaska (northern coastal ecoregion) ranked as the most diverse ecoregion for lake diatoms and phytoplankton, and was among the most diverse ecoregions for fish, zooplankton, and BMI (Table 51). The low Arctic in Greenland had diverse plankton assemblages (both phytoplankton and zooplankton), but ranked the lowest in diversity for diatoms (Table 5-1). Ecoregions in Canada and Iceland were found to be less diverse for many of the FECs, indicating lower diversity than was found overall in inland Fennoscandia, eastern Russia, and along the northern Alaskan coast. The warmer climate in Fennoscandia (particularly in the less mountainous taiga ecoregion) and strong connectivity to the mainland may play a role in the overall high diversity of the area. Similarly, connectivity of the Alaskan coastal region and lack of recent glaciation in that area may have contributed to high diversity of diatoms, phytoplankton, and fish.

Similar results were obtained when the diversity of river FECs was compared across ecoregions. The Scandinavian and Russian Taiga was overall the most diverse ecoregion across all three FECs, though the Arctic Coastal Tundra in Alaska ranked highest in diversity for diatoms and fish (Table 5-2). The Brooks-British Range Tundra in Alaska ranked low in diversity for both rivers and lakes, suggesting an effect of the presence of mountains which was also supported by the slightly lower diversity rankings for the Scandinavian Montane Birch Forest and Grasslands Table 5-1, Table 5-2). Interestingly, the Alaskan ecoregions south of the Brooks-British Range ranked low for fish diversity, which may have reflected barriers for dispersal of anadromous species south from the diverse Arctic Coastal Tundra. Diversity was lowest for diatoms and benthic macroinvertebrates in eastern and northern Canada, though patterns across these two FECs appeared to differ (i.e., ecoregions that ranked the lowest for benthic macroinvertebrates were not the lowest diversity ecoregions for diatoms; Table 5-2). Greenland ranked near the least diverse for both BMI and fish (Table 5-2).


Burbot
Photo: Stefanek Rostislav/Shutterstock.com

Table 5-1 Ranking of rarefied alpha diversity of lake diatoms, phytoplankton (Phyto), macrophytes, zooplankton (Zoo), benthic macroinvertebrates (BMI), and fish in ecoregions with a sufficient number of samples to rarefy to 25-40 stations. Ecoregion rankings are colored from green (highest diversity) through red (lowest diversity). Numbers in colored cells refer to ecoregion ranking, but are a different scale depending on how many ecoregions were compared for each FEC. Ecoregions are grouped by geographic region of the Arctic, and listed from west to east, as relevant.

Region	**Lake Ecoregions**	**FECs**					
		Diatoms	Phyto	Macrophytes	Zoo	BMI	Fish
Alaska	Arctic coastal region	1	1		3	3	3
	Brooks british Range Tundra				7		5
Canada	Low Arctic Tundra	5	3		5		
Greenland	Kalaallit Nunaat low arctic tundra	6	2		4		
Iceland	Iceland boreal birch forests and alpine tundra	2		3	6	4	6
Fennoscandia	Scandinavian Montane Bircg forest and grasslands	4		1		2	4
	Scandinavian and Russian taiga			2	2	1	1
Russia	Northwest Russian Novaya Zemlya Tundra				1		2
	Taimyr-Central Siberian Tundra	3	4				

Table 5-2 Ranking of rarefied alpha diversity of river diatoms, benthic macroinvertebrates (BMI), and fish in ecoregions with a sufficient number of samples to rarefy to 25-40 stations. Ecoregion rankings are colored from green (highest diversity) through red (lowest diversity). Numbers in colored cells refer to ecoregion ranking, but are a different scale depending on how many ecoregions were compared for each FEC. Ecoregions are grouped by geographic region of the Arctic, and listed from west to east, as relevant.

Region	**River Ecoregions**	**FECs**		
		Diatoms	BMI	Fish
Alaska	Arctic coastal tundra	1	6	1
	Brooks british Range Tundra		8	6
	Interior Alaska-Yukon lowland taiga		5	5
Canada	Northwest Territories taiga		2	4
	Low Arctic tundra	6	7	
	Middle Arctic tundra	4	10	
	High Arctic tundra	5	12	
	Eastern Canadian Shield taiga	7	3	
	Tongat Mountain tundra	8	9	
Greenland	Kalaallit Nunaat low arctic tundra		11	7
Fennoscandia	Scandinavian Montane Birch forest and grasslands	3	4	3
	Scandinavian and Russian taiga	2	1	2

Turnover was the dominant component of beta diversity for the majority of FECs, and only fish and zooplankton had a large number of ecoregions in which turnover and nestedness were approximately equal or nestedness dominated. Soininen et al. (2018) noted that turnover generally dominates across a number of ecosystems, but suggested that the relative importance of turnover and nestedness might differ as a function of latitude, with increased nestedness at higher latitudes. There was little support for a latitudinal shift in the importance of nestedness within ecoregions in this circumpolar analysis (when tested for some FECs, there was no evidence of a relationship between % nestedness and latitude). However, Culp et al. (In Press) found strong evidence for increased importance of nestedness in river BMI when more distant latitudes were compared, indicating that assemblage composition in high Arctic streams was a subset of (nested within) what was found at lower latitudes. Across the suite of FECs in our analysis, nestedness often played a larger role in spatially-isolated areas (e.g., islands such as Iceland and Svalbard). This result may be indicative of the smaller taxonomic pool on isolated islands (following from theories of island biogeography). However, nestedness was also low in ecoregions with few samples or with spatially-restricted sampling (e.g., ecoregions in Canada where many stations were grouped in a small geographic area), which indicated that beta diversity estimates and partitioning were affected by sample frequency and sample distribution. With greater spatial coverage of stations, it may be possible to conduct a more accurate assessment of the relative importance of turnover and nestedness across Arctic freshwaters.

5.2. Regional Comparisons Among FECs

5.2.1. North America

River data for North America showed high spatial coverage for diatoms, benthic macroinvertebrates, and fish allowing for assessment across multiple FECs. All FECs were sampled in northern coastal Alaska and the lower Arctic Archipelago, and other areas of North America had data for either benthic macroinvertebrates and fish or benthic macroinvertebrates and diatoms. Alpha diversity patterns were compared across ecoregions to evaluate whether trends across Canada and USA were similar for each of the FECs. We also examined whether there were latitudinal trends in alpha diversity by calculating family richness within hydrobasins. Finally, FEC patterns were compared with spatial patterns in abiotic variables to explore potential drivers of biodiversity for each of the FECs.

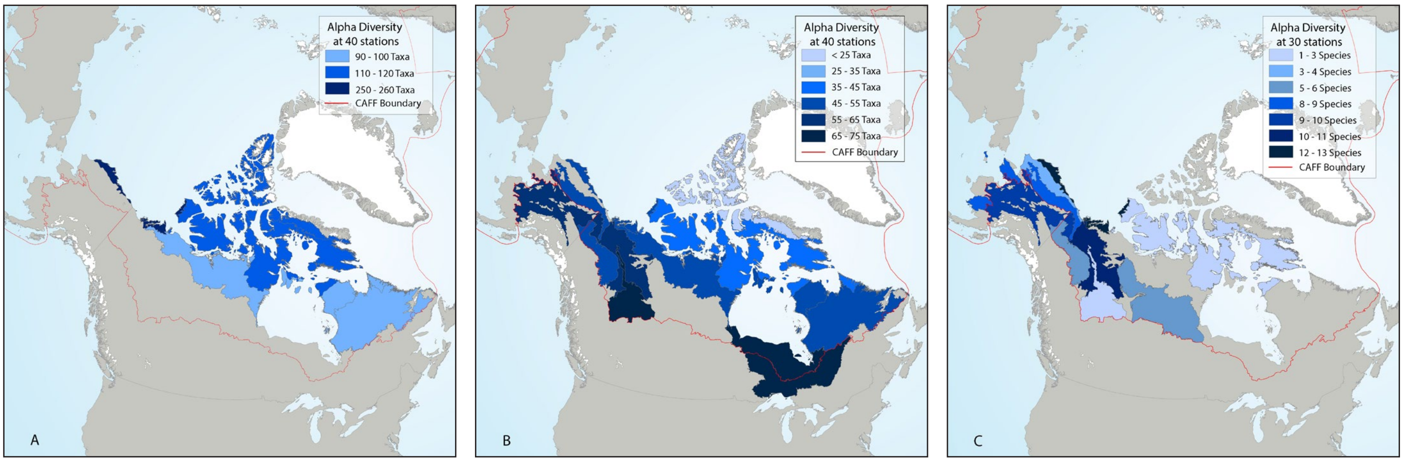


Figure 5-1 Rarefied alpha diversity of river (a) diatoms from benthic samples, (b) benthic macroinvertebrates, and (c) fish in ecoregions across North America.

Different spatial patterns of biodiversity were evident for river diatoms, benthic macroinvertebrates, and fish across North America (Figure 5-1). The lowest biodiversity of diatoms was in the more southern ecoregions of Canada, whereas biodiversity was higher towards the north, in the Arctic Archipelago (Figure 5-1a). In contrast, benthic macroinvertebrates showed a clear decline in alpha diversity towards the north in Canada, with the highest biodiversity south of Hudson Bay, and gradual declines in the number of families with increasing latitude (Figure 5-1b). Fish did not appear to display strong latitudinal trends in Canada in our limited data (Figure 5-1c), although it is known that the High Arctic ecoregion includes only one fish species (Arctic charr) and that there is a latitudinal decline in diversity of this FEC (see Scott and Crossman 1973). The highest biodiversity of both diatoms and fish was found in northern Alaska, in the Arctic Coastal Tundra ecoregion (Figure 5-1a, c), whereas there was only moderate diversity of benthic macroinvertebrates in that ecoregion (Figure 5-1b). However, benthic macroinvertebrates and fish displayed a similar latitudinal gradient in alpha diversity across the two southern Alaska ecoregions, with fish diversity further declining into the mountainous Brooks-British Range tundra (Figure 5-1b, c).

Latitudinal assessment of diversity across hydrobasins indicated no evidence of a latitudinal decline in either benthic diatoms or fish, though both showed a peak in diversity at around 70°N (stronger in diatoms), corresponding to hydrobasins in the Arctic Coastal Tundra ecoregion (Figure 5-2a, c). At other latitudes, diversity of diatoms remained similar for both eastern and western North America, whereas fish diversity (only tested in western hydrobasins) varied widely across remaining latitudes. Conversely, there was a clear decline in alpha diversity with increasing latitude for benthic macroinvertebrates (Figure 5-2b). Furthermore, western Arctic hydrobasins had consistently higher alpha diversity than eastern Arctic hydrobasins from similar latitudes (Figure 5-2b).

The contrasting spatial patterns of diversity among diatoms, benthic macroinvertebrates, and fish relate to differences in the response of each FEC to environmental drivers. For example, the latitudinal and longitudinal patterns in river benthic macroinvertebrates reflect temperature gradients across the North American Arctic. In addition to a strong latitudinal decline in temperatures, there is also a west-east temperature gradient in the North American Arctic, with higher temperatures in western North America than what is found in eastern North America at similar latitudes (Figure 5-3a). Benthic macroinvertebrates have thermal preferences and vary in their tolerance levels for extreme cold (Danks 1992, Danks et al. 1994, Wrona et al. 2013). As a result, several studies have noted declines in benthic macroinvertebrate diversity with increasing latitude that follow from a lower number of invertebrate taxa with the physiological tolerance levels for extreme cold conditions (Scott et al. 2011, Culp et al. In Press). Our results confirm these trends for the North American Arctic region and further indicate that benthic macroinvertebrate diversity also reflects the west-east temperature gradient, as diversity was consistently higher in the warmer western ecoregions than in the cooler eastern ecoregions at similar latitudes.

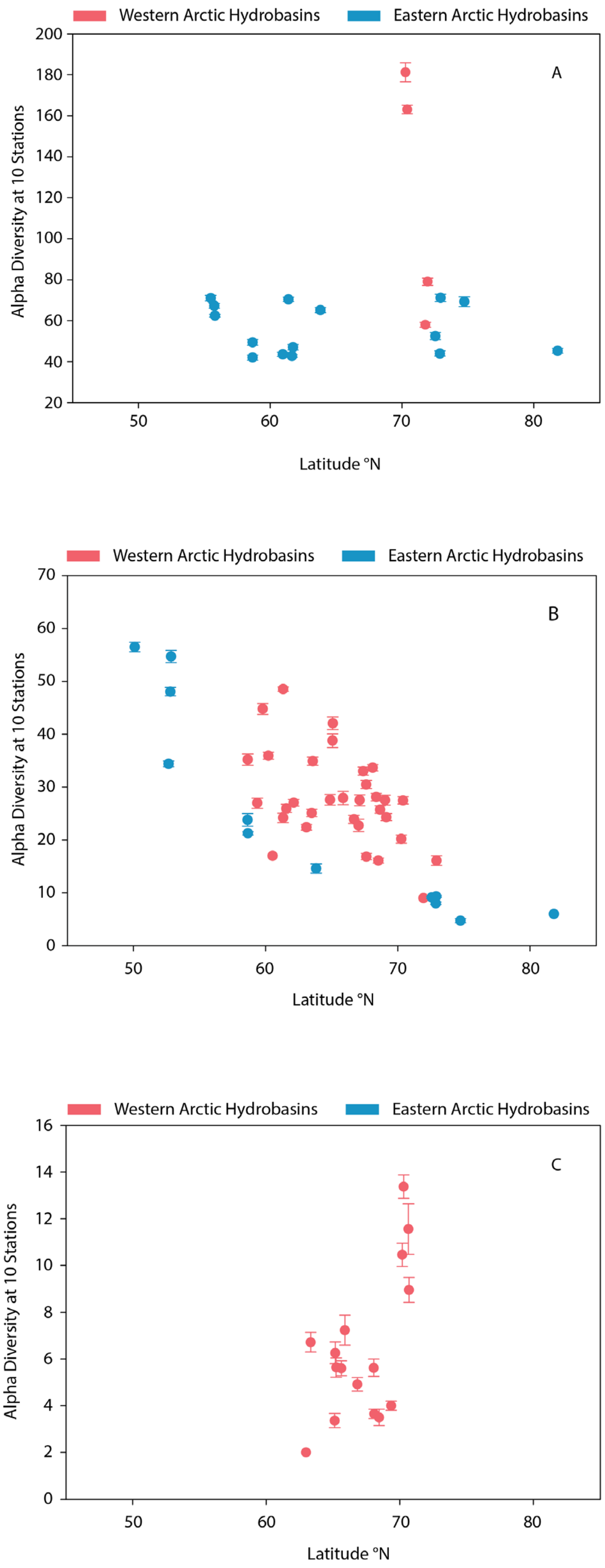


Figure 5-2 Alpha diversity (± standard error) of river (a) diatoms from benthic samples, (b) benthic macroinvertebrates, and (c) fish within hydrobasins in western and eastern North America plotted as a function of the average latitude in each hydrobasin. Alpha diversity is rarefied to 10 stations per hydrobasin, using size level 5 hydrobasins for all panels.

Although fish species have thermal preferences and tolerance levels, factors related to dispersal and glaciation may also play a predominant role in driving fish diversity patterns. For example, fish assemblages in the most northern latitudes of North America are limited to anadromous species that are able to access the productive marine environment for foraging (Wrona et al. 2013). Dispersal barriers in mountainous regions (e.g., Brooks-British Range Tundra) further limit species diversity of fish (Matthews 1998, Hugueny et al. 2010). In contrast, the areas of highest fish diversity, including northern and southern Alaska and the Northwest Territories Taiga, may reflect the lack of recent glaciation in these areas (Figure 5-3b), which would have eliminated the need for recolonization and maintained species diversity.

Similar to fish, the high diversity of river diatoms in coastal Alaska may have reflected patterns of glaciation in this area (Figure 5-3b). However, diatoms patterns did not appear to reflect temperature trends across North America, as higher diversity was noted in more northern ecoregions. Diatom assemblages are known to differ in response to underlying geology, due to its influence on water chemistry and nutrient availability (Grenier et al. 2006). Sampled areas of the southern ecoregions in Canada are underlain primarily by intrusive bedrock, whereas the northern ecoregions included sampling in areas of metamorphic, sedimentary, and volcanic bedrock (Figure 5-3c). This diversity in geological composition and the associated differences in water chemistry across the northern sampled areas may have contributed to the diversity of diatoms, as samples would have reflected different habitat conditions.

5.2.2 Fennoscandia

We analyzed a data set covering 13 Fennoscandian subarctic lakes that were situated between 62.1°N and 69.3°N, and had data for five FECs (phytoplankton, macrophytes, zooplankton, benthic macroinvertebrates and fish), covering both pelagic and benthic food webs and three trophic levels. These data were compared with a full set of abiotic and geospatial variables to study relationships between biodiversity and environmental drivers. The percentage taxa share (i.e., taxa richness in a lake relative to the total taxa richness in all Fennoscandian lakes) of individual FECs was calculated based on presence-absence data. This approach combines the summed information among all five organism groups (FECs) and not the traditional splitting of analyses for different organism groups. All FEC, abiotic and geospatial variables were averaged in the order: samples -> stations -> months -> years, to obtain inter-annual averages for each lake. Redundancy analysis (RDA) based on correlations was used to investigate the environmental drivers of the FEC patterns. Explanatory abiotic and geospatial variables were tested with permutational ANOVA, and only significant explanatory variables ($p < 0.05$) were included in the RDA.

The results showed that the FECs were strongly influenced by climatic drivers (e.g., latitude, temperature, precipitation) and vegetation cover (percent grasslands and woody savannas in hydrobasins) (Figure 5-4a and Figure 5-4b). Fish seemed to be more correlated with primary producers than with zooplankton and benthic macroinvertebrates. This correlation likely reflects the top-down trophic cascades in food chains and partly corresponds to a gradient between nutrient-poor

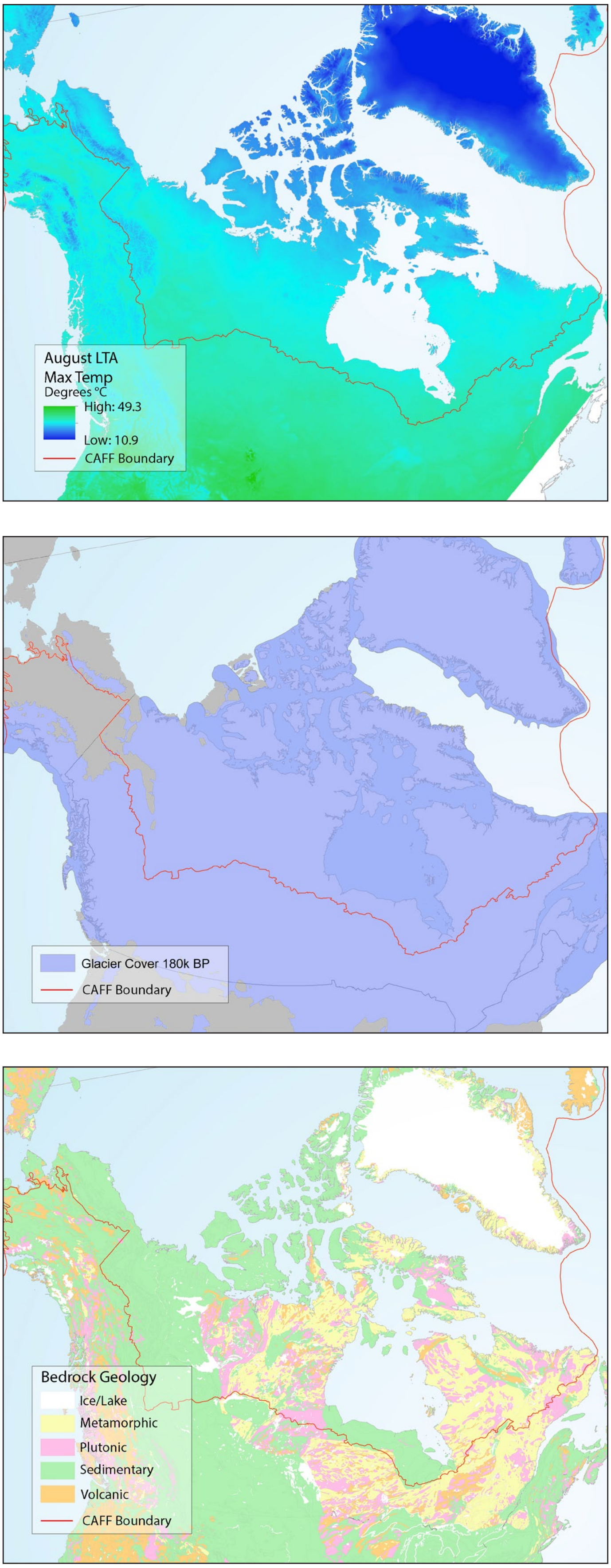


Figure 5-3 Abiotic drivers in North America, including (a) long-term average maximum August air temperature, (b) spatial distribution of ice sheets in the last glaciation of the North American Arctic region, and (c) geological setting of bedrock geology underlying North America. Panel (a) source Fick and Hijmans (2017). Panel (b) source Dyke et al. (2003). Panel (c) source: Garrity and Soller (2009).

and more nutrient-rich lakes. As the fish taxa could occupy various trophic positions in the Fennoscandian lakes, the correlation may also reflect that the diversity within and between trophic levels (i.e., horizontal and vertical diversity; Duffy et al. 2007) of the lake food webs were tightly coupled. A similar positive correlation in biodiversity index between fish and phytoplankton had been reported for Swedish boreal lakes that were either relatively pristine or subjected to long-term acidification with or without management interventions (Lau et al. 2017). Zooplankton and benthic macroinvertebrates taxa share increased with increasing altitude and decreasing relative cover of evergreen needle-leaf forests in hydrobasins. This result likely indicates climate effects on the intermediate trophic levels along the elevation gradient. Overall, the average species taxa share among FECs increased with increasing latitudes and altitudes. This analysis, however included relatively few lakes, largely due to a lack of data for primary producers for many of the lakes in our data set.

A second RDA analysis was run using 39 lakes situated between 62.1°N and 71.0°N with three FECs (zooplankton, benthic macroinvertebrates, and fish) and corresponding abiotic and geospatial variables. This analysis corroborated the weak correlation between fish, zooplankton, and benthic macroinvertebrates (Figure 5-4c and Figure 5-4d). In this analysis, fish and the average taxa share correlated strongly with lake total nitrogen concentrations, i.e., productivity, and the relative coverage of Scandinavian and Russian taiga vegetation, and negatively with open shrublands. The latter reflects the transition from evergreen pine forests to the tundra shrub vegetation along a latitudinal gradient and at higher altitudes in Fennoscandia. Results from our first (13 lakes) and second (39 lakes) RDAs together suggest that fish biodiversity is functionally important for supporting the overall biodiversity (i.e., average taxa share), and that fish can be an indicator FEC group to represent average taxa share in subarctic Fennoscandian lakes. Our second analysis also shows that the climate effects (e.g., latitude, annual mean precipitation) on fish and average taxa share could be strongly mediated by nutrients, and that zooplankton and benthic macroinvertebrates (%ShareBMI) were negatively correlated to latitude. The latitudinal trend in benthic macroinvertebrate diversity in Fennoscandian lakes is particularly consistent with that observed in North American hydrobasins (see section 5.2.1). Overall, these analyses reflect the biodiversity changes in the Fennoscandian lakes along latitudinal and nutrient gradients. Unfortunately, due to

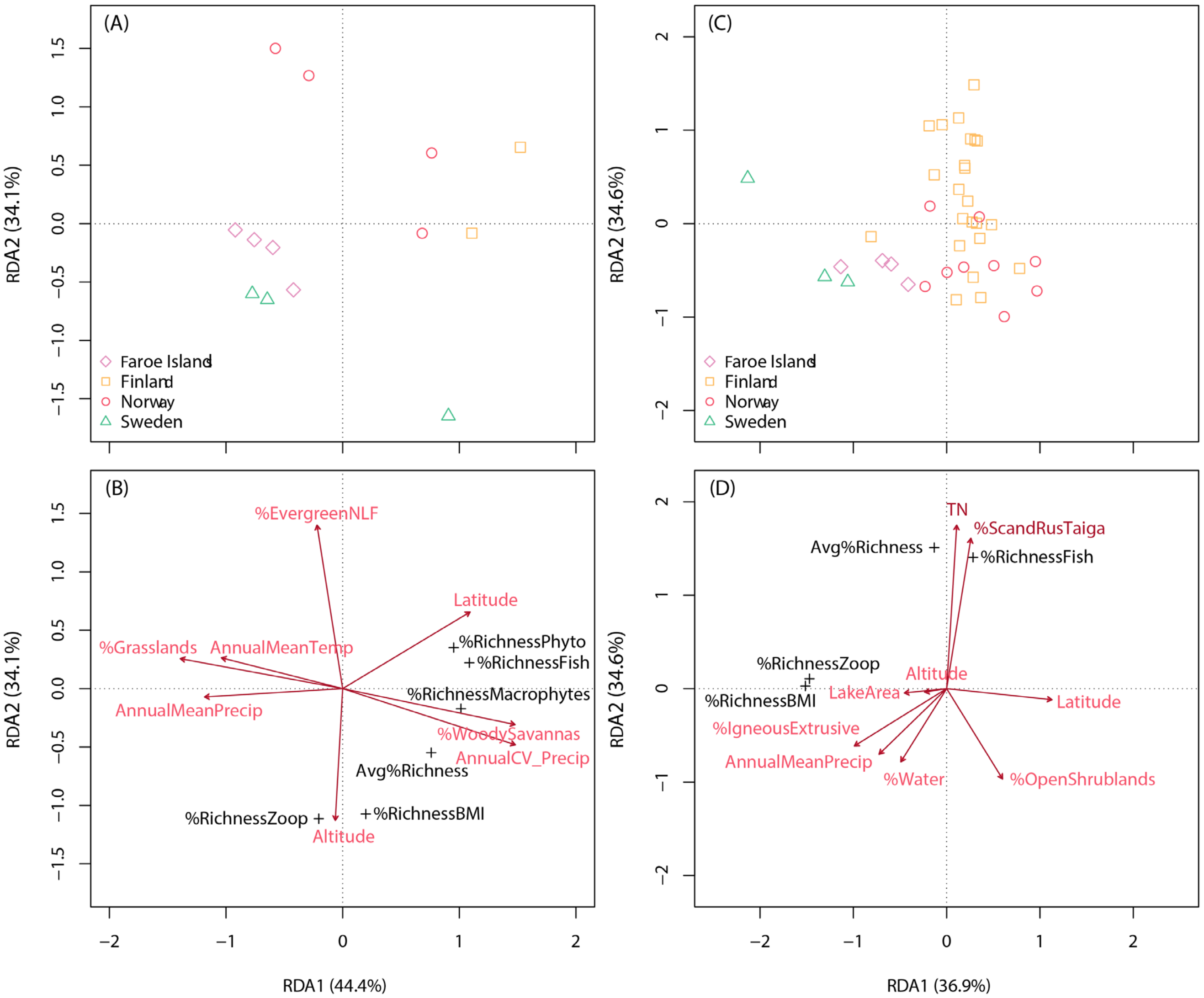


Figure 5-4 Redundancy analysis of percentage species taxa share among 5 FECs (phytoplankton, macrophytes, zooplankton, benthic macroinvertebrates and fish) in 13 Fennoscandian lakes (panels A and B) and among 3 FECs in 39 Fennoscandian lakes (panels C and D).The upper panels show lake ordinations, while the bottom panels show explanatory environmental variables (red arrows), as indicated by permutation tests ($p < 0.05$). Avg%Share: average percentage species taxa share calculated from all FECs (i.e., including benthic algae if present); %Share BMI: relative taxa share in benthic macroinvertebrates; %EvergreenNLF: percentage cover of evergreen needle-leaf forests.

data deficiency our analysis did not include local habitat variables (e.g., substratum type, vegetation), which are important descriptors for macroinvertebrate assemblages (Johnson and Goedkoop 2002). Although taxonomic composition is constrained by the size of the regional species pool, habitat heterogeneity and the outcome of biotic interactions are, along with climate, important descriptors of assemblage composition and diversity, both for benthic macroinvertebrates, zooplankton and fish.

5.3. Relation of Biodiversity to the Abiotic Template

Differences in biodiversity among ecoregions were hypothesized to be driven by temperature differences in many cases, and comparison of North American BMI diversity patterns with long-term average air temperature supported this idea (see section 5.2.1). To investigate this on a circumpolar scale, maximum long-term average August air temperatures were overlain on the ecoregions used in the SAFBR analysis (Figure 5-5). Clear patterns emerged with respect to temperature differences among ecoregions. For example, high latitude ecoregions such as the Kalaallit Nunaat High Arctic Tundra in Greenland, the Arctic Desert in Svalbard, the Wrangel Island Arctic Desert in Russia, and the eastern regions of the High Arctic Tundra and Middle Arctic Tundra in Canada were among the coldest areas. Unsurprisingly, several lower-latitude, high-altitude ecoregions were also colder than surrounding ecoregions, including the Scandinavian Montane Birch Forests and Grasslands, the Ogilvie-Mackenzie Alpine Tundra in western Canada, and the Brooks-British Range Tundra in Alaska. The biotic association with these climate patterns was most evident in the BMI, which generally showed lower diversity in these colder ecoregions than in neighbouring ecoregions (Figure 4-29 and Figure 4-30). However, fish diversity did also appear to be lower in many of these colder ecoregions (Figure 4-36), which indicated that both fish and BMI diversity relate to temperature. Biodiversity hotspots, such as the Scandinavian and Russian Taiga (lakes and rivers) and the Northwest Russia-Novaya Zemlya Tundra (lakes), have historically warmer temperatures (Figure 5-5) and are primarily not underlain by permafrost (Figure 5-6), which may have contributed to the higher diversity noted in these ecoregions for several FECs.


Sheenjek River, Alaska
Photo: Alexis Bonogofsky, USWFS

Warming water temperatures in Arctic rivers and lakes may lead to an increase in biodiversity, as southern benthic macroinvertebrate and fish species expand their range (Heino et al. 2009), but cold stenotherms may face extirpation from waters that are above their thermal tolerance (Wrona et al. 2006a). Changing temperatures in combination with increased human development have the potential to increase the risk of invasive species, which may significantly affect biodiversity of Arctic lakes and rivers through loss of native species. For example, *Bythotrephes longimanus*, also called the spiny water flea, is a native species in Eurasia, that has affected diversity in North America since its introduction to the Great Lakes in 1982, after which it rapidly dispersed (Yan et al. 1992, Strecker et al. 2006). Its distribution in the Arctic is restricted to Eurasia according to our dataset, but a northward dispersal on the North American continent could be possible in the future, particularly if climate change improves suitability of northern lake habitats and if human population growth in the north provides additional means for transport and introduction to these systems. In addition, a warmer and wetter climate will increase rates of mineral weathering, decomposition of soil organic matter, erosion and sedimentation as permafrost thaws and the permafrost extent shifts (Figure 5-6) that likely will lead to higher concentrations of dissolved organic matter (humic compounds), minerals (e.g. Ca, Mg), and nutrients (N and P) (Nadelhoffer et al. 1997). Such change in key drivers of the freshwater environment can affect large-scale processes (e.g., brownification, nutrient enrichment, sedimentation) and biodiversity in lake and river ecosystems as outlined in the impact hypotheses developed in the Freshwater Plan (Culp et al. 2012a).

Brownification of lakes was detected in the early 1990s by Forsberg (1992) and can be partly attributed to climate change (Graneli 2012), but is also a consequence of declines in acid precipitation that affect soil processes and lead to higher pH of run-off (Evans et al. 2006, de Wit et al. 2007, Monteith et al. 2007, de Wit et al. 2016). Brownification is a large-scale process in many areas of northern Europe (e.g., Erlandsson et al. 2008) and North America (e.g., Keller et al. 2008) that is related to permafrost thaw and the release of old carbon. This process reduces light penetration through the water column, thereby affecting primary production (Karlsson et al. 2009) and the distribution of submersed water plants (Mormul et al. 2012), with knock-on effects for consumers (i.e., reduced food resources for grazers with repercussions for fish production). Although brownification mostly has been described for temperate and boreal freshwater ecosystems, increased export of dissolved carbon from thawing soil layers has been observed in the Arctic (Wauthy et al. 2017). Increased nutrient transport from catchment soils to rivers and lakes contributes to increased algal production. This could affect biodiversity in rivers and lakes by changing the benthic habitat (e.g., through increased bryophyte growth) or through changes in the food web (Welch et al. 1989, Jorgenson et al. 1992, Slavik et al. 2004). In the pelagic zone of lakes, nutrient enrichment can result in the shading of benthic algae and water plants. While these effects are currently uncommon in Arctic lakes, they may become particularly severe in the future if, for example, cyanobacterial blooms are induced in N-limited lakes that receive increasing loads of nutrients. This is because cyanobacteria tend to accumulate in high densities at the water surface leading to shading primary producers in

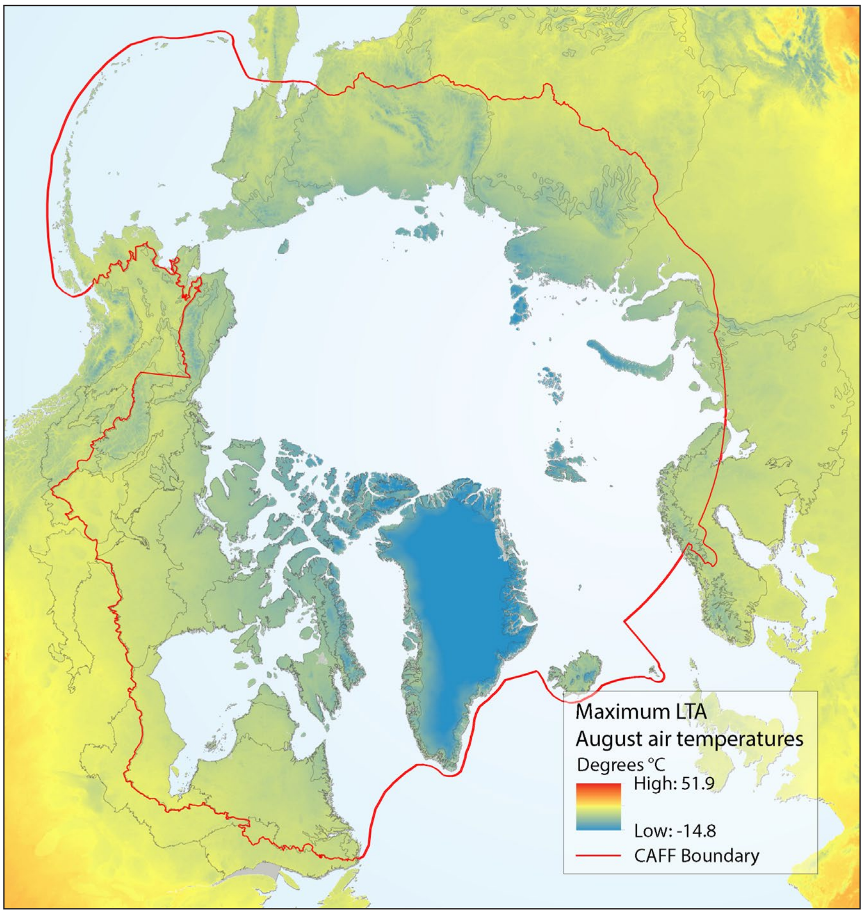


Figure 5-5 Maximum LTA (long-term average) August air temperatures for the circumpolar region, with ecoregions used in the analysis of the SAFBR outlined in black. Source for temperature layer: Fick and Hijmans (2017). Source for ecoregions: Olson et al. 2001

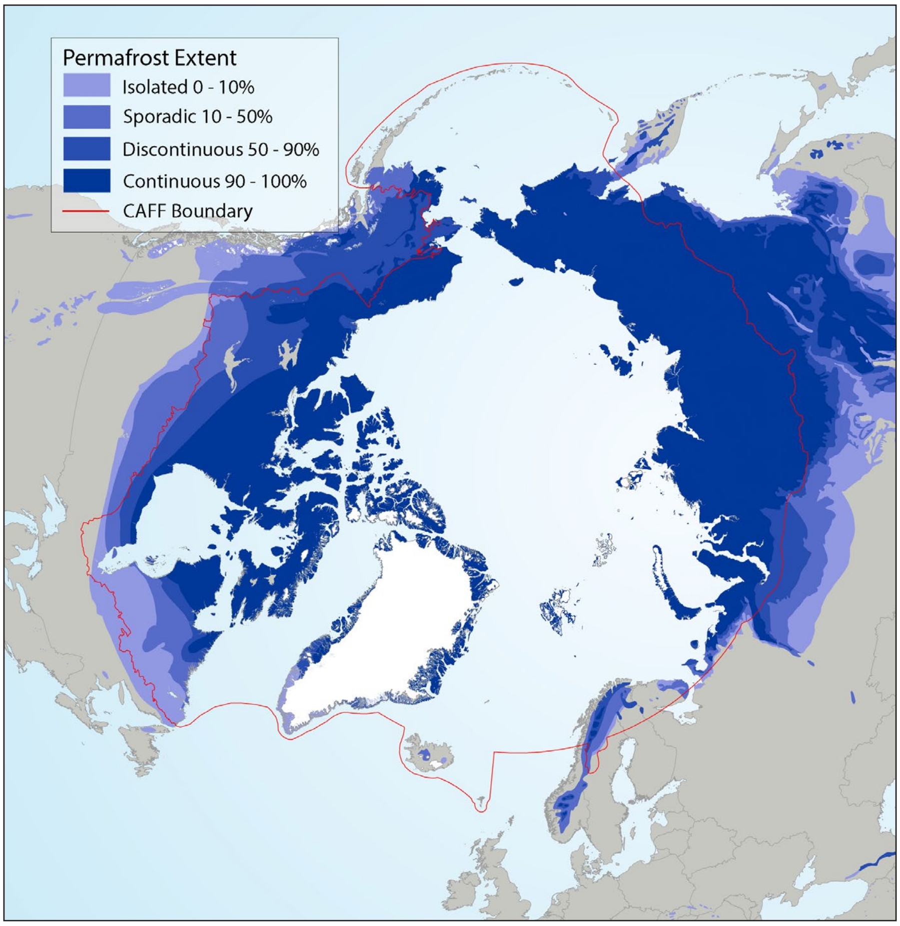


Figure 5-6 Circumpolar permafrost extent overlain on ecoregions used in SAFBR analysis, indicating continuous (90-100%), discontinuous (50-90%), sporadic (10-50%), and isolated (0-10%) permafrost extent. Source for permafrost layer: Brown et al. (2002).

deeper water strata and benthic habitats (cf. Scheffer 1989). Cyanobacteria further affect food webs as they are a poor food source (e.g., Ravet et al. 2003) and many taxa produce toxins (Christoffersen 1996).

Photo: Joseph Culp

In contrast to the effect of increased transport of nutrients from catchments to rivers and lakes, large-scale climate- and human-induced changes in landscapes can contribute to the documented oligotrophication of many northern lakes (Yan et al. 2008, Arvola et al. 2011, Huser et al. 2018) and potentially to rivers. For example, Huser et al. (2018) reported dramatic declines in total-P concentrations of many Swedish lakes since the mid-1990s. Large-scale catchment processes that contribute to reductions in nutrient runoff are (i) the observed changes in tundra vegetation cover, a.k.a. the "*Greening of the Arctic*" (Pouliot et al. 2009, Elmendorf et al. 2012) mediated by elevated N-mineralization and increased nutrient uptake by rooted plants (Aerts et al. 2006), (ii) the more efficient trapping of P that originates from soil pH increases (Gérard 2016), and (iii) low and declining trends in N-deposition (Karlsson et al. 2018). The concerted action of these large-scale changes contributes to the gradual transformation of northern freshwaters toward more nutrient-poor conditions and in lakes is expected to increase in the predominance of N2-fixing cyanobacteria. Furthermore, hydropower development and damming of large northern rivers can cause declines in nutrients downstream of dams and can affect species richness by providing barriers to dispersal.

Sediment load in freshwater systems is predicted to intensify with further permafrost degradation as temperatures and precipitation increase in many Arctic regions (Syvitski 2002, Kokelj et al. 2013, Kokelj et al. 2015). Hillslope thermokarst is common throughout northern Russia, northern Alaska (the Arctic Coastal Tundra and Arctic Foothills Tundra ecoregions), and in the northwestern Canadian Arctic (Figure 5-7). Among the most dramatic thermokarst features are retrogressive thaw slumps – large depressions of exposed permafrost

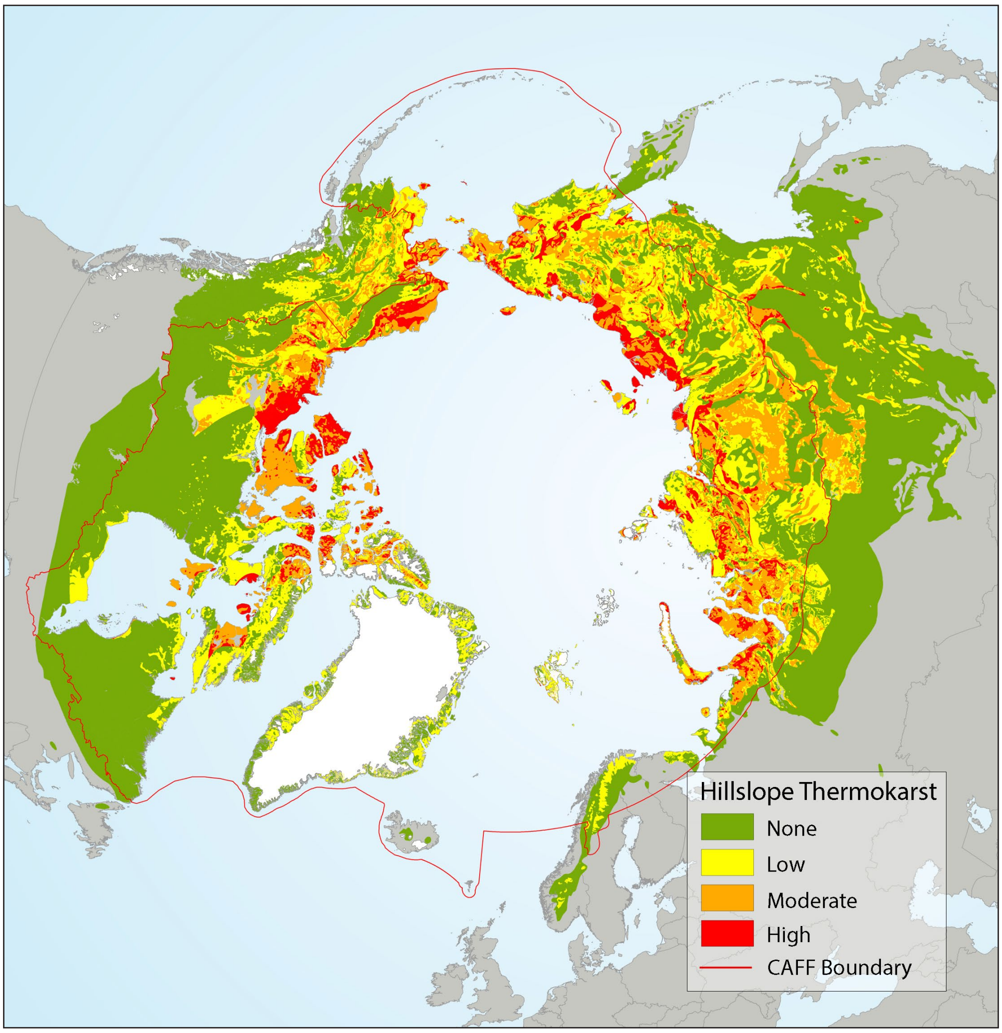


Figure 5-7 Spatial distribution of hillslope thermokarst across the circumpolar area, overlain with ecoregions used in the SAFBR analysis, showing no, low, moderate, and high thermokarst. Source for thermokarst layer: Olefeldt et al. (2016)

(see Figure 3-4a) that have increased in size and frequency on landscapes underlain by ice-rich permafrost (Lantz and Kokelj 2008). Retrogressive thaw slumps form in areas where permafrost is embedded with large masses of buried ice, such as the Peel Plateau,

Canada (Kokelj et al. 2013), the Brooks Range and foothills in Alaska (Jorgenson et al. 2006, Balser et al. 2014) and the northern Taymyr Peninsula in Siberia (Ulrich et al. 2010), with formation and growth driven primarily by precipitation events (Kokelj et al. 2015) and by warm air temperatures (Lacelle et al. 2010). Found along the shores of lakes and coastlines, river valleys, they can exceed 5 ha in area (Kokelj et al. 2013) and significantly impact aquatic ecosystems. As slump headwalls collapse, the resultant debris forms a mud slurry that can flow into nearby lakes and rivers (Figure 3-4). In lakes, this slurry settles to the bottom, increases pH and conductivity, and ultimately leads to higher water transparency (Mesquita et al. 2010). Biological effects observed in lakes include increased biomass of diatoms (Thienpont et al. 2013) and macrophytes (Mesquita et al. 2010), greater benthic invertebrate abundance and shifts in community composition (Moquin et al. 2014). In rivers, thaw slumps increase total nutrient and solute concentrations as well as sediment loads (Kokelj et al. 2013). Less intensive thermokarst disturbances may initially increase algal and invertebrate abundance in rivers if positive nutrient effects offset the negative impacts of sediments (Bowden et al. 2008, Levenstein 2016). However, large thaw slumps negatively impact riverine biodiversity by decreasing benthic algal biomass (Levenstein et al. 2018) and invertebrate abundance (Chin et al. 2016), and by increasing invertebrate drift (Levenstein 2016).

5.4. Conclusions

This chapter identifies and provides examples of how higher temperatures, as the overriding driver, and other environmental variables of ecological change affect the water quality and biodiversity of Arctic freshwaters. These examples show both large-scale, slowly progressing landscape-level processes that will have long-lasting effects, as well as rapid modifications that have more local and short-term effects. The concerted action of these environmental drivers and their subsequent and cumulative effects on biological assemblages will depend on regional conditions. Slow response times will make some of these processes progress for decades to come, while others may induce sudden biological shifts with strong repercussions on aquatic ecosystems when critical threshold levels are exceeded and functional redundancy is low. Examples of this are shifts to strong N-limitation, which will induce cyanobacterial blooms that will cause substantial shading for other primary producers, or permafrost degradation that may drain entire lakes. Furthermore, increases in the rate of glacier retreat cause changes to hydrologic, thermal, and sediment regimes that can lead to significant changes in taxonomic an functional diversity of freshwater systems receiving glacial inflows (Brown and Milner 2012, Milner et al. 2017). In addition, the loss of species with low physiological tolerances for higher temperatures and the northward movement of species tolerant of a broader range of temperature will affect alpha and beta biodiversity.


Photo: Daryl Halliwell

In our analysis, we did not directly identify distinct environmental gradients. However, some gradients, such as climatic and/or latitudinal and vegetation gradients, were inherent in our large dataset. Multivariate statistical approaches are useful tools to unravel relationships between biological assemblages and environmental gradients. Our analysis showed marked regional differences in biodiversity of several FECs and apparent changes in alpha diversity along latitudinal gradients. These analyses form the baseline to which future assessment can be compared and partly address some of the impact hypotheses in the freshwater biodiversity monitoring plan (Culp et al. 2012a).

Randy Brown, fish biologist, prepares to catch, tag, and track Dolly Varden in Arctic National Wildlife Refuge
Photo: Katrina Liebich, USFWS

6. State of Arctic Freshwater Monitoring

6.1. Introduction

In this chapter we describe the current status of freshwater monitoring in the Arctic countries, and address possible ways to improve future monitoring in the Arctic including community engagement. Building on the freshwater biodiversity plan published by CAFF in 2012 (Culp et al. 2012a), this first circumpolar assessment of freshwater biodiversity created and analyzed an expansive, circumpolar data set covering paleo, historic, and contemporary data on Arctic freshwater biodiversity. Biodiversity trends were evaluated at the circumpolar ecoregion level and also within regions defined by areas of similar geography, flora and fauna. The assessment also addresses knowledge gaps that limit our ability to conserve and protect freshwater biodiversity in the circum-Arctic countries, and forwards expert guidance on monitoring network design for Arctic freshwater biodiversity.

The availability and use of data for the SAFBR varied among the Focal Ecosystem Components (Table 61). For example, the Fish FEC had a number of parameters with available data (e.g., numbers, relative abundance, total biomass, presence/absence, age and size structure), but data availability and sampling methods varied for each parameter, and presence/absence offered the best and most consistent spatial and temporal coverage. The macrophyte FEC had the fewest potential parameters (as listed in Culp et al. 2012a), and differences across data sources similarly required the use of presence/absence data. Presence/absence information was available for all of the FECs, because other measured parameters could easily be converted to this parameter. Thus, biodiversity analysis for lakes and rivers were completed using relative abundance information when possible, or with presence/absence when required by inconsistencies in sampling methods or parameter measurements.

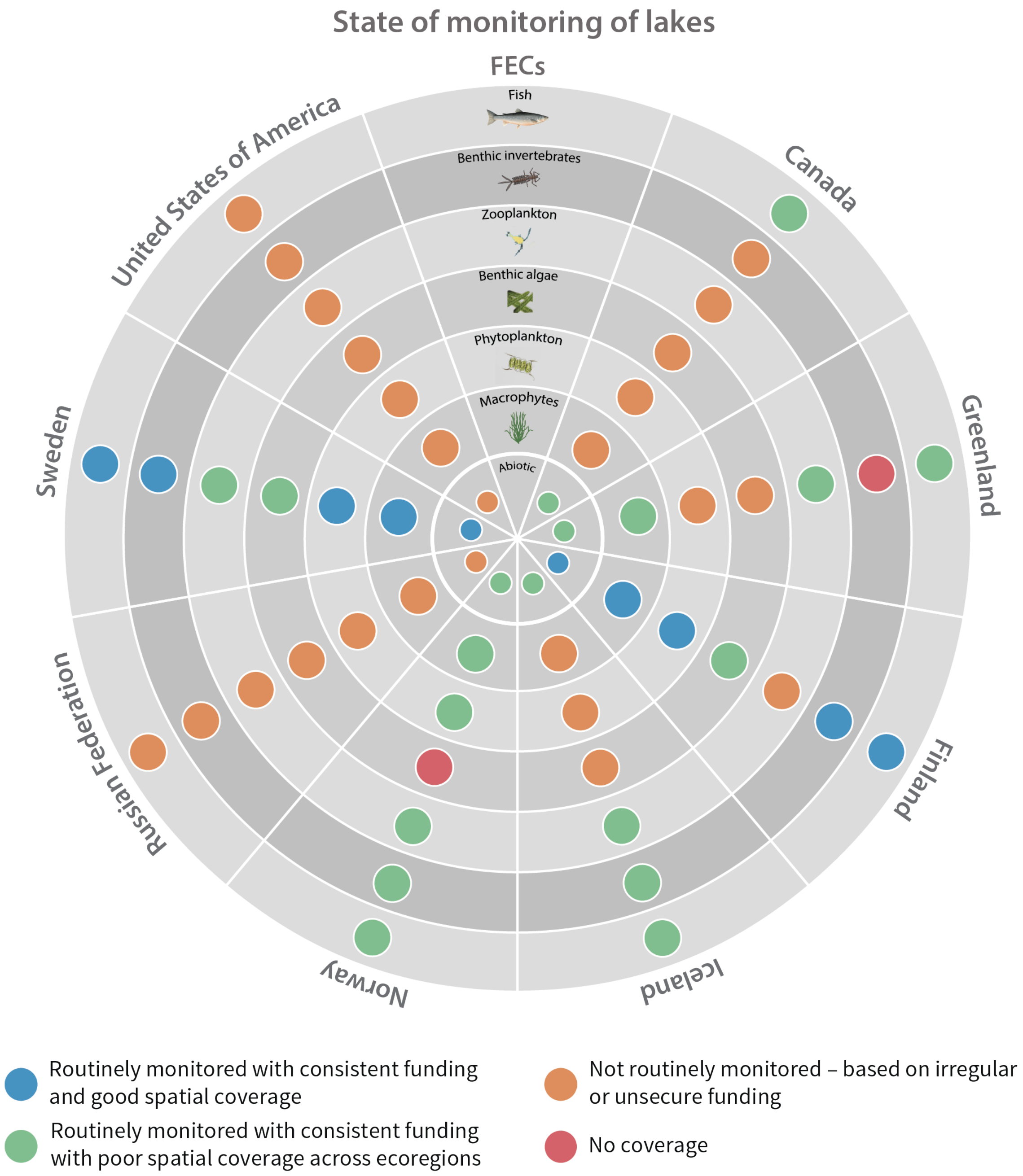


Figure 6-1 Current state of monitoring for lake FECs in each Arctic country.

6.2. Existing Monitoring in Arctic Countries

Although the circumpolar countries endeavor to support monitoring programs that provide good coverage of Arctic and subarctic regions, this ideal is constrained by the high costs associated with repeated sampling of a large set of lakes and rivers in areas that often are very remote. Consequently, freshwater monitoring has sparse, spatial coverage in large parts of the Arctic, with only Fennoscandia and Iceland having extensive monitoring coverage of lakes and streams (Table 6-2, Table 6-3, Figure 6-1, Figure 6-2). In many remote areas of the Arctic, monitoring is often associated with planned or ongoing development projects (e.g., hydropower, mining, oil and gas). Such monitoring can be short-term in nature and focused narrowly on point-source and/or regional effects. In contrast, Fennoscandian freshwaters have time-series information dating back to the 1960s for water chemistry, and to the 1980s for several biological variables; Iceland has long monitoring records of fish populations for some rivers. The more extensive data availability in Europe is, in part, the result of requirements by the European Union for regular reporting on the status of their inland waters, including sub-Arctic lakes and rivers within the CAFF-boundary. The availability of such higher-resolution data is critical for the development of predictive models of biodiversity change in other parts of the Arctic.

The following descriptions of monitoring of Lake and River FECs indicate how the current approaches and data coverage vary by country. Table 6-2 and Table 6-3 provide the details of the current monitoring status for lake and river FECs and their parameters within the eight Arctic countries including an assessment of spatial coverage and funding consistency.

State of monitoring of rivers FECs

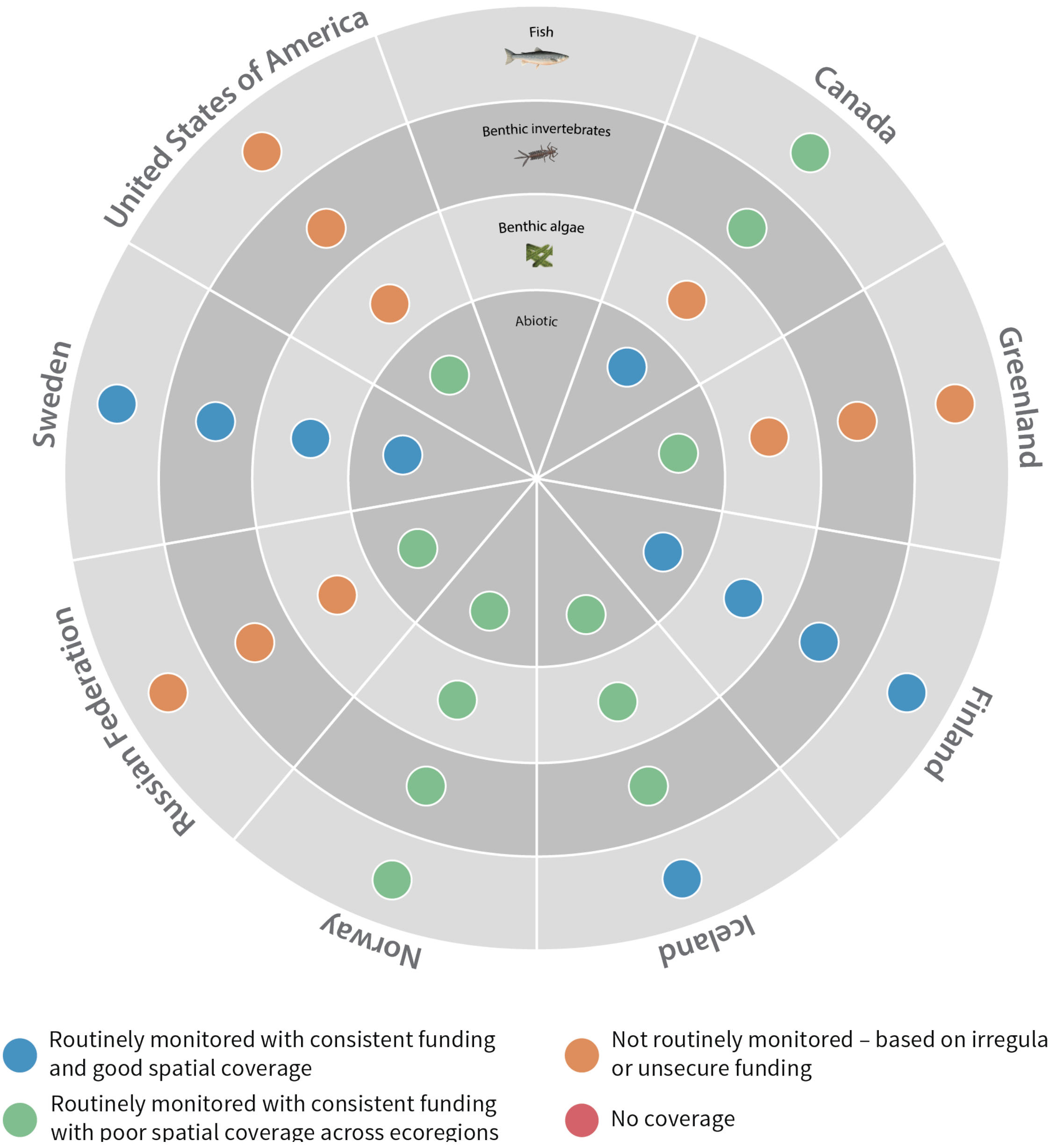


Figure 6-2 Current state of monitoring for river FECs in each Arctic country

Table 6-1 Overview of data availability and use for SAFBR analyses for lakes and rivers, including data from ongoing or past freshwater monitoring programs in the Arctic, as well as data from different sources (academia, industry, NGOs) to increase coverage for some parameters/regions. Values for each parameter indicate that: (1) data were available and were used; (2) some data were available but were not used for the assessment; or (3) there were few or no data available.

FECs and parameters as described in CBMP freshwater monitoring plan**		Data used in SAFBR
Fish	Numbers, relative abundance, total biomass	2
	Presence absence**	1
	Genetic diversity	3
	Size structure	2
	Age structure	2
	Timing of important life history events	3
	Contaminant concentration	3
Benthic invertebrates	Numbers, relative abundance, total biomass	1
	Presence absence**	1
	Size structure	3
	Timing of important life history events	3
	Contaminant concentration	3
Zooplankton	Numbers, relative abundance, total biomass	1
	Presence absence**	1
	Biomass of each taxon	2
	Timing of important life history events	3
Algae from benthic samples	Numbers, relative abundance, total biomass	1
	Presence absence**	1
	Biomass or biovolume of each taxon	2
	Bulk biomass (including chlorophyll a)	2
Phytoplankton	Numbers, relative abundance	2
	Presence absence**	1
	Biomass or biovolume of each taxon	1
	Bulk biomass (including chlorophyll a)	2
Macrophytes	Areal cover, distribution or number of individuals of each taxon	2
	Presence absence**	1

** Aquatic birds were originally included in the Freshwater Monitoring plan, but were taken out and instead included in the CBMP Terrestrial Monitoring Plan*

*** Not explicitly listed in the monitoring plan but necessitated by available data*

Photo: USWFS

6.2.1 USA

Researchers collect fish in the Agashashok River, Alaska
Photo: Michael Records, USGS

Freshwater monitoring in the Arctic region of the USA (i.e., Alaska) is limited in scope (Table 6-2, Table 6-3, Figure 6-1, Figure 6-2). Data are available for each of the FECs, yet little overlap in FEC data distribution occurs given the independence of projects and data collection. Monitoring data availability depends on the goals of the agency or group undertaking collection. Data are collected by the National Park Service Inventory and Monitoring network, the U.S. Geological Survey, U.S. Bureau of Land Management, U.S. Fish and Wildlife Service, Alaska state agencies, and academic programs.

Arctic biodiversity monitoring does take place in the country's National Parks located in the low Arctic zone in Alaska. The National Park Service Arctic and Central Alaska Inventory and Monitoring Networks (ARCN) monitor stream and lake communities and ecosystems in the Bering Land Bridge National Preserve, Cape Krusenstern National Monument, Noatak National Preserve, Kobuk Valley National Park, and Gates of the Arctic National Park and Preserve (science.nature.nps.gov/im/units/arcn/index.cfm). Since 2007, the ARCN has monitored water quality (temperature, turbidity, pH, and dissolved oxygen), macrophyte vegetation, and species richness/ abundance of benthic macroinvertebrates of shallow lakes. More recently (< 5 years), National Park Service initiated monitoring of water quality in streams, and in the coming decades they plan to begin monitoring long-term trends on chemical (e.g., pH, dissolved oxygen), physical (e.g., water levels, temperature), and biological (e.g., phytoplankton, zooplankton) characteristics of large lake ecosystems in Gates of the Arctic National Park and Preserve.

Outside of the National Parks, many FECs are not routinely monitored, and efforts to do so are often based on irregular or unsecure funding. Recent collaborative projects (www.fishcreekwatershed.org) between the University of Alaska Fairbanks and federal agencies aim to establish a baseline for fish habitat in streams and lakes, and include monitoring of biotic FECs (fish, benthic macroinvertebrates, zooplankton, and phytoplankton) and abiotic FECs (discharge, temperature, water quality). The Arctic Long-Term Ecological Research Site (LTER) uses long-term monitoring along with surveys and experiments with the goal of predicting Arctic ecosystem response to environmental change (arc-lter.ecosystems.mbl.edu). Numerous FECs have been sampled in the LTER site, including fish, benthic macroinvertebrates, zooplankton, phytoplankton, and physics/chemistry. Additional monitoring may also occur as part of mandated sampling conducted in conjunction with mining or energy extraction, but is localized in specific systems impacted by development.

6.2.2. Canada

Sampling in northern Canada
Photo: Daryl Hallowell

Monitoring of lakes and rivers in the Canadian Arctic is necessarily limited by the enormous spatial expanse of this sub-Arctic to high Arctic region encompassing drainage areas of the Pacific, Arctic, and Atlantic oceans as well as Hudson Bay. Freshwater biodiversity, water quality and water quantity data are collected to meet various federal commitments related to transboundary watersheds crossing international, inter-provincial and territorial borders, or under various other regulatory authorities. Data are collected by territorial governments (Yukon, Northwest Territories, Nunavut), provincial jurisdictions (Quebec, Labrador and Newfoundland) and federal departments (e.g., Fisheries and Oceans Canada, Environment and Climate Change Canada – ECCC, and Parks Canada). Other biodiversity information is collected opportunistically through industrial and academic research programs.

Ongoing freshwater biodiversity monitoring is generally limited to fish populations and benthic macroinvertebrates (Table 6-2, Table 6-3, Figure 6-1, Figure 6-2). Fish monitoring is restricted to federal and territorial stock assessments. Monitoring of macroinvertebrates occurs through development of the Canadian Aquatic Biomonitoring Network (CABIN) that has a standardized sampling protocol and assessment approach for assessing aquatic ecosystem condition. Arctic sampling is restricted to northern parts of the Yukon, the Northwest Territories and to locations where industry and academic research programs can provide monitoring synergies. This opportunistic sampling strategy increases spatial coverage of macroinvertebrate sampling but provides limited data for time series trends. Routine monitoring of macroinvertebrates is currently undertaken or under development by Parks Canada and some territorial governments (e.g., Government of Northwest Territories), though existing datasets have limited time series. Some additional monitoring of diatoms in lakes and rivers is currently ongoing in northern Quebec, but this represents an extremely limited geographic area of the Canadian Arctic.

Canada's long-term water quality and water quantify monitoring have contrasting spatial and temporal coverage. The water quality network is administered by ECCC along with territorial partners and Parks Canada, and only includes 46 sites across the North. Parameters regularly measured include temperature, pH, alkalinity, major ions, nutrients and metals. Sampling frequency varies from one to more than 6 times per year as sampling is adjusted according to a risk-based, adaptive management framework. Most sites have been monitored for the last 15 years or longer. In contrast, water quantity monitoring by the Water Survey of Canada is quite extensive with more than 100 sites where water levels are continuously recorded. While most sites are situated in the Subarctic region, several locations in the high Arctic are monitored. In the southern Arctic there are many sites with records exceeding 50 years; however, the relatively few sites in the high Arctic have been monitored for less than 25 years.

6.2.3. Kingdom of Denmark/Greenland/Faroe Islands

Zackenberg light measurement
Photo: Kirsten S. Christoffersen

Monitoring in Greenland focuses on lake ecosystems (Table 6-2, Table 6-3, Figure 6-1, Figure 6-2) as a component of the Greenland Ecosystem Monitoring (GEM) program (http://g-e-m.dk/). GEM is a joint effort of Denmark and Greenland institutions on behalf of the Kingdom of Denmark and includes a sub-program on Arctic biota (BioBasis). The BioBasis program monitors the dynamics of organisms and biological processes in terrestrial and freshwater ecosystems. These efforts have established a coherent and integrated understanding of the functioning of ecosystems in a highly variable climate based upon a comprehensive, long-term interdisciplinary data collection primarily at Kobbefjord (Nuuk) in low Arctic West Greenland and Zackenberg in high Arctic Northeast Greenland. Recently, a field site was added at the southern part of the Disko Island near the high Arctic. Each location has a state-of-the-art field station. The Faroe Islands has no biodiversity monitoring program for freshwaters, but freshwater sources used for drinking water and/or in fish farming are surveyed with respect to contaminants.

The GEM-BioBasis program incorporates monitoring and long-term research on ecosystems to understand climate change effects and related ecosystem feedbacks in the Arctic. Monitoring of freshwater includes biotic and abiotic dynamics, including biodiversity and phenology of phyto- and zooplankton, fish abundance and water chemistry. Sampling is typically performed on 2-6 dates during the ice-free period and occasionally during ice-cover. The longest time series exists for Zackenberg where two lakes have been sampled since 1997. Detailed protocols for the sampling and analysis program can be found on the home pages of the field sites, and data are publicly available through a data portal indexed on the GEM webpage.

6.2.4. Iceland

Sampling of macroinverterbrates in River Jokulsá á Dal, Iceland.
Photo: Guðni Guðbergsson

Freshwater monitoring in Iceland is chiefly based on three categories: (1) short term research projects including industrial or impact assessments, (2) long-term projects on productive and/or species rich areas, and (3) monitoring of rivers and lakes with harvested freshwater fish stocks. Freshwater monitoring is largely conducted by governmental institutions and has produced a good database with many parameters routinely collected (Table 6-2, Table 6-3, Figure 6-1, Figure 6-2). Iceland has a systematic monitoring of abiotic factors such as water discharge and glacier coverage. In the near future, Iceland will initiate monitoring of freshwater ecosystems to adopt the requirements set by the Water Framework Directive of the European Union. These new monitoring initiatives will add to existing monitoring and increase the number and geographical coverage of FECs. The monitoring programs in Iceland are almost exclusively dependent on governmental financing. Currently, no national monitoring database exists, but metadata compilation for freshwater research has been established in relation to the work in CBMP.

6.2.5. Norway

Monitoring station for waterflow
Photo: John Brittain

The Norwegian Arctic consists of 1) mainland Norway and the Norwegian portion of the Scandinavian peninsula north of the Polar circle, and (2) the remote islands Svalbard and Bjørnøya in the Barents Sea and Jan Mayen in the Norwegian Sea. Routine monitoring is undertaken but not on a broad spatial scale (Table 6-2, Table 6-3, Figure 6-1, Figure 6-2). In mainland Norway, monitoring includes physical, chemical, and biological FECs in both rivers and lakes. The primary purpose is to monitor pollution from industry and physical encroachments, including changes in discharge in connection with hydropower plants. In recent years, a number of reference sites and large lakes have also been monitored as a part of a national monitoring network. On Svalbard the only regular monitoring of freshwater includes two stations for discharge and transport of suspended matter. There are neither hydrochemical nor biological monitoring of freshwater sites in the Svalbard archipelago. However, research programs, specific studies, and student courses have generated some biological data from Svalbard freshwater sites. A few sites have data from multiple years, and these sites will be valuable for designing a future monitoring network on Svalbard. Except for the data collected by the hydrological stations, there are no time series from freshwater in Svalbard. There is also no ongoing monitoring of water chemistry or biota on Bjørnøya, although one lake on Bjørnøya has been studied for elevated levels of organic substances in fish and water birds.

During the last 4 years, monitoring of lakes and rivers has increased substantially in mainland Norway. New monitoring programs have been initiated, following the requirements of the EU Water Framework Directive (WFD). The FECs used in CBMP-Freshwater are similar to the WFD's Biological Quality Elements (phytoplankton, water plants, macroinvertebrates and fish), which are all included in Norwegian freshwater monitoring. The Svalbard archipelago is not a part of the EU-Norway EEA-agreement, but similar monitoring activities as on mainland Norway are also planned for this high-Arctic region. Linnévatnet in western Spitzbergen, Svalbard, was monitored in 2017 as test of traditional monitoring methods in the High Arctic. This lake has been sampled occasionally during the last 40 years, which makes it possible to document effects of climate change. Lake Linnévatnet, a couple of lakes in NY-Ålesund area and river sites in the same area are good candidates for future monitoring sites in the High Arctic. These sites would be good candidates for a circumpolar monitoring network for freshwater based on a hub-and-spoke principle as proposed in section 6.3.3.1.Within the low Arctic, the interior of Finnmark, in northern Norway, harbours an enhanced freshwater biodiversity compared to the coastal areas. This is due to the more continental climate, with a wide range in annual temperature, and dispersal from the east and southeast after the last Ice Age. This high biodiversity needs to be monitored in the light of the rapid environmental changes now taking place in the Arctic. Monitoring data from this region from the last decades make analyses of recent climate change effects possible.

6.2.6. Sweden


CBMP Freshwater co-lead Willem Goedkoop
Photo: Joseph Culp

National monitoring programs of lakes and streams/rivers, covering both biological and abiotic FECs, have been coordinated by the Swedish EPA (until 2011) and by the Swedish Agency for Marine and Water Management (2011–), while regional programs have been coordinated by the various county boards. These nationwide programs also cover the northern boreal forests, birch forests, and sub-Arctic mountainous regions of Sweden that all fall within the CAFF border. Swedish national monitoring has a good spatial and coverage within CAFF border of the Arctic (Table 6-2, Table 6-3, Figure 6-1, Figure 6-2). Monitoring of water quality in major rivers and the three largest lakes has been completed since the 1960s. The current national monitoring program started in 1988 and originally had a strong focus on the effects of acidification. The program successively grew to also incorporate streams and more lakes (in 1995). Sweden has also performed a number of synoptic, national surveys to map water quality of lakes across the country, starting in 1972 and repeated with approximately 5-y intervals. The two national surveys of lakes and streams, performed in 1995 and 2000, also included the sampling of benthic macroinvertebrates in some 700 lakes and 700 streams across the nation, among which many are in the Arctic/alpine ecoregion of Sweden.

In recent years, the national monitoring program has been gradually modified and adapted to better fit the requirements of the European Water Framework Directive. It now consists of a Trend Rivers program (67 watercourses evenly distributed across Sweden), and a Trend Lakes

Photo: USFWS

program (106 lakes) in which water chemistry and biotic samples are collected once or multiple times per year, though frequency of fish sampling is lower (29 Trend Rivers sampled annually, 45 Trend Lakes sampled at least every six years). Many of the water bodies in these programs have been sampled since 1988 or the mid 1990's. Sweden has a Rivers Outlet program (47 major rivers) that has monitored water chemistry since the 1960's. The Department of Aquatic Sciences and Assessment at the Swedish University of Agricultural Sciences has a responsibility for national monitoring, is the data host for freshwater data, and works closely with central authorities and many regional county boards. More detailed descriptions of the national programs can be found at the department's home page https://www.slu.se/en/departments/aquatic-sciences-assessment/ under the entry for Environmental Assessment. The history and current status of freshwater monitoring in Sweden has been described in detail by Fölster et al. (2014).

6.2.7. Finland


Benthic invertebrate sampling in Finnish Lapland.
Photo: Petri Liljaniemi

Freshwater monitoring is primarily carried out as part of national programs coordinated by the Ministry of the Environment, regional environmental administration and research institutes (Finnish Environment Institute and Natural Resources Institute) resulting in many parameters collected on a broad spatial scale (Table 6-2, Table 6-3, Figure 6-1, Figure 6-2). National programs for rivers and lakes are designed to monitor the long-term trends in different types of regions, including sub-Arctic headwaters. Monitoring programs include a set of biotic and abiotic FECs. Freshwater monitoring is also conducted by the private sector in order to fulfill the monitoring obligations of environmental permits (e.g., sewage treatment plants, industry, and mines). Research institutes and universities also produce data through short-term research projects.

The early biological data come from separate studies. More uniform long-term biological time-series exist from 1980-2000's for several components. Monitored biological components include fish, benthic macroinvertebrates, benthic diatoms, phytoplankton and macrophytes (e.g., species identification and abundance, community structure and some biomass estimates). The longest biological time series are for fish communities and these extend more than 30 years for large sub-Arctic rivers. At present, intensive, yearly fish monitoring includes population estimates and catch statistics for the border rivers Tornio and Teno, and Lake Inarijärvi.

Spatial coverage of especially biological monitoring was significantly expanded to meet the requirements of the EU Water Framework Directive during years 2006-2009. The largest rivers and lakes are monitored at least annually for water quality and biological components. Otherwise monitoring frequency alternates from once in every three to six years. In addition, some research programs collect yearly water quality data from small sub-Arctic rivers and lakes. The results are stored in national databases for water quality and species data. Data registers are designed to meet the needs of the reporting of ecological status for the EU. The Finnish monitoring program has a good coverage for most biological FEC's, and only zooplankton is not routinely monitored (Table 6-2, Table 6-3, Figure 6-1, Figure 6-2).

Water quality indicators include a variety of substances, from basic nutrients to heavy metals and toxins, with the longest time series extending from the 1960s. Additional abiotic monitoring covers hydrological and meteorological variables, such as discharge, water level, precipitation, and snow. Land cover variables including soil/bedrock properties, and human impact (e.g., loading, hydro-morphological alterations) are also available.

6.2.8. Russian Federation


Anadyr River, Eastern Russian Arctic
Photo: Andrei Stepanov/Shutterstock.com

Monitoring of the Arctic regions in the Russian Federation varies across the country, and is often associated with research activities of Russian Academy of Sciences institutions and State Nature Reserves; thus, routine sampling is not widely available (Table 6-2, Table 6-3, Figure 6-1, Figure 6-2). Where long-term monitoring activities exist, they are directed by the research interests of the scientific organizations, with the results published as scientific reports, publications, and less often, in open access sources. In addition, the state of natural aquatic ecosystems is monitored for potential impacts before and after human disturbances such as industrial activities. This impact-driven assessment sometimes includes longer-term monitoring activity. Monitoring of freshwaters includes both chemical and biological components of freshwater ecosystems.

Monitoring of Arctic freshwater ecosystems is directed by water quality standards of the All-Union State Standard (All-Union State Standard, 2018). This standard uses the concept of Maximum Allowable Concentrations (MAC) of chemical elements and their compounds in the environment that can affect human health or cause pathological changes or diseases. MACs have been established for more than 1300 substances, and can be specific for different climate zones. MACs for fishery purposes are intended for quality control of water in reservoirs of fishery (aquaculture) and agriculture (irrigation) purposes.

Table 6-2 Current monitoring status for Lake FECs and their parameters within the eight Arctic countries. The listed values indicate that monitoring of the FEC parameter is: (1: blue) routine with consistent funding and good spatial coverage across ecoregions; (2: green) routine with consistent funding but without good spatial coverage across ecoregions; (3: orange) not routine because funding is sporadic and not secure; (4: grey) unknown; or (5: red) not undertaken.

FECs and Parameters*		US	Canada	Greenland	Iceland	Norway	Sweden	Finland	Russia
Fish	Numbers, relative abundance, total biomass	2	2	2	2	2	1	1	3
	Presence absence**	2	2	2	2	2	1	1	3
	Genetic diversity	3	5	3	3	3	3	3	3
	Size structure	2	2	2	2	2	1	1	3
	Age structure	2	2	2	2	3	1	1	3
	Timing of important life history events	3	5	3	3	3	5	1	3
	Contaminant concentration	3	4	–	2	3	2	2	3
Benthic invertebrates	Numbers, relative abundance, total biomass	3	5	2	2	2	1	1	3
	Presence absence**	3	5	2	2	2	1	1	3
	Size structure	3	5	5	5	3	5	3	4
	Timing of important life history events	4	5	2	5	3	5	3	3
	Contaminant concentration	3	5	5	5	3	5	3	4
Zooplankton	Numbers, relative abundance, total biomass	3	2	2	2	2	2	3	3
	Presence absence**	3	2	2	2	2	2	3	3
	Biomass of each taxon	3	2	4	5	3	5	4	3
Algae from benthic samples	Numbers, relative abundance	3	3	3	5	2	2	1	3
	Presence absence**	3	3	3	5	2	2	1	3
	Biomass or biovolume of each taxon	3	3	5	5	3	3	3	3
	Bulk biomass (incl. chlorophyll a)	2	3	3	5	3	4	3	4
Phytoplankton	Numbers, relative abundance	3	3	3	2	2	1	1	3
	Presence absence**	3	2	3	2	2	1	1	3
	Biomass or biovolume of each taxon	3	2	5	2	2	1	2	3
	Bulk biomass (including chlorophyll a)	3	2	3	2	2	1	1	3
Macrophytes	Areal cover, distribution or number of individuals of each taxon	3	2	3	2	2	1	1	3
	Presence absence**	3	2	3	2	2	1	1	3
Physics and Chemistry	Physics	2	2	2	2	3	2	1	3
	Chemistry	2	2	2	2	2	1	1	3

** as described in CBMP freshwater monitoring plan,*
*** Not explicitly listed in the monitoring plan but necessitated by available data,*
**** Routinely monitored over very small spatial extents as part of mandatory monitoring by industry*

Table 6-3 Current monitoring status for River FECs and their parameters within the eight Arctic countries. The values indicate that monitoring of the FEC parameter is: (1) routine with consistent funding and good spatial coverage across ecoregions; (2) routine with consistent funding but without good spatial coverage across ecoregions; (3) not routine because funding is sporadic and not secure; (4) unknown; or (5) not undertaken.

FECs and Parameters*		US	Canada	Greenland	Iceland	Norway	Sweden	Finland	Russia
Fish	Numbers, relative abundance, total biomass	3	2	3	1	2	1	1	3
	Presence absence**	3	2	3	1	2	1	1	3
	Genetic diversity	3	3	3	3	2	3	3	3
	Size structure	3	2	3	1	1	1	1	3
	Age structure	3	2	3	1	1	1	1	3
	Timing of important life history events	3	3	5	4	3	4	1	3
	Contaminant concentration	4	3	2	4	2	4	2	3
Benthic invertebrates	Numbers, relative abundance, (total biomass)	3	2	3	2	2	1	1	3
	Presence absence**	3	2	3	2	2	1	1	3
	Size structure	3	3	3	5	5	5	3	3
	Timing of important life history events	3	4	5	3	3	5	3	3
	Contaminant concentration	4	3	5	5	5	5	3	4
Algae from benthic samples	Numbers, relative abundance	3	3	3	2	2	1	1	3
	Presence absence**	3	3	3	2	2	1	1	3
	Biomass or biovolume of each taxon	3	3	3	3	5	4	3	3
	Bulk biomass (incl. chlorophyll a)	3	2	3	2	4	3	3	4
Physics and Chemistry	Physics	2	2	2	1	1	2	1	2
	Chemistry	3	1	2	2	2	1	1	3

** as described in CBMP freshwater monitoring plan,*
*** Not explicitly listed in the monitoring plan but necessitated by available data,*
**** Routinely monitored over very small spatial extents as part of mandatory monitoring by industry*

6.3. Advice for Future Monitoring of Arctic Freshwater Biodiversity

This assessment has made substantial progress towards the development of a harmonized program for monitoring the state of Arctic freshwater biodiversity. However, future assessments will be improved through the adoption of the advice detailed below that identifies potential improvements in monitoring design, sampling methods and assessment approaches. Monitoring of Arctic freshwater biodiversity will benefit from such improvements and the use of an adaptive monitoring strategy.

6.3.1. Emerging Approaches

6.3.1.1. Incorporation of Traditional Knowledge

The incorporation of Traditional Knowledge (TK) and the engagement of Indigenous Peoples through observational networks and monitoring efforts can make an important contribution to freshwater biodiversity monitoring in the Arctic. A systematic literature review of TK indicated the prevalence of themes related to fish abundance, health, and quality, water quality, and changes to lake and river ice (Figure 63). Observations related to these themes, many of which are from the historical period prior to 1950 for which monitoring records are scarce, have the potential to fill gaps in our knowledge of biodiversity changes. Indigenous Peoples have a unique and holistic perspective on the landscape that spans multiple generations. The increased contribution of TK has enormous potential for future monitoring of biodiversity and the detection of ecological change in Arctic lakes and rivers. **We recommend the inclusion of Traditional Knowledge as an integral part of future circumpolar monitoring assessments and networks**.

6.3.1.2. Increased Use of Citizen Science

Citizen Science and the engagement of northern residents through observational networks and monitoring efforts has the potential to increase our capacity to monitor ongoing changes in Arctic freshwaters. Many of the barriers to monitoring in Arctic systems relate to logistical difficulties associated with travelling to the Arctic to conduct sampling. Engagement of local communities has the potential to reduce the financial burden associated with repeated visits to northern locations, Furthermore, this approach would provide opportunities to monitor the local/regional effects of human settlements and activities in the landscape (e.g., exploration, roads). For example, Finland uses educational programs involving children ranging from 7 to 17 years of age to collect various observations on freshwater ecosystems, and Canada is expanding the Canadian Aquatic Biomonitoring Network (CABIN) to include northern communities. Increased engagement of northern residents in Citizen Science has strong potential to support future monitoring activities and the detection of ecological change in Arctic lakes and rivers. Therefore, **we recommend the engagement of local communities in monitoring efforts through Citizen Science efforts as an integral part of future circumpolar monitoring networks**. These efforts could include adoption of existing platforms (e.g., iNaturalist) and programs (e.g., Globe program for schools) to quickly implement citizen science programs in northern regions.

6.3.1.3. Use of Remote Sensing Approaches

We also **recommend that this monitoring network includes an increased focus and use of remote sensing approaches (e.g., satellite imagery) and automated measuring devices (e.g. in situ data sensors)** for better coverage of chemical and physical supporting variables, and for estimating biodiversity in freshwaters, as allowed by future technological advances. Satellite images and aerial photography are efficient approaches to monitor indicators of landscape change and habitat variables (e.g., ice cover duration, snow cover, lake connectivity, turbidity of large lakes, terrestrial vegetation changes) across large spatial scales. In coming years, we expect an increased use of remote sensing in monitoring of freshwater, especially for lakes. This will

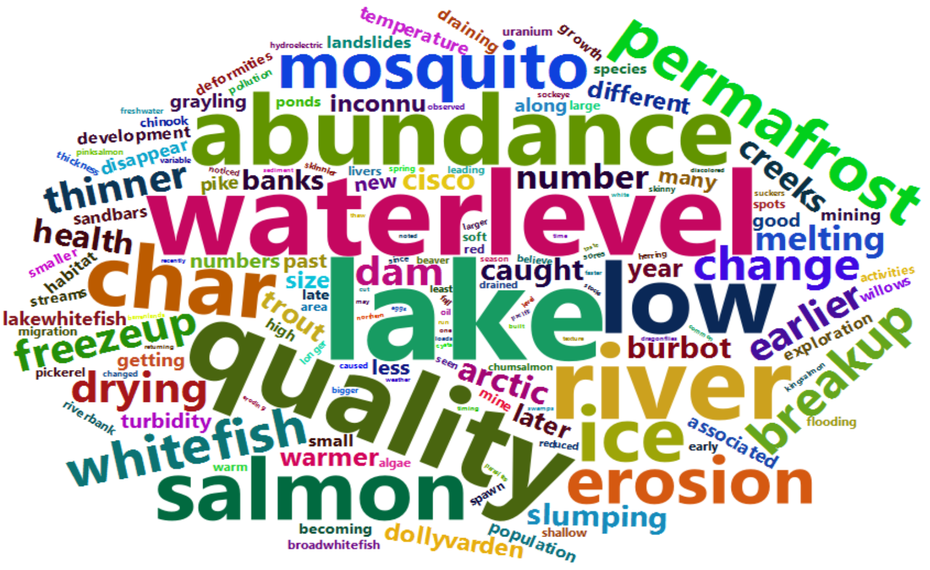


Figure 6-3 Word cloud of the common themes found in a systematic literature search of TK from Permanent Participants in Canada and Alaska, with word size indicating frequency of occurrence.

increase the collection of key parameters such as chlorophyll concentrations, water color, surface temperature and ice conditions, thereby improving environmental monitoring and the ability to estimate ecological status in remote areas of the Arctic.

6.3.1.4. Application of eDNA approaches

The Focal Ecosystem Components used in this report were necessarily restricted to those most likely to be commonly represented in existing databases for the circumpolar Arctic (Table 6-1). Indeed, important FECs such as microbial assemblages could not be assessed due to a lack of data. This deficiency may be corrected if future monitoring efforts **make use of recent advances in environmental DNA (eDNA) methods**, as these methods are particularly well advanced for microbial assemblages (Thomsen and Willerslev 2015). Furthermore, they provide a non-destructive way of monitoring presence/absence of fish. Thus, future monitoring activities should aim to build eDNA database information on freshwater assemblages, including microbial assemblages (i.e., bacteria, Archaea, fungi) as this FEC is pivotal to biogeochemistry processes and water quality, and likely will account for more alpha diversity than the other biotic groups combined. Furthermore, eDNA techniques can be applied to other FECs, such as macroinvertebrates, benthic algae and phytoplankton, to improve estimates of species richness. Clearly, the application of eDNA methods will require combination and calibration with traditional taxonomic and sampling methods to preserve the quality and continuity of long-term data series. Furthermore, it must be considered that while these techniques provide information about the presence/absence of taxa, they provide no information about lifestage or biomass/abundance.

6.3.2. Future Monitoring Methods

6.3.2.1. Sampling method harmonization

An important factor in the development of circumpolar monitoring is the consideration of using harmonized and intercalibrated monitoring approaches that are based upon intercalibrated international quality standards. In our analysis, differences in sample collection and processing methods were evident across the circumpolar region, reflecting the variety of sources from which data were obtained. For the purpose of the report, subsets of stations were selected to ensure comparability of data, controlling for differences in sampling methods, equipment, sampled habitats, and processing methods. However, **future sampling of Arctic freshwaters will require increased attention to harmonization of sampling approaches** (e.g., Culp et al. 2012a) to ensure broad-scale assessments can be completed. Such efforts ideally begins with harmonization of the suite of FECs that is sampled, to ensure that (1) multiple FECs are collected at each monitoring station (rather than only a single FEC, as was common in many areas) and (2) the full assemblage is sampled (e.g., species-specific sampling, which was common for fish in some areas, does not provide information about biodiversity) or a comparable portion of the assemblage is consistently sampled (e.g., if both diatoms and non-diatoms cannot be processed from benthic samples, ensure that diatoms are always processed so data are comparable with other countries). However, it will also be necessary to consider the different conditions that exist throughout the Arctic. For example, conditions in the high Arctic can be so different from low Arctic sites that specific or adapted methods are necessary. This can include specially-adapted field equipment, sampling effort, location of sampling sites (for example, sampling in the littoral or sub-littoral zones due to ice cover) and sampling time and frequency. Some adaptation may be required in these situations, though effort should be made to maintain as much continuity with harmonized methods as possible.

The selection of appropriate sampling methods and equipment must strike a balance between maintaining consistency and comparability with historical data and aligning with common methods used across the circumpolar region. Sampling approaches and sample processing are standardized to reduce observation variability and increase the ability to detect ecological changes (i.e., provide greater statistical power of assessments). The use of new methods will require calibration of the old and the new methods to preserve and guarantee the quality of long-term data series. Method comparison studies are available for several FECs including macroinvertebrates (Friberg et al. 2006, Buss et al. 2015, Poikane et al. 2016) and fish (Appelberg et al. 1995), and EU-countries have completed intercalibration assessments of ecological status using standardized methods for key FECs that are applicable to Arctic freshwaters. These studies can be used to inform the selection of harmonized sampling protocols, as outlined in Culp et al. (2012a). But additional effort is required to ensure sample processing is also broadly consistent across the circumpolar region. For example, large differences in magnification for algal sample processing could affect the accuracy of identification of small cells, and differences in methods used to estimate phytoplankton biovolume could affect comparability of data. Where sample collection and processing methods are not consistent across large spatial or temporal scales, analysis of data will be limited to qualitative or semi-quantitative assessments which, though informative, may not be sufficient to detect minor shifts in biodiversity.

Freshwater biomonitoring has traditionally focused on the assessment of ecosystem health rather than biodiversity, per se. Using a standardized sampling effort, this type of monitoring can provide a good estimate of the biodiversity of certain organism groups. However, these methods are not designed to measure biodiversity of a site because they underestimate the presence of rare species. Standardized biological samples of lakes and rivers can be modified to improve estimates of taxon richness and biodiversity. For example, Johnson and Goedkoop (2002) found that an additional 2-minute sample collection could increase taxa richness while not affecting the assessment of ecosystem health. Furthermore, the use of emerging technologies such as eDNA could provide additional information to better support the assessment of biodiversity patterns. We recognize that currently used, standardized monitoring efforts aim at assessing the ecological quality/integrity of freshwater and are not optimized to quantify biodiversity. Hence, we recommend that freshwater monitoring networks in the Arctic countries develop supplementary monitoring methods that provide better standardized estimates of biodiversity.

6.3.2.2. Sample distribution and replication

Analysis and comparison of diversity measures for each FEC was done using a regionalization approach based on ecologically-similar geographic regions. Such a regionalization approach reduces variability among data and increases statistical power as analyses compare areas that have similar climate and vegetation, and thus have similar climatic drivers. Furthermore, this approach supports the development and testing of impact hypotheses, particularly those related to changes in climate and vegetation. **We recommend that future monitoring uses such an ecoregion approach to guide the spatial distribution of sample stations.** The selection of ecoregions in a monitoring program could be driven partly by environmental conditions and predictions for expected change within ecoregions, and partly by the baseline diversity information presented in this report, including a selection of ecoregions with low and high alpha diversity, and with dominance of either nestedness or turnover components of beta diversity. Selection of ecoregions for monitoring should also recognize the distribution of existing or historic sampling stations for each FEC, to **ensure spatial coverage of sampled ecoregions is sufficient to address the overarching monitoring questions of the CBMP across the circumpolar region, maintain time series in key locations, and fill gaps where monitoring data are sparse.** For example, many FECs (including plankton and algae from benthic samples) had patchy distributions across the circumpolar region, which did not allow for a full assessment of spatial patterns in biodiversity.

Selection of stations for monitoring should also consider the spatial distribution within hydrobasins. Hydrobasins are standardly-derived geographic areas that relate directly to freshwater flow and sub-catchments, providing a smaller-scale geographic grouping of stations that can be used in combination with ecoregions. Within the SAFBR, stations were grouped by size level 5 or level 7 hydrobasins (see section 4.1.1), depending on sample replication. However, for many FECs, the stations in an ecoregion were found within a single hydrobasin, which indicated that there was inadequate spatial coverage of stations across the ecoregion. Estimates of alpha diversity and biodiversity in these cases were focused on individual sub-catchments within an ecoregion, and thus, may not provide an accurate picture of diversity patterns across the entire ecoregion. Future monitoring should ensure that multiple mid-level hydrobasins (size level 5 or level 7) are sampled within an ecoregion to improve the spatial distribution of stations.

In addition to sampling an adequate number of ecoregions and hydrobasins, it is necessary that **the number of monitoring stations should provide sufficient replication within chosen ecoregions**. In the SAFBR, alpha diversity was assessed across ecoregions by using rarefied taxonomic richness values to estimate the number of taxa found at a set number of stations. Where large numbers of stations were sampled within an ecoregion (e.g., 100 or more), rarefied alpha diversity estimates were more accurate, species accumulation curves reached or approached a plateau (e.g., Figure 6-4), and confidence intervals allowed for sound assessments of similarity among ecoregions with low variability. Even where sampling was more limited (e.g., 30-50 stations per ecoregion), alpha diversity could be compared among ecoregions with moderate confidence, though it was harder to distinguish differences among some ecoregions. However, comparison of alpha diversity at the rarefied level of only 10 stations per ecoregion, though necessary, was clearly inadequate, resulting in wide confidence intervals for poorly-sampled ecoregions (< 10 stations) and masking some differences among highly-sampled ecoregions that were evident when more stations were considered. For example, when three highly-sampled river BMI ecoregions were compared at approximately 40 stations or more,

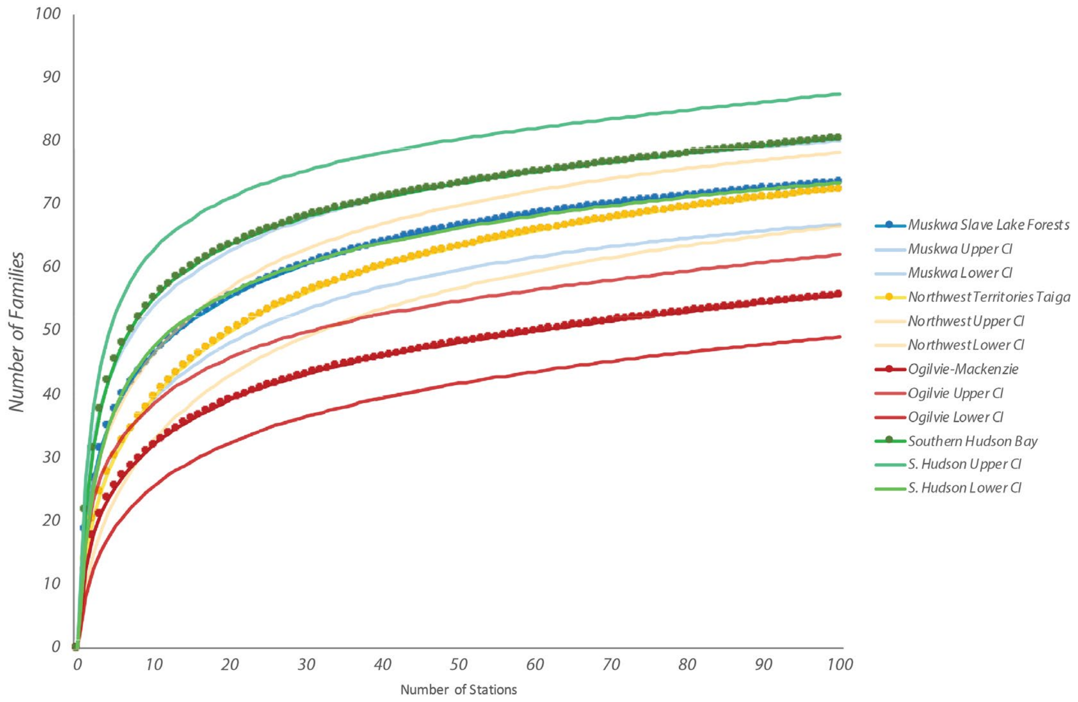


Figure 6-4 Rarefaction curves for river BMI in three ecoregions (Northwest Territories Taiga, Ogilvie-Mackenzie Alpine Tundra, and Southern Hudson Bay Taiga), showing the estimated number of families for each number of stations (up to 100 stations; thick lines with points) and 95% confidence intervals for diversity estimates (thin lines).

rarefaction curves indicated that alpha diversity was not significantly different between the Northwest Territories Taiga and the Southern Hudson Bay Taiga (confidence intervals overlapped), but that alpha diversity was significantly lower in the Ogilvie-Mackenzie Alpine Tundra than in either of the other two ecoregions (confidence intervals did not overlap; Figure 6-4). In contrast, when alpha diversity was compared at 10 stations on these rarefaction curves, the Southern Hudson Bay Taiga appeared to have significantly higher diversity than the other two ecoregions, and there was no difference between the Northwest Territories Taiga and Ogilvie-Mackenzie Alpine Tundra (Figure 6-4). Future monitoring should therefore increase replication within ecoregions to at least 30-40 stations to ensure more accurate assessments of alpha biodiversity patterns. As more targeted sampling designs are developed to address specific impact hypotheses, it may be possible to use estimates of variation from the CBMP database to inform sampling effort beyond the ecoregion-level recommendations.

6.3.3. Future Monitoring Design and Assessment

6.3.3.1. Integrated Experimental Design of Hub-and-Spoke Monitoring Networks

To provide better knowledge of the status and trends in Arctic freshwater biodiversity and the physico-chemical habitats supporting biodiversity, **we envision that Arctic countries develop joint efforts to establish a circumpolar monitoring network based on a hub-and-spoke principle in remote areas.** The hubs could provide the infrastructure platform required to monitor the effects of climate change and diffusive pollution on freshwaters in more remote Arctic areas and would include intensive sampling over time. Monitoring at secondary sites associated with the hub (i.e., spokes moving away from the central hub) would provide additional, more extensive baseline measures that would help generalize observations across larger spatial expanses. Good candidates for such platforms are existing Arctic monitoring and research stations such as the Canadian High Arctic Research Station (CHARS), Disko, Zackenberg, Longyearbyen/Ny Ålesund and Abisko. These locations could be linked to form a circumpolar network of hubs from which harmonized monitoring of lake and river biodiversity are undertaken. Such biological monitoring would be enhanced by incorporating remotely sensed data to improve the spatial applicability of models for environmental prediction across ecoregions. Several of the research locations listed above already have ongoing freshwater monitoring programs, while others are developing such programs.

The experimental design for the hub-and-spoke network should focus on addressing the Impact Hypotheses developed in the CBMP freshwater plan (Culp et al. 2012a), although regional and country-specific questions may also be considered. Many of the impact hypotheses require targeted study designs for detection of impacts and/or assessment of time series data. A future monitoring plan design will benefit from the use of large spatial analyses across gradients of expected change including those related to a warming climate (e.g., permafrost thaw, nutrient release, sediment loading). These gradients need to extend from reference (i.e., least impacted) areas to regions of high impact. An important consideration will be to examine the potential for climate change and development to impact areas of particular vulnerability (e.g., areas with low functional redundancy, important conservation areas). In addition, future monitoring should consider re-sampling previously visited sites to increase the potential to detect biodiversity changes over time and address the overarching CBMP monitoring questions that relate to changes in biodiversity and boundaries of Arctic zones (Culp et al. 2012a). Such a broad, integrated program will benefit from the use of harmonized monitoring protocols that can facilitate environmental and regulatory assessments, such as measuring the potential impact of industrial developments including mining and petroleum extraction. Moreover, a monitoring program that integrates biological variables with the drivers of biotic assemblage structure and function better identifies the primary drivers of biodiversity and contributes to our understanding of multiple stressors in this process (e.g., nutrient-contaminant interactions as impacted by warming).

We recommend that the Freshwater Steering Group of the CBMP continue to serve as the focal point for the development and implementation of pan-Arctic freshwater biodiversity monitoring. The CBMP steering group, which includes representatives of all Arctic countries with diverse expertise in science and decision making, should incorporate input from other key Arctic scientists to adjust and harmonize existing programs so that future freshwater biodiversity monitoring achieves the aims of the original CBMP freshwater plan (Culp et al. 2012a). A main objective of this steering group would be to optimize the circumpolar monitoring program to integrate the data flowing from the hub-and-spoke network of the Arctic countries. Finally, consideration needs to be given to how the Arctic freshwater biodiversity monitoring efforts can be linked to, contribute to and draw from the global Freshwater BON of GEO BON.

6.3.3.2. Maintaining and Building the Arctic Freshwater Biodiversity Database

A very important and unique output of this assessment is the creation of a pan-Arctic database of the Focal Ecosystem Components and supporting variables that were used to evaluate the status and trends in Arctic freshwater biodiversity. This database establishes a set of baseline data for future assessments of temporal and spatial change in biodiversity. It also represents an opportunity to derive a number of value-added outputs. For example, these baseline data can be used to produce indicators for monitoring and reporting on trends to support policy development in the Arctic. Furthermore, indicators can be aligned with those used in other programs (e.g., through development of Essential Biodiversity Variables, as used by GEO BON; Pereira et al. 2013) to support international efforts to monitor biodiversity. The database can also support future monitoring and research efforts by providing information about spatial and temporal variability within and among regions that can inform sampling design and monitoring extent.

To fully realize the benefits of this database, future resources must be provided to maintain and continue to build the database for future assessments. Building of the database must include not only the incorporation of future data from the proposed integrated, hub-and-spoke monitoring programs and from ongoing national monitoring activities, but also the incorporation of existing data from scientific studies that are complementary to monitoring efforts. Improved documentation of research data, and at a minimum appropriate metadata, needs to be catalogued in an appropriate database according to the "open data" strategies recently adopted by national funding agencies in many of the Arctic countries. Though extensive, the integration of research data into the CBMP database was not exhaustive as such data catalogues are not fully established in most countries. For example, there are a number of existing data sources that could improve spatial and temporal coverage of FECs, such as European research-based paleolimnological databases that could contribute to a more extensive assessment of temporal trends using top/bottom and downcore data. Another important data source is available in the "catch" information recorded for commercial, sustenance, and recreational fisheries. These catch statistics are usually coordinated by official authorities for regulatory purposes and often provide a unique, long-term record of the status and trend of species valued by humans. **We recommend that Arctic countries make efforts to document and preserve data from short-term research projects, research expeditions, industrial, university and government programs** because this broad range of activities can provide valuable information on Arctic freshwater biodiversity and the physico-chemical habitats supporting this biodiversity. Although many sites may have been visited only once, this suite of sites could provide a framework by which re-sampling visits could be planned based on an optimal sampling approach that allows for multiple environmental gradients to be covered (e.g., latitudinal transects) and the establishment of long-time series (albeit with low sampling frequency).

6.3.3.3. Assessment Methods

Rarefaction curves provide an effective way and a sound approach to estimate alpha diversity where irregular sampling has occurred, because these curves control for variation in sampling effort by comparing taxa richness at a set number of stations. Where many stations have been sampled in an ecoregion, the result is an estimate of richness based on repeatedly randomly selecting a subset for analysis, thus simulating the number of taxa that might have been collected with less sampling effort (in line with less-sampled ecoregions). The extraction of a full rarefaction curve for each station provides the opportunity to assess alpha diversity at different levels of sampling effort, as in this report, providing more accurate assessments of taxa richness in highly-sampled ecoregions. Rarefaction approaches also allow for the extrapolation of richness estimates to a higher number of stations than was sampled, to bring less-sampled ecoregions in line with those that had more sampling; however, large extrapolation or extrapolation from a very small number of stations (e.g., < 5) should be used with caution, as they result in large confidence intervals that make it difficult to compare alpha diversity estimates among ecoregions. Given the spatially patchy nature of existing data and of ongoing monitoring efforts, **future assessments will require the continued use of rarefaction curves to estimate alpha diversity for comparison across ecoregions.**

Spatial and temporal patterns in diversity across the circumpolar region should be assessed and compared among FECs to contribute to a whole-ecosystem understanding of the potential for change, but further application of this approach will require improvements to sample coverage. Each FEC responds to a different suite of environmental drivers, and assessment of multiple FECs provides the greatest potential to detect biotic shifts in response to stressors. However, limited sampling of multiple FECs at a station or even within an ecoregion (particularly in North America, where sampling efforts were more strongly research-based, focusing on specific questions related to a single FEC) often precluded such assessments, or masked some patterns in diversity. For example, the highest diversity for several FECs (e.g., macrophytes, plankton, lake BMI) was found in Fennoscandian ecoregions, which suggested that these were hot spots for diversity across multiple FECs. However, this was likely a reflection of the low or patchy availability of lake data for Canada, which led to overall lower diversity than in Fennoscandia. For example, when data with extensive spatial coverage in North America (e.g., river BMI) and Fennoscandia were compared, there was evidence of southern Canadian ecoregions that had higher alpha diversity than was found in Fennoscandia. Furthermore, areas of the Arctic that are known to have low diversity for a particular FEC (for example, low diversity of macroinvertebrates on Svalbard; Blaen et al. 2014, Chertoprud et al. 2017)may not have had a sufficient number of stations to draw broad conclusions across FECs and in comparison with other ecoregions. With increased sample coverage focused on filling gaps and improving replication within ecoregions, such assessments will be of high priority to inform management and policy.

An increased focus on assessing biotic-abiotic relationships in Arctic freshwater systems is necessary in order to effectively test impact hypotheses and address the overarching monitoring questions of the CBMP. This report begins to address these questions by relating biotic patterns to abiotic drivers, but more direct testing of these relationships is necessary to understand biodiversity change in the Arctic. Supporting abiotic data are not consistently recorded with biotic sampling data, nor are they always available or in a useable/comparable format. Thus, data on water chemistry, hydrology, water temperature, and site-level habitat structure were not available for a large share of monitoring stations, thus limiting the extent to which these relationships could be examined. Where possible, we have used geospatial variables (e.g., long-term air temperature and precipitation, ground ice content, thermokarst) by calculating summaries of parameters for the hydrobasin in which each station was found. The use of remote sensing and geospatial data allows for broad-scale assessments using abiotic variables that are inherently harmonized when they come from a single circumpolar data source. However, it was not always possible to access geospatial data that covered the entire area of interest (particularly where data were limited to above the Arctic Circle, e.g., Walker et al. 2005, Harrison et al. 2011). Despite these limitations, the use of geospatial data will continue to be necessary to provide standardized circumpolar measures of abiotic variables, particularly where in-stream measurements have not been collected or when variability within those measurements is too great.

6.3.4. Recommendations/Summary

The rapid change occurring in Arctic ecosystems highlights the need for the CAFF-CBMP initiative to establish baselines against which future biodiversity change can be assessed and promote the requirement of harmonizing monitoring efforts among Arctic countries. This report on Arctic freshwater biodiversity further emphasizes that status assessments of Arctic lakes and rivers must explore the association of biodiversity with spatial patterns of physico-chemical quality of aquatic habitats that can drive biological systems. Key recommendations for consideration in future biodiversity monitoring of freshwater ecosystems in the Arctic include the following:

Emerging Approaches

- Include Traditional Knowledge as an integral part of future circumpolar monitoring assessments and networks.
- Engage local communities in monitoring efforts through Citizen Science efforts as an integral part of future circumpolar monitoring networks.
- Include an increased focus and use of remote sensing approaches (e.g., satellite imagery, deployment of in situ data sensors).
- Make use of recent advances in environmental DNA (eDNA) methods

Future Monitoring Methods

- Employ a combination of traditional and novel approaches to improve monitoring efficiency, and further efforts focused on sampling approach harmonization among countries.
- Select appropriate sampling methods and equipment to balance between maintaining consistency and comparability with historical data and alignment with common methods used across the circumpolar region.
- Develop supplementary monitoring methods that provide better standardized estimates of biodiversity to maximize the likelihood of detecting new and/or invasive species.
- Use a regionalization approach based on ecoregions (Terrestrial Ecoregions of the World; TEOW) to guide the spatial distribution of sample stations and, ultimately, to provide better assessments.
- Ensure that spatial coverage of sampled ecoregions is sufficient to address the overarching monitoring questions of the CBMP across the circumpolar region, maintain time series in key locations, and fill gaps where monitoring data are sparse.
- Ensure that multiple mid-level hydrobasins (size level 5 or level 7) are sampled within an ecoregion to improve the spatial distribution of stations.
- Ensure the number of monitoring stations provides sufficient replication within chosen ecoregions.

Future Monitoring Design and Assessment

- Arctic countries should establish a circumpolar monitoring network based on a hub-and-spoke (intensive-extensive) principle in remote areas.
- Experimental design for the hub-and-spoke network should largely focus on addressing the Impact Hypotheses developed in the CBMP freshwater plan.
- An increased focus on assessing biotic-abiotic relationships in Arctic freshwater systems is necessary in order to effectively test impact hypotheses.
- The Freshwater Steering Group of the CBMP should continue to serve as the focal point for the development and implementation of pan-Arctic, freshwater biodiversity monitoring.
- Resources must be provided to maintain and build the freshwater database for future assessments in order to maximize the benefits of this database
- Arctic countries should make efforts to document and preserve data from short-term research projects, research expeditions, industrial, university and government programs.
- Due to the patchy nature of sampling, future assessments require the continued use of rarefaction curves for scientifically-sound comparisons of alpha diversity across ecoregions.
- Spatial and temporal diversity patterns across the circumpolar region should be assessed and compared among FECs to contribute to a whole-ecosystem understanding of the potential for change.

Fish drying
Photo: Steve Hillebrand, USFWS

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8. Appendix A

Appendix A. The fish species pools of seven geographic regions that were determined from historic geomorphic processes, current dispersal barriers and geographic size (ABA 2013). Lists are based on those compiled in the Arctic Biodiversity Assessment (2013), from literature (on alpha diversity, as quantified by estimates of species richness from reference texts (Scott and Crossman 1973; Mecklenburg, Mecklenburg, and Thorsteinson 2002; Muus and Dahlström 1993), and from academic and government biologists. Identification numbers (Figure 2 ID) are provided for the fishes included in Figure 2. Fish species included in the CBMP freshwater database referenced for this report are D – Arctic species in database and (D) – subarctic species in database. Other known Arctic species of fish are indicated with + and fish from the subarctic only with ++. Species marked * are introduced in that region. No. Regions, the number of geographic regions where a given species occurs.

Family	Scientific name	Common name	Figure 4-40 ID	Beringia and northwestern Canada	Northeastern North America	North Atlantic	Russia and Scandinavia	Western Siberia	Central Siberia	Eastern Siberia	No. Regions
Salmonidae	*Salvelinus alpinus*	Arctic charr	56	D	D	D	D	+	+	+	7
Salmonidae	*Coregonus lavaretus*	Common whitefish	46	+		+	D	+	+	+	6
Salmonidae	*Coregonus sardinella*	Least cisco	61	D	+		+	+	+	+	6
Esocidae	*Esox lucius*	Northern pike	67	D	+		D	+	+	+	6
Gadidae	*Lota lota*	Burbot	66	D	+		D	+	+	+	6
Salmonidae	*Oncorhynchys gorbuscha*	Pink salmon	57	D	+	+	D*		+	+	6
Gasterosteidae	*Pungitius pungitius*	Ninespine stickleback	55	D	+		D	+	+	+	6
Salmonidae	*Thymallus arcticus*	Arctic grayling	58	D	+		+	+	+	+	6
Salmonidae	*Coregonus autumnalis*	Arctic cisco	64	D			+	+	+	+	5
Salmonidae	*Coregonus nasus*	Broad whitefish	65	D			+	+	+	+	5
Salmonidae	*Coregonus pidschian*	Humpback whitefish	60	D			D	+	+	+	5
Gasterosteidae	*Gasterosteus aculeatus*	Threespine stickleback	54	D	D	+	D			+	5
Petromyzontidae	*Lethenteron kessleri*	Siberian brook lamprey	34	+			+	+	+	+	5
Osmeridae	*Osmerus mordax*	Rainbow smelt	69	D			+	+	+	+	5
Salmonidae	*Prosopium cylindraceum*	Round whitefish	74	D	+			+	+	+	5
Salmonidae	*Stenodus leucichthys*	Inconnu	63	D			+	+	+	+	5
Acipenseridae	*Acipenser baerii*	Siberian sturgeon	1				+	+	+	+	4
Cyprinidae	*Carassius carassius*	Crucian carp	12				+	++	++	++	4
Cyprinidae	*Carassius gibelio*	Prussian carp	10				+	++	++	++	4
Salmonidae	*Coregonus peled*	Peled	39				D	+	+	+	4
Cottidae	*Cottus poecilopus*	Alpine bullhead	4				(D)	++	+	+	4
Percidae	*Gymnocephalus cernuus*	Ruffe	32				D	+	+	+	4
Petromyzontidae	*Lethenteron camtschaticum*	Arctic lamprey	68	D			+	+		+	4

Family	Scientific name	Common name	Figure 4-40 ID	Beringia and northwestern Canada	Northeastern North America	North Atlantic	Russia and Scandinavia	Western Siberia	Central Siberia	Eastern Siberia	No. Regions
Cyprinidae	*Leuciscus leuciscus*	Common dace	20				D	+	+	+	4
Percidae	*Perca fluviatilis*	European perch	33				D	+	+	+	4
Cyprinidae	*Phoxinus phoxinus*	Eurasian minnow	27				D	+	++	+	4
Cyprinidae	*Rhynchocypris percnurus*	Lake minnow	25				+	+	+	+	4
Balitoridae	*Barbatula toni*	n/a						++	++	++	3
Salmonidae	*Brachymystax lenok*	Lenok						++	+	+	3
Catostomidae	*Catostomus catostomus*	Longnose sucker	81	D	+					+	3
Salmonidae	*Coregonus muksun*	Muksun	37					+	+	+	3
Cottidae	*Cottus cognatus*	Slimy sculpin	80	D	+					+	3
Cyprinidae	*Gobio gobio*	Gudgeon	23				++	++	++		3
Salmonidae	*Hucho taimen*	Taimen	41				++	+	+		3
Osmeridae	*Hypomesus olidus*	Pond smelt	70	+				+		+	3
Cyprinidae	*Leuciscus idus*	Ide	22				D	+	+		3
Salmonidae	*Oncorhynchus keta*	Chum salmon	59	D					+	+	3
Petromyzontidae	*Petromyzon marinus*	Sea lamprey	50		+	+	+				3
Cyprinidae	*Rhynchocypris czekanowskii*	Czekanowski's minnow	7					+	+	+	3
Cyprinidae	*Rutilus rutilus*	Roach	28				D	+	++		3
Salmonidae	*Salmo salar*	Atlantic salmon	49		+	D	D				3
Salmonidae	*Salvelinus fontinalis*	Brook trout	40	+	+		D*				3
Salmonidae	*Salvelinus namaycush*	Lake trout	62	D	D		D*				3
Cyprinidae	*Abramis brama*	Common bream	11				+	++			2
Acipenseridae	*Acipenser medirostris*	Green sturgeon		++						++	2
Acipenseridae	*Acipenser ruthenus*	Sterlet	0				+	+			2
Acipenseridae	*Acipenser sturio*	Sturgeon				++	+				2
Clupeidae	*Alosa sapidissima*	American shad		++						++	2
Anguillidae	*Anguilla anguilla*	European eel	51			+	D				2
Anguillidae	*Anguilla rostrata*	American eel	52		+	++					2
Cyprinidae	*Aspius aspius*	Asp	16				++	++			2
Catostomidae	*Catostomus commersonii*	White sucker	99	D	+						2
Cobitidae	*Cobitis sinensis*	Siberian spiny loach						++	++		2

Family	Scientific name	Common name	Figure 4-40 ID	Beringia and northwestern Canada	Northeastern North America	North Atlantic	Russia and Scandinavia	Western Siberia	Central Siberia	Eastern Siberia	No. Regions
Salmonidae	*Coregonus artedi*	Lake cisco	83	D	+						2
Salmonidae	*Coregonus clupeaformis*	Lake whitefish	76	D	D						2
Salmonidae	*Coregonus tugun tugun*	Tugun	38					+	+		2
Cottidae	*Cottus ricei*	Spoonhead sculpin	97	+	++						2
Cottidae	*Cottus sibiricus*	Siberian sculpin	2					+	+		2
Cyprinidae	*Couesius plumbeus*	Lake chub	79	D	+						2
Gasterosteidae	*Culaea inconstans*	Brook stickleback	90	D	++						2
Petromyzontidae	*Entosphenus tridentatus*	Pacific lamprey	82	+						++	2
Cyprinidae	*Gobio cynocephalus*	Siberian gudgeon	8				+	+			2
Petromyzontidae	*Lethenteron reissneri*	Far eastern brook lamprey		+						++	2
Cyprinidae	*Leuciscus baicalensis*	Siberian dace	6					+	+		2
Cyprinidae	*Margariscus margarita*	Pearl dace		++	++						2
Cyprinidae	*Notropis atherinoides*	Emerald shiner	94	++	++						2
Cyprinidae	*Notropis hudsonius*	Spottail shiner	92	+	++						2
Salmonidae	*Oncorhynchus mykiss*	Rainbow trout	42	+			D*				2
Salmonidae	*Oncorhynchus nerka*	Sockeye salmon	73	D						++	2
Salmonidae	*Oncorhynchus tshawytscha*	Chinook salmon	75	D						+	2
Percidae	*Perca flavescens*	Yellow perch	88	++	++						2
Percopsidae	*Percopsis omiscomaycus*	Trout-perch	86	+	++						2
Cyprinidae	*Phoxinus percnurus*	Swamp minnow	9				+	+			2
Cyprinidae	*Rhinichthys cataractae*	Longnose dace	91	++	++						2
Salmonidae	*Salmo trutta*	Sea trout (*trutta* form)	53			D	D				2
Salmonidae	*Salvelinus czerskii*	Cherskii's char							+	+	2
Salmonidae	*Salvelinus malma*	Dolly Varden	71	D						+	2
Salmonidae	*Salvelinus taranetzi*	Taranetz char		+						+	2
Percidae	*Sander vitreus*	Walleye	87	D	++						2
Salmonidae	*Thymallus pallasii*	East Siberian grayling		+						+	2
Salmonidae	*Thymallus thymallus*	Grayling	44				D	+			2
Cyprinidae	*Abramis ballerus*	Zope	15				++				1
Cyprinidae	*Abramis sapa*	White-eye bream	14				++				1

Family	Scientific name	Common name	Figure 4-40 ID	Beringia and northwestern Canada	Northeastern North America	North Atlantic	Russia and Scandinavia	Western Siberia	Central Siberia	Eastern Siberia	No. Regions
Acipenseridae	*Acipenser fulvescens*	Lake sturgeon			++						1
Cyprinidae	*Alburnus alburnus*	Bleak	21				(D)				1
Clupeidae	*Alosa agone*	Twaite shad				++					1
Balitoridae	*Barbatula barbatula*	Stone loach	26				+				1
Cyprinidae	*Blicca bjoerkna*	White bream	13				++				1
Cyprinidae	*Chrosomus eos*	Northern redbelly dace		++							1
Cyprinidae	*Chrosomus neogaeus*	Finescale dace	95	+							1
Cobitidae	*Cobitis taenia*	Spined loach	19				++				1
Salmonidae	*Coregonus albula*	Vendace	43				D				1
Salmonidae	*Coregonus laurettae*	Bering cisco	77	D							1
Salmonidae	*Coregonus zenithicus*	Shortjaw cisco		++							1
Cottidae	*Cottus aleuticus*	Coastrange sculpin		+							1
Cottidae	*Cottus bairdii*	Mottled sculpin			+						1
Cottidae	*Cottus gobio*	Bullhead	3				D				1
Cyprinidae	*Cyprinus carpio*	Common carp	5				+*				1
Umbridae	*Dallia pectoralis*	Alaska blackfish	78	D							1
Percidae	*Etheostoma nigrum*	Johnny darter			++						1
Gadidae	*Gadus morhua*	Atlantic cod			+						1
Hiodontidae	*Hiodon alosoides*	Goldeye	89	(D)							1
Petromyzontidae	*Lampetra fluviatilis*	European river lamprey	36				(D)				1
Petromyzontidae	*Lampetra planeri*	European brook lamprey	35				(D)				1
Cyprinidae	*Leucaspius delineatus*	Sunbleak	24					++			1
Cyprinidae	*Leuciscus cephalus*	Chub	17				++				1
Cyprinidae	*Mylocheilus caurinus*	Peamouth		++							1
Cottidae	*Myoxocephalus thompsonii*	Deepwater sculpin	98	+							1
Salmonidae	*Oncorhynchus kisutch*	Coho salmon	72	D							1
Osmeridae	*Osmerus eperlanus*	European smelt	30				D				1
Cottidae	*Paracottus knerii*	Stone sculpin						++			1
Odontobutidae	*Perccottus glenii*	Amur sleeper	29				++*				1
Percidae	*Percina caprodes*	Logperch			++						1
Cyprinidae	*Pimephales promelas*	Fathead minnow	96	++							1

Family	Scientific name	Common name	Figure 4-40 ID	Beringia and northwestern Canada	Northeastern North America	North Atlantic	Russia and Scandinavia	Western Siberia	Central Siberia	Eastern Siberia	No. Regions
Cyprinidae	*Platygobio gracilis*	Flathead chub	93	+							1
Salmonidae	*Prosopium coulterii*	Pygmy whitefish		+							1
Salmonidae	*Prosopium williamsoni*	Mountain whitefish	85	(D)							1
Cyprinidae	*Rhinichthys atratulus*	Blacknose dace			++						1
Salmonidae	*Salmo trutta*	Sea trout (*fario* form)	45				D				1
Salmonidae	*Salvelinus andriashevi*	Chukot char		+							1
Salmonidae	*Salvelinus boganidae*	Boganida char								+	1
Salmonidae	*Salvelinus confluentus*	Bull trout	84	(D)							1
Salmonidae	*Salvelinus drjagini*	Drjagin's char						+			1
Salmonidae	*Salvelinus elgyticus*	Small-mouth char								+	1
Salmonidae	*Salvelinus jacuticus*	Yakutian char							+		1
Salmonidae	*Salvelinus lepechini*	Lepechin's char					++				1
Salmonidae	*Salvelinus leucomaenis*	Whitespotted char								++	1
Salmonidae	*Salvelinus murta*	n/a	47			++					1
Salmonidae	*Salvelinus taimyricus*	Taymyr Lake char						+			1
Salmonidae	*Salvelinus thingvallensis*	n/a	48			++					1
Salmonidae	*Salvelinus tolmachoffi*	Esei Lake char						+			1
Salmonidae	*Salvethymus svetovidovi*	Long-finned char								+	1
Percidae	*Sander canadensis*	Sauger			++						1
Percidae	*Sander lucioperca*	Pike-perch	31					+			1
Cyprinidae	*Scardinius erythrophthalmus*	Common rudd	18				++				1
Cyprinidae	*Semotilus atromaculatus*	Creek chub			++						1
Cyprinidae	*Semotilus corporalis*	Fallfish			++						1
	Totals	Database		34	4	3	30	0	0	0	
		sub+low+high		62	38	13	61	50	40	49	
		Arctic (low + high)		28	11	2	29	19	12	14	

The Conservation of Arctic Flora and Fauna
Borgir, Nordurslod
600 Akureyri, Iceland
www.caff.is
www.arcticbiodiversity.is/freshwater
caff@caff.is
+354 462 3350