



Biomonitoring Using Diatoms and Paleolimnology in the Western Great Lakes National Parks

2011/2012 Summary

Natural Resource Technical Report NPS/GLKN/NRTR—2013/814



ON THE COVER

The diatom *Semiorbis hemicyclus* is characteristic of Outer Island lagoon at Apostle Islands National Lakeshore, Wisconsin. The cells grow in ring-shaped colonies.

Photograph by: Mark Edlund, St. Croix Watershed Research Station



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Contents

	Page
Figures.....	v
Tables.....	vii
Executive Summary	ix
Acknowledgments.....	xi
Acronyms Used in This Report.....	xi
Introduction.....	1
Methods.....	3
Water Quality Sampling.....	3
Surface Sediment Collection	3
Sediment Processing and Diatom Analysis	4
Numerical Analysis	5
Results.....	7
Water Quality, 2006–2012	7
Apostle Islands National Lakeshore (APIS).....	7
Indiana Dunes National Lakeshore (INDU).....	7
Isle Royale National Park (ISRO)	7
Mississippi National River and Recreation Area (MISS)	7
St. Croix National Scenic Riverway (SACN)	9
Surface Sediment Diatom Assemblages.....	9
Ecological Change in GLKN Lakes, 2006–2012	11
Diatom-inferred water quality measures	11
Diatom community change.....	16
Directionality of change using canonical correspondence analysis of the GLKN diatom calibration set	21
Environmental change in relation to cored APIS lagoons and ISRO lakes.....	24

Contents (continued)

	Page
Conclusions.....	31
Literature Cited	33

Figures

	Page
Figure 1. The nine national park units of the Great Lakes Inventory and Monitoring Network.....	2
Figure 2. Plot of squared chord distances for long sediment cores and surface repeat samples from GLKN park lakes.....	17
Figure 3. The 2011 repeat surface sediment samples from SACN and INDU passively plotted on the canonical correspondence analysis (CCA) of the GLKN diatom calibration set.	22
Figure 4. The 2012 repeat surface sediment samples from ISRO passively plotted on the canonical correspondence analysis (CCA) of the GLKN diatom calibration set.....	23
Figure 5. The 2012 repeat surface sediment samples from ISRO passively plotted on the canonical correspondence analysis (CCA) of the GLKN diatom calibration set.....	24
Figure 6. Detrended correspondence analysis (DCA) of downcore diatom assemblages from Outer Island lagoon core (APIS, 1836–2007) including the 2011 repeat surface sample (APOUTR_2011).....	26
Figure 7. Detrended correspondence analysis (DCA) of downcore diatom assemblages from Lake Ahmik core (ISRO, 1825–2007) including the 2012 repeat surface sample (ISAHMK_2012)	27
Figure 8. Detrended correspondence analysis (DCA) of downcore diatom assemblages from Lake Harvey core (ISRO, 1825–2007) including the 2012 repeat surface sample (ISHARV_2012)	28
Figure 9. Detrended correspondence analysis (DCA) of downcore diatom assemblages from Lake Richie core (ISRO, 1833–2005) including the 2012 repeat surface sample (ISRICH_2012).....	29

Tables

	Page
Table 1. Lakes at Apostle Islands National Lakeshore (APIS), Indiana Dunes National Lakeshore (INDU), St. Croix National Scenic Riverway (SACN), Mississippi National River and Recreation Area (MISS), and Isle Royale National Park (ISRO) where surface sediments were collected in 2011 and 2012.....	3
Table 2. Summary of minimum, maximum, mean and median water quality measures from 21 GLKN lakes, 2006-2012.	8
Table 3. Predominant diatoms (>5% relative abundance) in sediment samples collected from Apostle Islands National Lakeshore (APIS), Indiana Dunes National Lakeshore (INDU), Isle Royale National Park (ISRO), Mississippi National River and Recreation Area (MISS), and St. Croix National Scenic Riverway (SACN) in 2011 and 2012.....	9
Table 4A. Diatom-inferred (DI) estimates of specific conductance (EC25) based on transfer functions developed for the GLKN lakes by Edlund et al. (2011).	11
Table 4B. Diatom-inferred (DI) estimates of pH based on transfer functions developed for the GLKN lakes by Edlund et al. (2011).	12
Table 4C. Diatom-inferred (DI) estimates of total phosphorus (TP) based on transfer functions developed for the GLKN lakes by Edlund et al. (2011).	13

Executive Summary

A diatom biomonitoring program complements the National Park Service Great Lakes Inventory and Monitoring Network (GLKN) lakes and large rivers monitoring program. The diatom program is designed to: 1) provide a cost-effective means of monitoring a biological component common among all lakes, 2) be strongly integrated with current physical and chemical monitoring efforts developed for the lakes, and 3) detect biological changes that are not sampled during regular water quality monitoring. Biomonitoring involves sampling and analysis of diatoms in surface sediments that are collected from GLKN lakes every 3–5 years during regular water quality monitoring visits. Using the two monitoring strategies, we are able to detect biological change, estimate trends in water quality parameters (pH, total phosphorus [TP], specific conductance [EC25]) using diatom-inference models, interpret change, generate hypotheses on why change is occurring, and direct future research efforts and resources to better understand the drivers and mechanisms of ecological change in NPS lakes.

The 2011 and 2012 diatom biomonitoring activities targeted 21 lakes, lagoons, and flowages distributed among several GLKN parks including Apostle Islands (APIS) and Indiana Dunes (INDU) national lakeshores, St. Croix National Scenic Riverway (SACN), Mississippi National River and Recreation Area (MISS), and Isle Royale National Park (ISRO). Repeat surface sediments were collected with gravity cores from all sites and analyzed for diatom assemblages. The diatom communities were first characterized, and then the GLKN diatom transfer functions for estimating water quality parameters were applied to all repeat sediment samples to estimate total phosphorus (TP), pH, and specific conductance (EC25). The 2011/2012 surface sediment samples were compared to previous surface sediment diatom assemblages collected between 2006 and 2008 using the squared chord distance to determine the significance of community change between sampling events. Repeat diatom samples were passively plotted on the species-environment ordination developed for the GLKN lakes to determine recent ecological trajectories and explore what environmental factors may be driving changes in modern diatom communities. Last, to obtain a longer-term perspective on recent change in GLKN lakes, the most recent surface sediment collections were plotted on ordinations of diatom assemblages representing 150 years of change recorded in long sediment cores recovered in 2007 from three ISRO lakes and one lagoon at APIS.

A very diverse assemblage of diatoms was preserved in the 2011/2012 surface sediment samples from these five park units. Assemblages varied widely by park unit, reflecting the diversity of habitats and lake types within the GLKN. Over 650 diatom taxa were recorded in the samples.

The lakes, lagoons, and flowages were primarily circumneutral, varied in TP from oligotrophic to eutrophic levels, and had specific conductance values from $<20 \mu\text{S}/\text{cm}$ to over $500 \mu\text{S}/\text{cm}$. Diatom-inferred estimates of TP, pH, and EC25 generally showed good agreement with water quality monitoring data. Problem lakes were those that represented outliers (APIS and MISS sites) during the construction of the GLKN diatom calibration set or those near the ends of environmental gradients of GLKN lakes.

Among the APIS, INDU, SACN, MISS, and ISRO water bodies, changes in surface sediment diatom communities between the 2006–2008 and 2011/2012 sampling periods were often significant, based

on squared chord distances. Among the park units, lagoons at APIS, the large river sampling sites on SACN and MISS, INDU's Long Lake, and the shallow lakes of ISRO had significant change.

Surface sediment diatom communities from the 2011/2012 lakes were passively plotted on the species-environment ordination of the GLKN lakes to explore what environmental factors may be driving changes in modern diatom communities. Trajectories for the several of the sites at INDU and SACN suggested declines in nutrient availability, whereas diatom communities in the larger lakes of ISRO appear to be responding along the DOC and pH gradients.

Comparison of 2011/2012 repeat samples with sediment records from sites cored in 2007 at APIS (Outer lagoon) and ISRO (lakes Ahmik, Harvey, and Richie) show that recent samples from these water bodies continue the trend of greater biological variability among sediment samples deposited after the 1980s. The repeat samples further indicate that these new community trajectories are not a transitory period in lake history; none of the cored lakes show recent samples returning to historical communities.

The 2011/2012 repeat samples show that many of the GLKN lakes are undergoing biological change as recorded in surface sediment diatom assemblages. Some of the sites may have high levels of biological change because they do not represent depositional basins (e.g., the MISS sites and some lagoons at APIS). However, most sites sampled in 2011/2012 are depositional basins, and analyses support that diatom communities are changing rapidly across the GLKN. Diatom communities in shallow sites seem to be changing particularly quickly. Analysis of diatom communities preserved in surface sediments provides a measure of biological change that is difficult to capture with typical monitoring efforts that target physical and chemical parameters. Sediment assemblages provide an integrated record of biological variability in GLKN lakes and are serving as a bellwether of change in biological communities. Continued monitoring will be needed to fully understand the drivers and mechanisms of the unprecedented ecological changes in these protected lakes.

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Acronyms Used in This Report

APIS	Apostle Islands National Lakeshore (Wisconsin)
CCA	Canonical correspondence analysis
DCA	Detrended correspondence analysis
DOC	Dissolved organic carbon
EC25	Specific conductance normalized to 25°C
GLKN	Great Lakes Inventory and Monitoring Network
INDU	Indiana Dunes National Lakeshore (Indiana)
ISRO	Isle Royale National Park (Michigan)
MISS	Mississippi National River and Recreation Area (Minnesota)
PIRO	Pictured Rocks National Lakeshore (Michigan)
SACN	St. Croix National Scenic Riverway (Wisconsin/Minnesota)
SLBE	Sleeping Bear Dunes National Lakeshore (Michigan)
TN	Total nitrogen
TP	Total phosphorus
VOYA	Voyageurs National Park (Minnesota)

Introduction

National Park Service Great Lakes Inventory and Monitoring Network (GLKN) staff have been sampling lakes, lagoons, and rivers to establish baseline water quality conditions, monitor water quality for change, compare measured conditions with established criteria, and use these data to direct management efforts. In conjunction with the Network-wide water quality monitoring program, a program to use sediment diatom assemblages to document change in lakes, lagoons, and areas of sediment deposition in rivers (collectively referred to hereafter as “lakes”) is also in place. The sediment diatom monitoring program contains two facets. The first was a paleolimnology component to understand the ecological history of lakes across GLKN parks. A single piston core that preserved approximately 150-200 years of lake history was recovered from index lakes at Sleeping Bear Dunes (SLBE), Pictured Rocks (PIRO), and Apostle Islands (APIS) national lakeshores, and Isle Royale (ISRO) and Voyageurs (VOYA) national parks between 2005 and 2008 (Figure 1). Analysis of cores included radioisotopic dating, geochemistry, and analysis of preserved diatom communities to determine the sedimentation and biological changes in each lake. Most lakes showed increased rates of sediment accumulation and biological changes associated with initial logging and land clearance and more recent changes that could not be readily attributed to current land use patterns (Edlund et al. 2011).

The second aspect of using sediment diatom assemblages for lake monitoring relies on the integrative nature of sediment deposition in lakes to provide a spatially and temporally composited sample of diatoms from the last 2–5 years. Diatoms were initially collected in short sediment cores from the depositional zones of more than 60 GLKN lakes. These samples served two purposes: first as baseline biological assemblage with which to compare future samples, and second, as a training or calibration set of samples where diatom assemblages could be used to estimate water quality parameters. The calibration set was made by exploring the relationship between diatom assemblages and measured environmental parameters (Birks et al. 1990). When diatom assemblages were found to be significantly related to independent water quality variables, models called transfer functions were developed using weighted averaging regression and calibration (Birks et al. 1990) to reconstruct specific environmental variables within GLKN lakes. For GLKN parks, diatom-based inference models were developed to estimate pH, specific conductance (EC25), and total phosphorus (TP) (Edlund et al. 2011). These models were first applied to diatom assemblages preserved in the longer piston cores and showed that few GLKN lakes had any significant patterns of water quality change over the last 150-200 years (Edlund et al. 2011).

The full power of the initial sampling of the GLKN lakes is realized by repeatedly sampling surface sediments from the lakes over three-to-five-year intervals (Edlund et al. 2011). After microscopical analysis of subfossil diatom communities, numerical analysis includes applying diatom inference models to estimate recent TP, pH, and EC25 trends, followed by estimates of floristic change between the repeat and most recent surface sediment collection using the squared chord distance metric (Bennion et al. 2004, Simpson et al. 2005, Ramstack et al. 2008, Edlund et al. 2011). Passive plotting of repeat surface sediment assemblages on ordinations of the GLKN diatom calibration set provides trajectories of change between the repeat and previous surface sediments and includes the

opportunity to consider potential abiotic gradients that might be driving change in the diatom communities (Edlund et al. 2011).



Figure 1. The nine national park units of the Great Lakes Inventory and Monitoring Network. Grand Portage National Monument is not included in this report.

In this report, we present results from the 2011/2012 surface sediment collections made from 21 lakes. Among the GLKN parks, this set of samples represents repeat sediment collections from APIS, Indiana Dunes National Lakeshore (INDU), Mississippi National River and Recreation Area (MISS), St. Croix National Scenic Riverway (SACN), and ISRO, where surface sediment was last collected between 2006 and 2008 (Figure 1).

Methods

Water Quality Sampling

Physical and water chemistry sampling generally takes place three times each field season in GLKN index lakes (Elias et al. 2008). For lakes at APIS, INDU, MISS, and SACN, we report summary statistics for water quality data collected during field years 2007–2011 for most sampling sites. Summary statistics for ISRO lakes are reported for field years 2007–2012 for most lakes. Among this group of GLKN sampling sites, a few lakes were sampled at less frequent intervals, for fewer parameters, and not in every sampling year (Elias and VanderMeulen 2008, Elias 2009, Elias and Damstra 2011, and STORET 2013). Detailed water quality sampling and analytical methods are outlined in Elias et al. (2008).

Surface Sediment Collection

A single core was collected at the same site where water quality was measured using a line-operated gravity corer deployed from an anchored boat. Sediment samples were generally collected during the final visit of each field year (August to October) (Table 1). Surface sediments represent a spatially and temporally integrated assessment of biological remains and lake conditions during the past 3-5 years. Collection sites were located by global positioning system (GPS) to allow resampling in the same location in subsequent years; sampling sites represent central depositional basins in each lake (Ramstack et al. 2008). The upper 0-1 cm and 1-2 cm intervals of each sediment core were subsampled in the field; all sediment subsamples were refrigerated, transported to the St. Croix Watershed Research Station, Science Museum of Minnesota, and processed for diatoms.

Table 1. Lakes at Apostle Islands National Lakeshore (APIS), Indiana Dunes National Lakeshore (INDU), St. Croix National Scenic Riverway (SACN), Mississippi National River and Recreation Area (MISS), and Isle Royale National Park (ISRO) where surface sediments were collected in 2011 and 2012. Collection date and sample site UTM coordinates are shown. Year of previous surface sediment collection also noted. Core depths represent cm below sediment-water interface.

Lake	Park	Date	Depth (cm)	UTM_X	UTM_Y	UTM Zone	Last Sampled
Michigan	APIS	8/31/2011	0-2	689522	5194542	15N	2007
Outer	APIS	8/31/2011	0-2	693138	5209124	15N	2007
Stockton	APIS	8/29/2011	0-2	686672	5199772	15N	2007
Long	INDU	10/21/2011	0-2	482329	4607031	16N	2006
Middle Lagoon	INDU	10/21/2011	0-2	477277	4607079	16N	new
Trego	SACN	9/14/2011	0-2	586299	5088803	15N	2007
St. Croix Falls	SACN	9/14/2011	0-2	527379	5029450	15N	2007
Hudson	SACN	9/14/2011	0-2	519328	4977120	15N	2007
Prescott	SACN	9/14/2011	0-2	515256	4956046	15N	2007

Table 1. Lakes at Apostle Islands National Lakeshore (APIS), Indiana Dunes National Lakeshore (INDU), St. Croix National Scenic Riverway (SACN), Mississippi National River and Recreation Area (MISS), and Isle Royale National Park (ISRO) where surface sediments were collected in 2011 and 2012. Collection date and sample site UTM coordinates are shown. Year of previous surface sediment collection also noted. Core depths represent cm below sediment-water interface (continued).

Lake	Park	Date	Depth (cm)	UTM_X	UTM_Y	UTM Zone	Last Sampled
MS850/852	MISS	9/15/2011	0-2	484144	4973867	15N	2008
Spring Lake	MISS	9/15/2011	0-2	503264	4957001	15N	2006
Harvey	ISRO	8/17/2012	0-1	337146	5302488	16N	2007
Beaver	ISRO	8/20/2012	0-1	369123	5326674	16N	2007
Ahmik	ISRO	8/21/2012	0-2	385341	5333886	16N	2007
Desor	ISRO	8/15/2012	0-1	352187	5315216	16N	2007
George	ISRO	8/15/2012	0-1	374309	5318538	16N	2007
Feldtmann	ISRO	8/27/2012	0-1	337146	5302488	16N	2007
Sargent	ISRO	8/27/2012	0-1	376257	5327793	16N	2007
Richie	ISRO	8/24/2012	0-2	373252	5322262	16N	2007
Siskiwit	ISRO	8/23/2012	0-1	366041	5318296	16N	2008
LeSage	ISRO	8/23/2012	0-1	372652	5324151	16N	new

Sediment Processing and Diatom Analysis

For diatom analysis, sediment samples were prepared by placing approximately 0.25 cm³ of homogenized sediment in a 50 cm³ polycarbonate centrifuge tube and adding 2–5 drops of 10% v/v HCl solution to dissolve carbonates. Organic material was next oxidized by adding 10 ml of 30% hydrogen peroxide, and heating the material in a 85°C water bath for 3 hours. After cooling, material was sequentially rinsed with deionized water with centrifuging between rinses to remove oxidation byproducts. Cleaned material was dispersed onto 22 × 22 mm No. 1 cover glasses, allowed to dry, and then mounted on microscope slides using Zrax mountant (Ramstack et al. 2008).

For most lakes, the 0–1 cm sample was used to prepare diatom slides. For lakes with higher sedimentation rates as determined in Edlund et al. (2011), equal volumes of the 0–1 cm and 1–2 cm samples were combined to better represent the 3–5 year window of sediment accumulation (Edlund et al. 2011, 2012). Any remaining sample material was subsequently freeze-dried and archived in scintillation vials at the St. Croix Watershed Research Station.

Diatoms were identified along up to six random scribed transects of each microscope slide using oil immersion optics capable of 1250X magnification (NA 1.4) using brightfield or differential interference contrast illumination. A minimum of 400 valves was counted in each sample. Identification of diatoms relied on floras and monographs such as Hustedt (1927-1966, 1930), Patrick and Reimer (1966, 1975), Collins and Kalinsky (1977), Camburn et al. (1978, 1984-1986), Krammer and Lange-Bertalot (1986-1991), Cumming et al. (1995), Reavie and Smol (1998), Camburn and Charles (2000), Fallu et al. (2000), and the primary literature (e.g., Koppen 1975). For predominant

taxa present in each slide, digital micrographs were gathered to maintain taxonomic consistency. All diatom counts were converted to percentage abundances by taxon; abundances are reported relative to total diatom counts in each sample (Edlund et al. 2011).

Numerical Analysis

To determine extent of biological and water quality change between sediment sampling events, several numerical and modeling techniques were used to analyze the relationships between the 2011/2012 surface sediment diatom assemblages and information gathered during regular water quality monitoring, surface sediment diatoms sampled in 2006–2008, and historical diatom assemblages preserved in long sediment cores collected in 2007 at APIS and ISRO (Edlund et al. 2011).

Diatoms preserved in surface sediments were first used to quantitatively estimate three water quality parameters: pH, EC25, and TP. Diatom inference models are called transfer functions and were developed earlier for each of these parameters for the GLKN lakes (Edlund et al. 2011) using weighted averaging regression with inverse deshrinking, and bootstrap error estimation (Birks et al. 1990) with the software C2 (Juggins 2003). Transfer functions calculate each taxon's environmental optimum and tolerance for specific environmental variables. To reconstruct a water quality parameter from a diatom community, each taxon's environmental optimum was weighted by its relative abundance in the sample; the summed weighted optima of the entire species assemblage in the diatom community represents the diatom-inferred water quality estimate (Birks et al. 1990).

Next, to quantify the amount of biological change between surface sediment sampling events, we used a measure called the squared chord distance. Squared chord distance is a dissimilarity measure between samples with values ranging from 0 (samples are exactly the same) to 2 (samples are completely different). Meaningful change can be assessed using percentile cutoffs from randomly generated, approximately normally distributed data that range from the lower to upper limits of the squared chord distance (0 to 2) (Bennion et al. 2004, Simpson et al. 2005). As such, squared chord distances greater than 0.39 represent meaningful floristic change; this value was determined by Simpson et al. (2005) and Simpson (personal communication) and represents the fifth percentile in distribution of pairwise comparisons based on the above method. Squared chord distances less than the fifth percentile limit represent low floristic change (Bennion et al. 2004; Simpson, personal communication). Squared chord distances were estimated for surface sediments collected in 2011 (from APIS, INDU, SACN, and MISS) and in 2012 (from ISRO) against surface sediment diatom assemblages collected from corresponding lakes in these parks between 2006 and 2008 (Table 1) (Edlund et al. 2011). For lakes that have had long sediment cores collected from them, squared chord differences were previously calculated between samples dated from pre-Euro-American settlement and those dated from park formation, and between park formation and the uppermost sediments of each core to provide additional context to recent changes in GLKN diatom communities (Edlund et al. 2011)

Third, to explore potential environmental drivers of biological change, diatom assemblages from the 2011/2012 repeat samples were passively plotted on the GLKN calibration set canonical correspondence analysis (CCA). The trajectory between the 2006–2008 surface sediment sample and

the 2011/2012 repeat sample from each lake was identified and considered in relationship to the main environmental axes that explained independent variation of diatom abundances in the GLKN surface sediment calibration set (Ramstack et al. 2008; Edlund et al. 2011, 2012). These analyses were implemented with the software package R (Ihaka and Gentleman 1996). Several sites had to be removed from this analysis. Samples from MISS and APIS were shown to be outliers based on either water quality parameters or diatom assemblages during construction of the GLKN diatom calibration set (Edlund et al. 2011).

The final analysis on repeat surface sediment samples compared them with long sediment cores collected earlier from GLKN lakes at ISRO (Richie, Harvey, and Ahmik) and APIS (Outer lagoon). For each repeat sample from these lakes, we re-ran a detrended correspondence analysis (DCA) (R software; Ihaka and Gentleman 1996) using all downcore samples with the 2011/2012 repeat sample included. All core depths were plotted in ordination space (by ^{210}Pb date), and their relationships and variability were used to identify sample groups, ecological variability among core samples, and periods of change, including recent change (Edlund et al. 2011, 2012).

Results

Water Quality, 2006–2012

Many physical and water quality parameters are measured in conjunction with the inland lakes and large rivers water quality monitoring program for the GLKN parks. However, we consider here only those parameters that have been shown to have significant and independent relationships with diatom assemblages. These parameters, which were determined based on surface sediment sampling and water quality monitoring reported by Edlund et al. (2011), include pH, specific conductance (EC25), and total phosphorus (TP). Minimum, maximum, mean, and median values for these parameters are summarized in Table 2.

Apostle Islands National Lakeshore (APIS)

Three sites are regularly sampled at APIS: Outer, Michigan, and Stockton lagoons. These systems can be generally described as circumneutral to slightly acidic, with low conductance and low productivity. Michigan lagoon has slightly higher conductance, whereas Outer lagoon has the lowest conductance among these sites. Stockton lagoon, which has the greatest connectivity with Lake Superior waters, showed the greatest variation in water quality parameters among the APIS sites (Table 2).

Indiana Dunes National Lakeshore (INDU)

Long Lake and Middle Lagoon are routinely sampled for water quality. Long Lake has mesotrophic levels of total phosphorus, circumneutral pH, and median conductance of 270 $\mu\text{S}/\text{cm}$. Middle Lagoon has more alkaline pH, higher conductance (median 545 $\mu\text{S}/\text{cm}$), and similar mesotrophic levels of TP (Table 2).

Isle Royale National Park (ISRO)

Ten lakes on Isle Royale are included in the GLKN water quality monitoring program (Table 2). Lakes on ISRO are circumneutral and range from the oligotrophic Siskiwit Lake (median TP 4.7 $\mu\text{g}/\text{L}$) to the periodically eutrophic Lake Richie (median TP 23.3 $\mu\text{g}/\text{L}$, maximum TP 45 $\mu\text{g}/\text{L}$). Median values of EC25 in ISRO lakes range from 69 to 120 $\mu\text{S}/\text{cm}$, with the lowest values from Lake LeSage and highest values from Lake George. There are heightened concerns over water quality in ISRO lakes, as several have experienced unprecedented cyanobacterial blooms (Edlund et al. 2010) and deepening of the epilimnion (Saros et al. 2012).

Mississippi National River and Recreation Area (MISS)

Two water quality monitoring stations on the Mississippi River represent potential sediment depositional basins. Spring Lake below the Twin Cities and the reach upstream of the Ford Dam (MS850/852) are sampled regularly as part of the GLKN large rivers monitoring program. Water quality along MISS is highly variable. The upstream site at MS850/852 has lower, but eutrophic levels of TP (2006–2010 median TP 76.5 $\mu\text{g}/\text{L}$) than the Spring Lake site (median TP 145.5 $\mu\text{g}/\text{L}$). Similarly, EC25 is higher at the downstream Spring Lake site, with a median value of 636 $\mu\text{S}/\text{cm}$, compared to 405 $\mu\text{S}/\text{cm}$ at MS850/852. Both sites typically show slightly alkaline pH.

Table 2. Summary of minimum, maximum, mean and median water quality measures from 21 GLKN lakes, 2006-2012. Years of available data for each sampling site also noted. Data are presented for total phosphorus (TP), pH, and specific conductance (EC25). Data recovered from STORET (2013) and from Elias and Sieracki (2007), Elias and VanderMeulen (2008), VanderMeulen and Elias (2008), Elias (2009), and Elias and Damstra (2011, 2012).

Lake	Park	Total Phosphorus, TP (µg/L)				Specific conductance, EC25 (µS/cm)				pH				Years
		<i>min</i>	<i>max</i>	<i>mean</i>	<i>median</i>	<i>min</i>	<i>max</i>	<i>mean</i>	<i>median</i>	<i>min</i>	<i>max</i>	<i>mean</i>	<i>median</i>	
Michigan	APIS	8.0	18.2	12.9	13.0	75.0	171.0	103.3	102.0	6.4	7.8	7.4	7.4	07-11
Outer	APIS	6.0	43.9	12.8	11.0	19.0	41.0	24.7	23.0	5.8	7.7	6.8	6.8	07-11
Stockton	APIS	4.7	41.2	14.9	13.0	24.0	96.0	55.3	58.0	5.2	7.9	6.7	6.7	07-11
Long	INDU	5.0	32.0	21.2	23.0	212.1	658.8	291.7	269.6	5.8	8.1	7.0	6.9	06-11
Middle Lagoon	INDU	10.2	32.7	18.9	16.8	375.0	916.0	581.9	545.5	6.5	9.9	8.5	8.3	07-11
Trego	SACN	13.0	55.5	27.0	23.0	109.9	190.3	164.4	169.9	6.5	8.2	7.7	7.8	07-11
St. Croix Falls	SACN	22.4	96.0	42.7	38.0	82.5	204.7	160.2	157.4	6.8	8.6	7.7	7.8	07-11
Hudson	SACN	28.8	71.6	46.8	45.3	81.2	252.0	179.0	177.0	7.0	8.7	7.7	7.6	07-11
Prescott	SACN	18.8	133.0	44.8	41.7	106.4	321.1	194.7	197.0	6.7	9.1	7.5	7.5	07-11
MS850/852	MISS	22.0	137.7	76.9	76.5	286.2	468.4	399.6	404.9	7.8	9.2	8.3	8.3	06-10
Spring Lake	MISS	81.0	236.5	146.8	145.5	522.0	743.7	635.9	636.2	7.8	8.9	8.2	8.2	06-10
Ahmik	ISRO	13.0	30.3	19.1	17.4	85.0	203.0	105.2	102.0	6.8	8.0	7.4	7.5	07-12
Beaver	ISRO	12.0	21.5	17.3	17.7	89.4	143.7	100.3	99.0	6.7	8.1	7.6	7.7	07-12
Desor	ISRO	8.4	13.0	11.1	11.0	99.0	160.6	109.4	106.4	6.8	8.8	7.8	8.0	07-12
Feldtmann	ISRO	11.0	15.0	12.6	12.6	66.0	87.0	75.9	76.0	7.4	8.7	7.8	7.8	07-12
George	ISRO	7.5	12.4	9.9	10.1	103.8	134.3	121.0	120.0	6.8	8.9	8.0	7.9	07-12
Harvey	ISRO	11.0	21.0	14.4	14.4	84.0	235.3	93.1	89.9	6.9	9.1	8.4	8.5	07-12
LeSage	ISRO	11.8	16.3	14.7	15.9	59.5	103.3	72.4	69.1	6.7	7.7	7.1	7.1	11-12
Richie	ISRO	15.0	45.0	24.7	23.3	72.0	137.5	85.0	81.8	6.6	9.6	7.5	7.4	07-12
Sargent	ISRO	11.4	24.2	14.7	13.0	82.0	148.0	93.5	91.3	6.7	8.7	7.5	7.5	07-12
Siskiwit	ISRO	3.1	5.2	4.4	4.7	67.0	76.2	70.3	70.6	6.9	8.2	7.7	7.7	07-12

St. Croix National Scenic Riverway (SACN)

Four depositional basins along the St. Croix River are sampled for water quality and surface sediments: Trego Flowage, the impoundment at St. Croix Falls, and two sites in Lake St. Croix near Hudson and Prescott, Wisconsin. Water quality varies along the length of the St. Croix River. The most upstream site at Trego Flowage generally has mesotrophic levels of TP, whereas Lake St. Croix sites are typically more eutrophic and even experience periodic cyanobacteria blooms. Conductance of the St. Croix's circumneutral waters also increases slightly moving toward the downstream stations (Table 2).

Surface Sediment Diatom Assemblages

Diatom assemblages preserved in the 2011/2012 surface sediments from APIS, INDU, ISRO, MISS, and SACN cover a large range of habitat and lake types and contained over 650 diatom taxa. The predominant diatoms (>5% relative abundance in order from the most abundant) in each lake's sediment sample are shown in Table 3.

Table 3. Predominant diatoms (>5% relative abundance) in sediment samples collected from Apostle Islands National Lakeshore (APIS), Indiana Dunes National Lakeshore (INDU), Isle Royale National Park (ISRO), Mississippi National River and Recreation Area (MISS), and St. Croix National Scenic Riverway (SACN) in 2011 and 2012.

2011 Lakes	Diatoms (in order of abundance)
Michigan (APIS)	<i>Denticula kuetzingiana</i> , <i>Encyonema silesiacum</i> , <i>Achnantheidium minutissimum</i> , <i>Encyonopsis cesatii</i> , <i>Delicata delicatula</i>
Outer (APIS)	<i>Stauroforma exiguiiformis</i> , <i>Brachysira neoexilis</i> , <i>Frustulia rhomboides</i>
Stockton (APIS)	<i>Stauroforma exiguiiformis</i> , <i>Achnantheidium minutissimum</i> , <i>Brachysira neoexilis</i> , <i>Nitzschia perminuta</i>
Long (INDU)	<i>Staurosira construens</i> , <i>Staurosirella pinnata</i> , <i>Pseudostaurosira brevistriata</i> , <i>Staurosira venter</i> , <i>Navicula leptostriata</i> , <i>Staurosira elliptica</i> GLEI
Middle Lagoon (INDU)	<i>Achnantheidium minutissimum</i> , <i>Gomphonema angustum</i> , <i>Cocconeis placentula</i> var. <i>lineata</i> , <i>Fragilaria capucina</i> var. <i>mesolepta</i> , <i>Nitzschia amphibia</i>
Trego (SACN)	<i>Pseudostaurosira brevistriata</i> , <i>Cocconeis placentula</i> var. <i>lineata</i> , <i>Staurosirella pinnata</i> , <i>Staurosira venter</i>
St. Croix Falls (SACN)	<i>Aulacoseira ambigua</i> , <i>Cocconeis placentula</i> var. <i>lineata</i> , <i>Pseudostaurosira brevistriata</i> , <i>Staurosira venter</i>
Hudson (SACN)	<i>Aulacoseira ambigua</i> , <i>Aulacoseira granulata</i> , <i>Pseudostaurosira brevistriata</i>

Table 3. Predominant diatoms (>5% relative abundance) in sediment samples collected from Apostle Islands National Lakeshore (APIS), Indiana Dunes National Lakeshore (INDU), Isle Royale National Park (ISRO), Mississippi National River and Recreation Area (MISS), and St. Croix National Scenic Riverway (SACN) in 2011 and 2012 (continued).

2011 Lakes	Diatoms (in order of abundance)
Prescott (SACN)	<i>Aulacoseira ambigua</i> , <i>Aulacoseira granulata</i> , <i>Staurosira venter</i> , <i>Fragilaria crotonensis</i>
MS850/852 (MISS)	<i>Cocconeis placentula</i> var. <i>lineata</i> , <i>Aulacoseira granulata</i> , <i>Pseudostaurosira brevistriata</i>
Spring Lake (MISS)	<i>Aulacoseira granulata</i> , <i>Cyclotella meneghiniana</i> , <i>Aulacoseira ambigua</i> , <i>Cocconeis placentula</i> var. <i>lineata</i>
2012 Lakes	Diatoms (in order of abundance)
Harvey (ISRO)	<i>Staurosirella pinnata</i> , <i>Staurosira construens</i> , <i>Staurosira elliptica</i> GLEI, <i>Staurosira venter</i>
Beaver (ISRO)	<i>Diploneis marginestriata</i> , <i>Discostella stelligera</i> , <i>Staurosirella pinnata</i> , <i>Staurosira venter</i>
Ahmik (ISRO)	<i>Staurosira venter</i> , <i>Staurosirella pinnata</i> , <i>Staurosira construens</i>
Desor (ISRO)	<i>Cyclotella comensis</i> , <i>Aulacoseira ambigua</i>
George (ISRO)	<i>Staurosira venter</i> , <i>Achnanthes saccula</i> , <i>Achnantheidium minutissimum</i>
Feldtmann (ISRO)	<i>Achnantheidium minutissimum</i> , <i>Staurosira venter</i> , <i>Fragilaria vaucheriae</i>
Sargent (ISRO)	<i>Aulacoseira subarctica</i> , <i>Aulacoseira ambigua</i> , <i>Fragilaria crotonensis</i> , <i>Ulnaria delicatissima</i> , <i>Cyclotella</i> sp. <i>mich-like</i>
Richie (ISRO)	<i>Aulacoseira ambigua</i> , <i>Fragilaria crotonensis</i> , <i>Aulacoseira subarctica</i> , <i>Asterionella formosa</i> , <i>Synedra nana</i> , <i>Tabellaria flocculosa</i> IIIp
Siskiwit (ISRO)	<i>Cyclotella comensis</i> , <i>Cyclotella</i> sp. <i>mich-like</i> , <i>Discostella stelligera</i> , <i>Cyclotella delicatula</i> , <i>Cyclotella michiganiana</i>
LeSage (ISRO)	<i>Discostella stelligera</i> , <i>Aulacoseira tenella</i> , <i>Aulacoseira ambigua</i> , <i>Staurosira venter</i>

Ecological Change in GLKN Lakes, 2006–2012

Diatom-inferred water quality measures

Quantitative estimates of diatom-inferred (DI) TP, EC25, and pH were made using transfer functions developed for GLKN lakes by Edlund et al. (2011). We report reconstructions by park unit and compare inferred values with monitored values (Tables 4A–4C).

APIS (2011): Diatom-inferred TP estimates from the three APIS lagoons reconstruct very close to monitored values from 2007–2011, which generally measured as mesotrophic to oligotrophic. Median measured TP values among the three lagoons ranged from 11 to 13 µg/L, and DI-TP ranged from 8.5 to 10.0 µg/L. Reconstructed values of EC25 ranged from 25.4 to 69.7 µS/cm at APIS, closely matching the monitored values in these low conductance lagoons (Table 4A). The DI-EC25 value for Michigan lagoon was slightly lower than the measured range of EC25 values (Table 4B); however, error estimates based on model root mean squared error of predictions (RMSEP) captured the median EC25 at Michigan (102 µS/cm). Diatom-inferred pH estimates from APIS repeat surface sediments (pH 6.9–7.8) corresponded to monitoring data collected between 2007 and 2011 when median pH values were 6.7 to 7.4 (Table 4C).

Table 4A. Diatom-inferred (DI) estimates of specific conductance (EC25) based on transfer functions developed for the GLKN lakes by Edlund et al. (2011). Model estimates of EC25 are calculated in log units but have been back-transformed to µg/L EC25. Model error estimates are given for the low and high range of DI-EC25 based on model root mean squared error of predictions (RMSEP) (Edlund et al. 2011). The range and median measures of EC25 are also given for each lake based on monitoring data from 2006–2012 (Elias and Sieracki 2007, Elias and VanderMeulen 2008, VanderMeulen and Elias 2008, Elias 2009, Elias and Damstra 2011, 2012, STORET 2013).

Lake	Park	Specific Conductance (µS/cm)					
		measured EC25			diatom-inferred EC25		
		min	max	med	EC25	low range	high range
Michigan	APIS	75.0	171.0	102.0	69.7	46.3	104.9
Outer	APIS	19.0	41.0	23.0	25.4	16.9	38.3
Stockton	APIS	24.0	96.0	58.0	30.7	20.4	46.2
Long	INDU	212.1	658.8	269.6	159.2	105.8	239.6
Middle Lagoon	INDU	375.0	916.0	545.5	122.2	81.2	183.8
Trego	SACN	109.9	190.3	169.9	185.5	123.3	279.1
St. Croix Falls	SACN	82.5	204.7	157.4	154.1	102.4	231.9
Hudson	SACN	81.2	252.0	177.0	155.7	103.5	234.3
Prescott	SACN	106.4	321.1	197.0	161.7	107.5	243.3
MS850/852	MISS	286.2	468.4	404.9	242.5	161.2	364.9
Spring Lake	MISS	522.0	743.7	636.2	195.7	130.1	294.5
Ahmik	ISRO	85.0	203.0	102.0	130.5	86.7	196.4

Table 4A. Diatom-inferred (DI) estimates of specific conductance (EC25) based on transfer functions developed for the GLKN lakes by Edlund et al. (2011). Model estimates of EC25 are calculated in log units but have been back-transformed to µg/L EC25. Model error estimates are given for the low and high range of DI-EC25 based on model root mean squared error of predictions (RMSEP) (Edlund et al. 2011). The range and median measures of EC25 are also given for each lake based on monitoring data from 2006–2012 (Elias and Sieracki 2007, Elias and VanderMeulen 2008, VanderMeulen and Elias 2008, Elias 2009, Elias and Damstra 2011, 2012, STORET 2013) (continued).

Lake	Park	Specific Conductance (µS/cm)					
		<i>measured EC25</i>			<i>diatom-inferred EC25</i>		
		<i>min</i>	<i>max</i>	<i>med</i>	<i>EC25</i>	<i>low range</i>	<i>high range</i>
Beaver	ISRO	89.4	143.7	99.0	67.9	45.1	102.2
Desor	ISRO	99.0	160.6	106.4	149.6	99.4	225.0
Feldtmann	ISRO	66.0	87.0	76.0	90.6	60.2	136.3
George	ISRO	103.8	134.3	120.0	107.0	71.1	161.0
Harvey	ISRO	84.0	235.3	89.9	241.8	160.7	363.9
LeSage	ISRO	59.5	103.3	69.1	39.8	26.4	59.9
Richie	ISRO	72.0	137.5	81.8	57.5	38.2	86.5
Sargent	ISRO	82.0	148.0	91.3	83.4	55.4	125.4
Siskiwit	ISRO	67.0	76.2	70.6	167.5	111.3	252.0

Table 4B. Diatom-inferred (DI) estimates of pH based on transfer functions developed for the GLKN lakes by Edlund et al. (2011). Model error estimates are given for the low and high range of DI-pH based on model root mean squared error of predictions (RMSEP) (Edlund et al. 2011). The range and median measures of pH are given for each lake based on monitoring data from 2006–2012 (Elias and Sieracki 2007, Elias and VanderMeulen 2008, VanderMeulen and Elias 2008, Elias 2009, Elias and Damstra 2011, 2012, STORET 2013).

Lake	Park	pH					
		<i>measured pH</i>			<i>diatom-inferred pH</i>		
		<i>min</i>	<i>max</i>	<i>med</i>	<i>pH</i>	<i>low range</i>	<i>high range</i>
Michigan	APIS	6.4	7.8	7.4	7.8	7.4	8.2
Outer	APIS	5.8	7.7	6.8	6.9	6.5	7.3
Stockton	APIS	5.2	7.9	6.7	7.1	6.6	7.5
Long	INDU	5.8	8.1	7.0	7.9	7.5	8.3
Middle Lagoon	INDU	6.5	9.9	8.5	7.8	7.4	8.2
Trego	SACN	6.5	8.2	7.7	7.9	7.5	8.3
St. Croix Falls	SACN	6.8	8.6	7.7	7.8	7.4	8.2
Hudson	SACN	7.0	8.7	7.7	7.9	7.5	8.3
Prescott	SACN	6.7	9.1	7.5	7.9	7.5	8.3

Table 4B. Diatom-inferred (DI) estimates of pH based on transfer functions developed for the GLKN lakes by Edlund et al. (2011). Model error estimates are given for the low and high range of DI-pH based on model root mean squared error of predictions (RMSEP) (Edlund et al. 2011). The range and median measures of pH are given for each lake based on monitoring data from 2006–2012 (Elias and Sieracki 2007, Elias and VanderMeulen 2008, VanderMeulen and Elias 2008, Elias 2009, Elias and Damstra 2011, 2012, STORET 2013) (continued).

Lake	Park	pH					
		<i>measured pH</i>			<i>diatom-inferred pH</i>		
		<i>min</i>	<i>max</i>	<i>med</i>	<i>pH</i>	<i>low range</i>	<i>high range</i>
MS850/852	MISS	7.8	9.2	8.3	8.0	7.5	8.4
Spring Lake	MISS	7.8	8.9	8.2	7.9	7.5	8.3
Ahmik	ISRO	6.8	8.0	7.5	7.9	7.4	8.3
Beaver	ISRO	6.7	8.1	7.7	7.6	7.2	8.0
Desor	ISRO	6.8	8.8	8.0	8.3	7.9	8.7
Feldtmann	ISRO	7.4	8.7	7.8	7.7	7.3	8.1
George	ISRO	6.8	8.9	7.9	7.9	7.5	8.4
Harvey	ISRO	6.9	9.1	8.5	8.3	7.9	8.7
LeSage	ISRO	6.7	7.7	7.1	7.3	6.9	7.7
Richie	ISRO	6.6	9.6	7.4	7.5	7.1	8.0
Sargent	ISRO	6.7	8.7	7.5	7.9	7.5	8.3
Siskiwit	ISRO	6.9	8.2	7.7	8.4	8.0	8.8

Table 4C. Diatom-inferred (DI) estimates of total phosphorus (TP) based on transfer functions developed for the GLKN lakes by Edlund et al. (2011). Model estimates of TP are calculated in log units but have been back-transformed to µg/L TP. Model error estimates are given for the low and high range of DI-TP based on model root mean squared error of predictions (RMSEP) (Edlund et al. 2011). The range and median measures of TP are also given for each lake based on monitoring data from 2006–2012 (Elias and Sieracki 2007, Elias and VanderMeulen 2008, VanderMeulen and Elias 2008, Elias 2009, Elias and Damstra 2011, 2012, STORET 2013).

Lake	Park	Total Phosphorus (µg/L)					
		<i>measured TP</i>			<i>diatom-inferred TP</i>		
		<i>min</i>	<i>max</i>	<i>med</i>	<i>TP</i>	<i>low range</i>	<i>high range</i>
Michigan	APIS	8.0	18.2	13.0	10.0	6.5	15.5
Outer	APIS	6.0	43.9	11.0	8.5	5.5	13.1
Stockton	APIS	4.7	41.2	13.0	8.6	5.6	13.3
Long	INDU	5.0	32.0	23.0	17.2	11.2	26.5
Middle Lagoon	INDU	10.2	32.7	16.8	19.0	12.3	29.3
Trego	SACN	13.0	55.5	23.0	27.0	17.6	41.6

Table 4C. Diatom-inferred (DI) estimates of total phosphorus (TP) based on transfer functions developed for the GLKN lakes by Edlund et al. (2011). Model estimates of TP are calculated in log units but have been back-transformed to $\mu\text{g/L}$ TP. Model error estimates are given for the low and high range of DI-TP based on model root mean squared error of predictions (RMSEP) (Edlund et al. 2011). The range and median measures of TP are also given for each lake based on monitoring data from 2006–2012 (Elias and Sieracki 2007, Elias and VanderMeulen 2008, VanderMeulen and Elias 2008, Elias 2009, Elias and Damstra 2011, 2012, STORET 2013) (continued).

Lake	Park	Total Phosphorus ($\mu\text{g/L}$)					
		measured TP			diatom-inferred TP		
		min	max	med	TP	low range	high range
St. Croix Falls	SACN	22.4	96.0	38.0	24.9	16.1	38.3
Hudson	SACN	28.8	71.6	45.3	28.5	18.5	44.0
Prescott	SACN	18.8	133.0	41.7	36.4	23.7	56.1
MS850/852	MISS	22.0	137.7	76.5	38.2	24.8	58.8
Spring Lake	MISS	81.0	236.5	145.5	53.5	34.7	82.4
Ahmik	ISRO	13.0	30.3	17.4	18.2	11.8	28.1
Beaver	ISRO	12.0	21.5	17.7	12.8	8.3	19.8
Desor	ISRO	8.4	13.0	11.0	9.1	5.9	14.1
Feldtmann	ISRO	11.0	15.0	12.6	14.9	9.7	22.9
George	ISRO	7.5	12.4	10.1	14.5	9.4	22.4
Harvey	ISRO	11.0	21.0	14.4	18.4	11.9	28.3
LeSage	ISRO	11.8	16.3	15.9	10.0	6.5	15.4
Richie	ISRO	15.0	45.0	23.3	11.3	7.3	17.4
Sargent	ISRO	11.4	24.2	13.0	12.0	7.8	18.5
Siskiwit	ISRO	3.1	5.2	4.7	6.2	4.0	9.5

INDU (2011): The two sites monitored for water quality and surface sediments at INDU had DI-TP reconstructions of 17.2 $\mu\text{g/L}$ (Long) and 19.0 $\mu\text{g/L}$ (Middle Lagoon) that compare well with measured median TP values of 23.0 and 16.8 $\mu\text{g/L}$, respectively (Table 4A). In contrast, DI-EC25 returned estimates that were lower than measured values. Long Lake DI-EC25 was 159.2 $\mu\text{S/cm}$ and Middle Lagoon was 122.2 $\mu\text{S/cm}$ compared to median values from 2006–2011 of 270 and 545 $\mu\text{S/cm}$ in these two systems (Table 4B). Model error estimates do capture the lower range of measured EC25 values at Long Lake, but reconstructed values from Middle Lagoon are markedly lower than any measured values. The dominance of *Achnanthes minutissimum* in the 2011 sample, whose EC25 optimum is only 70.8 $\mu\text{S/cm}$, drives the reconstruction down. This diatom is common across a wide range of habitats, and its abundance is most likely determined by habitat availability, especially presence of aquatic macrophytes. Estimates of DI-pH at the INDU sites show good correspondence with monitored pH data; DI-pH at Long Lake was 7.9 and Middle Lagoon was 7.8 (Table 4C).

ISRO (2012): Ten lakes on ISRO were monitored for water quality between 2007 and 2012 (LeSage was only sampled in 2011 and 2012). Diatom-based transfer functions were applied to surface sediments collected in 2012 to generate estimates of TP, EC25, and pH (Tables 4A–C). Diatom-

inferred TP ranged from 6.2 to 18.4 $\mu\text{g/L}$, corresponding closely to median monitored TP values that ranged from 4.7 to 23.3 $\mu\text{g/L}$. Two lakes that reconstructed slightly low were LeSage and Richie, where DI-TP was lower than has ever been measured in these lakes. In both cases, however, model error estimates do capture much of the range of measured TP values. In contrast, George and Siskiwit lakes reconstructed a bit higher than measured values. These lakes exhibit some of the lowest productivity at ISRO and among the GLKN parks, which makes it difficult for the model to reconstruct the lower end of the TP gradient. Model error estimates for these two lakes do include much of the measured range of TP values, relieving concerns that these lakes may be increasing in productivity. Estimates of DI-EC25 ranged from 39.8 to 242 $\mu\text{S/cm}$, with LeSage and Harvey, respectively, representing the extremes and most lakes reconstructing close to their measured values. DI-EC25 from Beaver, LeSage, and Richie were slightly lower than monitored records. Error estimates from Beaver and Richie overlapped with the range of monitored values, but the LeSage estimate was well below the range of conductance measured in 2011–2012. Only one of the four most common diatoms in LeSage had an EC25 optimum in the range of measured values for this lake, which is why the reconstruction is too low. The predominant taxa, which are rather widespread among GLKN lakes and have broad EC25 tolerances, may also contribute to the poor reconstruction. Siskiwit Lake, on the other hand, had a high DI-EC25 (167.5 $\mu\text{S/cm}$) compared to measured values (median 70.6 $\mu\text{S/cm}$). Siskiwit Lake is dominated by several small *Cyclotella* taxa that are also common in PIRO and SLBE lakes where conductance values are significantly higher. These taxa (*C. comensis* and *C. sp. mich-like*) drive the elevated DI-EC25 reconstruction in Siskiwit Lake. The DI-pH estimates are very close to the circumneutral values that characterize the monitored ISRO lakes (Table 4C).

MISS (2011): Diatom-based reconstructions of water quality parameters were most disparate from measured parameters at the two Mississippi River sites (Table 4A–C). Part of the problem with these sites is that they were identified as outliers and removed during the construction of the original calibration set and transfer functions, based on species assemblages and/or water chemistry gradients (Edlund et al. 2011). The DI-TP for sediments collected in 2011 were 24.8 and 34.7 $\mu\text{g/L}$ at MS850/852 and Spring Lake, respectively (Table 4A). Both of these values are rather low compared to median monitored TP values of 76.5 and 145.5 $\mu\text{g/L}$ at the two sites. Model error estimates do capture much of the measured range of TP values at MS850/852. The underestimates of DI-TP at Spring Lake are an artifact of model construction; one of the dominant diatoms in the MISS samples, the eutrophic indicator species *Cyclotella meneghiniana*, is not included in our transfer function. Other predominant diatoms at this site (*Aulacoseira granulata*, *A. ambigua*, and *Cocconeis placentula* v. *lineata*) have modest TP optima of only 14.5 to 38.9 $\mu\text{g/L}$, which drive the DI-TP values too low. Similarly, the DI-EC25 at the two MISS sites reconstructs low (242 and 195 $\mu\text{S/cm}$ at MS850/852 and Spring Lake, respectively) compared to the fairly high measured values of specific conductance (median values of 404 and 636 $\mu\text{S/cm}$) at the two sites (Table 4B). Because the MISS sites represented outliers during the construction of the diatom calibration set, the diatom species optima for the GLKN lakes only range from 23 to 302 $\mu\text{S/cm}$ (Edlund et al. 2011), which eliminates the possibility of reconstructing near the monitored EC25 at MISS. In contrast, the DI-pH values of the MISS sites (pH 7.9–8.0) reconstruct close to monitored values (median pH range 8.2–8.3) (Tables 2 and 4C).

SACN (2011): Surface sediments collected at four sites in SACN in 2011 had DI-TP values ranging from 24.9 to 36.4 µg/L (Table 4A). Inferred values from Trego, St. Croix Falls, and Prescott were similar to monitored values, whereas the DI-TP of 28.5 µg/L at Hudson was a bit lower than the median TP of 45.3 µg/L recorded between 2007 and 2011. However, DI-TP error estimates of 18.5–44.0 µg/L do encompass much of the range of monitored TP values recorded at Hudson (Table 4A). Monitored EC25 is somewhat variable along the St. Croix River, but DI-EC25 values ranging from 154.1 to 185.5 µS/cm corresponded nicely with measured median values of 157.4 to 197.0 µS/cm. Diatom inferred estimates of pH were 7.8–7.9 at the SACN sites, closely aligned with measured values from 2007–2011 (Table 4C).

Diatom community change

Biological community change was quantified in the GLKN lakes and lagoons using the squared chord distance between the 2011/2012 samples and their corresponding surface sediment samples collected in 2006–2008 (Figure 2). In contrast to results of the 2010 surface sediment analysis from VOYA lakes that showed most lakes with low levels of biological change, there were many sites sampled in 2011/2012 that showed significant biological change (Figure 2). We highlight the lakes and lagoons that showed significant changes by park unit, giving consideration to diatom taxa that appeared, disappeared, or shifted significantly in abundance between samplings. We also provide ecological context for these changes.

APIS: All three sites in APIS showed significant biological change in sediment diatom assemblages between 2007 and 2011. Four predominant diatom taxa in 2011 were not found in the 2007 Michigan Island lagoon sample: *Delicata delicatula* (0% to 6%), *Encyonopsis microcephala* (0% to 2.89%), *Navicula salinarum* (0% to 2.89%), and *Eunotia praeurupta* (0% to 2%). One taxon found in 2007 was absent in 2011: *Encyonopsis subminuta* (4.79% to 0%). Other taxa that showed >2% changes in relative abundance between 2007 and 2011 were *Navicula cryptotenella* (17.3% to 4.67%), *N. cryptocephala* (5.18% to 2%), and *Encyonema silesiacum* (1.5% to 13.1%). The Michigan Island lagoon is a shallow site with strong wetland character, and the sediment assemblage is dominated by benthic diatom species. The species that changed in abundance were taxa that we expect in this region and type of habitat.

Between 2007 and 2011, Outer Island lagoon exhibited significant gains or appearances of several subdominant taxa including *Encyonema hebridicum* (0% to 2.6%), *Brachysira vitrea* (0% to 3.4%), *Navicula leptostriata* (0.4% to 2.6%), *E. neogracile* (1.6% to 3.6%), *B. neoexilis* (4.3% to 7.5%), and *Stauroforma exiguiiformis* (2.7% to 15.2%). Significant losses or declines in abundance were seen between 2007 and 2011 in *Pseudostaurosira microstriata* (23.1% to 0.9%), *Eunotia faba* (6.7% to 2.1%), *Semiorbis hemicyclus* (5.3% to 2.6%), *Nitzschia perminuta* (5.9% to 2.8%), *Navicula cryptotenella* (3.9% to 0.4%), *B. styriaca* (3.67% to 0%), *Psammothidium altaicum* (3.67% to 0.9%), *Navicula mediocris* (2.2% to 0.6%), and *Achnantheidium thermale* (2% to 0%). Outer Island lagoon has a unique diatom flora among the GLKN lakes (see cover image of *Semiorbis hemicyclus*) that is characterized by a soft water and slightly acidophilic flora. Its shallow nature supports a rich benthic diatom flora whose abundances seem to vary, sometimes dramatically (e.g. *Pseudostaurosira microstriata*) in recent years.

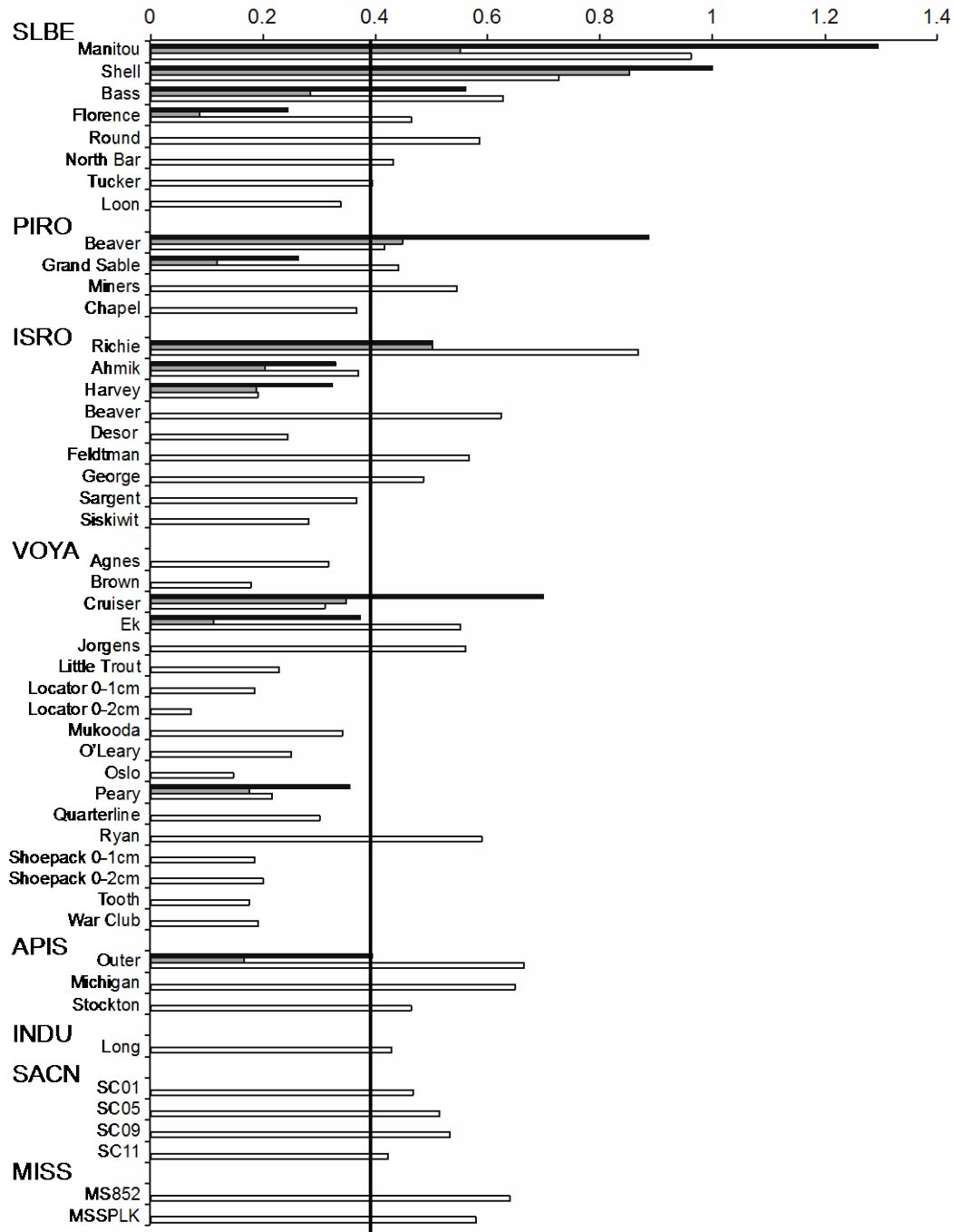


Figure 2. Plot of squared chord distances for long sediment cores and surface repeat samples from GLKN park lakes. *Black-filled bars* represent squared chord distance between pre-Euro-American settlement samples and top-of-core samples; *gray-filled bars* represent squared chord distance between samples deposited at park formation and top-of-core samples; *white bars* represent distance between calibration set samples collected in 2005 (SLBE and PIRO lakes), 2006 (VOYA, INDU, and MISS Spring Lake), 2007 (ISRO, APIS, and SACN) or 2008 (MISS MS850/852, ISRO Siskiwit) compared to surface sediment samples collected in 2008 (PIRO), 2009 (SLBE), 2010 (VOYA), 2011 (APIS, INDU, SACN, MISS), and 2012 (ISRO). The line at 0.39 represents the cutoff for significant floristic change (see text and Simpson et al. 2005).

In 2011 Stockton Island lagoon gained several new, but not unexpected taxa in its flora that were not present in 2007: *Eucoconeis flexella* v. *alpestris* (0% to 2.5%) and *Tabellaria flocculosa* (strain undetermined; 0% to 2.3%). Other taxa that increased >2% in abundance between 2007 and 2011 included *Stauroforma exiguiiformis* (5% to 15.7%) and *Nitzschia perminuta* (1.9% to 5.4%). Diatom taxa that decreased in abundance between sampling years or were notably absent in the 2011 sediment sample were *Pseudostaurosira microstriata* (9.6% to 0%), *Rossithidium linearis* (2.7% to 0%), *Nitzschia sublinearis* (2.3% to 0%), *Psammothidium rossii* (2.3% to 0%), and *Frustulia rhomboides* (4% to 1.8%). Future diatom monitoring will show if the large community changes seen between APIS repeat samples are characteristic of these lagoon habitats, which can be very dynamic systems due to their location and connectivity to Lake Superior.

INDU: Although water quality has been monitored at INDU's Middle Lagoon since 2007, the first sediment sample collected from Middle Lagoon was taken in 2011; this sample will serve as the basis for evaluating future change in its diatom communities. In contrast, the first surface sediment sample was taken from Long Lake in 2006, and the repeat sample collected in 2011 showed significant levels of biological change (see Figure 2). *Navicula leptostriata* appeared between 2006 and 2011 (0% to 5.25%). Other taxa decreased in abundance by more than 2%: *Staurosira construens* (22.4% to 18.6%), *Pseudostaurosira brevistriata* (9.5% to 1.8%), *S. elliptica* (9.0% to 5.3%), *Aulacoseira ambigua* (5.6% to 0%), *Fragilaria capucina* (2.3% to 0%), and *A. italica* (2% to 0%). The loss of the latter three taxa and declines in *Pseudostaurosira brevistriata* suggest that the amount of planktonic habitat in Long Lake decreased between 2006 and 2011, with only minor numbers of planktonic diatoms remaining in this lake.

ISRO: Lake LeSage was added to the GLKN sampling set in 2011, therefore its surface sediment collection from 2012 will serve as the basis for evaluating future change in this lake. Of the ten ISRO lakes sampled in 2012 for surface sediments, four lakes showed significant levels of change in their diatom communities since 2007. The four lakes that showed significant diatom community change between 2007 and 2012 were Beaver, Feldtmann, George, and Richie. Three of the four of these lakes are shallow systems, and changes in diatom assemblages are strongly driven by shifts in benthic species abundances. Lake Richie is over 10 m deep, but much of the lake is also shallow, with the deep area limited to its south-central basin. Vulnerability of wilderness lakes to environmental drivers of change such as climate has been linked to combined interactions of watershed-to-lake surface area ratios and the likelihood of persistent stratification (Rühland et al. 2008, Saros et al. 2012). Both the shallow lakes of ISRO (with high watershed:surface area) and lakes such as Richie that are susceptible to loss of stratification are suspected to be most vulnerable to change.

Beaver Lake had many diatom species, primarily benthic taxa, that shifted dramatically in abundance between 2007 and 2012. *Sellaphora bacilliformis* (0% to 2.4%) and *Pinnularia* cf. *nodosa* (0% to 4.2%) were added to the flora, and *Diploneis marginestriata* (4.3% to 11.7%) and *Staurosira venter* (2.1% to 6.1%) increased in abundance. The planktonic species *Discostella stelligera* (7.4% to 9.6%) and *Aulacoseira subarctica* (1.4% to 4.2%) also increased in abundance between samplings. Taxa

that declined in abundance included benthic species such as *Staurosirella pinnata* (12.5% to 7.8%), *Staurosirella pinnata* v. *intercedens* (7.0% to 0%), *Diploneis modica* (5.3% to 0%), *Pinnularia nodosa* v. *robusta* (3.0% to 0%), *Navicula cryptotenella* (2.8% to 0%), and the planktonic taxon *Cyclotella michiganiana* (2.55% to 0.2%).

Fewer diatoms changed in abundance at Feldtmann Lake between 2007 and 2012 compared to Beaver Lake. Two benthic species appeared in the flora—*Psammothidium sacculum* (0% to 3.6%) and *Gomphonema gracile* (0% to 2.9%)—and the widespread *Achnanthyidium minutissimum* (7.5% to 22.3%) dramatically increased in abundance. Taxa that decreased in abundance by >2% were the planktonic-to-tychoplanktonic *Fragilaria vaucheriae* (5.8% to 1.1%) and *Pseudostaurosira brevistriata* (5.1% to 1.1%), and the benthic taxa *Rossithidium lineare* (3.6% to 0.2%) and *Navicula (Eolimna) minima* (2.5% to 0.2%).

Significant diatom community change in Lake George was driven by seven primarily benthic taxa that had rather large shifts in abundance between 2007 and 2012. *Psammothidium sacculum* (0% to 12.3%) appeared in the flora, whereas *Staurosirella pinnata* v. *intercedens* (2.2% to 0%) was absent in 2012. Other benthic species whose abundance changed more than 2% include *Staurosira venter* (12.0% to 21.7%), which increased in abundance, and *Achnanthyidium minutissimum* (15.1% to 5.7%), *Rossithidium lineare* (5.0% to 0.2%), and *Navicula (Eolimna) minima* (5.4% to 0.9%), which declined in abundance. Only a single plankter, *Cyclotella michiganiana* (0.4% to 2.5%), shifted markedly in abundance between 2007 and 2012.

Many diatom species shifted in abundance between 2007 and 2012 in Lake Richie, and among ISRO lakes, the species shifts stand out because all the taxa that shifted are planktonic. Species that increased in abundance include *Aulacoseira ambigua* (9.7% to 14.9%), *Fragilaria crotonensis* (2.2% to 14.5%), *A. subarctica* (4.3% to 10.9%), *Asterionella formosa* (4.3% to 8.5%), *Fragilaria nanana* (0% to 5.9%), and a taxon tentatively identified as *Fragilaria nanana?* (0% to 4.8%). Planktonic species whose abundance declined are *Tabellaria flocculosa* IIIp (12.8% to 5.4%), *Cyclotella michiganiana* (4.9% to 2.0%), *Discostella stelligera* (4.1% to 1.8%), *Synedra tenera* (22.1% to 0%), and *Aulacoseira italica* (6.5% to 0%).

That significant biological changes in ISRO's shallow lakes are noteworthy. These shifts appear to be strongly linked to shifts primarily in benthic taxa and secondarily in planktonic or tycho-planktonic taxa. The species that shifted in abundance are all normal members of ISRO diatom communities. In Lake Richie, the shifts were exclusively among planktonic species, with greater contributions of araphid plankters (*Fragilaria crotonensis*, *Asterionella formosa*, *Fragilaria nanana*) and *Aulacoseira* spp., and continued downward trends in cyclotelloid species (*Cyclotella michiganiana*, *Discostella stelligera*). In wilderness lakes, these changes are thought to be indicative of changing patterns of stratification and mixing (Rühland et al. 2008, Saros et al. 2012). In Lake Richie the consequences seem to have pushed the lake over a threshold where periodic cyanobacteria blooms have become a regular component of the lake's ecology (Edlund et al. 2010, 2011). Enhanced monitoring efforts, nutrient addition experiments, and proposed research on these boreal systems are logical steps needed to fully understand how and why our wilderness lakes are changing so quickly.

MISS: Both sites sampled in 2011 showed significant levels of change in diatom communities between samplings. MS850/852 was first sampled in 2008; when resampled in 2011 there was one new species (*Aulacoseira ambigua*; 0% to 4.8%), and there were >2% increases in abundance of *Cocconeis placentula* v. *linearis* (2.4% to 17.7%), *A. granulata* (0.9% to 8.5%), *Amphora pediculus* (0.7% to 3.3%), *Nitzschia amphibia* (0.7% to 3.3%), and *Staurosira construens* (0.7% to 3.3%). The diatom *Planothidium hauckianum* v. *rostratum* was lost from the flora, while *Pseudostaurosira brevistriata* (9.0% to 5.4%), *Cyclotella atomus* (9.0% to 3.8%), *C. meneghiniana* (24.5% to 3.1%), and *Staurosirella pinnata* (2.4% to 0.2%) decreased in abundance between 2008 and 2011.

At Spring Lake (MSSPLK) many of the same taxa shifted in abundance between the 2006 and 2011 samplings. *Aulacoseira granulata* (2.9% to 20.9%), *A. ambigua* (2.7% to 11.1%), and *Cocconeis placentula* v. *lineata* (0.4% to 6.1%) increased in abundance, whereas *Cyclotella meneghiniana* (30% to 18%), *Cyclostephanos invisitatus* (5.4% to 2.6%), *Stephanodiscus minutulus* (5.8% to 2.0%), *Cyclostephanos tholiformis* (5.6% to 0.2%), *S. parvus* (5.6% to 0.2%), and *S. hantzschii* v. *tenuis* (4.1% to 0%) decreased in abundance. Notable among the biological changes in these two sites is the shift in planktonic dominants from the more eutrophic *Cyclotella*, *Cyclostephanos*, and *Stephanodiscus* species to more meso- to eutrophic *Aulacoseira* species. There were also increases in some benthic taxa (e.g., *Cocconeis placentula* v. *lineata*, *Amphora pediculus*) that would traditionally indicate some improvement in water quality along these river reaches. Alternatively, the true depositional nature of these two sites has been questioned and we may be sampling transient deposits that reflect only recent deposition between scouring events.

SACN: All four sampling stations showed significant biological change between 2007 and the most recent surface sediment collections made in 2011. At the Trego Flowage, diatoms that increased in abundance between 2007 and 2011 included *Pseudostaurosira brevistriata* (5.3% to 10.7%), *Cocconeis placentula* v. *lineata* (4.7% to 10.2%), *Navicula submuralis* (1.3% to 4.7%), and *Karayevia clevei* (1.7% to 4.5%). Three taxa found in 2007 were not present in the 2011 sample: *Fragilaria crotonensis* (6.4% to 0%), *Fragilaria capucina* (3.0% to 0%), and *Geissleria cummerowii* (2.3% to 0%). Two other species declined in abundance between samplings: *Fragilaria capucina* v. *mesolepta* (5.3% to 1.3%) and *Amphora subcostulata* (3.4% to 0.4%) dropped in abundance by >2%. Changes at Trego point toward greater numbers of benthic or attached forms and declines in more eutrophic plankters such as *Fragilaria capucina* v. *mesolepta*.

Only one species, *Aulacoseira ambigua* (0.2% to 12.7%), increased in abundance by >2% at St. Croix Falls between 2007 and 2011. Others decreased in relative abundance including *Pseudostaurosira brevistriata* (8.8% to 5.8%), *Staurosira venter* (9.3% to 5.0%), *Planothidium rostratum* (8.0% to 3.2%), *Achnantheidium minutissimum* (4.3% to 1.9%), *Karayevia clevei* (4.1% to 1.3%), and *Amphora subcostulata* (3.2% to 0.7%). The notable increase in abundance of the mesotrophic indicator *Aulacoseira ambigua* is shared among the lower St. Croix sampling stations.

Similar to St. Croix Falls, sediment diatom assemblages collected at Hudson had few taxa that increased in abundance between 2007 and 2011. *Aulacoseira ambigua* (5.7% to 20.8%) and *Pseudostaurosira brevistriata* (1.1% to 5.9%) both increased more than 2% in relative abundance. In contrast, several dominant taxa in 2007 decreased significantly in abundance including *A. granulata*

(26.6% to 5.9%), *Staurosira venter* (6.5% to 2.7%), *Discostella stelligera* (7.1% to 0.8%), *Aulacoseira granulata* v. *angustissima* (3.7% to 0.8%), *Fragilaria crotonensis* v. *oregonica* (3.3% to 0.6%), and *Staurosirella pinnata* v. *lancettula* (3.0% to 0%). Decreases in planktonic taxa that are more eutrophic indicators (e.g., *Aulacoseira granulata*, *A. granulata* v. *angustissima*, and *Fragilaria crotonensis* v. *oregonica*) are an encouraging shift in diatom communities of the lower St. Croix.

Changes in sediment diatom assemblages between 2007 and 2011 at the Prescott sampling station were primarily planktonic species. *Aulacoseira ambigua* (10.1% to 24.6%), *A. granulata* (16.2% to 20.7%), *Pseudostaurosira brevistriata* (0% to 3.5%), and the benthic taxon *Staurosira construens* v. *venter* (1.7% to 8.6%) all increased in abundance by more than 2%. The plankters *Fragilaria crotonensis* (17.4% to 7.0%), *Stephanodiscus hantzschii* (4.5% to 1.4%), *F. crotonensis* v. *oregonica* (3.5% to 0.8%), and *A. granulata* v. *angustissima* (3.1% to 0.4%) decreased in abundance in the lower St. Croix River between 2007 and 2011. Many of the planktonic taxa that have declined in abundance near Prescott are indicators of eutrophic conditions. Hopefully their changes are connected to efforts to reduce nutrient loading to the St. Croix River.

Directionality of change using canonical correspondence analysis of the GLKN diatom calibration set

We can further investigate links between changes in diatom communities and potential environmental drivers by passively plotting the 2011/2012 surface sediment assemblages onto the CCA of species-environment relationships among all GLKN lakes (Figures 3–6) (Edlund et al. 2011, 2012). Note that some of the 2011/2012 sediment samples could not be tested with this analysis; all lakes at APIS and MISS were removed earlier as species-environment outliers during construction of the GLKN diatom calibration set (Edlund et al 2011). Middle Lagoon (INDU) and Lake LeSage (ISRO) have been added to the GLKN monitoring program and this report represents the first analysis of diatom assemblages at those locations.

The 2011 surface sediment samples from INDU and SACN generally show similar direction of change when passively plotted on the diatom species-environment CCA for GLKN lakes (Figure 3). Long Lake, Trego, St. Croix Falls, and Hudson all show negative movement along CCA Axis 2, whereas Prescott (SC11) shows positive movement along that same axis. The nutrient vectors for TP and NO₃-NO₂ are strongly loaded on Axis 2, which suggests that changes in diatom communities at sites with negative movement along Axis 2 may reflect changes in nutrient availability. The diatom community changes at Long Lake (INDU) suggested loss of planktonic habitat and likely increased habitat availability for attached diatoms. However, there is high variability among TP measures from Long Lake, making it difficult to detect significant declines in nutrient availability. At SACN, the Trego, St. Croix Falls, and Hudson stations similarly show trends suggesting nutrient decreases. Benthic diatoms increased at the Trego Flowage, which is a likely response to improved water clarity. At St. Croix Falls and Hudson, increased abundance of more mesotrophic species such as *Aulacoseira ambigua* and decreased abundance of more eutrophic plankters (*A. granulata*, *Pseudostaurosira brevistriata*) may be an early response to nutrient reduction efforts. In contrast, SACN's 2011 Prescott sample suggests slightly increased nutrients; increased abundance of *A. granulata* and *P. brevistriata* may provide evidence for this downriver trend. As expected in a large

river system, monitoring data from GLKN show high variability between 2007 and 2011 and are equivocal on recent trends in nutrient availability along the St. Croix.

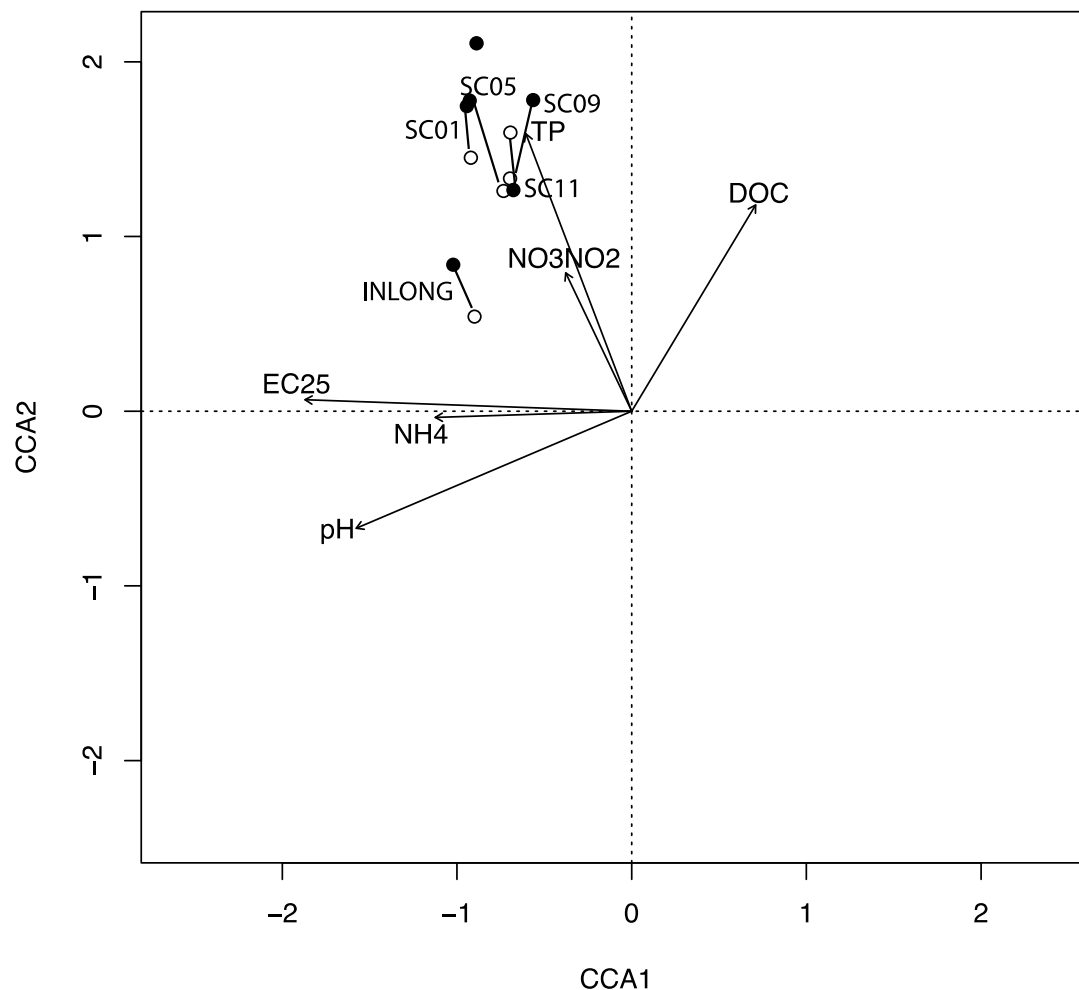


Figure 3. The 2011 repeat surface sediment samples from SACN and INDU passively plotted on the canonical correspondence analysis (CCA) of the GLKN diatom calibration set. Vectors show environmental variables that explain significant independent variation in diatom abundance. *Filled circles* represent surface sediment samples taken in 2007 from SACN stations. *Open circles* represent repeat surface sediment samples taken in 2011 from SACN and INDU stations (SC01-Trego, SC05-St. Croix Falls, SC09-Hudson, SC11-Prescott, INLONG-Long Lake). Chords connect the 2007 and 2011 samples from labeled stations.

Diatom communities for the four ISRO lakes that showed significant community change (Beaver, Feldtmann, George, and Richie) (see Figure 2) show similar directional change along the CCA environmental gradients, with positive trajectories along Axis 1 and slightly negative trajectories along Axis 2 (Figure 4). Although the directional changes are minor given the full environmental gradients present in the GLKN lakes, their trajectories suggest that a slight shift downward in nutrients, pH, and EC25, and perhaps a slight increase in dissolved organic carbon (DOC) may be

driving changes in these diatom communities. Watershed monitoring at ISRO's Wallace Lake may support these inferences as there has been greater export of DOC from this shallow lake watershed in recent years (Toczydlowski and Stottlemeyer 2009).

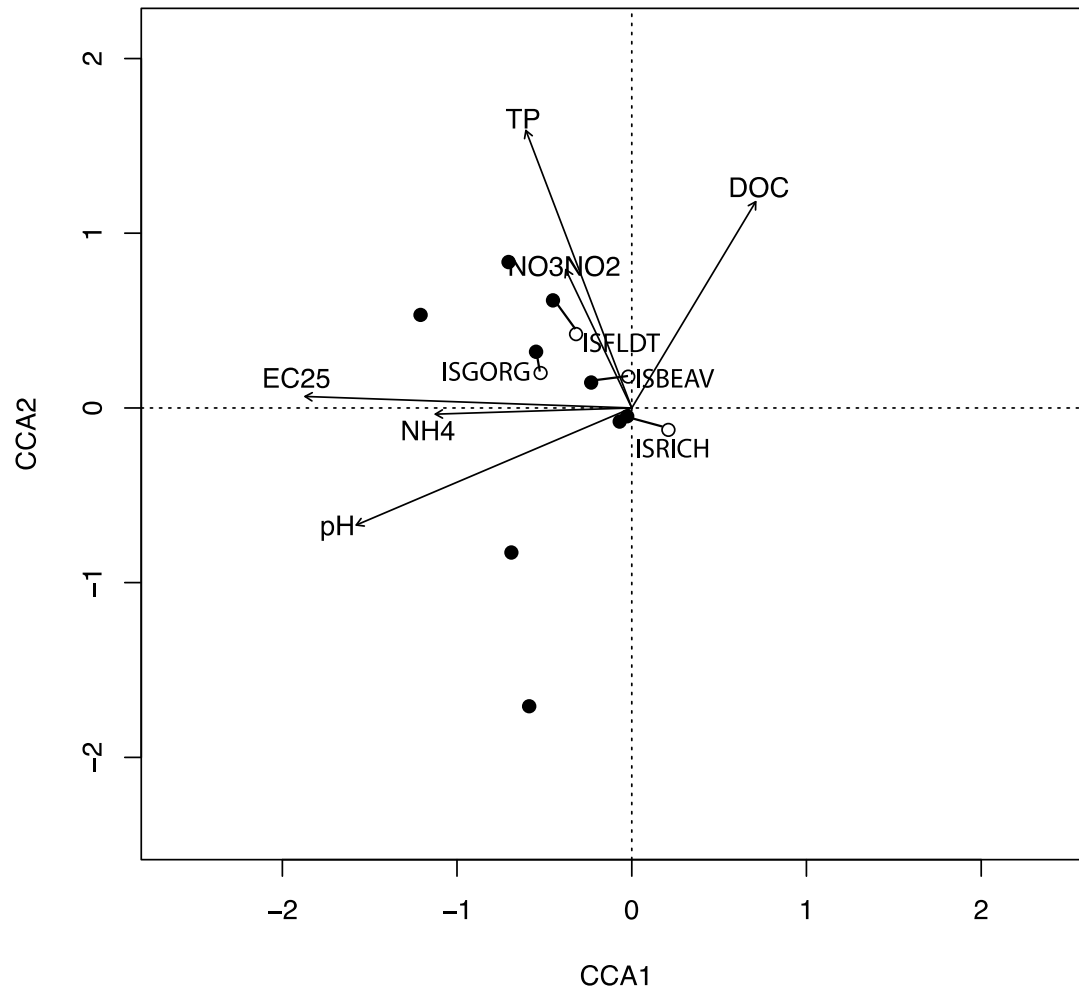


Figure 4. The 2012 repeat surface sediment samples from ISRO passively plotted on the canonical correspondence analysis (CCA) of the GLKN diatom calibration set. Vectors show environmental variables that explain significant independent variation in diatom abundance. *Filled circles* represent surface sediment samples taken in 2007 from all ISRO lakes (2008 for Siskiwit). *Open circles* represent repeat surface sediment samples taken in 2012 from ISRO lakes (Richie, Beaver, Feldtman, George). Chords connect the 2007 and 2012 samples from labeled lakes.

In contrast, the other ISRO lakes without significant community change (Ahmik, Desor, Harvey, Sargent, and Siskiwit) (Figure 5) show negative directional change along both CCA axes 1 and 2. Changes in ISRO's large lakes (including Sargent, Siskiwit, and Desor) move negatively along the DOC axis, perhaps indicating that DOC levels may be falling in these larger lake systems, and positively along the pH axis, perhaps as a result of increasing productivity. Other physical drivers

(stratification, mixing, ice-free season) may also be at play in how the diatom communities are changing (Edlund et al. 2011, Saros et al. 2012).

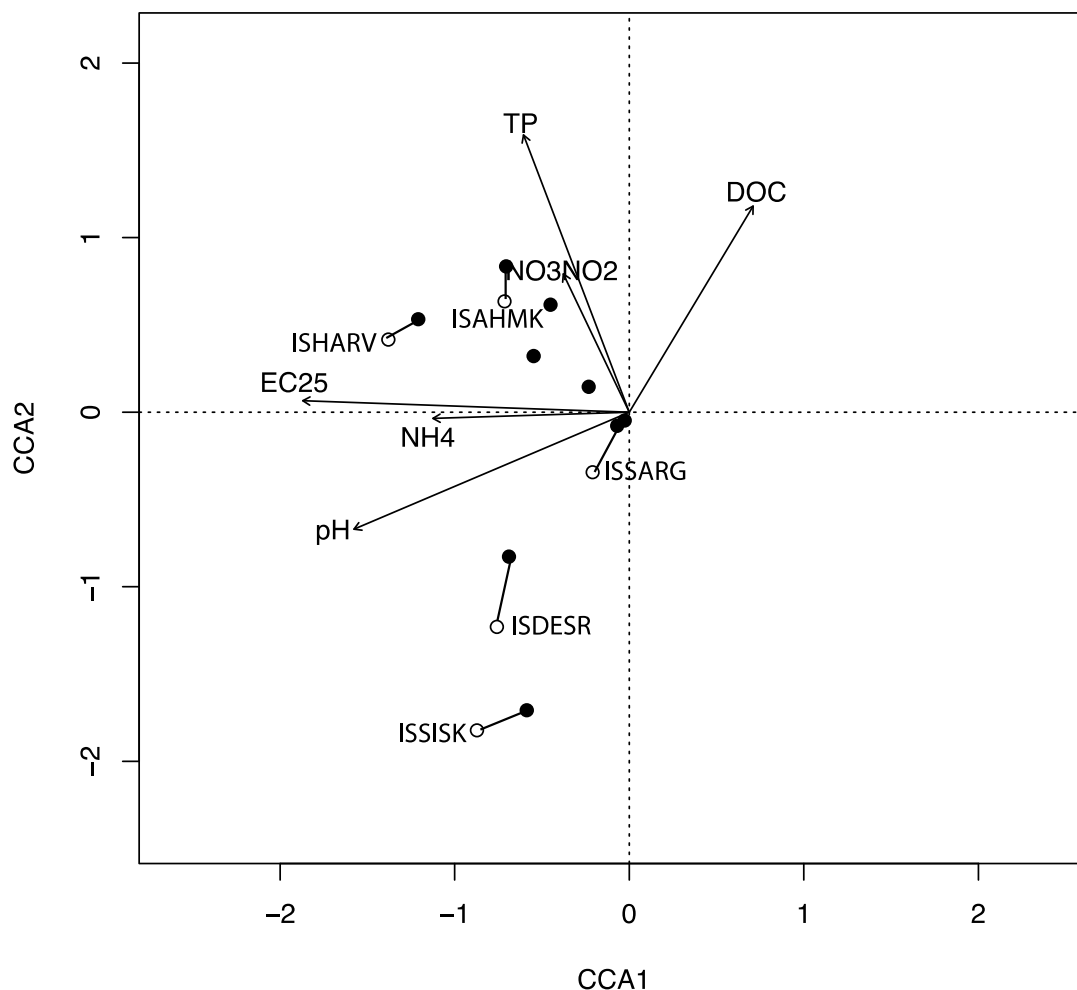


Figure 5. The 2012 repeat surface sediment samples from ISRO passively plotted on the canonical correspondence analysis (CCA) of the GLKN diatom calibration set. Vectors show environmental variables that explain significant independent variation in diatom abundance. *Filled circles* represent surface sediment samples taken in 2007 from all ISRO lakes (2008 for Siskiwit). *Open circles* represent repeat surface sediment samples taken in 2012 from ISRO lakes (Ahmik, Sargent, Desor, Harvey, Siskiwit). Chords connect the 2007 and 2012 samples from labeled lakes.

Environmental change in relation to cored APIS lagoons and ISRO lakes

Between 2005 and 2008, long sediment cores were recovered from a subset of lakes across five GLKN parks to provide greater temporal context to changes in the lakes (Edlund et al. 2011). For the parks where surface sediment cores were collected in 2011/2012 (this report), long cores had been collected at ISRO (lakes Ahmik, Harvey, and Richie) and APIS (Outer Island lagoon) in 2007 and analyzed for records of biogeochemical change covering the last 150 years. These records are

available to provide long-term context to changes noted in recent surface sediment diatom assemblages. Squared chord distances were earlier estimated between pre-Euro-American sediment assemblages and those deposited at the time of park formation, and between the time of park formation and the first round of surface sediment collection in 2007 (see Figure 1) (Edlund et al. 2011). These data showed that among these lakes, only ISRO's Lake Richie showed significant diatom community change across these two time periods. Detrended Correspondence Analyses and constrained cluster analyses were also run on all the downcore diatom assemblages to determine the timing of diatom community changes. In these lakes, most showed two periods of change; one corresponded to major land use changes (primarily logging), and the other was a characteristic recent period of change in the 1980s and 1990s that may be related to climate drivers (Edlund et al. 2011).

The most recent repeat surface sediment assemblages were also plotted with the long sediment core assemblages using DCA to see how recent changes in diatom communities are related to historical changes. At Outer Island lagoon (APIS), the 2011 repeat sample plots quite distant from the past assemblages and along the first axis of variation, suggesting that this recent sample represents a new community trajectory for the lagoon (Figure 6). The large number of species that showed significant changes (see above) point to a diatom community and lake that appears to be undergoing rapid change that was initiated between 1995 and 2005 and continues in the most recent samples.

At ISRO, the 2012 sample from Lake Ahmik plots at the end of a trajectory of samples that began between 1994 and 2001 (Figure 7). We note that neither the 2012 sample nor other recent core samples from Lake Ahmik (2001, 2007) represent a short excursion of diatom assemblages; rather, Lake Ahmik assemblages appear to be moving directionally along DCA Axis 2 (Figure 7) and orthogonal to the longer term change recorded along DCA Axis 1 in sediments deposited between 1825 and 1994. The 2012 sample from Lake Harvey, similar to Lake Ahmik, plots along DCA Axis 2 (Figure 8), further suggesting that recent diatom assemblages in these wilderness lakes represent new community trajectories that have not been seen in nearly two centuries. The Lake Richie record provides the most dramatic record of recent change (Figure 9). There is increased variability in the diatom assemblages since 1992, culminating with the 2012 sample plotting distant from any earlier assemblages. All three ISRO core records support the point that recent changes in these lakes are not transitory but represent broad-scale community shifts within these protected wilderness lakes.

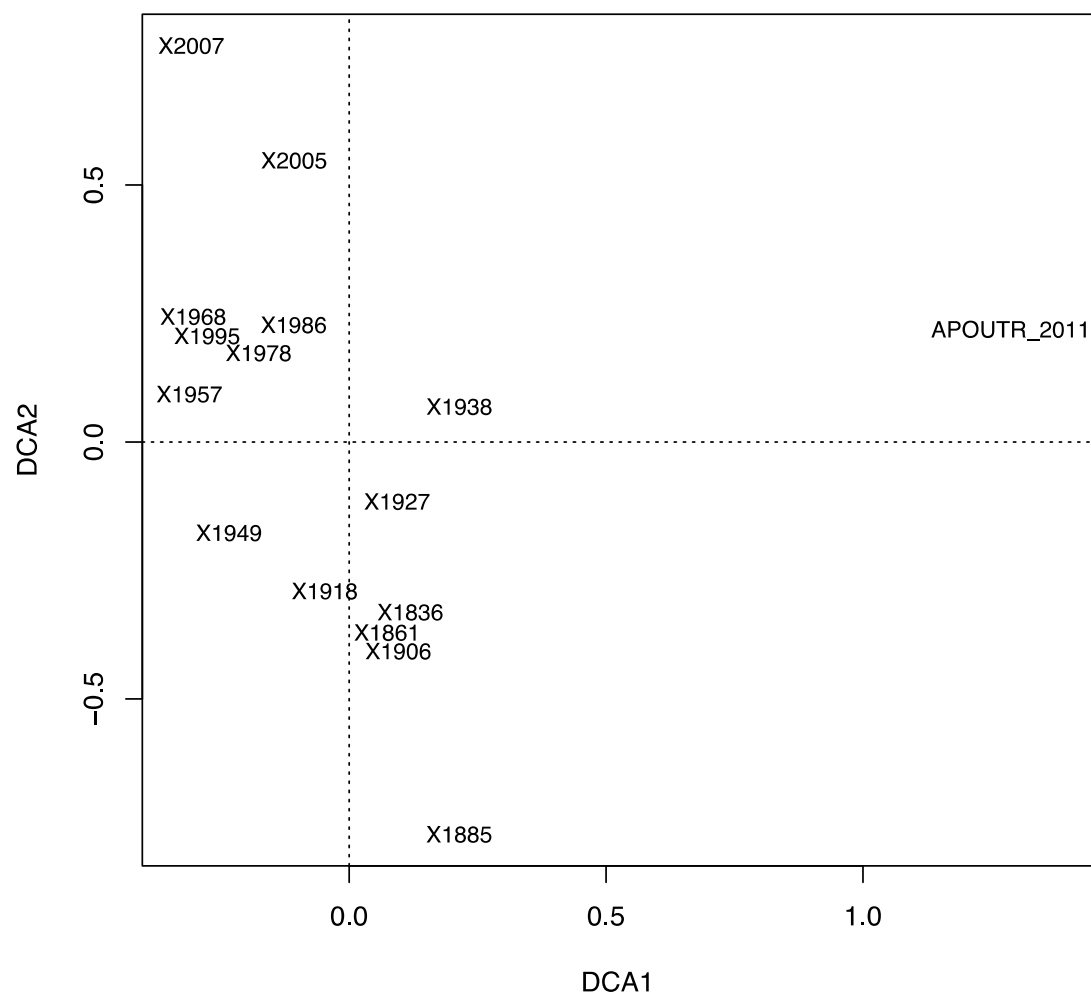


Figure 6. Detrended correspondence analysis (DCA) of downcore diatom assemblages from Outer Island lagoon core (APIS, 1836–2007) including the 2011 repeat surface sample (APOUTR_2011); axes 1 and 2 loadings plotted for core depths-dates.

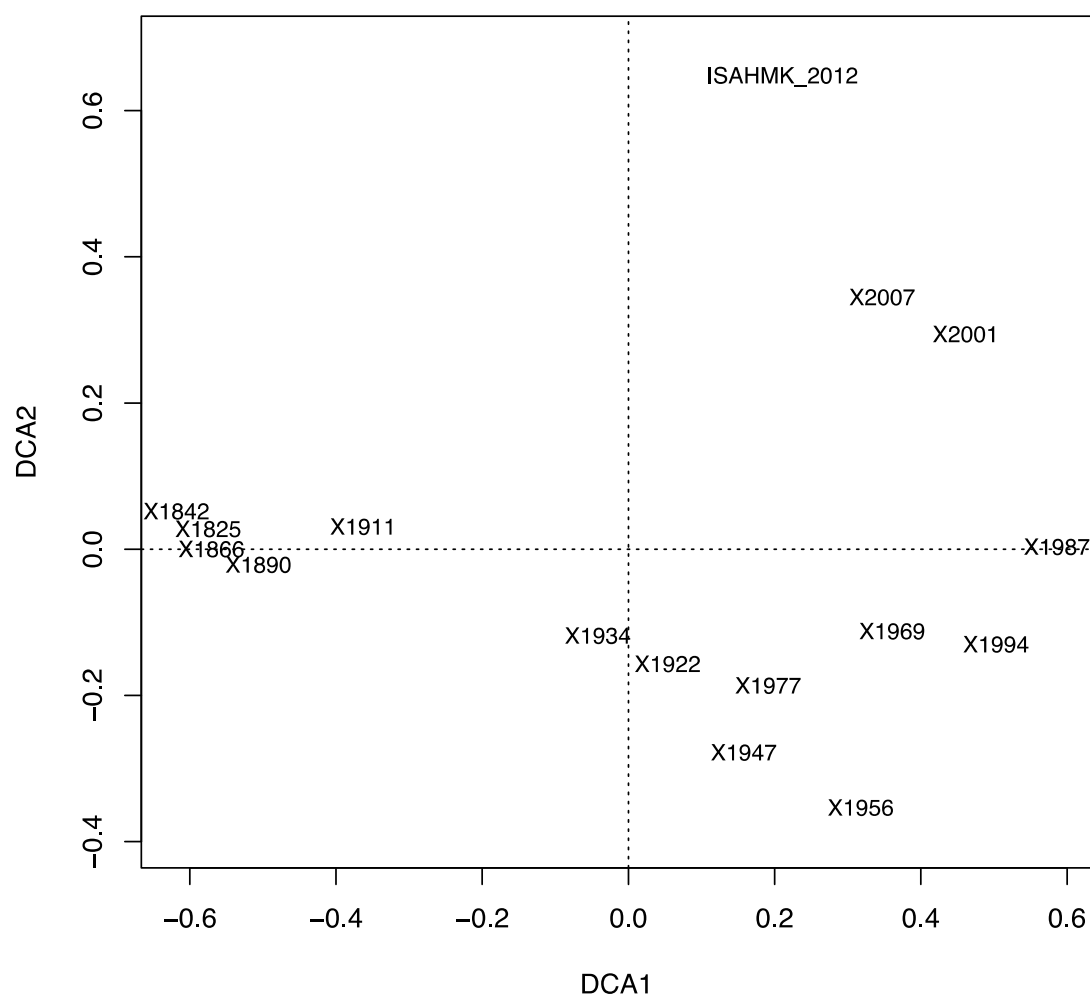


Figure 7. Detrended correspondence analysis (DCA) of downcore diatom assemblages from Lake Ahmik core (ISRO, 1825–2007) including the 2012 repeat surface sample (ISAHMK_2012); axes 1 and 2 loadings plotted for core depths-dates.

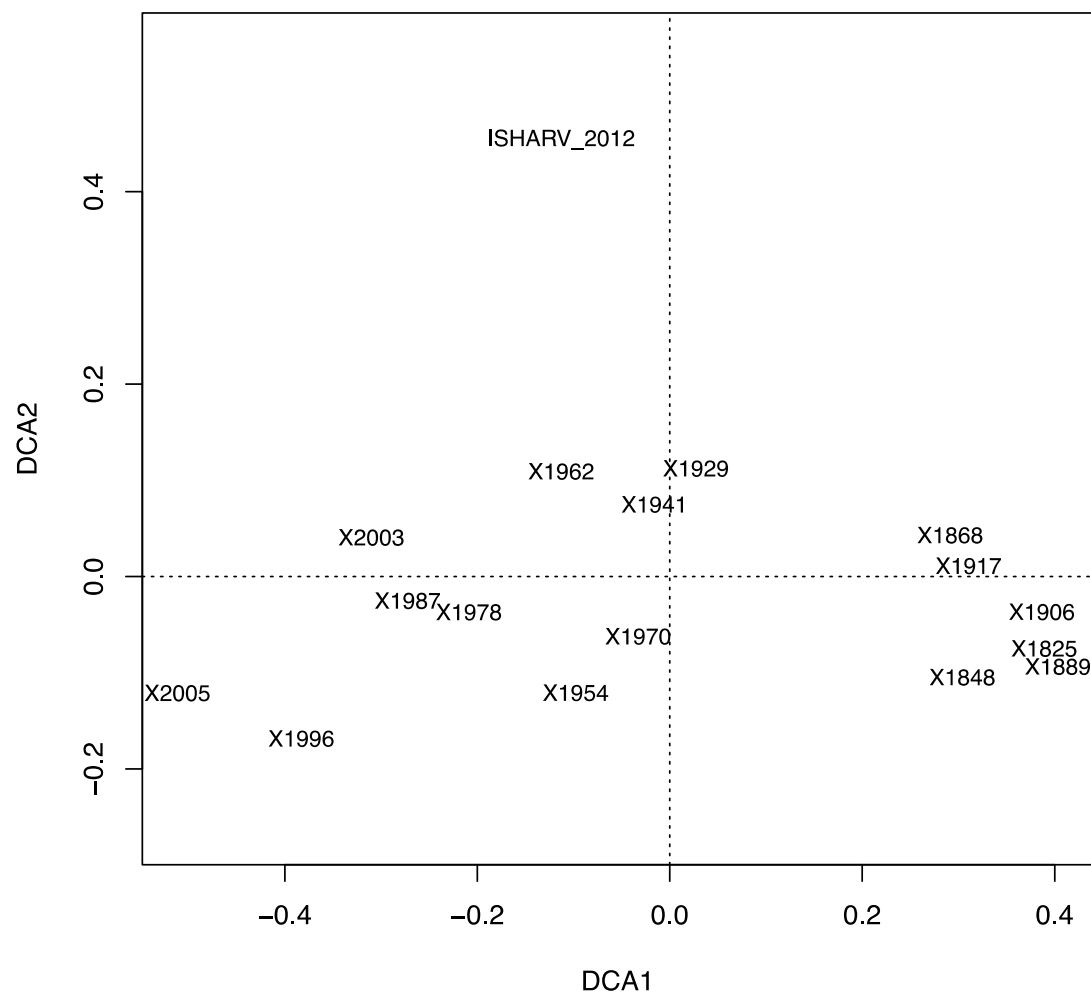


Figure 8. Detrended correspondence analysis (DCA) of downcore diatom assemblages from Lake Harvey core (ISRO, 1825–2007) including the 2012 repeat surface sample (ISHARV_2012); axes 1 and 2 loadings plotted for core depths-dates.

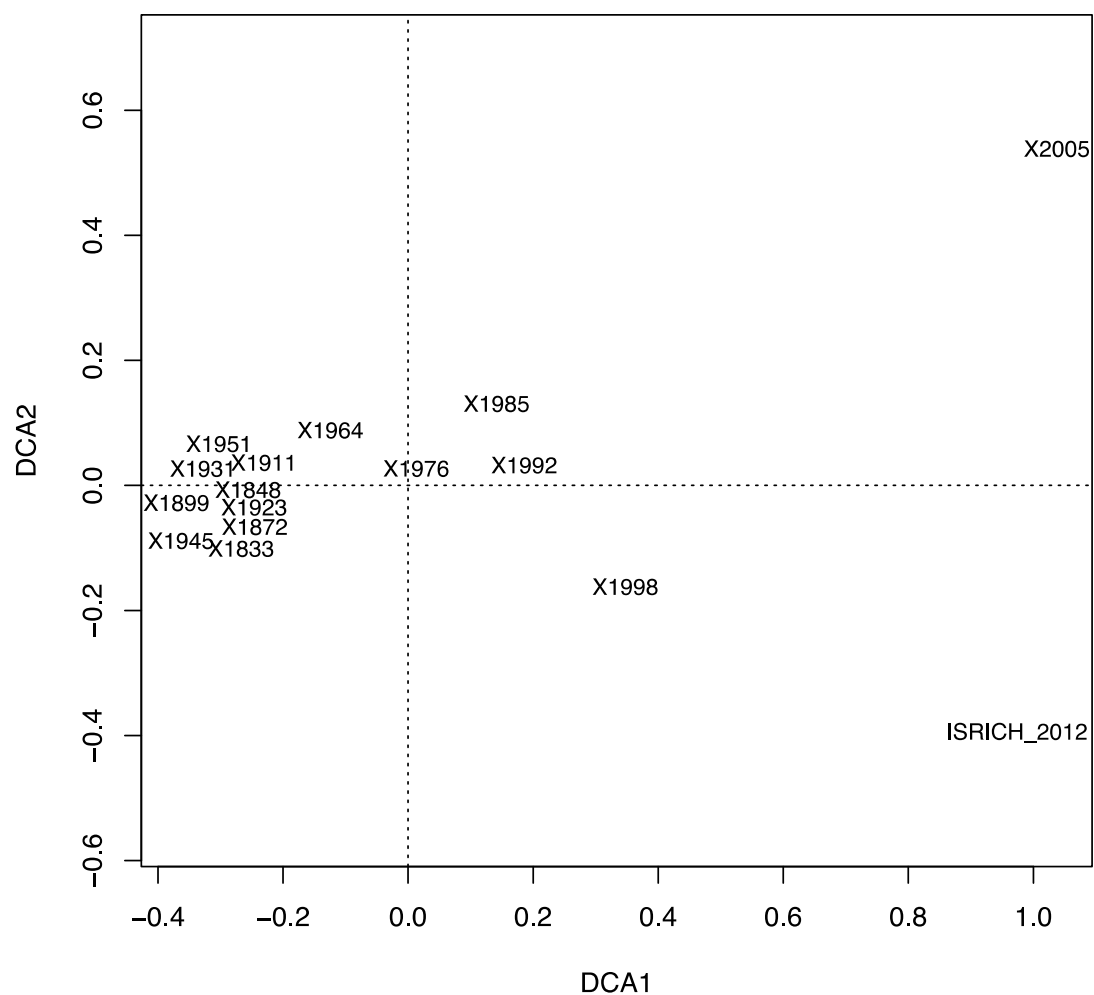


Figure 9. Detrended correspondence analysis (DCA) of downcore diatom assemblages from Lake Richie core (ISRO, 1833–2005) including the 2012 repeat surface sample (ISRICH_2012); axes 1 and 2 loadings plotted for core depths-dates.

Conclusions

Using diatoms to conduct biomonitoring complements the GLKN lakes and large rivers monitoring program and is designed to 1) provide a cost-effective means of monitoring a biological component common among all lakes, 2) be strongly integrated with current physical and chemical monitoring efforts developed for the lakes, and 3) detect biological changes that are not sampled during regular water quality monitoring. Biomonitoring involves sampling and analysis of diatoms in surface sediments that are collected from GLKN lakes every three-to-five years during regular water quality monitoring visits. Pairing the two monitoring techniques, we are able to detect biological change, estimate trends in several water quality parameters (pH, TP, EC25) using diatom-inference models, interpret change, generate hypotheses on why change is occurring, and direct future research efforts and resources to better understand the drivers and mechanisms of ecological change that are impacting NPS lakes.

In this report we analyzed diatom assemblages from 21 surface sediments collected in 2011/2012 from lakes, lagoons, and flowages at APIS, INDU, ISRO, MISS, and SACN. We first applied diatom-inference models developed for the GLKN lakes to estimate or reconstruct measures of TP, pH, and EC25 for each lake based on recent diatom assemblages. Reconstructed water quality parameters generally agreed with monitored water quality data. Problem lakes were those that represented outliers (APIS and MISS sites) during the construction of the GLKN diatom calibration set, or those near the ends of environmental gradients of GLKN lakes.

The most recent surface sediment diatoms were then compared using the squared chord distance with samples collected from the same lakes between 2006 and 2008 to quantify biological change within each lake. Most of the lakes showed significant biological change, which we interpreted based on shifts in species abundances between sampling events. Among the park units, lagoons at APIS, the large river sampling sites at SACN and MISS, INDU's Long Lake, and the shallow lakes of ISRO had significant biological change between sampling events.

Biological changes were also used to identify potential environmental drivers by passively plotting the new surface sediment collections on ordinations of diatom species-environment relationships. Although diatom community changes were fairly small in the context of species and environmental gradients across the GLKN lakes, trajectories for several of the sites at INDU and SACN suggested declines in nutrient availability, and diatom communities in the larger lakes of ISRO seem to be responding along the DOC and pH gradients.

Finally, we ran similar passive plots of the 2011/2012 repeat samples on DCA ordinations of diatom assemblages preserved in long sediment cores that were collected from one APIS lagoon (Outer) and three ISRO lakes (Ahmik, Harvey, Richie) in 2007. This analysis compares recent diatom community changes with long-term changes in these lakes. In all four lakes the most recent surface sediment collection continued a trend toward greater biological variability among sediment samples deposited after the 1980s. Furthermore, in no case did recent diatom communities represent a return to previous diatom communities that had been present in the lakes; recent biological changes in the

lakes represent new diatom species associations and indicate that recent changes are not transitory but have been a characteristic of these lakes since the 1980s and 1990s.

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