ANALYSIS, EVALUATION AND REPORTING OF ENVIRONMENTAL MONITORING DATA FOR THE LOWER ATHABASCA RIVER

FINAL REPORT
JANUARY 22, 2024

EXECUTIVE SUMMARY

There is growing concern that there may be an environmental requirement to return treated oil sands process waters (OSPWs) to the Athabasca River. Treated process waters would be complex, with potential residual uncertainties regarding their environmental effects in the river. Environmental Effects Monitoring (EEM) is a process for verifying predictions related to the release of treated, complex effluents. EEM is used to assess effluents from Canada's pulp mills and metal and diamond mines. There is the potential that Site-specifically designed EEM programs may be used to support and assess the release of treated OSPW to the Athabasca River in the future.

To support a potential EEM, Alberta Environment and Protected Areas (AEPA) completed three years of baseline monitoring of the mainstem of the Lower Athabasca River (LAR), from downstream of Fort McMurray to the confluence of the Ells River, under what was called the Enhanced Monitoring Program of the LAR (EMP). The EMP monitored conventional EEM parameters including sentinel adult fish populations (Trout-perch), benthic invertebrate communities, benthic algae communities, water and sediment quality variables, and body and tissue burdens of contaminants of potential concern in fish and benthic invertebrates in the vicinity of Syncrude operations on the Athabasca River. The EMP was initially designed to test for effects associated with a pilot release of treated effluent. The pilot release ultimately was not permitted. As such, the EMP provides three years of baseline data on which to assess spatial and temporal variation in the mainstem of the River.

The EMP is not the only monitoring program in the mainstem Athabasca River. Canada's Oil Sands Monitoring (OSM) program has also been collecting EEM-type data in the mainstem, but from a broader spatial area, and for a longer time. There are obvious synergies between the EMP and OSMP that have not been examined.

This report, therefore, has four main objectives:

(1) summarize sources of variation in measured aquatic environment monitoring variables collected under the OSMP and EMP;
(2) determine if (and how) data collected under the OSMP can be used to support future EEM programs designed to assess point-source release of OSPW;
(3) quantify the ability (power) to statistically detect changes in the various EMP monitoring response endpoints;
(4) consider the information provided by the analyses addressing objectives 1 through 3 above, and provide recommendations on how EEM programs could be designed to assess the point-source release of treated OSPW.

Under the EMP, river flow volume (discharge, quantified as the daily average flow in the 60 days prior to biological sampling, or Q60) explained significant variations in most water quality variables, sediment quality variables, benthic algae and invertebrate community indices, and sentinel fish population variables. After controlling for the effects of discharge, there were generally modest variations in the various endpoints among years and Sites (upstream and downstream of the proposed pilot release of OSPW).

General linear models (GLMs) were used to generate predictive 'models' for the various EMP endpoints. OSMP data were sufficient for building predictive models (and estimating normal ranges) for water and sediment quality variables, and fish population responses (for Trout-perch). The OSMP benthic community samples were collected from cobble substrates, which were coarser than the sand/silt that EMP benthic community samples were collected from. As such, the OSMP data were not suitable for developing appropriate predictive models for the EMP benthic community data. Normal ranges for benthic algae and benthic invertebrates were developed using EMP data.

GLMs were used to quantify within-Site, among-replicate variability, which was used in statistical power analyses to compute sample sizes for all the various EEM components. Within-Site noise was also used to support calculation of Site-specific (local) normal ranges.

The models developed, using OSMP and EMP, can be used to support the calculation of normal ranges for EEM endpoints for assessing variations in relation to future OSPW release. Normal ranges for several conventional EEM variables are illustrated graphically. A general methodology for estimating Site-specific and regional (that incorporate among-Site and among-year variability) normal ranges is provided.

The design of future EEM programs focused on specific OSPW releases can make use of the existing baseline data developed under the OSMP and EMP. A generic EEM design with one exposure Site, one reference Site and at least one baseline year of data can use data from the OSMP and EMP to support viable estimates of normal baseline ranges for classic EEM biological responses.

EEM for OSPW should be designed to test anticipated/predicted conditions. It can be expected that OSPW will not be permitted for release unless it is predicted that biological effects will be negligible relative to the background variability. As such, any observed variations that exceed the normal range of variation should be used to trigger confirmation of effects, and / or investigation of cause of those effects. It can also be anticipated that engineering, mass-balance models will make it possible to predict concentrations of constituents of concern in water and sediment. As such, EEM should be designed to test those predictions. Any observed concentrations in the Athabasca River that are inconsistent with the engineering models would justify confirmation, and / or investigation of cause of the deviation from prediction.

Single reference and exposure Sites, upstream and downstream of release points would initially suffice in an EEM program assessing effects of an OSPW. Each Site would contain replicate samples of benthic algae and benthos, sentinel fish, water and sediment, and fish and benthos tissues (burdens). Those data would be used to test the generic predictions. OSMP and EMP data would be useful for establishing confidence in estimated normal ranges for biological, physical and chemical responses in the LAR.

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List of Acronyms and Abbreviations

A	Alberta Environment and Protected Areas
ANOVA	Analysis of Variance
BC	Bray Curtis
BGA	Blue Green Algae
BIC	Benthic Invertebrate Community
CCME	Canadian Council of Ministers of the Environment
CES	Critical Effect Size
Chl-*a*	Chlorophyll-*a*
CI	Confidence Interval
CSM	Conceptual Site Models
C-WQI	Cytotoxicity Water Quality Index
df	Degrees of Freedom
DL	Detection Limit
DS	Downstream
ECCC	Environment and Climate Change Canada
EEM	Environmental Effects Monitoring
EMP	Enhanced Monitoring Program
EPT	Ephemeroptera, Plecoptera, and Trichoptera
EROD	Ethoxyresorufin-*O*-deethylase
F	Female
FL	Fork length
GLM	General Linear Model
GSI	Gonadosomatic index
IOC	Investigation of Cause
ISQG	Intermediate Sediment Quality Guidelines
JOSMP	Joint Oil Sands Monitoring Program
K	Condition Factor
LAR	Lower Athabasca River
LL	Lower Level
LPL	Lowest Practical Taxonomic Level
LSI	Liver Somatic Index
M	Male
MDL	Method Detection Limits
MDMER	Metal and Diamond Mining Effluent Regulations
MSE	Mean Square Error
ND	Not Detected
NDR	Peak detected but did not meet quantification criteria
NMDS	Non-metric Multidimensional Scaling
NQ	Not Quantifiable
NR	Normal Range
OSM	Oil Sands Monitoring
OSPW	Oil Sands Process-affected Waters
OSR	Oil Sands Region
PAC	Polycyclic Aromatic Compound

PAH	Polycyclic Aromatic Hydrocarbon
PC	Principal Component
PCA	Principal Components Analysis
PPER	Pulp and Paper Effluent Regulations
PTI	Pollution Tolerance Index
Q	Discharge/Flow Volume (on day of sampling)
Q60	Discharge (60 day mean prior to sampling)
QA/QC	Quality Assurance/Quality Control
RAMP	Regional Aquatic Monitoring Program
RDL	Reported Detection Limit
Ref	Reference
SD	Standard Deviation
$SD_{regional}$	Regional Standard Deviation
S^2_A	Variation among sites
S^2_W	Variation among samples within sites
SE	Standard Error
SPMD	Semipermeable Membrane Device
SS	Sums of Squares
ST	Science Team
TOC	Total Organic Carbon
TRPR	Trout-perch
TU	Toxic Unit
UL	Upper Level
US	Upstream
US EPA	United States Environmental Protection Agency
VC	Valued Components
WHSC	White Sucker
WQG	Water Quality Guideline
WQI	Water Quality Index
%VE	Percent of Variance Explained
ΣPAC4	Sum of Benzo[a]anthracene, Chrysene, Benzo[b]fluoranthene, and Benzo[a]pyrene

1.0 INTRODUCTION

1.1 Background

Operations in the Oil Sands Region (OSR) of northeastern Alberta began in the late 1960’s, generating large volumes of Oil Sands Process-affected Waters (OSPW). Primary stressors in the OSR are land disturbances, hydrological alterations, atmospheric emissions, treated sewage effluents, and depositions of particulates and gases from surface mining activities, along with the release of some non-industrial and, although rare, industrial wastewater (Hazewinkel & Westcott, 2015). In contrast to other industries such as pulp and paper mills and metal mines, release of treated effluents is largely absent from the OSR. OSPW have been, and continue to be, stored in “out-of-pit” and “in-pit” tailing facilities, although the discharge of OSPW may occur in the future (Hicks & Scrimgeour, 2019b). The treatment and release of OSPW into the environment would allow for a reduction of long-term containment requirements, minimize landscape disturbances, expedite terrestrial and aquatic reclamation activities, mitigate OSPW salinization, and achieve minor closure outcomes. OSPW would need to be treated prior to being discharged into the environment, and studies would need to be conducted to identify potential effects caused by these discharges.

The Canadian Environmental Effect Monitoring (EEM) programs are a science-based performance measurement tool used to evaluate the adequacy of Pulp and Paper and Metal and Diamond Mining Effluent Regulations (Munkittrick et al., 2002). In the EEM programs, biological responses in the exposure area (i.e., exposed to effluent) are assessed via comparison to biological responses in a reference or baseline condition. Effects are defined as statistically significant differences in exposure area endpoints relative to what is observed in reference areas (Environment Canada, 2012a). The magnitude of the effect (e.g., percent difference) is evaluated relative to critical effect sizes (CES), thresholds above which an effect may be indicative of a higher risk to the environment and effects are deemed to be sufficient to justify follow up studies (Environment Canada, 2012a). Currently, monitoring for effects of oil sands operations is unlike the other industries where there are distinct points of release of significant volumes of treated wastewater that are known to clearly change the chemistry of the receiver. In the Lower Athabasca River, changes in water quality have been subtle (Arciszewski et al., 2018), and changes in biota even more so (Arciszewski et al., 2017; Arciszewski, 2021), in part because of the difficulty in identifying clearly exposed areas or conditions and due to the erosion of bitumen deposits by natural processes. Similar to identifying impacts of OSPW, natural changes in groundwater quality also occur throughout the region (Gibson et al., 2013; Birks et al., 2018) challenging the clear attribution of any changes to oil sands operators.

A substantial challenge when assessing environmental changes in the OSR are the key information gaps related to the understanding of baseline environmental conditions and quantifying exposed conditions. The compounds found in OSPW are also naturally occurring and source attribution has historically been a challenge (Hewitt et al., 2020; Roy et al., 2016; Sun et al., 2017). Another substantial challenge when assessing changes in the OSR is discerning the role of anthropogenic stressors from confounding natural processes. It is not uncommon that differences between baseline and exposed conditions can be large and that factors other than exposure to a stressor of interest may partially or completely drive these differences. For example, it has been demonstrated that flow volumes (i.e., river discharge) and mean summer air temperature, a surrogate for water temperature, affect year-to-year fish population performances in the OSR (Hatfield Consultants et al., 2016; Kilgour et al., 2019). As such, to improve our interpretation of fish population performance in the OSR there is a need to first understand the magnitude of baseline variation that occurs and the causes of that baseline variation. Baseline variations can put into context changes in water quality and biotic indicators, providing potential benchmarks for understanding the potential consequences of large changes.

Environmental monitoring studies in the OSR began in the 1970s with the Alberta Oil Sands Environmental Research Program which was followed later by the Northern River Basins Study and the Northern Rivers Ecosystem Initiative, although these programs had a broad focus. In 1997, the Regional Aquatic Monitoring Program (RAMP) was developed and rebranded in 2012 as the Canada-Alberta Joint Oil Sands Monitoring Program (JOSMP, Environment Canada, 2012), and again in 2016 as the Oil Sands Monitoring (OSM) program. As part of RAMP and then JOSM/OSM, environmental data were collected each year from the Athabasca River and its tributaries, with a focus on water and sediment quality, benthic invertebrate communities, and fish populations. The science-based monitoring programs were designed to further our understanding of aquatic ecosystems in the OSR and to monitor the aquatic environment for potential effects related to industrial development.

Alberta Environment and Protected Areas (AEPA) developed the Enhanced Monitoring Program (EMP) in 2018. The EMP was initially conceived (by the OSPW Science Team, OSPW-ST) to quantify the effects of a proposed pilot-scale release of treated OSPW to biophysical responses in the Athabasca River. The EMP was built through: (1) the integration and augmentation of sampling at existing monitoring sites deployed through the OSMP; and (2) the supplemental sampling at new monitoring sites situated in the vicinity of the proposed pilot-scale effluent release. The EMP focused on seven key components including (1) surface water chemistry, (2) benthic sediment chemistry, (3) epipelic algae, (4) benthic macroinvertebrates, (5) fish communities, (6) small and large-bodied fish health, and (7) burden of contaminants and Stable Isotope Ratios (SIRs) within the foodweb. The EMP focused on the mainstem of the Athabasca River, with data collected from 14 stations contained within the OSMP stations M3 through M7 (Figure 1, Figure 2, Figure 3, and Figure 4). Although the EMP sampled a smaller spatial scale than in OSMP, the stations included those upstream and downstream of the proposed pilot-scale release point of treated OSPW, which is located on the left (west) bank near Syncrude operations. The EMP was funded for three years by the OSPW-ST and deployed by AEPA 2018, 2019, and 2021 during the open water season. Developed and initially implemented in 2018, the EMP was expected to consist of two years of baseline data collection (2018 and 2019), followed by two years of pilot effluent release and exposure in 2020 and 2021. Ultimately, the proposed pilot release of treated OSPW was cancelled. No sampling took place in 2020 due to COVID-19. A third year of baseline monitoring under the EMP was executed in 2021.

Three government reports are available describing the study design, standard operating procedures, and activities completed to date (Hicks & Scrimgeour, 2019b, 2019a, 2020).

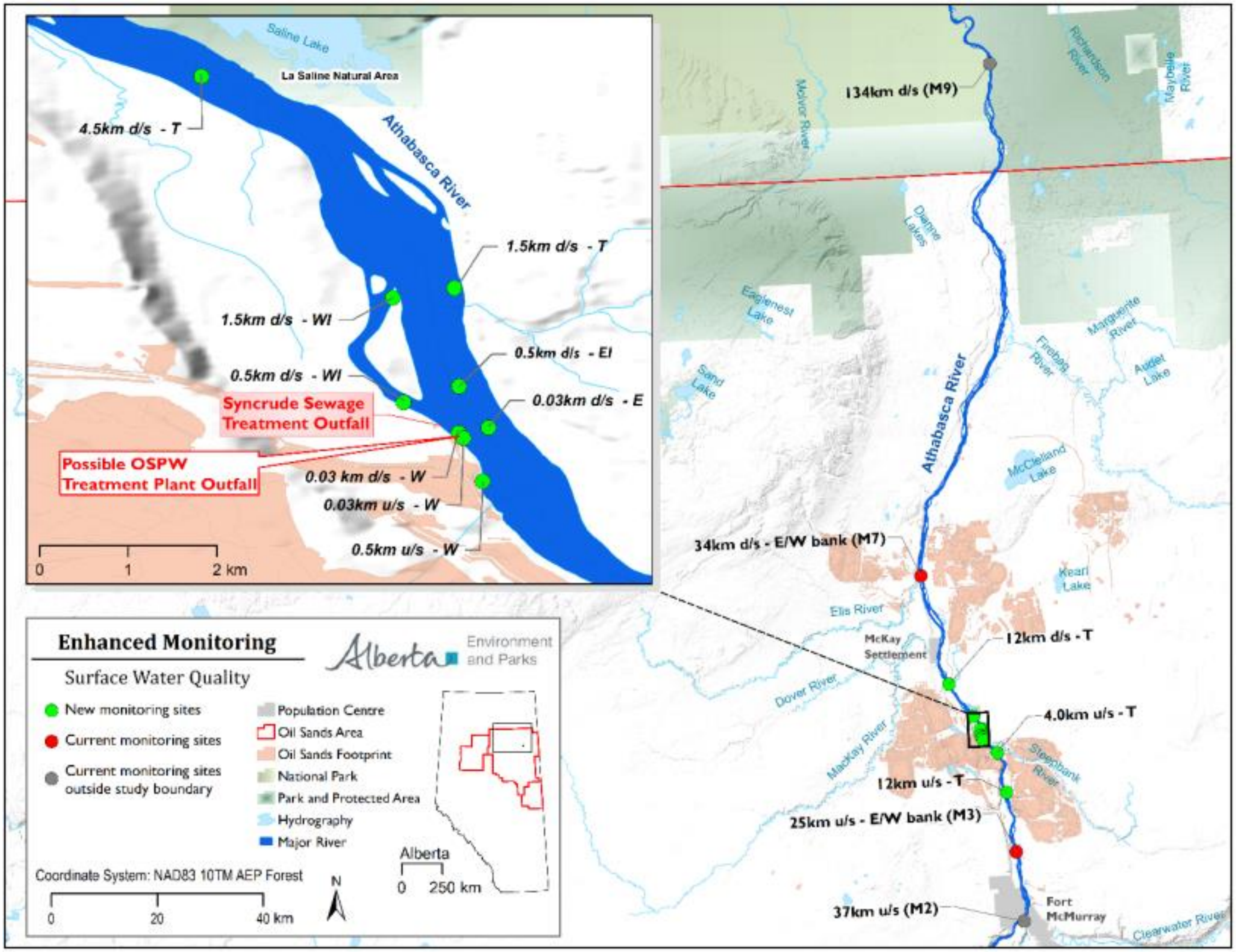

Figure 1 Oil Sands Monitoring (OSM) Program and Enhanced Monitoring Program (EMP) water sampling stations

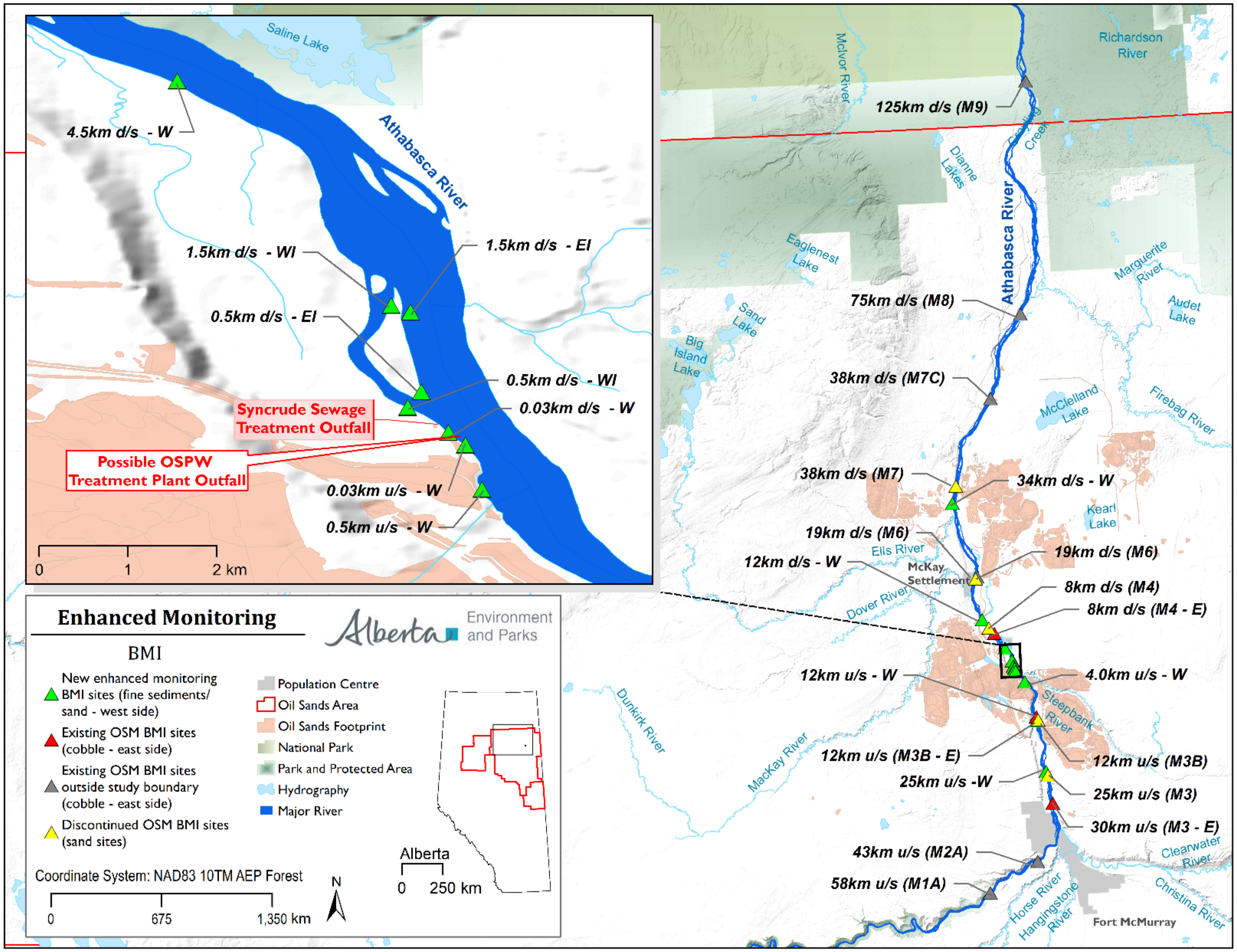

Figure 2 Oil Sands Monitoring (OSM) Program and Enhanced Monitoring Program (EMP) sediment, algae, and benthic sampling stations

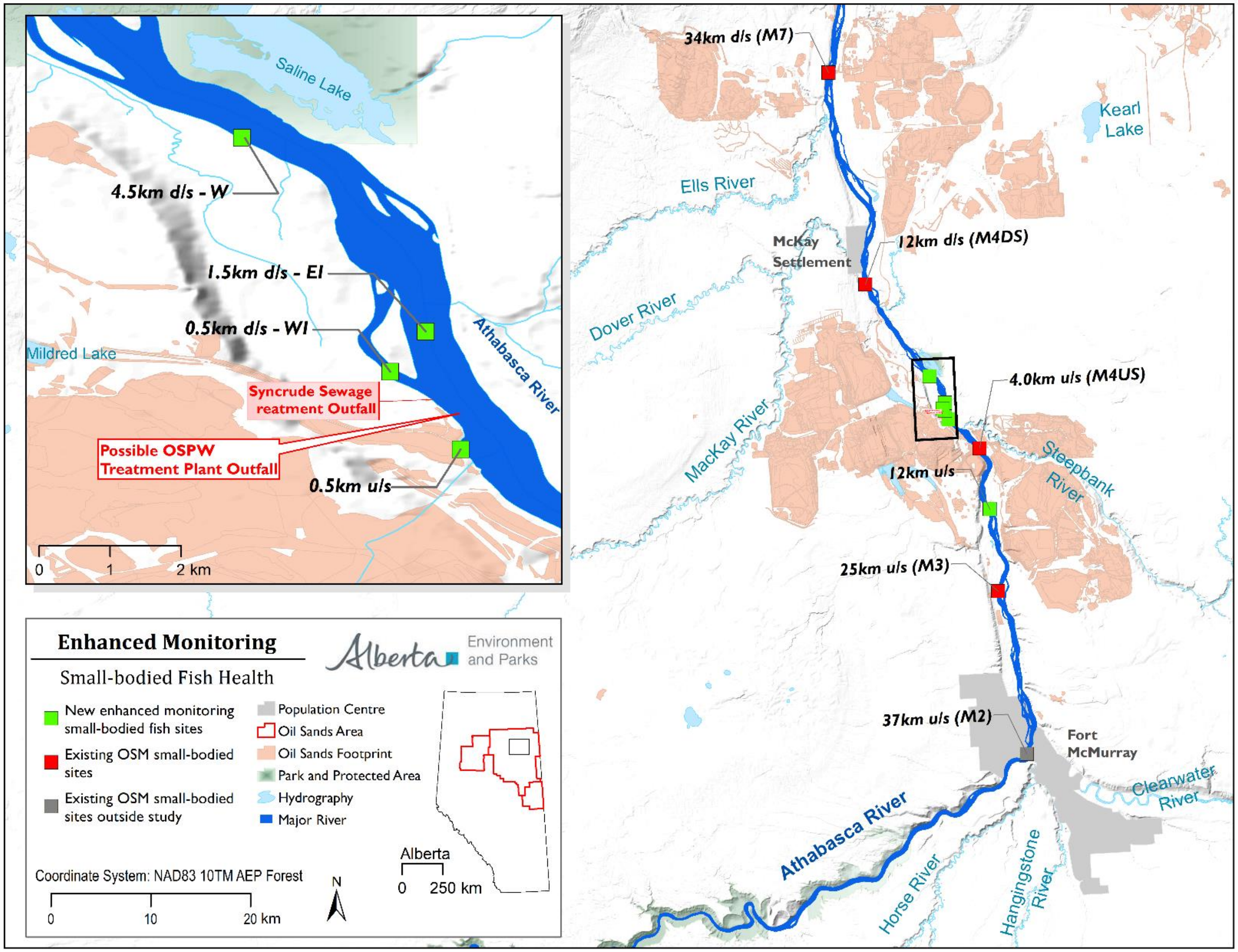

Figure 3 Oil Sands Monitoring (OSM) Program and Enhanced Monitoring Program (EMP) small-bodied fish sampling stations

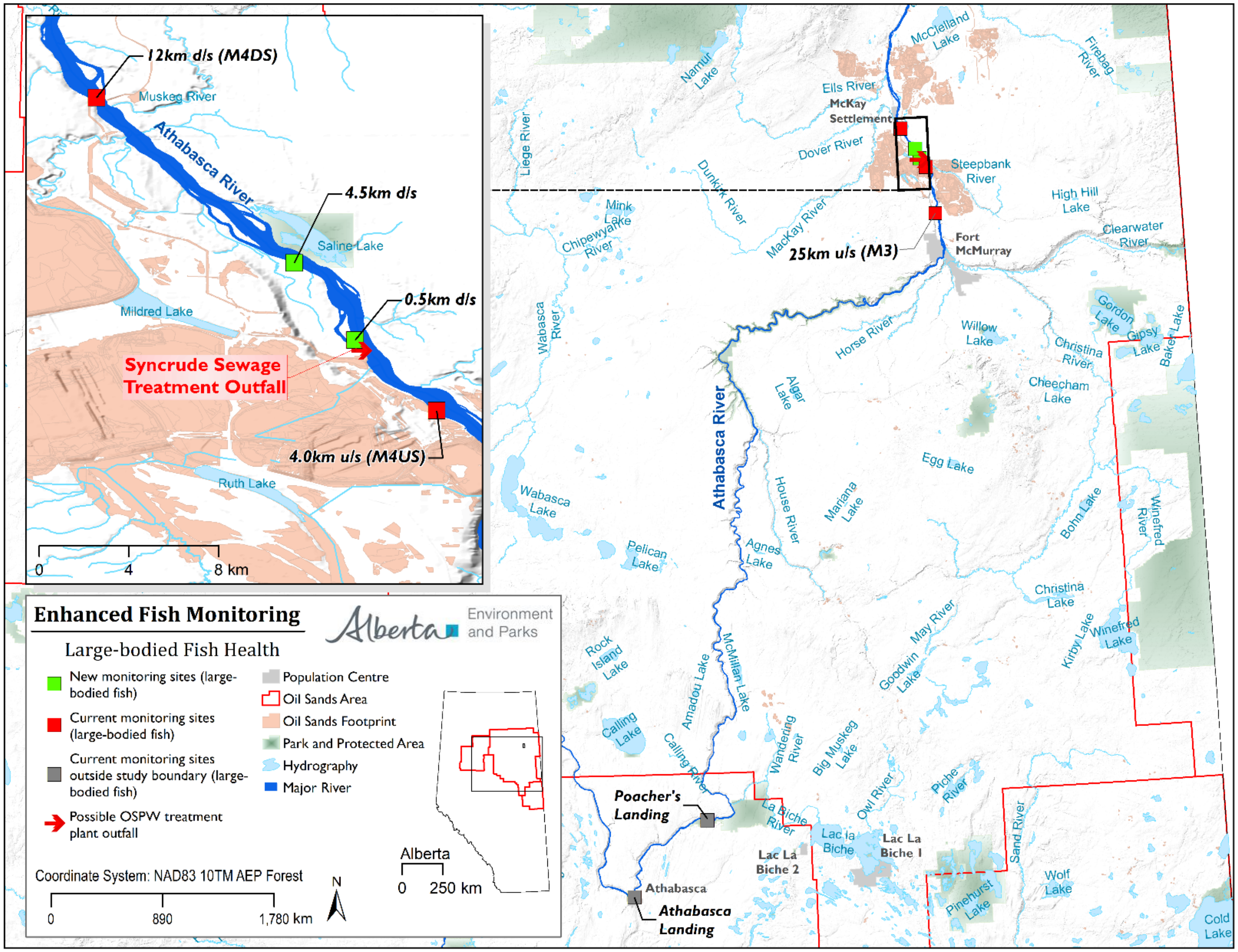

Figure 4 Oil Sands Monitoring (OSM) Program and Enhanced Monitoring Program (EMP) large-bodied fish sampling stations

1.2 Objectives and Report Structure

This report has four main objectives:

(1) summarize sources of variation in measured aquatic environment monitoring endpoints collected under the OSMP and EMP;

(2) determine if (and how) data collected under the OSMP can be used to support future EEM programs designed to assess point-source release of OSPW;

(3) quantify the ability (power) to statistically detect changes in the various EMP monitoring response endpoints;

(4) consider the information provided by the analyses addressing objectives 1 through 3 above, and provide recommendations on how EEM programs could be designed to assess the point-source release of treated OSPW.

This report is, therefore, broken down into four major tasks or sections, addressing each objective.

Task 1 (Section 2) presents an analysis and evaluation of data (water, sediment, algae, invertebrates, and fish) collected as part of the EMP for the Lower Athabasca River (LAR). Statistical analyses of the three years of monitoring data (i.e., 2018, 2019 and 2021) will define the variability in physical-chemical parameters and ecological indicators in the reach of the LAR where discharges of OSPW are considered the most likely to occur (at some time in the future).

Task 2 (Section 3) presents an analysis of relevant data from the OSMP to support the interpretation of the EMP data and more fully characterize the variability in the LAR.

Task 3 (Section 4) presents the results of a power analysis completed to assess the ability to statistically detect changes in the various EMP endpoints.

Task 4 (Section 5) provides recommendations towards the design of a generic Site-specific EEM program that would assess potential impacts of future releases of OSPW to the Lower Athabasca River.

2.0 TASK 1: ENHANCED MONITORING PROGRAM EVALUATION

2.1 Overview

The purpose of this Task 1 was to summarize the monitoring data collected under the EMP. In this summary, general statistical comparisons to environmental quality guidelines are provided. Further, data are statistically analyzed to quantify sources of variability in measured environmental responses. The overall objective of the data analysis was to identify factors to consider when designing a subsequent point-source focused aquatic-environment monitoring program.

2.2 Methodology

In the Sections that follow, we describe the data handling and analytical procedures used to address Task 1.

2.2.1 Data Assembly

Data were obtained for 14 stations located in the LAR, either downstream or upstream of the proposed pilot discharge point for OSPW (Table 1). A combination of raw and summary datasets were obtained from AEPA including data on water quality (major ions, minor elements, nutrients, carbon, hydrocarbons, naphthenic acids, polycyclic aromatic compounds (PAHs), cytotoxicity, etc.), sediment quality (metals and major ions), algal communities (densities, biomass, chlorophyll-*a*), benthic communities (taxa enumerations), fish health indicators (length and weight measurements), fish community assessment, and body burdens (benthos and fish). The frequency and timing of samples varied from year to year. Water samples were collected multiple times per year, while biotic samples were collected in the fall. The preliminary assembly of data consisted of assembling similar data types from multiple years into a single square matrix with variables (analytes, responses, taxa, etc.) as columns and observations (i.e., individual samples collected from a station-year) as rows.

In the context of the EMP, “Stations” were “areas” or “locations” where samples were collected, and where “sample” here is used as the lowest level of replication. Further, variation among samples is used in the various tests below to judge the significance of variations among Stations.

Table 1 Enhanced Monitoring Program (EMP) stations in the Lower Athabasca River (LAR)

Station Name	Station Description
AB07DA0800	Athabasca River 34 km downstream OSPW
AB07DA3008	Athabasca River 12 km downstream OSPW – Thalweg
AB07DA3009	Athabasca River 4.5 km downstream OSPW – Thalweg
AB07DA3015	Athabasca River 1.5 km downstream OSPW – Thalweg
AB07DA3016	Athabasca River 1.5 km downstream OSPW - West of island
AB07DA3017	Athabasca River 0.5 km downstream OSPW - East of island
AB07DA3018	Athabasca River 0.5 km downstream OSPW - West of island
AB07DA3019	Athabasca River 0.03 km downstream OSPW – Right bank
AB07DA3020	Athabasca River 0.03 km downstream OSPW – Left bank
AB07DA3021	Athabasca River 0.03 km upstream OSPW – Left bank

Station Name	Station Description
AB07DA3022	Athabasca River 0.5 km upstream OSPW – Left bank
AB07DA3023	Athabasca River 4.0 km upstream OSPW – Thalweg
AB07DA3024	Athabasca River 12 km upstream OSPW – Thalweg
AB07DA0062	Athabasca River 25 km upstream OSPW

2.2.2 Water Quantity

River flow (i.e., discharge) was anticipated to be a significant predictor of environmental and biological responses in the Athabasca River. Flow data for the Athabasca River station 07DA001 near Fort McMurray were obtained from the Water Survey of Canada (Government of Canada, 2022b). Data available at the time of download included those from 1957 through 2021. Daily average river discharge values (m^3/s) were compiled for all of 1957 through 2021. Those data were used to determine flow volumes on the day environmental samples were collected (Q). Additionally, the average discharge for the 60 days prior to sampling (Q60) were computed for use as potential covariables for the various biotic indices, following Kilgour et al. (2019a).

Hydrographs were used to describe flows in 2018, 2019 and 2021 relative to historical flow levels, giving context to the flow periods during the EMP. The graphs also demonstrated patterns of increase/decrease over various seasons, in which different data were collected for the EMP. Finally, summary statistics (i.e., minimum, maximum, and mean) for the three years of the EMP were also calculated.

2.2.3 Surface Water Quality Data

Surface water in the Athabasca River will be the recipient of any released treated OSPW. Quantifying the spatial and temporal variability in surface water parameters (i.e., metals, nutrients, polycyclic aromatic compounds (PAHs), and naphthenic acids (NAs)) and identifying potential drivers of this variability is crucial to understanding how water quality may be affected by released OSPW.

The purpose of analysis of discrete surface water quality data was to quantify sources of variability. The analysis of water quality data involved the following steps:

1. Data Processing;
2. Data Summary;
3. Explanatory Models; and,
4. Visualization of Trends.

Each step is described below.

2.2.3.1 Data Processing

2.2.3.1.1 General Data Cleaning

A variety of data processing steps were caried out to generate a single, workable, balanced EMP surface water quality dataset that encompassed all sampling years, stations, and parameters. These general steps are outlined below:

- Three bulk datasets were provided from AEPA, one for each sampling years (2018, 2019, and 2021) in "long format", the first step involved adding a "Parameter Group" column to the 2021 dataset which was missing, as to remain consistent with the other two sampling years;

- Individual parameter names were summarized from each dataset and compared to ensure continuity, differences in bracket use, i.e. [] vs. (), primarily for individual polycyclic aromatic compound (PAC) analytes, were corrected;

- Specifically for PAHs, any deuterated surrogate data were removed from the datasets, as it was assumed all data had already been previously QA/QC'd;

- Station numbers and long names were summarized for each dataset to confirm overlap, any errors between datasets in station number/name were confirmed with AEPA staff and corrected;

- Any QA/QC samples such as trip blanks, field blanks, or lab blanks were removed from the datasets to ensure only environmental data were to be considered in further analysis;

- The data was provided from AEP in "long format", which is the ideal framework for data analysis, and did not require any additional re-formatting steps.

2.2.3.1.2 Dealing with Detection Limits

The percentage of samples below detection limits (DL) were summarized at the station x year level for each individual analyte. Analytes that were below DL in over 80% of occurrences were removed from future analysis.

2.2.3.1.3 Complex Parameter Groups

The high-resolution polycyclic aromatic compounds (PAHs) data comprised of 75 individual analytes belonging to two main groups: alkylated and parent PAHs. A new term (total PAC) was calculated which sums the alkylated and parents PAHs. Analytes that were < MDL were replaced with zero.

Naphthenic acids (NAs) were measured in the EMP using HPLC-Orbitrap-MS, which produces estimated concentrations of NAs that are classified as to the number of carbons and double bonds. The high-resolution NAs data in the EMP dataset were used to compute NAs Toxic Units (TUs) for Rainbow Trout and Fathead Minnow. NAs data were used to calculate the sum of percentage by carbon number (Double bond energies - DBEs 1 to 7), which were then converted to mg/L and used to compute toxic units (TU's) for NAs with 9 to 21 carbons. For Rainbow Trout, toxic units (i.e., multiples of the anticipated LC50) are defined by Hughes et al. (2017a) per the following equation:

$$TU = \sum [O_2^-]_i \times e^{0.591i-12.53}$$

Where, $[O_2^-]_i$ is the concentration of NAs with i carbons. This equation was used to compute Rainbow Trout toxicity unit for each sample. For Fathead Minnow, the following toxic unit model from Scarlett et al. (2012) was also applied:

$$TU = \sum \frac{[O_2^-]_i}{e^{11.06-0.591i}}$$

In the case of the Scarlett et al equation for Fathead Minnow, the denominator is the LC50 for the i^{th} carbon number, such that the result is a multiple of the LC50.

Results from a water cytotoxicity test, developed to assess environmental water quality by Alberta Health and the Alberta Center for Toxicology (ACFT) at the University of Calgary, were included in the surface water quality dataset. The output of the test, the Cytotoxicity Water Quality Index (C-WQI), is a measure of potential human health risk, as testing utilizes human hepatocarcinoma (HepG2) cells. C-WQI values having been shown to correlate well with results of Microtox testing, a widely used microbiological effects-based assay for environmental water quality monitoring (Pan et al., 2013). Lower C-WQI values indicate lower cytotoxicity and higher values indicate higher cytotoxicity.

2.2.3.1.4 Data Amalgamation

After each EMP surface water quality dataset (i.e., 2018, 2019, and 2021) were summarized following the steps above, we ensured all column names matched. All three datasets were combined using the bind_row() function available in the *dplyr* (Wickham et al., 2015) R package (R Studio ver. 4.1.3).

2.2.3.2 Parameter Selection

To simplify the list of over 200 analytes included in EMP dataset, as well as to focus on elements of concern from mining and upgrading bitumen in the OSR, a subset of parameters were selected for more detailed study. These included concentrations of nickel (Ni) and vanadium (V) per (Klemt et al., 2020). These also included concentrations of chloride (Cl-), sulphate (SO_4^{2}), sodium (Na^+), calcium (Ca^{2+}), magnesium (Mg^{2+}), and total concentrations of aluminum (Al), molybdenum (Mo), thallium (Tl) and iron (Fe) per Arciszewski et al. (2018). Additionally, total alkalinity (measured as mg $CaCO_3$), copper (Cu^{2+}), lead (Pb^{2+}), total phosphorous (TP), and zinc (Zn) were included as well as these have associated WQG's which can be used to inform on power analyses and study design considerations. Finally, total NAs concentration, NAs toxic units, total PAHs, cytotoxicity WQI's, total nitrites and nitrates (as N), and total nitrogen were also included in the modelling exercises. This subset of 22 parameters became the primary focus of further surface water quality analysis.

2.2.3.3 Data Summary

The number of samples below DLs were summarized for each parameter. The number of sampling events along with the type of samples collected (i.e., discrete samples, triplicate samples, vertical samples, and blank samples) each year at each station were summarized. Summary statistics (i.e., sample size, minimum, maximum, mean, median, standard deviation (SD), standard error (SE), and the 90^{th} percentile) were computed for select analytes. Exceedances of Canadian Council of Ministers of the Environment (CCME) Long-Term and Short-Term Water Quality Guidelines (WQG) as well as Alberta's Provincial WQG's were investigated and summarized for each sampling year.

2.2.3.4 Explanatory Models

General linear models (GLMs) were used to explore potential sources of variation in measured analyte concentrations.

The GLM equation was:

$$Y = constant + Q + Year + Dist_{OSPW} + Dist_{Shore} + Q * Year$$

where, Y is the log_{10} of the analyte concentration, Q is the log_{10} of flow volume on the date that the water sample was collected, Year is the year of sampling treated as a continuous covariable, $Dist_{OSPW}$ is the distance from the proposed future OSPW discharge point (with stations upstream assigned a negative distance, stations downstream assigned a positive distance), and $Dist_{Shore}$ is the distance measured from the eastern shoreline.

Flow volume (i.e., river discharge, Q) was included as a covariable as it can have a significant influence on concentrations of metals and nutrients (and other constituents) in the Athabasca River (Hatfield Consultants et al., 2016; Arciszewski et al., 2017). Discharge at the time of surface water sampling has also been shown to be an important parameter for interpreting temporal and spatial patterns in water quality (Glozier et al., 2018). Flow values (i.e., discharge on the day of sampling) were assigned to each observation (i.e., station x year) in the EMP dataset. Historical hydrometric data were obtained online for the Fort McMurray area (Government of Canada, 2022b).

Year was included as a covariable because there has been some evidence that some water quality variables in the Athabasca River are increasing over time (Arciszewski, Munkittrick, Scrimgeour, et al., 2017a). It is important to know if there are temporal trends in water quality variables over time, prior to the release of treated ground or process water, so that release of treated waters is not implicated as the cause of future increasing trends. The interaction term between Year and Q was also included in the model.

Regardless of the significance of the various terms in the model, the GLM were used to compute coefficients for the various terms, which were assembled and could be used to estimate expected surface water analyte concentrations. Variables that are significant will have model coefficients that are significantly different from zero. Variables that were non-significant will produce model coefficients that are not significantly different from zero, while estimates for those coefficients will be very close to zero (reflecting their non-significance). For the purpose of the modeling exercise here, all models (and associated coefficients) were retained so that readers could explore for themselves the influence of significant and non-significant covariables on estimated analyte concentrations.

2.2.3.5 Visualization of Trends

Temporal and spatial variability in surface water quality was explored and illustrated using various graphing techniques including scatterplots and boxplots. A Principal Component Analysis (PCA), and associated biplots, was used to illustrate variation and covariation of analytes among surface water samples for the various years and stations. Analyte concentrations were normalized to a common river flow (if the analyte is significantly affected by flow) prior to graphing to improve the interpretation of potential temporal and spatial variations by accounting for variations due to natural conditions. Data were normalized to a discharge of 900 m^3/s, which was the average of the 3 years of sampling for the EMP.

Data normalization was done in 4 steps. First, the relationship between each individual analyte and discharge was determined using a GLM. Second, the coefficients (i.e., slope and intercept) from the GLM were used to calculate the predicted concentration of the analyte, given the observed discharge on the day of sampling (estimated using data from the flow station 07DA001). Third, residuals were computed as the difference between observed (i.e., measured) concentrations and predicted concentrations. Finally, analyte concentrations were normalized to a discharge of 900 m^3/s, as follows:

$$\log(concentration)_{obs} = \text{slope} * \log(Q_{obs}) + intercept$$

$$residual = observed - predicted$$

$$normalized(concentration) = (slope * \log(900) + residuals) + intercept$$

2.2.4 Semipermeable Membrane Devices (SPMDs)

Semipermeable Membrane Devices (SPMDs) were deployed at each of the 12 EMP stations between 6 and 10 times throughout the 3 years of monitoring (2018, 2019, and 2021). The SPMDs remained in the river for 28 to 30 days and then collected. Paired surface water grab samples were collected at both the time of SPMD deployment and collection. The EMP surface water grab sample dataset was combined with SPMD data and filtered for grab samples collected on the final day of SPMD sampling and a subsequent linear regression was carried out.

The purpose of analysis of SPMD data was to assess its relationship with grab samples and quantify sources of variability. The analysis of SPMD data involved the following steps:

1. Data Processing;
2. Relationship with Grab Samples; and,
3. Explanatory Models.

Each step is described below.

2.2.4.1 Data Processing

Contrary to surface water grab samples previously discussed, the SPMD data did not require rigorous data processing.

2.2.4.2 Relationship with Grab Samples

Linear regression was used to compare the two datasets and explore the relationship between total PAC concentrations in the SPMDs, which are made up by the dissolved fraction, and grab sample, which are made up of both dissolved and particulate PAC fractions.

The linear model structure was:

$$\log{(Grab)} = \log{(SPMD)} + b$$

where Grab is the $\log_{10}$ of total PAC concentrations in the grab sample and SPMD is the total PAC concentration in the SPMD sample. Metrics such as R-square, slope, and intercept were used to assess the fit and predictability of the data.

Following the approach outlined by Kim et al., (2014), total PAC calculations included PAC analytes that had detection frequencies of over 50% and relative standard deviations below 25% across within sampling station and year to ensure abundant analytes were being considered and to take into account differences in detection limits.

2.2.4.3 Explanatory Models

General linear models (GLMs) were used to explore potential sources of variation in measured analyte concentrations.

The GLM equation was:

$$Y = constant + Qavg + Year + Dist_{OSPW} + Qavg * Year$$

where, Y is the $\log_{10}$ of the analyte concentration, Qavg is the $\log_{10}$ of the average flow volume over the period that the SPMD was deployed, Year is the year of sampling treated as a continuous covariable, and $Dist_{OSPW}$ is the distance from the proposed future OSPW discharge point (with stations upstream assigned a negative distance, stations downstream assigned a positive distance).

Average flow values were assigned to each observation (i.e., station x year) in the EMP dataset. Historical hydrometric data were obtained online for the Fort McMurray area (Government of Canada, 2022b) and deployment times for each individual SPMD were provided by AEPA. We used the runMean() function available in the *TTR* (Ulrich et al., 2023) R package (R Studio ver. 4.1.3) to calculate the running mean discharge based on the length of SPMD deployment (26 to 50 days).

Year was included as a covariable because there has been some evidence that some water quality variables in the Athabasca River are increasing over time (Arciszewski, Munkittrick, Scrimgeour, et al., 2017a). It is important to know if there are temporal trends in water quality variables over time, prior to the release of treated ground or process water, so that release of treated waters is not implicated as the cause of future increasing trends. The interaction term between Year and Q was also included in the model.

Regardless of the significance of the various terms in the model, the GLM were used to compute coefficients for the various terms, which were assembled and could be used to estimate expected surface water analyte concentrations. Variables that are significant will have model coefficients that are significantly different from zero. Variables that were non-significant will produce model coefficients that are not significantly different from zero, while estimates for those coefficients will be very close to zero (reflecting their non-significance). For the purpose of the modeling exercise here, all models (and associated coefficients) were retained so that readers could explore for themselves the influence of significant and non-significant covariables on estimated analyte concentrations.

2.2.5 Sediment Quality Data

The purpose of analysis of sediment quality data was to quantify sources of variability. The analysis of sediment quality data involved the following steps:

1. Data Processing;
2. Data Summary;
3. Explanatory Models; and,
4. Visualization of Trends.

Each step is described below.

2.2.5.1 Data Processing

2.2.5.1.1 General Data Cleaning

Five bulk sediment datasets were provided from AEPA, one for each parameter group (i.e., mercury, naphthenic acids, nutrients, PAHs, and metals), in “wide format”. The datasets were received in the general order as exhibited in the example in Table 2.

Table 2 Example of the original format of sediment PAC concentration data as received.

Station Name	Sample ID	Date	Flag	RDL	Analyte 1	Flag	RDL	Analyte 2	…	Flag	RDL	Analyte X
					ng/g			ng/g				ng/g
AR34d/sOSPW-1	18SWE06890	2018-09-24	NA	0.8	3.15	<	0.595	4.19	…	NA	0.113	3.18
AR34d/sOSPW-3	18SWE06891	2018-09-24	<	2.2	15	<	0.643	12.9	…	NA	0.102	9.17
AR34d/sOSPW-5	18SWE06892	2018-09-24	NA	1.5	10.4	<	0.68	8.23	…	<	0.103	4.99

This format requires multiple manual steps to get the data in a workable format for data analysis (i.e., long format). First, each of the five datasets were loaded into R (R Studio ver. 4.1.3) and converted into long format using the gather() function from the *tidyr* R package (Wickham & Henry, 2020), creating a column that encompassed the analyte names, flags, and Reportable Detection Limits (RDLs). The datasets were manually inspected to ensure they were of identical length, and merged using the full_join() function in the *dplyr* R package (Wickham et al., 2015), thus converting the wide format data into long format as exhibited in the example in Table 3.

Table 3 Example of the reconfiguration of sediment PAC concentration data that was undertaken in order to make data usable for analysis.

Station Name	Sample ID	Date	Flag	RDL	Analyte name	Concentration	Unit
AR34d/sOSPW-1	18SWE06890	2018-09-24	NA	0.8	Analyte 1	3.15	ng/g
AR34d/sOSPW-3	18SWE06891	2018-09-24	<	2.2	Analyte 1	15	ng/g
AR34d/sOSPW-5	18SWE06892	2018-09-24	NA	1.5	Analyte 1	10.4	ng/g
AR34d/sOSPW-1	18SWE06890	2018-09-24	<	0.595	Analyte 2	4.19	ng/g
AR34d/sOSPW-3	18SWE06891	2018-09-24	<	0.643	Analyte 2	12.9	ng/g
AR34d/sOSPW-5	18SWE06892	2018-09-24	<	0.68	Analyte 2	8.23	ng/g
AR34d/sOSPW-1	18SWE06890	2018-09-24	NA	0.113	Analyte X	3.18	ng/g
AR34d/sOSPW-3	18SWE06891	2018-09-24	NA	0.102	Analyte X	9.17	ng/g
AR34d/sOSPW-5	18SWE06892	2018-09-24	<	0.103	Analyte X	4.99	ng/g

These specific steps outlined above had to be repeated for each of the five datasets, after which, the following data processing steps were carried out to generate a single, workable, balanced EMP sediment quality dataset that encompassed all sampling years, stations, and parameters. These general steps are outlined below:

- Individual parameter names were summarized from each dataset and compared to ensure continuity, differences in bracket use, i.e. [] vs. (), primarily for individual PAC analytes, were corrected;
- Specifically for PAHs, any deuterated surrogate data were removed from the datasets, as we assumed all data had already been previously QA/QC'd;
- Station number's and long names were summarized for each dataset to confirm overlap, any errors between datasets in station number/name were confirmed with AEPA staff and corrected;
- Any QA/QC samples such as trip blanks, field blanks, or lab blanks were removed from the datasets to ensure only environmental data were to be considered in further analysis;

2.2.5.1.2 Dealing with Detection Limits

The percentage of samples below DLs were summarized at the station x year level for each individual analyte. Analytes that were below DL in over 80% of occurrences were removed from future analysis.

2.2.5.1.3 Data Amalgamation

After each EMP sediment quality dataset (i.e., 2018, 2019, and 2021) were summarized following the steps above, we ensured all column names matched. The datasets were provided in "long format", with a parameter column and a concentration column. All three datasets were combined using the bind_row() function available in the *dplyr* (Wickham et al., 2015) R package (R Studio ver. 4.1.3).

2.2.5.2 Data Summary

The number of samples below DLs were summarized for each parameter. The number of sampling events along with the number of grab samples collected each year at each station were summarized. Exceedances of Canadian Sediment Quality Guidelines for the Protection of Aquatic Life were investigated and summarized for each sampling year.

2.2.5.3 Explanatory Models

GLMs were used to explore potential sources of variation in measured sediment analyte concentrations. The GLM structure was:

$$Y = constant + Al + Year + Dist_{OSPW} + Al * Year$$

where, Y is the $\log_{10}$ of the analyte concentration, Al is the $\log_{10}$ of aluminum concentration in the sediment sample, Year is the year of sampling treated as a continuous covariable, and $Dist_{OSPW}$ is the distance from the proposed future OSPW discharge point (with stations upstream assigned a negative distance, stations downstream assigned a positive distance). Year and the interaction term between Year and Al was also included in the model.

Aluminum was included as a covariables because is it commonly used, along with Fe and Li, as a reference analyte with which to normalize sediment data (Charlesworth & Service, 2000; Ho et al., 2012; Loring, 1991). Sedimentary loads will vary naturally with grain-size and the mineral composition (UNEP, 1995). Normalizing can help differentiate between inputs derived from natural variation and anthropogenic activities (Loring, 1991; Yau & Gray, 2005).

Regardless of the significance of the various terms in the model, the GLM were used to compute coefficients for the various terms, which were assembled and could be used to estimate expected sediment analyte concentrations.

2.2.5.4 Visualization of Trends

Temporal and spatial variability in sediment quality was explored using various graphing techniques including scatterplots and boxplots. A PCA and associated biplots was used to illustrate variation and covariation of analytes among sediment samples for the various years and stations. Analyte concentrations were normalized to a common aluminum concentration prior to graphing to improve the interpretation of potential temporal and spatial variations by accounting for variations aluminum concentrations in the samples. Data were normalized to an aluminum concentration of 6000 µg/g, which was the rounded average of the 3 years of sampling for the EMP.

Data normalization was done in 4 steps. First, the relationship between each individual analyte and aluminum concentrations was determine using a GLM. Second, the coefficients (i.e., slope and intercept) from the GLM were used to calculate the predicted concentration of the analyte, given the observed aluminum concentration in the sediment sample. Third, residuals were computed as the difference between observed (i.e., measured) concentrations and predicted concentrations. Finally, analyte concentrations were normalized to an aluminum concentration of 6000 µg/g, as follows:

$$\log(concentration)_{obs} = \text{slope} * \log(Al_{obs}) + intercept$$

$$residual = observed - predicted$$

$$normalized(concentration) = (slope * \log(6000) + residuals) + intercept$$

2.2.6 Algal Community Data

Algae assemblages are not studied as part of a typical EEM program to assess the presence of effluent related effects under the Metal and Diamond Mining Effluent Regulations (MDMER). Algal communities were, however, sampled as part of the EMP. The analysis of algal community data involved the following steps:

1. Calculation of indices of community composition;
2. Data Summary
3. Explanatory Models; and,
4. Data Visualization.

Each step is described below.

2.2.6.1 Indices of Community Composition

The assessment of the algae assemblages involved computing the following indices of community composition:

- Total Density (number of organisms per m^2);
- Taxa Richness (number of groups at the lowest practical level (LPL) per sample);
- Simpson's Diversity (at the LPL);
- Simpson's Evenness (at the LPL);
- Bray-Curtis (BC) Index of Composition;
- Chlorophyll-*a* (Chl-*a*) levels (µg/cm^2);
- Biomass g/m^2; and,
- Non-metric Multidimensional Scaling (NMDS) Axis 1 and 2 scores

Simpson's Evenness is a measure of the distribution of abundance among taxa (B. Smith & Wilson, 1996). It is a core EEM index under the MDMER and was calculated as recommended by the EEM guidance document (Environment Canada, 2012a), as:

$$E = \frac{\frac{1}{\sum (p_i)^2}}{S}$$

Where p_i is the proportion that taxon *i* contributes to the total number of organisms in a sample, and S is the number of families.

Simpson's Diversity is a measure of diversity which considers both richness and evenness. This index is calculated by determining the proportion of individuals for each taxonomic group that contributes to the total number of individuals in the sample. The value of the Simpson's diversity index ranges between 0 and 1 where the greater the value, the greater the sample diversity suggesting a more stable biological community. Simpson's Diversity was computed as:

$$D = 1 - \sum (p_i)^2$$

The **Bray-Curtis distance measure** quantifies similarity in composition of taxa between pairs of samples, and is a conventional measure used as input into NMDS. The Bray-Curtis distance measure has a maximum value of 1 when two samples have entirely different fauna, and a minimum value of 0 when two samples have identical fauna in the exact same abundances. The formula for the Bray-Curtis distance measure is from Legendre and Legendre (1998).

$$BC = \frac{\sum |y_{i1} - y_{i2}|}{\sum |y_{i1} + y_{i2}|}$$

where, BC is the Bray Curtis distance between samples 1 and 2, y_{i1} is the abundance of species *i* in sample 1, y_{i2} is the abundance of species *i* in sample 2, and n is the number of species present in the two samples. Bray-Curtis distances were computed between all possible pairs of community samples (for algae and benthos, separately) to create a triangular matrix of ecological distances. Taxa densities were $\log_{10}$ transformed to provide interpretable Bray-Curtis distances. The triangular Bray-Curtis distance matrix was ordinated using NMDS to compute synthetic ordination axes that are intended to interpret the inter-sample distances in a reduced dimensionality (in this case, two ordination axes). Prior to carrying out the NMDS, rare taxa that accounted for less than 0.5% of the total abundance across all samples were removed.

Chlorophyll-*a* levels were measured in waters collected using a syringe from the interface of fine benthic sediments and water (i.e., epipelic algae).

2.2.6.2 Data Summary

The relative abundance (%) of the various taxa identified to LPL in the algal community samples were calculated across stations. Summary statistics (minimum, maximum, mean, SD) of algal community indicators were computed by station and year. Variability of observations (station x year) in relation to potential environmental covariables (i.e., discharge and air temperature) were explored using a variety of tools including Spearman Rank correlation analysis, scatterplots and PCA.

2.2.6.3 Explanatory Models

GLMs were computed for algal indices of community composition, along with flow volume, year of sampling, and the distance from the potential OSPW discharge as predictors. Models had the following structure:

$$Y = constant + \ Q_{60} + Year + Dist_{OSPW}$$

where, Y is an indicator of algal community, Q_{60} is the log_{10} of flow volume averaged over the 60 days prior to sampling, Year is the year of sampling treated as a continuous covariable, and $Dist_{OSPW}$ is the distance from the proposed future OSPW discharge point (with stations upstream assigned a negative distance, stations downstream assigned a positive distance).

Discharge was included in the models as it has been demonstrated that this natural environmental factor impacts the variability in other biotic indices, such as benthic community and fish health endpoints (Kilgour et al., 2019a). Flow (Q60) values were assigned to each observation (i.e., station x year combination) in the EMP dataset. Historical hydrometric data were obtained online for the Fort McMurray area (Government of Canada, 2022b) (Government of Canada, 2022a). Air temperature, a surrogate for water temperature, has also been shown to impact biotic variability (Kilgour et al., 2019a). However, with only three years of data, the covariable was dropped in the modelling process.

2.2.6.4 Visualization of Trends

Temporal and spatial variability in algal indices of community composition was explored and illustrated using various graphing techniques including scatterplots and boxplots. Algae indices were normalized to a common river flow prior to graphing to improve the interpretation of potential temporal and spatial variations by accounting for variations due to natural conditions. Data were normalized to a discharge of 900 m^3/s, which was the average of the 3 years of sampling for the EMP, as described in Section 2.2.3.4.

2.2.7 Benthic Invertebrate Community Data

Benthic invertebrate community studies are used in EEM to assess the presence of effluent related effects (Environment Canada, 2012a) and will be an important biological receptor of potential effects of discharged treated OSPW. The analysis of benthic community data involved the following steps:

1. Calculation of indices of community composition;
2. Data Summary;
3. Explanatory Models; and,
4. Visualization of Trends.

Each step is described below.

2.2.7.1 Indices of Community Composition

The assessment of the benthic invertebrate community involved computing the following indices of community composition:

- Total Density (number of organisms per sample);
- Taxa Richness (number of groups at the LPL per sample);
- Simpson's Evenness;

- Simpson's Diversity;
- Bray-Curtis Index of Composition;
- Chironomidae Pollution Tolerance Index (PTI); and
- Ephemeroptera-Plecoptera-Trichoptera (EPT) Index.

Simpson's Evenness, Simpson's Diversity and Bray-Curtis Index were calculated as described in Section 2.2.6.1, using taxa at the LPL.

Since the majority of benthic invertebrates in depositional zones of the LAR are Chironomidae larvae (i.e., non-biting midges), a **Benthic PTI** was computed using chironomid pollution tolerance data compiled by Namayandeh and Culp (2016). A weighted mean PTI value was calculated for each benthic sample using assigned tolerance values for each chironomid genera. For each sample, the weighted mean PTI was calculated using the equation:

$$PTI = \frac{\sum yi * PT_i}{\sum yi}$$

where, PTI is the pollution tolerance index for any given sample, y_i is the density of genus *i* in sample and PT_i is the associated pollution tolerance of each genus. The PT_i values were taken from Namayandeh and Culp (2016). The PTI index ranges between 0 and 10, with higher values representing a chironomid assemblage that is overall more tolerant to pollution.

The **Benthic EPT Index** is a measure of the proportion of freshwater organisms belonging to the taxa Ephemeroptera, Plecoptera, and Trichoptera. These species are considered sensitive to pollution and are a measure of good environmental condition.

2.2.7.2 Data Summary

The relative abundance (%) of the various taxa identified to LPL in the benthic samples were calculated across stations. Summary statistics (minimum, maximum, mean, SD) of benthic invertebrate community indicators were computed by station and year. Variability of observations (station x year) in relation to potential environmental covariables were explored using a variety of tools including Spearman Rank correlation analysis, scatterplots and PCA.

2.2.7.3 Explanatory Models

GLMs were computed for benthic indices of community composition, along with flow volume, particle size, year of sampling, and the distance from the potential OSPW discharge as predictors. Models had the following structure:

$$Y = constant + \ Q_{60} + PS + Year + Dist_{OSPW}$$

where, Y is an indicator of benthic community, Q_{60} is the $\log_{10}$ of flow volume averaged over the 60 days prior to sampling, PS is the mean particle size (mm) of the sample, Year is the year of sampling treated as a continuous covariable, and $Dist_{OSPW}$ is the distance from the proposed future OSPW discharge point (with stations upstream assigned a negative distance, stations downstream assigned a positive distance).

Discharge and mean particle size were included in the models as it has been demonstrated that these natural environmental factors impact the variability in indices of community composition (Kilgour et al., 2019a). Flow and mean particle size values were assigned to each observation (i.e., station x year combination) in the EMP dataset. Historical hydrometric data were obtained online for the Fort McMurray area (Government of Canada, 2022b). Mean particle size was not directly measured, but was inferred based on the dominant substrate assigned to each sample using the following equation:

$$PS = (1.03125 * \%sand + 0.0332 * \%silt + 0.00198 * \%clay)/100$$

Where *%sand, %silt, and %clay* represent the percentage of each substrate type as determined by grain size analysis and the constants represent the mean particle size of each substrate class based on Wentworth (1922)'s grain size classification for particle size. For example, clay is considered any particle that measures between 0.00006 and 0.0039 mm, the average size of a clay particle is therefore 0.00198 mm.

2.2.7.4 Visualization of Trends

Temporal and spatial variability in benthic indices of community composition was explored and illustrated using various graphing techniques including scatterplots and boxplots. Benthic indices were normalized to a common river flow and sediment particle size prior to graphing to improve the interpretation of potential temporal and spatial variations by accounting for variations due to natural conditions and habitat. Data were normalized to a discharge of 900 m^3/s and to a particle size value of 0.7 mm, which were both the average of the covariables for the 3 years of sampling for the EMP.

Data normalization was done in 4 steps. First, the relationship between each individual benthic index and both flow and particle size were determine using a GLM. Second, the coefficients (i.e., slope and intercept) from the GLM were used to calculate the predicted index value, given the observed flow (Q60) and particle size. Third, residuals were computed as the difference between observed (i.e., measured) concentrations and predicted concentrations. Finally, benthic indices were normalized to discharge and particle size, as follows:

$$benthic\ index_{obs} = \text{slope} * \log(Q60_{obs}) + slope * \log(PS_{obs}) + intercept$$

$$residual = observed - predicted$$

$$normalized\ index = (slope * \log(900) + slope * \log(0.7) + residuals) + intercept$$

2.2.8 Fish Community Assessment

Fish community assemblages are typically reported during EEM activities and are important to consider when discussing potential effects of discharged treated OSPW. The analysis of fish community data involved the following steps:

5. Calculation of indices of community composition;

6. Data Summary;

7. Explanatory Models; and,

8. Visualization of Trends.

Each step is described below.

2.2.8.1 Indices of Community Composition

The assessment of the fish community involved computing the following indices of community composition:

- Total abundance (number of fish caught);
- Species Richness (number of species caught);
- Simpson's Evenness;
- Simpson's Diversity; and,
- Bray-Curtis Index of Composition.

Simpson's Evenness, Simpson's Diversity and Bray-Curtis Index were calculated as previously described in Section 2.2.6.1.

2.2.8.2 Data Summary

The relative abundance (%) of the various species were calculated across stations. Summary statistics (minimum, maximum, mean, SD) of fish community indicators were computed by station and year.

2.2.8.3 Explanatory Models

GLMs were computed for benthic indices of community composition, along with flow volume, particle size, year of sampling, and the distance from the potential OSPW discharge as predictors. Models had the following structure:

$$Y = constant + \ Q_{60} + Year + Dist_{OSPW} + effort$$

where, Y is an indicator of fish community, Q_{60} is the log_{10} of flow volume averaged over the 60 days prior to sampling, Year is the year of sampling treated as a continuous covariable, $Dist_{OSPW}$ is the distance from the proposed future OSPW discharge point (with stations upstream assigned a negative distance, stations downstream assigned a positive distance), and effort is the total electrofishing time spent to catch a given fish assemblage.

Historical hydrometric data were obtained online for the Fort McMurray area (Government of Canada, 2022b). Effort was provided in the EMP dataset, along with sampling date.

2.2.8.4 Visualization of Trends

Temporal and spatial variability in fish indices of community composition was explored and illustrated using various graphing techniques including scatterplots and boxplots. Fish community indices were normalized to a common river flow (if Q60 was deemed a significant predictor) prior to graphing to improve the interpretation of potential temporal and spatial variations by accounting for variations due to natural conditions and habitat. Data were normalized to a discharge of 600 m^3/s.

Data normalization was done in 4 steps. First, the relationship between each individual fish community index and flow were determine using a GLM. Second, the coefficients (i.e., slope and intercept) from the GLM were used to calculate the predicted index value, given the observed flow (Q60). Third, residuals were computed as the difference between observed (i.e., measured) concentrations and predicted concentrations. Finally, fish community indices were normalized to flow as follows:

$$fish\ comm\ index_{obs} = \text{slope} * \log(Q60_{obs}) + intercept$$

$$residual = observed - predicted$$

$$normalized\ index = (slope * \log(600) + residuals) + intercept$$

2.2.9 Sentinel Fish Populations Health

Sentinel fish population health are a main component of EEM programs and will be an important biological receptor of potential effects of discharged treated OSPW. Fish health endpoints in EEM include those which reflect survival (age), energy use (growth, gonad size) and energy storage (condition, liver size). The data collected for the EMP were similarly used to compute endpoints which reflect energy use (gonadosomatic index, or GSI) and energy storage (condition and liver somatic index, or LSI). No survival endpoint was included in our analysis, as discussed below.

The analysis of fish population data involved the following steps:

1. Calculation of fish health indicators;
2. Filter for mature fish based on outlier GSI values (i.e., GSI > 1%) as per Environment Canada (2012a)
3. Data Summary;
4. Explanatory Models; and,
5. Visualization of Trends.

Each step is described below.

2.2.9.1 Indicators of Fish Health

The objective of the analysis here was to quantify sources of variability in fish population variables for Trout-perch collected under the EMP. The typical analysis of fish population data involves use of analysis of variance (ANOVA) to quantify and test for the significance of variations among locations and times in mean age, growth, gonad size, condition and liver size (Environment Canada, 2012a). In an analysis of

Trout-perch data completed for Kilgour et al (2019a) demonstrated that GSI, condition factor (K) and LSI related strongly to size normalized equivalents. That is, size (body weight) normalized gonad size correlated strongly with GSI, size (fork length) normalized body weight correlated strongly with K, and size (body weight) normalized liver size correlated strongly with LSI. GSI, K, and LSI were therefore examined herein for Trout-perch and Common White Sucker data from the EMP. These variables were the focus, because they tend to be the variables that are used to influence sample sizes.

The conventional indicators of K, GSI, and LSI are straightforward to calculate and easy to communicate as they are known to most fisheries practitioners. These indicators reflect energy use (relative gonad size) and energy storage (condition, relative liver size). There are no conventional indicators for survival or growth (size at age) in the fisheries literature analogous to GSI, K or LSI. Mean age is an endpoint in Canadian EEM programs that is meant to reflect survival by comparing the relative ages of exposed and reference fish populations. There are often, however, problems associated with the assignment of ages in fish population studies, such as inconsistencies associated age determination and size-selective gear (Munkittrick et al., 2010), especially with small-bodied fish. As such, the survival endpoint was not investigated as part of this present study.

Fish population indicators were calculated as follows.

Condition (K) was calculated using the formula:

$$K = \frac{1000 * total\ weight}{fork\ length^3}$$

where total weight is measured in grams (g) and fork length (FL) in millimeters (mm).

Gonadosomatic Index (GSI) was calculated using the formula:

$$GSI = \frac{100 * gonad\ weight}{total\ weight}$$

Liver Somatic Index (LSI) was calculated using the formula:

$$LSI = \frac{100 * liver\ weight}{total\ weight}$$

2.2.9.2 Data Summary

Summary statistics of fish health indicators were computed by station and year for females and males separately. Variability of observations (station x year) in relation to potential environmental covariables were explored using a variety of tools including Spearman Rank correlation analysis, scatterplots and PCA.

2.2.9.3 Explanatory Models

GLMs were computed for females and males separately, using observations of GSI, K and LSI from the EMP dataset, along with flow volume, year of sampling, and the distance from the potential OSPW discharge as predictors. Models had the following structure:

$$Y = constant + Q_{60} + Year + Dist_{OSPW}$$

where, Y is a fish index (GSI, K or LSI), Q_{60} is the $\log_{10}$ of flow volume averaged over the 60 days prior to sampling, Year is the year of sampling treated as a continuous covariable, and $Dist_{OSPW}$ is the distance from the proposed future OSPW discharge point (with stations upstream assigned a negative distance, stations downstream assigned a positive distance).

Discharge was included in the models as it has been demonstrated that this natural environmental factor impacts the variability in fish population performances in the OSR (Hatfield Consultants et al., 2016; Kilgour et al., 2019a). Flow values were assigned to each observation (i.e., station x year combination) in the EMP dataset. Historical hydrometric data were obtained online for the Fort McMurray area (Government of Canada, 2022b). Air temperature, a surrogate for water temperature, has also been shown to impact fish population performance variability (Kilgour et al., 2017a). However, with only three years of data, the covariable was dropped in the modelling process.

2.2.9.4 Visualization of Trends

Temporal and spatial variability in fish indicators was explored and illustrated using various graphing techniques including scatterplots and boxplots. Fish health indicators were normalized to a common river flow prior to graphing to improve the interpretation of potential temporal and spatial variations by accounting for variations due to natural conditions. Data were normalized to a discharge of 900 m^3/s, which was the average of the 3 years of sampling for the EMP, as described in Section 2.2.3.4.

2.2.10 Fish Body and Tissue Burdens

The purpose of the analysis of fish body and tissue burden data was to quantify sources of variability. The analysis of fish body and tissue burden data involved the following steps:

1. Data Processing;
2. Data Summary;
3. Explanatory Models; and
4. Visualization of Trends.

Each step is described below. The focus of the analysis was on total PAHs, total mercury and methylmercury, total selenium, liver Ethoxyresorufin-*O*-deethylase (EROD) activity, along with stable isotope ratios (SIR; $\delta^{15}N$ and $\delta^{13}C$). These are important indicators of biological health and/or have useful benchmarks and guidelines to which the data could be compared. Pearson’s correlation matrix was used to investigate analytes concentrations that may covary with liver EROD activity.

2.2.10.1 Data Processing

2.2.10.1.1 General Data Cleaning

Four fish body and tissue burden datasets were provided from AEPA, one for each parameter group (i.e., mercury, SIRs, metals, and PAHs), in “wide format”. The same data cleaning steps as outlined for the sediments in section 2.2.5.1 were also carried out for the fish body burden datasets.

After the above QA/QC steps were carried out, the following general steps were undertaken:

- Individual parameter names were summarized from each dataset and compared to ensure continuity, differences in bracket use, i.e. [] vs. (), primarily for individual PAC analytes, were corrected;
- Specifically for PAHs, any deuterated surrogate data were removed from the datasets, as we assumed all data had already been previously QA/QC'd;
- Station number's and long names were summarized for each dataset to confirm overlap, any errors between datasets in station number/name were confirmed with AEPA staff and corrected;
- Any QA/QC samples such as trip blanks, field blanks, or lab blanks were removed from the datasets to ensure only environmental data were to be considered in further analysis.

2.2.10.1.2 Dealing with Detection Limits

The percentage of samples below DLs were summarized at the station x year level for each individual analyte. Analytes that were below DL in over 80% of occurrences were removed from future analysis.

2.2.10.1.3 Data Amalgamation

After each EMP fish body burden dataset (i.e., 2018, 2019, and 2021) was summarized following the steps above, we ensured all column names matched. The datasets were provided in "long format", with a parameter column and a concentration column. All three datasets were combined using the bind_row() function available in the *dplyr* (Wickham et al., 2015) R package (R Studio ver. 4.1.3).

2.2.10.2 Data Summary

The number of samples available for each species-sex combination were summarized for each main analyte, along with the number of non-detects, summary statistics (min, max, mean, SD), and guideline exceedances.

2.2.10.3 Explanatory Models

GLMs were used to explore potential sources of variation in measured tissue and body burden concentrations. Model development for the analytes of focus were sex dependent and had the following structures.

The general GLM structure was:

$$Y = constant + Q_{60} + FL + Year + Dist_{OSPW} + Q_{60}xYear$$

where, Y is the $\log_{10}$ of the analyte concentration, Q_{60} is the $\log_{10}$ of flow volume averaged over the 60 days prior to sampling, FL is $\log_{10}$ of the fish fork length in mm, Year is the year of sampling treated as a continuous covariable, and $Dist_{OSPW}$ is the distance from the proposed future OSPW discharge point (with stations upstream assigned a negative distance, stations downstream assigned a positive distance).

Discharge was included in the models as it has been demonstrated that this natural environmental factor impacts the variability in fish population performances in the OSR (Hatfield Consultants et al., 2016; (Kilgour et al., 2019a). Flow values were assigned to each observation (i.e., station x year combination) in the EMP dataset. Historical hydrometric data were obtained online for the Fort McMurray area (Government of Canada, 2022b).

Year was included as a covariable because there has been some evidence that some water quality variables in the Athabasca River are increasing over time (Arciszewski, Munkittrick, Scrimgeour, et al., 2017a).

Walleye and White Sucker GLM structure was similar to Trout-perch model above but did not include the Year variable as data were only collected in 2019 and 2021.

2.2.10.4 Visualization of Trends

PCA plots were used as an initial step to visualize similarities and differences in the chemical profiles of fish body and tissues collected across the EMP sampling years and stations. Concentrations were normalized to a common fork length prior to graphing to improve the interpretation of potential temporal and spatial variations. Trout-perch data were normalized to a fork length of 60 mm and both Walleye and White Sucker were normalized to a fork length of 450 mm. Temporal and spatial variability in body burden variables, with a focus on Trout-perch, were explored and illustrate using various graphing techniques including scatterplots and boxplots.

2.2.11 Benthic Invertebrate Body Burdens

The purpose of the analysis of benthic body burden data was to quantify sources of variability. The analysis of benthic body burden data involved the following steps:

1. Data Processing;
2. Data Summary;
3. Explanatory Models; and
4. Visualization of Trends.

Each step is described below. A similar approach was taken as with the fish body burden data, with a focus on total PAHs, total mercury and methylmercury, total selenium, and SIRs (i.e., $\delta^{15}N$ and $\delta^{13}C$).

2.2.11.1 Data Processing

2.2.11.1.1 General Data Cleaning

Three benthic body burden datasets were provided from AEPA, one for each parameter group (i.e., mercury, SIRs, and metals), in “wide format”. Data for four separate benthic families were provided; Ametropodidae, Chironomidae, Gomphidae, and Pteronarcydiae, however the Chironomid data only included isotope concentrations from 2018, and did not include the expanded list of compounds and sampling years, therefore the data were omitted from further analysis. The same issues with the datasets as was outlined for the sediments in section 2.2.5.1 and fish body burdens in section 2.2.10.1 was also documented for the benthic body burden datasets and therefore the same data cleaning and reorganization steps were followed.

After the above QA/QC steps were carried out, the following general steps were undertaken:

- Station number's and long names were summarized for each dataset to confirm overlap, any errors between datasets in station number/name were confirmed with AEPA staff and corrected;

- Any QA/QC samples such as trip blanks, field blanks, or lab blanks were removed from the datasets to ensure only environmental data were to be considered in further analysis.

2.2.11.1.2 Dealing with Detection Limits
The percentage of samples below DLs were summarized at the station x year level for each individual analyte. Analytes that were below DL in over 80% of occurrences were removed from future analysis.

2.2.11.1.3 Data Amalgamation
After each EMP benthic body burden dataset (i.e., 2018, 2019, and 2021) were summarized following the steps above, we ensured all column names matched. The datasets were provided in "long format", with a parameter column and a concentration column. All three datasets were combined using the bind_row() function available in the *dplyr* (Wickham et al., 2015) R package (R Studio ver. 4.1.3).

2.2.11.2 Data Summary

The number of samples available for each benthic LPL were summarized for each main analyte, along with the number of non-detects, and summary statistics (min, max, mean, SD).

2.2.11.3 Explanatory Models

GLMs were used to explore potential sources of variation in measured benthic body burden concentrations. Model development for the analytes of focus had the following GLM structure:

$$Y = constant + Year + Dist_{OSPW}$$

where, Y is the $\log_{10}$ of the analyte concentration, Year is the year of sampling treated as a continuous covariable, and $Dist_{OSPW}$ is the distance from the proposed future OSPW discharge point (with stations upstream assigned a negative distance, stations downstream assigned a positive distance). River discharge (Q60) was not included in the model, primarily because the benthic body burden dataset provided did not facilitate merging with the historical discharge dataset.

2.2.11.4 Visualization of Trends

PCA plots were used as an initial step to visualize similarities and differences in the chemical profiles of benthic body burdens collected across the EMP sampling years and stations. Temporal and spatial variability were explored and illustrated using various graphing techniques including scatterplots and boxplots.

2.3 Results

2.3.1 Historical Flow in the Athabasca River

Flow in the Athabasca River is a principal driver of the physical, chemical, and biological environment, and as such can aid with the interpretation of potential spatial and temporal trends in environmental data. Flow data for the Athabasca River (station 07DA001) were compiled, plotted (Figure 5) and summary statistics calculated for the three EMP sampling years (Table 4).

The individual hydrographs for 2018, 2019, and 2021 are superimposed, along with historical values for the period of 1957 to 2017 represented by grey lines (i.e., minimum, mean, and maximum flows in ascending order). Specific sampling events for the EMP are also represented on the graph (circles); there were a total of 3 sampling events in 2018 (August, September, and October), 6 sampling events in 2019 (May, June, July, August, September, and October), and 5 sampling events in 2021 (June, July, August, September, and October). The annual hydrograph demonstrates the expected pattern of rapid increase in flow volumes in the spring/summer months, followed by a decrease in the fall, and a plateau in the winter. Flow volumes during the EMP generally fall within the historic minima and maxima ranges.

Flow volumes varied between sampling years of the EMP. For example, average flow volumes at the time of the July sampling events were 2041 m^3/s in 2019 and 983 m^3/s in 2021 (a range of ~ 2x, Figure 5). This stresses the need to include flow volume as a normalization method when comparing physical and chemical parameters between sampling years.

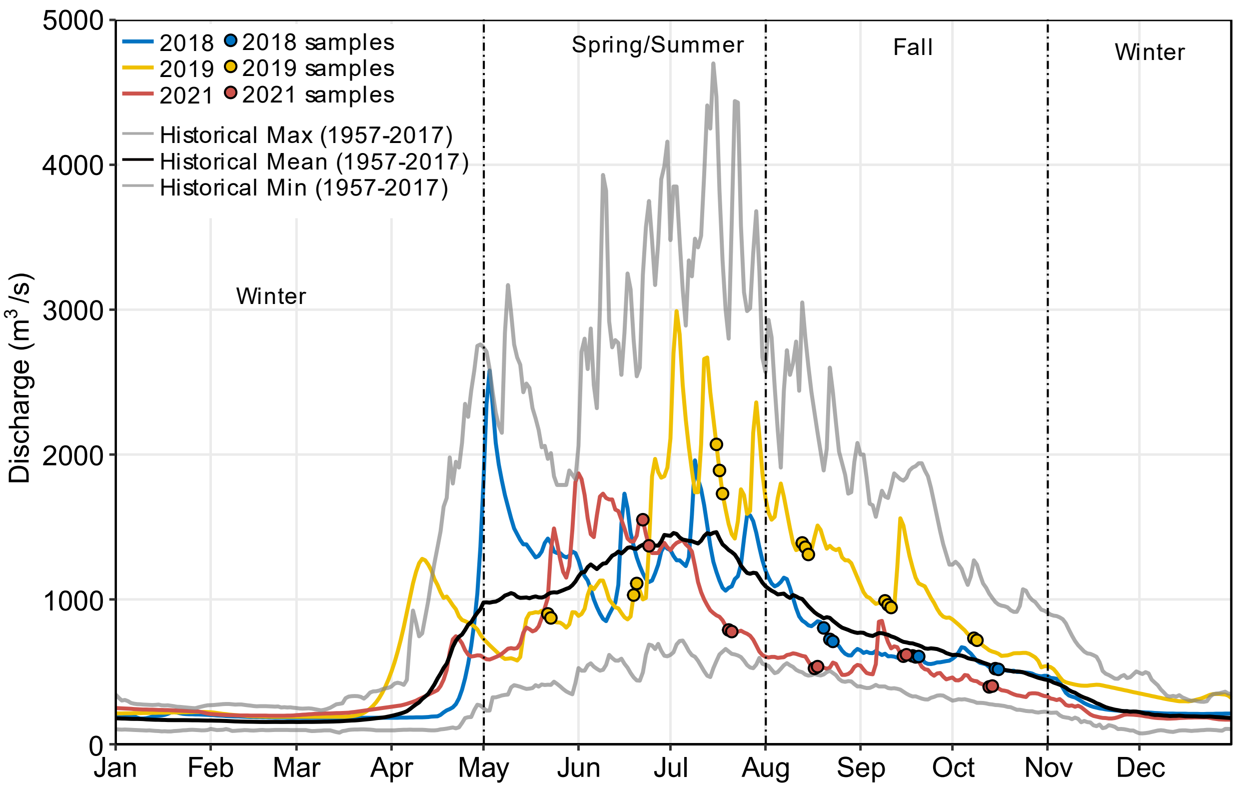

Figure 5 Water discharge (Q, m^3/s) from the water survey station (Fort McMurray 07DA001) along with historical values. Instantaneous sampling timepoints throughout each year (circles) are shown.

Table 4 Summary statistics for water flowing (m^3/s) through the water survey station 07DA001 in Fort McMurray during the EMP (2018, 2019 and 2021).

Month	2018 Flow (m^3/s)		2019 Flow (m^3/s)		2021 Flow (m^3/s)	
	Mean	SD	Mean	SD	Mean	SD
May	1556.1	356.2	769.6	129.7	907.2	336.4
June	1185.8	222.0	1186.8	369.3	1530.0	158.5
July	1362.9	232.0	2041.3	434.0	982.6	267.1
August	870.7	192.2	1426.5	160.8	563.9	46.9
September	599.8	27.9	1070.0	177.4	578.1	109.6
October	541.5	60.1	669.9	86.9	390.2	47.7

2.3.2 Water Quality Variables

2.3.2.1 Data Summary

The EMP surface water quality dataset includes 218 samples collected on 162 sampling events in 2018, 2019 and 2021. In 2018, there were a total of 36 sampling events across all 12 stations at which 6 discrete samples, 6 triplicate samples, 30 vertically integrated samples, and 6 blank samples (3 trip 3 field blanks) were collected (Table 5). In 2019, there were a total of 72 sampling events across all 12 stations at which 12 discrete samples, 12 triplicate samples, 60 vertically integrated samples were collected, and 12 blank samples (6 trip 6 field blanks). Finally, in 2021, there were a total of 54 sampling events across all 12 stations at which 1 discrete sample, 10 triplicate samples, 53 vertically integrated samples and 10 blank samples (5 trip 5 field blanks) were collected. A summary of VMV codes, method detection limits, and non-detects can be found in Appendix A Table A1.

There were many surface water variables for which there were a large proportion of non-detects, and these were not considered for further analysis. However, it remains important to document the occurrence of non-detectable concentrations to establish existing conditions in the LAR. Occurrences of non-detects in 2018, 2019 and 2021 were summarized for eight major groups of analytes: carbon, petroleum hydrocarbons, PAHs, major ions, minor element, NAs, nutrients, and phenols (Figure 6). Approximately 40, 41, and 35% of carbon analytes (i.e., dissolved organic/inorganic, total organic, and carbonate) were below detection limits in 2018, 2019, and 2021, respectively. Approximately 100, 84, and 91% of hydrocarbon fractions (i.e., F1, F2, F3, and F4) were below detection limits in 2018, 2019, and 2021, respectively. Since this parameter category was dominated by non-detections, it was not considered in further analysis. Approximately 16, 13, and 21% of PAHs were below detection limits in 2018, 2019, and 2021, respectively. Approximately 14, 14, and 17% of major ions below detection limits in 2018, 2019, and 2021, respectively. Approximately 22, 22, and 27% of minor elements, including metals, were below detection limits in 2018, 2019, and 2021, respectively. Approximately 28, 38, and 61% of naphthenic acids, were below detection limits in 2018, 2019, and 2021, respectively, here we observe a clear increasing trend of non-detection with sampling year (Figure 6). Approximately 43, 31, and 40% of nutrients (i.e., ammonia, nitrogen, nitrate, nitrite, and phosphorous) were below detection limits in 2018, 2019, and 2021, respectively. Finally, 63, 85, and 38% of phenols were below detection limits in 2018, 2019, and

2021, respectively.

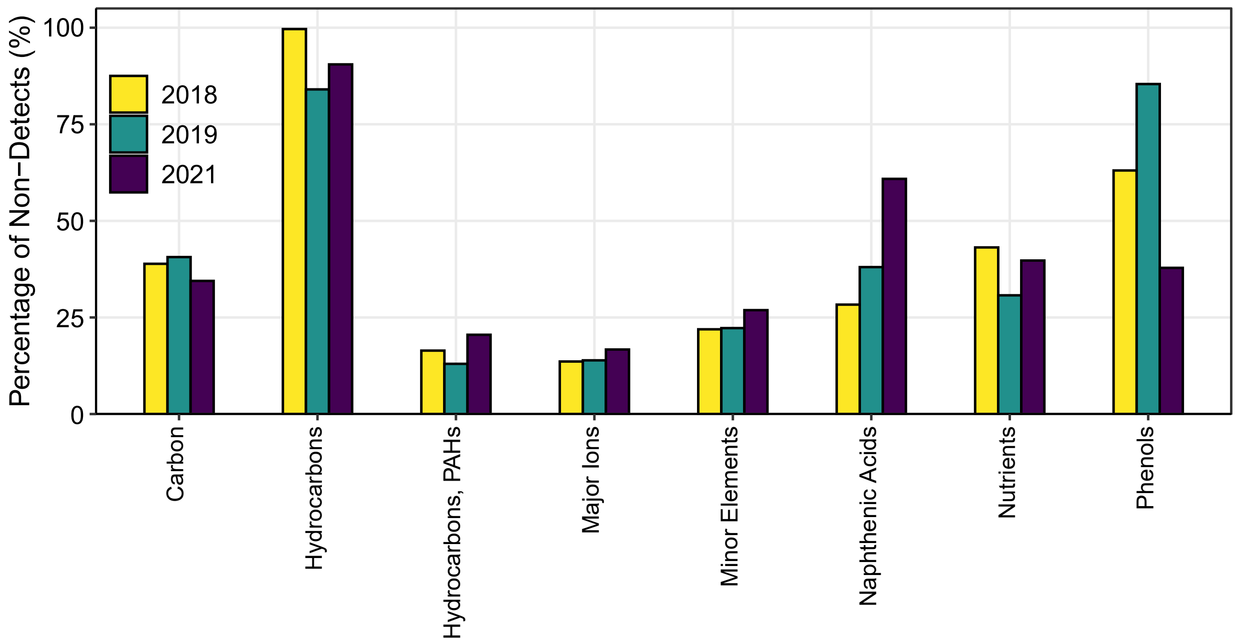

Figure 6 Percentage of non-detects observed in surface water samples by parameter category in the EMP dataset (2018, 2019, and 2021).

Table 5 Lower Athabasca River (LAR) sampling events for surface water over the Enhanced Monitoring Program period (2018, 2019, and 2021).

Station Name	2018						2019						2021						Total					
	Sampling Events	Discrete Samples	Triplicate Samples	Vertical Integrated	Trip Blank	Field Blank	Sampling Events	Discrete Samples	Triplicate Samples	Vertical Integrated	Trip Blank	Field Blank	Sampling Events	Discrete Samples	Triplicate Samples	Vertical Integrated	Trip Blank	Field Blank	Sampling Events	Discrete Samples	Triplicate Samples	Vertical Integrated	Trip Blank	Field Blank
12 km DS -Th	3	0	0	3	0	0	6	0	2	6	1	1	5	0	2	5	1	1	14	0	4	14	2	2
4.5 km DS -Th	3	0	0	3	0	0	6	0	0	6	0	0	5	0	2	5	1	1	14	0	2	14	1	1
1.5 km DS -Th	3	0	0	3	0	0	6	0	2	6	1	1	5	0	0	5	0	0	14	0	2	14	1	1
1.5 km DS -WI	3	3	0	0	0	0	6	6	0	0	0	0	2	0	0	2	0	0	11	9	0	2	0	0
0.5 km DS -EI	3	0	0	3	0	0	6	0	0	6	0	0	5	0	0	5	0	0	14	0	0	14	0	0
0.5 km DS -WI	3	3	0	0	0	0	6	6	0	0	0	0	2	1	0	1	0	0	11	10	0	1	0	0
0.03 km DS -RB	3	0	2	3	1	1	6	0	0	6	0	0	5	0	0	5	0	0	14	0	2	14	1	1
0.03 km DS -LB	3	0	2	3	1	1	6	0	4	6	2	2	5	0	4	5	2	2	14	0	10	14	5	5
0.03 km US -LB	3	0	0	3	0	0	6	0	0	6	0	0	5	0	0	5	0	0	14	0	0	14	0	0
0.5 km US -LB	3	0	0	3	0	0	6	0	2	6	1	1	5	0	2	5	1	1	14	0	4	14	2	2
4.0 km US -Th	3	0	2	3	1	1	6	0	2	6	1	1	5	0	0	5	0	0	14	0	4	14	2	2
12 km US -Th	3	0	0	3	0	0	6	0	0	6	0	0	5	0	0	5	0	0	14	0	0	14	0	0
Totals	**36**	**6**	**6**	**30**	**3**	**3**	**72**	**12**	**12**	**60**	**6**	**6**	**54**	**1**	**10**	**53**	**5**	**5**	**162**	**19**	**28**	**143**	**14**	**14**

Table Notes: Sampling approaches and locations differed and are indicated as follows: Th – thalweg, E – east of island, W – west of island, Right – rightbank, and Left – leftbank.

2.3.2.1.1 Guideline Exceedances

The current EMP surface water quality dataset includes concentrations for over 200 individual parameters (e.g., nutrients, metals, major ions, minor elements, petroleum hydrocarbons, PAHs, naphthenic acids, phenols). Individual parameter concentrations were compared to a variety of different water quality guideline's (WQGs) including both CCME WQGs for the protection of aquatic life and Alberta's provincial WQGs. Occurrences of WQG exceedances in 2018, 2019 and 2021 were summarized for the 7 major groups for which WQGs exist: petroleum hydrocarbons, PAHs, major ions, minor elements, NAs, nutrients, and routine observations (i.e., pH) (Figure 7).

Exceedances of the CCME long-term WQGs were only observed for metals and nutrients (Figure 7A). Approximately 25% of samples exceeded the CCME long -term WQG for metals in all sampling years of the EMP (i.e., copper, iron, nickel, and lead; Figure 7). Finally, approximately 17, 33, and 20% of samples exceeded the CCME long -term WQG for nutrients in 2018, 2019, and 2021, respectively (i.e., ammonia and nitrite; Figure 7). No parameters exceeded CCME's Short-Term WQGs at any point throughout the EMP monitoring period.

Exceedances of Alberta's provincial WQGs were observed for minor elements and nutrients (Figure 7B). Approximately 8, 17, and 8% of samples exceeded Alberta's provincial WQG for minor elements in 2018, 2019, and 2021, respectively (i.e., copper, lead, and zinc; Figure 7). Finally, approximately 17, 33, and 20% of samples exceeded Alberta's provincial WQG for nutrients in 2018, 2019, and 2021, respectively (i.e., nitrogen and phosphorus; Figure 7).

Guidelines for drinking water quality were also considered. Total organic carbon (TOC) and turbidity values typically exceeded the British Columbia Source Drinking Water Quality guidelines of 4 mg/L and 1 NTU. TOC in surface water quality samples ranged between 0.5 mg/L and 16 mg/L, with 83% of samples > 4 mg/L. Turbidity in surface water quality samples ranged between 0.1 and 200 NTU, with 88% of samples > 1 NTU. Manganese concentrations in surface water samples ranged between 0.04 and 347 µg/L, exceeding the Canadian Drinking Water Quality guideline of 120 µg/L in 68% of samples. Aluminum and lead concentrations in surface water quality samples exceeded the Canadian Drinking Water Quality guidelines of 2.9 mg/L and 5 µg/L in 6% and 4% of samples, respectively. However, It is important to note that Canadian Drinking Water Quality guidelines are applicable to detected concentrations in finished drinking water, not source waters. Drinking water treatment processes are designed (and mandated) to reduce contaminant load below guideline value concentrations to ensure safety of human consumption.

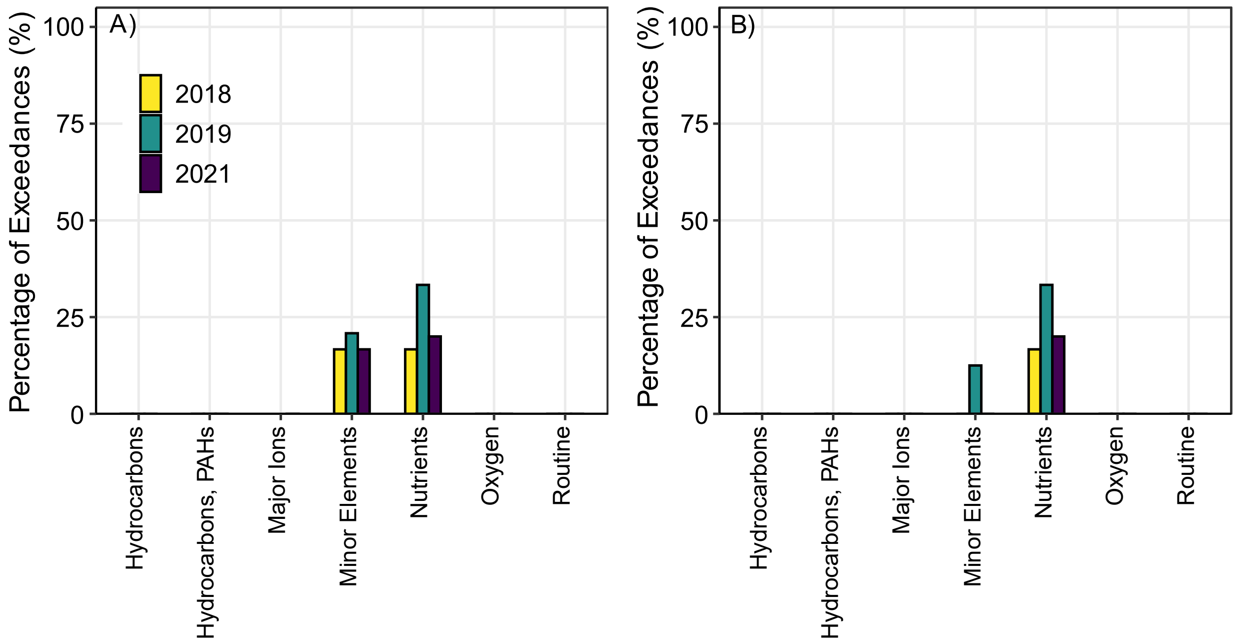

Figure 7 Percentage of observed exceedances of federal CCME (A) and provincial Alberta (B) water quality guideline (WQG), by analyte category in the EMP dataset (2018, 2019, and 2021).

Table 6 Summary of the number of Water Quality Guideline (WQG) exceedances by parameter over each EMP sampling year.

Group	Parameter	WQG	WQG Value	Units	Sampling Year		
					2018	2019	2021
Minor Elements	Copper (Total)	CCME Long-Term	Variable (2 to 3)	µg/L	2	44	8
	Iron (Total)		300	µg/L	37	84	57
	Nickel (Total)		Variable (2.5 to 4)	µg/L	3	35	8
	Lead (Total)		Variable (3 to 4)	µg/L	-	13	-
Nutrients	Ammonia (Total)		19	µg/L	48	96	74
	Nitrite as Nitrogen		60	µg/L	-	1	-
Minor Elements	Copper (Total)	Alberta	7	µg/L	-	14	-
	Lead (Total)		Variable (3 to 4)	µg/L	-	13	-
	Zinc (Total)		Variable (12.75 to 22.5)	µg/L	-	3	-
Nutrients	Nitrogen (Total)		1000	µg/L	-	2	-
	Phosphorus (Total)		50	µg/L	8	70	18

Table Notes: Repeated rows of the same compound represent WQG's that differ on a sample-by-sample basis according to the following equations:

CCME guideline for cadmium is related to hardness; 0.04 µg/L when hardness is < 17 mg/L, calculated using equation $CWQG_{cadmium}$ (µg/L) = 10{0.83(log[hardness])-2.46} at hardness ≥ 17 to ≤ 280 mg/L, and 0.37 µg/L when hardness is > 280 mg/L

CCME guideline for ammonia (as N) is dependent on water temperature and pH, here a guideline of 2.22 mg/L for 15°C and a pH of 7.5 were used as a more conservative value

CCME guideline for copper is related to hardness, 2 µg/L when hardness < 82 mg/L, calculated using equation $CWQG_{copper}$ (µg/L) = 0.2*{e(0.8545[ln(hardness)]-1.465} at hardness ≥ 82 to ≤ 180 mg/L, and 4 µg/L when hardness is > 180 mg/L

CCME guideline for lead is related to hardness, 1 µg/L for hardness ≤ 60 mg/L, calculated using equation $CWQG_{lead}$ (µg/L) = e{1.273[ln(hardness)]-4.705} at hardness > 60 to ≤ 180 mg/L, and 7 µg/L for hardness > 180 mg/L.

CCME guideline for nickel is related to hardness, 25 µg/L when hardness is ≤ 60 mg/L, calculated using equation $CWQG_{nickel}$ (µg/L) = e{0.76[ln(hardness)]+1.06} when hardness is > 60 and ≤ 180 mg/L, and 150 µg/L when hardness is > 180 mg/L

Alberta guidelines for zinc is related to hardness; when hardness < 90 mg/L, $CWQG_{zinc}$ (µg/L) = 7.5, when hardness > 90 mg/L, $CWQG_{zinc}$ (µg/L) = 7.5 + 0.75(hardness-90)

2.3.2.1.2 Naphthenic Acids

Naphthenic acids (NAs) are a chemical group of concern in OSPW because of the uncertainties associated with biological risks (Kilgour et al., 2018; Scott et al., 2020). Concentrations of total NAs in 2018 ranged in value from 1.7 to 51.1 µg/L, which concentrations in 2019 ranged in value from 1.4 to 204 µg/L (Figure 9). In 2021, most samples had non-detectable concentrations of total NAs suggesting values of < 4 µg/L, but with reported values of up to 23.9 µg/L. The major difference in total NAs concentrations from year to year is highlighted in the proportion of collected samples that were below method detection limits (s). In 2018, 100% of collected samples produced concentrations that were > MDL (all 42 samples), in 2019 95% of the collected samples were > MDL, and finally in 2021 only 4% of samples were > MDL. The low frequency of NAs detected in 2021 (when compared to 2018 and 2019) suggests a methodological variation.

Overall, concentrations of NAs measured using orbitrap were very low. Concentrations, when re-expressed as toxic unit equivalents (i.e., relative to LC50s for Rainbow Trout and Fathead Minnow) were well below 1 (in fact they were below 0.05 for both species of fish; Figure 8) indicating negligible likelihood of lethality to fish. NAs toxicity to aquatic organisms typically occurs when concentrations are well into the mg/L range (Kilgour et al., 2018; Scott et al., 2020), in contrast to the concentrations reported here which were in the µg/L range. As such, NAs concentrations in the LAR pose negligible risks of harm to fish (Hughes et al., 2017b) or other (non-fish) aquatic organisms.

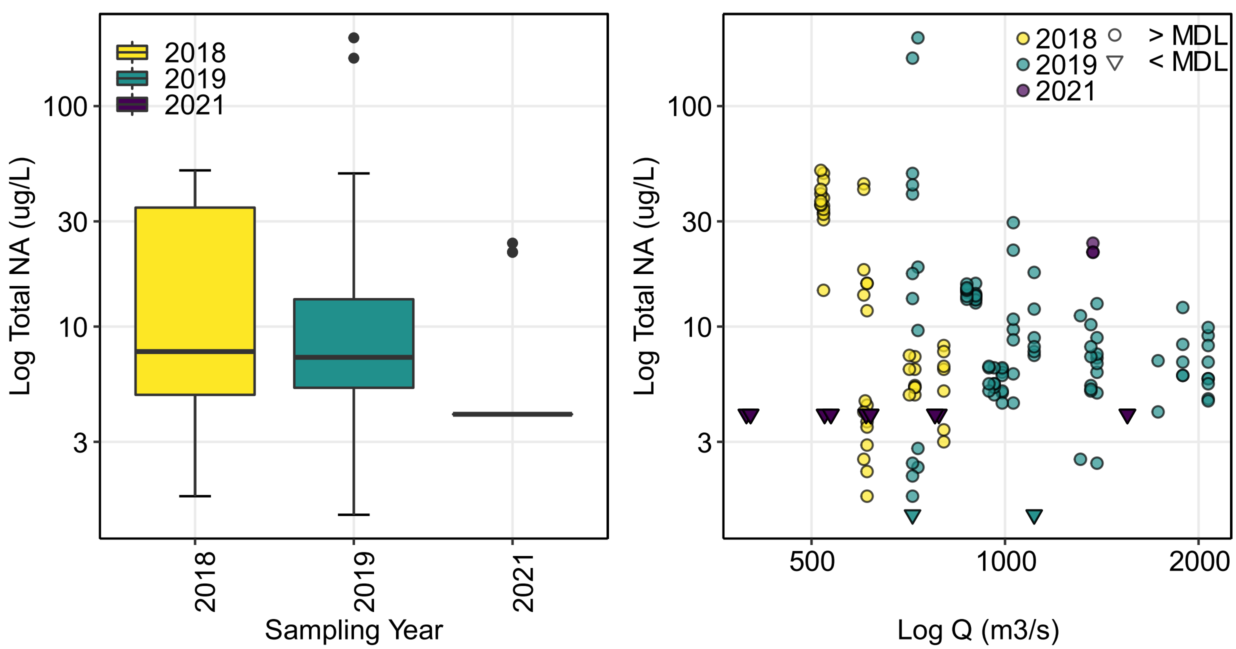

Figure 8 Relationship between log transformed total NAs concentration across EMP sampling years (A) and log transformed discharge (B).

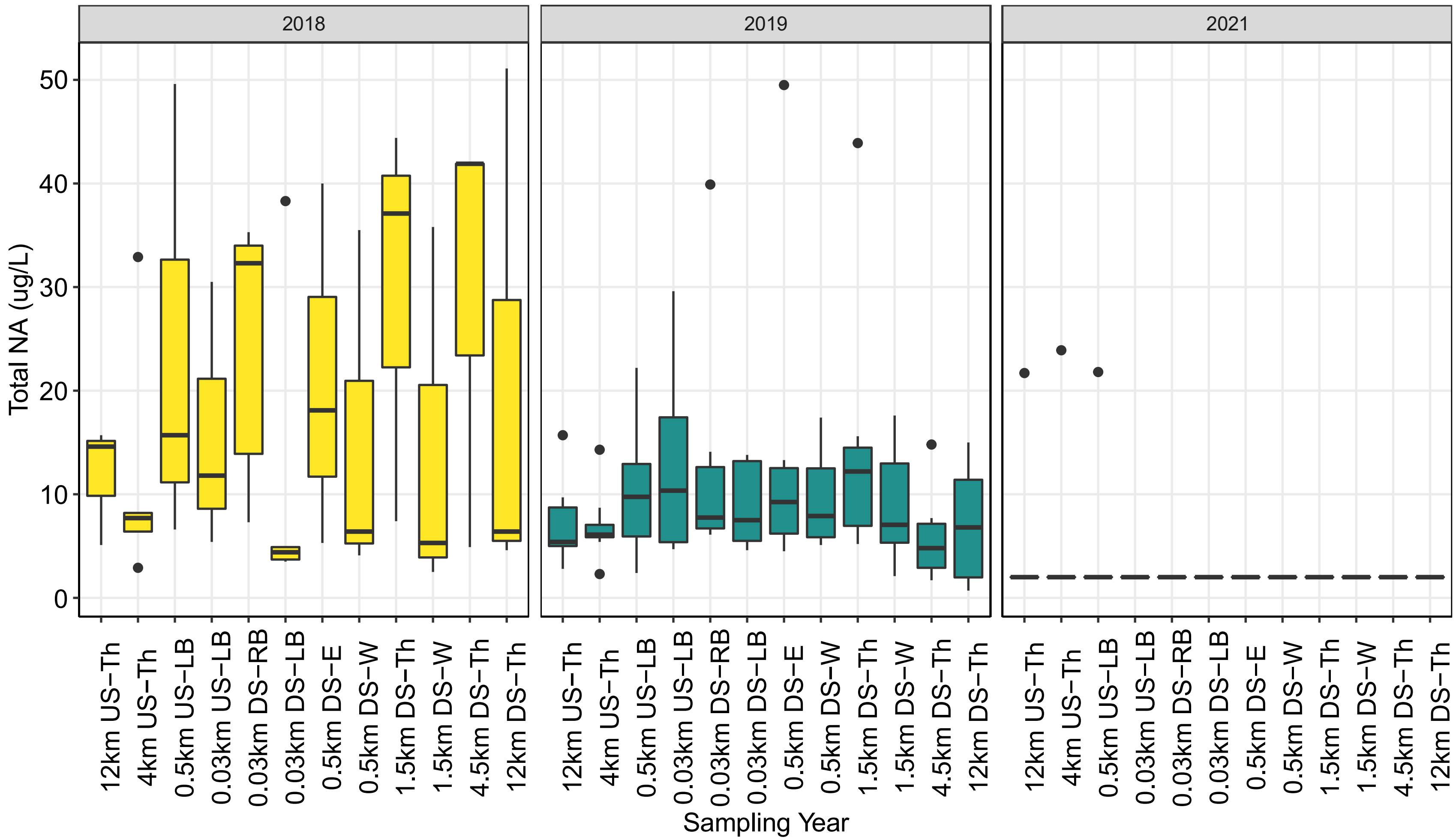

Figure 9 Concentrations of total Naphthenic Acids (NAs) in samples collected from the various EMP stations in 2018, 2019 and 2021.

Figure Notes: Values < MDL were replaced by MDL/2 for the purpose of plotting

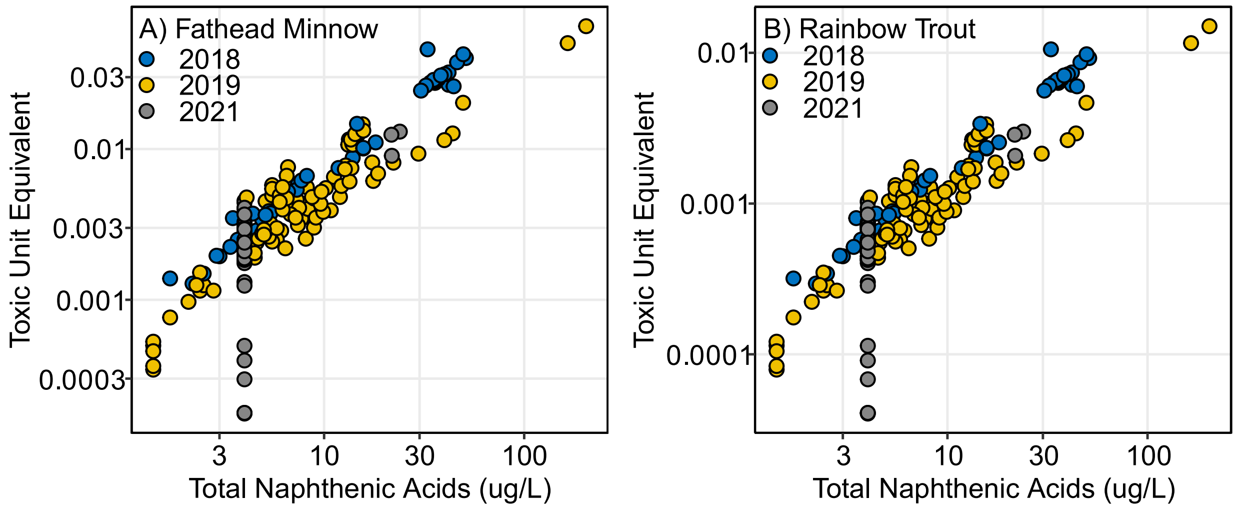

Figure 10 Relationship between predicted Toxic Unit Equivalent and Total Naphthenic Acid concentration in (A) Fathead Minnow (*Pimephales promelas*) and (B) Rainbow Trout (*Oncorhynchus mykiss*).

2.3.2.1.3 Cytotoxicity WQI

Based on the perturbation of cellular growth of human hepatocarcinoma cells, the cytotoxicity WQI (C-WQI) is a measure of potential human health risk, with lower values representing lower cytotoxicity and higher values representing higher cytotoxicity. A C-WQI less than 1.0 is considered non-cytotoxic, while a C-WQI greater than 1.0 may warrant further investigation or chemical testing. C-WQI values ranged from 0 to 4.1 across all sampling years and stations during the EMP (Table 7), which is consistent with the range of C-WQI values reported in a previous ACFT study evaluating water samples from the Athabasca River and its tributaries (Kinniburgh et al., 2021; Kinniburgh et al., 2023a; Kinniburgh et al., 2023b).

Figure 11 demonstrates the relationship between WQI and potential covariables discharge and turbidity. First, there is a clear significant relationship between turbidity and discharge, where turbidity increases with increasing discharge ($R^2 = 0.77$, $p < 0.001$, slope = 1.98; Figure 11A). Therefore, to avoid issues of multicollinearity, only one variable (i.e., discharge or turbidity) can be chosen as a covariable in future models. While WQI is positively correlated with both discharge (Figure 11B) and turbidity (Figure 11C), discharge was chosen to normalize the data to remain consistent with other sections of this report, where discharge is often included as a covariable. When interpreting the measured C-WQI values from the EMP, it is important that water quality stressors independent of discharge are not overlooked by relying on interpretation of normalized C-WQI values alone. As demonstrated previously, non-normalized C-WQI values can offer insight into temporal and spatial cytotoxicity trends for effects-based environmental water quality monitoring (Kinniburgh et al., 2021).

After standardizing the WQI values to a discharge of 900 m^3/s, Pearson's correlations among the cytotoxicity WQI and other water quality analytes were computed, and the most correlated parameters are provided in Table 8. Methyl acenaphthene yielded the highest Pearson's correlation coefficient (R = 0.64).

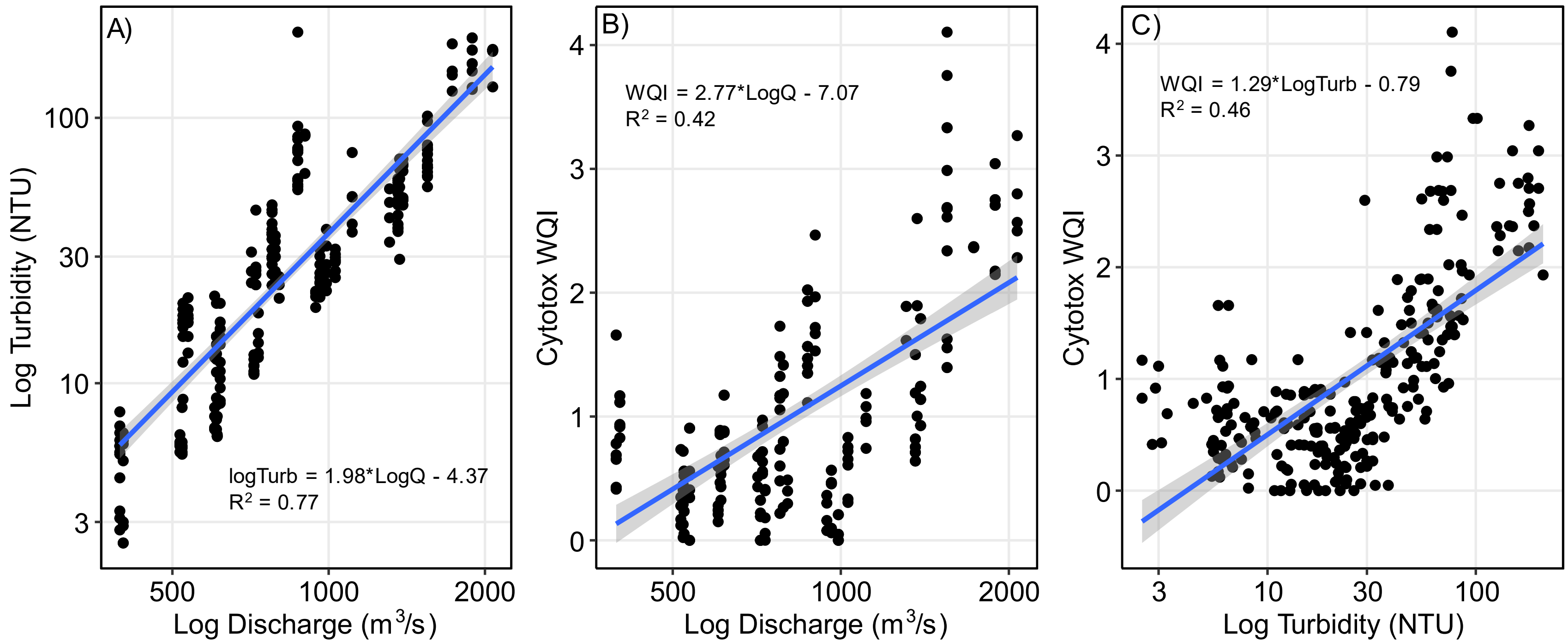

Figure 11 Relationship between turbidity and discharge (A), C-WQI and discharge (B), and C-WQI and turbidity (C) during the EMP (2018, 2019, and 2021). Solid blue line represents the regression line and the shaded gray area represents the 95% confidence interval around the regression line.

Table 7 Summary statistics of C-WQI for samples collected during the EMP program (2018, 2019 and 2021).

Year	Description	Mean	SD	Min	Max	N
2018	12km DS-Th	0.62	0.16	0.51	0.73	2
	4.5km DS-Th	0.40	0.24	0.24	0.68	3
	1.5km DS-Th	0.35	0.08	0.28	0.43	3
	1.5km DS-WI	0.45	0.46	0.12	0.97	3
	0.5km DS-EI	0.34	0.26	0.17	0.63	3
	0.5km DS-WI	0.70	0.19	0.58	0.92	3
	0.03km DS-RB	0.33	0.20	0.15	0.54	3
	0.03km DS-LB	0.71	0.02	0.68	0.72	3
	0.03km US-LB	0.51	0.06	0.45	0.56	3
	0.5km US-LB	0.28	0.14	0.13	0.40	3
	4km US-Th	0.42	0.14	0.30	0.57	3
	12km US-Th	0.31	0.25	0.02	0.49	3
2019	12km DS-Th	1.05	0.89	0.00	2.36	6
	4.5km DS-Th	1.23	0.91	0.00	2.37	6
	1.5km DS-Th	0.87	0.82	0.00	2.15	6
	1.5km DS-WI	1.20	0.84	0.22	2.17	6
	0.5km DS-EI	1.19	0.97	0.47	3.04	6
	0.5km DS-WI	1.16	0.97	0.19	2.71	6
	0.03km DS-RB	1.02	0.85	0.06	2.28	6
	0.03km DS-LB	1.45	1.16	0.41	3.27	6
	0.03km US-LB	1.20	1.05	0.06	2.57	6
	0.5km US-LB	1.00	0.95	0.00	2.50	6
	4km US-Th	0.93	1.00	0.05	2.75	6
	12km US-Th	1.00	1.06	0.00	2.80	6
2021	12km DS-Th	1.09	0.71	0.65	2.68	7
	4.5km DS-Th	0.97	0.86	0.00	2.69	7
	1.5km DS-Th	0.59	0.39	0.22	1.11	4
	1.5km DS-WI	1.85	1.38	0.60	3.33	3
	0.5km DS-EI	0.74	0.40	0.34	1.40	5
	0.5km DS-WI	0.93	0.77	0.07	1.56	3
	0.03km DS-RB	1.08	0.76	0.40	2.34	5
	0.03km DS-LB	1.78	1.22	0.52	4.10	9
	0.03km US-LB	1.57	1.33	0.23	3.75	5
	0.5km US-LB	1.13	0.67	0.56	2.60	7
	4km US-Th	0.70	0.53	0.05	1.42	5
	12km US-Th	0.72	0.67	0.27	1.89	5

Table Notes: Th = Thalweg, E = East of Island, W = West of Island, Right = Right Bank, and Left = Left Bank.

Table 8 Correlation between C-WQI and surface water quality parameter after normalizing to discharge. The top 5 most correlated parameters are shown.

Parameter	R	P-value	Lower CI	Upper CI
Methyl Acenaphthene	0.64	<0.001	0.43	0.78
Fluoranthene	0.49	<0.001	0.39	0.58
1,7-Dimethylfluorene	0.48	<0.001	0.33	0.61
7-Methylbenzo(a)Pyrene	0.48	<0.001	0.29	0.63
2-Methylfluorene	0.46	<0.001	0.33	0.57

Table Notes : CI = Confidence Interval

2.3.2.2 EXPLANATORY MODELS

Results from the GLMs for the 22 water quality variables of concern are summarized in Table 9. The models determined that daily discharge (Q) was a significant predictor (p -value < 0.05) of variation in concentrations of alkalinity, total Al, dissolved Ca, dissolved Cl, total Cu, total Fe, Total Pb, dissolved Mg, total Ni, total P, dissolved Na, dissolved SO_4, total Tl, total PAHs, total V, total Zn, nitrite + nitrate, and total nitrogen (TN). Daily discharge was also a significant predictor of variation in the Cytotoxicity WQI. Discharge typically explained >20% variation in analyte concentrations (% variance explained between 6 and 72%). An examination of the relationships between the analytes and daily discharge indicated that Al, Cu, Fe, Pb, Ni, P, Tl, PAC, V, Zn, NO_2+NO_3, TN, and C-WQI generally increased with river flow volume, while Ca, Cl, Mg, Na, and SO_4, generally decreased with river flow volume (Table 9; Figure 12). There was no apparent association between discharge and total Mo, total NAs, or NA Toxic Units. An example of each type of relationship is illustrated in Figure 12.

Total NA concentrations did not vary significantly with river discharge, but did vary significantly across years, and significantly (but modestly) with distance from shore. The low frequency of NAs detected in 2021 (when compared to 2018 and 2019) suggests laboratory inconsistencies which may have masked the effect of discharge. The detection limit for total NA did increase in 2021 to 4 µg/L (from 1 µg/L) possibly resulting in more non-detects.

Linear trends over time as well as variations in trends over time that depended on discharge (i.e., the interaction term between Q x Year) were statistically significant for all analytes, except for dissolved Cl and total Mo. There was no statistically significant variation associated with the distance from the proposed OSPW. There were, however, statistically significant variations associated with the distance from shore for alkalinity, total Al, dissolved Ca, dissolved Cl, total Cu, total Fe, dissolved Mg, total Mo, total Ni, dissolved Na, dissolved SO_4, and nitrite + nitrate.

Temporal and spatial differences were further investigated below in Section 2.3.2.2 on flow normalized water quality variables. Flow normalizing equations are provided in Appendix C Table C1.

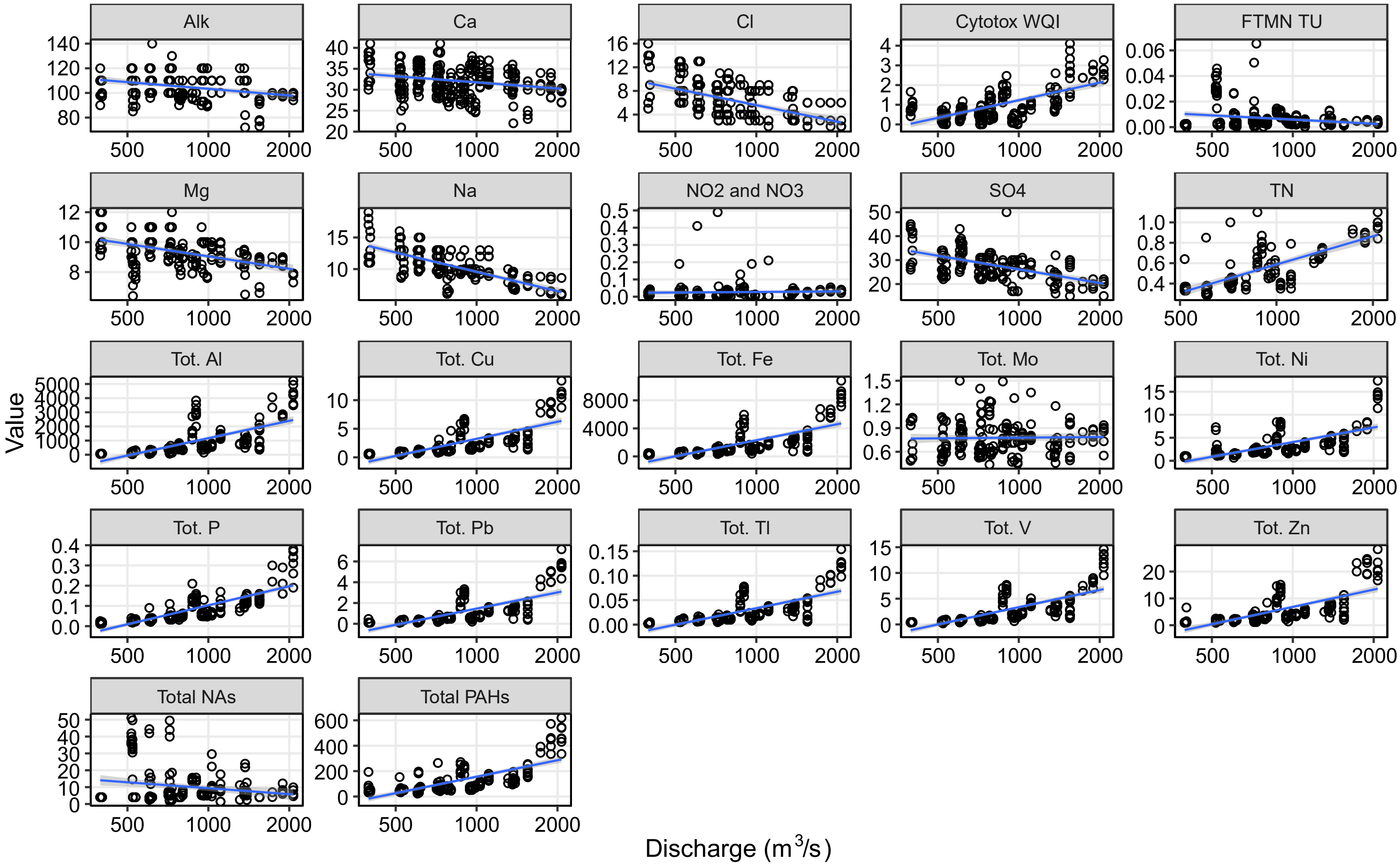

Figure 12 Relationship between river discharge and parameter concentrations showing in the EMP dataset (2018, 2019, and 2021).

Table 9 Significance (p-value) and percent of variance explained (%VE) for predictors of concentrations of analytes in surface waters collected in the Lower Athabasca River, EMP (2018, 2019, 2021)..

Analytes	Discharge		Year		Distance (US/DS)		Shoreline Distance		Q x Year	
	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE
Alkalinity Total $CaCO_3$	**<0.001**	9.9	**<0.001**	20.17	0.449	0.19	**<0.001**	8.95	**0.027**	1.63
Aluminum Total Recoverable	**<0.001**	70.0	0.057	0.56	0.603	0.04	**0.358**	0.13	**<0.001**	1.93
Calcium Dissolved	**<0.001**	6.1	**<0.001**	3.41	0.298	0.27	**<0.001**	14.24	**0.012**	1.55
Chloride Dissolved	**<0.001**	36.1	0.773	0.02	0.068	0.83	**<0.001**	18.61	0.415	0.16
Copper Total Recoverable	**<0.001**	72.2	**0.008**	0.86	0.792	0.01	**0.001**	1.26	**<0.001**	4.43
Iron Total Recoverable	**<0.001**	69.1	0.112	0.37	0.519	0.06	**0.728**	0.02	**<0.001**	4.52
Lead Total Recoverable	**<0.001**	71.8	0.586	0.04	0.679	0.02	0.256	0.17	**<0.001**	4.91
Magnesium Dissolved	**<0.001**	21.2	**<0.001**	5.25	0.300	0.38	**<0.001**	8.71	0.345	0.32
Molybdenum Total Recoverable	0.480	0.2	0.284	0.53	0.330	0.44	**<0.001**	15.59	0.812	0.03
Total Naphthenic Acids	0.195	0.6	**<0.001**	24.62	0.362	0.32	0.666	0.07	**<0.001**	6.36
Nickel Total Recoverable	**<0.001**	62.5	**<0.001**	3.72	0.165	0.34	**0.025**	0.90	0.017	1.01
Phosphorus Total	**<0.001**	83.5	0.435	0.05	0.860	0.00	0.203	0.14	**0.011**	0.58
Sodium Dissolved/Filtered	**<0.001**	52.3	**<0.001**	6.43	0.080	0.55	**<0.001**	8.69	0.870	0.00
Sulphate Dissolved	**<0.001**	33.7	**0.046**	1.129	0.803	0.02	**<0.001**	13.27	**0.024**	1.45
Thallium Total Recoverable	**<0.001**	71.3	**0.006**	1.10	0.711	0.02	0.165	0.27	**<0.001**	2.00
Total PAHs	**<0.001**	59.9	**0.001**	2.33	0.374	0.15	0.235	0.28	**<0.001**	2.43
Vanadium Total Recoverable	**<0.001**	73.9	0.746	0.01	0.799	0.01	0.441	0.08	**<0.001**	2.97
Zinc Total Recoverable	**<0.001**	67.1	0.119	0.39	0.815	0.01	0.073	0.51	**<0.001**	3.49
NAs Toxic Unit	0.370	0.2	**<0.001**	20.10	0.148	0.65	0.561	0.10	**<0.001**	14.35
C-WQI	**<0.001**	14.5	**<0.001**	6.57	0.504	0.18	0.843	0.02	0.257	1.10
NO2 and NO3 as N	**<0.001**	6.07	**0.001**	5.53	0.099	1.30	**0.042**	1.98	0.457	0.26
Total Nitrogen	**<0.001**	56.29	**0.004**	2.92	0.572	0.11	0.301	0.36	0.087	0.99

Table Notes: Significant values (i.e., p-value < 0.05) are in bold
%VE represents the percentage of total variance explained by each predictor within the individual models
Shaded cells highlight the %VE that corresponds to significant p-values

2.3.2.3 Visualization of Trends

2.3.2.3.1 Temporal

The PCA of flow-normalized water quality variables revealed that the surface water quality was similar in 2018 and 2021, as indicated by the overlapping ellipses in (Figure 13A), whereas 2019 samples were associated with higher PCA axis 1 scores. The 2019 samples seem to be influenced heavily by total Ni, Cu, Tl, Fe, P, V, P, Zn, and to a lesser extent total PAHs. However, when the sampling months were overlaid on the PCA (Figure 13B), sampling month seems to be the principal driver of PCA axis 1 scores. As shown in Figure 5, water samples were only collected during the month of May in 2019. No water sampling occurred in May 2018 or 2021.

An example of the overall temporal trend and differences amongst sampling years can be found in Figure 14AB for total naphthenic acids and Figure 15AB for total PAHs. Temporal trends in total naphthenic acid concentrations indicated a decreasing trend over time, while trends in total PAC concentrations indicated an increasing trend over time in the LAR. As was previously discussed, the observed decrease of NAs over time was likely due to laboratory inconsistencies rather than a true temporal trend and therefore caution should be used when interpreting these results.

The cytotoxicity WQI values were generally more variable in 2019 compared to the other sampling years (Figure 17). Temporal increases from 2018 to 2021 was observed at some stations, such as the station located 1.5 km downstream, west of the island, where values incrementally increased from 2018 to 2019, and from 2019 to 2021 (Figure 17).

2.3.2.3.2 Spatial

The PCA plot (Figure 18) depicts sampling location within the channel overlaid with the compounds driving the separation. The most notable separation of ellipses is between left bank (i.e., west) and right bank (i.e., east), where left bank samples produce positive PC axis 2 scores and right bank samples produce negative PC axis 2 scores. These differences are driven primarily by elevated alkalinity, Mg, Ca, SO_4, and Mo in left bank samples and elevated concentrations of Cl and Na in right bank samples. These results are further supported by linear model results found in Table 9, where shoreline distance (a measurement of the samples distance from shore) was a significant predictor of alkalinity, Ca, Cl, Cu, Mg, Mo, Ni, Na, and SO_4. Examples where concentration are consistently higher on the left bank compared to the right bank can be found in Figure 19 for Ca, Mg, SO_4, Mo, and alkalinity. This cross-channel variation in analytes may suggest the consistent presence of an anthropogenic input along the west side of the LAR. Potential sources could include the Fort McMurray Wastewater Treatment Plant located on the west side of the LAR roughly 30km upstream of the proposed OSPW discharge point. Examples where analyte concentrations are consistently higher along the right bank of the LAR can be found in Figure 20 for Na and Cl, this cation and anion may be originating from the outflow of the Clearwater River, which flows into the LAR roughly 33km upstream of the proposed OSPW discharge point. These cross-channel differences have been reported previously in the regional OSMP (Glozier et al., 2018). Regardless of the source, further study would be required to isolate the source of these cross-channel variations in the EMP dataset.

Finally, no clear distinguishable patterns were observed when comparing sampling stations upstream and downstream of the potential OSPW release point (Figure 1) as the 95% confidence ellipses in Figure 21 were highly overlapped. Further, sampling station distance upstream and/or downstream of the proposed

OSPW discharge point was not a significant predictor for any of the surface water quality variables (Table 9). This exercise is an important step in developing baseline conditions, if the future release of treated OSPW does in fact cause a shift in surface water quality downstream of the discharge, it will likely become evident in repeat of this multivariate analysis.

Water quality samples were collected at various locations upstream and downstream of the proposed effluent release point, as well as at various locations across the channel. Samples were collected near the right bank, left bank and mid channel (thalweg) because it is well documented that the Clearwater River confluence with the Athabasca River (and associated mixing) causes horizontal variability in water quality. Here, examples of the spatial (horizontal or across the river) variability in water quality were observed for concentrations of total naphthenic acids (Figure 14D) and total PAHs (Figure 15D).

An important spatial comparison to consider is the difference between samples collected on the east and west side of the island located 0.5 km downstream of the proposed OSPW discharge point, as these stations are directly downstream of the Syncrude sewage treatment outfall (Figure 1). Differences among samples collected from the east and west side of the island were explored using a Tukey's post-hoc test for individual comparisons. At sampling stations located 0.5 km downstream of the proposed OSPW discharge, concentrations of molybdenum were significantly higher on the west side of the island compared to the east side in both 2018 and 2019 (p-value < 0.001 in both years; Table 10). Further, in 2019, concentrations of both nitrate and nitrate (as N) as well as total nitrogen were significantly higher on the west side of the island compared to the east side of the island ($p<0.001$ and $p = 0.02$, respectively; Table 10). While Figure 16D does show higher phosphorous levels on the west side of the island compared to the east side in both 2018 and 2019, the differences were surprisingly not significant ($p = 0.16$ and 0.117 in 2018 and 2019, respectively), likely due to the high degree of variation among samples. Note that this comparison did not include data from 2021 as the site on the west side of the island was dried up for most of the year due to low discharge. Considering the Syncrude sewage treatment outfall is located on the west side of the LAR, it is likely that these spatial differences across the width of the river at 0.5 km downstream are driven by sewage inputs.

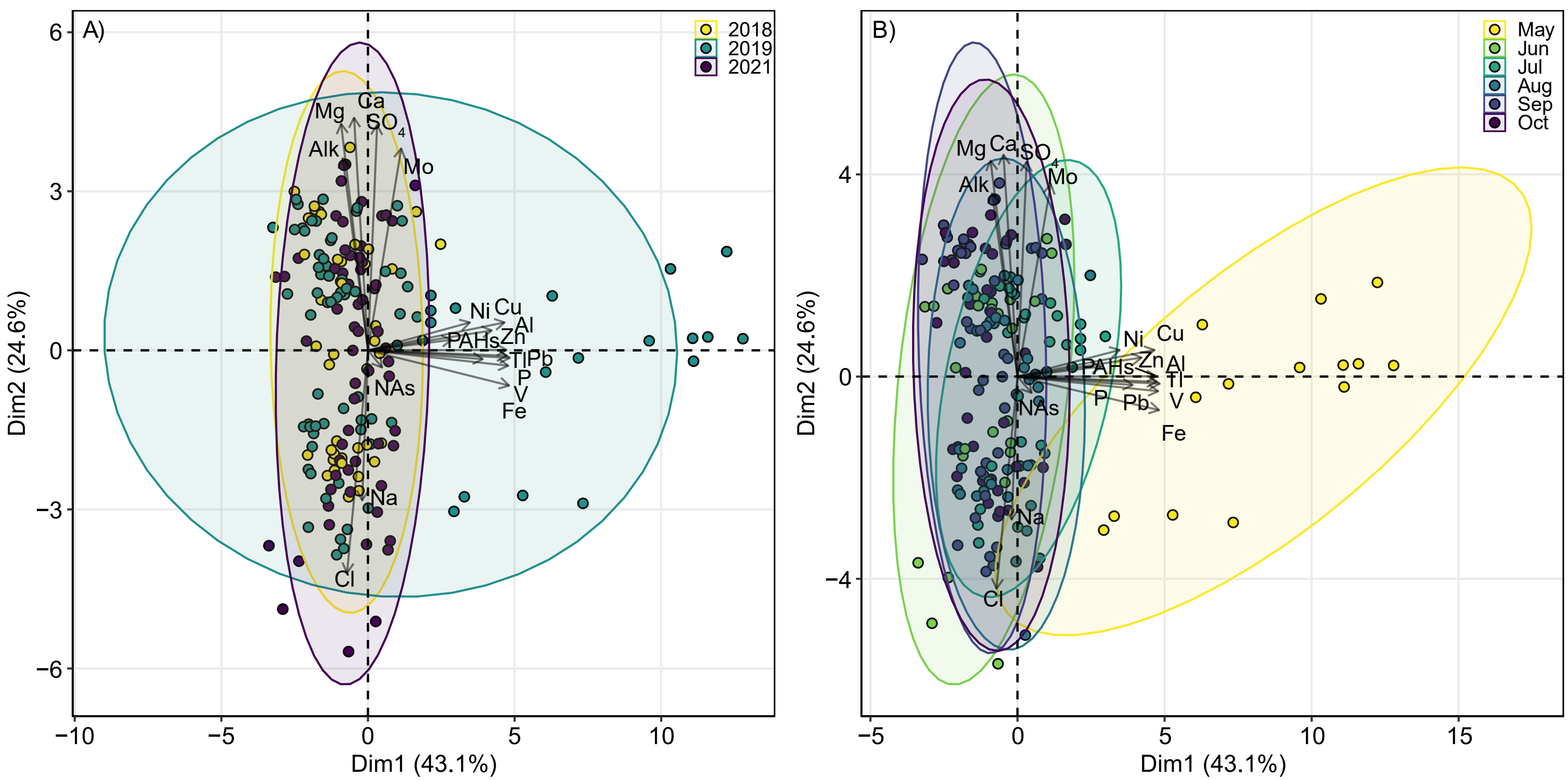

Figure 13 Principal Component Analysis depicting temporal (A) and seasonal (B) patterns in water quality based on a subset of 18 parameters (Alk, Al, Ca, Cu, Cl, Fe, Pb, Mg, Mo, NA, Ni, P, Na, SO_4, Tl, PAC, V, Zn). Parameter concentrations were standardized to a discharge of 900 m^3/s prior to evaluation. Ellipses represent 95% confidence intervals.

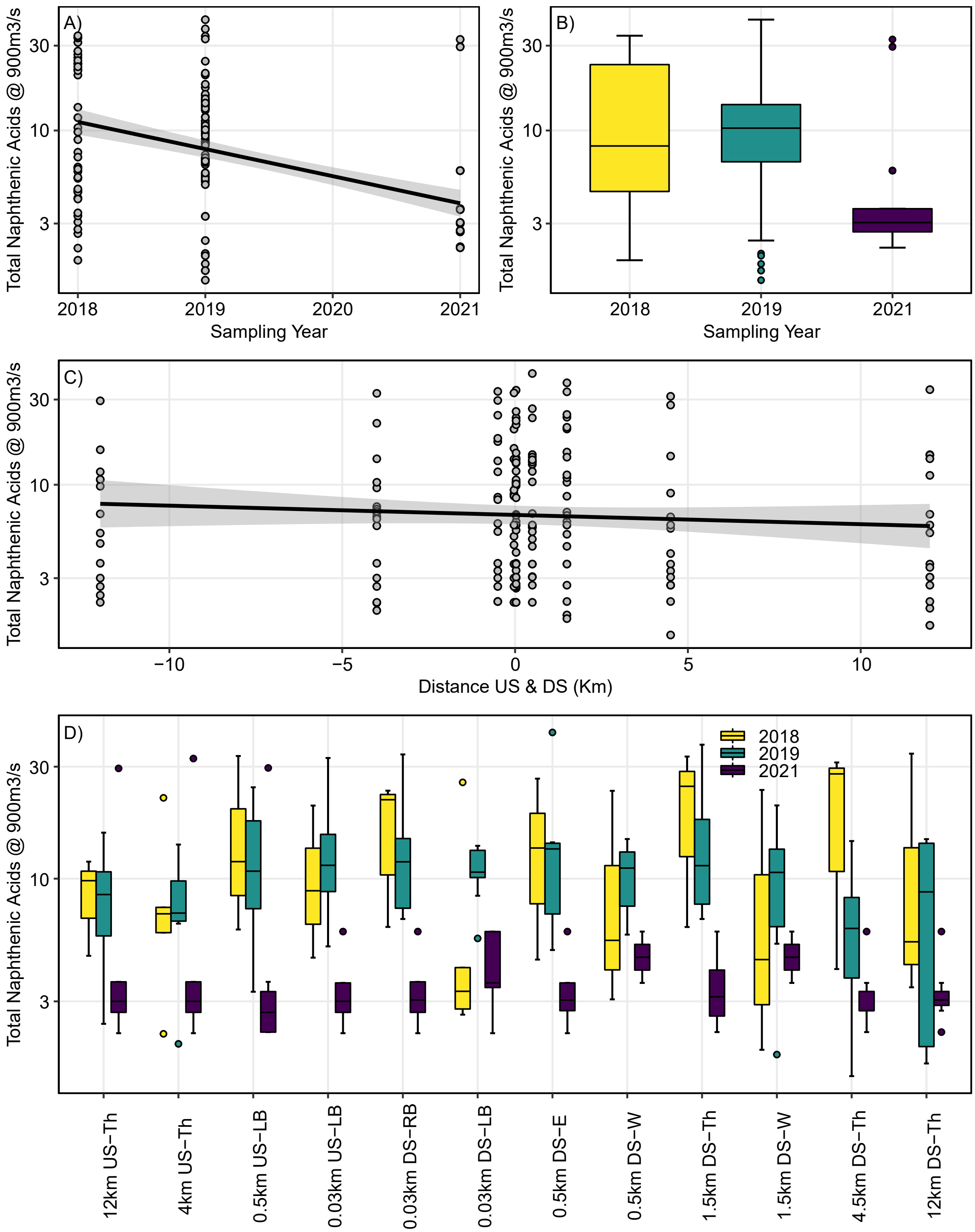

Figure 14 Range and variation in total naphthenic acids over time and across sampling stations. Concentrations standardized to a discharge of 900 m^3/s.

Figure Notes: Th = Thalweg; LB = Left Bank: RB = Right Bank, W = West of Island, and E = East of Island

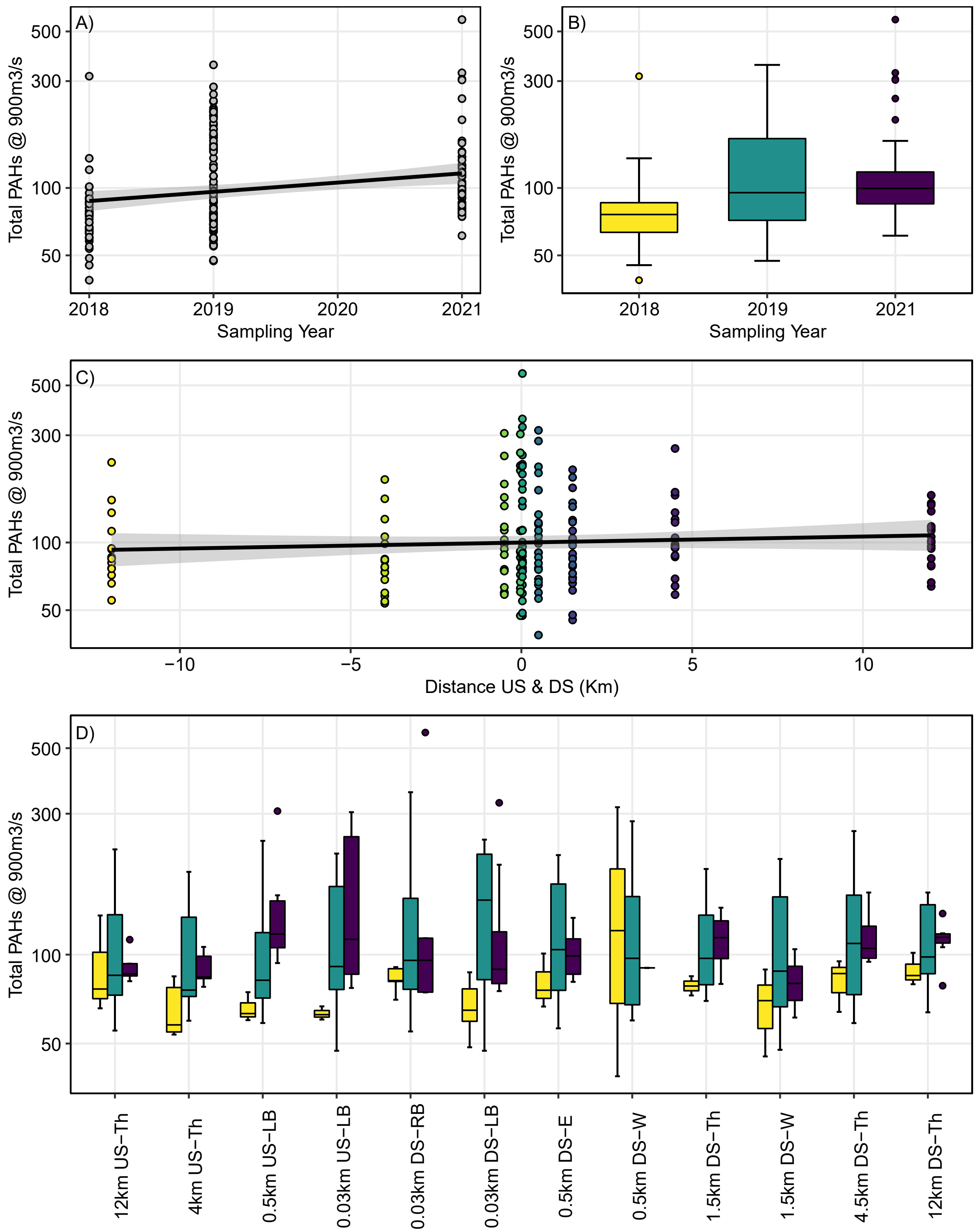

Figure 15 Range and variation in total PAHs over time and across sampling stations. Concentrations standardized to a discharge of 900 m³/s.

Figure Notes: Th = Thalweg; LB = Left Bank: RB = Right Bank, W = West of Island, and E = East of Island

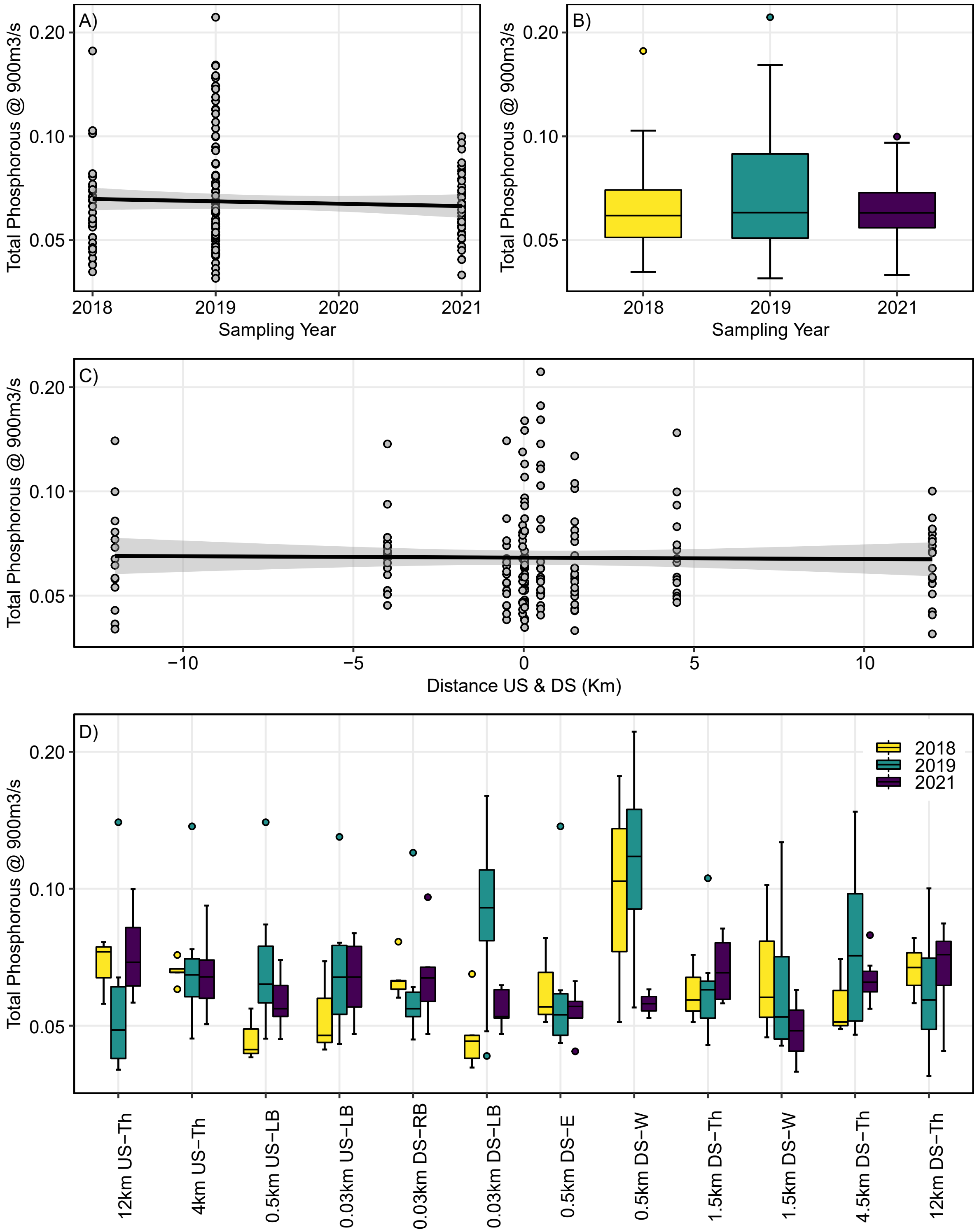

Figure 16 Range and variation in phosphorous over time and across sampling stations. Concentrations standardized to a discharge of 900 m³/s.

Figure Notes: Th = Thalweg; LB = Left Bank: RB = Right Bank, W = West of Island, and E = East of Island

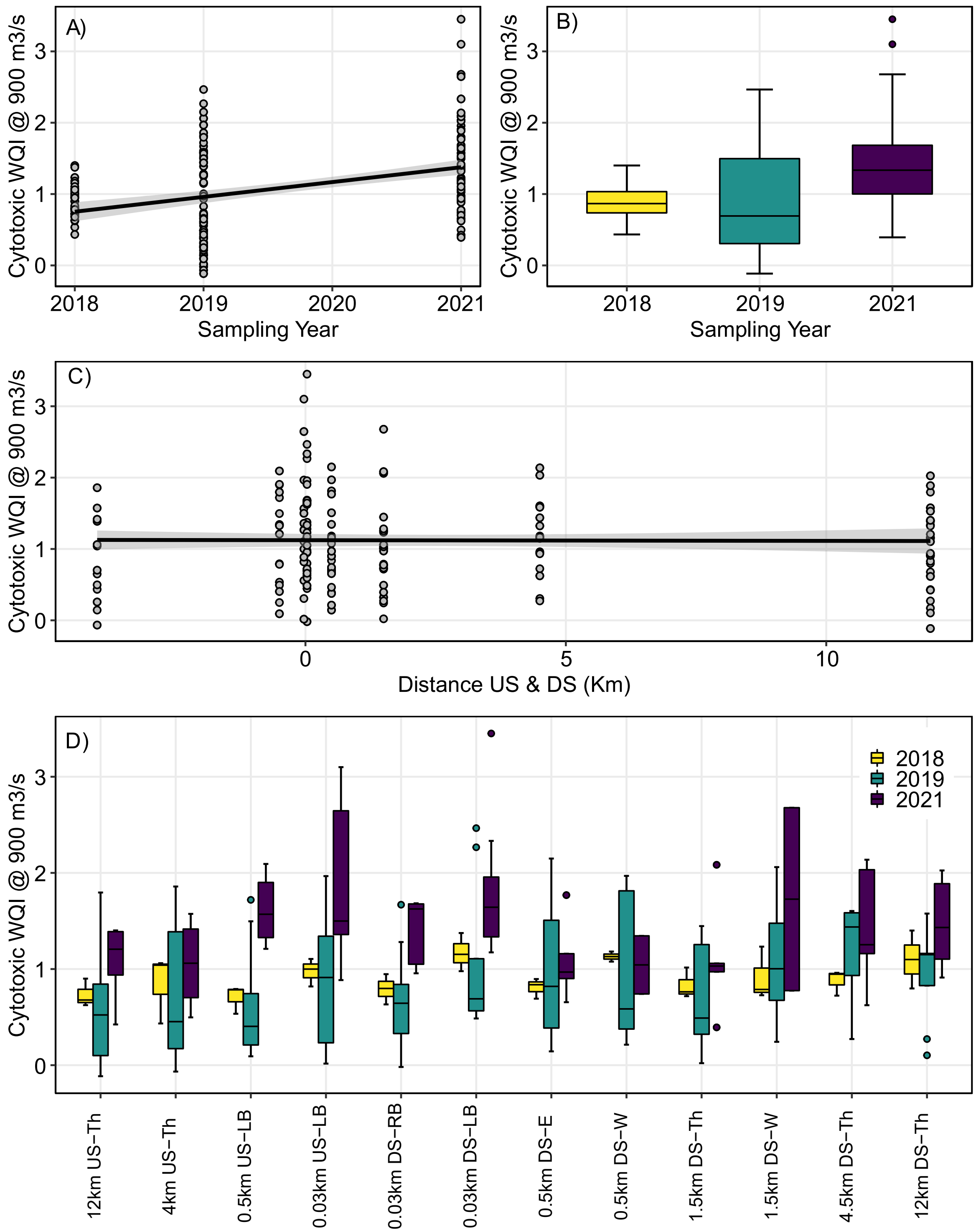

Figure 17 Range and variation in cytotoxic water quality indicator (WQI) over time and across sampling stations. Concentrations standardized to a discharge of 900 m^3/s.

Figure Notes: Th = Thalweg; LB = Left Bank: RB = Right Bank, W = West of Island, and E = East of Island

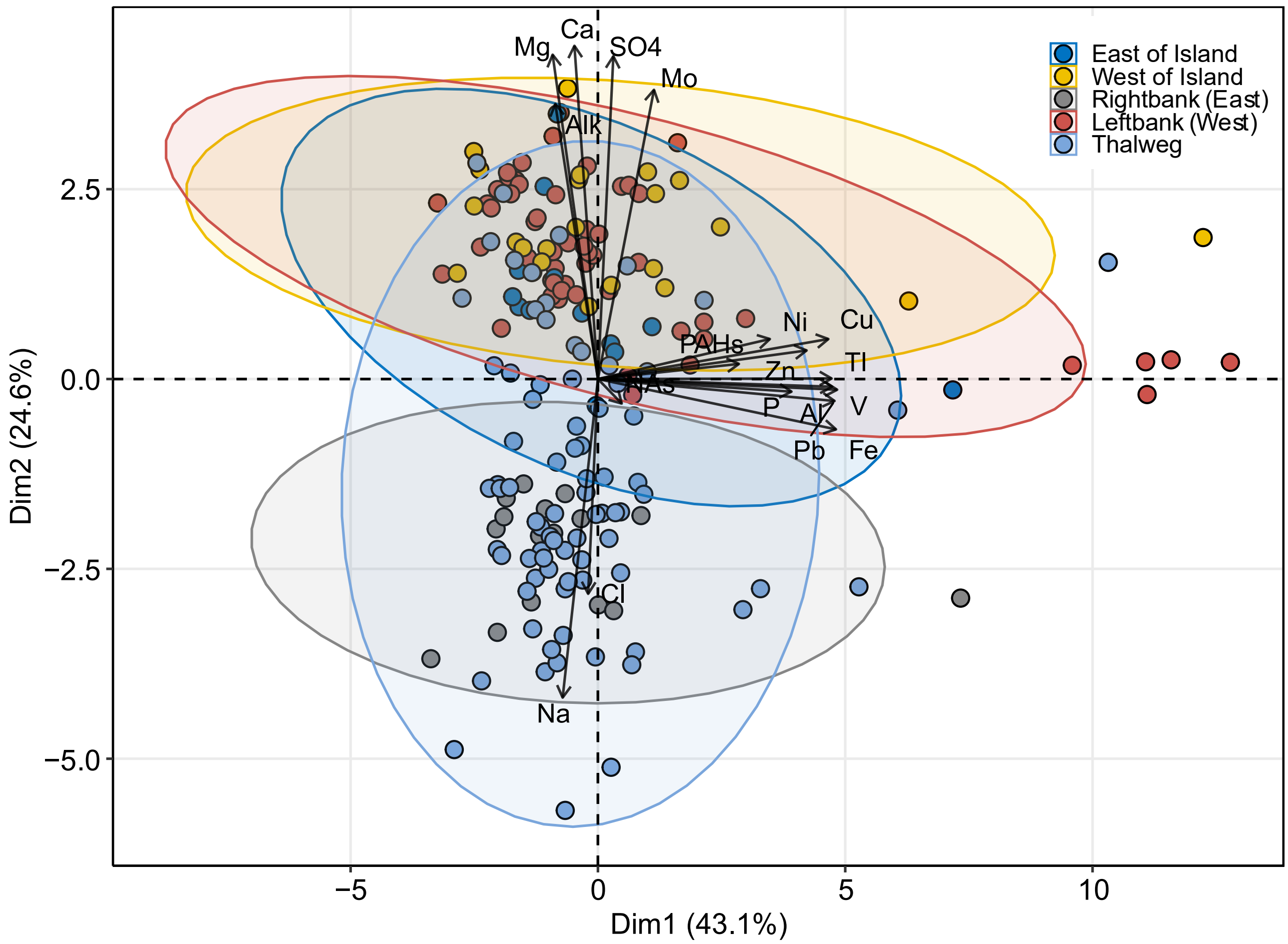

Figure 18 Principal Component Analysis depicting spatial patterns across the LAR in water quality based on a subset of 18 parameters (Alk, Al, Ca, Cu, Cl, Fe, Pb, Mg, Mo, NA, Ni, P, Na, SO_4, Tl, PAH, V, Zn). Parameter concentrations were standardized to a discharge of 900 m^3/s prior to evaluation. The ellipses represent 95% confidence intervals.

Figure Notes: Th = Thalweg; LB = Left Bank; RB = Right Bank; W = West of Island; E = East of Island

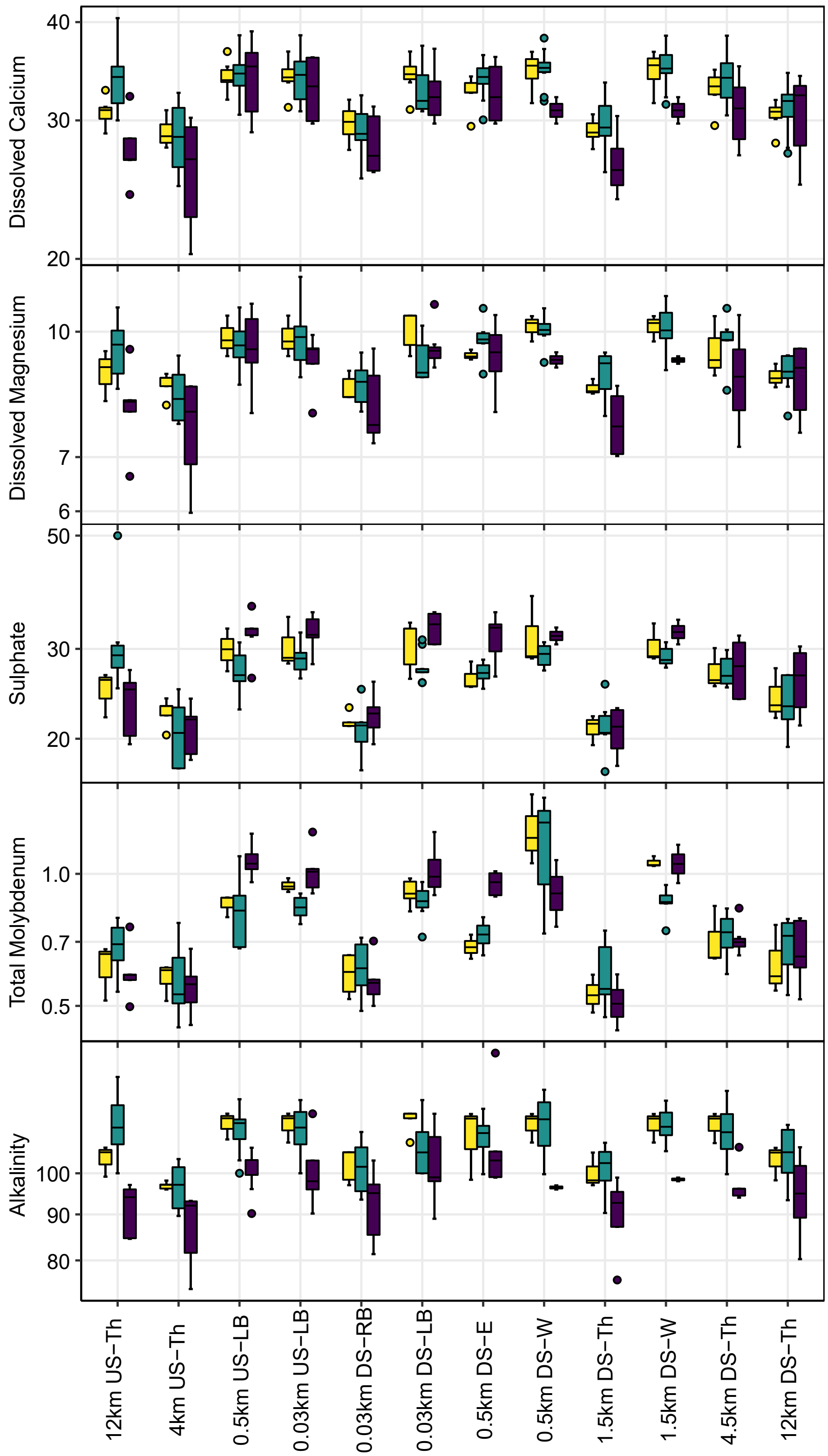

Figure 19 Range and variation in Ca, Mg, SO_4, Mo, and Alkalinity standardized to a discharge of 900 m^3/s from EMP sampling stations.

Figure Notes: Th = Thalweg; LB = Left Bank; RB – Right Bank; W = West of Island; E = East of Island

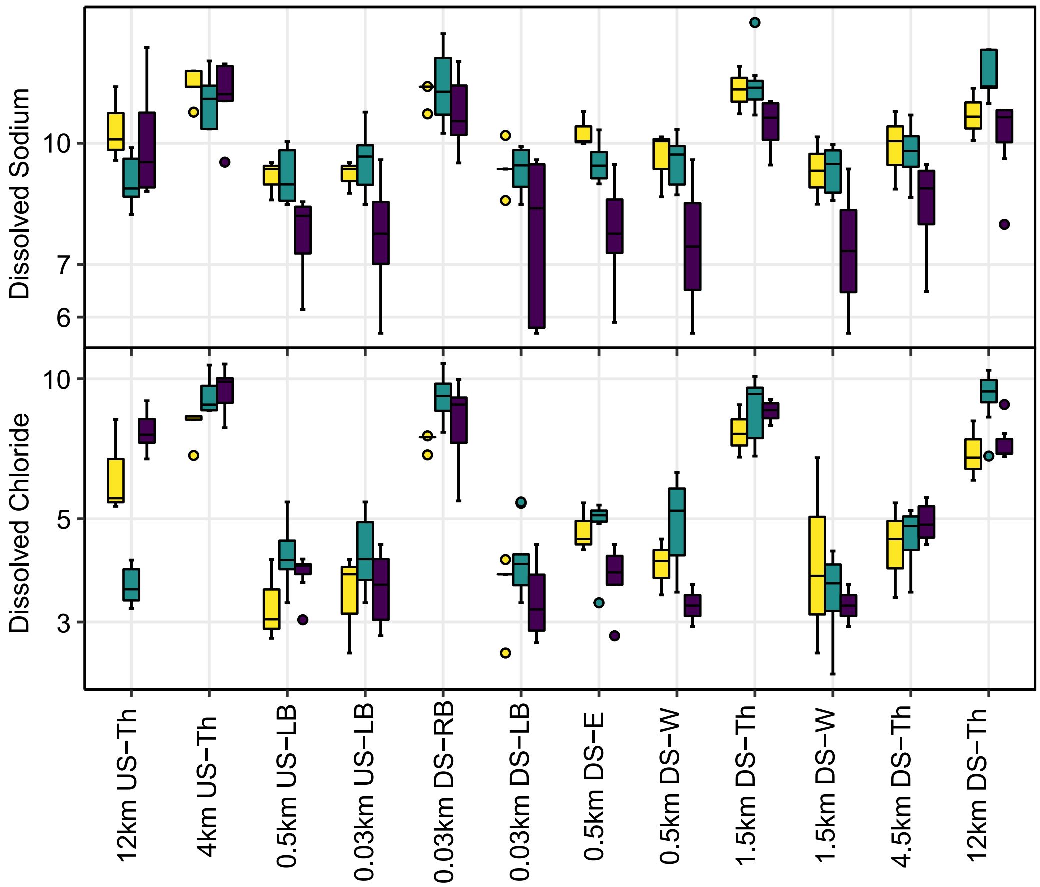

Figure 20 Range and variation in Na and Cl[-] standardized to a discharge of 900 m^3/s from EMP sampling stations.

Figure Notes: Th = Thalweg; LB = Left Bank; RB – Right Bank; W = West of Island; E = East of Island

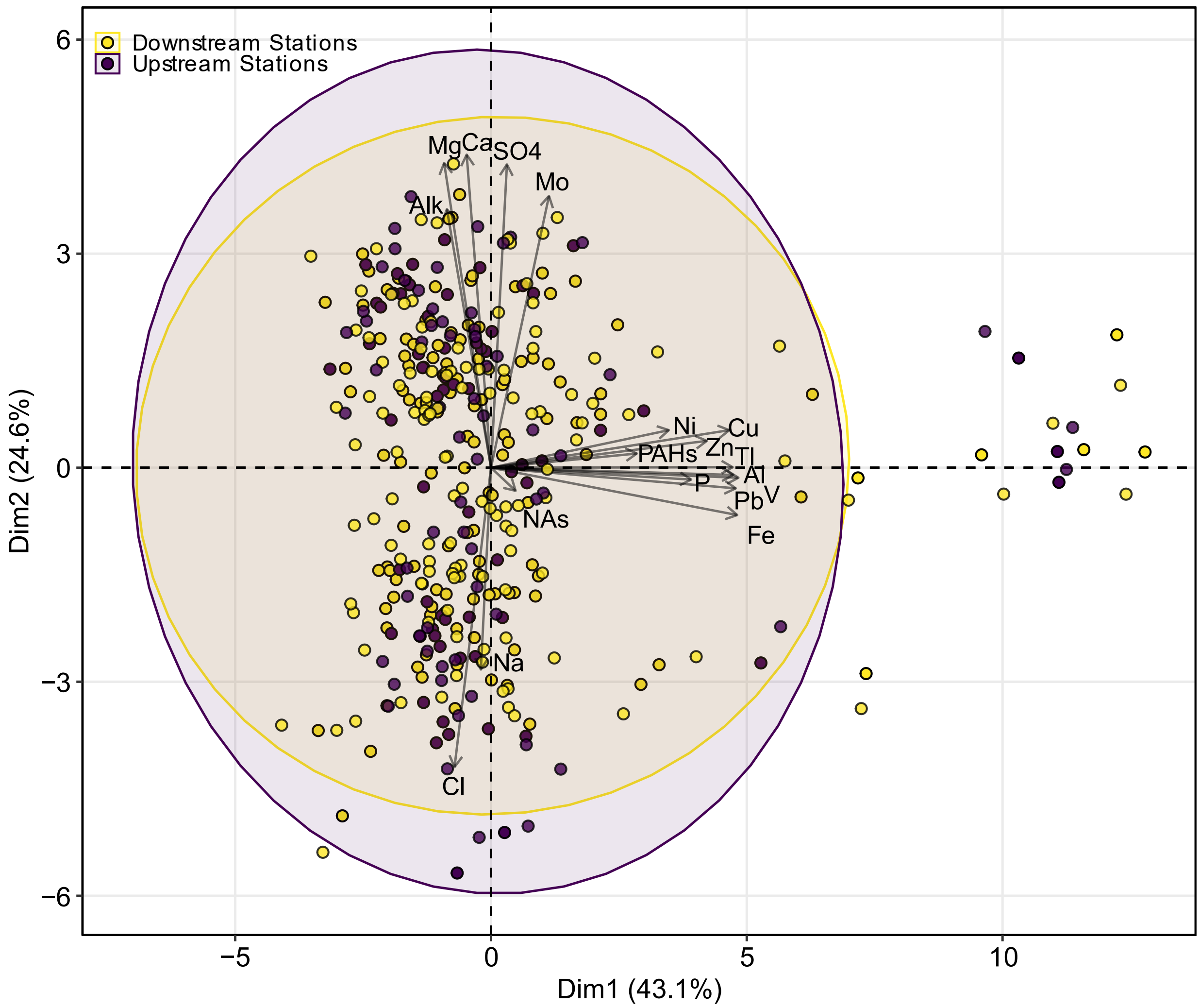

Figure 21 Principal Component Analysis depicting spatial patterns across upstream and downstream locations in water quality based on a subset of 18 parameters (Alk, Al, Ca, Cu, Cl, Fe, Pb, Mg, Mo, NAs, Ni, P, Na, SO_4, Tl, PAH, V, Zn). Analyte concentrations were standardized to a discharge of 900 m^3/s prior to evaluation. The ellipses represent 95% confidence intervals.

Table 10 Tukey's post-hoc test comparing flow normalized analyte concentrations measured both East and West of the island at 0.5 km downstream, EMP dataset (2018 and 2019).

Analyte	0.5 km DS – E vs. 0.5 km DS – W			
	2018		2019	
	Direction	P-value	Direction	P-value
Alkalinity Total CaCO3	E < W	0.999	E < W	1
Aluminum Total	E < W	1	E < W	1
Calcium Dissolved	E < W	0.368	E < W	0.996
Chloride Dissolved	E > W	0.998	E < W	1
Copper Total	E < W	0.995	E < W	1
Iron Total	E < W	1	E < W	1
Lead Total	E < W	0.996	E < W	1
Magnesium Dissolved	E < W	0.629	E < W	1
Molybdenum Total	E < W	**<0.001**	E < W	**<0.001**
Total Naphthenic acids	E > W	1	E > W	0.988
Nickel Total	E > W	1	E < W	1
Phosphorus Total	E < W	0.16	E < W	0.117
Sodium Dissolved/Filtered	E > W	0.982	E < W	1
Sulphate Dissolved	E < W	0.365	E < W	0.99
Thallium Total	E < W	0.977	E < W	1
Total PAHs	E < W	0.456	E < W	1
Vanadium Total	E < W	1	E < W	1
C-WQI	E < W	0.377	E > W	1
Zinc Total	E < W	0.962	E < W	1
NO2 and NO3 as N	E < W	0.080	E < W	**<0.001**
Total Nitrogen	E < W	0.224	E < W	**0.020**

Table Notes: W represents samples collected on the West side of the island, I represent samples collected on the east side of the island. Significant p-values (i.e., $p < 0.05$) are shown in bold. No data from 2021 is presented as the West channel was dry during this year and no comparison can be made.

2.3.3 Semi-permeable Membrane Devices

2.3.3.1 Data Summary

SPMDs were deployed at all EMP stations in August and September in 2018, May, June, July, August, and September in 2019, and July, August, and September in 2021 to provide multiple exposure trials. This technique enables the measurement of dissolved parent and alkylated PAHs, particularly those with logK_{ow} values below 3 (Harman et al., 2009). Water grab samples were also collected following the deployment (day 0) and the collection (day 28–- 30) of the SPMDs, which allows us to compare PAH results between the two sampling techniques. A summary of VMV codes, method detection limits, and non-detections can be found in Appendix A Table A1.

2.3.3.2 Covariation of grab and SPMD PAH data

Figure 22 demonstrates the relationship between total PAHs in paired SPMD and grab samples across all stations. In general, 99% of the grab samples had higher total PAH measurements than the SPMD samples. It is important to note that the SPMD method only quantifies PAHs in the dissolved state, whereas grab samples do not separate dissolved and particulate PAHs and therefore quantify a "total" value, which is the likely explanation for the differences in concentration magnitude when comparing grab and SPMD samples. An important question that can inform future monitoring is whether or not grab samples (i.e., dissolved + particulate PAHs) can accurately predict dissolved PAH fractions, which are important contributors to toxicity due to their higher bioavailability (Huckins et al., 1990). The overall model fit, as represented by the R^2 is 0.25, therefore only 25% of the variability in the dissolved PAH SPMD concentrations can be explained by the grab samples (Figure 22). Additional sampling years during EMP are likely required to further study the relationship between grab and SPMD samples and assess the use of grab samples to predict dissolved PAH fractions.

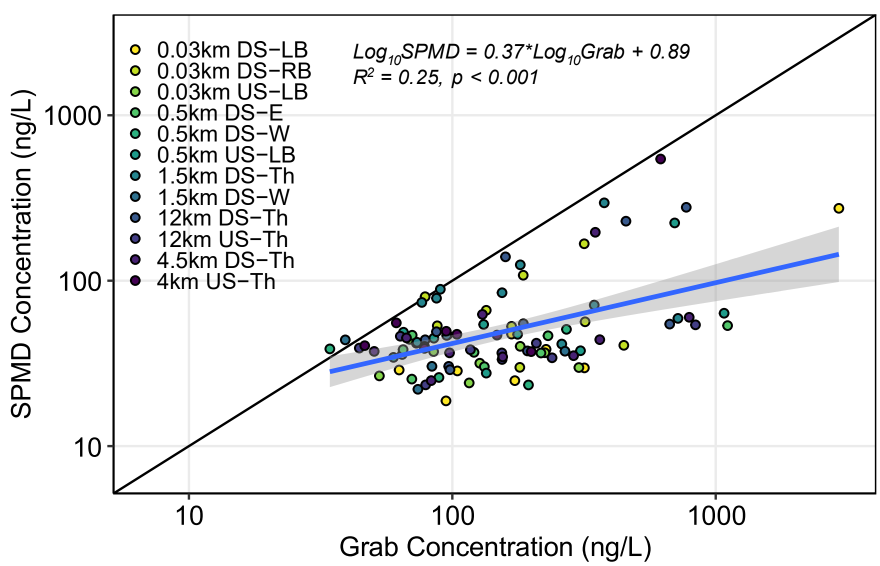

Figure 22 Relationship between PAH concentrations in paired grab and SPMD water samples during EMP (2018, 2019, and 2021).

Figure Notes: The black line represents a 1:1 line, the blue line represents the regression equation of the relationship between grab concentration and SPMD concentration.

2.3.3.3 EXPLANATORY MODELS

Results from the GLM for 76 individual PAH concentrations are summarized in Table 11. The model determined that average discharge (Q_{avg}) was a significant predictor of variation among 38 of the 76 PAHs, with the highest amount of variance explained by discharge in perylene concentrations (Figure 23A) and the least amount of variation explained by discharge in biphenyl concentrations (Figure 23B). Sampling year and/or the interaction between Q_{avg} and Year was determined to be a significant predictor of variation among 58 of the 76 PAHs, sampling distance upstream or downstream was a significant predictor for 16 of 76 PAHs, and shoreline distance was a significant predictor for 43 of 76 PAHs (Table 11). An example of these relationships for total PAHs can be found in Figure 24.

An important spatial comparison to consider is the difference between samples collected on the east and west side of the island located 0.5 km downstream of the proposed OSPW discharge point, as these stations are directly downstream of the Syncrude sewage treatment outfall (Figure 1). Differences among samples collected from the east and west side of the island were explored using a Tukey's post-hoc test for individual comparisons. No significant differences in SPMD concentrations of any of the 76 PAH analytes were observed between the East and West sides of the LAR in either 2018 or 2019, except for phenanthrene in 2018, which was significantly higher on the West side compared to the East side (p = 0.045; Table 12).

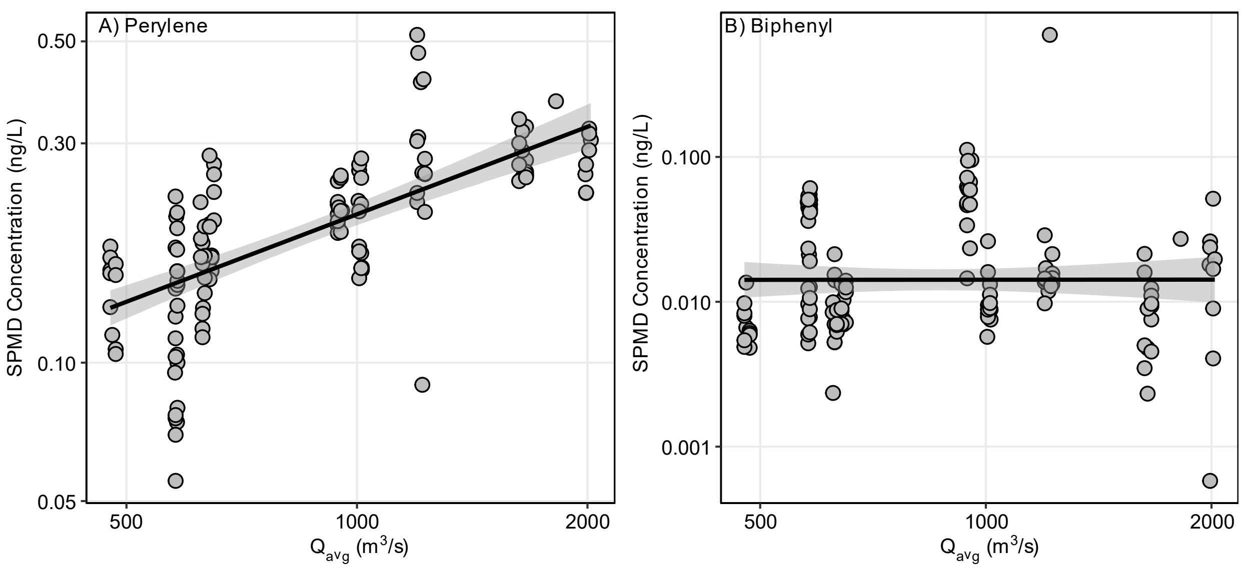

Figure 23 Example of a significant relationship between Q_{avg} and PAH concentration for perylene (A) and a non-significant relationship between Q_{avg} and PAH concentration for biphenyl in EMP dataset (2018, 2019, and 2021).

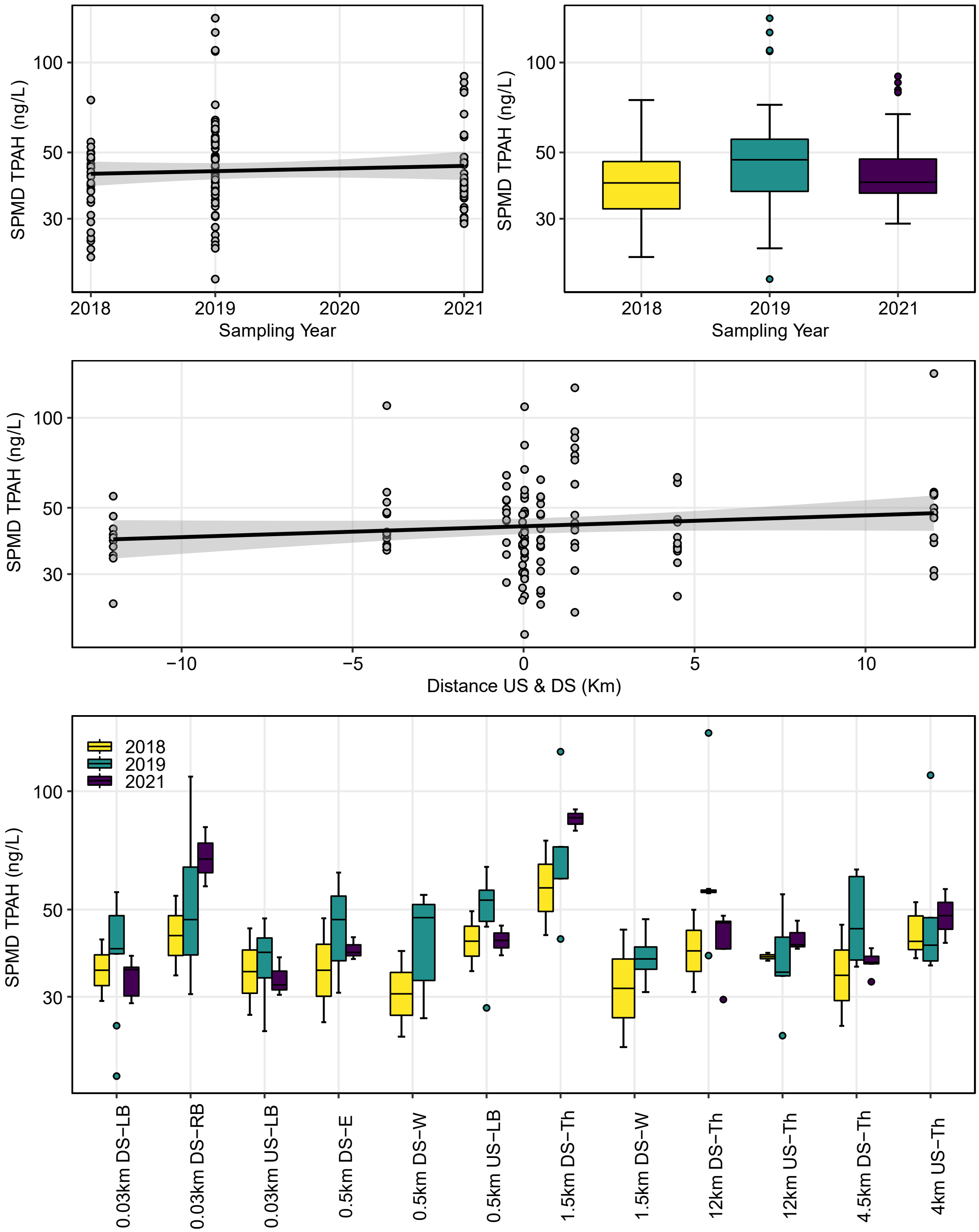

Figure 24 SPMD total PAH concentrations over time (A & B), across sampling distances downstream of proposed OSPW discharge point (C), and among sampling station x year combinations (D).

Table 11 Significance (p-value) and percent of variance explained (%VE) for predictors of concentrations of SPMD PAHs in surface waters collected in the Lower Athabasca River, EMP (2018, 2019, 2021).

Compound	Q_{avg}		Year		Distance (US/DS)		Shoreline Distance		Qavg x Year	
	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE
1,2,6-Trimethylphenanthrene	0.165	1.39	**0.007**	5.43	0.110	1.84	**0.008**	5.16	0.488	0.34
1,2-Dimethylnaphthalene	**0.003**	6.46	0.488	0.35	0.444	0.42	0.064	2.50	**0.024**	3.75
1,4,6,7-Tetramethylnaphthalene	**<0.001**	34.37	0.263	0.60	0.817	0.03	**0.001**	5.32	**0.025**	2.43
1,7-Dimethylfluorene	**<0.001**	33.94	0.324	0.50	0.672	0.09	**0.017**	2.98	0.262	0.65
1,7-Dimethylphenanthrene	0.776	0.06	0.065	2.74	0.561	0.27	0.186	1.40	0.698	0.12
1,8-Dimethylphenanthrene	0.090	2.23	**0.022**	4.13	0.198	1.28	0.913	0.01	0.808	0.05
1-Methylchrysene	0.878	0.01	**<0.001**	16.34	0.093	1.75	**0.001**	6.61	0.129	1.42
1-Methylnaphthalene	**0.010**	3.58	**<0.001**	25.56	0.370	0.43	**0.008**	3.82	**0.019**	2.98
1-Methylphenanthrene	**0.060**	2.62	**0.002**	7.57	0.786	0.05	0.375	0.58	0.298	0.80
2,3,5-Trimethylnaphthalene	**<0.001**	30.33	**0.020**	2.60	0.592	0.13	0.288	0.53	**<0.001**	10.21
2,3,6-Trimethylnaphthalene	**<0.001**	26.20	0.239	0.65	0.755	0.05	0.540	0.18	**<0.001**	16.59
2,4-Dimethyldibenzothiophene	0.003	6.60	0.135	1.66	0.419	0.48	0.073	2.41	0.838	0.03
2,6-Dimethylnaphthalene	**<0.001**	17.77	**0.005**	4.91	0.212	0.93	0.982	0.00	**0.005**	4.89
2,6-Dimethylphenanthrene	**<0.001**	13.73	0.232	0.97	0.481	0.33	0.316	0.68	**0.032**	3.17
2/3-Methyldibenzothiophenes	0.439	0.43	**0.007**	5.30	**0.018**	4.05	0.643	0.15	**0.013**	4.46
2-Methylanthracene	0.364	0.58	0.330	0.67	**0.002**	7.04	**0.005**	5.65	0.242	0.97
2-Methylfluorene	**0.001**	7.30	**<0.001**	11.32	0.500	0.28	**0.006**	4.86	0.073	2.01
2-Methylnaphthalene	**0.002**	5.52	**<0.001**	25.48	0.936	0.00	**0.027**	2.70	0.239	0.76
2-Methylphenanthrene	0.094	1.92	**0.033**	3.12	0.898	0.01	0.227	0.99	**<0.001**	12.60
3,6-Dimethylphenanthrene	**<0.001**	9.42	0.439	0.43	0.576	0.22	**0.014**	4.40	0.922	0.01
3-Methylfluoranthene/Benzo(a)Fluorene	0.534	0.29	**0.027**	3.69	0.248	0.99	**0.007**	5.54	0.559	0.25
3-Methylphenanthrene	**<0.001**	14.97	0.290	0.70	0.666	0.12	0.258	0.80	**<0.001**	8.21
5,9-Dimethylchrysene	**0.005**	5.43	**0.024**	3.41	**0.018**	3.74	**0.001**	7.56	0.311	0.68
5/6-Methylchrysene	0.825	0.03	**<0.001**	17.84	0.049	2.37	**0.003**	5.63	0.089	1.76
7-Methylbenzo(a)Pyrene	0.549	0.27	0.946	0.00	0.167	1.43	0.124	1.77	**0.002**	7.21
9/4-Methylphenanthrene	**<0.001**	18.69	0.987	0.00	0.920	0.01	0.101	1.77	0.166	1.26
Acenaphthene	**<0.001**	10.96	0.172	1.29	0.497	0.32	**0.014**	4.24	0.371	0.55
Acenaphthylene	0.829	0.03	0.438	0.42	0.961	0.00	**0.008**	5.10	**<0.001**	10.62
Anthracene	0.071	2.10	0.001	7.74	**<0.001**	8.16	**0.027**	3.19	0.064	2.21
Benz(a)Anthracene/Chrysene-C1	0.190	1.22	**0.002**	6.98	**0.047**	2.83	**0.023**	3.71	0.635	0.16
Benz(a)Anthracene/Chrysene-C3	0.763	0.07	0.115	1.85	**0.006**	5.77	**0.025**	3.76	0.760	0.07
Benz(a)Anthracene/Chrysene-C4	0.057	2.84	0.705	0.11	0.299	0.84	0.165	1.51	0.176	1.43
Benzo(a)Anthracene	0.354	0.69	0.310	0.83	0.267	0.99	0.231	1.15	0.854	0.03
Benzo(a)Anthracene/Chrysene-C2	0.840	0.03	**0.002**	6.36	**0.012**	4.27	**0.001**	7.73	0.077	2.09

Compound	Q_{avg}		Year		Distance (US/DS)		Shoreline Distance		Qavg x Year	
	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE
Benzo(a)Pyrene	**0.001**	7.27	**0.014**	4.23	0.135	1.54	**0.033**	3.18	0.141	1.50
Benzo(B)Fluoranthene	**<0.001**	13.39	**0.030**	2.88	0.357	0.51	**0.008**	4.41	**0.002**	6.31
Benzo(b,k)Fluoranthene/Benzo(a)Pyrene-C2	**0.004**	6.12	**0.034**	3.29	0.097	2.01	0.423	0.46	0.219	1.10
Benzo(e)Pyrene	0.370	0.49	**<0.001**	16.91	**0.035**	2.74	**0.007**	4.50	**0.046**	2.44
Benzo(g,h,i)Perylene	**<0.001**	16.06	**<0.001**	9.01	0.103	1.54	**0.029**	2.81	0.156	1.17
Benzo(J,K)Fluoranthenes	0.666	0.14	**0.021**	4.15	0.370	0.62	0.177	1.40	0.184	1.36
Benzofluoranthene/Benzopyrene-C1	**0.021**	3.59	**<0.001**	8.41	**0.044**	2.70	**0.039**	2.84	**0.022**	3.54
Benzophenanthrene	0.582	0.21	**0.024**	3.60	**0.035**	3.14	**<0.001**	9.48	0.836	0.03
Biphenyl	0.990	0.00	**<0.001**	11.98	0.309	0.72	0.911	0.01	**0.023**	3.65
C1-Fluoranthenes/Pyrenes	0.359	0.58	**0.001**	7.74	0.087	2.03	**0.002**	6.65	0.526	0.28
C1-Fluorenes	**0.009**	4.98	0.171	1.35	0.172	1.34	**0.022**	3.83	0.072	2.35
C1-Naphthalenes	**0.001**	5.50	**<0.001**	29.11	0.581	0.15	**0.018**	2.83	**0.018**	2.83
C1-Phenanthrenes/Anthracenes	**0.049**	2.98	0.061	2.68	0.991	0.00	0.405	0.52	**0.047**	3.03
C2-Fluoranthenes/Pyrenes	0.281	0.79	**0.010**	4.70	0.053	2.59	**0.001**	8.20	0.123	1.63
C2-Fluorenes	**<0.001**	17.31	**<0.001**	9.69	0.713	0.08	**0.014**	3.57	0.461	0.31
C2-Naphthalenes	0.229	0.98	**0.022**	3.63	**0.009**	4.79	0.192	1.16	**0.001**	7.96
C2-Phenanthrenes/Anthracenes	0.060	2.77	0.190	1.33	0.764	0.07	**0.040**	3.30	0.823	0.04
C3-Fluorenes	**<0.001**	10.83	**0.007**	5.02	0.417	0.44	**0.045**	2.75	0.747	0.07
C3-Naphthalenes	**<0.001**	16.87	**0.026**	3.02	0.547	0.22	0.058	2.17	**0.001**	6.26
C3-Phenanthrenes/Anthracenes	0.507	0.33	**0.029**	3.65	0.247	1.01	**0.011**	4.95	0.823	0.04
C4-Naphthalenes	**<0.001**	24.82	0.261	0.73	0.815	0.03	**0.033**	2.68	**0.049**	2.27
C4-Phenanthrenes/Anthracenes	0.667	0.15	0.971	0.00	0.832	0.04	0.061	2.85	0.330	0.76
Chrysene	**0.001**	7.95	0.084	1.92	**0.036**	2.86	**0.001**	7.55	**0.039**	2.78
Dibenzo(a,h)Anthracene	0.435	0.47	**0.026**	3.89	0.597	0.22	0.667	0.14	0.096	2.16
Dibenzothiophene	0.785	0.05	**0.002**	6.41	**0.001**	7.13	0.436	0.40	**0.002**	6.36
Dibenzothiophene-C1	0.054	2.81	0.280	0.87	0.081	2.30	0.272	0.90	**0.035**	3.39
Dibenzothiophene-C2	**0.021**	3.90	**0.005**	5.75	0.407	0.49	**0.027**	3.58	0.725	0.09
Dibenzothiophene-C3	0.346	0.64	**0.004**	6.34	0.154	1.47	**0.012**	4.62	0.621	0.18
Dibenzothiophene-C4	0.181	1.30	**0.020**	4.00	0.062	2.54	**0.008**	5.22	0.904	0.01
Dimethyl Biphenyl	**0.015**	4.14	0.068	2.29	0.051	2.63	**<0.001**	9.17	0.585	0.20
Fluoranthene	**<0.001**	9.88	**<0.001**	31.48	0.077	1.44	0.126	1.08	0.089	1.33
Fluoranthene/Pyrene-C3	**0.036**	2.96	**0.005**	5.47	**0.012**	4.32	**0.001**	7.51	0.840	0.03
Fluoranthene/Pyrene-C4	0.806	0.04	**0.003**	5.99	**0.014**	4.04	**0.002**	6.34	**0.008**	4.68
Fluorene	**0.023**	3.91	0.088	2.19	0.130	1.72	0.087	2.20	0.448	0.43
Indeno(1,2,3-c,d)Pyrene	0.221	1.13	0.131	1.73	0.474	0.39	**0.013**	4.71	0.156	1.53
Methyl Acenaphthene	**<0.001**	24.16	0.816	0.03	0.169	1.05	0.345	0.49	**<0.001**	8.05

Compound	Q_{avg}		Year		Distance (US/DS)		Shoreline Distance		Qavg x Year	
	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE
Methyl Biphenyl	**0.001**	7.34	**0.002**	6.62	0.788	0.05	0.124	1.60	**0.015**	4.04
Naphthalene	0.244	0.84	**<0.001**	20.62	0.085	1.85	0.255	0.80	0.083	1.87
Perylene	**<0.001**	47.27	**<0.001**	9.21	0.674	0.06	**<0.001**	4.21	0.402	0.23
Phenanthrene	0.713	0.10	**0.002**	7.68	0.847	0.03	0.140	1.64	0.229	1.08
Retene	0.060	2.64	**0.002**	7.09	0.991	0.00	0.933	0.01	0.110	1.89
Total PAH	**0.028**	3.59	0.117	1.80	0.165	1.41	**0.005**	5.79	0.861	0.02

Table Notes: Significant values (i.e., p-value < 0.05) are in bold, %VE represents the percentage of total variance explained by each predictor within the individual models, Shaded cells highlight the %VE that corresponds to significant p-values, Q_{avg} represents the average discharge over the entire SPMD deployment period.

Table 12 Tukey's post-hoc test comparing SPMD TPAH concentrations measured both East and West of the island at 0.5 km downstream, EMP dataset (2018 and 2019).

Parameter	0.5 km–DS - E vs. 0.5 km–DS - W			
	2018		2019	
	Direction	P-value	Direction	P-value
1-Methylchrysene	E > W	1.000	E > W	1.000
1-Methylnaphthalene	E < W	1.000	E > W	1.000
1-Methylphenanthrene	E < W	1.000	E < W	1.000
1,2-Dimethylnaphthalene	E < W	1.000	E > W	1.000
1,2,6-Trimethylphenanthrene	E > W	1.000	E > W	1.000
1,4,6,7-Tetramethylnaphthalene	E > W	1.000	E < W	1.000
1,7-Dimethylfluorene	E > W	0.996	E < W	1.000
1,7-Dimethylphenanthrene	E > W	1.000	E > W	1.000
1,8-Dimethylphenanthrene	E > W	1.000	E > W	1.000
2-Methylanthracene	E > W	0.997	E > W	1.000
2-Methylfluorene	E < W	0.999	E < W	1.000
2-Methylnaphthalene	E < W	1.000	E > W	1.000
2-Methylphenanthrene	E < W	0.257	E < W	0.999
2,3,5-Trimethylnaphthalene	E < W	0.996	E > W	1.000
2,3,6-Trimethylnaphthalene	E < W	0.982	E > W	1.000
2,4-Dimethyldibenzothiophene	E > W	0.251	E > W	1.000
2,6-Dimethylnaphthalene	E < W	1.000	E > W	1.000
2,6-Dimethylphenanthrene	E > W	1.000	E < W	1.000
2/3-Methyldibenzothiophenes	E < W	0.998	E < W	1.000
3-Methylfluoranthene/Benzo(a)Fluorene	E > W	1.000	E > W	1.000
3-Methylphenanthrene	E < W	0.323	E < W	1.000
3,6-Dimethylphenanthrene	E > W	1.000	E > W	1.000
5,9-Dimethylchrysene	E > W	1.000	E > W	1.000
5/6-Methylchrysene	E > W	0.999	E > W	1.000
7-Methylbenzo(a)Pyrene	E > W	1.000	E > W	1.000
9/4-Methylphenanthrene	E < W	1.000	E < W	1.000
Acenaphthene	E < W	0.798	E < W	1.000
Acenaphthylene	E < W	1.000	E > W	1.000
Anthracene	E > W	0.996	E > W	1.000
Benz(a)Anthracene/Chrysene-C1	E > W	0.999	E > W	1.000
Benz(a)Anthracene/Chrysene-C3	E > W	1.000	E > W	0.996
Benz(a)Anthracene/Chrysene-C4	E < W	1.000	E > W	0.971
Benzo(a)Anthracene	E < W	1.000	E < W	1.000
Benzo(a)Anthracene/Chrysene-C2	E > W	1.000	E > W	1.000
Benzo(a)Pyrene	E > W	1.000	E > W	1.000
Benzo(B)Fluoranthene	E < W	1.000	E > W	1.000
Benzo(b,k)Fluoranthene/Benzo(a)Pyrene-C2	E < W	1.000	E > W	0.97
Benzo(e)Pyrene	E > W	1.000	E > W	1.000
Benzo(g,h,i)Perylene	E < W	1.000	E > W	1.000
Benzo(J,K)Fluoranthenes	E > W	1.000	E > W	1.000
Benzofluoranthene/Benzopyrene-C1	E > W	1.000	E > W	1.000
Benzophenanthrene	E > W	0.998	E > W	1.000
Biphenyl	E < W	1.000	E < W	1.000

Parameter	0.5 km–DS - E vs. 0.5 km–DS - W			
	2018		2019	
	Direction	P-value	Direction	P-value
C1-Fluoranthenes/Pyrenes	E > W	0.99	E > W	1.000
C1-Fluorenes	E < W	1.000	E < W	1.000
C1-Naphthalenes	E < W	1.000	E > W	1.000
C1-Phenanthrenes/Anthracenes	E < W	0.346	E < W	1.000
C2-Fluoranthenes/Pyrenes	E > W	0.975	E > W	1.000
C2-Fluorenes	E > W	0.944	E > W	1.000
C2-Naphthalenes	E < W	0.999	E > W	1.000
C2-Phenanthrenes/Anthracenes	E > W	1.000	E > W	1.000
C3-Fluorenes	E > W	0.999	E > W	1.000
C3-Naphthalenes	E < W	1.000	E > W	1.000
C3-Phenanthrenes/Anthracenes	E > W	1.000	E > W	1.000
C4-Naphthalenes	E > W	1.000	E < W	1.000
C4-Phenanthrenes/Anthracenes	E > W	1.000	E > W	1.000
Chrysene	E > W	1.000	E > W	1.000
Dibenzo(a,h)Anthracene	E < W	1.000	E > W	1.000
Dibenzothiophene	E < W	0.715	E > W	1.000
Dibenzothiophene-C1	E < W	0.842	E < W	1.000
Dibenzothiophene-C2	E < W	0.997	E > W	1.000
Dibenzothiophene-C3	E > W	0.998	E > W	1.000
Dibenzothiophene-C4	E > W	0.989	E > W	1.000
Dimethyl Biphenyl	E < W	1.000	E < W	0.624
Fluoranthene	E < W	1.000	E < W	1.000
Fluoranthene/Pyrene-C3	E > W	0.999	E > W	1.000
Fluoranthene/Pyrene-C4	E > W	1.000	E > W	1.000
Fluorene	E < W	0.571	E > W	1.000
Indeno(1,2,3-c,d)Pyrene	E > W	1.000	E < W	1.000
Methyl Acenaphthene	E < W	1.000	E < W	1.000
Methyl Biphenyl	E < W	1.000	E < W	1.000
Naphthalene	E < W	0.998	E > W	1.000
Perylene	E > W	1.000	E > W	1.000
Phenanthrene	E < W	**0.046**	E < W	1.000
Retene	E > W	1.000	E > W	1.000
Total PAH	E > W	1.000	E > W	1.000

Table Notes: Significant p-values (i.e., $p < 0.05$) are shown in bold. No data from 2021 is presented as the West channel was dry during this year and no comparison can be made.

2.3.4 Sediment Quality Variables

2.3.4.1 Data Summary

The EMP sediment quality dataset includes 119 samples collected at 37 sampling events in 2018, 2019 and 2021. A map of sampling locations can be found in Figure 2. In 2018, there were a total of 13 sampling events, representing one at each station (Table 13). Samples were collected in triplicate at all stations, except for stations located 0.03 km upstream and downstream, where five samples were collected on each sampling event. In 2019, there were a total of 13 sampling events, representing one at each station.

During each sampling event, five samples were collected from each sampling station (Table 13). Finally, in 2021, there were a total of 11 sampling events, with stations 1.5 km DS – W and 0.5 km DS – W not sampled, because the channel on the west side was dried up. In 2021, a total of 37 grab samples were collected. A summary of VMV codes, method detection limits, and non-detections can be found in Appendix A Table A1.

Non-detects for sediment quality parameters were compiled as they were for surface water (Figure 25). For all the analyses, non-detects (including PAHs not meeting quantification criteria; NDR and cases of matrix interference; M) were not included, and no substitutions were made. Less than 25% of PAHs, minor elements, and nutrients were below detection limits across all sampling years and stations (Figure 25). Naphthenic acids were below detection limits in less than 25% of samples in 2018 and 2019 but had roughly 30% non-detection in 2021. Finally, nearly 50% of the phenols measured in 2018 and 2019 were below detection limit in sediments across all sampling stations.

No individual analytes surpassed long-term or short-term CCME sediment quality guidelines for the protection of aquatic life within the EMP.

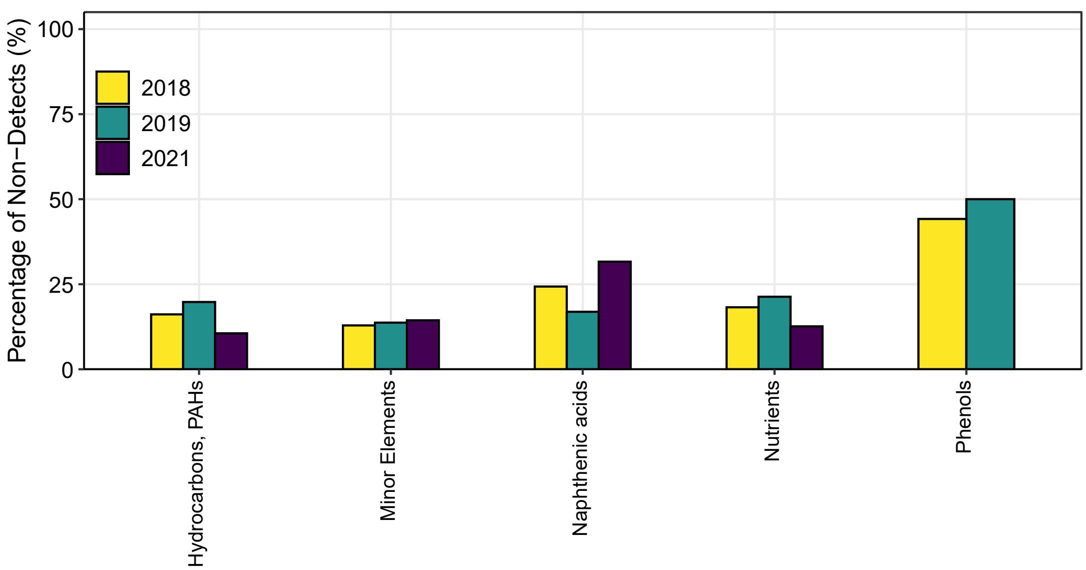

Figure 25 Percentage of non-detects observed in sediment samples by parameter category in the EMP dataset (2018, 2019, and 2021).

Table 13 Lower Athabasca River (LAR) sediment sampling events for sediment over the EMP period (2018, 2019, and 2021).

Station Name	2018		2019		2021		Total	
	Sampling Events	Grab Samples	Sampling Events	Grab Samples	Sampling Events	Grab Samples	Sampling Events	Grab Samples
25 km US	1	3	1	3	1	3	3	9
34 km DS	1	3	1	3	1	3	3	9
12 km DS	1	3	1	3	1	3	3	9
4.5 km DS	1	3	1	3	1	3	3	9
1.5–km DS - E	1	3	1	3	1	3	3	9
1.5–km DS - W	1	3	1	3	-	-	2	6
0.5–km DS - E	1	3	1	3	1	3	3	9
0.5–km DS - W	1	3	1	3	-	-	2	6
0.03 km DS	1	5	1	5	1	3	3	13
0.03 km US	1	5	1	5	1	3	3	13
0.5 km US	1	3	1	3	1	3	3	9
4 km US	1	3	1	3	1	3	3	9
12 km US	1	3	1	3	1	3	3	9
Totals	**13**	**43**	**13**	**43**	**11**	**37**	**37**	**119**

Table Notes: Sampling approaches and locations differed and are indicated as follows: Th = Thalweg, E = East of Island, W = West of Island, Right = Right Bank, and Left = Left Bank.

2.3.4.2 Explanatory Models

Results from the GLMs for the sediment quality variables are summarized in Table 15. The models determined that aluminum concentrations in sediment was a significant predictor (p value < 0.05) of variation in concentrations of all analytes, except for total Na. An examination of the relationships between the analytes and concentrations of aluminum indicated that concentrations of analytes in sediment typically increased with concentration of aluminum (Figure 26).

Linear trends over time (i.e., Year) were statistically significant for ammonium, naphthenic acids, and total concentrations of B, Fe, Li, Ni, Ag, Sr, Sn, U, V and Na. Variations in trends over time that depend on aluminum concentrations (i.e., the interaction term between Al X Year) were statistically significant for total concentrations of B, Ca, Mn, Ni, Sn, PAHs, along with Total Kjeldahl Nitrogen and Total Organic Nitrogen. Spatial trends, when considering the distance upstream (US) or downstream (DS) from the proposed OSPW discharge point, were significant for methyl mercury, total mercury, total Mo, total organic nitrogen, total P, total Sn, and total PAHs.

Temporal and spatial differences were further investigated below and in Section 2.3.4.3 on aluminum normalized sediment quality variables. Aluminum normalizing equations are provided in Appendix C Table C2.

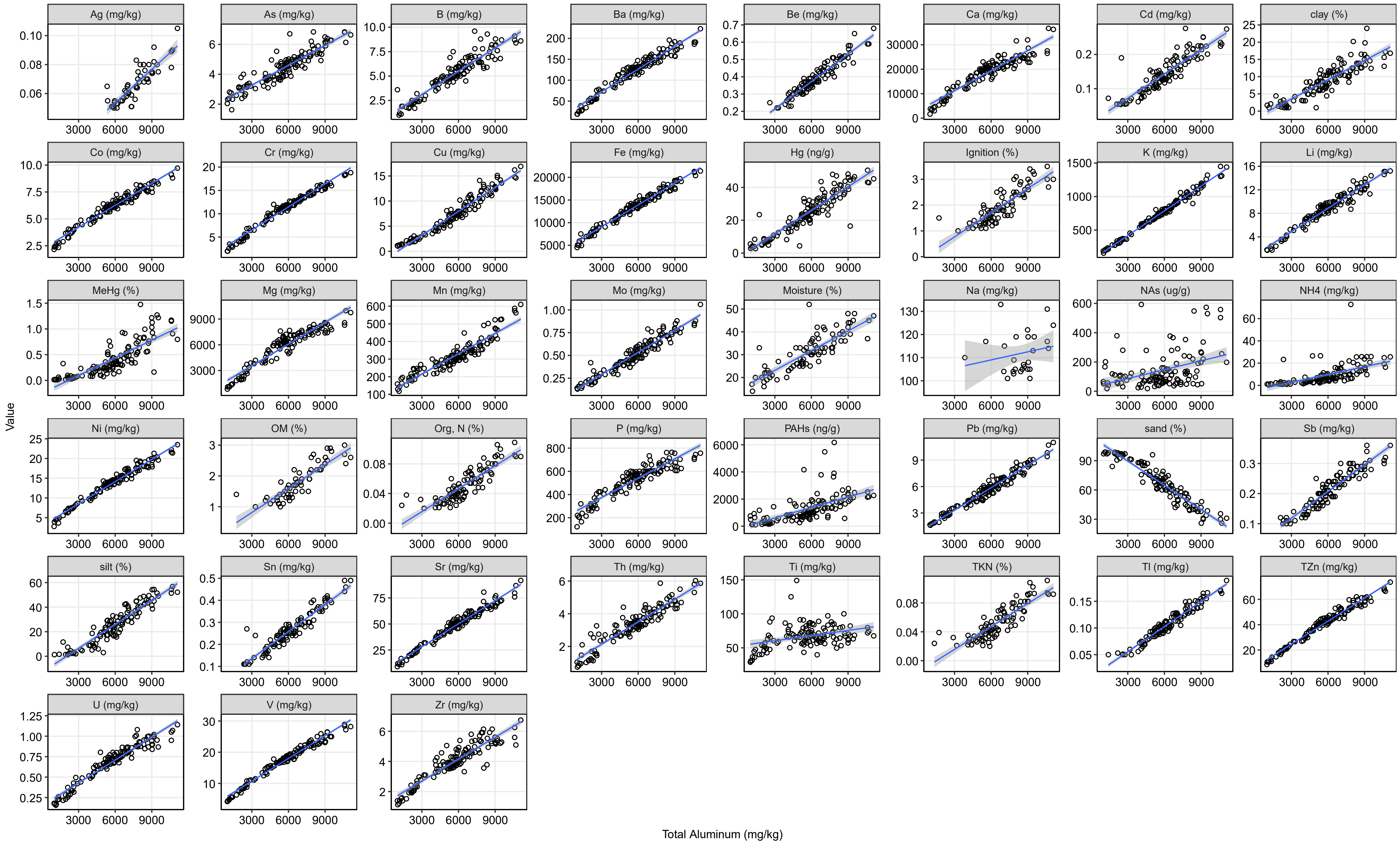

Figure 26 Relationship between total aluminum concentration and the subset of sediment quality parameters measured under the EMP (2018, 2019, and 2021).

Table 14 Significance (p-value) and percent of variance explained (%VE) for predictors of concentrations of analytes in sediment collected the Lower Athabasca River, EMP (2018, 2019, 2021).

Analyte	Aluminum		Year		Distance (US/DS)		Al x Year	
	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE
Available Ammonium	**<0.001**	62.5	**0.024**	1.7	0.956	<0.1	0.530	0.1
Methyl Mercury	**<0.001**	82.0	0.122	0.4	**0.024**	0.8	0.577	<0.1
Naphthenic acids	**<0.001**	9.9	**<0.001**	46.4	0.921	0.004	0.148	0.8
Total Antimony	**<0.001**	87.3	0.305	0.1	0.243	0.2	0.145	0.3
Total Arsenic	**<0.001**	82.8	0.082	0.4	0.825	0.01	0.673	<0.1
Total Barium	**<0.001**	96.4	0.437	0.02	0.684	<0.1	0.100	0.08
Total Beryllium	**<0.001**	90.3	0.094	0.3	0.961	<0.1	0.248	0.1
Total Boron	**<0.001**	88.4	**<0.001**	1.8	0.323	0.1	**0.002**	0.8
Total Cadmium	**<0.001**	78.2	0.161	0.4	0.135	0.5	0.819	<0.1
Total Calcium	**<0.001**	92.3	0.820	<0.1	0.168	0.1	**0.001**	0.7
Total Chromium	**<0.001**	98.0	0.875	<0.1	0.082	0.1	0.069	0.1
Total Cobalt	**<0.001**	97.5	0.234	0.03	0.330	0.0	0.755	<0.1
Total Copper	**<0.001**	96.5	0.442	0.02	0.057	0.11	0.156	0.06
Total Iron	**<0.001**	97.3	**0.044**	0.1	0.642	<0.1	0.152	<0.1
Total Kjeldahl Nitrogen	**<0.001**	57.7	0.706	0.1	0.096	1.2	**0.007**	3.3
Total Lead	**<0.001**	96.4	0.686	0.01	0.728	0.004	0.126	0.07
Total Lithium	**<0.001**	96.6	**<0.001**	1.0	0.346	0.02	0.770	0.002
Total Magnesium	**<0.001**	94.3	0.100	0.1	0.191	0.1	0.134	0.1
Total Manganese	**<0.001**	85.9	0.876	0.003	0.004	0.9	**0.001**	1.1
Total Mercury	**<0.001**	83.9	0.657	0.03	**0.012**	0.8	0.183	0.2
Total Molybdenum	**<0.001**	92.5	0.825	0.003	**0.018**	0.4	0.510	<0.1
Total Nickel	**<0.001**	98.0	**0.013**	0.1	0.075	<0.1	**<0.001**	0.2
Total Organic Nitrogen	**<0.001**	59.7	0.766	0.04	**0.014**	2.5	**0.007**	3.0
Total Phosphorus	**<0.001**	90.4	0.415	0.05	**0.010**	0.5	0.548	0.03
Total Potassium	**<0.001**	99.0	0.256	0.01	0.747	0.001	0.773	0.0007
Total Silver	**<0.001**	73.5	**0.026**	2.5	0.473	0.3	0.731	0.06
Total Strontium	**<0.001**	97.9	**0.007**	0.1	0.133	0.04	0.068	0.05
Total Thallium	**<0.001**	91.0	0.084	0.3	0.623	0.02	0.745	0.01
Total Thorium	**<0.001**	87.1	0.408	0.1	0.208	0.17	0.614	0.03
Total Tin	**<0.001**	85.4	**0.011**	0.8	**0.030**	0.61	**0.027**	0.6
Total Titanium	**<0.001**	32.7	0.364	0.5	0.703	0.1	0.239	0.8
Total Uranium	**<0.001**	95.9	**0.014**	0.2	0.797	0.002	0.114	0.084
Total Vanadium	**<0.001**	98.8	**0.001**	0.1	0.223	0.01	0.632	0.002
Total Zinc	**<0.001**	98.6	0.390	0.01	0.177	0.02	0.748	0.001
Total Zirconium	**<0.001**	91.9	0.699	0.01	0.447	0.04	0.489	0.03
Total PAHs	**<0.001**	53.8	0.619	0.08	**0.036**	1.4	**<0.001**	8.0
Total Sodium	0.501	1.5	**0.044**	14.9	0.343	3.1	0.212	5.4

Table Notes: Significant values (i.e., p-value < 0.05) are in bold.
%VE represents the percentage of total variance explained by each predictor within the individual models
Shaded cells highlight the %VE that corresponds to significant p-values.

2.3.4.3 Visualization of Trends

A PCA plot of aluminum-normalized sediment quality variables across EMP sampling stations (n = 13 through 2018 and 2019, n = 11 in 2021) and years (n = 3) is provided in Figure 27.

2.3.4.3.1 Temporal

The PCA revealed that the overall sediment quality profiles were generally similar across all three sampling years (i.e., 2018, 2019, and 2021), as indicated by the overlapping ellipses (Figure 28A). An example of the overall temporal trend and differences amongst sampling years can be found in Figure 29for naphthenic acids. In this case, we observed NAs concentrations decreasing with time.

2.3.4.3.2 Spatial

The PCA plot depicting spatial variability among sediment quality variables (Figure 28B) demonstrates general overlap of ellipses among the 12 sampling stations. There are instances where individual samples collected from the 25km upstream station are more positively correlated with PC axis 1 and negatively correlated with PC axis 2 (Figure 28B). An example of the decreasing trend of Mo sediment concentrations downstream of the proposed OSPW discharge point can be found in Figure 30B.

No clear distinguishable patterns were observed when comparing sampling stations upstream and downstream of the potential OSPW release point as the 95% confidence ellipses in Figure 28C are nearly overlapped. This exercise is an important step in developing baseline conditions. If the future release of treated OSPW does in fact cause a shift in sediment quality downstream of the discharge, it will likely become evident in repeat of this multivariate procedure.

An important spatial comparison to consider is the difference between samples collected on the east and west side of the island located 0.5 km downstream of the proposed OSPW discharge point, as these stations are directly downstream of the Syncrude sewage treatment outfall (Figure 1). Differences among samples collected from the east and west side of the island were explored using a Tukey's post-hoc test for individual comparisons. Potassium was the only compound measured in the sediment that differed significantly between E and W sampling locations at the 0.5km downstream station, where E samples were significantly higher than W samples (p = 0.002; Table 15). Considering the Syncrude sewage treatment outfall is located along the western bank of the LAR, the increase in potassium in sediments from the eastern bank is likely unrelated.

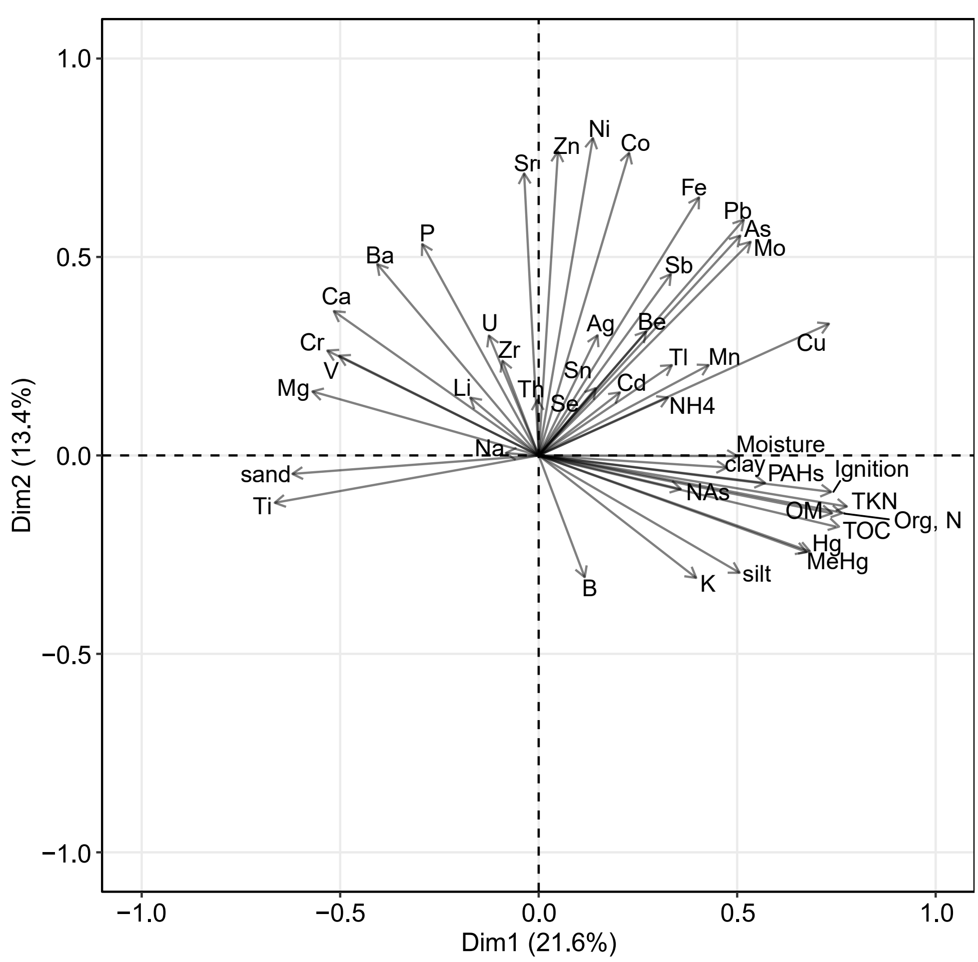

Figure 27 Principal Component Analysis depicting sediment parameter loadings. Parameter concentrations were normalized to total aluminum prior to evaluation.

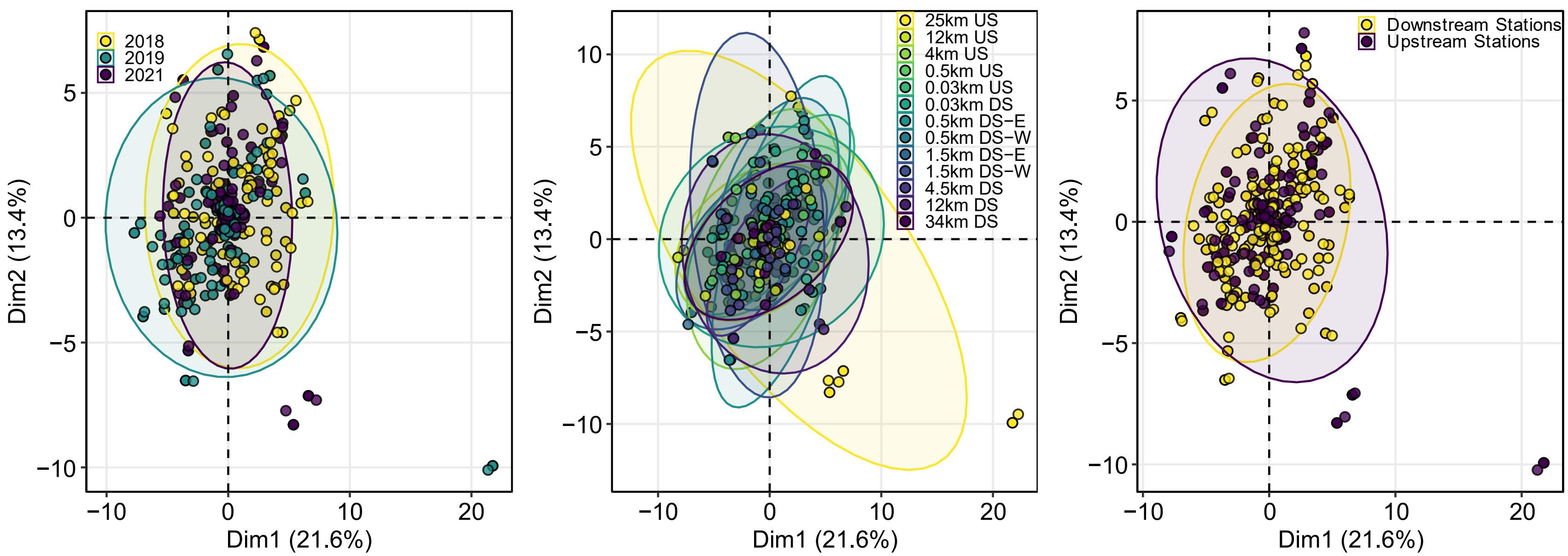

Figure 28 Principal Component Analysis depicting temporal (A) and spatial patterns (across stations (B) and upstream vs. downstream (C)) in sediment quality. Parameter concentrations were normalized to aluminum (6000 µg/g) prior to evaluation and correspond to the parameter loadings in Figure 27. The ellipses represent 95% confidence intervals.

Figure Notes: W = West of island, and E = East of Island

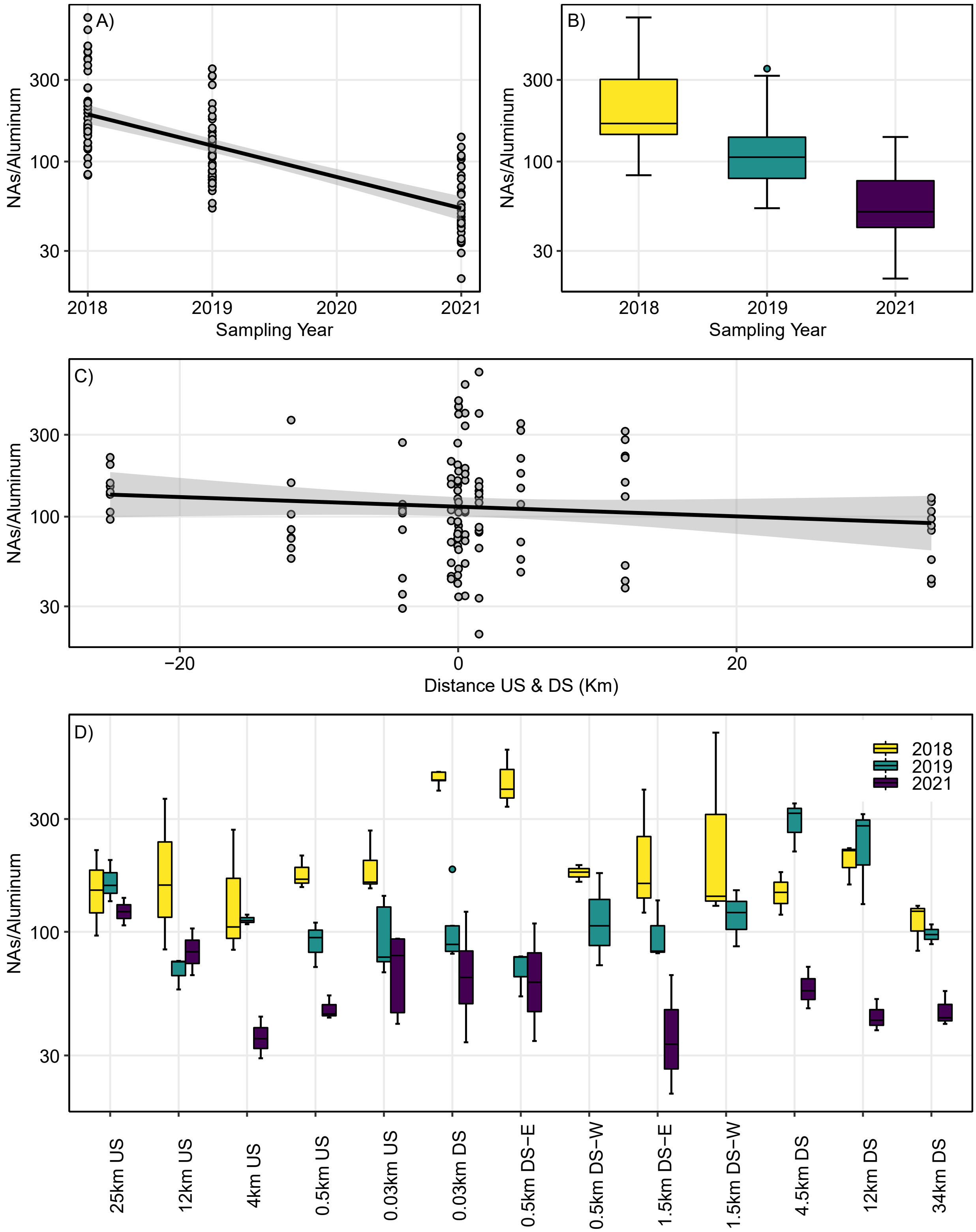

Figure 29 Range and variation in total naphthenic acids over time and across sampling stations. Concentrations standardized to total aluminum 6000 µg/g.

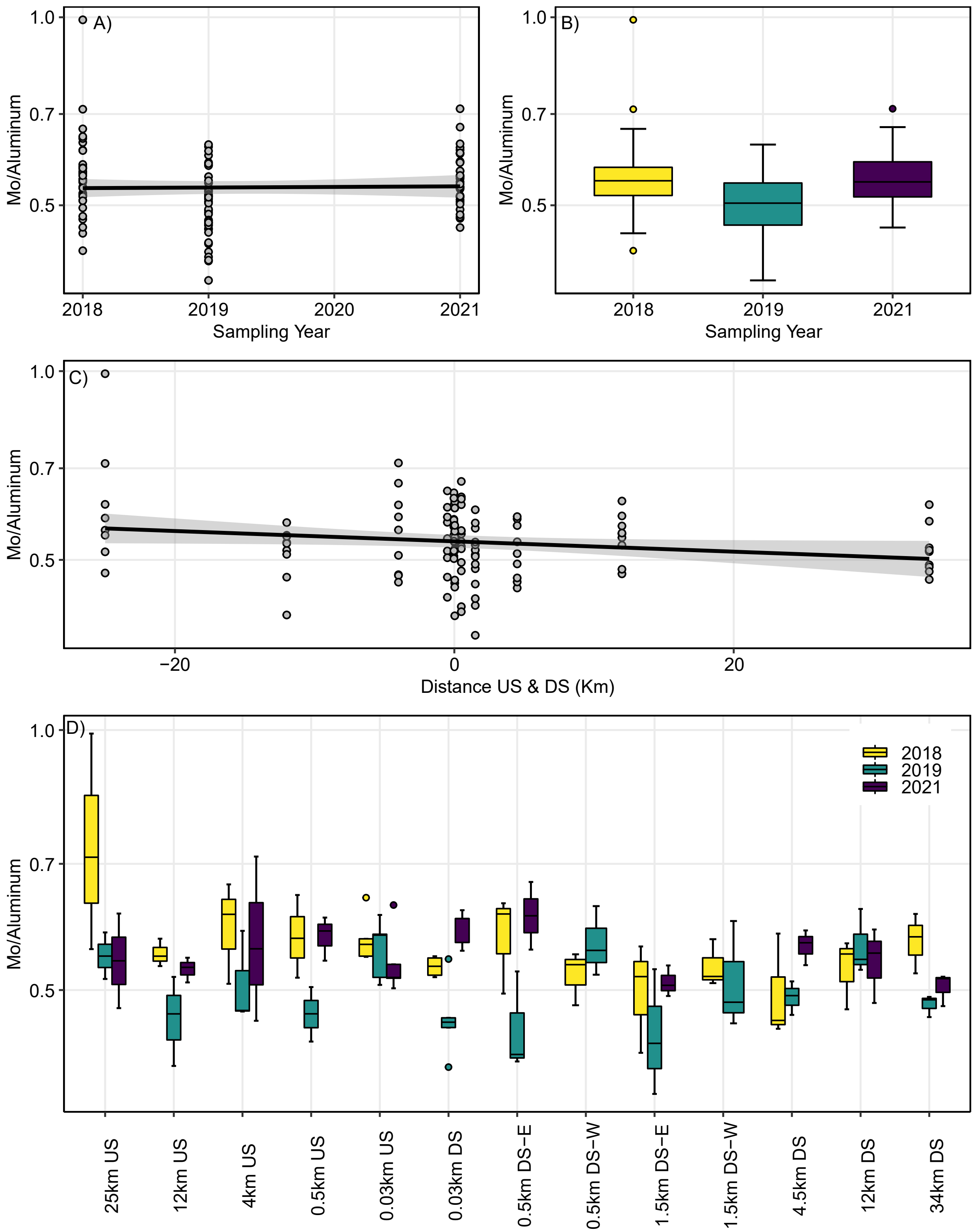

Figure 30 Range and variation in molybdenum over time and across sampling stations. Concentrations standardized to total aluminum concentration of 6000 µg/g.

Table 15 Tukey's post-hoc test comparing aluminum normalized analyte concentrations measured at the stations (0.5 km DS) located both east (E) and West (W) of the island, EMP dataset (2018 and 2021).

Analyte	–.5 km DS - E vs. –.5 km DS - W			
	2018		2019	
	Direction	P-value	Direction	P-value
Ammonium	E > W	1	E < W	0.32
Methyl Mercury	E > W	0.988	E < W	1
Total Naphthenic Acids	E > W	0.221	E < W	0.967
Total Antimony	-	-	-	-
Total Arsenic	E > W	0.942	E < W	0.984
Total Barium	E < W	0.987	E < W	0.451
Total Beryllium	-	-	-	-
Total Boron	E > W	1	E > W	1
Total Cadmium	-	-	-	-
Total Calcium	E < W	0.992	E < W	1
Total Chromium	E > W	0.999	E < W	0.575
Total Cobalt	E < W	0.914	E < W	0.263
Total Copper	E > W	0.617	E < W	0.307
Total Iron	E > W	1	E < W	0.517
Total Kjeldahl Nitrogen	-	-	-	-
Total Lead	E < W	1	E < W	0.579
Total Lithium	-	-	E < W	0.0987
Total Magnesium	E < W	0.968	E < W	0.992
Total Manganese	E < W	1	E > W	1
Total Mercury	E > W	1	E < W	0.983
Total Molybdenum	E > W	0.999	E < W	0.295
Total Nickel	E > W	1	E < W	0.242
Total Organic Carbon	E > W	1	E < W	0.996
Total Phosphorous	E < W	1	E < W	0.0735
Total Potassium	E > W	**0.002**	E < W	1
Total Silver	-	-	-	-
Total Sodium	-	-	-	-
Total Strontium	E > W	1	E < W	0.14
Total Thallium	-	-	E > W	0.637
Total Thorium	E < W	1	E > W	0.96
Total Tin	-	-	-	-
Total Titanium	E < W	0.999	E > W	1
Total Uranium	E < W	0.986	E < W	0.752
Total Vanadium	E < W	0.981	E < W	0.0804
Total Zinc	E < W	0.577	E < W	0.23
Total Zirconium	E < W	1	E < W	0.91
Total PAHs	E > W	1	E < W	1

Table Notes: Significant p-values (i.e., $p < 0.05$) are shown in bold. No data from 2021 is presented as the West channel was dry during this year and no comparison can be made.

2.3.5 Benthic Algae Communities

The algae data collected during the EMP were used to calculate several indices of community composition at the LPL. A map of sampling locations can be found in Figure 2. Summary statistics were determined and trends, both spatial and temporal, were investigated and discussed below.

2.3.5.1 Data Summary

Similar to sediments, algae communities were collected from 13 stations in 2018 and 2019 while only 11 stations were sampled in 2021 (Table 16) due to two stations located at 0.5 and 1.5 km DS drying up on the west side of the island. Algae communities were dominated by diatoms such as *Diatoma moniliformis* (11%), *Nitzschia dissipata* (10%), *Nitzschia acicularis* (10%), *Stephanodiscus parvus* (8%), *Diatoma tenuis var. moniliformis* (8%), *Nitzschia fonticola* (7%), and *Achnanthidium minutissimum* (7%). The relative abundances of dominant taxa are illustrated in Figure 31 for stations located upstream and downstream of the proposed OSPW discharge point.

Summary statistics for algal indices of community composition are provided in Table 17 and broken down by sample-year in Appendix B Table B1. Stations located upstream from the proposed OSPW discharge point had mean biomasses ranging between 295 and 1,017 g/m^2, while stations located downstream had mean biomasses ranging between 233 and 804 g/m^2. Summary statistics also highlighted similarities in algal indices of richness (US: 2-69, DS: 1-68), diversity (US: 0.50-0.96, DS: 0-0.96), and evenness (–S: 0.046 - 1, DS: 0.054-1.00,) when comparing stations located upstream and downstream of the potential OSPW discharge point. These were examined further in Section 2.3.4.2.

The ordination of the algal community data is presented in Figure 32. Samples collected in 2018 and 2021 were clustered around 0 for both axes, while 2019 samples had negative NMDS axis 1 scores and NMDS axis 2 scores ranging between -1.5 and 3.0. Pearson correlations between raw taxa (LPL) densities and sample scores on each of the NMDS axes are illustrated. An overlay of either top panel (A or B) with the lower panel (panel C) indicates which taxa were more abundant in which samples. For example, the skew of 2019 samples into the negative region of NMDS1 is driven by the low relative abundance across all benthic algae groups compared to years 2018 and 2021.

Spearman Rank correlations among indices and potential covariables (i.e., discharge Q60 and summer air temperature) are provided in Table 18. There was a correlation (i.e., r > critical r of 0.18) between discharge (Q60) and all algal indices. Density (r = -0.40), richness (r = -0.58), Simpson's Diversity (r = -0.37), Chlorophyll-*a* (r = -0.40), biomass (r = -0.33), and NMDS axis 1 scores (r = -0.61) were negatively associated with discharge, while Simpson's Evenness (r = 0.30) and NMDS2 scores (r = 0.21) were positively associated with discharge. Similarly, there was a correlation between air temperature, a surrogate for water temperature, and many of the algal indices including Density (r = -0.26), Chlorophyll-*a* (r = 0.36), and NMDS axis 1 and 2 scores (r = -0.22 and 0.51, respectively). Based on this, discharge was chosen as the covariable to include in the explanatory models considering it correlates with more of the algal indices of community composition and is an important predictor for other sample types including water quality, benthos, and fish.

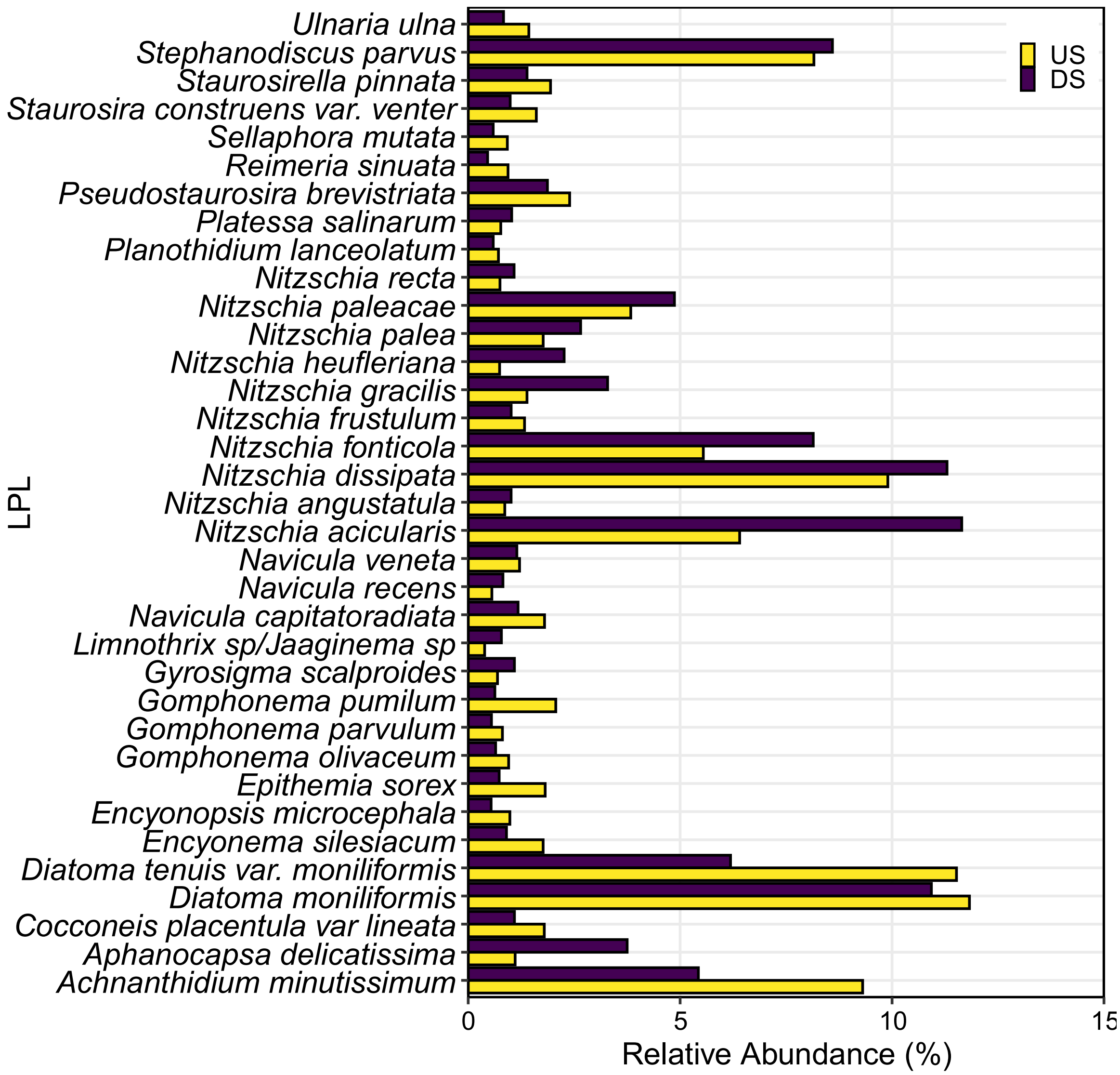

Figure 31 Relative abundance of non-rare taxa (> 0.5%) in samples collected both upstream and downstream of the proposed OSPW discharge point, EMP program (2018, 2019 and 2021).

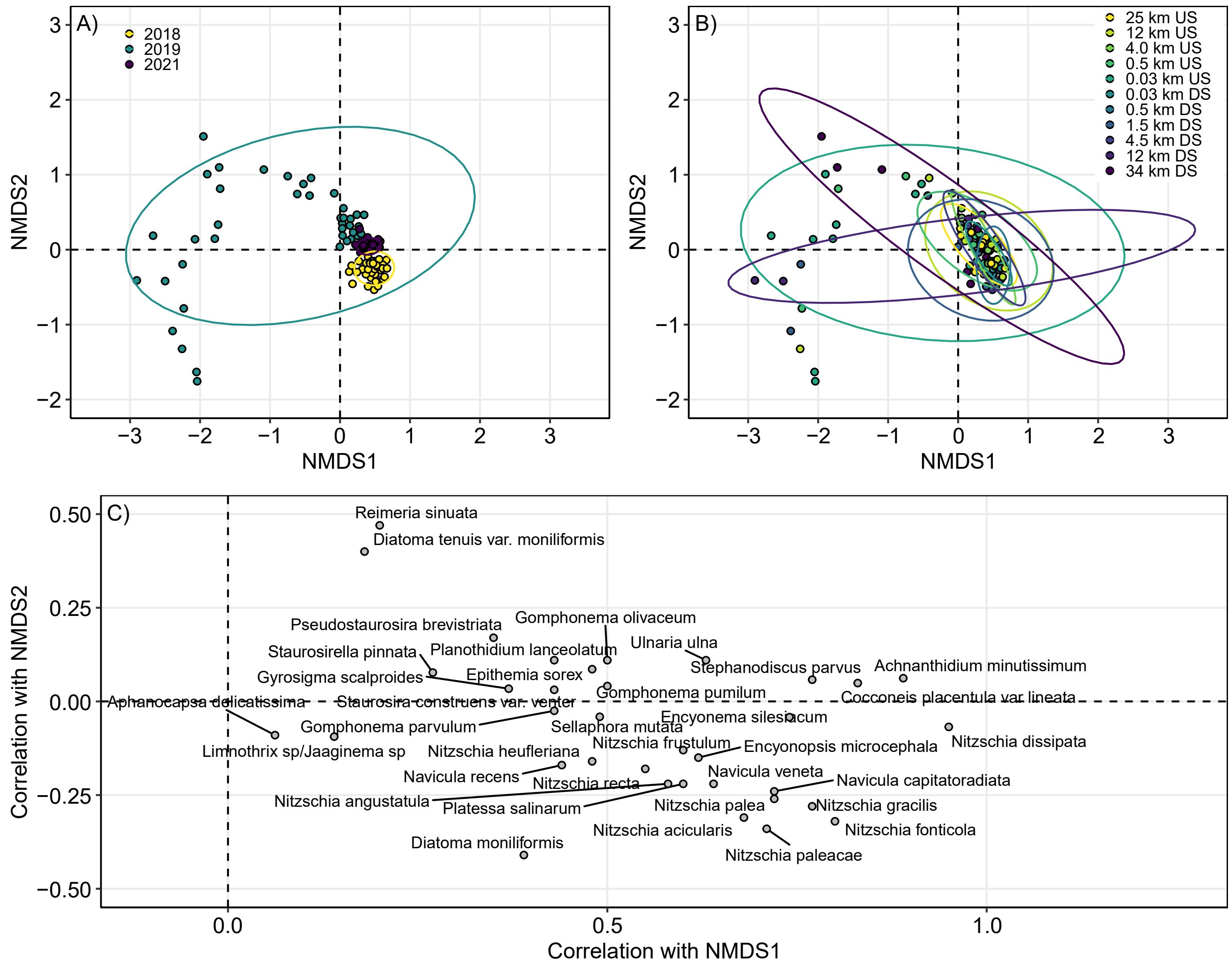

Figure 32 NMDS Axis 1 and 2 scores by year (A), sampling station (B), and correlation with LPL (C) for the EMP dataset (2018, 2019, 2021).

Table 16 Benthic algae community samples collected as part of the EMP (2018, 2019 and 2021).

Station	2018	2019	2021
25 km US	3	3	3
34 km DS	3	3	3
12 km DS	3	3	3
4.5 km DS	4	3	3
1.5 km DS - E	2	3	3
1.5 km DS - W	3	3	-
0.5 km DS - E	3	3	3
0.5 km DS - W	3	3	-
0.03 km US	4	5	5
0.03 km US	5	5	5
0.5 km US	3	3	3
4.0 km US	3	3	3
12 km US	3	3	3
Total Events	**13**	**13**	**11**

Table 17 Summary statistics of benthic algal community indices of community composition for samples collected during the EMP program (2018, 2019 and 2021).

Statistic	Density (# of organisms / m^2)	Richness (# of LPL/sample)	Diversity	Evenness	Chlorophyll-*a* ($\mu g/cm^2$)	Biomass (g/m^2)
N	122	122	122	122	122	122
Min	385	1.0	0.0	0.05	0.02	0.06
Max	6,773,363	69	0.96	1.00	2.3	4,959
Mean	936,940	47	0.87	0.33	0.7	508
SD	946,169	18.7	0.13	0.25	0.6	617
SE	85,662	1.7	0.01	0.02	0.05	56

Table 18 Spearman Rank correlations among covariables and benthic algal indices of community composition for samples collected during the EMP program (2018, 2019 and 2021).

Parameters		Discharge	Air Temp	Density	Richness	Simpson's Diversity	Simpson's Evenness	Chl-*a*	Biomass	NMDS1	NMDS2
Covariables	Discharge	1	-	-	-	-	-	-	-	-	-
	Air Temperature	**-0.43**	1	-	-	-	-	-	-	-	-
Algae Indices	Density	**-0.40**	**-0.26**	1	.	-	-	-	-	-	-
	Richness	**-0.58**	0.17	**0.48**	1	-	-	-	-	-	-
	Simpson's Diversity	**-0.37**	0.15	-0.04	**0.53**	1	-	-	-	-	-
	Simpson's Evenness	**0.30**	0.09	**-0.74**	**-0.40**	**0.33**	1	-	-	-	-
	Chlorophyll-*a*	**-0.40**	**0.36**	**0.20**	**0.36**	**0.36**	-0.06	1	-	-	-
	Biomass	**-0.33**	0.00	**0.73**	**0.51**	**0.51**	**-0.58**	**0.27**	1	-	-
	NMDS1	**-0.61**	**-0.22**	**0.76**	**0.56**	**0.56**	**-0.45**	**0.26**	**0.49**	1	-
	NMDS2	**0.21**	**0.51**	**-0.36**	-0.01	-0.01	0.12	0.10	-0.02	**-0.41**	1

Table Notes: Values in bold exceed the critical r of 0.18, calculated as 1.96/(SQRT(N-1)), where N is the sample size of the dataset (i.e., 122).

2.3.5.2 Explanatory Models

Results from the GLMs for the algal indices of community composition are summarized in Table 19. The models determined that discharge for the 60-day period prior to algae sampling (Q60) was a significant predictor (p-value < 0.05) of variation for algal density, richness, diversity, evenness, chlorophyll-*a*, biomass, and NMDS axis 1 scores. Discharge explained a significant amount of variation (i.e., > 20%) for density (30%), NMDS axis 1 scores (25%) and richness (20%). There was no apparent association between discharge and NMDS axis 2 scores.

Linear trends over time were statistically significant for all algal indices of community composition, except for Simpson's Diversity and NMDS axis 2 scores. There were also statistically significant variations associated with the distance from the proposed OSPW discharge point for density, Simpsons' Diversity, Simpson's Evenness, and biomass.

Temporal and spatial differences were investigated below (Section 2.3.5.3) on flow-normalized indices of algal community. Flow normalizing equations are provided in Appendix C Table C3.

Table 19 Significance (p-value) and percent of variance explained (%VE) for predictors of benthic algal indices of community composition for samples collected in the Lower Athabasca River, EMP (2018, 2019, 2021).

Index	Q60 (m^3/s)		Year		Distance (US/DS)	
	P-value	%VE	P-value	%VE	P-value	%VE
log Density	**<0.001**	30.1	**<0.001**	13.9	**0.011**	3.0
log Richness	**<0.001**	29.4	**0.004**	5.2	0.177	1.2'
Simpson's Diversity	**0.001**	17.6	0.178	3.8	**0.334**	0.6'
Simpson's Evenness	**<0.001**	23.2	**<0.001**	7.2	**0.014**	4.1
log Chl *a*	**<0.001**	12.9	**0.021**	3.8	0.253	0.9
log Biomass	**<0.001**	15.9	**0.021**	3.6	**0.045**	2.7
NMDS1	**<0.001**	36.1	**0.001**	11.3	0.206	2.2
NMDS2	0.091	3.2	0.402	16.1	0.996	1.8

Table Notes: Significant values (i.e., p-value < 0.05) are in bold.
%VE represents the percentage of total variance explained by each predictor within the individual models
Shaded cells highlight the %VE that corresponds to significant p-values.
Data was log-transformed (base 10) where indicated.

2.3.5.3 Visualization of Trends

Temporal and spatial variations in flow-normalized algal community indices are provided in Figure 33 to Figure 40. Density, richness, diversity, biomass and NMDS axis 1 scores appear to be decreasing over time, while Simpson's Evenness, chlorophyll-*a*, and NMDS axis 2 scores appear to be increasing over time. Density, richness, chlorophyll-*a* levels, biomass and NMDS axis 1 scores also appear to be decreasing as you move further downstream, away from the proposed OSPW discharge point. Simpson's Diversity, Evenness and NMDS axis 2 scores, however, appear to be increasing as you move further downstream, away from the proposed OSPW discharge point.

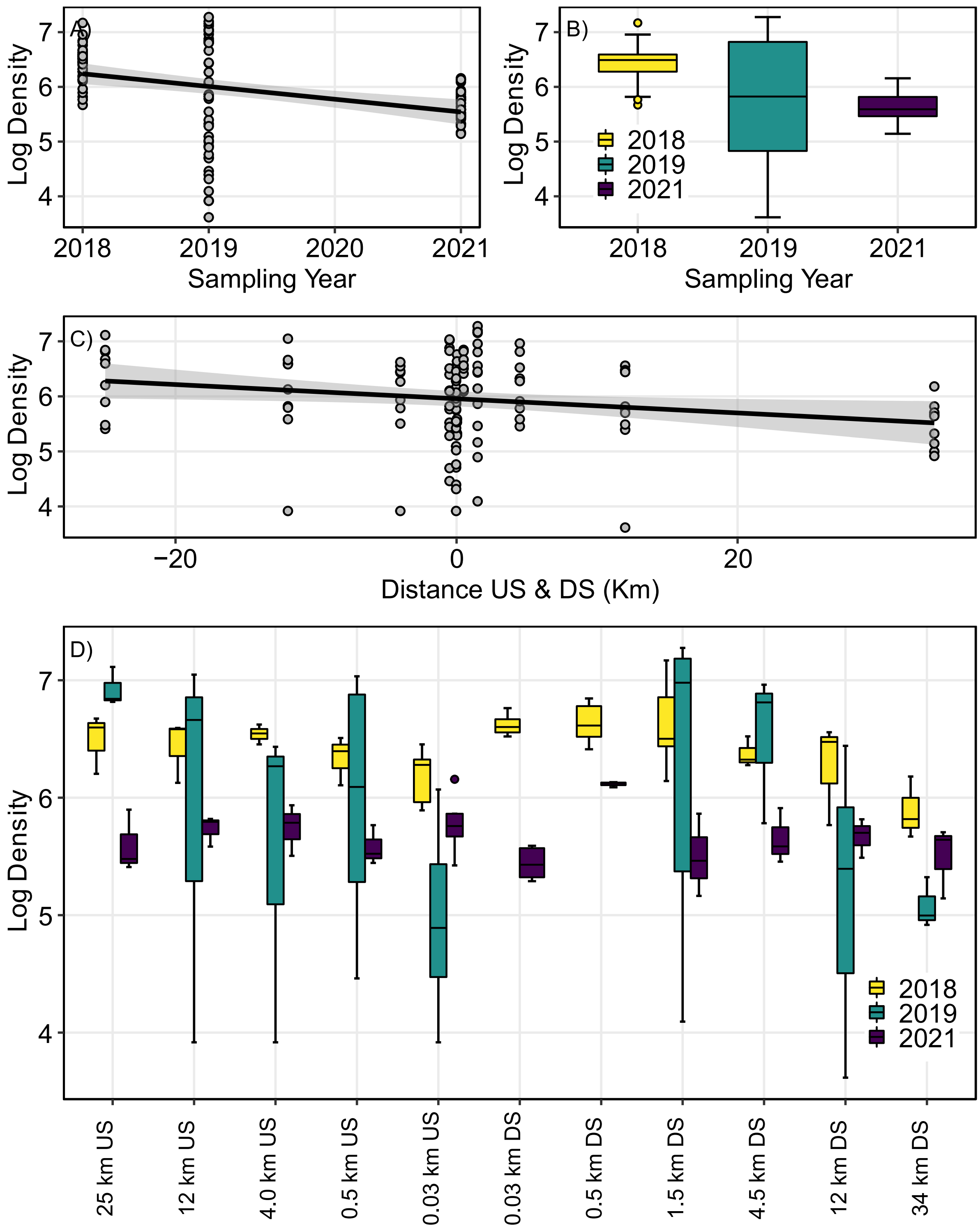

Figure 33 Variation of total algal density over time (A, B) and over distance upstream/downstream of proposed OSPW discharge point (C).

Figure Notes: Data are normalized to a Q60 of 900 m^3/s.

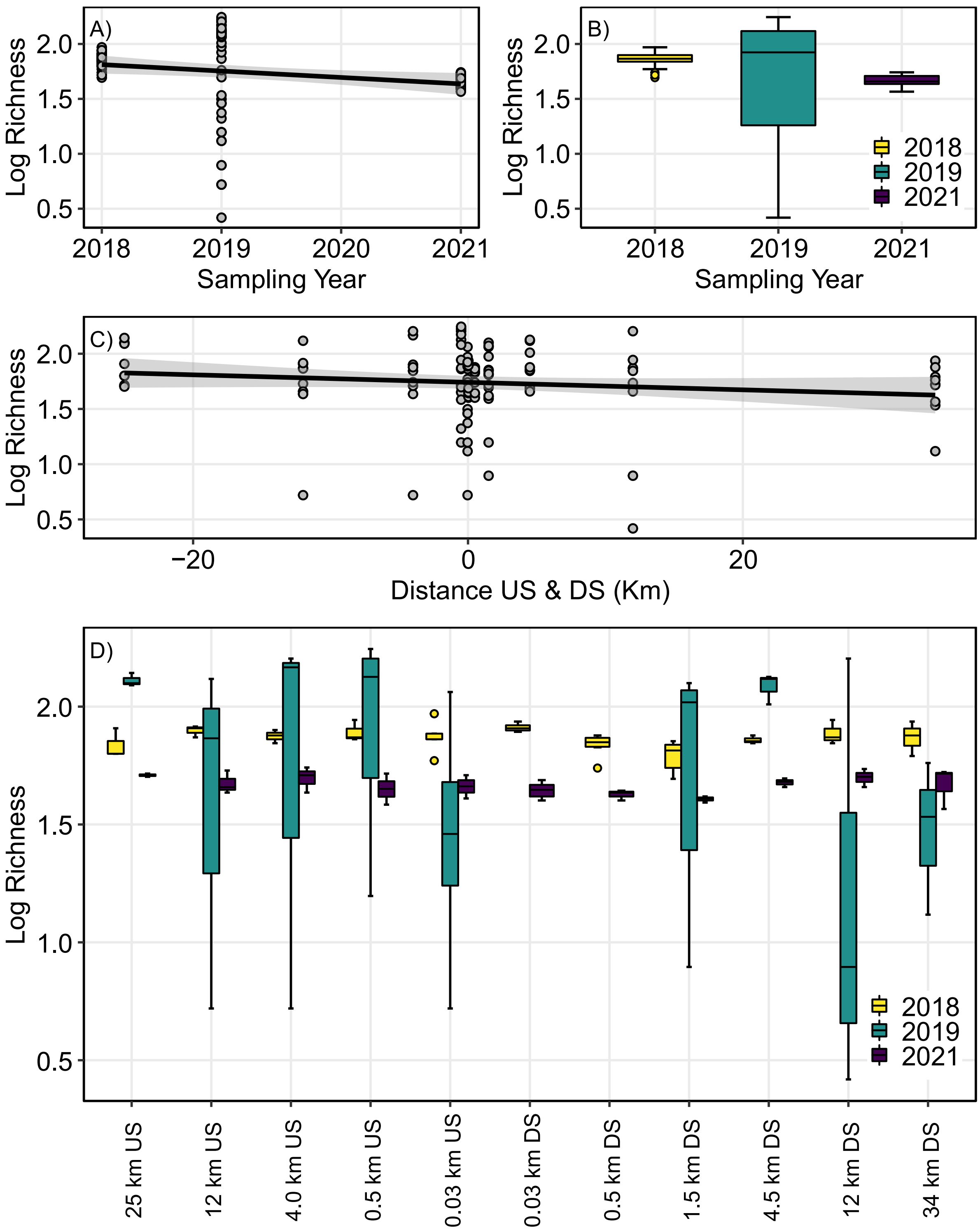

Figure 34 Variation of algal LPL richness over time (A, B) and over distance upstream/downstream of proposed OSPW discharge point (C)

Figure Notes: Data are normalized to a Q60 of 900 m^3/s.

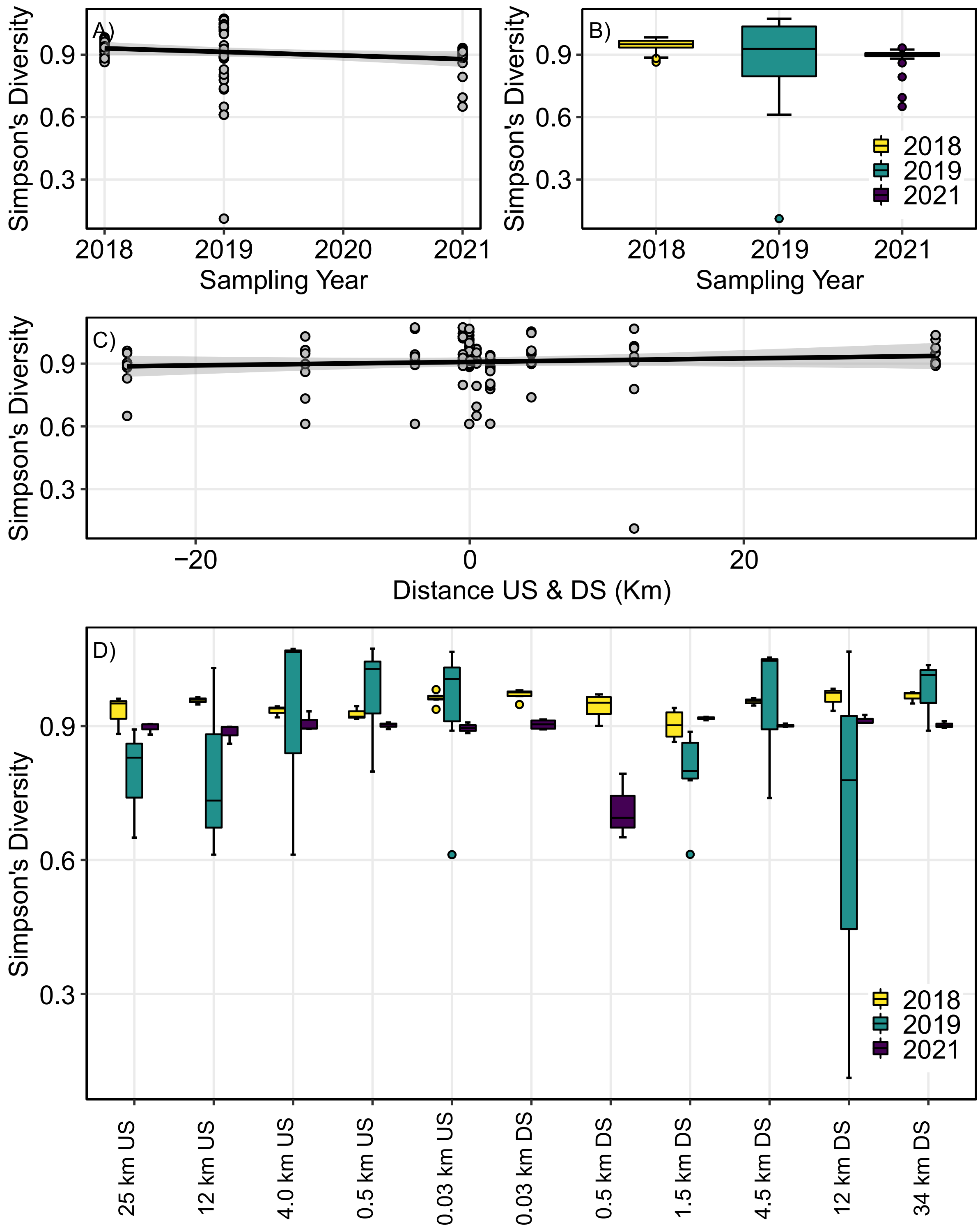

Figure 35 Variation of algal Simpson's Diversity over time (A, B) and over distance upstream/downstream of proposed OSPW discharge point (C).

Figure Notes: Data are normalized to a Q60 of 900 m^3/s.

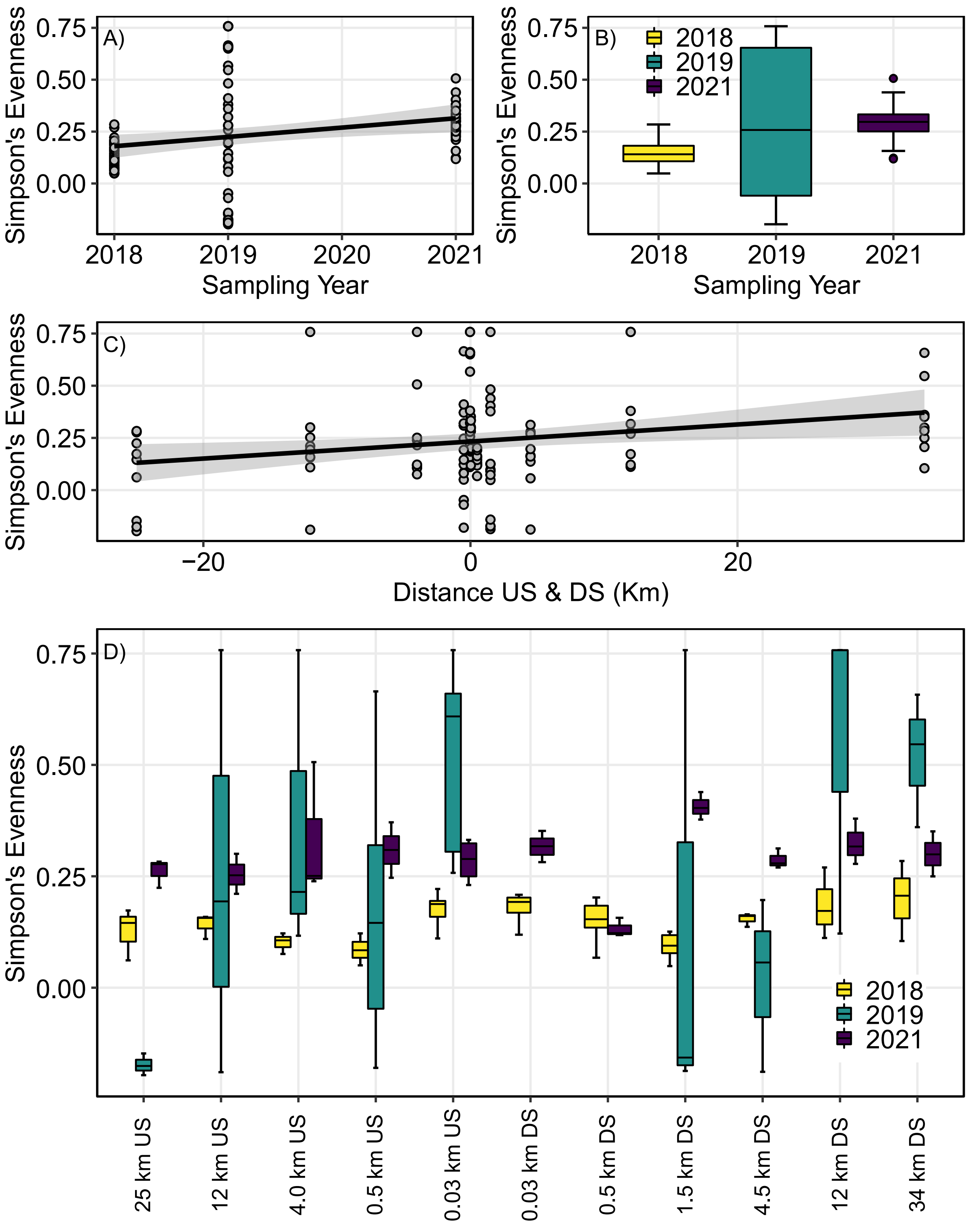

Figure 36 Variation of algal Simpson's Evenness over time (A, B) and over distance upstream/downstream of proposed OSPW discharge point (C).

Figure Notes: Data are normalized to a Q60 of 900 m^3/s.

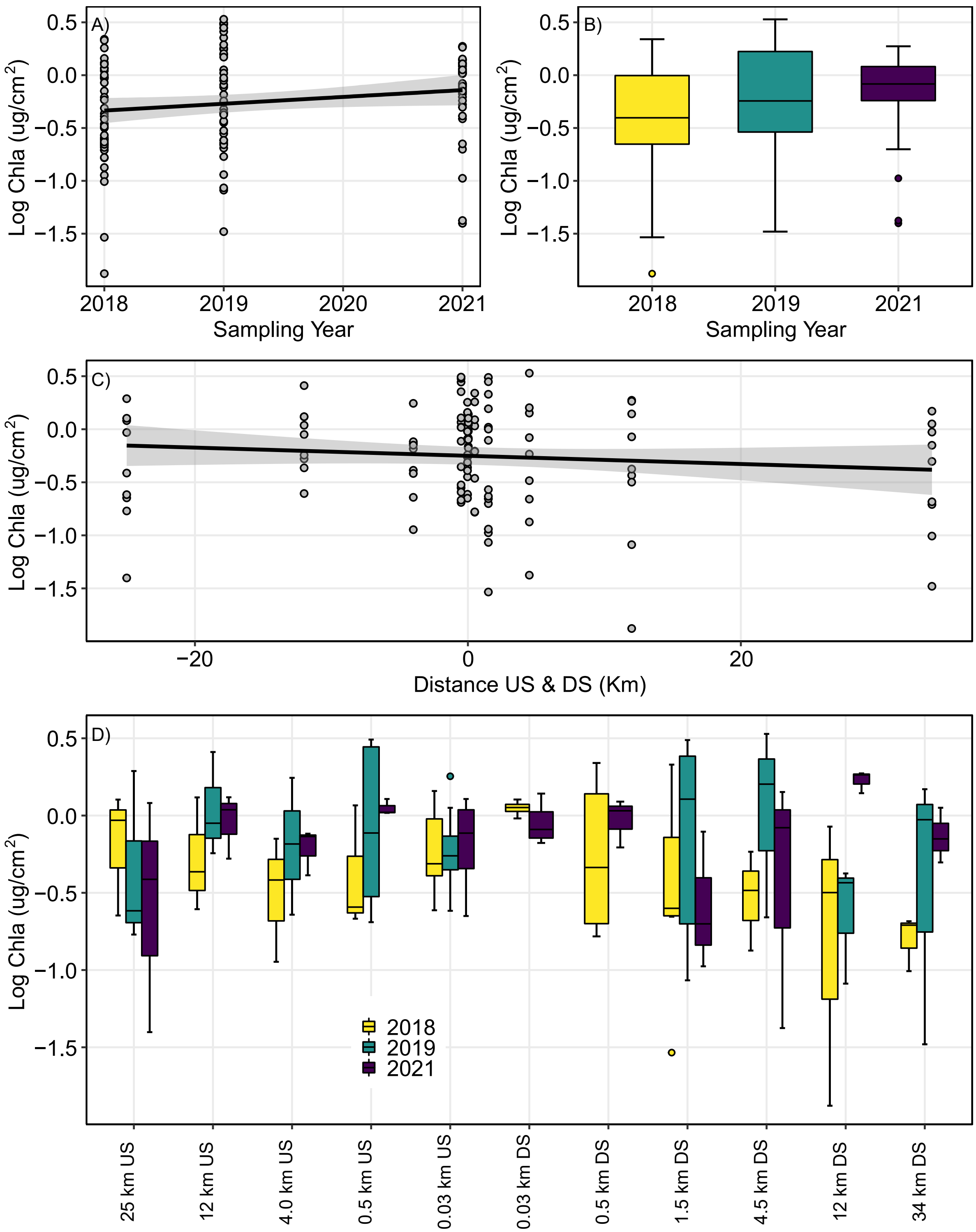

Figure 37 Variation of chlorophyll-a over time (A, B) and over distance upstream/downstream of proposed OSPW discharge point (C).

Figure Notes: Data are normalized to a Q60 of 900 m^3/s.

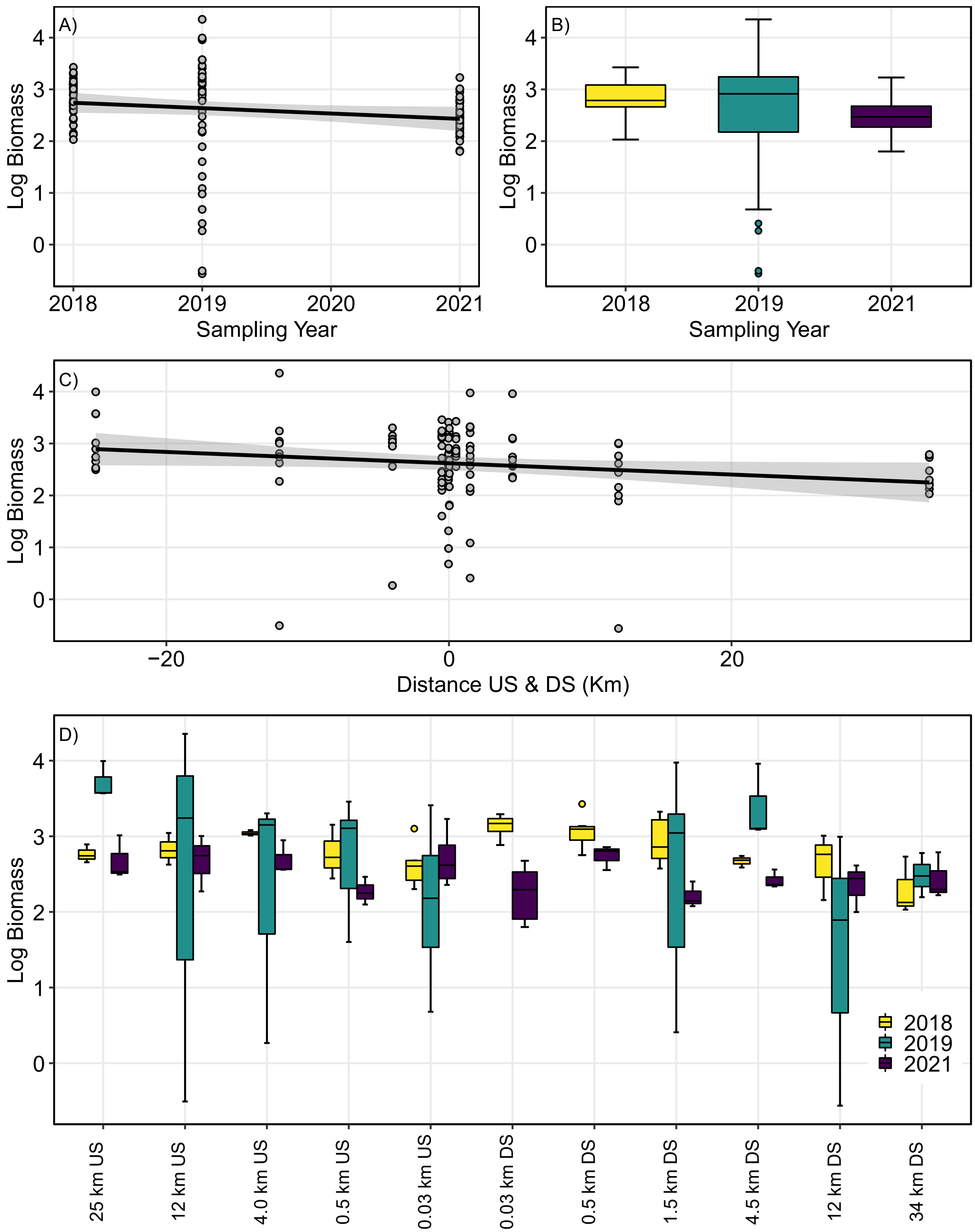

Figure 38 Variation of total algal biomass over time (A, B) and over distance upstream/downstream of proposed OSPW discharge point (C).

Figure Notes: Data are normalized to a Q60 of 900 m^3/s.

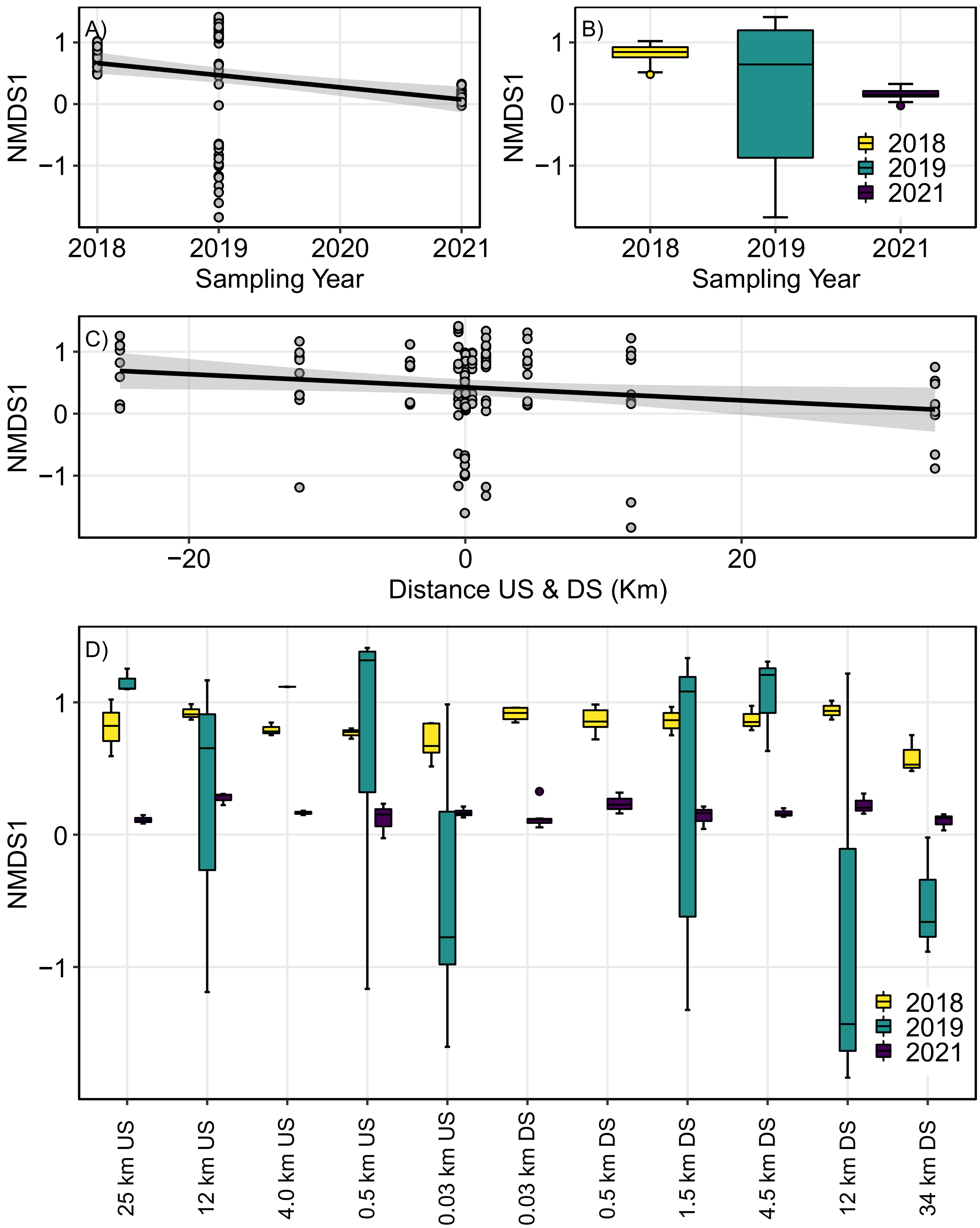

Figure 39 Variation of algal NMDS axis 1 scores over time (A, B) and over distance upstream/downstream of the proposed OSPW discharge point (C).

Figure Notes: Data are normalized to a Q60 of 900 m^3/s.

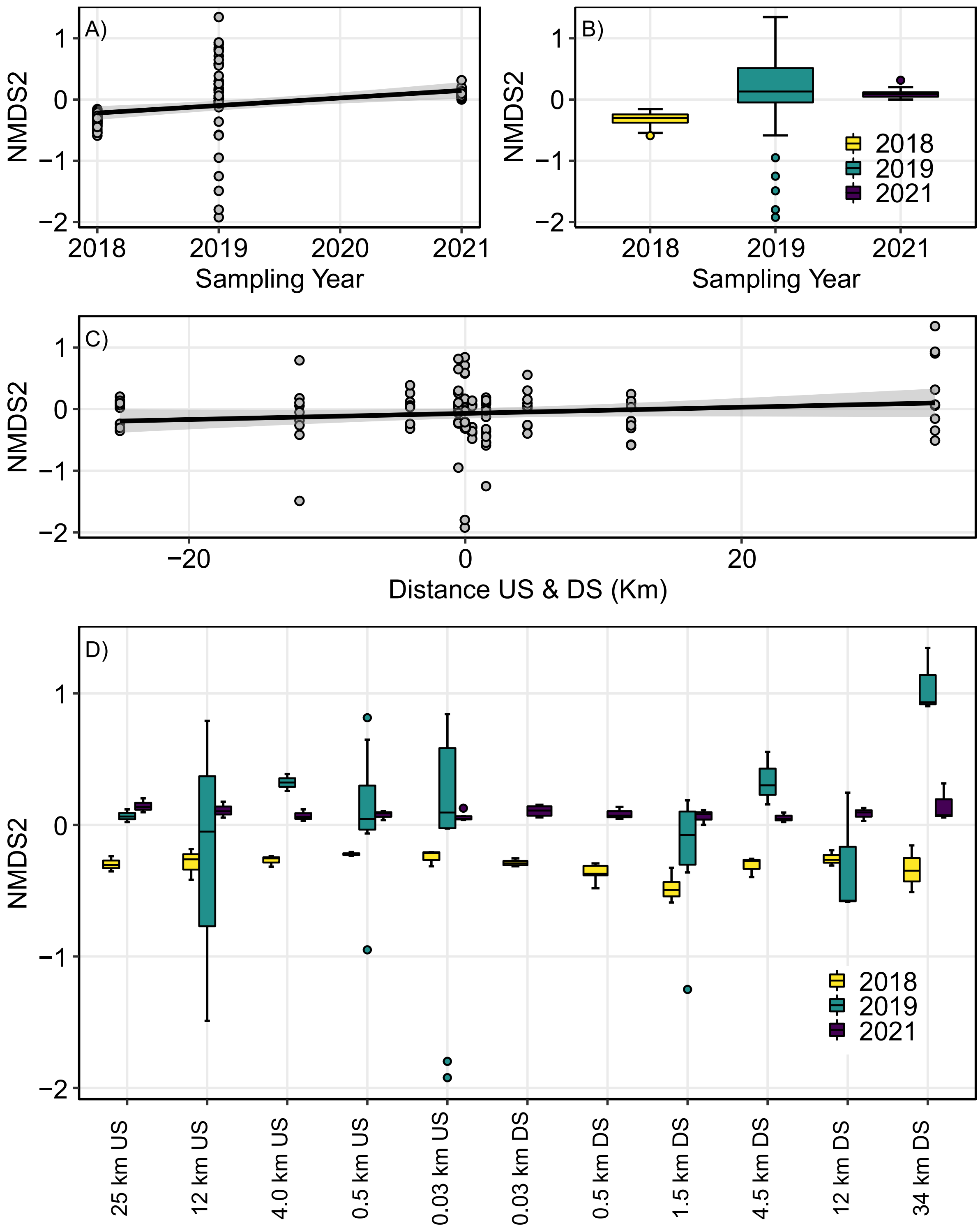

Figure 40 Variation of algal NMDS axis 2 scores over time (A, B) and over distance upstream/downstream of the proposed OSPW discharge point (C).

Figure Notes: Data are normalized to a Q60 of 900 m^3/s.

2.3.6 Benthic Invertebrate Communities

The data collected during the EMP were used to determine several effect indicators used in EEM programs (i.e., density, richness, evenness, Bray-Curtis index of similarity), along with other indices of community composition. A map of sampling locations can be found in Figure 2. Summary statistics were determined and trends, both spatial and temporal, were investigated and discussed below.

2.3.6.1 DATA SUMMARY

Benthic invertebrate communities were sampled in the fall of 2018, 2019 and 2021 from 13 stations in depositional reaches. A total of 5 samples were collected from each station using the CABIN traveling-kick methodology (Environment Canada, 2012a) for a total of 65 samples per year. Two of the stations sampled in 2018 and 2019 (i.e., 0.5 km DS WI, 1.5 km DS WI) were not sampled in 2021 since the west side of the channel had dried up, reducing the total sample size that year to N = 55.

Benthic communities were dominated by non-biting midges, the Chironomidae family, at all stations (32 to 90%), with 54 different genera identified. Nine of these genera, however, made up over 90% of the Chironomidae captured in samples: *Polypedilum* (63%, PTI 5.3), *Chironomus* (8%; PTI 8.3), *Procladius* (8%; PTI 6.3), *Monodiamesa* (4%; no PTI assigned), *Paracladopelma* (3%; PTI 6.0), *Paralauterborniella* (3%; PTI 5.3), *Stempellinella* (3%; PTI 4.7), *Cryptochromes'* (1%; PTI 6.0) and *Micropsectra* (1%, PTI 5.0). Subdominant taxa in the benthic samples included oligochaete worms Naididae (<1 to 56%), the stonefly *Perlodidae* (<1 to 10%), and the mayflies *Ametropodidae* (1 to 25%), *Metretopodidae* (<1 to 8%), *Baetidae* (<1 to 5%) and *Ephemerellidae* (<1 to 5%). The relative abundance of dominant taxa is illustrated in Figure 41 for stations located upstream and downstream of the proposed OSPW discharge point.

Summary statistics for benthic community indices are provided in Table 20 and broken down by station-year in Appendix B Table B2. Stations located upstream from the proposed OSPW discharge point had mean densities ranging between 360 individuals to 4,354 individuals, while stations located downstream had mean densities ranging between 229 individuals to 2,192 individuals. Summary statistics also highlighted potential differences in benthic indices of richness (US: 12-21, DS: 17-33), diversity (US: 0.59-0.74, DS: 0.60-0.79), and evenness (US: 0.18-0.54, DS: 0.16-0.30) when comparing stations located upstream and downstream of the potential OSPW discharge point. These were examined further in Section 2.3.5.2.

The ordination of the benthic community data is presented in Figure 42. Pearson correlations between LPL densities and sample scores on each of the NMDS axes are illustrated. An overlay of either top panel (A or B) with the lover panel (panel C) indicates which taxa were more abundant in which samples. For example, Panel A demonstrates that samples collected in 2021 have higher NMDS axis 1 scores which are associated with higher densities of Polypedilum. In Panel B, we see that samples collected from the stations 25 km U/S from the proposed OSPW discharge point had lower NMDS axis 1 scores, which are associated with higher densities of Ephemerellidae.

Spearman Rank correlations among indices and potential covariables (i.e., discharge Q60 and particle size) are provided in Table 21. There was a correlation (i.e., r > critical r of 0.14) between discharge (Q60) and all benthic indices, except for %EPT. Density (r = -0.30), richness (r = -0.15), PTI (r = -0.57), and the two NMDS axis scores (r = -0.28 and -0.43, respectively) were negatively associated with discharge, while both Simpson's Diversity (r = 0.21) and Evenness (r = 0.39) were positively associated with discharge. Similarly, there was a correlation between particle size and all benthic indices, except for Simpson's Diversity. Density (r = -0.51), richness (r = -0.24), and NMDS axis 1 scores (r = -0.56) were negatively associated with particle size, while Simpson's Evenness (r = 0.23), %EPT (r = 0.47), PTI (r = 0.20) and NMDS axis 2 scores (r = 0.42) were positively associated with particle size.

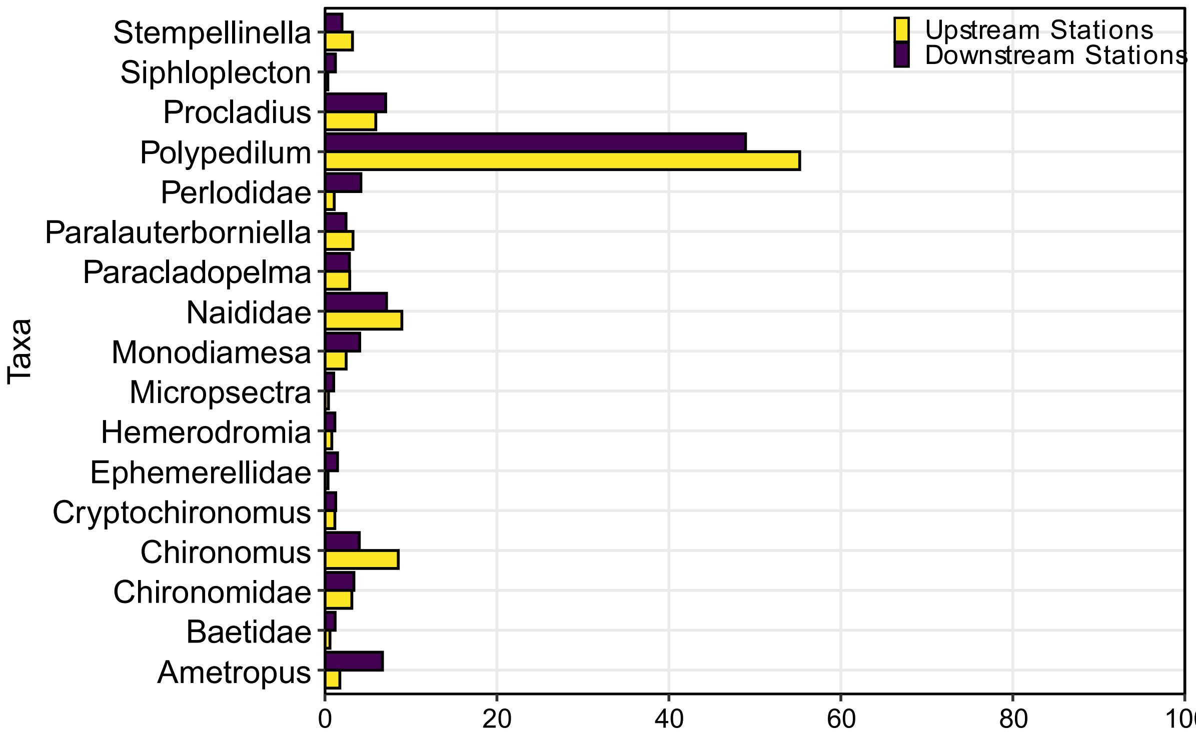

Figure 41 Relative abundance of non-rare taxa (> 0.5%) in samples collected both upstream and downstream of the proposed OSPW discharge point, EMP program (2018, 2019 and 2021)

Figure Notes: Family Chironomidae included due to high number of non-identifiable chironomids, generally consisting of early instars.

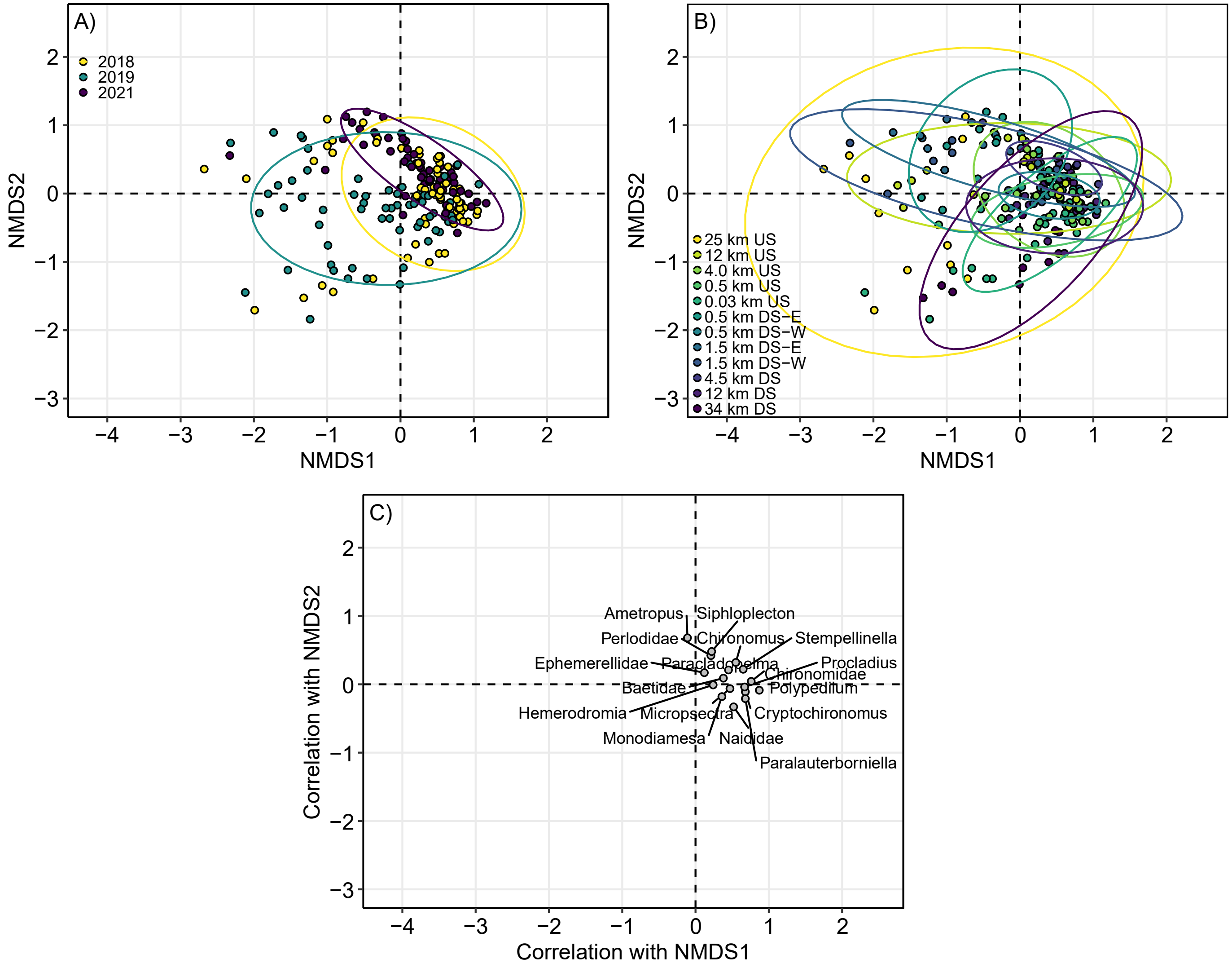

Figure 42 NMDS Axis 1 and 2 scores by year (A), sampling station (B), and correlation with Lowest Practical Taxonomic Level (LPL) (C) for the EMP dataset (2018, 2019 and 2021).

Table 20 Summary statistics of benthic indices of community for samples collected during the EMP program (2018, 2019 and 2021).

Statistic	Density	Richness	Diversity	Evenness	EPT	PTI
Min	3	2	0.00	0.08	0.0	5.02
Max	21,420	49	0.89	1.00	88.9	8.30
Mean	1,372	19	0.43	0.31	20.6	5.88
SD	2,172	9	0.22	0.19	24.7	0.71

Table 21 Spearman Rank correlations among covariables and benthic indices of community composition for samples collected during the EMP program (2018, 2019 and 2021).

Parameters		Discharge	Particle Size	Density	Richness	Simpson's Diversity	Simpson's Evenness	% EPT	PTI	NMDS1	NMDS2
Covariables	Discharge	1	-	-	-	-	-	-	-	-	-
	Particle Size	0.02	1	-	-	-	-	-	-	-	-
Benthic Indices	Density	**-0.30**	**-0.51**	1	-	-	-	-	-	-	-
	Richness	**-0.15**	**-0.24**	**0.50**	1	-	-	-	-	-	-
	Simpson's Diversity	**0.21**	0.02	-0.10	**0.41**	1	-	-	-	-	-
	Simpson's Evenness	**0.39**	**0.23**	**-0.62**	**-0.50**	**0.50**	1	-	-	-	-
	% EPT	-0.06	**0.47**	**-0.38**	0.12	0.13	0.03	1	-	-	-
	PTI	**-0.57**	**0.20**	-0.14	**-0.09**	0.05	0.11	**0.22**	1	-	-
	NMDS1	**-0.28**	**-0.56**	**0.87**	**0.46**	0.03	**-0.47**	**-0.51**	**0.91**	1	-
	NMDS2	**-0.43**	**0.42**	0.01	**0.24**	-0.03	**-0.25**	**0.70**	-0.05	**-0.16**	1

Table Notes: Values in bold exceed the critical r of 0.14, calculated as $1.96/\sqrt{N-1}$), where N is the sample size of the dataset (i.e., 185)

2.3.6.2 Explanatory Models

Results from the GLMs for the indices of benthic communities are summarized in Table 22. The models determined that discharge for the 60-day period prior to benthic sampling (Q60) was a significant predictor (p-value < 0.05) of variations in all indices of benthic invertebrate community, except for Simpson's Diversity and % EPT. Similarly, the models determined that substrate particle size was a significant predictor of variations in all indices of benthic invertebrate community, except for Simpson's Diversity. Discharge (Q60) explained a significant amount of variation (i.e., > 20%) for PTI (31%), while substrate particle size explained a significant amount of variation for NMDS axis 1 scores.

Linear trends over time were statistically significant for PTI only. Statistically significant variations associated with the distance from the proposed OSPW discharge point were present for Simpson's Evenness, PTI, and NMDS axis 1 and 2 scores.

Temporal and spatial differences were further investigated below in Section 2.3.6.3 on both flow and particle size normalized indices of benthic community. Normalization equations are provided in Appendix C Table C3.

Table 22 Significance (p-value) and percent of variance (%VE) explained for predictors of benthic communities for samples collected in the Lower Athabasca River, EMP (2018, 2019, 2021).

Index	Q60 (m^3/s)		Particle Size (mm)		Year		Distance (US/DS)	
	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE
log Abundance	**<0.001**	9.6	**<0.001**	16.4	0.684	0.1	0.217	0.6
log Richness	**0.005**	4.1	**0.004**	4.3	0.551	0.2	0.230	0.7
Simpson's Evenness	**<0.001**	11.5	**0.002**	4.4	0.575	0.1	**0.010**	3.1
Simpson's Diversity	0.046	2.1	0.826	0.0	0.115	1.3	0.126	1.3
log EPT	0.356	0.4	**<0.001**	16.9	0.518	0.2	0.499	0.2
log PTI	**<0.001**	31.2	**<0.001**	4.4	**<0.001**	4.8	**<0.001**	5.7
NMDS1	**<0.001**	8.1	**<0.001**	24.0	0.205	0.6	**0.006**	2.8
NMDS2	**<0.001**	16.1	**<0.001**	11.6	0.256	0.5	**0.039**	1.7

Table Notes: Significant values (i.e., p-value < 0.05) are in bold.
%VE represents the percentage of total variance explained by each predictor within the individual models.
Shaded cells highlight the %VE that corresponds to significant p-values.
Data was log-transformed (base 10) where indicated.

2.3.6.3 Visualization of Trends

Temporal and spatial variations in flow- and particle size-normalized benthic community indices are provided in Figure 43 to Figure 50.

Density, Simpson's Evenness, Simpson's Diversity and PTI appear to be increasing over time, while NMDS axis 1 and 2 scores appear to be decreasing over time. Once taking into account the influence of both discharge (Q60) and particle size, there appears to be no temporal variation in richness or % EPT.

Density, richness, and NMDS axis 1 scores also appear to be increasing as you move further downstream, away from the proposed OSPW discharge point. Simpson's Evenness, Simpson's Diversity, %EPT and PTI appear to be decreasing as you move further downstream, away from the proposed OSPW discharge point. Once taking into account the influence of both discharge (Q60) and particle size, there appears to be no spatial variation in NMDS axis 2 scores.

An important spatial comparison to consider is the difference between samples collected on the east and west side of the island located 0.5 km downstream of the proposed OSPW discharge point, as these stations are directly downstream of the Syncrude sewage treatment outfall (Figure 1). Differences among samples collected from the east and west side of the island were explored using a Tukey's post-hoc test for individual comparisons. Results are provided in Table 23. There were no statistically significant differences in any of the benthic indices of community composition between the east and west sides of the island in either 2018 or 2019. Data from 2021 were not included in the analysis as the West side of the island was dry.

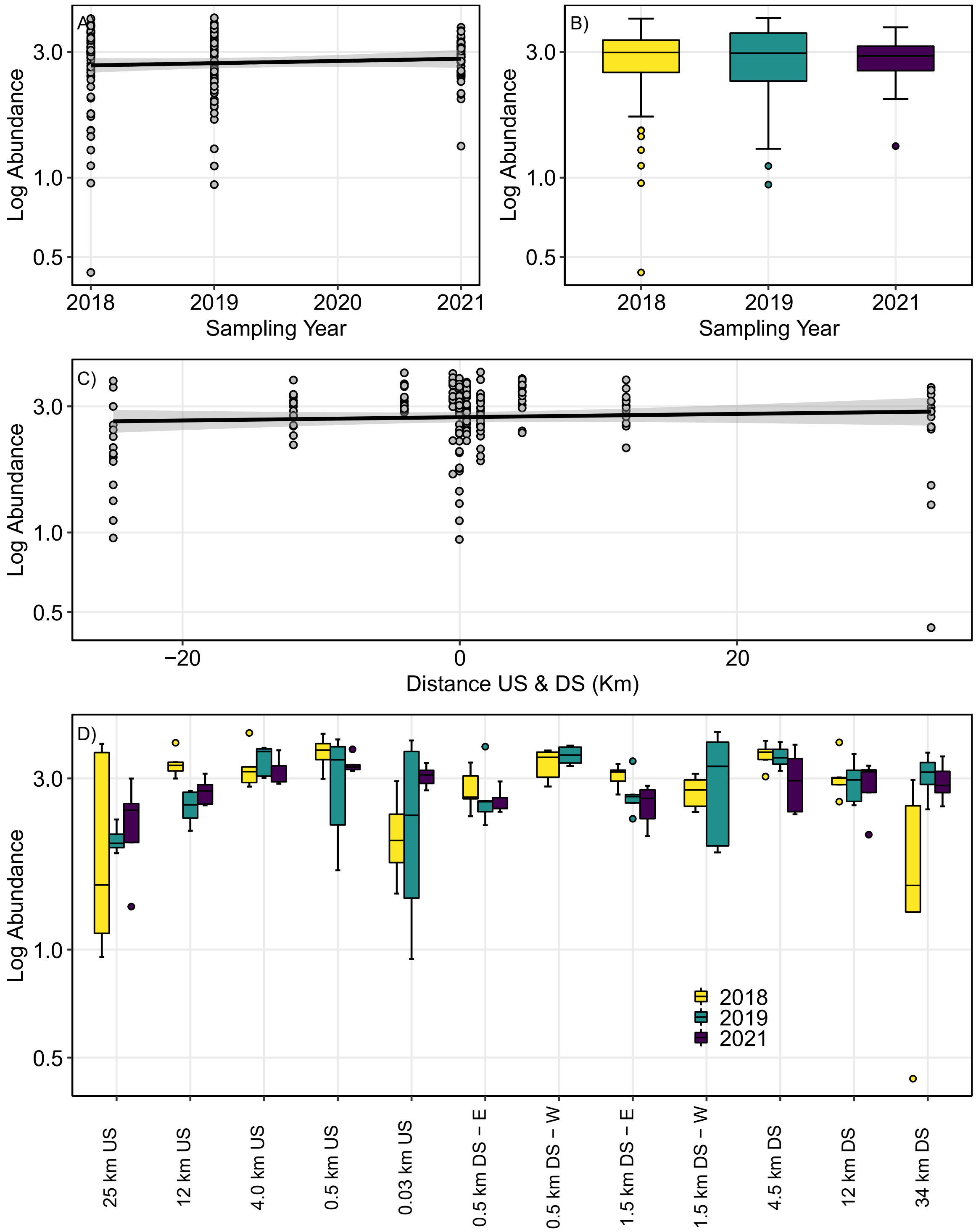

Figure 43 Variation of total benthic density (log-transformed) over time (A-linear trend, B-ANOVA), over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 900 m^3/s and particle size of 0.7 mm.

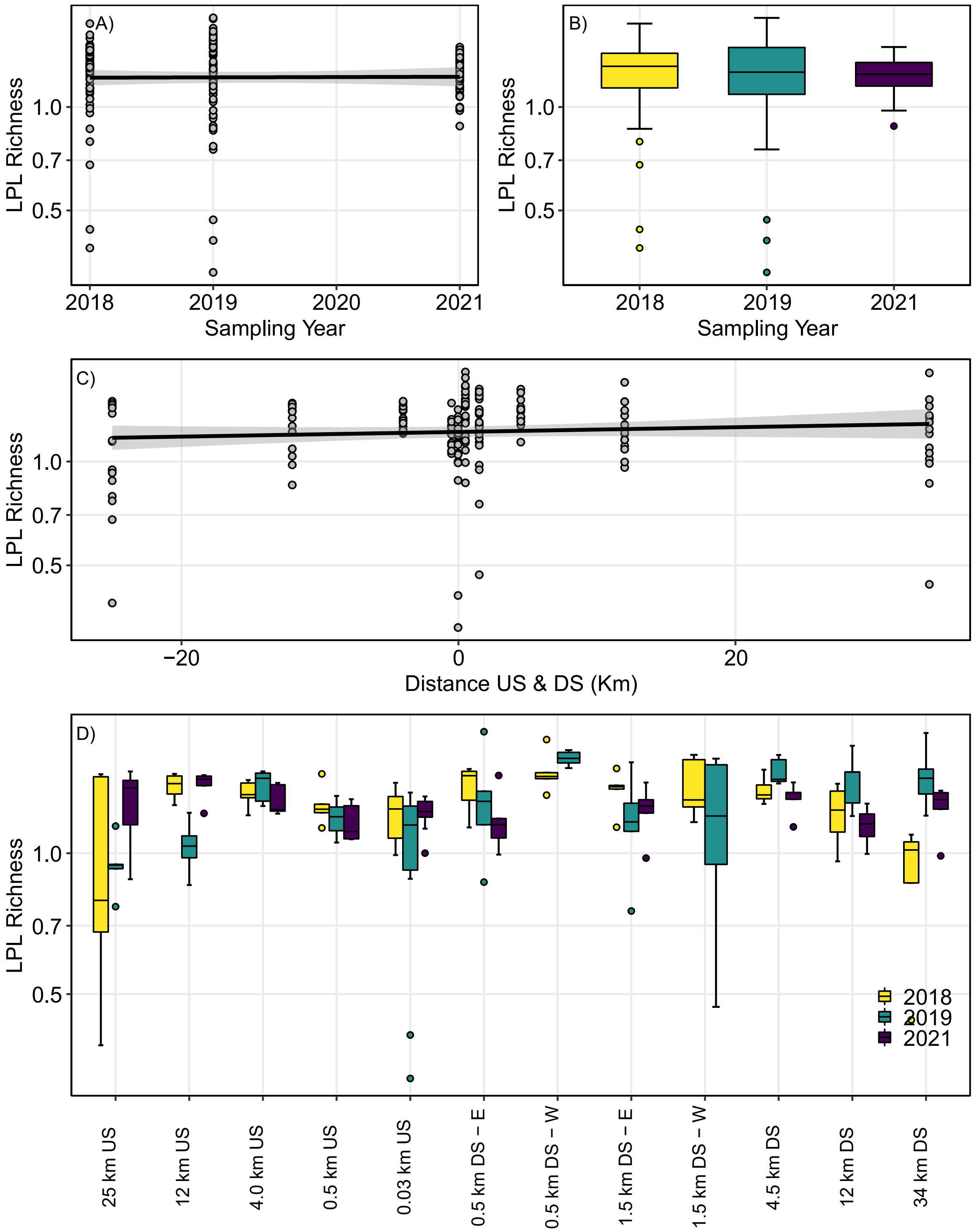

Figure 44 Variation of LPL richness (log-transformed) over time (A-linear trend, B-ANOVA) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 900 m^3/s and particle size of 0.7 mm.

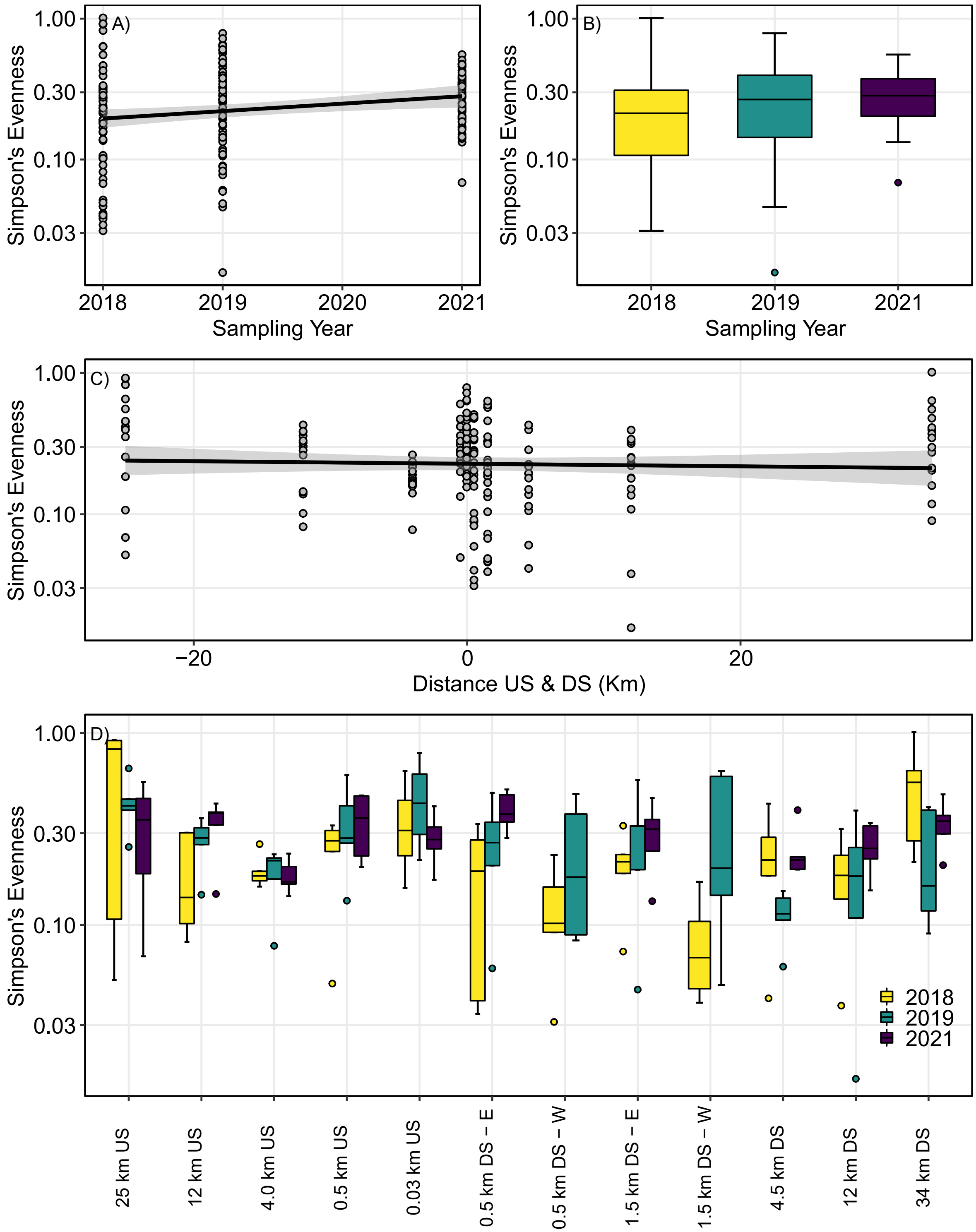

Figure 45 Variation of Simpson's Evenness over time (A-linear trend, B-ANOVA) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 900 m^3/s and particle size of 0.7 mm.

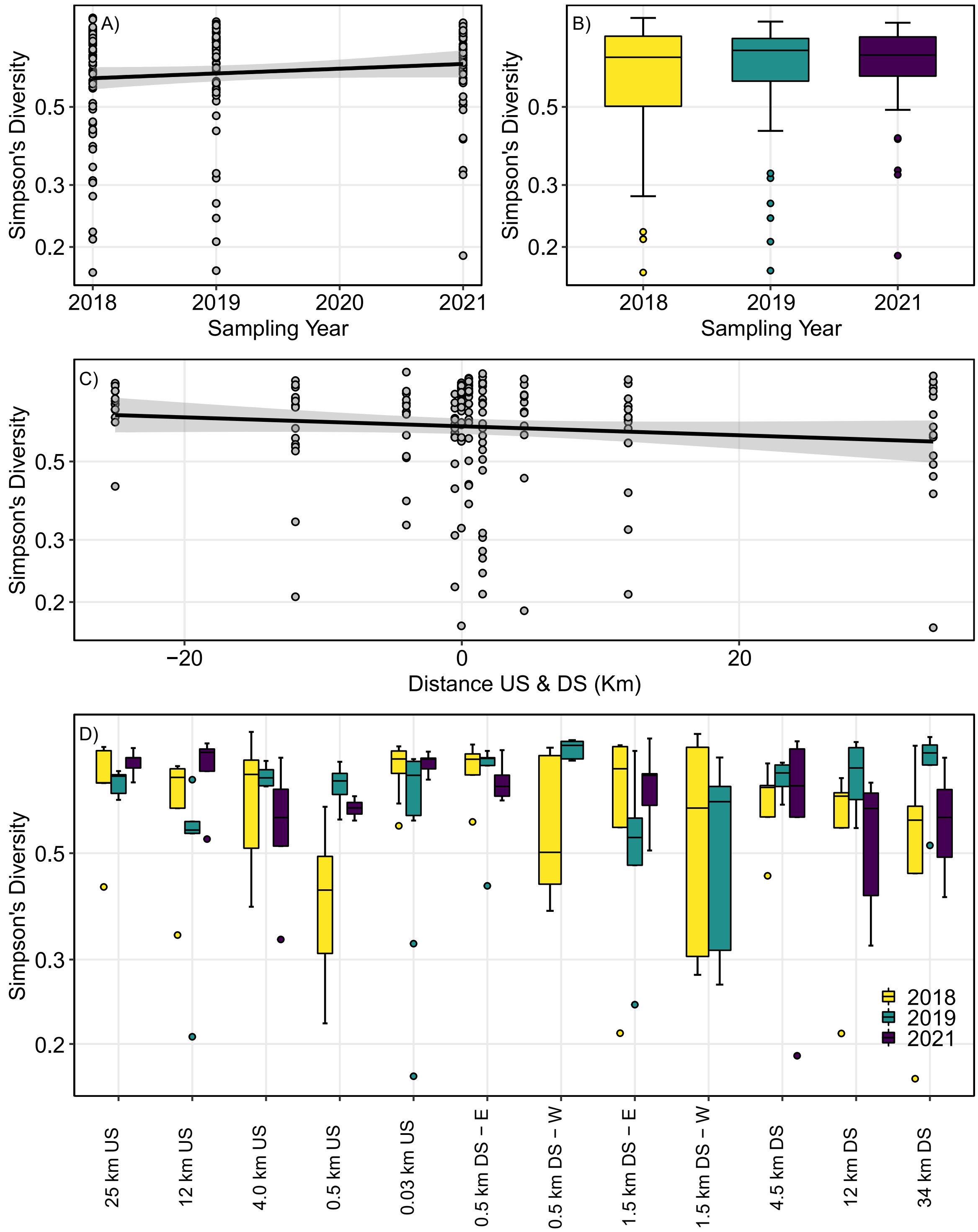

Figure 46 Variation of Simpsons' Diversity over time (A-linear trend, B-ANOVA) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 900 m^3/s and particle size of 0.7 mm.

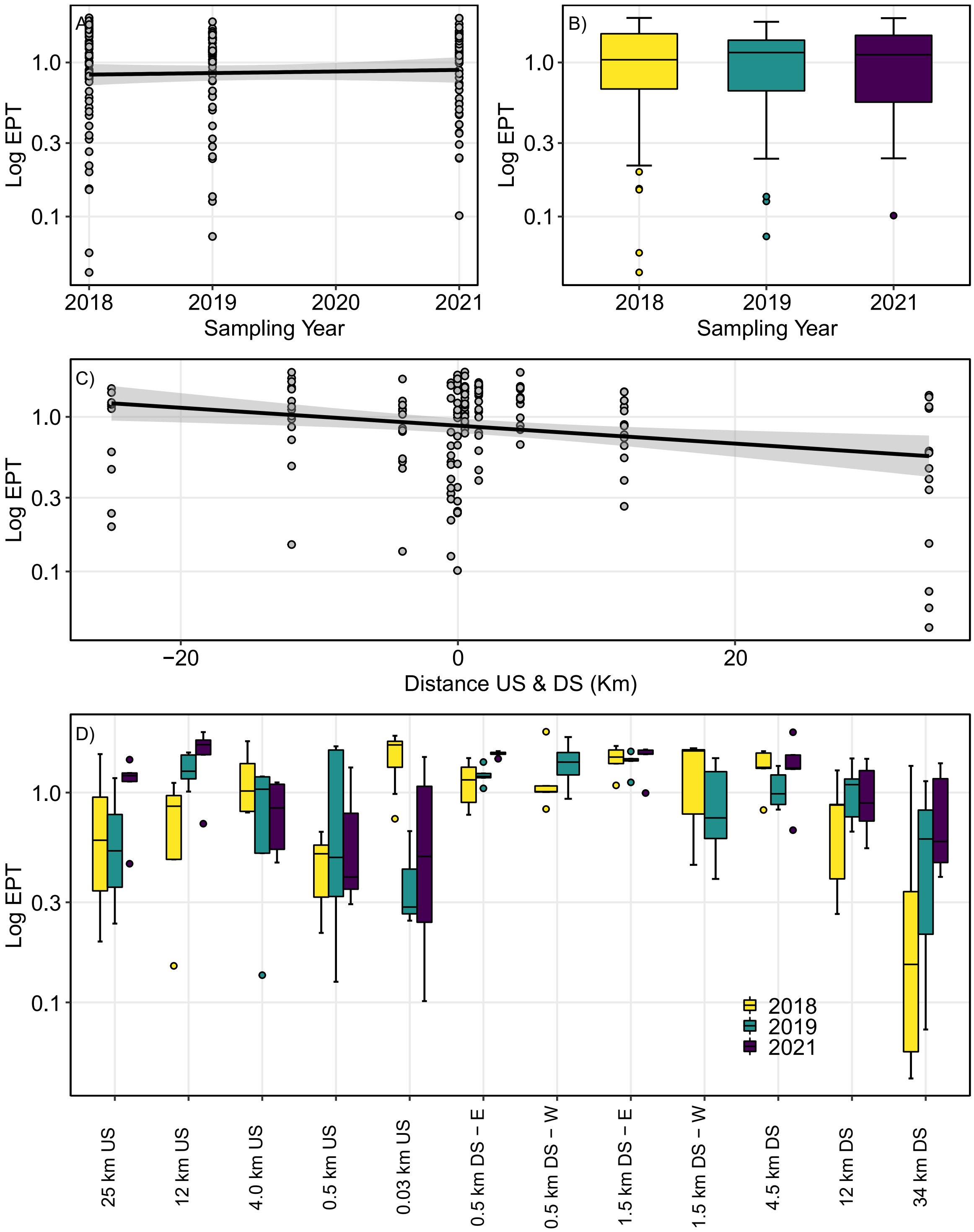

Figure 47 Variation of %EPT (log-transformed) over time (A-linear trend, B-ANOVA) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 900 m^3/s and particle size of 0.7 mm.

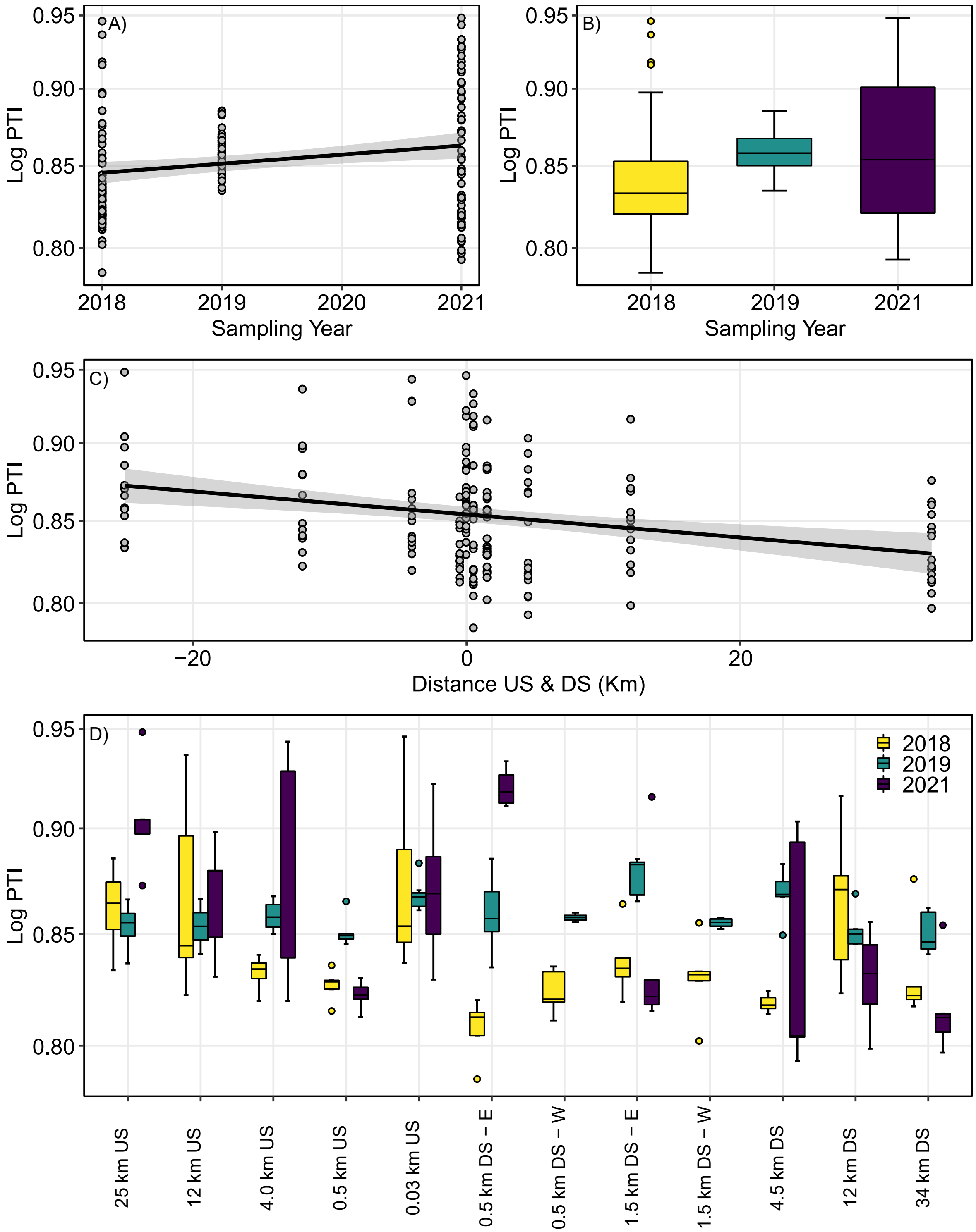

Figure 48 Variation of PTI (log-transformed) over time (A-linear trend, B- ANOVA) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 900 m^3/s and particle size of 0.7 mm.

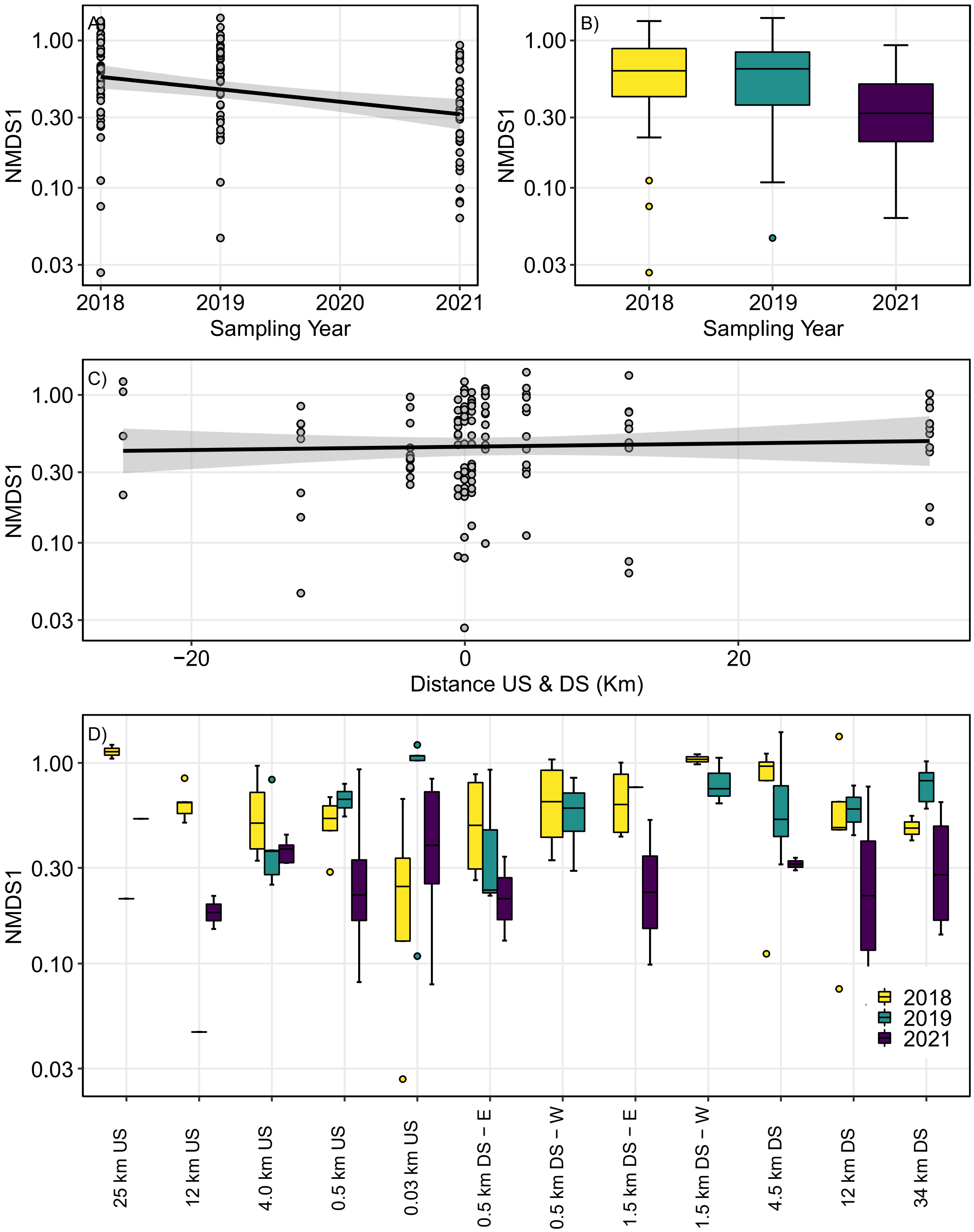

Figure 49 Variation of NMDS1 scores over time (A-linear trend, B- ANOVA) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 900 m^3/s and particle size of 0.7 mm.

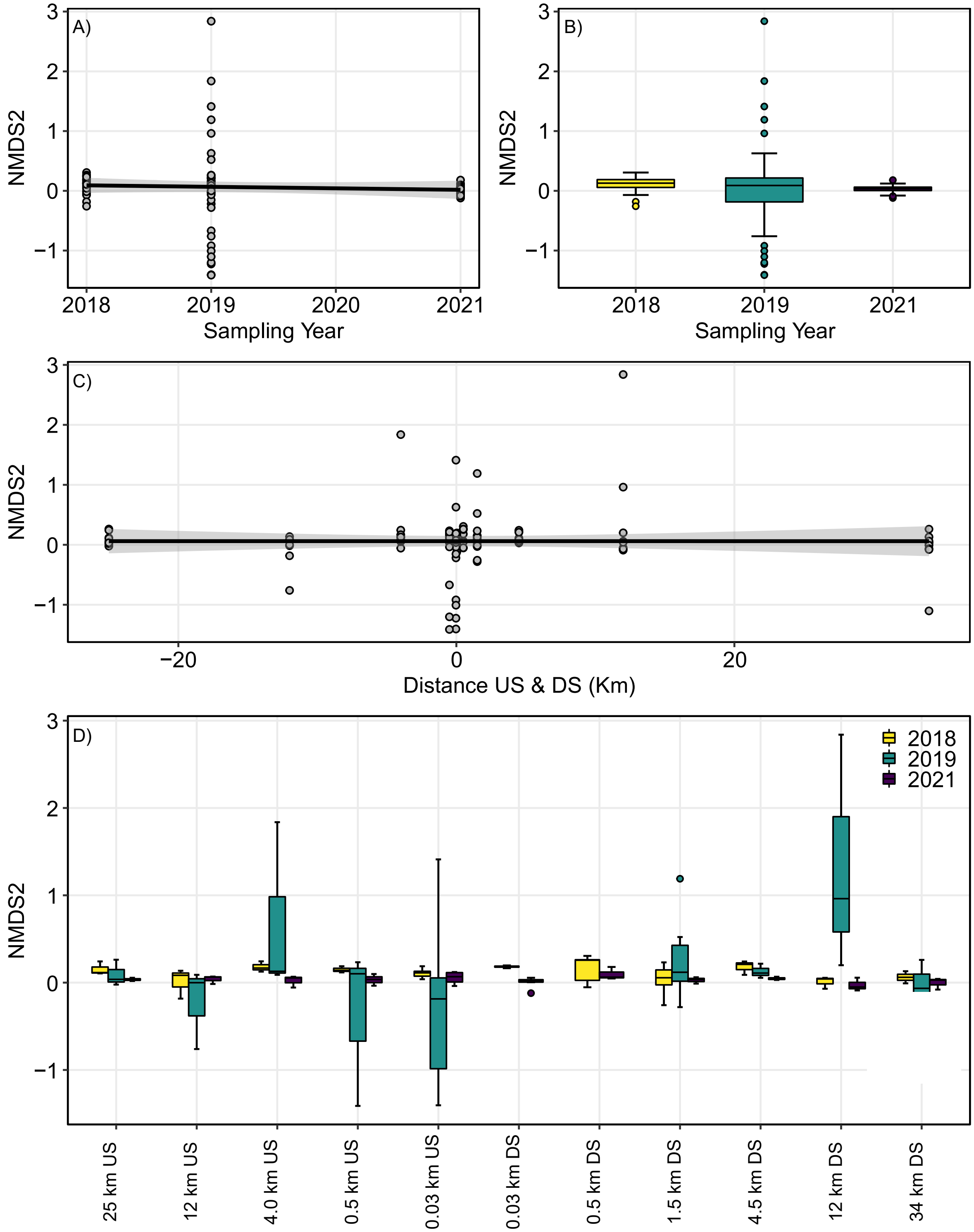

Figure 50 Variation of NMDS2 scores over time (A-linear trend, B- ANOVA) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 900 m^3/s and particle size of 0.7 mm.

Table 23 Tukey's post-hoc test comparing flow and particle size normalized benthic indices for samples collected at the stations located both East and West of the island, EMP dataset (2018 and 2019).

Index	0.5 km DS - E vs. 0.5 km DS - W			
	2018		2019	
	Direction	P-value	Direction	P-value
Log Density	E < W	0.973	E < W	0.791
Log Richness	E < W	0.999	E < W	0.794
Evenness	E > W	1	E > W	1
Simpson's Diversity	E > W	0.968	E < W	0.993
Log EPT	E < W	1	E < W	1
Log PTI	E < W	0.997	E > W	1
NMDS1	E < W	1	E < W	1
NMDS2	E > W	0.912	E < W	0.878

2.3.7 Fish Community Assessment

The EMP produced boat electrofishing community data that included components such as collection date, site, species, sex, length, body weight, transect information, and total shocking time. The data collected during the EMP were used to determine several effect indicators (i.e., abundance, richness, evenness, Bray-Curtis index of similarity), along with other indices of community composition. Summary statistics were determined and trends, both spatial and temporal, were investigated and discussed below.

2.3.7.1 DATA SUMMARY

Fish communities were sampled in the September of 2018, 2019 and 2021 from 8 stations in the LAR (Table 24). A total of 483 fish across 15 different species were collected in 2018, 879 fish across 10 different species were collected in 2019, and 866 fish across 14 different species were collected in 2021.The catch per unit effort (CPUE) ranged from 2.41 to 5.01 fish/min and was 2.83 fish/min for the entire program across all three sampling years.

The relative abundance of the different fish species is illustrated in Figure 51 for stations located upstream and downstream of the proposed OSPW discharge point. The most abundant species caught at both upstream and downstream sites was the Goldeye, followed by Emerald Shiner. Flathead Chub, Lake Whitefish, and Trout Perch all had similar relative abundances at both upstream and downstream sites. Less abundant species include Yellow Perch, White Sucker, Walleye, Spottail Shiner, Spoonhead Sculpin, Northern Pike, Longnose Sucker, Lake Chub, Cisco, and Burbot.

Summary statistics for fish community indices are provided in Table 24 and Table 25 and are broken down by station in Appendix B Table B3. Overall, stations located upstream produce similar fish community indices as stations located downstream. Stations located upstream from the proposed OSPW discharge point had abundances ranging between 20 individuals to 157 individuals, while stations located downstream had abundances ranging between 20 individuals to 159 individuals. Stations located upstream from the proposed OSPW discharge point had Simpson’s Evenness values ranging between 0.24 and 0.72, while stations located downstream ranged from 0.18 to 0.75. Species Richness values ranged from 4 to 13 across the upstream stations and 3 to 12 across the downstream stations. Finally, Simpson’s Diversity values ranged from 0.33 to 0.88 upstream and 0.45 to 0.84 downstream.

The ordination of the fish community data is presented in Figure 52. Pearson correlations between abundances and sample scores on each of the NMDS axes are illustrated. An overlay of either top panel (A or B) with the lover panel (panel C) indicates which fish species were more abundant in which samples. For example, Panel A demonstrates that samples collected in 2019 have higher NMDS axis 1 scores which are associated with higher abundances of Goldeye and Lake Whitefish. In Panel B, we see an even spread of samples across the two NMDS axis, suggesting there are little differences in community assemblages across the different sampling stations.

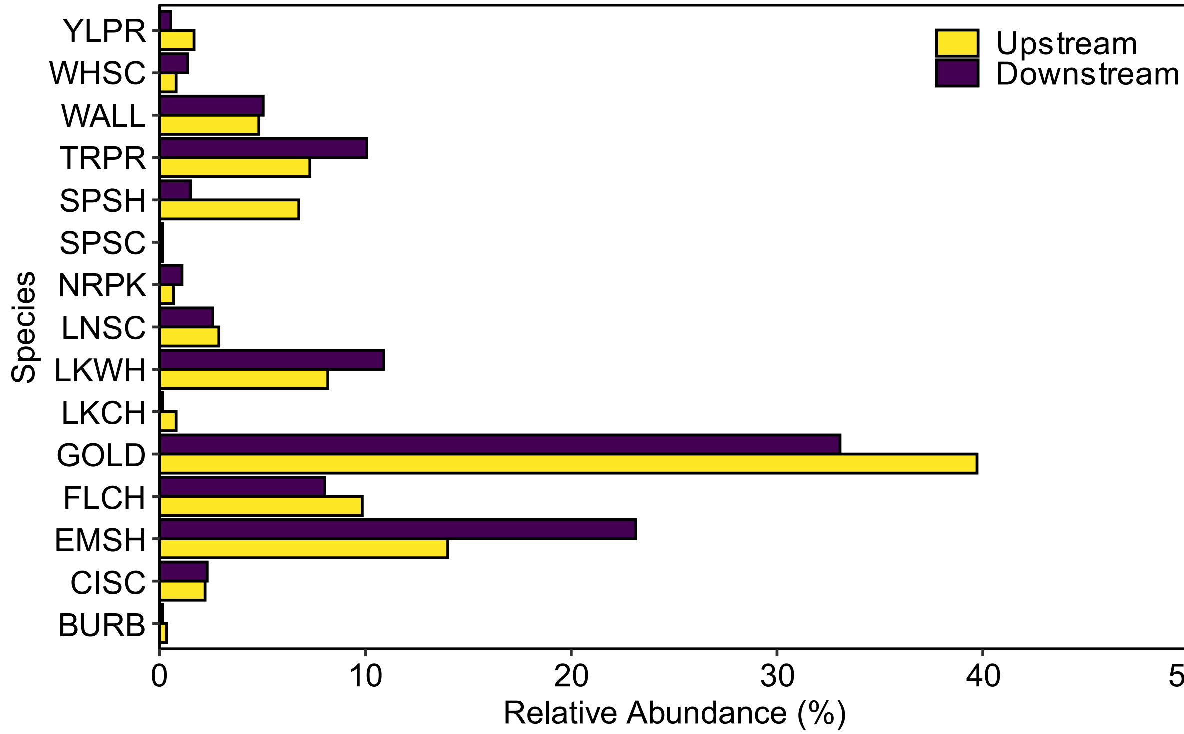

Figure 51 Relative abundance of fish species collected both upstream and downstream of the proposed OSPW discharge point, EMP program (2018, 2019 and 2021)

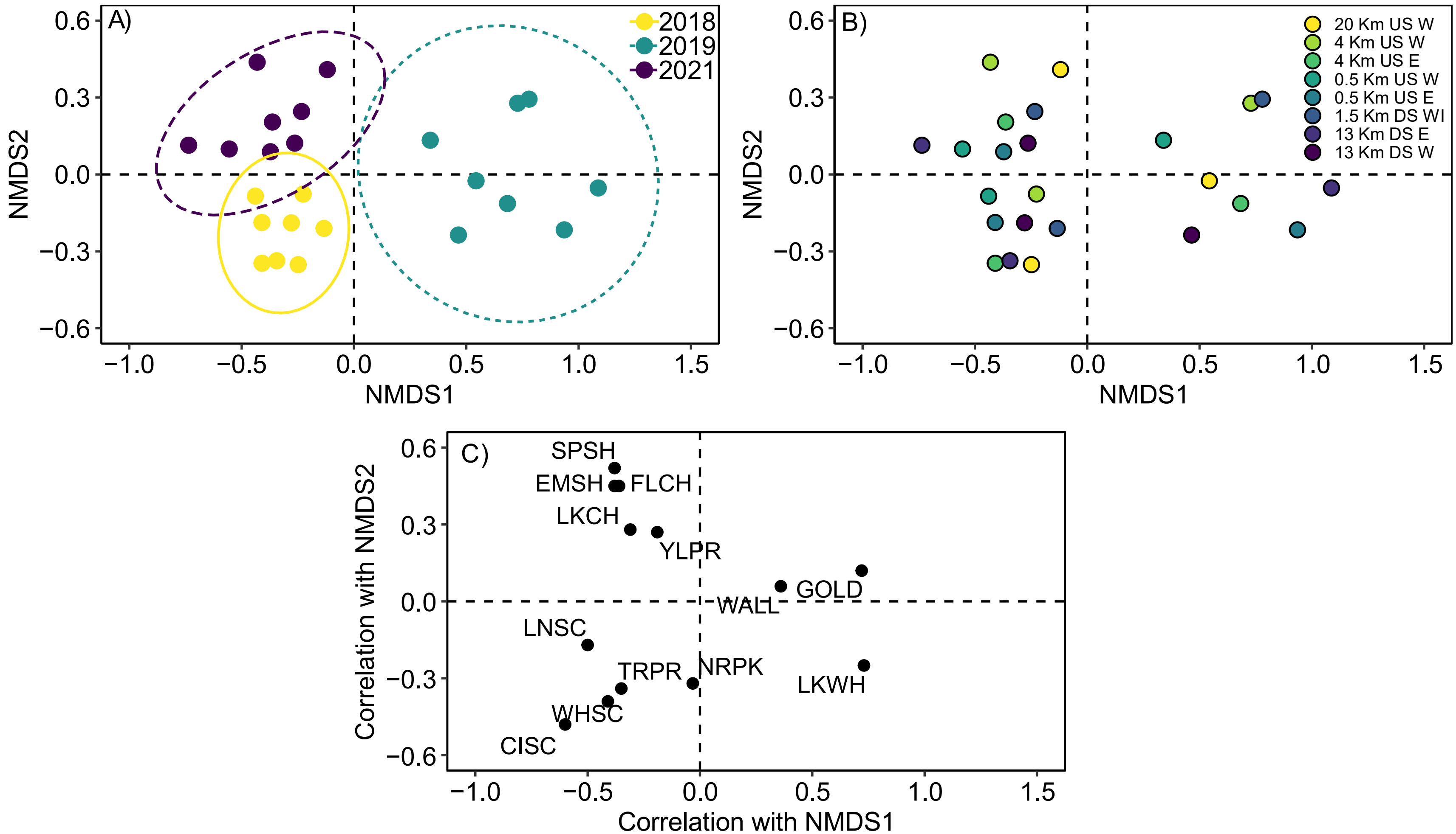

Figure 52 NMDS Axis 1 and 2 scores by year (A), sampling station (B), and correlation with fish species (C) for the EMP dataset (2018, 2019 and 2021)

Figure Note: Fish abbreviations are provided in Table 24

Table 24 Fish community catch assemblage in the LAR under EMP over three years (2018, 2019, and 2021)

Common name	Species Abbrv.	Scientific name	2018	2019	2021	Total
Burbot	BURB	*Lota lota*	2		4	
Cisco	CISC	*Coregonus artedii*	36		14	
Emerald Shiner	EMSH	*Notropis atherinoides*	45	3	331	379
Flathead Chub	FLCH	*Platygobio gracilis*	68	45	93	206
Goldeye	GOLD	*Hiodon alsoides*	88	574	174	836
Lake Chub	LKCH	*Couesius plumbeus*	6	1	6	13
Lake Whitefish	LKWH	*Coregonus clupeaformis*	22	174	6	202
Longnose Sucker	LNSC	*Catostomus catostomus*	40		22	
Northern Pike	NRPK	*Esox lucius*	8	7	3	18
Spoonhead Sculpin	SPSC	*Cottus ricei*	1			
Spottail Shiner	SPSH	*Notropis hudonius*	1		111	
Trout Perch	TRPR	*Percopsis omiscomaycus*	114	23	46	183
Walleye	WALL	*Sander vitreus*	34	50	25	109
White Sucker	WHSC	*Catostomus commersoni*	16	1	5	22
Yellow Perch	YLPR	*Perca flavescens*	2	1	26	29
Catch Summary Statistics						
Number of Species			15	10	14	15
Total Catch			483	879	866	1997
Effort (minutes)			200.1	331.7	172.8	704.6
CPUE (fish/min)			2.41	2.65	5.01	2.83

Table 25 Summary statistics of fish indices of community collected during the EMP program (2018, 2019 and 2021)

Location	Index	Min	Max	Mean	Sd
US	Abundance	20	157	99.53	41.17
	Evenness	0.24	0.72	0.49	0.15
	Richness	4	13	8.67	2.74
	Diversity	0.33	0.88	0.70	0.18
DS	Abundance	20	159	81.67	47.86
	Evenness	0.18	0.75	0.50	0.22
	Richness	3	12	8.11	3.10
	Diversity	0.45	0.84	0.67	0.15

2.3.7.2 Explanatory Models

Results from the GLMs for the indices of fish communities are summarized in Table 24. The models determined that discharge for the 60-day period prior to efishing (Q60) was a significant predictor (p-value < 0.05) of variations in Simpson’s Diversity, species richness, and NMDS axis 1 and 2 scores as well, but not the other indices. Sampling year was a significant predictor of variation in all indices except for abundance. Both distances upstream and/or downstream and sampling effort (i.e., efishing time) were not significant for any of the indices of fish community. Q60 explained a significant amount of variation (i.e., > 20%) for NMDS axis 1 scores (75.7%), species richness (56.6%), and diversity (44.1%), while sampling year explained a significant amount of variation for NMDS axis 2 scores (33.1%) and Simpson's Diversity (26.1%).

Temporal and spatial differences were further investigated below in Section 2.3.7.3 on flow normalized indices of fish community composition. Normalization equations are provided in Appendix C Table C4.

Table 26 Significance (p-value) and percent of variance (%VE) explained for predictors of fish communities caught in the Lower Athabasca River, EMP (2018, 2019, 2021).

Index	Q60 (m3/s)		Year		Distance (US/DS)		Effort (sec)	
	P-Val	%VE	P-Val	%VE	P-Val	%VE	P-Val	%VE
Log Abundance	0.796	0.3	0.073	13.8	0.350	3.6	0.132	9.5
Simpson's Evenness	0.940	0.0	**0.048**	16.2	0.323	4.2	0.109	10.3
Simpson's Diversity	**<0.001**	44.1	**<0.001**	26.1	0.468	1.4	0.649	0.3
Species Richness	**<0.001**	56.6	**0.010**	11.5	0.766	0.4	0.087	4.6
NMDS1	**<0.001**	75.7	**0.001**	11.3	0.833	0.2	0.461	0.4
NMDS2	**0.035**	13.0	**0.002**	33.1	0.183	5.0	0.608	0.7

Table Notes: Significant values (i.e., p-value < 0.05) are in bold.
%VE represents the percentage of total variance explained by each predictor within the individual models.
Shaded cells highlight the %VE that corresponds to significant p-values.
Data was log-transformed (base 10) where indicated

2.3.7.3 Visualization of Trends

Temporal and spatial variations in fish community indices are provided in Figure 53 to Figure 58. When adjusted for flow, abundance, NMDS1, and NMDS2 appear to be increasing over time, while Simpson's Diversity and Evenness, and species richness appear to be decreasing with time.

NMDS1 appears to be slightly increasing with increasing distance downstream of the proposed OSPW discharge point, while the remaining indices show little to no change with increasing distance. However, as previously noted, distance from the proposed OSPW discharge point was not a significant predictor for any of the indices.

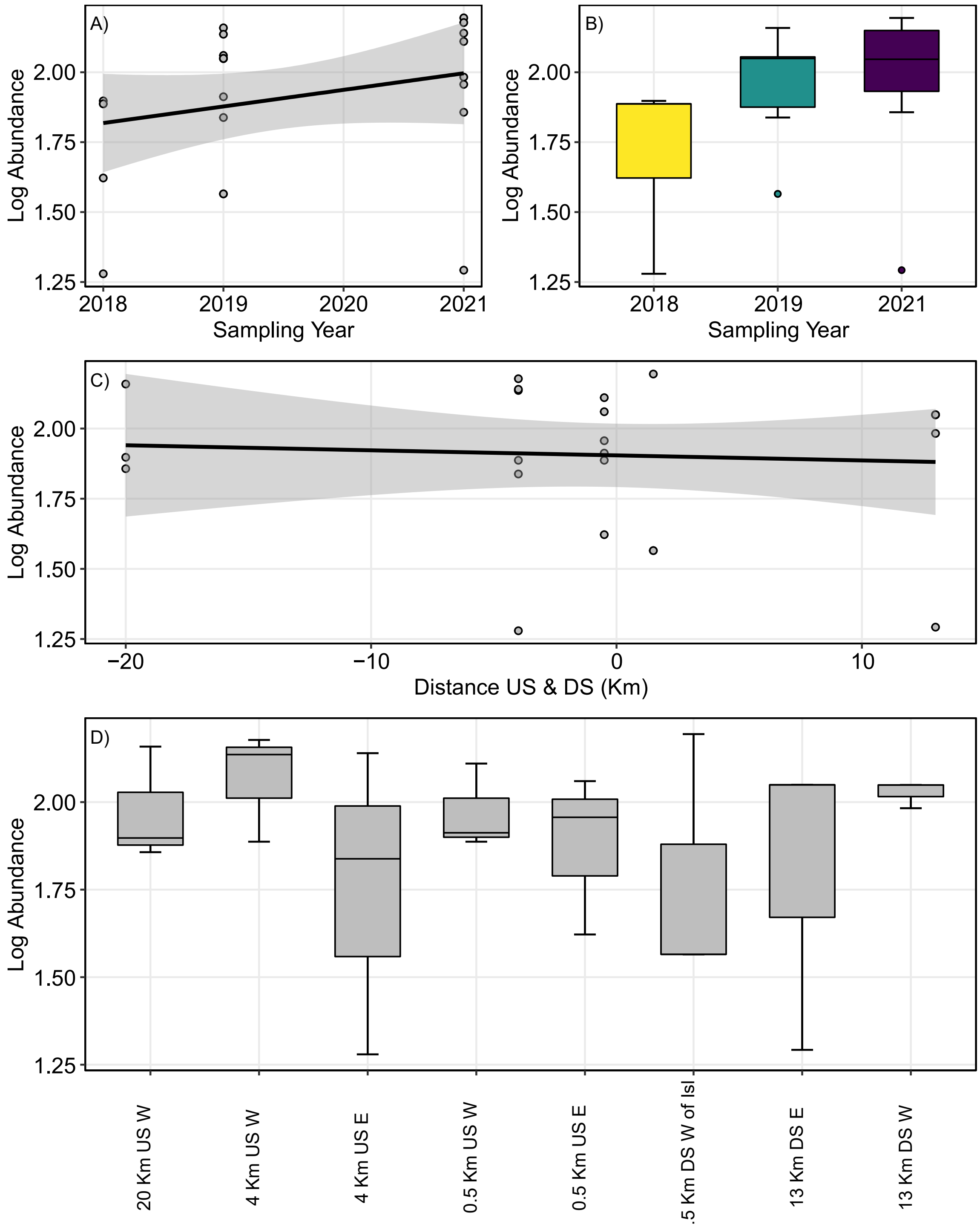

Figure 53 Variation of total fish abundance (log-transformed) over time (A-linear trend, B-ANOVA), over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 600 m^3/s.

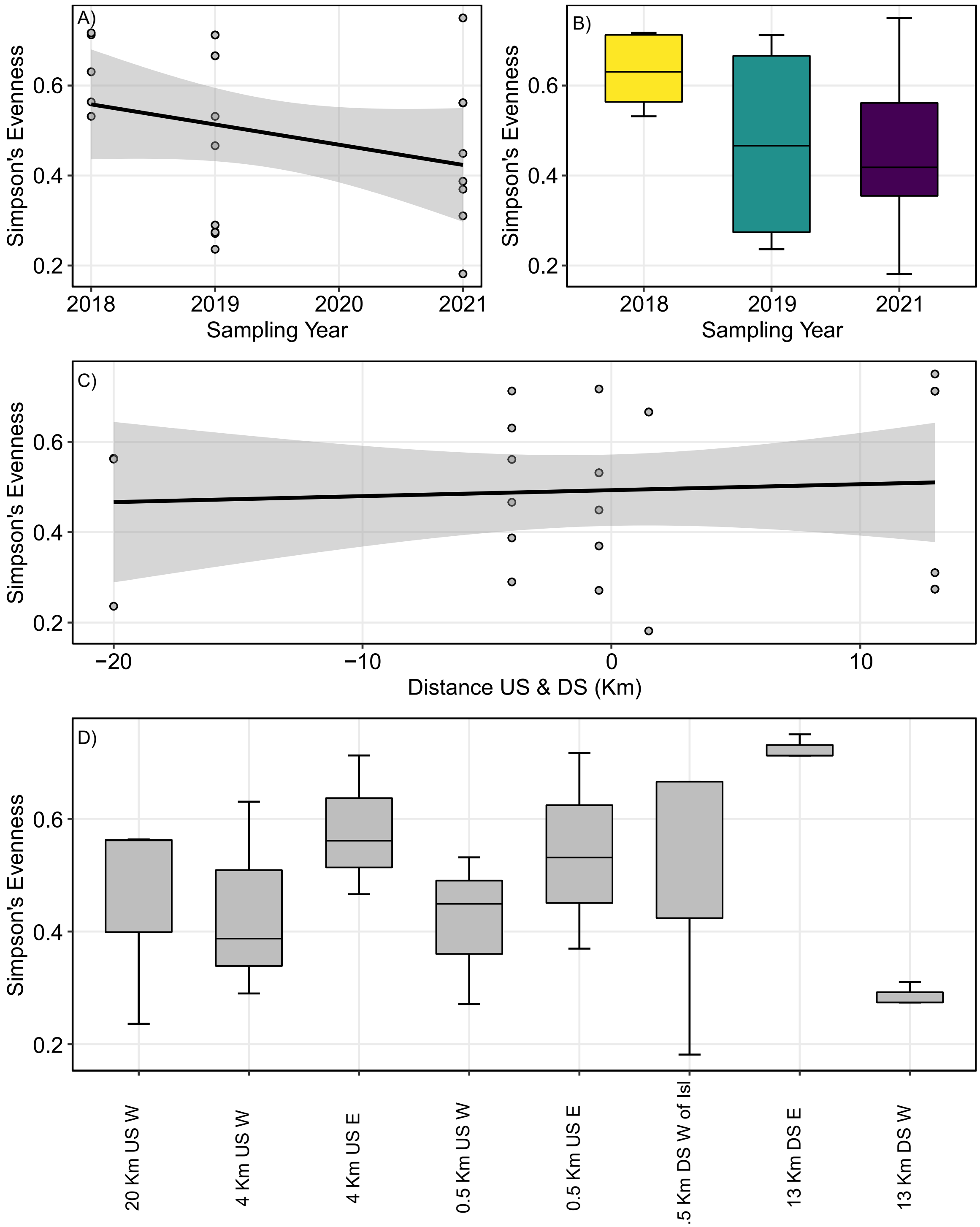

Figure 54 Variation of Simpson's Evenness over time (A-linear trend, B-ANOVA) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 600 m^3/s

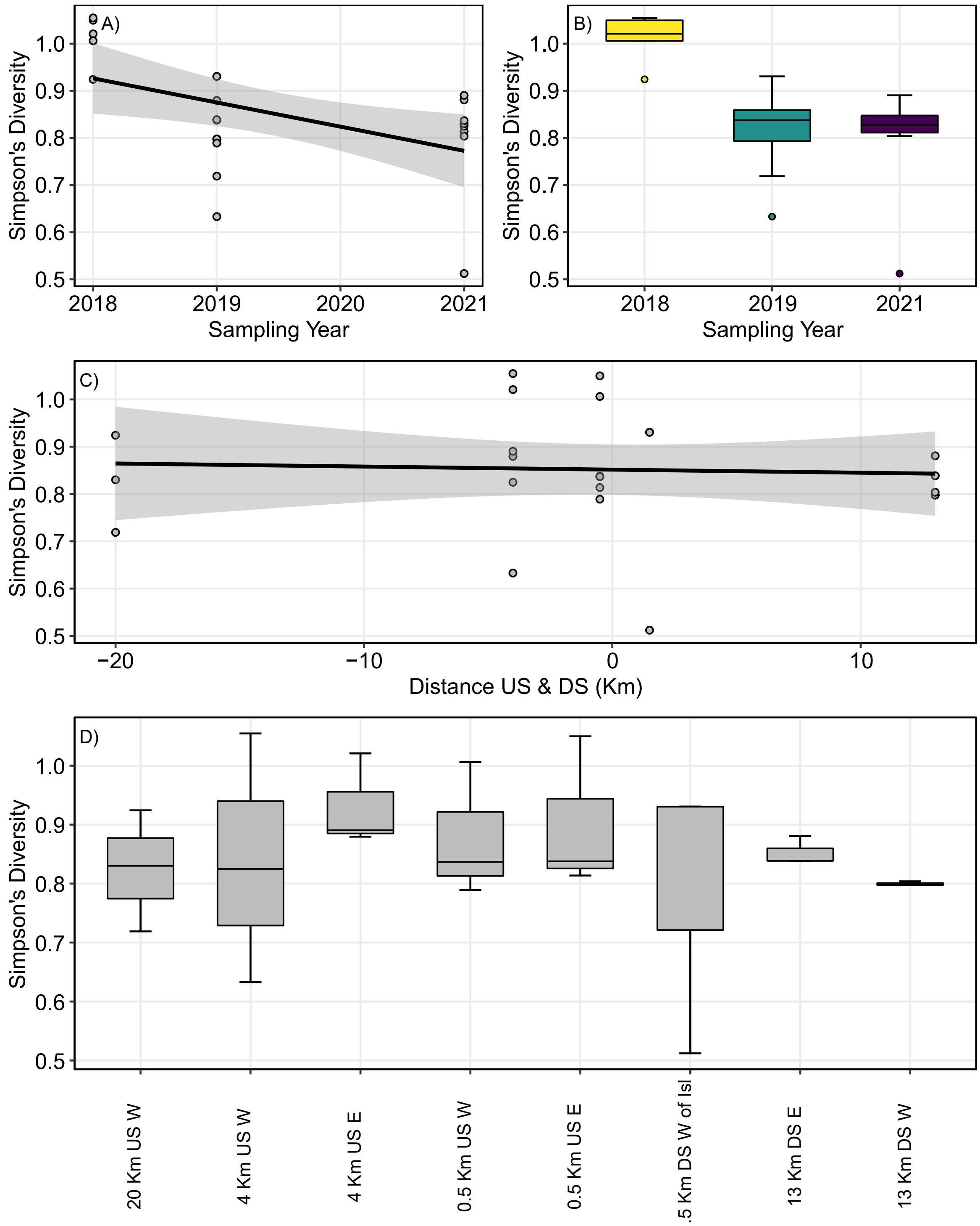

Figure 55 Variation of Simpsons' Diversity over time (A-linear trend, B-ANOVA) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 600 m^3/s.

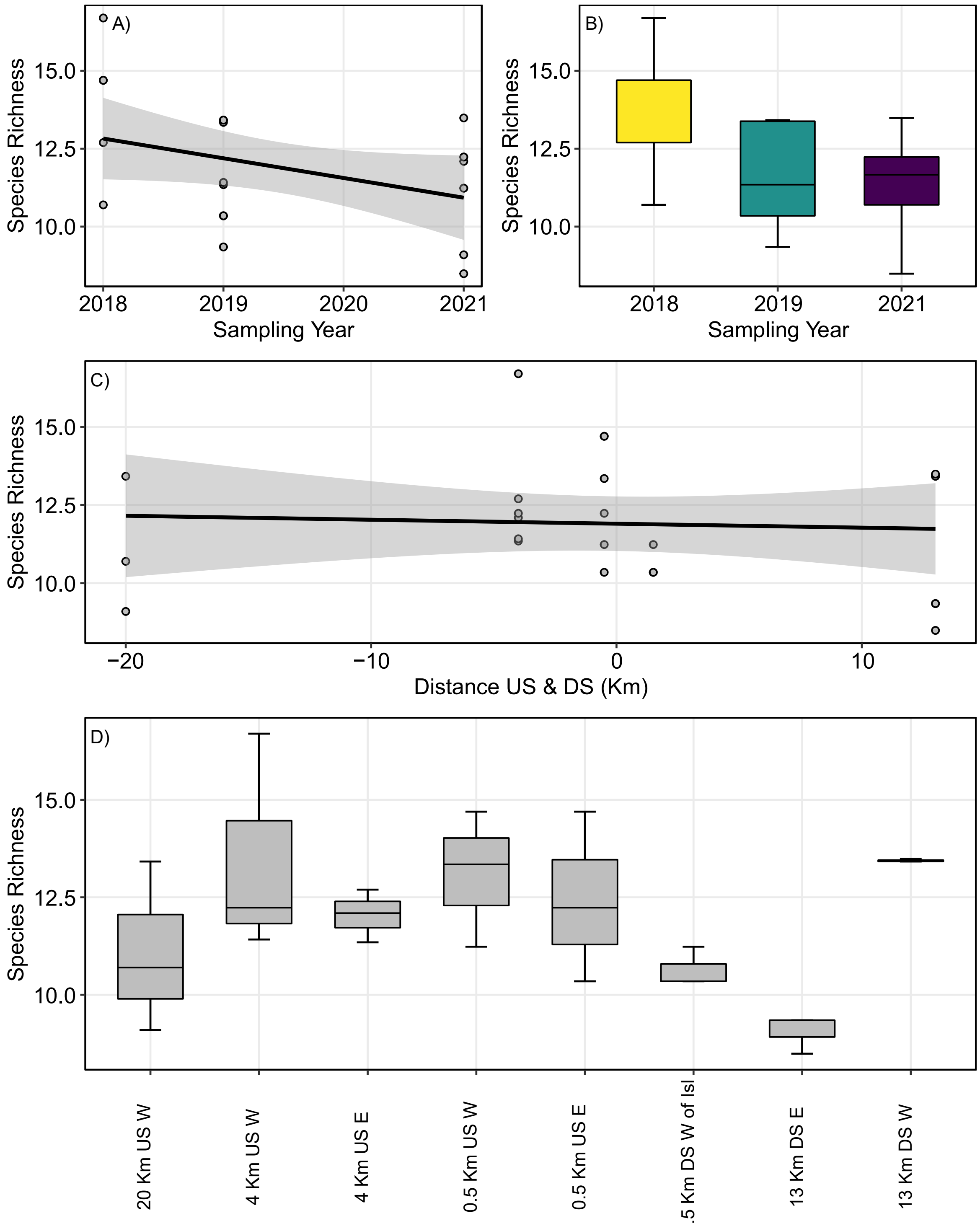

Figure 56 Variation of species richness over time (A-linear trend, B- ANOVA) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 600 m^3/s.

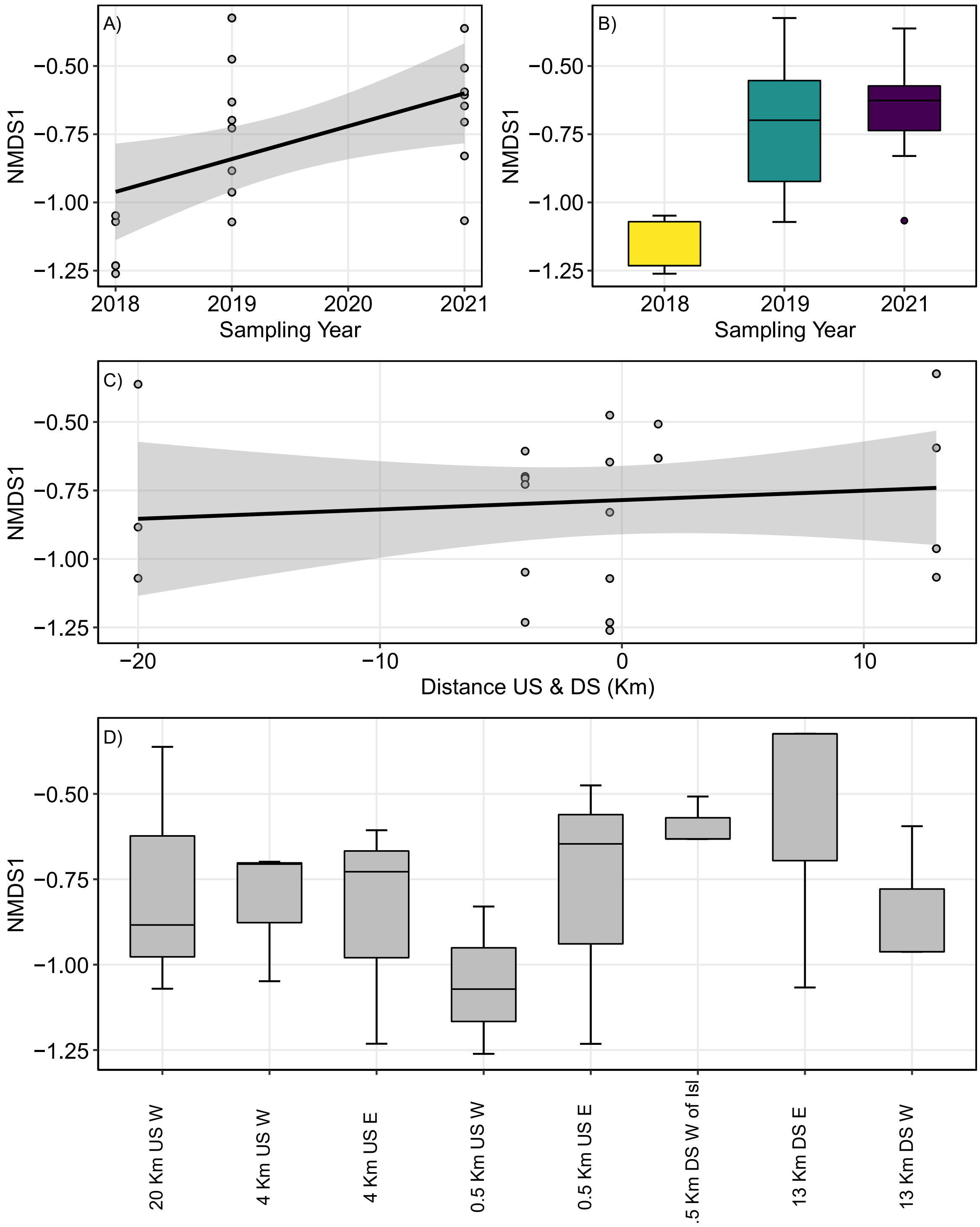

Figure 57 Variation of NMDS1 over time (A-linear trend, B- ANOVA) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 600 m^3/s.

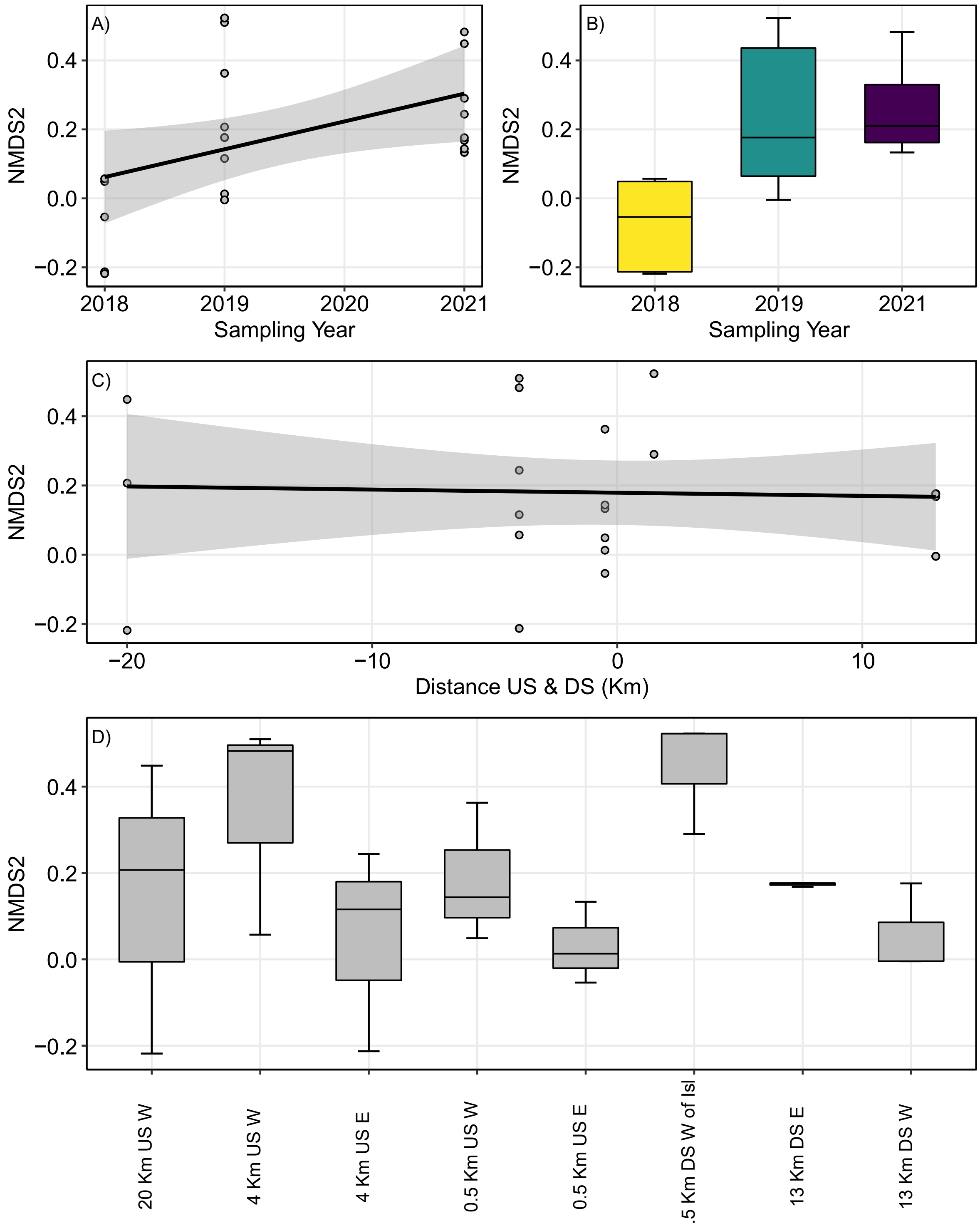

Figure 58 Variation of NMDS2 over time (A-linear trend, B- ANOVA) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Data are normalized to a Q60 of 600 m^3/s.

2.3.8 Sentinel Fish Populations Health

The EMP data consisted of measurements for both Trout-perch (*Percopsis omiscomaycus*, TRPR) and White Sucker (*Catostomus commersonii*, WHSC). Only Trout-perch were examined here as more data were available (Table 27) and given that they are commonly used as a sentinel species in the OSR. Only data for mature individuals were included in the analysis.

Table 27 Number of mature female (F) and male (M) Trout-perch and White Sucker captured from the stations sampled as part of the EMP.

Station	Description	Trout-perch						White Sucker			
		2018		2019		2021		2019		2021	
		F	M	F	M	F	M	F	M	F	M
AB07DA0062	25 km US	20	20	20	20	20	21	16	15	20	12
AB07DA3024	12 km US-Th	20	20	21	20	20	25	—	—	—	—
AB07DA3023	4 km US-Th	20	20	22	20	20	20	2	2	18	10
AB07DA3022	0.5 km US-LB	20	20	24	20	20	21	—	—	—	—
AB07DA3018	0.5 km DS-WI	21	20	20	21	—	—	5	4	16	11
AB07DA3017	0.5 km DS-EI	10	20	20	21	49	61	—	—	—	—
AB07DA3015	1.5 km DS-EI	20	20	20	20	20	23	—	—	—	—
AB07DA3009	4.5 km DS-Th	20	20	19	21	18	20	1	2	2	2
AB07DA3008	12 km DS-Th	20	20	24	20	21	20	7	12	18	16
AB07DA0800	34 km DS	21	20	20	21	30	39	—	—	—	—

Table Notes: Th = Thalweg, E = East of Island, W = West of Island, Right = Right Bank, and Left = Left Bank.

2.3.8.1 Data Summary

Summary statistics are provided in Table 28 and broken down by station-year in Appendix B Table B4. Potential differences among stations located upstream and downstream of the potential OSPW discharge point are investigated further in Section 2.3.6.2.

Spearman Rank correlations among fish health endpoints and potential covariables (i.e., discharge Q60 and air temperature) are provided in Table 29. There was a correlation (i.e., $r >$ critical r of 0.08) in the negative direction between discharge (Q60) and both condition for both females ($r = -0.29$) and males ($r = -0.32$). There was also a correlation between discharge and GSI for both females ($r = 0.36$) and males ($r = 0.47$), as well as LSI for females only ($r = 0.19$). There was a negative correlation between air temperature and GSI (female $r = -0.14$, male $r = -0.71$) and LSI (female $r = -0.40$, male $r = -0.13$). Discharge was chosen as the covariable to include in the explanatory models.

Table 28 Summary statistics for mature female and male Trout-perch collected during the EMP program (2018, 2019 and 2021)

Statistic	Females			Males		
	K	**GSI**	**LSI**	**K**	**GSI**	**LSI**
Min	0.86	1.01	0.59	0.69	1.00	0.52
Max	1.78	7.24	3.65	1.42	3.40	2.45
Mean	1.13	4.44	1.57	1.09	1.52	1.31
SD	0.10	1.13	0.31	0.09	0.38	0.23

Table Notes: Condition (K), Gonadosomatic index (GSI) and Liver Somatic Index (LSI)

Table 29 Spearman Rank correlations among fish health metrics for mature Trout-perch collected during the EMP program (2018, 2019 and 2021)

Parameters			Q60	K	GSI	LSI
Covariable	Discharge		1	-	-	-
Fish Population Metric	Females	Condition	**-0.29**	1	-	-
		Gonadosomatic Index	**0.36**	**-0.17**	1	-
		Liver Somatic Index	**0.19**	**0.10**	**0.30**	1
	Males	Condition	**-0.32**	1	-	-
		Gonadosomatic Index	**0.47**	**-0.09**	1	-
		Liver Somatic Index	0.02	0.06	**0.14**	1

Table Notes: Values in bold exceed the critical r of 0.08, calculated as $1.96/\sqrt{N-1}$, where N is the sample size of the dataset (i.e., 620 females and 648 males)

2.3.8.2 Explanatory Models

Results from the GLMs for mature female and male Trout-perch are summarized in Table 30. The models determined that discharge for the 60-day period prior to fish collections (Q60) was a significant predictor (p-value < 0.05) of variations in female and male Trout-perch GSI and condition (K), along with female LSI. Discharge (Q60) explained between 1.2 and 15.6% of the variation in these indices of fish population health when significant.

Linear trends over time were statistically significant for female and male condition (K) and LSI, along with male GSI. There were no statistically significant variations associated with the distance from the proposed OSPW discharge point.

Temporal and spatial differences were further investigated below in Section 2.3.7.3 on flow-normalized fish population metrics. Normalization equations are provided in Appendix C Table C5.

Table 30 Results of statistical analyses assessing variation in different metric of fish health indicators in both female and male Trout-perch collected along the LAR under the EMP (2018, 2019, 2021).

Sex	Index	Q60 (m^3/s)		Year		Distance (US/DS)	
		P-value	%VE	P-value	%VE	P-value	%VE
F	GSI	**<0.001**	15.6	0.868	0.0	0.598	0.0
	K	**<0.001**	7.6	**0.001**	1.8	0.116	0.4
	LSI	**<0.001**	2.3	**<0.001**	14.8	0.435	0.1
M	GSI	**0.004**	1.2	**<0.001**	31.7	0.801	0.0
	K	**<0.001**	8.4	**<0.001**	3.1	0.678	0.0
	LSI	0.093	0.6	**0.035**	1.0	0.567	0.1

Table Notes: Significant values (i.e., p-value < 0.05) are in bold.
%VE represents the percentage of total variance explained by each predictor within the individual models
Shaded cells highlight the %VE that corresponds to significant p-values.

2.3.8.3 Visualization of Trends

Temporal and spatial variations in flow-normalized fish population indices are provided in Figure 59 to Figure 64. There appears to be a slight decrease over time in flow normalized GSI (males), K (females), LSI (females and males). After removing the influence of flow volume averaged over the 60 days prior to sampling (Q60), there appears to be no spatial variations in female and male GSI, K and LSI values.

An important spatial comparison to consider is the difference between samples collected on the east and west side of the island located 0.5 km downstream of the proposed OSPW discharge point, as these stations are directly downstream of the Syncrude sewage treatment outfall (Figure 1). Differences among samples collected from the east and west side of the island were explored using a Tukey's post-hoc test for individual comparisons. Results are provided in Table 31. Only GSI values in female Trout-perch were significantly higher on the west side of the island compared to the east side (p-value = 0.001; Table 31; Figure 59).

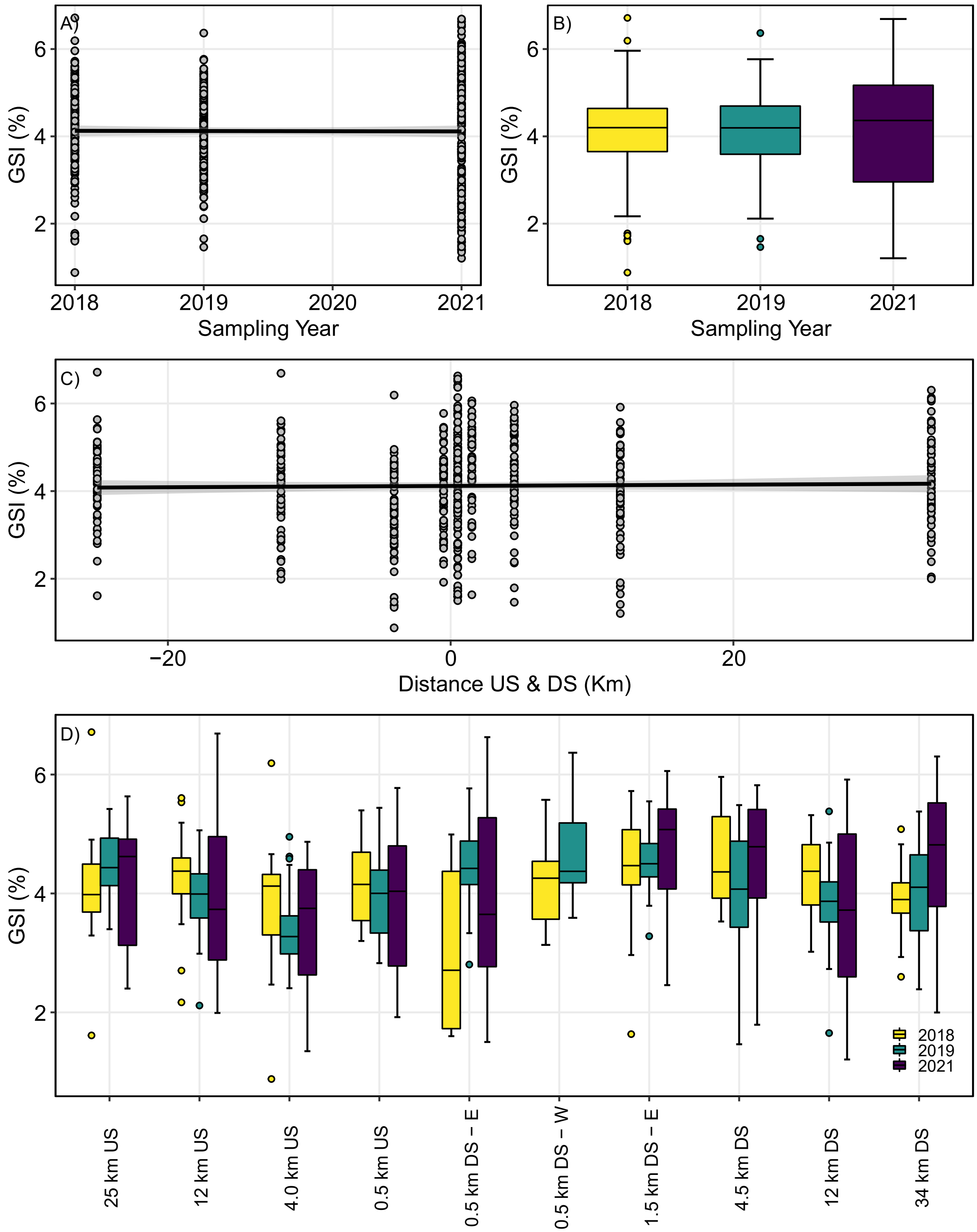

Figure 59 Variation of Gonadosomatic Index (GSI) in female Trout-perch over time (A), between sampling years (B), over distance upstream/downstream of proposed OSPW discharge point (C), and between EMP sampling stations (D).

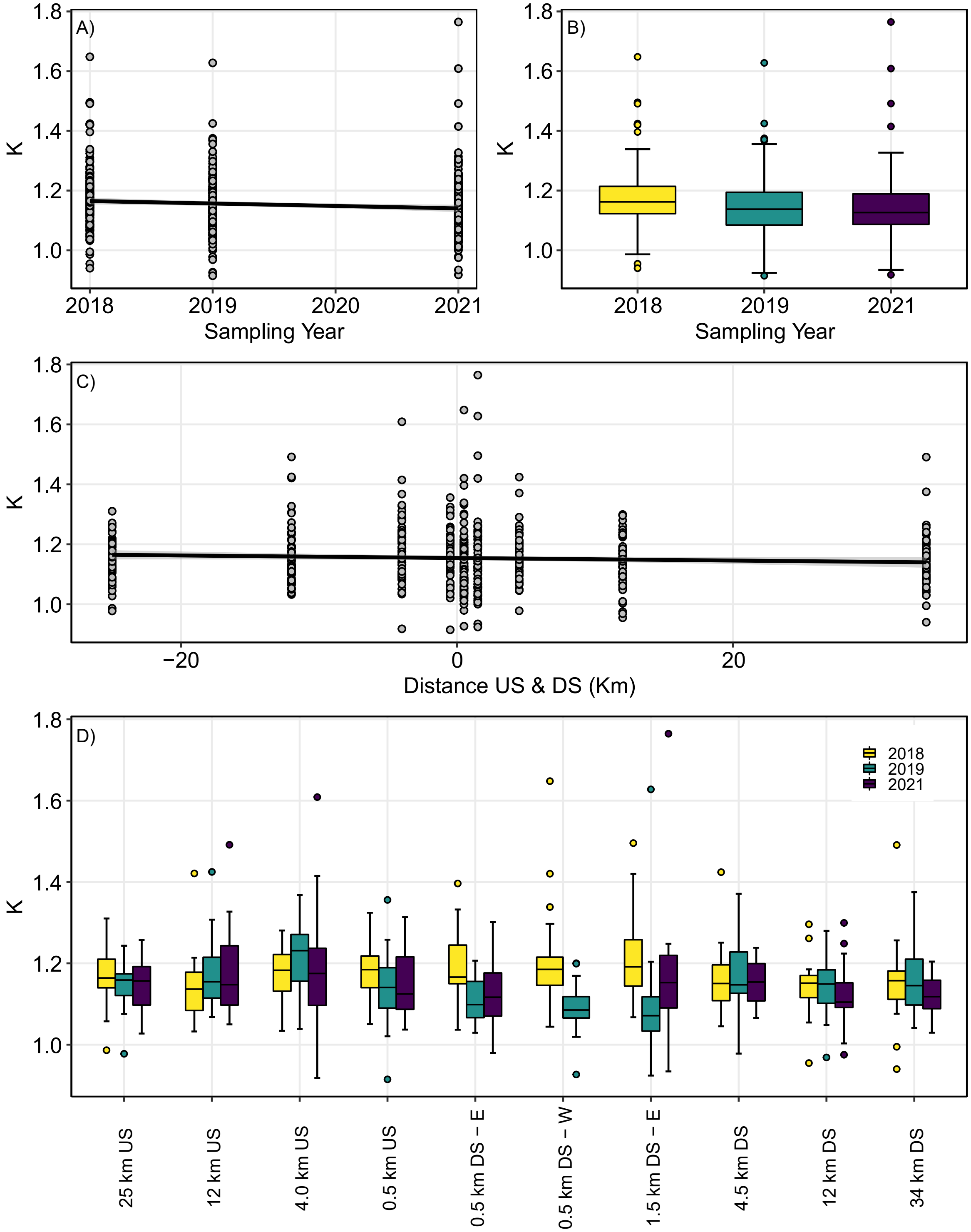

Figure 60 Variation of Condition Factor (K) in female Trout-perch over time (A), between sampling years (B), over distance upstream/downstream of proposed OSPW discharge point (C), and between EMP sampling stations (D).

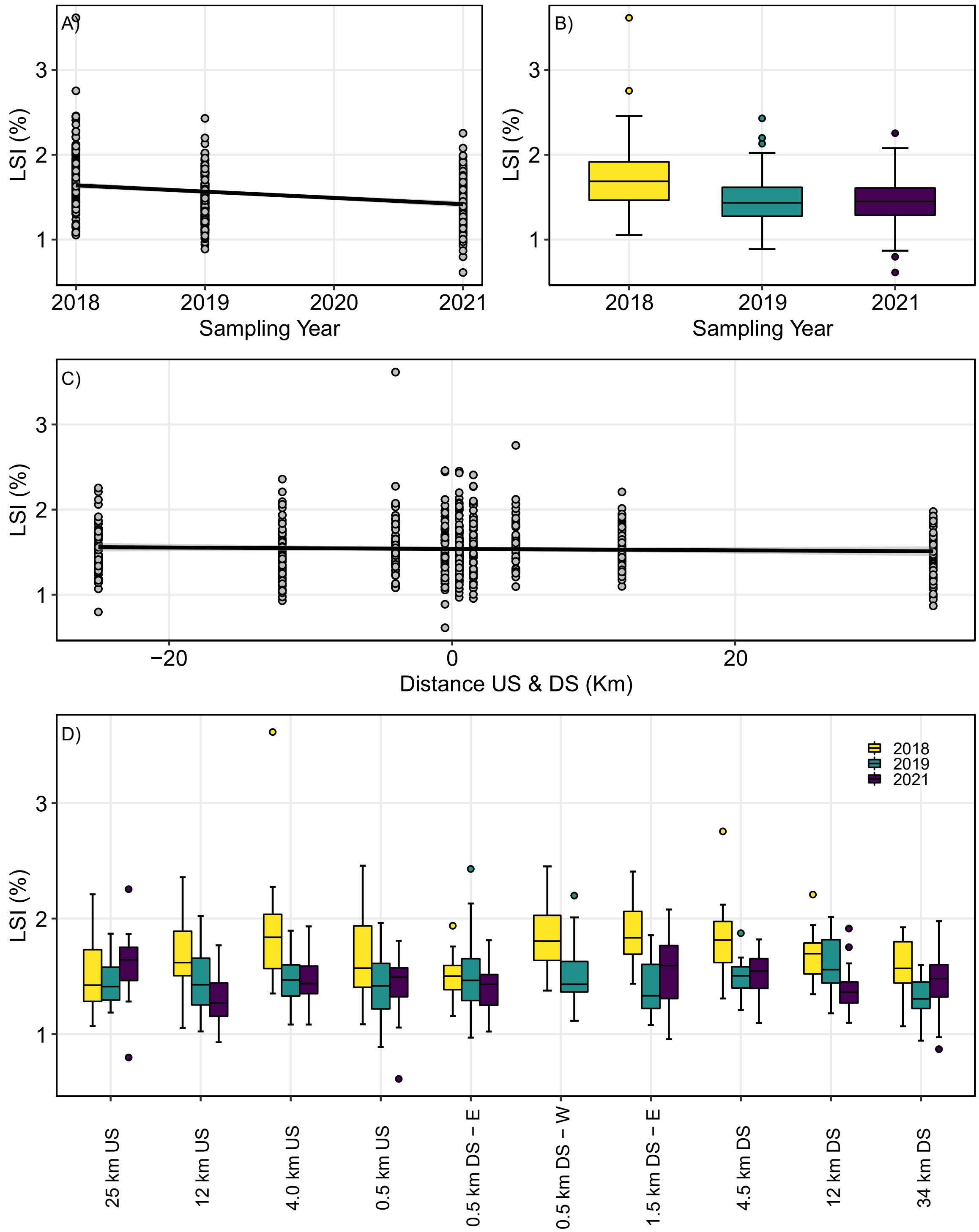

Figure 61 Variation of Liver Somatic Index (LSI) in female Trout-perch over time (A), between sampling years (B), over distance upstream/downstream of proposed OSPW discharge point (C), and between EMP sampling stations (D).

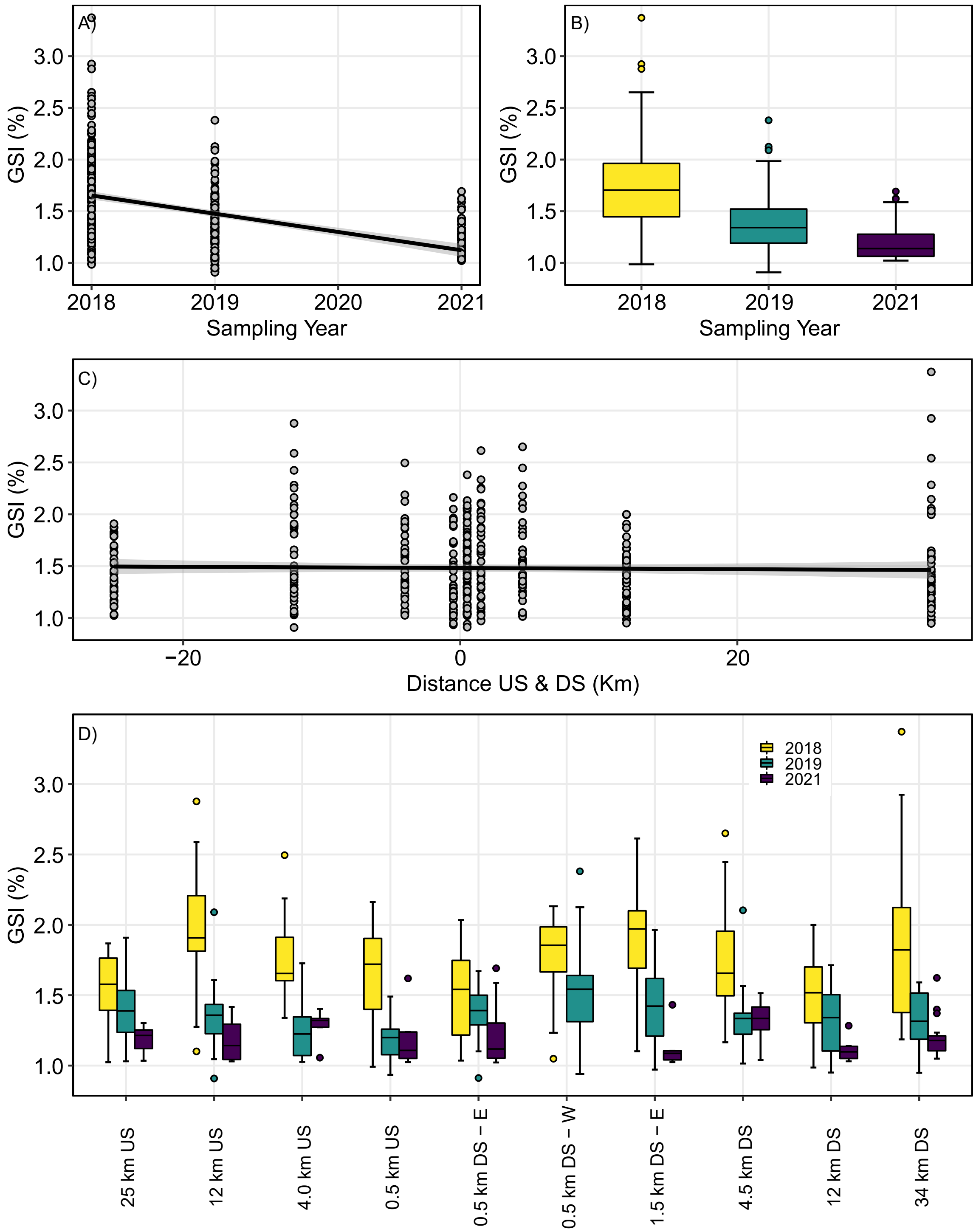

Figure 62 Variation of Gonadosomatic Index (GSI) in male Trout-perch over time (A), between sampling years (B), over distance upstream/downstream of proposed OSPW discharge point (C), and between EMP sampling stations (D).

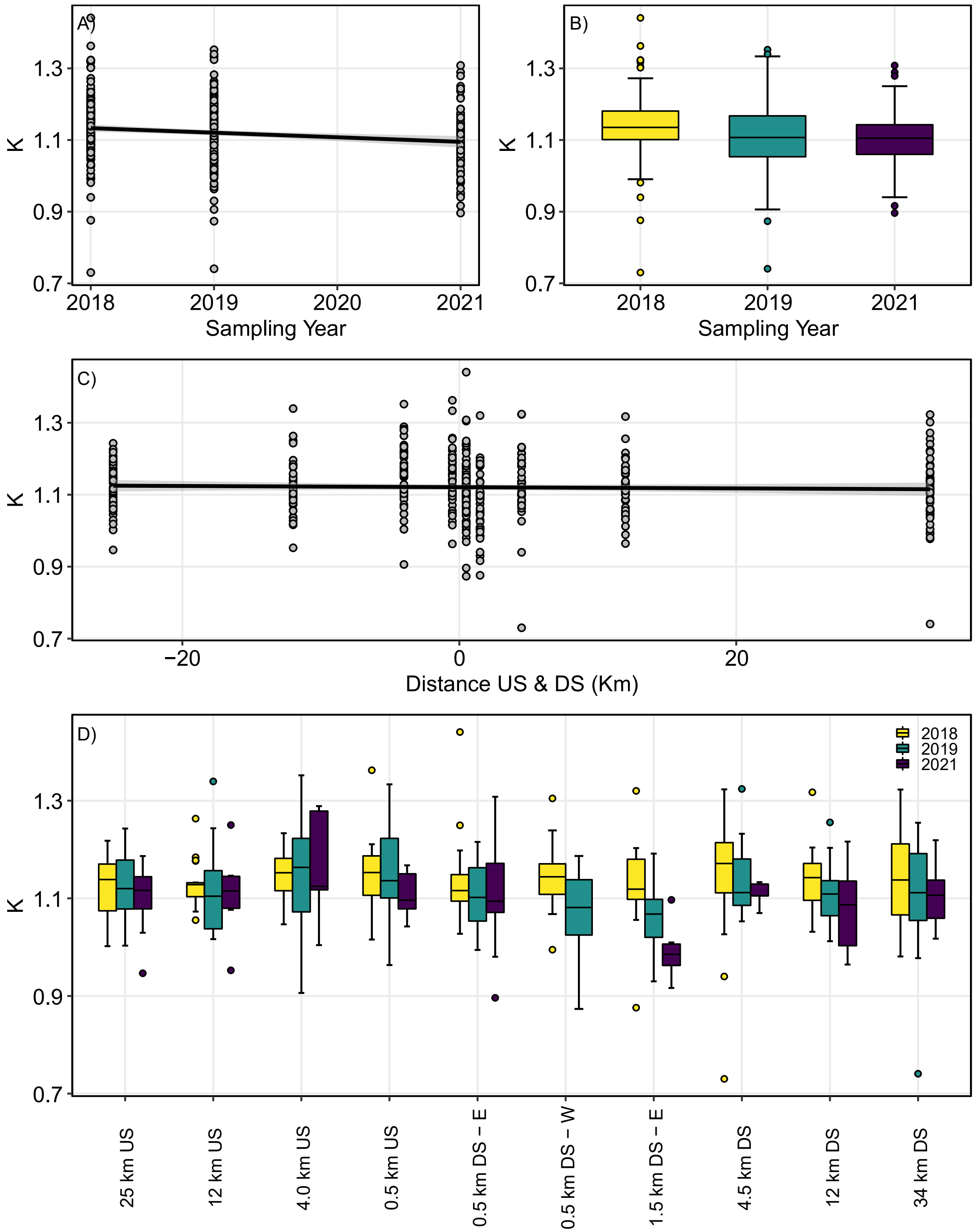

Figure 63 Variation of Condition Factor (K) in male Trout-perch over time (A), between sampling years (B), over distance upstream/downstream of proposed OSPW discharge point (C), and between EMP sampling stations (D).

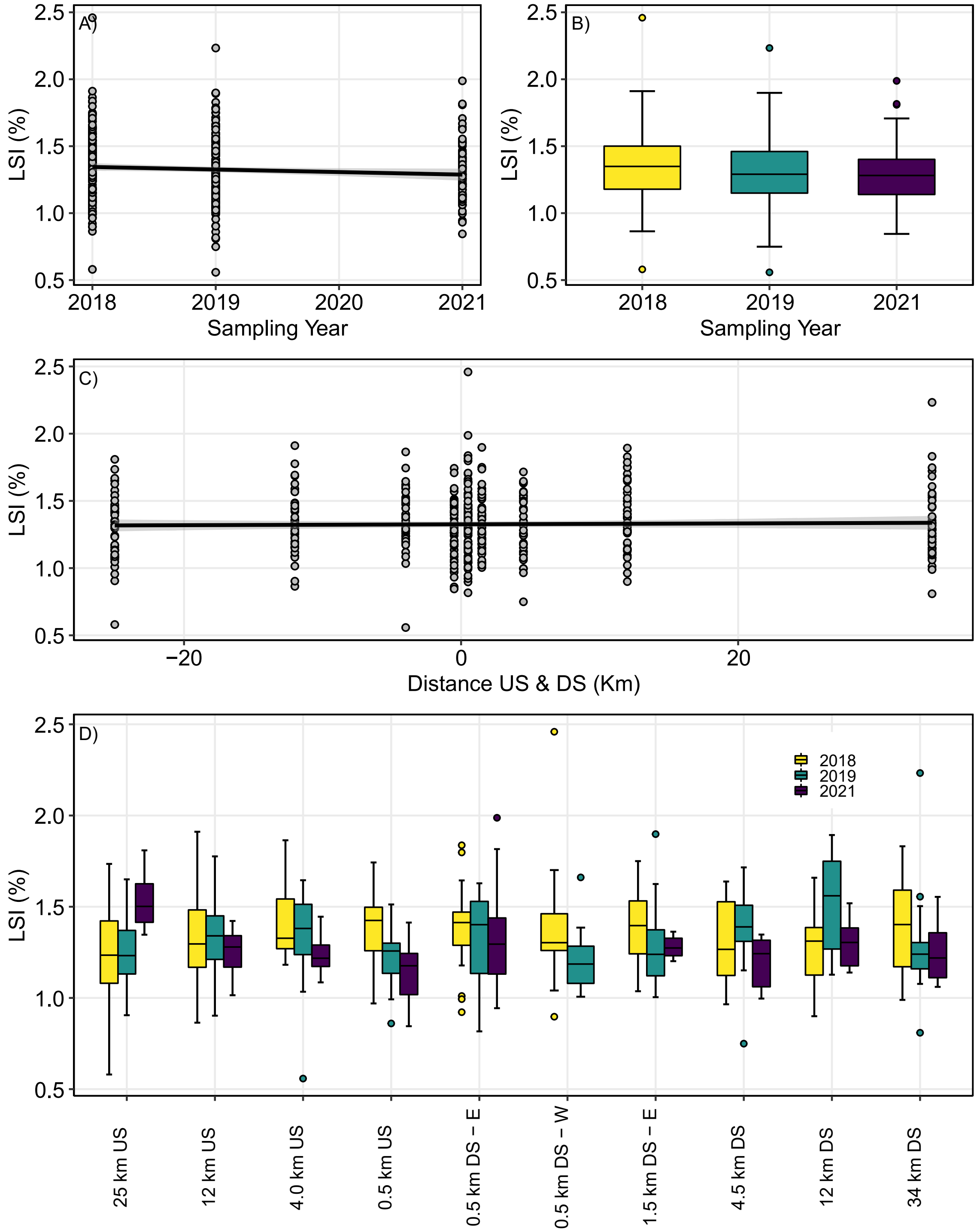

Figure 64 Variation of Liver Somatic Index (LSI) in male Trout-perch over time (A), between sampling years (B), over distance upstream/downstream of proposed OSPW discharge point (C), and between EMP sampling stations (D).

Table 31 Tukey's post-hoc test comparing flow normalized fish health indices for individuals collected at the stations (0.5km DS) located both East (E) and West (W) of the island, EMP dataset (2018, 2019, and 2021).

Sex	Index	0.5 km DS - E vs. 0.5 km DS - W			
		2018		2019	
		Direction	P-value	Direction	P-value
M	GSI	E < W	0.211	E < W	0.71
	k	E < W	1	E > W	0.985
	LSI	E < W	1	E > W	0.861
F	GSI	E < W	**0.001**	E < W	0.998
	k	E < W	1	E > W	0.999
	LSI	E < W	0.299	E > W	1

Table Notes: Significant values ($p < 0.05$) are in bold.

2.3.9 Fish Body and Tissue Burden

The EMP data included in this section of the report consisted of measurements of metals, mercury, PAHs, phenols, and SIRs in fish whole body and tissues (i.e., dorsal muscle) collected from female and male Trout-perch, White Sucker and Walleye. Fish were typically collected from 10 different EMP stations, which varied slightly from year to year. Samples consisted of both composite (i.e., PAHs) and non-composite samples. Analytes measured in each sample also differed across species, sex, station, and year, as discussed below. Measured analytes were classified into four groups: PAHs, metals, phenols, mercury (both total, and methyl mercury) and SIRs. EROD activity was also measured, but only in Trout-perch liver samples.

2.3.9.1 Data Summary

2.3.9.1.1 Data Availability

For Trout-perch, PAHs were analyzed in females across all years and stations, and males were reserved for the other analyte groups (i.e., metals, SIR, and mercury). Females had to be pooled (3-5 fish per sample to achieve enough sample for the PAH analysis). Metals, Hg, and SIR were completed on individual fish. On occasion (in 2021), PAHs, metals, and mercury were measured in both female and male Trout-perch for comparative purposes; see Appendix B Table B5. Consequently, temporal and spatial trends discussed in upcoming sections 2.3.9.2 and 0 will focus only on PAHs for female Trout-perch and the other analyte groups for male Trout-perch. PAHs, metals, and mercury were measured in whole body samples (with liver removed), while SIRs were measured in skinless dorsal epaxial muscle tissue.

For Walleye, all parameter groups (i.e., PAHs, SIRs, phenols, metals, and mercury) were measured in muscle tissue of both females and males. Walleye samples were only collected in 2019 and 2021 from a total of 5 stations (Appendix B Table B6). Similarly, for White Sucker, all parameter groups (i.e., PAHs, SIRs, phenols, metals, and mercury) were measured in muscle tissue of both females and males. White Sucker samples were only collected in 2019 and 2021 from a total of 6 stations (Appendix B Table B7).

2.3.9.1.2 Non-Detects

A summary of VMV codes, method detection limits, and non-detections can be found in Appendix A Table A1. Female Trout-perch whole body samples were below detection limits 43 to 54% of the time across all individual PAHs, while male Trout-perch whole body samples were below detection limits 20 to 50% of the time across all individual metals (Figure 65). Male Trout-perch whole body samples were not below detection limit for mercury.

Female and male Walleye tissue samples were below detection limits 49 to 56% of the time across all individual PAHs, 41 to 48% of the time across all individual metals, 70 to 79% of the time across all individual phenols, and not below detection for the other analytes (Figure 66). Female and male White Sucker tissue samples were below detection limits 0 to 50% of the time across all individual PAHs, 42 to 47% of the time across all individual metals, 69 to 78% of the time across all individual phenols, and not below detection for the other analytes (Figure 67).

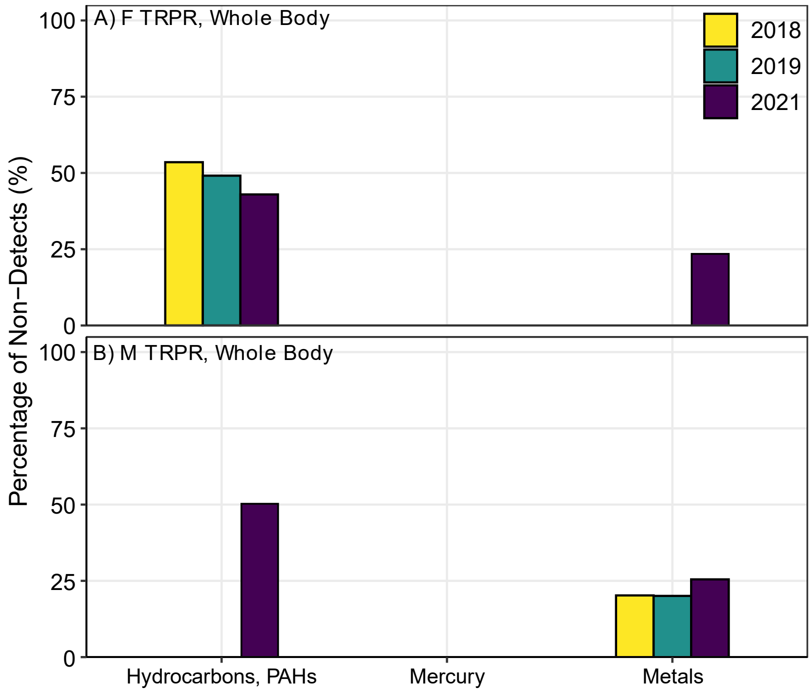

Figure 65 Percentage of non-detects observed in Trout-perch body burden samples by parameter category in the EMP dataset (2018, 2019, and 2021)

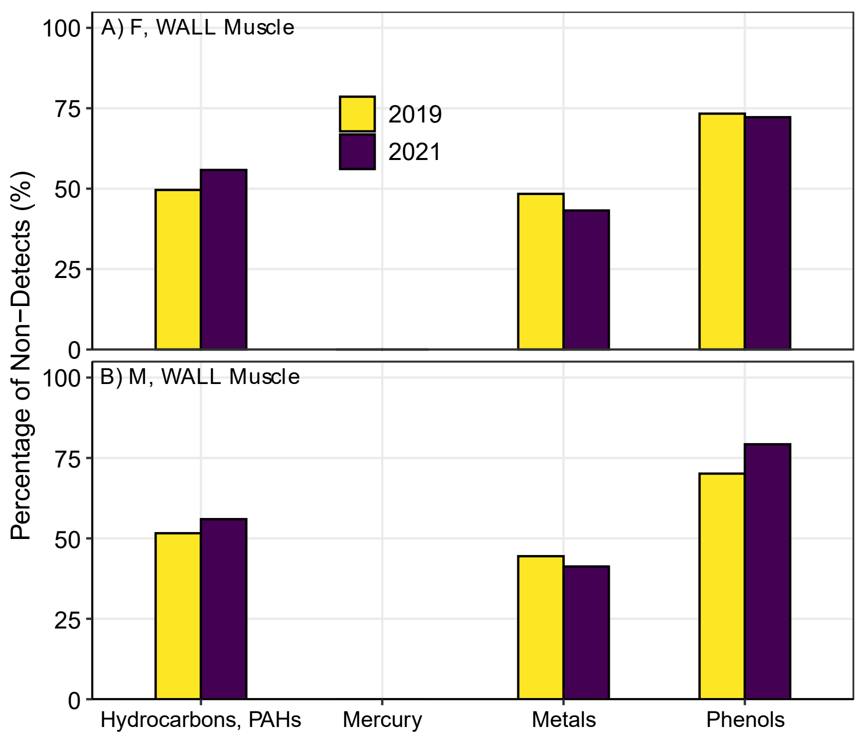

Figure 66 Percentage of non-detects observed in Walleye body burden samples by parameter category in the EMP dataset (2018, 2019, and 2021)

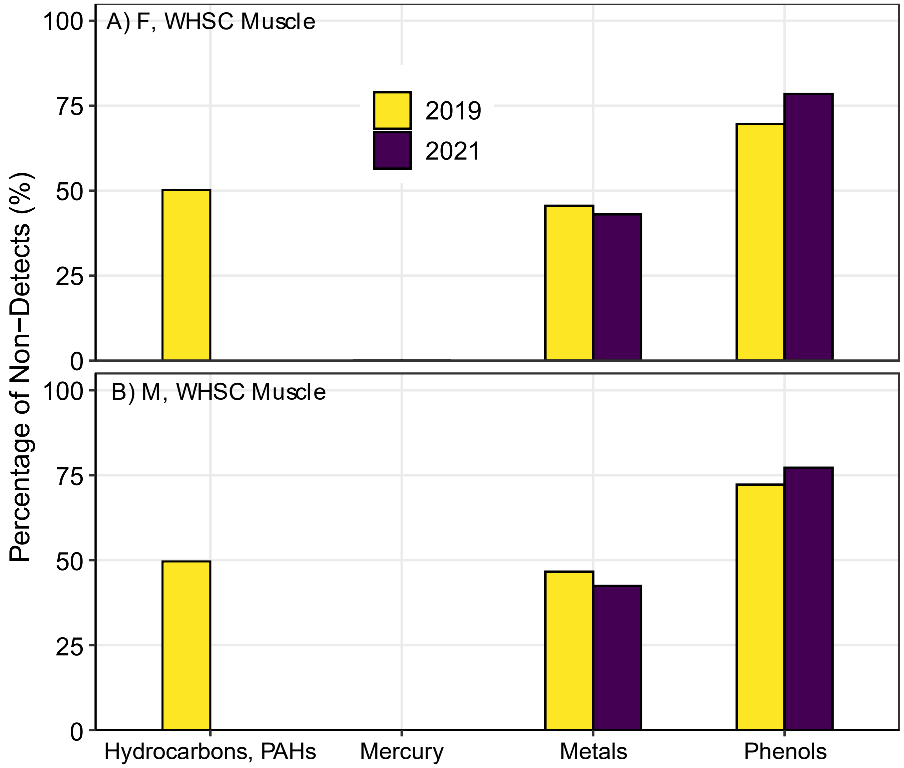

Figure 67 Percentage of non-detects observed in White Sucker body burden samples by parameter category in the EMP dataset (2018, 2019, and 2021)

2.3.9.1.3 Summary Statistics

Summary statistics for key analytes, by species and sex, pooled for all stations and years are provided in Table 32. Detailed EROD and SIR summary statistics, broken down by station and year, are provided in Appendix B Table B8 and A9, respectively.

In female fish, mean total PAH values were 105, 10.1, and 9 ng/g w.w. in Trout-perch, Walleye, and White Sucker, respectively. In male fish, mean total PAH values were 169, 11,2 and 10 ng/g w.w. in Trout-perch, Walleye, and White Sucker, respectively.

Mean total mercury concentrations were 447 and 199 ng/g w.w. while mean methyl mercury concentrations were 357 and 136 in female Walleye and White Sucker, respectively. Mean total mercury concentrations in male fish were 26, 532, and 184 ng/g w.w. while mean methyl mercury concentrations were 40, 435, and 153 ng/g w.w. in Trout-perch, Walleye, and White Sucker, respectively.

In female fish, mean total selenium concentrations were 0.5, 0.3, and 0.2 ng/g w.w. in Trout-perch, Walleye, and White Sucker, respectively. In male fish, mean total selenium concentrations were 0.6, 0.3, and 0.3 ng/g w.w. in Trout-perch, Walleye, and White Sucker, respectively.

In female fish, mean total phenol concentrations were 118 and 70.7 ng/g w.w. in Walleye and White Sucker, respectively, while male fish mean total phenol concentrations were 148 and 53.8 ng/g w.w. in Walleye and White Sucker, respectively.

Mean (± SD) EROD activity values ranged from 0.9 ± 1.2 pmol/min/mg (n = 620) in female Trout-perch, and 1.0± 1.4 pmol/min/mg (n = 620) in males. Pearson correlations of liver EROD activity and other analytes measured in Trout-perch tissues indicated there was a significant positive correlation between liver EROD activity and total PAH ($r = 0.42$), and a significant negative correlation between liver EROD activity and total silver ($r = -0.41$). Measurement of EROD induction in fish has been commonly used as an indicator of exposure to PAHs (Grøsvik et al., 1997; Whyte et al., 2000). PAHs tend to be metabolized and excreted quickly in fish and other vertebrate tissues, making their detection harder to capture (Dwiyitno et al., 2016; Livingstone, 1998; Whyte et al., 2000). EROD induction can be used in conjunction with other indicators of PAH exposure such as measurement of PAH metabolites and detoxifying enzyme activity (i.e., CYP1A activity) which can be a powerful technique to assess exposure. Heavy metals (i.e., Cd, Cr, Cu, Fe, Hg, Zn) on the other hand have been shown to inhibit liver EROD activity in aquatic species (Oliveira et al., 2004; Viarengo et al., 1997; Whyte et al., 2000). However, research on the mechanisms behind EROD inhibition is limited. Aside from total silver, there were no other significant relationships between liver EROD activity and heavy metal concentrations in Trout-perch tissues. Mean $\delta^{13}C$ values were around -27‰ for females and males of all species, while mean $\delta^{15}N$ values were around 9‰.

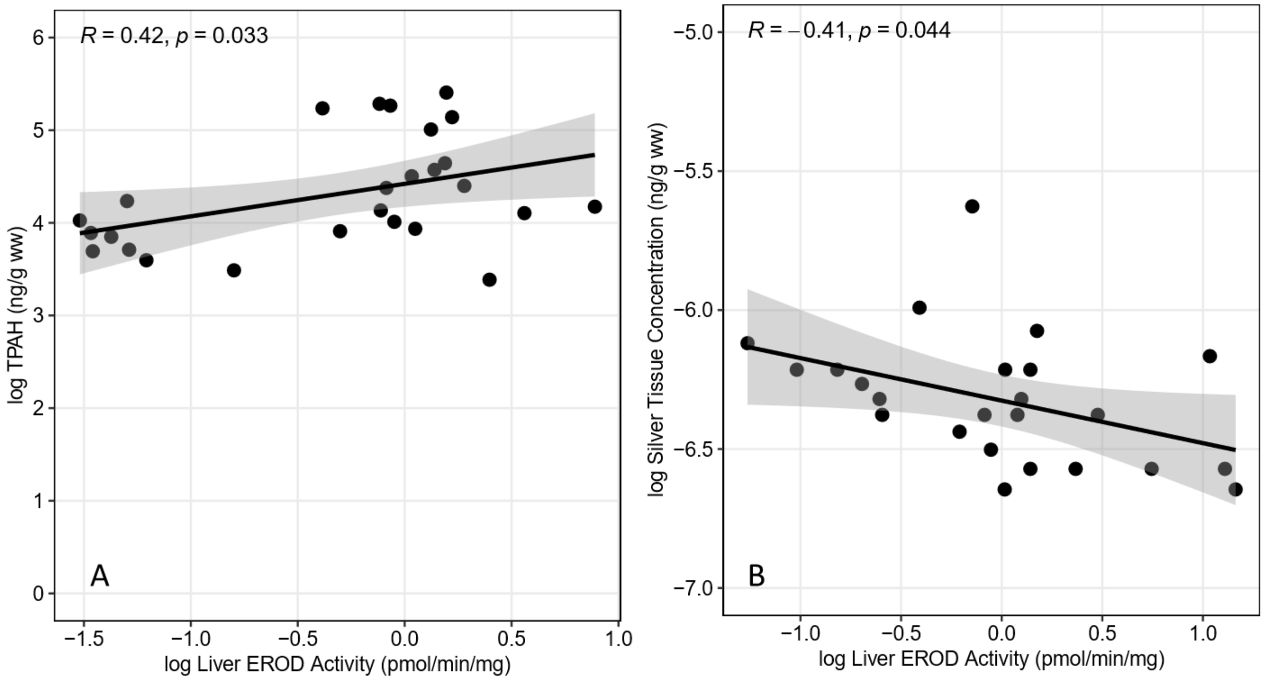

Figure 68 Relationship between total PAH tissue concentrations (A) and total silver tissue concentrations (B) with liver EROD activity in Trout-perch, each datapoint represents an individual fish collected during EMP (2018, 2019, and 2021)

Figure Note: Raw data are presented, pooled across the entire EMP dataset (all years and sites) for data that could be paired by Sample ID, Sample Date, and Sampling Station (i.e., N = 26 for TPAH vs. EROD and N = 24 for Ag vs. EROD)

Table 32 Summary statistics for body burden concentrations in mature female and male Trout-perch, Walleye, and White Sucker collected during the EMP program (2018, 2019 and 2021). A subset list of compounds of concern is presented.

Species	Variable	Units	Females				Males			
			Min	Max	Mean	SD	Min	Max	Mean	SD
Trout-perch	Total PAH	ng/g w.w.	6.7	1836.7	105.1	173.6	70.1	296.8	168.7	102.7
	Total Mercury	ng/g w.w.	-	-	-	-	17.6	75.8	39.8	12.8
	Methyl Mercury	ng/g w.w.	-	-	-	-	15.9	69.7	35.7	10.7
	Total Selenium	ng/g w.w.	0.4	0.7	0.5	0.1	0.3	1.7	0.6	0.2
	EROD	pmol/ min/mg	0	10.4	0.9	1.2	0	10.9	1	1.4
	$\delta^{15}N$	‰	9	10.3	9.6	0.4	6.2	11.2	9.4	0.7
	$\delta^{13}C$	‰	-29.3	-25.8	-26.7	0.8	-30.1	-24	-26.8	0.7
Walleye	Total PAH	ng/g w.w.	3.5	26.4	10.1	5.6	4.2	153	11.2	20.3
	Total Mercury	ng/g w.w.	220.2	681.7	447.2	116.6	62.9	1160.8	532.3	230.7
	Methyl Mercury	ng/g w.w.	298.4	414.9	356.7	82.3	165.2	688.1	435.1	176.3
	Total Selenium	ng/g w.w.	0.1	0.4	0.3	0.1	0.1	0.5	0.3	0.1
	Total Phenols	ng/g w.w.	0.843	656	118	215	7.18	148	33.3	32.3
	$\delta^{15}N$	‰	9.9	12.3	10.8	0.6	9.8	12.8	11.1	0.7
	$\delta^{13}C$	‰	-30.1	-24.5	-27.5	1.6	-31.6	-24.6	-26.8	1.7
White Sucker	Total PAH	ng/g w.w.	4.9	15.1	9.3	3.3	4.2	25.5	10.1	5.1
	Total Mercury	ng/g w.w.	26.6	462	198.6	98.4	52.6	393.6	184.2	90.7
	Methyl Mercury	ng/g w.w.	22.9	223.2	136	72.4	73.2	251.8	153.2	57.4
	Total Selenium	ng/g w.w.	0.1	0.5	0.2	0.1	0.1	0.5	0.3	0.1
	Total Phenols	ng/g w.w.	<0.1	430	70.7	111	3.7	136	53.8	38.4
	$\delta^{15}N$	‰	4.3	10.3	8.7	0.9	6.4	11.1	9	0.8
	$\delta^{13}C$	‰	-35.5	-24.6	-28.5	1.7	-32.3	-24.1	-28.2	1.6

Table Notes: w.w. = Wet Weight; SD = Standard Deviation

2.3.9.1.4 Mercury

Methylmercury was only measured in 2018 and 2019 for Trout Perch and only in 2019 for Walleye and White Sucker and was removed from the 2021 analyte list due high cost and the assumption that most of the total mercury measurement would be made up of methyl mercury. Comparison of paired methylmercury and total mercury data suggests that between 56.6 and 99.8% for trout perch, 55.4 to 97.1% for Walleye, and 58.7 to 97.9% for White Sucker of the total mercury measurement is made up of methylmercury (Table 33). Further, when regressing the log transformed concentration of methylmercury against the log transformed concentration of total mercury as shown in Figure 69, the relationship yields a nearly perfect 1:1 line (slope of 1.01) and an R^2 of 0.98. This suggests that methylmercury in fish tissue can be predicted from total mercury concentrations. Thus, it is recommended that future monitoring of mercury involving the analysis of only total mercury is sufficient.

Table 33 Summary statistics for the percentage of methylmercury accounted for in total mercury measurements among mature fish, pooled across both male and females.

Species	% of MeHg in Total Hg			
	Min	Max	Mean	SD
Trout perch	56.6	99.8	80.4	10.4
Walleye	55.4	97.1	83.6	9.91
White Sucker	58.7	97.9	83.1	12.2

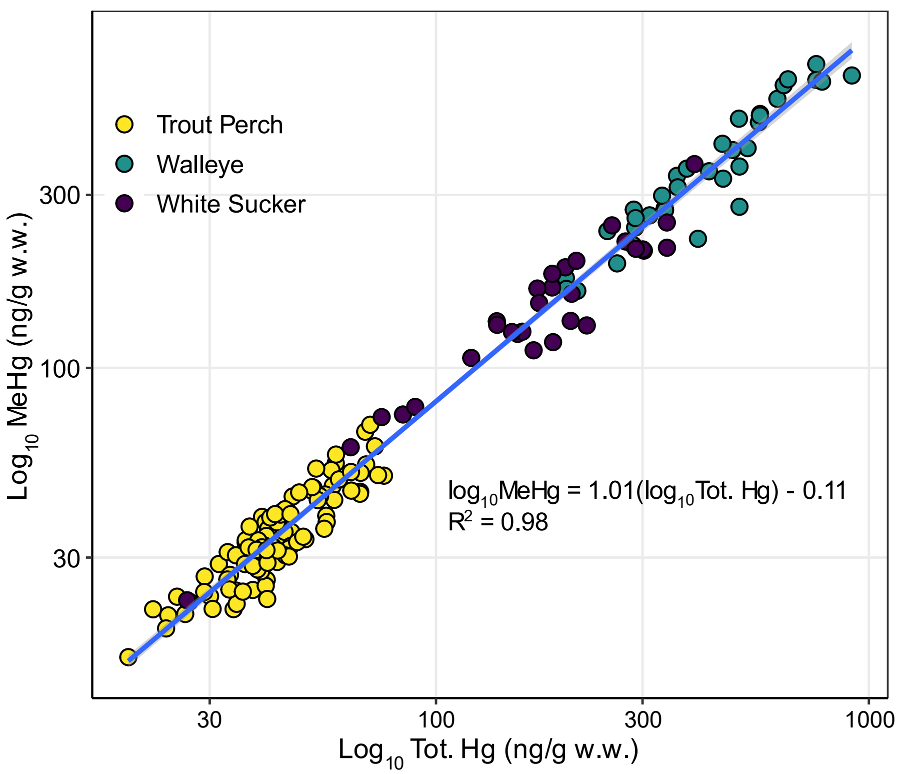

Figure 69 Relationship between log transformed methylmercury and total mercury in paired datasets for Trout-perch, Walleye, and White Sucker.

2.3.9.1.5 Guideline Exceedances

Fish muscle and whole-body burdens guidelines and benchmarks were compiled for the subset of compounds of concern:

- Total mercury concentrations of 500 ng/g w.w. in muscle samples for human consumption of all commercial fish and seafood and 200 ng/g w.w. in muscle samples for subsistence consumers (Alberta Health and Wellness, 2009);

- Benzo[a]pyrene (2 ng/g w.w.) in muscle samples of smoked fish (EU, 2015);

- ΣPAH4 (sum of benzo[a]anthracene, chrysene, benzo[b]fluoranthene, and benzo[a]pyrene; 12 ng/g w.w.) in muscle samples of smoked fish (EU, 2015);

- Methylmercury (33 ng/g w.w.) in whole-body samples for protection of wildlife consumers (CCME, 2000)

- Selenium (6700 ng/g d.w.) in whole-body samples (Environment and Climate Change Canada, 2021)

Approximately 48% of Walleye muscle samples exceeded the guideline for total mercury of 500 ng/g w.w. and 97% exceeded the guideline for subsistence consumers of 200 ng/g w.w, across all sampling years and stations (Figure 70A), while no White Sucker sampled exceeded 500 ng/g w.w. and 37% exceeded the guideline of 200 ng/g w.w. (Figure 70B). No Walleye or White Sucker samples exceeded the guideline of 2 ng/g w.w. for Benzo[a]pyrene, considering only 3 of 132 samples were above the detection limit that ranged from 0.011 – 0.245 ng/g w.w. (Figure 70CD). No Walleye or White Sucker samples exceeded the guideline of 12 ng/g w.w for ΣPAH4 (Figure 70EF). A total of 52% of Trout-perch whole body samples exceeded the methylmercury guideline of 33 ng/g w.w. (Figure 70G). Finally, no fish whole body sample exceeded the selenium consumption guideline of 6700 ng/g d.w. (Figure 70H).

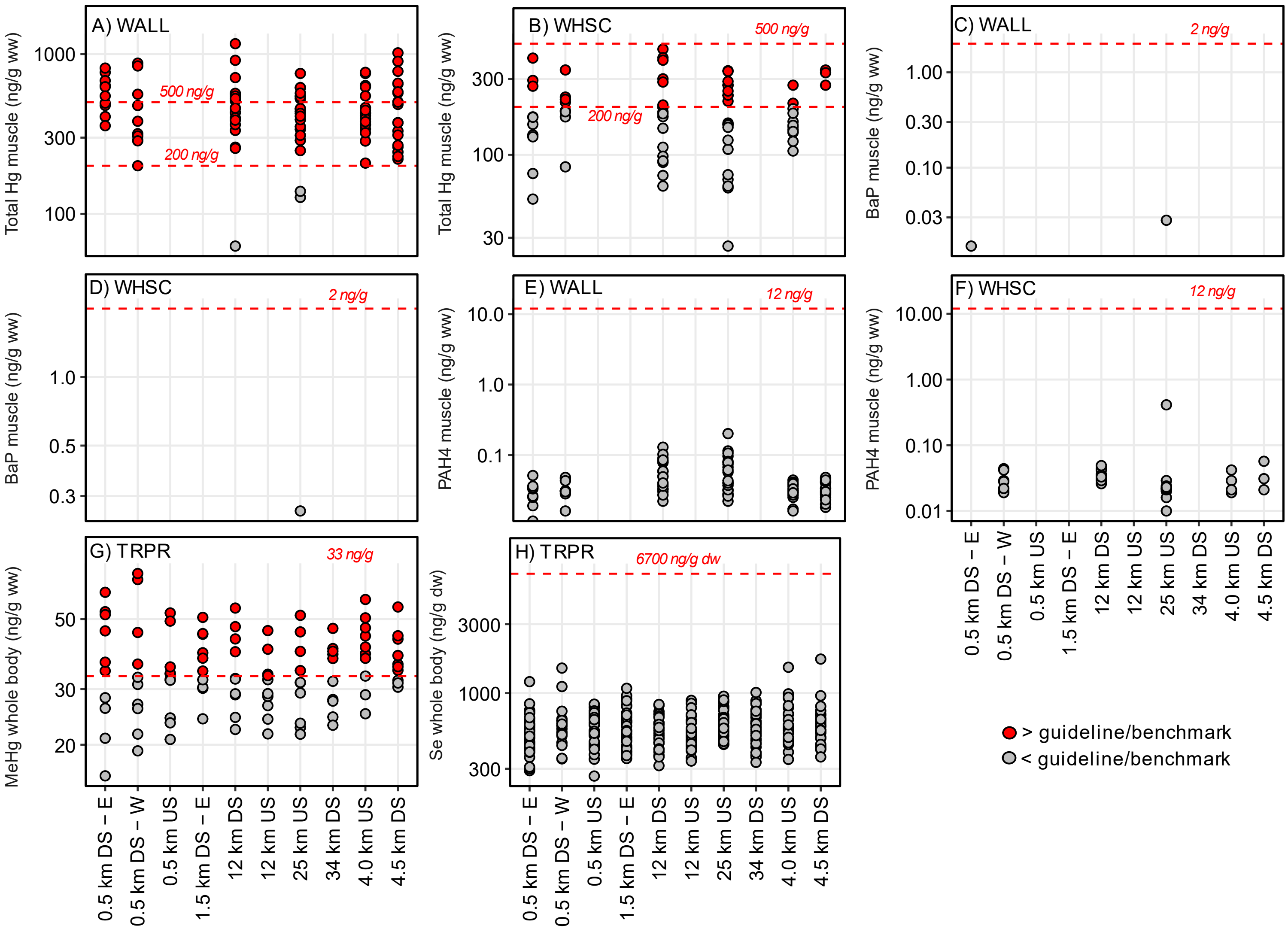

Figure 70 Tissue body burden benchmarks for compounds of concern in Walleye (A, C, E), White Sucker (B, D, F), and Trout-perch (G, H) measured under the EMP (2018, 2019, and 2021). Horizontal red lines represent the benchmark value/guideline.

2.3.9.2 Explanatory Models

Results from the GLMs for fish body and tissue burdens are summarized in Table 34. For Trout-perch, the models determined that discharge was a significant predictor (p-value < 0.05) of variations in concentrations of EROD and total PAH in female whole-body samples, $\delta^{15}N$ in male muscle tissues, as well as EROD, MeHg, Hg, and Se in male whole-body samples. Fork length was a significant predictor of variations of $\delta^{15}N$ in female muscle tissues, EROD, ΣPAH4, and total PAH levels in female whole-body samples, $\delta^{15}N$ and $\delta^{13}C$ in male muscle tissues, and EROD, MeHg, Hg, and Se in male whole-body samples. Year and/or the interaction between year and discharge was a significant predictor of variations of EROD and Total PAH levels in female whole-body samples, $\delta^{15}N$ in male muscle tissues, and EROD and Se in male whole-body samples. Finally, distance from the proposed OSPW discharge point was a significant predictor of EROD and total PAH levels in female whole-body samples, $\delta^{15}N$ in male muscle tissues, and EROD in male whole-body samples.

For Walleye, discharge was a significant predictor (p-value < 0.05) of variations of $\delta^{13}C$, Hg, and total PAH in female muscle samples and $\delta^{13}C$ and Hg in male muscle samples. Fork length was a significant predictor of Hg in female muscle samples and $\delta^{15}N$ and Hg in male muscle samples. Finally, distance from the proposed OSPW discharge point was not a significant predictor for any of the focused subset of compounds. Year and the interaction between year and discharge was not included in the models for Walleye since sampling did not occur over a minimum of three years.

For White Sucker, discharge was a significant predictor of variation of $\delta^{13}C$ and ΣPAH4 levels in female muscle samples and was not significant for any of the focused subset of compounds for male muscles samples. Fork length was a significant predictor of $\delta^{15}N$, MeHg, and Hg in female muscle samples and $\delta^{15}N$, MeHg, Hg, and Se in male muscle samples. Finally, distance from the proposed OSPW discharge point was a significant predictor for total PAH levels in female muscles samples only. Year and the interaction between year and discharge was not included in the models for Walleye since sampling did not occur over a minimum of three years.

Temporal and spatial differences were further investigated below in Section 2.3.8.3 on fork length and Q60 normalized fish body burden concentrations. Normalization equations are provided in Appendix C Table C6.

Table 34 Results of statistical analyses assessing variation in different fish tissue body burdens for species collected along the LAR under the EMP (2018, 2019, 2021).

Species	Sample Matrix	Sex	Analyte	Q60 (m3/s)		FL (mm)		Year		Distance (US/DS)		Q60xYear	
				P-Val	%VE	P-Val	%VE	P-Val	%VE	P-Val	%VE	P-Val	%VE
Trout Perch	F	Muscle	13C	0.493	2.8	0.843	0.2	-	-	-	-	-	-
			15N	0.709	0.6	**0.025**	26.0	-	-	-	-	-	-
		Whole Body	BaP	0.154	28.8	-	-	0.804	0.7	0.671	2.2	0.591	3.5
			EROD	**<0.001**	28.4	**<0.001**	1.7	**<0.001**	7.6	**0.013**	0.6	0.141	0.2
			PAH4	0.13	1.9	-	-	0.184	1.4	0.390	0.6	**0.010**	5.5
			Total (Wet Wt) Mercury (Hg)	0.117	15.4	0.626	1.4	-	-	-	-	-	-
			Total (Wet Wt) Selenium (Se)	0.566	2.2	0.975	<0.1	-	-	-	-	-	-
			Total PAH	**<0.001**	46.2	-	-	**<0.001**	9.8	**0.010**	2.4	**0.005**	2.8
	M	Muscle	13C	0.212	0.5	**0.002**	3.1	0.109	0.8	0.441	0.2	0.687	0.1
			15N	**<0.001**	7.9	0.168	0.5	**<0.001**	3.4	**0.009**	2.0	**0.007**	2.1
		Whole Body	EROD	**<0.001**	16.9	**<0.001**	1.9	**<0.001**	9.8	**0.010**	0.7	**<0.001**	6.0
			Methyl Mercury (wet weight)	**0.031**	4.5	**0.015**	5.7	-	-	0.538	0.4	-	-
			Total (Wet Wt) Mercury (Hg)	**<0.001**	23.9	**0.002**	2.3	0.127	0.6	0.582	0.1	0.181	0.4
			Total (Wet Wt) Selenium (Se)	**0.048**	1.0	**0.001**	3.0	**<0.001**	21.1	0.824	<0.1	0.606	0.1
Walleye	F	Muscle	13C	**<0.001**	41.2	0.780	0.2	-	-	0.086	8.6	-	-
			15N	0.689	0.9	0.998	<0.1	-	-	0.773	0.4	-	-
			PAH4	0.177	9.5	0.545	1.8	-	-	0.536	1.9	-	-
			Total (Wet Wt) Mercury (Hg)	**<0.001**	33.5	**0.003**	24.6	-	-	0.619	0.6	-	-
			Total (Wet Wt) Selenium (Se)	0.958	<0.1	0.356	4.3	-	-	0.306	5.3	-	-
			Total PAH	**0.013**	29.4	0.742	0.4	-	-	0.695	0.6	-	-
	M		13C	**<0.001**	44.0	0.592	0.3	-	-	0.731	0.1	-	-
			15N	0.688	0.3	**0.039**	8.1	-	-	0.512	0.8	-	-
			Methyl Mercury (wet weight)	0.514	2.3	0.102	16.0	-	-	0.222	8.5	-	-
			PAH4	0.302	2.1	0.222	3.0	-	-	0.489	1.0	-	-
			Total (Wet Wt) Mercury (Hg)	**0.003**	13.6	**0.001**	17.0	-	-	0.556	0.5	-	-

Species	Sample Matrix	Sex	Analyte	Q60 (m3/s)		FL (mm)		Year		Distance (US/DS)		Q60xYear	
				P-Val	%VE	P-Val	%VE	P-Val	%VE	P-Val	%VE	P-Val	%VE
			Total (Wet Wt) Selenium (Se)	0.558	0.7	0.822	0.1	-	-	0.854	0.1	-	-
			Total PAH	0.144	4.2	0.262	2.4	-	-	0.268	2.4	-	-
White Sucker	Muscle	F	13C	**0.005**	7.6	0.464	0.5	-	-	0.229	1.3	-	-
			15N	0.353	0.7	**<0.001**	20.8	-	-	0.576	0.2	-	-
			Methyl Mercury (wet weight)	0.516	2.2	**0.016**	56.1	-	-	0.090	19.5	-	-
			PAH4	**0.02**	30.6	0.457	2.6	-	-	0.157	9.9	-	-
			Total (Wet Wt) Mercury (Hg)	0.418	1.2	**3E-05**	40.6	-	-	0.092	5.2	-	-
			Total (Wet Wt) Selenium (Se)	0.672	0.5	0.055	10.2	-	-	0.055	10.2	-	-
			Total PAH	0.192	6.7	0.166	7.6	-	-	**0.005**	39.6	-	-
		M	13C	0.074	3.9	0.355	1.0	-	-	0.342	1.1	-	-
			15N	0.700	0.2	**0.027**	5.9	-	-	0.355	1.0	-	-
			Methyl Mercury (wet weight)	0.284	8.5	**0.042**	38.7	-	-	0.277	8.8	-	-
			PAH4	0.848	0.3	0.780	0.6	-	-	0.947	<0.1	-	-
			Total (Wet Wt) Mercury (Hg)	0.877	0.1	**0.012**	17.7	-	-	0.950	<0.1	-	-
			Total (Wet Wt) Selenium (Se)	0.397	1.4	**<0.001**	29.7	-	-	0.058	7.2	-	-
			Total PAH	0.285	7.0	0.217	9.4	-	-	0.361	5.0	-	-

Table Notes: Significant values (i.e., $p < 0.05$) are in bold
FL represents fork length (mm)
%VE represents the percentage of the total variance explained by each predictor within the individual models
Shaded cells highlight the %VE that corresponds to significant p-values
ΣPAH4 is the sum of benzo[a]anthracene, chrysene, benzo[b]fluoranthene, and benzo[a]pyrene concentrations.

2.3.9.3 Visualization of Trends

PCA plots of body and tissue burden profiles used to identify potential temporal and spatial trends across EMP sampling years and stations are provided in Figure 71, Figure 72 and Figure 73.

The Trout-perch PCA plots indicate that the tissue burden profiles for female and male Trout-perch were relatively similar in 2019 and 2021, as indicated by the overlapping ellipses (Figure 71). The PCA plots also indicate no clear spatial separations of samples collected across the different stations. Trends in male Trout-perch data were explored further using scatterplots and boxplots of select parameters with relevant consumption guidelines (i.e., total mercury and selenium), implications for toxicity (i.e., EROD), and implications for food web dynamics (i.e., δ13C and δ15N), provided in Figure 74 to Figure 78. Male Trout-perch data was selected to display spatial and temporal trends, as this dataset is the most complete in terms of years and stations sampled (Appendix B Table B9). Data was normalized to fork length and Q60 prior to plotting, the normalization equations can be found in Appendix C (Table C6). There appears to be a slight decreasing temporal trend in normalized δ15N and total selenium values, while the remaining compounds do not show any apparent difference over time. No compounds demonstrated an apparent trend with distance.

The Walleye PCA plots indicate that the tissue burden profiles for female and male Walleye differed in 2019 and 2021, as indicated by the separation of the ellipses (Figure 72). Samples collected in 2019 are more negatively correlated with PC axis 2, being driven by the presence of Al and As, whereas 2021 samples seem to be more positively correlated with PC axis 2, driven by Zn, Na, and Hg. The PCA plots also indicate no clear spatial separation of samples collected across the different stations, with overlapping ellipses suggesting similarities in tissue burdens across the EMP sampling stations.

The White Sucker PCA plots indicate that the tissue burden profiles for female and male White Sucker differed in 2019 and 2021, as indicated by the separation of the ellipses (Figure 73).Samples collected in 2019 tended to group tightly near the origin, showing some negatively correlation with PC axis 2, this appears to be driven by the presence of Zn, P, and Ti in the samples. Samples collected in 2021 were much more dispersed along the PCA axes. The PCA plots also indicate no clear spatial separation of samples collected across the different stations, with overlapping ellipses suggesting similarities in tissue burdens across the EMP sampling stations.

An important spatial comparison to consider is the difference between samples collected on the east and west side of the island located 0.5 km downstream of the proposed OSPW discharge point, as these stations are directly downstream of the Syncrude sewage treatment outfall (Figure 1). Differences among samples collected from the east and west side of the island were explored using a Tukey’s post-hoc test for individual comparisons. Results are provided in Table 35. The only analyte that varied significantly between the two sides of the island within the LAR was $\delta^{13}C$, which was significantly higher on the west side of the island in male whole body Walleye samples.

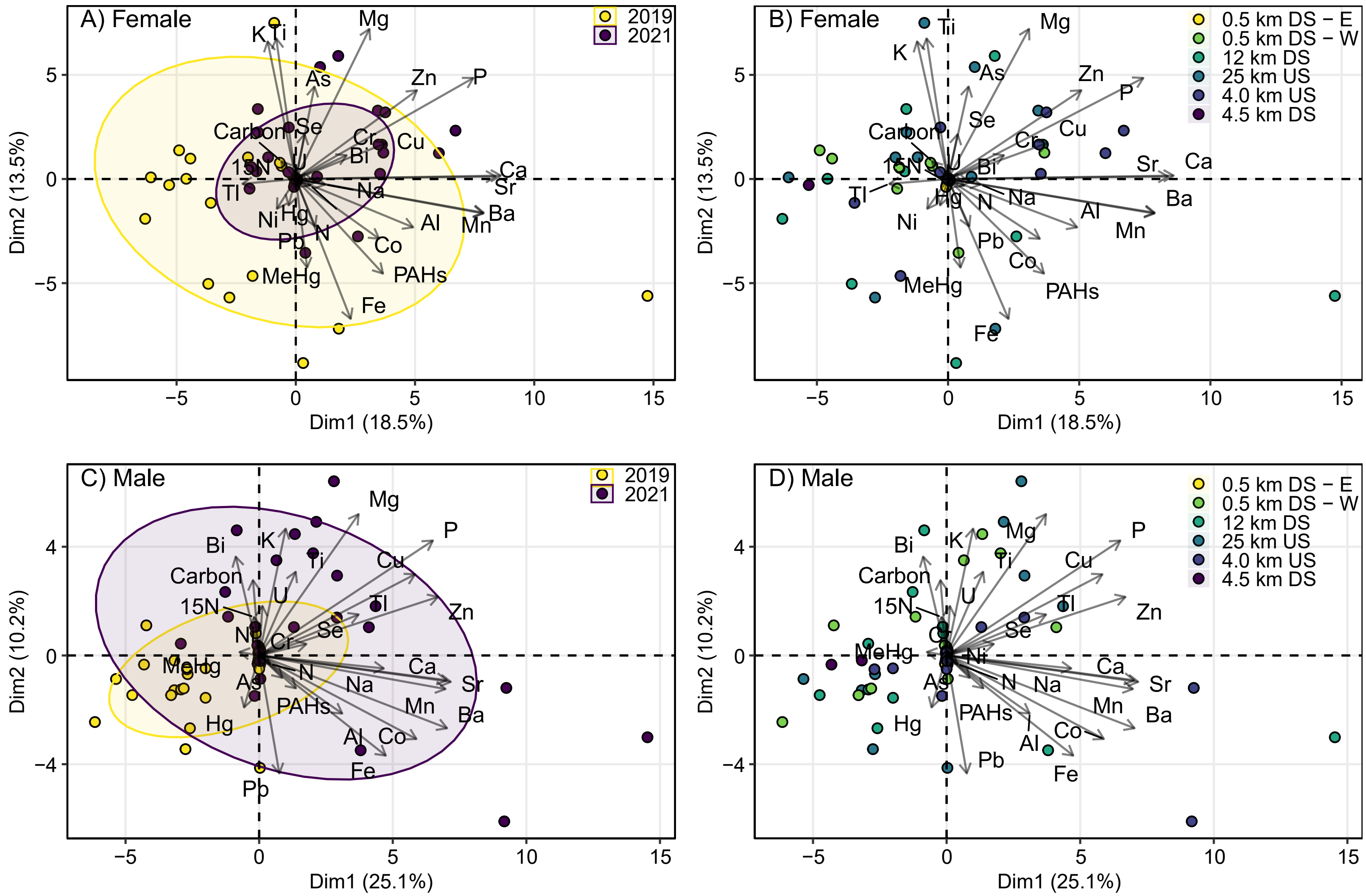

Figure 71 PCA for Trout-perch demonstrating temporal and spatial trends across EMP sampling years and stations.

Figure Notes: Concentrations were adjusted to a fork length of 60mm for each sample

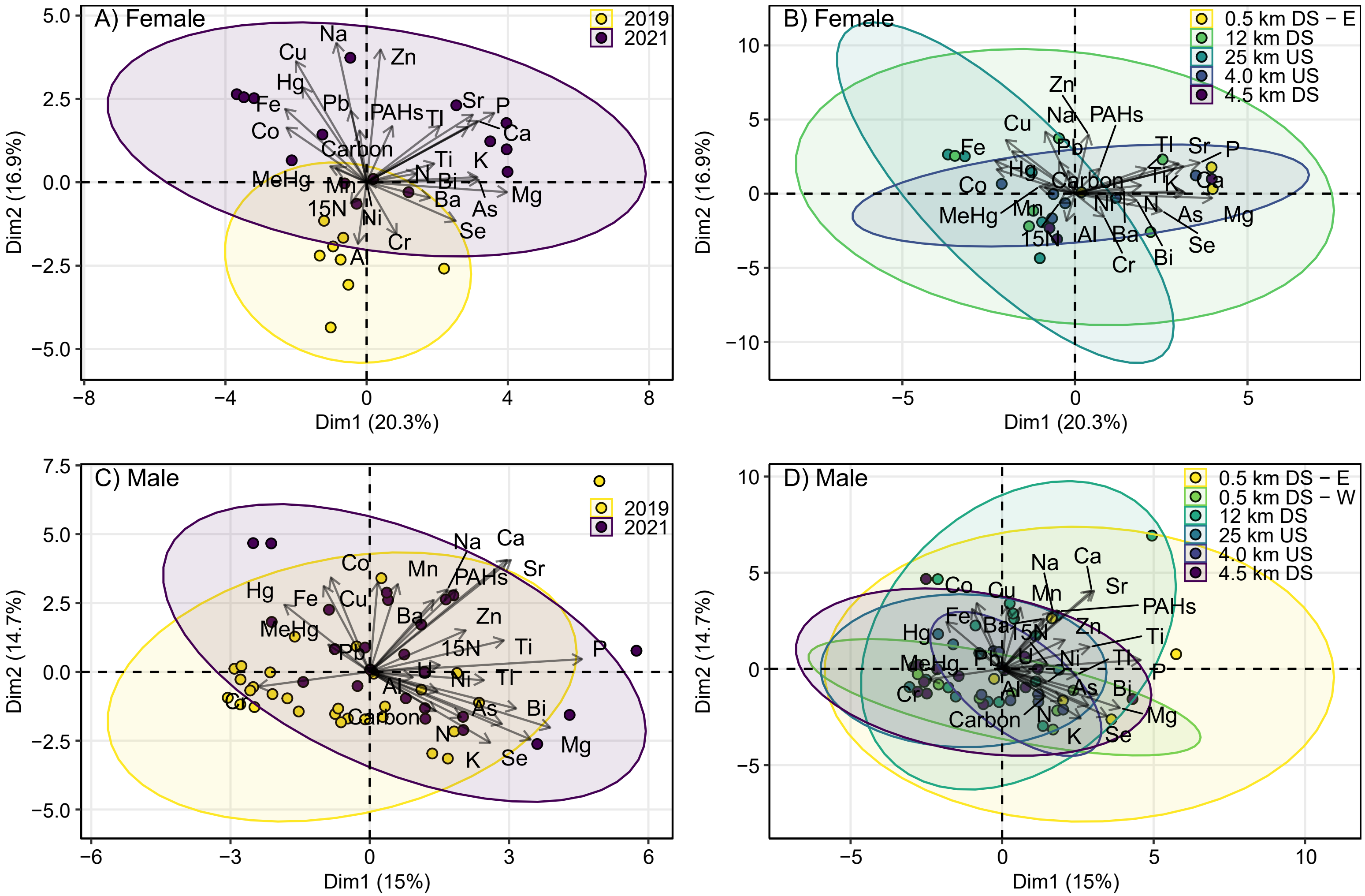

Figure 72 PCA for Walleye demonstrating temporal (A-female, B-male) and spatial (C-male, D-female) trends across EMP sampling years and stations.

Figure Notes: Concentrations were adjusted to a fork length of 450mm for each sample

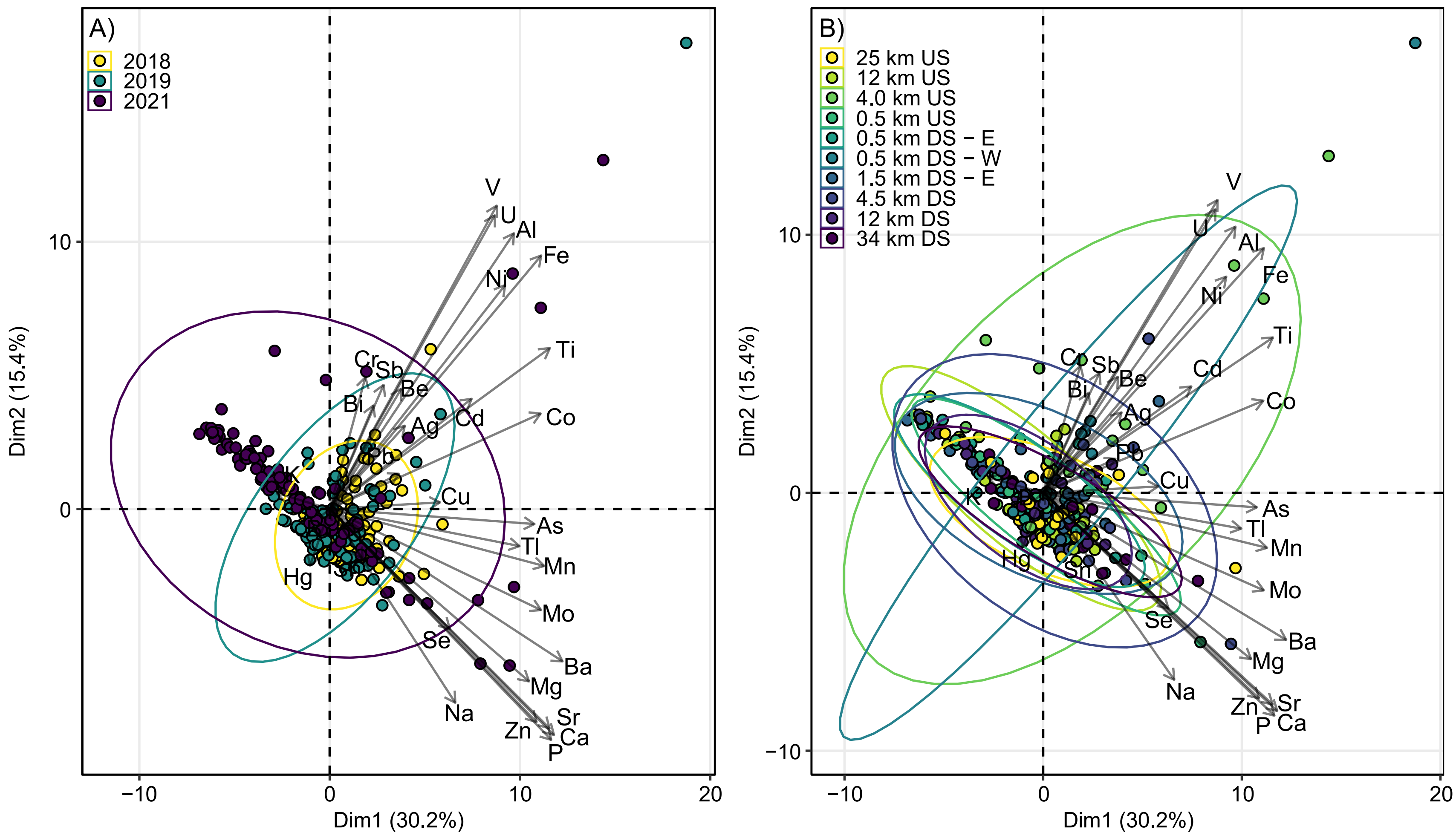

Figure 73 White Sucker demonstrating temporal (A-female, B-male) and spatial (C-male, D-female) trends across EMP sampling years and stations.

Figure Notes: Concentrations were adjusted to a fork length of 450mm for each sample

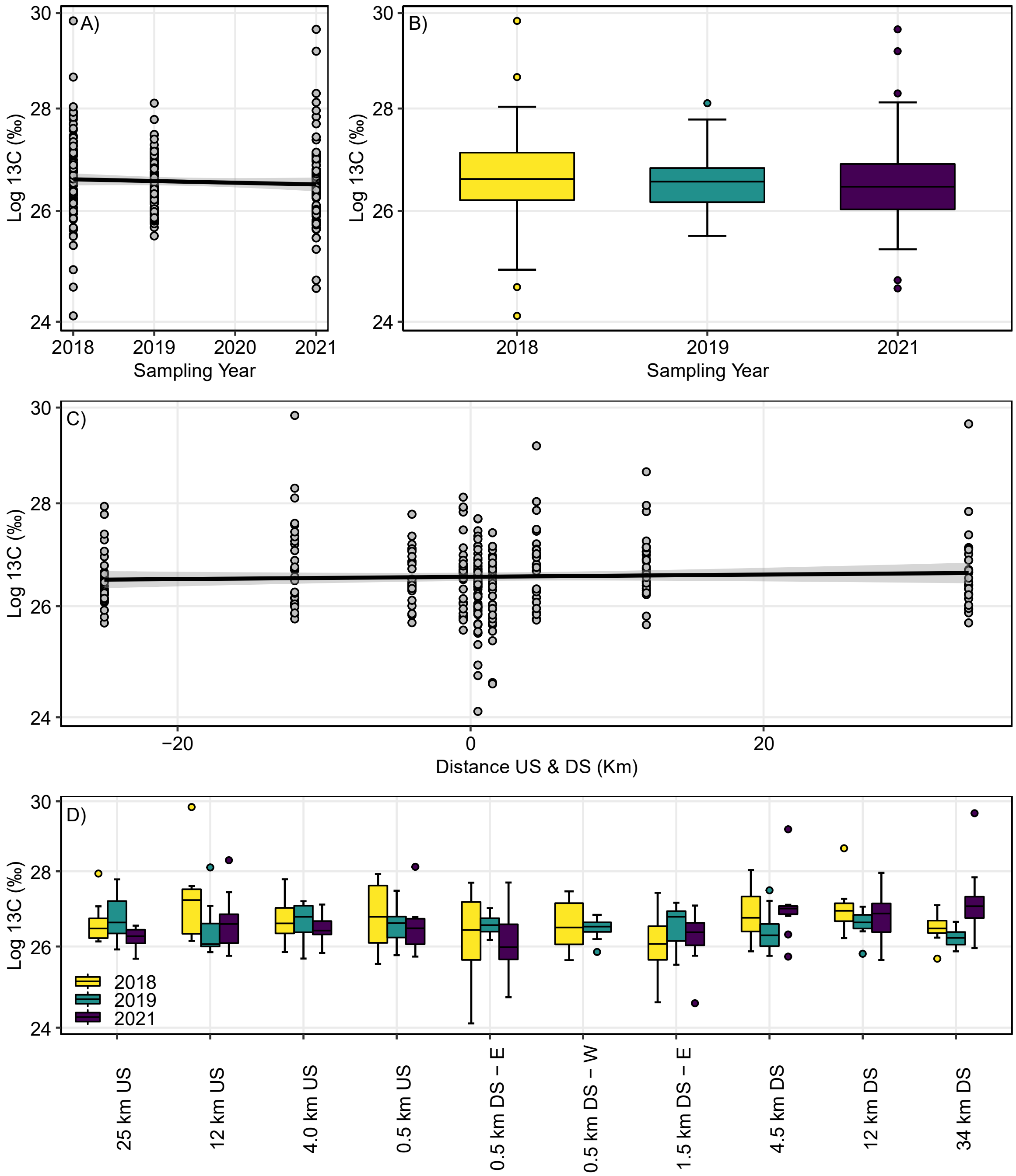

Figure 74 Variations in δ[13]C ratio in male Trout-perch muscle samples over time (A,B) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Concentrations were standardized to a fork length of 60 mm and a Q60 of 600 m^3/s.

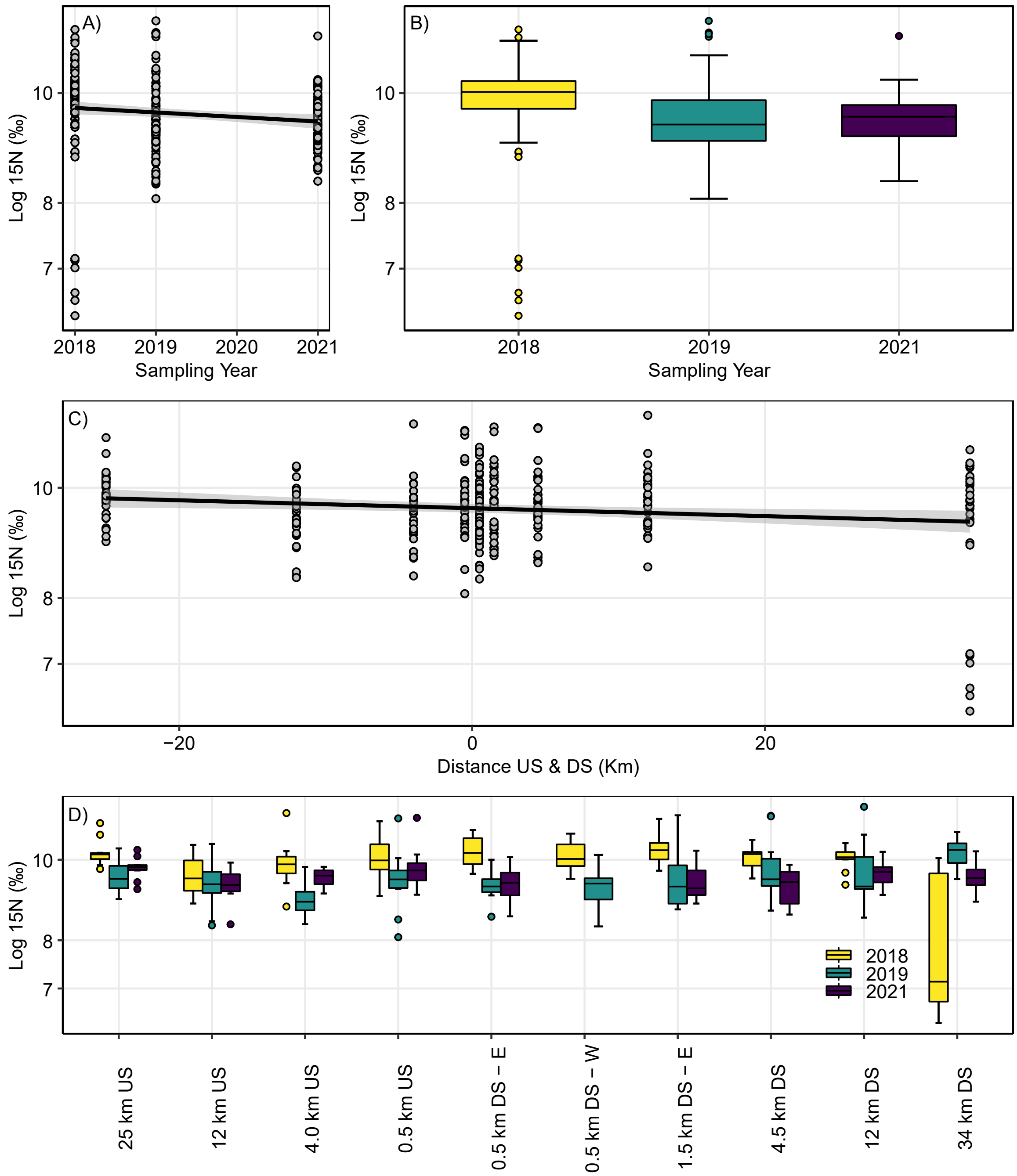

Figure 75 Variations in δ^{15}N ratio in male Trout-perch muscle samples over time (A,B) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Concentrations were standardized to a fork length of 60 mm and a Q60 of 600 m^3/s.

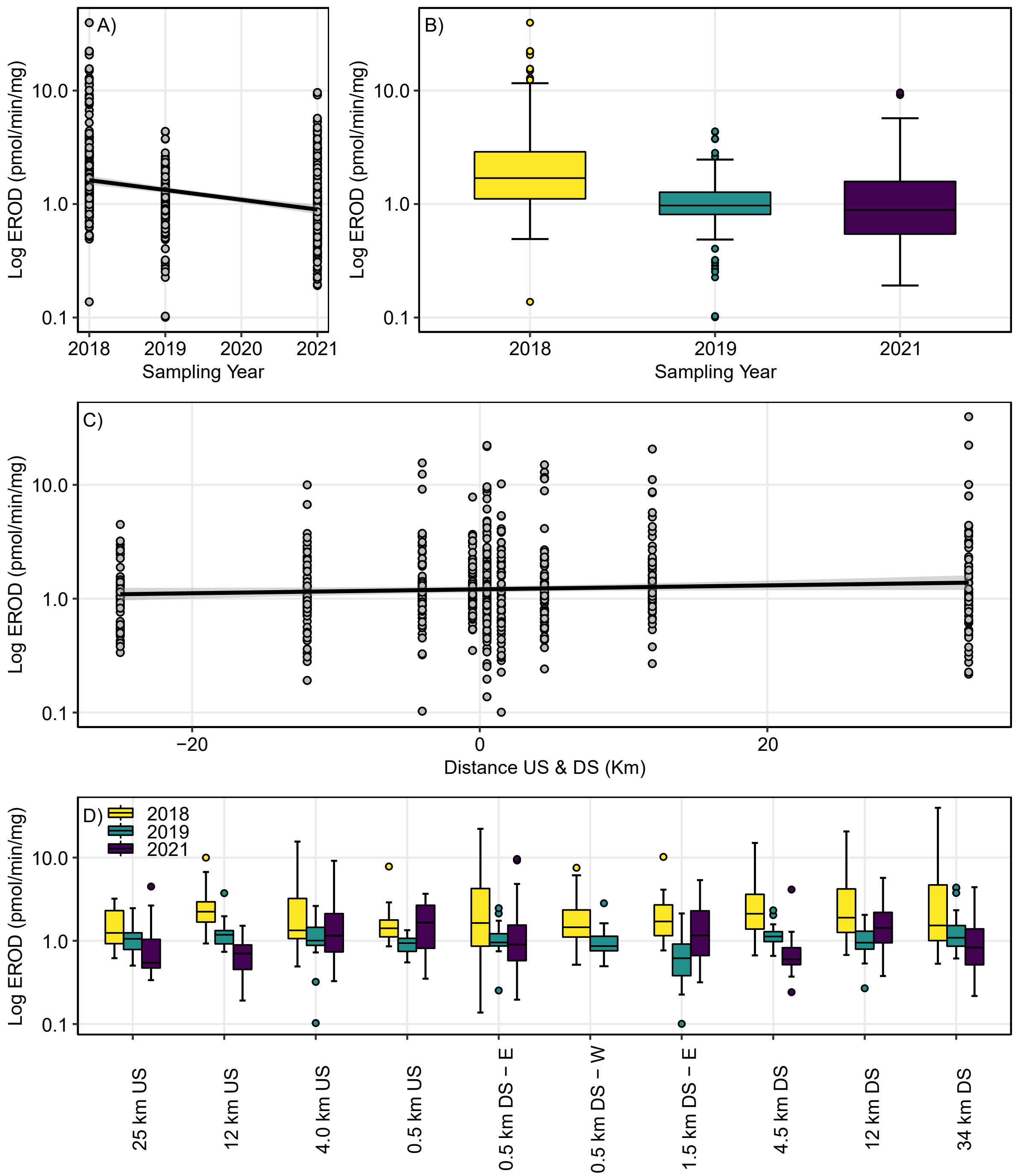

Figure 76 Variations in EROD in male Trout-perch whole-body samples over time (A,B) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Concentrations were standardized to a fork length of 60 mm and a Q60 of 600 m^3/s.

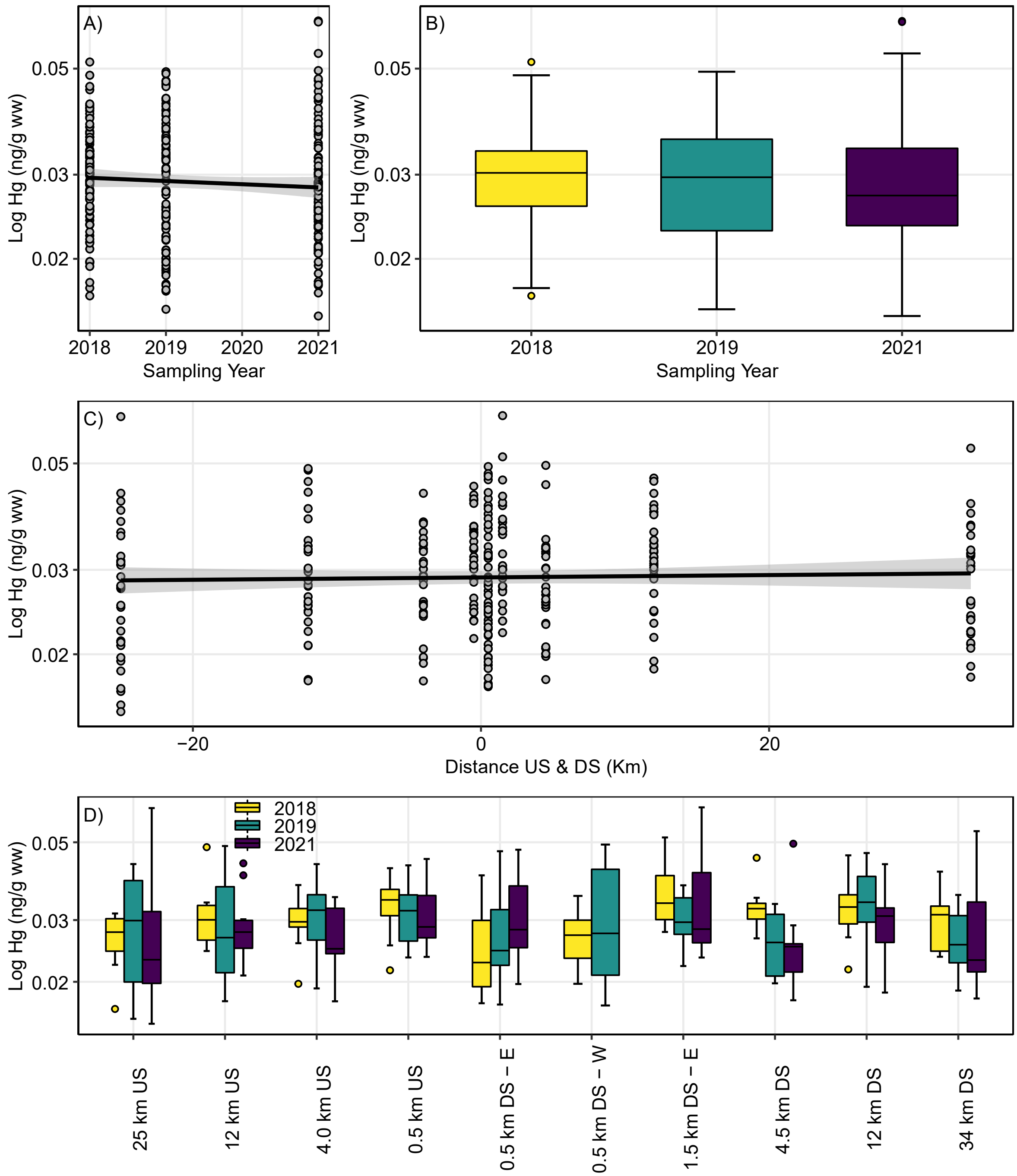

Figure 77 Variations in total mercury in male Trout-perch whole-body samples over time (A,B) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Concentrations were standardized to a fork length of 60 mm and a Q60 of 600 m^3/s.

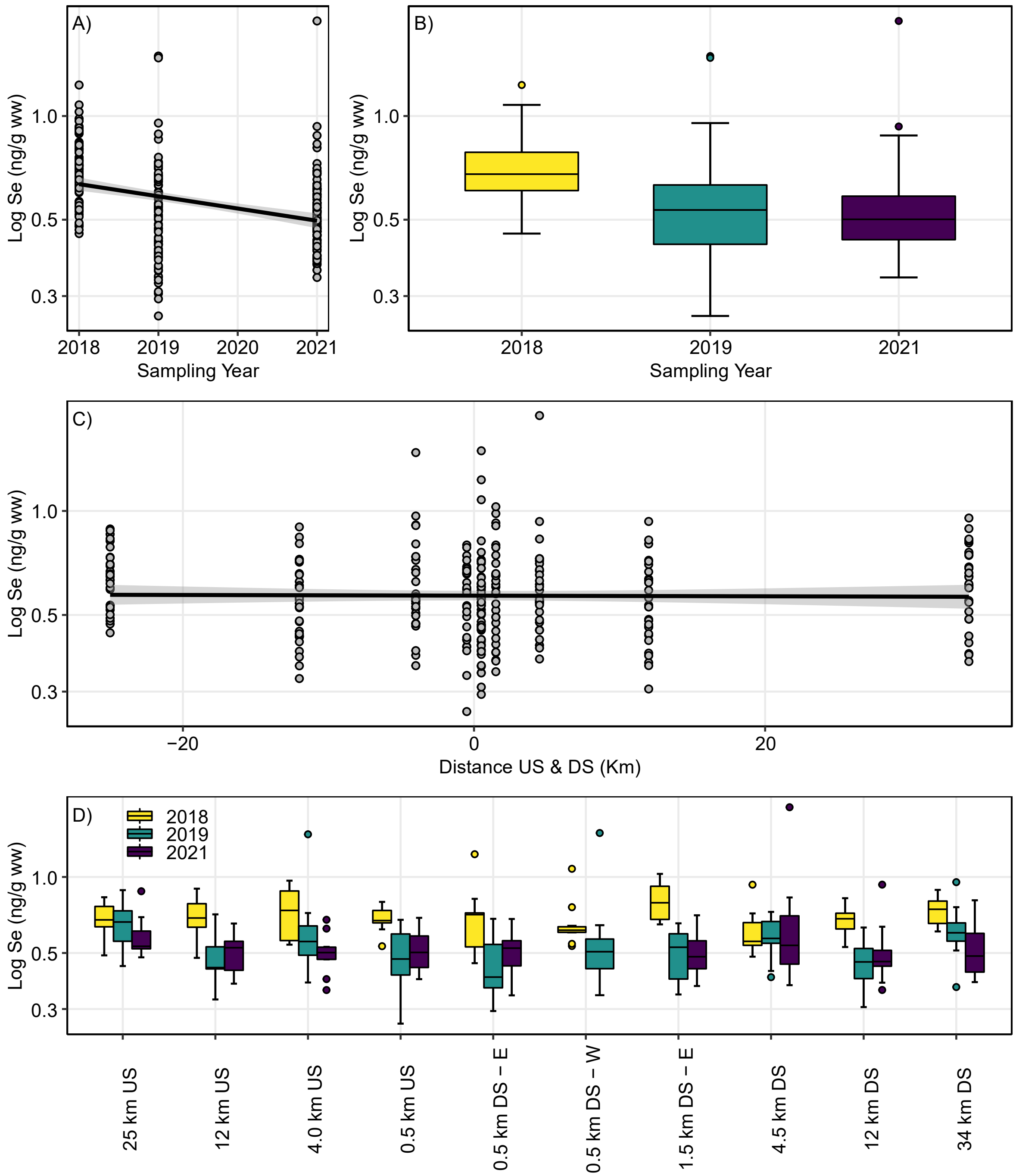

Figure 78 Variations in total selenium in male Trout-perch whole-body samples over time (A,B) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Figure Notes: Concentrations were standardized to a fork length of 60 mm and a Q60 of 600 m^3/s.

Table 35 Tukey's post-hoc test comparing fork length normalized body burden parameters measured in fish (female and male data pooled) captured at the stations located both East and West of the island, EMP dataset (2018, 2019, and 2021).

Species	Sex	Sample matrix	Analyte	0.5 E vs. 0.5 W	P-value
Trout-perch	F	Liver	EROD	E > W	0.309
	M	Muscle	$\delta^{13}C$	E > W	0.233
			$\delta^{15}N$	E > W	0.975
		Whole Body	EROD	E > W	0.0817
			Methyl Mercury	E > W	0.982
			Total Mercury	E < W	0.218
			Total Selenium	E < W	0.172
Walleye		Muscle	$\delta^{13}C$	E < W	**<0.001**
			$\delta^{15}N$	E < W	0.534
			Total Mercury	E > W	0.138
			Total Selenium	E < W	0.645
			ΣPAH4	E > W	0.602
			Total PAH	E > W	0.109

Table Notes: Missing analytes and/or species/sexes represent cases where the specific sites were not samples and/or there was not sufficient data to calculate a mean value for the comparison.
ΣPAH4 is the sum of benzo[a]anthracene, chrysene, benzo[b]fluoranthene, and Benzo[a]pyrene concentrations.
Significant p-values (i.e., < 0.05) are in bold.

2.3.10 Benthic Body Burden

The EMP data consisted of measurements of metals, mercury, and SIRs in benthic samples consisting of Ametropodidae, Chironomidae, Gomphidae, and Pteronarcyidae families. Benthic samples were collected from a total of 10 different EMP stations over three different sampling years (2018, 2019, and 2021), however analytes measured in each sample differed across family, station, and year, as discussed below. Measured analytes were classified into three groups: SIRs, mercury (total and methyl mercury) and metals.

2.3.10.1 Data Summary

2.3.10.1.1 Data Availability

Ametropodidae was the most frequently sampled benthic family across both sampling stations and years. All three analyte groups were consistently measured across the three sampling years and most stations (aside from stations located 0.03 km upstream and downstream in 2021). For Gomphidae, all three analyte groups were measured at two stations in 2018, a single station in 2019, and three stations in 2021. Pteronarcyidae was the most seldom sampled benthic family, with no stations sampled in 2018, two stations sampled in 2019, and two stations sampled in 2021.

2.3.10.1.2 Non-Detects

A summary of VMV codes, method detection limits, and non-detects can be found in Appendix A Table A1. Ametropodidae body burden samples were below detection limits for metals in 0.2 to 0.7 % of samples across all sampling years and stations, no samples were below detection limits for mercury (Figure 79A). Gomphidae body burden samples were below detection limits for metals in 0 to 3.3 % of samples across all sampling years and stations, no samples were below detection limits for mercury (Figure 79B). Finally, Pteronarcyidae body burden samples were below detection limits for metals in 1.1 to 1.7 % of samples across all sampling years and stations, no samples were below detection limits for mercury (Figure 79C).

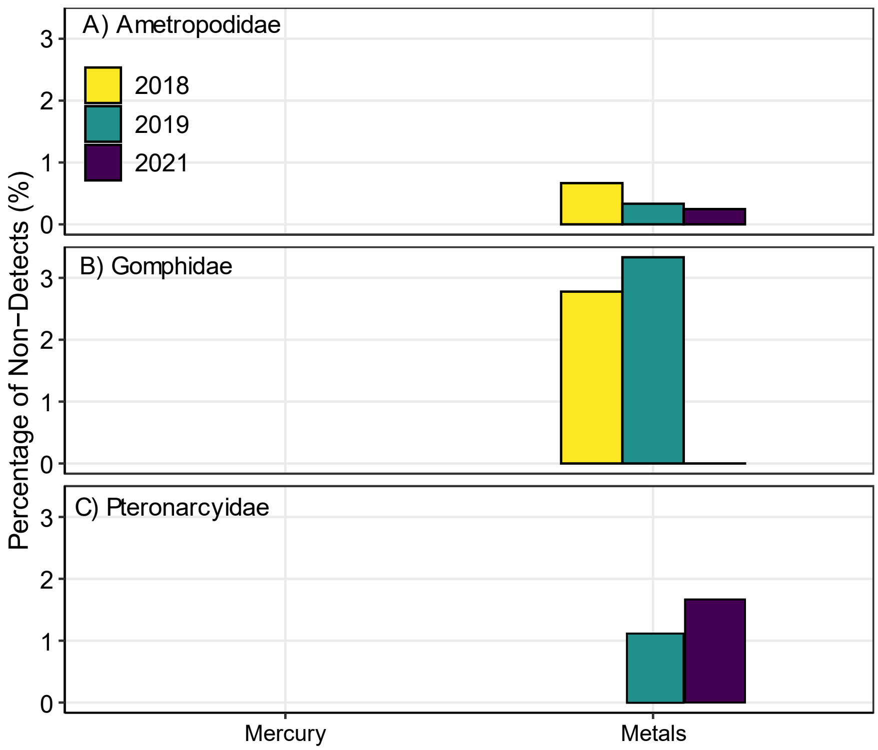

Figure 79 Percentage of non-detects observed in Ametropodidae (A), Gomphidae (B), and Pteronarcyidae (C) body burden samples by parameter category in the EMP dataset (2018, 2019, and 2021)

2.3.10.1.3 Summary Statistics

Summary statistics for key analytes, by benthic family, pooled for all stations and years are provided in Table 36.

Detailed isotope summary statistics, broken down by station and year, are provided in Appendix B Table B10. Mean $\delta^{13}C$ values ranged from -30.2 ± 1.7 ‰ (± SD, n = 52) in Ametropodidae, -28.5 ± 1.5 ‰ (± SD, n = 13) in Gomphidae, and -28.8 ± 1.2 ‰ (± SD, n = 9) in Pteronarcyidae. Mean $\delta^{15}N$ values ranged from 5.2 ± 1.3 ‰ (± SD, n = 52) in Ametropodidae, 6.5 ± 1.3 ‰ (± SD, n = 13) in Gomphidae, and 6.4 ± 2.1 ‰ (± SD, n = 9) in Pteronarcyidae. Mean methyl mercury values ranged from 1.6 ± 0.2 ng/g w.w. (± SD, n = 56) in Ametropodidae, 6.7 ± 1.4 ng/g w.w. (± SD, n = 21) in Gomphidae, and 2.3 ± 1.5 ng/g w.w. (± SD, n = 10) in Pteronarcyidae. Mean total mercury values ranged from 3.8 ± 0.8 ng/g w.w. (± SD, n = 56) in Ametropodidae, 11.3 ± 4.9 ng/g w.w. (± SD, n = 21) in Gomphidae, and 6.6 ± 2.6 ng/g w.w. (± SD, n = 10) in Pteronarcyidae. Finally, mean total selenium values ranged from 0.4 ± 0.3 ng/g w.w. (± SD, n = 56) in Ametropodidae, 0.8 ± 0.6 ng/g w.w. (± SD, n = 21) in Gomphidae, and 0.3 ± 0.2 ng/g w.w. (± SD, n = 10) in Pteronarcyidae.

Table 36 Summary statistics for body burden concentrations in benthic families collected during the EMP program pooled across sampling years (2018, 2019 and 2021). A subset list of compounds is presented

Family	Analyte	Units	Min	Max	Mean	SD	N
Ametropodidae	$\delta^{13}C$	‰	-33.0	-25.8	-30.2	1.7	52
	$\delta^{15}N$	‰	3.5	9.1	5.2	1.3	52
	Methyl Mercury	ng/g w.w.	1.1	1.9	1.6	0.2	29
	Total Mercury	ng/g w.w.	2.1	5.2	3.8	0.8	56
	Total Selenium	ng/g w.w.	0.2	0.4	0.3	0.1	57
Gomphidae	$\delta^{13}C$	‰	-31.9	-26.3	-28.5	1.5	13
	$\delta^{15}N$	‰	4.3	8.6	6.5	1.3	13
	Methyl Mercury	ng/g w.w.	4.7	8.7	6.7	1.4	16
	Total Mercury	ng/g w.w.	6.5	26.3	11.3	4.9	21
	Total Selenium	ng/g w.w.	0.2	2.7	0.8	0.6	21
Pteronarcyidae	$\delta^{13}C$	‰	-31.2	-27.6	-28.8	1.2	9
	$\delta^{15}N$	‰	3.7	10.6	6.4	2.1	9
	Methyl Mercury	ng/g w.w.	1.2	5.3	2.3	1.5	6
	Total Mercury	ng/g w.w.	3.4	11.3	6.6	2.6	10
	Total Selenium	ng/g w.w.	0.2	0.9	0.3	0.2	10

Table Notes: w.w. represents wet weight

2.3.10.1.4 Mercury

Comparison of paired methylmercury and total mercury data suggests that between 24.9 and 51.7% for Ametropodidae, 66.6 to 83.6% for Gomphidae, and 22.2 to 49.4% for Pteronarcyidae of the total mercury measurement is made up of methylmercury (Table 37). Further, when regressing the log transformed concentration of methylmercury against the log transformed concentration of total mercury as shown in Figure 80, the relationship yields a slope of 1.70 and an R^2 of 0.87. This suggests that similar to fish body burden, methylmercury in benthic macroinvertebrates can be predicted reasonably well from total mercury concentrations and the analysis of only total mercury is would be sufficient to predict levels of methylmercury.

Table 37 Summary statistics for the percentage of methylmercury accounted for in total mercury measurements among benthic families

Species	% of MeHg in Total Hg			
	Min	Max	Mean	SD
Ametropodidae	24.9	51.7	39.0	5.45
Gomphidae	66.6	83.6	74.0	4.68
Pteronarcyidae	22.2	49.4	49.5	9.78

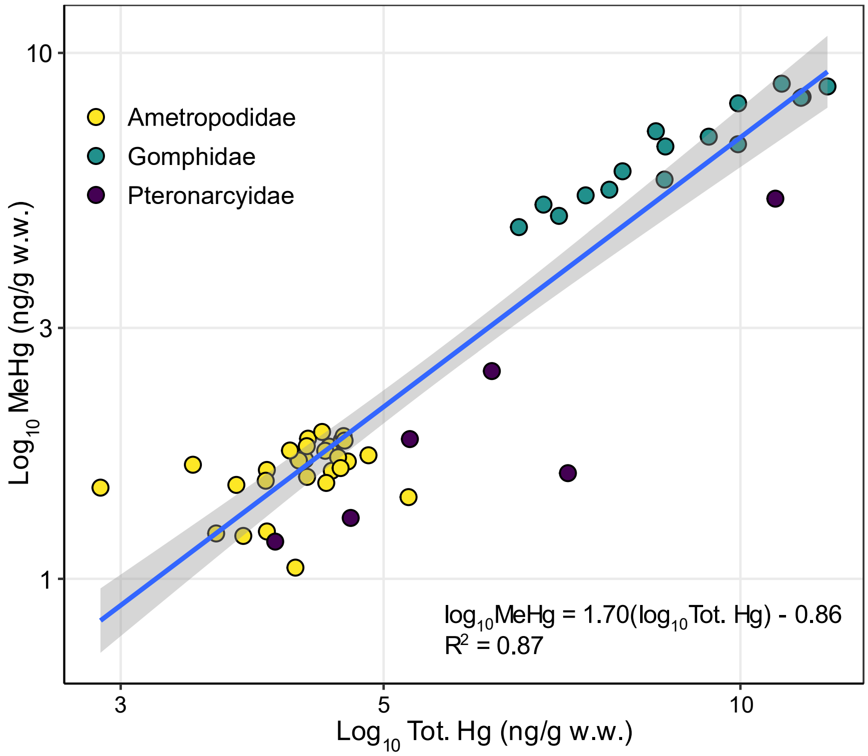

Figure 80 Relationship between log transformed methylmercury and total mercury in paired datasets for Ametropodidae, Gomphidae, and Pteronarcyidae.

2.3.10.2 Explanatory Models

Results from the GLMs for benthic body burdens are summarized in Table 38. Benthic body burden dataset produced during EMP sampling did not provide a unique sampling date for each sample to be used to pair with discharge data, therefor discharge could not be used as a predictor. For Ametropodidae, $\delta^{13}C$, $\delta^{15}N$, total mercury, methyl mercury, and total selenium varied significantly with sampling year ($p < 0.001$ for all analytes; Table 38), accounting for between 17 and 74 % of the explained variance in the models. The only analytes that varied significantly with distance upstream to downstream of the proposed OSPW discharge point were $\delta^{13}C$ and $\delta^{15}N$ (p-value = 0.026 and 0.003, respectively; Table 38). For Gomphidae, total mercury was the only analyte that varied significantly with sampling year ($p < 0.001$; Table 38), accounting for 78 % of the explained variance in the models. The only analyte that varied significantly with distance upstream to downstream of the proposed OSPW discharge point was $\delta^{15}N$ (p-value = 0.009; Table 38), accounting for 39 % of the explained variance in the models. For Pteronarcyidae, none of the subset list of analytes varied significant with sampling year or distance (Table 38). Temporal and spatial differences were further investigated below in Section 2.3.10.3.

Table 38 Results of statistical analyses assessing variation in different benthic body burdens for samples collected along the LAR under the EMP (2018, 2019, 2021).

Analyte	Ametropodidae				Gomphidae				Pteronarcyidae			
	Year		Distance (US/DS)		Year		Distance (US/DS)		Year		Distance (US/DS)	
	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE
$\delta^{13}C$	**<0.001**	19.8	**0.026**	5.8	0.250	6.1	**0.009**	39.1	0.345	12.8	0.908	0.2
$\delta^{15}N$	**<0.001**	73.9	**0.003**	3.3	0.074	22.2	0.620	1.5	0.153	26.8	0.990	<0.1
Total Mercury	**<0.001**	41.7	0.538	0.3	**<0.001**	77.8	0.306	1.1	0.841	0.5	0.423	8.1
Methylmercury	**<0.001**	32.8	0.308	1.9	0.105	17.1	0.209	9.8	-	-	0.079	58.0
Selenium	**<0.001**	17.2	0.183	2.0	0.119	10.7	0.321	4.2	0.604	3.4	0.543	4.6

Table Notes: Significant values (i.e., p-value < 0.05) are in bold.
%VE represents the percentage of total variance explained by each predictor within the individual models.
Shaded cells highlight the %VE that corresponds to significant p-values.

2.3.10.3 Visualization of Trends

PCA plots of body burden profiles for metals used to identify potential temporal and spatial trends across EMP sampling years and stations are provided in Figure 81 and Figure 82.

The PCA revealed that the body burden profiles for Ametropodidae were relatively similar in 2018 and 2019, with each respective ellipse showing considerable overlap and positive correlation with PC axis 2, driven by similarities Co and Mo body burdens (Figure 81A). Samples collected in 2021 tend to be more negatively correlated with PC axis 2 compared to 2018 and 2019 samples, driven largely by differences in K, As, and Ca concentrations (Figure 81A). Body burden profiles for Gomphidae were different between 2018 and 2021, with limited overlap between PCA ellipses (Figure 81B) driven by generally elevated concentrations of metals in 2018 compared to 2021. Body burden profiles for Pteronarcyidae were only assessed in 2019 and 2021 and showed nearly complete overlap between PCA ellipses (Figure 81C).

The PCA revealed that the body burden profiles for Ametropodidae were relatively similar across all sampling stations, showing clear overlap between ellipses in all cases except 0.03 km upstream (Figure 82A). Samples collected in 2021 tend to be more negatively correlated with PC axis 2 compared to 2018 and 2019 samples, driven largely by differences in K, As, and Ca concentrations (Figure 82A). Body burden profiles for Gomphidae were different between 2018 and 2021, with limited overlap between PCA ellipses (Figure 81B) driven by generally elevated concentrations of metals in 2021 compared to 2021. Body burden profiles for Pteronarcyidae were only assessed in 2019 and 2021 and showed nearly complete overlap between PCA ellipses (Figure 81C).

Ametropodidae data were explored further using scatterplots and boxplots of select parameters with relevant consumption guidelines (i.e., total mercury, methyl mercury, and selenium) and implications for food web dynamics (i.e., $\delta^{13}C$ and $\delta^{15}N$) provided in Figure 83 to

Figure 89. Ametropodidae data was selected to display spatial and temporal trends, as this is the only benthic family dataset with three years of data. There appears to be a clear decreasing temporal trend in $\delta^{13}C$ values (Figure 83AB) and decreasing trend with distance (Figure 83CD). There is a clear increasing temporal trend in $\delta^{15}N$ values (Figure 84AB) and increasing trend with distance (Figure 84CD). Only two years (2018 and 2019) of methyl mercury data was available, where methyl mercury concentrations were higher in 2019 compared to 2018 (Figure 85AB), with no clear trend with distance (Figure 85CD).Total mercury values demonstrate a clear decreasing temporal trend (Figure 86AB) and no apparent trend with increasing distance downstream of the proposed OSPW discharge point (Figure 86CD). Finally, total selenium concentrations demonstrated an increasing temporal trend (Figure 87AB) but no clear apparent trend with increasing distance (Figure 87CD).

An important spatial comparison to consider is the difference between samples collected on the east and west side of the island located 0.5 km downstream of the proposed OSPW discharge point, as these stations are directly downstream of the Syncrude sewage treatment outfall (Figure 1). Differences among samples collected from the east and west side of the island were explored using a Tukey's post-hoc test for individual comparisons. Results are provided in Table 39. Significant differences between the two sides of the island wat 0.5 km downstream were only observed in the Gomphidae family, where $\delta^{15}N$ and selenium values were significantly higher on the west side (p-value = 0.031 and 0.019, respectively; Table 39). Conversely, total mercury values were significantly lower on the west side compared to the east ($p < 0.001$; Table 39).

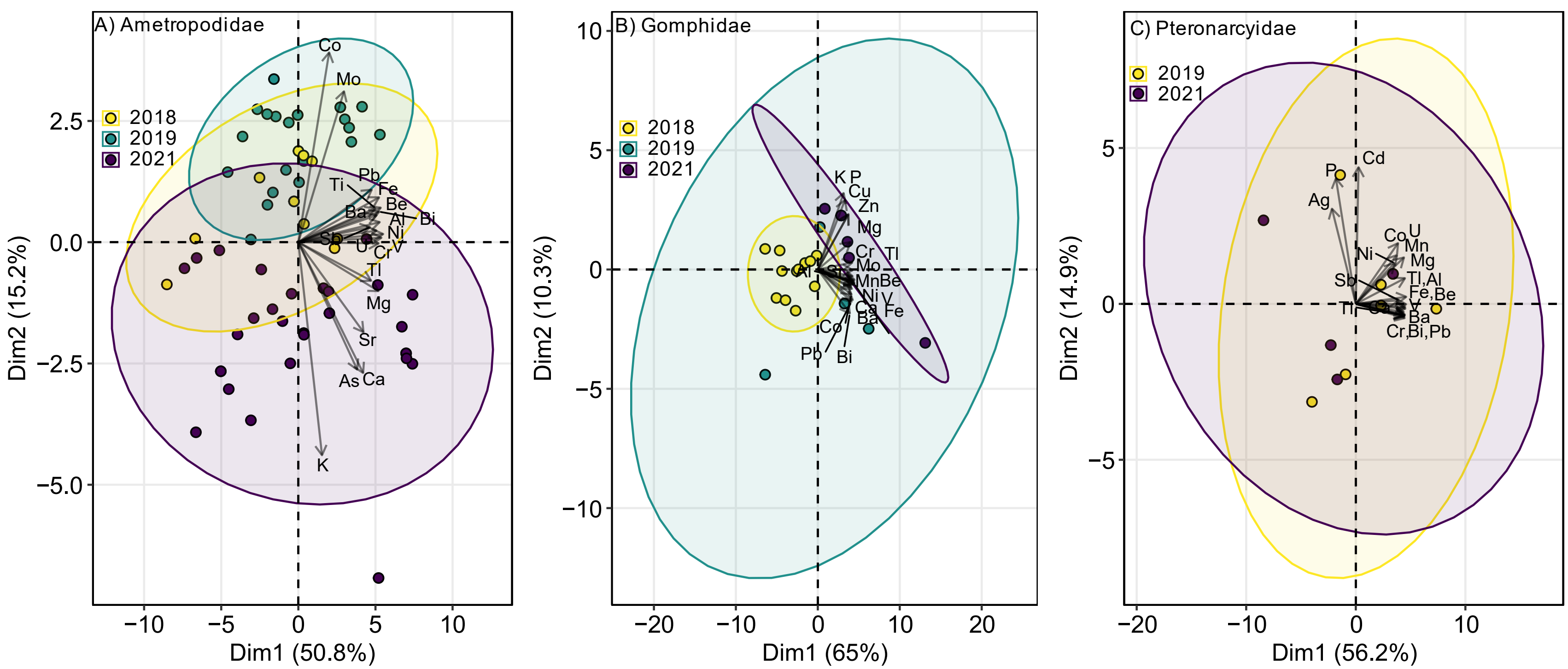

Figure 81 Benthic body burden temporal and spatial PCA, metals only.

Figure Notes: Only the top 20 contributing analytes to principal component axes 1 and 2 are presented for ease of visualization

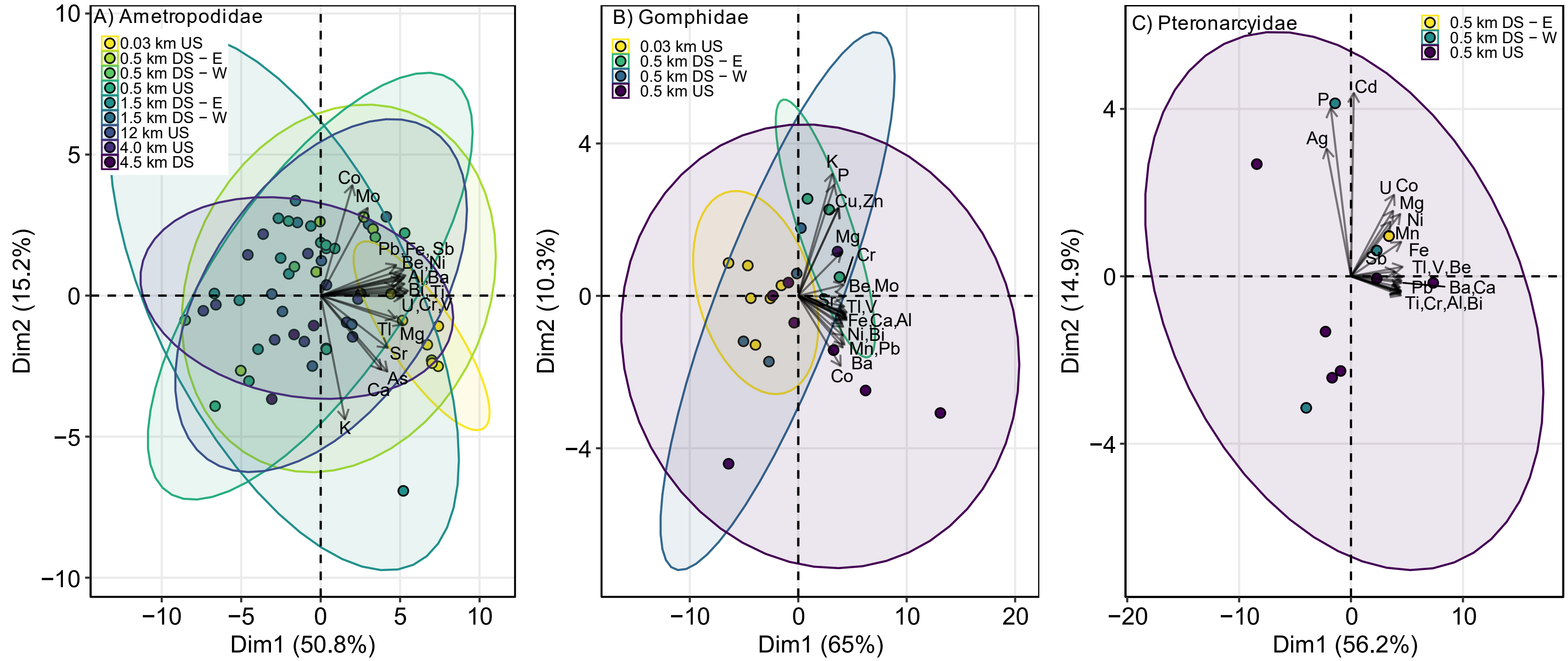

Figure 82 Benthic body burden spatial PCA, metals only.

Figure Notes: Only the top 20 contributing analytes to principal component axes 1 and 2 are presented for ease of visualization.

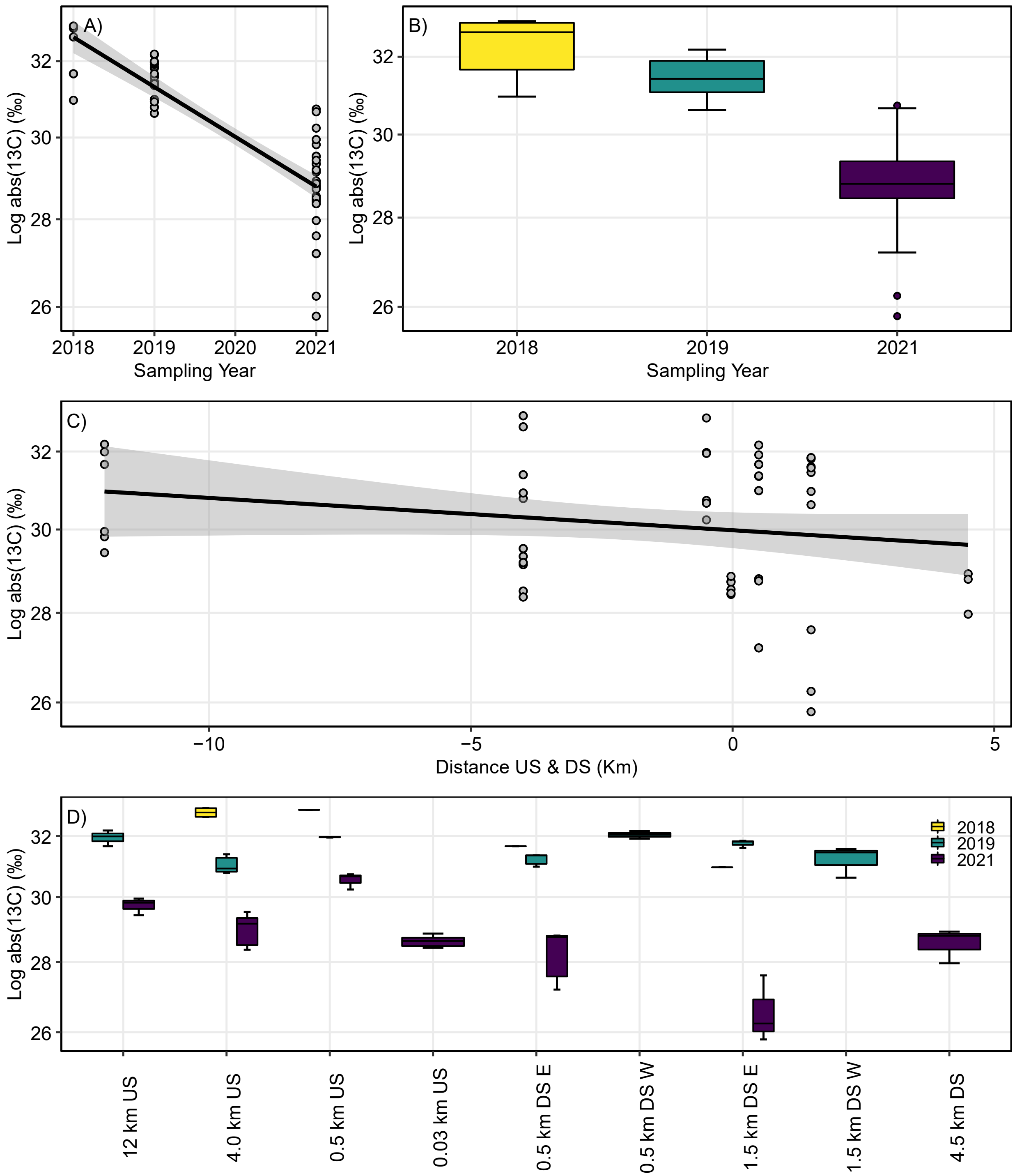

Figure 83 Variations in δ^{13}C ratio in Ametropodidae tissues over time (A,B) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

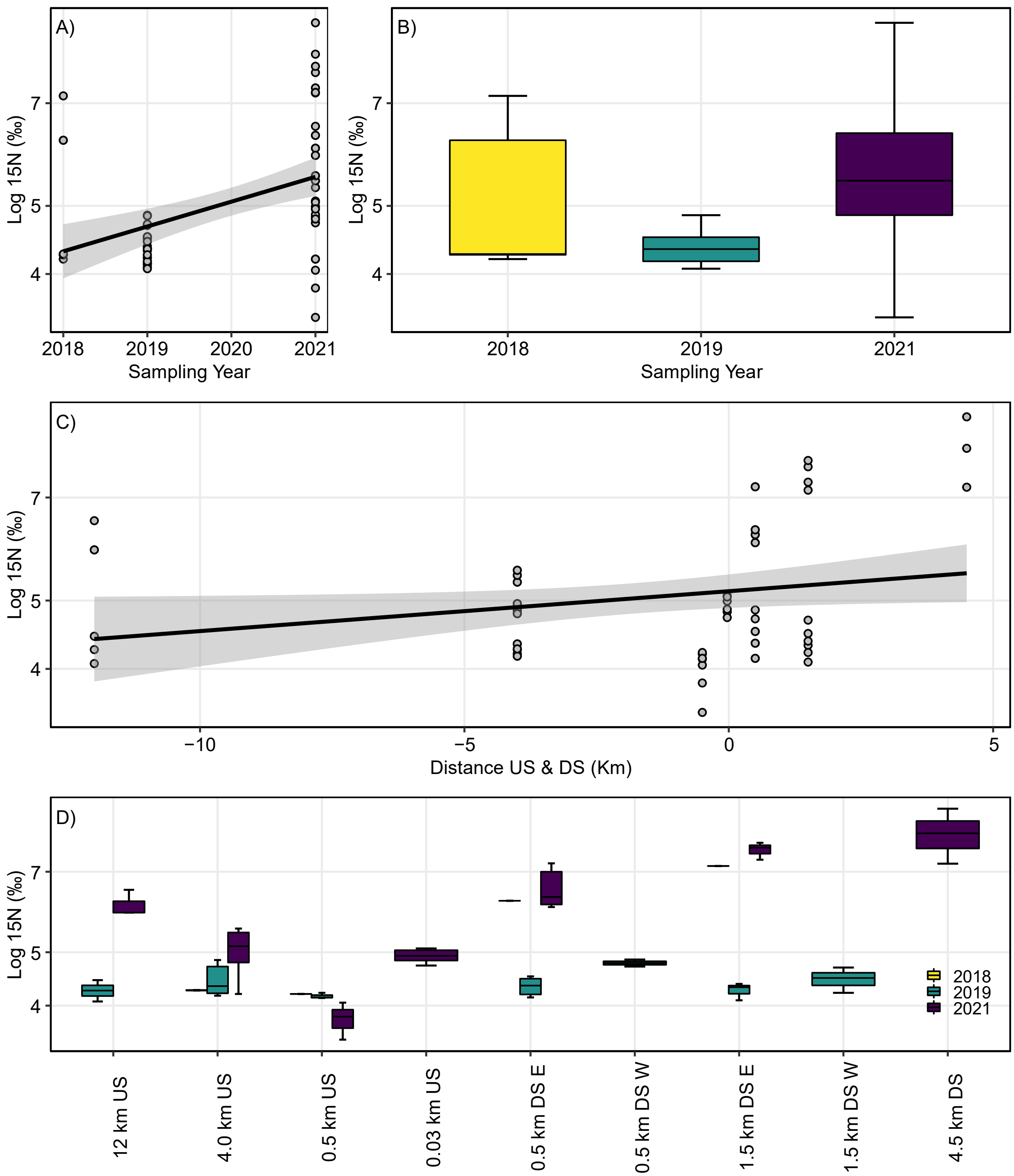

Figure 84 Variations in δ15N ratio in Ametropodidae tissues over time (A,B) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

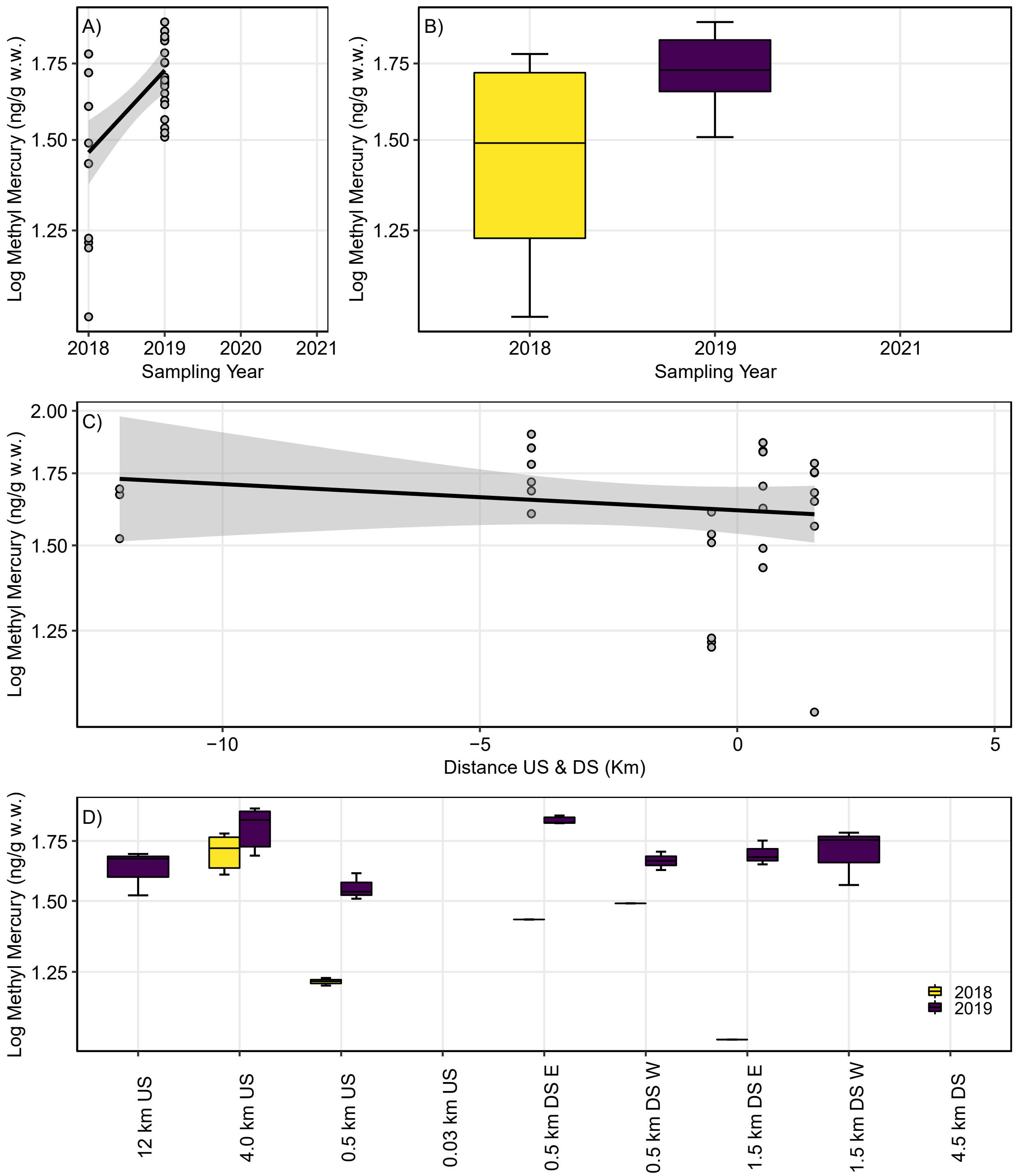

Figure 85 Variations in Methyl Mercury in Ametropodidae tissues over time (A,B) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

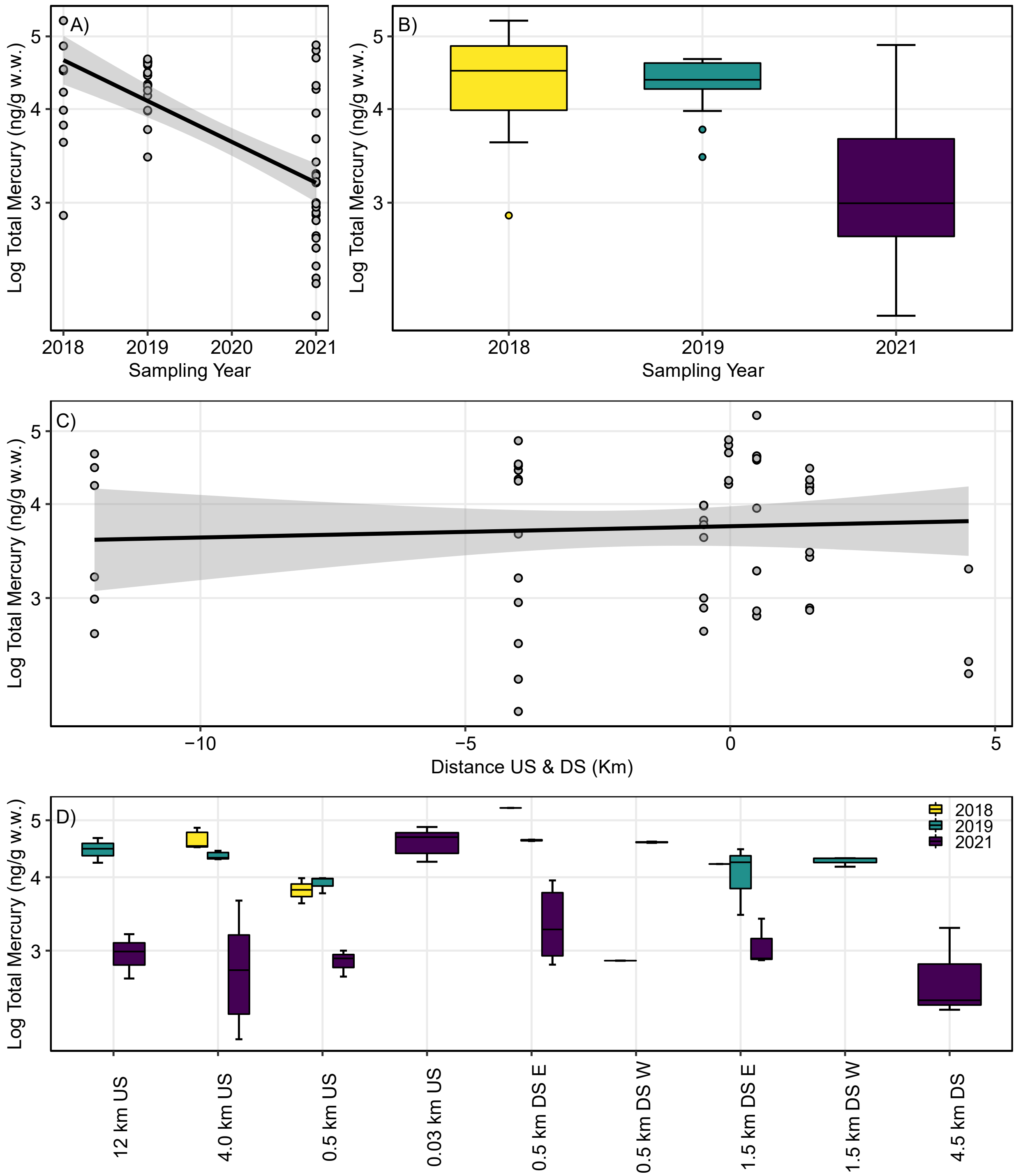

Figure 86 Variations in Total Mercury in Ametropodidae tissues over time (A,B) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

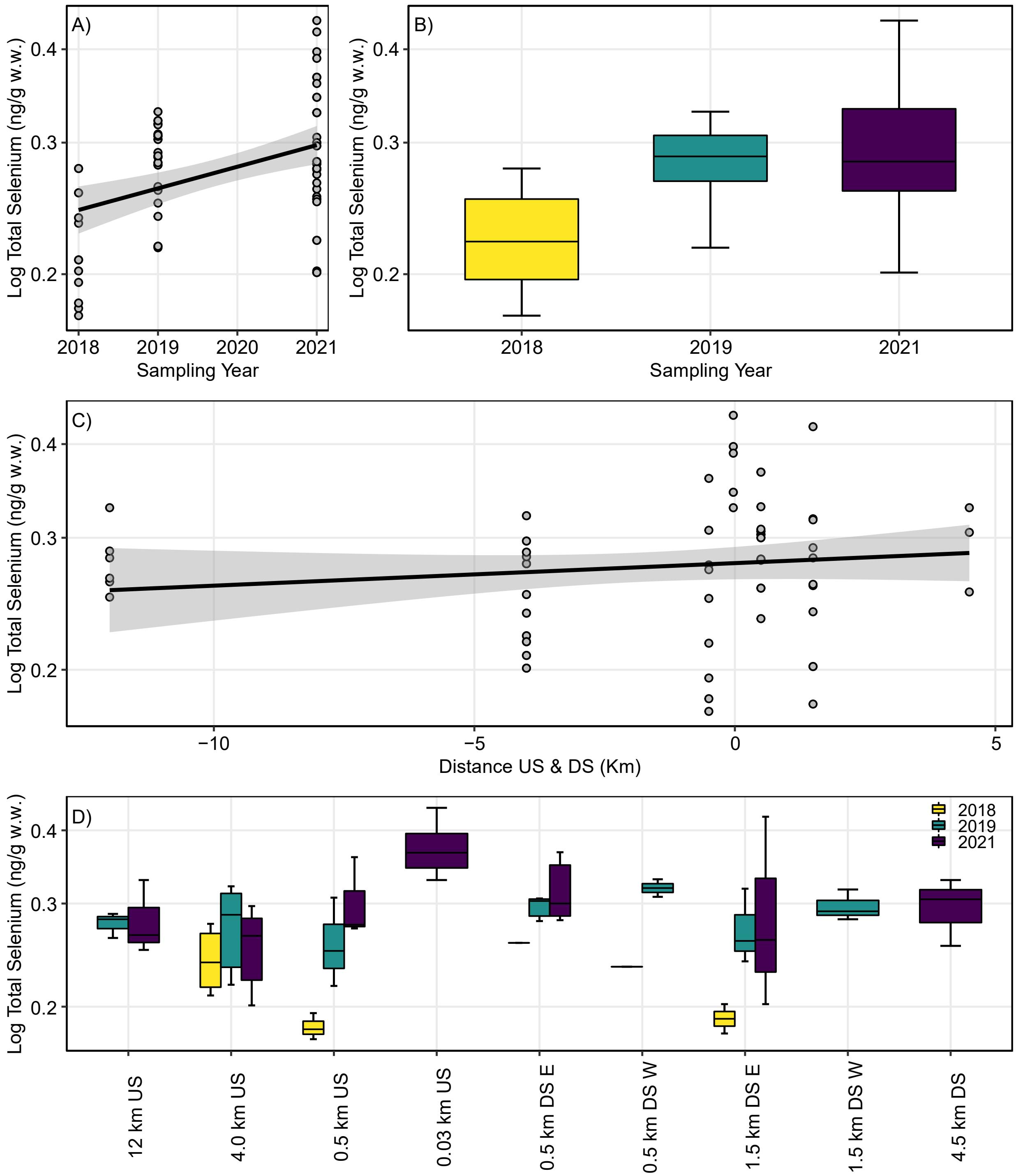

Figure 87 Variations in Total Selenium in Ametropodidae tissues over time (A,B) over distance upstream/downstream (C), and between sampling stations (D) during EMP.

Table 39 Tukey's post-hoc test benthic body burden analytes measured in benthic samples captured at the stations (0.5 km DS) located both East (E) and West (W) of the island, EMP dataset (2018, 2019, and 2021).

Family	Analyte	0.5 km DS E vs W	P-value
Ametropodidae	Carbon	E < W	0.763
	$\delta^{13}C$	E < W	0.553
	$\delta^{15}N$	E > W	0.881
	MeHg	E > W	0.859
	Se	E > W	0.990
	Total Hg	E > W	0.990
Gomphidae	Carbon	E < W	0.447
	$\delta^{13}C$	E > W	0.124
	$\delta^{15}N$	E < W	**0.031**
	Se	E < W	**0.019**
	Total Hg	E > W	**< 0.001**
Pteronarcyidae	Carbon	E < W	0.557
	$\delta^{13}C$	E < W	0.917
	$\delta^{15}N$	E < W	0.037
	Se	E < W	0.447
	Total Hg	E > W	0.497

Table Notes: Significant p-values (i.e., < 0.05) are in bold

2.4 Summary of Task 1

Water Quality

Explanatory models indicated that the water quality in the Athabasca River varied significantly with river flow volume (Q). After normalizing for flow, data analysis indicated that the majority of analytes varied significantly with sampling year and distance from the shoreline, but not distance downstream from the proposed OSPW discharge point. Further, the use of SPMDs to assess whether surface water grab samples can be used to predict SPMD concentrations (i.e., dissolved PAH phase) was deemed to not be an accurate approach of prediction.

Sediment Quality

Explanatory models indicated that the sediment quality in the Athabasca River varied significantly with sediment aluminum concentrations. After normalizing for aluminum, some analytes, including ammonium, naphthenic acids, and total concentrations of B, Ca, Fe, Li, Mn, Ni, Ag, Sr, Sn, U, V, Na, PAHs, and total organic nitrogen varied significantly with sampling year, while methyl mercury, total mercury, total Mo, total organic nitrogen, total P, total Sn, and total PAHs varied with distance from the proposed discharge point.

Benthic Algae Communities

The data analysis indicated that the algal community composition in the Athabasca River varied significantly with river flow volume (Q60). After taking variations related to flow into account, variations in algal community composition (i.e., density, richness, diversity, evenness, chlorophyll-*a*, biomass, NMDS scores) among years, stations or sampling areas were significant.

Benthic Invertebrate Communities

The data analysis indicated that the benthic community composition in the Athabasca River is dominated by Chironomidae and varies significantly with river flow volume (Q60) and sediment particle size. After taking variations related to those variables into account, variations in indicators of benthic community composition (i.e., density, richness, diversity, evenness, %EPT, PTI, NMDS scores) among years, stations or sampling areas were not significant.

Sentinel Fish Populations

The data analysis indicated that fish population health metrics varied significantly with river flow volume (Q60). After taking variations related to flow into account, variations in fish health indicators (i.e., condition, GSI and LSI) among stations or sampling areas were not significant. Fish health indicators did, however, seem to vary among years.

Fish Body and Tissue Burdens

The data analysis indicated that fish body burdens varied significantly with fork length for the majority of the compounds of interest. After taking variations related to fork length into account, fish body burdens did seem to vary among sampling years in Trout-perch, but limited evidence of variation among sampling stations was observed.

Benthic Invertebrate Body Burdens

The data analysis indicated that benthic body burdens of a subset of compounds of interest tended to vary with both sampling year and area, primarily in Ametropodidae, but not the other families.

3.0 TASK 2: OIL SANDS MONITORING PROGRAM EVALUATION AND DEVELOPMENT OF NORMAL RANGES

3.1 Overview

One of the major concerns expressed during the design of the Enhanced Monitoring Plan was to ensure that the number of sites and sampling frequency was sufficient to define upstream reference conditions and local reference conditions prior to a potential pilot discharge. It has been assumed that a minimum of three years of data are needed to characterize recent baseline variability (AEP, 2022), which is captured under the three years of partial sampling during the EMP. Examination of the EMP dataset provided in Section 2.0 suggests some variability in the different environmental samples both spatially and temporally. The intent of this analysis is therefore to evaluate the variability in the OSMP data (focusing on sites from M3 to M7) and determine whether the conditions documented in the local study area (during EMP) are reflected at a regional scale (under OSMP). More than 10 years of data are available for some sites along the LAR, including M3 and M7, and data are sufficient to provide an estimate of variability and normal ranges across an annual cycle. The analyses here were designed to quantify sources of background variability, and to estimate normal ranges of surface water chemistry, sediment chemistry, algal community composition, benthic invertebrate community composition, sentinel fish population variables, and fish/benthic body burdens.

With data collected across several years, and from several locations along the length of the LAR , the analysis involved various procedures to harmonize the data (OMSP and EMP) so that they were comparable across years and stations. After taking into consideration the potential influences of variability, such as river flow volume, aluminum concentrations in sediments, particle size, sampling year, distance upstream/downstream etc., the analysis tested for trends in these potential predictor variables. For those variables for which there was evidence of a significant effect, normal ranges of variability were computed that considered that observed variation. The normal ranges provide ‘triggers’ or benchmarks that can be used as one tool to assess variations in areas potentially exposed to the release of treated oil sands ground or process water. For those variables that produced a linear time trend, normal ranges were computed across monitoring time series. Trends over time and space suggest that conditions are changing, potentially naturally, or otherwise. The model that describes these trends can subsequently be used to forecast a normal range of conditions unrelated to exposure to the release of treated oil sands ground or process water.

There were two different approaches used to develop normal ranges:

1. Sufficient OSMP data exist and there is clear overlap and coincidence with EMP data, in terms of the locations sampled, the endpoints assessed, and the methods employed. In this case, normal

range models were developed using OSMP data, and then were tested against the EMP data to assess how well OSMP data can predict the variability in the EMP data, and instances where EMP data exceed normal ranges. This approach was employed for surface water quality, sediment quality, and fish health indicators.

2. Insufficient OSMP data exist to provide clear overlap with the EMP dataset in terms of locations sampled, endpoints measured, and/or methods used. In this case, the normal ranges were developed using the last 3 years of sampling data during EMP. This approach was employed for algae assemblages, benthic invertebrates, and fish/benthic body burdens.

Models were constructed using GLM. GLM determined how much of the variation in various response variables was 'due' or related to the assortment of categorical factors (e.g., sampling location/time) and continuous variables (discharge, particle size etc.) considered. GLM developed quantitative model coefficients that best explained variations in response variables. Unexplained (error) variation, or mean-squared errors (MSEs), was used to inform / support estimates of normal ranges. Here, the MSE is a measure of variability around samples within Sites. The square root of the MSE term is a pooled estimate of the within-Site standard deviation among samples.

3.2 Methodology

3.2.1 Water Quality Variables

3.2.1.1 Step 1 – General Data Processing

Water Quality data were obtained from Environment and Climate Change Canada (ECCC) for sampling areas M2, M3, M4, M5, M6, and M7 for the years 2012 through 2021. Longitude and latitude of each sampling location (station and panel #) were loaded into Google Earth and the shorter distance to the eastern and western shores of the Athabasca River were measured. Further, concentration data were paired with daily average flow volumes (m^3/s) for the Athabasca River at the Fort McMurray station. Flow volumes at that station are not considered exact flow volumes for any of the sample areas M3 through M7 because of the physical separation. Flow volumes for Fort McMurray, however, are assumed to provide relative estimates of flow volume. A similar assumption was made by Arciszewski et al. (2018) and is assumed to be relatively robust for the purpose of the modelling exercise here.

For the purpose of the modelling exercise, non-detects from both the Regional and Enhanced datasets were removed. Typically, a non-detect value would be replaced by half the specific compounds detection limit, however, this would produce an arbitrary value that is not impacted by model predictors and thus skew the model performance.

3.2.1.2 Step 2 – Building Predictive Models

3.2.1.2.1 Discrete Water Samples

Mixed-model GLM was used to explore potential sources of variation in measured analyte concentrations. The GLM structure was:

Y = constant + Q + Year + Distance(US/DS) + Distance(Shoreline) + Year x Q

where:

- Y = $\log_{10}$ of the analyte concentration;
- Q = $\log_{10}$ of flow volume on the day the water sample was collected (continuous - covariable);
- Year = year of the sample (continuous - covariable);
- Distance(US/DS) = distance upstream (-) or downstream (+) (continuous - covariable);
- Distance(Shoreline) = distance from the sampling point to the direct Eastern shoreline of the Athabasca River (continuous - covariable).

Year was included as a covariable because there has been evidence that at least some water quality parameters in the Athabasca River are increasing (Kilgour et al., 2019a). It is important to know if there are temporal trends in water quality variables over time, prior to the release of treated process water, so that release of treated waters is not implicated as the cause of future trends.

Regardless of the significance of the various terms in the model, the GLM was used to compute coefficients for the various terms, which were assembled and could be used to estimate expected analyte concentrations. Variables that are significant will have model coefficients that are significantly different from zero. Variables that were non-significant will produce model coefficients that are not significantly different from, and are very close to, zero (reflecting their non-significance). For the purpose of the modeling exercise here, all models (and associated coefficients) were retained so that readers could explore for themselves the influence of significant and non-significant covariables on estimated analyte concentrations.

3.2.1.2.2 SPMD Samples

Again, mixed-model GLM was used to explore potential sources of variation in measured SPMD (dissolved) PAH concentrations. The GLM model structure was:

$$Y = \text{constant} + Q_{avg} + \text{Year} + \text{Distance(US/DS)} + \text{Year} \times Q_{avg}$$

where:

- Y = $\log_{10}$ of the analyte concentration;
- Q_{avg} = $\log_{10}$ of the average flow volume measured over the period in which the SPMD device was deployed (continuous - covariable);
- Year = year of the sample (continuous - covariable);
- Distance(US/DS) = distance upstream (-) or downstream (+) (continuous - covariable);

Year was included as a covariable because there has been evidence that at least some water quality parameters in the Athabasca River are increasing. For the purpose of the modeling exercise here, all models (and associated coefficients) were retained so that readers could explore for themselves the influence of significant and non-significant covariables on estimated analyte concentrations, as justified in Section 3.3.1.2.

3.2.1.3 Step 3 – Computing Normal Ranges

The linear models fit in Step 2 were used to estimate normal ranges for a variety of conditions, but were generally of the following form:

$$\text{Normal Range} = \text{Model Prediction} \pm 2SD$$

where, SD was estimated from $\sqrt{MSE}$ for the linear model.

3.2.2 Sediment Quality Variables

3.2.2.1 Step 1 – General Data Processing

Sediment Quality data were obtained from ECCC for sampling areas M2, M3, M4, M5, M6, and M7 for the years 2012 through 2021. Preliminary analysis of the OSMP sediment data indicated no clear relationship between discharge and analyte concentrations, however a clear relationship was yielded when concentrations were regressed against total aluminum concentrations. Total aluminum concentrations were filtered from the bulk dataset and merged as a separate column, paired by Sample ID, to be used as a predictor in explanatory models.

For the purpose of the modelling exercise, non-detects from both the Regional and Enhanced datasets were removed. Typically, a non-detect value would be replaced by half the specific compounds detection limit, however, this would produce an arbitrary value that is not impacted by model predictors and thus skew the model performance.

3.2.2.2 Step 2 – Building Predictive Models

Mixed-model GLM was used to explore potential sources of variation in measured analyte concentrations. The GLM model structure was:

$$Y = \text{constant} + Al + Year + Distance(US/DS) + Year \times Al$$

where:

- Y = $\log_{10}$ of the analyte concentration;
- Al = $\log_{10}$ of the total aluminum (Al) concentration on the day the water sample was collected (continuous - covariable);
- Year = year of the sample (continuous - covariable);
- Distance(US/DS) = distance upstream (-) or downstream (+) (continuous - covariable).

Year was included as a covariable since it is important to know if there are temporal trends in sediment quality variables over time, prior to the release of treated process water, so that release of treated waters is not implicated as the cause of future trends.

The GLM was used to compute coefficients for the various terms, which were assembled and could be used to estimate expected analyte concentrations. Variables that are significant will have model coefficients that are significantly different from zero. Variables that were non-significant will produce model coefficients that are not significantly different from, and are very close to, zero (reflecting their non-significance). For the purpose of the modeling exercise here, all models (and associated coefficients) were retained so that readers could explore for themselves the influence of significant and non-significant covariables on estimated analyte concentrations.

3.2.2.3 Step 3 – Computing Normal Ranges

The linear models fit in Step 2 were used to estimate normal ranges for a variety of conditions, but were generally of the following form:

Normal Range = Model Prediction ± 2SD

Where, SD was estimated from $\sqrt{MSE}$ for the linear model.

3.2.3 Benthic Algae Community Composition

3.2.3.1 Step 1 – General Data Processing

Since no OSMP Algae Assemblage datasets were provided, the EMP datasets, the EMP data was used to determine normal ranges for algal indices of community composition within the LAR. The general data processing steps have already been outlined in section 2.2.6.

3.2.3.2 Step 2 – Building Predictive Models

Mixed-model GLM was used to explore potential sources of variation in measured analyte concentrations. The GLM model structure was:

Y = constant + Q60 + Year + Distance(US/DS)

where,

- Y is the index of community composition value, log transformed for density, richness, chlorophyl-*a*, and biomass and raw values for Simpson's Diversity/Evenness, and NMDS1 and 2 scores;
- Q60 = $\log_{10}$ of the average discharge over the previous 60 days prior to sample collection (continuous - covariable);
- Year = year of the sample (continuous - covariable);
- Distance(US/DS) = distance upstream (-) or downstream (+) (continuous - covariable).

Year was included as a covariable since it is important to know if there are temporal trends in algal community composition over time, prior to the release of treated process water, so that release of treated waters is not implicated as the cause of future trends.

The GLM was used to compute coefficients for the various terms, which were assembled and could be used to estimate expected variation in algal community composition. Variables that are significant will have model coefficients that are significantly different from zero. Variables that were non-significant will produce model coefficients that are not significantly different from, and are very close to, zero (reflecting their non-significance). For the purpose of the modeling exercise here, all models (and associated coefficients) were retained so that readers could explore for themselves the influence of significant and non-significant covariables on estimated analyte concentrations.

3.2.3.3 Step 3 – Computing Normal Ranges

The linear models fit in Step 2 were used to estimate normal ranges for a variety of conditions, but were generally of the following form:

$$\text{Normal Range} = \text{Model Prediction} \pm 2\text{SD}$$

Where, SD was estimated from $\sqrt{MSE}$ for the linear model.

3.2.4 Benthic Invertebrate Community Composition

3.2.4.1 Step 1 – General Data Processing

The OSMP has produced CABIN kick and sweep benthic samples for sample areas M3 through M7, for both gravel/cobble (2011 – 2017) and sand (2011 – 2015) substrata through the years 2011 through 2017. First, to determine whether there is sufficient overlap between the OSMP and EMP datasets, PCA's were carried out comparing the following indices of composition: Total density, taxa richness (S), Simpson's Diversity (D), Simpson's Evenness (E), NMDS axis 1 and 2 scores (NMDS1 & NMDS2), EPT taxa, and PTI. This step is important considering OSMP has targeted areas along the Athabasca River that were characteristics of a riverine area and included both cobble and sand as the primary substrate types. Whereas EMP samples depositional areas. Therefore this initial step will deem whether or not OSMP data can be used to predict EMP data and develop normal ranges that are applicable to future samples collected under EMP.

Index values were averaged by Sample Area – Year – Substrate combination. There were a total of 75 Area– Year – Substrate combinations, 39 gravel and 36 sand. Average index values were used in the subsequent Step 2 below.

3.2.4.2 Step 2 – Building Predictive Models

Mixed-model GLM was used to explore potential sources of variation in measured analyte concentrations. The GLM model structure was:

$$Y = \text{constant} + Q60 + PS + \text{Year} + \text{Distance(US/DS)}$$

where,

- Y = index of community composition value, log transformed for density and richness and raw values for Simpson's Diversity/Evenness, and NMDS1 and 2 scores
- Q60 = $\log_{10}$ of the average discharge over the previous 60 days prior to sample collection (continuous - covariable);
- PS = $\log_{10}$ of the particle size (mm) of the benthic sediment sample (continuous - covariable)
- Year = year of the sample (continuous - covariable)

- Distance(US/DS) = distance upstream (-) or downstream (+) (continuous - covariable)

Particle size was included in this model as it allows for the correction of benthic indices of community composition data for sediment type. Year was included as a covariable since it is important to know if there are temporal trends in sediment quality variables over time, prior to the release of treated process water, so that release of treated waters is not implicated as the cause of future trends.

The GLM was used to compute coefficients for the various terms, which were assembled and could be used to estimate expected analyte concentrations. Variables that are significant will have model coefficients that are significantly different from zero. Variables that were non-significant will produce model coefficients that are not significantly different from, and are very close to, zero (reflecting their non-significance). For the purpose of the modeling exercise here, all models (and associated coefficients) were retained so that readers could explore for themselves the influence of significant and non-significant covariables on estimated analyte concentrations.

3.2.4.3 Step 3 – Computing Normal Ranges

The linear models fit in Step 2 were used to estimate normal ranges for a variety of conditions, but were generally of the following form:

$$\text{Normal Range} = \text{Model Prediction} \pm 2SD$$

Where, SD was estimated from $\sqrt{MSE}$ for the linear model.

3.2.5 Fish Community Assessment

3.2.5.1 Step 1 – General Data Processing

Historic fish community data were available from the Regional Aquatic Monitoring Program (RAMP) from 1987 to 2014 in which normal range models could be developed. Data from 8 sites from the RAMP dataset were used to build the normal range models and are those sites that overlapped with data collected under the EMP.

3.2.5.2 Step 2 – Building Predictive Models

Mixed-model GLM was used to explore potential sources of variation in measured analyte concentrations. The GLM model structure was:

$$Y = \text{constant} + Q60 + \text{Year} + \text{Distance(US/DS)} + \text{effort}$$

where,

- Y = index of fish community composition value, log transformed for abundance and raw values for richness, Simpson’s Diversity/Evenness, and NMDS1 and 2 scores

- Q60 = $\log_{10}$ of the average discharge over the previous 60 days prior to sample collection (continuous - covariable);

- Year = year of the sample (continuous - covariable)

- Distance(US/DS) = distance upstream (-) or downstream (+) (continuous - covariable)
- Effort = electrofishing time measured in seconds (continuous – covariable)

The GLM was used to compute coefficients for the various terms, which were assembled and could be used to estimate expected analyte concentrations. Variables that are significant will have model coefficients that are significantly different from zero. Variables that were non-significant will produce model coefficients that are not significantly different from, and are very close to, zero (reflecting their non-significance). For the purpose of the modeling exercise here, all models (and associated coefficients) were retained so that readers could explore for themselves the influence of significant and non-significant covariables on estimated analyte concentrations.

3.2.5.3 Step 3 – Computing Normal Ranges

The linear models fit in Step 2 were used to estimate normal ranges for a variety of conditions, but were generally of the following form:

Normal Range = Model Prediction ± 2SD

Where, SD was estimated from $\sqrt{MSE}$ for the linear model.

3.2.6 Sentinel Fish Populations Health

3.2.6.1 Step 1 – General Data Processing

OSMP fish population health data were obtained from AEPA and ECCC for sampling areas M3 to M7 for the years 2009 through 2019 for two species of fish: Trout-perch and White Sucker. The conventional indices of K, GSI, and LSI were calculated as outlined in section 2.2.9.1.

Comparisons between OSMP and EMP data were carried out using boxplots to ensure there were similarities between the two datasets, both in terms of species sampled, and the magnitude of the indices. The OSMP data was then used to generate a predictive model as discussed below.

3.2.6.2 Step 2 – Building Predictive Models

Mixed-model GLM was used to explore potential sources of variation in measured fish health indicator. The GLM model structure was:

Y = constant + Q60 + Year + Distance(US/DS) + Q60xYear

where,

- Y = fish health indicator (K, GSI, LSI);
- Q60 = $\log_{10}$ of the average discharge over the previous 60 days prior to sample collection (continuous - covariable);
- Year = year of the sample (continuous - covariable)
- Distance(US/DS) = distance upstream (-) or downstream (+) (continuous - covariable)

Year was included as a covariable because it is important to know if there are temporal trends in water quality variables over time, prior to the release of treated process water, so that release of treated waters is not implicated as the cause of future trends.

Regardless of the significance of the various terms in the model, the GLM was used to compute coefficients for the various terms, which were assembled and could be used to estimate expected analyte concentrations. Variables that are significant will have model coefficients that are significantly different from zero. Variables that were non-significant will produce model coefficients that are not significantly different from, and are very close to, zero (reflecting their non-significance). For the purpose of the modeling exercise here, all models (and associated coefficients) were retained so that readers could explore for themselves the influence of significant and non-significant covariables on estimated analyte concentrations.

3.2.6.3 Step 3 – Computing Normal Ranges

The linear models fit in Step 2 were used to estimate normal ranges for a variety of conditions, but were generally of the following form:

$$\text{Normal Range} = \text{Model Prediction} \pm 2\text{SD}$$

Where, SD was estimated from $\sqrt{MSE}$ for the linear model.

3.2.7 Fish Body and Tissue Burdens

3.2.7.1 Step 1 – General Data Processing

OSMP fish population health data were obtained from AEPA for sampling areas M3 to M7 for the years 2011 through 2016 for four species of fish: Trout-perch, Longnose Sucker, Walleye, and White Sucker.

Comparisons between OSMP and EMP data were carried out by inspecting differences and similarities between the two datasets in terms of fish species samples, analytes measured, and sample matrix (i.e., liver, muscle, whole body etc.). The EMP data was then used to generate a predictive model as discussed below.

3.2.7.2 Step 2 – Building Predictive Models

Mixed-model GLM was used to explore potential sources of variation in measured analyte concentrations in fish tissue. The GLM model structure was:

$$Y = \text{constant} + FL + Q_{60} + \text{Year} + \text{Distance(US/DS)} + Q_{60} \times \text{Year}$$

where,

- Y = measured analyte concentration in fish tissue;
- FL = fork length of the fish (mm);
- Q60 = $\log_{10}$ of the average discharge over the previous 60 days prior to sample collection (continuous - covariable);

- Year = year of the sample (continuous - covariable)
- Distance(US/DS) = distance upstream (-) or downstream (+) (continuous - covariable)

3.2.7.3 Step 3 – Computing Normal Ranges

The linear models fit in Step 2 were used to estimate normal ranges for a variety of conditions, but were generally of the following form:

Normal Range = Model Prediction ± 2SD

Where, SD was estimated from $\sqrt{MSE}$ for the linear model.

3.2.8 Benthic Invertebrate Tissue Burdens

3.2.8.1 Step 1 – General Data Processing

Since no OSMP benthic body burden datasets were provided, the EMP data was used to determine normal ranges for benthic body burdens within the LAR. The general data processing steps have already been outlined in section 2.2.11.

3.2.8.2 Step 2 – Building Predictive Models

Mixed-model GLM was used to explore potential sources of variation in measured analyte concentrations. The GLM model structure was:

Y = constant + Year + Distance(US/DS)

where,

- Y = measured analyte concentration in benthic tissue;
- Year = year of the sample (continuous - covariable)
- Distance(US/DS) = distance upstream (-) or downstream (+) (continuous - covariable)

It is important to note that the original dataset did not contain a " (day-month-year)" column, but instead a month-year column. Without a unique identifier specifying the exact date in which the sample was collected, we could not include Q60 as a predictor.

3.2.8.3 Step 3 – Computing Normal Ranges

The linear models fit in Step 2 were used to estimate normal ranges for a variety of conditions, but were generally of the following form:

Normal Range = Model Prediction ± 2SD

Where, SD was estimated from $\sqrt{MSE}$ for the linear model.

3.3 Results

3.3.1 Water Quality Variables

3.3.1.1 Data QA/QC

The OSMP and EMP surface water quality datasets provided sufficient overlap in the analytes measured both spatially and temporally. Therefore, for the purpose of developing normal ranges, the OSMP data was used and compared to the EMP results in the following section.

3.3.1.2 Normal Ranges

3.3.1.2.1 Discrete Water Samples

Visual examination of the 22 selected surface water quality parameters (selection explained in Section 2.2.3.2) measured throughout OSMP indicated that concentrations of many of the analytes were related to river flow volume (discharge) in a log-linear form (Appendix C Table C1). Mixed-model GLM's determined that daily discharge was a significant predictor of variation in concentrations of all parameters except total nitrogen (Table 40), similar to what was observed with the EMP data.

Linear trends over time as well as variations in trends over time that depended on discharge (i.e., Year x Discharge) were statistically significant for all parameters except chloride and nitrite + nitrate (Table 40). There were statistically significant variations among numeric distance upstream and/or downstream for several variables (Table 40; i.e., Al, Cl, Pb, Mo, P, Na, and SO_4). Spatial variations, however, were generally negligible.

An example of the predicted normal ranges for copper generated based on OSMP data as described in Section 3.2.1 can be found in Table 41 with an illustration of the model performance in Figure 88. The models for water quality variables have retained all the components for discharge (Q), Year, distance, and Year x Discharge regardless of the statistical significance of those terms, in part for simplicity. Normally in model construction like this, terms that are not significant are removed from the model in order that the sums of squares (SS) and degrees of freedom (df) associated with the term can be put into the Error term, thus increasing the statistical power of the test. Here, and with over 900 samples in the overall analysis, there is no problem with statistical power. Thus, including the non-significant terms in the model does not diminish model significance. The coefficients associated with 'non-significant' terms, further are 'not different from zero', and here with a very high Error df, are not very different from zero and have essentially negligible effect on estimated concentrations of the respective constituents.

For copper, the model constant was 290.2, while the slope for discharge was -91.4, the year term was -0.145, etc. (see model breakdown in Table 41). The model MSE was 0.056 (for $\log_{10}$ of [Cu]), the square root of which is 0.236. The SD among samples, for any modeled scenario is therefore 0.236. The table provides three scenarios for which we desire an estimate of the normal range for copper concentrations. All three scenarios attempt to predict normal ranges in 2022 at 4km upstream of the potential OSPW release point in the thalweg under three different discharge conditions (i.e., 100, 600, and 1200 m^3/s). summarizes the model predicted surface water concentrations of each analyte against the observed values from field samples collected under the EMP. The model performance can be assessed based the deviation of the points from the 1:1 line (i.e., a perfect fit). While the data points at each sampling station do tend to fall on the 1:1 line, there are instances variation within the experimental results that are not captured by the predictive model. For example, total aluminum concentrations tend to vary more widely along the observed data (y-axis in

Figure 89) whereas total alkalinity clusters more tightly to the 1:1 line.

Normal range model coefficients and exceedances in the EMP data are summarized in Table 42 for each individual analyte. Overall, between 0 and 22.6 % (but typically 5 to 10%) of EMP samples fell below the normal range lower limit, while between 0 and 10.2 % (but typically ~ 5%) of EMP samples fell above the normal range upper limit. The rest of the EMP samples (typically 90 to 95) were within the normal ranges. The overlap between the OSMP predictions with the EMP data sets suggest that the OSMP data could be used to support estimates of normal ranges for EMP studies.

When dealing with surface water quality, our primary concern is exceedances of the upper normal range limit. The compound that exceeded this value the most often was sulphate (10.2%).

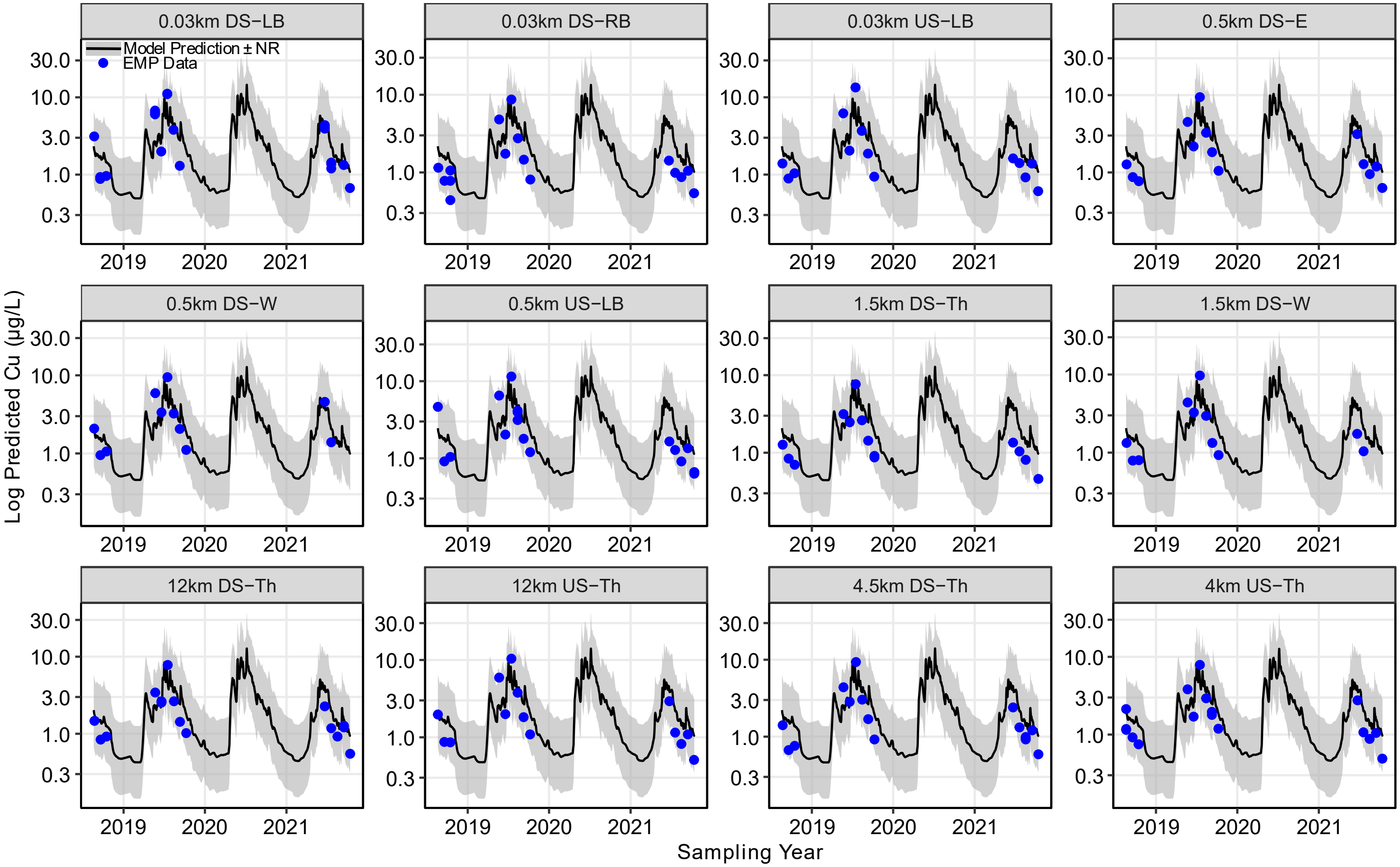

Figure 88 Variations in model prediction of total copper concentrations built with OSMP data in relation to sampling year compared to observed measurements measured during EMP (2018, 2019, and 2021) across each of the 12 sampling stations.

Figure Notes: NR = Normal Range (±2SD); Th = Thalweg; LB = Left Bank; RB = Right Bank; W = West of Island; E = East of Island

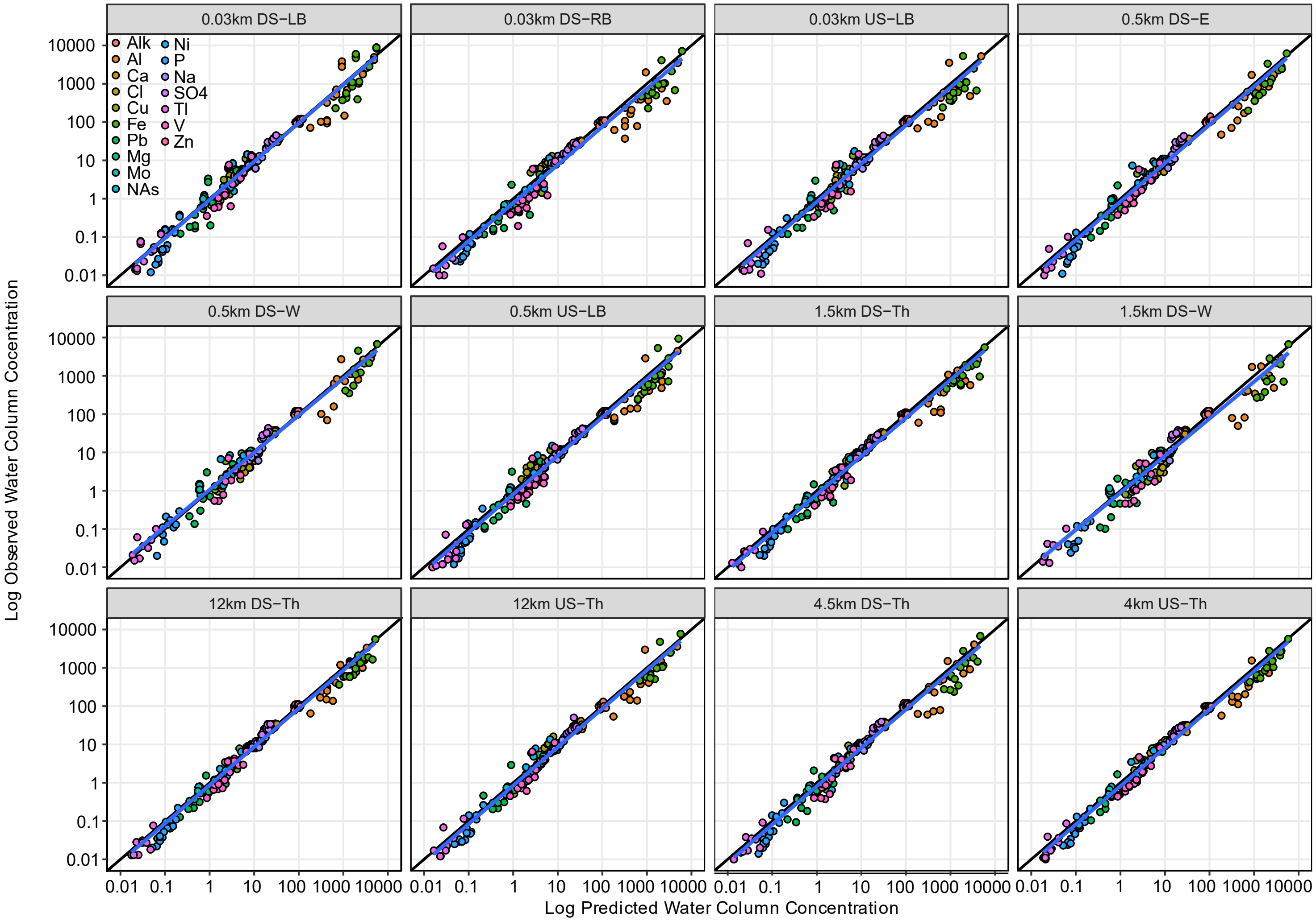

Figure 89 Model performance for each of the 17 individual analytes, y-axis represents the actual measurements during EMP and the x-axis represents predicted values based on OSMP data.

Figure Notes: The black line represents a 1:1 line, while the blue line represents the overall goodness of fit.

Table 40 Significance (p-value) and percent of variance explained (%VE) for discharge, year, distance upstream/downstream, distance to shoreline, and year x discharge as predictors of concentrations of water quality analytes in the Lower Athabasca River, under OSMP 2011 to 2021.

Compound	Discharge (Q)		Year		Distance (US/DS)		Shoreline Distance		Q x Year	
	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE
Alkalinity	**<0.001**	61.0	**0.014**	0.2	0.792	<0.1	**<0.001**	7.1	**0.002**	0.3
Aluminum (Total)	**<0.001**	88.3	**<0.001**	0.5	**0.002**	0.1	0.566	<0.1	**<0.001**	0.4
Calcium (Dissolved)	**<0.001**	46.3	**<0.001**	0.6	0.147	0.1	**<0.001**	13.4	**0.018**	0.2
Chloride (Dissolved)	**<0.001**	54.8	0.726	<0.1	**<0.001**	2.3	**<0.001**	16.6	**0.027**	0.1
Copper (Total)	**<0.001**	65.4	**<0.001**	0.9	0.193	0.1	**0.001**	0.4	**<0.001**	1.3
Iron (Total)	**<0.001**	75.0	**<0.001**	0.6	0.588	<0.1	**<0.001**	0.6	**<0.001**	0.5
Lead (Total)	**<0.001**	80.2	**<0.001**	0.5	**0.003**	0.2	0.412	<0.1	**<0.001**	1.1
Magnesium (Dissolved)	**<0.001**	58.2	**<0.001**	1.4	0.275	<0.1	**<0.001**	9.6	**0.016**	0.2
Molybdenum (Total)	**<0.001**	3.9	**<0.001**	4.6	**0.003**	0.7	**<0.001**	21.9	0.434	<0.1
Naphthenic acids	**<0.001**	4.8	**<0.001**	63.2	0.719	<0.1	0.768	<0.1	**<0.001**	8.5
Nickel (Total)	**<0.001**	71.7	**<0.001**	0.4	0.146	0.1	**<0.001**	0.7	**0.007**	0.2
Phosphorus (Total)	**<0.001**	69.3	**<0.001**	1.1	**0.678**	0.0	**0.008**	0.2	**0.001**	0.3
Sodium (Dissolved)	**<0.001**	84.3	**<0.001**	0.3	**0.004**	0.1	**<0.001**	2.8	**0.035**	0.1
Sulphate (Dissolved)	**<0.001**	47.7	**0.010**	0.3	**0.006**	0.3	**<0.001**	15.0	0.795	<0.1
Thallium (Total)	**<0.001**	79.8	**<0.001**	0.6	**0.049**	0.1	**<0.001**	0.4	**<0.001**	0.3
Vanadium (Total)	**<0.001**	29.2	**<0.001**	6.2	0.823	0.0	0.716	0.0	**<0.001**	2.5
Zinc (Total)	**<0.001**	62.9	**0.014**	0.2	**0.001**	0.5	**0.021**	0.2	**0.003**	0.4
Nitrite + Nitrate as N	**<0.001**	48.12	0.054	0.29	0.108	0.20	**0.011**	0.51	**0.004**	0.65
Nitrogen (Total)	0.423	18.61	**-**	-	0.468	14.94	0.817	1.38	**-**	-

Table Notes: Shaded cells represent the percent of variance explained by the predictor in cases where the p-value is significant (i.e., < 0.05)

Table 41 Example calculation of predicted normal ranges using the model for total copper at the sampling station that is 4km upstream of the potential OSPW release point in the thalweg.

Model Component	Description	Coefficient	Scenario		
			1	2	3
Constant	Intercept	290.21	1	1	1
Discharge	Slope for linear relation with Q	-91.37	100	600	1200
Year	Slope for linear trend across years	-0.145	2022	2022	2022
Distance (US/DS)	Slope for linear relationship with distance US/DS	-0.0006	4	4	4
Shoreline Distance	Slope for linear relationship with distance to shoreline	0.0002	43.2	43.2	43.2
Year x Discharge	Term accounting for the different slope for year effect, depending on Q	0.0458	4044	5617	6226
Copper estimate in logarithms			-0.80	0.13	0.49
MSE in logarithms		0.056	—		
SD in logarithms		0.236	—		
Lower limit of normal range in logarithms			-1.27	-0.34	0.02
Upper limit of normal range in logarithms			-0.33	0.60	0.96
Copper estimate in real units			0.16	1.35	3.09
Lower limit of normal range in real units			0.05	0.46	1.04
Upper limit of normal range in real units			0.47	4.00	9.15

Table Notes: Normal Ranges were calculated as the estimate in real units ± 2SD

Table 42 Model coefficients used for the prediction of surface water quality analyte normal ranges in the EMP dataset and the percentage of normal range (NR) exceedances (2018, 2019, and 2021).

Compound	Int	Q	Year	Distance (US/DS)	Distance to Shoreline	Q x Year	MSE	EMP NR Exceedance (%)		
								< LL	Inside NR	> UL
Alkalinity Total $CaCO_3$	-30.6	11.2	0.016	-0.0002	0.0002	-0.0057	0.003	1.1	94.6	4.3
Aluminum Total Recoverable	304.3	-92.2	-0.152	-0.0010	0.0000	0.0466	0.059	22.6	74.7	2.7
Calcium Dissolved	-29.0	9.8	0.015	-0.0001	0.0002	-0.0049	0.004	0.6	96.8	2.6
Chloride Dissolved	59.5	-30.9	-0.028	0.0036	-0.0010	0.0150	0.041	2.2	97.8	0.0
Copper Total Recoverable	290.2	-91.4	-0.145	-0.0006	0.0002	0.0458	0.056	1.6	98.4	0.0
Iron Total Recoverable	230.3	-71.7	-0.114	0.0001	-0.0002	0.0361	0.059	8.6	90.9	0.5
Lead Total Recoverable	378.8	-122.1	-0.190	-0.0010	0.0000	0.0613	0.068	5.9	91.9	2.2
Magnesium Dissolved	-30.0	8.7	0.016	-0.0001	0.0002	-0.0044	0.003	1.1	92.5	6.5
Molybdenum Total Recoverable	39.2	-5.7	-0.020	0.0000	0.0004	0.0029	0.012	0.0	93.5	6.5
Nickel Total Recoverable	122.6	-35.4	-0.062	-0.0006	0.0002	0.0180	0.039	1.1	95.7	3.2
Phosphorus Total	147.6	-43.4	-0.075	0.0000	-0.0001	0.0219	0.041	19.4	80.6	0.0
Sodium Dissolved/Filtered	20.2	-12.0	-0.009	0.0006	-0.0002	0.0057	0.007	7.0	93.0	0.0
Sulphate Dissolved	-6.2	1.6	0.004	0.0000	0.0004	-0.0009	0.012	0.0	89.8	10.2
Thallium Total Recoverable	179.6	-53.0	-0.092	-0.0007	0.0002	0.0269	0.044	6.9	92.0	1.1
Vanadium Total Recoverable	83.4	-15.5	-0.043	-0.0006	0.0000	0.0084	0.044	11.8	86.0	2.2
Zinc Total Recoverable	190.7	-57.7	-0.096	-0.0015	0.0001	0.0291	0.086	0.0	100.0	0.0
Nitrite + Nitrate as N	-195.6	66.2	0.097	-0.0012	0.0002	-0.0332	0.115	0.0	100.0	0.0
Nitrogen (Total)	82.4	-28.3	-	0.0064	-0.00004	-	0.006	0.0	100.0	0.0

Table Notes: Normal Ranges were calculated as the estimate in real units ± 2SD. Int = Intercept; Q = daily average discharge; LL = Lower Level; UL = Upper Level

3.3.1.2.2 SPMD Samples

Mixed-model GLM's determined that average discharge was a significant predictor of variation among the majority of PAH compounds, with 56 of 72 compounds varying significantly with discharge. Further, 42 of 72 varying significantly with Year and/or the interaction between year and discharge, and 28 of 72 compounds varying significantly with distance (Table 43).

An example of the predicted normal ranges for SPMD derived total PAH concentrations generated based on OSMP data as described in Section 3.2.1.2.2 can be found in

Table 44 with an illustration of the model performance in Figure 90 and Figure 91. The models SPMD derived PAHs have retained all the components for discharge, Year, and distance regardless of the statistical significance of those terms, in part for simplicity. Normally in model construction like this, terms that are not significant are removed from the model in order that the sums of squares (SS) and degrees of freedom (df) associated with the term can be put into the Error term, thus increasing the statistical power of the test. Here, and with over 127 samples in the overall analysis, there is no problem with statistical power. Thus, including the non-significant terms in the model does not diminish model significance. The coefficients associated with 'non-significant' terms, further are 'not different from zero', and here with a very high Error df, are not very different from zero and have essentially negligible effect on estimated concentrations of the respective constituents.

For total PAH, the model constant was -1.8, while the discharge term was 16.4, the year term was 0.001, etc. (see model breakdown in Table 44). The model MSE was 0.05 (for $\log_{10}$ of [Total PAH]), the square root of which is 0.223. The SD among samples, for any modeled scenario is therefore 0.223. The table provides three scenarios for which we desire an estimate of the normal range for total PAH SPMD concentrations. The three scenarios attempt to predict normal ranges at the station located 4km upstream of the potential OSPW release point in the thalweg in 2022, 2023, and 2024. Figure 91 summarizes the model predicted surface water concentrations of each analyte against the observed values from field samples collected under the EMP. The model performance can be assessed based on the deviation of the points from the 1:1 line (i.e., a perfect fit). While the data points at each sampling station do tend to fall on the 1:1 line, there are instances of variation within the experimental results that are not captured by the predictive model.

Normal range model coefficients and exceedances in the EMP data are summarized in Table 45 for each individual analyte. Overall, between 0 and 30 % (but typically 5 to 10%) of EMP samples fell below the normal range lower limit, while between 0 and 7.1 % of EMP samples fell above the normal range upper limit. The rest of the EMP samples (typically 90 to 95) were within the normal ranges. The overlap between the OSMP predictions with the EMP data sets, therefore, suggest that the OSMP data could be used to support estimates of SPMD normal ranges for EMP studies.

When dealing with surface water quality, our primary concern is exceedances of the upper normal range limit. The PAH that exceeded this value the most often was Benz(a)Anthracene/Chrysene-C4 (7.1%).

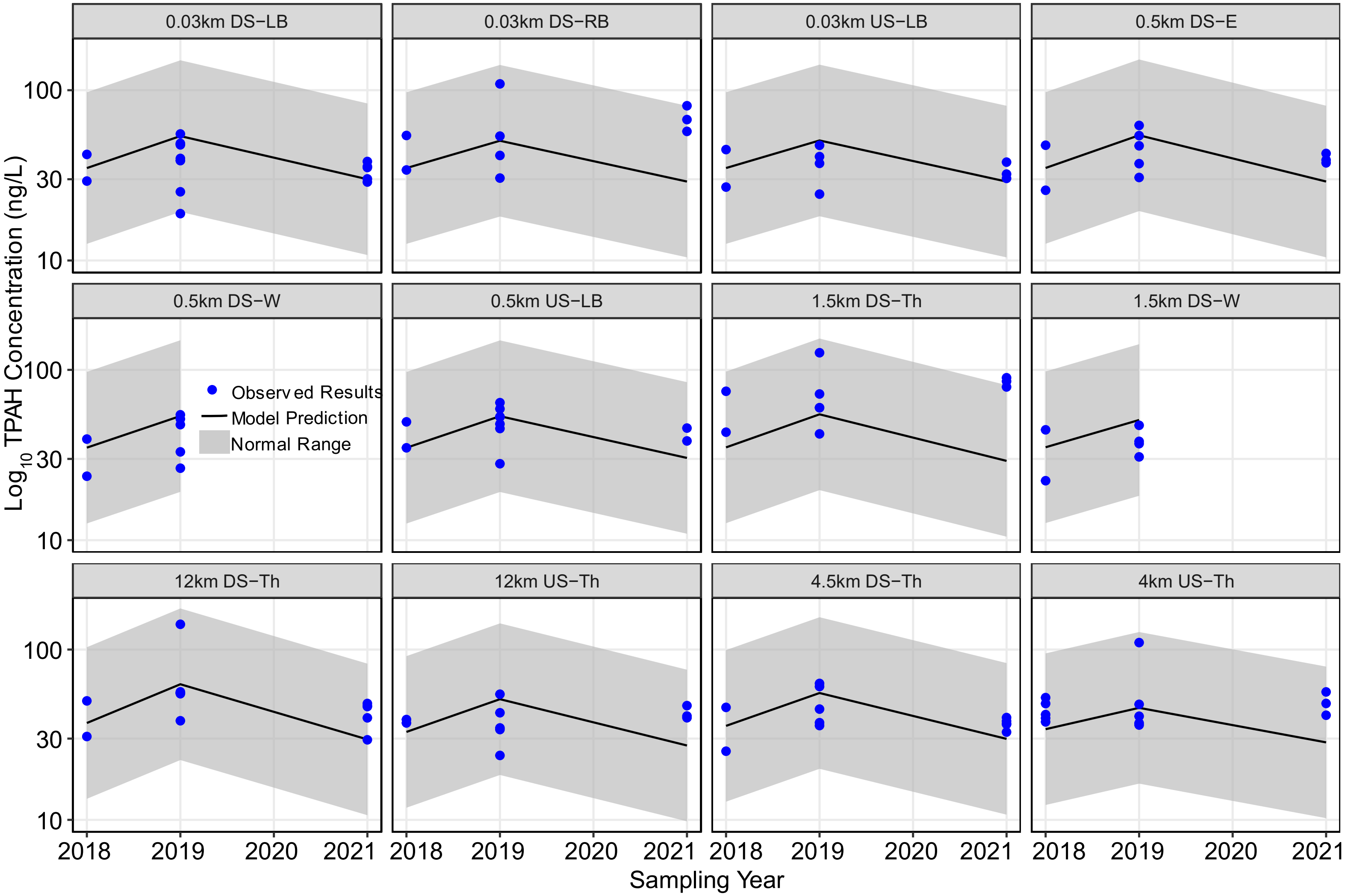

Figure 90 Variations in model prediction of total PAH SPMD concentrations built with OSMP data in relation to sampling year compared to observed measurements measured during EMP (2018, 2019, and 2021) across each of the 12 sampling stations. The missing data in 2021 on the West side of the LAR is due to the sampling station drying up.

Figure Notes: NR = Normal Range(±2SD); Th = Thalweg; LB = Left Bank; RB = Right Bank; W = West of Island; E = East of Island.

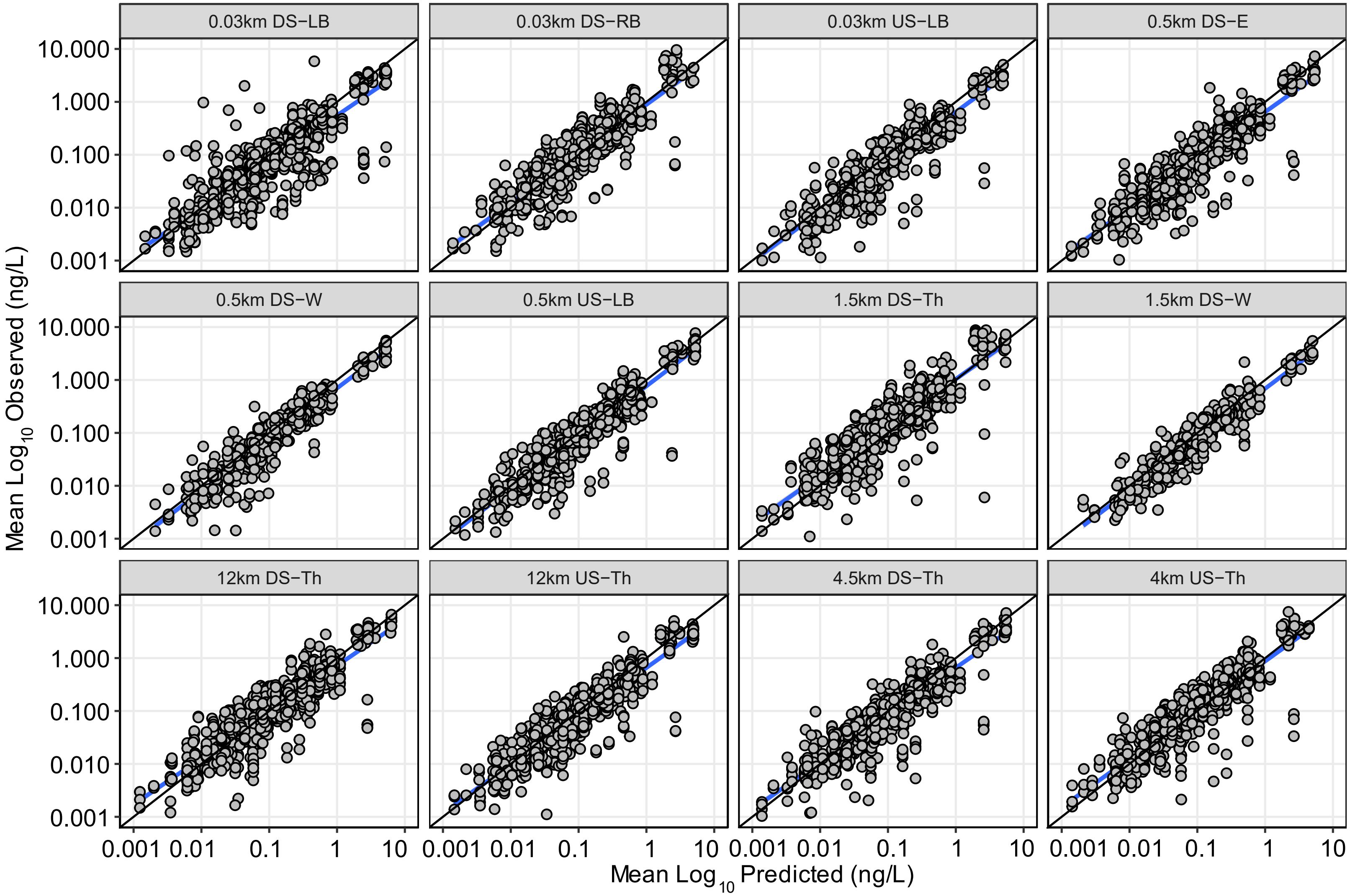

Figure 91 Model performance for each of the 71 individual PAH analytes among SPMD samples, y-axis represents the actual measurements during EMP and the x-axis represents predicted values based on OSMP data.

Figure Notes: The black line represents a 1:1 line, while the blue line represents the overall goodness of fit.

Table 43 Significance (p-value) and percent of variance explained (%VE) for discharge, year, distance upstream/downstream, and year x discharge as predictors of concentrations of SPMD PAHs in the Lower Athabasca River, under OSMP 2011 to 2021.

Compound	Q_{avg}		Year		Distance (US/DS)		Q x Year	
	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE
1-Methylchrysene	**<0.001**	18.61	**0.024**	3.58	**<0.001**	9.71	0.066	2.34
1-Methylnaphthalene	0.932	0.01	0.054	3.54	0.285	1.08	0.074	3.04
1-Methylphenanthrene	**<0.001**	35.02	0.406	0.46	0.702	0.10	0.884	0.01
1,2-Dimethylnaphthalene	**<0.001**	27.88	**0.006**	5.25	0.288	0.74	**0.040**	2.83
1,2,6-Trimethylphenanthrene	**<0.001**	28.75	0.528	0.26	**<0.001**	9.13	0.586	0.19
1,4,6,7-Tetramethylnaphthalene	**<0.001**	44.91	0.051	1.93	0.284	0.57	**0.003**	4.74
1,7-Dimethylfluorene	**<0.001**	37.52	**0.002**	5.39	0.534	0.22	**0.026**	2.84
1,7-Dimethylphenanthrene	**<0.001**	36.07	0.777	0.05	0.112	1.64	0.441	0.38
1,8-Dimethylphenanthrene	**<0.001**	23.45	**<0.001**	9.86	0.129	1.52	0.070	2.18
2-Methylanthracene	**<0.001**	22.76	0.236	1.01	**0.005**	5.84	0.195	1.21
2-Methylfluorene	**<0.001**	26.98	**0.029**	3.28	0.048	2.69	0.095	1.91
2-Methylnaphthalene	0.793	0.07	0.213	1.50	0.354	0.83	0.077	3.04
2-Methylphenanthrene	**<0.001**	27.01	0.272	0.90	0.502	0.34	0.791	0.05
2,3,5-Trimethylnaphthalene	**<0.001**	31.38	0.155	1.33	0.485	0.32	**0.011**	4.36
2,3,6-Trimethylnaphthalene	**<0.001**	25.47	0.605	0.19	0.769	0.06	**0.002**	7.36
2,4-Dimethyldibenzothiophene	**<0.001**	36.88	0.086	1.82	0.047	2.45	0.505	0.27
2,6-Dimethylnaphthalene	0.636	0.23	0.665	0.19	0.308	1.07	0.938	0.01
2,6-Dimethylphenanthrene	**<0.001**	39.16	0.640	0.13	0.108	1.59	0.369	0.49
2/3-Methyldibenzothiophenes	**<0.001**	31.22	0.107	1.72	**0.017**	3.86	0.500	0.30
3-Methylfluoranthene/Benzo(a)Fluorene	**<0.001**	9.72	**<0.001**	17.26	**0.012**	4.49	0.053	2.61
3-Methylphenanthrene	**<0.001**	42.19	0.100	1.57	0.697	0.09	0.182	1.03
3,6-Dimethylphenanthrene	**<0.001**	42.16	0.077	1.80	0.156	1.15	0.669	0.10
5,9-Dimethylchrysene	**<0.001**	20.98	0.570	0.22	**<0.001**	13.47	0.289	0.76
5/6-Methylchrysene	**<0.001**	17.76	**0.019**	3.89	**<0.001**	10.00	0.109	1.80
7-Methylbenzo(a)Pyrene	0.855	0.03	**0.036**	4.37	0.196	1.63	0.479	0.49
9/4-Methylphenanthrene	**<0.001**	42.91	**0.036**	2.50	0.339	0.51	0.347	0.49
Acenaphthene	**<0.001**	19.06	**<0.001**	12.10	0.581	0.21	0.620	0.17
Acenaphthylene	0.086	2.88	0.104	2.56	0.822	0.05	0.913	0.01
Anthracene	0.122	2.28	0.213	1.48	0.277	1.12	0.123	2.27
Benz(a)Anthracene/Chrysene-C1	**0.001**	8.39	**0.001**	8.94	**<0.001**	10.62	**0.030**	3.41
Benz(a)Anthracene/Chrysene-C3	**<0.001**	15.98	0.100	1.72	**<0.001**	17.09	**0.008**	4.62
Benz(a)Anthracene/Chrysene-C4	**0.008**	5.74	**<0.001**	13.60	0.096	2.23	0.121	1.93
Benzo(a)Anthracene	**0.014**	5.79	0.167	1.79	0.280	1.09	0.714	0.12
Benzo(a)Anthracene/Chrysene-C2	**<0.001**	14.04	**0.024**	3.46	**<0.001**	15.43	**0.029**	3.25
Benzo(a)Pyrene	**0.010**	6.33	0.083	2.79	0.356	0.78	0.855	0.03
Benzo(B)Fluoranthene	**<0.001**	45.23	0.142	1.00	**0.046**	1.86	**<0.001**	6.96
Benzo(e)Pyrene	**<0.001**	27.25	**0.022**	2.79	**<0.001**	11.70	**<0.001**	7.54

Compound	Q_{avg}		Year		Distance (US/DS)		Q x Year	
	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE
Benzo(g,h,i)Perylene	**0.006**	7.03	0.082	2.80	0.399	0.65	0.915	0.01
Benzo(J,K)Fluoranthenes	**0.009**	6.27	**0.030**	4.32	0.498	0.41	0.492	0.43
Benzofluoranthene/Benzopyrene-C1	**<0.001**	36.94	0.992	0.00	**<0.001**	7.05	**0.007**	4.00
Biphenyl	0.988	0.00	**0.048**	3.86	0.504	0.43	0.397	0.69
Chrysene	0.665	0.18	**0.015**	5.77	0.184	1.67	0.946	0.00
Dibenzo(a,h)Anthracene	0.500	0.43	0.515	0.40	0.213	1.49	0.051	3.70
Dibenzothiophene	**0.012**	5.95	0.059	3.27	0.232	1.30	0.473	0.47
Dibenzothiophene-C1	**<0.001**	19.21	0.162	1.45	**0.005**	6.14	0.325	0.72
Dibenzothiophene-C2	**<0.001**	36.71	0.146	1.26	**0.014**	3.68	0.569	0.19
Dibenzothiophene-C3	**<0.001**	28.60	0.071	2.09	**0.001**	7.25	0.854	0.02
Dibenzothiophene-C4	**<0.001**	17.22	**0.048**	2.66	**<0.001**	13.83	0.279	0.78
Dimethyl Biphenyl	**<0.001**	24.79	0.066	2.44	0.063	2.48	0.320	0.70
Fluoranthene	**<0.001**	20.14	**<0.001**	8.55	**<0.001**	9.77	**<0.001**	8.53
Fluoranthene/Pyrene-C1	**0.036**	3.53	**<0.001**	10.28	**0.002**	7.80	0.320	0.78
Fluoranthene/Pyrene-C2	0.095	2.05	**<0.001**	12.87	**<0.001**	11.62	0.087	2.15
Fluoranthene/Pyrene-C3	0.256	0.93	**0.001**	8.45	**<0.001**	17.70	0.059	2.59
Fluorene	**<0.001**	21.74	**0.003**	6.49	0.288	0.80	0.108	1.84
Fluorene-C1	**<0.001**	22.25	**<0.001**	18.05	**0.046**	2.34	0.377	0.45
Fluorene-C2	**<0.001**	38.12	**0.002**	5.41	0.092	1.53	**0.031**	2.53
Fluorene-C3	**<0.001**	39.46	**0.012**	3.52	**0.042**	2.29	0.097	1.51
Indeno(1,2,3-c,d)Pyrene	0.135	2.19	0.138	2.16	0.944	0.00	0.903	0.01
Methyl Acenaphthene	**<0.001**	22.55	**0.005**	5.77	**0.037**	3.10	0.584	0.21
Methyl Biphenyl	0.420	0.64	0.461	0.53	0.512	0.42	0.126	2.31
Naphthalene	0.423	0.42	**<0.001**	34.25	0.683	0.11	0.345	0.59
Naphthalene-C1	0.922	0.01	0.090	2.73	0.326	0.90	**0.010**	6.41
Naphthalene-C2	**0.008**	6.64	0.245	1.26	0.568	0.30	0.326	0.90
Naphthalene-C3	**<0.001**	35.71	**0.010**	3.84	0.334	0.53	**0.004**	4.82
Naphthalene-C4	**<0.001**	49.09	**0.041**	1.94	0.374	0.36	**0.005**	3.82
Perylene	**<0.001**	62.81	0.482	0.17	0.176	0.63	**0.006**	2.69
Phenanthrene	0.189	1.48	**<0.001**	14.20	0.463	0.46	0.581	0.26
Phenanthrene/Anthracene-C1	**<0.001**	29.30	**0.034**	3.13	0.671	0.12	0.510	0.30
Phenanthrene/Anthracene-C2	**<0.001**	39.55	**0.025**	2.86	**0.019**	3.12	0.870	0.01
Phenanthrene/Anthracene-C3	**<0.001**	30.52	0.058	2.29	**0.004**	5.39	0.862	0.02
Phenanthrene/Anthracene-C4	**<0.001**	19.90	0.539	0.26	**<0.001**	11.34	0.180	1.24
Total PAH	**<0.001**	32.14	**0.046**	2.48	**0.004**	5.18	0.867	0.02

Table Notes: Shaded cells represent the percent of variance explained by the predictor in cases where the p-value is significant (i.e., < 0.05).

Table 44 Example calculation of predicted normal ranges using the model for total PAH in the SPMD samples at station the EMP station 4km upstream of the potential OSPW release point.

Model Component	Description	Coefficient	Scenario		
			1	2	3
Constant	Intercept	-1.8	1	1	1
Qavg	Slope for linear trend with discharge	16.4	600	600	600
Year	Slope for linear trend across years	0.001	2022	2023	2024
Distance (US/DS)	Slope for linear trend with distance US/DS	0.0023	-4	-4	-4
Q60 x Year	Slope for interaction between Q60 and Year	-0.0078	5617	5620	5623
Total PAH estimate in logarithms			1.45	1.43	1.41
MSE in logarithms		0.050			
SD in logarithms		0.223			
Lower limit of normal range in logarithms			1.00	0.98	0.96
Upper limit of normal range in logarithms			1.89	1.87	1.85
Total PAH estimate in real units			28.03	26.71	25.45
Lower limit of normal range in real units			10.06	9.58	9.13
Upper limit of normal range in real units			78.10	74.43	70.93

Table 45 Model coefficients used for the prediction of SPMD derived PAH ranges in the EMP dataset and the percentage of normal range (NR) exceedances (2018, 2019, and 2021).

Compound	Int	Qavg	Year	Distance (US/DS)	Qavg x Year	MSE	EMP NR Exceedance (%)		
							< LL	Inside NR	> UL
1-Methylchrysene	-532.7	200.2	0.263	0.003	-0.099	0.064	0.0	100.0	0.0
1-Methylnaphthalene	-1303.2	401.0	0.647	-0.002	-0.199	0.277	18.1	81.9	0.0
1-Methylphenanthrene	59.9	-13.4	-0.031	0.000	0.007	0.050	0.8	99.2	0.0
1,2-Dimethylnaphthalene	899.7	-270.7	-0.449	0.001	0.135	0.094	6.3	88.2	5.5
1,2,6-Trimethylphenanthrene	-157.1	58.2	0.077	0.003	-0.029	0.061	0.0	100.0	0.0
1,4,6,7-Tetramethylnaphthalene	1146.1	-365.2	-0.571	0.001	0.182	0.077	3.1	92.1	4.7
1,7-Dimethylfluorene	914.2	-274.5	-0.456	0.001	0.137	0.082	0.0	93.7	6.3
1,7-Dimethylphenanthrene	-245.7	85.1	0.121	0.001	-0.042	0.066	0.8	99.2	0.0
1,8-Dimethylphenanthrene	-519.3	214.1	0.256	0.001	-0.106	0.075	10.2	84.3	5.5
2-Methylanthracene	581.9	-181.8	-0.291	0.003	0.091	0.108	5.5	92.9	1.6
2-Methylfluorene	621.4	-187.9	-0.310	0.002	0.094	0.069	0.8	97.6	1.6
2-Methylnaphthalene	-1427.8	452.2	0.708	-0.002	-0.225	0.360	7.9	91.3	0.8
2-Methylphenanthrene	108.2	-26.7	-0.055	0.001	0.014	0.058	0.8	98.4	0.8
2,3,5-Trimethylnaphthalene	896.0	-287.7	-0.446	0.001	0.143	0.068	1.6	96.9	1.6
2,3,6-Trimethylnaphthalene	1031.7	-352.3	-0.513	0.000	0.175	0.064	2.4	96.9	0.8
2,4-Dimethyldibenzothiophene	287.9	-79.0	-0.144	0.002	0.040	0.079	0.8	95.3	3.9
2,6-Dimethylnaphthalene	65.5	-15.0	-0.033	-0.002	0.008	0.207	3.1	96.1	0.8
2,6-Dimethylphenanthrene	278.0	-89.8	-0.139	0.001	0.045	0.055	0.8	98.4	0.8
2/3-Methyldibenzothiophenes	276.4	-77.0	-0.139	0.002	0.039	0.073	0.8	96.9	2.4
3-Methylfluoranthene/Benzo(a)Fluorene	-439.6	190.5	0.217	0.002	-0.094	0.052	1.6	98.4	0.0
3-Methylphenanthrene	421.5	-128.0	-0.211	0.000	0.064	0.051	0.8	98.4	0.8
3,6-Dimethylphenanthrene	169.5	-41.8	-0.086	0.001	0.021	0.054	0.8	96.9	2.4
5,9-Dimethylchrysene	-454.6	145.3	0.224	0.005	-0.072	0.102	0.0	100.0	0.0
5/6-Methylchrysene	-474.3	182.3	0.234	0.004	-0.090	0.070	0.0	99.2	0.8
7-Methylbenzo(a)Pyrene	-218.8	98.8	0.107	0.001	-0.049	0.106	3.9	94.5	1.6
9/4-Methylphenanthrene	320.8	-90.1	-0.161	0.001	0.045	0.051	0.8	98.4	0.8
Acenaphthene	-427.2	84.4	0.211	0.001	-0.042	0.159	0.8	99.2	0.0
Acenaphthylene	-149.9	22.3	0.073	0.000	-0.011	0.228	3.1	96.9	0.0
Anthracene	926.3	-333.6	-0.461	0.002	0.166	0.258	0.0	100.0	0.0
Benz(a)Anthracene/Chrysene-C1	-599.4	234.5	0.297	0.003	-0.116	0.063	2.4	95.3	2.4
Benz(a)Anthracene/Chrysene-C3	-1223.0	433.7	0.605	0.007	-0.215	0.140	0.0	100.0	0.0

Compound	Int	Qavg	Year	Distance (US/DS)	Qavg x Year	MSE	EMP NR Exceedance (%)		
							< LL	Inside NR	> UL
Benz(a)Anthracene/Chrysene-C4	-584.2	255.0	0.288	0.002	-0.126	0.146	0.0	92.9	7.1
Benzo(a)Anthracene	-230.3	58.1	0.113	0.001	-0.029	0.139	0.0	100.0	0.0
Benzo(a)Anthracene/Chrysene-C2	-662.0	244.3	0.328	0.004	-0.121	0.066	0.8	97.6	1.6
Benzo(a)Pyrene	-200.7	37.1	0.098	0.002	-0.018	0.227	0.0	100.0	0.0
Benzo(B)Fluoranthene	-949.6	327.7	0.470	0.001	-0.162	0.039	1.6	98.4	0.0
Benzo(e)Pyrene	-951.4	334.5	0.471	0.003	-0.166	0.042	3.1	96.9	0.0
Benzo(g,h,i)Perylene	-145.9	20.5	0.071	0.001	-0.010	0.199	0.0	100.0	0.0
Benzo(J,K)Fluoranthenes	-527.5	139.3	0.260	0.001	-0.069	0.227	0.0	100.0	0.0
Benzofluoranthene/Benzopyrene-C1	-1197.3	399.8	0.592	0.004	-0.198	0.117	0.8	99.2	0.0
Biphenyl	-658.3	184.3	0.326	-0.001	-0.092	0.264	3.9	95.3	0.8
Chrysene	-111.4	-18.2	0.055	0.003	0.009	0.395	0.0	100.0	0.0
Dibenzo(a,h)Anthracene	1049.6	-342.9	-0.522	-0.002	0.170	0.169	7.1	92.1	0.8
Dibenzothiophene	352.6	-152.5	-0.176	0.002	0.076	0.252	0.0	100.0	0.0
Dibenzothiophene-C1	526.4	-158.6	-0.263	0.004	0.079	0.145	0.8	96.9	2.4
Dibenzothiophene-C2	229.7	-63.6	-0.115	0.002	0.032	0.071	1.6	95.3	3.1
Dibenzothiophene-C3	-14.3	20.8	0.006	0.003	-0.010	0.066	0.8	96.9	2.4
Dibenzothiophene-C4	-305.4	119.7	0.151	0.004	-0.059	0.067	0.8	96.9	2.4
Dimethyl Biphenyl	554.4	-161.7	-0.277	0.003	0.081	0.148	0.8	99.2	0.0
Fluoranthene	-734.1	227.4	0.364	-0.002	-0.113	0.018	18.9	81.1	0.0
Fluoranthene/Pyrene-C1	-245.9	116.8	0.122	0.003	-0.058	0.076	0.0	0.0	0.0
Fluoranthene/Pyrene-C2	-477.7	199.7	0.237	0.004	-0.099	0.074	0.0	0.0	0.0
Fluoranthene/Pyrene-C3	-594.6	232.9	0.295	0.005	-0.115	0.083	0.8	98.4	0.8
Fluorene	-654.4	189.4	0.324	0.001	-0.094	0.076	3.1	96.9	0.0
Fluorene-C1	382.8	-84.4	-0.191	0.002	0.042	0.052	0.0	0.0	0.0
Fluorene-C2	779.5	-233.2	-0.388	0.002	0.116	0.064	0.0	0.0	0.0
Fluorene-C3	615.0	-183.1	-0.306	0.002	0.091	0.067	0.0	0.0	0.0
Indeno(1,2,3-c,d)Pyrene	-129.8	21.3	0.063	0.000	-0.011	0.168	0.8	98.4	0.8
Methyl Acenaphthene	400.2	-94.0	-0.201	0.003	0.047	0.168	0.0	99.2	0.8
Methyl Biphenyl	895.2	-313.4	-0.445	0.001	0.155	0.231	0.8	98.4	0.8
Naphthalene	-999.0	205.0	0.496	-0.001	-0.102	0.262	29.9	69.3	0.8
Naphthalene-C1	-2067.2	657.6	1.026	-0.002	-0.327	0.345	0.0	0.0	0.0
Naphthalene-C2	684.2	-254.3	-0.341	-0.001	0.126	0.373	0.0	0.0	0.0
Naphthalene-C3	979.1	-305.7	-0.487	0.001	0.152	0.061	0.0	0.0	0.0
Naphthalene-C4	1024.4	-324.2	-0.510	0.001	0.161	0.070	0.0	0.0	0.0
Perylene	-737.1	251.8	0.364	0.001	-0.124	0.045	2.4	97.6	0.0

Compound	Int	Qavg	Year	Distance (US/DS)	Qavg x Year	MSE	EMP NR Exceedance (%)		
							< LL	Inside NR	> UL
Phenanthrene	147.7	-127.4	-0.073	-0.001	0.063	0.296	0.8	99.2	0.0
Phenanthrene/Anthracene-C1	305.4	-80.4	-0.153	0.000	0.040	0.084	0.0	0.0	0.0
Phenanthrene/Anthracene-C2	105.6	-16.2	-0.053	0.002	0.008	0.060	0.0	0.0	0.0
Phenanthrene/Anthracene-C3	-8.3	19.0	0.003	0.003	-0.009	0.062	0.0	0.0	0.0
Phenanthrene/Anthracene-C4	-407.2	131.1	0.202	0.003	-0.065	0.053	0.0	0.0	0.0
Total PAH	-1.8	16.4	0.001	0.002	-0.008	0.050	0.8	96.9	2.4

3.3.2 Sediment Quality Variables

3.3.2.1 Data QA/QC

The primary difference between OSMP and EMP sediment quality datasets was in the PAH data. There were clear differences in PAH concentrations, detection limits, and proportions of non-detects between the two programs. Sediment samples collected under OSMP were analyzed for PAHs at the analytical laboratories of ECCC, while samples collected under the EMP were analyzed at SGS AXYS Ltd. commercial laboratory. The laboratory RDLs varied ~10-40 ng/g, with the OSMP detection limits being higher (Figure 92). Because of this, the percentage of individual analytes that were below detection limits in each sediment sample was much higher in the OSMP data when compared to the EMP (Figure 93). The clear differences in PAH sediment concentration results between the two programs, as stated above, did not facilitate the use of OSMP data to predict EMP data as was previously carried out for the other analytes (i.e., metals). Because of this, normal ranges for PAHs developed on the OSMP dataset could not be calculated.

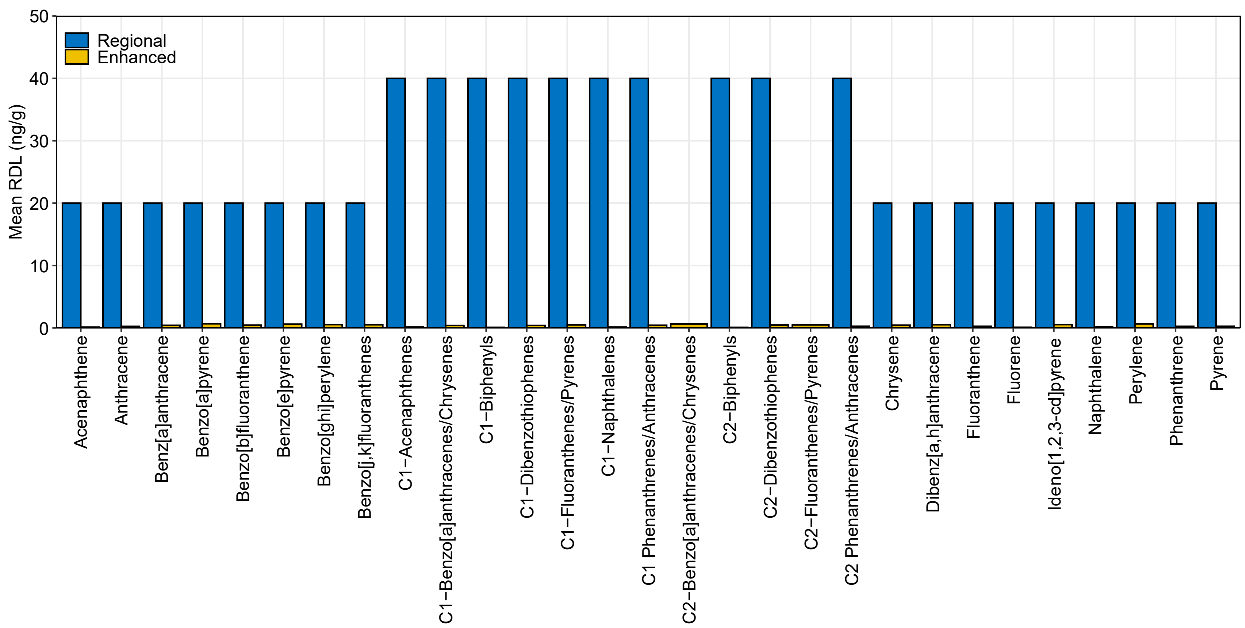

Figure 92 Differences in mean Reporting Detection Limits (RDL) between the Regional and Enhanced monitoring programs for a subset of PAHs analyzed in each program.

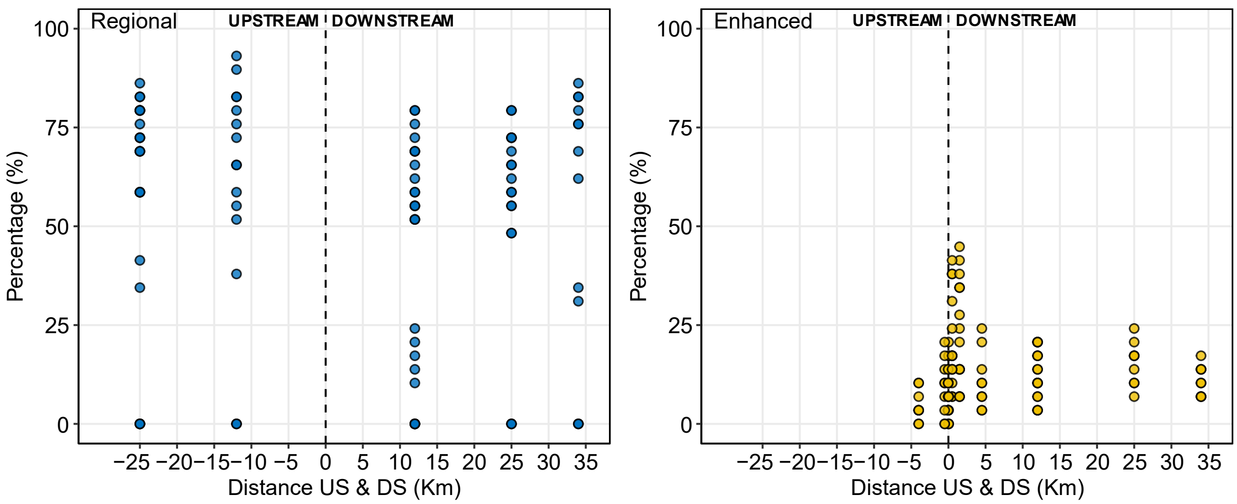

Figure 93 Percentage of non-detects observed within the subsetted PAH dataset in both the Regional and Enhanced monitoring programs from upstream to downstream stations.

3.3.2.2 Normal Ranges

Visual examination of the 34 selected sediment quality parameters measured throughout OSMP indicated that concentrations of many of the analytes covaried with total aluminum (surrogate for suspended sediments) in a log-linear form (Appendix C Table C2). Mixed-model GLM's determined that total aluminum was a significant predictor of variation in concentrations for 32 of the 34 parameters (Table 46).

Linear trends over time as well as variations in trends over time that depended on aluminum (i.e., Year x aluminum) were statistically significant for 25 of the 34 parameters (Table 46). There were statistically significant variations among numeric distance upstream and/or downstream for 12 of the 34 analytes (Table 46; i.e., Tl, K, Mn, Ni, Ba, Pb, B, Sr, Mg, Cu, Cd, and Na).

An example of the predicted normal ranges for copper generated based on OSMP data as described in Section 3.2.2.2 can be found in

Table 47 with an illustration of the model performance in Figure 94. The models in Table 46 included Aluminum, Year, distance, and Year x Aluminum as predictors of analyte concentration, regardless of the statistical significance of those terms, in part for simplicity. Here, and with over 100 samples in the overall analysis, statistical power was high, and the various predictors were generally "significant". Thus, including the non-significant terms in the model does not diminish model significance. The coefficients associated with 'non-significant' terms, further are 'not different from zero', and here with a very high Error df, are not very different from zero and have essentially negligible effect on estimated concentrations of the respective constituents.

For copper for example, the model constant was -129.8, while the slope for aluminum was 42.3, the year term was -0.06, etc. (see model breakdown in Table 47). The model MSE was 0.006 (for $\log_{10}$ of [Cu]), the square root of which is the standard deviation of observations within Sites, or here 0.075. The table provides three scenarios for which we desire an estimate of the normal range for copper concentrations. All three scenarios attempt to predict normal ranges in 2022 at Station 12km DS of the potential OSPW release point under three different aluminum conditions (i.e., 1000, 4000, and 9000 mg/kg).

Figure 95 summarizes the model predicted sediment concentrations of each analyte against the observed values at each station from field samples collected under the EMP. The model performance can be assessed based on the deviation of the points from the 1:1 line (i.e., a perfect fit). While the data points at each sampling station do tend to fall on the 1:1 line, there were instances when EMP observations were not captured by the OSM-derived predictive model. For example, total magnesium concentrations tended to vary more widely along the observed data (y-axis in Figure 95) whereas total phosphorous clusters more tightly to the 1:1 line.

Normal range model coefficients and exceedances in the EMP data are summarized in Table 48 for each individual analyte. Overall, across all analytes, sampling stations, and sampling years, between 0 and 49.6 % of EMP samples fell below the normal range lower limit, while between 0 and 29 % of EMP samples fell above the normal range upper limit. The rest of the EMP samples remained within the normal ranges. When dealing with sediment quality, our primary concern is exceedances of the upper normal range limit. The compound that exceeded this value the most often was thallium, where 29% of the EMP samples fell above the upper limit of the modeled normal range.

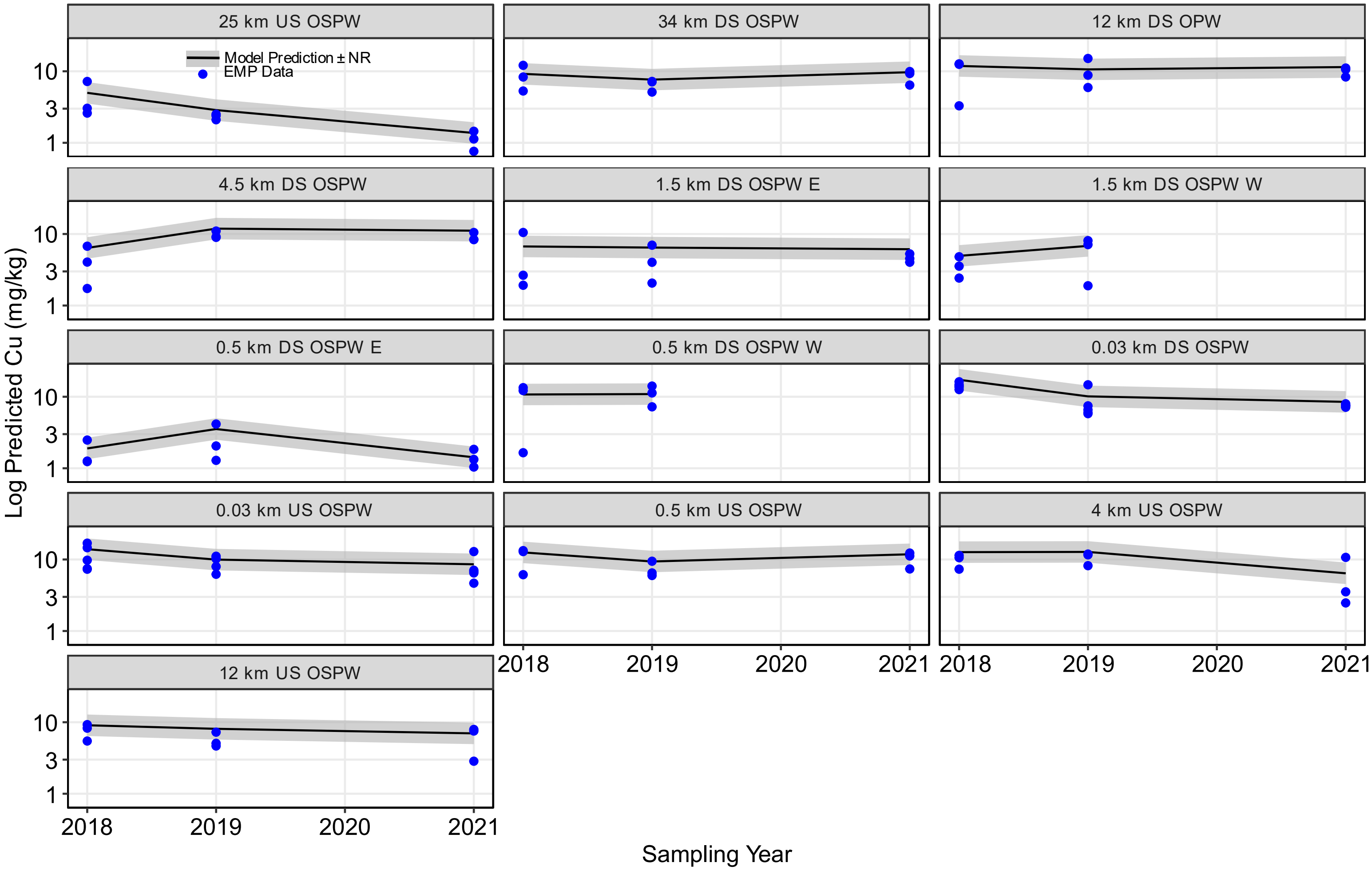

Figure 94 Variations in sediment model prediction of total copper concentrations built with OSMP data in relation to sampling year compared to observed measurements measured during EMP (2018, 2019, and 2021) across each of the 12 sampling stations.

Figure Notes: Gaps in the data represent periods when samples were not collected. NR = Normal Range (±2SD); W = West of island; E = East of Island

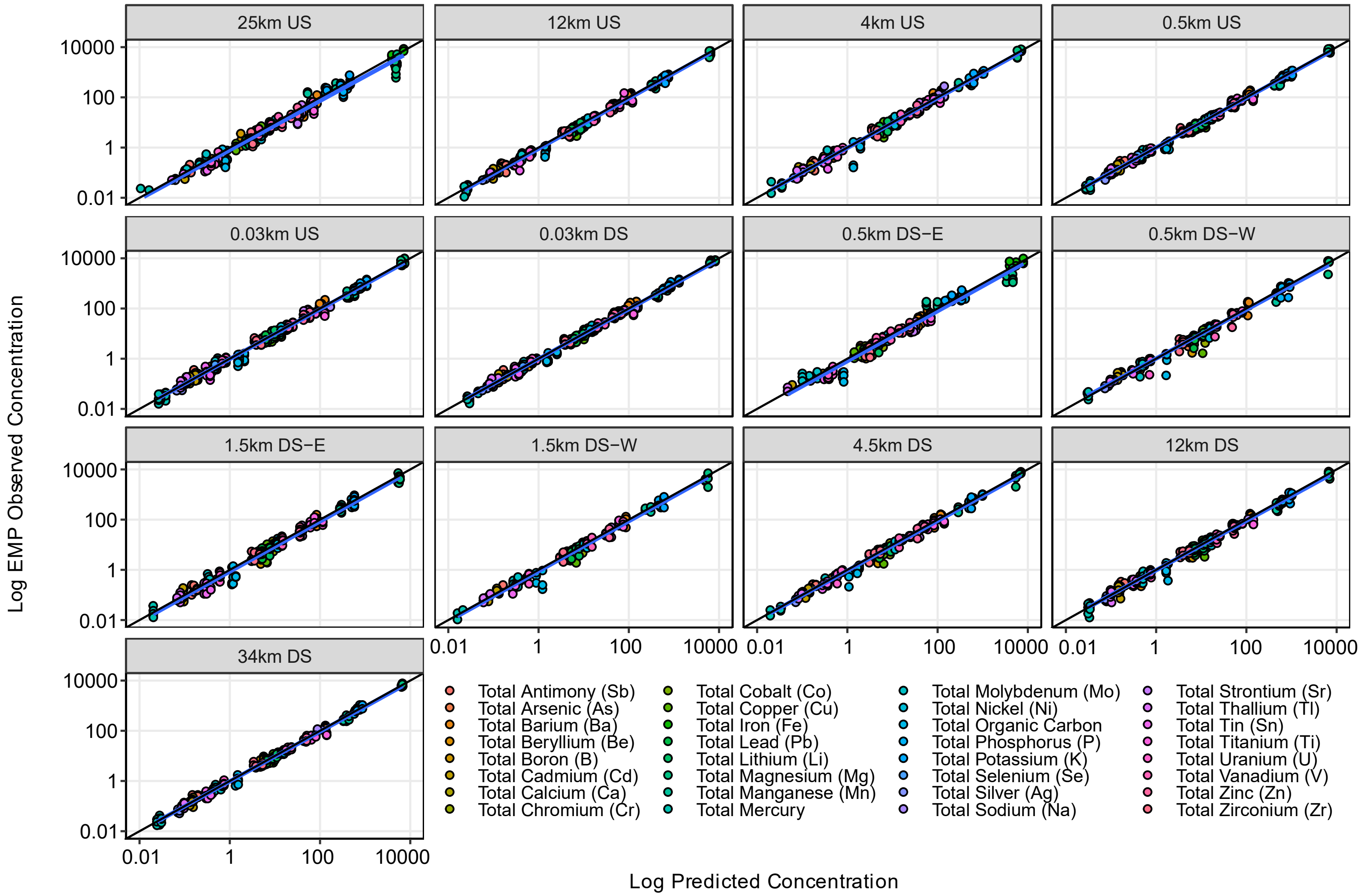

Figure 95 Sediment model performance for each of the 32 individual analytes, y-axis represents the actual measurements during EMP, and the x-axis represents predicted values based on OSMP data.

Figure Notes: The black line represents a 1:1 line, while the blue line represents the overall goodness of fit.

Table 46 Significance (p-value) and percent of variance explained (%VE) for aluminum, year, distance upstream/downstream, and year x aluminum as predictors of concentrations of sediment quality analytes in the Lower Athabasca River, under OSMP 2011 to 2021.

Parameter	Aluminum		Year		Distance (US/DS)		Aluminum x Year	
	p-value	%VE	p-value	%VE	p-value	%VE	p-value	%VE
Total Molybdenum	**<0.001**	37.4	0.072	2.5	0.084	2.29	0.228	1.1
Total Thallium	**<0.001**	83.2	**<0.001**	3.1	**0.005**	1.4	0.449	0.1
Total Phosphorus	**<0.001**	80.4	0.102	0.7	0.188	0.423	0.201	0.4
Total Potassium	**<0.001**	94.1	0.670	0.0	**<0.001**	1.20	0.104	0.2
Total Manganese	**<0.001**	67.1	0.214	0.58	**0.005**	3.0	0.079	1.17
Total Lithium	**<0.001**	96.9	0.225	0.1	0.090	0.11	0.104	0.1
Total Organic Carbon	**<0.001**	76.0	0.561	0.1	0.732	0.0	**0.004**	2.5
Total Selenium	**<0.001**	50.4	**<0.001**	10.6	0.780	0.0	0.821	0.0
Total Iron	**<0.001**	91.1	**<0.001**	1.4	0.073	0.3	**0.048**	0.4
Total Nickel	**<0.001**	89.0	**<0.001**	2.2	**0.007**	0.8	0.372	0.1
Total Barium	**<0.001**	64.6	**<0.001**	3.69	**<0.001**	11.3	0.577	0.1
Total Antimony	**<0.001**	20.7	**0.001**	11.7	0.893	0.02	0.631	0.21
Total Lead	**<0.001**	89.6	**0.001**	1.2	**0.042**	0.5	0.087	0.3
Total Zinc	**<0.001**	89.9	**<0.001**	2.8	0.079	0.3	0.225	0.1
Total Boron	**<0.001**	20.7	0.909	0.02	**<0.001**	24.0	0.708	0.2
Total Vanadium	**<0.001**	95.2	**<0.001**	0.8	0.337	0.04	**0.004**	0.4
Total Mercury	**<0.001**	75.9	0.480	0.2	0.793	0.0	0.548	0.1
Total Titanium	0.205	1.1	**<0.001**	47.3	0.845	0.0	0.199	1.1
Total Uranium	**<0.001**	85.5	**0.003**	1.4	0.059	0.6	**0.019**	0.9
Total Strontium	**<0.001**	69.9	**0.004**	2.5	**0.001**	3.6	**0.017**	1.8
Total Magnesium	**<0.001**	88.9	0.122	0.23	**0.021**	0.5	**<0.001**	3.1
Total Silver	**<0.001**	85.8	**0.006**	1.33	0.088	0.5	0.753	0.02
Total Chromium	**<0.001**	98.1	0.122	0.05	0.100	0.06	**0.008**	0.16
Total Copper	**<0.001**	89.3	**0.001**	1.3	**0.040**	0.5	0.410	0.08
Total Zirconium	**0.003**	8.6	**<0.001**	21.7	0.910	0.01	0.747	0.1
Total Tin	**<0.001**	19.5	**<0.001**	26.3	0.253	1.16	0.334	0.8
Total Calcium	**<0.001**	56.9	0.142	0.9	0.100	1.17	**<0.001**	8.9
Total Cobalt	**<0.001**	84.6	**0.001**	2.2	0.725	0.02	0.723	0.0
Total Arsenic	**<0.001**	56.0	**0.003**	4.7	0.899	0.0	0.512	0.2
Total Cadmium	**<0.001**	58.9	**<0.001**	7.9	**0.016**	3.1	0.692	0.08
Total Beryllium	**<0.001**	91.7	**0.001**	1.1	0.944	0.00	0.865	0.003
Total Sodium	**<0.001**	31.6	0.326	0.8	**0.002**	7.9	0.372	0.6

Table Notes: Significant values (i.e., p-value < 0.05) are in bold.
%VE represents the percentage of total variance explained by each predictor within the individual models
Shaded cells highlight the %VE that corresponds to significant p-values.

Table 47 Example calculation of predicted normal ranges using the model for total copper in the sediments at station 12km DS of the potential OSPW release point.

Model Component	Description	Coefficient	Scenario		
			1	2	3
Constant	Intercept	-129.8020	1	1	1
Aluminum	Slope for linear relation with Q	42.2909	1000	4000	9000
Year	Slope for linear trend across years	0.0626	2022	2022	2022
Distance (US/DS)	Slope for linear relationship with distance US/DS	-0.0009	12	12	12
Year x Aluminum	Term accounting for the different slope for year effect, depending on Q	-0.0204	6066	7283	7995
Copper estimate in logarithms			0.07	0.73	1.12
MSE in logarithms		0.006			
SD in logarithms		0.075			
Lower limit of normal range in logarithms			-0.08	0.58	0.97
Upper limit of normal range in logarithms			0.22	0.88	1.27
Copper estimate in real units			1.18	5.38	13.06
Lower limit of normal range in real units			0.84	3.81	9.24
Upper limit of normal range in real units			1.67	7.61	18.46

Table Notes: Normal Ranges were calculated as the estimate in real units ± 2SD

Table 48 Model coefficients used for the prediction of sediment quality analyte normal ranges in the EMP dataset and the percentage of NR exceedances (2018, 2019, and 2021).

Analyte	Int	log Al	Year	Distance (US/DS)	log Al x Year	MSE	EMP NR Exceedance (%)		
							< LL	Inside	> UL
Total Molybdenum	429.2	-123.8	-0.214	0.00153	0.062	0.024	0.8	96.7	2.4
Total Thallium	-88.5	29.1	0.042	0.00084	-0.014	0.003	2.9	68.0	29.1
Total Phosphorus	-141.7	40.4	0.071	0.00030	-0.020	0.002	21.1	76.4	2.4
Total Potassium	159.7	-40.6	-0.080	-0.00087	0.021	0.001	23.3	55.8	20.8
Total Manganese	434.3	-111.0	-0.216	-0.00135	0.056	0.009	16.3	75.6	8.1
Total Lithium	-124.6	34.9	0.061	-0.00031	-0.017	0.001	49.6	46.0	4.4
Total Organic Carbon	-1554.8	413.3	0.769	-0.00071	-0.204	0.043	22.0	78.0	0.0
Total Selenium	121.8	-17.7	-0.062	-0.00017	0.009	0.016	0.0	100.0	0.0
Total Iron	186.8	-45.0	-0.092	-0.00029	0.023	0.001	36.6	52.8	10.6
Total Nickel	-79.6	26.2	0.039	0.00061	-0.013	0.002	21.1	70.7	8.1
Total Barium	-90.7	29.8	0.045	0.00269	-0.014	0.006	13.0	71.5	15.4
Total Antimony	-345.6	71.2	0.170	-0.00022	-0.035	0.050	0.0	100.0	0.0
Total Lead	-186.1	54.6	0.091	-0.00056	-0.027	0.002	36.6	56.9	6.5
Total Zinc	-109.6	35.6	0.054	0.00038	-0.017	0.002	26.0	65.9	8.1
Total Boron	157.1	-40.2	-0.079	-0.00251	0.020	0.007	18.0	79.5	2.5
Total Vanadium	202.9	-57.0	-0.101	-0.00010	0.029	0.001	38.2	49.6	12.2
Total Mercury	-180.1	50.0	0.087	-0.00023	-0.024	0.011	18.7	78.9	2.4
Total Titanium	585.1	-210.9	-0.289	0.00042	0.105	0.061	0.8	99.2	0.0
Total Uranium	-256.9	71.8	0.126	0.00040	-0.035	0.002	22.0	65.0	13.0
Total Strontium	-462.7	129.0	0.229	0.00136	-0.064	0.006	19.5	73.2	7.3
Total Magnesium	-711.3	192.8	0.353	0.00049	-0.095	0.003	27.6	67.5	4.9
Total Silver	69.2	-14.1	-0.037	0.00067	0.007	0.005	11.1	88.9	0.0
Total Chromium	-156.5	41.4	0.077	-0.00023	-0.020	0.001	37.4	40.7	22.0
Total Copper	-129.8	42.3	0.063	-0.00086	-0.020	0.006	35.0	60.2	4.9
Total Zirconium	79.4	-52.9	-0.039	0.00019	0.026	0.062	0.0	100.0	0.0
Total Tin	-577.1	132.7	0.286	0.00072	-0.066	0.019	12.4	87.6	0.0
Total Calcium	-1314.9	356.6	0.654	0.00075	-0.177	0.014	19.5	77.2	3.3
Total Cobalt	-25.5	11.5	0.012	-0.00010	-0.005	0.002	18.7	77.2	4.1
Total Arsenic	126.7	-28.4	-0.063	-0.00002	0.014	0.004	9.8	88.6	1.6
Total Cadmium	-99.5	39.1	0.047	0.00197	-0.019	0.017	1.9	94.4	3.7
Total Beryllium	-6.9	5.8	0.002	-0.00002	-0.002	0.002	24.2	69.7	6.1
Total Sodium	-378.5	109.7	0.188	-0.00319	-0.054	0.034	0.0	100.0	0.0

Table Notes: Normal Ranges (NR) were calculated as the estimate in real units ± 2SD, LL and UL represent the upper and lower level, respectively, of the normal range.

3.3.3 Benthic Algae Communities

3.3.3.1 Data QA/QC

The models to establish normal ranges for algal indices of community composition were developed using EMP data only (2018, 2019, and 2021) as there was no OSMP dataset available to inform the models.

3.3.3.2 Normal Ranges

The models were developed with flow volume averaged over the 60 days prior to sampling (Q60), sampling year, and distance upstream/downstream from the proposed OSPW discharge point as predictors. The relationship between different algal indices of community composition with time and distance upstream/downstream has already been discussed and presented in section 2.3.5.2 (Table 19). In summary, density, richness, evenness, chlorophyll-a, biomass, and NMDS1 scores varied significantly with sampling year and density, evenness, and biomass varied significantly with distance (after controlling for discharge).

An example of the predicted normal ranges for algal density at the 12 km downstream EMP station can be found in Table 49 with an example of the model performance for each index at that station in Figure 96. The GLM models for algae used Year and distance, regardless of the statistical significance of those terms, in part for simplicity. Here, and with over 120 samples in the analysis, statistical power was high. Thus, including the non-significant terms in the model does not diminish model significance.

For algal density, the model constant was 738, while the slope for Q60 was -5.4, the slope for the linear trend with year was -0.36 and the slope for distance was -0.0143 (see full model breakdown in Table 49). The model MSE was 0.482 (for $\log_{10}$ of density), the square root of which is 0.694. The SD among samples, for any modeled scenario is therefore 0.694.

Figure 97 summarizes the EMP algal index models as evaluated against the observed values from field samples collected under the EMP. The model performance can be assessed based the deviation of the points from the 1:1 line (i.e., a perfect fit). While the data points at each sampling station do tend to fall on the 1:1 line, there are instances variation within the experimental results that are not captured by the predictive model. For example, total density measurements tend to vary more widely along the observed data (y-axis in Figure 97) whereas indices such as Simpson’s Diversity cluster more tightly to the 1:1 line. This can be attributed to general variability among the field samples, as was demonstrated in Section 2.3.5.2 where total density varied by up to 3 orders of magnitude within a specific sampling station (Figure 33).

Normal range model coefficients and exceedances in the EMP data are summarized in Table 50 for each index of community composition. Normal ranges were computed as the model predicted average index value ± 2 SDs. In this context, the normal range was anticipated to capture ~ 95% of potential future observations (Kilgour et al., 1998). Normal range model coefficients can be found in Table 50. Between 0 and 5.7 % of EMP samples fell below the normal range lower limit and above the normal range upper limit, the rest of the samples (>90%) fell within the predicted normal range. These exceedance probabilities are about as predicted given that the EMP data were used to compute normal ranges, and the normal range region was designed to cover 95% of the data.

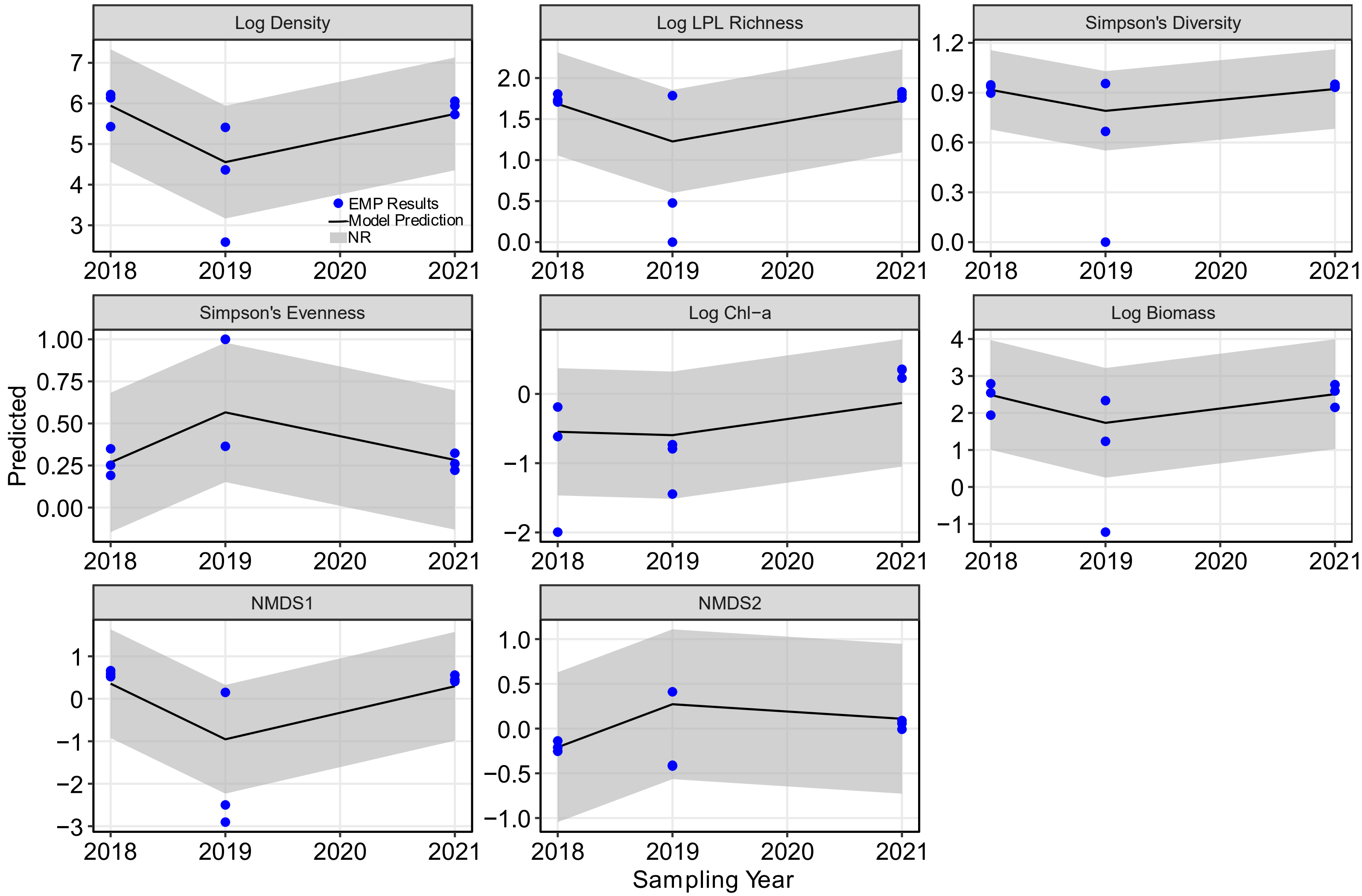

Figure 96 Variations in algal indices of community composition model predictions generated with EMP data in relation to sampling year overlayed with observed measurements during EMP (2018, 2019, and 2021).

Figure Note: NR = Normal Range (±2SD); Model predictions from the EMP sampling station located at 12 km downstream are provided as an example.

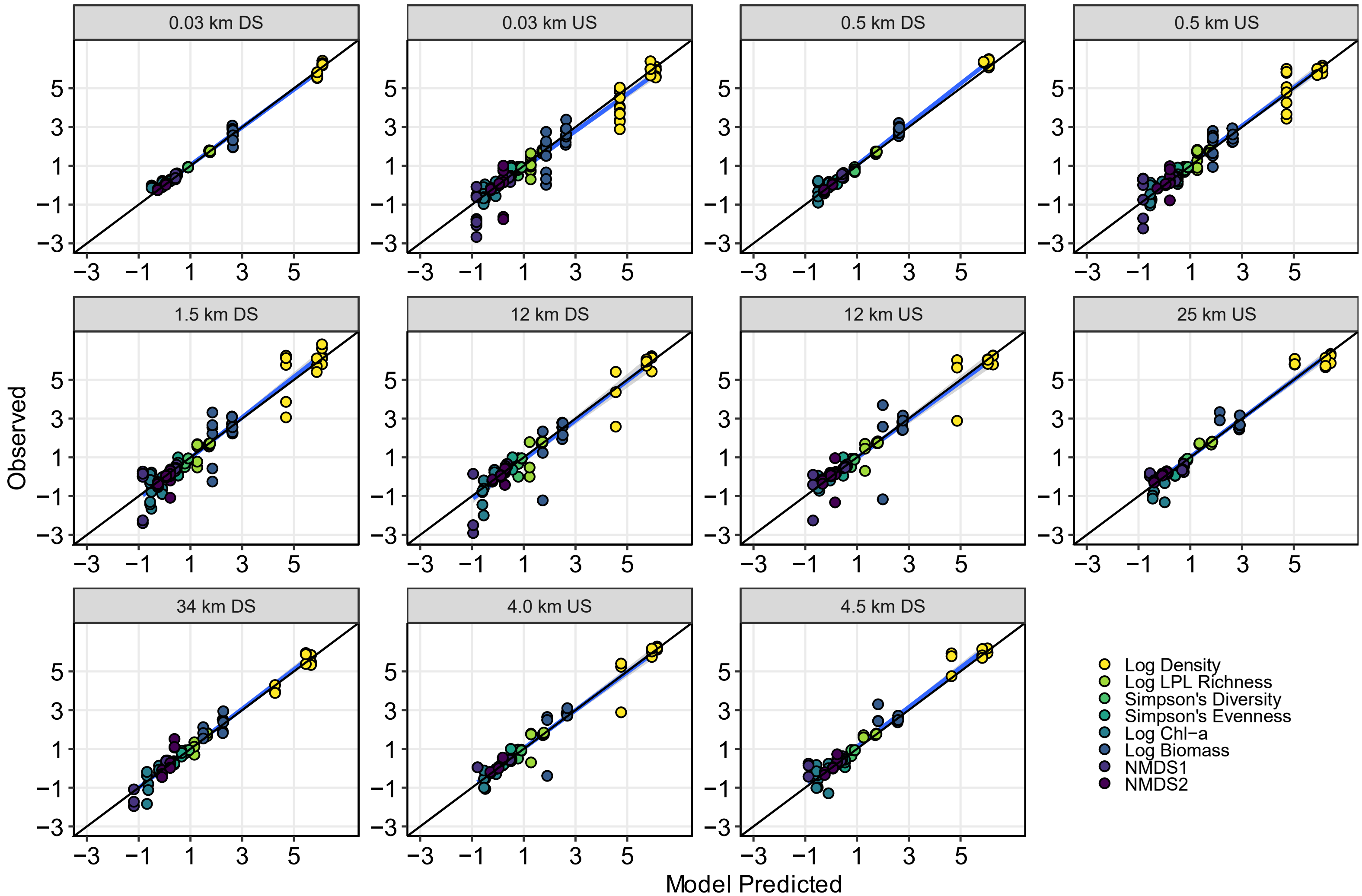

Figure 97 Algal indices of community composition model performance for each of the 9 modelled indices, x-axis represents the actual measurements during EMP, and the y-axis represents normal range model predictions.

The black line represents a 1:1 line, while the blue line represents the overall goodness of fit at each sampling station.

Table 49 Example normal range model output for algal density under different scenarios of discharge.

Model Component	Description	Coefficient	Scenario		
			1	2	3
Constant	Intercept	738.387	1	1	1
Q60	Slope for linear relation with Q	-5.406	600	900	1200
Year	Slope for linear trend across years	-0.355	2022	2022	2022
Distance (US/DS)	Slope for linear relationship with distance US/DS	-0.013	12	12	12
MSE (logarithm)		0.482	—		
SD (logarithm)		0.694	—		
Density estimate in log units			5.40	4.45	3.77
Lower limit of normal range in log units			4.01	3.06	2.39
Upper limit of normal range in log units			6.79	5.84	5.16
Density estimate in real units			**252531**	**28208**	**5956**
Lower limit of normal range in real units			**10323**	**1153**	**243**
Upper limit of normal range in real units			**6177870**	**690064**	**145704**

Table Notes: Normal Ranges were calculated as the estimate in real units ± 2SD

Table 50 Model coefficients used for the prediction of benthic indices of community composition normal ranges in the EMP dataset and the percentage of NR exceedances (2018, 2019, and 2021).

Index	Int	Q60	Year	Distance	MSE	EMP NR Exceedance (%)		
						< LL	Inside NR	> UL
log Density	738.387	-5.406	-0.355	-0.013	0.482	4.9	92.6	2.5
log Richness	187.629	-1.919	-0.089	-0.003	0.099	4.9	95.1	0
Simpson's Diversity	55.398	-0.524	-0.026	0.001	0.014	4.1	95.9	0
Simpson's Evenness	-141.664	1.197	0.069	0.004	0.043	5.7	88.5	5.7
log Chl-*a*	-196.052	-0.767	0.098	-0.004	0.211	4.1	95.9	0
log Biomass	332.421	-3.115	-0.159	-0.011	0.550	4.9	93.4	1.6
NMDS1	623.057	-5.271	-0.301	-0.011	0.409	5.7	94.3	0
NMDS2	-383.188	1.532	0.187	0.005	0.175	4.1	95.1	0.82

Table Notes: Normal Ranges were calculated as the estimate in real units ± 2SD. Int = Intercept; Q = Q = average discharge over previous 60 days; LL = Lower Level; UL = Upper Level

3.3.4 Benthic Invertebrate Communities

3.3.4.1 Data QA/QC

There are inherent differences in the OSMP (2011 – 2015) and EMP (2018 – 2021) benthic community indices datasets in terms of habitats sampled, which make the comparison of the two programs difficult. Historically, the OSMP sampled areas along the Athabasca River that were characteristics of a riverine area and included both cobble and sand as the primary substrate types. In the case of the EMP benthic datasets, depositional habitats were the dominant habitats available. Because of the differences in substrate type, we observed significant differences in the distribution of common benthic indices of community composition (i.e., density, richness, diversity, evenness, and NMDS 1 & 2 scores) between the two programs.

A useful exploratory step in comparing of benthic indices of community composition between OSMP and EMP involves the use of PCA (Figure 98A). The PCA plot demonstrated clear separation between the data points, with EMP samples clustering primarily in the bottom right quadrat of the PCA, corresponding to positive PC axis 1 scores and negative PC axis 2 scores, whereas OSMP samples show two distinct groups, one of which is negatively correlated with PC axis 1 and the other positively correlated with PC axis 1 (Figure 98A). It is expected that the primary cause of the two clusters in the OSMP dataset is due to the sample split between sand and cobble samples. Figure 98B separates data points based on the substrate sampled between the two programs and it is clear that the cobble samples are driving the cluster separation in the OSMP dataset. However, even after removing cobble samples and rerunning the PCA to compare the indices between programs, there is limited overlap in ellipses (Figure 99). Because the OSMP benthic community data set obviously captured a fundamentally different benthic community than that of the EMP, the OSMP data were not used to compute normal ranges.

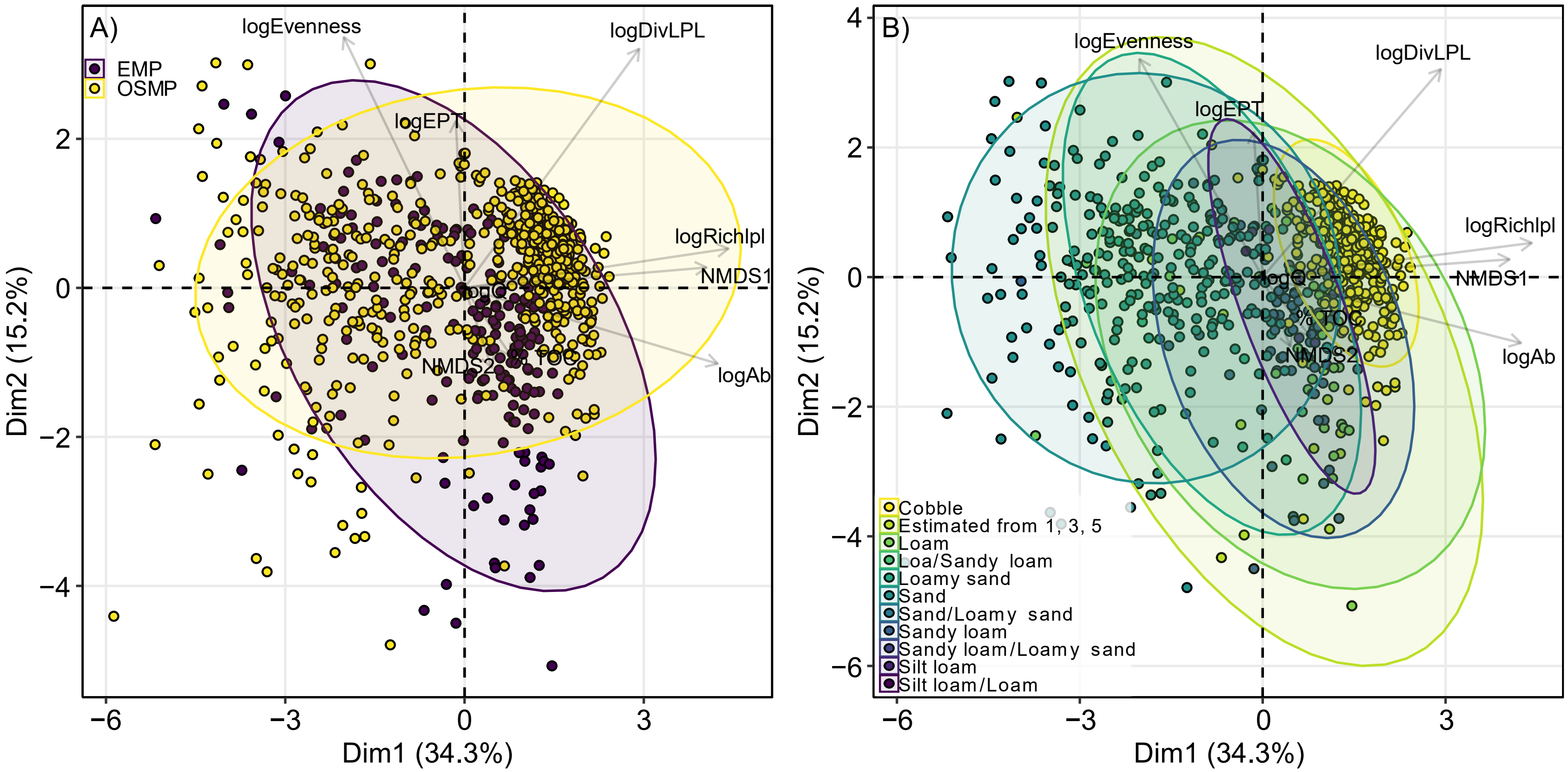

Figure 98 PCA of BIC in both OSMP and EMP by dataset (A) and substrate type (B).

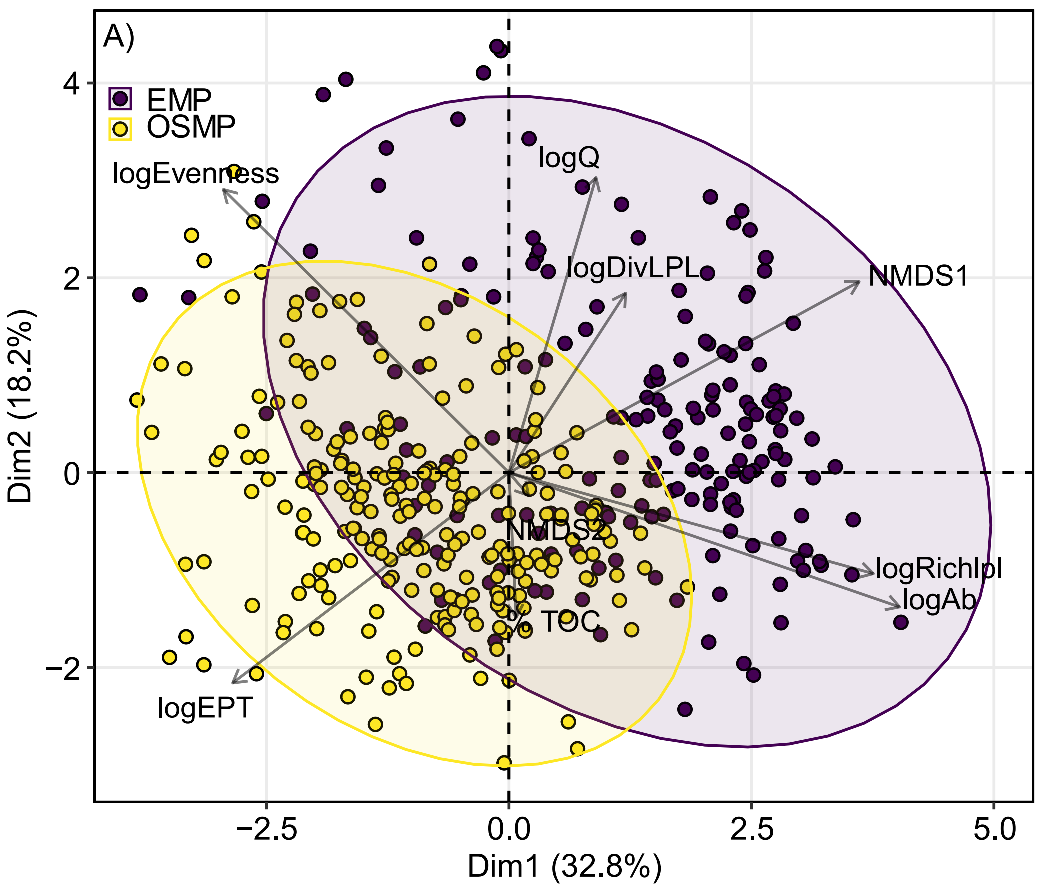

Figure 99 PCA of BIC comparing datasets (OEMP and EMP) with cobble samples removed.

3.3.4.2 Spatial and Temporal Variation in OSMP Data

Mixed-model GLM's determined that Q60 was a significant predictor of variation in benthic indices of composition for NMDS2, density, and richness (Table 51).

Linear trends over time as well as variations in trends over time that depended on discharge (i.e., Year x Q60) were statistically significant for NMDS1, NMDS2, density, richness, EPT, and evenness (Table 51). Visual inspection of scatterplots of benthic indices of composition over time (Figure 100) indicate a general increasing trend over time for NMDS2, density, richness, EPT, and evenness, and a decreasing trend over time for NMDS1 scores. While these temporal trends were statistically significant, the magnitude of change over time across all sampling stations is very small (Figure 100).

There were statistically significant variations among numeric distance upstream and/or downstream NMDS1, NMDS2, richness, and EPT (Table 51). Figure 101 demonstrates no observable change in density, diversity, or evenness with increasing distance downstream, while EPT and NMDS2 scores show signs of increasing, and richness and NMDS1 scores show signs of decreasing.

Table 51 Significance (p-value) and percent of variance explained (%VE) for discharge, year, distance upstream/downstream, and year x discharge as predictors of benthic indices of community composition in the Lower Athabasca River, under OSMP 2011 to 2015.

Indices	Q60		Year		Distance (US/DS)		Q60 x Year	
	P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE
NMDS1	0.387	0.4	**0.011**	3.3	**0.001**	5.6	0.221	0.7
NMDS2	**<0.001**	8.4	**0.002**	3.4	**<0.001**	17.4	**<0.001**	6.1
log Density	**0.014**	2.8	0.160	0.9	0.710	0.1	**<0.001**	13.6
log Richness	**0.002**	4.9	0.469	0.3	**0.026**	2.5	0.014	3.0
log EPT	0.055	1.7	0.528	0.2	**<0.001**	9.3	**0.001**	5.6
log Diversity	0.190	0.9	0.899	<0.1	0.321	0.5	0.227	0.8
log Evenness	0.160	1.0	0.293	0.6	0.558	0.2	**0.014**	3.2

Table Notes: Shaded cells represent percent variance explained by the predictor when the p-value is significant (i.e. p-value < 0.05). Data was log-transformed (base 10) where indicated.

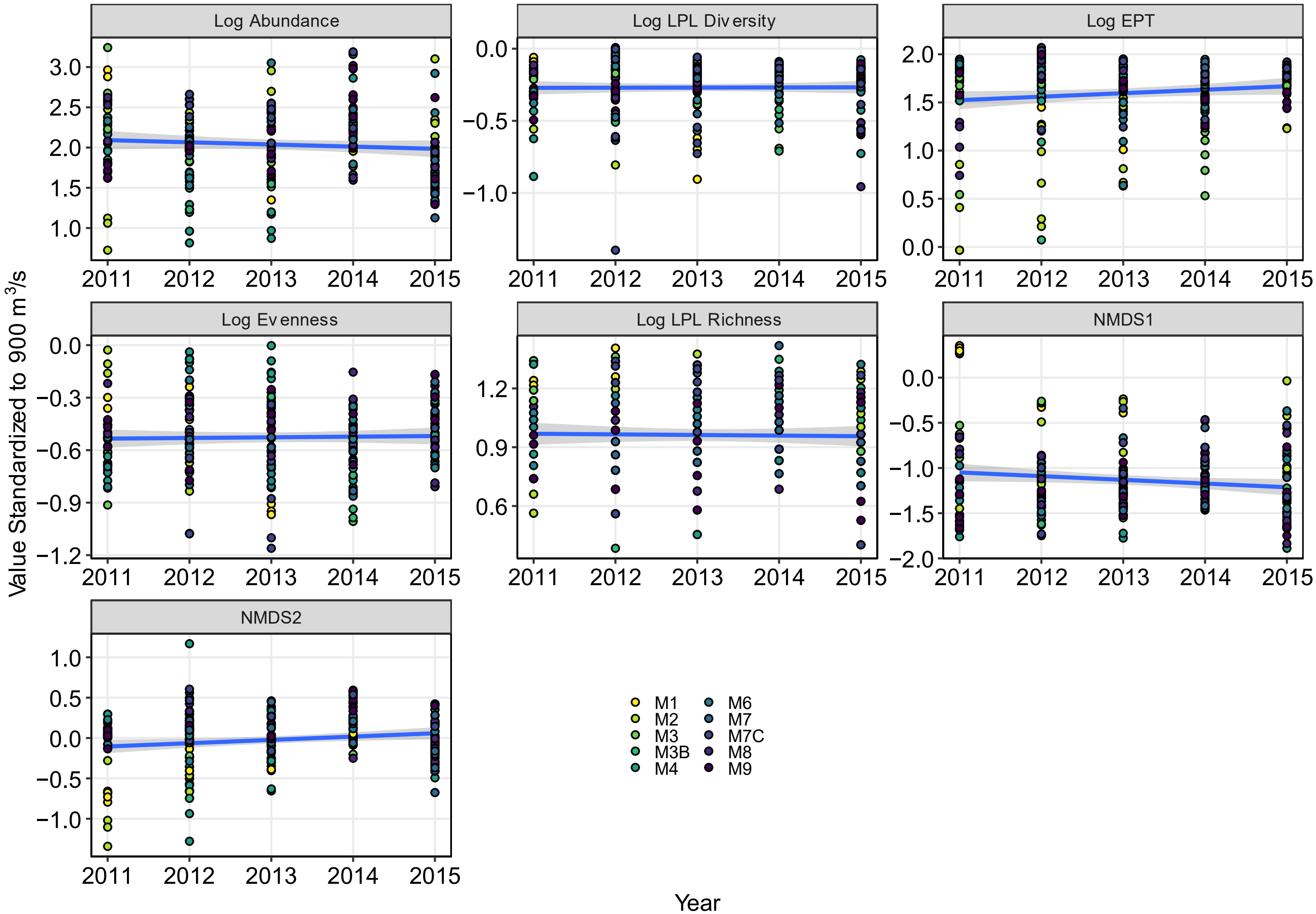

Figure 100 Scatterplot of benthic indices of community composition (standardized to Q of 900 m^3/s) in relation to sampling year under OSMP (2011 – 2015).

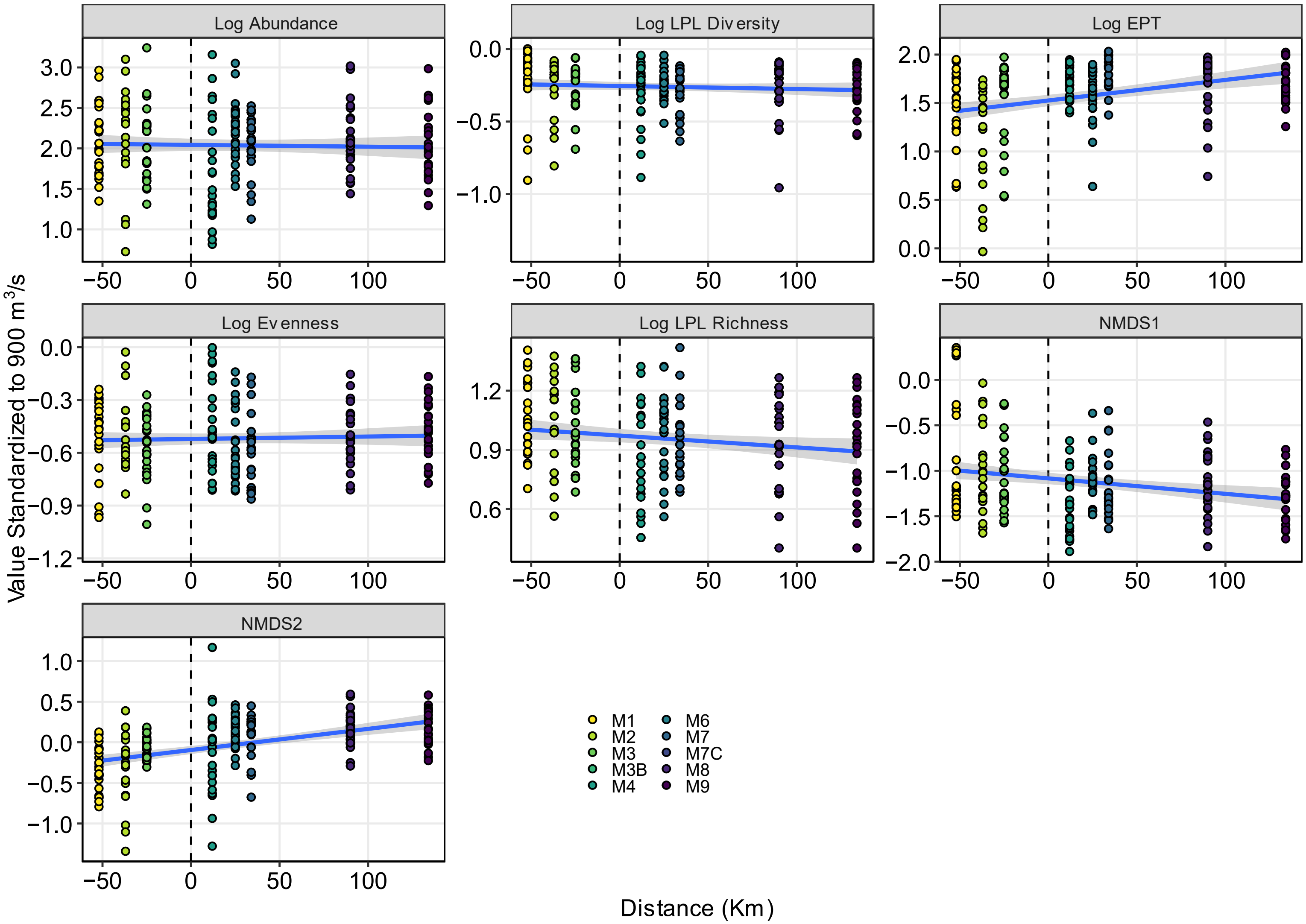

Figure 101 Scatterplot of benthic indices of community composition (standardized to Q of 900 m³/s) in relation to distance US/DS from the proposed OSPW discharge under OSMP (2011 – 2015).

3.3.4.3 Normal Ranges

Because of the inherent differences in the two datasets (OSMP and EMP), in terms of substrate type and areas sampled as described above, it was determined that the OSMP data cannot be used to develop normal ranges for the EMP dataset. Normal ranges were generated using only the EMP dataset to inform on future sampling during EMP.

The model used to generate normal ranges was developed with flow volume averaged over the 60 days prior to sampling (Q60), particle size, sampling year, and distance upstream/downstream from the proposed OSPW discharge point as predictors. The relationship between different benthic indices of community composition with time and distance upstream/downstream has already been discussed and presented in section 2.2.7.3 (Table 22). In summary, no index varied with sampling year, while Simpson's evenness, NMDS1, and NMDS2 scores varied significantly with distance (after controlling for Q60 and particle size).

An example of the predicted normal ranges for the benthic invertebrate communities (BIC) at the 12 km downstream EMP station can be found in Table 52 with an illustration of the model performance for that station in Figure 102. The models for benthic indices of community composition have retained all of the components Year and distance, regardless of the statistical significance of those terms, in part for simplicity. Here, and with over 185 samples in the overall analysis, there is no problem with statistical power. Thus, including the non-significant terms in the model does not diminish model significance. The coefficients associated with 'non-significant' terms, further are 'not different from zero', and here with a very high Error df, are not very different from zero and have essentially negligible effect on estimated concentrations of the respective constituents.

For total density, the model constant was -31, while the slope for Q60 was -1.6, the slope for particle size was -2.3, the slope for the linear trend with year was 0.019, and the slope for distance was 0.0049 (see full model breakdown in Table 52). The model MSE was 0.472 (for $\log_{10}$ of Q60), the square root of which is 0.687. The SD among samples, for any modeled scenario is therefore 0.687.

Figure 103 summarizes the EMP BIC models as evaluated against the observed values from field samples collected under the EMP. The model performance can be assessed based on the deviation of the points from the 1:1 line (i.e., a perfect fit). While the data points at each sampling station do tend to fall on the 1:1 line, there are instances of variation within the experimental results that are not captured by the predictive model. For example, NMDS 1 and 2 scores, as well as total density, tend to vary more widely along the observed data (y-axis in Figure 103) whereas indices such as Simpson's Diversity cluster more tightly to the 1:1 line. This can be attributed to general variability among the field samples, as was demonstrated in Section 2.3.6.2 where total density varied by up to 3 orders of magnitude within a specific sampling station (Figure 43), and NMDS 1 and 2 axes scores varied by a full unit within a single site, primarily at 0.03km downstream (Figure 49 & Figure 50). Normal range model coefficients and exceedances in the EMP data are summarized in Table 53 for each BIC. Normal ranges for benthic indices of composition were computed as the predicted index mean ± 2 SDs. Between 0 and 6.5 % of EMP samples fell below the normal range lower limit and between 0 and 5.9 % of EMP samples fell above the normal range upper limit, the rest of the samples (>90%) fell within the predicted normal range. As with water quality and sediment quality variables, these exceedance probabilities are about as expected, given the construct of the normal range.

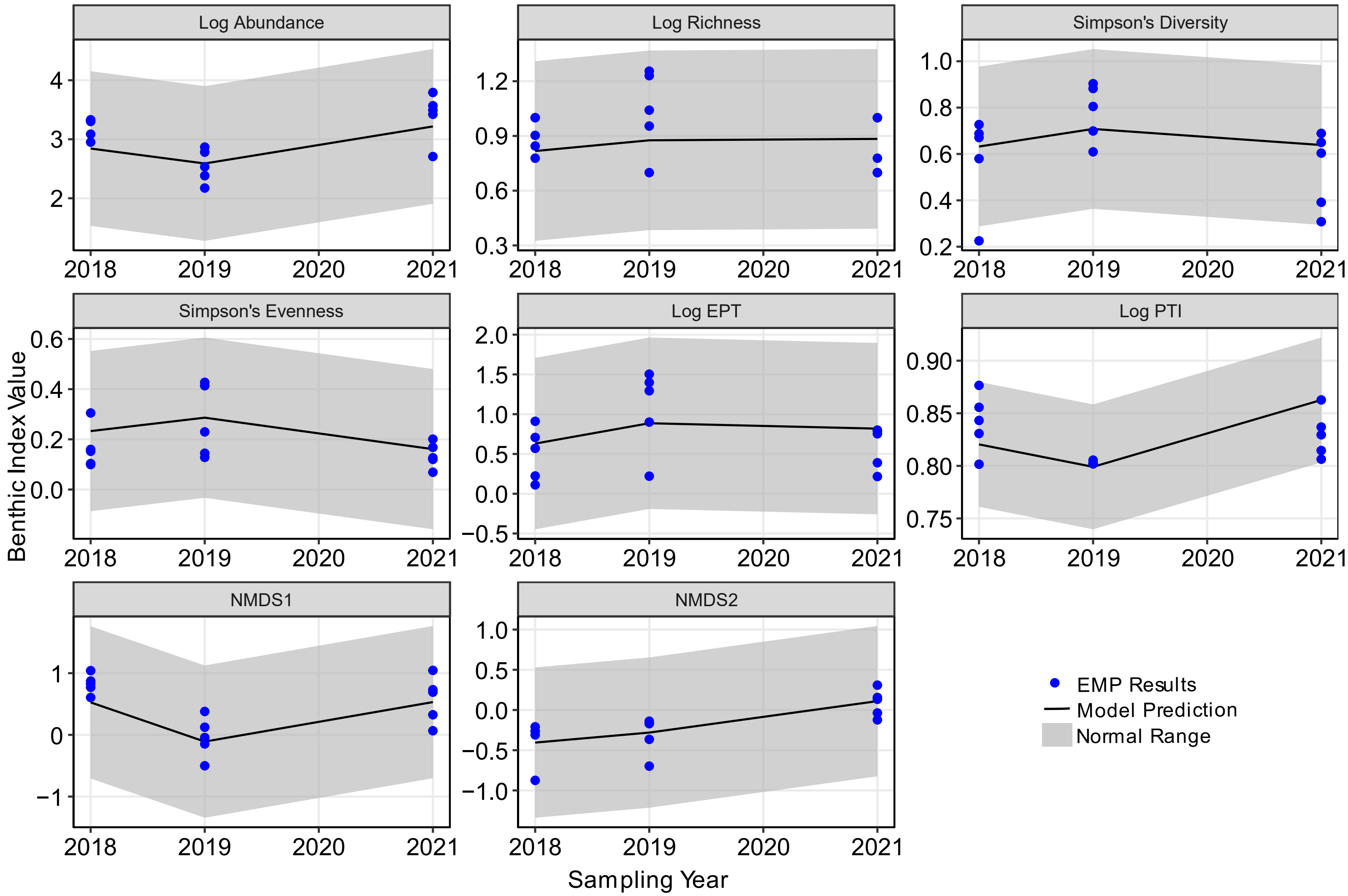

Figure 102 Variations in benthic index values of community composition model predictions generated with EMP data in relation to sampling year overlayed with observed measurements during EMP (2018, 2019, and 2021).

Figure Notes: NR = Normal Range (±2SD); Model predictions from the EMP sampling station located at 12 km downstream are provided as an example.

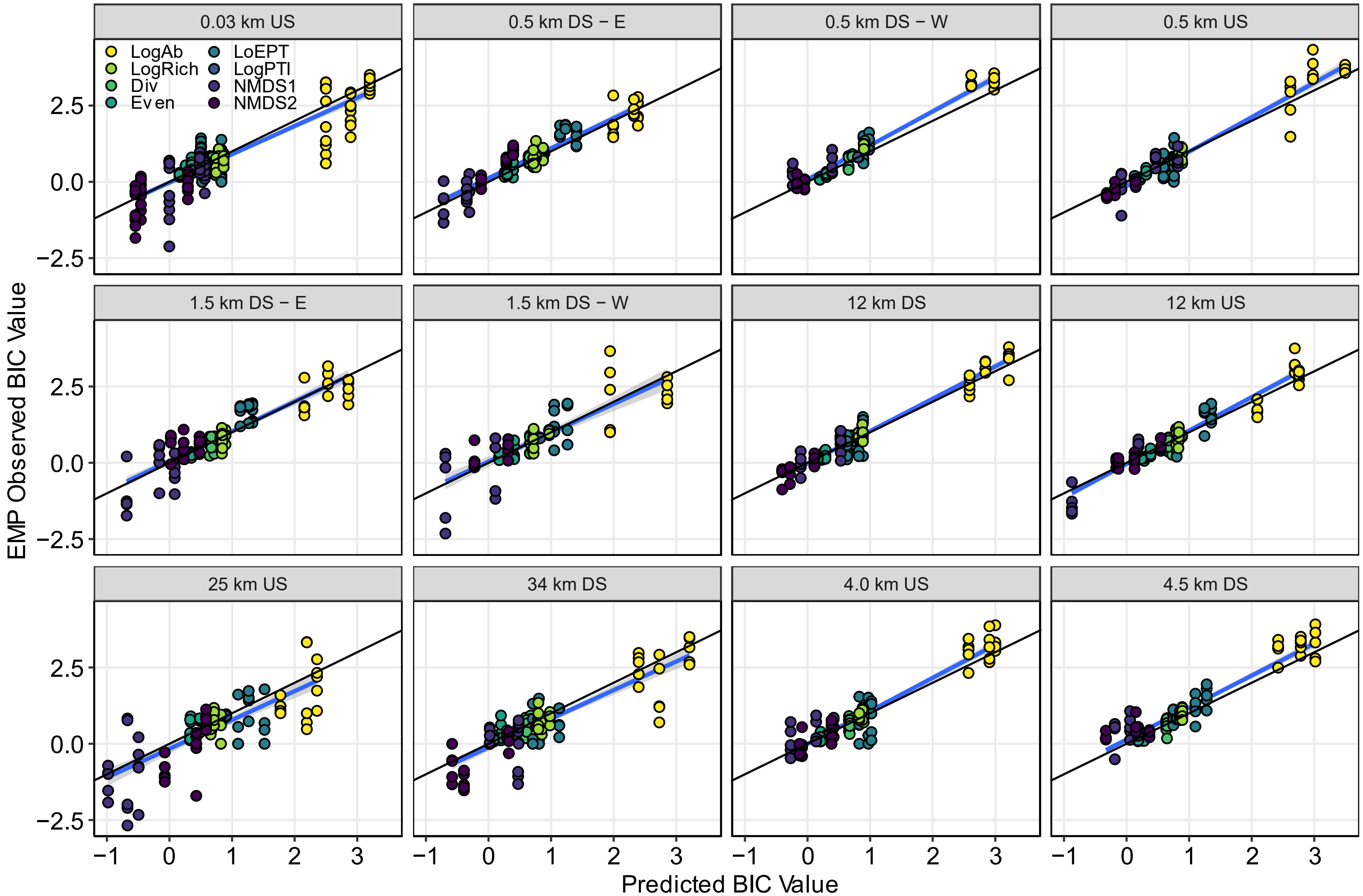

Figure 103 Benthic indices of community composition model performance for each of the 6 modelled indices, x-axis represents the actual measurements during EMP, and the y-axis represents normal range model predictions.

Figure Notes: The black line represents a 1:1 line, while the blue line represents the overall goodness of fit at each sampling station

Table 52 Example normal range model output for the different benthic indices of community composition across under a specific scenario of discharge and particle size.

Model Component	Description	Density		Richness		Diversity		Evenness		EPT		PTI	
		Coeff	Scenario	Coeff	Scenario	Coeff	Scenario	Coeff	Scenario	Coeff	Scenario	Coeff	Scenario
Constant	Intercept	-31	1	26	1	-36	1	-42	1	58	1	74	1
Discharge	Slope for linear relation with Q	-1.6	1200	-0.4	1200	0.4	1200	0.3	1200	-0.4	1200	-2.5	1200
Particle Size	Slope for linear relation with PS	-2.3	0.4	-0.3	0.4	0.4	0.4	0.0	0.4	1.9	0.4	-2.9	0.4
Year	Slope for linear trend across years	0.019	2022	-0.012	2022	0.017	2022	0.021	2022	-0.027	2022	-0.032	2022
Distance (US/DS)	Slope for linear relationship with distance US/DS	0.0049	12	0.0017	12	-0.0009	12	-0.0015	12	-0.0023	12	0.0033	12
BIC estimate in logarithms		—	3.28	—	1.28	—	—	—	—	—	0.41	—	2.04
MSE in logarithms		0.472	—	0.063	—	0.033	—	0.030	—	0.344	—	0.467	—
SD in logarithms		0.687	—	0.251	—	0.181	—	0.172	—	0.586	—	0.684	—
Lower limit of normal range in logarithms		—	1.91	—	0.78	—	—	—	—	—	-0.76	—	0.68
Upper limit of normal range in logarithms		—	4.66	—	1.78	—	—	—	—	—	1.58	—	3.41
BIC estimate in real units		—	**1912.03**	—	**18.92**	—	**0.26**	—	**0.73**	—	**2.56**	—	**110.65**
Lower limit of normal range in real units		—	**80.70**	—	**5.96**	—	**-0.10**	—	**0.39**	—	**0.17**	—	**4.75**
Upper limit of normal range in real units		—	**45303.96**	—	**60.08**	—	**0.63**	—	**1.07**	—	**38.07**	—	**2576.63**

Table Notes: Normal Ranges were calculated as the estimate in real units ± 2SD. Data was log-transformed (base 10) where indicated.

Table 53 Model coefficients used for the prediction of benthic indices of community composition normal ranges in the EMP dataset and the percentage of NR exceedances (2018, 2019, and 2021).

Index	Int	log Q	log PS	Year	Distance (US/DS)	MSE	EMP NR Exceedance (%)		
							< LL	Inside NR	> UL
log Density	-31.3	-1.63	-2.25	0.019	0.005	0.472	4.9	95.1	0.0
log Richness	23.4	-0.18	-0.17	-0.011	0.002	0.066	2.2	96.8	1.1
Simpson's Evenness	-17.1	0.47	0.22	0.008	-0.002	0.027	0.0	94.1	5.9
Simpson's Diversity	-42.4	0.28	-0.03	0.021	-0.002	0.030	6.5	93.5	0.0
log EPT	58.0	-0.42	1.91	-0.027	-0.002	0.344	3.2	96.8	0.0
log PTI	-18.0	-0.11	0.04	0.009	-0.001	0.001	0.5	94.1	5.4
NMDS1	139.6	-1.96	-2.60	-0.066	0.011	0.432	5.4	94.6	0.0
NMDS2	-86.1	-1.45	1.33	0.045	-0.006	0.252	5.9	94.1	0.0

Table Notes: Normal Ranges were calculated as the estimate in real units ± 2SD. Int = Intercept; Q = average discharge over previous 60 days; PS = Particle Size; LL = Lower Level; UL = Upper Level. Values were log-transformed (base 10) where indicated.

3.3.5 Fish Community Assessment

3.3.5.1 Data QA/QC

In this instance, the long term regional data that was provided was from RAMP which included over 20 years of boat electrofishing community data that recorded components such as collection date, site, species, sex, length, body weight, transect information, and total shocking time along the LAR. Table 54 demonstrates reasonable overlap between both RAMP and EMP programs in terms of the sites sampled along the LAR, with the RAMP covering a more expanded reach of the river.

Table 54 Comparison of sampling sites in both the EMP and RAMP fish community assessment programs.

Site Code	Description	EMP	RAMP
01A	20 Km US W	x	x
04A	4 Km US W	x	x
04B	4 Km US E	x	x
05A	0.5 Km US W	x	x
05B	0.5 Km US E	x	x
06A	1.5 Km DS W of Island	x	x
10B	13 Km DS E	x	x
11A	13 Km DS W	x	x
16A	31 Km DS W		x
17A	33 Km DS W		x
19B	40 Km DS E		x
19A	42 Km DS W		x
00B	22 Km US E		x
-03B	42 Km US E		x

The data collected during these programs were used to determine several effect indicators (i.e., abundance, richness, evenness, Bray-Curtis index of similarity), along with other indices of community composition. Summary statistics were determined and trends, both spatial and temporal, were investigated and discussed below.

A total of 62,725 fish across 27 different species were collected in from 1987 to 2014 (Table 55). The catch per unit effort (CPUE) ranged from 0.56 to 4.88 fish/min and was 2.78 fish/min for the entire program across all sampling years. The relative abundance of the different fish species is illustrated in Figure 104 for stations located upstream and downstream of the proposed OSPW discharge point. The most abundant species caught during RAMP were Goldeye and Troutperch. Other abundant species include Walleye and White Sucker. The remaining 23 species all had lower relative abundances (i.e., < 10%).

The nature of this dataset is very similar to that which was produced under EMP and summarized in Section 2.3.7, therefore no data QA /QC was required to use OSMP data to predict the EMP data.

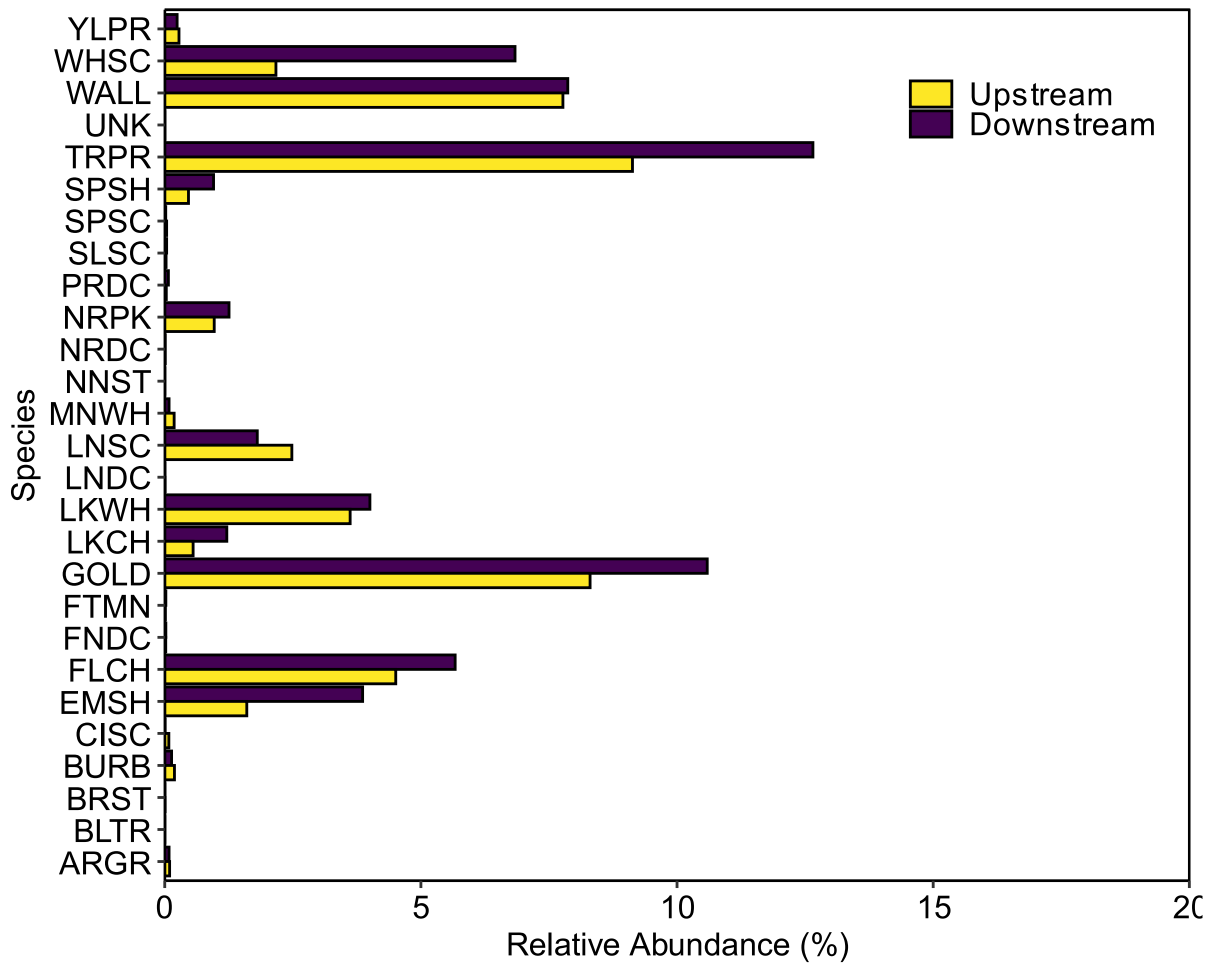

Figure 104 Relative Abundance fish species collected both upstream and downstream of the proposed OSPW discharge point over the 23 years of RAMP.

Table 55 Fish community catch assemblage in the LAR under RAMP over 23 years.

Common name	Abbv	Scientific name	1987	1989	1990	1991	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	Total
Flathead Chub	FLCH	*Platygobio gracilis*	26	51	2	1	26	138	99	29	60	29	66	60	243	204	225	153	388	272	589	384	857	501	510	4913
Goldeye	GOLD	*Hiodon alsoides*	27	106	87	75	80	265	159	83	48	22	65	110	304	360	133	215	1164	666	346	1566	1611	1164	638	9294
Lake Whitefish	LKWH	*Coregonus clupeaformis*	1	40	17	3	30	56	132	43	20	2	25	45	76	154	127	298	579	458	447	396	387	252	32	3620
Longnose Sucker	LNSC	*Catostomus catostomus*	139	22	3		19	87	103	35	45	17	54	113	90	105	160	185	232	249	439	417	450	523	228	3715
Northern Pike	NRPK	*Esox lucius*	5	68	11	15	46	60	33	71	23	10	27	192	138	195	249	208	222	251	221	145	203	192	55	2640
Walleye	WALL	*Sander vitreus*	95	132	110	99	128	351	222	172	43	39	266	313	319	707	515	410	713	549	745	737	910	509	325	8409
White Sucker	WHSC	*Catostomus commersoni*	7	32	3		69	78	89	43	19	20	56	232	274	408	488	909	1185	846	801	1023	912	1013	579	9086
Arctic Grayling	ARGR	*Thymallus arcticus*		35	1			4	13				1	17	14	7	19	13	3	13	75	7	17	5	1	245
Burbot	BURB	*Lota lota*		2	2	5		3	3	3			2	1		8	4	20	11	8	18	14	20	35	20	179
Emerald Shiner	EMSH	*Notropis atherinoides*		2			11	3	63	4	7	38	23	6	29	14	52	40	109	116	206	296	462	931	140	2552
Mountain Whitefish	MNWH	*Prosopium williamsoni*		1	2		5	19	5	3			3	47	28	29	18	27	38	19	104	25	24	2	1	400
Trout Perch	TRPR	*Percopsis omiscomaycus*		1		2	16	62	89	10	14	11	47	95	441	206	1059	815	1278	873	2670	1717	1330	389	485	11610
Brook Stickleback	BRST	*Culaea inconstans*					1								1	1		1	8		2			1		15
Lake Chub	LKCH	*Couesius plumbeus*					28	40	6	1	1		4	25	31	249	295	58	402	111	135	254	113	200	40	1993
Spottail Shiner	SPSH	*Notropis hudonius*					2	10	31	1	2	4	2	44	41	162	151	134	282	218	190	496	560	758	54	3142
Yellow Perch	YLPR	*Perca flavescens*					4	10	3	3			4	6	10	4	40	27	18	24	109	19	32	40	18	371
Spoonhead Sculpin	SPSC	*Cottus ricei*								1			3	2	1	3	18	10	4	3	8	8	3	7	1	72
Longnose Dace	LNDC	*Rhinicthys cataractae*											1				3	4	12		1	4	5			30
Slimy Sculpin	SLSC	*Cottus cognatus*											2	5	1	7		5	41	23	32	94	22	26	1	259
Cisco	CISC	*Coregonus artedii*												4		34		1					3			42
Pearl Dace	PRDC	*Semotilus margarita*												2				3	19		15	17		5	2	63
Fathead Minnow	FTMN	*Pimephales promelas*													8		7	6	3			1	3			28
Bull Trout	BLTR	*Salvelinus confluentus*														1										1
Lake Trout	LKTR	*Salvelinus namaycush*															1						1			2
Ninespike Stickleback	NNST	*Pungitius pungitius*																1					1			2
Finescale Dace	FNDC	*Phoxinus neogaeus*																	9		1	3	4	10		27
Northern Redbelly Dace	NRDC	*Phoxinus eos*																		1			4	10		15
Catch Summary Statistics																										
Number of Species			7	12	10	7	14	15	15	15	11	10	18	19	18	20	19	23	22	18	21	21	24	21	18	27
Total Catch			300	492	238	200	465	1186	1050	502	282	192	651	1319	2049	2858	3564	3543	6720	4700	7154	7623	7934	6573	3130	62725
Effort (minutes)			248.3	655.0	279.4	189.7	417.4	836.6	678.5	435.6	432.8	342.9	521.2	1371.4	1199.1	1442.9	1295.1	1227.0	1546.6	1645.2	1806.7	1603.6	1626.8	1499.0	1275.3	22575.9
CPUE (fish/min)			1.21	0.75	0.85	1.05	1.11	1.42	1.55	1.15	0.65	0.56	1.25	0.96	1.71	1.98	2.75	2.89	4.34	2.86	3.96	4.75	4.88	4.39	2.45	2.78

3.3.5.2 Spatial and Temporal Variation in RAMP Data

Figure 105 demonstrates the relationships between fish community indices and proposed predictors of variation. In general, total abundance, species richness, and NMDS axis 2 scores appear to increase with increasing distance downstream, while evenness decreases. There are no apparent changes with distance downstream in diversity and NMDS axis 1 scores. Total abundance, species richness, diversity and NMDS axis 1 and 2 scores increase with increasing effort, whereas evenness decreases with increasing effort. There are no apparent changes in any of the fish indices of community composition and Q60. Finally, total abundance, species richness, diversity, and NMDS axis 1 and 2 scores all increase over time, whereas evenness decreases over time.

Mixed-model GLM's determined that Q60 was a significant predictor of variation in total abundance, evenness, and NMDS axis 1 scores, however the percent of variance explained (%VE) was very low in all three cases <1%. Year was significant for all indices (%VE: 9 to 46%), distance was only a significant predictor of total abundance (%VE = 0.6%), and fishing effort was a significant predictor of all indices except NMDS axis 2 scores (%VE = 1.7 to 14.5%; Table 56).

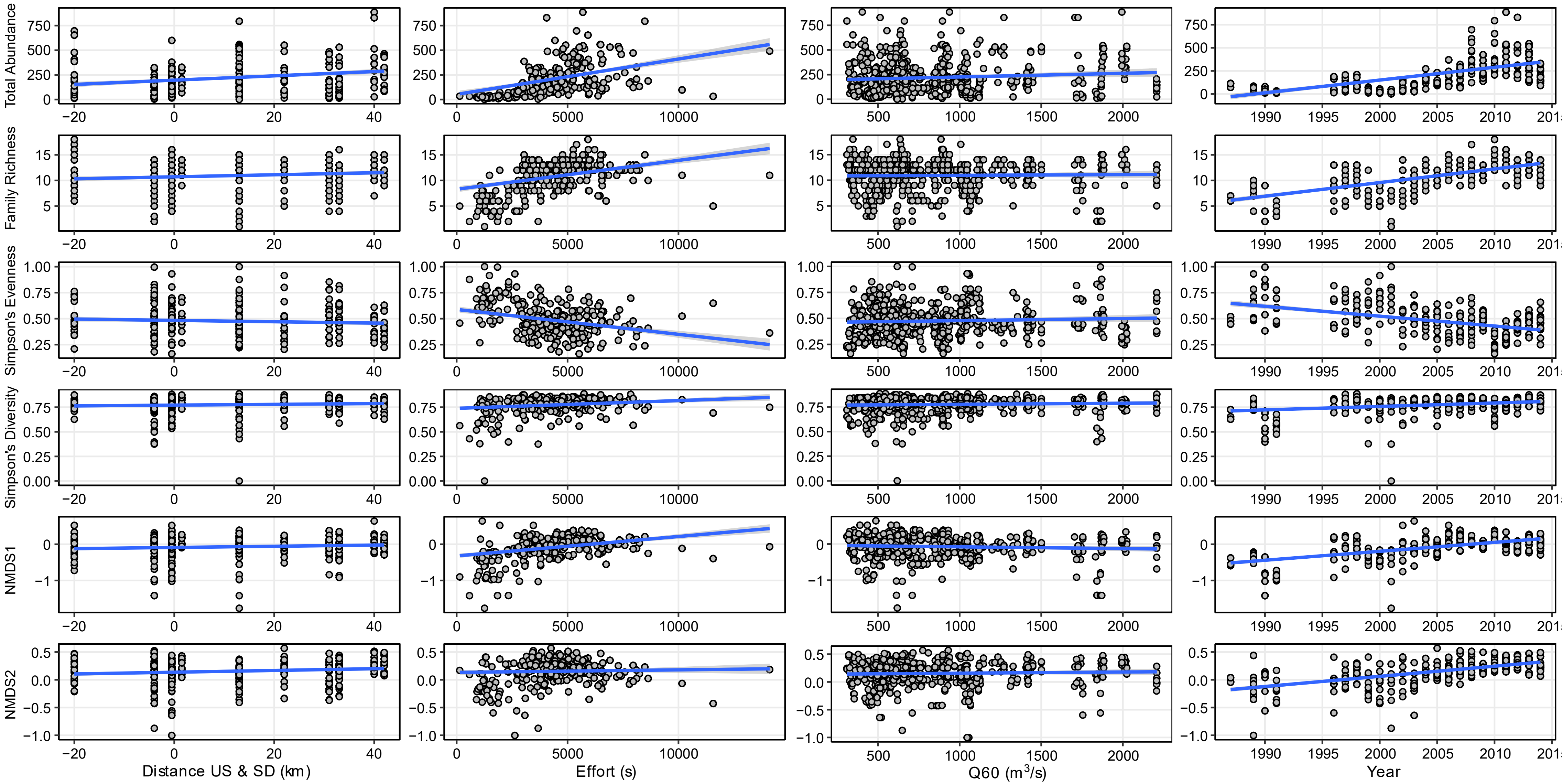

Figure 105 Variations in fish community indices with different predictors (distance, effort, discharge, and sampling year) under RAMP

Table 56 Significance (p-value) and percent of variance explained (%VE) for discharge, year, distance upstream/downstream, and effort as predictors of fish indices of community composition in the Lower Athabasca River, under RAMP

Indice	Q60 (m3/s)		Year		Distance (US/DS)		Effort (sec)	
	P-Val	%VE	P-Val	%VE	P-Val	%VE	P-Val	%VE
Abundance	**0.007**	0.5	**<0.001**	45.8	**0.002**	0.6	**<0.001**	14.5
Species Richness	0.605	<0.1	**<0.001**	44.3	0.753	0.0	**<0.001**	7.6
Simpson's Evenness	**0.025**	0.6	**<0.001**	22.6	0.700	0.0	**<0.001**	6.3
Simpson's Diversity	0.047	0.6	**<0.001**	9.0	0.525	0.1	**0.001**	1.7
NMDS1	**0.032**	0.5	**<0.001**	36.0	0.532	0.0	**<0.001**	6.8
NMDS2	0.227	0.2	**<0.001**	36.3	0.747	0.0	0.157	0.2

3.3.5.3 Normal Ranges

An example of the predicted normal ranges for fish indices of community composition generated based on RAMP data as described in Section 3.2.5 can be found in Table 57 with an illustration of the model performance in Figure 106. The models for fish community indices have retained all the components for discharge, Year, distance, and effort. Table 57 provides a forecasted normal range scenario, where predicted normal range values are provided for the year 2022, where Q60 is 620 m3/s, the distance downstream is 13 km (i.e., Site 10B), and the amount of effort is 1098 seconds. Using total fish abundance as an example, the model predicts the normal range (i.e., baseline mean ± 2SD) to be 87 (lower level) to 969 (upper level) with a baseline of 291.

Figure 107 summarizes the model predicted value of each fish community indices against the observed values from field samples collected under the EMP. By inspecting the graph, one observes that abundance, Simpson's Diversity and Richness, and NMDS axis 1 and 2 scores all fall along the 1:1 Line, suggesting reasonable model fit. Species richness, however, falls well below the 1:1 line at each EMP site, indicating that the RAMP data does not accurately predict fish species richness along the mainstem of the LAR. The RAMP dataset used to model the EMP data covers a span of 84 km of the LAR whereas the EMP dataset only covers a span of 33 km (Table 54 Comparison of sampling sites in both the EMP and RAMP fish community assessment programs (Table 54). By covering a large span of the LAR, sampling crews would be encountering a more diverse range of habitat types, thus increasing chances of catching more species that may otherwise not be present at the more localized scale of the EMP. Further, the RAMP collected data from 1987 to 2014 whereas EMP has only collected data from 2018 to 2021, therefore the inherent differences in spatial and temporal scale of the two programs support the differences observed in species richness.

Normal range model coefficients are summarized in Table 58 for each indices of fish community. No EMP data exceeded the normal range for total abundance, species richness, Simpson's Diversity, or NMDS axis 2 scores. A total of 21% of EMP data exceeded the predicted normal range for Simpson's Evenness while 29% of EMP data exceeded the predicted normal range for NMDS axis 1 scores. For indicators of community composition, it is also important to consider occurrences where values fell below the modelled normal range. A total of 63% of species richness values in the EMP dataset fell below the modelled normal range, followed by 58% of both NMDS2 scores, 50% of Simpson's Diversity values, 33% of NMDS1 scores, 33% of abundance values, and 0% of Simpson's Evenness values.

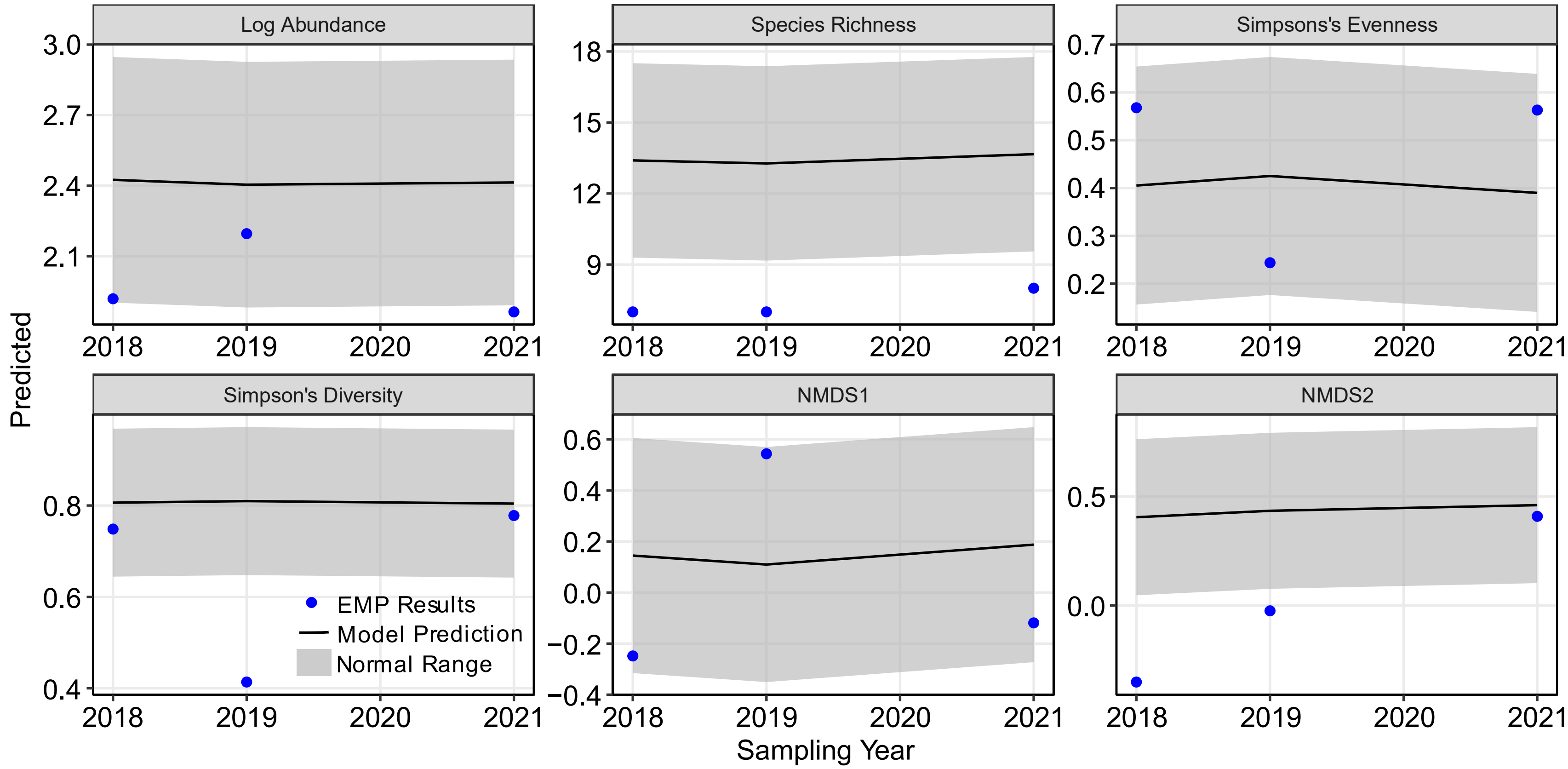

Figure 106 Variations in fish community indices model predictions generated with RAMP data in relation to sampling year overlayed with observed measurements during EMP (2018, 2019, and 2021).

Figure Notes: NR = Normal Range (±2SD); Model predictions from the EMP sampling station located at 12 km downstream are provided as an example.

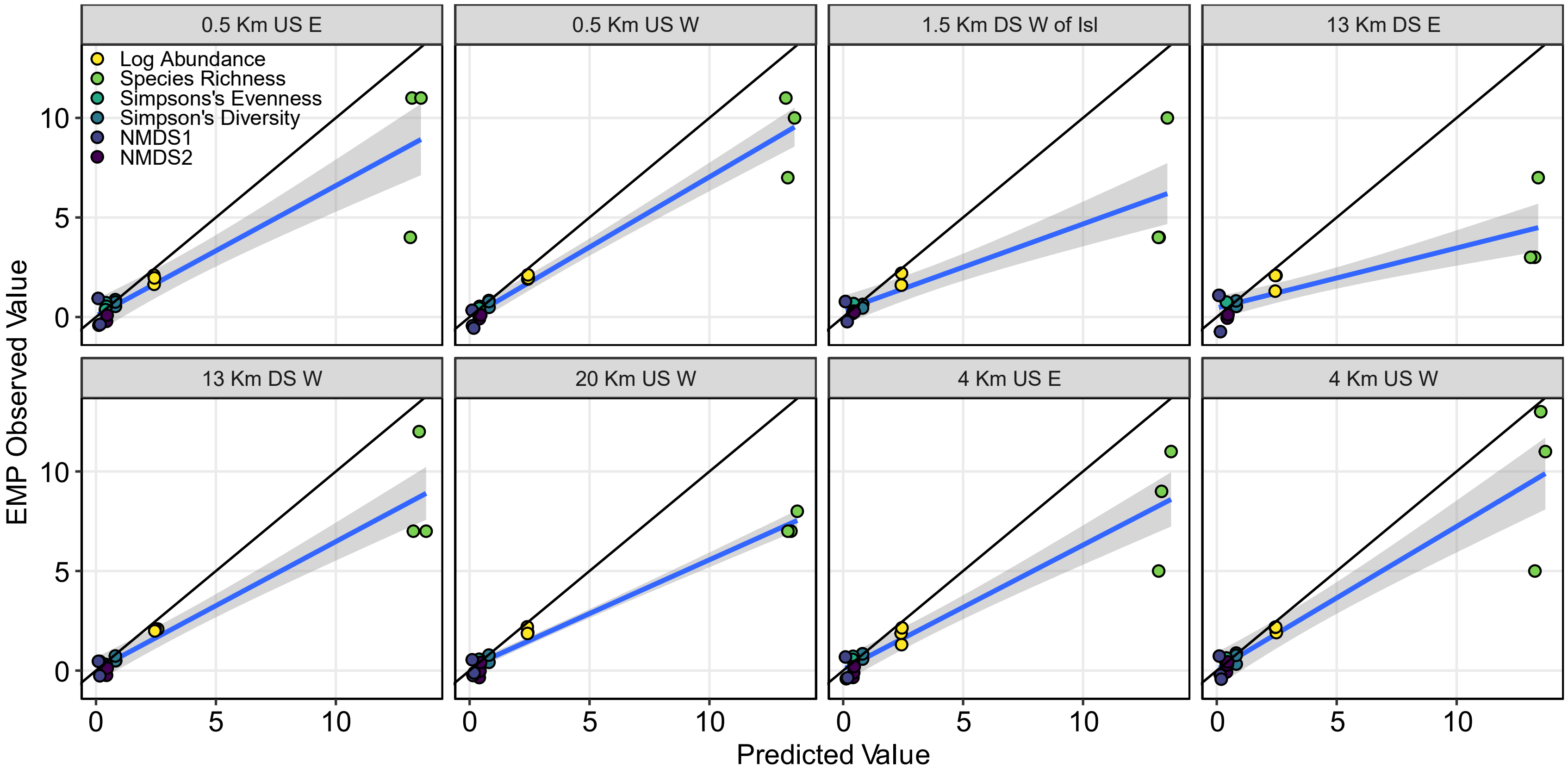

Figure 107 Fish community indices model performance for each of the 6 modelled indices, x-axis represents the actual measurements during EMP, and the y-axis represents normal range model predictions.

Figure Notes: The black line represents a 1:1 line, while the blue line represents the overall goodness of fit at each sampling station

Table 57 Example normal range model output for the different fish community indices under a specific forecasting scenario.

Model Component	Description	Abundance		Species Richness		Simpson's Diversity		Simpson's Evenness		NMDS1		NMDS2	
		Coeff	Scenario	Coeff	Scenario	Coeff	Scenario	Coeff	Scenario	Coeff	Scenario	Coeff	Scenario
Constant	Intercept	-68.48	1	-499.15	1	-5.85	1	18.33	1	-47.22	1	-36.83	1
Discharge (Q60)	Slope for linear relation with Q60	0.09	620	-0.06	620	0.03	620	0.07	620	-0.12	620	0.04	620
Year	Slope for linear trend across years	0.0349	2022	0.2535	2022	0.0032	2022	-0.0090	2022	0.0236	2022	0.0184	2022
Distance (US/DS)	Slope for linear relationship with distance US/DS	0.0010	13	-0.0058	13	0.0001	13	0.0003	13	-0.0008	13	0.0002	13
Effort	Slope for linear relationship with effort	0.00009	1098	0.00043	1098	0.00001	1098	-0.00002	1098	0.00004	1098	-0.00001	1098
Estimate in logarithms/real units			2.46		13.66		0.81		0.39		0.19		0.48
MSE in logarithms		0.068		4.214		0.007		0.016		0.053		0.032	
SD in logarithms		0.261		2.053		0.081		0.125		0.230		0.179	
Estimate in real units			291.04										
Lower limit of normal range in real units			87.41		9.55		0.64		0.14		-0.27		0.13
Upper limit of normal range in real units			969.07		17.77		0.97		0.64		0.65		0.84

Table Notes: Normal Ranges were calculated as the estimate in real units ± 2SD. Data was log-transformed (base 10) where indicated.

Table 58 Model coefficients used for the prediction of fish community indices normal ranges in the EMP dataset and the percentage of NR exceedances (2018, 2019, and 2021).

Indicator	Int	Q60	Year	Distance (US/DS)	Effort	MSE	EMP NR Exceedance (%)		
							< LL	Inside NR	> UL
Log Abundance	-68.5	0.1	0.0349	0.0010	8.5E-05	0.068	33	67	0
Species Richness	-499.1	-0.1	0.2535	-0.0058	4.3E-04	4.214	63	38	0
Simpson's Evenness	18.3	0.1	-0.0090	0.0003	-2.0E-05	0.016	0	79	21
Simpson's Diversity	-5.8	0.0	0.0032	0.0001	5.9E-06	0.007	50	50	0
NMDS1	-47.2	-0.1	0.0236	-0.0008	4.2E-05	0.053	33	38	29
NMDS2	-36.8	0.0	0.0184	0.0002	-5.6E-06	0.032	58	42	0

Table Notes: Normal Ranges were calculated as the estimate in real units ± 2SD. Int = Intercept; Q = average discharge over previous 60 days; Effort = electrofishing effort; LL = Lower Level; UL = Upper Level. Values were log-transformed (base 10) where indicated.

3.3.6 Sentinel Fish Populations Health

3.3.6.1 Data QA/QC

In order to inform on whether the OSMP data can be used to predict EMP data, fish health indictors were compared between the two programs. Both programs collected the same species of fish, White Sucker and Trout-perch, at similar sample sizes over each sampling years (Table 59). Fish were collected at the same time of year and with the same AEPA/ECCC field team. Fish health indicators GSI (%), LSI (%), and K were compared between the two sampling programs for each species across each sampling year. GSI (Figure 108A), LSI (Figure 108B), and K (Figure 108C) are similar between the two sampling programs for Trout-perch, aside from a select few GSI and LSI outliers. The three fish health indicator values are also similar across the two programs for White Sucker (Figure 109).

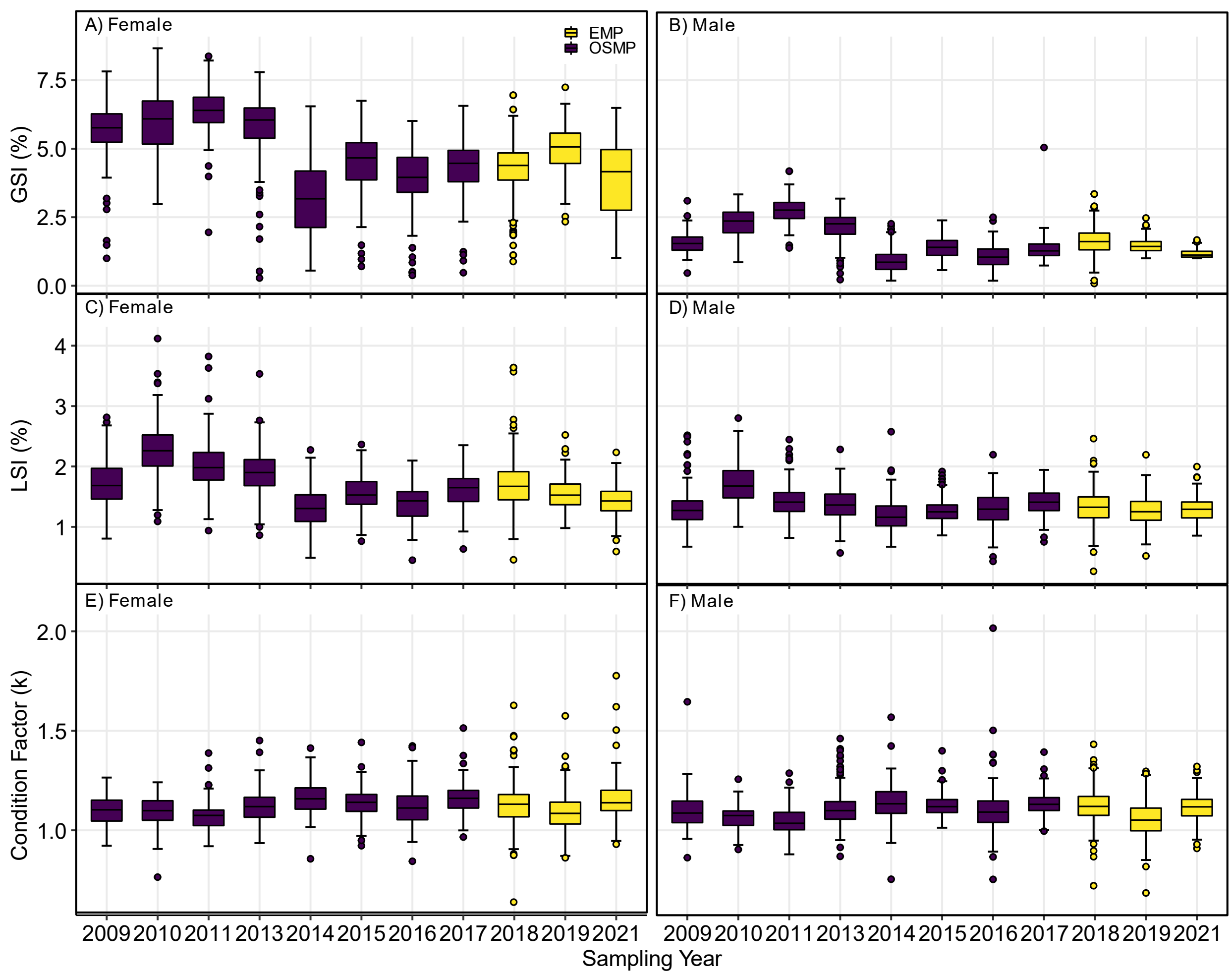

Figure 108 Variation in GSI (A, B), LSI (C, D), and K (E, F) in female and male Trout-perch from both EMP and OSMP, data pooled across all sampling stations.

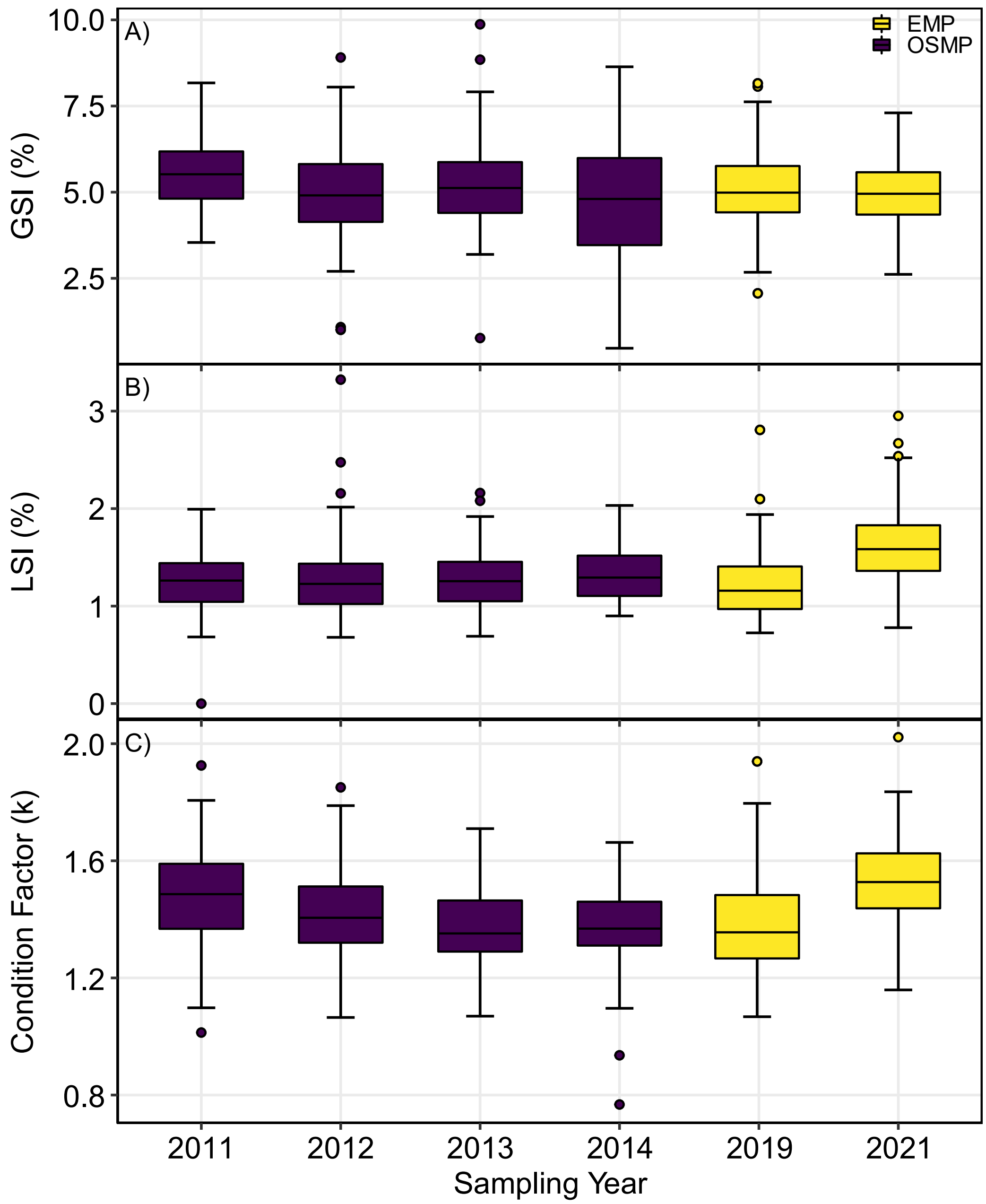

Figure 109 Variation in GSI (A), LSI (B), and K (C) in White Sucker from both EMP and OSMP data pooled across both sexes and all sampling stations.

Table 59 Sample numbers for fish health EMP and OSMP.

Study	Species	Sex	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2021
EMP	TRPR	F	—	—	—	—	—	—	—	—	—	192	210	218
		M	—	—	—	—	—	—	—	—	—	200	204	250
	WHSC	F	—	—	—	—	—	—	—	—	—	—	71	74
		M	—	—	—	—	—	—	—	—	—	—	72	51
OSMP	TRPR	F	104	76	121	—	135	180	200	99	100	—	—	—
		M	101	76	121	—	142	187	200	100	100	—	—	—
	WHSC	F	—	—	77	87	86	34	—	—	—	—	—	—
		M	—	—	79	88	67	22	—	—	—	—	—	—

3.3.6.2 Normal Ranges

Figure 110 demonstrates that discharge is an important predictor of fish health in Trout-perch, where k shows a slight increase with increasing discharge (Figure 110A), while GSI and LSI decrease with increasing discharge (Figure 110BC) and therefore discharge was included as a predictor in the normal ranges models for Trout-perch. For White Suckers, no clear relationships between discharge and fish health indicators were observed (Figure 110DEF) and therefore discharge was not included as a predictor in the normal range models for White Sucker. Kilgour et al (2019a) performed a similar exercise in 2019 and found a general increase in condition factor with discharge (Q60dp) and a decrease in the GSI and LSI with discharge for both female and male trout-perch, which is in agreement with what is being observed in this report.

For Trout-perch, GLM results demonstrated that discharge was a significant predictor for all health indicators in both males and females (i.e., $p < 0.05$; Table 60). Linear trends over time as well as variations in trends over time that depended on discharge (i.e., Year x Q60) were statistically significant for all indicators (Table 60). The only fish health indicator that showed significant variation from upstream to downstream stations was K in female Trout-perch (p-value = 0.028; Table 60), all other indicators in both females and males did not vary significantly with distance.

For White Sucker, GLM results demonstrated that linear trends over time were only statistically significant for K in female White Suckers (p-value = 0.039; Table 60), all other indicators in both female and male White Sucker did not vary significantly with sampling year. Significant spatial variation from upstream to downstream of the LAR was observed for k and LSI in female White Sucker (p=0.006 & 0.012, respectively; Table 60) and LSI in male White Sucker (p-value = 0.014; Table 60).

An example of the predicted normal ranges for GSI in female Trout-perch generated based on OSMP data as described in Section 3.2.6.2 can be found in Table 61 with an illustration of the model performance in Figure 111. The models for fish health indicators in Trout-perch have retained all the components for discharge, Year, distance, and Year x discharge. For GSI in female Trout-perch, the model constant was 11448, while the slope for discharge was -3729, the year term was -5.7, the distance term was 0.0036, and the year x discharge terms was 1.85. The model MSE was 1.675, the square root of which is 1.294. The SD among samples, for any modeled scenario is therefore 1.294 (Table 61). The table provides three scenarios for which we desire an estimate of the normal range for GSI in female Trout-perch. All three scenarios attempt to predict normal ranges in 2022 at the station that is located 12km downstream of the potential OSPW release point under three different discharge conditions (i.e., 100, 600, and 1200 m^3/s).

Figure 112 summarizes the model predicted value of each fish health indicator against the observed values from field samples collected under the EMP for Trout-perch. The model performance can be assessed based the deviation of the points from the 1:1 line (i.e., a perfect fit). While the data points at each sampling station do tend to fall on the 1:1 line, there are instances of variation within the experimental results that are not captured by the predictive model. For example, GSI values tend to vary more widely along the observed data (y-axis in Figure 112) reflective of the variability in the field collected data that is not capture by the model.

An example of the predicted normal ranges for GSI in female White Sucker generated based on OSMP data can be found in Table 61 with an illustration of the model performance in Figure 113. The models for fish health indicators in White Sucker have only retained all the components for Year and distance as predictors as there was no evident relationship between health indicators and discharge in White Sucker (Figure 110). For GSI in female White Sucker (as an example), the model constant was 64.4, while the slope for year was -0.03 and the distance term was 0.0071. The model MSE was 2.23, the square root of which is 1.493. The SD among samples, for any modeled scenario is therefore 1.493 (Table 62). The table provides three scenarios for which we desire an estimate of the normal range for GSI in female White Sucker. The scenarios attempt to predict normal ranges in sampling years: 2022, 2023, and 2024 and at sampling stations located 12, 4, and 0.5 km downstream of the potential OSPW release, resistively.

Normal range model coefficients and exceedances in the EMP data are summarized in Table 63 for each individual fish health indicators across each species and sex. Overall, across all indicators, species, sexes, sampling stations, and sampling years, between 0 and 22 % of EMP samples fell below the normal range lower limit, while between 0 and 29 % of EMP samples fell above the normal range upper limit. The rest of the EMP samples remained within the normal ranges.

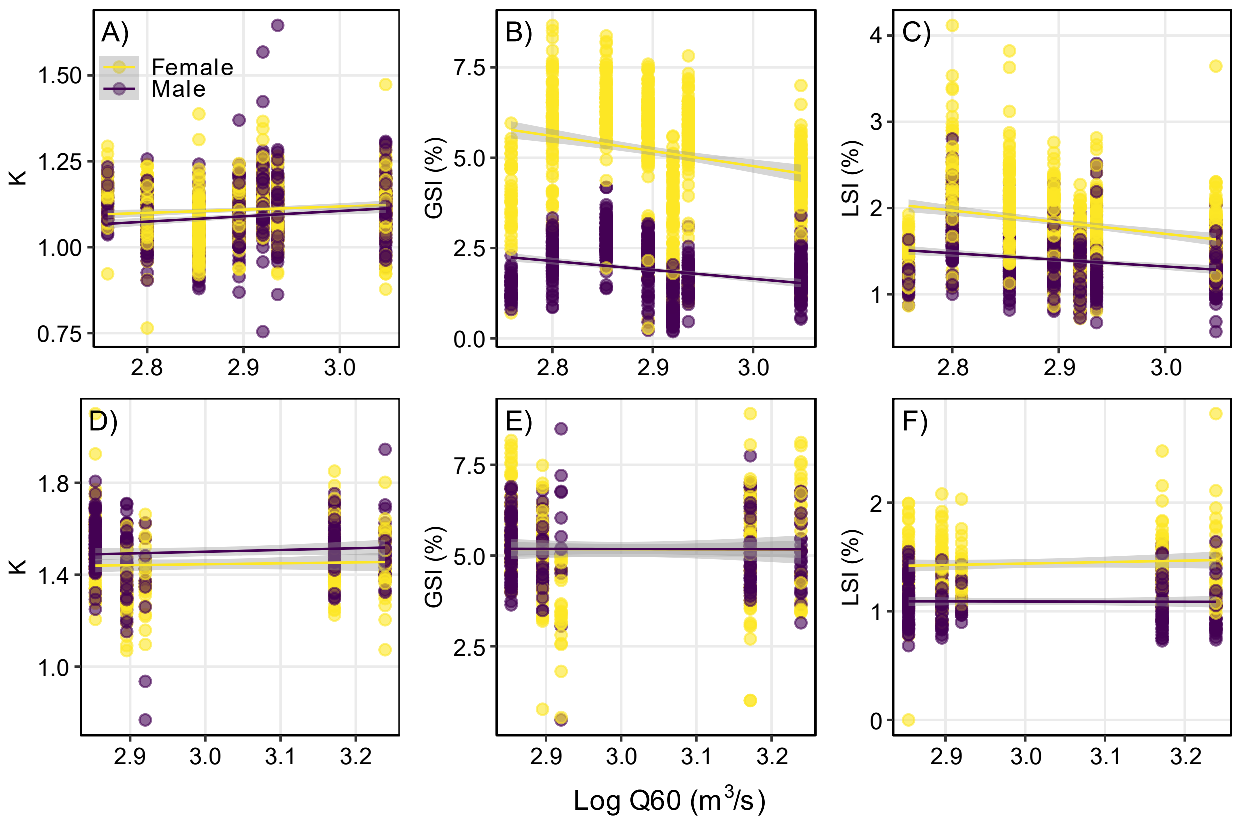

Figure 110 Relationship between fish health indicators and Q60 among Trout-perch (A, B, C) and White Sucker (D, E, F) from EMP (2018, 2019, and 2021).

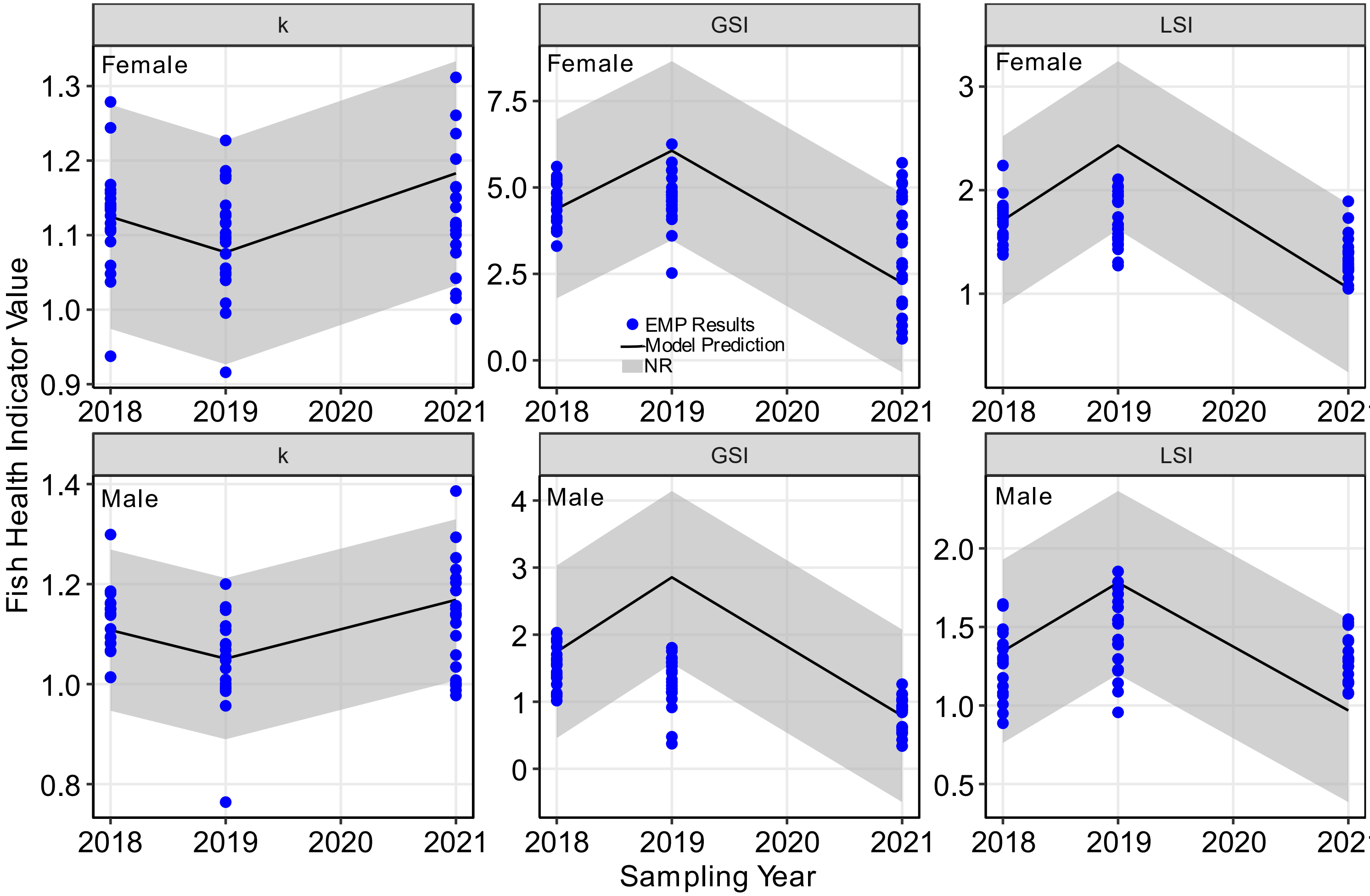

Figure 111 Variations in female and male Trout-perch health indicator model predictions built with OSMP data in relation to sampling year compared to observed measurements measured during EMP (2018, 2019, and 2021).

Figure Notes: NR = Normal Range (±2SD); Model predictions from the EMP sampling station located at 12 km downstream are provided as an example.

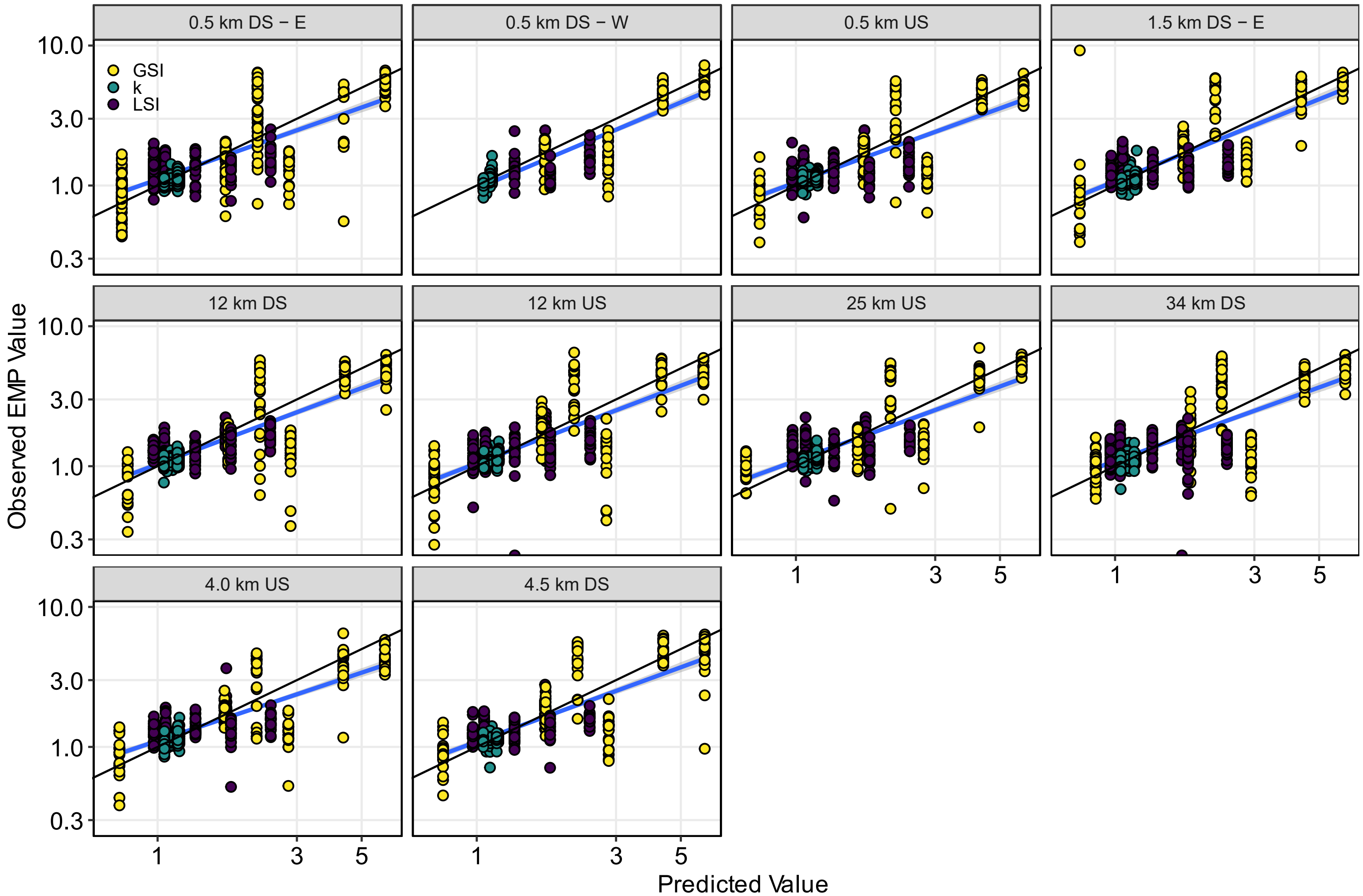

Figure 112 Trout-perch fish health indicator model performance, the y-axis represents the actual measurements during EMP, and the x-axis represents normal range model predictions.

Figure Notes: The black line represents a 1:1 line, while the blue line represents the overall goodness of fit at each sampling station. SD = Standard Deviation, W = West of island, and E = East of Island

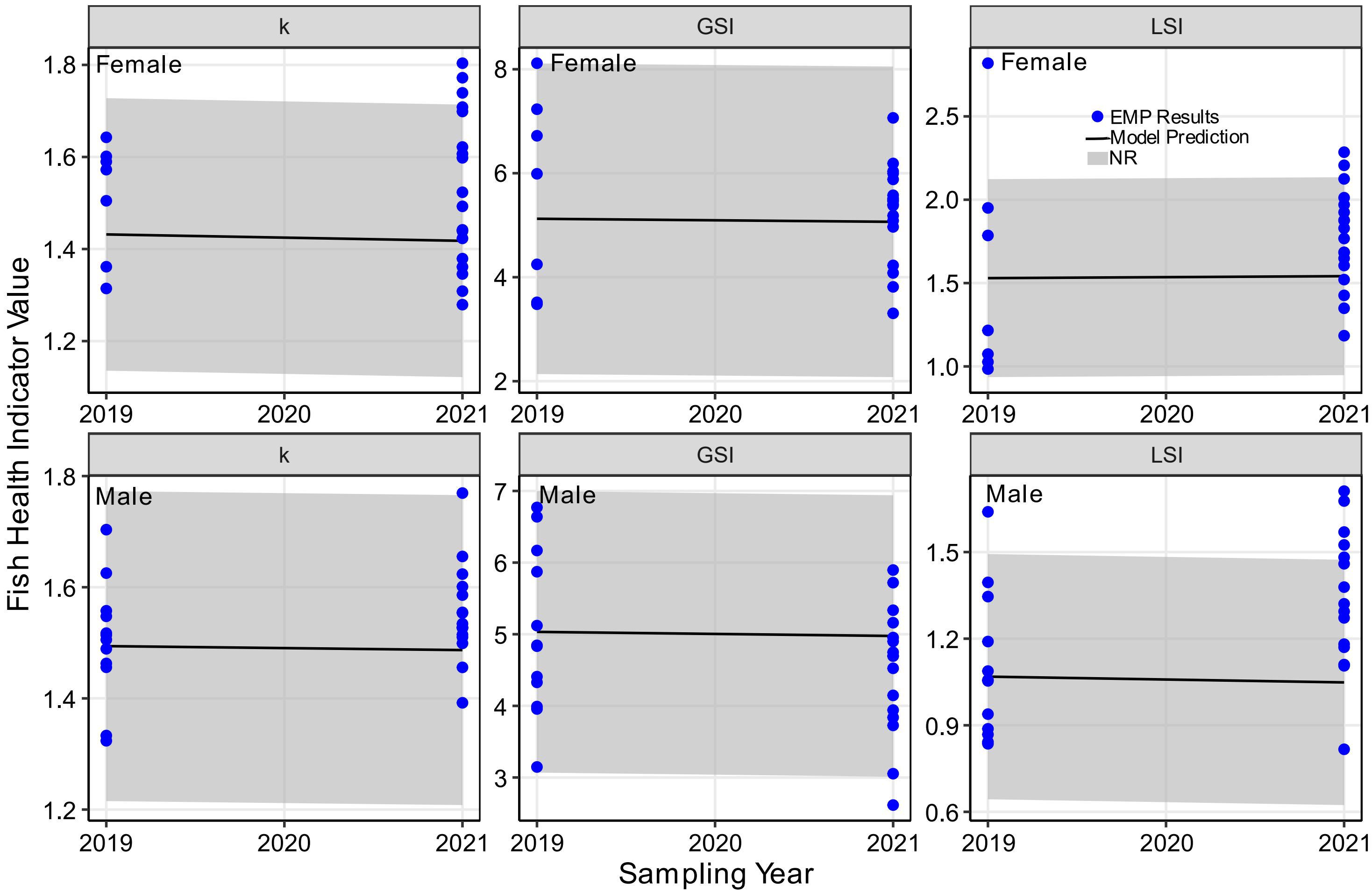

Figure 113 Variations in female and male White Sucker health indicator model predictions built with OSMP data in relation to sampling year compared to observed measurements measured during EMP (2018, 2019, and 2021).

Figure Notes: NR = Normal Range (±2SD); Model predictions from the EMP sampling station located at 12 km downstream are provided as an example.

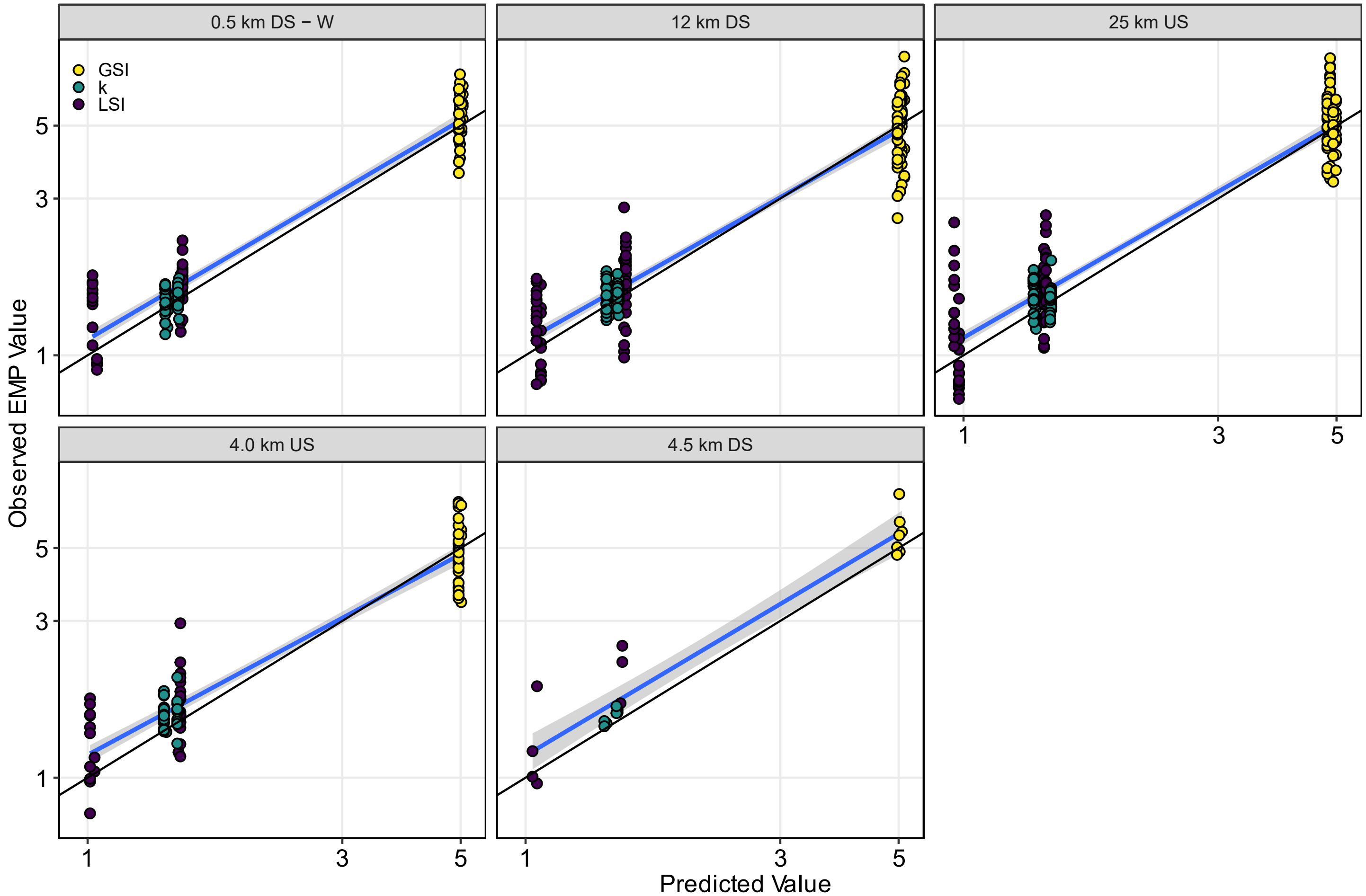

Figure 114 White Sucker fish health indicator model performance, the y-axis represents the actual measurements during EMP, and the x-axis represents normal range model predictions.

Figure Notes: The black line represents a 1:1 line, while the blue line represents the overall goodness of fit at each sampling station. SD = Standard Deviation, W = West of island, and E = East of Island

Table 60 Significance (p-value) and percent of variance explained (%VE) for Q60, year, distance upstream/downstream, and year x Q60 as predictors of fish health indicators in the Lower Athabasca River, under OSMP 2009 to 2018.

Species	Sex	Indicator	Q60 (m^3/s)		Year		Distance (US/DS)		Q60 x Year	
			P-value	%VE	P-value	%VE	P-value	%VE	P-value	%VE
TRPR	F	k	**0.013**	1.0	**<0.001**	3.6	**0.026**	0.8	**<0.001**	2.0
		GSI	**<0.001**	5.2	**<0.001**	14.9	0.381	0.1	**<0.001**	7.2
		LSI	**<0.001**	5.6	**<0.001**	3.0	**0.006**	0.9	**<0.001**	17.1
	M	k	**<0.001**	2.3	**0.0003**	2.1	0.091	0.4	**<0.001**	3.7
		GSI	**<0.001**	7.2	**<0.001**	1.6	0.075	0.4	**<0.001**	18.5
		LSI	**<0.001**	3.9	**0.002**	1.4	0.076	0.4	**<0.001**	13.4
WHSC	F	k	—	—	0.057	1.7	**0.005**	3.9	—	—
		GSI			0.426	0.3	0.251	0.7		
		LSI			0.660	0.1	**0.012**	3.1		
	M	k			0.313	0.5	0.141	1.1		
		GSI			0.285	0.6	0.771	0.0		
		LSI			0.067	1.7	**0.014**	3.0		

Table Notes: Shaded cells represent percent variance explained by each predictor when the p-value is significant (i.e., $p < 0.05$)

Table 61 Example calculation of predicted normal ranges using the model for GSI in female Trout-perch (OSMP 2009 to 2018) under different scenarios.

Model Component	Description	Coefficient	Scenario		
			1	2	3
Constant	Intercept	11448.2953	1	1	1
Q60	Slope for linear relation with Q	-3729.6995	600	900	1200
Year	Slope for linear trend across years	-5.6817	2022	2022	2022
Distance (US/DS)	Slope for linear relationship with distance US/DS	0.0036	12	12	12
Year x Q60	Term accounting for the different slope for year effect, depending on Q	1.8518	5617	5973	6226
MSE		1.675	—		
SD		1.294	—		
GSI Estimate			0.31	2.87	4.69
Lower limit of normal range in real units			-2.28	0.28	2.10
Upper limit of normal range in real units			2.90	5.46	7.28

Table Notes: Normal Ranges were calculated as the estimate in real units ± 2SD.

Table 62 Example calculation of predicted normal ranges using the model for GSI in female White Sucker under different scenarios.

Model Component	Description	Coefficient	Scenario		
			1	2	3
Constant	Intercept	64.4368	1	1	1
Year	Slope for linear trend across years	-0.0294	2022	2023	2024
Distance (US/DS)	Slope for linear relationship with distance US/DS	0.0071	12	4	0.5
MSE		2.231			
SD		1.493			
GSI Estimate			5.04	4.95	4.90
Lower limit of normal range in real units			2.05	1.96	1.91
Upper limit of normal range in real units			8.02	7.94	7.88

Table Notes: Normal Ranges were calculated as the estimate in real units ± 2SD.

Table 63 Resulting models for fish health indicators and a summary of the normal range exceedances when compared to the EMP data (2018, 2019, and 2021).

Species	Sex	Indicator	Int	log Q	Year	Distance (US/DS)	log Q x Year	MSE	Emp NR Exceedances (%)		
									< LL	Inside NR	> UL
TRPR	F	K	-308.3	101.1	0.15	-0.0003	-0.05	0.006	3.4	91.5	5.1
		GSI	11448.3	-3729.7	-5.68	0.0036	1.85	1.675	1.0	87.9	11.1
		LSI	5357.0	-1797.5	-2.66	-0.0007	0.89	0.165	21.8	76.9	1.3
	M	K	-447.0	148.9	0.22	-0.0003	-0.07	0.006	3.8	93.4	2.9
		GSI	8911.6	-3000.5	-4.42	0.0031	1.49	0.414	25.0	74.8	0.2
		LSI	3254.3	-1095.0	-1.61	-0.0002	0.54	0.085	15.0	83.0	2.0
WHSC	F	K	15.5	—	-0.01	0.0017	—	0.022	0.0	83.7	16.3
		GSI	64.4		-0.03	0.0071		2.231	0.0	98.1	1.9
		LSI	-10.4		0.01	0.0031		0.088	0.0	87.5	12.5
	M	K	8.7		0.00	0.0009		0.019	0.0	96.5	3.5
		GSI	62.3		-0.03	0.0013		0.965	1.2	98.8	0.0
		LSI	21.2		-0.01	0.0024		0.045	0.0	70.9	29.1

Table Notes: Normal Ranges were calculated for Trout-perch (TRPR) and White Sucker (WHSC) as the estimate in real units ± 2SD**,** LL and UL represent the upper and lower level, respectively, of the normal range. Data was log transformed (base 10) where indicated.

3.3.7 Fish Body and Tissue Burdens

3.3.7.1 Data QA/QC

To develop fish body burden normal ranges, we rely on the development of a predictive model trained on the regional data set and applied to the enhanced dataset that incorporates temporal and spatial differences. Comparison of EMP and OSMP fish body burden data suggested that there is reasonable overlap in terms of analytes analyzed, however differences in sample numbers at the station and year scale, as well as the types of fish tissues analyzed, between the two programs suggest that the OSMP dataset cannot be used to accurately predict the EMP data in a similar fashion as it was used in sections 3.3.1 and 3.3.1.2.2.

There are a variety of discrepancies and differences between the two datasets highlighted in Table 64 and Table 65. First, to compare the datasets, we must be comparing the same type of sample matrix. The OSMP analyzed carcass, liver, and muscle tissues across sampled species, whereas the EMP analyzed only muscle and whole-body tissues. The only area of overlap between the two programs is in muscle tissues for both male and female Walleye and female White Sucker (Table Table 64). Secondly, we assume the sampling station (i.e., distance upstream or downstream from the proposed OSPW discharge point) to have been an important predictor of fish body burden data. To inform a meaningful relationship between distance upstream and/or downstream, we require a minimum of 3 data points at the spatial scale to carry out a linear regression analysis with body burden data. As shown in Table 65, there are two instances in the OSMP dataset where less than 3 stations were sampled (metals in male Walleye and female White Sucker), and therefore distance cannot be used as a numerical predictor.

Normal ranges have therefore been developed for the Trout-perch, Walleye, and White Sucker based on the EMP data only, primarily due to the increased number of stations sampled. No OSMP data was considered in the development of normal ranges. Further, to produce accurate predictions, we sufficient sample size is required when determining linear temporal and spatial trends. We therefore only include a year and/or distance predictor in our model if there are at least 3 data points associated with each predictor.

Table 66 summarizes the number of sampling years and stations for each analyte measured in each fish species at the sex and tissue level. Based on these summary counts, the following four predictive models were utilized:

1. Conc = FL + Year + Distance (US/DS) when n>3 years and n>3 stations
2. Conc = FL + Year when n>3 years and n<3 stations
3. Conc = FL + Distance (US/DS) when n<3 years and n>3 stations
4. Conc = FL when n<3 years and n<3 stations

Table 64 Comparison of fish body burden sample matrix data between OSMP and EMP.

Species	Sex	Sample Matrix	OSMP	EMP
Trout-perch	M	Whole Body		x
		Carcass		
		Liver		
		Muscle		x
	F	Whole Body		x
		Carcass	x	
		Liver		
		Muscle		x
Longnose Sucker	M	Whole Body		
		Carcass		
		Liver	x	
		Muscle		
	F	Whole Body		
		Carcass		
		Liver	x	
		Muscle	x	
Walleye	M	Whole Body		
		Carcass		
		Liver	x	
		Muscle	x	x
	F	Whole Body		
		Carcass		
		Liver	x	
		Muscle	x	x
White Sucker	M	Whole Body		
		Carcass		
		Liver	x	
		Muscle		x
	F	Whole Body		
		Carcass		
		Liver	x	
		Muscle	x	X

Table Notes: Cells in bold represent similarities between OSMP and EMP datasets that can be used for future comparison.

Table 65 Comparison of sampling stations by parameter category included in the fish body burden data (muscle tissue only) between the OSMP and EMP.

Species	Sex	Parameter Category	# Of Sampling Stations	
			OSMP	EMP
Walleye	F	Hydrocarbons, PAHs	4	5
	F	Metals	-	5
	M	Hydrocarbons, PAHs	4	6
	M	Metals	2	6
White Sucker	F	Hydrocarbons, PAHs	2	5
	F	Metals	3	5

Table Notes: Only meaningful comparisons as determined in Table 64 are included

Table 66 Number of sampling years (# Y) and sampling stations (# S) included in the EMP fish body burden dataset.

Analyte	Trout Perch								Walleye				White Sucker			
	Whole Body				Muscle				Muscle							
	Female		Male		Female		Male		Female		Male		Female		Male	
	# Y	# S	# Y	# S	# Y	# S	# Y	# S	# Y	# S	# Y	# S	# Y	# S	# Y	# S
Aluminum	1	2	3	9	-	-	-	-	2	3	2	5	2	4	2	5
Antimony	1	1	3	9	-	-	-	-	-	-	-	-	-	-	-	-
Arsenic	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Barium	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Beryllium	1	1	3	3	-	-	-	-	-	-	-	-	-	-	-	-
Bismuth	1	1	3	9	-	-	-	-	2	5	2	5	1	3	2	5
Boron	1	1	1	1	-	-	-	-	-	-	-	-	-	-	-	-
Cadmium	1	2	3	9	-	-	-	-	-	-	1	1	-	-	-	-
Calcium	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Chromium	1	1	3	9	-	-	-	-	2	2	2	4	2	2	1	1
Cobalt	1	2	3	9	-	-	-	-	2	4	2	5	2	5	2	5
Copper	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Iron	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Lead	1	2	3	9	-	-	-	-	2	2	2	4	2	2	2	3
Magnesium	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Manganese	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Total Mercury	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Methyl Mercury	-	-	2	9	-	-	-	-	1	1	1	5	1	4	1	5
Molybdenum	1	2	3	9	-	-	-	-	-	-	-	-	1	1	-	-
Nickel	1	2	3	9	-	-	-	-	1	1	2	4	1	3	1	1
Phosphorus	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Potassium	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Selenium	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Silver	1	2	3	9	-	-	-	-	-	-	-	-	-	-	-	-
Sodium	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5

Analyte	Trout Perch								Walleye				White Sucker			
	Whole Body				Muscle				Muscle							
	Female		Male		Female		Male		Female		Male		Female		Male	
	# Y	# S	# Y	# S	# Y	# S	# Y	# S	# Y	# S	# Y	# S	# Y	# S	# Y	# S
Strontium	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Thallium	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
Tin	1	2	2	9	-	-	-	-	-	-	1	1	-	-	-	-
Titanium	1	2	3	9	-	-	-	-	2	5	2	5	2	4	2	4
Uranium	1	2	3	9	-	-	-	-	-	-	1	3	1	1	1	2
Vanadium	1	2	3	9	-	-	-	-	-	-	-	-	-	-	-	-
Zinc	1	2	3	9	-	-	-	-	2	5	2	5	2	5	2	5
$\delta^{15}N$	-	-	-	-	1	2	3	9	2	5	2	5	2	5	2	5
Total PAHs	-	-	-	-	-	-	-	-	2	5	2	5	1	5	1	5

3.3.7.2 Normal Ranges

The model used to generate normal ranges for fish body burden was developed with Q60, fork length, sampling year, and distance upstream/downstream from the proposed OSPW discharge point as predictors. The relationship between the different analytes and these predictors in the EMP dataset has been previously discussed and presented in section 2.3.9.2 (Table 34). In summary, for Trout-perch, the models determined that discharge was a significant predictor (p-value < 0.05) of variations in concentrations of EROD and total PAH in female whole-body samples, $\delta^{15}N$ in male muscle tissues, as well as EROD, MeHg, Hg, and Se in male whole-body samples. Fork length was a significant predictor of variations of $\delta^{15}N$ in female muscle tissues, EROD, ΣPAH4, and total PAH levels in female whole-body samples, $\delta^{15}N$ and $\delta^{13}C$ in male muscle tissues, and EROD, MeHg, Hg, and Se in male whole-body samples. Year and/or the interaction between year and discharge was a significant predictor of variations of EROD and Total PAH levels in female whole-body samples, $\delta^{15}N$ in male muscle tissues, and EROD and Se in male whole-body samples. Finally, distance from the proposed OSPW discharge point was a significant predictor of EROD and total PAH levels in female whole-body samples, $\delta^{15}N$ in male muscle tissues, and EROD in male whole-body samples. For Walleye, discharge was a significant predictor (p-value < 0.05) of variations of $\delta^{13}C$, Hg, and total PAH in female muscle samples and $\delta^{13}C$ and Hg in male muscle samples. Fork length was a significant predictor of Hg in female muscle samples and $\delta^{15}N$ and Hg in male muscle samples. Finally, distance from the proposed OSPW discharge point was not a significant predictor for any of the focused subset of compounds. Year and the interaction between year and discharge was not included in the models for Walleye since sampling did not occur over a minimum of three years. For White Sucker, discharge was a significant predictor of variation of $\delta^{13}C$ and ΣPAH4 levels in female muscle samples and was not significant for any of the focused subset of compounds for male muscles samples. Fork length was a significant predictor of $\delta^{15}N$, MeHg, and Hg in female muscle samples and $\delta^{15}N$, MeHg, Hg, and Se in male muscle samples. Finally, distance from the proposed OSPW discharge point was a significant predictor for total PAH levels in female muscles samples only. Year and the interaction between year and discharge was not included in the models for Walleye since sampling did not occur over a minimum of three years.

An example of the predicted normal ranges for total mercury in male and female Trout-perch, Walleye, and White Sucker collected from the EMP station located 12 km downstream, generated based on EMP data as described in Section 3.2.7.2 in Figure 115. Figure 116 summarizes the model predicted for each fish species against the observed values from field samples collected under the EMP for Trout-perch, Walleye, and White Sucker. The model performance can be assessed based on the deviation of the points from the 1:1 line (i.e., a perfect fit). While the data points at each sampling station do tend to fall on the 1:1 line, there are instances of variation within the experimental results that are not captured by the predictive model. This is primarily observed in Trout-perch, where a likely explanation for the deviation of the model is the lack of recorded fork length data in the EMP dataset, which does not allow the use of fork length as a predictor for certain analytes.

The model coefficients used for the prediction of fish body burden normal range for total mercury in male Trout-perch is provided in Table 67. For total mercury in male Trout-perch (as an example), the model constant was 3734, the Q60 term was -1280, the fork length term was 0.5424, the year term was -1.8511, the distance term was 0.0002, and the interaction term (Q60 x Year) was 0.6345. The model MSE was 0.014, the square root of which is 0.117. The SD among samples, for any modeled scenario is therefore

0.117. The full set of model coefficients used for the prediction of fish body burden normal range for a subset of compounds of interest is provided in Table 67.

Normal ranges were computed as the model predicted average value ± 2 SDs. In this context, the normal range was anticipated to capture ~ 95% of potential future observations (Kilgour et al., 1998a). Normal range model coefficients and exceedances in the EMP data are summarized in Table 58 for each species, analyte, sex, and tissue sample type. In Trout-perch muscle samples, 100% of female samples fell within the normal range, for each analyte, while male samples ranged between 96 to 100 % of EMP samples falling within the normal ranges across all analytes. For Trout-perch whole body samples, between 94 and 100 % of female samples fell within the normal range, while male samples ranged between 95 to 96 % of EMP samples falling within the normal range. For Walleye muscle samples, between 96 and 100 % of both female and male samples fell within the normal range. Finally, for White Sucker muscle samples, between 77 and 100 % of female samples fell within the normal range, while between 94 and 100 % of male samples fell within the normal range.

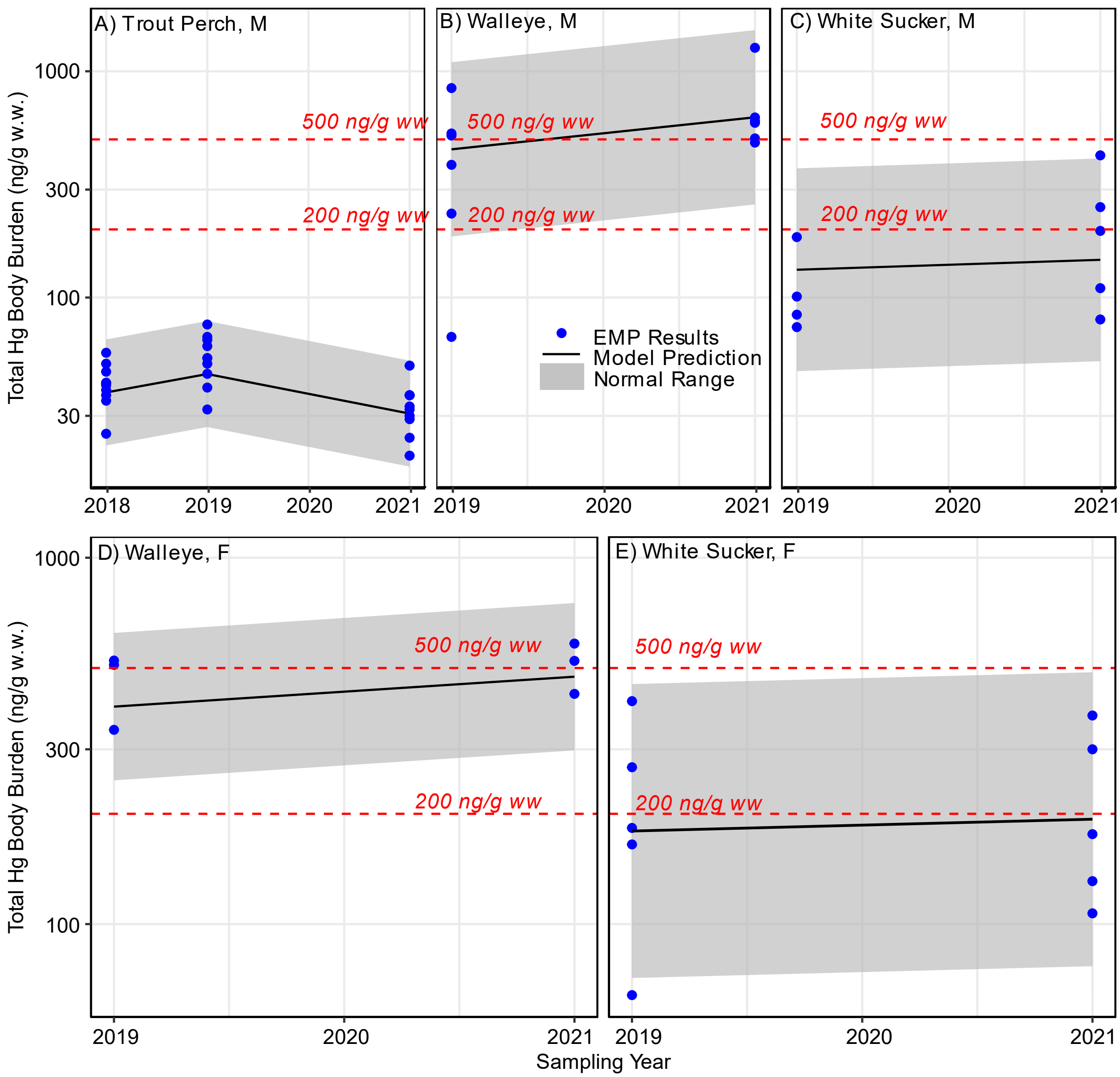

Figure 115 Variations in total mercury body burden model generated with EMP data in relation to sampling year overlayed with observed measurements during EMP (2018, 2019, and 2021).

Figure Notes: Normal Range (±2SD); Example data are presented from the EMP station located 12 km downstream.

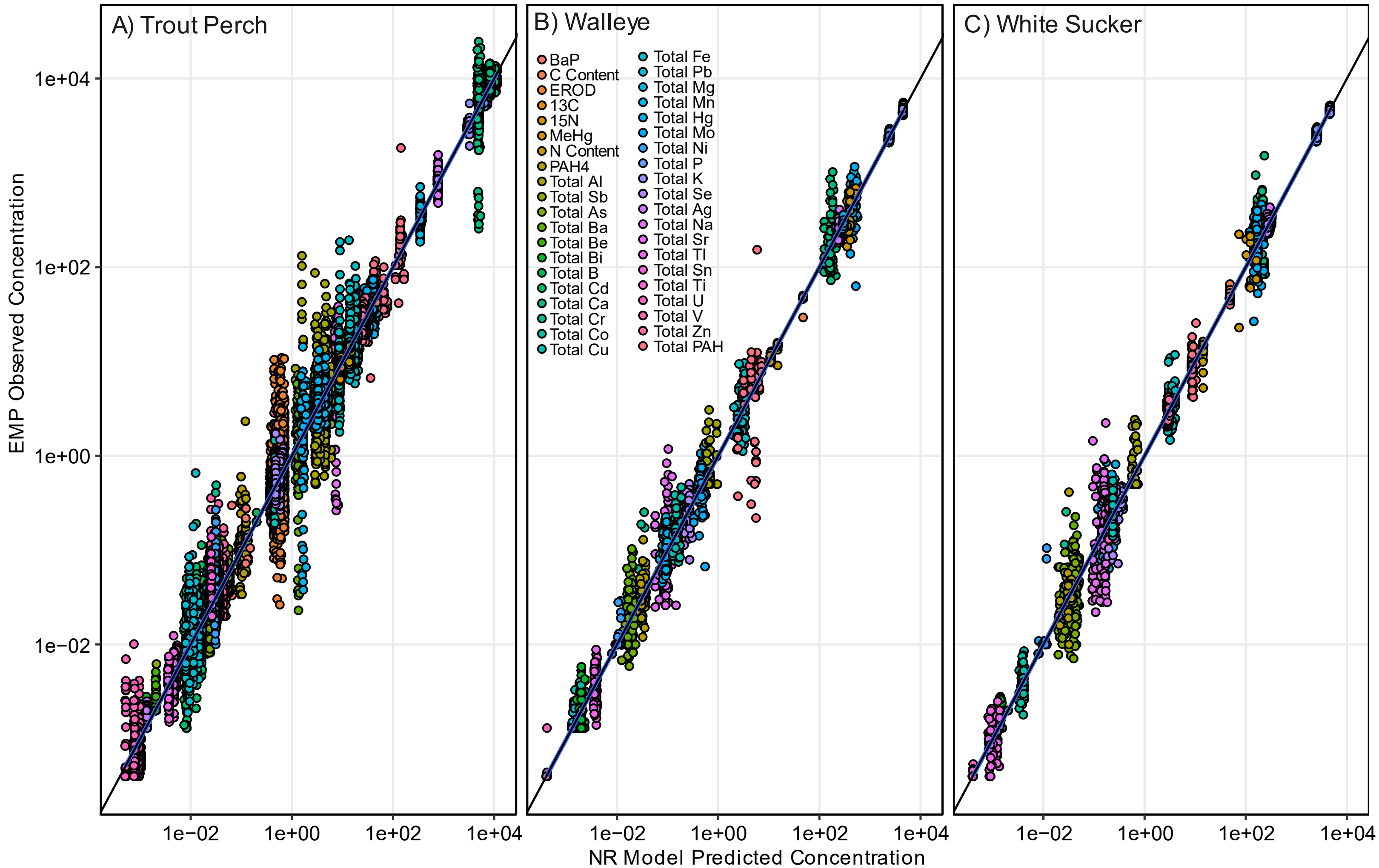

Figure 116 Trout-perch, Walleye, and White Sucker body burden model performance for each analyte, x-axis represents the actual measurements during EMP, and the y-axis represents normal range model predictions.

Figure Notes: The black line represents a 1:1 line, while the blue line represents the overall goodness of fit of the data.

Table 67 Example normal range model output for total mercury in male Trout-perch under different temporal and spatial scenarios.

Model Component	Description	Coefficient	Scenario		
			1	2	3
Constant	Intercept	3734.6502	1	1	1
Q60	Slope for linear relation to Q60	-1280.8899	600	600	600
logFL	Slope for linear relation to FL	0.5424	50	60	70
Year	Slope for linear trend across years	-1.8511	2022	2022	2022
Distance (US/DS)	Slope for linear relationship with distance US/DS	0.0002	12	12	12
Q60xYear	Interaction between Q60 and Year	0.6345	5617	5617	5617
Tot. Hg estimate in log units			-1.72	-1.68	-1.64
MSE in logarithms		0.014			
SD in logarithms		0.117			
Lower limit of normal range in logarithms			-1.96	-1.91	-1.88
Upper limit of normal range in logarithms			-1.49	-1.44	-1.41
Tot. Hg estimate in real units			19.0	20.9	22.8
Lower limit of normal range in real units (ng/g)			11.06	12.21	13.27
Upper limit of normal range in real units (ng/g)			32.53	35.91	39.04

Table 68 Model coefficients used for the prediction of body burden normal ranges in the EMP dataset and the percentage of NR exceedances (2018, 2019, and 2021).

Species	Sex	Matrix	Analyte	Int	log Q60	log FL	Year	Distance (US/DS)	log Q60 x Year	MSE	Emp NR Exceedances (%)		
											<LL	Inside NR	> UL
Trout Perch	F	Muscle	$\delta^{13}C$	5.47	-1.42	-0.0166	-	-	-	0.0002	0	100	0
			$\delta^{15}N$	-1.95	0.94	0.1515	-	-	-	0.0003	0	100	0
		Whole Body	BaP	25929.05	-8705.08	-	-12.8389	-0.0021	4.3102972	0.1412	0	100	0
			EROD	-4783.74	1734.20	-0.4104	2.3721	0.0021	-0.859795	0.1029	1.5	94.4	4.1
			ΣPAH4	24825.70	-8583.27	-	-12.2935	0.0011	4.2503055	0.0627	0.8	94.1	5.1
			Total Hg	-46.12	15.62	0.3980	-	-	-	0.0142	0.0	100	0.0
			Total PAH	21074.90	-7332.94	-	-10.4335	-0.0035	3.6307342	0.0384	1.7	97.5	0.8
			Total Se	11.33	-4.11	-0.0437	-	-	-	0.0079	0.0	100	0.0
	M	Muscle	$\delta^{13}C$	-106.71	38.06	0.0580	0.0535	0.0000	-0.018843	0.0001	0.0	100.0	0.0
			$\delta^{15}N$	-2186.09	759.89	0.1151	1.0830	-0.0004	-0.376344	0.0011	2.3	96.0	1.7
			EROD	-17967.51	6290.41	-0.6784	8.9004	0.0011	-3.115875	0.0987	0.9	94.7	4.3
			MeHg	-0.86	0.25	0.8895	-	0.0005	-	0.0151	2.0	96.0	2.0
			Total Hg	3734.65	-1280.89	0.5424	-1.8511	0.0002	0.6344859	0.0137	2.0	96.0	2.0
			Total Se	-1212.56	459.61	-0.2118	0.6008	-0.0001	-0.227709	0.0119	1.3	96.0	2.7
Walleye	F	Muscle	$\delta^{13}C$	1.62	-0.09	0.0275	-	-0.0006	-	0.0004	0.0	100	0.0
			$\delta^{15}N$	0.99	0.01	0.0005	-	-0.0001	-	0.0006	0.0	95.7	4.3
			MeHg	-2.90	-	1.9983	-	-	-	-	0.0	100	0.0
			ΣPAH4	1.72	-0.40	-0.7234	-	-0.0026	-	0.0671	0.0	95.5	4.5
			Total Hg	-2.91	-0.48	1.4722	-	-0.0008	-	0.0101	4.3	95.7	0.0
			Total PAH	2.07	-0.65	0.2877	-	0.0012	-	0.0362	0.0	100	0.0
			Total Se	-2.48	-0.01	0.7169	-	0.0030	-	0.0324	0.0	100	0.0
	M		$\delta^{13}C$	1.80	-0.10	-0.0298	-	0.0001	-	0.0004	0.0	100	0.0
			$\delta^{15}N$	0.63	0.01	0.1453	-	0.0002	-	0.0007	0.0	98.1	1.9
			MeHg	-58.76	18.69	0.8315	-	0.0138	-	0.0348	0.0	100	0.0
			ΣPAH4	-2.51	-0.16	0.5960	-	-0.0015	-	0.0322	0.0	98.1	1.9
			Total Hg	-3.56	-0.46	1.7479	-	0.0014	-	0.0371	3.7	96.3	0.0
			Total PAH	-0.11	-0.22	0.6453	-	0.0030	-	0.0499	0.0	98.1	1.9
			Total Se	-0.54	0.08	-0.1082	-	0.0004	-	0.0325	1.9	98.1	0.0
White Sucker	F		$\delta^{13}C$	1.71	-0.03	-0.0582	-	0.0002	-	0.0006	0.0	100	0.0
			$\delta^{15}N$	-0.81	-0.01	0.6740	-	-0.0002	-	0.0021	3.5	96.5	0.0
			MeHg	-136.38	38.05	6.7764	-	0.0240	-	0.0381	11.1	77.8	11.1
			ΣPAH4	-15.59	3.76	0.8156	-	0.0089	-	0.0161	0.0	100	0.0
			Total Hg	-11.97	-0.20	4.4634	-	0.0041	-	0.0402	8.6	88.6	2.9
			Total PAH	-84.84	25.87	1.4580	-	0.0177	-	0.0129	0.0	100	0.0
			Total Se	-5.97	0.03	1.9638	-	-0.0044	-	0.0348	2.9	97.1	0.0
	M		$\delta^{13}C$	1.70	-0.02	-0.0678	-	0.0002	-	0.0005	0.0	100	0.0
			$\delta^{15}N$	0.27	0.002	0.2583	-	-0.0002	-	0.0012	2.1	94.7	3.2
			MeHg	48.84	-17.76	3.6495	-	-0.0089	-	0.0189	0.0	100	0.0
			ΣPAH4	-10.34	3.34	-0.6715	-	0.0011	-	0.1267	0.0	94.4	5.6
			Total Hg	-8.45	0.09	2.8084	-	-0.0002	-	0.0502	5.4	94.6	0.0
			Total PAH	-44.41	15.61	-1.5449	-	0.0081	-	0.0348	0.0	94.4	5.6
			Total Se	-7.34	-0.05	2.6564	-	-0.0034	-	0.0208	5.4	95	0

Table Notes: Cells marked with a "—" refer to species, sexes, sample matrix, and compounds that did not have n > 2 sampling years and/or stations, Normal Ranges (NR) were calculated as the estimate in real units ± 2SD, LL and UL represent the upper and lower level, respectively, of the normal range.

3.3.8 Benthic Invertebrate Body Burden

3.3.8.1 Data QA/QC

There is no benthic body burden data associated with the OSMP included in this report, there are no comparisons to draw on between the OSMP and EMP datasets as it relates to benthic body burdens.

3.3.8.2 Normal Ranges

Since there is no OSMP benthic body burden dataset to use in generating normal ranges and inform on its predictability of the EMP data (similar to sections 3.3.3, 3.3.4, and 3.3.7), normal ranges were instead generated using only the EMP dataset.

The model used to generate normal ranges was developed with sampling year and distance upstream/downstream from the proposed OSPW discharge point as predictors. The relationship between different tissue level parameters with time and distance upstream/downstream has already been discussed and presented in section 2.3.10.2 (Table 38). In summary, body burden concentrations of 16 of the 34 analytes in Ametropodidae varied significantly with time and 2 analytes varied significantly with both time and distance. For Gomphidae, 25 of the 34 analytes varied significantly with time, 2 varied significantly with distance from the proposed OSPW discharge point, and 2 varied significantly with both time and distance. Finally, only one of the 34 analytes varied significantly with time and one analyte varied significantly with distance.

An example of the predicted normal ranges for total mercury in Ametropodidae can be found in Table 69 with an illustration of the model performance for each compound at that station in Figure 117. The models for benthic body burden data have retained all the components Year and distance, regardless of the statistical significance of those terms, in part for simplicity. Here, and with over 165 samples for Ametropodidae, 55 samples for Gomphidae, and 29 samples for Pteronarcyidae in the overall analysis, there is no problem with statistical power. Thus, including the non-significant terms in the model does not diminish model significance. The coefficients associated with 'non-significant' terms, further are 'not different from zero', and here with a very high Error df, are not very different from zero and have essentially negligible effect on estimated concentrations of the respective constituents.

For total mercury body burdens in Ametropodidae (as an example), the model constant was 111 while the slope for year was -0.055, and the slope for distance was 0.0015 (see model breakdown in Table 69). The model MSE was 0.006 (for $\log_{10}$ of [Hg]), the square root of which is 0.080. The SD among samples, for any modeled scenario is therefore 0.080.

Figure 118 summarizes the predicted analyte values models as evaluated against the observed values from field samples collected under the EMP. The model performance can be assessed based on the deviation of the points from the 1:1 line (i.e., a perfect fit). While the data points at each sampling station do tend to fall on the 1:1 line, there are instances of variation within the experimental results that are not captured by the predictive model. This was most evident in the model performance of Pteronarcyidae, where there was more variation in observed values (y-axis in Figure 118) compared to predicted values. There were less body burden samples included in the modelling exercise for Pteronarcyidae compared to the other families, which would reduce our model's ability to accurately predict values.

Normal range model coefficients and exceedances in the EMP data are summarized in Table 70 for a subset of analytes of interest. Normal ranges were computed as the model predicted average value ± 2 SDs. In this context, the normal range was anticipated to capture ~ 95% of potential future observations (Kilgour et al., 1998). Between 94 and 100 % of EMP samples fell within the normal range for Ametropodidae, between 95 and 100 % of EMP samples fell within the normal range for Gomphidae, and between 93 and 100 % of EMP samples fell within the normal range for Pteronarcyidae. These exceedance probabilities are about as predicted given that the EMP data were used to compute normal ranges, and the normal range region was designed to cover 95% of the data.

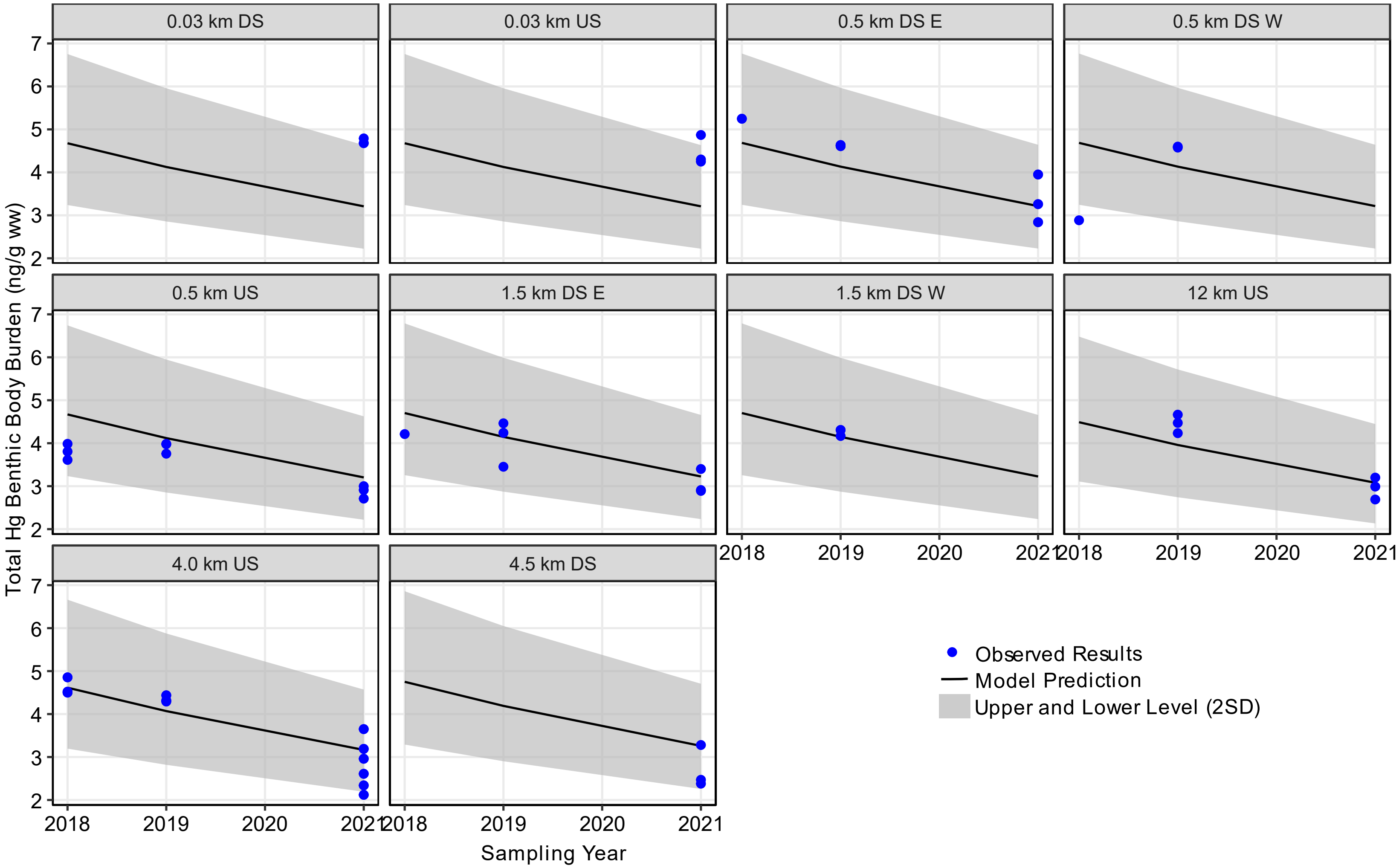

Figure 117 Variations in benthic body burden (Ametropodidae) model prediction of total mercury concentrations generated with EMP data in relation to sampling year compared to observed measurements measured during EMP (2018, 2019, and 2021) across each of the 9 stations sampled.

Figure Notes: Gaps in the data represent periods where samples were not collected.

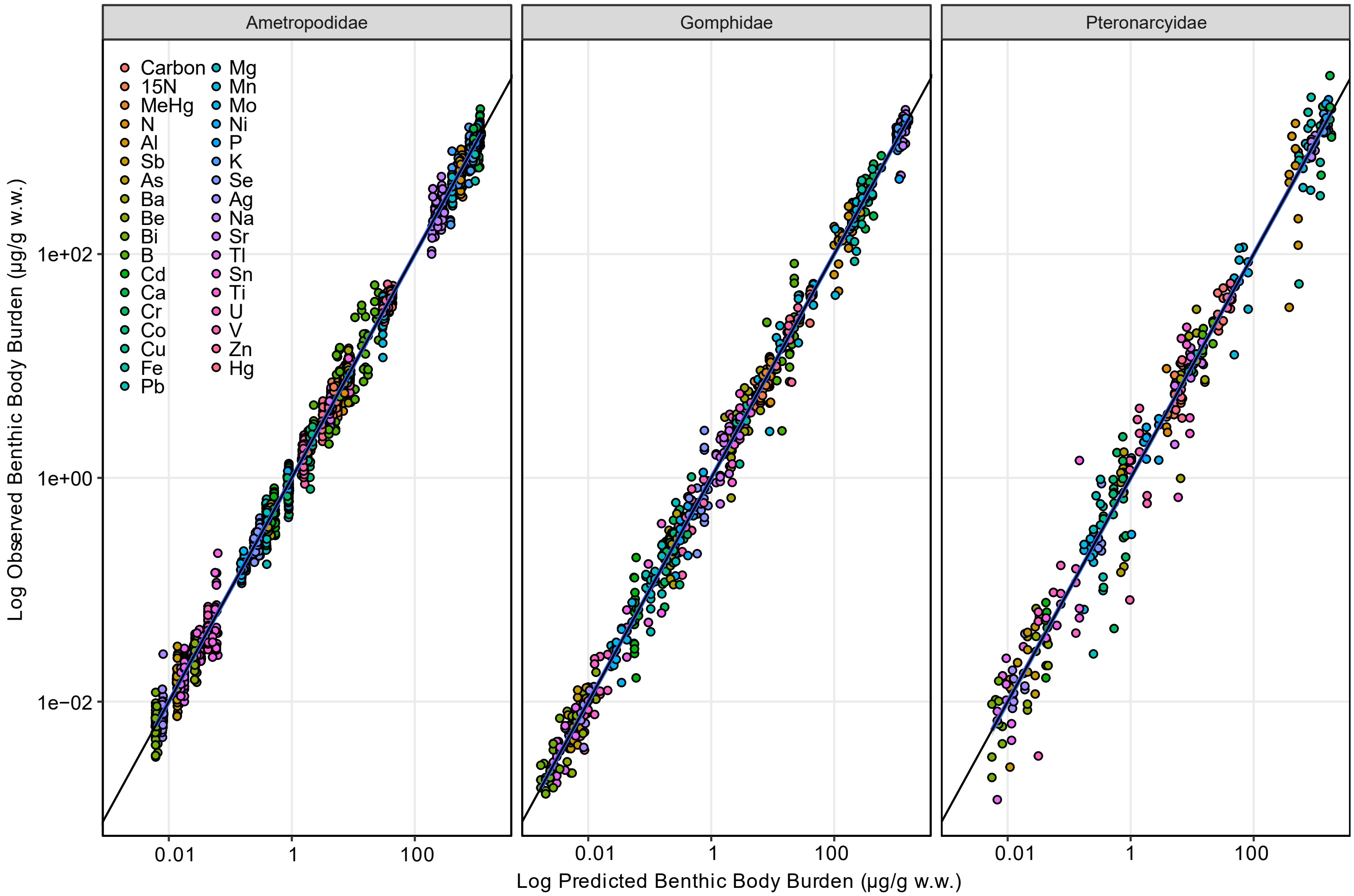

Figure 118 Benthic body burden model performance for each of the 35 individual analytes, x-axis represents the actual measurements during EMP, and the y-axis represents normal range model predictions.

Figure Notes: The black line represents a 1:1 line, while the blue line represents the overall goodness of fit for each benthic family.

Table 69 Example normal range model output for total mercury in Ametropodidae under different temporal and spatial scenarios.

Model Component	Description	Ametropodidae			
		Coefficient	Scenario		
			1	2	3
Constant	Intercept	111	1	1	1
Year	Slope for linear trend across years	-0.055	2022	2024	2026
Distance (US/DS)	Slope for linear relationship with distance US/DS	0.0015	-25	0.5	12
Total Hg estimate in logarithms			0.41	0.34	0.25
MSE in logarithms		0.006	—	—	—
SD in logarithms		0.080	—	—	—
Lower limit of normal range in logarithms		—	0.25	0.18	0.09
Upper limit of normal range in logarithms		—	0.57	0.50	0.41
Total Hg estimate in real units		—	2.60	2.21	1.78
Lower limit of normal range in real units		—	1.80	1.53	1.24
Upper limit of normal range in real units		—	3.75	3.18	2.58

Table 70 Model coefficients used for the prediction of body burden normal ranges in the EMP dataset and the percentage of NR exceedances (2018, 2019, and 2021).

Family	Analyte	Int	Year	Distance (US/DS)	MSE	NR Exceedances (%)		
						< LL	Inside	> UL
Ametropodidae	$\delta^{13}C$	38.3	-0.0183	-0.0011	0.0001	0	100.0	0
	$\delta^{15}N$	-70.5	0.0353	0.0056	0.0066	2.0	96.1	2.0
	Hg	110.7	-0.0545	0.0015	0.0064	2.8	93.5	3.7
	MeHg	-144.4	0.0716	-0.0023	0.0025	2.0	98.0	0.0
	Se	-59.4	0.0291	0.0032	0.0061	2.8	96.3	0.9
Gomphidae	$\delta^{13}C$	-12.6	0.0070	0.0071	0.0005	0.0	100.0	0.0
	$\delta^{15}N$	-15.8	0.0082	-0.1308	0.0051	0.0	94.7	5.3
	Hg	-222.5	0.1107	-0.0418	0.0066	0.0	100.0	0.0
	MeHg	-99.9	0.0499	-0.0843	0.0071	0.0	100.0	0.0
	Se	131.5	-0.0652	0.1113	0.0492	2.5	95.0	2.5
Pteronarcyidae	$\delta^{13}C$	17.9	-0.0081	0.0001	0.0003	0	100.0	0.0
	$\delta^{15}N$	105.6	-0.0519	-0.0123	0.0250	0	100.0	0.0
	Hg	-12.0	0.0063	-0.0872	0.0288	0	100.0	0.0
	MeHg	0.3	-	-0.3321	0.0300	0	100.0	0.0
	Se	-53.3	0.0261	-0.0701	0.0328	0	92.9	7.1

Table Notes: Normal Ranges (NR) were calculated as the estimate in real units ± 2SD, LL and UL represent the upper and lower level, respectively, of the normal range.

4.0 TASK 3 (PART I): STATISTICAL POWER

4.1 Approach

The ability to statistically detect change is a function of data variability (within sampling locations/times) and the magnitude of the true underlying difference (between treatments, say reference and exposure). Required sample sizes increase when the true differences are small, and the noise (within locations and times) is high. The ability to detect effects with the various enhanced monitoring response endpoints was determined through a power analysis carried out using PASS2020 software per Chow, et al. (2017), Julious (2010), Zar (1984), and Machin, et al. (1997). For each response parameter (water quality, sediment quality, benthos, fish, etc.), the among-replicate or within-location/time variability (standard deviation) was estimated from the residual error term (mean square error) from models that documented sources of variability. In short, this section assumes the discharge of OSPW is underway, and therefore all EMP sampling sites upstream of the OSPW discharge point are considered "reference" and all sites downstream from the OSPW discharge point are considered "exposure". In the subsequent sections, mean values are calculated using pooled upstream data.

4.1.1 Water Quality Variables

4.1.1.1 Grab Samples

Power calculations were used to determine sample size requirements for 26 water quality variables that have water quality objectives (Table 71). For each water quality variable, the CES was computed as the ½ way point between the typical upper limit of the normal range and the lowest of the available water quality guidelines (typically the CCME chronic exposure guideline). The upper limit of the normal range was approximated from $\bar{x}_{US} + 2SD_{US}$, where $\bar{x}_{US}$ is the mean of the upstream data and SD_{US} is the standard deviation of the upstream data. Power calculations were carried out with the standard deviation and differences between upstream (typical upper limit of the normal range) and the water quality guideline (see the column in Table 71 labelled "CES", which provides the critical effect size used in the power calculations). The next column over is the "N to detect CES", or the number of samples that would be required to detect a change in water concentration if the concentration was ½ way to the guideline.

For metals (and arsenic), 2 to 21 samples per station, per year, would be required in order to have sufficient likelihood of detecting when the concentrations are ½ way to the guideline, whereas for PAHs, the number of samples to detect when concentrations are ½ way to the guideline is 2 (Table 71). A total of 19 samples per area would be required to have sufficient likelihood of detecting when selenium concentrations are ½ way to the guideline. In EEM-style monitoring, one or two water samples are collected within sampling areas, as "supporting variables". In that sense, water quality data are not typically evaluated like benthic community data or fish population data, that is with the intention to determine if there are significant differences in concentrations between reference and exposure areas. In the table below, power calculations indicate that n=2 samples will provide sufficient power to detect changes in concentrations that are ½ way between the typical upper range and the lowest water quality guideline. In those cases, the typical upper range is a small fraction (in terms of concentration) of the ½ way point to the guideline (i.e., the CES). Some variables were noisier relative to the guideline. For example, the typical upper range of normal for total recoverable arsenic was 1.65 µg/L, with a guideline of 5 µg/L and CES of 3.33 µg/L (i.e., ½ way between 1.65 and 5), the typical upper normal range is about

50% of the CES. For total arsenic, power calculations suggest 11 samples would be required to provide sufficient likelihood of detecting a change in concentration from 1.65 to 3.33 µg/L. Other variables that would require higher sample sizes included lead (n=21), total mercury (n=10) and total selenium (n=19). Per the power calculations, these sample numbers are numbers required per sampling area/time.

Table 71 Results of power analysis for water quality variables that have water quality objectives.

Variable	Pooled SD	Typical Upper	WQG	CES	N to detect CES	Typical Upper / CES (%)
Acenaphthene	0.00033	0.00038	5.8	2.9002	2	0.01
Anthracene	0.00016	0.00018	0.012	0.0061	2	3.0
Benzo(a)anthracene	0.00038	0.00043	0.018	0.0092	2	4.7
Benzo(a)pyrene	0.00056	0.00065	0.015	0.0078	2	8.3
Fluoranthene	0.00122	0.00141	0.04	0.0207	2	6.8
Fluorene	0.00044	0.00050	3.0	1.5003	2	0.03
Naphthalene	0.0026	0.0030	1.0	0.5015	2	0.6
Phenanthrene	0.00290	0.00335	0.30	0.1517	2	2.2
Pyrene	0.00166	0.00191	0.025	0.0135	2	14
Aluminum Dissolved	9.44	10.9	100	55.45	2	20
Arsenic Total Recoverable	1.434	1.656	5	3.33	11	50
Boron Total Recoverable	19.6	22.66	1500	761.3	2	3.0
Copper Total Recoverable	3.481	4.02	2.96	10.98	4	37
Iron Total Recoverable	2548	2942	300	8038	4	37
Lead Total Recoverable	1.604	1.852	4.443	3.15	21	59
Manganese Total Recoverable	98.0	113.2	8381	4247	2	2.7
Mercury Total	0.00728	0.00841	0.0260	0.0172	10	49
Methyl Mercury	0.00014	0.00016	0.0040	0.0021	2	7.8
Molybdenum Total Recoverable	0.944	1.09	73	37.05	2	3
Nickel Total Recoverable	4.568	5.274	4.44	14.41	4	37
Nitrate as Nitrogen	35.3	40.8	13000	6520	2	0.6
Phosphorus Total	113	130	50	355	4	37
Selenium Total Recoverable	0.348	0.400	1.000	0.700	19	57
Silver Total Recoverable	0.016	0.018	0.250	0.134	2	13
Uranium Total Recoverable	0.524	0.605	15	7.80	2	7.8
Zinc Total Recoverable	8.539	9.86	37.5	23.68	6	42

Table Notes: SD = Standard Deviation; CES = Critical Effect Size; N = Number of Samples; List of water quality guidelines (WQG) are provided in Government of Alberta (2018) and CCME (2023)

4.1.1.2 SPMDs

Power calculations were used to determine sample size requirements for the PAHs (for which SPMD data exists) as was done in Section 4.1.1.1 for surface water grab samples (Table 72). For each PAHs, the CES was computed as the ½ way point between the typical upper limit of the normal range and the lowest of the available water quality guidelines (typically the CCME chronic exposure guideline). We acknowledge that guideline values are for total concentrations, not dissolved as captured by SPMD's, however in order to facilitate comparison between the power of grab samples and SPMD's, the same guidelines were used. The upper limit of the normal range was approximated from $\bar{x}_{US} + 2SD_{US}$, where $\bar{x}_{US}$ is the mean of the upstream data and SD_{US} is the standard deviation of the upstream data. Power calculations were carried out with the standard deviation and differences between upstream (typical upper limit of the normal range) and the water quality guideline (see the column in Table 72 labelled "CES", which provides the critical effect size used in the power calculations). The next column over is the "N to detect CES", or the number of samples that would be required to detect a change in water concentration if the concentration was ½ way to the guideline.

For all SPMD PAHs, 2 samples per area would be required to have sufficient likelihood of detecting when the concentrations are ½ way to the guideline. These sample sizes are so low because the CES's are roughly 50 to >4000X higher than the upper normal range of the reference sites ($\bar{x}_{US} + 2SD_{US}$). Therefore, any change of that magnitude would be easily detectable with a very low sample size, as is also the case for grab samples described in Section 4.1.1.1.

Table 72 Results of power analysis for SPMD PAHs that have water quality objectives.

Variable	Pooled SD	Typical Upper	WQG	CES	N to detect CES	Typical Upper / CES (%)
Acenaphthene	0.000184	0.000654	5.8	2.9003	2	0.02
Anthracene	0.000026	0.000101	0.012	0.0061	2	1.67
Benzo(a)anthracene	0.000009	0.000040	0.018	0.0090	2	0.44
Benzo(a)pyrene	0.000004	0.000019	0.015	0.0075	2	0.25
Fluoranthene	0.000068	0.000357	0.04	0.0202	2	1.77
Fluorene	0.000056	0.000217	3	1.5001	2	0.01
Naphthalene	0.000588	0.001575	1	0.5008	2	0.31
Phenanthrene	0.000186	0.000856	0.3	0.1504	2	0.57

4.1.2 Sediment Quality Variables

Power calculations were used to determine sample size requirements for 17 sediment quality variables that have sediment quality objectives (Table 73). For each sediment quality variable, the CES was computed as the ½ way point between the upper limit of the normal range and the interim sediment quality guideline. The upper limit of the normal range was approximated from $\bar{x}_{US} + 2SD_{US}$, where $\bar{x}_{US}$ is the mean of the upstream data and SD_{US} is the standard deviation of the upstream data. For sample size calculations, this approximation of the upper limit of the normal range is considered sufficient. Sources of variability in sediment quality variables involved analysis and assessment of the logarithm (base 10) transformed concentrations. Therefore, power calculations were carried out with the standard deviation and differences between reference (upper limit) and Intermediate Sediment Quality Guidelines (ISQG) expressed as logarithms (see the column in Table 73 labelled "log of ½ way point (CES), which provides the critical effect size used in the power calculations). The next column over is the "N to detect CES", or the number of samples that would be required to detect a change in sediment concentration if the concentration was ½ way to the guideline (i.e., the ISQG).

Because sediments are often synoptically sampled with benthos, and because benthos community sampling typically involves the collection of *n*=5 samples per sampling area (Environment Canada, 2012a), the detectable effect size, for an *n* of 5, was also computed (Table 73).

For most metals (and mercury), only 2 samples per area would be required to have sufficient likelihood of detecting when the concentrations are ½ way to the ISQG (Table 73). More samples would however be required to have sufficient likelihood of detecting when the concentration of Arsenic (29 samples) and Cadmium (3 samples) reaches the CES. More samples would also be required to have sufficient likelihood to detect PAH concentrations when they reach the ½ way point to the guideline (i.e., CES) where sample numbers vary between 4 and 78 depending on the PAH (Table 73). Power curves were also developed for the metals (i.e., aluminum and thallium) and sediment quality variables (i.e., selenium, phosphorus, TOC) that do not have sediment quality objectives as well as total PAHs and naphthenic acids (Figure 119) to detect changes of specific magnitudes. A relatively low number of samples would be required to have sufficient likelihood to detect small changes in aluminum, (2 samples), thallium (5 samples to detect 20% increase in concentration), selenium (2 samples), phosphorus (3 samples to detect 20% increase in concentration), and TOC (3 samples to detected 20% increase) but more samples would be required to have sufficient likelihood to detected meaningful changes in Total PAH (26 samples to detect a 50% increase) and naphthenic acids (27 samples to detect a 50% increase) concentrations. In general, larger changes in concentrations could be detected with even fewer sediment samples.

Table 73 Results of power analysis for sediment quality variables that have sediment quality objectives.

Variable	MSE	SD	Reference Mean (Upstream Sites)			ISQG			% Change Increase at ISQG	log Trigger (CES)	Trigger (CES)	N to detect CES	Detectable Concentration Change with n=5			
			Units	Values	log Values	Units	Values	log Values					Units	log Values	Values	% Increase
Total Arsenic (As)	0.0024	0.049	mg/kg	4.94	0.69	mg/kg	5.9	0.77	19	0.73	5.40	29	mg/kg	0.17	1.48	30
Total Cadmium (Cd)	0.0067	0.082	mg/kg	0.16	-0.80	mg/kg	0.6	-0.22	280	-0.51	0.31	3	mg/kg	-1.1	0.08	50
Total Chromium (Cr)	0.0007	0.027	mg/kg	12.83	1.11	mg/kg	37.3	1.57	191	1.34	21.88	2	mg/kg	0.28	1.92	15
Total Copper (Cu)	0.0026	0.051	mg/kg	9.17	0.96	mg/kg	35.7	1.55	289	1.26	18.09	2	mg/kg	0.44	2.75	30
Total Lead (Pb)	0.0009	0.029	mg/kg	6.60	0.82	mg/kg	35.0	1.54	430	1.18	15.20	2	mg/kg	0.12	1.32	20
Total Mercury	0.0131	0.114	ng/g	30.17	1.48	mg/kg	0.17	-0.77	-99	0.36	2.26	2	mg/kg	1.38	24.14	80
Total Zinc (Zn)	0.0004	0.021	mg/kg	48.03	1.68	mg/kg	123.0	2.09	156	1.89	76.86	2	mg/kg	0.68	4.80	10
2-Methylnaphthalene	0.0361	0.190	ng/g	13.60	1.13	ng/g	20.2	1.31	49	1.22	16.57	78	ng/g	1.31	20.40	150
Acenaphthene	0.0324	0.180	ng/g	0.80	-0.10	ng/g	6.7	0.83	739	0.36	2.32	4	ng/g	0.049	1.12	140
Anthracene	0.0552	0.235	ng/g	2.70	0.43	ng/g	46.9	1.67	1637	1.05	11.25	4	ng/g	0.75	5.67	210
Benz[a]anthracene	0.0605	0.246	ng/g	10.40	1.02	ng/g	31.7	1.50	205	1.26	18.16	19	ng/g	1.360	22.88	220
Benzo[a]pyrene	0.0497	0.223	ng/g	12.70	1.10	ng/g	31.9	1.50	151	1.30	20.13	23	ng/g	1.38	24.13	190
Chrysene	0.0930	0.305	ng/g	18.00	1.26	ng/g	57.1	1.76	217	1.51	32.06	27	ng/g	1.76	57.60	320
Fluoranthene	0.1498	0.387	ng/g	7.70	0.89	ng/g	111.0	2.05	1342	1.47	29.24	9	ng/g	1.97	93.60	520
Naphthalene	0.1232	0.351	ng/g	5.10	0.71	ng/g	34.6	1.54	578	1.12	13.28	14	ng/g	1.33	21.42	420
Phenanthrene	0.1616	0.402	ng/g	15.00	1.18	ng/g	41.9	1.62	179	1.40	25.07	58	ng/g	1.92	84.00	560
Pyrene	0.1369	0.370	ng/g	11.80	1.07	ng/g	53.0	1.72	349	1.40	25.01	23	ng/g	1.70	55.46	470

Table Notes: MSE = Mean Squared Error; SD = Standard Deviation; ISQG = Intermediate Sediment Quality Guidelines; N = Sample Size; SD = Standard Deviation; CES = Critical Effect Size; N = Number of Samples; List of water quality guidelines (WQG) are provided in Government of Alberta (2018) and (CCME, 2023). Data was log-transformed (base 10) where indicated.

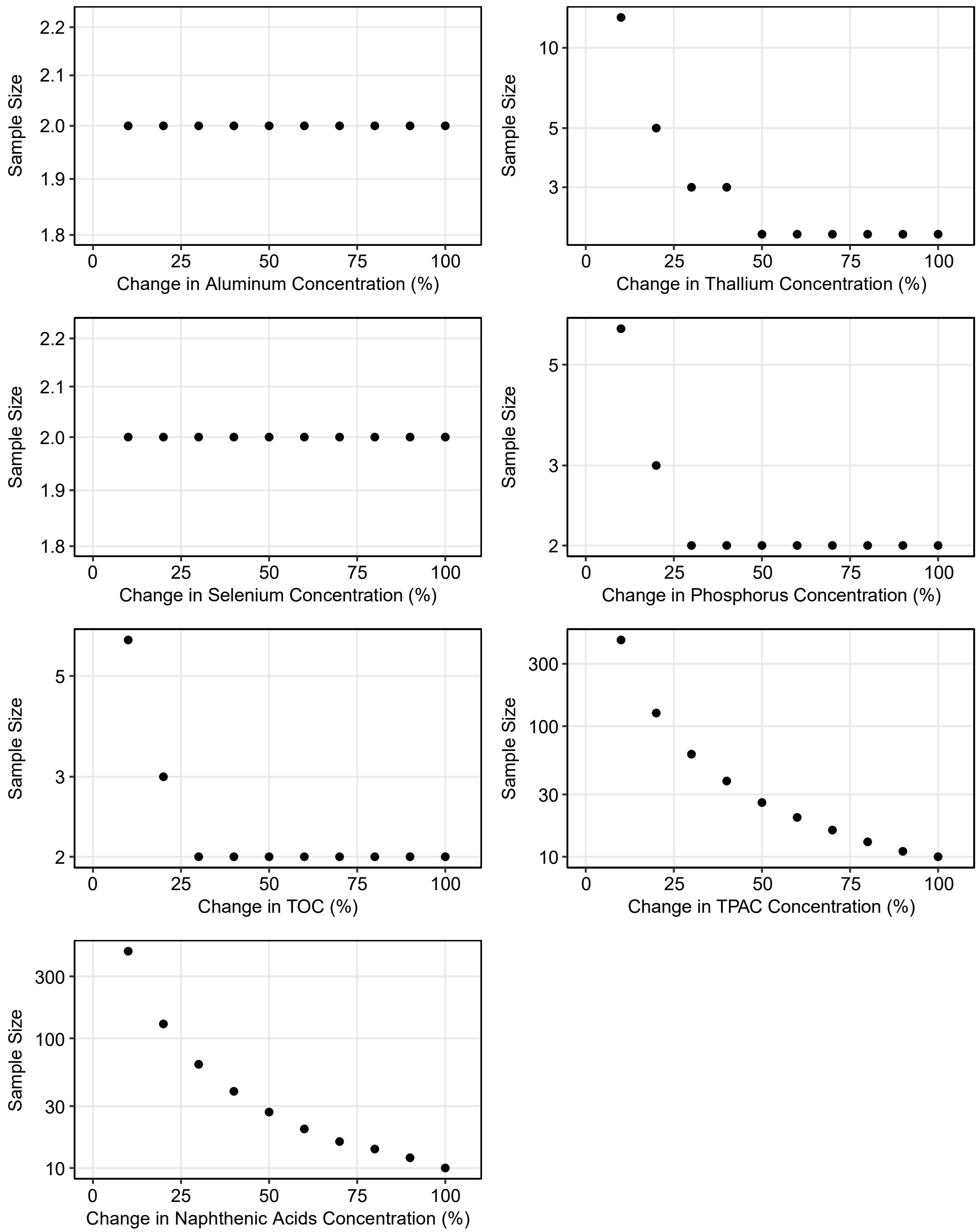

Figure 119 Power curves illustrating the number of samples required to detect changes in sediment concentrations of specific magnitudes of aluminum, thallium, selenium, phosphorus, total organic carbon (TOC), total polycyclic aromatic compounds (TPAH), and naphthenic acids.

4.1.3 Algae Community Composition Indices

The federal environmental effects monitoring programs for pulp and paper (i.e., Pulp and Paper Effluent Regulations; PPER) and metal mining (i.e., MDMER) commonly use the CESs for surveys of BIC and fish populations. With the data available as part of the enhanced monitoring program, the same composition indices that are used for benthic invertebrates were used here for the algae community. Power calculations were completed to determine the within-area sample size requirements, for the various algal community composition indices and with effect sizes expressed in the real units. For each response variable, the within-area standard deviation was estimated from the square root of the mean square error term from the analysis of variance provided in Table 74. Power calculations were conducted to develop power curves to illustrate the number of samples required to detect changes of specific magnitudes for density, family richness, Simpson's evenness and diversity, Chlorophyll-*a* (Chl-*a*) biomass, total biomass, as well as scores on NMDS axes 1 and 2. The required sample sizes, in relation to effect size are summarized/illustrated in Figure 120. The Nordin (2001) benchmark of 100 mg/m^2 Chl-*a* was used to derive a critical effect size for that Chl-*a*.

Density, Chl-*a* biomass, and total biomass were the most variable algae indices, resulting in a relatively large number of samples (> 17) per Site required to have reasonable likelihood of detecting interpretable changes. Upstream average taxa richness was ~13. Power calculations indicate 9 samples per Site would provide reasonable power to detect changes in richness equal to baseline mean (Figure 120).

Other variables were more sensitive. The upstream mean Chl-*a* level was 4.1 mg/m^2. Approximately four samples would provide sufficient power to detect a change equal to the ½ way point to the 100 mg/m^2 benchmark. Approximately five samples would detect changes of 0.3 units in diversity with 90% likelihood, and Type I error of 10%: seven samples would be needed for evenness.

NMDS axis 1 scores varied from approximately -3 to +1, while axis 2 scores varied from roughly -2 to +3. Approximately 10 and six samples per Site will generally be sufficient to detect changes in NMDS axis 1 or 2 scores, respectively, of about 1 unit. Statistical power as it relates to NMDS axis scores is somewhat understated. NMDS axis scores are 'calibrated' with the data provided, which here were essentially in a baseline condition: that is, there were no highly degraded Sites with which to 'calibrate' the ordination scores.

Table 74 Algae indices of community composition value standard deviations (SDs) used in power calculations.

Algae Indices	Upstream Reference Means	Unit	SD within sampling areas/times after adjusting for modifying factors
log Density	$10^{5.4}$	Organisms/m^2	0.69
log Richness	$10^{1.5}$	LPL/sample	0.31
Simpson's Diversity	0.86	unitless	0.12
Simpson's Evenness	0.36	Unitless	0.21
log Chlorophyll A (Chl-*a*)	$10^{-0.38}$	mg/m^2	0.46
log Biomass	2.31	g/m^2	0.74
NMDS Axis 1	-0.11	Unitless	0.64
NMDS Axis 2	0.035	Unitless	0.42

Table Notes: Data were log-transformed where indicated;
Upstream Reference Means refer to sites located upstream of the proposed OSPW release point.

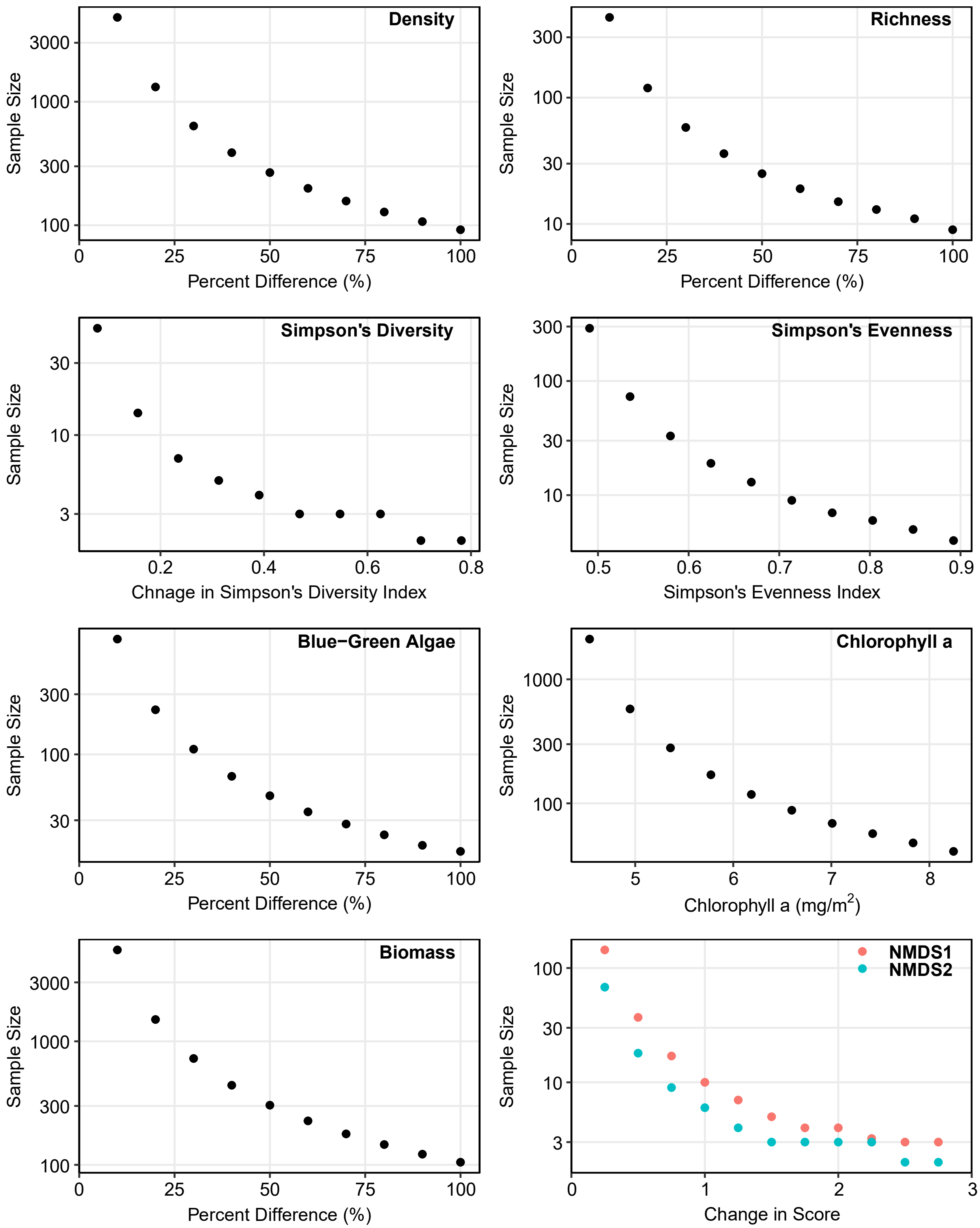

Figure 120 Power curves for indices of algae community composition for the EMP data set.

4.2 Monitoring Components

In EEM, the selection of monitoring components is ideally determined after consideration of Valued Components (VCs) and Conceptual Site Models (CSMs). VCs are typically identified through stakeholder processes such as environmental assessment. In the case of the EMP, components were selected in consultation with nationally recognized scientists with expertise in aquatic environment monitoring, including those working on behalf of Indigenous communities. The design of the EMP followed the conventional designs that have developed and evolved in Canada since the mid-1990s. Aquatic EEM programs in Canada (generally) consist of the following components (Hatfield Consultants, 2022):

1. Biological responses
    a. Benthic communities;
    b. Fish populations; and,
    c. Fish tissue contaminant levels (mercury, selenium).
2. Water Quality; and,
3. Sediment Quality.

Under the federal EEM programs for pulp & paper (Pulp and Paper Effluent Regulations, PPER) and metal mining (Metal and Diamond Mining Effluent Regulations, MDMER), surveys of adult fish populations have been a keystone for EEM because they relate obviously and directly to the goals of the federal Fisheries Act (Fisheries and Oceans Canada, 2019; Hatfield Consultants, 2022; Kilgour et al., 2005). Benthic communities, in contrast, are justified in the federal regulations (PPER, MDMER) on the basis that they represent potential effects on fish habitat. Benthic communities in that context are ‘surrogate’ indicators (Cairns et al., 1993) of the potential for effects to fish. Contaminant levels in fish tissue, again per the federal regulations and approaches to EEM, are monitored to indicate the potential effects of effluents on use of fisheries resources. In federal mining EEM programs, selenium is monitored because it is regularly observed as a risk to fish populations receiving mining effluents. Mercury in fish flesh is also monitored in mining EEM programs because of enhanced risks particularly with effluents from gold mines. Mercury in fish flesh poses risks wildlife and human consumers (Alberta Environment, 2018; CCME, 2000). Elevated levels in of selenium in fish tissues poses risks to fish (Alberta Environment, 2018), not persons (CCME, 1999). These components were incorporated into the enhanced monitoring program (Table 65 and Table 75).

The enhanced monitoring program collected additional kinds of data, to reflect concerns raised by various stakeholders. The EMP additionally collected:

1. Periphytic algae community data;
2. Water chemistry variables (hydrocarbons, naphthenic acids);
3. Sediment chemistry (nutrients, hydrocarbons, PAHs, naphthenic acids);
4. Fish tissue chemistry (EROD, PAHs, SIRs of $\delta^{13}C$ and $\delta^{15}N$); and,

5. Benthic tissue chemistry (SIRs of $\delta^{13}C$ and $\delta^{15}N$).

These data types were added to the EMP because they are anticipated to respond to one or more of the anticipated constituents of oil sands process water, and because they have the potential to provide early warning of potential effects on fish or human health, and/or to support interpretation of the cause of change in biological effects. Treated process waters are anticipated to contribute nutrients (nitrogen and phosphorus), hydrocarbons, PAHs and naphthenic acids to the Athabasca River (Four Elements Consulting, 2022).

Algal communities (as documented by community composition, chlorophyll *a* biomass and total biomass) are anticipated to change quickly to changes in nutrient levels (phosphorus principally) and as such can potentially provide early warning of potential effects on benthos and the fish community (Cairns et al., 1993; Kilgour et al., 2005). There is potential that periphyton will also respond to other constituents such as metals, hydrocarbons, etc., still providing early warning of potential effects that may cascade to benthos and fish.

Water and sediment quality indicators in the EMP included those constituents that are likely to be contributed by treated process water, including hydrocarbons, PAHs, and naphthenic acids. Those data can be used to confirm exposure conditions and will support interpretation of cause of effects (diagnostic indicators per (Cairns et al., 1993). Water and sediment quality data can also be used to confirm predictions made from mass-balance models such as those provided by Four Elements Consulting (2022). Four Elements estimated contributions from treated process waters to the Athabasca River for various constituents including nutrients, major ions, hydrocarbons. For any regulated release of process water, engineering (mass-balance) models can be expected to be required to estimate constituent concentrations in treated effluent, as well as the assimilative capacity of the Athabasca River. Monitoring of these constituents in the receiver can subsequently verify the mass-balance models (Somers et al., 2018).

EROD is a classic indicator used in monitoring programs to demonstrate fish have been exposed to organic chemicals (Whyte et al., 2000), perhaps from an effluent. EROD therefore is valuable in the EMP context, because it will provide evidence that monitored fish have been exposed (or not) to released process water. Given their body size, White Sucker can be anticipated to have a home range of 1,000 to 10,000 m^2 (Minns, 1995) during their spawning season (Doherty et al., 2010). The home range of White Suckers, therefore, may be sufficiently large that individual fish may move between reference and exposure Sites (depending on how far apart they are), complicating their use as sentinel organisms. Trout-perch, being considerably smaller, can be anticipated to have home ranges that are between about 10 and 200 m^2 (Minns, 1995) based on body size. As such, individual Trout-perch are less likely to move significantly between reference and exposure Sites and are subsequently a more suitable sentinel species for EEM programs focused on point-source releases. However, for both sucker and Trout-perch, measured EROD would provide an additional line of evidence of effluent exposure (the other line of evidence being location of capture).

Stable isotope ratios are a common tool for determining food source (Hobson, 2007; Trueman & Moore, 2007), including those caused anthropogenically (Bannon & Roman, 2008; Corbett et al., 2015; Cunjak et al., 2005). The carbon and nitrogen in oil sands process water has a unique isotopic signature relative to surface waters derived from natural overland flows (Chad et al., 2022; Gibson et al., 2011). Aquatic organisms (algae, benthos, fish) residing upstream of process-water release points should therefore have a different isotopic signature compared to organisms living/feeding in Sites influenced by oil sands process waters. Isotope ratios may also then provide additional evidence of exposure history.

Some of the variables monitored under the EMP relate to potential human health risks. Water samples, for example, were processed by the ACFT at the University of Calgary using the human hepatocarcinoma (HepG2) cell-based water cytotoxicity assay. By measuring the perturbation of cellular growth by the mixture components in water samples, the water cytotoxicity assay is considered a human-health relevant test and a screening tool for whole mixtures (Kinniburgh et al., 2021; Pan et al., 2013), with C-WQI values greater than 1.0 indicating a significant biological response. C-WQI values have also been shown to correlate with lethal doses in rats (Pan et al., 2013), thereby indicating relevance to mammalian and therefore human health. Mercury levels in fish tissues were also monitored under the EMP. Mercury levels in fish tissue have obvious relevance to human health (Gaudet et al., 1995), with the Government of Alberta (2019) setting 0.2 mg/kg w.w. as a guideline for subsistence consumers, and 0.5 mg/kg w.w. as a guideline for commercial sale. PAHs were measured in tissues of White Sucker and Trout-perch. Benzo(a)pyrene levels of 2 ng/g w.w., and ∑PAH4 (i.e., sum of benzo(a)pyrene, benz(a)anthracene, benzo(b)fluoranthene, chrysene) levels of 12 ng/g w.w. pose risks to human health (European Union Commission Regulations 835/2011, 2015/1125 amending 1881/2006). The C-WQI and fish tissue concentrations related to human health risks are not meant as the direct outcome; it is anticipated that they would be considered as inputs to the comprehensive human health risk assessment. Those results can be thought of as a screening-level risk assessment for the hazard identification step.

Table 75 Aquatic environment monitoring components incorporated into oil sands monitoring (OSM) and the enhanced monitoring program (EMP) in the mainstem of the Athabasca River.

Component		Rationale	Measured Indicators	Calculated Indicators	Trigger Thresholds		
					Normal Range	Predictions	Guidelines
Biological (Effect Indicator)	Periphytic algae	Early-warning indicator for effects on fish. Expected to be the first biological response to change.	Periphyton community composition	Density, richness, evenness, multivariate ordination	x		
			Chlorophyll a biomass		x		
			Biomass		x		x
	Benthic communities, CABIN protocol	Early-warning indicator for effects on fish	Community composition	Density, richness, evenness, EPT, multivariate ordination	x		
	(Adult) Fish populations (e.g., White Sucker, Trout-perch)	Surrogate for effects on fish communities	Mean age		x		
			Size at age		x		
			Liver size	LSI	x		
			Gonad size	GSI	x		
			Condition factor	K	x		
Environmental Quality (Potential Predictors)	Water	(1) Supporting data to confirm reference/exposure condition (2) Verify water quality predictions (3) Supports interpretation of biological effects (4) Support interpretation of potential risks to persons	General limnological variables		x	x	x
			Nutrients		x	x	x
			Major ions (e.g., chloride, sodium)		x	x	x
			Metals (V, Ni, As, Se)		x	x	x
			Hydrocarbons (TPAH)		x	x	x
			Naphthenic acids	Fathead minnow TUs	x	x	x
			Cytotoxicity test	Cytotoxicity index	x	x	x
	Sediments	(1) Supporting data to confirm reference/exposure condition (2) Verify sediment quality predictions	Grain size, TOC		x	x	
			Nutrients		x	x	
			Major ions (e.g., chloride, sodium)		x	x	x
			Metals (V, Ni, As, Se)		x	x	x

Component		Rationale	Measured Indicators	Calculated Indicators	Trigger Thresholds		
					Normal Range	Predictions	Guidelines
		(3) Supports interpretation of biological effects	Hydrocarbons		x	x	
			PAHs		x	x	
			Naphthenic acids		x	x	
	Fish tissues	(1) Confirms exposure (2) Provides levels relevant to human health exposure	EROD		x		
			Mercury		x	x	
			PAHs		x	x	
			Selenium		x		x
			Stable SIRs ($\delta^{13}C$ and $\delta^{15}N$)		x	x	
	Benthos tissues	(1) Early warning of potential change in fish body burdens (2) Supports interpretation of fish body burdens	Mercury		x	x	
			PAHs		x	x	
			Selenium		x	x	x
			Stable SIRs ($\delta^{13}C$ and $\delta^{15}N$)		x		

4.2.1 Benthic Invertebrate Indices of Composition

The federal environmental effects monitoring programs for pulp and paper (i.e., PPER) and metal mining (i.e., MDMER) use the CESs for surveys of BIC and fish populations as indicated in Table 76 below. CESs for benthos density, family richness and evenness are: $CES = \bar{x}_{US} \pm 2SD_{US}$, where $\bar{x}_{US}$ is the mean of the reference data and SD_{US} is the standard deviation of the upstream data. For benthic community endpoints, and with simple before vs after, control vs impact, or reference vs exposure designs, and with one sampling area in each treatment, five samples per area is sufficient to detect effect equal to the CES with Type I and II errors equal to 0.1 (i.e., 90% power, with 10% likelihood of declaring a difference to be significant when there is no difference) (Environment Canada, 2012a). Sample sizes of n=5, therefore, provide sufficient power to detect differences between reference and exposure areas when the differences are approximately equal to the background noise (Kilgour et al., 1998a). Additional sample size calculations, further, for this case are not required (Environment Canada, 2012a).

However, there remains interest in understanding what the detectable effect sizes are for benthic community indices, expressed in the real units (i.e., not standard deviations, but things like actual density or percent EPT). Therefore, power calculations were completed to determine the within-area sample size requirements, for the various benthic indices and with effect sizes expressed in the real units.

With the enhance monitoring program data, power calculations were completed for:

- Density;
- LPL richness;
- Evenness;
- Diversity;
- % EPT;
- PTI; and,
- NMDS axes 1 and 2.

For each response variable, the within-area standard deviation was estimated from the square root of the mean square error term from the analysis of variance provided in Table 76. From the power calculations, power curves were developed (Figure 121) to illustrate the number of samples required to detect changes of specific magnitudes for the calculated benthic indices.

Densities are the most variable benthic indices of community composition considered, requiring 100 or more samples per Site to detect changes of 2000 organisms per sample (i.e., the typical number).

Site average taxa richness was typically 20 to 40 (Figure 44). To detect changes in richness of that magnitude, would require > 10 samples per Site (Figure 121).

Five samples will detect (with 90% likelihood, and Type I error of 10%) changes of 0.3 units for both diversity and evenness.

Percent tolerant chironomids (i.e., PTI) produced relatively tighter estimates, such that only two kick samples per Site would be needed to detect a change in PTI scores of 1 unit. Finally, five samples per Site will generally be sufficient to detect changes in NMDS axis 1 or 2 scores of about 1 unit. NMDS axis 1 scores varied from approximately -3 to +1, while axis 2 scores varied from about -2 to +1. Those ranges are for existing conditions in the mainstem Athabasca River, which are generally (and currently) in a baseline (minimally influenced) condition.

Percent EPT was also somewhat variable such that upwards of 100 samples per Site would be required to detect changes of 20%. EPT accounted, typically, for about 10% of the fauna (Figure 47), such that a reduction in %EPT would be difficult to detect with n=5. It is unlikely that %EPT will increase significantly with any perturbance.

Total densities, taxon richness and %EPT were sufficiently variable within Sites, that a relatively large number of samples (> 10) per Site would be required before it would be likely that effects equal to the average condition would be detectable. However, diversity and evenness, as well as ordination axis scores and PTI can be anticipated to be sufficiently powerful to detect more subtle effects. PTI is demonstrated in the power analysis to be exceptionally powerful, requiring only 2 samples per Site to detect changes of a single index unit.

Understanding, or quantifying, the power of the benthic program in the case of the EMP is somewhat challenging since there were no highly degraded Sites with which to 'calibrate' the ordination scores. NMDS scores will generally scale between -3 and +3, almost regardless the nature of the variability in the communities. As such, the ability to detect change in benthic communities, using multivariate ordination techniques can be anticipated to be greater, if meaningful changes occur.

Table 76 Benthic indices of community composition standard deviations (SDs) used in power calculations.

Benthic Indices	Upstream Reference Means	Units	SD within sampling areas/times after adjusting for modifying factors
log Density	0.69	Organisms/m^2	0.59
log LPL Richness	0.76	LPL/sample	0.23
log %EPT	0.74	%	0.49
log PTI	0.84	Unitless	0.03
Simpson's Diversity	0.69		0.17
Simpson's Evenness	0.30		0.14
NMDS Axis 1	0.003		0.55
NMDS Axis 2	-0.15		0.44

Table Notes: Data were log-transformed (base 10) where indicated.

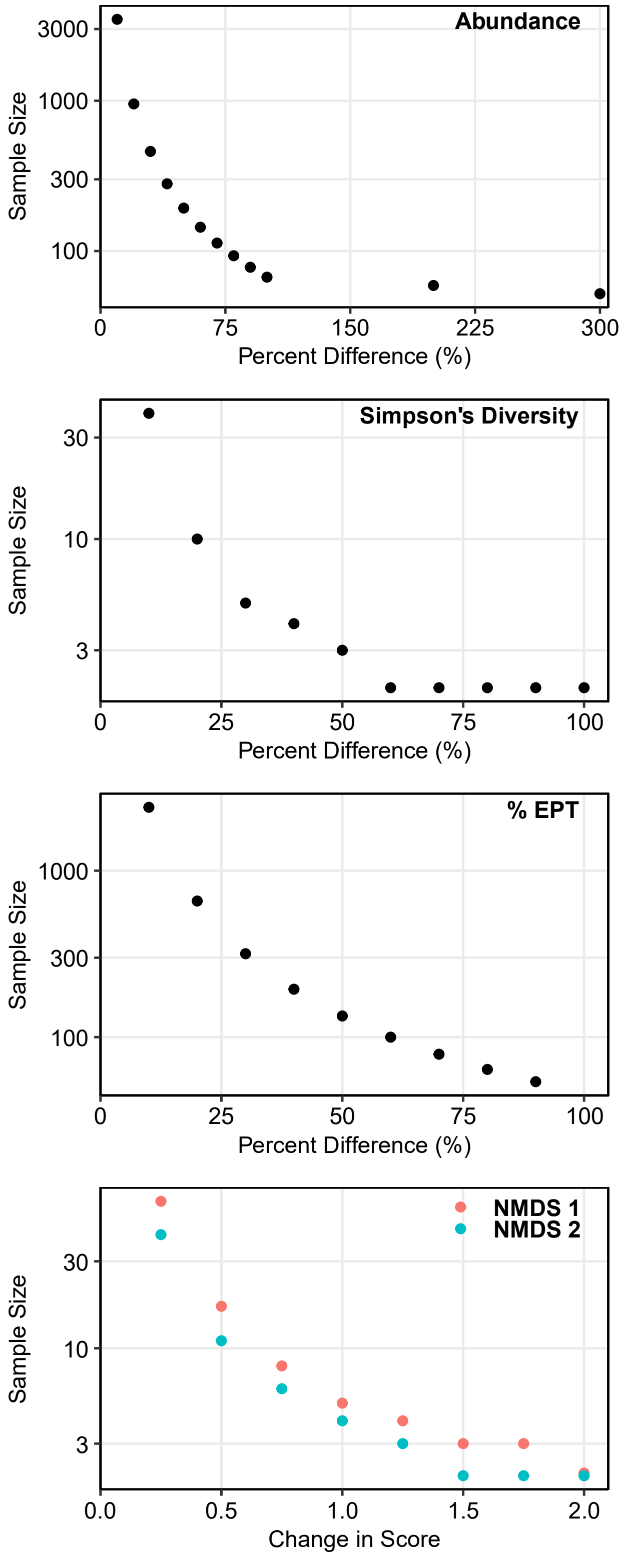

Figure 121 Power curves for indices of benthic community composition for the EMP data set.

4.2.2 Sentinel Adult Fish Population Indices

For fish population parameters, power analyses were used to compute sample sizes required to achieve 95% likelihood of detecting CESs. Here, the CESs were the reference mean ±25% for liver size, gonad size and growth, and the reference mean ± 10% for condition factor, as provided by ECCC's metal mining EEM guidance document (Environment Canada, 2012a).

Table 77 below provides mean-square error (MSE) terms for GSI, K and LSI for Trout-perch and Common White Sucker. Those MSE values are from tables summarizing sources of variation for those indices (Table 34) and were used to estimate the within-area/within time standard deviation (i.e., $SD = \sqrt{MSE}$).

The results of the power calculations are provided in Table 77. The number of fish required to detect effects equal in magnitude to the critical effect size varies between 7 (White Sucker, male GSI) to 29 (Trout-perch, male GSI). For each species and sex, the critical n that would achieve desired power for all three indices is shaded light grey. For Trout-perch females, 23 fish per sampling area/time would achieve desired power for all indices, while for male Trout-perch the critical number is 29, for female White Sucker critical sample size is 14, and for male White Sucker critical sample size is 13. These estimated sample sizes are generally consistent with what is anticipated given fish population sampling programs elsewhere (Environment Canada, 2012a).

Table 77 Results of power calculations for fish population indices.

Species	Sex	Indicator	Mean Square Error	SD	Ref Mean	CES	Critical n
Trout-perch	F	GSI	1.93	1.39	4.27	5.34	23
		k	0.01	0.09	1.15	1.26	10
		LSI	0.08	0.29	1.56	1.94	9
	M	GSI	0.22	0.47	1.31	1.63	29
		k	0.01	0.09	1.12	1.23	10
		LSI	0.06	0.24	1.28	1.60	9
Common White Sucker	F	GSI	1.30	1.14	5.19	6.48	11
		k	0.02	0.15	1.54	1.69	14
		LSI	0.13	0.36	1.66	2.08	11
	M	GSI	0.67	0.82	4.81	6.01	7
		k	0.02	0.13	1.51	1.66	11
		LSI	0.09	0.30	1.23	1.54	13

Table Notes: SD = Standard Deviation, GSI = Gonadosomatic Index, K = Condition Factor, LSI = Liver Somatic Index, Ref= Reference, CES = Critical Effect Size, n = Number of Samples (Fish)

4.2.3 Benthic Tissues

4.2.3.1 Methylmercury and Selenium

Power calculations were used to determine sample size requirements to achieve 90% likelihood of detecting the CES for methylmercury and selenium concentrations in benthic organisms. The CESs for benthic body burdens here were set to the ½ way point between the upstream reference means and the tissue guidelines for the protection of wildlife (33 ng/g w.w. for methylmercury and 4 µg/g d.w. for selenium) (Alberta Environment, 2018; CCME, 2000). The background upstream concentrations and standard deviations of methylmercury and selenium in Ametropodidae, Gomphidae, and Pteronarcyidae are listed in Table 78. The within-area/time standard deviations were estimated from the MSE terms (i.e., $SD = \sqrt{MSE}$) derived from the GLMs used to explore potential sources of variation in measured methylmercury and selenium tissue concentrations in benthic invertebrates.

Methylmercury concentrations detected in benthic invertebrates were relatively low, ranging from 1.63 ng/g w.w. (detected in Ametropodidae) to 7.17 ng/g w.w. (detected in Gomphidae) (Table 78). For tissue concentrations to reach the CES, 961%, 180%, and 465% increase in Ametropodidae, Gomphidae, and Pteronarcyidae methylmercury tissue concentrations, respectively, would be required to achieve the desired power (90%). Methylmercury tissue concentrations produced relatively tight estimates, such that three benthic community samples would be required to have the likelihood of detecting benthic tissue burdens reaching the CES.

Of the taxonomic families analyzed for selenium concentrations, Gomphidae were the most variable within Sites resulting in a relatively large number of samples (72) per Site to have desired power for detecting tissue concentrations reaching the CES (Table 78). However, selenium concentrations detected in Ametropodidae and Pteronarcyidae were less variable within Sites and can be anticipated to have the desired power to detect more subtle effects within the benthic community. Six benthic community samples containing Ametropodidae and Pteronarcyidae would be required to have the desired power for detecting benthic tissue burdens reaching the CES. Larger changes in concentration could be detected with even fewer samples of benthos tissue per Site.

Table 78 Results of power analysis for methyl mercury and selenium fish tissue guideline for the protection of aquatic life.

Variable	Units	Taxonomic Family	log SD	Reference		Tissue Guideline	% Change Increase at Guideline	½ Way to Guideline	log of ½ Way Point (CES)	N to Detect CES
				Value	log Value					
Methylmercury	ng/g w.w.	Ametropodidae	0.050	1.63	0.213	33	1921	17.3	1.24	2
		Gomphidae	0.084	7.17	0.856	33	360	20.1	1.30	2
		Pteronarcyidae	0.173	3.20	0.506	33	930	18.1	1.26	3
Selenium	µg/g d.w.	Ametropodidae	0.078	1.85	0.268	4	116	2.93	0.466	4
		Gomphidae	0.222	2.54	0.406	4	57	3.27	0.515	72
		Pteronarcyidae	0.181	1.16	0.412	4	243	2.58	0.412	6

Table Notes: SD = standard deviation, log = logarithm to the base 10, CES = critical effect size

4.2.3.2 $\delta^{15}N$

Power calculations were used to calculate sample size requirements to achieve 90% likelihood (desired power) of detecting changes of various magnitudes in benthic invertebrate $\delta^{15}N$ tissue content at specific magnitudes. $\delta^{15}N$ can be used in conjunction with $\delta^{13}C$ to determine shifts in food-web compartments and trophic levels. Table 79 provides the MSE terms for $\delta^{15}N$ tissue content derived from the GLMs used to explore potential sources of variation in measured $\delta^{15}N$ tissue content in benthic invertebrates. The within-area/time standard deviation were estimated from the MSE terms (i.e., $SD = \sqrt{MSE}$).

The $\delta^{15}N$ was more variable in Pteronarcyidae than the other families. For Ametrophididae, Gomphidae, Pteronarcyidae and Chironomidae, between 2 and 10 samples per Site would be sufficient to have the desired power (90% likelihood) of detecting changes equivalent to 50% of the baseline (Figure 122). Considerably more samples of Pteronarcyidae per Site would be required for the same power.

Table 79 $\delta^{15}N$ detected in benthic invertebrates and the standard deviations used in power calculations.

Variable	Taxonomic Family	Upstream Reference Mean (‰)	Standard Deviation
$\delta^{15}N$	Ametropodidae	7.1	0.073
	Gomphidae	9.5	0.094
	Pteronarcyidae	6.5	0.213
	Chironomidae	10.2	0.019

Table Notes: Normal Ranges (NR) were calculated as the estimate in real units ± 2SD, LL and UL represent the upper and lower level, respectively, of the normal range.

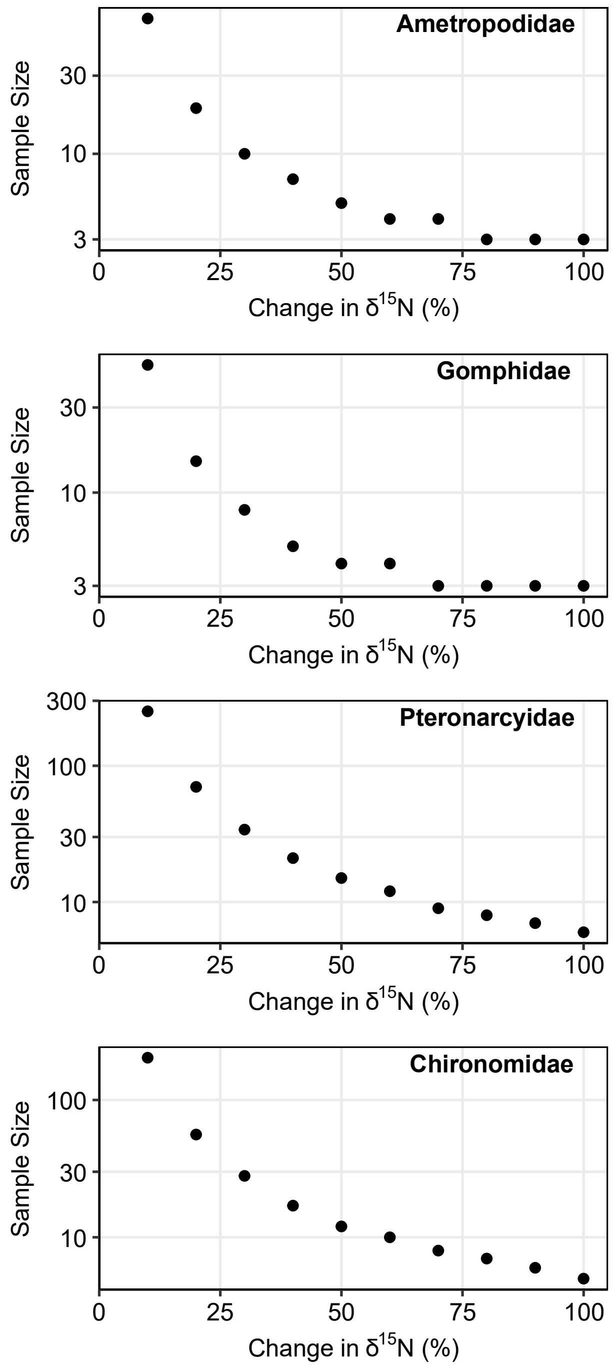

Figure 122 Power curve illustrating the number of samples required to detect changes in $\delta^{15}N$ of specific magnitudes for Ametropodidae, Gomphidae, Pteronarcyidae, and Chironomidae.

4.2.4 Fish Tissues

4.2.4.1 Total Mercury and Methylmercury

Power calculations were used to determine the sample sizes required to achieve 90% likelihood of detecting changes in mercury content to the CES for mercury tissue concentrations in Walleye and White Sucker, and in mercury and methylmercury tissue concentrations in Trout-perch. The CES in this case is the ½ way point between the upstream mean mercury and methylmercury tissue concentrations and the tissue guidelines for the protection of human health (in the case of total mercury) and aquatic life (in the case of methylmercury). Walleye and White Suckers are important species to monitor as they can be consumed for subsistence by the local residence, especially Indigenous communities, while Trout-perch is also widely distributed in North American waters and are an important source of food for other species in their habitat. The Canadian guideline for food safety recommends that the total concentration of mercury in the edible portion of commercial fish must not exceed 500 ng/g w.w. (Environment Canada, 2001) while the Canadian tissue residue guideline for methylmercury for the protection of aquatic biota is set at 33 ng/g w.w. (Canadian Council of Ministers of the Environment, 2000). Table 80 provides the MSE terms for total mercury for Walleye and White Sucker as well as methylmercury tissue concentrations for Trout-perch. These MSE values are derived from the sex dependent GLMs used to explore potential sources of variation in measured tissue and body burden concentrations and were used to estimate the within-area/time standard deviation (i.e., $SD = \sqrt{MSE}$).

The upstream reference means for total mercury detected in Walleye and White Sucker tissue ranged from 260 ng/g w.w. (female White Sucker) to 498 ng/g w.w. (male Walleye) while the upstream reference means for methylmercury detected in male Trout-perch was 34.6 ng/g w.w. Since total mercury (detected in Walleye) and methylmercury (detected in Trout-perch) fish tissue concentrations were close or surpassed the guidelines, power curves were developed (Figure 123 and Figure 125) to determine sample size requirements to detect changes at specific magnitudes. Power curves were also developed for White Suckers, but upstream mean tissue concentrations were low enough to determine the sample sizes required to have the desired power for detecting changes in tissue concentrations reaching the CES (Figure 124).

Total mercury concentrations detected in male Walleye were sufficiently variable within Sites resulting in eight fish required to have the desired power for detecting a 100% increase in total mercury tissue concentrations (Figure 123). However, total mercury concentrations detected in female Walleye were less variable within Sites and requires smaller sample sizes (7) to have the desired power for detecting a 50% increase in total mercury tissue concentrations. Total mercury concentrations detected in male and female White Suckers also varied considerably within the Site where 68 males and 26 female White Suckers would be required to have the desired power for detecting changes in tissue concentrations reaching the CES (30% and 46% increase in total mercury tissue concentrations in male and female White Suckers, respectively; Figure 124). Methylmercury concentrations detected in the upstream male Trout-perch were relatively high and the power curve revealed that 10 Trout-perch would be required to have the desired power to detect a 50% increase in methylmercury tissue concentrations (Figure 125). Generally, larger changes in concentration could be detected with even fewer samples of fish tissue per Site with the desired power.

Table 80 Total mercury and methylmercury tissue concentrations in fish and standard deviations used in power calculations.

Variable	Species	Sex	Upstream mean (ng/g w.w.)	Standard Deviation
Total Mercury	Walleye	Female	428	0.099
		Male	498	0.193
	White Sucker	Female	260	0.200
		Male	314	0.222
Methylmercury	Trout-perch	Male	34.6	0.125

Table Notes: Standard deviation is expressed in logarithms (base 10)

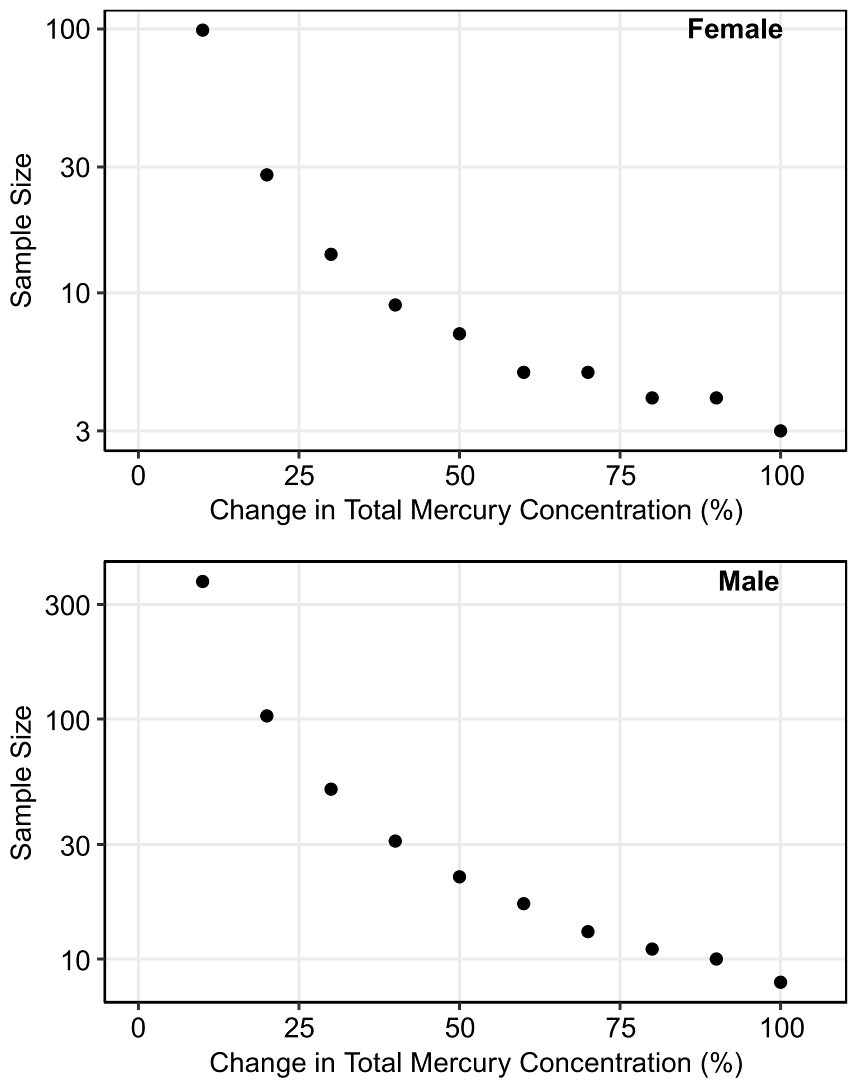

Figure 123 Power curves illustrating the number of samples required to detect changes in Total Mercury concentration in female (top panel) and male (bottom panel) Walleye.

Figure Notes: Mean baseline Total Mercury concentrations in female and male Walleyes are 498 ng/g w.w. and 429 ng/g w.w., respectively.

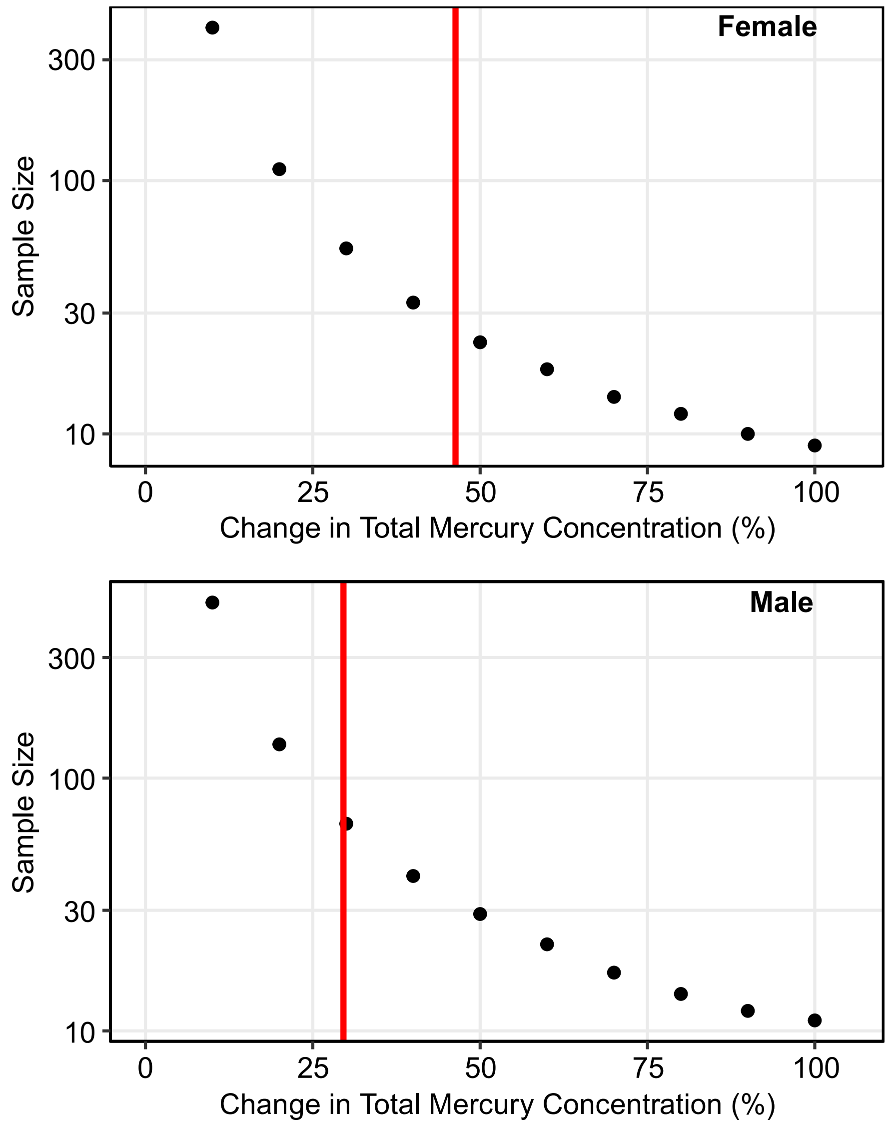

Figure 124 Power curves illustrating the number of samples required to detect changes in Total Mercury concentration in female (top panel) and male (bottom panel) White Suckers.

Figure Notes: Mean baseline Total Mercury concentrations in female and male White Suckers are 314 ng/g w.w. and 259 ng/g w.w., respectively. 68 males and 26 female White Suckers would be required to have sufficient likelihood of detecting when Total Mercury tissue concentrations reach the critical effect size (CES; red vertical line) which is set as the ½ point between mean baseline concentrations and the Health Canada tissue guidelines set at 500 ng/g w.w.

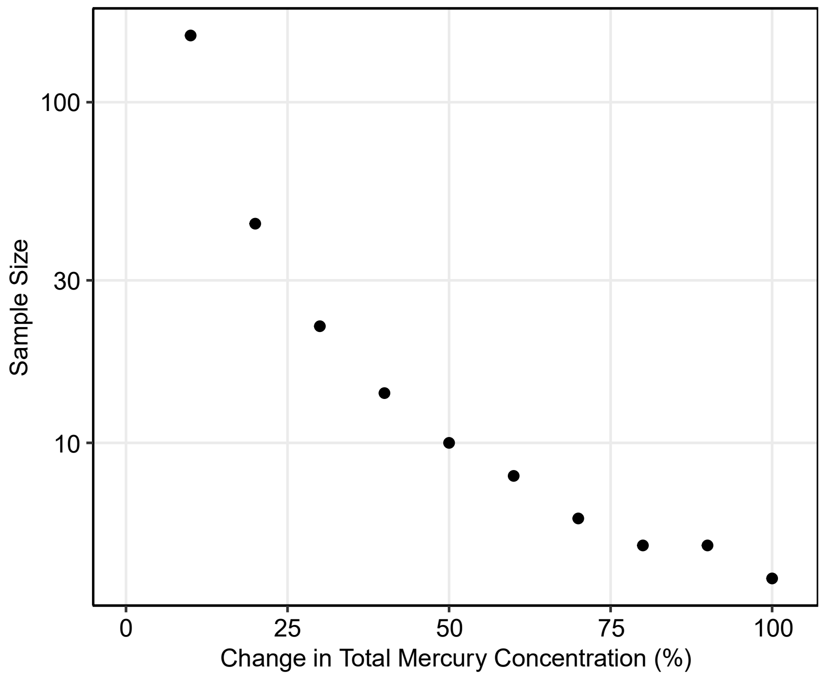

Figure 125 Power curve illustrating the number of samples required to detect changes in Methylmercury concentrations in male Trout-perch.

Figure Notes: Mean baseline methylmercury concentrations in male Trout-perch were 34.5 ng/g w.w.

4.2.4.2 Selenium

For selenium fish tissue burdens, power calculations were used to compute the sample sizes required to achieve 90% likelihood of detecting changes to a CES. The CES in this case is the ½ way point between the upstream mean selenium tissue concentration (ranging from 1.24 µg/g d.w. to 2.38 µg/g d.w. in Walleye, White Sucker, and Trout-perch; Table 81) and the tissue guidelines for the protection of aquatic life which is set at 4 µg/g d.w. (Alberta Environment, 2018). Male Trout-perch had the highest selenium tissue concentrations while female Walleye had the lowest. Table 81 provides the MSE terms for selenium tissue concentrations for Walleye, White Sucker, and Trout-perch. These MSE values are derived from the sex dependent GLMs used to explore potential sources of variation in measured tissue and body burden concentrations and were used to estimate the within-area/time standard deviation (i.e., $SD = \sqrt{MSE}$). The results of the power calculations are also provided in Table 81. The number of fish required to have the desired power for detecting changes in tissue concentrations reaching the CES ranged between 6 (female Walleye) to 14 (male Trout-perch). These estimated sample sizes are generally consistent with what is anticipated given fish population sampling programs elsewhere (Environment Canada, 2012a).

Table 81 Results of power analysis for selenium fish tissue guideline for the protection of aquatic life.

Variable	Species	Sex	log SD	Reference (µg/g d.w.)		Fish Tissue Guideline (µg/g d.w.)	% Change Increase at Guideline	½ to Guideline	log of ½ Point (CES)	N to Detect CES
				Raw	Logarithm					
Selenium	Walleye	F	0.175	1.24	0.0924	4	233	2.61	0.418	6
	Walleye	M	0.179	1.46	0.164	4	174	2.73	0.436	8
	White Sucker	F	0.184	1.33	0.125	4	200	2.67	0.426	12
	White Sucker	M	0.142	1.72	0.236	4	132	2.86	0.457	9
	Trout-perch	M	0.109	2.38	0.377	4	68	3.19	0.504	14

Table Notes: Data was log-transformed (base 10) where indicated.

4.2.4.3 EROD

Power calculations were used to calculate sample size requirements to achieve 90% likelihood (desired power) of detecting changes of various magnitudes in Trout-perch liver EROD activity. The standard deviation of EROD (0.31 and 0.32 pmol/min/mg for male and female Trout-perch, respectively) was calculated from the MSE terms derived from the sex dependent GLMs used to explore potential sources of variation in measured liver EROD activity and used to estimate the within-area/time standard deviation (i.e., $SD = \sqrt{MSE}$). Even if the upstream mean liver EROD activity was similar for both sexes (0.81 pmol/min/mg), significant gender effects have been previously detected in EROD activity, especially during spawning events where activity in reproductively active female fish was significantly suppressed compared to male fish (Schreiber et al., 2006; Whyte et al., 2000). Therefore, power was calculated separately for female and male fish separately. Table 82 provides the upstream reference mean and MSE terms used for the power calculations and power curves were developed (Figure 126), illustrating the number of samples required to detect changes of specific magnitudes. Roughly 50 samples from female and male Trout-perch would be required to detect a 0.1 pmol/min/mg change (or 55% increase) in activity with the desired power. Larger changes in activity could be detected with even fewer samples of Trout-perch liver tissue per Site.

Table 82 Measured EROD levels in Trout-perch and standard deviations used in power calculations.

Variable	Species	Sex	Upstream mean (pmol/min/mg)	Standard Deviation
EROD	Trout-perch	Female	0.81	0.320
		Male	0.81	0.312

Table Notes: Standard deviation is in logarithms (base 10)

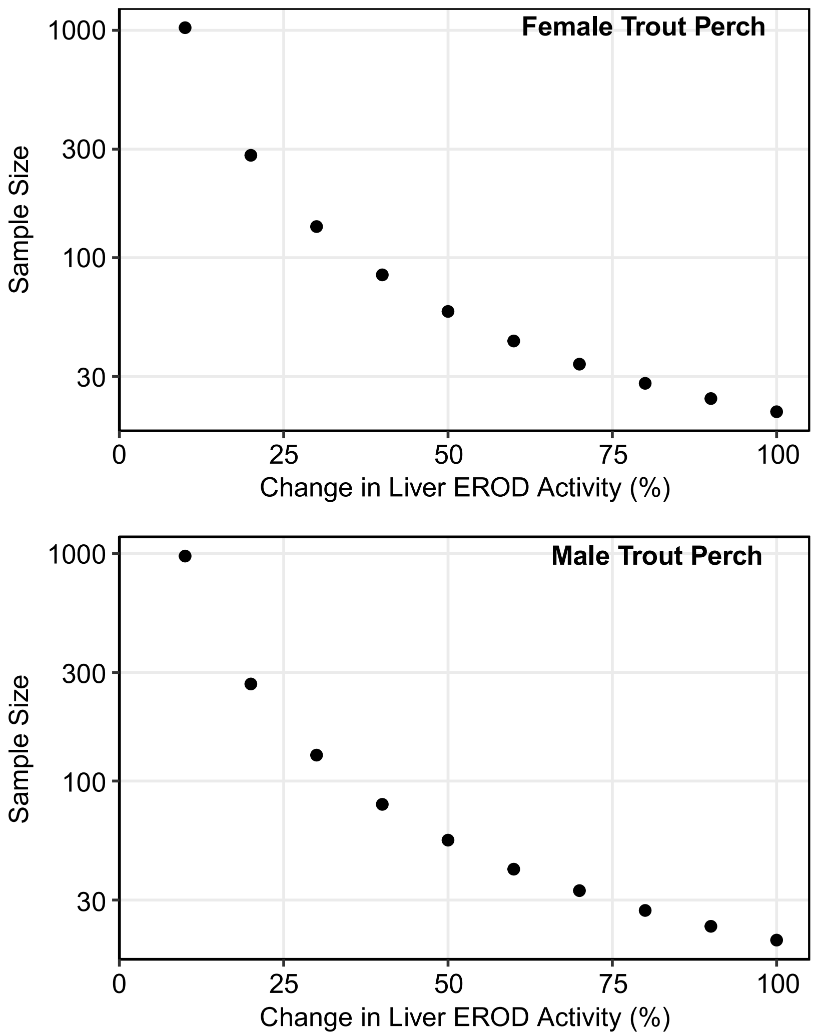

Figure 126 Power curves to detect change in female and male Trout-perch liver EROD activity based on the number of samples collected.

4.2.4.4 PAHs

There are two fish consumption guidelines in place for the protection of Human Health. Fish tissue containing ΣPAH4 (sum of benzo[a]anthracene, chrysene, benzo[b]fluoranthene, and benzo[a]pyrene) concentrations greater than 12 ng/g w.w. should not be consumed while fish tissue having benzo[a]pyrene concentrations higher than 2 ng/g w.w. should be avoided (EU, 2015). It should be noted that these consumption guidelines are not endorsed or adopted by the Government of Canada or the Government of Alberta and are used solely for the purpose Critical Effect Size (CES) calculation only. With the limited number of detectable PAHs in Walley and White Sucker, power calculations could not be conducted to determine the sample size required to have the likelihood of detecting tissue concentrations reaching the CES (½ point between the upstream mean reference and tissue consumption guideline). The average reporting detection limit of benzo[a]pyrene as part of the enhanced monitoring program is 0.032 ± 0.021 ng/g w.w. (±SD, n= 124), thus, a 6,150% increase in fish tissue concentrations would be required to reach the benzo[a]pyrene consumption guidelines if benzo[a]pyrene concentrations in Walleye and White Sucker tissues were at the detection limit. Trout-perch are not likely to be consumed by humans so power analysis to determine sample sizes required to detect the likelihood of Trout-perch tissue concentrations to reach the consumption guidelines were not conducted.

4.2.4.5 $\delta^{13}C$ and $\delta^{15}N$ Isotope Ratios

Power calculations were used to calculate sample size requirements to achieve 90% likelihood (desired power) of detecting changes of various magnitudes in $\delta^{13}C$ (Figure 127) and $\delta^{15}N$ (Figure 128) tissue concentrations in male and female Walleye, White Sucker, and Trout-perch. The upstream reference means, and the within-area/time standard deviations used for the development of the power curves are presented in Table 83. The standard deviations were calculated from the MSE terms (i.e., $SD = \sqrt{MSE}$) derived from the sex dependent GLMs used to explore potential sources of variation in measured tissue concentrations.

$\delta^{13}C$ detected in male and female fish did not vary considerably within Sites and a low number of samples (2) is required to have the desired power to detect anything from 5% to a 100% change in $\delta^{13}C$. $\delta^{15}N$ was modestly more variable within Sites but still can be anticipated to be sufficiently powerful to detect subtle effects. Four samples would be required to have the desire power to detect a 30% increase in $\delta^{15}N$ concentrations in fish tissue.

Table 83 Stable isotope levels in fish and standard deviations used in power calculations.

Variable	Species	Sex	Upstream mean (‰)	Standard Deviation
$\delta^{13}C$	Walleye	Female	-27.8	0.025
		Male	-26.5	0.027
	White Sucker	Female	-28.3	0.024
		Male	-28.4	0.023
	Trout-perch	Female	-26.8	0.013
		Male	-26.9	0.011
$\delta^{15}N$	Walleye	Female	10.8	0.025
		Male	11.0	0.027
	White Sucker	Female	8.6	0.046
		Male	9.1	0.034
	Trout-perch	Female	9.6	0.016
		Male	9.4	0.034

Table Notes: Standard deviation is in logarithms (base 10)

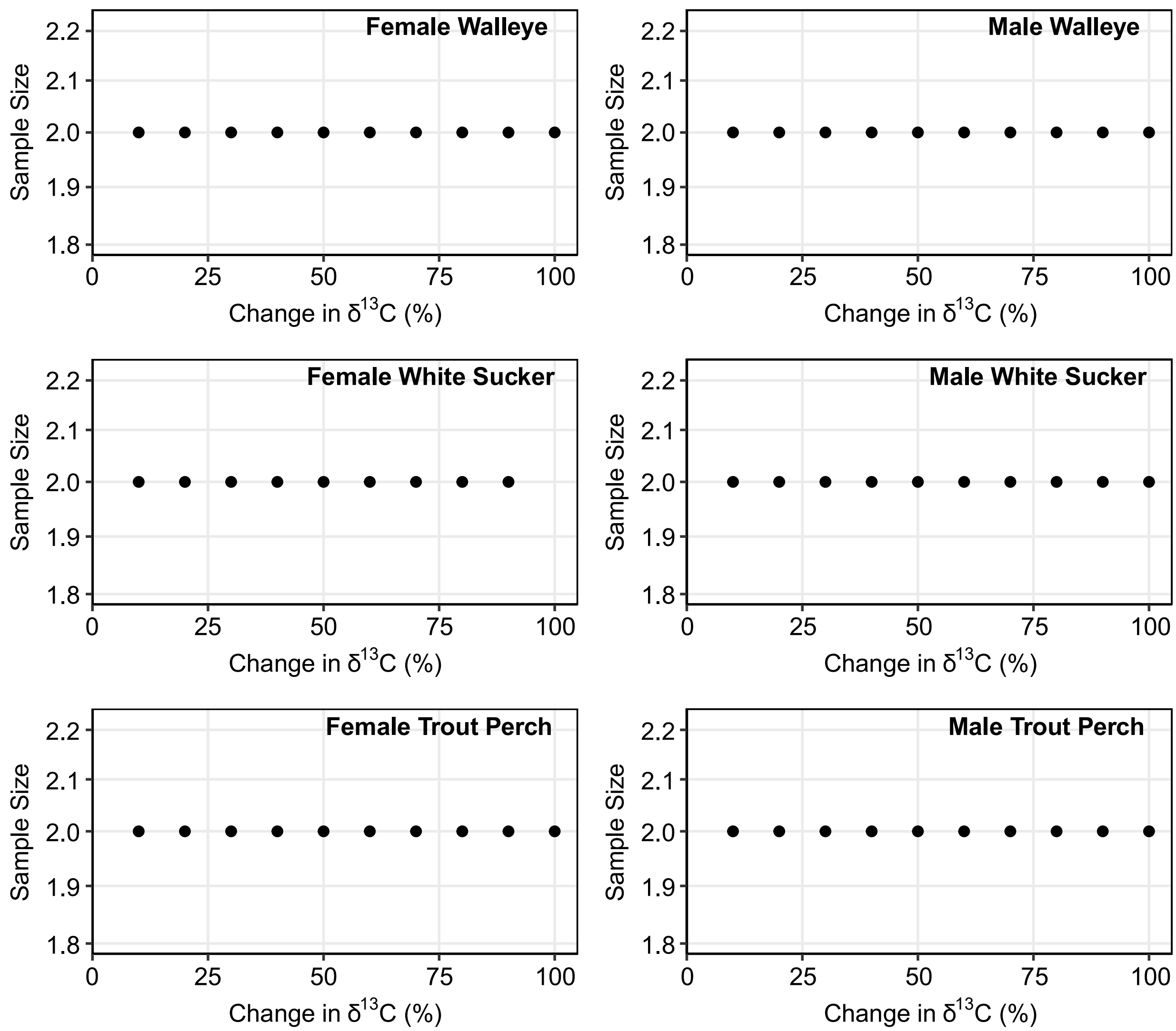

Figure 127 Power curves illustrating the number of samples required to detect the percent changes in $\delta^{13}C$ in Walleye (top panels), White Sucker (middle panels), and Trout-perch (bottom panels).

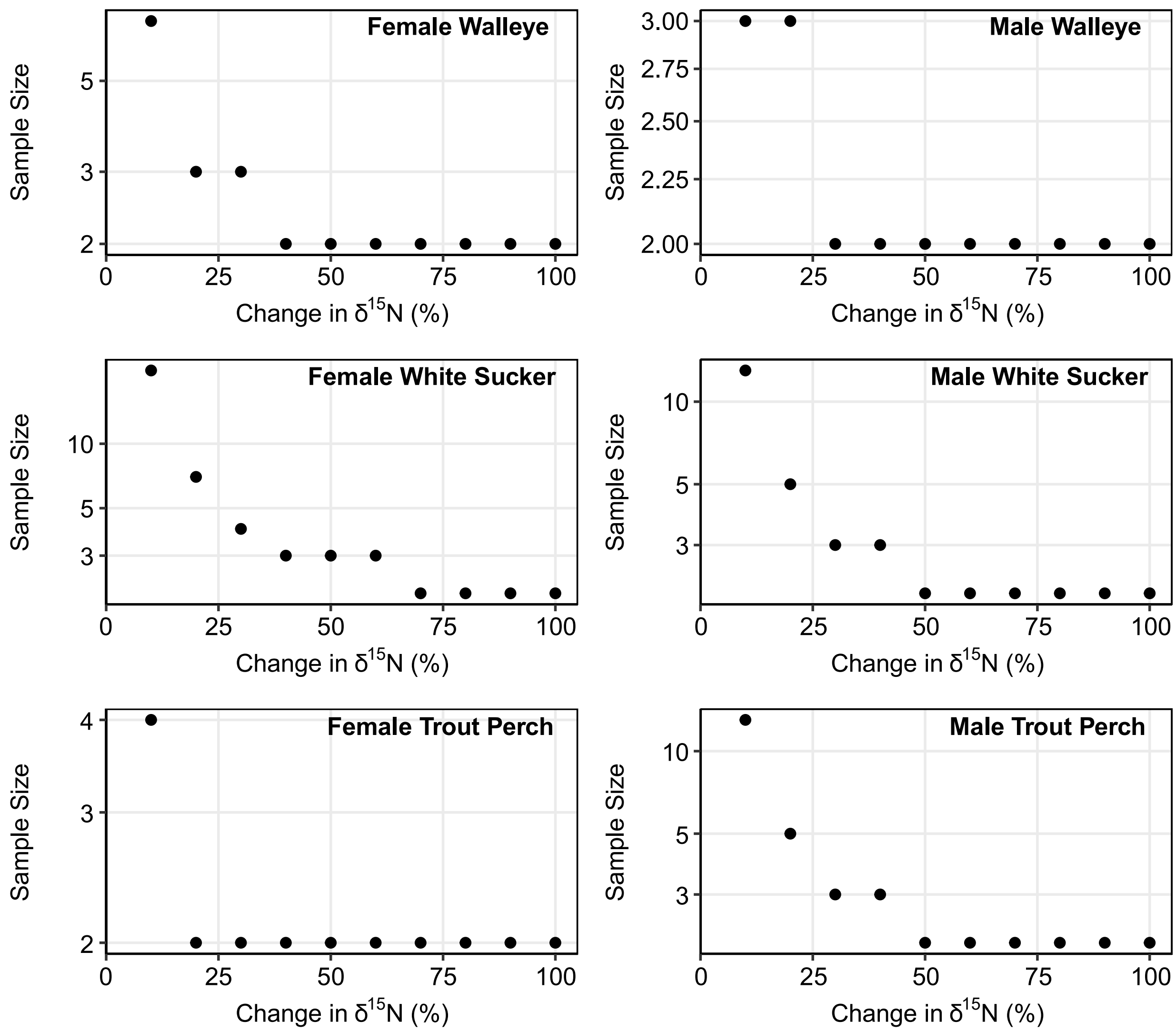

Figure 128 Power curves illustrating the number of samples required to detect the percent changes in $\delta^{15}N$ in Walleye (top panels), White Sucker (middle panels), and Trout-perch (bottom panels)

5.0 TASK 3 (PART III): STUDY DESIGN GUIDANCE

The purpose of this section is to provide general guidance to AEPA for the future design of EEM programs for individual releases of treated oil sands process water. The sub-sections below provide:

(1) a general rationale for EEM;

(2) recommendations for measured endpoints;

(3) a discussion of EEM tiers that can be applied in an adaptive context;

(4) suggestions for statistical design;

(5) suggested thresholds that can be applied to the various endpoints to trigger changes in the EEM tier.

Recommendations herein are derived from consideration of published literature, as well as the results of the various statistical analyses of the EMP and OSMP data sets in the previous sections. The recommendations here are intended to be specific to the potential application of EEM for assessing point-source releases of oil sands process water to the mainstem of the Athabasca River.

5.1 EEM Rationale

Regulated effluents are not permitted to be released to surface waters unless effects in the receiving environment are predictable and acceptable. For example, the MDMER permit the release of complex mine effluents, so long as the effluents achieve a minimum standard of quality (specified by limits for specific metals and other constituents such as BOD and total suspended solids) that ensure a minimum of environmental protection. However, mining effluents can contain more than metals, such as various polymers (associated with effluent treatment, or other operational processes), or other constituents posing risks (potentially not well understood) to aquatic organisms. As such, the environmental outcomes of released mine effluents have residual uncertainty. EEM is a process for validating effects predictions and testing for unexpected effects (Somers et al., 2018). Observed receiving-environment effects that are consistent with predictions demonstrate that we have sufficient understanding of (1) the constituents in the effluent, and (2) receiving-environment conditions including tolerances of biological receptors. Observed receiving-environment effects that differ from predictions indicate insufficient understanding of (1) effluent constituents, and/or (2) receiving-environment conditions including tolerances of biological receptors. In this context, EEM informs adaptive management of effluent quality (Somers et al., 2018).

The purpose of this overall assignment has been to support the development of an EEM program to assess potential releases of treated oil sands process waters. Alberta will not permit the release of oil sands process waters without compelling evidence that the various constituents in the waters will pose low and acceptable risks of biological effects in the receiving environment. EEM is the proposed mechanism for verifying predictions and testing for unexpected effects.

5.2 Tiers in Adaptive Monitoring

There are several peer reviewed papers and government reports laying out the adaptive monitoring process (Arciszewski, Munkittrick, Scrimgeour, et al., 2017b; Cairns et al., 1993; Environment Canada, 2012b; Hatfield Consultants, 2022; Hodson et al., 1996; Kilgour & Associates Ltd., 2022; Somers et al., 2018). In general, EEM contains the following steps:

1. Routine monitoring;
2. Confirmation monitoring;
3. Investigation of Cause.

Routine monitoring is designed to verify the predictions associated with the discharge, which may be (Somers et al., 2018):

1. no change relative to the normal range of baseline conditions (for biological receptors);
2. changes in chemistry not exceeding what is predicted from mass-balance or other models.

When routine monitoring verifies predictions, routine monitoring at the same or a reduced schedule is justified. When routine monitoring detects unexpected change, confirmation monitoring or investigation of cause (IOC) monitoring is potentially justified. Transitioning from routine to confirmation or an IOC would be a potential recommendation from a monitoring committee, that would be considered by a management committee (or regulator) (Kilgour & Associates Ltd., 2022). It will invariably be a management committee (or regulator) that dictates the next tier in monitoring, as is the situation in Canadian pulp & paper and metal mining EEM programs (Hatfield Consultants, 2022).

All the variables used in the EMP (and listed in Table 75) can be used in an adaptive EEM process to trigger changes in monitoring tiers.

5.3 Thresholds Triggering Adaptive Monitoring

Adaptive monitoring programs require numeric (or narrative) thresholds that trigger changes in monitoring. Somers et al. (2018) proposed an approach that involved the use of monitoring, forecast and management triggers. Here, the term ‘threshold’ will be used to express the numeric or narrative value (level) which when met or exceeded will trigger a change in monitoring. Figure 129, presented below, is from (Kilgour & Associates Ltd., 2022) and illustrates various thresholds that could be built into a monitoring program to trigger different follow-up actions. In the example, we are using concentration of selenium in water. There are water quality guidelines for selenium in water (Beatty & Russo, 2014). There may also be Site-specifically derived benchmarks used to anticipate deleterious effects in fish (Teck, 2014). The baseline threshold is a representation of the historical background or reference condition, which would be derived from historical or present-day data. Given installation of a proposed engineered facility, it is reasonable to assume that concentrations in water could be predicted for a future operational condition (see Teck, 2014, for an example). Ideally, the engineered facility will be designed to ensure that concentrations in the receiving environment (i.e., forecast threshold) are < guidelines for the protection of aquatic life (Figure 129).

In the subsections below, we discuss how (1) baseline thresholds (normal ranges), (2) forecast thresholds, and (3) environmental quality guidelines can be derived and used in an adaptive monitoring program.

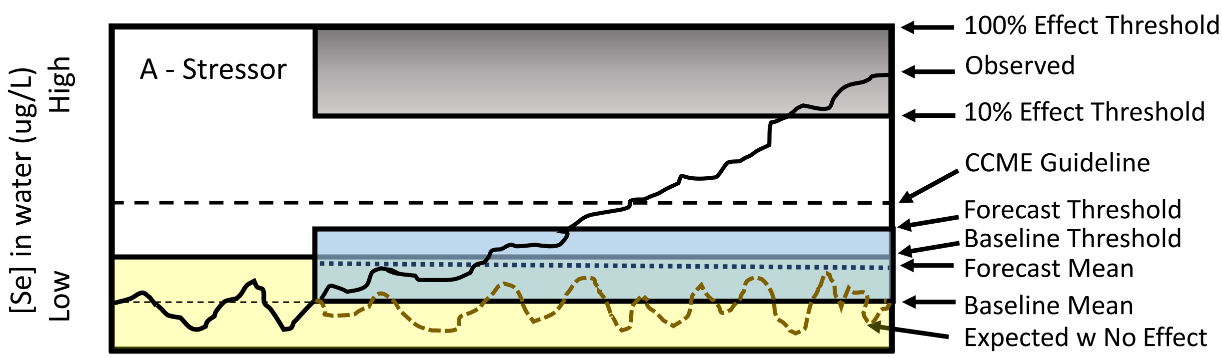

Figure 129 Schematic illustration of various thresholds for triggering changes in monitoring.

5.3.1 Baseline Thresholds (Normal Ranges)

5.3.1.1 General Description

Here, baseline thresholds are limits on normal ranges for measured chemical and/or biological responses. Normal ranges can be computed for Site-specific conditions, or regionally. These variations are discussed below.

5.3.1.2 Site-Specific Normal Ranges

As it relates to normal baseline ranges, there are multiple options for how they can be derived. Normal ranges are typically constructed (estimated) based on reference data, where the reference data are from an upstream reference Site or are from a historical baseline period prior to the engineered facility operating. Normal ranges can be simply estimated as:

$$\text{Site mean} \pm 2 \text{ x } SDs,$$

where, the Site mean will be the actual measured (or flow adjusted estimated) mean condition for a Site at a specific time, and the SD in this case is based on the variation among replicates within sites (S^2_w). The value *2* is rounded up from 1.96 (or the standard normal deviate for a 95% region under a normal curve; (Kilgour et al., 1998b, 2017b). The use of “mean ±2*SD*s” provides an estimate of the normal range limits. With small sample size, these estimated limits can be substantially inaccurate resulting in inflated Type I error rates (concluding that samples from a Site are in a degraded condition when they are not).

Tolerance limits are an alternative approach to estimating normal range values when sample sizes are low and can improve Type 1 errors. Tolerance limits are equal to:

$$\text{Site mean} \pm k \text{ x } SDs, \qquad [1]$$

where k is a tabled value that depends on number of reference observations: smaller samples sizes require a larger k, for example (R. W. Smith, 2002). Tolerance limits represent the range of values for which there is some likelihood (say 95%) that a new sample will fall within, if the sample has a value that actually falls within the reference condition.

Site-specific normal ranges can reflect either historical pristine conditions, or recent degraded conditions, and care should be taken to be clear what they are intended to be. Hatfield Consultants (2022) (and see Figure 130) illustrates the conundrum when upstream reference sites are in a degraded condition. If we use a degraded reference normal range as a threshold indicating acceptable levels of change, monitoring runs the risk of accepting additional (cumulative) levels of degradation.

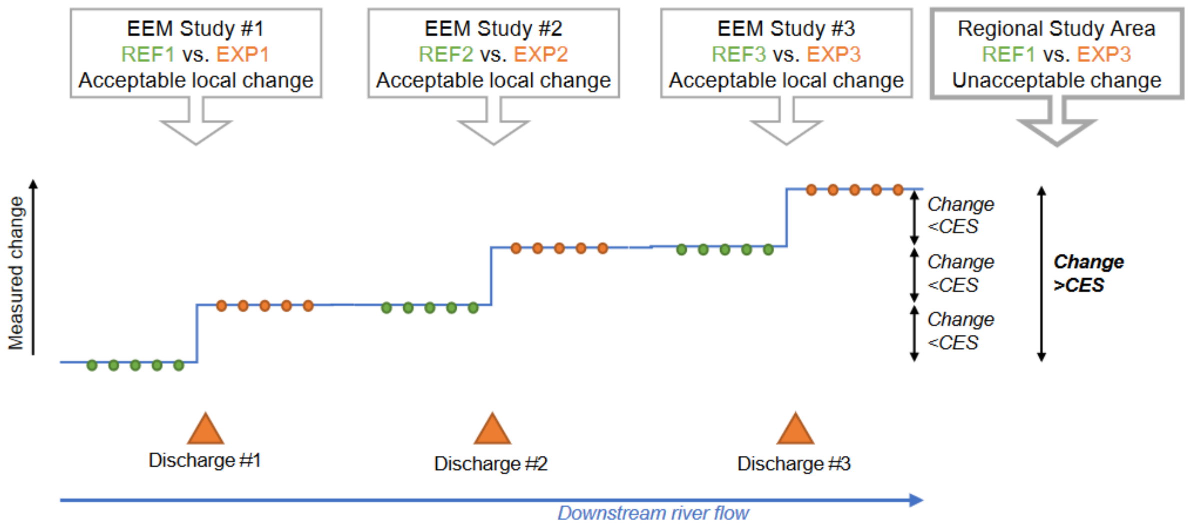

Figure 130 Schematic illustrating the potential consequence of a sliding baseline in environmental effects monitoring.

Figure Notes: Adapted from Figure 3 in (Hatfield Consultants, 2022).

5.3.1.3 Regional Normal Ranges

Tolerance limits are typically estimated using a SD derived from the among-sample variance term, and where the samples are taken from one reference Site at one time (per Environment Canada, 2012). However, there is increasing recognition that there are normal, and random, variations among reference areas and times. Recognition of this additional random variability can arguably (Kilgour et al., 2017b) improve interpretation of the significance of Site-specific differences between reference and exposure conditions.

The analyses in Section 3 of this report clearly illustrated that, even after considering the effects of discharge, there remains significant variability in algal, benthic and fish responses that are related to sampling year and location. Those year and location variances are probably random and can arguably (Kilgour et al., 2017b; Kilgour & Somers, 2017) be included in an estimate of the normal range of reference/baseline conditions.

Smith (2002) described multiple methods for quantifying tolerance limits 'random-effects' models. That is, he described methods for incorporating random variability due to time and location into estimates of tolerance limits, wherein this case time and location are deemed "random effects". Regional normal ranges could be estimated using a regional standard deviation ($SD_{regional}$) that includes not only variation among samples within sites (i.e., S^2_w), but also includes random variation among sites (S^2_A). The appropriate term for computing a regional normal range then would be based on the within and among variance terms:

$$SD_{regional} = \sqrt{S_A^2 + S_W^2} \qquad [2]$$

The $SD_{regional}$ term could also include a temporal component if it was deemed significant, and random. The $SD_{regional}$ term can be incorporated into equations 1 or 2 above. If included in equation 2, the value k would be 'estimated' using computational or bootstrapping (simulation) methods as described by Smith (2002) and others.

5.3.1.4 Layering of Normal Ranges

As above, there are multiple 'normal' ranges that can be computed for the assessment of biological, physical or chemical responses. Site-specific normal ranges would apply to the Site being assessed. The Site-specific normal range can be derived for a baseline period condition that may reflect pristine, or maybe least-impaired conditions. The OSMP and EMP data presented in this report do not reflect a pristine condition, because the LAR has been under the influence of the town of Fort McMurray, oil sands operations, and other anthropogenic influences (e.g. pulp and paper mills). The OSMP and EMP data examined here, however, may reflect a minimally impaired condition. There is limited indication that fish populations or benthic communities are under undue stress. As such, the biological, physical and chemical conditions in the Athabasca River today (i.e., from 2009 to 2021) may be deemed a minimally impaired, appropriate baseline condition.

Going into the future, it may be 10 (or more) years before OSPW is licensed for release to the LAR. In the intervening 10 (or more) years, there is the potential for changes in chemical, physical and biological conditions in the mainstem of the river. As such, the Site-specific and regional normal ranges for this mainstem of the LAR may shift as illustrated in Figure 130. There are, therefore, multiple normal ranges that may be considered, and layered, for the purpose of assessing variations in the various EEM components, an illustration of which is provided below. Site-specific normal ranges could be computed from upstream control/reference Sites. There may be a shift in condition from 2020 to 2030 resulting from influences upstream of the potential OSPW release point. The % EPT in the exposure Site in 2030 may be impaired (outside of the normal range from a baseline period, i.e., 2020), but within the upstream reference normal range based on 2030 data. The conclusion would be that the exposure Site is impaired relative to a baseline period normal range. In the example illustrated there is no obvious difference in %EPT between reference and exposure Sites, and so the impairment in the exposure Site in 2030 could not be associated with the release of OSPW.

This notion of layering of normal ranges has been implemented in the Elk Valley in British Columbia. Teck Coal has been monitoring benthic invertebrates in the Elk River downstream of five coal mines (Environmental Monitoring Committee, 2022). Models have been developed that relate variations in benthic community indices to natural factors (e.g., substrate texture) and to mining-related factors (i.e., water quality). Those models are used to estimate multiple Site-specific normal ranges: (1) natural, nonmining related; and (2) mining influenced. The natural normal range is the range of values expected in the absence of mining. The mining-related normal ranges are expected, given that there is mining. Both normal ranges can be further put into context with a regional normal range, which is the range of variation for regional reference sites.

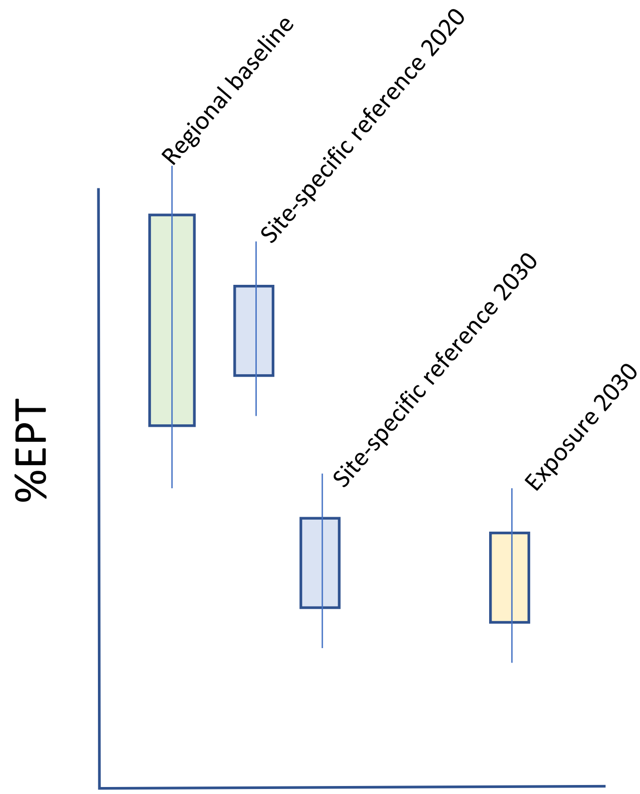

Figure 131 Schematic illustrating potential 'layering' of normal ranges.

Figure Notes: In this schematic, the Site-specific reference is a Site upstream of OSPW release, and the Exposure is a Site within the mixing zone of OSPW release.

5.3.1.5 Sample Size Implications

The main issue with small sample sizes is error in estimating the mean and variance. As sample size increases, the accuracy of the parameter estimates increases. With limited data sets, tolerance limits can be used to provide confidence limits on the normal range (Kilgour et al., 2017b). Munkittrick (1992) demonstrated that variance estimates for fish population variables tends to stabilize at about 16 fish. That conclusion on sample size reflected normal statistical process: that is, as sample sizes approach n=10 to 20, variances tend to stabilize (see for example how Student's *t* values change with degrees of freedom; they tend to asymptote at between 10 and 20 df). Our ability to estimate random noise within Sites is improved when there are multiple Sites (reference and exposure) and times (say multiple times before and after release of a treated effluent). That is, data from other Sites and times will also have 'within-Site/time' variance, which can support a 'pooled' estimate of variance. So, even if there is a more limited data set (for estimating variance terms) in the first year of a program, data from subsequent years can be used to augment the degrees of freedom associated with the normal range. In that sense, the historical data from OSMP and the data collected under this EMP will be valuable for estimating within-Site/time variance. Further, considering that the EMP has approximately 10 sites x 3 years = 30 site/years of data, the program arguably has sufficient data (now historically) to support estimating among-site variance terms.

5.3.2 Forecast Thresholds

Forecast thresholds are numeric or narrative statements that indicate the level of change anticipated given that an engineered facility (or another project) is about to influence a receiving environment. Per Somers et al., (2018), on the basis of facility operation knowledge, emissions (effluent) quality, and dilution ratios in a receiving environment, it is normally an easy exercise to predict (using mass-balance models) the expected outcomes (concentrations) in the receiver. The predicted concentrations can be used as a Forecast Threshold, that would then be assessed via monitoring. As with any forecast, there will be uncertainty in the estimate: that is the forecast concentrations will have an associated confidence that should be incorporated into the forecast.

The forecast concentrations will typically be below some guideline or other benchmark that agencies are using to manage the system, because the agencies will enforce effluent limits to achieve receiving-environment quality.

Four Elements Consulting (2022) recently used mass-balance models to estimate the likely concentrations of major ions and nutrients in surface water and sediment of the Athabasca River resulting from the release of OSPW. That exercise exemplifies the notion of Forecast Thresholds. If release of OSPW to the Athabasca River is permitted, it is recommended that mass-balance models be developed (facility specifically) to estimate concentrations to be expected in the receiver. Those concentrations would then be used to evaluate monitoring data.

Within adaptive monitoring programs, it is also important to have predictions for biological responses. With complex effluents, however, it can be challenging to make formally testable predictions. For example, biological impacts associated with release of treated effluent at AREVA's McClean Lake Operations are narrative. Significant impacts to benthic communities are predicted (in the environmental assessment) and are associated with the release of treated uranium mining effluent, while non-significant impacts are anticipated in the third receiver (McClean Lake) (Canadian Nuclear Safety Commission et al., 2012). Regular (routine) monitoring has confirmed that impacts on the benthic community are significant in the first two receivers (Sink Reservoir, Vulture Lake), and within background normal ranges in McClean Lake East Basin (AREVA Resources Canada Inc. et al., 2009). Effects in McClean Lake East Basin that are within the background noise of reference conditions are considered consistent with the predictions from the environmental assessment. Effects in McClean Lake East Basin that exceed background variability would trigger additional follow-up.

In Canada's pulp & paper and metal and diamond mining EEM programs, changes in monitoring tiers are triggered by confirmed effects that exceed CESs. For benthic communities, variations in indices of composition in exposure areas that fall outside the range of values defined by the reference area mean ±2SD trigger confirmation (or investigation of cause if the effect is already confirmed). For variations in sentinel fish populations, exposure-area mean liver size or gonad size values that fall outside the range of values defined by the reference area mean ± 25% are considered large, justifying confirmation or investigation of cause: for condition factor, exposure areas mean falling outside of the reference mean ±10% are considered large enough to trigger confirmation or IOC.

For benthic community indices, values falling outside of the reference data normal range may be one threshold to consider, per the classic Canadian EEM (Environment Canada, 2012b). However, spatial extent of effects in benthic community variables can also be incorporated as thresholds. Green (1979) recommends that the spatial extent of effects is an important consideration when assessing the significance of effects on benthic (and other) communities. Spatial extent was a factor in the predictions provided in the McClean Lake environmental assessment predictions (Canadian Nuclear Safety Commission et al., 2012). In another example, DeBlois et al. (2014) predicted degraded benthic communities within a 1-km radius out from offshore oil drilling platforms. Follow-up monitoring was designed to test those predictions (DeBlois et al., 2014).

5.4 Design Considerations

The purpose of this section is to consider the results of Sections 2.0 and 3.0 and how they instruct a future EEM program designed to assess the effects of released, treated OSPW. The sections that follow address design considerations for benthic algae communities, benthic invertebrate communities, sentinel fish populations, water and sediment quality, and tissues of benthic invertebrates and fish. Within each subsection, we provide some general commentary on the data that were available, sources of variability, recommended sample sizes, thresholds that could be used to trigger changes in monitoring tiers, and a comment as to the potential utility of historical OSMP data for interpreting EMP data.

5.4.1 Benthic Algae Communities

5.4.1.1 General Comments

Algae communities were sampled in the fall of 2018, 2019 and 2021 from 13 stations (section 2.2.6 Table 16). Algae communities were dominated by diatoms such as Bacillariaceae (37%), Tabellariaceae (17%), Stephanodiscaceae (9%), Naviculaceae (8%), Achnanthaceae (7%), and Fragilariaceae (7%). Other than diatoms, non-rare algal groups included the cyanobacteria Chroococcaceae (3%), Oscillatoriaceae (0.6%), and Pseudanabaenaceae (0.5%). The relative densities of dominant taxa were illustrated in section 2.2.6 Figure 31 for stations located upstream and downstream of the proposed OSPW discharge point.

5.4.1.2 Sources of Background Variability

River discharge was consistently the most significant source of variability in variations in algal community measures (Table 19). Algal cell density, taxonomic richness, diversity, biomass, and chlorophyll a were all lower when flows were higher, while taxonomic evenness was higher in response to higher flows (Table 19). After removing the effects of river discharge there remained significant variation in algal community measures among years and spatially. Cell densities and richness decreased from 2018 to 2021 (Figure 33 and Figure 34), while evenness and chlorophyll-*a* biomass increased over the same period (Figure 36 and Figure 37). Finally, densities and total algal biomass decreased from upstream to downstream (Figure 33 and Figure 38), while diversity and evenness increased from upstream to downstream (Figure 35 and Figure 36). This data indicate that multiple data from multiple reference Sites over multiple years would be required in order to characterize baseline variability in algal community composition. Another important consideration is the proximity of the upstream sites to the Fort McMurray (FMM) Wastewater Treatment Plant (WWTP) outfall, which discharges along the west side of the LAR roughly 20km further upstream than the most upstream algal sampling site in EMP. As was described in Section 2.3.2.3.2, there were instances where elevated levels of Ca, Mg, SO_4, Mo, and alkalinity in surface water grab samples on the west side of the LAR suggested an effect of the upstream FMM WWTP outfall (located on the west bank of the LAR) impacting downstream water quality. The outfall is located roughly 10km further upstream than the most upstream benthic algae sampling site (25 km upstream of proposed OSPW discharge point). Therefore, it is possible that inputs from the WWTP may be driving the decreasing density and biomass and increasing evenness and diversity from upstream to downstream in EMP. Regardless, further study would be required to isolate the source trends in benthic algae data.

5.4.1.3 Sample Sizes

Some of the measures of algal community composition were sufficiently noisy that relatively large differences would be required before there would be sufficient likelihood of detection. Those variables include density, richness, and chlorophyll-*a* (Figure 120). For those variables, more than 10 samples per Site were estimated as being required to detect changes of ± 100%. However, other compositional variables had greater sensitivities, more consistent with what would be acceptable in an EEM program. Between about 5 and 10 replicate samples per Site was estimated in Section 4.0 to have sufficient power to detect reasonably small changes in NMDS scores, i.e., a multivariate descriptor. As discussed in Chapter 4, the ability to detect compositional changes will be greater than indicated in the analyses, because the calibration data in the EMP does not contain algal communities from degraded conditions. The program would require only n=4 samples to detect changes in benthic chlorophyll-*a* equivalent to the suggested trigger of 50 mg/m^2. A sampling program, therefore, with between 5 and 10 samples per Site can be anticipated to have sufficient statistical power to detect interpretable changes.

Sampling of benthic algal communities should be synoptic with benthic samples to maximize utility.

5.4.1.4 Thresholds to Trigger Changes in Monitoring

Two thresholds are possible for benthic algae data: (1) guideline; and (2) normal range. Nordin (2001) provide a guideline of 100 mg/m^2 for benthic chlorophyll *a*. Levels higher than that guideline can be anticipated to co-occur with degraded water quality and fish habitat. Normal ranges (local and regional) can be computed for all indices of benthic community composition.

5.4.1.5 Comparison of Monitoring Programs: EMP and OSMP

There was no OSMP benthic algae community data provided on which to derive a regional model. A regional model was therefore constructed using the EMP data only. The decision as to whether algae should be included as an indicator in the regional OSMP must be based on formal consideration of a conceptual site model. Algae are known indicators of the degree of deterioration of water quality within aquatic systems and can be used, along with other indicator, to assess environmental status (Omar, 2010). If decision-makers deem changes in algae communities as a valued ecosystem component (VEC) response along the adverse outcome pathway, then it should be considered in other core monitoring programs such as OSMP.

5.4.2 Benthic Communities

5.4.2.1 General Comments

The EMP used the CABIN protocol for collecting and processing benthic community samples. The sampling process involves a three-minute traveling kick. The methodology was originally derived for sampling in cobble-based lotic habitats and was never intended for use in sand or muck-bottomed habitats. The sampling technique, however, does an excellent job of characterizing the benthic community in a stretch of river. The EMP program in this case, however, sampled benthic habitats with sediments that were principally very fine-grained sand/silt/clay. The OSMP in contrast sampled coarser sands. The benthic communities collected by both programs were significantly different, because of the underlying sediments that the samples were representing. In Section 3.3.4, the OSMP data were demonstrated to be of little value in predicting benthos associated EMP sites. However, the OSMP data were only unable to predict benthos in EMP sites because of the lack of sites with similar fine-grained sediments. Going forward, the combined EMP/OSMP data set has the potential to be used to support the interpretation of CABIN samples collected as part of EEM programs assessing OSPW release.

5.4.2.2 Sources of Background Variability

Sediment grain size, as in the paragraph above, is a significant source of variability in indices of benthic community composition in the Lower Athabasca River. The influence of grain size was observed when considering the EMP data alone (Table 22). As such, any EEM using benthos in the Lower Athabasca River will need to be sufficiently designed to consider grain size as a covariable. It is expected that for future sampling, the receiving environment will likely be a mixture of depositional and erosional habitat. The modeling approach described can be used to predict a reference condition for both depositional and erosional, by accounting for particle size, hopefully providing assurance for practitioners.

In addition to grain size, river discharge also explained significant variability in indices of benthic community composition. Discharge can be anticipated to influence grain size, development of periphyton and macrophytes on the surficial sediments, each of which will influence the kinds of benthos occurring. Distance upstream/downstream explained significant variability in some indices of composition, with uncertainty in the underlying cause. Benthic communities may have been responding to localized variations in water or sediment quality (neither of which was similarly spatially varying). However, variations among Sites may also have been simply random. Underwood (1991, 1992, 1994) explains that communities (and in his experience benthic communities) differ significantly (and randomly) naturally from place to place and from time to time. Underwood therefore argues for collecting community samples from multiple reference areas to support an EEM program. Underwood’s preferred sampling design was what he referred to as an asymmetric design, that had multiple reference areas against which to judge variations in an exposure area.

5.4.2.3 Sample Sizes

Environment Canada (2012) proposes five samples be collected within reference and exposure sampling areas. A sample size of n=5 results in a 90% likelihood of detecting effects when effects are equivalent to the mean reference value ± 2SDs (Environment Canada, 2012b). The notion of using the reference mean ± 2SD as a generic critical effect size for benthic community surveys now has a long 30-year history of use in Canada, going back to the pulp & paper EEM program guidance documents of the mid-1990s. It should be noted that sample size recommendations and the EEM is based on a different sampling protocol (Surber sampler), and not a CABIN traveling kick.

Sampling areas should be spatially large enough to contain samples that are separated sufficiently to ensure a minimum of spatial autocorrelation. With benthic sampling, what should be obvious is that the closer two samples are spatially, the more likely they are to have the same taxa; and the more separated they are, the more likely they will contain different taxa. When the community in one sample can be used to predict the community in an adjacent sample, they are no-longer statistically independent, violating a foundation of statistical analysis and interpretation. It is therefore important that samples be separated spatially to the extent possible. Environment Canada (2012) has proposed that replicate stations (the unit of replication) in lakes should be at least 10m x 10m in area, and be separated by 20 m. For lotic habitats, Environment Canada (2012) recommends stations be separated by distances equivalent to a pool/riffle sequence or 3x the bankfull width. In the Lower Athabasca River (with a bankfull width of several hundred meters), it is inconceivable that sampling should be separated by any factor of the bankfull width.

In the EMP, CABIN samples were separated on average by 50 m (range 10-100 m) and by 10 m at Sites closest to the proposed OSPW release point. Consistency in habitat (substrate, depth, flow) was prioritized over distance between sites, hence the variability in distance between sites. Separation by 50m is sufficient, considering Environment Canada's (2012b) advice for lakes. The separation distance should be maintained throughout the design to ensure that the degree of spatial autocorrelation is similar among sampling areas. More discussion of sample sizes is provided in the section below on thresholds triggering change.

5.4.2.4 Thresholds to Trigger Changes in Monitoring

Site-specific and regional normal ranges can be used to interpret variations in benthic invertebrate community data. The use of regional (or even local) normal ranges, however, may not have appeal for some of the indices of composition. Benthic density, richness, and percent EPT were three indices for which variability within sampling sites (SD_w) was high enough that computed normal ranges will be relatively broad. The typical density was about 2000 individuals, with an estimated normal range of between 130 and 30,000 individuals per sample. For density, for example, n=14 samples would be required to detect changes in density of about 6000 individuals per sample. Clearly, the ability of an EEM using CABIN kick samples will be unlikely to detect meaningful change in density of benthic invertebrates in the Lower Athabasca River. Percent EPT was also variable within Sites. With an average reference condition of about 20% EPT, the estimate of the normal range, given the withing-site variability, extends from about 2 to 100%; clearly a normal range of that magnitude is not acceptable for detection of change. In these cases, values other than the normal range can be adopted. For example, some other factor of SD_w could be used, such as 1x, but in the case of %EPT, the normal range (for a mean of 20%) would still range from 6 to 62%.

Although density and %EPT are potentially more variable than desired, the benthic community data can be used to estimate other indices. The PTI, based on tolerances of chironomids, was demonstrated to be significantly more sensitive. Changes in PTI of 1 unit were demonstrated in section 4.0 to require only 2 CABIN kick samples. The statistical power of NMDS (multivariate) ordination scores was also demonstrated to be reasonably high, such that the typical n=5 samples was deemed sufficient to detect change in benthic community composition that could be interpreted.

5.4.2.5 Comparison of Monitoring Programs: EMP and OSMP

The benthic invertebrate data collected under the OSMP were collected from cobble substrates, consistent with what is the normal expectation for CABIN kick samples (Reynoldson et al., 2001). The EMP samples, in contrast, were collected from finer-grained (sand/silt) sediments. As a result, the OSMP data were not used here for modeling normal ranges. However, the OSMP data could be used to model normal ranges for EEM programs for which the dominant substrate is coarser cobble material.

5.4.3 Sentinel Fish Populations

5.4.3.1 General Comments

The EMP used both Trout-perch and White Sucker as sentinel species, consistent with the regional OSMP. Both “sentinel” species can be considered surrogate indicators of potential changes in the broader fish community (Kilgour et al., 2005). Trout-perch are an obvious sentinel species for use in EEM programs assessing released oil sands process water. Sufficient sample sizes of male and female fish were obtained in all sampled areas in all years. Further, the species can be anticipated to have a more restricted home range, given its small-bodied size, thereby more likely integrating local environmental conditions than White Sucker.

White Sucker are considered less suitable as a sentinel species because of their greater mobility (compared to Trout-perch). The EMP, further, did not collect sufficient minimum numbers for robust statistical analysis. Typically, numbers of adult sucker were < 20 for males or females. A sample size of 20 fish is generally considered the minimum number required to detect changes of ±10% in condition factor, or ± 25% in liver and gonad size (Environment Canada, 2012b).

5.4.3.2 Sources of Background Variability

The analysis of Trout-perch indicated significant variability in GSI, LSI and K related to river discharge, and sampling year. Sampling location (i.e., station) upstream/downstream of the proposed point-source release did not explain significant variability in those endpoints. The data suggest that the Trout-perch population somatic indices (i.e., GSI, LSI and K) are relatively homogenous along the LAR, while responding significantly to discharge. It is uncertain how the term “year” functionally relates to and explains variations in Trout-perch LSI, GSI and K. There is the potential that the “year” term reflects environmental influences related to oil sands operations. For example, as oil sands operations continue and expand, there is the potential that the associated land-cover change is causing variability in physico-chemical responses in the river that have not been characterized by the sampling carried out in OSMP or EMP. Arciszewski (2021) recently demonstrated that land-cover change in tributaries to the LAR significantly explained variability in indices of benthic community composition. In the EMP, discharge varied among the years. Therefore, it’s possible that the “year” term is simply additional non-linear variation in LSI, GSI and K related to discharge. These models of EMP Trout-perch response variability, however, are consistent with what was described in a previous analysis of the OSMP Trout-perch data by Kilgour et al. (2019).

The absence of variation in somatic indices (LSI, GSI, K) longitudinally along the length of the LAR is interesting. This suggests the possibility that Trout-perch do not have limited home ranges, but integrate conditions across broad areas. Per the analysis by Minns (1995), based on adult Trout-perch being relatively small, we predict a limited home range. The data here, indicating a relatively homogeneous response along the river length within years, suggests a more mobile population. Other data on Trout-perch from the LAR have similarly not indicated longitudinal variation. For example, Sinnatamby et al. (2019) demonstrated Trout-perch in the LAR were relatively invariant in otolith metals chemistry with distance along the channel. Blanar et al. (2016) however, demonstrated that parasite loads in Trout-perch varied spatially, with those spatial variations relating well to land use within 5 km of the study site. There is therefore some uncertainty regarding the mobility/home range of Trout-perch. As such, consideration for distance between reference and exposure areas is required when designing the sampling program for Trout-perch.

5.4.3.3 Sample Sizes

Environment Canada (2012b) proposes an initial sample size of 20 males and 20 females from each reference and exposure area sampled, for each sentinel species. Those numbers were derived from an analysis of White Sucker data from the Moose River (Munkittrick et al., 2000) that demonstrated that (1) variance terms in key indicators tended to stabilize after about 18 fish were collected, and (2) power analysis supported a conclusion that a sample size of about 20 fish (per sex) was sufficient to detect 10% changes in condition factor, and 25% changes in liver and gonad size. Here, the analysis demonstrated similar sample sizes for Trout-perch were sufficient to detect those critical effect sizes.

5.4.3.4 Thresholds to Trigger Changes in Monitoring

Per Munkittrick & Dixon (1989a,b) and Gibbons & Munkittrick (1994), variations in sentinel species condition are general indications of condition of the broader fish community. Significant changes in population performance measures of the sentinel fish population, therefore, can be used to suggest that the broader fish community is at some risk of potential change. In the context of a local EEM focused on a point-source release of OSPW, observed changes in Trout-perch in a nearfield exposure area may be localized. As such, changes in Trout-perch in a nearfield study area should provide early warning of impending effects at the broader fish community level. However, if Trout-perch are more mobile than their size would suggest, then changes in the nearfield study area may be reflecting larger-scale influences. If larger-scale influences are present, then it is unlikely that differences between populations collected in reference and exposure areas would differ significantly in condition. The use of EROD and potentially $\delta^{13}C$ and $\delta^{15}N$ SIRs could assist in demonstrating the site fidelity of Trout-perch, and in interpreting the significance of observed variability in condition.

For Trout-perch, the recommended threshold to trigger a change in monitoring, is the normal range in a reference population. “Reference” here is defined by what is expected for the exposure area in the absence of exposure to treated OSPW. The predicted reference condition can be obtained from data collected upstream (in the same year), or from data collected in the exposure area in years prior to OSPW exposure. Given the importance of discharge in naturally modifying population performance of Trout-perch, any pre-exposure baseline data collected from the exposure area would need to be adjusted (using a statistical model like those in Section 3) for the discharge conditions observed during the exposure period.

The normal range for LSI, GSI and K (as well as mean age, and size at age), would be determined from the upstream reference population (or the statistical model). If derived from a survey of an upstream population, the normal range will be estimated from a minimum number of fish (~ 20). If normal ranges are derived from models of existing historical baseline data, they will be based on several hundred fish (that have already been collected, over the years, and processed), and as such will be more reliable. It is therefore recommended that AEPA make the existing Trout-perch data available for future use in developing/quantifying normal ranges for assessing exposure-area Trout-perch.

5.4.3.5 Comparison of Monitoring Programs: EMP and OSMP

Normal ranges developed using OSMP Trout-perch data provided reasonable approximations of EMP data. Results in Table 63 illustrate that most (typically 95%) EMP values for K, GSI, and LSI fell within expected normal ranges. Therefore, going forward, OSMP Trout-perch data can be confidently used to inform normal range calculations for EMP, or Site-specific EEM programs on the LAR.

5.4.4 Water Quality

5.4.4.1 Sources of Background Variability

River discharge, year and distance from shore were the most significant sources of variability for most of the water quality variables in the EMP data (Table 9). Distance from shore is a significant factor, as it relates to the mixing of tributaries upstream (significantly the Athabasca River upstream of Fort McMurray and the Clearwater River flowing in from the East) as well as the influence of the Fort McMurray and Syncrude sewage treatment outfall located on the west side of the river. The collection and reporting of water quality, relative to baseline normal ranges and forecast thresholds, therefore, needs to take into account distance from shore. Further, because water quality varies among years (irrespective of discharge), appropriate reference data will need to be collected synoptically with exposure area data.

5.4.4.2 Sample Sizes

Within a Site-year combination, water quality variables were generally measured precisely. Triplicate samples demonstrate that the relative percent difference among samples is well < 20% on the typical analyte (except for some hydrocarbons). For most water quality variables that will be influenced by OSPW, a maximum of two grab samples were demonstrated to be easily sufficient to detect a change equivalent to ½ the difference between the baseline average and the most conservative guideline for the protection of aquatic life: smaller effects are easily detectable for most variables. That outcome is sensible. Water sampling in a typical EEM context involves the collection of 1 or 2 grab samples per Site, per sampling period (Environment Canada, 2012b).

5.4.4.3 Thresholds to Trigger Changes in Monitoring

Water quality should be monitored for two purposes: (1) to support the interpretation of biological responses; and (2) to verify the predictions from mass-balance models of the anticipated concentrations in the receiving environment. To support the interpretation of biological responses typically requires a modest data set. In conventional EEM, a single sampling event synoptic with the collections of fish and benthos suffices (Environment Canada, 2012b). If the objective is to verify predictions, it may be necessary to collect samples monthly, or at some increased frequency, depending on the temporal variability of release of treated OSPW.

Like with other monitored variables, Site-specific and regional baseline normal ranges can be computed from the existing data. However, the release of treated OSPW will de facto change the chemistry of the receiving environment. The question is not if there is change, but if the change is consistent with what we anticipated (and agreed to). Therefore, we can anticipate water quality variables in exposure areas exceeding Site-specific and potentially baseline normal ranges. Those changes, however, would be uninformative because they are predicted. Water quality data should not be used to test for change, they should instead be used to test that the changes in the receiving environment are less than the prediction. Where changes exceed the prediction would indicate that the mass-balance model has an insufficient understanding of the receiving environment.

5.4.4.4 Comparison of Monitoring Programs: EMP and OSMP

Normal ranges developed using OSMP water quality data provided reasonable approximations of EMP data. The results in Table 42 illustrate that most (typically 95%) EMP values fell within expected normal ranges. Therefore, going forward, OSMP water quality data can be confidently used to inform normal range calculations for EMP, or Site-specific EEM programs on the Lower Athabasca River. It is recommended that compounds of concern related to oil sands and OSPW, namely naphthenic acids (NAs), be monitored in OSMP as part of regular water quality sampling. Monitoring NAs at the regional scale will become even more important if treated OSPW is discharged into the LAR.

5.4.5 Sediment Quality

5.4.5.1 General Comments

Sediment quality variables are monitored in EEM primarily to support the interpretation of biological responses, typically benthic community data. Recommendations below are based on that general intent.

5.4.5.2 Sources of Background Variability

Concentrations of metals, nutrients, NAs, and PAHs varied strongly with aluminum levels, indicating a strong influence of grain size. Almost half of the variables varied significantly annually, and only a few variables varied longitudinally. Annual and spatial variations, however, were trivial relative to the variance explained by grain size (aluminum).

Concentrations of NAs (adjusted for to an aluminum level of 6000 µg/g) varied significantly among years with a strong decreasing trend from 2018 to 2021 (Figure 22). The reduction in sediment NAs over time was consistent with what was observed in water (Figure 11).

5.4.5.3 Sample Sizes

For all metals except arsenic, four or fewer samples per Site would be sufficient to detect changes equivalent to the difference between the average baseline and ½ way to the most conservative sediment quality guideline with acceptable power. In several cases, only 2 samples are required (e.g., chromium, copper, lead, mercury, zinc; Table 73 and Figure 119). For PAHs, considerably larger sample sizes would be required in order to have desired likelihood of detection when the concentrations in sediment are ½ way to the sediment quality guideline. For all PAHs, except phenanthrene and pyrene, n=5 samples would be sufficient to detect change before their respective guideline is reached.

Assuming that n=5 benthic samples were collected per Site per time, and that sediment quality samples are collected synoptically with benthic community samples, then there would generally be sufficient statistical power with the sediment quality data to detect change before sediment quality guidelines are exceeded.

5.4.5.4 Thresholds to Trigger Changes in Monitoring

Sediments can be monitored for two purposes: (1) to support the interpretation of biological responses; and (2) to verify the predictions of engineering models that estimate precipitation of constituents of concern into sediments (e.g., Four Elements Consulting, 2022). If there is a reasonable expectation that sediment quality will change, under the influence of a released oil sands process water, then monitoring should be designed to verify the predictions. Concentrations that exceed Forecast Thresholds should trigger follow up monitoring, either confirmation or investigation of cause.

Local and regional normal ranges of baseline concentrations can be incorporated, however, as with water quality we should be anticipating changes in sediment quality, and not be surprised if changes are occurring. If baseline normal ranges are considered, then limited data sets will probably satisfactorily define the range of values. Concentrations of NAs, however, have been significantly variable over time, such that additional annual sampling is probably required for that analyte. There was limited spatial variability, within years, for NAs such that samples from a single reference Site may suffice to capture that annual variability.

Several metals and PAHs have sediment quality guidelines. Unexpected changes that exceed ½ way to the guideline (or the guideline itself) could prompt additional concern and associated follow-up including confirmation monitoring and/or investigation of cause.

5.4.5.5 Comparison of Monitoring Programs: EMP and OSMP

Normal ranges developed using OSMP sediment quality data provided reasonable approximations of EMP data. The results in Table 48 illustrate that most (typically 95%) EMP values fell within expected normal ranges. Therefore, going forward, OSMP sediment quality data can be confidently used to inform normal range calculations for EMP, or Site-specific EEM programs on the LAR. Building on what was described in Section 5.4.4.4, it is recommended that NAs be monitored in sediment samples in OSMP as part of regular sediment quality sampling. Research in the Athabasca Oil Sands Region has shown that NAs are likely to persist in the water column and accumulate in the sediments over time (Headley & McMartin, 2004), therefore monitoring sediment borne NAs at the regional scale will become even more important if treated OSPW is discharged into the LAR.

5.4.6 Fish Tissue

5.4.6.1 General Comments

Comparison of EMP and OSMP fish body burden data suggested that there was reasonable overlap in terms of analytes analyzed, however differences in sample numbers at the station and year scale, as well as the types of fish tissues analyzed, between the two programs suggest are not comparable. The discrepancies between the two datasets were summarized in section 3.3.7.1.

5.4.6.2 Sources of Background Variability

After considering species, sex and fish size, variations in tissue levels of $\delta^{13}C$ and $\delta^{15}N$ SIRs, mercury, selenium and PAHs did not significantly vary over time or spatially (Table 34). This potentially indicates that the species monitored (Trout-perch, Walleye, White Sucker) were mobile, or that sources of constituents of concern were homogenously distributed through the study area, and temporally. EROD measured in Trout-perch, however, illustrated spatial variability, indicating fish were resident, and responding to localized influences.

5.4.6.3 Sample Sizes

For selenium, EROD and SIRs $\delta^{13}C$ and $\delta^{15}N$, relatively modest samples sizes of fish are required to detect meaningful changes. With Walleye or White Sucker, a sample size of n=4 will provide sufficient power to detect changes in body burdens before levels exceed ½ way to tissue guidelines (Se, Hg). Similar sample sizes for Walleye, White Sucker and Trout-perch will provide sufficient power to detect modest (<25%) changes in SIRs $\delta^{13}C$ and $\delta^{15}N$. With Trout-perch, higher sample sizes (of ~ n=20) will be required to have sufficient power to detect relatively large changes (75%) in EROD activity.

With mercury, larger sample sizes are required. Mercury levels in adult 45 cm Walleye are near (498 ng/g for male fish and 429 ng/g for female fish) the consumption guideline of 500 ng/g w.w.. As such, even with a modest sample size of fish, the probability of detecting a change that exceeds the guideline will be low. With White Sucker, mercury levels in 45 cm fish are 314 ng/g in males and 259 ng/g in females. Approximately 30 female and ~ 60 male sucker (per Site or time) would be needed to have reasonable power for detecting a changes equivalent to the difference between the baseline and the guideline.

The EMP did not provide any data that supports sample-size estimation for PAHs in fish tissue, given that most of the PAHs in tissue were below guidelines.

5.4.6.4 Thresholds to Trigger Changes in Monitoring

Normal ranges can be applied to all measured fish tissue constituents. For selenium and mercury, the data would suggest that the site-specific and regional normal ranges will be highly similar. The data also suggest that EROD will vary spatially (for at least Trout-perch), such that there may be Site-specific and regional normal ranges for that variable. There are also guidelines for tissue levels of selenium, mercury, and PAHs. Exceedance of those guidelines (Figure 70), or the point ½ way to the guideline, can be used to trigger confirmation or investigation of cause.

5.4.6.5 Comparison of Monitoring Programs: EMP and OSMP

There was insufficient overlap in fish tissue data in terms of species or sex between OSMP and EMP data sets, to justify the development of OSMP regional models. Therefore, in this context, OSMP data were not (cannot) be used to support interpretation of the EMP fish tissue data. Further, it is recommended that fish tissue analysis in OSMP begin to monitor SIRs ($\delta^{13}C$ and $\delta^{15}N$) as they can be used as indicators of site fidelity.

5.4.7 Benthic Tissue

5.4.7.1 General Comments

Benthic tissue data were only available for this report from the EMP. Data were provided for three common families: a mayfly (Ametropodidae); a dragonfly (Gomphidae); and a stonefly (Pteronarcyidae).

5.4.7.2 Sources of Background Variability

Variations in benthic tissue levels of SIRs ($\delta^{13}C$ and $\delta^{15}N$), mercury and selenium, depended on the benthic family, sampling year and sampling Site (Table 38). As such, variations among years and Sites should be considered for the computation of normal ranges.

5.4.7.3 Sample Sizes

Relatively modest sample sizes of between 2 and 6 tissues per Site/time would be required in order to have reasonable likelihood of detecting changes in tissue levels (of selenium and mercury) equivalent to a change from baseline levels to ½ way to a conservative tissue guideline.

5.4.7.4 Thresholds to Trigger Changes in Monitoring

Normal ranges can be applied to all measured benthic tissue constituents. For selenium and mercury, the data suggest that regional normal ranges will be modestly, but significantly larger than site-specific normal ranges.

There are guidelines for tissue levels of selenium and mercury. Exceedance of those guidelines (Figure 70), or the point ½ way to the guideline, could be used to trigger confirmation or investigation of cause.

5.4.7.5 Comparison of Monitoring Programs: EMP and OSMP

There was no OSMP benthic body burden data provided on which to derive a regional model. A model was therefore constructed using the EMP data only. Because of the stationary nature of benthic invertebrate populations, it is recommended that tissues be analyzed for various constituents of potential concern and markers of exposure in the OSMP program as well. Burdens of contaminants in benthic invertebrates can also provide a direct measure of condition of an aquatic assemblage and can be used to support interpretation of benthic invertebrate community data, assuming benthic body burden samples should be collected synoptically with BIC.

5.5 Recommended Generic EEM

5.5.1 Components

The components to include in an EEM program should be based on the goals and objectives of the resource management (Cairns et al., 1993). The study design supporting the EMP did not justify the included components, relative to goals and objectives related to use of the LAR. However, the LAR supports fish and other aquatic organisms, and is a source of fish tissue for wildlife and human consumers. The LAR can also be anticipated to be a source of potable water, but any consumption of water can be assumed to be supported by some form of water treatment to remove solids (and various associated contaminants) and bacteria. So, for the purpose here, it is assumed that the main purpose of EEM is to assure that environmental quality is sufficient to support fish and other aquatic organisms, and to assure that contaminants in fish tissues remain within acceptable levels for wildlife and human consumers.

5.5.1.1 Adult Fish Populations

5.5.1.1.1 Sentinel Fish Populations

The components listed in Table 65, then, are justifiable for an EEM program related to assessing the potential effects of oil sands process water. Sentinel fish populations were justified for EEM by Munkittrick et al. (2000), Munkittrick & Dixon (1989a, 1989b) and Kilgour et al. (2005) on the basis that variations in mean age (of adult fish), condition, liver size, and gonad size relates reasonably well to the overall fish community condition. The relationships between population-level performance and community are not numerically quantified, but are rather narrative (Munkittrick & Dixon, 1989b). Regardless, variations in fish population performance provides some ability to infer effects at a higher organizational level. One other reason to use sentinel fish populations is because it can be challenging to characterize the fish community, particularly in large systems like the LAR (Kilgour et al., 2005).

The use of Trout-perch as a key sentinel species for the LAR is logical. The species is small-bodied and likely has a limited home range (Minns, 1995), although home range has not be explicitly described for the species. Variations in body tissue burdens (mercury, selenium, EROD) suggest that Trout-perch in the Athabasca River keep a relatively small home range. The collection of n=20 male and female fish per sampling Site per time is justified from power analysis. That sample size will also address sampling requirements for most tissue contaminant-level analyses. The optimal time for the collection of Trout-perch is in the fall, when water levels in the LAR are low. Ovary and testes tissues collected in the fall will be sufficiently developed to provide potentially meaningful differences between reference and exposure conditions (Gibbons et al., 1998).

5.5.1.1.2 Fish Community Assemblages

Fish community surveys evaluate whether there are differences between areas in the diversity and abundance of fish species present and is an important indicator that is monitored in EEM’s for the metal mining industry (Environment Canada, 2012a). When the fish community composition has changed because of the presence of an effluent (treated OSPW in this case), there will also likely be measurable changes in the fish populations that remain. Monitoring changes in fish community assemblages, both before and after the release of OSPW, is therefore an important indicator to include in future EEM programs along the LAR.

5.5.1.2 Benthic Invertebrates

BIC composition provides a direct measure of the condition of an aquatic assemblage. Benthic communities are also justified components on the basis that variations in composition can be used to infer variations in fish community composition, and thereby relate to values relevant to the Fisheries Act (Kilgour et al., 2005). The CABIN benthic community sampling protocol is a suitable method to be applied to the LAR. The sampling technique results in the collection of benthos in the nearshore habitat from 0 to ~ 1 m of water depth. Samples were (in the EMP) and should continue to be collected in the fall when water levels are low. Sampling at other times of the year would have the potential (with this method) of collecting from habitats that were underwater for only a brief period, and not truly representative of longer-term conditions. CABIN samples were ‘nested’ within sampling Sites, and separated by a distance of 10 m in Sites close to the proposed pilot release, and by 10-100 m in Sites further from the release point. Technically, the spatial separation between samples (within Sites) should be the same, so that the spatial autocorrelation is the same from Site to Site, and time to time. Spatial separation of samples within Sites will determine the overall size of sampling Sites.

BIC sample size should be a minimum n=5 replicate CABIN kicks per Site per year. That sample size is as per Environment Canada (2012) recommendations, and will ensure sufficient statistical power to detect changes equivalent to the reference Site mean ±2SD.w. If the upstream reference Site is degraded, then a regional reference condition can be modeled.

The CABIN kick method is classically applied to cobble-bottomed substrates, as applied in the OSMP. However, CABIN samples can also be collected in softer materials. Models can be used subsequently to factor out, or predict, BIC composition with substrate as one of the predictors. The regional model developed here did not include substrate as a predictor, but it could have. In the event that future OSPW releases are in areas of the LAR dominated with cobble, there will be the added benefit of utilizing baseline OSMP data from cobble habitats.

5.5.1.3 Benthic Algae

Benthic algal data, like BIC data, also provide a direct measure of condition of an aquatic assemblage. Benthic chlorophyll *a* can also indicate risks to fish and BIC (Nordin, 2001), and can support interpretation of BIC data. To support BIC interpretation, benthic algae samples should be collected synoptically with BIC. Assuming that n=5 benthic transects are sampled per Site, then the same number of benthic algae samples would be appropriate.

5.5.1.4 Water and Sediment Quality

Water and sediment quality data are required in EEM as supporting variables to interpret biological variations.

5.5.1.4.1 Continuous Water Quality Monitoring

Continuous water quality monitoring at specific sites that are relevant to other important biological indicators (i.e., fish, algae, and benthos) would aid in the refining and development of normal range models as described throughout this report. Producing data on water quality parameters such as temperature, pH, specific conductivity, and dissolved oxygen that can be easily paired and linked to fish, algae, and benthic datasets would enable these parameters to be used to account for sources of variation in normal range models, thus leading to better defined thresholds of change.

5.5.1.4.2 Regular Water Grab Sampling

Sufficient resolution in water quality data can be obtained with a relatively limited number of samples. For metals, nutrients, etc., single (or duplicate) samples will suffice. For PAHs or other organics, a higher number of samples may be required, depending on the threshold that AEPA adopts. Within a sampling Site, there is limited spatial variation in concentrations of constituents of concern. The main value in collecting multiple samples is to characterize the modest ‘among-sample’ error.

5.5.1.4.3 SPMDs

The power analysis carried out on PAHs with established guidelines in Section 4.1.1 indicated that the sample size required to detect when concentrations are ½ way to the guideline is N = 2 for both grab samples (Section 4.1.1.1) and SPMDs (Section 4.1.1.2). Thus, if both methods have the same capacity to detect change, the decision to include grab sampling and/or SPMDs in future EEM framework must be dependent on logistics, efficiency, and the desired outcome.

Ultimately, the decision to include SPMD’s should be based on formal consideration of a conceptual site model. Conceptual models are typically stressor-based vector diagrams showing the hypothesized pathways associated with the fate and transport of specific stressors or effects-based flowcharts identifying the stressors and associated responses of selected ecosystem indicators (Ankley et al., 2010; Davidson et al., 2020). If decision-makers can connect SPMD-derived stressors to a valued ecosystem component (VEC) response along the adverse outcome pathway or use SPMD-derived PAH concentrations to provide early warning of impacts on the VEC, then they should be considered in future monitoring.

Water quality criteria and threshold limit values are often based on dissolved concentrations, and SPMDs provide estimates of the dissolved PAH phase, whereas grab samples provide a total concentration, including particulates. If the main goal is to use water quality data to estimate the potential for toxicity, then SPMDs may be a more direct approach. Alternatively, if water quality data is solely meant to be used as supporting variable in EEM, SPMDs may not be warranted.

5.5.1.4.4 Cytotoxicity

The C-WQI values derived from the water cytotoxicity assay may be useful for identifying hotspots that may contain constituents posing a human health risk. As an effects-based assay, it measures the total cytotoxicity caused by the mixture components in a water sample but does not reveal the identity of the causal agent(s). Additional research would appear to be warranted to understand the linkage between the index values and human health risks. If the causal linkage between elevated C-WQI in the LAR can be established, whether it is driven by a single compound or a suite of compounds, then it may be beneficial for future EEM to focus on the compounds driving increased C-WQI, rather than C-WQI itself.

5.5.1.4.5 Sediments

A limited number of 2 to 5 sediment samples also provides sufficient statistical power for detecting reasonable change in sediment quality. Sediment samples should be collected synoptically with benthic invertebrate and benthic algal community samples (if chosen to be included), to improve upon the ability to interpret variations in those biological responses.

5.5.1.5 Tissues

In terms of mercury in fish tissues, the most relevant fish species to consider is Walleye. At the top of the food chain, mercury levels in Walleye will be higher than those of White Sucker or Trout-perch. Walleye, further, are more consistently consumed by anglers and subsistence consumers. Walleye and White Sucker migrate considerable distances in the LAR (Arciszewski, Munkittrick, Kilgour, et al., 2017) and such that body burdens of mercury in both species will vary with regional influences. As such, it would not be appropriate to use either Walleye or White Sucker for monitoring changes in mercury level relative to a future proposed release of OSPW. However, Trout-perch appears to have smaller home ranges, and if the risk of mercury contamination is deemed possible, via the release of process water, then Hg in Trout-perch tissue would be a reasonable measurement.

Because of the migration potential, tissue levels of other contaminants (Se) or markers (EROD) in Walleye and White Sucker would not be appropriate for a Site-specific EEM program. However, measurement of those analytes in Trout-perch does make sense. Required sample sizes for Trout-perch tissue are generally consistent with those required for population performance measures. That is sample sizes of n=20 male and female fish (each) can be expected to generally be sufficient to detect changes in markers such as EROD, and in mercury or selenium. Larger numbers of fish may be required in order to have sufficient power to detect small changes in PAHs.

Because of the stationary nature of benthic invertebrate populations, it also makes sense that tissues be analyzed for various constituents of potential concern and markers of exposure. Selenium levels in benthic tissue for example can be reasonably used to predict changes in selenium levels in fish (Orr et al., 2006). Benthic sample sizes appear to be reasonable (n=2 to ~6) to detect reasonable changes in mercury and selenium. There is some uncertainty of the number of samples required to detect changes in SIRs, just because there is uncertainty as to the magnitude of change in SIRs that matters.

5.5.2 Spatial and Temporal Design

Given that there are historical OSMP data, and now historical EMP data, the design of a site-specific EEM program for a potential future release of treated OSPW becomes relatively straightforward. Assuming that collection and processing methods do not change over time, the historical data can be used to model this historical baseline period and derive Site-specific and regional normal ranges for this period of data. With typically 10 Sites per each of three years in the EMP, there are ~ 30 Site x year combinations of EMP data from which within-site, among-site and among-year error terms can be estimated. The OSMP data expands that basic data matrix, and significantly enhances the ability to quantify normal ranges. As such, any future Site-specific EEM program need not expend energy quantifying spatial variation in the reference condition and can be designed with a single downstream exposure area and a single upstream reference area.

Within the reference and exposure areas, benthic algae, benthic invertebrates and fish population (Trout-perch) data should be collected, with the minimum replication for each component. Water and sediment quality in addition to fish and benthic invertebrate tissue data should also be collected.

Ideally, sampling of reference and exposure areas would include a minimum of three years of site-specific baseline data collected from each of the reference and exposure areas. For most monitoring components, variations among years were demonstrated to be principally associated with river flow volumes (discharge). As such, the historical OSMP and EMP data will be valuable in quantifying variability that may be temporal in nature. Regardless, having an upstream reference Site will provide the gold-standard for accounting for discharge (and other) year-related influences.

5.6 Data Processing

Throughout sections 2.2 and 3.2, a variety of inconsistencies and the method in which they were dealt with within the different datasets were presented. When dealing with environmental datasets that span across a variety of different environmental media (i.e., water, sediments, benthos, fish, algae etc.), parameters (i.e., concentrations, fish health indicators, indicators of community composition etc.), sampling years (i.e., 2009 – 2021), sampling programs (i.e., EMP vs. OSMP), it is unsurprising that inconsistencies were flagged during the amalgamation of the datasets. When the datasets are large, a considerable amount of time, effort, and resources must be devoted to the quality assurance and quality control of the datasets, and this can ultimately take away time otherwise slated for data analysis.

5.6.1 Data QA/QC

While the overall goal of this report was not to focus on data processing, it is still important to acknowledge shortcomings in the datasets and propose solutions to enhance future dataset management. Below is a summarized list of common data inconsistencies encountered within the EMP and OSMP datasets (across all sampling years):

- Date formats:
  - Ex: 2018-01-01 vs. 01-01-2018
  - In the case of the benthic body burden data, only month and year were provided in the date column (ex: 01/2018), and therefore samples could not be paired with daily discharge values to enhance the normal range model performance.
- Analyte nomenclature;
  - Spaces between names (ex: benzo [a] pyrene vs. benzo[a]pyrene)
  - Use of parentheses (ex: benzo[a]pyrene vs. benzo(a)pyrene)
- Repeated columns with the same column name but different rows of data.
  - Ex: a different column for benzo(a)pyrene data for each sampling year
- “Flag” columns and “RDL” columns arranged in wide format;
  - Issue highlighted in section 2.2.5.1.1
- Different identifiers for instances when a value is below a method detection limit.
  - Ex: < vs. L vs. M vs. NR
- Copy and paste errors in the raw datasets resulting in scrambled data columns;
  - Ex: fish body weight, fork length, and age columns swapping mid-way down the datasets

- Inconsistent EMP or OSMP sample station names;
- Clear data recording errors that produced outliers 1000x larger or smaller than expected values;
- Inconsistent units (i.e., ng/g vs. mg/kg vs. µg/g) or lack of units altogether.

The above list is not an all-encompassing list of data inconsistencies observed throughout the conception of this report, but it does however highlight clear shortcoming in the provided datasets that could be easily negated with additional Quality Assurance and Quality Control (QAQC) steps.

5.6.2 Database Management

The use of an Environmental Database Management System (EDMS) is recommended for future data compilation along the LAR under both OSMP and EMP. While shifting historical data into an EDMS is a resource intensive endeavor, the two primary benefits are inseparable for any high functioning data-driven work environment: capacity and accessibility.

Using an EDMS would allow for:

- Collection and storage of environmental data on a consistent basis;
- Enable the user to pose detailed data queries;
- Ensure consistent and proper formatting of datasets prior to inclusion in the database to avoid error as listed in section 5.6.1;
- Automatically locate and highlight anomalies and errors;
- Automatically convert metrics and units for consistency;
- Pull data that is already formatted for use in modern statistical software such as R and/or Python.

These recommendations would likely improve the efficiency in which data-driven reports that deal with historical environmental data within the LAR are performed and completed.

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Respectfully submitted,

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Appendix A VMV Codes and Detection Limits

Appendix A - VMV's and DL's

Table A1. Detection limit and analytical code summary for samples collected during the EMP program (218, 2019, and 2021)

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Surface Water Grab	1-Methylchrysene	4081	0.014 - 0.573	ng/L	218	12.844
Surface Water Grab	1-Methylnaphthalene	4081	0.029 - 0.532	ng/L	218	-
Surface Water Grab	1-Methylphenanthrene	4081	0.022 - 0.756	ng/L	218	10.092
Surface Water Grab	1,2-Dimethylnaphthalene	4081	0.043 - 0.802	ng/L	218	46.789
Surface Water Grab	1,2,6-Trimethylphenanthrene	4081	0.027 - 0.839	ng/L	218	32.110
Surface Water Grab	1,4,6,7-Tetramethylnaphthalene	4081	0.05 - 0.815	ng/L	218	17.890
Surface Water Grab	1,7-Dimethylfluorene	4081	0.055 - 0.703	ng/L	218	61.009
Surface Water Grab	1,7-Dimethylphenanthrene	4081	0.018 - 0.518	ng/L	218	10.550
Surface Water Grab	1,8-Dimethylphenanthrene	4081	0.018 - 0.527	ng/L	218	37.615
Surface Water Grab	2-Methylanthracene	4081	0.024 - 0.828	ng/L	218	61.927
Surface Water Grab	2-Methylfluorene	4081	0.038 - 0.322	ng/L	218	34.404
Surface Water Grab	2-Methylnaphthalene	4081	0.027 - 0.5	ng/L	218	-
Surface Water Grab	2-Methylphenanthrene	4081	0.022 - 0.763	ng/L	218	6.881
Surface Water Grab	2,3,5-Trimethylnaphthalene	4081	0.023 - 0.938	ng/L	218	5.046
Surface Water Grab	2,3,6-Trimethylnaphthalene	4081	0.022 - 0.902	ng/L	218	3.670
Surface Water Grab	2,4-Dimethyldibenzothiophene	4081	0.027 - 0.67	ng/L	218	22.018
Surface Water Grab	2,6-Dimethylnaphthalene	4081	0.024 - 0.677	ng/L	218	2.752
Surface Water Grab	2,6-Dimethylphenanthrene	4081	0.018 - 0.527	ng/L	218	19.725
Surface Water Grab	2/3-Methyldibenzothiophenes	4081	0.03 - 0.889	ng/L	218	22.018
Surface Water Grab	3-Methylfluoranthene/Benzo(a)Fluorene	4081	0.022 - 0.81	ng/L	218	9.174
Surface Water Grab	3-Methylphenanthrene	4081	0.023 - 0.781	ng/L	218	8.716
Surface Water Grab	3,6-Dimethylphenanthrene	4081	0.018 - 0.536	ng/L	218	12.844
Surface Water Grab	5,9-Dimethylchrysene	4081	0.03 - 0.71	ng/L	218	15.596
Surface Water Grab	5/6-Methylchrysene	4081	0.014 - 0.58	ng/L	218	17.431
Surface Water Grab	7-Methylbenzo(a)Pyrene	4081	0.051 - 1.82	ng/L	218	73.853
Surface Water Grab	9/4-Methylphenanthrene	4081	0.023 - 0.781	ng/L	218	11.927
Surface Water Grab	Acenaphthene	4081	0.031 - 0.363	ng/L	218	11.468
Surface Water Grab	Acenaphthylene	4081	0.013 - 0.285	ng/L	218	27.982
Surface Water Grab	Aluminum Total Recoverable	1847	0.4 - 4	µg/l	215	1.860
Surface Water Grab	Ammonia Total	439	0.015 - 0.075	mg/l	218	82.569
Surface Water Grab	Anthracene	4081	0.025 - 0.6	ng/L	218	55.505
Surface Water Grab	Benz(a)Anthracene/Chrysene-C1	4081	0.014 - 0.577	ng/L	218	9.174
Surface Water Grab	Benz(a)Anthracene/Chrysene-C3	4081	0.022 - 0.585	ng/L	218	17.431
Surface Water Grab	Benz(a)Anthracene/Chrysene-C4	4081	0.031 - 0.478	ng/L	218	64.679
Surface Water Grab	Benzene	3319	5e-04 - 0.03	µg/l	217	100.000
Surface Water Grab	Benzene	3642	5e-04 - 0.03	mg/l	217	100.000
Surface Water Grab	Benzo(a)Anthracene	4081	0.014 - 0.747	ng/L	218	18.349
Surface Water Grab	Benzo(a)Anthracene/Chrysene-C2	4081	0.03 - 0.71	ng/L	218	10.092
Surface Water Grab	Benzo(a)Pyrene	4081	0.038 - 1.82	ng/L	218	33.028
Surface Water Grab	Benzo(B)Fluoranthene	4081	0.019 - 1.09	ng/L	218	13.303
Surface Water Grab	Benzo(b,k)Fluoranthene/Benzo(a)Pyrene-C2	4081	0.039 - 2.25	ng/L	218	18.349
Surface Water Grab	Benzo(e)Pyrene	4081	0.035 - 1.71	ng/L	218	13.761

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Surface Water Grab	Benzo(g,h,i)Perylene	4081	0.032 - 1.27	ng/L	218	11.009
Surface Water Grab	Benzo(J,K)Fluoranthenes	4081	0.027 - 1.28	ng/L	218	38.073
Surface Water Grab	Benzofluoranthene/Benzopyrene-C1	4081	0.051 - 1.82	ng/L	218	10.550
Surface Water Grab	Biphenyl	4081	0.013 - 0.261	ng/L	218	-
Surface Water Grab	C1-Fluoranthenes/Pyrenes	4081	0.022 - 0.81	ng/L	218	8.257
Surface Water Grab	C1-Fluorenes	4081	0.038 - 0.322	ng/L	218	1.835
Surface Water Grab	C1-Naphthalenes	4081	0.027 - 0.5	ng/L	218	-
Surface Water Grab	C1-Phenanthrenes/Anthracenes	4081	0.022 - 0.756	ng/L	218	7.339
Surface Water Grab	C2-Fluoranthenes/Pyrenes	4081	0.021 - 0.84	ng/L	218	6.881
Surface Water Grab	C2-Fluorenes	4081	0.055 - 0.703	ng/L	218	5.046
Surface Water Grab	C2-Naphthalenes	4081	0.043 - 0.802	ng/L	218	0.459
Surface Water Grab	C2-Phenanthrenes/Anthracenes	4081	0.018 - 0.527	ng/L	218	3.211
Surface Water Grab	C3-Fluorenes	4081	0.059 - 1.16	ng/L	218	16.514
Surface Water Grab	C3-Naphthalenes	4081	0.022 - 0.92	ng/L	218	-
Surface Water Grab	C3-Phenanthrenes/Anthracenes	4081	0.027 - 0.839	ng/L	218	5.963
Surface Water Grab	C4-Naphthalenes	4081	0.05 - 0.815	ng/L	218	8.257
Surface Water Grab	C4-Phenanthrenes/Anthracenes	4081	0.05 - 3.84	ng/L	218	6.422
Surface Water Grab	Calcium Dissolved	1516	0.03 - 0.3	mg/l	359	11.699
Surface Water Grab	Calcium Dissolved	1849	0.03 - 0.3	mg/l	359	11.699
Surface Water Grab	Carbon Dissolved Organic	119	0.5 - 2.5	mg/l	218	9.633
Surface Water Grab	Carbon Total Organic (TOC)	119	0.5 - 1	mg/l	218	11.927
Surface Water Grab	Chrysene	4081	0.014 - 0.816	ng/L	218	-
Surface Water Grab	Colour True	3535	02-Feb	---	218	11.927
Surface Water Grab	Colour True	3535	02-Feb	Pt/Co units	218	11.927
Surface Water Grab	Dibenzo(a,h)Anthracene	4081	0.049 - 1.44	ng/L	218	65.596
Surface Water Grab	Dibenzothiophene	4081	0.021 - 0.396	ng/L	218	6.422
Surface Water Grab	Dibenzothiophene-C1	4081	0.03 - 0.889	ng/L	218	18.349
Surface Water Grab	Dibenzothiophene-C2	4081	0.027 - 0.67	ng/L	218	5.046
Surface Water Grab	Dibenzothiophene-C3	4081	0.022 - 8.66	ng/L	218	6.422
Surface Water Grab	Dibenzothiophene-C4	4081	0.024 - 1.88	ng/L	218	5.046
Surface Water Grab	Dimethyl Biphenyl	4081	0.022 - 0.553	ng/L	218	0.917
Surface Water Grab	Dissolved Methyl Mercury (MeHg)	3740	-	ng/L	212	14.151
Surface Water Grab	Dissolved Methyl Mercury (MeHg)	3740	-	ng/L	212	14.151
Surface Water Grab	Ethylbenzene	3319	5e-04 - 0.05	µg/l	217	100.000
Surface Water Grab	Ethylbenzene	3642	5e-04 - 0.05	mg/l	217	100.000
Surface Water Grab	F1 Hydrocarbons (BTEX)	3319	0.1 - 0.5	µg/l	121	91.736
Surface Water Grab	F1 Hydrocarbons (BTEX)	3642	0.1 - 0.5	mg/l	121	91.736
Surface Water Grab	F1 Hydrocarbons (C6-C10)	3319	0.1 - 0.5	µg/l	217	88.940
Surface Water Grab	F1 Hydrocarbons (C6-C10)	3642	0.1 - 0.5	mg/l	217	88.940
Surface Water Grab	F2 Hydrocarbons (C10-C16)	3522	0.1 - 8	mg/l	216	96.759
Surface Water Grab	F2 Hydrocarbons (C10-C16)	3696	0.1 - 8	µg/l	216	96.759
Surface Water Grab	F3 Hydrocarbons (C16-C34)	3522	0.25 - 12	mg/l	216	66.667
Surface Water Grab	F3 Hydrocarbons (C16-C34)	3696	0.25 - 12	µg/l	216	66.667
Surface Water Grab	F4 Hydrocarbons (C34-C50)	3522	0.25 - 5	mg/l	216	75.463

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Surface Water Grab	F4 Hydrocarbons (C34-C50)	3696	0.25 - 5	µg/l	216	75.463
Surface Water Grab	Fluoranthene	4081	0.015 - 0.44	ng/L	218	-
Surface Water Grab	Fluoranthene/Pyrene-C3	4081	0.02 - 0.622	ng/L	218	10.550
Surface Water Grab	Fluoranthene/Pyrene-C4	4081	0.019 - 0.546	ng/L	218	11.468
Surface Water Grab	Fluorene	4081	0.019 - 0.353	ng/L	218	4.587
Surface Water Grab	Indeno(1,2,3-c,d)Pyrene	4081	0.041 - 1.41	ng/L	218	20.642
Surface Water Grab	Iron Dissolved	1849	0.06 - 2	µg/l	431	27.842
Surface Water Grab	Iron Dissolved	2725	0.06 - 2	mg/l	431	27.842
Surface Water Grab	Iron Total Recoverable	1847	0.6 - 6	µg/l	215	8.837
Surface Water Grab	M- + P-Xylene	3319	5e-04 - 0.06	µg/l	217	94.931
Surface Water Grab	M- + P-Xylene	3642	5e-04 - 0.06	mg/l	217	94.931
Surface Water Grab	Manganese Dissolved	1849	0.004 - 0.01	µg/l	433	44.111
Surface Water Grab	Manganese Dissolved	2725	0.004 - 0.01	mg/l	433	44.111
Surface Water Grab	Mercury Dissolved	3734	-	ng/L	212	11.321
Surface Water Grab	Mercury Dissolved	3734	-	ng/L	212	11.321
Surface Water Grab	Mercury Total	3737	-	ng/L	212	10.377
Surface Water Grab	Mercury Total	3737	-	ng/L	212	10.377
Surface Water Grab	Methyl Acenaphthene	4081	0.029 - 0.265	ng/L	218	83.028
Surface Water Grab	Methyl Biphenyl	4081	0.016 - 0.261	ng/L	218	-
Surface Water Grab	Methyl Mercury	3735	-	ng/L	212	13.208
Surface Water Grab	Methyl Mercury	3735	-	ng/L	212	13.208
Surface Water Grab	Naphthalene	4081	0.028 - 1.3	ng/L	218	-
Surface Water Grab	Naphthenic acids	3672	1.43 - 4	µg/l	217	35.023
Surface Water Grab	Nitrogen Kjeldahl Dissolved	442	0.05 - 0.25	mg/l	218	12.385
Surface Water Grab	Nitrogen Kjeldahl Total	442	0.05 - 0.25	mg/l	218	12.385
Surface Water Grab	Nitrogen Total	301	0.02 - 0.055	mg/l	144	11.806
Surface Water Grab	O-Xylene	3319	5e-04 - 0.04	µg/l	217	91.244
Surface Water Grab	O-Xylene	3642	5e-04 - 0.04	mg/l	217	91.244
Surface Water Grab	Perylene	4081	0.038 - 1.73	ng/L	218	11.927
Surface Water Grab	Phenanthrene	4081	0.025 - 0.596	ng/L	218	-
Surface Water Grab	Phosphorus Total	443	0.003 - 0.015	mg/l	218	10.092
Surface Water Grab	Phosphorus Total Dissolved	443	0.003 - 0.006	mg/l	218	16.514
Surface Water Grab	Pyrene	4081	0.015 - 0.432	ng/L	218	-
Surface Water Grab	Reactive Silica	512	0.05 - 0.1	mg/l	183	12.568
Surface Water Grab	Residue Nonfilterable	430	0.99 - 3	mg/l	218	11.009
Surface Water Grab	Retene	4081	0.05 - 3.84	ng/L	218	9.174
Surface Water Grab	Silicon	1847	0.02 - 20	mg/l	148	13.514
Surface Water Grab	Silicon	1849	0.02 - 20	mg/l	148	13.514
Surface Water Grab	Styrene	3642	5e-04 - 0.001	mg/l	44	100.000
Surface Water Grab	Sulphate Dissolved	405	02-Jan	mg/l	218	12.844
Surface Water Grab	Toluene	3319	5e-04 - 0.04	µg/l	217	93.548
Surface Water Grab	Toluene	3642	5e-04 - 0.04	mg/l	217	93.548
Surface Water Grab	Total PAHs	4081	0.014 - 0.161	ng/L	218	-
Surface Water Grab	Turbidity	409	-	NTU	296	3.716
Surface Water Grab	Turbidity	409	-	NTU	296	3.716

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Surface Water Grab	Alkalinity Phenolphthalein CaCO3	620	1	mg/l	218	100.000
Surface Water Grab	Alkalinity Total CaCO3	620	1	mg/l	218	12.844
Surface Water Grab	Aluminum Dissolved	1849	0.4	µg/l	215	5.581
Surface Water Grab	Ammonia Dissolved	439	0.015	mg/l	74	81.081
Surface Water Grab	Anions Total	4	-	me/L	214	-
Surface Water Grab	Antimony Dissolved	1849	0.09	µg/l	215	89.767
Surface Water Grab	Antimony Total Recoverable	1847	0.008	µg/l	215	11.628
Surface Water Grab	Arsenic Dissolved	1849	0.01	µg/l	215	11.628
Surface Water Grab	Arsenic Total Recoverable	1847	0.01	µg/l	215	11.163
Surface Water Grab	Barium Dissolved	1849	0.05	µg/l	215	10.698
Surface Water Grab	Barium Total Recoverable	1847	0.05	µg/l	215	11.163
Surface Water Grab	Beryllium Dissolved	1849	0.004	µg/l	215	61.860
Surface Water Grab	Beryllium Total Recoverable	1847	0.003	µg/l	215	11.628
Surface Water Grab	Bicarbonate (Calcd.)	620	1	mg/l	218	12.844
Surface Water Grab	Bismuth Dissolved	1849	0.003	µg/l	215	89.302
Surface Water Grab	Bismuth Total Recoverable	1847	0.003	µg/l	215	22.326
Surface Water Grab	Boron Dissolved	1849	0.2	µg/l	215	11.628
Surface Water Grab	Boron Total Recoverable	1847	0.2	µg/l	215	11.628
Surface Water Grab	C10H16O2 (Z = -4, DBE = 3)	3672	0.01	%	217	70.968
Surface Water Grab	C10H18O2 (Z = -2, DBE = 2)	3672	0.01	%	217	48.387
Surface Water Grab	C10H20O2 (Z = 0, DBE = 1)	3672	0.01	%	217	29.032
Surface Water Grab	C11H14O2 (Z = -8, DBE = 5)	3672	0.01	%	217	52.535
Surface Water Grab	C11H16O2 (Z = -6, DBE = 4)	3672	0.01	%	217	77.880
Surface Water Grab	C11H18O2 (Z = -4, DBE = 3)	3672	0.01	%	217	42.396
Surface Water Grab	C11H20O2 (Z = -2, DBE = 2)	3672	0.01	%	217	28.571
Surface Water Grab	C11H22O2 (Z = 0, DBE = 1)	3672	0.01	%	217	51.152
Surface Water Grab	C12H16O2 (Z = -8, DBE = 5)	3672	0.01	%	217	76.498
Surface Water Grab	C12H18O2 (Z = -6, DBE = 4)	3672	0.01	%	217	41.935
Surface Water Grab	C12H20O2 (Z = -4, DBE = 3)	3672	0.01	%	217	41.014
Surface Water Grab	C12H22O2 (Z = -2, DBE = 2)	3672	0.01	%	217	42.857
Surface Water Grab	C12H24O2 (Z = 0, DBE = 1)	3672	0.01	%	217	37.788
Surface Water Grab	C13H16O2 (Z = -10, DBE = 6)	3672	0.01	%	217	96.774
Surface Water Grab	C13H18O2 (Z = -8, DBE = 5)	3672	0.01	%	217	83.871
Surface Water Grab	C13H20O2 (Z = -6, DBE = 4)	3672	0.01	%	217	38.710
Surface Water Grab	C13H22O2 (Z = -4, DBE = 3)	3672	0.01	%	217	23.502
Surface Water Grab	C13H24O2 (Z = -2, DBE = 2)	3672	0.01	%	217	24.424
Surface Water Grab	C13H26O2 (Z = 0, DBE = 1)	3672	0.01	%	217	58.986
Surface Water Grab	C14H16O2 (Z = -12, DBE = 7)	3672	0.01	%	217	86.636
Surface Water Grab	C14H18O2 (Z = -10, DBE = 6)	3672	0.01	%	217	59.908
Surface Water Grab	C14H20O2 (Z = -8, DBE = 5)	3672	0.01	%	217	44.700
Surface Water Grab	C14H22O2 (Z = -6, DBE = 4)	3672	0.01	%	217	5.991
Surface Water Grab	C14H24O2 (Z = -4, DBE = 3)	3672	0.01	%	217	10.138
Surface Water Grab	C14H26O2 (Z = -2, DBE = 2)	3672	0.01	%	217	29.954
Surface Water Grab	C14H28O2 (Z = 0, DBE = 1)	3672	0.01	%	217	41.014
Surface Water Grab	C15H14O2 (Z = -16, DBE = 9)	3672	0.01	%	217	95.392

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Surface Water Grab	C15H16O2 (Z = -14, DBE = 8)	3672	0.01	%	217	86.636
Surface Water Grab	C15H18O2 (Z = -12, DBE = 7)	3672	0.01	%	217	44.700
Surface Water Grab	C15H20O2 (Z = -10, DBE = 6)	3672	0.01	%	217	25.806
Surface Water Grab	C15H22O2 (Z = -8, DBE = 5)	3672	0.01	%	217	17.972
Surface Water Grab	C15H24O2 (Z = -6, DBE = 4)	3672	0.01	%	217	9.217
Surface Water Grab	C15H26O2 (Z = -4, DBE = 3)	3672	0.01	%	217	14.286
Surface Water Grab	C15H28O2 (Z = -2, DBE = 2)	3672	0.01	%	217	27.650
Surface Water Grab	C15H30O2 (Z = 0, DBE = 1)	3672	0.01	%	217	40.092
Surface Water Grab	C16H14O2 (Z = -18, DBE = 10)	3672	0.01	%	217	83.410
Surface Water Grab	C16H16O2 (Z = -16, DBE = 9)	3672	0.01	%	217	59.908
Surface Water Grab	C16H18O2 (Z = -14, DBE = 8)	3672	0.01	%	217	51.613
Surface Water Grab	C16H20O2 (Z = -12, DBE = 7)	3672	0.01	%	217	23.963
Surface Water Grab	C16H22O2 (Z = -10, DBE = 6)	3672	0.01	%	217	14.286
Surface Water Grab	C16H24O2 (Z = -8, DBE = 5)	3672	0.01	%	217	7.834
Surface Water Grab	C16H26O2 (Z = -6, DBE = 4)	3672	0.01	%	217	13.364
Surface Water Grab	C16H28O2 (Z = -4, DBE = 3)	3672	0.01	%	217	18.894
Surface Water Grab	C16H30O2 (Z = -2, DBE = 2)	3672	0.01	%	217	28.111
Surface Water Grab	C16H32O2 (Z = 0, DBE = 1)	3672	0.01	%	217	49.309
Surface Water Grab	C17H18O2 (Z = -16, DBE = 9)	3672	0.01	%	217	26.267
Surface Water Grab	C17H20O2 (Z = -14, DBE = 8)	3672	0.01	%	217	16.590
Surface Water Grab	C17H22O2 (Z = -12, DBE = 7)	3672	0.01	%	217	6.452
Surface Water Grab	C17H24O2 (Z = -10, DBE = 6)	3672	0.01	%	217	9.217
Surface Water Grab	C17H26O2 (Z = -8, DBE = 5)	3672	0.01	%	217	13.825
Surface Water Grab	C17H28O2 (Z = -6, DBE = 4)	3672	0.01	%	217	15.668
Surface Water Grab	C17H30O2 (Z = -4, DBE = 3)	3672	0.01	%	217	19.816
Surface Water Grab	C17H32O2 (Z = -2, DBE = 2)	3672	0.01	%	217	29.032
Surface Water Grab	C17H34O2 (Z = 0, DBE = 1)	3672	0.01	%	217	70.507
Surface Water Grab	C18H20O2 (Z = -16, DBE = 9)	3672	0.01	%	217	21.659
Surface Water Grab	C18H22O2 (Z = -14, DBE = 8)	3672	0.01	%	217	8.295
Surface Water Grab	C18H24O2 (Z = -12, DBE = 7)	3672	0.01	%	217	7.834
Surface Water Grab	C18H26O2 (Z = -10, DBE = 6)	3672	0.01	%	217	10.599
Surface Water Grab	C18H28O2 (Z = -8, DBE = 5)	3672	0.01	%	217	11.521
Surface Water Grab	C18H30O2 (Z = -6, DBE = 4)	3672	0.01	%	217	22.120
Surface Water Grab	C18H32O2 (Z = -4, DBE = 3)	3672	0.01	%	217	24.885
Surface Water Grab	C18H34O2 (Z = -2, DBE = 2)	3672	0.01	%	217	34.101
Surface Water Grab	C18H36O2 (Z = 0, DBE = 1)	3672	0.01	%	217	31.797
Surface Water Grab	C19H20O2 (Z = -18, DBE = 10)	3672	0.01	%	217	11.982
Surface Water Grab	C19H22O2 (Z = -16, DBE = 9)	3672	0.01	%	217	10.599
Surface Water Grab	C19H24O2 (Z = -14, DBE = 8)	3672	0.01	%	217	7.373
Surface Water Grab	C19H26O2 (Z = -12, DBE = 7)	3672	0.01	%	217	7.373
Surface Water Grab	C19H28O2 (Z = -10, DBE = 6)	3672	0.01	%	217	16.590
Surface Water Grab	C19H30O2 (Z = -8, DBE = 5)	3672	0.01	%	217	14.286
Surface Water Grab	C19H32O2 (Z = -6, DBE = 4)	3672	0.01	%	217	18.894
Surface Water Grab	C19H34O2 (Z = -4, DBE = 3)	3672	0.01	%	217	27.650
Surface Water Grab	C19H36O2 (Z = -2, DBE = 2)	3672	0.01	%	217	54.378

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Surface Water Grab	C19H38O2 (Z = 0, DBE = 1)	3672	0.01	%	217	60.369
Surface Water Grab	C20H22O2 (Z = -18, DBE = 10)	3672	0.01	%	217	9.677
Surface Water Grab	C20H24O2 (Z = -16, DBE = 9)	3672	0.01	%	217	12.442
Surface Water Grab	C20H26O2 (Z = -14, DBE = 8)	3672	0.01	%	217	9.677
Surface Water Grab	C20H28O2 (Z = -12, DBE = 7)	3672	0.01	%	217	22.120
Surface Water Grab	C20H30O2 (Z = -10, DBE = 6)	3672	0.01	%	217	14.286
Surface Water Grab	C20H32O2 (Z = -8, DBE = 5)	3672	0.01	%	217	8.756
Surface Water Grab	C20H34O2 (Z = -6, DBE = 4)	3672	0.01	%	217	21.198
Surface Water Grab	C20H36O2 (Z = -4, DBE = 3)	3672	0.01	%	217	35.023
Surface Water Grab	C20H38O2 (Z = -2, DBE = 2)	3672	0.01	%	217	66.820
Surface Water Grab	C20H40O2 (Z = 0, DBE = 1)	3672	0.01	%	217	55.760
Surface Water Grab	C21H24O2 (Z = -18, DBE = 10)	3672	0.01	%	217	15.668
Surface Water Grab	C21H26O2 (Z = -16, DBE = 9)	3672	0.01	%	217	14.747
Surface Water Grab	C21H28O2 (Z = -14, DBE = 8)	3672	0.01	%	217	20.276
Surface Water Grab	C21H30O2 (Z = -12, DBE = 7)	3672	0.01	%	217	7.373
Surface Water Grab	C21H32O2 (Z = -10, DBE = 6)	3672	0.01	%	217	20.276
Surface Water Grab	C21H34O2 (Z = -8, DBE = 5)	3672	0.01	%	217	45.161
Surface Water Grab	C21H36O2 (Z = -6, DBE = 4)	3672	0.01	%	217	32.258
Surface Water Grab	C21H38O2 (Z = -4, DBE = 3)	3672	0.01	%	217	43.779
Surface Water Grab	C21H40O2 (Z = -2, DBE = 2)	3672	0.01	%	217	53.456
Surface Water Grab	C21H42O2 (Z = 0, DBE = 1)	3672	0.01	%	217	35.945
Surface Water Grab	C22H32O2 (Z = -12, DBE = 7)	3672	0.01	%	217	11.982
Surface Water Grab	C22H34O2 (Z = -10, DBE = 6)	3672	0.01	%	217	18.433
Surface Water Grab	C22H36O2 (Z = -8, DBE = 5)	3672	0.01	%	217	35.945
Surface Water Grab	C22H38O2 (Z = -6, DBE = 4)	3672	0.01	%	217	32.719
Surface Water Grab	C22H40O2 (Z = -4, DBE = 3)	3672	0.01	%	217	61.290
Surface Water Grab	C22H42O2 (Z = -2, DBE = 2)	3672	0.01	%	217	38.249
Surface Water Grab	C22H44O2 (Z = 0, DBE = 1)	3672	0.01	%	217	52.535
Surface Water Grab	C23H32O2 (Z = -14, DBE = 8)	3672	0.01	%	217	33.641
Surface Water Grab	C23H34O2 (Z = -12, DBE = 7)	3672	0.01	%	217	40.092
Surface Water Grab	C23H36O2 (Z = -10, DBE = 6)	3672	0.01	%	217	73.272
Surface Water Grab	C23H38O2 (Z = -8, DBE = 5)	3672	0.01	%	217	68.664
Surface Water Grab	C23H40O2 (Z = -6, DBE = 4)	3672	0.01	%	217	82.488
Surface Water Grab	C23H42O2 (Z = -4, DBE = 3)	3672	0.01	%	217	85.253
Surface Water Grab	C23H44O2 (Z = -2, DBE = 2)	3672	0.01	%	217	80.184
Surface Water Grab	C23H46O2 (Z = 0, DBE = 1)	3672	0.01	%	217	61.290
Surface Water Grab	C24H36O2 (Z = -12, DBE = 7)	3672	0.01	%	217	72.811
Surface Water Grab	C24H38O2 (Z = -10, DBE = 6)	3672	0.01	%	217	86.175
Surface Water Grab	C24H40O2 (Z = -8, DBE = 5)	3672	0.01	%	217	99.078
Surface Water Grab	C24H42O2 (Z = -6, DBE = 4)	3672	0.01	%	217	80.184
Surface Water Grab	C24H44O2 (Z = -4, DBE = 3)	3672	0.01	%	217	94.470
Surface Water Grab	C24H46O2 (Z = -2, DBE = 2)	3672	0.01	%	217	58.986
Surface Water Grab	C24H48O2 (Z = 0, DBE = 1)	3672	0.01	%	217	73.272
Surface Water Grab	C25H38O2 (Z = -12, DBE = 7)	3672	0.01	%	217	95.392
Surface Water Grab	C25H40O2 (Z = -10, DBE = 6)	3672	0.01	%	217	99.078

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Surface Water Grab	C25H42O2 (Z = -8, DBE = 5)	3672	0.01	%	217	98.157
Surface Water Grab	C25H44O2 (Z = -6, DBE = 4)	3672	0.01	%	217	80.184
Surface Water Grab	C25H46O2 (Z = -4, DBE = 3)	3672	0.01	%	217	98.157
Surface Water Grab	C25H48O2 (Z = -2, DBE = 2)	3672	0.01	%	217	95.392
Surface Water Grab	C25H50O2 (Z = 0, DBE = 1)	3672	0.01	%	217	73.272
Surface Water Grab	C5H10O2 (Z = 0, DBE = 1)	3672	0.01	%	217	86.636
Surface Water Grab	C6H12O2 (Z = 0, DBE = 1)	3672	0.01	%	217	41.475
Surface Water Grab	C7H12O2 (Z = -2, DBE = 2)	3672	0.01	%	217	89.862
Surface Water Grab	C7H14O2 (Z = 0, DBE = 1)	3672	0.01	%	217	64.977
Surface Water Grab	C8H14O2 (Z = -2, DBE = 2)	3672	0.01	%	217	94.009
Surface Water Grab	C8H16O2 (Z = 0, DBE = 1)	3672	0.01	%	217	24.424
Surface Water Grab	C9H14O2 (Z = -4, DBE = 3)	3672	0.01	%	217	92.627
Surface Water Grab	C9H16O2 (Z = -2, DBE = 2)	3672	0.01	%	217	73.733
Surface Water Grab	C9H18O2 (Z = 0, DBE = 1)	3672	0.01	%	217	74.654
Surface Water Grab	Cadmium Dissolved	1849	0.002	µg/l	215	11.163
Surface Water Grab	Cadmium Total Recoverable	1847	0.01	µg/l	215	13.023
Surface Water Grab	Calcium Total	1847	0.01	mg/l	141	9.929
Surface Water Grab	Carbon Dissolved Inorganic	3142	1	mg/l	74	13.514
Surface Water Grab	Carbonate (calcd.)	620	1	mg/l	218	100.000
Surface Water Grab	Cations Total	4	-	me/L	214	-
Surface Water Grab	Chloride Dissolved	426	1	mg/l	218	12.385
Surface Water Grab	Chlorine Dissolved	1849	0.2	mg/l	141	10.638
Surface Water Grab	Chlorine Total Recoverable	1847	0.03	mg/l	141	4.255
Surface Water Grab	Chromium Dissolved	1849	0.3	µg/l	215	96.744
Surface Water Grab	Chromium Total Recoverable	1847	0.1	µg/l	215	12.093
Surface Water Grab	Cloud Cover	2234	-	%	103	-
Surface Water Grab	Cobalt Dissolved	1849	0.006	µg/l	215	14.419
Surface Water Grab	Cobalt Total Recoverable	1847	0.002	µg/l	215	9.767
Surface Water Grab	Colour (at Site)	3364	-	-	93	-
Surface Water Grab	Conductance (Field)	2732	-	µS/cm	180	-
Surface Water Grab	Copper Dissolved	1849	0.08	µg/l	215	10.698
Surface Water Grab	Copper Total Recoverable	1847	0.08	µg/l	215	10.233
Surface Water Grab	Euphotic depth	3448	-	m	74	-
Surface Water Grab	Flow Estimate	3364	-	-	93	-
Surface Water Grab	Foam (Visual) At Site	3364	-	-	93	-
Surface Water Grab	Hardness Total (Calcd.) CaCO3	423	0.5	mg/l	218	12.844
Surface Water Grab	Hydroxide (Calcd.)	620	1	mg/l	218	100.000
Surface Water Grab	Ice Cover	3365	-	%	74	-
Surface Water Grab	Ice Thickness Estimate	3366	-	m	74	-
Surface Water Grab	Ion balance	-	-	-	218	26.606
Surface Water Grab	Ionic Balance Difference (Calcd.)	3518	-	%	218	13.761
Surface Water Grab	Lead Dissolved	1849	0.02	µg/l	215	41.395
Surface Water Grab	Lead Total Recoverable	1847	0.004	µg/l	215	8.837
Surface Water Grab	Lithium Dissolved	1849	0.02	µg/l	215	11.628
Surface Water Grab	Lithium Total Recoverable	1847	0.007	µg/l	215	10.233

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Surface Water Grab	Magnesium Dissolved	1516	0.2	mg/l	218	12.844
Surface Water Grab	Manganese Total Recoverable	1847	0.04	µg/l	215	9.302
Surface Water Grab	Molybdenum Dissolved	1849	0.005	µg/l	215	11.628
Surface Water Grab	Molybdenum Total Recoverable	1847	0.002	µg/l	215	9.302
Surface Water Grab	Nickel Dissolved	1849	0.03	µg/l	215	11.163
Surface Water Grab	Nickel Total Recoverable	1847	0.03	µg/l	215	9.767
Surface Water Grab	Nitrate and Nitrite as Nitrogen	2902	0.0042	mg/l	218	36.239
Surface Water Grab	Nitrate as Nitrogen	2902	0.003	mg/l	218	33.486
Surface Water Grab	Nitrite as Nitrogen	2902	0.003	mg/l	218	96.330
Surface Water Grab	Odour (In Sample)	3364	-	-	92	-
Surface Water Grab	Orthophosphate Dissolved	1117	0.003	mg/l	218	29.817
Surface Water Grab	Oxygen Dissolved (% Saturation)	3620	-	%	181	-
Surface Water Grab	Oxygen Dissolved (Field Meter)	1788	-	mg/L	181	-
Surface Water Grab	pH	389	-	---	218	-
Surface Water Grab	pH (Field)	2731	-	-	181	-
Surface Water Grab	Phenolic material	154	0.001	mg/l	46	63.043
Surface Water Grab	Phenols total	3107	0.001	mg/l	170	64.706
Surface Water Grab	Phenols total	3607	0.001	mg/l	170	64.706
Surface Water Grab	Potassium Dissolved/Filtered	1516	0.3	mg/l	218	12.844
Surface Water Grab	Redox Potential	31	-	mV	181	-
Surface Water Grab	Rhenium dissolved	1849	0.005	µg/l	74	58.108
Surface Water Grab	Rhenium total	1847	0.005	µg/l	74	58.108
Surface Water Grab	River Width	2920	-	m	181	-
Surface Water Grab	Secchi Depth	51	-	m	74	-
Surface Water Grab	Selenium Dissolved	1849	0.2	µg/l	215	44.186
Surface Water Grab	Selenium Total Recoverable	1847	0.2	µg/l	215	18.605
Surface Water Grab	Silver Dissolved	1849	0.003	µg/l	215	96.744
Surface Water Grab	Silver Total Recoverable	1847	0.001	µg/l	215	11.628
Surface Water Grab	Snow Cover In Immediate Area	3365	-	%	74	-
Surface Water Grab	Snow Cover On Ice	3365	-	%	74	-
Surface Water Grab	Snow Depth On Ice Estimate	3366	-	m	74	-
Surface Water Grab	Sodium Adsorption Ratio (Calcd.)	453	0.1	---	130	3.077
Surface Water Grab	Sodium Dissolved/Filtered	1516	0.5	mg/l	218	12.844
Surface Water Grab	Specific Conductance	32	2	µS/cm	218	11.927
Surface Water Grab	Strontium Dissolved	1849	0.07	µg/l	215	9.767
Surface Water Grab	Strontium Total Recoverable	1847	0.07	µg/l	215	9.767
Surface Water Grab	Temperature	2733	-	C	181	-
Surface Water Grab	Temperature (Air)	2182	-	C	177	-
Surface Water Grab	Thallium Dissolved	1849	0.002	µg/l	215	19.070
Surface Water Grab	Thallium Total Recoverable	1847	0.002	µg/l	215	16.279
Surface Water Grab	Thorium Dissolved	1849	0.002	µg/l	215	11.163
Surface Water Grab	Thorium Total Recoverable	1847	0.002	µg/l	215	10.698
Surface Water Grab	Tin Dissolved	1849	0.06	µg/l	215	100.000
Surface Water Grab	Tin Total Recoverable	1847	0.06	µg/l	215	100.000
Surface Water Grab	Titanium Dissolved	1849	0.03	µg/l	215	11.163

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Surface Water Grab	Titanium Total Recoverable	1847	0.03	µg/l	215	8.372
Surface Water Grab	Total Dissolved Solids (Filterable Residue)	8	10	mg/l	436	11.468
Surface Water Grab	Total Dissolved Solids (Filterable Residue)	427	10	mg/l	436	11.468
Surface Water Grab	Turbidity (Visual) At Site	3364	-	-	92	-
Surface Water Grab	Uranium Dissolved	1849	0.002	µg/l	213	11.737
Surface Water Grab	Uranium Total Recoverable	1847	0.002	µg/l	215	11.163
Surface Water Grab	Vanadium Dissolved	1849	0.006	µg/l	215	10.233
Surface Water Grab	Vanadium Total Recoverable	1847	0.007	µg/l	215	8.372
Surface Water Grab	Water Depth Estimate	3366	-	m	176	-
Surface Water Grab	Xylene	3642	0.00071	mg/l	47	100.000
Surface Water Grab	Zinc Dissolved	1849	0.3	µg/l	215	33.488
Surface Water Grab	Zinc Total Recoverable	1847	0.2	µg/l	215	5.116
SPMD	1,2-Dimethylnaphthalene	-	0.0162 - 0.05	ng/L	127	4.724
SPMD	1,2,6-Trimethylphenanthrene	-	0.0609 - 0.1301	ng/L	127	1.575
SPMD	1,8-Dimethylphenanthrene	-	0.013 - 0.0496	ng/L	127	20.472
SPMD	2-Methylanthracene	-	0.0033 - 0.1434	ng/L	127	81.102
SPMD	2-Methylfluorene	-	0.0258 - 0.1673	ng/L	127	14.961
SPMD	7-Methylbenzo(a)Pyrene	-	0.0035 - 0.1181	ng/L	127	18.110
SPMD	Acenaphthylene	-	0.0168 - 0.1325	ng/L	127	35.433
SPMD	Anthracene	-	0.0208 - 0.19	ng/L	127	42.520
SPMD	Benz(a)Anthracene/Chrysene-C4	-	0.0037 - 0.1471	ng/L	127	70.866
SPMD	Benzo(a)Anthracene	-	0.0099 - 0.0359	ng/L	127	4.724
SPMD	Benzo(a)Pyrene	-	0.0061 - 0.0926	ng/L	127	7.087
SPMD	Benzo(b,k)Fluoranthene/Benzo(a)Pyrene-C2	-	0.0052 - 0.0196	ng/L	127	7.874
SPMD	Benzo(g,h,i)Perylene	-	0.0042 - 0.0163	ng/L	127	1.575
SPMD	Benzo(J,K)Fluoranthenes	-	0.0265 - 0.0741	ng/L	127	1.575
SPMD	Dibenzo(a,h)Anthracene	-	0.0015 - 0.0961	ng/L	127	58.268
SPMD	Dibenzothiophene	-	0.0401 - 0.1632	ng/L	127	18.110
SPMD	Fluorene	-	0.0617 - 0.2822	ng/L	127	2.362
SPMD	Indeno(1,2,3-c,d)Pyrene	-	0.0051 - 0.0155	ng/L	127	8.661
SPMD	Methyl Acenaphthene	-	0.0069 - 0.1472	ng/L	127	6.299
SPMD	1-Methylchrysene	-	0.037319615	ng/L	127	0.787
SPMD	1,7-Dimethylfluorene	-	0.115351401	ng/L	127	0.787
SPMD	2/3-Methyldibenzothiophenes	-	0.194069028	ng/L	127	0.787
SPMD	5,9-Dimethylchrysene	-	0.08336699	ng/L	127	0.787
SPMD	5/6-Methylchrysene	-	0.0377561	ng/L	127	0.787
SPMD	Benz(a)Anthracene/Chrysene-C3	-	0.112440221	ng/L	127	0.787
SPMD	Benzo(a)Anthracene/Chrysene-C2	-	0.08336699	ng/L	127	0.787
SPMD	Benzo(e)Pyrene	-	0.089801244	ng/L	127	0.787
SPMD	Benzofluoranthene/Benzopyrene-C1	-	0.118118003	ng/L	127	0.787
SPMD	Dibenzothiophene-C1	-	0.153465226	ng/L	127	0.787

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
SPMD	Dibenzothiophene-C3	-	0.139315262	ng/L	127	0.787
SPMD	Dibenzothiophene-C4	-	0.111754321	ng/L	127	0.787
SPMD	Fluoranthene/Pyrene-C4	-	0.083604991	ng/L	127	0.787
SPMD	Perylene	-	0.089514993	ng/L	127	0.787
SPMD	1-Methylnaphthalene	-	-	ng/L	127	-
SPMD	1-Methylphenanthrene	-	-	ng/L	127	-
SPMD	1,4,6,7-Tetramethylnaphthalene	-	-	ng/L	127	-
SPMD	1,7-Dimethylphenanthrene	-	-	ng/L	127	-
SPMD	2-Methylnaphthalene	-	-	ng/L	127	-
SPMD	2-Methylphenanthrene	-	-	ng/L	127	-
SPMD	2,3,5-Trimethylnaphthalene	-	-	ng/L	127	-
SPMD	2,3,6-Trimethylnaphthalene	-	-	ng/L	127	-
SPMD	2,4-Dimethyldibenzothiophene	-	-	ng/L	127	-
SPMD	2,6-Dimethylnaphthalene	-	-	ng/L	127	-
SPMD	2,6-Dimethylphenanthrene	-	-	ng/L	127	-
SPMD	3-Methylfluoranthene/Benzo(a)Fluorene	-	-	ng/L	127	-
SPMD	3-Methylphenanthrene	-	-	ng/L	127	-
SPMD	3,6-Dimethylphenanthrene	-	-	ng/L	127	-
SPMD	9/4-Methylphenanthrene	-	-	ng/L	127	-
SPMD	Acenaphthene	-	-	ng/L	127	-
SPMD	Benz(a)Anthracene/Chrysene-C1	-	-	ng/L	127	-
SPMD	Benzo(B)Fluoranthene	-	-	ng/L	127	-
SPMD	Benzophenanthrene	-	-	ng/L	127	-
SPMD	Biphenyl	-	-	ng/L	127	-
SPMD	C1-Fluoranthenes/Pyrenes	-	-	ng/L	127	-
SPMD	C1-Fluorenes	-	-	ng/L	127	-
SPMD	C1-Naphthalenes	-	-	ng/L	127	-
SPMD	C1-Phenanthrenes/Anthracenes	-	-	ng/L	127	-
SPMD	C2-Fluoranthenes/Pyrenes	-	-	ng/L	127	-
SPMD	C2-Fluorenes	-	-	ng/L	127	-
SPMD	C2-Naphthalenes	-	-	ng/L	127	-
SPMD	C2-Phenanthrenes/Anthracenes	-	-	ng/L	127	-
SPMD	C3-Fluorenes	-	-	ng/L	127	-
SPMD	C3-Naphthalenes	-	-	ng/L	127	-
SPMD	C3-Phenanthrenes/Anthracenes	-	-	ng/L	127	-
SPMD	C4-Naphthalenes	-	-	ng/L	127	-
SPMD	C4-Phenanthrenes/Anthracenes	-	-	ng/L	127	-
SPMD	Chrysene	-	-	ng/L	127	-
SPMD	Dibenzothiophene-C2	-	-	ng/L	127	-
SPMD	Dimethyl Biphenyl	-	-	ng/L	127	-
SPMD	Fluoranthene	-	-	ng/L	127	-
SPMD	Fluoranthene/Pyrene-C3	-	-	ng/L	127	-
SPMD	Methyl Biphenyl	-	-	ng/L	127	-
SPMD	Naphthalene	-	-	ng/L	127	-
SPMD	Phenanthrene	-	-	ng/L	127	-

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
SPMD	Retene	-	-	ng/L	127	-
Sediments	Extractable Phenols	-	0.02 - 0.2	mg/kg	86	94.186
Sediments	Moisture	-	0.3	%	209	-
Sediments	Total Aluminum (Al)	103475	100	mg/kg	123	1.626
Sediments	Total Antimony (Sb)	103501	0.1	mg/kg	123	13.821
Sediments	Total Arsenic (As)	103476	0.2 - 0.5	mg/kg	123	-
Sediments	Total Barium (Ba)	103478	0.1	mg/kg	123	-
Sediments	Total Beryllium (Be)	103479	0.2	mg/kg	123	19.512
Sediments	Total Bismuth (Bi)	103480	0.1	mg/kg	123	83.740
Sediments	Total Boron (B)	103477	1	mg/kg	123	0.813
Sediments	Total Cadmium (Cd)	103482	0.05	mg/kg	123	12.195
Sediments	Total Calcium (Ca)	-	100	mg/kg	123	-
Sediments	Total Chromium (Cr)	103485	0.1 - 0.3	mg/kg	123	-
Sediments	Total Cobalt (Co)	103484	0.1 - 0.3	mg/kg	123	-
Sediments	Total Copper (Cu)	103486	0.5	mg/kg	123	-
Sediments	Total Iron (Fe)	103487	100	mg/kg	123	-
Sediments	Total Lead (Pb)	103498	0.1	mg/kg	123	-
Sediments	Total Lithium (Li)	103489	0.5 - 5	mg/kg	123	8.130
Sediments	Total Magnesium (Mg)	-	100	mg/kg	123	-
Sediments	Total Manganese (Mn)	103491	0.2	mg/kg	123	-
Sediments	Total Mercury (Hg)	1620	0.05	mg/kg	NA	-
Sediments	Total Molybdenum (Mo)	103492	0.1	mg/kg	123	-
Sediments	Total Nickel (Ni)	103494	0.5 - 0.8	mg/kg	123	-
Sediments	Total Phosphorus (P)	-	10	mg/kg	123	-
Sediments	Total Potassium (K)	-	100	mg/kg	123	2.439
Sediments	Total Selenium (Se)	103502	0.05 - 0.5	mg/kg	123	96.748
Sediments	Total Silver (Ag)	103474	0.05	mg/kg	123	56.098
Sediments	Total Sodium (Na)	-	100	mg/kg	123	77.236
Sediments	Total Strontium (Sr)	103505	0.1	mg/kg	123	-
Sediments	Total Thallium (Tl)	103508	0.05	mg/kg	123	16.260
Sediments	Total Thorium (Th)	103506	0.1	mg/kg	123	-
Sediments	Total Tin (Sn)	103504	0.1	mg/kg	123	14.634
Sediments	Total Titanium (Ti)	103507	1	mg/kg	123	-
Sediments	Total Tungsten (W)	-	0.5	mg/kg	123	100.000
Sediments	Total Uranium (U)	103509	0.05	mg/kg	123	-
Sediments	Total Vanadium (V)	103510	02-Jan	mg/kg	123	-
Sediments	Total Zinc (Zn)	103511	1	mg/kg	123	-
Sediments	Total Zirconium (Zr)	-	0.5	mg/kg	123	0.813
Sediments	Total Mercury	2092	1.6	ng/g	123	-
Sediments	Methyl Mercury	5008	0.01	ng/g	123	-
Sediments	Loss on Ignition @ 375 C	-	1	%	123	31.707
Sediments	Organic Matter	-	1	%	123	21.138
Sediments	Total Kjeldahl Nitrogen	109071	2.00%	%	123	24.390
Sediments	Total Organic Nitrogen	109526	2.00%	%	123	23.577
Sediments	Total Organic Carbon	74471	5.00%	%	123	-

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Sediments	Available Ammonium-N	109037	1	mg/kg	123	4.878
Sediments	C10H16O2 (Z = -4; DBE = 3)	5753	0.01	%	123	67.480
Sediments	C10H18O2 (Z = -2; DBE = 2)	5754	0.01	%	123	5.691
Sediments	C10H20O2 (Z = 0; DBE = 1)	5755	0.01	%	123	53.659
Sediments	C11H14O2 (Z = -8; DBE = 5)	5756	0.01	%	123	49.593
Sediments	C11H16O2 (Z = -6; DBE = 4)	5757	0.01	%	123	86.179
Sediments	C11H18O2 (Z = -4; DBE = 3)	5758	0.01	%	123	54.472
Sediments	C11H20O2 (Z = -2; DBE = 2)	5759	0.01	%	123	47.154
Sediments	C11H22O2 (Z = 0; DBE = 1)	5760	0.01	%	123	34.959
Sediments	C12H16O2 (Z = -8; DBE = 5)	5761	0.01	%	123	52.033
Sediments	C12H18O2 (Z = -6; DBE = 4)	5762	0.01	%	123	83.740
Sediments	C12H20O2 (Z = -4; DBE = 3)	5763	0.01	%	123	37.398
Sediments	C12H22O2 (Z = -2; DBE = 2)	5764	0.01	%	123	-
Sediments	C12H24O2 (Z = 0; DBE = 1)	5765	0.01	%	123	29.268
Sediments	C13H16O2 (Z = -10; DBE = 6)	5766	0.01	%	123	78.049
Sediments	C13H18O2 (Z = -8; DBE = 5)	5767	0.01	%	123	71.545
Sediments	C13H20O2 (Z = -6; DBE = 4)	5767	0.01	%	123	38.211
Sediments	C13H22O2 (Z = -4; DBE = 3)	5769	0.01	%	123	44.715
Sediments	C13H24O2 (Z = -2; DBE = 2)	5770	0.01	%	123	25.203
Sediments	C13H26O2 (Z = 0; DBE = 1)	5771	0.01	%	123	21.138
Sediments	C14H16O2 (Z = -12; DBE = 7)	5772	0.01	%	123	96.748
Sediments	C14H18O2 (Z = -10; DBE = 6)	5773	0.01	%	123	70.732
Sediments	C14H20O2 (Z = -8; DBE = 5)	5774	0.01	%	123	34.959
Sediments	C14H22O2 (Z = -6; DBE = 4)	5775	0.01	%	123	1.626
Sediments	C14H24O2 (Z = -4; DBE = 3)	5776	0.01	%	123	2.439
Sediments	C14H26O2 (Z = -2; DBE = 2)	5777	0.01	%	123	0.813
Sediments	C14H28O2 (Z = 0; DBE = 1)	5778	0.01	%	123	-
Sediments	C15H14O2 (Z = -16; DBE = 9)	5779	0.01	%	123	79.675
Sediments	C15H16O2 (Z = -14; DBE = 8)	5780	0.01	%	123	55.285
Sediments	C15H18O2 (Z = -12; DBE = 7)	5781	0.01	%	123	73.984
Sediments	C15H20O2 (Z = -10; DBE = 6)	5782	0.01	%	123	27.642
Sediments	C15H22O2 (Z = -8; DBE = 5)	5783	0.01	%	123	26.016
Sediments	C15H24O2 (Z = -6; DBE = 4)	5784	0.01	%	123	1.626
Sediments	C15H26O2 (Z = -4; DBE = 3)	5785	0.01	%	123	1.626
Sediments	C15H28O2 (Z = -2; DBE = 2)	5786	0.01	%	123	-
Sediments	C15H30O2 (Z = 0; DBE = 1)	5787	0.01	%	123	17.073
Sediments	C16H14O2 (Z = -18; DBE = 10)	5788	0.01	%	123	64.228
Sediments	C16H16O2 (Z = -16; DBE = 9)	5789	0.01	%	123	86.179
Sediments	C16H18O2 (Z = -14; DBE = 8)	5790	0.01	%	123	69.106
Sediments	C16H20O2 (Z = -12; DBE = 7)	5791	0.01	%	123	30.081
Sediments	C16H22O2 (Z = -10; DBE = 6)	5792	0.01	%	123	16.260
Sediments	C16H24O2 (Z = -8; DBE = 5)	5793	0.01	%	123	-
Sediments	C16H26O2 (Z = -6; DBE = 4)	5794	0.01	%	123	-
Sediments	C16H28O2 (Z = -4; DBE = 3)	5795	0.01	%	123	-
Sediments	C16H30O2 (Z = -2; DBE = 2)	5796	0.01	%	123	-

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Sediments	C16H32O2 (Z = 0; DBE = 1)	5797	0.01	%	123	29.268
Sediments	C17H18O2 (Z = -16; DBE = 9)	5798	0.01	%	123	49.593
Sediments	C17H20O2 (Z = -14; DBE = 8)	5799	0.01	%	123	47.154
Sediments	C17H22O2 (Z = -12; DBE = 7)	5800	0.01	%	123	9.756
Sediments	C17H24O2 (Z = -10; DBE = 6)	5801	0.01	%	123	9.756
Sediments	C17H26O2 (Z = -8; DBE = 5)	5802	0.01	%	123	4.878
Sediments	C17H28O2 (Z = -6; DBE = 4)	5803	0.01	%	123	2.439
Sediments	C17H30O2 (Z = -4; DBE = 3)	5804	0.01	%	123	3.252
Sediments	C17H32O2 (Z = -2; DBE = 2)	5805	0.01	%	123	-
Sediments	C17H34O2 (Z = 0; DBE = 1)	5806	0.01	%	123	17.073
Sediments	C18H20O2 (Z = -16; DBE = 9)	5807	0.01	%	123	36.585
Sediments	C18H22O2 (Z = -14; DBE = 8)	5808	0.01	%	123	21.138
Sediments	C18H24O2 (Z = -12; DBE = 7)	5809	0.01	%	123	26.829
Sediments	C18H26O2 (Z = -10; DBE = 6)	5810	0.01	%	123	5.691
Sediments	C18H28O2 (Z = -8; DBE = 5)	5811	0.01	%	123	-
Sediments	C18H30O2 (Z = -6; DBE = 4)	5812	0.01	%	123	-
Sediments	C18H32O2 (Z = -4; DBE = 3)	5813	0.01	%	123	-
Sediments	C18H34O2 (Z = -2; DBE = 2)	5814	0.01	%	123	19.512
Sediments	C18H36O2 (Z = 0; DBE = 1)	5815	0.01	%	123	-
Sediments	C19H20O2 (Z = -18; DBE = 10)	5816	0.01	%	123	33.333
Sediments	C19H22O2 (Z = -16; DBE = 9)	5817	0.01	%	123	31.707
Sediments	C19H24O2 (Z = -14; DBE = 8)	5818	0.01	%	123	7.317
Sediments	C19H26O2 (Z = -12; DBE = 7)	5819	0.01	%	123	8.130
Sediments	C19H28O2 (Z = -10; DBE = 6)	5820	0.01	%	123	1.626
Sediments	C19H30O2 (Z = -8; DBE = 5)	5821	0.01	%	123	-
Sediments	C19H32O2 (Z = -6; DBE = 4)	5822	0.01	%	123	2.439
Sediments	C19H34O2 (Z = -4; DBE = 3)	5823	0.01	%	123	-
Sediments	C19H36O2 (Z = -2; DBE = 2)	5824	0.01	%	123	32.520
Sediments	C19H38O2 (Z = 0; DBE = 1)	5825	0.01	%	123	64.228
Sediments	C20H22O2 (Z = -18; DBE = 10)	5826	0.01	%	123	37.398
Sediments	C20H24O2 (Z = -16; DBE = 9)	5827	0.01	%	123	8.130
Sediments	C20H26O2 (Z = -14; DBE = 8)	5828	0.01	%	123	20.325
Sediments	C20H28O2 (Z = -12; DBE = 7)	5829	0.01	%	123	21.138
Sediments	C20H30O2 (Z = -10; DBE = 6)	5830	0.01	%	123	29.268
Sediments	C20H32O2 (Z = -8; DBE = 5)	5831	0.01	%	123	-
Sediments	C20H34O2 (Z = -6; DBE = 4)	5832	0.01	%	123	2.439
Sediments	C20H36O2 (Z = -4; DBE = 3)	5833	0.01	%	123	-
Sediments	C20H38O2 (Z = -2; DBE = 2)	5834	0.01	%	123	35.772
Sediments	C20H40O2 (Z = 0; DBE = 1)	5835	0.01	%	123	28.455
Sediments	C21H24O2 (Z = -18; DBE = 10)	5836	0.01	%	123	17.073
Sediments	C21H26O2 (Z = -16; DBE = 9)	5837	0.01	%	123	71.545
Sediments	C21H28O2 (Z = -14; DBE = 8)	5838	0.01	%	123	34.146
Sediments	C21H30O2 (Z = -12; DBE = 7)	5839	0.01	%	123	1.626
Sediments	C21H32O2 (Z = -10; DBE = 6)	5840	0.01	%	123	2.439
Sediments	C21H34O2 (Z = -8; DBE = 5)	5841	0.01	%	123	4.878

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Sediments	C21H36O2 (Z = -6; DBE = 4)	5842	0.01	%	123	-
Sediments	C21H38O2 (Z = -4; DBE = 3)	5843	0.01	%	123	1.626
Sediments	C21H40O2 (Z = -2; DBE = 2)	5844	0.01	%	123	25.203
Sediments	C21H42O2 (Z = 0; DBE = 1)	5845	0.01	%	123	-
Sediments	C22H32O2 (Z = -12; DBE = 7)	5846	0.01	%	123	-
Sediments	C22H34O2 (Z = -10; DBE = 6)	5847	0.01	%	123	-
Sediments	C22H36O2 (Z = -8; DBE = 5)	5848	0.01	%	123	-
Sediments	C22H38O2 (Z = -6; DBE = 4)	5849	0.01	%	123	8.943
Sediments	C22H40O2 (Z = -4; DBE = 3)	5850	0.01	%	123	-
Sediments	C22H42O2 (Z = -2; DBE = 2)	5851	0.01	%	123	2.439
Sediments	C22H44O2 (Z = 0; DBE = 1)	5852	0.01	%	123	2.439
Sediments	C23H32O2 (Z = -14; DBE = 8)	5853	0.01	%	123	21.951
Sediments	C23H34O2 (Z = -12; DBE = 7)	5854	0.01	%	123	24.390
Sediments	C23H36O2 (Z = -10; DBE = 6)	5855	0.01	%	123	17.886
Sediments	C23H38O2 (Z = -8; DBE = 5)	5856	0.01	%	123	6.504
Sediments	C23H40O2 (Z = -6; DBE = 4)	5857	0.01	%	123	-
Sediments	C23H42O2 (Z = -4; DBE = 3)	5858	0.01	%	123	1.626
Sediments	C23H44O2 (Z = -2; DBE = 2)	5859	0.01	%	123	4.065
Sediments	C23H46O2 (Z = 0; DBE = 1)	5860	0.01	%	123	-
Sediments	C24H36O2 (Z = -12; DBE = 7)	5861	0.01	%	123	31.707
Sediments	C24H38O2 (Z = -10; DBE = 6)	5862	0.01	%	123	8.943
Sediments	C24H40O2 (Z = -8; DBE = 5)	5863	0.01	%	123	8.130
Sediments	C24H42O2 (Z = -6; DBE = 4)	5864	0.01	%	123	-
Sediments	C24H44O2 (Z = -4; DBE = 3)	5865	0.01	%	123	-
Sediments	C24H46O2 (Z = -2; DBE = 2)	5866	0.01	%	123	1.626
Sediments	C24H48O2 (Z = 0; DBE = 1)	5867	0.01	%	123	18.699
Sediments	C25H38O2 (Z = -12; DBE = 7)	5868	0.01	%	123	43.089
Sediments	C25H40O2 (Z = -10; DBE = 6)	5869	0.01	%	123	15.447
Sediments	C25H42O2 (Z = -8; DBE = 5)	5870	0.01	%	123	14.634
Sediments	C25H44O2 (Z = -6; DBE = 4)	5871	0.01	%	123	14.634
Sediments	C25H46O2 (Z = -4; DBE = 3)	5872	0.01	%	123	0.813
Sediments	C25H48O2 (Z = -2; DBE = 2)	5873	0.01	%	123	0.813
Sediments	C25H50O2 (Z = 0; DBE = 1)	5874	0.01	%	123	29.268
Sediments	C5H10O2 (Z = 0; DBE = 1)	5875	0.01	%	123	56.911
Sediments	C6H12O2 (Z = 0; DBE = 1)	5876	0.01	%	123	45.528
Sediments	C7H12O2 (Z = -2; DBE = 2)	5877	0.01	%	123	61.789
Sediments	C7H14O2 (Z = 0; DBE = 1)	5878	0.01	%	123	54.472
Sediments	C8H14O2 (Z = -2; DBE = 2)	5879	0.01	%	123	28.455
Sediments	C8H16O2 (Z = 0; DBE = 1)	5880	0.01	%	123	32.520
Sediments	C9H14O2 (Z = -4; DBE = 3)	5881	0.01	%	123	75.610
Sediments	C9H16O2 (Z = -2; DBE = 2)	5882	0.01	%	123	52.846
Sediments	C9H18O2 (Z = 0; DBE = 1)	5883	0.01	%	123	29.268
Sediments	Naphthenic acids	5752	1	ug/g	123	-
Sediments	1,2,6-Trimethylphenanthrene	1550	0.105 - 2.8	ng/g	123	79.675
Sediments	1,2-Dimethylnaphthalene	1523	0.053 - 0.328	ng/g	123	30.894

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Sediments	1,4,6,7-Tetramethylnaphthalene	1527	0.064 - 1.54	ng/g	123	86.179
Sediments	1,7-Dimethylfluorene	1568	0.071 - 0.846	ng/g	123	60.976
Sediments	1,7-Dimethylphenanthrene	1547	0.058 - 1.11	ng/g	123	0.813
Sediments	1,8-Dimethylphenanthrene	1548	0.058 - 1.11	ng/g	123	33.333
Sediments	1-Methylchrysene	1584	0.095 - 1.74	ng/g	123	-
Sediments	1-Methylnaphthalene	1519	0.037 - 0.297	ng/g	123	0.813
Sediments	1-Methylphenanthrene	1541	0.108 - 1.72	ng/g	123	1.626
Sediments	2,3,5-Trimethylnaphthalene	1525	0.052 - 0.868	ng/g	123	3.252
Sediments	2,3,6-Trimethylnaphthalene	1524	0.049 - 0.816	ng/g	123	-
Sediments	2,4-Dimethyldibenzothiophene	1574	0.066 - 2.13	ng/g	123	74.797
Sediments	2,6-Dimethylnaphthalene	1521	0.043 - 0.268	ng/g	123	2.439
Sediments	2,6-Dimethylphenanthrene	1546	0.058 - 1.11	ng/g	123	5.691
Sediments	2/3-Methyldibenzothiophenes	1572	0.113 - 1.3	ng/g	123	16.260
Sediments	2-Methylanthracene	1539	0.112 - 1.78	ng/g	123	51.220
Sediments	2-Methylfluorene	1532	0.058 - 0.677	ng/g	123	23.577
Sediments	2-Methylnaphthalene	1518	0.035 - 0.281	ng/g	123	-
Sediments	2-Methylphenanthrene	1538	0.107 - 1.71	ng/g	123	2.439
Sediments	3,6-Dimethylphenanthrene	1545	0.059 - 1.11	ng/g	123	100.000
Sediments	3-Methylfluoranthene/Benzo[a]fluorene	1578	0.063 - 1.97	ng/g	123	1.626
Sediments	3-Methylphenanthrene	1537	0.109 - 1.74	ng/g	123	59.350
Sediments	4,6-Dimethyldibenzothiophene	-	0.084 - 1.67	ng/g	123	8.130
Sediments	5,9-Dimethylchrysene	1586	0.074 - 4.1	ng/g	123	-
Sediments	5/6-Methylchrysene	1583	0.097 - 1.76	ng/g	123	0.813
Sediments	7-Methylbenzo[a]pyrene	1590	0.144 - 2.73	ng/g	123	10.569
Sediments	9/4-Methylphenanthrene	1540	0.109 - 1.74	ng/g	123	0.813
Sediments	Acenaphthene	1531	0.04 - 0.264	ng/g	123	16.260
Sediments	Acenaphthylene	1530	0.024 - 0.147	ng/g	123	95.935
Sediments	Anthracene	1535	0.034 - 1.21	ng/g	123	78.049
Sediments	Benz[a]anthracene	-	0.089 - 1.84	ng/g	123	23.577
Sediments	Benzo[a]pyrene	1559	0.119 - 4.44	ng/g	123	11.382
Sediments	Benzo[b]fluoranthene	1556	0.079 - 2.71	ng/g	123	23.577
Sediments	Benzo[e]pyrene	1558	0.109 - 4.17	ng/g	123	0.813
Sediments	Benzo[ghi]perylene	1563	0.139 - 5.32	ng/g	123	0.813
Sediments	Benzo[j,k]fluoranthenes	1557	0.084 - 3.2	ng/g	123	56.098
Sediments	Biphenyl	1529	0.023 - 0.189	ng/g	123	1.626
Sediments	C1 Phenanthrenes/Anthracenes	1542	0.108 - 1.72	ng/g	123	-
Sediments	C1-Acenaphthenes	-	0.031 - 0.307	ng/g	123	52.033
Sediments	C1-Benzo[a]anthracenes/Chrysenes	1582	0.096 - 1.75	ng/g	123	-
Sediments	C1-Benzofluoranthenes/Benzopyrenes	1589	0.144 - 2.73	ng/g	123	-
Sediments	C1-Biphenyls	-	0.024 - 0.151	ng/g	123	-
Sediments	C1-Dibenzothiophenes	1571	0.113 - 1.3	ng/g	123	5.691
Sediments	C1-Fluoranthenes/Pyrenes	1577	0.063 - 1.97	ng/g	123	-
Sediments	C1-Fluorenes	1567	0.058 - 0.677	ng/g	123	1.626
Sediments	C1-Naphthalenes	1520	0.035 - 0.281	ng/g	123	-
Sediments	C2 Phenanthrenes/Anthracenes	1549	0.058 - 1.11	ng/g	123	-

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Sediments	C2-Benzo[a]anthracenes/Chrysenes	1585	0.074 - 4.1	ng/g	123	-
Sediments	C2-Benzofluoranthenes/Benzopyrenes	1591	0.183 - 2.96	ng/g	123	-
Sediments	C2-Biphenyls	-	0.023 - 0.151	ng/g	123	-
Sediments	C2-Dibenzothiophenes	1573	0.066 - 2.13	ng/g	123	-
Sediments	C2-Fluoranthenes/Pyrenes	1579	0.088 - 2.09	ng/g	123	-
Sediments	C2-Fluorenes	1569	0.071 - 0.846	ng/g	123	0.813
Sediments	C2-Naphthalenes	1522	0.053 - 0.328	ng/g	123	-
Sediments	C3-Benzo[a]anthracenes/Chrysenes	1587	0.115 - 1.41	ng/g	123	0.813
Sediments	C3-Dibenzothiophenes	1575	0.094 - 2.17	ng/g	123	0.813
Sediments	C3-Fluoranthenes/Pyrenes	1580	0.106 - 7.75	ng/g	123	0.813
Sediments	C3-Fluorenes	1570	0.164 - 2.95	ng/g	123	1.626
Sediments	C3-Naphthalenes	1526	0.051 - 0.841	ng/g	123	0.813
Sediments	C3-Phenanthrenes/Anthracenes	1551	0.105 - 2.8	ng/g	123	0.813
Sediments	C4-Benzo[a]anthracenes/Chrysenes	1588	0.106 - 1.3	ng/g	123	11.382
Sediments	C4-Dibenzothiophenes	1576	0.14 - 3.95	ng/g	123	0.813
Sediments	C4-Fluoranthenes/Pyrenes	1581	0.111 - 1.54	ng/g	123	0.813
Sediments	C4-Naphthalenes	1528	0.064 - 1.54	ng/g	123	0.813
Sediments	C4-Phenanthrenes/Anthracenes	1553	0.29 - 11.6	ng/g	123	2.439
Sediments	Chrysene	1555	0.089 - 2.16	ng/g	123	6.504
Sediments	Dibenz[a,h]anthracene	1561	0.149 - 2.17	ng/g	123	60.976
Sediments	Dibenzothiophene	1536	0.036 - 0.85	ng/g	123	73.171
Sediments	Fluoranthene	1543	0.051 - 1.53	ng/g	123	0.813
Sediments	Fluorene	1533	0.032 - 0.295	ng/g	123	8.943
Sediments	Indeno[1,2,3-cd]pyrene	1562	0.155 - 2.69	ng/g	NA	-
Sediments	Naphthalene	1517	0.048 - 0.33	ng/g	123	8.130
Sediments	Perylene	1560	0.117 - 4.24	ng/g	123	0.813
Sediments	Phenanthrene	1534	0.035 - 1.2	ng/g	123	-
Sediments	Pyrene	1544	0.05 - 1.51	ng/g	123	-
Sediments	Retene	1552	0.29 - 11.6	ng/g	123	2.439
Sediments	% clay	-	-	%	163	4.294
Sediments	% Moisture	-	-	%	123	-
Sediments	% sand	-	-	%	163	-
Sediments	% silt	-	-	%	163	6.135
Sediments	Ideno[1,2,3-cd]pyrene	-	-	ng/g	123	19.512
Sediments	Texture	-	-	-	163	-
Sediments	Total PAHs	-	-	ng/g	123	-
Trout Perch - Body Burden	Total (Wet Wt) Aluminum (Al)	-	0.5	mg/kg	319	3.135
Trout Perch - Body Burden	Total (Wet Wt) Antimony (Sb)	-	0.002	mg/kg	319	87.147
Trout Perch - Body Burden	Total (Wet Wt) Arsenic (As)	-	0.005	mg/kg	319	-
Trout Perch - Body Burden	Total (Wet Wt) Barium (Ba)	-	0.01	mg/kg	319	1.254
Trout Perch - Body Burden	Total (Wet Wt) Beryllium (Be)	-	0.002	mg/kg	319	97.179
Trout Perch - Body Burden	Total (Wet Wt) Bismuth (Bi)	-	0.0013	mg/kg	319	86.520
Trout Perch - Body Burden	Total (Wet Wt) Boron (B)	-	0.2	mg/kg	319	99.373

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Trout Perch - Body Burden	Total (Wet Wt) Cadmium (Cd)	-	0.0013	mg/kg	319	0.940
Trout Perch - Body Burden	Total (Wet Wt) Calcium (Ca)	-	4	mg/kg	319	0.313
Trout Perch - Body Burden	Total (Wet Wt) Chromium (Cr)	-	0.025	mg/kg	319	72.414
Trout Perch - Body Burden	Total (Wet Wt) Cobalt (Co)	-	0.0013	mg/kg	319	-
Trout Perch - Body Burden	Total (Wet Wt) Copper (Cu)	-	0.013	mg/kg	319	-
Trout Perch - Body Burden	Total (Wet Wt) Iron (Fe)	-	0.25	mg/kg	319	-
Trout Perch - Body Burden	Total (Wet Wt) Lead (Pb)	-	0.0013	mg/kg	319	1.254
Trout Perch - Body Burden	Total (Wet Wt) Magnesium (Mg)	-	0.4	mg/kg	319	-
Trout Perch - Body Burden	Total (Wet Wt) Manganese (Mn)	-	0.1	mg/kg	319	0.940
Trout Perch - Body Burden	Total (Wet Wt) Mercury (Hg)	-	0.013	mg/kg	319	-
Trout Perch - Body Burden	Total (Wet Wt) Molybdenum (Mo)	-	0.008	mg/kg	319	1.254
Trout Perch - Body Burden	Total (Wet Wt) Nickel (Ni)	-	0.01	mg/kg	319	0.313
Trout Perch - Body Burden	Total (Wet Wt) Phosphorus (P)	-	2	mg/kg	319	0.313
Trout Perch - Body Burden	Total (Wet Wt) Potassium (K)	-	2.5	mg/kg	319	-
Trout Perch - Body Burden	Total (Wet Wt) Selenium (Se)	-	0.01	mg/kg	319	-
Trout Perch - Body Burden	Total (Wet Wt) Silver (Ag)	-	0.0013	mg/kg	319	81.505
Trout Perch - Body Burden	Total (Wet Wt) Sodium (Na)	-	2.5	mg/kg	319	-
Trout Perch - Body Burden	Total (Wet Wt) Strontium (Sr)	-	0.013	mg/kg	319	0.313
Trout Perch - Body Burden	Total (Wet Wt) Thallium (Tl)	-	0.0004	mg/kg	319	-
Trout Perch - Body Burden	Total (Wet Wt) Tin (Sn)	-	0.02	mg/kg	319	85.266
Trout Perch - Body Burden	Total (Wet Wt) Titanium (Ti)	-	0.13	mg/kg	319	-
Trout Perch - Body Burden	Total (Wet Wt) Uranium (U)	-	0.0004	mg/kg	319	31.348
Trout Perch - Body Burden	Total (Wet Wt) Vanadium (V)	-	0.02	mg/kg	319	31.975
Trout Perch - Body Burden	Total (Wet Wt) Zinc (Zn)	-	0.2	mg/kg	319	-
Walleye - Body Burden	Total (Wet Wt) Aluminum (Al)	-	0.5	mg/kg	120	63.333
Walleye - Body Burden	Total (Wet Wt) Antimony (Sb)	-	0.002	mg/kg	120	98.333
Walleye - Body Burden	Total (Wet Wt) Arsenic (As)	-	0.005	mg/kg	120	-
Walleye - Body Burden	Total (Wet Wt) Barium (Ba)	-	0.01	mg/kg	120	31.667
Walleye - Body Burden	Total (Wet Wt) Beryllium (Be)	-	0.002	mg/kg	120	100.000
Walleye - Body Burden	Total (Wet Wt) Bismuth (Bi)	-	0.0013	mg/kg	120	44.167
Walleye - Body Burden	Total (Wet Wt) Boron (B)	-	0.2	mg/kg	120	100.000
Walleye - Body Burden	Total (Wet Wt) Cadmium (Cd)	-	0.0013	mg/kg	120	98.333
Walleye - Body Burden	Total (Wet Wt) Calcium (Ca)	-	4	mg/kg	120	-
Walleye - Body Burden	Total (Wet Wt) Chromium (Cr)	-	0.025	mg/kg	120	90.833
Walleye - Body Burden	Total (Wet Wt) Cobalt (Co)	-	0.0013	mg/kg	120	31.667
Walleye - Body Burden	Total (Wet Wt) Copper (Cu)	-	0.013	mg/kg	120	-
Walleye - Body Burden	Total (Wet Wt) Iron (Fe)	-	0.25	mg/kg	120	-
Walleye - Body Burden	Total (Wet Wt) Lead (Pb)	-	0.0013	mg/kg	120	84.167
Walleye - Body Burden	Total (Wet Wt) Magnesium (Mg)	-	0.4	mg/kg	120	-

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Walleye - Body Burden	Total (Wet Wt) Manganese (Mn)	-	0.01	mg/kg	120	-
Walleye - Body Burden	Total (Wet Wt) Mercury (Hg)	-	0.013	mg/kg	120	-
Walleye - Body Burden	Total (Wet Wt) Molybdenum (Mo)	-	0.008	mg/kg	120	100.000
Walleye - Body Burden	Total (Wet Wt) Nickel (Ni)	-	0.01	mg/kg	120	91.667
Walleye - Body Burden	Total (Wet Wt) Phosphorus (P)	-	2	mg/kg	120	-
Walleye - Body Burden	Total (Wet Wt) Potassium (K)	-	2.5	mg/kg	120	-
Walleye - Body Burden	Total (Wet Wt) Selenium (Se)	-	0.01	mg/kg	120	-
Walleye - Body Burden	Total (Wet Wt) Silver (Ag)	-	0.0013	mg/kg	120	100.000
Walleye - Body Burden	Total (Wet Wt) Sodium (Na)	-	2.5	mg/kg	120	-
Walleye - Body Burden	Total (Wet Wt) Strontium (Sr)	-	0.013	mg/kg	120	-
Walleye - Body Burden	Total (Wet Wt) Thallium (Tl)	-	0.0004	mg/kg	120	-
Walleye - Body Burden	Total (Wet Wt) Tin (Sn)	-	0.02	mg/kg	120	98.333
Walleye - Body Burden	Total (Wet Wt) Titanium (Ti)	-	0.13	mg/kg	120	44.167
Walleye - Body Burden	Total (Wet Wt) Uranium (U)	-	0.0004	mg/kg	120	95.000
Walleye - Body Burden	Total (Wet Wt) Vanadium (V)	-	0.02	mg/kg	120	99.167
Walleye - Body Burden	Total (Wet Wt) Zinc (Zn)	-	0.2	mg/kg	120	-
White Sucker - Body Burden	Total (Wet Wt) Aluminum (Al)	-	0.5	mg/kg	92	72.826
White Sucker - Body Burden	Total (Wet Wt) Antimony (Sb)	-	0.002	mg/kg	92	100.000
White Sucker - Body Burden	Total (Wet Wt) Arsenic (As)	-	0.005	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Barium (Ba)	-	0.01	mg/kg	92	1.087
White Sucker - Body Burden	Total (Wet Wt) Beryllium (Be)	-	0.002	mg/kg	92	100.000
White Sucker - Body Burden	Total (Wet Wt) Bismuth (Bi)	-	0.0013	mg/kg	92	79.348
White Sucker - Body Burden	Total (Wet Wt) Boron (B)	-	0.2	mg/kg	92	100.000
White Sucker - Body Burden	Total (Wet Wt) Cadmium (Cd)	-	0.0013	mg/kg	92	100.000
White Sucker - Body Burden	Total (Wet Wt) Calcium (Ca)	-	4	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Chromium (Cr)	-	0.025	mg/kg	92	94.565
White Sucker - Body Burden	Total (Wet Wt) Cobalt (Co)	-	0.0013	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Copper (Cu)	-	0.013	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Iron (Fe)	-	0.25	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Lead (Pb)	-	0.0013	mg/kg	92	89.130
White Sucker - Body Burden	Total (Wet Wt) Magnesium (Mg)	-	0.4	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Manganese (Mn)	-	0.01	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Mercury (Hg)	-	0.013	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Molybdenum (Mo)	-	0.008	mg/kg	92	98.913
White Sucker - Body Burden	Total (Wet Wt) Nickel (Ni)	-	0.01	mg/kg	92	95.652
White Sucker - Body Burden	Total (Wet Wt) Phosphorus (P)	-	2	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Potassium (K)	-	2.5	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Selenium (Se)	-	0.01	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Silver (Ag)	-	0.0013	mg/kg	92	100.000

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
White Sucker - Body Burden	Total (Wet Wt) Sodium (Na)	-	2.5	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Strontium (Sr)	-	0.013	mg/kg	92	-
White Sucker - Body Burden	Total (Wet Wt) Thallium (Tl)	-	0.0004	mg/kg	92	4.348
White Sucker - Body Burden	Total (Wet Wt) Tin (Sn)	-	0.02	mg/kg	92	100.000
White Sucker - Body Burden	Total (Wet Wt) Titanium (Ti)	-	0.13	mg/kg	92	51.087
White Sucker - Body Burden	Total (Wet Wt) Uranium (U)	-	0.0004	mg/kg	92	96.739
White Sucker - Body Burden	Total (Wet Wt) Vanadium (V)	-	0.02	mg/kg	92	100.000
White Sucker - Body Burden	Total (Wet Wt) Zinc (Zn)	-	0.2	mg/kg	92	-
Trout Perch - Body Burden	Total Mercury	2092	1.6	ng/g	NA	-
Trout Perch - Body Burden	Methyl Mercury	5008	0.01	ng/g	NA	-
Walleye - Body Burden	Total Mercury	2092	1.6	ng/g	NA	-
Walleye - Body Burden	Methyl Mercury	5008	0.01	ng/g	NA	-
White Sucker - Body Burden	Total Mercury	2092	1.6	ng/g	NA	-
White Sucker - Body Burden	Methyl Mercury	5008	0.01	ng/g	NA	-
Trout Perch - Body Burden	1,2,6-Trimethylphenanthrene	1456	0.014 - 0.257	ng/g	124	97.581
Trout Perch - Body Burden	1,2-Dimethylnaphthalene	1429	0.028 - 0.313	ng/g	124	15.323
Trout Perch - Body Burden	1,4,6,7-Tetramethylnaphthalene	1433	0.026 - 0.516	ng/g	124	99.194
Trout Perch - Body Burden	1,7-Dimethylfluorene	1474	0.018 - 0.348	ng/g	124	98.387
Trout Perch - Body Burden	1,7-Dimethylphenanthrene	1453	0.015 - 0.730	ng/g	124	92.742
Trout Perch - Body Burden	1,8-Dimethylphenanthrene	1454	0.015 - 0.739	ng/g	124	99.194
Trout Perch - Body Burden	1-Methylchrysene	1490	0.009 - 0.116	ng/g	124	81.452
Trout Perch - Body Burden	1-Methylnaphthalene	1425	0.019 - 0.164	ng/g	124	1.613
Trout Perch - Body Burden	1-Methylphenanthrene	1447	0.018 - 0.555	ng/g	124	70.161
Trout Perch - Body Burden	2,3,5-Trimethylnaphthalene	1431	0.022 - 0.176	ng/g	124	43.548
Trout Perch - Body Burden	2,3,6-Trimethylnaphthalene	1430	0.021 - 0.167	ng/g	124	5.645
Trout Perch - Body Burden	2,4-Dimethyldibenzothiophene	1480	0.012 - 0.581	ng/g	124	99.194
Trout Perch - Body Burden	2,6-Dimethylnaphthalene	1427	0.022 - 0.254	ng/g	124	1.613
Trout Perch - Body Burden	2,6-Dimethylphenanthrene	1452	0.015 - 0.739	ng/g	124	95.968
Trout Perch - Body Burden	2/3-Methyldibenzothiophenes	1478	0.016 - 0.645	ng/g	124	96.774
Trout Perch - Body Burden	2-Methylanthracene	1445	0.019 - 0.603	ng/g	124	72.581
Trout Perch - Body Burden	2-Methylfluorene	1438	0.016 - 0.189	ng/g	124	60.484
Trout Perch - Body Burden	2-Methylnaphthalene	1424	0.018 - 0.156	ng/g	124	1.613
Trout Perch - Body Burden	2-Methylphenanthrene	1444	0.018 - 0.561	ng/g	124	70.161
Trout Perch - Body Burden	3,6-Dimethylphenanthrene	1451	0.015 - 0.748	ng/g	124	100.000
Trout Perch - Body Burden	3-Methylfluoranthene/Benzo[a]fluorene	1484	0.016 - 0.322	ng/g	124	74.194
Trout Perch - Body Burden	3-Methylphenanthrene	1443	0.018 - 0.572	ng/g	124	62.903
Trout Perch - Body Burden	4,6-Dimethyldibenzothiophene	-	0.010 - 0.460	ng/g	124	84.677

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Trout Perch - Body Burden	5,9-Dimethylchrysene	1492	0.014 - 0.089	ng/g	124	89.516
Trout Perch - Body Burden	5/6-Methylchrysene	1489	0.009 - 0.119	ng/g	124	91.129
Trout Perch - Body Burden	7-Methylbenzo[a]pyrene	1496	0.025 - 0.151	ng/g	124	100.000
Trout Perch - Body Burden	9/4-Methylphenanthrene	1446	0.018 - 0.572	ng/g	124	45.968
Trout Perch - Body Burden	Acenaphthene	1437	0.020 - 0.122	ng/g	124	4.032
Trout Perch - Body Burden	Acenaphthylene	1436	0.007 - 0.087	ng/g	124	48.387
Trout Perch - Body Burden	Anthracene	1441	0.015 - 0.495	ng/g	124	38.710
Trout Perch - Body Burden	Benz[a]anthracene	-	0.010 - 0.101	ng/g	124	89.516
Trout Perch - Body Burden	Benzo[a]pyrene	1465	0.025 - 0.160	ng/g	124	95.968
Trout Perch - Body Burden	Benzo[b]fluoranthene	1462	0.015 - 0.101	ng/g	124	92.742
Trout Perch - Body Burden	Benzo[e]pyrene	1464	0.023 - 0.147	ng/g	124	92.742
Trout Perch - Body Burden	Benzo[ghi]perylene	1469	0.016 - 0.126	ng/g	124	83.065
Trout Perch - Body Burden	Benzo[j,k]fluoranthenes	1463	0.017 - 0.117	ng/g	124	99.194
Trout Perch - Body Burden	Biphenyl	1435	0.010 - 0.086	ng/g	124	0.806
Trout Perch - Body Burden	C1 Phenanthrenes/Anthracenes	1448	0.018 - 0.555	ng/g	124	32.258
Trout Perch - Body Burden	C1-Acenaphthenes	1472	0.011 - 0.169	ng/g	124	34.677
Trout Perch - Body Burden	C1-Benzo[a]anthracenes/Chrysenes	1488	0.009 - 0.117	ng/g	124	35.484
Trout Perch - Body Burden	C1-Benzofluoranthenes/Benzopyrenes	1495	0.025 - 0.151	ng/g	124	41.129
Trout Perch - Body Burden	C1-Biphenyls	1470	0.009 - 0.167	ng/g	124	0.806
Trout Perch - Body Burden	C1-Dibenzothiophenes	1477	0.016 - 0.645	ng/g	124	65.323
Trout Perch - Body Burden	C1-Fluoranthenes/Pyrenes	1483	0.016 - 0.322	ng/g	124	22.581
Trout Perch - Body Burden	C1-Fluorenes	1473	0.016 - 0.189	ng/g	124	0.806
Trout Perch - Body Burden	C1-Naphthalenes	1426	0.018 - 0.156	ng/g	124	1.613
Trout Perch - Body Burden	C2 Phenanthrenes/Anthracenes	1455	0.015 - 0.739	ng/g	124	3.226
Trout Perch - Body Burden	C2-Benzo[a]anthracenes/Chrysenes	1491	0.014 - 0.089	ng/g	124	23.387
Trout Perch - Body Burden	C2-Benzofluoranthenes/Benzopyrenes	1497	0.017 - 0.145	ng/g	124	35.484
Trout Perch - Body Burden	C2-Biphenyls	1471	0.008 - 0.091	ng/g	124	0.806
Trout Perch - Body Burden	C2-Dibenzothiophenes	1479	0.012 - 0.581	ng/g	124	6.452
Trout Perch - Body Burden	C2-Fluoranthenes/Pyrenes	1485	0.014 - 0.180	ng/g	124	21.774
Trout Perch - Body Burden	C2-Fluorenes	1475	0.018 - 0.348	ng/g	124	-
Trout Perch - Body Burden	C2-Naphthalenes	1428	0.028 - 0.313	ng/g	124	0.806
Trout Perch - Body Burden	C3-Benzo[a]anthracenes/Chrysenes	1491	0.017 - 0.073	ng/g	124	63.710
Trout Perch - Body Burden	C3-Dibenzothiophenes	1481	0.013 - 0.422	ng/g	124	19.355
Trout Perch - Body Burden	C3-Fluoranthenes/Pyrenes	1486	0.015 - 0.124	ng/g	124	56.452
Trout Perch - Body Burden	C3-Fluorenes	1476	0.030 - 0.618	ng/g	124	5.645
Trout Perch - Body Burden	C3-Naphthalenes	1432	0.021 - 0.172	ng/g	124	0.806
Trout Perch - Body Burden	C3-Phenanthrenes/Anthracenes	1457	0.014 - 0.257	ng/g	124	24.194

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Trout Perch - Body Burden	C4-Benzo[a]anthracenes/Chrysenes	1494	0.014 - 0.080	ng/g	124	56.452
Trout Perch - Body Burden	C4-Dibenzothiophenes	1482	0.009 - 0.265	ng/g	124	1.613
Trout Perch - Body Burden	C4-Fluoranthenes/Pyrenes	1487	0.014 - 0.095	ng/g	124	79.032
Trout Perch - Body Burden	C4-Naphthalenes	1434	0.026 - 0.516	ng/g	124	4.839
Trout Perch - Body Burden	C4-Phenanthrenes/Anthracenes	1459	0.035 - 0.849	ng/g	124	8.065
Trout Perch - Body Burden	Chrysene	1461	0.011 - 0.105	ng/g	124	20.161
Trout Perch - Body Burden	Dibenz[a,h]anthracene	1467	0.019 - 0.114	ng/g	124	97.581
Trout Perch - Body Burden	Dibenzothiophene	1442	0.013 - 0.323	ng/g	124	92.742
Trout Perch - Body Burden	Fluoranthene	1449	0.010 - 0.216	ng/g	124	23.387
Trout Perch - Body Burden	Fluorene	1439	0.009 - 0.115	ng/g	124	-
Trout Perch - Body Burden	Indeno[1,2,3-cd]pyrene	1468	0.019 - 0.151	ng/g	124	98.387
Trout Perch - Body Burden	Naphthalene	1423	0.023 - 0.240	ng/g	124	1.613
Trout Perch - Body Burden	Perylene	1466	0.023 - 0.157	ng/g	124	89.516
Trout Perch - Body Burden	Phenanthrene	1440	0.015 - 0.494	ng/g	124	-
Trout Perch - Body Burden	Pyrene	1450	0.010 - 0.213	ng/g	124	36.290
Trout Perch - Body Burden	Retene	1458	0.035 - 0.849	ng/g	124	85.484
Walleye - Body Burden	1,2,6-Trimethylphenanthrene	1456	0.005 - 0.081	ng/g	117	100.000
Walleye - Body Burden	1,2-Dimethylnaphthalene	1429	0.015 - 0.168	ng/g	117	76.923
Walleye - Body Burden	1,4,6,7-Tetramethylnaphthalene	1433	0.005 - 0.092	ng/g	117	88.034
Walleye - Body Burden	1,7-Dimethylfluorene	1474	0.005 - 0.163	ng/g	117	100.000
Walleye - Body Burden	1,7-Dimethylphenanthrene	1453	0.003 - 0.103	ng/g	117	87.179
Walleye - Body Burden	1,8-Dimethylphenanthrene	1454	0.003 - 0.104	ng/g	117	100.000
Walleye - Body Burden	1-Methylchrysene	1490	0.002 - 0.039	ng/g	117	94.872
Walleye - Body Burden	1-Methylnaphthalene	1425	0.007 - 0.080	ng/g	117	5.983
Walleye - Body Burden	1-Methylphenanthrene	1447	0.005 - 0.094	ng/g	117	49.573
Walleye - Body Burden	2,3,5-Trimethylnaphthalene	1431	0.006 - 0.123	ng/g	117	35.043
Walleye - Body Burden	2,3,6-Trimethylnaphthalene	1430	0.005 - 0.117	ng/g	117	12.821
Walleye - Body Burden	2,4-Dimethyldibenzothiophene	1480	0.004 - 0.058	ng/g	117	99.145
Walleye - Body Burden	2,6-Dimethylnaphthalene	1427	0.012 - 0.137	ng/g	117	24.786
Walleye - Body Burden	2,6-Dimethylphenanthrene	1452	0.003 - 0.104	ng/g	117	93.162
Walleye - Body Burden	2/3-Methyldibenzothiophenes	1478	0.005 - 0.070	ng/g	117	94.017
Walleye - Body Burden	2-Methylanthracene	1445	0.005 - 0.102	ng/g	117	96.581
Walleye - Body Burden	2-Methylfluorene	1438	0.005 - 0.063	ng/g	117	49.573
Walleye - Body Burden	2-Methylnaphthalene	1424	0.007 - 0.076	ng/g	117	1.709
Walleye - Body Burden	2-Methylphenanthrene	1444	0.005 - 0.095	ng/g	117	77.778
Walleye - Body Burden	3,6-Dimethylphenanthrene	1451	0.003 - 0.106	ng/g	117	100.000
Walleye - Body Burden	3-Methylfluoranthene/Benzo[a]fluorene	1484	0.003 - 0.056	ng/g	117	56.410
Walleye - Body Burden	3-Methylphenanthrene	1443	0.005 - 0.097	ng/g	117	37.607
Walleye - Body Burden	4,6-Dimethyldibenzothiophene	-	0.003 - 0.046	ng/g	117	97.436
Walleye - Body Burden	5,9-Dimethylchrysene	1492	0.005 - 0.060	ng/g	117	99.145
Walleye - Body Burden	5/6-Methylchrysene	1489	0.002 - 0.039	ng/g	117	95.726

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Walleye - Body Burden	7-Methylbenzo[a]pyrene	1496	0.013 - 0.073	ng/g	117	100.000
Walleye - Body Burden	9/4-Methylphenanthrene	1446	0.005 - 0.097	ng/g	117	29.060
Walleye - Body Burden	Acenaphthene	1437	0.006 - 0.083	ng/g	117	10.256
Walleye - Body Burden	Acenaphthylene	1436	0.004 - 0.074	ng/g	117	60.684
Walleye - Body Burden	Anthracene	1441	0.002 - 0.204	ng/g	117	58.974
Walleye - Body Burden	Benz[a]anthracene	-	0.002 - 0.027	ng/g	117	98.291
Walleye - Body Burden	Benzo[a]pyrene	1465	0.008 - 0.055	ng/g	117	100.000
Walleye - Body Burden	Benzo[b]fluoranthene	1462	0.005 - 0.032	ng/g	117	82.051
Walleye - Body Burden	Benzo[e]pyrene	1464	0.007 - 0.051	ng/g	117	99.145
Walleye - Body Burden	Benzo[ghi]perylene	1469	0.005 - 0.042	ng/g	117	97.436
Walleye - Body Burden	Benzo[j,k]fluoranthenes	1463	0.005 - 0.037	ng/g	117	100.000
Walleye - Body Burden	Biphenyl	1435	0.005 - 0.027	ng/g	117	17.949
Walleye - Body Burden	C1 Phenanthrenes/Anthracenes	1448	0.005 - 0.094	ng/g	117	17.094
Walleye - Body Burden	C1-Acenaphthenes	1472	0.006 - 0.337	ng/g	117	83.761
Walleye - Body Burden	C1-Benzo[a]anthracenes/Chrysenes	1488	0.002 - 0.039	ng/g	117	66.667
Walleye - Body Burden	C1-Benzofluoranthenes/Benzopyrenes	1495	0.013 - 0.073	ng/g	117	60.684
Walleye - Body Burden	C1-Biphenyls	1470	0.004 - 0.046	ng/g	117	8.547
Walleye - Body Burden	C1-Dibenzothiophenes	1477	0.005 - 0.070	ng/g	117	70.940
Walleye - Body Burden	C1-Fluoranthenes/Pyrenes	1483	0.003 - 0.056	ng/g	117	14.530
Walleye - Body Burden	C1-Fluorenes	1473	0.005 - 0.063	ng/g	117	-
Walleye - Body Burden	C1-Naphthalenes	1426	0.007 - 0.076	ng/g	117	1.709
Walleye - Body Burden	C2 Phenanthrenes/Anthracenes	1455	0.003 - 0.104	ng/g	117	-
Walleye - Body Burden	C2-Benzo[a]anthracenes/Chrysenes	1491	0.005 - 0.060	ng/g	117	30.769
Walleye - Body Burden	C2-Benzofluoranthenes/Benzopyrenes	1497	0.007 - 0.077	ng/g	117	47.863
Walleye - Body Burden	C2-Biphenyls	1471	0.005 - 0.046	ng/g	117	-
Walleye - Body Burden	C2-Dibenzothiophenes	1479	0.004 - 0.058	ng/g	117	4.274
Walleye - Body Burden	C2-Fluoranthenes/Pyrenes	1485	0.003 - 0.054	ng/g	117	44.444
Walleye - Body Burden	C2-Fluorenes	1475	0.005 - 0.163	ng/g	117	1.709
Walleye - Body Burden	C2-Naphthalenes	1428	0.015 - 0.168	ng/g	117	6.838
Walleye - Body Burden	C3-Benzo[a]anthracenes/Chrysenes	1491	0.006 - 0.045	ng/g	117	68.376
Walleye - Body Burden	C3-Dibenzothiophenes	1481	0.005 - 0.055	ng/g	117	41.026
Walleye - Body Burden	C3-Fluoranthenes/Pyrenes	1486	0.004 - 0.035	ng/g	117	92.308
Walleye - Body Burden	C3-Fluorenes	1476	0.013 - 0.159	ng/g	117	8.547
Walleye - Body Burden	C3-Naphthalenes	1432	0.005 - 0.120	ng/g	117	-
Walleye - Body Burden	C3-Phenanthrenes/Anthracenes	1457	0.005 - 0.081	ng/g	117	29.915
Walleye - Body Burden	C4-Benzo[a]anthracenes/Chrysenes	1494	0.006 - 0.054	ng/g	117	54.701
Walleye - Body Burden	C4-Dibenzothiophenes	1482	0.004 - 0.041	ng/g	117	23.077
Walleye - Body Burden	C4-Fluoranthenes/Pyrenes	1487	0.004 - 0.040	ng/g	117	94.017
Walleye - Body Burden	C4-Naphthalenes	1434	0.005 - 0.092	ng/g	117	4.274
Walleye - Body Burden	C4-Phenanthrenes/Anthracenes	1459	0.005 - 0.258	ng/g	117	22.222
Walleye - Body Burden	Chrysene	1461	0.002 - 0.031	ng/g	117	21.368
Walleye - Body Burden	Dibenz[a,h]anthracene	1467	0.007 - 0.110	ng/g	117	96.581
Walleye - Body Burden	Dibenzothiophene	1442	0.003 - 0.053	ng/g	117	83.761
Walleye - Body Burden	Fluoranthene	1449	0.002 - 0.039	ng/g	117	2.564
Walleye - Body Burden	Fluorene	1439	0.004 - 0.033	ng/g	117	15.385

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
Walleye - Body Burden	Indeno[1,2,3-cd]pyrene	1468	0.006 - 0.050	ng/g	117	100.000
Walleye - Body Burden	Naphthalene	1423	0.010 - 0.070	ng/g	117	11.111
Walleye - Body Burden	Perylene	1466	0.007 - 0.053	ng/g	117	95.726
Walleye - Body Burden	Phenanthrene	1440	0.002 - 0.204	ng/g	117	0.855
Walleye - Body Burden	Pyrene	1450	0.002 - 0.039	ng/g	117	0.855
Walleye - Body Burden	Retene	1458	0.005 - 0.258	ng/g	117	55.556
White Sucker - Body Burden	1,2,6-Trimethylphenanthrene	1456	0.007 - 0.068	ng/g	55	100.000
White Sucker - Body Burden	1,2-Dimethylnaphthalene	1429	0.010 - 0.236	ng/g	55	81.818
White Sucker - Body Burden	1,4,6,7-Tetramethylnaphthalene	1433	0.009 - 0.135	ng/g	55	100.000
White Sucker - Body Burden	1,7-Dimethylfluorene	1474	0.010 - 0.047	ng/g	55	100.000
White Sucker - Body Burden	1,7-Dimethylphenanthrene	1453	0.006 - 0.049	ng/g	55	89.091
White Sucker - Body Burden	1,8-Dimethylphenanthrene	1454	0.006 - 0.050	ng/g	55	100.000
White Sucker - Body Burden	1-Methylchrysene	1490	0.004 - 0.027	ng/g	55	100.000
White Sucker - Body Burden	1-Methylnaphthalene	1425	0.008 - 0.063	ng/g	55	-
White Sucker - Body Burden	1-Methylphenanthrene	1447	0.007 - 0.056	ng/g	55	23.636
White Sucker - Body Burden	2,3,5-Trimethylnaphthalene	1431	0.008 - 0.178	ng/g	55	30.909
White Sucker - Body Burden	2,3,6-Trimethylnaphthalene	1430	0.007 - 0.169	ng/g	55	9.091
White Sucker - Body Burden	2,4-Dimethyldibenzothiophene	1480	0.008 - 0.065	ng/g	55	98.182
White Sucker - Body Burden	2,6-Dimethylnaphthalene	1427	0.008 - 0.192	ng/g	55	7.273
White Sucker - Body Burden	2,6-Dimethylphenanthrene	1452	0.006 - 0.050	ng/g	55	100.000
White Sucker - Body Burden	2/3-Methyldibenzothiophenes	1478	0.009 - 0.049	ng/g	55	94.545
White Sucker - Body Burden	2-Methylanthracene	1445	0.007 - 0.060	ng/g	55	100.000
White Sucker - Body Burden	2-Methylfluorene	1438	0.005 - 0.055	ng/g	55	43.636
White Sucker - Body Burden	2-Methylnaphthalene	1424	0.008 - 0.060	ng/g	55	-
White Sucker - Body Burden	2-Methylphenanthrene	1444	0.007 - 0.056	ng/g	55	74.545
White Sucker - Body Burden	3,6-Dimethylphenanthrene	1451	0.007 - 0.050	ng/g	55	100.000
White Sucker - Body Burden	3-Methylfluoranthene/Benzo[a]fluorene	1484	0.006 - 0.042	ng/g	55	52.727
White Sucker - Body Burden	3-Methylphenanthrene	1443	0.007 - 0.057	ng/g	55	40.000
White Sucker - Body Burden	4,6-Dimethyldibenzothiophene	-	0.006 - 0.051	ng/g	55	94.545
White Sucker - Body Burden	5,9-Dimethylchrysene	1492	0.006 - 0.034	ng/g	55	100.000
White Sucker - Body Burden	5/6-Methylchrysene	1489	0.004 - 0.028	ng/g	55	100.000
White Sucker - Body Burden	7-Methylbenzo[a]pyrene	1496	0.010 - 0.056	ng/g	55	100.000
White Sucker - Body Burden	9/4-Methylphenanthrene	1446	0.007 - 0.057	ng/g	55	25.455
White Sucker - Body Burden	Acenaphthene	1437	0.006 - 0.050	ng/g	55	-
White Sucker - Body Burden	Acenaphthylene	1436	0.004 - 0.041	ng/g	55	32.727
White Sucker - Body Burden	Anthracene	1441	0.005 - 0.060	ng/g	55	34.545
White Sucker - Body Burden	Benz[a]anthracene	-	0.003 - 0.027	ng/g	55	100.000
White Sucker - Body Burden	Benzo[a]pyrene	1465	0.008 - 0.049	ng/g	55	98.182

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
White Sucker - Body Burden	Benzo[b]fluoranthene	1462	0.005 - 0.028	ng/g	55	100.000
White Sucker - Body Burden	Benzo[e]pyrene	1464	0.007 - 0.045	ng/g	55	98.182
White Sucker - Body Burden	Benzo[ghi]perylene	1469	0.005 - 0.036	ng/g	55	98.182
White Sucker - Body Burden	Benzo[j,k]fluoranthenes	1463	0.006 - 0.034	ng/g	55	100.000
White Sucker - Body Burden	Biphenyl	1435	0.004 - 0.041	ng/g	55	1.818
White Sucker - Body Burden	C1 Phenanthrenes/Anthracenes	1448	0.007 - 0.056	ng/g	55	10.909
White Sucker - Body Burden	C1-Acenaphthenes	1472	0.006 - 0.046	ng/g	55	63.636
White Sucker - Body Burden	C1-Benzo[a]anthracenes/Chrysenes	1488	0.004 - 0.027	ng/g	55	49.091
White Sucker - Body Burden	C1-Benzofluoranthenes/Benzopyrenes	1495	0.010 - 0.056	ng/g	55	25.455
White Sucker - Body Burden	C1-Biphenyls	1470	0.004 - 0.083	ng/g	55	-
White Sucker - Body Burden	C1-Dibenzothiophenes	1477	0.009 - 0.049	ng/g	55	56.364
White Sucker - Body Burden	C1-Fluoranthenes/Pyrenes	1483	0.006 - 0.042	ng/g	55	18.182
White Sucker - Body Burden	C1-Fluorenes	1473	0.005 - 0.055	ng/g	55	-
White Sucker - Body Burden	C1-Naphthalenes	1426	0.008 - 0.060	ng/g	55	-
White Sucker - Body Burden	C2 Phenanthrenes/Anthracenes	1455	0.006 - 0.050	ng/g	55	1.818
White Sucker - Body Burden	C2-Benzo[a]anthracenes/Chrysenes	1491	0.006 - 0.034	ng/g	55	34.545
White Sucker - Body Burden	C2-Benzofluoranthenes/Benzopyrenes	1497	0.009 - 0.065	ng/g	55	12.727
White Sucker - Body Burden	C2-Biphenyls	1471	0.005 - 0.042	ng/g	55	-
White Sucker - Body Burden	C2-Dibenzothiophenes	1479	0.008 - 0.065	ng/g	55	16.364
White Sucker - Body Burden	C2-Fluoranthenes/Pyrenes	1485	0.005 - 0.050	ng/g	55	36.364
White Sucker - Body Burden	C2-Fluorenes	1475	0.010 - 0.047	ng/g	55	-
White Sucker - Body Burden	C2-Naphthalenes	1428	0.010 - 0.236	ng/g	55	-
White Sucker - Body Burden	C3-Benzo[a]anthracenes/Chrysenes	1491	0.007 - 0.039	ng/g	55	49.091
White Sucker - Body Burden	C3-Dibenzothiophenes	1481	0.010 - 0.062	ng/g	55	36.364
White Sucker - Body Burden	C3-Fluoranthenes/Pyrenes	1486	0.004 - 0.023	ng/g	55	89.091
White Sucker - Body Burden	C3-Fluorenes	1476	0.015 - 0.078	ng/g	55	-
White Sucker - Body Burden	C3-Naphthalenes	1432	0.007 - 0.174	ng/g	55	-
White Sucker - Body Burden	C3-Phenanthrenes/Anthracenes	1457	0.007 - 0.068	ng/g	55	34.545
White Sucker - Body Burden	C4-Benzo[a]anthracenes/Chrysenes	1494	0.006 - 0.036	ng/g	55	50.909
White Sucker - Body Burden	C4-Dibenzothiophenes	1482	0.008 - 0.054	ng/g	55	23.636
White Sucker - Body Burden	C4-Fluoranthenes/Pyrenes	1487	0.004 - 0.027	ng/g	55	98.182
White Sucker - Body Burden	C4-Naphthalenes	1434	0.009 - 0.135	ng/g	55	12.727
White Sucker - Body Burden	C4-Phenanthrenes/Anthracenes	1459	0.017 - 0.098	ng/g	55	9.091
White Sucker - Body Burden	Chrysene	1461	0.003 - 0.031	ng/g	55	43.636
White Sucker - Body Burden	Dibenz[a,h]anthracene	1467	0.007 - 0.055	ng/g	55	100.000
White Sucker - Body Burden	Dibenzothiophene	1442	0.006 - 0.038	ng/g	55	96.364
White Sucker - Body Burden	Fluoranthene	1449	0.004 - 0.075	ng/g	55	14.545

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
White Sucker - Body Burden	Fluorene	1439	0.004 - 0.022	ng/g	55	-
White Sucker - Body Burden	Indeno[1,2,3-cd]pyrene	1468	0.005 - 0.045	ng/g	55	100.000
White Sucker - Body Burden	Naphthalene	1423	0.008 - 0.173	ng/g	55	-
White Sucker - Body Burden	Perylene	1466	0.008 - 0.046	ng/g	55	100.000
White Sucker - Body Burden	Phenanthrene	1440	0.005 - 0.060	ng/g	55	-
White Sucker - Body Burden	Pyrene	1450	0.004 - 0.074	ng/g	55	9.091
White Sucker - Body Burden	Retene	1458	0.017 - 0.098	ng/g	55	70.909
Trout Perch - Body Burden	Carbon content	-	-	%	320	-
Trout Perch - Body Burden	δ13C	-	-	‰	320	-
Trout Perch - Body Burden	δ15N	-	-	‰	320	-
Trout Perch - Body Burden	Methyl Mercury (wet weight)	-	-	ng/g	300	-
Trout Perch - Body Burden	Nitrogen content	-	-	%	320	-
Trout Perch - Body Burden	Total Mercury (wet weight)	-	-	ng/g	300	-
Walleye - Body Burden	2-methylphenol	-	1	ng/g	73	31.507
Walleye - Body Burden	2,3-dimethylphenol	-	1	ng/g	73	98.630
Walleye - Body Burden	2,4- & 2,5-dimethylphenol	-	2	ng/g	73	98.630
Walleye - Body Burden	2,6-dimethylphenol	-	1	ng/g	73	98.630
Walleye - Body Burden	3 and 4 methylphenol	-	2	ng/g	73	36.986
Walleye - Body Burden	3,4-dimethylphenol	-	1	ng/g	73	98.630
Walleye - Body Burden	3,5-dimethylphenol	-	1	ng/g	73	97.260
Walleye - Body Burden	Benzothiophene	-	1	ng/g	73	94.521
Walleye - Body Burden	Carbon content	-	-	%	120	-
Walleye - Body Burden	δ13C	-	-	‰	120	-
Walleye - Body Burden	δ15N	-	-	‰	120	-
Walleye - Body Burden	Methyl Mercury (wet weight)	-	-	ng/g	120	-
Walleye - Body Burden	Nitrogen content	-	-	%	120	-
Walleye - Body Burden	Phenol	-	-	ng/g	73	5.479
Walleye - Body Burden	Total Mercury (wet weight)	-	-	ng/g	120	-
White Sucker - Body Burden	2-methylphenol	-	1	ng/g	62	37.097
White Sucker - Body Burden	2,3-dimethylphenol	-	1	ng/g	62	98.387
White Sucker - Body Burden	2,4- & 2,5-dimethylphenol	-	2	ng/g	62	98.387
White Sucker - Body Burden	2,6-dimethylphenol	-	1	ng/g	62	96.774
White Sucker - Body Burden	3 and 4 methylphenol	-	2	ng/g	62	40.323
White Sucker - Body Burden	3,4-dimethylphenol	-	1	ng/g	62	100.000
White Sucker - Body Burden	3,5-dimethylphenol	-	1	ng/g	62	96.774
White Sucker - Body Burden	Benzothiophene	-	1	ng/g	62	98.387
White Sucker - Body Burden	Carbon content	-	-	%	245	-
White Sucker - Body Burden	δ13C	-	-	‰	245	-
White Sucker - Body Burden	δ15N	-	-	‰	245	-

Appendix A - VMV's and DL's

Media	Parameter	VMV Code	DL	Units	Total # of Samples	% Below DL
White Sucker - Body Burden	Methyl Mercury (wet weight)	-	-	ng/g	95	-
White Sucker - Body Burden	Nitrogen content	-	-	%	245	-
White Sucker - Body Burden	Phenol	-	-	ng/g	62	6.452
White Sucker - Body Burden	Total Mercury (wet weight)	-	-	ng/g	95	-
Benthos Body Burden	Carbon content	-	-	%	89	-
Benthos Body Burden	Î´13C	-	-	%	89	-
Benthos Body Burden	Î´15N	-	-	%	89	-
Benthos Body Burden	Methyl Mercury (ng/g) Wet Weight	-	-	ng/g ww	87	-
Benthos Body Burden	Nitrogen content	-	-	%	89	-
Benthos Body Burden	Total (Wet Wt) Aluminum (Al)	ICPMS*	0.5	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Antimony (Sb)	ICPMS*	0.002	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Arsenic (As)	ICPMS*	0.005	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Barium (Ba)	ICPMS*	0.01	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Beryllium (Be)	ICPMS*	0.002	mg/kg	88	1.136
Benthos Body Burden	Total (Wet Wt) Bismuth (Bi)	ICPMS*	0.0013	mg/kg	88	2.273
Benthos Body Burden	Total (Wet Wt) Boron (B)	ICPMS*	0.2	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Cadmium (Cd)	ICPMS*	0.0013	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Calcium (Ca)	ICPMS*	4	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Chromium (Cr)	ICPMS*	0.025	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Cobalt (Co)	ICPMS*	0.0013	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Copper (Cu)	ICPMS*	0.013	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Iron (Fe)	ICPMS*	0.25	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Lead (Pb)	ICPMS*	0.0013	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Magnesium (Mg)	ICPMS*	0.4	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Manganese (Mn)	ICPMS*	0.01	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Molybdenum (Mo)	ICPMS*	0.008	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Nickel (Ni)	ICPMS*	0.01	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Phosphorus (P)	ICPMS*	2	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Potassium (K)	ICPMS*	2.5	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Selenium (Se)	ICPMS*	0.01	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Silver (Ag)	ICPMS*	0.0013	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Sodium (Na)	ICPMS*	2.5	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Strontium (Sr)	ICPMS*	0.013	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Thallium (Tl)	ICPMS*	4.00E-04	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Tin (Sn)	ICPMS*	0.02	mg/kg	88	23.864
Benthos Body Burden	Total (Wet Wt) Titanium (Ti)	ICPMS*	0.13	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Uranium (U)	ICPMS*	4.00E-04	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Vanadium (V)	ICPMS*	0.02	mg/kg	88	-
Benthos Body Burden	Total (Wet Wt) Zinc (Zn)	ICPMS*	0.2	mg/kg	88	-
Benthos Body Burden	Total Mercury (ng/g) Wet Weight	ICPMS*	-	ng/g ww	87	-

Table Note: ICPMS*- no VMV code exists for benthic body burden samples, however they were ran on ICPMS similar to fish body burden samples

Appendix B Summary Statistics

Appendix B - Summary Statistics

Table B1. Summary statistics of algal indices of community for samples collected during the EMP program (2018, 2019 and 2021)

Site	Year	Statistic	Density	Richness (LPL)	Diversity	Evenness	Chlorophyll-*a*	NMDS1	NMDS2
AB07DA0062	2018	Minimum	733575	46	0.85	0.14	0.17	0.24	-0.30
		Maximum	2165288	59	0.92	0.25	0.97	0.67	-0.18
		Arithmetic Mean	1572467	50	0.89	0.21	0.62	0.46	-0.24
		Standard Deviation	746903	8	0.04	0.06	0.41	0.21	0.06
	2019	Minimum	610022	47	0.54	0.05	0.07	0.03	0.19
		Maximum	1209783	53	0.78	0.09	0.85	0.19	0.28
		Arithmetic Mean	822358	49	0.68	0.07	0.34	0.09	0.23
		Standard Deviation	336037	3	0.13	0.02	0.44	0.09	0.05
	2021	Minimum	445193	63	0.91	0.17	0.05	0.33	0.06
		Maximum	1371588	65	0.93	0.23	1.46	0.40	0.16
		Arithmetic Mean	779513	64	0.92	0.21	0.66	0.36	0.11
		Standard Deviation	514179	1	0.01	0.03	0.73	0.03	0.05
AB07DA0800	2018	Minimum	214458	45	0.91	0.18	0.08	0.13	-0.46
		Maximum	695415	63	0.94	0.36	0.16	0.40	-0.10
		Arithmetic Mean	403886	54	0.93	0.28	0.13	0.24	-0.28
		Standard Deviation	256220	9	0.01	0.09	0.05	0.14	0.18
	2019	Minimum	7701	5	0.78	0.60	0.01	-1.95	1.07
		Maximum	19609	22	0.92	0.90	0.65	-1.09	1.51
		Arithmetic Mean	12185	13	0.87	0.76	0.36	-1.59	1.23
		Standard Deviation	6475	9	0.08	0.15	0.32	0.45	0.25
	2021	Minimum	240780	46	0.92	0.19	0.60	0.28	0.02
		Maximum	881142	66	0.94	0.29	1.36	0.40	0.28
		Arithmetic Mean	627398	59	0.93	0.24	0.94	0.35	0.11
		Standard Deviation	340232	11	0.01	0.05	0.39	0.06	0.14
AB07DA3008	2018	Minimum	268391	51	0.90	0.19	0.01	0.52	-0.25
		Maximum	1656702	64	0.95	0.35	0.65	0.66	-0.14
		Arithmetic Mean	1098895	56	0.93	0.26	0.30	0.59	-0.20

Appendix B - Summary Statistics

Site	Year	Statistic	Density	Richness (LPL)	Diversity	Evenness	Chlorophyll-*a*	NMDS1	NMDS2
		Standard Deviation	733229	7	0.03	0.08	0.32	0.07	0.06
	2019	Minimum	385	1	0.00	0.36	0.04	-2.90	-0.42
		Maximum	257553	61	0.95	1.00	0.19	0.15	0.41
		Arithmetic Mean	93684	22	0.54	0.79	0.13	-1.75	-0.14
		Standard Deviation	142369	34	0.49	0.37	0.08	1.66	0.48
	2021	Minimum	534236	57	0.93	0.22	1.69	0.41	-0.01
		Maximum	1135283	68	0.95	0.32	2.27	0.56	0.09
		Arithmetic Mean	846747	63	0.94	0.27	2.06	0.47	0.05
		Standard Deviation	301240	6	0.01	0.05	0.32	0.08	0.05
AB07DA3009	2018	Minimum	868798	51	0.90	0.20	0.10	0.44	-0.34
		Maximum	1645150	55	0.93	0.24	0.45	0.62	-0.20
		Arithmetic Mean	1252642	52	0.91	0.23	0.24	0.54	-0.26
		Standard Deviation	390814	2	0.01	0.02	0.15	0.08	0.06
	2019	Minimum	56486	39	0.63	0.05	0.10	-0.44	0.32
		Maximum	853786	51	0.94	0.44	1.48	0.24	0.72
		Arithmetic Mean	504621	47	0.83	0.26	0.76	-0.02	0.50
		Standard Deviation	407760	7	0.18	0.20	0.69	0.36	0.20
	2021	Minimum	495067	57	0.92	0.21	0.05	0.38	-0.02
		Maximum	1413972	62	0.93	0.26	1.72	0.45	0.06
		Arithmetic Mean	858947	60	0.93	0.23	0.93	0.41	0.02
		Standard Deviation	488363	3	0.00	0.02	0.84	0.03	0.04
AB07DA3015	2018	Minimum	1245752	36	0.84	0.17	0.02	0.49	-0.54
		Maximum	1291979	47	0.88	0.18	0.21	0.54	-0.49
		Arithmetic Mean	1268866	42	0.86	0.17	0.11	0.52	-0.51
		Standard Deviation	32687	8	0.03	0.01	0.13	0.04	0.03
	2019	Minimum	1155	3	0.67	0.10	0.04	-2.39	-1.09
		Maximum	593325	44	0.78	1.00	1.23	0.03	0.35
		Arithmetic Mean	200599	18	0.74	0.61	0.44	-1.54	-0.31
		Standard Deviation	340124	23	0.06	0.46	0.68	1.36	0.73

Appendix B - Summary Statistics

Site	Year	Statistic	Density	Richness (LPL)	Diversity	Evenness	Chlorophyll-*a*	NMDS1	NMDS2
	2021	Minimum	252984	49	0.94	0.32	0.13	0.29	-0.04
		Maximum	1267568	52	0.95	0.38	0.95	0.46	0.07
		Arithmetic Mean	674656	51	0.94	0.35	0.44	0.39	0.03
		Standard Deviation	528524	2	0.00	0.03	0.45	0.09	0.06
AB07DA3016	2018	Minimum	636993	38	0.83	0.13	0.17	0.40	-0.49
		Maximum	6773363	52	0.90	0.21	1.63	0.62	-0.38
		Arithmetic Mean	3852014	46	0.86	0.16	0.85	0.49	-0.42
		Standard Deviation	3078708	7	0.04	0.04	0.73	0.11	0.06
	2019	Minimum	1327938	36	0.50	0.06	0.46	0.00	0.04
		Maximum	1756886	48	0.69	0.07	1.35	0.27	0.31
		Arithmetic Mean	1513734	43	0.63	0.06	0.83	0.14	0.16
		Standard Deviation	220151	6	0.11	0.01	0.46	0.14	0.14
AB07DA3017	2018	Minimum	1668268	40	0.86	0.15	0.13	0.45	-0.43
		Maximum	3213283	53	0.92	0.22	0.47	0.51	-0.31
		Arithmetic Mean	2339959	48	0.89	0.19	0.24	0.49	-0.35
		Standard Deviation	791997	7	0.03	0.04	0.20	0.03	0.06
	2019	Minimum	17836	19	0.69	0.06	0.09	-0.75	0.18
		Maximum	972829	67	0.96	0.65	1.22	0.34	0.98
		Arithmetic Mean	368567	46	0.86	0.37	0.50	-0.05	0.54
		Standard Deviation	525558	24	0.15	0.30	0.62	0.60	0.40
	2021	Minimum	2126761	50	0.68	0.06	0.75	0.41	0.01
		Maximum	2350228	55	0.82	0.10	1.49	0.57	0.10
		Arithmetic Mean	2264180	53	0.74	0.08	1.18	0.48	0.05
		Standard Deviation	120265	3	0.07	0.02	0.38	0.08	0.05
AB07DA3018	2018	Minimum	1186658	49	0.92	0.24	0.26	0.37	-0.33
		Maximum	3012933	55	0.93	0.28	1.67	0.63	-0.24
		Arithmetic Mean	1889169	53	0.93	0.26	1.10	0.54	-0.27
		Standard Deviation	983316	3	0.01	0.02	0.74	0.15	0.05
	2019	Minimum	62009	46	0.91	0.17	0.13	0.01	0.13

Appendix B - Summary Statistics

Site	Year	Statistic	Density	Richness (LPL)	Diversity	Evenness	Chlorophyll-*a*	NMDS1	NMDS2
		Maximum	1006858	64	0.96	0.44	1.36	0.32	0.42
		Arithmetic Mean	564980	56	0.93	0.31	0.94	0.21	0.26
		Standard Deviation	475378	9	0.03	0.13	0.70	0.17	0.15
AB07DA3020	2018	Minimum	1525705	57	0.91	0.20	0.73	0.50	-0.26
		Maximum	2658461	63	0.94	0.29	0.97	0.61	-0.20
		Arithmetic Mean	1966865	60	0.93	0.26	0.86	0.56	-0.23
		Standard Deviation	499052	3	0.01	0.04	0.10	0.06	0.03
	2019	Minimum	770	2	0.50	0.62	0.16	-2.67	-1.76
		Maximum	67805	21	0.92	1.00	0.79	-0.52	0.88
		Arithmetic Mean	16873	9	0.79	0.88	0.37	-1.81	-0.40
		Standard Deviation	28798	8	0.17	0.15	0.27	0.79	1.21
	2021	Minimum	337496	50	0.92	0.24	0.83	0.30	0.02
		Maximum	675512	61	0.94	0.30	1.68	0.57	0.12
		Arithmetic Mean	466510	55	0.93	0.27	1.17	0.39	0.08
		Standard Deviation	154277	4	0.01	0.02	0.35	0.11	0.04
AB07DA3021	2018	Minimum	358296	43	0.90	0.19	0.19	0.17	-0.26
		Maximum	1306083	68	0.94	0.30	1.10	0.49	-0.15
		Arithmetic Mean	785856	55	0.92	0.25	0.54	0.35	-0.19
		Standard Deviation	396438	9	0.02	0.04	0.37	0.14	0.05
	2019	Minimum	4760	9	0.79	0.50	0.11	-2.07	0.14
		Maximum	109399	44	0.95	0.90	0.36	-0.08	1.01
		Arithmetic Mean	32433	21	0.89	0.65	0.23	-1.29	0.56
		Standard Deviation	44540	16	0.07	0.19	0.09	0.89	0.39
	2021	Minimum	459749	51	0.91	0.17	0.27	0.36	0.00
		Maximum	2488927	64	0.93	0.28	1.55	0.46	0.09
		Arithmetic Mean	1120775	58	0.92	0.23	0.91	0.40	0.02
		Standard Deviation	798201	5	0.01	0.04	0.50	0.04	0.04
AB07DA3022	2018	Minimum	585614	53	0.88	0.13	0.16	0.38	-0.18
		Maximum	1479468	64	0.91	0.20	0.89	0.45	-0.15

Appendix B - Summary Statistics

Site	Year	Statistic	Density	Richness (LPL)	Diversity	Evenness	Chlorophyll-*a*	NMDS1	NMDS2
		Arithmetic Mean	1069360	57	0.89	0.16	0.42	0.42	-0.17
		Standard Deviation	451454	6	0.02	0.04	0.41	0.04	0.01
	2019	Minimum	2695	6	0.78	0.20	0.12	-2.23	-0.78
		Maximum	704287	61	0.92	0.91	0.99	0.33	0.81
		Arithmetic Mean	237201	25	0.84	0.55	0.48	-1.21	0.04
		Standard Deviation	404509	31	0.07	0.36	0.45	1.35	0.80
	2021	Minimum	482869	48	0.92	0.19	1.26	0.22	0.00
		Maximum	1011996	65	0.93	0.31	1.55	0.48	0.07
		Arithmetic Mean	690923	56	0.93	0.25	1.36	0.37	0.04
		Standard Deviation	282089	9	0.01	0.06	0.16	0.13	0.04
AB07DA3023	2018	Minimum	1306087	51	0.88	0.16	0.09	0.40	-0.26
		Maximum	1926407	58	0.91	0.20	0.54	0.50	-0.18
		Arithmetic Mean	1616893	55	0.90	0.18	0.31	0.44	-0.22
		Standard Deviation	310162	4	0.01	0.02	0.23	0.05	0.04
	2019	Minimum	770	2	0.50	0.36	0.10		
		Maximum	252731	61	0.96	1.00	0.77		
		Arithmetic Mean	142027	40	0.81	0.61	0.39		
		Standard Deviation	128729	33	0.26	0.35	0.35		
	2021	Minimum	554787	54	0.92	0.18	0.50	0.40	-0.01
		Maximum	1494876	69	0.96	0.45	0.92	0.43	0.08
		Arithmetic Mean	1036820	62	0.93	0.28	0.77	0.41	0.03
		Standard Deviation	470503	8	0.02	0.15	0.24	0.02	0.04
AB07DA3024	2018	Minimum	614503	54	0.91	0.19	0.19	0.52	-0.36
		Maximum	1803114	60	0.93	0.24	1.00	0.64	-0.13
		Arithmetic Mean	1391500	58	0.92	0.22	0.51	0.57	-0.23
		Standard Deviation	673296	3	0.01	0.03	0.43	0.06	0.12
	2019	Minimum	770	2	0.50	0.05	0.25	-2.26	-1.32
		Maximum	1040255	50	0.92	1.00	1.13	0.10	0.96
		Arithmetic Mean	489559	27	0.68	0.50	0.59	-0.86	-0.08

Appendix B - Summary Statistics

Site	Year	Statistic	Density	Richness (LPL)	Diversity	Evenness	Chlorophyll-*a*	NMDS1	NMDS2
		Standard Deviation	522500	24	0.22	0.48	0.47	1.24	1.15
	2021	Minimum	665240	54	0.89	0.15	0.64	0.47	0.02
		Maximum	1144263	67	0.92	0.24	1.59	0.56	0.14
		Arithmetic Mean	963403	59	0.91	0.20	1.18	0.52	0.07
		Standard Deviation	260165	7	0.02	0.04	0.49	0.05	0.06

Appendix B - Summary Statistics

Table B2. Summary statistics of benthic indices of community for samples collected during the EMP program (2018, 2019 and 2021)

Site	Year	Statistic	Abundance	Richness (LPL)	Diversity	Evenness	EPT	PTI	NMDS1	NMDS2
AB07DA0800	2018	Minimum	5	3	0.19	0.11	0	5.30	-1.32	-1.53
		Maximum	832	11	0.86	0.93	29	6.30	0.53	-0.88
		Arithmetic Mean	232	8	0.55	0.44	6	5.52	-0.48	-1.24
		Standard Deviation	356	3	0.25	0.34	13	0.44	0.87	0.28
	2019	Minimum	72	12	0.57	0.19	0	5.24	-0.01	-1.33
		Maximum	950	49	0.93	0.44	20	5.54	0.54	0.00
		Arithmetic Mean	468	25	0.81	0.30	5	5.38	0.26	-0.70
		Standard Deviation	353	14	0.14	0.09	8	0.13	0.25	0.52
	2021	Minimum	391	11	0.39	0.07	1	5.49	0.00	-0.31
		Maximum	3,160	24	0.78	0.27	14	6.25	0.77	0.41
		Arithmetic Mean	1,678	20	0.58	0.15	7	5.72	0.40	0.16
		Standard Deviation	1,313	5	0.15	0.08	6	0.30	0.36	0.30
AB07DA3008	2018	Minimum	892	12	0.23	0.10	0	5.33	0.61	-0.88
		Maximum	2,138	23	0.73	0.31	7	6.53	1.04	-0.21
		Arithmetic Mean	1,663	18	0.58	0.16	3	5.95	0.82	-0.38
		Standard Deviation	569	5	0.20	0.08	3	0.45	0.16	0.28
	2019	Minimum	149	12	0.61	0.13	1	5.34	-0.50	-0.70
		Maximum	736	37	0.90	0.43	31	5.39	0.38	-0.14
		Arithmetic Mean	414	23	0.78	0.27	16	5.35	-0.04	-0.31
		Standard Deviation	247	9	0.12	0.14	12	0.02	0.33	0.24
	2021	Minimum	509	13	0.31	0.07	1	5.40	0.07	-0.12
		Maximum	6,200	21	0.69	0.20	5	6.29	1.05	0.31
		Arithmetic Mean	3,235	18	0.53	0.14	3	5.77	0.57	0.09
		Standard Deviation	2,049	3	0.17	0.05	2	0.35	0.38	0.17
AB07DA3009	2018	Minimum	312	15	0.47	0.10	11	5.38	-0.51	0.39
		Maximum	2,556	27	0.79	0.23	88	5.54	0.65	1.04

Appendix B - Summary Statistics

Site	Year	Statistic	Abundance	Richness (LPL)	Diversity	Evenness	EPT	PTI	NMDS1	NMDS2
		Arithmetic Mean	1,429	20	0.66	0.17	45	5.45	0.34	0.59
		Standard Deviation	800	5	0.12	0.05	31	0.06	0.48	0.26
	2019	Minimum	764	22	0.68	0.09	3	5.31	0.13	0.14
		Maximum	3,218	35	0.82	0.22	12	5.70	1.07	0.43
		Arithmetic Mean	1,987	28	0.77	0.17	8	5.51	0.59	0.28
		Standard Deviation	937	6	0.06	0.05	4	0.15	0.36	0.14
	2021	Minimum	496	15	0.18	0.08	3	5.34	0.26	0.22
		Maximum	8,200	29	0.84	0.26	43	7.06	0.55	0.45
		Arithmetic Mean	3,162	24	0.62	0.15	18	6.05	0.38	0.34
		Standard Deviation	3,225	5	0.27	0.08	15	0.87	0.15	0.10
AB07DA3015	2018	Minimum	153	11	0.23	0.12	19	5.50	-1.00	0.12
		Maximum	1,467	28	0.86	0.34	89	6.21	0.59	1.09
		Arithmetic Mean	644	21	0.66	0.20	52	5.72	0.07	0.52
		Standard Deviation	520	6	0.26	0.10	31	0.29	0.64	0.38
	2019	Minimum	36	4	0.30	0.16	19	5.61	-1.73	-0.05
		Maximum	616	28	0.87	0.35	83	6.00	0.21	0.89
		Arithmetic Mean	169	13	0.59	0.27	63	5.87	-1.10	0.63
		Standard Deviation	250	9	0.21	0.07	25	0.18	0.75	0.39
	2021	Minimum	81	9	0.50	0.16	15	5.82	-1.03	0.28
		Maximum	527	25	0.86	0.28	71	7.61	0.31	0.86
		Arithmetic Mean	290	17	0.68	0.21	51	6.24	-0.34	0.56
		Standard Deviation	184	6	0.13	0.05	22	0.77	0.49	0.25
AB07DA3016	2018	Minimum	89	12	0.30	0.09	3	5.23	-1.18	0.07
		Maximum	646	37	0.91	0.33	86	5.96	0.81	0.78
		Arithmetic Mean	277	22	0.60	0.18	52	5.64	-0.35	0.46
		Standard Deviation	229	11	0.29	0.09	42	0.26	0.92	0.27
	2019	Minimum	10	2	0.32	0.12	2	5.41	-2.32	-0.15
		Maximum	4,571	27	0.85	0.74	80	5.60	0.30	0.74
		Arithmetic Mean	1,151	14	0.59	0.36	31	5.52	-0.76	0.10

Appendix B - Summary Statistics

Site	Year	Statistic	Abundance	Richness (LPL)	Diversity	Evenness	EPT	PTI	NMDS1	NMDS2
		Standard Deviation	1,947	11	0.24	0.27	33	0.08	1.21	0.37
AB07DA3017	2018	Minimum	69	11	0.60	0.16	13	5.02	-1.00	0.57
		Maximum	603	26	0.86	0.32	64	5.49	0.26	0.80
		Arithmetic Mean	265	20	0.77	0.25	34	5.33	-0.23	0.69
		Standard Deviation	226	7	0.10	0.06	22	0.19	0.52	0.09
	2019	Minimum	29	5	0.48	0.14	29	5.27	-1.34	-0.24
		Maximum	680	45	0.87	0.64	69	6.05	0.02	0.22
		Arithmetic Mean	177	18	0.77	0.41	40	5.62	-0.70	-0.04
		Standard Deviation	282	16	0.16	0.18	16	0.30	0.53	0.20
	2021	Minimum	144	9	0.64	0.20	53	7.54	-0.66	0.88
		Maximum	505	27	0.81	0.32	73	8.00	0.01	1.20
		Arithmetic Mean	245	15	0.70	0.25	66	7.73	-0.38	1.04
		Standard Deviation	150	7	0.07	0.05	8	0.19	0.27	0.13
AB07DA3018	2018	Minimum	1,055	24	0.39	0.07	3	5.31	0.28	-0.25
		Maximum	3,630	45	0.85	0.15	31	5.41	1.05	0.09
		Arithmetic Mean	2,246	32	0.60	0.10	15	5.36	0.59	-0.07
		Standard Deviation	964	8	0.21	0.04	13	0.04	0.33	0.16
	2019	Minimum	1,328	26	0.83	0.16	10	5.31	-0.12	-0.21
		Maximum	3,145	40	0.91	0.33	40	5.45	0.60	0.25
		Arithmetic Mean	1,815	34	0.87	0.25	21	5.37	0.28	-0.02
		Standard Deviation	755	6	0.04	0.06	12	0.05	0.27	0.17
AB07DA3020	2018	Minimum	207	19	0.65	0.12	13	5.61	0.34	-0.36
		Maximum	855	24	0.85	0.29	26	7.13	0.72	0.17
		Arithmetic Mean	490	23	0.78	0.22	17	6.30	0.49	-0.03
		Standard Deviation	301	2	0.08	0.07	5	0.68	0.15	0.21
	2019	Minimum	437	12	0.81	0.28	0	5.54	0.45	-0.41
		Maximum	1,836	21	0.84	0.50	0	5.63	0.70	-0.13
		Arithmetic Mean	1,296	16	0.82	0.38	0	5.59	0.62	-0.30
		Standard Deviation	576	3	0.01	0.09	0	0.03	0.10	0.10

Appendix B - Summary Statistics

Site	Year	Statistic	Abundance	Richness (LPL)	Diversity	Evenness	EPT	PTI	NMDS1	NMDS2
	2021	Minimum	933	17	0.77	0.21	1	6.62	0.28	0.05
		Maximum	2,485	22	0.80	0.27	23	7.53	0.58	0.34
		Arithmetic Mean	1,682	19	0.78	0.24	11	6.98	0.41	0.20
		Standard Deviation	574	2	0.01	0.02	8	0.34	0.11	0.10
AB07DA3021	2018	Minimum	29	9	0.58	0.15	7	5.60	-0.38	-1.25
		Maximum	787	25	0.84	0.54	14	5.79	0.64	-0.23
		Arithmetic Mean	260	15	0.76	0.36	10	5.69	0.18	-0.72
		Standard Deviation	314	6	0.11	0.19	3	0.08	0.37	0.40
	2019	Minimum	4	2	0.22	0.28	0	5.38	-2.12	-1.84
		Maximum	63	10	0.74	0.80	0	5.60	-0.46	-1.09
		Arithmetic Mean	23	6	0.53	0.53	0	5.51	-1.08	-1.35
		Standard Deviation	24	4	0.23	0.21	0	0.12	0.65	0.31
	2021	Minimum	764	13	0.70	0.16	0	5.96	0.62	-0.58
		Maximum	3,200	21	0.79	0.30	2	6.53	0.97	0.04
		Arithmetic Mean	1,690	17	0.74	0.24	1	6.24	0.80	-0.23
		Standard Deviation	991	4	0.04	0.05	1	0.21	0.12	0.23
AB07DA3022	2018	Minimum	2,314	14	0.23	0.06	0	5.30	0.44	-0.44
		Maximum	21,420	27	0.64	0.15	5	5.40	0.92	0.00
		Arithmetic Mean	7,358	20	0.43	0.10	1	5.35	0.74	-0.28
		Standard Deviation	8,141	5	0.16	0.04	2	0.04	0.18	0.18
	2019	Minimum	30	10	0.64	0.23	1	5.20	-1.11	-0.54
		Maximum	1,953	18	0.82	0.46	27	5.37	0.27	-0.37
		Arithmetic Mean	880	13	0.74	0.32	10	5.31	-0.17	-0.45
		Standard Deviation	786	4	0.07	0.09	11	0.06	0.56	0.07
	2021	Minimum	3,722	13	0.57	0.09	0	5.61	0.50	-0.17
		Maximum	6,867	25	0.64	0.19	7	5.79	1.17	0.10
		Arithmetic Mean	4,824	18	0.61	0.16	2	5.70	0.71	-0.04
		Standard Deviation	1,264	5	0.03	0.05	3	0.08	0.27	0.11
AB07DA3023	2018	Minimum	1,110	15	0.40	0.07	0	5.36	0.22	-0.41

Appendix B - Summary Statistics

Site	Year	Statistic	Abundance	Richness (LPL)	Diversity	Evenness	EPT	PTI	NMDS1	NMDS2
		Maximum	7,680	29	0.91	0.40	34	5.57	0.85	0.55
		Arithmetic Mean	2,817	23	0.68	0.20	9	5.45	0.59	-0.02
		Standard Deviation	2,741	6	0.21	0.13	14	0.09	0.25	0.35
	2019	Minimum	209	16	0.74	0.17	0	5.35	-0.48	-0.39
		Maximum	2,777	25	0.83	0.30	32	5.65	0.71	0.00
		Arithmetic Mean	1,312	21	0.78	0.23	13	5.44	0.08	-0.20
		Standard Deviation	952	4	0.04	0.05	13	0.13	0.48	0.17
	2021	Minimum	470	17	0.32	0.09	0	5.71	0.08	0.00
		Maximum	7,060	24	0.78	0.22	24	8.11	0.93	0.77
		Arithmetic Mean	2,443	21	0.57	0.13	11	7.09	0.40	0.45
		Standard Deviation	2,717	3	0.17	0.06	10	1.17	0.40	0.32
AB07DA3024	2018	Minimum	893	17	0.35	0.06	1	5.35	0.53	-0.20
		Maximum	5,633	29	0.78	0.20	10	7.28	0.77	0.15
		Arithmetic Mean	2,245	24	0.65	0.14	5	6.10	0.63	-0.02
		Standard Deviation	1,918	5	0.18	0.05	4	0.79	0.10	0.13
	2019	Minimum	31	5	0.26	0.21	20	5.30	-1.67	-0.21
		Maximum	120	12	0.76	0.47	85	5.67	-0.63	0.34
		Arithmetic Mean	74	8	0.58	0.33	44	5.48	-1.33	0.09
		Standard Deviation	42	3	0.19	0.10	27	0.26	0.42	0.21
	2021	Minimum	347	16	0.53	0.13	7	6.03	-0.17	0.39
		Maximum	1,066	34	0.83	0.18	61	7.10	0.15	0.81
		Arithmetic Mean	732	27	0.74	0.16	41	6.58	0.03	0.60
		Standard Deviation	296	7	0.13	0.02	21	0.40	0.12	0.20
AB07DA0062	2018	Minimum	3	2	0.44	0.23	0	5.60	-2.68	-1.71
		Maximum	2,160	27	0.85	0.90	60	6.58	0.83	0.36
		Arithmetic Mean	857	12	0.71	0.60	13	6.14	-1.03	-0.25
		Standard Deviation	1,165	12	0.16	0.33	26	0.41	1.69	0.84
	2019	Minimum	10	4	0.70	0.36	0	5.30	-1.93	-1.25
		Maximum	39	10	0.80	0.83	40	5.68	-0.72	-0.29

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Site	Year	Statistic	Abundance	Richness (LPL)	Diversity	Evenness	EPT	PTI	NMDS1	NMDS2
		Arithmetic Mean	19	6	0.76	0.70	9	5.52	-1.22	-0.89
		Standard Deviation	12	2	0.04	0.20	18	0.16	0.50	0.38
	2021	Minimum	12	7	0.69	0.11	4	6.81	-2.33	0.44
		Maximum	588	29	0.82	0.79	53	8.30	0.21	1.13
		Arithmetic Mean	205	19	0.76	0.32	29	7.44	-0.80	0.76
		Standard Deviation	228	9	0.05	0.28	17	0.54	0.94	0.27

Appendix B - Summary Statistics

Table B3. Summary statistics of indices of fish community sampled during the EMP program, pooled over sampling years (2018, 2019 and 2021)

Site Code	Station Name	Statistic	Abundance	Richness	Diversity	Evenness	NMDS1	NMDS2
01A	20 Km US W	Min	73	7	0.41	0.24	-0.25	-0.35
		Max	157	8	0.78	0.57	0.54	0.41
		Mean	104	7	0.65	0.46	0.06	0.01
		Sd	46	1	0.20	0.19	0.42	0.38
04A	4 Km US W	Min	81	5	0.33	0.30	-0.43	-0.08
		Max	153	13	0.88	0.63	0.73	0.44
		Mean	128	10	0.66	0.44	0.02	0.21
		Sd	40	4	0.29	0.17	0.62	0.26
04B	4 Km US E	Min	20	5	0.58	0.47	-0.41	-0.35
		Max	140	11	0.85	0.72	0.68	0.20
		Mean	78	8	0.75	0.58	-0.03	-0.09
		Sd	60	3	0.15	0.12	0.62	0.28
05A	0.5 Km US W	Min	81	7	0.49	0.28	-0.56	-0.08
		Max	131	11	0.83	0.54	0.34	0.13
		Mean	100	9	0.70	0.42	-0.22	0.05
		Sd	27	2	0.18	0.13	0.49	0.12
05B	0.5 Km US E	Min	44	4	0.54	0.37	-0.41	-0.22
		Max	125	11	0.87	0.72	0.94	0.09
		Mean	87	9	0.72	0.54	0.05	-0.10
		Sd	41	4	0.17	0.18	0.77	0.17
06A	1.5 Km DS W of Isl	Min	26	4	0.45	0.18	-0.23	-0.21
		Max	159	10	0.82	0.67	0.78	0.29
		Mean	75	8	0.63	0.49	0.14	0.11
		Sd	73	3	0.18	0.27	0.56	0.28
10B	13 Km DS E	Min	20	3	0.54	0.61	-0.74	-0.34
		Max	122	10	0.84	0.75	1.09	0.11

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Site Code	Station Name	Statistic	Abundance	Richness	Diversity	Evenness	NMDS1	NMDS2
		Mean	70	7	0.73	0.69	0.00	-0.09
		Sd	51	4	0.17	0.07	0.96	0.23
11A	13 Km DS W	Min	80	7	0.49	0.28	-0.28	-0.24
		Max	122	12	0.74	0.35	0.46	0.12
		Mean	100	10	0.66	0.31	-0.03	-0.10
		Sd	21	3	0.14	0.03	0.43	0.19

Appendix B - Summary Statistics

Table B4. Summary statistics of fish health indicators for samples collected during the EMP program (2018, 2019 and 2021)

Site	Description	Statistic	Females									Males								
			2018			2019			2021			2018			2019			2021		
			K	GSI	LSI	K	GSI	LSI	K	GSI	LSI	K	GSI	LSI	K	GSI	LSI	K	GSI	LSI
AB07DA0062	25 km US	Minimum	0.97	1.90	1.10	0.93	4.27	1.28	1.04	0.50	0.78	0.98	0.92	0.57	0.95	0.70	0.87	0.96	0.64	0.99
		Maximum	1.29	7.00	2.24	1.19	6.29	1.96	1.27	5.43	2.23	1.20	1.90	1.72	1.19	2.00	1.61	1.53	1.28	1.82
		Arithmetic Mean	1.15	4.37	1.55	1.09	5.39	1.54	1.16	3.85	1.57	1.11	1.51	1.25	1.07	1.46	1.21	1.19	0.99	1.43
		Standard Deviation	0.07	0.94	0.35	0.06	0.54	0.20	0.06	1.25	0.29	0.06	0.31	0.26	0.07	0.30	0.19	0.13	0.19	0.20
AB07DA3024	12 km US	Minimum	1.02	2.46	1.08	1.02	2.99	1.12	1.06	1.79	0.91	1.04	1.13	0.00	0.96	0.41	0.86	0.93	0.28	0.51
		Maximum	1.40	5.89	2.39	1.37	5.93	2.11	1.50	6.49	1.75	1.25	2.91	1.90	1.28	2.18	1.74	1.29	1.40	1.67
		Arithmetic Mean	1.12	4.53	1.73	1.13	4.82	1.55	1.19	3.79	1.28	1.11	1.99	1.25	1.06	1.27	1.27	1.13	0.83	1.21
		Standard Deviation	0.09	0.85	0.32	0.08	0.72	0.29	0.11	1.28	0.23	0.05	0.42	0.38	0.09	0.44	0.22	0.09	0.28	0.27
AB07DA3023	4 km US	Minimum	1.02	1.17	1.38	0.99	3.28	1.17	0.93	1.14	1.06	1.03	1.37	1.17	0.00	0.53	0.52	1.02	0.38	0.99
		Maximum	1.26	6.48	3.65	1.32	5.83	1.99	1.62	4.67	1.91	1.22	2.53	1.85	1.30	1.82	1.61	1.42	1.38	1.65
		Arithmetic Mean	1.16	4.10	1.89	1.16	4.30	1.54	1.20	3.21	1.46	1.13	1.80	1.39	1.03	1.30	1.30	1.20	0.90	1.29
		Standard Deviation	0.07	1.07	0.49	0.09	0.72	0.20	0.15	1.21	0.23	0.05	0.28	0.19	0.27	0.29	0.25	0.11	0.28	0.18
AB07DA3008	12 km DS	Minimum	0.94	3.31	1.38	0.92	2.52	1.27	0.99	0.62	1.05	1.01	1.02	0.89	0.76	0.37	0.96	0.98	0.34	1.07
		Maximum	1.28	5.60	14.42	1.23	6.25	2.11	1.31	5.71	1.89	1.30	2.03	1.65	1.20	1.81	1.85	1.39	1.26	1.55
		Arithmetic Mean	1.13	4.59	2.33	1.09	4.74	1.70	1.13	3.23	1.36	1.12	1.53	1.28	1.05	1.29	1.45	1.13	0.83	1.28
		Standard Deviation	0.07	0.61	2.85	0.07	0.76	0.24	0.08	1.66	0.20	0.07	0.32	0.21	0.09	0.38	0.26	0.11	0.25	0.14
AB07DA3022	0.5 km US Left	Minimum	1.03	3.49	1.12	0.86	3.70	0.98	1.00	0.76	0.59	1.00	1.02	0.96	0.88	0.64	0.82	1.05	0.39	0.85
		Maximum	1.31	5.68	2.49	1.30	6.31	2.05	1.33	5.57	1.79	1.34	2.19	1.73	1.28	1.58	1.47	1.24	1.60	2.03
		Arithmetic Mean	1.17	4.43	1.70	1.09	4.79	1.52	1.16	3.55	1.37	1.13	1.65	1.38	1.08	1.21	1.17	1.14	0.90	1.25
		Standard Deviation	0.07	0.69	0.40	0.09	0.66	0.27	0.09	1.33	0.28	0.07	0.34	0.21	0.10	0.21	0.16	0.05	0.27	0.25
AB07DA3017	0.5 km DS East	Minimum	1.02	0.56	1.19	0.98	3.68	1.06	0.99	0.74	0.93	1.01	0.60	0.83	0.91	0.74	0.78	0.91	0.43	0.79
		Maximum	1.38	5.28	1.97	1.15	6.64	2.52	1.31	6.43	1.79	1.42	2.06	1.82	1.16	1.76	1.59	1.37	1.67	2.00
		Arithmetic Mean	1.18	3.06	1.55	1.06	5.32	1.61	1.13	3.75	1.38	1.12	1.42	1.36	1.05	1.37	1.31	1.13	0.94	1.27
		Standard Deviation	0.11	1.60	0.22	0.06	0.72	0.34	0.07	1.59	0.21	0.09	0.43	0.28	0.07	0.30	0.21	0.09	0.27	0.23
AB07DA3015		Minimum	1.05	1.92	1.47	0.87	4.15	1.17	0.95	2.26	0.94	0.86	1.13	1.02	0.87	1.06	0.96	0.93	0.39	0.97

Appendix B - Summary Statistics

Site	Description	Statistic	Females									Males								
			2018			2019			2021			2018			2019			2021		
			K	GSI	LSI	K	GSI	LSI	K	GSI	LSI	K	GSI	LSI	K	GSI	LSI	K	GSI	LSI
	1.5 km DS East	Maximum	1.48	6.01	2.44	1.58	6.42	1.95	1.78	5.86	2.06	1.30	2.64	1.74	1.14	2.06	1.86	2.78	9.23	1.64
		Arithmetic Mean	1.19	4.69	1.90	1.05	5.40	1.50	1.18	4.46	1.53	1.11	1.93	1.38	1.01	1.52	1.24	1.13	1.22	1.25
		Standard Deviation	0.11	0.99	0.26	0.14	0.54	0.25	0.17	1.13	0.31	0.09	0.38	0.20	0.06	0.28	0.22	0.37	1.81	0.16
AB07DA3018	0.5 km DS West	Minimum	1.03	3.42	1.41	0.87	4.46	1.21	-	-	-	0.98	0.94	0.88	0.82	0.83	0.97	-	-	-
		Maximum	1.63	26.90	2.48	1.15	7.24	2.29	-	-	-	1.29	2.16	2.45	1.13	2.47	1.63	-	-	-
		Arithmetic Mean	1.19	5.57	1.88	1.04	5.52	1.60	-	-	-	1.13	1.76	1.36	1.02	1.56	1.20	-	-	-
		Standard Deviation	0.13	4.93	0.30	0.06	0.72	0.28	-	-	-	0.07	0.35	0.31	0.07	0.41	0.19	-	-	-
AB07DA3009	4.5 km DS	Minimum	1.03	3.81	1.34	0.93	0.97	1.30	1.08	1.59	1.07	0.71	1.20	0.95	1.00	0.79	0.71	0.93	0.45	1.00
		Maximum	1.41	6.25	2.79	1.32	6.36	1.97	1.25	5.62	1.80	1.31	2.68	1.63	1.27	2.20	1.68	1.26	1.49	1.78
		Arithmetic Mean	1.14	4.83	1.85	1.12	4.68	1.58	1.16	4.25	1.49	1.12	1.79	1.30	1.09	1.30	1.35	1.13	0.94	1.26
		Standard Deviation	0.09	0.77	0.31	0.09	1.34	0.16	0.06	1.24	0.21	0.13	0.39	0.22	0.07	0.33	0.22	0.07	0.26	0.20
AB07DA0800	34 km DS	Minimum	0.92	2.88	0.00	0.99	3.26	1.03	1.04	1.80	0.85	0.96	0.76	0.98	0.69	0.61	0.63	0.92	0.59	0.87
		Maximum	1.47	5.37	1.96	1.32	6.25	1.69	1.22	6.10	1.96	1.31	3.40	1.82	1.24	1.68	2.19	1.47	1.60	1.57
		Arithmetic Mean	1.14	4.23	1.53	1.11	4.88	1.40	1.13	4.40	1.44	1.12	1.80	1.39	1.07	1.24	1.20	1.12	1.01	1.22
		Standard Deviation	0.11	0.59	0.42	0.08	0.77	0.18	0.05	1.18	0.26	0.10	0.67	0.25	0.12	0.34	0.33	0.10	0.21	0.18

Table B5. Sample sizes of Trout-perch fish body burden sample summary (# of samples) by sex, sampling year, and station under EMP (2018, 2019, and 2021)

Sex	Year	Station Number	Station Name	Parameter Category			
				PAHs	Metals	Isotopes	Mercury
F	2018	AB07DA0062	25 km US	4	-	-	-
		AB07DA0800	34 km DS	4	-	-	-
		AB07DA3008	12 km DS	8	-	-	-
		AB07DA3015	1.5 km DS - E	4	-	-	-
		AB07DA3017	0.5 km DS - E	2	-	-	-
		AB07DA3018	0.5 km DS - W	4	-	-	-
		AB07DA3022	0.5 km US	4	-	-	-
		AB07DA3023	4.0 km US	4	-	-	-
		AB07DA3024	12 km US	4	-	-	-
	2019	AB07DA0062	25 km US	4	-	-	-
		AB07DA0800	34 km DS	4	-	-	-
		AB07DA3009	4.5 km DS	8	-	-	-
		AB07DA3015	1.5 km DS - E	4	-	-	-
		AB07DA3017	0.5 km DS - E	4	-	-	-
		AB07DA3018	0.5 km DS - W	4	-	-	-
		AB07DA3022	0.5 km US	4	-	-	-
		AB07DA3023	4.0 km US	4	-	-	-
		AB07DA3024	12 km US	4	-	-	-
	2021	AB07DA0062	25 km US	4	-	-	-
		AB07DA0800	34 km DS	4	10	-	-
		AB07DA3008	12 km DS	4	-	-	-
		AB07DA3009	4.5 km DS	3	-	-	-
		AB07DA3015	1.5 km DS - E	4	-	-	-
		AB07DA3017	0.5 km DS - E	8	8	10	-
		AB07DA3022	0.5 km US	4	-	10	-
		AB07DA3023	4.0 km US	4	-	-	-
		AB07DA3024	12 km US	5	-	-	-
M	2018	AB07DA0062	25 km US	-	10	10	10
		AB07DA0800	34 km DS	-	10	10	10
		AB07DA3008	12 km DS	-	10	10	10
		AB07DA3009	4.5 km DS	-	10	10	10
		AB07DA3015	1.5 km DS - E	-	10	10	10
		AB07DA3017	0.5 km DS - E	-	10	10	10
		AB07DA3018	0.5 km DS - W	-	10	10	10
		AB07DA3022	0.5 km US	-	10	10	10
		AB07DA3023	4.0 km US	-	10	10	10
		AB07DA3024	12 km US	-	10	10	10

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Sex	Year	Station Number	Station Name	Parameter Category			
				PAHs	Metals	Isotopes	Mercury
	2019	AB07DA0062	25 km US	-	10	10	10
		AB07DA0800	34 km DS	-	10	10	10
		AB07DA3008	12 km DS	-	10	10	10
		AB07DA3009	4.5 km DS	-	10	10	10
		AB07DA3015	1.5 km DS - E	-	10	10	10
		AB07DA3017	0.5 km DS - E	-	10	10	10
		AB07DA3018	0.5 km DS - W	-	10	10	10
		AB07DA3022	0.5 km US	-	10	10	10
		AB07DA3023	4.0 km US	-	10	10	10
		AB07DA3024	12 km US	-	10	10	10
	2021	AB07DA0062	25 km US	-	10	10	10
		AB07DA0800	34 km DS	3	10	10	10
		AB07DA3008	12 km DS	-	10	10	10
		AB07DA3009	4.5 km DS	-	10	10	10
		AB07DA3015	1.5 km DS - E	-	10	10	10
		AB07DA3017	0.5 km DS - E	3	20	20	20
		AB07DA3022	0.5 km US	-	10	10	10
		AB07DA3023	4.0 km US	-	10	10	10
		AB07DA3024	12 km US	-	11	10	10

Table B6. Sample sizes of Walleye fish body burden sample summary (# of samples) by sex, sampling year, and station under EMP (2019, and 2021)

Sex	Year	Station Number	Station Name	Parameter Category			
				PAHs	Metals	Isotopes	Mercury
F	2019	AB07DA0062	25 km US	2	2	2	2
		AB07DA3008	12 km DS	3	3	3	3
		AB07DA3009	4.5 km DS	2	2	2	2
		AB07DA3023	4.0 km US	1	1	1	1
	2021	AB07DA0062	25 km US	3	3	3	3
		AB07DA3008	12 km DS	3	3	3	3
		AB07DA3009	4.5 km DS	1	1	1	1
		AB07DA3017	0.5 km DS - E	2	3	3	3
		AB07DA3023	4.0 km US	5	5	5	5
M	2019	AB07DA0062	25 km US	6	6	6	6
		AB07DA3008	12 km DS	6	6	6	6
		AB07DA3009	4.5 km DS	8	8	8	8
		AB07DA3018	0.5 km DS - W	7	7	7	7
		AB07DA3023	4.0 km US	3	3	3	3
	2021	AB07DA0062	25 km US	2	2	2	2
		AB07DA3008	12 km DS	6	6	6	6
		AB07DA3009	4.5 km DS	5	6	6	6
		AB07DA3017	0.5 km DS - E	5	5	5	5
		AB07DA3023	4.0 km US	4	5	5	5

Table B7. Sample sizes of White Sucker fish body burden sample summary (# of samples) by sex, sampling year, and station under EMP (2019, and 2021)

Sex	Year	Station Number	Station Name	Parameter Category			
				PAHs	Metals	Isotopes	Mercury
F	2019	AB07DA0062	25 km US	5	5	16	5
		AB07DA3008	12 km DS	5	5	8	5
		AB07DA3009	4.5 km DS	1	1	1	1
		AB07DA3018	0.5 km DS - W	4	2	4	4
		AB07DA3023	4.0 km US	2	2	2	2
	2021	AB07DA0062	25 km US	-	5	20	5
		AB07DA3008	12 km DS	-	5	18	5
		AB07DA3009	4.5 km DS	-	-	2	-
		AB07DA3017	0.5 km DS - E	-	-	-	5
		AB07DA3018	0.5 km DS - W	-	4	16	-
		AB07DA3023	4.0 km US	-	6	18	5
M	2019	AB07DA0062	25 km US	5	5	15	5
		AB07DA3008	12 km DS	5	4	12	5
		AB07DA3009	4.5 km DS	2	2	2	2
		AB07DA3018	0.5 km DS - W	4	4	4	4
		AB07DA3023	4.0 km US	2	2	2	2
	2021	AB07DA0062	25 km US	-	5	11	5
		AB07DA3008	12 km DS	-	5	16	5
		AB07DA3009	4.5 km DS	-	-	2	-
		AB07DA3017	0.5 km DS - E	-	-	-	5
		AB07DA3018	0.5 km DS - W	-	5	11	-
		AB07DA3023	4.0 km US	-	5	10	5

Table B8. Mean (±SD) liver EROD activity detected in Trout-perch (pooled across both male and female) collected during the EMP (2018, 2019, and 2021)

Station	Description	Year	Mean Liver EROD Activity (pmol/min/mg)	Standard Deviation
AB07DA0062	25 km US	2018	0.74	0.39
		2019	0.26	0.11
		2021	0.82	0.77
AB07DA3024	12 km US	2018	1.91	2.35
		2019	0.41	0.38
		2021	0.61	0.6
AB07DA3023	4.0 km US	2018	1.26	1.66
		2019	0.3	0.17
		2021	1.11	1.12
AB07DA3022	0.5 km US	2018	0.9	0.78
		2019	0.25	0.08
		2021	1.16	0.84
AB07DA3017	0.5 km DS - E	2018	1.96	2.93
		2019	0.3	0.15
		2021	1.17	1.36
AB07DA3018	0.5 km DS - W	2018	1.02	1.14
		2019	0.27	0.12
AB07DA3015	1.5 km DS - E	2018	1.06	0.94
		2019	0.24	0.13
		2021	1.27	1.04
AB07DA3009	4.5 km DS	2018	1.65	1.86
		2019	0.28	0.1
		2021	0.94	1.12
AB07DA3008	12 km DS	2018	1.93	2.49
		2019	0.27	0.11
		2021	1.32	1.02
AB07DA0800	34 km DS	2018	1.7	2.08
		2019	0.37	0.26
		2021	1.06	0.88

Table B9. Mean (±SD) carbon and nitrogen stable isotope ratios (δ13C and δ15N, respectively) detected in the fish captured at the EMP survey stations (2018, 2019, and 2021).

Species	Station	Description	Year	$\delta^{13}C$ (%)		$\delta^{15}N$	
				Mean	SD	Mean	SD
Trout-perch	AB07DA0062	25 km US	2018	-26.8	0.57	9.94	0.4
			2019	-27.06	0.61	9.07	0.42
			2021	-26.32	0.33	9.79	0.32
	AB07DA3024	12 km US	2018	-27.4	1.09	9.39	0.56
			2019	-26.66	0.67	8.84	0.57
			2021	-26.88	0.82	9.36	0.43
	AB07DA3023	4.0 km US	2018	-26.89	0.63	9.67	0.64
			2019	-26.99	0.5	8.48	0.38
			2021	-26.66	0.44	9.5	0.25
	AB07DA3022	0.5 km US	2018	-27.02	0.89	9.82	0.57
			2019	-26.84	0.53	8.95	0.8
			2021	-26.77	0.84	9.7	0.55
	AB07DA3017	0.5 km DS - E	2018	-26.4	1.18	9.96	0.46
			2019	-26.89	0.27	8.84	0.34
			2021	-26.38	0.65	9.44	0.38
	AB07DA3018	0.5 km DS - W	2018	-26.76	0.61	9.88	0.43
			2019	-26.79	0.27	8.82	0.53
	AB07DA3015	1.5 km DS - E	2018	-26.29	0.85	10.08	0.41
			2019	-26.85	0.58	9.05	0.78
			2021	-26.35	0.74	9.43	0.55
	AB07DA3009	4.5 km DS	2018	-27.05	0.7	9.83	0.32
			2019	-26.74	0.57	9.26	0.83
			2021	-27.18	0.91	9.33	0.52
	AB07DA3008	12 km DS	2018	-27.14	0.67	9.79	0.32
			2019	-26.97	0.36	9.21	0.88
			2021	-27	0.72	9.65	0.38

Appendix B - Summary Statistics

Species	Station	Description	Year	δ¹³C (%)		δ¹⁵N	
				Mean	SD	Mean	SD
	AB07DA0800	34 km DS	2018	-26.62	0.37	7.81	1.54
			2019	-26.51	0.26	9.65	0.38
			2021	-27.41	0.98	9.61	0.49
Walleye	AB07DA0062	25 km US	2019	-25.95	1.23	10.97	0.52
			2021	-27.97	1.44	10.96	0.72
	AB07DA3023	4.0 km US	2019	-25.91	0.81	10.85	0.96
			2021	-27.61	1.41	10.84	0.65
	AB07DA3017	0.5 km DS - E	2021	-28.15	1.24	10.96	0.64
			2019	-25.75	0.81	11	0.79
	AB07DA3009	4.5 km DS	2019	-26.23	1.43	10.82	0.85
			2021	-28.21	1.33	11.2	0.79
	AB07DA3008	12 km DS	2019	-25.85	1.04	11.09	1.09
			2021	-27.97	1.53	10.83	0.44
White Sucker	AB07DA0062	25 km US	2019	-27.74	1.18	9	0.6
			2021	-28.78	1.52	8.77	0.8
	AB07DA3023	4.0 km US	2019	-28.44	0.55	9.05	0.88
			2021	-28.45	1.34	8.62	1.16
	AB07DA3017	0.5 km DS - E	2021	-27.59	1.64	8.85	1.15
			2019	-28.59	1.66	8.79	0.78
	AB07DA3009	4.5 km DS	2019	-28.51	2.4	8.86	0.61
			2021	-29.81	1.49	8.76	0.26
	AB07DA3008	12 km DS	2019	-28.37	1.54	8.8	0.83
			2021	-29.01	1.8	8.65	1.29

Table B10. Mean (±SD) carbon and nitrogen stable isotope ratios (δ13C and δ15N, respectively) detected in the benthos captured at the EMP survey stations (2018, 2019, and 2021).

Family	Station	Description	Year	$\delta^{13}C$ (%)		$\delta^{15}N$	
				Mean	SD	Mean	SD
Ametropodidae	AB07DA3023	4.0 km US	2018	-32.81	0.21	4.27	0.01
			2019	-31.03	0.32	4.45	0.35
			2021	-29.01	0.47	5.03	0.5
	AB07DA3022	0.5 km US	2018	-32.9	—	4.2	—
			2019	-31.96	0.01	4.16	0.05
			2021	-30.54	0.27	3.78	0.29
	AB07DA3020	0.03 km US	2021	-28.63	0.17	4.92	0.14
	AB07DA3017	0.5 km DS - E	2018	-31.66	—	6.2	NA
			2019	-31.23	0.22	4.34	0.19
			2021	-28.25	0.91	6.53	0.64
	AB07DA3018	0.5 km DS - W	2019	-32.04	0.18	4.78	0.1
	AB07DA3015	1.5 km DS - E	2018	-30.96	—	7.17	—
			2019	-31.75	0.13	4.26	0.15
			2021	-26.55	0.94	7.67	0.28
	AB07DA3016	1.5 km DS - W	2019	-31.21	0.52	4.47	0.24
	AB07DA3009	4.5 km DS	2021	-28.56	0.52	8.19	0.94
	AB07DA3008	12 km DS	2019	-31.95	0.27	4.26	0.19
			2021	-29.73	0.27	6.1	0.34
Chironomidae	AB07DA3022	0.5 km US	2018	-28.61	0.61	5.83	0.16
	AB07DA3020	0.03 km US	2018	-26.74	0.42	6.58	0.29
	AB07DA3018	0.5 km DS - W	2018	-23.98	1.01	15.11	2.14
	AB07DA3015	1.5 km DS - E	2018	-28.34	—	6.76	—
Gomphidae	AB07DA3022	0.5 km US	2018	-28.39	0.6	6.09	0.68
			2019	-28.53	0.68	7	0.03
			2021	-27.81	0.83	8.56	0.03
	AB07DA3020	0.03 km US	2018	-27.28	—	5.7	—

Appendix B - Summary Statistics

Family	Station	Description	Year	δ^{13}C (%)		δ^{15}N	
				Mean	SD	Mean	SD
	AB07DA3018	0.5 km DS - W	2018	-26.33	—	7.29	—
	AB07DA3015	0.5 km DS - E	2021	-30.03	2.15	5.06	0.63
Pteronarcyidae	AB07DA3022	0.5 km US	2019	-29.97	1.68	4.81	0.21
			2021	-27.88	0.44	6.52	1.53
	AB07DA3018	0.5 km DS - W	2019	-28.96	1.15	8.13	2.31
	AB07DA3015	0.5 km DS - E	2021	-28.85	—	3.68	—

Appendix C Normalization GLMs

Appendix C - Normalization GLM's

Table C1. Discharge (Q) normalization models for a subset of water quality parameters measured in the EMP program (2018, 2019 and 2021)

Water Quality Parameters		Model Parameters	
		Intercept	Slope (Q)
Alkalinity Total $CaCO_3$		2.24	-0.08
Metals (Total Recoverable)	Aluminum	-4.25	2.35
	Copper	-4.29	1.55
	Iron	-2.11	1.76
	Lead	-6.28	2.07
	Molybdenum	-0.21	0.03
	Nickel	-3.29	1.26
	Thallium	-6.86	1.73
	Vanadium	-5.36	1.90
	Zinc	-4.15	1.60
Nutrients	Calcium Dissolved	1.69	-0.06
	Chloride Dissolved	2.81	-0.71
	Magnesium Dissolved	1.34	-0.13
	Phosphorus Total	-6.17	1.69
	Sodium Dissolved/Filtered	2.29	-0.44
	Sulphate Dissolved	2.32	-0.30
Total Polycyclic aromatic hydrocarbons (PAHs)		-1.85	1.30
Naphthenic acids (NA)	Total NA	3.14	-0.73
	Rainbow Trout Toxic Unit Equivalent	-2.68	-0.13
	Fathead Minnow Toxic Unit Equivalent	-2.04	-0.13
Cytotoxicity Water Quality Index (WQI)		-3.57	1.17

Appendix C - Normalization GLM's

Table C2. Total aluminum (Al) normalization models for a subset of sediment quality parameters measured in the EMP program (2018, 2019 and 2021)

Sediment Quality Parameters		Model Parameters	
		Intercept	Slope (Al)
Properties	Ignition	-3.11	0.88
	Moisture	-0.19	0.45
	Organic Matter	-2.75	0.78
Metals (Total Recoverable)	Antimony	-3.84	0.84
	Arsenic	-1.09	0.47
	Barium	-1.34	0.91
	Beryllium	-3.59	0.84
	Boron	-2.46	0.85
	Cadmium	-4.39	0.94
	Chromium	-2.27	0.88
	Cobalt	-1.37	0.58
	Copper	-3.70	1.21
	Iron	1.85	0.61
	Lead	-2.15	0.77
	Lithium	-2.54	0.92
	Manganese	0.51	0.53
	Mercury	-2.98	1.16
	Methyl Mercury	-7.62	1.90
	Molybdenum	-3.37	0.82
	Nickel	-1.70	0.76
	Selemium	-2.04	0.44
	Silver	-4.52	0.86
	Strontium	-1.58	0.87
	Thallium	-4.19	0.85
	Thorium	-2.32	0.76
	Tin	-4.04	0.91
	Titanium	0.82	0.27
	Uranium	-3.17	0.80
	Vanadium	-1.71	0.79
	Zinc	-1.50	0.83
	Zirconium	-1.91	0.67
Nutrients	Ammonium	-4.51	1.43
	Calcium	0.70	0.95
	Magnesium	0.24	0.94
	Nitrogen	-5.07	0.99
	Phosphorus	0.49	0.59
	Potasium	-0.46	0.89
	Sodium	2.48	-0.11

Appendix C - Normalization GLM's

Sediment Quality Parameters		Model Parameters	
		Intercept	**Slope (Al)**
	Total kjeldahl nitrogen	-5.07	0.99
	Total Organic Carbon	-4.32	1.13
Total Polycyclic aromatic hydrocarbons (PAHs)		-1.06	1.09
Total Naphthenic acids (NA)		0.46	0.42
Particle Size	Sand	3.68	-0.51
	Silt	-6.09	1.97
	Clay	-3.79	1.25

Appendix C - Normalization GLM's

Table C3. Discharge (Q60) and particle size (PS) normalization models for algal and benthic indicators of community measured in the EMP program (2018, 2019 and 2021)

Biotic Index		Intercept	Slope (Q60)	Slope (PS)
Algal Community Indices	Density	16.64	-3.62	-
	Richness (LPL)	6.08	-1.47	-
	Diversity (LPL)	2.07	-0.39	-
	Evenness (LPL)	-2.28	0.85	-
	%Blue-Green Algae	0.34	-0.07	-
	Chlorophyll-*a*	3.46	-1.26	-
	Biomass	9.44	-2.31	-
	NMDS1	11.5	-3.75	-
	NMDS2	-1.78	0.58	-
Benthic Community Indices	Abundance	7.59	-1.73	-2.34
	Richness (LPL)	2.29	-0.37	-0.39
	Diversity (LPL)	0.13	0.18	0.02
	Evenness (LPL)	-0.44	0.27	0.41
	%EPT	0.17	0.42	2.08
	PTI	1.35	-0.17	0.06
	NMDS1	4.55	-1.64	-2.88
	NMDS2	5.38	-1.68	1.50
Female Trout-perch	Condition	1.70	-0.19	-
	GSI	-4.93	3.06	-
	LSI	0.59	0.32	-
Male Trout-perch	Condition	1.70	-0.20	-
	GSI	0.53	0.32	-
	LSI	1.74	-0.14	-

Table C4. Discharge (Q60) normalization models for fish population health indicators measured in the EMP program (2018, 2019 and 2021)

Fish Community Index	Intercept	Slope (Q60)
Total Abundance	1.66	0.0875
Species Richness	53.6	-15
Simpson’s Diversity	2.84	-0.714
Simpson’s Evenness	0.444	0.0175
NMDS1	-10.1	3.34
NMDS2	1.69	-0.542

Table C5. Discharge (Q60) normalization models for fish population health indicators measured in the EMP program (2018, 2019 and 2021)

Biotic Index		Intercept	Slope (Q60)
Female Trout-perch	Condition	1.70	-0.19
	GSI	-4.93	3.06
	LSI	0.59	0.32
Male Trout-perch	Condition	1.70	-0.20
	GSI	0.53	0.32
	LSI	1.74	-0.14

Appendix C - Normalization GLM's

Table C6. Fork length (FL) normalization models for a subset of fish body burden concentrations in the EMP program (2018, 2019 and 2021)

Species	Sample Matrix	Parameter	Intercept	Slopes	
				Fork Length (FL)	Fork Length (FL)
Trout-perch	Muscle	δ13C	1.33	0.04	0.01
		δ15N	1.03	0.07	-0.06
	Whole Body	EROD	5.68	-1.13	-1.29
		MeHg	-0.83	0.87	0.25
		Total Hg	-3.63	0.46	0.46
		Total Se	0.34	-0.45	0.08
Walleye	Muscle	δ13C	1.73	0.00	-0.10
		δ15N	0.86	0.05	0.02
		MeHg	-11.78	1.25	3.49
		Total Hg	-2.02	1.09	-0.40
		Total Se	-0.88	0.04	0.07
		BaP	148.31	-56.49	-
		PAH4	-1.17	0.16	-0.22
		Total PAH	0.78	0.46	-0.35
White Sucker		δ13C	1.68	-0.05	-0.03
		δ15N	-0.21	0.43	0.01
		MeHg	1.59	3.64	-2.86
		Total Hg	-10.01	3.54	-0.04
		Total Se	-4.87	1.63	0.00
		PAH4	7.65	0.05	-2.94
		Total PAH	-11.33	-0.25	4.08