

# Use of Arcellinida (testate lobose amoebae) Arsenic tolerance limits as novel tool for biomonitoring Arsenic contamination in lakes

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Use of Arcellinida (testate lobose amoebae) Arsenic tolerance limits as a novel

#### **ABSTRACT GRAPHIC**



#### 52 ABSTRACT

Arsenic (As) contamination from legacy gold mining in subarctic Canada poses an ongoing 53 threat to lake biota. With climatic warming expected to increase As bioavailability in lake 54 waters, developing tools for monitoring As variability becomes essential. Arcellinida (testate 55 lobose amoebae) is an established group of lacustrine bioindicators that are sensitive to changes 56 in environmental conditions and lacustrine ecological health. In this study, As-tolerance of 57 Arcellinida (testate lobose amoebae) in lake sediments (n = 93) in subarctic Northwest 58 Territories, Canada was investigated. Arcellinida assemblage dynamics were compared with the 59 60 intra-lake As distribution to delineate the geospatial extent of legacy As contamination related to the former Giant Mine (Yellowknife). Cluster analysis revealed five Arcellinida assemblages 61 that correlate strongly with ten variables (variance explained = 40.4%), with As (9.4%) and S1-62 carbon (labile organic matter; 8.9%) being the most important (*p*-value = 0.001, n = 84). 63 Stressed assemblages characterized proximal lakes <10 km from the former mine site, consistent 64 with a recently identified, geochemically-based zone of high As impact. The weighted average 65 tolerance and optima (WATO) analysis led to identification of three arcellinidan groups based 66 on the As-sensitivity: Low-Moderate Tolerance Group (As=0-350 ppm); High Tolerance Group 67 (As=350–760 ppm); and, Extreme Tolerance Group (As>750 ppm). The predictive capability of 68 the Low-Moderate and Extreme tolerance groups is particularly strong, correlating with As 69 70 concentrations in 66.6% (n = 20/30) of a test dataset. We propose that As influences the spatial 71 distribution of the more nutrient-sensitive Arcellinida taxa (e.g., Cucurbitella tricuspis and 72 Difflugia oblonga strain "oblonga") through suppression of preferred microbial food sources. These findings, which indicate that there is a variable species-level arcellinidan response to As 73 contamination, showcases the potential of using the group as a reliable tool for inferring 74

historical variability in As concentrations in impacted lakes, not possible using As itself due to
the redox driven sensitivity of the metalloid to post-depositional remobilization. Arcellinida can
also provide insight into the overall impact of As contamination on the ecological health of lakes,
a metric not readily captured using instrumental analyses. Lakes with As-stressed arcellinidan
faunas and high As concentrations may then be targeted for further As speciation analysis to
provide additional information for risk assessment.

Key words: Arsenic contamination, subarctic, gold mining, lake sediments, Arcellinida, As
 tolerance limits

### 84 1. INTRODUCTION

Arsenic (As) is a ubiquitous metal(loid), averaging 5 mg/kg<sup>-1</sup> in the Earth's crust (USDHHS, 85 2007). While As has various industrial applications (Wang and Mulligan, 2006), it is also 86 87 globally recognized as an element of environmental concern and is often linked to several ecological and human health hazards (Caussy and Priest, 2008). Due to the mineralization of 88 gold with As-rich sulfides, gold mining mineral processing activities are a primary 89 anthropogenic source of As in lake sediments and waters in mining districts worldwide (e.g., 90 Borba et al., 2003; Oyarzun et al., 2004; Palmer et al., 2015; Galloway et al., 2017). 91 Contamination of lake sediments and waters by As is of particular concern due to the substantial 92 recreational, ecological, and traditional values of lakes. Lake sediments can serve as a repository 93 for As that can be liberated to the overlying water column under certain environmental 94 conditions (e.g. seasonably variable redox conditions; Martin and Pedersen, 2002; Palmer et al., 95 2019). 96

The latent risk of gold mining-associated As contamination in lakes has been the impetus 97 behind numerous studies focused on characterizing As in lake sediments and waters (Azcue et 98 99 al., 1994; Bright et al., 1996; Andrade et al., 2010; Palmer et al., 2015; Galloway et al., 2015, 2017). Several instrumental techniques, like Instrumental Neutron Activation Analysis (e.g. 100 Salzsauler et al., 2005), Inductively Coupled Plasma (ICP) - Atomic Emission Spectrometry 101 (e.g., Ryu et al., 2002), and ICP - Mass Spectrometry (e.g., Galloway et al., 2017), are routinely 102 used as means to quantify the spatio-temporal variability of total As concentrations in sediments 103 104 of impacted lakes. However, results generated by such methods do not provide information on 105 the ecological response of lacustrine ecosystem to As contamination. Additionally, several 106 studies have highlighted limitations associated with utilizing elemental concentration profiles of redox-sensitive elements such as As, due to the potential of post-depositional remobilization 107

(Couture et al., 2008; Andrade et al., 2010; Schu et al., 2017). The long-term stability of As in 108 lake systems is dependent on its interaction with Fe-, Mn-, and Al-(oxy)hydroxides, organic 109 matter (OM), and sulfides, which in turn are mediated by factors like seasonal ice cover, pH, 110 redox condition, and biotic functions (e.g. microbial activity; Toevs et al., 2006; Du Ling et al., 111 2009). Limnological drivers can remobilize sedimentary As, and due to associated seasonal 112 changes in redox conditions, can result in cycling between the highly toxic inorganic species 113 (e.g.  $As^{+3}$  and  $As^{+5}$ ) and compounds of As (e.g. arsenic trioxide (As<sub>2</sub>O<sub>3</sub>), arsenite (AsO<sub>3</sub><sup>3-</sup>), and 114 arsenate (As $O_4^{3-}$ ); Martin and Pedersen, 2002; Palmer et al., 2019). Climate variability has also 115 been shown to have a profound impact on the stability of physical, chemical and biological 116 properties of lake systems (Rosenzweig et al., 2007). Computer model-predicted climate 117 warming and associated changes in redox conditions may lead to the release of As from 118 sediments into overlying surface waters in impacted lakes. There is thus a need to better 119 understand the long-term spatio-temporal variability of As in lake ecosystems to better predict 120 121 future geochemical trajectories and the potential impact on biota. Benthic meiofaunal communities are sensitive to environmental change in lakes. The 122

subfossil remains of such communities can be used to document the impact of contamination on 123 lacustrine ecosystems through time (e.g., Dixit et al., 1989; Cattaneo et al., 2004). Arcellinida 124 (i.e. testate lobose amoebae) are well established freshwater benthic bioindicators and are 125 routinely used to provide insights into both sediment character and the general ecological health 126 of lakes (Patterson and Kumar, 2002). This cosmopolitan group of shelled protists is found 127 across a wide geographical range that extends from the tropics to the poles (Beyens and Chardz, 128 1995; Dalby et al., 2000) living in fresh and brackish aquatic systems (Charman, 2001; Patterson 129 and Kumar, 2002; van Hengstum et al., 2008). The importance of Arcellinida as bioindicators is 130

mainly attributed to their: 1) reproduction (1–11 days) that enables rapid community response to 131 ecological change (Medioli and Scott, 1983); 2) high preservation potential owing to their decay-132 resistant tests (shells); and, 3) sensitivity to a wide range of environmental parameters (e.g., 133 Kumar and Patterson, 2000; Neville et al., 2011; Patterson et al., 2013; Roe et al., 2010; Prentice 134 et al., 2017). Such attributes offer a high degree of resolution of environmental interpretation, 135 which is imperative for monitoring the ecological health of lakes impacted by anthropogenic 136 contamination (Neville et al., 2011; Patterson et al., 2013). Several recent studies have applied 137 statistical techniques to evaluate the response of Arcellinida to mine-induced contamination (e.g., 138 Kumar and Patterson, 2000; Kihlman and Kauppila, 2012), but only a few have considered the 139 140 impact of As contamination on assemblage composition (Patterson et al., Reinhardt et al., 1998; Nasser et al., 2016; Patterson et al. 2019). In 2016, As was identified as a significant control on 141 142 arcellinidan distribution in lakes differentially impacted by As contamination associated with 143 legacy gold mining operations and mineral processing at the Giant Mine site (1948-1999) in the Yellowknife area, Northwest Territories, Canada (Nasser et al., 2016; Figure 1). The findings of 144 this proof-of-concept study provided new insight into the sensitivity of Arcellinida to As 145 contamination and identified the potential of using the group as a tool for monitoring changes in 146 As concentrations and ecological health in impacted lakes. 147

This study aims to further develop Arcellinida as a tool for biomonitoring variability in As concentrations and lacustrine ecological health by determining the tolerance limits of various taxa to varying As concentrations to identify specific As-indicator taxa or assemblages. The study represents the largest inter-lake assessment of the influence of mine-induced As contamination on Arcellinida assemblage dynamics in lake systems under take to date, as well as the first attempt to quantify the species-level response of Arcellinida to As concentration

- variability. A secondary objective of this study is to assess whether As-controlled arcellinidan
- assemblage dynamics vary as a function of distance from Giant Mine, likely due to downwind
- roaster stack-derived As aerial fallout. To achieve these objectives, the spatial distribution of
- 157 Arcellinida in 93



- 160 Figure 1. Map of sampling sites showing the locations of the 93 near-surface sediment samples examined in the study (colored
- 161 circles). The color-coding of the circles reflects the spatial distribution of As in the Yellowknife area, which is divided into three
- 162 categories: high (red circles), moderate (yellow circles), and low (green circles). The dashed circle represents the outer limits of the
- 163 Giant Mine zone of immediate airborne As contamination impact.

sediment-water interface samples from 90 lakes within a radius of ~30 km around the Giant 164 Mine site was examined. The publication of several recent studies on the regional distribution of 165 As in lake waters (Palmer et al., 2015) and sediments (Galloway et al., 2012, 2015) in the 166 Yellowknife area has provided valuable information about the geospatial extent of the As 167 contamination zone of impact. Palmer et al. (2015) delineated the limit of the zone of significant 168 aerial As fallout to be ~17 km away from the historic roaster stack at Giant Mine, based on an 169 170 assessment of the As concentration in 98 lake surface water samples. In an investigation of As levels in near-surface sediment samples, Galloway et al. (2017) identified a similar zone of 171 influence as well as a zone of immediate influence within a radius of 11 km around Giant Mine. 172 The dataset (n=93) used in this study is a subset of the data used by Galloway et al. (2017; n =173 105) and is thus directly comparable to that research. 174

175

#### 176 **2. Study Area**

The study was carried out in lakes within a radius of ~30 km around the Giant Mine, a former 177 gold mine located ~5 km northeast of the city of Yellowknife (Figure 1). Detailed information 178 179 pertaining to the history of gold mining in the Yellowknife area is provided in **Supplementary Document 1**. The study area is characterized by a gradual change in elevation, from 157 m above 180 mean sea level (MASL) close to Great Slave Lake to 350-400 m above MSAL to the north of 181 182 Thistlethwaite Lake (Kerr and Wilson, 2000). The primary drainage in the river catchment is via the Yellowknife River, which flows southward into Yellowknife Bay, Great Slave Lake. 183 184 Yellowknife has a subarctic, continental climate characterized by short, dry, cool summers with a mean annual temperature of -4.3° C and a mean annual precipitation of 170.7 mm 185

(Environmental Canada, 2019). The wind direction is variable throughout the year but blowsprimarily from the east and south (Pinard et al., 2007).

Lakes investigated in this study are underlain by rocks assigned to the Yellowknife
Supergroup of the southern Slave structural province of the Canadian Shield. These include
Archean metavolcanic and metasedimentary rocks intruded by younger granitoids and diabase
dykes (Yamashita et al., 1999; Cousens, 2000). The most prevalent surficial sediments in the
study region are fine clastic lacustrine sediments from Glacial Lake McConnell and glacigenic
sediments that form a thick (<2 m) discontinuous veneer (Kerr and Wilson, 2000).</li>
Accumulations of Holocene-aged peat also occur in the study region and can be greater than 1 m

thick in bogs and wetlands (Kerr and Wilson, 2000).

196

### **3. MATERIALS AND METHODS**

#### **3.1 Field methods**

A total of 93 surface sediment samples (upper ~ 1 cm) were collected from 90 lakes around the 199 sites of Giant and Con Mines in 2012 (sample ID: B12; n = 61) and 2014 (sample ID: Y14; n =200 32; Figure 1). Lakes located within a radius of 30 km from the mines were targeted to ensure 201 202 coverage of areas beyond the Airborne As fallout zones of impact (~17km) and immediate 203 impact (~11 km) proposed by Palmer et al. (2015) and Galloway et al. (2017). Lakes were accessed via a pontoon-equipped Bell Long Ranger helicopter. Surface sediment samples were 204 collected by Ekman Grab approximately 1 cm of sediment from the top of each grab, where 205 Arcellinida populations are often abundant, was retained using an inert plastic laboratory spoon 206 for arcellinidan, sedimentological and geochemical analyses. The location of each sampling 207 station was recorded by Global Positioning System (GPS; accuracy  $\pm 3$  m) (Galloway et al., 208

209 2015, 2017). The water sampling depth at each station was determined using a HONDEX Honda portable handheld depth sounder (model: PS-7; optimal depth range: 0.6-80 m; beam 210 angle: 24°; Galloway et al., 2015, 2017). Where possible, muddy substrates from the middle of 211 each lake were selected for sampling because arcellinidan populations are typically reduced on 212 nutrient-poor silt to sand substrates (Patterson and Kumar, 2002). Water property data (pH, water 213 temperature, dissolved oxygen and conductivity) was collected from each sample site using a 214 215 YSI Professional Plus handheld multi-parameter unit with quatro-cable (Galloway et al., 2015, 216 2017).

217

### 218 **3.2 Laboratory methods**

Samples used in this study were subsampled and analyzed for element and organic geochemical, 219 sedimentological, and micropaleontological analysis. Elemental concentrations of the sediment 220 221 subsamples were analyzed using ICP-MS following aqua regia digestion (ICP-MS 1F/AQ250 package) at Bureau Veritas, Vancouver (Supplementary Table 1). Aqua regia digestion was 222 used instead of complete digestion as the former provides the total concentration of metal(loids) 223 that could potentially become bioavailable, while the latter can volatilize As (Parsons et al., 224 2012). Analytical precision was assessed using three Pulp duplicates. Calculated Relative 225 Percent Difference (RPD) was less than 5% for As (RPD range = 1.47%-4.31%). Analytical 226 accuracy was assessed using three standard reference materials: 1) STD DS9 (n = 9); 2) STD 227 D10 (n = 2); and, 3) STD OREAS45EA (n = 11). Mean As concentration measured in STD DS9 228 was 27.4 ppm  $\pm$  1.42 (n = 9) compared to an expected *aqua regia* concentration of 229 25.5 ppm (mean RPD =  $7.806\% \pm 3.95$ ). Mean As concentration measured in STD DS10 was 230 45.6 ppm  $\pm$  0.1 (n = 2) compared to an expected *aqua regia* concentration of 46.2 ppm (mean 231

232 RPD =  $1.307\% \pm 0.3101$ ). Mean measured As concentration for STD OREAS45EA was

233 9.7 ppm  $\pm$  1.16 (n = 11) compared to an expected *aqua regia* As concentration of 10.3 ppm

234 (mean RPD =  $11.1\% \pm .7.27$ ). Analyzing eleven laboratory methods blanks resulted in detecting

As in only two blanks (detected As concentrations = 0.2 ppm and 0.1 ppm).

Particle size analysis (PSA) was performed on the sediment subsamples to recognize 236 sedimentological patterns across the study area that may influence the distribution of Arcellinida 237 238 and element concentrations. Subsamples were prepared for PSA by digesting subsamples in a 239 heated bath (70 C°) with 10% HCl and 30% H<sub>2</sub>O<sub>2</sub> to remove carbonate and organic content, respectively (Murray 2002; van Hengstum et al., 2007). Following digestion, sedimentary grain 240 241 size in each subsample was analyzed using a Beckman Coulter LS13 320 laser diffraction analyzer fitted with a universal liquid medium (ULM) sample chamber over a measurement 242 243 range between 0.4 and 2000lm. Samples were loaded into the instrument until an obscuration level of 10±3% was attained. GRADISTAT (Version 8; Blott and Pye, 2001) was used to 244 245 compile the results (Supplementary Table 1). Garnet 15 (mean diameter 15  $\mu$ m:  $\pm 2 \mu$ m), an accuracy standard supplied by Beckman Coulter, was run once per month. An in-house mud 246 247 sample (Cushendun Mud; mean diameter =  $20.5 \ \mu\text{m}$ :  $\pm 0.76 \ \mu\text{m}$ ) was run at the start of every session as a precision control. 248

Sediment subsamples were also analyzed for organic matter content using the Rock-Eval® 6 instrument at the Geological Survey of Canada, Calgary. Rock-Eval® 6 Analysis uses heat to break down large organic matter molecules to smaller and chemically more identifiable molecules (Lafargue et al., 1998). Quantitative measurements of total organic carbon (TOC) and other organic geochemical variables, including S1 carbon, S2 carbon, and S3 carbon were produced (Supplementary Table 1). S1-carbon represents the quantity of free hydrocarbon in

sediments (mg hydrocarbons/g) that is devolatilized during pyrolysis at 300 °C. In sediment-255 water interface sample, S1 mainly consists of readily degradable geolipids 256 and pigments predominantly derived from autochthonous organic matter such as algal-257 derived lipids (Carrie et al., 2012). S2-carbon represents the quantity of large molecules, 258 kerogen-derived hydrocarbons released through thermal cracking of the organic matter, in 259 sediment samples (mg hydrocarbons/g) near 650 °C. The S2 compounds in sediment generally 260 261 correspond to highly aliphatic biomacromolecule structures of algal cell walls (Meyers and 262 Teranes, 2001. S3 represents the amount of carbon dioxide released during pyrolysis of kerogen, while in sediment samples it represents lignins, terrigenous plant materials, humic and fulvic 263 264 acids (Carrie et al., 2012). The quantity of all organic matter released during pyrolysis and oxidation heating accounts for TOC (wt.%) in sediment samples. Analyses of standard reference 265 material (IFP 160000, Institut Français du Pétrole and internal 9107 shale standard, Geological 266 267 Survey of Canada, Calgary; Ardakani et al., 2016) show accuracy and precision to be greater than 5% relative standard deviation. 268 Sediment subsamples (3 cm<sup>3</sup>) were used for micropaleontological analysis. Subsamples 269

were first wet sieved through a coarse (297  $\mu$ m) and fine (37  $\mu$ m) sieves to remove any coarse 270 debris (e.g. grass and sticks) and retain Arcellinida tests, respectively. A wet splitter (Scott and 271 272 Hermelin, 1993) was used to subdivide each subsample into six aliquots for quantitative analysis. 273 Aliquots were identified and enumerated wet for total Arcellinida tests (live plus dead) on a gridded petri dish using an Olympus SZH dissecting binocular microscope (7.5-64X 274 magnification) until, whenever possible, a statistically significant number of specimens were 275 quantified (Supplementary Table 1; Patterson and Fishbein, 1989). Although living Arcellinida 276 specimens may have been present at the time of sampling the samples were not stained so the 277

enumerated Arcellinida analysis was carried out on live plus dead Arcellinida specimens (i.e., 278 arcellinidan tests). Identification of Arcellinida primarily followed the illustrations and 279 descriptions found in various key papers where specimens are well illustrated (e.g. Reinhardt et 280 al., 1998; Roe et al., 2010; Patterson et al., 2013). Arcellinidan species can display considerable 281 environmentally controlled infraspecific morphological variability (e.g., Medioli and Scott, 282 1983). To deal with this phenotypic plasticity, the accepted practice has been to designate 283 284 informal infrasubspecific "strain" names for these ecophenotypes (Asioli et al., 1996; Reinhardt et al., 1998; Patterson and Kumar, 2002). While infrasubspecific level designations have no 285 status under the International Zoological Code of Nomenclature (art. 45.5; 4th edition, 1999; 286 287 ICZN, 1999), they have been extensively used in the literature for defining environmentally significant populations within lacustrine environments (e.g. Reinhardt et al. 1998; Kumar and 288 289 Patterson, 2000; Patterson and Kumar, 2002; Roe et al., 2010; Steele et al., 2018). Scanning 290 electron microscope images of common species and strains were obtained using a Tescan Vega-II XMU VP scanning electron microscope (SEM) in the Carleton University Nano Imaging 291 Facility. All SEM plates were digitally produced using Adobe Photoshop<sup>TM</sup> CC 2018 (Figure 2; 292 Figure 3). 293

294

## 295 **3.3 Data screening, variables reduction**

The data were screened to remove samples or variables characterized by >25% missing values and values below the lower method detection limit (MDL; e.g., As lower MDL = 0.1 ppm) or above the upper MDL (e.g., As upper MDL = 10000 ppm; Reimann et al., 2008). Samples with geochemical results below the lower MDL were converted to  $\frac{1}{2}$  lower MDL (e.g., 0.05 ppm for As). Values that exceeds the upper MDL are changed to upper MDL (e.g., 10000 ppm for As; applicable only to sample BC19; Reimann et al., 2008). These criteria resulted in the removal of
five samples from the analyses (B44, B56, B59, Y56, Y59).

Because the inclusion of all measured variables in ordination analyses (e.g. redundancy 303 analysis) creates clutter that can mask meaningful patterns generated by these methods, we used 304 305 the Spearman's Rank correlation and Variance Inflation Factor (VIF) to reduce the number of variables used in the analyses. Spearman's Rank correlation served to remove highly correlated 306 307 variables ( $r_s > 0.7$ ), while VIF was employed to ensure the removal of highly collinear variables 308 (VIF >10) (Supplementary Table 2). Although TOC had collinear features with a number of variables (e.g. As and S1-carbon) it was retained for statistical analyses as this variable is known 309 310 to influence the distribution of several key arcellinidan taxa as well as sediment chemistry 311 (Patterson and Kumar, 2002).

312

#### 313 **3.4 Statistical analyses**

Thirty arcellinidan species and strains were identified in this study. Statistical analysis carried out on the Arcellinida dataset is described in Nasser et al (2016). Based on calculated Probable Error (pe) and Standard Error (Sxi), six samples (B10, B20, B48, B55, B59, and Y69) containing statistically insignificant populations and five statistically insignificant species were excluded from subsequent multivariate data analyses (**Supplementary Table 1**).

RStudio statistical software (version 0.98.1028; R Core Team, 2014) was used to carry out several statistical and multivariate analyses on measured parameters and species data. As recommended by by Fishbein and Patterson (1993) Q-and R- mode cluster analysis, using Ward's Minimum variance method and Euclidean distance (Ward, 1963), was used to group samples containing similar Arcellinida assemblages and to determine which species were most closely associated (R packages: stats, cluster, and gplots). Non-metric multidimensional scaling

325	(NMDS; Kruskal, 1964) was used to further investigate the results of cluster analysis by
326	assessing the similarity between identified assemblages in multidimensional space (R package:
327	vegan). Redundancy analysis (RDA; van den Wollenberg, 1977) of the post-screening data sets
328	(84 samples and 25 species and strains) was used to evaluate the relationship between
329	arcellinidan assemblages and measured environmental variables (R package: stats). A series of
330	partial RDAs (pRDA), coupled with variance partitioning tests, were carried out to identify the
331	significance of the RDA axes and measured variables (R package: stats). Variables with a $p$ <
332	0.05 were considered to be significant contributors to variance in the arcellinidan assemblage.
333	Analysis of Arcellinida tolerance and optima to As spatial variability was carried out using
334	Weighted Average Tolerance and Optima (WATO; Ter Braak and Barendregt, 1986) methods
335	performed through the package 'analogue' in RStudio. The method produced ecological optima
336	values and tolerance limits (upper and lower limits) for each



Figure 2. Scanning electron microscope of selected arcellinidan tests from the study lakes. Formore taxonomic information see Supplementary Materials Section 3.



Figure 3. Scanning electron microscope of selected Arcellinidan tests from the study lakes. For
 more taxonomic information see Supplementary Materials Section 3.

348

identified taxa, which is necessary for the identification of indicator-species and/or assemblages

## 349 (Supplementary Table 3).

350

## 351 4. RESULTS AND DISCUSSION

#### 352 4.1 Spatial Distribution of As

Measured sedimentary As concentrations were significantly higher than the levels proposed by 353 the interim sediment quality guidelines (ISQG; 5.9 ppm; CCME, 2002) and probable effect level 354 guidelines (PEL; 17 ppm; CCME, 2002) in 94% (n=84) of the samples (n=89; Supplementary 355 Table 1). This is particularly evident in lakes to the west (median As = 290.4 ppm; range = 356 30.2-4778.2 ppm n=27) and north (median As = 147 ppm; range = 16.1-10000 ppm n=24) of the 357 Giant Mine. Median As levels in lakes to the east (As = 36.3 ppm; range = 9.7-553.9 ppm n=19) 358 359 and south (median As = 31.3 ppm; range = 6.3-317.8 ppm n=13) of the mine were comparatively lower, yet remain above the ISQG and PEL guidelines. 360 A negative Spearman's Rank coefficient between sedimentary As concentration and the 361 362 distance from the roaster site ( $r_s = -0.5$ ) indicates decreasing As concentrations in distal lakes (Supplementary Table 3). This spatial pattern reflects the influence of the prevailing 363 364 southeasterly winds (Pinard et al., 2007), which transported As-bearing stack emissions from the 365 Giant and Con mines toward the northwest (Palmer et al., 2015; Galloway et al., 2017). The influence of prevailing wind direction may also explain the persistence of elevated levels of As 366 in distal lakes to the north (B2; distance = 17 km, As=905.2 ppm) and west of the historic mining 367 operations (Y15; distance = 21.5 km; As = 689.9). Recent studies confirm the persistence of 368 As<sub>2</sub>O<sub>3</sub> in lake sediments downwind of Giant Mine (Galloway et al., 2017; Schuh et al., 2017; 369 Van den Berghe et al., 2017). Geogenic As is elevated in the Yellowknife area (background 370 concentration = 150 ppm; Risklogic, 2002), yet its contribution of As to these lakes is minor. 371

#### 372 4.2 Arcellinida Assemblages

373 The results of Q-mode cluster analysis (Figure 4), and NMDS (Figure 5) revealed five distinct arcellinidan assemblages: 1) "High As Contamination Assemblage (HAC)"; 2) "As 374 contamination Assemblage (AC)"; 3) "Centropyxis aculeata Assemblage (CA)"; 4) "Transitional 375 Assemblage (T)"; and 5) "Healthy Assemblage (H)". Results of the R-mode cluster analysis 376 suggests that only seven out of the 25 identified arcellinidan taxa (Difflugia elegans Penard, 377 378 1890, Centropyxis constricta (Ehrenberg, 1843) "constricta", Centropyxis constricta (Ehrenberg, 379 1843) "aerophila", Centropyxis aculeata (Ehrenberg, 1843) "aculeata", Cucurbitella tricuspis (Carter, 1856), Difflugia oblonga Ehrenberg, 1832 "oblonga" and Difflugia glans Penard, 1902 380 381 "glans") contributed significantly to defining the derived faunal assemblages (Figures 4, 5; Supplementary Table 1). The unique faunal structure of each assemblage reflects ecological 382 conditions characteristic of stressed (e.g. HAC, AC, and CA; SDI values 1.3-2.2), transitional 383 384 (e.g. T; SDI values 1.6-2.5) and relatively healthy lacustrine systems (e.g. H; SDI values 1.7-2.4). The variability reflected by the assemblages developed mostly in response to ten significant 385 environmental parameters (As, S1 carbon, sulfur [S], sodium [Na], calcium [Ca], distance to 386 Giant Mine, phosphorous [P], barium [Ba], mercury [Hg], and total organic content 387 [TOC])identified by using partial RDA analysis that explain ~40% of the variance in the 388 arcellinidan distribution (Figure 6). Arsenic (9.4%) and S1-carbon (8.9%) exert the most 389 390 influence over the composition of the identified assemblages and collectively explain 18.3% of the total variance. A detailed description of the location, taxonomic composition and primary 391 environmental controls for each assemblage is provided in Supplementary Material. 392



Figure 4. Combined Q-mode and R-mode cluster dendrogram for the 84 samples and 25
statistically significant species and strains. Five faunal assemblages are indicated.



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Figure 5. Non-Metric Multidimensional Scaling (NMDS) bi-plot. AV - Arcella vulgaris, CAA -398 Centropyxis aculeata "aculeata", CAD - Centropyxis aculeata "discoides", CCA - Centropyxis 399 constricta "aerophila", CCC – Centropyxis constricta "constricta", CCS – Centropyxis constricta 400 "spinosa", CP – Conicocassis pontigulasiformis, CT – Cucurbitella tricuspis, MC – Mediolus 401 corona, DB – Difflugia bidens, DOO – Difflugia oblonga "oblonga", DOS – Difflugia oblonga 402 "spinosa", DOT – Difflugia oblonga "tenuis", DGG – Difflugia glans "glans", DGD – Difflugia 403 glans "distenda", DU – Difflugia urens, DE – Difflugia elegans, DPP – Difflugia protaeiformis 404 "protaeiformis", DPAC – Difflugia protaeiformis "acuminata", DPCl – Difflugia protaeiformis 405 406 "claviformis", DCUR – Difflugia protaeiformis "curvicaulis", DPSC – Difflugia protaeiformis "scalpellum", LS – Lesquereusia spiralis, LV – Lagenodifflugia vas, PC – Pontigulasia 407 408 compressa

#### 410 **4.3** Controls over the Distribution of Arcellinida

The RDA and pRDA results in **Figure 6** show that As is the dominant control on the distribution 411 of Arcellinida. Stress-tolerant taxa such as D. elegans, and C. constricta strains 'aerophila' and 412 'constricta' dominate in HAC and AC assemblages (SDI range = 1.4-2.2). This is likely a 413 response to the elevated sediment As levels (median As = 290.4 ppm; range = 21.1-10000 ppm; 414 n = 31), which in turn is attributed to the downwind location (93.5% of lakes located to the west 415 or north; n = 28) and close proximity to the historic Giant Mine roaster stack as a point source of 416 417 As contamination (median distance to Giant Mine roaster stack = 9.85 km; n = 31). Levels of As were notably lower in samples associated with the CA assemblage (median As = 258.1 ppm; 418 419 range = 33.4-921.1 ppm; n = 14), even though such lakes were relatively close to the mine (median distance to Giant Mine roaster stack = 6.6 km; n = 14). Most (57%; n=8) of the lakes 420 421 hosting the CA assemblage were located upwind of the roaster, while the remaining lakes (43%; 422 n=6) are situated downwind from the mine site (Figure 1). The reduction in As levels is associated with a moderately diverse assemblage (SDI = 1.3-2.5; median SDI = 2) that is 423 characterized by the emergence of C. aculeata "aculeata" as a dominant member, a notably 424 lower number of stress-indicating taxa, and a slight elevation in the number of healthy-lake 425 species and strains (e.g. C. tricuspis, D. oblonga "oblonga" and D. glans "glans"). With greater 426 427 distance from the Giant Mine site, the stress-indicating HAC and AC assemblages were less 428 common and the lakes became dominated by a transitional assemblage (T; SDI = 1.6-2.5; median SDI = 2.1) that is comprised of lower proportions of stress-indicating species and higher 429



Figure 6. Redundancy Analysis (RDA) species-environment-sample tri-plots for the 84
sediment-water-interface samples that yielded statistically significant arcellinidan populations
and had no missing values. The five identified Arcellinida assemblages are: 1) the high As
contamination assemblage (red square); 2) As contamination assemblage (light blue diamond); *Centropyxis aculeata* assemblage (purple pentagon); 4) transitional assemblage (blue
triangle); and 5) healthy assemblage (green circle).

numbers of healthy-lake indicating Arcellinida taxa. This result was expected since the 444 transitional assemblage was observed most commonly in relatively distal lakes (median distance 445 to Giant Mine roaster stack = 12.6 km; n = 14) and characterized by moderate to low As levels 446 (median As = 76.5 ppm; range = 16.1-740.7 ppm; n = 14). The healthiest arcellinidan 447 assemblage (H; SDI = 1.7-2.4; median SDI = 2.1) was found in lakes >10 km from the mine site 448 (median distance = 19.6 km; n = 25) in lakes characterized by the lowest As concentrations 449 (median As = 30.3 ppm; range = 6.3-905.2 ppm; n = 25). The observed faunal shift from 450 stressed to healthy assemblages suggest well defined zones of impact (radius of ~ 20 km around 451 the Giant Mine) and immediate impact (radius of  $\sim 10$  km around the Giant Mine), which are 452 453 consistent with the geospatial extent of the zones delineated by Palmer et al., (2015) and Galloway et al., (2017) (17 km and 11 km, respectively). 454

455 The RDA and pRDA results indicated that the labile fraction of total organic matter (S1-456 carbon) is also a significant control over the distribution of Arcellinida taxa in area lakes (Figure 6). The RDA tri-plot shows that S1-carbon and As are closely correlated, which is corroborated 457 by a significant positive Spearman's Rank correlation between the two variables ( $r_s = 0.5$ ; 458 Figure 6, Supplementary Table 2). It is notable that the arcellinidan response to the spatial 459 variability of S1-carbon was similar to that observed with the As concentrations. The highest 460 461 average levels of S1-carbon in HAC (median S1-carbon = 50.4 HC/g rock; range = 33.9-66.5462 HC/g rock; n = 15) and AC assemblages (median S1-carbon = 53.2 HC/g rock; range = 34.8-59.8 HC/g rock; n = 16) are associated with elevated levels of As and the dominance of stress-463 indicating taxa. The lowest median S1-carbon values are associated with low As levels and 464 healthy lake taxa (HA; median S1-carbon = 18.9 HC/g rock; range = 0.4-36.4 HC/g rock; n =465 25). Previous studies have similarly reported high organic matter content in metal-contaminated 466

soils (Valsecchi et al., 1995; Kelly and Tate, 1998) and lakes (Gough et al., 2008; Gough et al., 467 2010). The relationship between organic matter and As may reflect: 1) organic matter 468 mediatiation reduction of As<sup>5+</sup> to As<sup>3+</sup> and subsequent release of sediment-bound As into 469 sediment pore water and the overlying water column (Van den berghe et al., 2017); 2) 470 competition with As over for sorption sites (Redman et al., 2002); or 3) enhancement of As 471 sequestration by providing an organic substrate with a large surface area for metal(loid)-organic 472 473 matter complexation (Grafe et al., 2001). Galloway et al. (2017) reported a significant 474 association between As, S1-carbon and S in Yellowknife area lakes. The relationship was interpreted to reflect S1-carbon as an organic substrate suitable for microbial growth, which in 475 476 turn mediated the authigenic precipitation of As derived from roaster emission to As sulfides. The similar arcellinidan response to both variables in this study suggests that both influence 477 478 faunal ecology.

479 While phosphorus (P) explains a small portion of the variance in Arcellinida distribution (1.8%), spatial variability of P in the study area seems to follow a relatively modest trend of 480 increasing concentrations in distal lakes (median P = 1060 ppm; median distance = 16.7 km; n =481 39) compared to lakes closer to the Giant Mine (median P = 880 ppm; median distance = 8.1 km; 482 n = 45). As expected, the RDA tri-plot shows that P is positively associated with distance from 483 484 the Giant Mine, and negatively with As and S1-carbon (Figure 6). While organic matter is a 485 limiting factor for bacterial growth, an increasing number of studies have documented the importance of several elements, particularly P, in controlling bacterial growth efficiency and 486 attainable biomass within a wide range of aquatic systems (Toolan et al., 1991; Elser et al., 1995; 487 Gurung and Urabe, 1999). In addition, the availability of P has been shown to influence As 488 toxicity to primary producers in freshwater systems (Levy et al., 2005; Wang et al., 2013). 489

Arsenate  $(AsO_4^{3-})$  and phosphate  $(PO_4^{3-})$  are chemically analogous. This similarity allows arsenate to substitute for phosphate, when the availability of the latter is low, and pass into the cell via phosphate transporters and inhibit phosphorylation, which consequently impacts several protein functions and cellular growth (Meharg and Macnair, 1991). Therefore, the slightly reduced proportions of P in lakes closer to the Giant Mine may play a role in intensifying As toxicity to microbial and arcellinidan communities, influencing their distribution in the process.

## 497 4.4 Interaction between Arcellinida, As and S1

Healthy and active microbial communities thrive within organic substrates, especially the labile 498 fraction (i.e. S1-carbon; Sanei et al., 2005). The development of such communities may provide 499 500 an adequate source of nourishment for Arcellinida, which feed on bacteria, algae and fungi 501 (Nikolaev et al., 2005). However, As is known to be toxic to most bacteria, except As-tolerant strains, as it can inhibit basic cellular functions linked to energy metabolism, basal respiration 502 and enzyme activities (Baath, 1989; Walker et al., 2000). The effects of As toxicity have also 503 been shown to be associated with a significant reduction in the microbial biomass in soil 504 (Maliszewska et al., 1985; Hiroki, 1993; Simon, 2000), and, to a lesser extent, in lacustrine 505 environments (Gough et al., 2008; Gough et al., 2010). Such a reduction in microbial biomass 506 may induce sufficient environmental stress to impact nutrient-sensitive arcellinidan taxa (e.g. C. 507 tricuspis and D. oblonga "oblonga") as competition for dwindling food resources intensifies. An 508 experimental study by Burnskill et al (1980) suggests that As may inhibit the activity of organic 509 matter-reducing bacteria during the winter, while exerting no influence on bacterial productivity 510 in the summer. A recent study by Palmer et al. (2019) in the Yellowknife area reported elevated 511 levels of As<sup>5+</sup> in well-mixed surface waters during the summer when oxic conditions dominate, 512

and higher concentrations of As<sup>3+</sup> during the winter when ice cover results in reducing 513 conditions. Such seasonal inhibition may lead to a reduction in microbial biomass and a 514 concurrent buildup of organic matter during the winter, which may decompose during the 515 summer upon the recovery of microbial activities. A similar positive correlation between higher 516 proportions of organic matter, represented by TOC, and elevated concentrations of heavy metal 517 contamination have been reported by several studies (Kelly and Tate, 1998; Valsecchi et al., 518 519 1995; Gough et al., 2008). TOC is relatively high in our study (median = 25.6%), which may 520 explain the association of stress-tolerant taxa with high levels of As, S1-carbon, and TOC, and less tolerant species with relatively lower As, S1-carbon, and TOC levels in our study. Based on 521 522 our results we propose a mechanism whereby As has an indirect influence over the spatial distribution of Arcellinida in the study area through suppression of growth of their food resource 523 524 (i.e. microbial communities).

525 The introduction of a contaminant into an aquatic system may directly or indirectly impact biota. Direct influences can increase mortality rates through toxicity and reduce 526 populations, while indirect effects may either decrease (e.g., by limiting sources of nourishment) 527 or increase (e.g., by lowering competition over food sources) the population. Unfortunately, 528 studies designed specifically to assess the direct and indirect influence of As on Arcellinida are 529 530 lacking. The results of this study provide information that explains indirect impacts of As on arcellinidan trophic function but it is important to note that the hypothesized relationship does 531 not preclude any direct impact of As toxicity on arcellinidan species in the Yellowknife area 532 lakes. 533

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#### 536 4.5 Tolerance of Arcellinida Taxa to As

In environmentally stressed lakes, the biotic components often display a range of tolerances in 537 response to the introduction of contaminants into the system (Negro and de Hoyos, 2005). For 538 instance, contamination is often associated with a massive reduction in the biomass of intolerant 539 species, while resilient taxa may show little indication of ecological stress, and may in fact 540 expand to fill the ecological void. Such variation in tolerance was captured by the WATO 541 analysis performed on the 25 Arcellinida taxa, which exhibited a wide tolerance to changes in As 542 543 concentration (Figure 7, Supplementary Table 3). The optima values and upper tolerance limits exhibited a modest increasing trend accompanied by a taxonomic gradient from steno-544 545 metalloid (As) species (e.g. healthy lake taxa) to highly eury-metalloid (As) Arcellinida (e.g. stressed lake taxa). This association is in agreement with the findings of studies linking the 546 abundance of healthy-lake and stressed-lake arcellinidan taxa to low or high levels of As, 547 548 respectively (Patterson et al., 1996; Reinhardt et al., 1998; Nasser et al., 2016). However, the identification of robust As indicator species depends on quantitative characterization of species 549 with both well-defined ecological optima and narrow tolerance ranges (Negro and de Hoyos, 550 2005). While the results of the analysis define the optima and upper tolerance As values for each 551 taxon. The lower As tolerance limit for all the identified Arcellinida species was 0 ppm. This 552 553 result is not surprising because the 25 arcellinidan taxa as would also be expected to be found in 554 substrates where As is present in low concentrations, or is even absent.

While it was not possible to identify any individual high concentration steno-metalloid (As) indicator species in our study, the results of the WATO method reveal three groupings of taxa that reflect certain As concentration ranges: (1) "low-moderate tolerance group" (LMTG; As range = 0-350 ppm); (2) "high tolerance group" (HTG; As range = 360-760 ppm); and (3)

<sup>559</sup> "extreme tolerance group" (ETG; As range = >750 ppm; Figure 7). The LMTG includes 12 arcellinidan species and strains with a relatively low to moderate As optima (99.3-225 ppm) and upper tolerance As range of 156.7-355.5 ppm. The species composition of the LMTG is primarily represented by *Difflugia* (n= 8) species and *C. tricuspis*, which are known to be abundant in relatively healthy lakes (Patterson et al., 1996; Neville et al., 2011). Therefore, members of this group are likely to be characteristic of assemblages from lakes were As levels do not exceed 360 ppm.

Species comprising the ETG are characterized by well-known stress-tolerant taxa (e.g. 566 centropyxid species and stains and *D. elegans*), elevated optima range (507-1433.6 ppm) and 567 568 upper tolerance range (1382.5-2613.5 ppm). Centropyxid species and strains are known for their opportunistic nature and ability to withstand a variety of severely stressed environmental 569 conditions (Medioli and Scott, 1983; Kihlman and Kaupplia, 2012; Nasser et al., 2016; Gavel et 570 571 al., 2018). In addition, D. elegans was previously reported (identified as D. protaeiformis "amphoralis") as being abundant in substrates with high As concentration (300-2100 ppm; 572 573 Reinhardt et al., 1998). Therefore, arcellinidan assemblages dominated by these species are likely to be present under a wide range of As levels but expected to dominate when levels of As 574 are extremely elevated (>750 ppm). The HTG (360-760 ppm), spanning As concentrations 575 between the LMTG and ETG, is composed of six species (A. vulgaris, L. vas and D. urens) and 576 strains (D. glans "distenda", D. oblonga "tenuis" and C. aculeata "aculeata"). These taxa 577 include representatives of both the Difflugia-dominated LMTG and Centropyxid-dominated 578 ETG. This explains why there is an overlap of As tolerance between the HTG [As optima (241-579 387.3 ppm] and upper tolerance limits [540.7-765.7 ppm]) and both the LMTG and ETG. Since 580 the HTG is found both in lakes where As levels are below 360 ppm and those with 581



Figure 7. Results of the Weighted Average Tolerance and Optima Analysis (WATO) on the 25
statistically significant arcellinidan taxa.

As concentrations between 360-760 ppm, it is a generally less useful As indicator than theLMTG and ETG.

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## 588 4.6 Paleoenvironmental Assessment Tool

The reliability of using the identified Arcellinida As tolerance groups to infer temporal changes 589 in As levels in impacted lakes was assessed on a test dataset comprising arcellinidan assemblages 590 591 and As concentrations (measured by ICP-MS) from 30 subsamples from a freeze core collected from Frame Lake, Northwest Territories, Canada (Gavel et al., 2018). We hypothesized that As 592 ranges inferred by the relative abundance of taxa in the three tolerance groups derived from the 593 594 inter-lake data set would represent measured total As concentrations from each core subsample independently measured using ICP-MS. Frame Lake sediments represented ideal test material as 595 596 the well documented system was initially impacted by As contamination, then nutrient loading, 597 and overprinted by redox-influenced remobilization of As upwards the stratigraphic column (Gavel et al., 2018). 598

Our results show that the measured As levels in 66.6% of the samples (n=20) where 599 remobilization of As fell within the As range suggested by the bioindicator groupings (Figure 8; 600 Supplementary Table 3). The identification of an LMTG assemblage (0-350 ppm As) in 16 601 samples was in line with measured As concentrations of 121.5-278 ppm. In addition, four 602 603 samples characterized by very high levels of As (913.2-1473.5 ppm) were dominated by the ETG members. The two model mismatches (FL21, FL22) were dominated by high proportions of As-604 tolerant taxa from the HTG and ETG assemblages, even though the measured As concentrations 605 of both samples is below 350 ppm. These results may be indicative of post-depositional 606

- 607 remobilization of As out of these horizons (Supplementary Table 3). Unidentified confounding
- 608 environmental stressors



- **Figure 8.** The correlation between Arcellinida As tolerance group and As concentrations of 30
- 611 freeze core samples from Frame Lake. The strength of the correlation is represented by three
- 612 colored symbols, with the green square representing a strong correlation, the orange triangle
- 613 representing a mismatch (weak correlation) attributed to the influence of As remobilization, and
- the red circle representing a mismatch attributed to the influence of other confounding variables
- 615 (modified after Gavel et al., 2018).

may have also contributed to the observed fauna in these samples. The mismatches between the arcellinidan fauna and As concentrations in the other eight samples were likely attributable to variability in redox conditions, which resulted in As remobilization (Gavel et al., 2018). This result highlights the utility of these bioindicators as a tool to reconstruct As concentrations in sedimentary records, since the non-mobile arcellinidan assemblages provide a faithful paleoenvironmental record of As contamination at the time of deposition, regardless of any postdepositional remobilization of As.

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#### 624 **5.** Conclusions

625 Arcellinidan taxa (n = 25) in 84 sediment-water interface samples from the Yellowknife area responded to a decline in As concentrations further from the Giant Mine site by shifting from 626 627 stressed assemblages near the mine to healthier assemblages in distant lakes (>10 km), thus 628 corroborating the geographic extent of the airborne As contamination zone of immediate impact delineated by Palmer et al. (2015) and Galloway et al. (2017). The results also show that 629 arcellinidan groups, based on As-tolerance limits, can be used to successfully infer As 630 concentrations in 20 out of 30 freeze core samples impacted by mine-induced As regardless of 631 As post-depositional remobilization. These results establish the utility of arcellinidan 632 633 bioindicators as an independent proxy for monitoring changes in As concentrations and the 634 ecological health in lakes impact by mine-induced As contamination. While the findings of this study confirm a strong relationship between Arcellinida and mine-derived As contamination in 635 high latitude lakes (Yellowknife area), similar relationships are expected in As-contaminated 636 lower latitude lakes. However, the impact of As contamination on the arcellinidan assemblage 637 dynamics is likely to be more intense in low latitude lakes due to the warmer water conditions, 638

which in turn increases the likelihood of As post-depositional mobility. More research assessing
the relationship between Arcellinida and mine-induced As contamination in both higher and low
latitude lakes is required to assess the consistency of the Arcellinidan response to As.

642 Faunal changes may provide insight into the nature of prevailing species of As, with highly stressed assemblages likely thriving when the more toxic tri-valent As<sup>3+</sup> is dominant, 643 while less-stressed assemblages likely associated with relatively less toxic penta-valent As<sup>5+</sup> and 644 more inert organic forms. Such insight is significant because accurate geochemical determination 645 of concentrations of particular As species is a metric not captured during typical industry 646 647 standard ICP-MS analysis. However, more research investigating direct and indirect effects of As contamination on Arcellinida and identify different As uptake mechanisms and pathways is 648 required to validate the use of group as a tool for inferring the dominant As species in lakes 649 sediments. Nevertheless, Arcellinida show great potential as a robust reconnaissance tool for 650 identifying impacted lakes where As concentrations may be elevated prior to conducting As 651 speciation geochemical analysis. The findings generated in this study have broad application to 652 other As-impacted lacustrine systems and will provide valuable information to policy makers, 653 654 environmental planners, mine developers, as well as potentially facilitating rehabilitation efforts in lakes impacted by As contamination. 655

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