Shifts in organic matter character and microbial assemblages from glacial headwaters to downstream reaches in the Canadian Rocky Mountains

Hayley F. Drapeau[1,2], Suzanne E. Tank[1], Maria A. Cavaco[2], Jessica A. Serbu[1], Vincent L. St. Louis[1], Maya P. Bhatia[2,3]

[1]Department of Biological Sciences, University of Alberta, Edmonton, T6G 2R3, Canada
[2]Department of Earth and Atmospheric Sciences, University of Alberta, Edmonton, T6G 2R3, Canada
[3]Deceased
Correspondence to: Suzanne Tank (suzanne.tank@ualberta.ca)

Abstract. Climate change is causing mountain glacial systems to warm rapidly, leading to increased water fluxes and concomitant export of glacially-derived sediment and organic matter (OM). Glacial OM represents an aged, but potentially bioavailable carbon pool that is compositionally distinct from OM found in non-glacially sourced waters. Despite this, the composition of riverine OM from glacial headwaters to downstream reaches and its possible role in structuring microbial assemblages have rarely been characterized in the Canadian Rockies. Over three summers (2019–2021) we collected samples before, during, and after glacial ice melt along stream transects ranging 0 – 100 km downstream of glacial termini on the eastern slopes of the Canadian Rocky Mountains. We quantified dissolved and particulate organic carbon (DOC, POC) concentration, and used isotopes (Δ^{14}C-OC, δ^{13}C-OC) and dissolved OM (DOM) absorbance and fluorescence to assess OM age, source, and character. Environmental data were combined with microbial 16S rRNA gene sequencing to assess controls on the composition of streamwater microbial assemblages. From glacial headwaters to downstream reaches, OM showed a clear transition from being aged and protein-like with an apparent microbial source to being relatively younger and humic-like. Indicator microbial species for headwater sites included chemolithoautotrophs and taxa known to harbour adaptations to cold temperatures and nutrient-poor conditions, suggesting some role for glacial seeding of microbial taxa to headwaters of this connected riverine gradient. However, physical and chemical conditions (including water temperature; POC concentration; protein-like DOM; and deuterium excess, an additional indicator of water source) could only significantly explain ~9% of the observed variation in microbial assemblage structure. This finding, paired with the identification of a ubiquitous core microbial assemblage that comprised a small proportion of all identified amplicon sequence variants (ASVs), but was present in large relative abundance at all sites, suggests that mass effects (i.e., whereby high dispersal rates causes homogenization of adjacent communities) largely overcome species sorting to enable a connected microbial assemblage along this strong environmental gradient. Our findings suggest that a loss of novel glacial and microbial inputs with climate change, coupled with catchment terrestrialization, could change OC cycling and microbial assemblage structure across the evolving mountain-to-downstream continuum in glacierized systems.

1 Introduction

Anthropogenic climate change is causing glaciers to retreat at unprecedented rates (Zemp et al., 2015). Rapid glacial retreat has the potential to impact downstream hydrology (Clarke et al., 2015), carbon cycling (Hood et al., 2015; Hood et al., 2020) and microbial assemblage dynamics (Bourquin et al., 2022; Hotaling et al., 2017), as source water contributions to headwater streams undergo fundamental change. In the Canadian Rocky Mountains, glacial meltwater contributions to downstream rivers is presently increasing but is projected to peak by 2040 (Clarke et al., 2015; Pradhananga and Pomeroy, 2022). Over the coming decades, the eventual disappearance of glacial meltwater additions to glacially-fed rivers is predicted to lead to decreased summer flows and water availability, impacting millions of individuals in communities downstream (Anderson and Radić, 2020).

Changing contributions of glacial meltwater to headwater streams could disrupt downstream riverine carbon dynamics, because glaciers provide, and internally cycle, unique sources of organic carbon (OC) (Hood et al., 2015; Musilova et al., 2017; Wadham et al., 2019). As glaciers melt, the formation of supra- and sub-glacial channels can cause meltwater to be routed on top of, through, and/or underneath, glaciers (Nienow et al., 1998). At the beginning of the melt season, water is typically sourced from snowmelt which is inefficiently routed through a distributed hydrological network at the ice–bed interface or along the ice margin, and is characterized by high rock:water contact times (Arendt et al., 2016). As the melt season progresses an efficient subglacial network forms facilitating rapid transit through the subglacial environment (Arendt et al., 2016) or routing of water laterally via marginal streams. This meltwater is then exported into headwater streams where it provides an important seasonal control on stream hydrology via augmentation of summer discharge (Campbell et al., 1995), and biogeochemistry via its unique chemical signature (Milner et al., 2017). The contribution of glacially sourced water to headwater streams enables movement of novel organic matter (OM) from overridden vegetation at the glacier bed (Bhatia et al., 2010), anthropogenic aerosols on the ice surface (Stubbins et al., 2012), and in situ microbial communities (Stibal et al., 2012) to downstream systems. Previous studies have shown that glacially derived dissolved OM (DOM) is aged (Bhatia et al., 2013; Hood et al., 2009; Stubbins et al., 2012), exhibits protein-like fluorescence (Barker et al., 2009; Kellerman et al., 2020), and can serve as a labile substrate for downstream microorganisms (Hood et al., 2009; Singer et al., 2012). In contrast, glacially derived particulate OM (POM, and its OC subset POC), which has been less studied, can be predominantly derived from comminuted rock and sediment, and can persist within rivers (Cui et al., 2016, Hood et al., 2020). During transit through fluvial networks, glacially exported OM mixes with non-glacial terrestrially derived OM, which typically exhibits a humic-like fluorescent signature (McKnight et al., 2001a), is relatively younger (Raymond and Bauer, 2001), and has generally been shown to be less accessible for microbial consumption (D'Andrilli et al., 2015).

River ecosystems are overwhelmingly heterotrophic, with food webs sustained by microbial consumption and mineralization of OM inputs (Bernhardt et al., 2022). OM composition has been found to be an important determinant of microbial assemblage

65   structure (Judd et al., 2006) alongside environmental factors such as light, river flow regimes (Bernhardt et al., 2022; Milner et al., 2017), temperature, and nutrient availability (Elser et al., 2020). Thus, the increase and eventual decline of glacial meltwater inputs to fluvial networks, with their entrained glacial OM, may impact microbial assemblages. In addition, headwater assemblages can shift as a result of the direct loss of novel glacially-sourced microbes (Wilhelm et al., 2013) as glaciers have been found to host unique microbial taxa in both the supraglacial and subglacial environments (Bourquin et al.,

70   2022; Hotaling et al., 2017; Stibal et al., 2012). These taxa are often uniquely adapted to glacial conditions, such as seasonally fluctuating and overall low nutrient concentrations, cold temperatures, and the presence of reduced chemical species in low oxygen subglacial environments, which can select for chemosynthetic metabolisms (Bourquin et al., 2022; Hotaling et al., 2017). Recent work at European, Greenlandic, and North American glaciers indicates that microbes from supraglacial environments are readily entrained in meltwaters, enabling these flow paths to seed downstream microbial communities

75   (Stevens et al., 2022). Overall, if changes in glacial loss drive changes in microbial assemblage composition, this could lead to a loss of unique metabolic pathways (Bourquin et al., 2022), changes in respiratory efflux of CO_2 (e.g., Singer et al., 2012), and food web perturbations, given that OM incorporated into microbial biomass is assimilated by higher trophic level organisms (e.g., Fellman et al., 2015).

80   In this study, we pair measurements of stream OM and microbial assemblage composition along transects from glacial headwaters to downstream reaches, with sampling across three years and encompassing conditions prior to, during, and after the summer glacial melt season. We undertake this work in glacially-fed fluvial networks draining eastward from the Canadian Rocky Mountains, to assess how glacial loss will impact downstream microbial diversity and carbon cycling in this region. The icefields (Wapta and Columbia) that feed these alpine glaciers are rapidly shrinking and are projected to be reduced by

85   80–100% of their 2005 area by 2100 (Clarke et al., 2015). Our objectives in this rapidly changing landscape were three-fold: first, to investigate spatial and temporal variation in stream OM age, source, and character across different hydrological periods in three Rocky Mountain rivers with a distance ranging from 0–100 km downstream from glacial termini; second, to characterize microbial assemblage composition and its variation along this same gradient; and finally, to identify environmental drivers of variation in microbial assemblage structure, with a particular focus on OM composition. We

90   hypothesized that: (1) OM character, apparent source, and age would transition along our 100 km transects; (2) microbial assemblages would similarly vary along our transects; and (3) these shifts would be tied, such that changes in OM and other chemical parameters would structure microbial assemblages from headwaters to downstream reaches.

2 Methods

2.1 Site description

95   Samples were collected from the Sunwapta-Athabasca, North Saskatchewan (NSR) and Bow rivers located within Banff and Jasper National Parks of the Canadian Rocky Mountains (Fig. 1). Each of these rivers are sourced from glacial headwaters with theSunwapta-Athabasca and NSR receiving inputs predominately from the Columbia Icefield, and the Bow River receiving inputs from the Wapta Icefield. At ~216 km^2 (Tennant and Menounos, 2013; Bolch et al., 2010), the Columbia Icefield is the largest icefield in the Canadian Rockies, while the Wapta Icefield is smaller at ~80 km^2 (Ommanney, 2002).

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Along each river we chose three to four sampling sites at which we examined how stream OM and microbial characteristics transitioned with increasing distance from glacial input. Sampling sites ranged from glacial headwaters (0 km downstream of glacial inflow) to 100 km downstream of glacial termini. For certain statistical analyses (predominantly, associated with microbial assemblage structure), sites were binned into four distance ranges, selected a priori based on variation in the upstream

105   catchment between sites: headwater (0.2 km downstream), near (2–6 km downstream), mid (18–50 km downstream) and far (>50 km downstream) (Fig. 1a). These bins were selected a priori, to enable a gradient of sites within each sampled river, and based on variation in the upstream catchment between sites (Table S1). For example, the "headwater" site had no catchment forest cover, "near" sites ranged from 0.3 – 5% forest and 42-55% snow and ice, and "far" sites had greater than 25% forest, and 6-12% snow and ice (Table S1). Sites were restricted to locations within Banff and Jasper National Parks to enable a

110   comparison of study sites that were minimally impacted by direct anthropogenic landscape disturbance. Throughout, we generally use "stream" to refer to specific sites, although we acknowledge that our transects span from glacial headwaters to relatively large riverine reaches.

Samples were collected over a three-year study period (Fig. 1b). Each of the sample sites were visited every three to four weeks

115   throughout the months of May to October during 2019, 2020, and 2021. Opportunistic samples were also collected at a subset of sites in December 2019 and January 2021. This sampling design enabled us to cover the three main hydrological stages in a glacially sourced river: prior to the seasonal glacial ice melt (pre-melt; n=2 visits, 24 samples total), the glacial melt period (melt; n=13 visits, 177 samples), the post-glacial ice melt period (post-melt; n=1 visit; 14 samples), and during winter (n=2 visits; 7 samples). Hydrological periods were determined using the Environment and Climate Change Canada (ECCC) stream

120   discharge hydrographs from Athabasca Glacier (station ID: 07AA007) (ECCC, 2022) (Fig. 1b). We defined the pre-melt stage as the period when monthly average stream discharge at the Athabasca glacier headwater was low (< 1 m^3 s^{-1}; assessed as the inflection point on the Athabasca Glacier hydrograph); a likely indication that glacial meltwater channels were not yet established (Arendt., 2015). The glacial melt period was defined as the period when monthly average discharge was high (>1, but rapidly transitioning to >2.5 m^3 s^{-1}; discharge typically peaked during July–August); at this time glacial meltwater channels

125 were likely well established, with melting glacier ice contributing substantially to headwater stream flow. Finally, the post-melt period was characterized by, once again, low average monthly discharge at the glacier headwater (< 1 m^3 s^{-1}; typically, October and onwards); likely indicating closed glacial channels and cessation of ice melt input. In 2019, all three hydrological stages (pre-melt, melt, post-melt) were sampled; in 2020, sampling was delayed due to the COVID-19 pandemic, and samples were only collected during the melt and post-melt stages; in 2021, samples were collected during the pre-melt and melt stages

130 only. Over our three-year study period, 2019 exhibited the lowest melt-period discharge and 2021 the highest (Fig. 1b).

2.2 Field sampling and field laboratory processing

At each site samples were collected for: DOM absorbance and fluorescence, DOC concentration, POC concentration, particulate and dissolved OC isotopes (δ^{13}C-DOC, δ^{13}C-POC, Δ^{14}C-DOC, Δ^{14}C-POC), and 16S rRNA gene sequencing. To further assess water source and hydrochemical controls on microbial diversity in our analyses, we used data for water

135 isotopes (δ^{18}O-H$_2$O, δ^{2}H-H$_2$O), nutrients (ammonia (NH$_4$$^+$), nitrate and nitrite (NO$_3$$^-$+NO$_2$$^-$), total dissolved nitrogen (TDN), total nitrogen (TN), soluble reactive phosphorous (SRP), total dissolved phosphorus (TDP), total phosphorous (TP), and dissolved silica (SiO$_2$)), major anions and cations (Mg^{2+}, Cl$^-$, Na^{2+}, SO$_4$$^{2-}$, Ca^{2+}, K$^+$), trace metals (Al^{3+}, Ba^{2+}, dCr, dMn, dMo, dNi, Sr^{2+}), temperature, and specific conductance that were collected at each trip (see also Serbu et al. 2024). All sample bottles, except those for anions and nutrients, were soaked overnight in a dilute acid bath (1.2 mol L^{-1} trace metal

140 grade HCl), rinsed at least 3 times with 18.2 MΩ MilliQ water, and rinsed three times with sample water before collection at the field site. Bottles for anion and nutrient analyses were new, following the tested protocol in the CALA (Canadian Association for Laboratory Accreditation) certified laboratory where these analyses were run (University of Alberta Biogeochemical Analytical Services Laboratory; BASL). Trace metal bottles were pre-soaked in a 0.01% Citranox (phosphate free detergent) bath prior to the acid bath step. Following the HCl soak and MilliQ water rinse, all glassware

145 (bottles, EPA vials, and filtration apparatus) were combusted at 560°C for a minimum of 4 h, and high-density polycarbonate (HDPC) bottles for microbial analysis were autoclaved. Glass microfibre filters (grade GF/F, Whatman) were combusted at 460°C for a minimum of 4 h prior to use. High-density polyethylene (HDPE) bottles for ion collection were pre-washed with Citranox prior to the acid cleaning procedure.

150 Samples for DOM absorbance and fluorescence, DOC concentration and δ^{13}C-DOC were sub-sampled from a 250 mL amber glass collection bottle and filtered streamside through 0.45 μm polyethersulfone (PES) filters (Fisherbrand Basix; pre-rinsed with 60 mL Milli-Q and 15 mL river water) into combusted 40 mL amber EPA vials. Nutrient samples were sub-sampled from a 500 mL HDPE collection bottle and filtered streamside using a pre-rinsed sterile plastic syringe and a pre-rinsed 0.45 μm cellulose acetate filter (Sartorius) into polypropylene collection bottles. Water isotope samples were filtered streamside using

155 either 0.45 μm cellulose acetate (in 2019, 2020) or PES (in 2021) filters into 2mL glass (2019, 2020) or 25mL HDPE scintillation (2021) vials filled with no headspace. Major ions and trace metals were sub-sampled from a 250 mL HDPE

collection bottle and filtered streamside into 20 mL scintillation vials using rubber free syringes and a 0.45 µm PES filter. Water temperature and specific conductance were measured onsite using a YSI EXO sonde. Bulk water samples were collected for microbial assemblage analyses in prepared (see above) HDPC bottles, radiocarbon (Δ^{14}C-DOC, Δ^{14}C-POC) in Teflon bottles, and POC concentration and δ^{13}C-POC samples in 4L HDPE plastic bottles. All bulk water chemistry samples (for Δ^{14}C-DOC, POC concentration, Δ^{14}C-POC, and δ^{13}C-POC) were filtered within 24 h of collection off site. Samples for 16S rRNA gene sequencing were generally filtered within 4–12 h of collection, apart from the Bow River samples that were filtered after 24 h of collection. To minimize the effects of differential processing times on assessing microbial assemblage dynamics, Bow River samples were excluded from our microbial analysis but retained in the hydrochemical and carbon analyses.

Two litre bulk water samples collected for 16S rRNA gene sequencing were filtered "until refusal" using a 0.22 µm Sterivex (MilliporeSigma) filter and a peristaltic pump operated at 50–60 mL min^{-1} to minimize cell breakage. A field blank, consisting of a clean 2L polycarbonate bottle filled with milliQ water, left opened at each site, was also processed in the same way to control for any outside contamination during processing. Dissolved radiocarbon samples were filtered through a 0.7 µm GF/F filter using a glass filter tower, collected into 1L amber glass bottles, and acidified to pH 2 using HPLC grade H_3PO_4. Material retained on the 0.7 µm GF/F filter was used for Δ^{14}C-POC analysis. Samples for POC concentration and δ^{13}C-POC were collected on 0.7 µm GF/F filters using plastic filter towers. Samples for DOC concentration and δ^{13}C-DOC were acidified to pH 2 using trace metal grade HCl. Cation and trace metal samples were acidified to pH 2 using trace metal grade HNO_3. Samples for DOC concentration, δ^{13}C-DOC, Δ^{14}C-DOC, DOM absorbance and fluorescence, anions, TDN, TDP and dSi were stored at 4°C; samples for NH_4^+, NO_3^-+NO_2^-, δ^{13}C-POC, Δ^{14}C-POC were stored at -20°C, and sterivex filters were flash frozen in a liquid nitrogen dry shipper and upon return to the laboratory stored at -80°C until analysis.

2.3 Laboratory analyses

2.3.1 DOM absorbance and fluorescence, DOC, POC and OC isotopes

DOM absorbance and fluorescence were analysed using a HORIBA Scientific Aqualog with a 1 cm path length quartz cuvette. Absorbance scans were collected over a 240–800 nm wavelength range in 1 nm increments with a 0.5 s integration time. Excitation emission matrices (EEMs) were constructed over a 230–500 nm excitation wavelength range with an increment of 5 nm and a 5 s integration time, with an emission coverage increment of 2.33 nm with a 230–500 nm emission wavelength range.

DOC samples were analysed using a Shimadzu Total Organic Carbon analyser. Samples with less than 1 mg L^{-1} DOC were analysed on a Shimadzu TOC-V equipped with a high sensitivity catalyst, using a 2 mL injection and five minute sparge time. Samples estimated (from absorbance) to have more than 1 mg L^{-1} DOC were analysed using a Shimadzu TOC-L fitted with a

regular sensitivity catalyst, using a 150 μL injection and five minute sparge time. For analyses on the TOC-V, a five point (0-1 mg L^{-1}) or six point (0-0.5 mg L^{-1}) calibration curve (R^2 > 0.98) was created daily using dilution from a 5 mg L^{-1} stock solution (SCP Science). For analyses on the TOC-L, a five point (0-1 mg L^{-1} or 0-2 mg L^{-1}) calibration curve (R^2 > 0.98) was created through dilution of either 5 mg L^{-1} stock solution (SCP Science) or a 10 mg L^{-1} stock solution (SCP Science). Reference waters were created using dilution of a 5 mg L^{-1} stock solution (SCP Science) or from a 1 mg C L^{-1} caffeine solution. Reference waters and MilliQ blanks were run every ten samples and were within 10% of accepted values. Samples were blank corrected via the subtraction of mean blank concentrations of MilliQ samples run prior to ten sample groups to account for instrument drift.

POC, δ^{13}C-POC and Δ^{14}C-POC samples were subjected to a heated acid fumigation following procedures outlined in Whiteside et al. (2011). Briefly, this involved heating the filters at 60°C for 24 h in a desiccator with 20 mL of concentrated trace metal grade HCl, then neutralizing the filters at room temperature for 24 h in a desiccator with NaOH pellets. Samples for POC concentration and δ^{13}C-POC were packaged into tin capsules before being measured using an Elementar VarioEL Cube Elemental Analyser a DeltaPlus Advantage isotope ratio mass spectrometer with a ConFlo III interface (δ^{13}C-POC only) at the Environmental Isotope Laboratory at the University of Waterloo (in 2019) or the Jan Veizer Stable Isotope Laboratory (Ottawa, ON, Canada, in 2020 and 2021). δ^{13}C-DOC samples were analysed using the wet oxidation method on an OI Analytical Aurora model 1030 wet TOC analyzer connected to an Isotope Ratio Mass Spectrometer (IRMS) at the Jan Veizer Stable Isotope Laboratory. Δ^{14}C-POC samples were processed via organic combustion and Δ^{14}C-DOC samples were processed using UV-oxidation, prior to graphitization and accelerator mass spectrometry analysis at the André E. Lalonde AMS Laboratory at the University of Ottawa (in 2019) or the National Ocean Sciences Accelerator Mass Spectrometry laboratory (NOSAMS; WHOI, Woods Hole, MA, USA, in 2020 and 2021; Batch numbers = 67580, 67380, 67069, 66828).

2.3.2 Hydrochemical samples

Nutrient (NH$_4^+$, NO$_3^-$+NO$_2^-$, TDN, TDP, TN, TP, dSi, SRP), trace metal (Al^{3+}, Ba^{2+}, dCr, dMn, dMo, dNi, Sr^{2+}), and ion (Mg^{2+}, Cl$^-$, Na^{2+}, SO$_4^{2-}$, Ca^{2+}, K$^+$) samples were analysed at the CALA accredited BASL facility at the University of Alberta (Edmonton, AB, Canada) (see Table S2 for detection limits and method details). Nutrient samples were analysed using flow injection analysis on a Lachat QuikChem 8500 FIA automated ion analyser and anions were analysed via ion chromatography on a Dionex DX600. Cations and trace metals were analysed using Inductively Coupled Plasma-Mass Spectrometry, on a Thermo Scientific iCAP Q (2019) and Agilent 7900 (2020-21). Water isotopes (δ^{18}O-H$_2$O and δ^2H-H$_2$O) were analysed using a Picarro L2130 isotope and gas concentration analyser calibrated using standard references obtained from ice core water (USGS46) and Lake Louise drinking water (USGS47) (United States Geological Survey). Reference waters and MilliQ were run every 20 samples and were within 20% of accepted values. Sample values were calculated from an average of three injections where the standard deviation of δ^{18}O-H$_2$O was less than 0.2 and the standard deviation for δ^2H-H$_2$O was less than

220   1; the first five injections were typically excluded due to memory effects. $\delta^{18}O$-H_2O and δ^2H-H_2O were used to calculate deuterium excess, which we use as an indicator of water source given its known increase with elevation, and therefore water sourced from glaciers and high elevation snow (Bershaw et al., 2019; Bershaw et al., 2020; Boral et al., 2019)

2.3.3 Lab and bioinformatic processing of microbial samples

Bulk genomic DNA collected on the Sterivex filter was extracted using a DNeasy PowerWater Sterivex kit (Qiagen) according
225   to the manufacturer's instructions (Qiagen, 2019) with an amendment of a 1 h at 72°C for the initial incubation time, rather than 90°C for 5 mins, to increase DNA yield. Extracted samples, including field blank samples and a separate negative control containing PCR grade water as template, were amplified using the 515F (5'GTGYCAGCMGCCGCGGTAA'3) and 926R (5'CCGYCAATTYMTTTRAGTTT'3) primers (Parada et al., 2016) targeting the V4–V5 hypervariable region of the 16S rRNA gene with the following protocol: 3 min initial denaturation at 98°C, 35 cycles of: 30 s denaturation, 30 s primer
230   annealing at ~ 60°C and 30 s extension at 72°C; and finally, 10 mins of final extension at 72°C. Polymerase Chain Reaction (PCR) products were visualized on a 1.5% agarose gel and those samples showing product on the gel were subsequently purified using Nucleomag beads (ThermoFisher Scientific) at a 0.8 ratio of beads: sample. We only purified and pooled samples if our negative control did not amplify (no band on the gel), demonstrating no introduction of exogenous contamination during library preparation steps. Unique indexes were then added to each sample using i7 and i5 adapters (Illumina) to construct
235   the subsequent library. All amplicon and barcoded products from each respective year of sampling (2019, 2020, 2021) were verified on a 1.5% agarose gel and those that could be successfully visualized were then pooled (12.4 ng μL^{-1}, 20.17 ng μL^{-1}, 14.3 ng μL^{-1}). The final quality of each pool was determined on an Agilent 2100 Bioanalyzer at the Molecular Biology Service Unit (University of Alberta, Edmonton, AB, Canada) using a high sensitivity DNA assay prior to submitting the library for 16S rRNA gene sequencing. The final prepared 2–4 nM libraries, containing up to 50% PhiX Control v3 (Illumina, Canada
240   Inc., NB, Canada) were sequenced on an Illumina MiSeq (Illumina Inc., CA, USA) using a 2 × 250 cycle MiSeq Reagent Kit v3 submitted at the Molecular Biological Services Unit (Univ. of Alberta) in 2019 and The Applied Genomics Core (Univ. of Alberta) in 2020 and 2021. Measured genomic DNA extracted from filters ranged from below detection (<0.5 ng μL^{-1}) to 20.8 ng μL^{-1}; when DNA concentrations quantified below 10 ng μL^{-1} on the Qubit following DNA extraction, we added 5uL of template DNA (default is 2.5uL), along with bovine serum albumin BSA to maximize DNA available for PCR, adjusting for
245   total water added to the reaction.

Sequence data was demultiplexed using MiSeq Reporter software (version 2.5.0.5) and Miseq Local Run Manager GenerateFastQ Analysis Module 3.0. The assembled data was then processed using the Quantitative Insights into Microbial Ecology (QIIME2) pipeline (Boylen et al., 2020, version 2021.11). Sequences were clustered into amplicon sequence variants
250   (ASVs) with chimeric sequences and singletons removed using DADA2 (Callahan et al., 2019). All representative sequences were classified with the SILVA v138 taxonomic database (Quast et al., 2013, Yilmaz et al., 2014), at the default 0.8 similarity

cut-off. ASV sequences that were identified as eukaryotes or chloroplasts were removed. Samples were decontaminated by removing ASVs present in a sequenced field blank using the prevalence method with a threshold of 0.5 (Karstens et al., 2019; Parada et al., 2016b). No positive controls were included in our sequencing run, because we have spiked in PhiX quite heavily
255 into our pool (~50%). A large proportion of PhiX was added to increase the diversity of our oligotrophic pool, such that the reads obtained will generally correspond with the most abundant microbial taxa in the sample, as a function of this and the read coverage of the MiSeq. Following inspection of rarefaction curves, samples with less than 5,000 reads were excluded from further analysis because they had not reached a plateau, indicating that sequencing might not have captured a good representation of species.

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2.4 Data analysis

2.4.1 DOM absorbance and fluorescence calculations and PARAFAC analysis

EEMs were corrected for inner filter effects, Raman normalized, blank corrected and had Rayleigh and Raman scatter bands removed prior to parallel factor (PARAFAC) analysis (Murphy et al., 2013). PARAFAC analysis was performed in Matlab
265 (Version 9.12.0) using the drEEM toolbox (Version 0.6.5) (Murphy et al., 2013). PARAFAC models of up to seven components were assessed and samples with high leverage were removed. A four-component model was validated using split-half analysis and was ultimately selected based on a low sum of square error and visual confirmation that only random noise remained in the residuals. Absorbance scans were utilized to calculate specific UV absorbance at 254 nm ($SUVA_{254}$; a measure of DOM aromaticity) (Weishaar et al., 2003) and the spectral slope coefficient between 275 and 295 nm ($S_{275-295}$; a measure
270 of DOM molecular weight) (Helms et al., 2008). Corrected fluorescence data were utilized to calculate the fluorescence index (FI) (McKnight et al., 2001b), humification index (HIX) and biological index (BIX) (Huguet et al., 2008), and the proportional contribution of common fluorescence peaks (peaks A, B, C, M, T; (Coble, 1996)). Further description of absorbance and fluorescence-based metrics is provided in Table S3.

2.4.2 Geochemistry

275 We ran linear mixed-effects models with distance (as km downstream) and season (comparing winter, pre-melt, melt, and post-melt) as fixed effects, and river and year as random effects, to explore variation in geochemical parameters (OC concentration, OM character, OC isotopic composition, and water isotopes). We ran a Type II ANOVA on model outputs to assess significant main effects of distance and season, and their interaction. When the effect of season was significant, the analysis was followed by Tukey-adjusted contrasts using estimated marginal means. A linear model was used to assess the relationship between
280 Δ^{14}C-DOC and protein-like PARAFAC components. We use deuterium excess as proxy for glacial meltwater because it is known to increase with elevation (Bershaw et al., 2019) and is thus typically higher in glacial ice (Souchez et al., 1983) and

water contributions from high elevation snow (Bershaw et al., 2020) when compared to lower elevation (downstream) water sources. To explore how DOM parameters varied across hydrologic seasons and with distance downstream a principal component analysis (PCA) was performed. Deuterium excess was fitted onto the ordinated PCA as a passive overlay (i.e., fit post-ordination) to not affect the analytical output, using envfit (Oksanen, 2007).

2.4.3 Microbial analyses

To enable contrasts between sites most strongly influenced by glaciers (i.e., the headwater and near sites; 42 – 55% snow and ice coverage in catchment; Table S1) and those further downstream (here "far" sites, with 6 – 12% snow and ice coverage in catchment), the mid distance range sites were excluded from statistical analysis of the microbial samples. Alpha diversity was calculated using the Shannon index (Hill, 1973) and beta diversity was visualized using non-parametric multi-dimensional scaling (NMDS), using a Bray Curtis distance matrix created from Hellinger-transformed ASV abundance data (Ramette, 2007; Legendre and Gallagher, 2001). Significant differences between a priori determined clusters on the NMDS were assessed using permutational multivariate ANOVAs (perMANOVA), set at 999 permutations and using the Bray Curtis distance matrix. The clusters used for comparison included distance (headwater, mid, and far), year, and river. We additionally constructed a Venn Diagram in R using packages from *dplyr* and *vegan* to explore ASV overlap between headwater, mid, and far sites, and refer to ASVs present across all distance bins as a "core" community, below. To explore the possible role of environmental control on microbial assemblage structure, we performed a backward selection distance based redundancy analysis (RDA) on Hellinger-transformed ASV abundance data and non-correlated normally scaled environmental parameters. To do this, we first considered a broad suite of environmental parameters in a non-stepped RDA including temperature, pH, specific conductance, turbidity, DOC concentration and DOM composition (% humic-like [see Section 3.1.2], FI, BIX, HIX, peak A, peak B, peak C, peak M, peak T, $S_{275-295}$, $SUVA_{254}$), nutrients (TP, TDN, dSi), major ions, and trace metals , and used a variance inflation factor (VIF) of 10 as a benchmark to omit highly correlated parameters to avoid an over-parameterized model. The backward selection RDA was run on this streamlined set of parameters, and percent variance of the model output was adjusted following the approach of Peres-Neto et al. (2006). An indicator species analysis (ISA) was performed on raw (non-transformed) ASV abundance data (Dufrêne and Legendre, 1997). Indicator values were compared to the Spearman rank correlation coefficient for the relationship between ASV abundance and deuterium excess, to explore how the presence of indicator species varied across gradients of water source. We re-ran NMDS and perMANOVA analyses using rarefied, rather than Hellinger-transformed, datasets, to confirm the suitability of the Hellinger transformation for our analyses.

All data were analysed within the R statistical language, using *vegan* (Oksanen, 2007), *lme4* (Bates et al. 2015), *car* (Fox and Weisberg 2019), *emmeans* (Lenth, 2024), *stardom* (Pucher et al., 2019), *decontam* (Davis et al., 2018), *indicspecies* (Cáceres and Legendre, 2009), and base packages (R Core Team, 2022). Variance adjustment within the RDA used the function

"RsquareAdj", and the perMANOVA was performed using the function "adonis", both in *vegan*. Data visualization utilized the package *ggplot2* (Wickham, 2016).

3 Results

3.1 Spatial and temporal variation in stream organic matter characteristics

3.1.1 Water isotopes, DOC concentration and DOM composition

In a model with sample year and river as random effects, deuterium excess declined with distance downstream (x^2_1=58.75; p<0.001) and varied seasonally (x^2_3=58.75; p<0.001) with values greater during melt than pre-melt seasons (p<0.001) (Fig. S1b, Tables S4, S5). $\delta^{18}O$-H_2O also varied seasonally (x^2_3=30.81; p<0.001), becoming more enriched from pre-melt to melt (p=0.001), and melt to post-melt seasons (p < 0.023) (Fig. S1a, Tables S4, S5). A two-way ANOVA to assess differences in water source between the *a priori* selected distance bins showed deuterium excess to vary significantly between bins ($F_{3,186}$ = 44.74, p<0.001), with Tukey post-hoc comparisons showing headwater and near sites to be similar to each other (p=0.15), but all other bins to be significantly different from one another (p<0.001).

DOC concentrations were universally low closest to the glacier (0.09–0.76 mg L^{-1}) and increased with distance downstream (x^2_1=44.87; p<0.001) (Fig. 2a, Table S4). Concentrations also varied seasonally (x^2_3=58.75; p<0.001), with pre-melt concentrations greater than those during melt (p<0.001), post-melt (p<0.001) and winter (p<0.001). However, a significant interaction suggests that these effects were not necessarily consistent (x^2_3=17.06; p<0.001) (Fig. 2a, Tables S4, S5). Across all sites and dates, POC concentrations ranged from 0.04 mg L^{-1} to 2.8 mg L^{-1}. POC was typically the dominant component of the total OC (TOC) pool at glacial headwaters (POC:TOC = 0.32–0.89; median value > 0.5 for all but pre-melt 2021; data not shown), and showed a proportional decline with distance downstream (x^2_1=18.31; p<0.001) (Fig. 2e; Table S4).

From the suite of PARAFAC models considered, a four component model was selected (Fig. S2). Using the OpenFluor database (Murphy et al., 2014), components 1 (C1) and 2 (C2) were identified as humic-like components, component 3 (C3) was identified as a tryptophan-like (protein-like) component, and component 4 (C4) was identified as a tyrosine-like (protein-like) component (Table S6). The proportion of protein- (sum of C3 and C4) and humic- (sum of C1 and C2) like components transitioned along transects, with humic-like DOM increasing proportionately with distance downstream, while protein-like DOM proportionally decreased (x^2_1=43.96; p<0.001) (Fig. 2b–c, Table S4). Overall, the proportion of humic-like DOM was greater in the pre-melt than melt (p=0.005) and post-melt (p<0.001) periods (Table S5).

3.1.2 Dissolved and particulate OC isotopes

δ^{13}C-DOC ranged from -35 to -20.2‰ (Figs. 3 and 4), showing a marginal enrichment with distance downstream (χ^2_1=3.57; p=0.059; Table S4), and significant variation across season (χ^2_3=9.68; p=0.022; Table S4), with depleted values during the pre-melt (p=0.050) and melt (p=0.035) seasons when compared to the post-melt season (Table S5). δ^{13}C-POC ranged from -38.8 to -20.8‰ (Fig. 4b), and increased with distance downstream (χ^2_1=6.23; p=0.013), but with a significant interaction between distance and season (χ^2_3=10.70; p=0.013; Table S4). Δ^{14}C-DOC became more enriched with increasing distance downstream (χ^2_1=6.03; p=0.014) but with high variability at the headwater site, particularly during the melt season (-554‰ to +7‰; n = 8) (Figs. 4a and 5a). Δ^{14}C-DOC also varied (χ^2_3=23.70; p=0<001), with values most depleted during winter compared to other seasons (p=0.020 to 0.001 for various comparisons) (Fig. 5a, Tables S4, S5). In addition, Δ^{14}C-DOC showed a modest negative association with protein-like DOM (linear model R^2_{adj} = 0.17, $F_{1,39}$ = 8.212, p = 0.007) (Fig. 5b). Similar to Δ^{14}C-DOC, Δ^{14}C-POC exhibited a large range in values, from -550‰ to -108‰ (Fig. 4b). However, in contrast to Δ^{14}C-DOC the large variation in Δ^{14}C-POC values was not associated with significant seasonal or spatial differences between samples (Fig. 4, Table S4).

3.1.3 An integrative assessment of DOM composition across sites

A principal component analysis (PCA) exploring DOM composition across sites identified eight principal components with 64% of the observed variation being explained by the first two principal component axes (Fig. 6). Principal component 1 (PC1) explained 49% of sample variance, with humic-like PARAFAC components, DOC concentration and HIX (humification) loading positively on this axis, and protein-like PARAFAC components, BIX (biological origin), FI (microbial origin), and $S_{275-295}$ (declining molecular weight) loading negatively. Sites generally plotted along PC1 according to their distance from glacier termini, with headwater and near sites more frequently having negative PC1 scores, and far sites having more positive PC1 scores (Fig. 6). PC2 explained 14% of sample variance and was positively associated with δ^{13}C-DOC and negatively associated with SUVA$_{254}$ (aromaticity). A passive overlay of deuterium excess was negatively associated with PC1, and positively associated with PC2 (Fig. 6).

3.2 Microbial assemblage characterization and its controls over space and time

Of our target sample set, we successfully obtained sequences from 72 samples, with ASV richness ranging from 81 to 9,634, and read counts ranging from 5,066 to 409,740; inspection of Rarefaction curves confirms that a plateau were reached for all included samples (i.e., those with greater than 5,000 reads; Fig. S3). Generally, microbial assemblages shared similar dominant phyla and orders across sites and seasons with the top ten most abundant phyla across all samples being: Acidobacteriota, Actinobacteriota, Bacteroidota, Bdellovibrionota, Chloroflexi, Cyanobacteria, Firmicutes, Plantomycetota, Proteobacteria, and Verrucomicrobiota (Fig. S4). At near and far sites in the Sunwapta-Athabasca and North Saskatchewan rivers, these ten phyla

collectively described between 80–88% of the resolved microbial assemblage composition (Fig. S4). Major microbial orders were dominated by Burkholderiales, Chitinophagales, Cytophagales, Flavobacteriales, Frankiales, Microtrichales, Sphingobacteriales, Synechococcales and Vicinamibacterales) (Fig. S5), with microorganisms related to Burkholderiales appearing most dominant, particularly in August at sites closer to the headwaters. Cumulatively, the top 10 orders accounted for 56 % of the total microbial assemblage composition. To explore taxonomic variation at a finer scale, we investigated shared ASVs across distance ranges (Fig 7a). This assessment revealed a core microbial assemblage, shared among all our samples, comprised of 1409 ASVs total, with the top 10 most abundant ASVs belonging to families Comamonadaceae, Cyanobiaceae, Gallionellaceae, Ilumatobacteraceae, Sporichthyaceae, Alcaligenaceae, Methylophilaceae and Flavobacteriaceae. While this core assemblage only represented 3.4% of all identified ASVs (Fig. 7a), it made up a large fraction of relative abundance at each site (between 25–80%; Fig. 7b), with a general decreased prevalence of the core assemblage at the far sites, when compared to headwaters and near. Accordingly, microbial assemblage diversity was highest at the far sites during all seasons (Fig. S5), and a comparison of microbial diversity between samples (beta diversity) showed that assemblage composition shifted with movement from headwater and near to far sites ($R^2 = 0.09$, $p < 0.001$) (Fig. 8, Table S7). Communities were also distinct across rivers ($R^2 = 0.08$, $p < 0.001$; comparing the Athabasca, Sunwapta, and North Saskatchewan) and different sampling years ($R^2 = 0.06$, $p < 0.001$; comparing 2019, 2020, 2021) with each river and year being significantly different from one other ($p<0.01$) (Fig. S7, Table S7).

We used a distance-based RDA to assess how microbial assemblages varied across environmental gradients, to better understand the role of environment in structuring assemblages at each of our sites. Removal of environmental variables with VIF>10 reduced the initially-considered factors described in the Methods (n=32) to: % protein-like DOM, water temperature, DOC concentration, POC concentration, δ^{13}C-DOC, δ^{18}O-H$_2$O, deuterium excess, $S_{275-295}$, SUVA$_{254}$, TN, TP, and specific conductance (n=12). Following backward selection, the final RDA described 9.1% (adjusted; see Methods) of microbial assemblage structure, with % protein-like DOM, water temperature, specific conductance, POC and deuterium excess identified as significant predictor variables ($p < 0.001$) (Fig. 9). RDA axis 1 (RDA1) described a gradient of apparent high elevation (e.g., glacial ice, snow) water source inputs, with deuterium excess and protein-like DOM negatively correlated, and specific conductance and water temperature positively correlated with this axis (Fig. 9). RDA axis 2 (RDA2) showed water temperature loading negatively on this axis and POC loading positively (Fig. 9). Headwater sites plotted negatively on RDA1 and positively on RDA2, whereas far sites tended to plot positively on RDA1. Overall, headwater microbial communities were associated with relatively higher proportions of protein-like DOM, deuterium excess, and POC but lower specific conductance and water temperature compared to communities at the far sites (Fig. 9).

3.2.1 Indicator species analysis

An ISA resolved three strong (indicator value (IV) > 0.8, p < 0.05) indicator species for headwater sites and two strong indicator species for far sites. There were no strong indicator species for near sites at this IV threshold. Indicator species for the headwater sites were identified as a species from the genus *Cryobacterium* (IV = 0.89), a species from family Beggiatoaceae (IV = 0.82), and a species from family Microbacteriaceae (IV = 0.87). Far indicators were a species from family Sporichthyaceae (IV = 0.83), and a species in the genus *Cyanobium_PCC-6307* (IV = 0.83). Correlations between ASV abundance and deuterium excess revealed that taxa indicative of headwater environments were strongly positively correlated with deuterium excess (positive ρ values), while taxa indicative of far environments showed a negative correlation (negative ρ values; Fig. 10).

4 Discussion

4.1 Changing OM quantity and character driven by increased soil development downstream

Stream OM originates from two primary sources: 1) in situ primary production of OM from autotrophic microbes or macrophytes (autochthonous carbon); or 2) OM transported into the stream system from outside sources (allochthonous carbon) such as nearby vegetation and soils (Hinton et al., 1997, 1998; Tanentzap et al., 2014; Table S8). In glacially-fed streams, glacially sourced OM also contributes to the allochthonous carbon pool (Hood et al., 2015a). However, glaciers are associated with a series of variable OM sources, as a result of in situ microbial metabolism on the glacier surface (supraglacial) and at the glacier bed (subglacial) (Hotaling et al., 2017; Stibal et al., 2012), as well as from windblown deposition on the snow and ice surface and comminuted sediments and bedrock at the glacier bed (Hood et al., 2020; Stubbins et al., 2012). Thus, the changing water source contributions to streams is a critical determinant of fluvial OM composition, both in alpine environments and elsewhere (Fasching et al., 2016).

This study identified a shift in OC quantity and OM character with increasing distance downstream from glaciers and thus declining relative contributions of glacial meltwater and associated OM to streamflow. Low DOC concentrations in our mountain headwater streams, which drain catchments lacking developed soils and where dilute glacial meltwaters are the primary source of allochthonous carbon, are in agreement with previous work from the Alps (Singer et al., 2012), Tibetan Plateau (Zhang et al., 2018; Zhou et al., 2019), Canadian Rockies (Lafrenière et al., 2004), and the Greenland Ice sheet (Kellerman et al., 2020). In contrast, the large relative contribution of POC to the headwater TOC pool is consistent with increased erosional processes in high slope glacial margin environments, export of glacially derived sediments (Hood et al., 2020), and higher rates of particle re-suspension in these high gradient, turbulent, reaches (Marcus et al., 1992). At the headwaters, the protein-like character of the DOM pool suggests dominant contributions from microbial production or

reworking of DOM, either on the glacier surface or at the bed, as has been found in many other glacierized regions (Zhou et al., 2019; Dubnick et al., 2010; Hood et al., 2009; Stubbins et al., 2012; Pain et al., 2020; Singer et al., 2012). Increasing DOC concentrations and a shift from protein-like to humic-like DOM with increasing distance downstream supports increasing contributions of allochthonous carbon from surrounding vegetation and more developed soils (see also, Zhou et al., 2019). The

435   conclusion that headwater sites receive a majority of their OM directly from the glacier, and downstream locations gain additional OM contributions derived from catchment soils, is supported by declines in deuterium excess with movement downstream (Fig. S1, Table S4), and thus an inferred switch to increasing water contributions of rainfall and lower-elevation sourced precipitation (Bershaw and Lechler, 2019; Bershaw et al., 2020).

4.2 Stream POC sourced from plant carbon with potential additions from glacier microbes

440   Compared to the DOC pool, the POC pool displayed less isotopic variation, pointing to a more consistent POC source to streams. Across seasons and with increasing distance downstream, the POC pool was consistently aged (Δ^{14}C-POC = -550‰ to -108‰). At the headwaters, this depletion likely indicates sourcing from glacial export of overridden vegetation (Bhatia et al., 2013), or fossil fuel products (Stubbins et al., 2012) and the heterotopic consumption of fossil fuels by cryoconite microbes (Margesin et al., 2002). Further downstream, radiocarbon depleted POC could be sourced from the addition of water from

445   aged soil margins, or glacially exported aged POM that has persisted through the stream network (Hood et al., 2020). However, save for the headwater site during the pre-melt period, δ^{13}C-POC indicated POM primarily of terrestrial origin (δ^{13}C‰ = -26 to -28‰), with occasional slight depletions at headwater and near sites (δ^{13}C‰ = -29 to -30‰) pointing to possible contribution from subglacial chemosynthetic biofilms (Table S8). Indeed, if some of the aged POM is sourced from subglacial microbes, this fraction of POM may be more accessible to recipient food webs than aged terrestrial POM (Brett et al., 2017), though

450   relative contributions through our transects is likely small. The pre-melt headwater sample was uniquely δ^{13}C-enriched compared to other POM samples, which may suggest microbial re-working of depleted C sources as a more dominant POM source at this time.

4.3 Seasonal patterns in DOM from headwaters to downstream reaches

455   *Glacial headwaters*

Over the past two decades, studies from polar and alpine regions across the globe have led to the characterization of glacial DOM as universally dilute, of microbial origin, and old (i.e. radiocarbon depleted) (Bhatia et al., 2013; Hood et al., 2009). Similarly, we found that DOM was consistently most protein-like in headwater reaches (Table S4), but with compositional variation during the melt season (Fig. 2). Our results contrast somewhat with other studies where melt-season shifts in glacial

460   stream DOM character from relatively more humic-like to more protein-like have been associated with efficient subglacial channels during the peak melt season (Kellerman et al., 2020). For example, DOM fluorescence in the outflow draining Leverett Glacier, a large land-terminating glacier of the Greenland Ice Sheet, was relatively more humic-like in the early

season, when waters draining subglacial flow paths access terrestrial material from previously overridden soils and bedrock dominated stream contributions (Kellerman et al., 2020). In comparison, during peak melt flow, when supraglacial inputs

465   dominated, Leverett Glacier outflow DOM had a distinctly more protein-like fluorescence signature (Kellerman et al., 2020). Similarly, protein-like fluorescence increased from the pre-melt to the melt period in waters draining the Mendenhall Glacier, Alaska (Spencer et al. 2014). At these Rocky Mountain glacier sites, the variability in DOM source during the melt season may point to temporally patchy interactions with the overridden vegetation and paleosols known to be prevalent in this region (Luckman et al. 2020).

470   During the pre-melt period, water at the Athabasca glacier terminus is predominately sourced from the distributed subglacial drainage network (Arendt, 2015), with DOC that was aged (Δ^{14}C-DOC = -175‰), with δ^{13}C-DOC of -25.5‰ (Fig. 4). These carbon isotopic signatures, in conjunction with the consistent protein-like DOM fluorescence, suggest microbial reworking of previously overridden vegetation and soils at the glacier bed (Bhatia et al., 2013; Dubnick et al., 2010; see also Table S8). During the melt season, headwater δ^{13}C-DOC shows substantial variation, in line with the DOM compositional variation

475   described above. Four of six Δ^{14}C-DOC samples from this period were relatively aged (Fig. 4); together with the relatively protein-like composition of the DOM pool, this isotopic variation is consistent with a range of sources including: a) chemolithoautotrophy tied to microbial heterotrophic consumption of ancient vegetation and soil at the glacier bed (Hood et al., 2009); b) microbial re-working of over-ridden vegetation; or c) microbial incorporation of radiocarbon-dead fossil fuel combustion products deposited on the glacier surface (McCrimmon et al., 2018). Previous work in alpine glacial systems in

480   both Alaska (Stubbins et al., 2012) and Tibet (Spencer et al., 2014) have found the latter to substantially contribute to supraglacial DOC pools during peak melt periods. In a subset of melt season samples, however, headwater Δ^{14}C-DOC was enriched (Fig. 4). This could be indicative of contributions from recent photosynthetic supraglacial microbial communities (Kellerman et al., 2021; Smith et al., 2017; Stibal et al., 2012) or wildfire-derived soot present on the glacier surface (Aubry-Wake et al., 2022; Nizam et al., 2020). Overall, the range of DOM composition, δ^{13}C-DOC, and Δ^{14}C-DOC points to striking

485   heterogeneity of water sources draining to the glacial outflow at this site.

Transit downstream

With transit downstream, DOM becomes overall more humic like (Table S4), but with seasonal variability in character. During the pre-melt period, inputs of enriched Δ^{14}C-DOC were coupled with δ^{13}C-DOC values consistent with terrestrial OM inputs

490   (-25‰ to -27‰) at the near and mid sites, likely indicative of snowpack melt draining surficial soils, leading to pulses of concentrated modern, terrestrial-origin DOM delivery to streams (Fig. 4). This flushing of concentrated DOM from soils was the likely cause of the spike in DOC concentration observed during the pre-melt season at downstream sites in 2019 (Fig. 2), when sampling captured the spring freshet. This process has been observed previously at Bow River from 1998–2000, when the highest DOC concentrations were found to occur during the initial snowpack melt and, as in this study, were followed by

495   a decline in river DOC concentrations (Lafrenière and Sharp, 2004). These isotopic signatures were also tied to higher proportional inputs of humic-like DOM (Fig. 4).

During the melt season, a proportional increase in protein-like fluorescence at the near, mid, and far sites was coupled with DOM pools at near and mid sites switching from being consistent with allochthonous terrestrial inputs (δ^{13}C-DOC = -25 ‰ to -29 ‰) to showing a clear autochthonous signature reflective of chemo- or photosynthetic OM production (δ^{13}C-DOC ranging from -35‰ to -20‰; Figs. 3 and 4). Near and mid sites were also characterized by DOM that was often Δ^{14}C-DOC depleted. This trend may reflect increasing in situ production by microbial photoautotrophs, as evidenced by the presence of cyanobacteria DNA in the microbial amplicon libraries from these sites (Fig. S3), or microbial re-working of humic-like DOM, either within the stream or in catchment soils (Fig. 4). Aged carbon downstream likely represents the mixing of either: a) ancient glacially exported material from the headwaters persisting downstream or b) additions from deeper, older soils (Shi et al., 2020). During the post-melt period, stream water at the near and mid distances was either aged (Δ^{14}C-DOC = -630‰, -400‰) with a δ^{13}C-DOC terrestrial signature (-28‰, -29‰), consistent with OM contributions from deeper soils when discharge is declining, or slightly more modern (Δ^{14}C-DOC = -193‰, -110‰) and δ^{13}C-enriched (-22‰), consistent with incorporation of autochthonous primary production into the DOM pool (Fig. 4). Future studies could consider a mixing model to add more specificity to these assessments, via the inclusion of additional tracers to resolve the multiple endmembers of this system.

4.4 Implications of inferred composition for OM lability

The transition from recently produced, low molecular weight, protein-like DOM to high molecular weight, humic-like DOM with movement downstream has implications for DOM lability. Protein-like DOM is associated with free amino acids and proteinaceous compounds which are generally considered to be more available for heterotrophic microbial consumption (Coble, 1996), while factors such as decreasing molecular weight ($S_{275-295}$) (Fig. 6, Table S3) are also associated with increasing OM lability (Moran and Zepp, 1997; Patriarca et al., 2021). In previous studies of glacial DOM, paired high resolution mass spectrometry and UV-Vis spectrometry have shown that protein-like fluorescence is associated with unsaturated aliphatic compounds, identified as being peptide- and lipid-like (Kellerman et al., 2020, 2021; Stubbins et al., 2012; Zhou et al., 2019), which have been found to be a common contributor to glacier DOM pools in many regions (Singer et al., 2012; Bhatia et al., 2013; Kellerman et al., 2021; Stubbins et al., 2012; Zhou et al., 2019). In contrast, humic-like DOM is associated with complex material (e.g., increased lignin content; Mann et al., 2016; Zhou et al., 2019) and thus is generally considered to be a more recalcitrant fraction of DOM. This shift in lability has important implications for stream ecosystems, because microbial consumption of DOM can provide an energetic transfer to higher trophic levels (e.g., Fellman et al., 2015), with labile DOM being linked to increased microbial productivity (Pontiller et al., 2020). Labile OM released at the headwater site could also initiate priming of the OM pool, resulting in increased microbial consumption of recalcitrant soil-derived OM (Bingeman et al., 1953; Guenet et al., 2010). Notably, the importance of priming has yet to be assessed in glacially fed streams (Bengtsson et al., 2018). However, despite the shift in apparent bulk DOM lability with movement downstream, one additional

consideration is the low overall DOC concentration at headwater sites. Although downstream sites had lower proportions of
530 protein-like DOM, these sites had higher DOC concentrations overall. A rough calculation to compare between sites yields an estimated mean 0.28 mg L^{-1} of protein-like DOM at downstream sites, compared to ~0.21 mg L^{-1} at the headwaters (using the mean DOC and % protein-like DOM for each distance bin). Therefore, although DOM may have been more accessible to microbes on a proportional basis in the headwaters, overall OM processing may still have been higher downstream.

4.5 Controls on microbial assemblage structure with transit downstream

535 Microbial communities and OM pools are inherently linked since microbes contribute both to the creation of OM and its consumption (Kujawinski, 2011). To explore possible coupling between microbial assemblages and stream OM, we investigated how assemblages shifted with increasing distance from glaciers and coupled changes in OM source, OM character, and other physicochemical characteristics. Across all sites and seasons, 10 phyla were found to account for 80–88% of assemblage composition (Fig. S3). While some of these phyla (Actinobacteria, Acidobacteriota, Cyanobacteria, Bacteroidota,
540 and Proteobacteria) are ubiquitous in freshwater (Tamames et al., 2010), many (Proteobacteria, Bacteroidota, Acidobacteriota, Actinobacteriota, Chloroflexi, Cyanobacteria and Firmicutes, Plantomycetota) are also commonly identified as being dominant in glacial ecosystems and in glacially-fed streams (Boetius et al., 2015; Bourquin et al., 2022; Hotaling et al., 2019). These glacially associated phyla are known to include taxa that are adapted to the environmental conditions typical of glacial environments, such as oligotrophy and cold temperatures (Bourquin et al., 2022). At a finer taxonomic resolution, the orders
545 Cytophagales, Chitinophagales, Vicinamibacterales, Burkholderiales, and Flavobacteriales are also commonly found in glacially fed waters (Kohler et al., 2020, Brandani et al., 2023, Guo et al., 2024). Burlholderiales and Flavobacteriales are known for their contributions to organic matter cycling in oligotrophic systems such as these (Guo et al., 2024). Interestingly, the order Sphingomonadales has also been shown to be significantly correlated with suspended sediment concentrations released from the Greenland Ice Sheet (Dubnick et al., 2017). Although our system is situated within the Canadian Rocky
550 Mountains, similar controls on the incorporation of sediment occur as glaciers melt, seeding downstream rivers with sediment-associated microorganisms.

Poor evidence for species sorting along glacier to downstream gradients

At the ASV level, we see some evidence for structured changes in microbial assemblages along the connected headwater to
555 downstream reaches that we investigated. First, within site (alpha) and between site (beta) diversity increased with increasing distance from the glacier (Fig. 7, Fig. S6). This result is consistent with a previous study in the Alps, where lower alpha diversity near glaciers was attributed to a decrease in the diversity of microbial source pools at higher elevations (e.g., lower groundwater contributions) and/or harsher upstream environmental conditions (Wilhelm et al., 2013). Second, we identified specific microbial indicator species for specific to each of the headwater and downstream sites. Headwater indicators
560 Microbacteriaceae *sp.* and Microbacteriaceae *Cryobacterium sp.* are heterotrophs that are commonly found within glacial

cryoconite holes and are believed to possess specific adaptations for cold temperatures and nutrient-poor conditions (Liu et al., 2020). The third headwater microbial indicator was a sulphur oxidizing chemolithotroph (Beggiatoaceae *sp.*), likely sourced from the subglacial environment where chemosynthesis is common (Anesio et al., 2017). The identification of these indicator species suggests that unique microbes (sourced from both the supraglacial and subglacial environment) may be seeding the fluvial microbial assemblage at the glacial headwaters (Wilhelm et al. 2013; Bhatia et al., 2006). Third, redundancy analysis identified a series of physicochemical variables – including protein-like DOM, water temperature, POC concentration, specific conductance, and deuterium excess – that were significantly associated with microbial assemblage composition across sites (Fig. 9). Of these environmental controls, DOM character can select for different specialized heterotrophic microbes (Judd et al., 2006), water temperature broadly controls rates of microbial activity (D'Amico et al., 2006), and POC can shape bacterial extracellular enzyme requirements and thus, microbial assemblage composition (Kellogg and Deming, 2014). From these lines of evidence, we conclude that glaciers are providing a biotic source of novel microbes to glacially-fed streams in the Rocky Mountains (see also Wilhelm et al., 2013; Bhatia et al., 2006), but that the association between assemblages structure and environment is modest, with the RDA explained only a small amount (9%) of the observed variation in microbial assemblages across sites (Fig. 9).

Dispersal and increasing cell sources with transit downstream

In addition to the overall modest RDA results, perMANOVA tests of species divergence among site bins (i.e., beta diversity) indicated that spatial variation only explained a small proportion of assemblage composition, suggesting a high degree of similarity between microbial communities ($R^2 = 0.09$, $p < 0.001$, Table S7). This low overall explanatory power of specific environmental controls, combined with a lack of strong assemblage divergence between environmentally disparate sites, is consistent with a recent synthesis effort that concluded environmental drivers are not always good predictors for microbial assemblage composition in streams (Zeglin, 2015).

This lack of strong control from among our suite of environmental variables is also consistent with patterns observed for the "core" group of microbial ASVs that was ubiquitous across all distance ranges (headwater, near, far) examined in this study. While this core ASV group accounted for only ~3% of the total identified ASVs, it represented a large proportion of average relative ASV abundance across sites, ranging from a median of 70% of the total assemblage relative abundance at headwater locations to 55% at downstream sites (Fig. 8). Across all sites, the most abundant (top 10) core ASVs, identified to the lowest taxonomic level available, display diverse metabolisms, with some heterotopic taxa that are considered to be generalists (e.g., *Flavobacterium sp.* (Zheng et al., 2019), Sporichthyaceae), specialists for methanol consumption (Methylophilaceae (Beck et al., 2014)), specialists for consumption of terrestrial carbon (*Alcaligenaceae sp.*) and autotrophic taxa, including both photoautotrophs (e.g., cyanobacteria) and chemoautotrophs (e.g., iron oxidizing Gallionellaceae (Hallbeck and Pedersen, 2014)). Increases in alpha diversity and declining relative importance of the "core" assemblage from headwaters to downstream sites indicates more diverse communities downstream, consistent with an increased variety of sources of microbial

595 communities from the surrounding terrestrial environment (Figs. 7, S6). However, the strong persistence of the core assemblage across our transects also reinforces that rivers are highly connected environments that can facilitate high rates of dispersal (Tonkin et al., 2018), albeit in a unidirectional fashion. As such, mass effects can be strong drivers of microbial assemblage composition in fluvial networks resulting in assemblage homogenization and, overall, decreased species sorting (Crump et al., 2007; Evans et al., 2017; Leibold et al., 2004; Pandit et al., 2009). These findings contrast with those from

600 studies of more isolated environments such as lakes (Logue et al., 2010; Van der Gucht et al., 2008) and soils (Fierer et al., 2006), where longer residence times better enables environmental divergence between sites to drive divergence in microbial assemblage composition. In our glacially-fed systems, particularly, we find that while local environmental conditions show some association with microbial assemblage composition, mass effects are likely more influential, suggesting a clear ecological connection from glacial outflows to 100-km downstream sites, despite striking environmental change.

605 **5 Conclusions**

This study illustrates that fluvial OM quantity and quality changes along stream transects from glacier headwaters to ~100 km downstream in the Rocky Mountains. These changes occur as a result of shifting inputs from different OM sources, with headwater sites adjacent to the glacier containing DOM that is compositionally consistent with predominantly autotrophic inputs (chemosynthesis and photosynthesis) and microbial reworking of terrestrially sourced OM that is aged and likely labile.

610 Comparatively, the downstream DOM pool was predominately humic-like, but exhibited seasonal shifts in apparent OM source with some allochthonous inputs in the pre-glacial melt and post-glacial melt seasons, and autochthonous sources increasing at the height of the melt season. This study also identified that a portion of glacially exported POC may be sourced from microbial activity, and thus is potentially accessible to downstream food webs. In contrast to these clear environmental gradients, we found that, despite evidence for glacial seeding of headwater microbial communities, mass effects appeared to enable dispersal

615 of a persistent core microbial group throughout our stream transects from the glacial headwaters to ~100 km downstream. Increasing microbial assemblage diversity with movement downstream, in the absence of a strong correlation with in-stream physical and chemical condition, points to a greater diversity of cell sources (i.e., increasing lateral inputs) along our study continuum.

620 Overall, findings from this work will enable better predictions about the impacts of glacial retreat on stream ecosystems, and the loss of labile OM inputs in alpine headwater regions. In the future, with increased glacier retreat in the Canadian Rockies, glacially-fed rivers will shift from a glacier ice melt regime towards one dominated by seasonal snow and rainwater inputs (Pradhananga and Pomeroy, 2022). Necessarily, these changes will be accompanied by increases in surrounding soil development and vegetation cover. The likely result is a higher proportion of humic-like DOM at headwater sites, a potential

625 loss of glacially-derived POM bioavailable to downstream primary consumer organisms (Cotner and Biddanda, 2002), and changes in other environmental parameters (temperature, chemistry) identified in this study as being important correlates for

stream microbial assemblages. At the same time, a loss of glacially exported microbes will alter the seed pool at headwater sites, which appears to persist far downstream, thus potentially altering the core assemblage that is characteristic of this glacially-fed river system. Overall, glacier retreat and eventual deglaciation appears poised to cause a loss of glacial seeding

630 of microbial assemblages in headwaters of the Canadian Rockies, similar to predictions for other glacierized Alpine regions (Hotaling et al., 2017). This loss of a unique source pool to the base of stream food webs could have a negative impact on overall biodiversity and ecosystem function, particularly if this loss of inputs decreases the functional breadth of these systems.

Data availability

635 All water chemistry and organic matter datasets generated and analysed in this study are available on PANGAEA repository (Felden et al., 2023)(PDI-35664 and Serbu et al., 2024). Microbial 16S rRNA gene sequences generated and used in this study can be accessed from the NCBI database, using accession number PRJNA995204.

Author contributions

640 VLStL, JAS, SET, and MPB designed the field program, and SET and MPB designed this study. HFD and JAS led the field sampling campaigns, and HFD, MAC, and JAS conducted the laboratory analysis. MAC built the 16S rRNA gene libraries and conducted the bioinformatic analyses. HFD conducted all other data and statistical analyses, and visualized the results. HFD wrote the paper with input from SET and MPB, and all authors provided comments on the manuscript.

Competing interests

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The authors declare that they have no conflict of interest.

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660   stahkoii ᗪᒧᒋᐧ ᐢᑊᕁᐟ (Blackfoot / Niitsítapi ᗪᒧᐟᗡ), Secwepemcúl'ecw (Secwépemc), Tsuut'ina, Michif Piyii (Métis), and Mountain Métis people. Finally, we are grateful to Martin Sharp for illuminating discussions.

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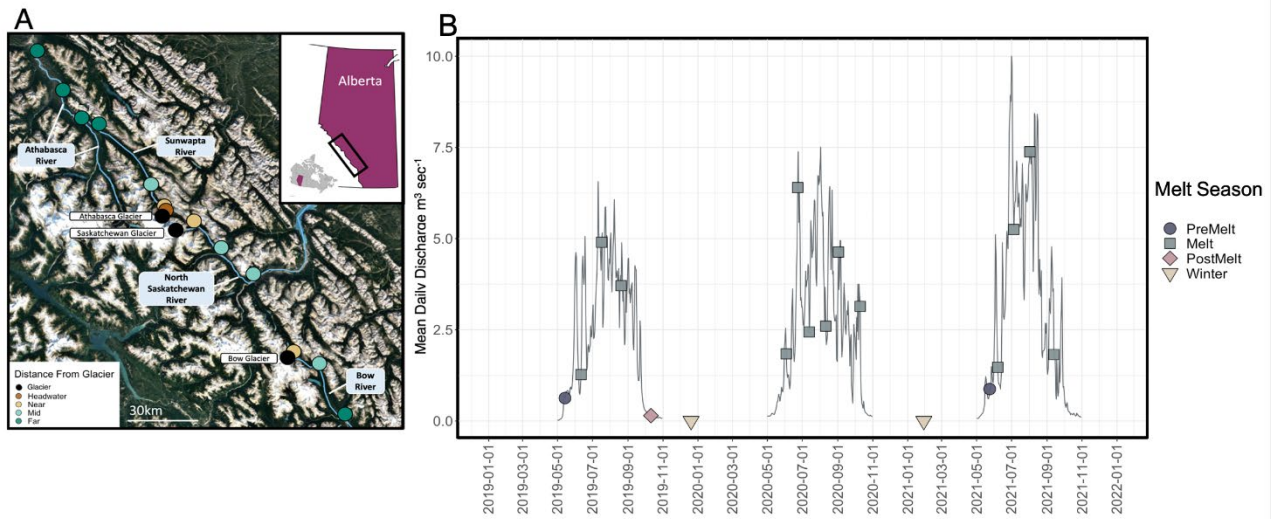
Figure 1: (A) Sampling locations of study sites on the Bow, North Saskatchewan, Sunwapta, and
Athabasca Rivers, in Alberta, Canada (Google Earth © 2022). Locations are coloured by distance range,
binned as headwater (0.2 km downstream), near (3-6 km downstream), mid (16-50 km downstream) and
far (>50 km downstream). The inset map shows the location of the sampling region (black box) within
Alberta. (B) Hydrograph of open-water discharge measured at the gauging station 3 km downstream of
Athabasca glacier (station 07AA007, maintained by Environment and Climate Change Canada) from
2019–2021; note that data are not collected November-May. The location of the hydrologic station
corresponds to the Sunwapta "near" site. Sample collection dates are demarcated, with corresponding
melt season indicated.
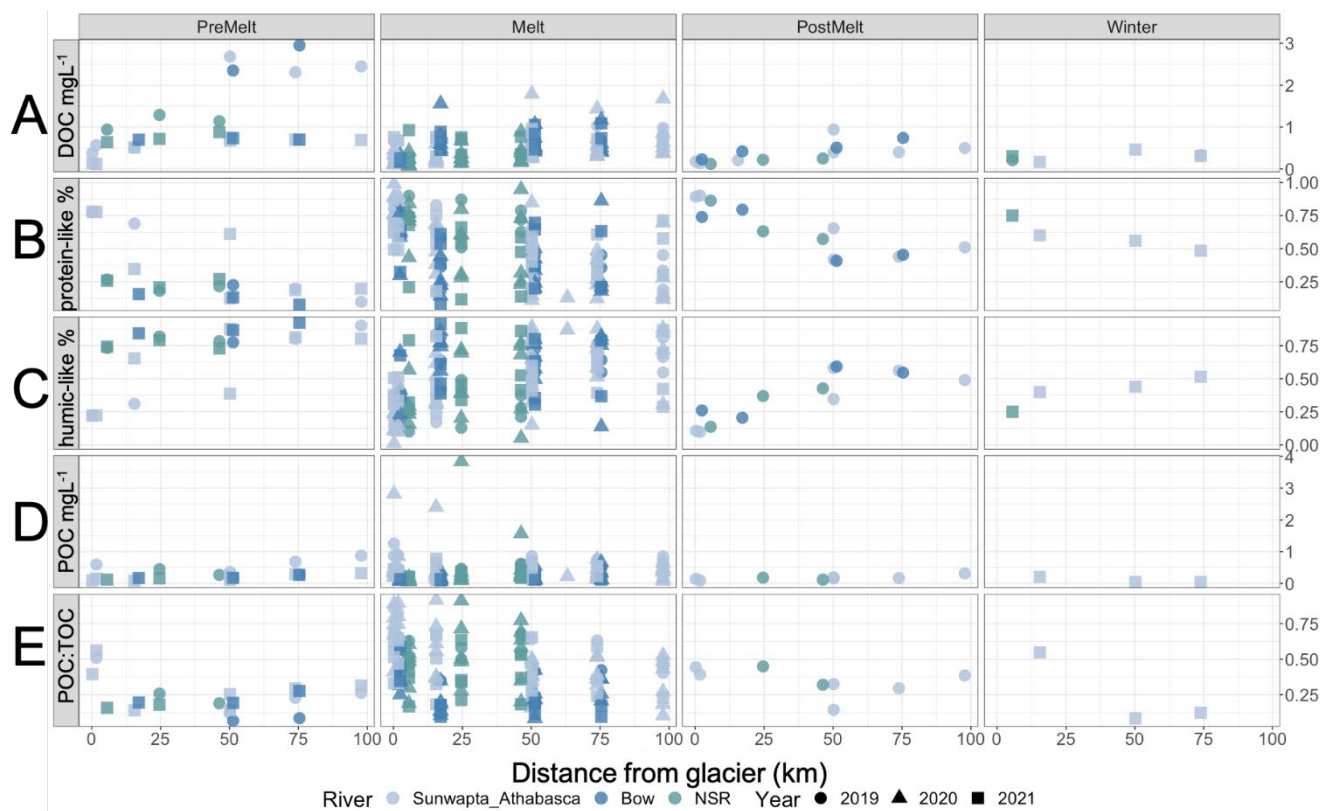
Figure 2: Plots of: (a) DOC concentration, in mg L^{-1} (b) relative percentage (%) of protein-like DOM (c) relative percentage (%) of humic-like DOM (d); POC concentration in mg L^{-1}; and (E) the ratio of POC to TOC (POC + DOC). Data are split by melt season and plotted against distance downstream, with river and year indicated.
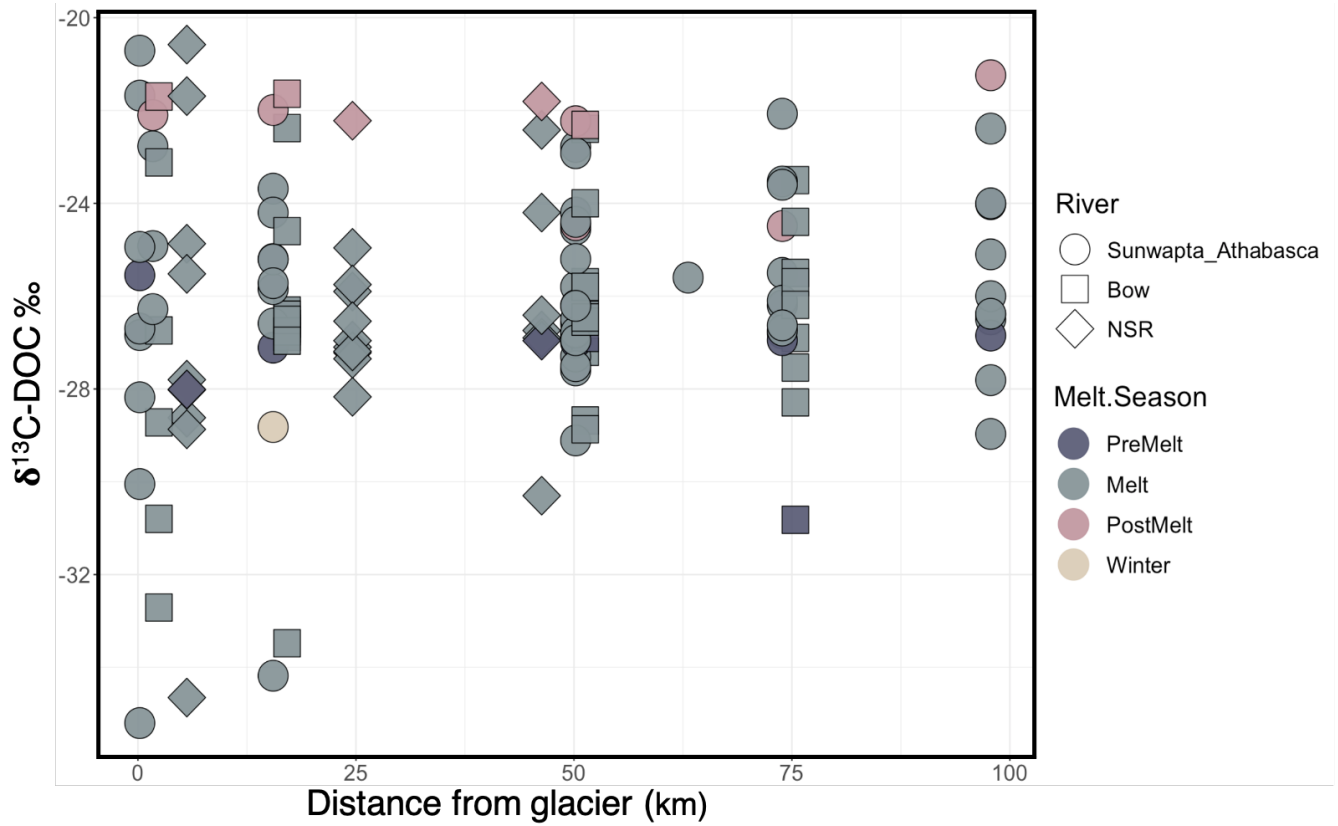
Figure 3: Stream δ¹³C-DOC as a function of increasing distance from glacier terminus. River and melt season are indicated by point shape and shading, respectively.
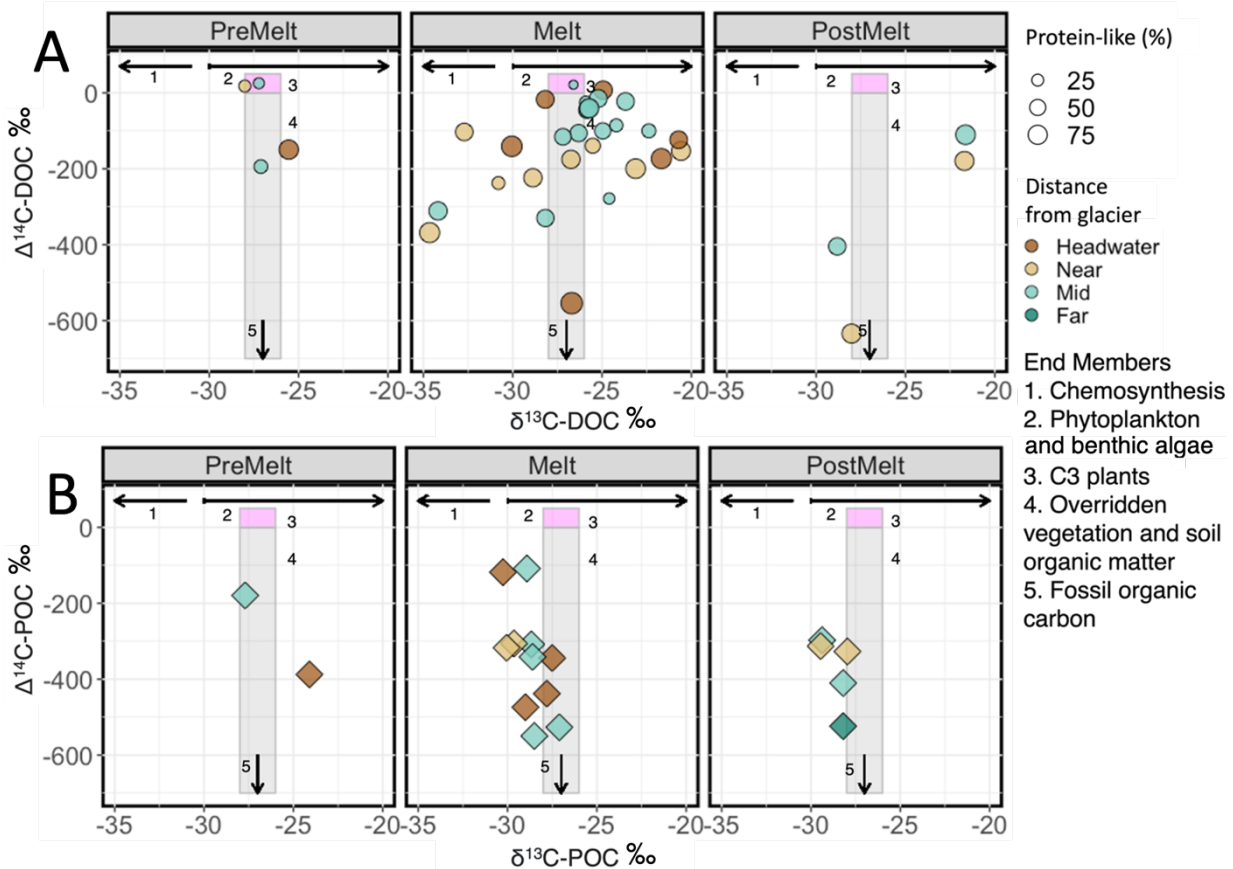
Figure 4: Δ^{14}C versus δ^{13}C values for (A) dissolved OC and (B) particulate OC. In panel A, symbol size indicates the percentage protein-like DOM in each sample. Symbol size variation is not applicable in panel B. Different colours represent distance ranges. Numbered (1–5) pink and grey boxes and black lines represent literature isotopic ranges for various endmembers: (1) right arrow; (2) right arrow; (3) pink box; (4) grey box; (5) downwards arrow (Table S8).
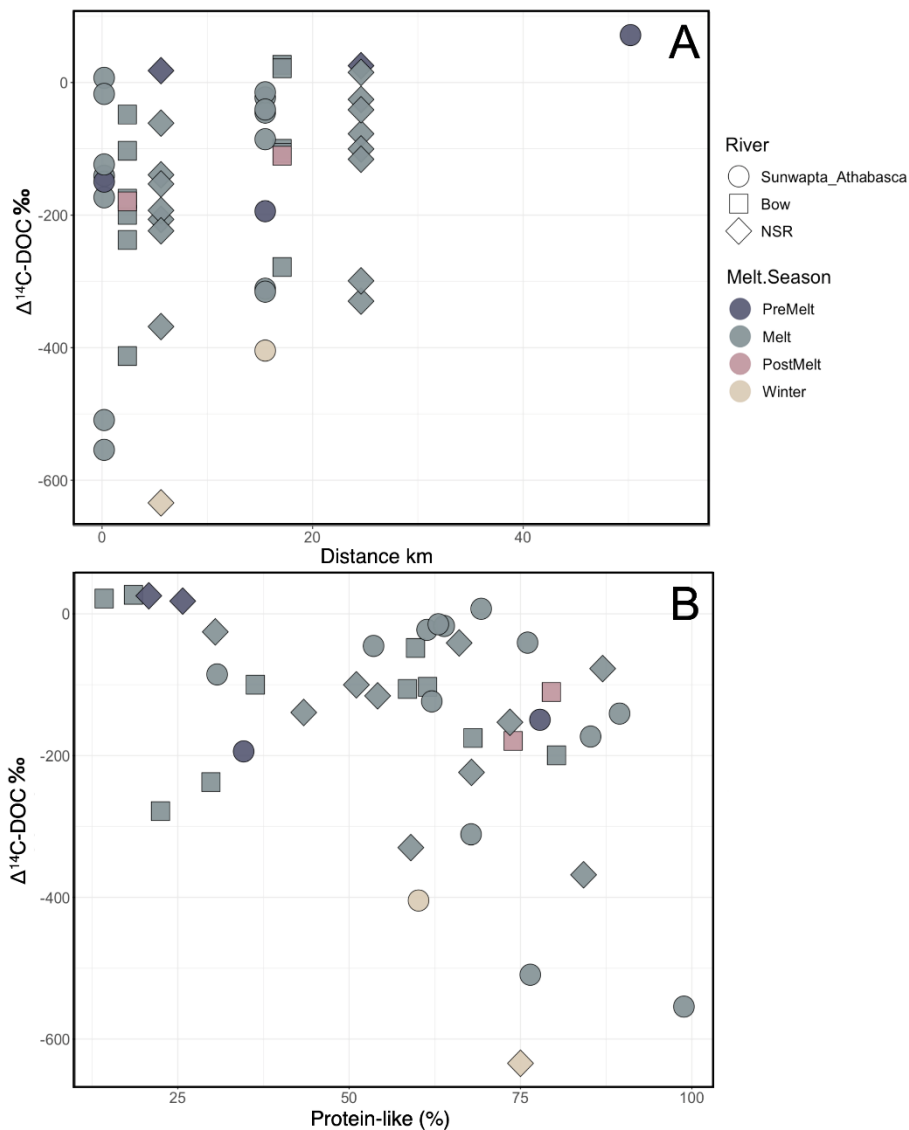
Figure 5: Plots of (A) stream Δ^{14}C-DOC with increasing distance from glacier terminus and (B) the relationship between stream Δ^{14}C-DOC and the relative percentage of protein-like DOM (%). River and melt season are indicated by point shape, and shading, respectively. Statistics for the relationship between Δ^{14}C-DOC and distance downstream are provided in Table S3. A linear model of the relationship between Δ^{14}C-DOC and protein like DOM shows a significant negative slope (slope = -3.037; $R2_{adj}$ = 0.17, $F_{1,39}$ = 8.212, p = 0.007).
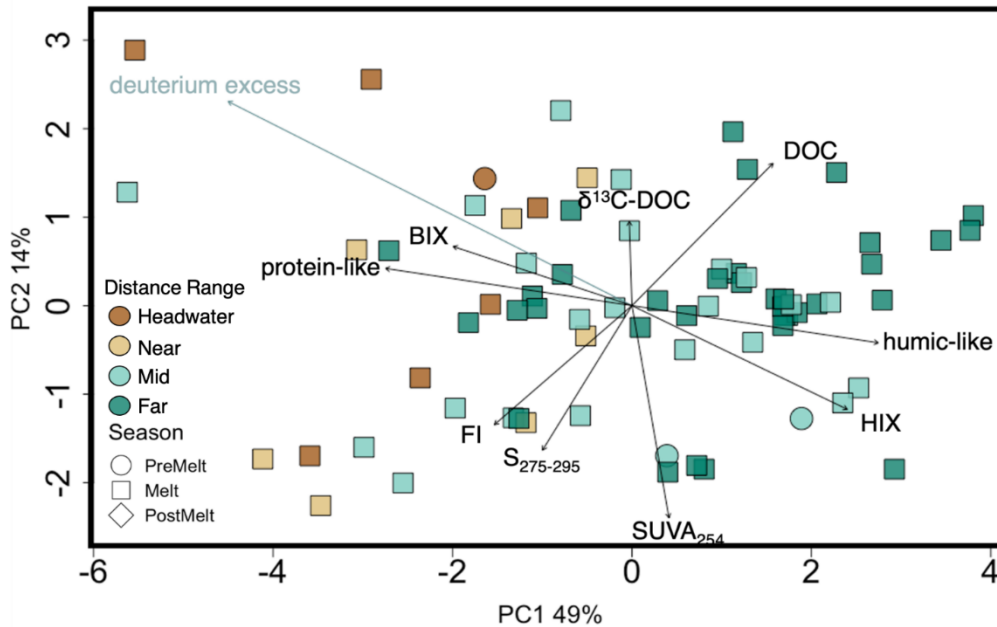
Figure 6: Principal component analysis of dissolved OM parameters (DOC concentration, δ^{13}C-DOC, $S_{275-295}$, SUVA$_{254}$, BIX, FI, HIX, protein-like DOM, and humic-like DOM). Deuterium excess (blue text) is a passive overlay. Colours represent distance range, and shapes represent hydrological period. Percentages along the axes are explained variance.

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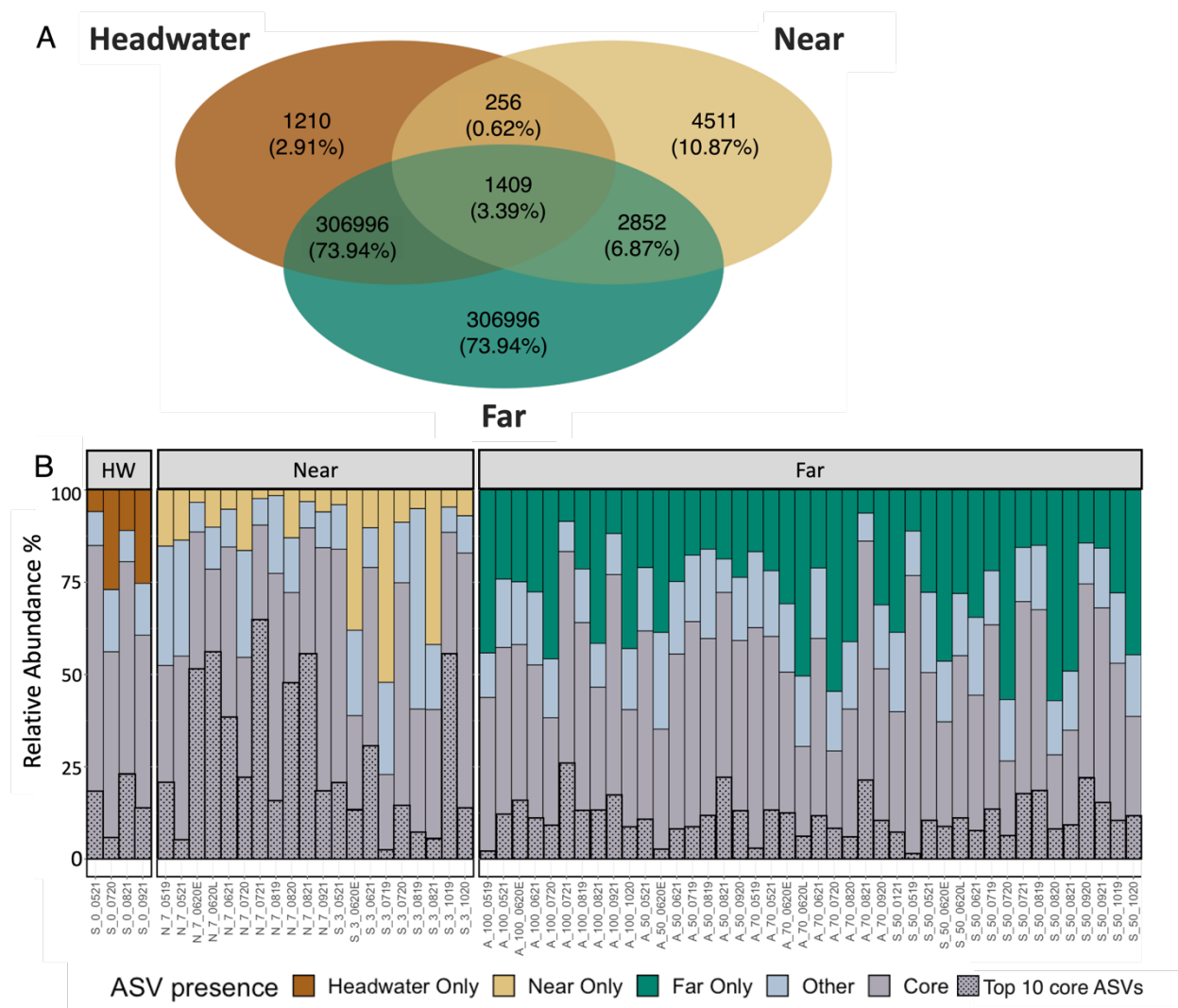
Figure 7: (A) Venn diagram showing the number of identified ASVs at the headwater (n = 4 samples), near (n = 20) and far sites (n = 42). (B) Bar plot showing the relative abundance of ASVs that are unique to each distance range, those ASVs identified in all three (headwater, near, far) distance bins ("core") or those that are shared between two distance bins ("other") in each sample. Sample abbreviations on the x-axis indicates river (Sunwapta (S), North Saskatchewan (N), Athabasca (A)), distance downstream (km), and sample date (as mmyy); suffixes "E" and "L" correspond to samples early, and late, in the associated sample month.
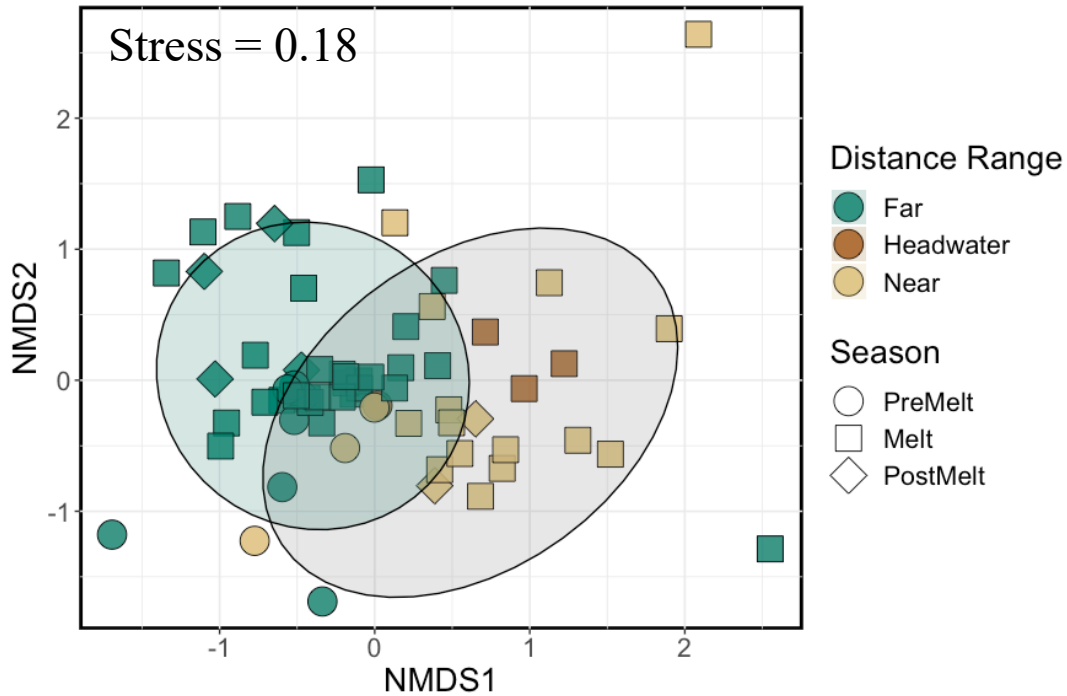
Figure 8: NMDS of microbial assemblage composition across sites and sample dates. Shapes represent different hydrological periods while colour indicates distance range. Shaded circles represent the 95% confidence interval for significantly different groupings, where the far sites were significantly different from headwater and near sites (pairwise perMANOVA p < 0.001 (holms adjusted), R2 = 0.09; see Table S7 for comparisons).
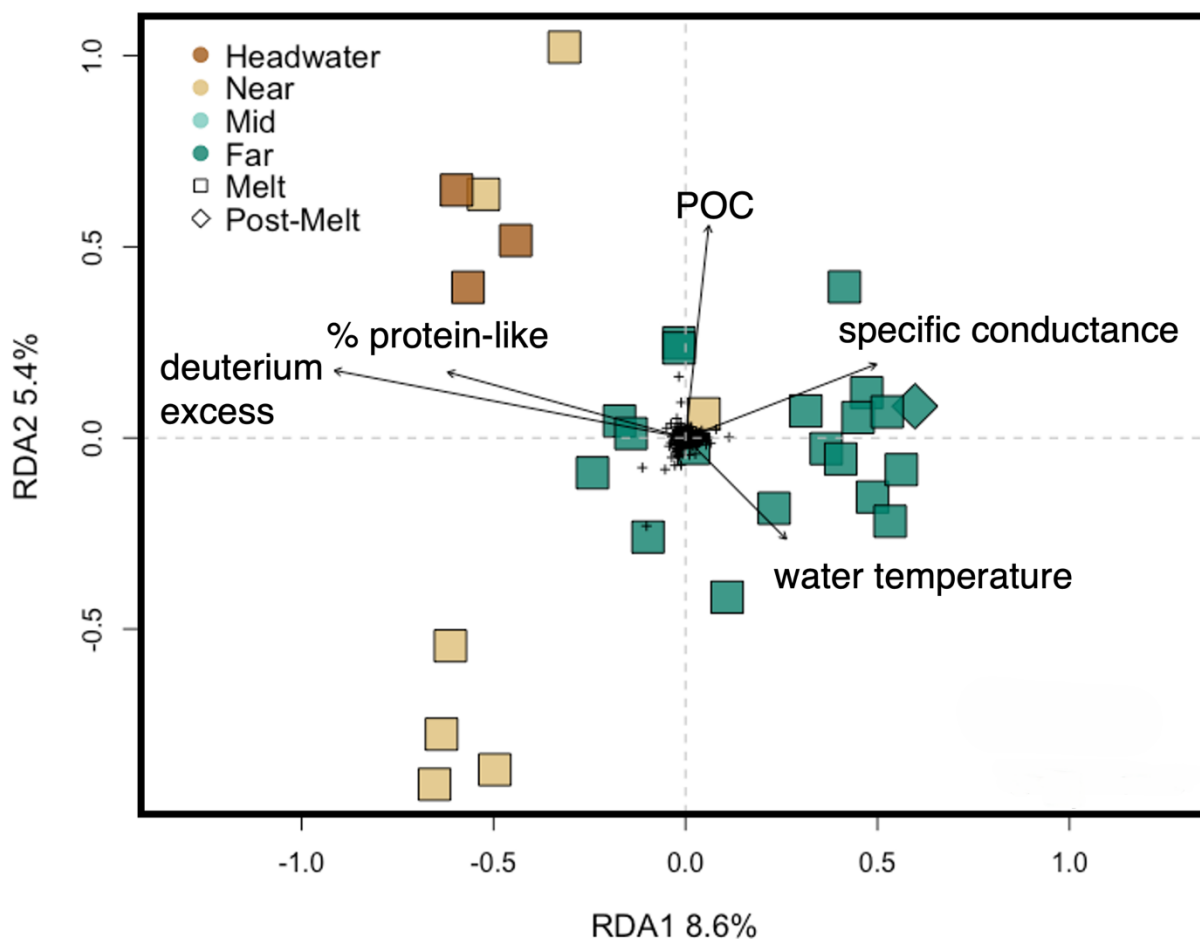
Figure 9: Redundancy analysis of microbial assemblage structure at the headwater, near, and far sites constrained by environmental variables, with black arrows showing significant explanatory variables as determined by a backward selection analysis (see methods). The percent variance explained by the first two RDA axes is shown on the axis labels. The full RDA has an adjusted R2 of 0.09, and unadjusted R2 of 0.25. Colours represent distance range and shapes represent hydrological period. Black crosses in the centre represent individual ASV scores.
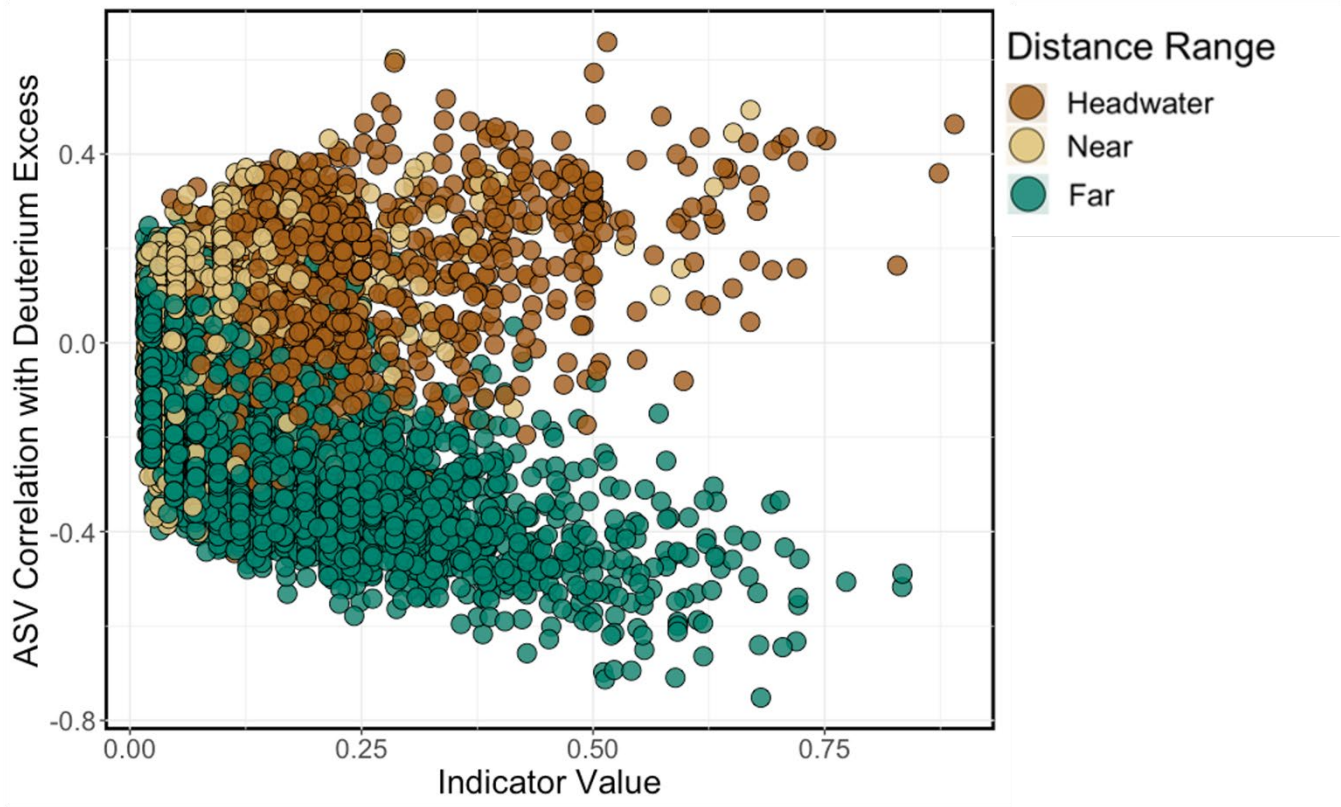
Figure 10: A scatterplot of the relationship between the Spearman's rank correlation coefficient (ρ) for ASV relative abundance and deuterium excess, and ASV indicator species value. Brown, beige, and green circles represent the ASVs resolved in the headwater, near and far samples, respectively.