Seasonal and spatial pattern of dissolved organic matter bio- and photodegradation in boreal humic waters

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Synopsis:

In boreal humic waters of a forest lake and a bog, the rate of dissolved organic matter photodegradation is four times higher than that of biodegradation. However, given the shallow (0.5 m) light-penetrating layer, the biodegradation provides the largest contribution to CO_2 emission from water surfaces. A few trace metals were partially removed (1-10 %) during photo- and biodegradation, via precipitation of Fe(III) hydroxides after destabilization of organo-ferric colloids and organic complexes

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Abstract

Studying competitive effects of microbial and light-induced degradation of dissolved organic matter (DOM) and trace metals is crucially important for understanding the factors controlling aquatic carbon (C), micronutrients and toxicants transformation in boreal waters. Here we characterized the degree of DOM and related major and trace element transformation under biotic activity and sunlight using conventional incubations of humic surface waters from the European subarctic: an ombrotrophic peatbog continuum (subsurface water – peatland pool – stream) and a stratified forest lake across seasons. Along the bog water continuum, biodegradation rate was the highest in subsurface waters collected via piezometer and the lowest in the acidic peatland pool. Photodegradation was similar for piezometrically collected subsurface waters and the stream, but was not detectable in the peatland pool. The waters of forest lake exhibited a strong seasonal effect of biodegradation, which was the highest in October and the lowest in June. Overall, the biodegradation was capable of removing between 1 and 7 % of initial DOC, being the highest in the forest lake in October and in peatland pool in summer. The photolysis was capable of degrading a much higher proportion of the initial DOC (10-25 %), especially in the forest lake during June and the bog stream during July. Only a few trace metals (TM) were sizably affected by both photo- and biodegradation of DOM (Fe, Al, Ti, Nb and light REE), whereas V, Mn, Co, Cu and Ba were affected solely by biodegradation. A likely mechanism of metal removal was their coprecipitation with coagulating Fe(III) hydroxides. Compared to typical CO_2 emissions from inland waters of the region, biodegradation of DOM can provide the totality of C-CO_2 evasion from lake water surfaces whereas bio- and photodegradation are not sufficient to explain the observed fluxes in bog water continuum. Overall, these results demonstrated strong spatial and seasonal variability in bio- and photodegradation of DOM and organic TM complexes, and call for the need of a systematic assessment of both processes across seasons with high spatial resolution.

1. Introduction

Organic Carbon (OC) processing via metabolic biological (heterotrophic bacteria uptake and respiration) and inorganic physico-chemical (photolysis) pathways is considered to be one of the major source of CO_2 supersaturation in surface waters and related C emissions (Lapierre et al., 2013; Tranvik et al., 2009), although the relative role of dissolved vs particulate organic carbon (DOC and POC, respectively) remains poorly quantified (e.g. Attermeyer et al., 2018; Lau et al., 2021; Shirokova et al., 2021; Keskitalo et al., 2022; Raudina et al., 2022). Given sizable CO_2 emissions in boreal and subarctic waters (Karlsson et al., 2021), together with high concentrations of DOC (Cole et al., 2007; Vonk et al., 2015), and fast ongoing and predicted environmental changes in high latitude aquatic and terrestrial ecosystems (Wauthy et al., 2018; Chaudhary et al., 2020; Harris et al., 2022), the surface waters of subarctic regions are at the forefront of studies on the biogeochemical cycle of C. Although emissions from these waters are significantly lower than those in the 10 °S – 10 °N equatorial belt (e.g., Borges et al., 2015), the magnitude of possible changes in C flux from northern waters to the atmosphere remains much less known. Further, there are still important geographical biases linked to insufficient knowledge of rates and mechanisms of DOC transformation in certain regions. An example is wetland-dominated northern aquatic settings, where high concentrations of soil organic C surrounding the bogs provide high concentrations of DOC but also some related trace metals, whose concentration and migration can be strongly controlled by processes of dissolved organic matter (DOM) transformaion.

Thorough laboratory and field work on DOM bio- and photolability conducted over the past decades have demonstrated both phenomena are important, and, depending on environmental setting (nutrient regime, photic layer depth, nature of DOM, etc.), one or another may dominate overall DOM removal in surface waters (Vachon et al., 2016, 2017; Vähätalo and Wetzel, 2008; Obernosterer and Benner, 2004). In addition to DOM, trace elements (limiting

micronutrients, toxicants and geochemical tracers) present in the form of organic and organo-mineral (Fe, Al) colloids and organic matter comlexes may be subjected to strong transformations during microbioological and photoltic degradations of DOM. This in turn may impact the bioavailability, toxicity and export fluxes of trace metals in from terrestrial to aquatic continental and finally coastal environments.

Recently, specific attention was devoted to the aquatic systems of permafrost peatlands given their high vulnerability to climate warming and huge potential for release of soil organic C to surfaces waters (Vonk et al., 2015; Shirokova et al., 2019; Payandi-Rolland et al., 2020; Prijac et al., 2022; Rosset et al., 2022; Taillardet et al., 2022). These studies provided a range of DOM susceptibility to biotic degradation. Thus, between 10 and 40 % of the DOC in lakes, rivers and soil waters of the boreal zone may be available for bacterial uptake over a time frame of several weeks (Berggren et al., 2010; Roehm et al., 2009). This range is consistent with 14-16% of biodegradable DOC (BDOC) assessed globally (Begum et al. 2022). The necessity for further studies was also indicated, most notably with regard to *i*) seasonal aspects, given that the overwhelming majority of available studies were performed during Arctic summer (see discussions in Vonk et al., 2015; Laurion et al., 2021), and *ii*) increased spatial resolution, given that sizable variations of BDOC can be observed within quite short distances of a hydrological continuum (Payandi-Rolland et al., 2020; Raudina et al., 2022). Another poorly studied aspect is DOM photo- and biolability across the depth of the water column, especially in seasonally stratified lakes which are subject to spring and autumn overturn.

Based on a compilation of available studies on BDOC and their own research, Vonk et al. (2015) argued there is a negligible amount of biodegradable DOC in aquatic systems without permafrost. This is, however, contradictory to available assessments on biodegradation of aquatic DOM as major driver of CO_2 emission in general (Amaral et al., 2021; Liu and Wang, 2022) and in boreal waters in particular (Ask et al., 2012; Lapierre et al., 2013). Furthermore, among all

Arctic rivers, the highest annual (20%) and winter (ca. 45%) biodegradable DOC (BDOC) was reported for the Ob River, which drains through peatlands with minimal permafrost influence (Wickland et al., 2012). These non-exhaustive examples illustrate certain inconsistency in current estimations of DOC biodegradability in surface organic-rich waters of high latitudes, which precludes quantitative modeling of future C fluxes between land, water and atmosphere in these environmentally important regions. Towards addressing these inconsistences, in this study, we chose a typical hydrological continuum in a boreal ombrotrophic bog in a glacial lake-ridge complex that includes subsurface water, a small peatland pool in the central part of the bog and an outlet stream. Further, we selected a well-studied deep stratified humic lake in the same region (Lake Temnoe; Chupakov et al., 2017) where we sampled surface and deep horizons for the incubation experiments. The chosen waters represent subarctic non-permafrost regions that exhibit sizable organic C pool in their soils and high concentrations of DOC in their surface waters. In contrast to previous studies of permafrost peatlands (Shirokova et al., 2019; Laurion et al., 2021; Payandi-Rolland et al., 2020; Mazoyer et al., 2022) where the main source of DOM is peat or ground vegetation like mosses and lichens, in this highly productive southern taiga region, DOC may be more vulnerable to microbial activity due to the presence of forest leachates (i.e., Don and Kalbitz, 2005; Kalbitz et al., 2003; Kawahigashi et al., 2004; Kiikkilä et al., 2013) and much higher bioproductivity for both the terrestrial and aquatic parts of the lake-river ecosystems. A broad importance of DOM bio- and photodegradation dynamics is that these processes can contribute to CO_2 emissions from water surfaces thereby directly controlling the C cycling between the land and the atmopshere.

The first working hypothesis behind our study design is that the DOC-rich subsurface water and deep horizons of the humic lake are mostly sensitive to sunlight impact (Stubbins et al., 2010), and that maximal impact of photodegradation is expected during allochthonous aromatic DOM input (high surface inflow to lakes and bogs in June and October). In contrast,

maximal biodegradation of DOM is expected during periods of possible phytoplankton bloom in August, when autochthonous organic material is generated in the water column. An important novelty of the present study is adressing trace metal (TM) partitioning during bio- and photodegradation. The link between DOM and TE is straightforward: in humic waters of peatlands, most TE (except probably some alkalis and oxyanions) are strongly (> 80%) associated to DOM in the form of organic and organo-mineral (Fe, Al) colloids (Pokrovsky et al., 2005, 2012, 2016). As a result, any DOM transformation processes may directly control the pattern of TE. From the other hand, some TE may be photosensitive (Mn, Fe), toxic (Al, Cu, As, Cd, Pb), or limiting micronutrients (Zn, Co, Ni, Mo) for the bacteria. Our second working hypothesis here is that removal of DOM via photo- or bio-degradation will change the partitioning of trace elements which are either *i*) strongly bound to DOM, such as divalent transition metals, or *ii*) incorporated into organo-mineral (Fe, Al) colloids, such as trivalent and tetravalent hydorlysates. The TE of 1st group might either remain in solution (during photodegradation), hence not modifying their total dissolved concentration, or being taken up by growing bacteria during bio-degradation of TE-bound organic matter (Shirokova et al., 2017a, c). The elements of the second group are capable of co-precipitating with Fe and Al hydroxides hence being scanvenged from the aqueous solution.(e.g., Kopacek et al., 2005, 2006). To test these hypotheses, we examined DOM and related trace metals bio- and photodegradability aiming to assess 1) spatial variations along a hydrological continuum of non-permafrost peatland and different horizons of a neighboring deep stratified lake located in the forest, and 2) temporal variability during 3 main hydrological seasons (high flow in June, baseflow in August and autumn rain season in October) in the forest lake. Achieving these objectives should allow quantifying the relative share of bio- and photodegradation on overall DOC and TM removal from surface waters via biotic and physico-chemical mechanisms. The third aim of this study

was to relate the measured rates of DOM photo and biodegradation to CO_2 emissions observed in the studied water bodies.

2. Materials and Methods

2.1. Natural settings of subarctic bog and stratified lake

The study site is in the NE part of the European boreal zone (Arkhangelsk region), **Fig. 1**. The mean annual air temperature is 0 °C and average annual precipitation is 700 ± 50 mm. The pristine ombrotrophic Ilasskoe Bog is located 30 km SE of Arkhangelsk, and is a typical lake-ridge complex formed from the last glaciation approximately 10,000 years ago. Its total surface area is 89 km², with an average peat thickness of 3 m. The hydrological continuum of the Ilasskoe Bog includes subsurface water collected via piezometer (2-2.5 m depth), a small lake (Severnoe) and a stream outlet (**Fig. 1**). Lake Severnoe, located in the central part of the bog, is a typical peatland pool with an average depth of 1.5 m and a surface area of 0.013 km². The Chernyi Stream is an outlet for the eastern part of the bog. The stream is 0.7-2.0 m wide, 10 km long and it flows in a forested (taiga) zone in the shade of tree canopy. The waters of the Ilasskoe Bog are acidic (pH ranges from 3.9-4.0 in piezometer and peatland pool to 5.7 in stream Chernyi), organic-rich (DOC is equal to 88, 13 and 38 mg L^{-1} in the piezometer, lake and stream, accordingly) and low mineralized (Electrical Conductivity is 17-46 µS cm^{-1}), as listed in **Table 1**.

Lake Temnoe is located in a pristine forest 100 km NNE of the town of Arkhangelsk, an area that does not receive any direct anthropogenic impact (**Fig. 1**). The watershed area is 3.08 km² and the lake surface area is 0.091 km², with a maximum depth of 37 m and a Secchi disk depth of 3.5±0.5 m. The water residence time in the lake is 394 days. Bogs constitute 31% of lake's watershed area, which is represented by carbonate-free loamy moraine atop the peat, podzol and gley soils. The lake water is slightly acidic (pH = 5.1 to 6.0), humic (DOC = 13-20

mg L^{-1}) and dominated by allochthonous DOM with a low concentration of total dissolved ions (Electrical Conductivity of 20 µS cm^{-1}). Similar to other deep boreal and subarctic lakes, the lake exhibits 2 main periods of pronounced stratification (November to April and June to September) and two periods of lake overturn (October and May). Maximal winter stratification occurs in March; the highest water temperature typically occurs in July (see Chupakov et al., 2017 for details).

The surface waters were collected from the shore (peatland pool and stream) or a PVC boat (Lake Temnoe). Surface (30-50 cm depth) waters were sampled in the Ilasskoe bog and 3 water horizons (0.5, 5 and 10 m) were sampled in the Temnoe Lake using a pre-cleaned polycarbonate horizontal water sampler (Aquatic Research Co, ID, USA). The water samples were placed into 2-L Milli-Q pre-cleaned PVC jars and kept refrigerated (4 °C) until arrival at the laboratory within 2-3 hours of collection.

2.2. Experiments

2.2.1. Biodegradation

For DOM biodegradation assessments we followed the recommended protocol and used the appropriate type of labware for assessing biodegradable DOC of Arctic waters without external nutrient addition (Vonk et al., 2015; Payandi-Rolland et al., 2020) and applied a slight modification from Shirokova et al. (2019) to assess maximal possible biodegradation. Initial water samples brought to the laboratory within 2-3 hours after sampling were filtered through 3 µm sterilized Nylon Sartorius membranes (47 mm diameter); these were used because 'conventional' 0.8-1.2 µm (GF/F) filtration membranes might remove too many microbial cells (Dean et al., 2018).

Duplicate 30 mL aliquots of 3 µm-filtered water were placed into pre-combusted (4.5 hours at 450°C) dark borosilicate 40 mL glass bottles wrapped in Al foil to prevent any

photolysis, without nutrient amendment and incubated at 22±1°C in the dark. The bottles were closed with loosened sterilized PVC caps to allow air exchange. The bottles were shaken manually once a day avoiding the liquid touching the cap. The entire reactor was used for sampling after 0, 2, 5, 8, 12, and 21 days of exposure. Sampled solutions were filtered through sterile, MilliQ-cleaned Sartorius 0.22 µm filters. The DOC blanks for these filters did not exceed 1% of DOC concentrations in experimental samples. Sterilized control reactors were filled with natural water that was filtered through a 0.22 µm sterile filter and incubated together with experimental reactors following the approach of Köhler et al (2002).

All handling and sampling of bottles was performed in the laminar hood box in a sterilized workspace. Filtered samples were acidified with 30 µL of concentrated (8.1 M) double distilled HCl, tightly capped and stored in the refrigerator before DOC analyses. The non-acidified portion of filtrate was used for pH, Specific Conductivity, DIC and $UV_{254\ nm}$ and optical spectra measurement. Control runs were 0.22 µm sterile-filtered water which was incubated in parallel with experiments and re-filtered through 0.22 µm filters the day of sampling. To ensure minimized release from sterilized Nylon membrane, we ran blank (Milli-Q) filtrations through both GF/F and 0.22 µm Nylon filters; in both cases the DOC blank was below 0.1-0.2 mg/L which is less than 1% of DOC concentration in our samples. The glass bottles were incubated in duplicates at 22±1°C and agitated manually at least once a day over the 16 days of exposure.

2.2.2. Photodegradation

For photodegradation incubations, water samples were collected in Al-foil covered pre-cleaned polypropylene jars and sterile filtered (0.22 µm Nalgene Rapid-Flow Sterile Systems) within 2 hours of sampling and refrigerated. The filtrates were transferred under laminar hood box into sterilized, acid-washed quartz tubes (150 mL volume, 20% air headspace) with silicate stoppers and placed at 3 ± 2 cm depth into an outdoor pool which was filled by river water having

the light transparency similar to that of the Ilasskoe and Temnoe lakes. The outdoor pools were placed in an unshaded area with a latitude similar to the sampling sites (< 30 km from Ilasskoe Bog and Temnoe Lake). Slight wind movement and regular manual shaking allowed for sufficient mixing of reactor interiors during exposure. All photodegradation experiments were run in duplicates. The water temperature (EBRO EBI 20) and light intensity (Luxmeter Testo 545) were continuously recorded every 3 hours.

For DOM photodegradation experiments, we followed conventional methods requiring exposure of 0.2 µm-sterile filtered samples in quartz reactors in the outdoor pool (Vähätalo et al., 2003; Chupakova et al., 2018; Gareis and Lesack, 2018), solar simulator (Lou and Xie, 2006; Amado et al., 2014) or directly in the lake water (Laurion and Mladenov, 2013; Groeneveld et al., 2016). Note that the 0.22 µm sterile filtration is the only way of conducting photodegradation experiments, given that autoclave sterilization of DOM-rich natural waters would coagulate humic material and thereby would not be suitable (Andersson et al., 2018). Filtration through a smaller pore size, however, would decrease the concentration of DOC and trace metals (i.e., Ilina et al., 2014; Vasyukova et al., 2010). We have chosen a 16 day exposure time for logistical constraints, which is still consistent with biodegradation experiments described above and with the duration used in previous studies on photodegradation under sunlight, from 15 to 70 days (Moran et al., 2000; Vähätalo and Wetzel, 2004; Mostofa et al., 2007; Chupakova et al., 2018). Dark control experiments were conducted also in duplicates, using sterilized glass tubes filled with sterile 0.22 µm-filtered water, wrapped in Al foil and placed in the same outdoor pool as the experiments. The headspace (approx. 20% of total reaction volume) was similar in experimental and control reactors. The individual reactors were sterile sampled at the beginning and after the 0, 2, 5, 8, 12, and 16 days of exposure. Each sampling sacrificed the entire reactor. The Milli-Q blanks were collected and processed to monitor for any potential sample

contamination introduced by our filtration, incubation, handling and sampling procedures. The organic carbon blanks of the filtrates did not exceed 0.2 mg/L.

2.3. Analyses

The temperature, pH, O_2 and specific conductivity in surface waters were measured in the field. The dissolved CO_2 concentration in the studied bodies of water was measured in-situ using submersible Vaissala Carbocap® GM70 handheld carbon dioxide meter with GMP222 probes (accuracy 1.5%; see Serikova et al. (2018, 2019) for methodological details). The diffusional CO_2 flux was calculated using a wind-based model (Cole and Caraco, 1998) with $k_{600} = 2.07 + 0.215 \times u_{10}^{1.7}$, where u_{10} is the wind speed at 10 m height, following the approaches developed for surface waters of peatlands (Zabelina et al., 2021).

The DOC and DIC were analyzed by high-temperature catalytic oxidation using a Shimadzu® TOC-VCSN (uncertainty ± 2%, 0.1 mg L^{-1} detection limit). DIC was measured after sample acidification with HCl and DOC was analyzed in acidified samples after sparging it with C-free air for 3 min at 100 mL min^{-1} as non-purgable organic carbon (NPOC). Internationally certified water samples (MISSISSIPPI-03 and Pérade-20) were used to check validity and reproducibility of the analysis. Filtered sampled collected from photodegradation experiments were acidified with ultrapure nitric acid and analyzed for major and TE following the procedures employed by Geoscience and Environment Toulouse Laboratory for analyses of boreal humic waters (Oleinikova et al., 2017, 2018).

The UV- and visual absorbance of water samples was measured using a 10 mm quartz cuvette on a CARY-50 UV-vis spectrophotometer to assess the aromaticity of pore fluids via specific UV absorbance ($SUVA_{254}$). In the filtrates, we measured optical density at 254 nm and at selected wavelengths (365, 436, 470, and 665 nm) as well as the entire UV-visible spectrum. The specific UV-absorbency ($SUVA_{254}$, L mg^{-1} m^{-1}) and E_{470}:E_{665} ratios are used as a proxy for

281 degree of condensation of aromatic groups of DOM, or humification (Chin et al., 1994; Weishaar
282 et al., 2003; Hur et al., 2006; Peacock et al., 2013). The ratio E_{254}:E_{436} is useful for evaluation
283 of contributions of autochtonous (aquatic) DOM compared to terrestrial (soil) C (Hur et al., 2006;
284 Ilina et al., 2014). The ratio E_{254}:E_{365} also allows approximating the mean molecular weight of
285 DOM (Hiriart-Baer et al., 2008; Berggren et al., 2007). For better visualization of the differences
286 in spectral parameters between experimental and control reactors, we calculated the difference
287 (ΔA) between the absorbance of the photo- or bio-reactor and that of the control reactor at each
288 sampling time.

289 Major cations, Si, P and ~40 TE were measured with a quadrupole ICP-MS (Agilent 7500
290 ce) using In and Re as internal standards. The international geo-standard SLRS-6 (Riverine Water
291 Reference Material for Trace Metals) was used to check validity and reproducibility of analyses.
292 Note that for both bio- and photodegradation experiments, ICP MS analyses were performed
293 over 16 days of incubation time.

294 To check for possible microbial development in biodegradation experiments, we
295 performed oligotrophic and eutrophic bacteria counts over the course of incubation, following
296 the standard methodology used in biodegradation experiments of peat waters (Stutter et al., 2013)
297 and also described previously (Shirokova et al., 2017b; Chupakova et al., 2018). Specifically,
298 active bacteria number count (colony forming units, CFU mL^{-1}) was performed using Petri dishes
299 inoculation (0.1 to 1.0 mL of lake water in three replicates) performed in a laminar hood box
300 immediately prior the experimental incubation start and upon each sampling. Samples were
301 inoculated on Nutrient Agar (5 g L^{-1} beef extract, 5 g L^{-1} gelatine peptone, 15 g L^{-1} bacteriological
302 agar, pH=6.8±0.2 at 25 °C) to determine the total number of heterotrophic bacteria. Difco@ agar
303 (granulated powder, Lot No 6290083) inoculation was used to assess the number of oligotrophic
304 bacteria. Inoculation of blanks was routinely performed to assure the absence of contamination
305 from external environments.

306 *2.4. Data treatment*

307 The bio- and photodegradable DOC and trace metals were calculated as percent loss
308 relative to control in similar fashion with other studies (Vonk et al., 2015; Chupakova et al.,
309 2018; Shirokova et al., 2017b, 2019). However, previous works in similar environmental
310 contexts of high-DOC humic waters demonstrated that the effects of DOC and element decrease
311 are rather low and often comparable to uncertainties of duplicates (Shirokova et al., 2019). To
312 assess the net effect of bio- or photodestruction during the experiment, we used the integral
313 values of concentration change, estimated as the difference between the experiment and the
314 control, while taking into account the standard deviation of replicates. For this, we first calculated
315 the mean of replicates at the *i*-th time of sampling for the experiment and the control of *X*
316 component ($^{mean}X_i$ and $^{control}X_i$, respectively). We next calculated the sum of mean concentration
317 of replicates and its standard deviation ($^{mean}X_i+SD_i$). Thus, we obtained 3 values characterizing
318 the bio- or photo-degradation process: 1) the change of concentration in the experimental reactor
319 (^{mean}S), 2) the change of concentration not linked to the studied process (^{control}S), and 3) the
320 maximal uncertainty of the concentration change in the reactor ($^{mean+SD}$S). This allowed
321 calculating, in percentages, the efficiency of bio or photodegradation of *X* component relative to
322 the control, taken into account relevant uncertainties as following:

323 $$X\ (\%) = 100 \times (|^{mean}X| - |^{control}X|)/|^{control}X| \quad (1)$$

324 $$SD\ (\%) = 100 \times (|^{mean+SD}X| - |^{mean}X|)/|^{control}X| \quad (2)$$

325 where *X* is biodegradable DOC or trace element (BDOC and BTE, respectively) or
326 photodegradable DOC and trace element (PDOC and PTE, respectively). The sign of *X*
327 designates either a decrease («–») or an increase (« + ») of solute concentration during the
328 experiment. We considered the decrease of concentration significant when *X* (%) > SD (%). In
329 other cases, the change was non-systematic over the course of experiment or non-measurable
330 using the experimental technique employed in the present study.

331 The mean rate of bio- or photodegradation of X component (V_X) was calculated based on
332 the overall change (ΔX, in %) between the initial (X_0) and final value normalized to overall
333 duration of the experiment t (22 and 16 days for bio- and photodegradation, respectively):

334 $$V_X = ((\Delta X / X_0) / t \quad (3)$$

335 The SD for rates of component change were calculated in a similar way.

336 The spectral differences between experimental and control reactors were presented as X-
337 Y-Z diagrams where X is elapsed time, Y is wavelength, and Z is ΔA. The data were plotted in
338 a Surfer software package using triangulation with a linear interpolation method. Statistical
339 treatment included the least squares method and the Pearson correlation, as the data were
340 normally distributed. All calculations were performed in STATISTICA ver. 10 (StatSoft
341 Inc.,Tulsa) at $p = 0.05$).

342

343 **3. Results**

344 *3.1. Field measured C concentration and calculated CO_2 fluxes*

345 The DOC concentration ([DOC]) ranged from 13 to 21 mg L^{-1} in Lake Temnoe,
346 depending on depth and season. The CO_2 concentrations and fluxes increased from June to
347 October and varied from 99 to 220 µmol L^{-1} and 32 to 71 mmol CO_2 m^{-2} d^{-1}, respectively (**Table**
348 **1**). In Ilasskoe Bog hydrological continuum, the DOC decreased from 88 mg L^{-1} in the peat soil
349 water to 38 mg L^{-1} in the outlet stream. The DOC concentration was generally similar (within ±
350 5 %) between 3, 0.8 (GFF), 0.45 and 0.22 µm pore size filtration of the initial sample, which is
351 in agreement with former size fractionation measurements for Arctic and subarctic systems
352 (Vasyukova et al., 2010; Pokrovsky et al., 2012, 2016, Shirokova et al., 2019). The waters of
353 Ilasskoe Bog continuum exhibited CO_2 supersaturation with respect to atmosphere (from 55 to
354 3300 µmol L^{-1}) and calculated CO_2 emission (diffusion) flux ranging from 22 mmol CO_2 m^{-2} d^{-1}
355 in the peatland pool to 1600 mmol CO_2 m^{-2} d^{-1} in the piezometer (**Table 1**).

356 *3.2. Biodegradation of DOM*

357 3.2.1. DOC concentration evolution in the experiments

358 In the Temnoe Lake, the range of [DOC] change during 2-3 week incubation in the
359 experimental reactors did not exceed 2 mg L^{-1} and remained within +0.5 to -1.5 mg L^{-1}, which is
360 less than 10% of the initial DOC amount (**Fig. 2 A and Fig. S1** of the Supplement). The
361 biodegradable DOC was both season and depth dependent and ranged from 2 to 6 % (**Table 2**).
362 The integral 2-week rates of biodegradation (**Table 3, Fig. 3 A**) demonstrated the highest values
363 during autumn at depths of 0.5 m and 10 m and the lowest values during June at all depths. The
364 final 0-10 m water column- and season-averaged biodegradation rate in Lake Temnoe ranged
365 from 0.02 to 0.04 mg DOC L^{-1} d^{-1}. Integral rates of bio-degradation in the 0-10 m layer
366 demonstrated an increase from May to October, over the entire open-water period (**Fig. 4 A**).

367 For Ilasskoe Bog, the BDOC was highest in the peatland pool (4.9 ± 1.4 %) and lowest
368 in the outlet stream (3.1 ± 2.4 %; **Fig. 2 A** and **Fig. S1**). The integral rate of DOC biodegradation
369 followed the order 'piezometer >> stream > peatland pool' and ranged from 0.03 to 0.17 mg C
370 L^{-1} d^{-1} (**Table 3, Fig. 3 A**).

371

372 3.2.2. Optical parameters of DOM

373 In Lake Temnoe, the $SUVA_{254}$ remained relatively constant (4.2 to 4.6 L mg C^{-1} m^{-1})
374 across seasons and depths (**Table 1 B**). Over the course of biodegradation, the $SUVA_{254}$ did not
375 change significantly (i.e., less than 0.2 units, which is comparable to the variability of duplicates;
376 **Fig. S2**). The ratio E_{254}:E_{436}, which is an indicator of humification, increased with incubation
377 time in Lake Temnoe waters; the magnitude of this increase across depth followed the order "0.5
378 m > 5 m > 10 m" (**Fig. S3**). The ratio E_{254}/E_{365} also increased over the course of biodegradation,
379 corresponding to an increase of mean molecular weight of DOM (Hiriart-Baer et al., 2008;
380 Berggren et al., 2007). The ratio E_{365}/E_{470} also demonstrated the strongest increase in surface

horizons and virtually no change in the deepest horizon (**Fig. S3**). An increase in the ratio E_{470}:E_{665} corresponds to a decrease in the degree of aromaticity (humification). An increase in the ratio E_{254}:E_{436} signifies a decrease in contribution of autochthonous (aquatic) DOM compared to terrestrial (soil) C, whereas an increase in the E_{254}:E_{365} ratio characterizes removal of low molecular weights compounds.

In Ilasskoe Bog samples, the highest SUVA was observed in the water of the piezometer and the lowest in the stream, but the evolution of this parameter in the course of biodegradation was rather weak (**Fig. S3**). The E_{254}:E_{365} and E_{254}:E_{436} ratios increased with incubation time in the piezometer and decreased with time in the stream (**Fig. S3**). The optical ratios (E_{254}:E_{436}, E_{365}:E_{470}, E_{470}:E_{665}) increased in the peatland pool, suggesting an increase in the molecular weight and an increase in the ratio of aromatic to aliphatic compounds.

Complete spectral differences between the experimental and control samples demonstrated rather weak ($\Delta A \leq 0.04$) changes of spectral parameters, mostly detectable after 10-12 days of incubation (**Fig. S4**). These results were generally consistent with the discrete spectral parameters presented above and demonstrated maximal effects in the piezometer and bog outlet stream. In Lake Temnoe, the maximal impact of biodegradation on spectral parameters was observed in June, at 0.5 m depth.

3.2.3. Bacterial number evolution during biodegradation experiments

The number of cultivable eutrophic bacteria (EB) sizably (ca., 2 orders of magnitude) increased during biodegradation of Lake Temnoe waters. However, this evolution was not systematic in the course of incubation; there was a pronounced decrease after 2 weeks of exposure in June and August and rather stable concentration in waters of all horizons sampled in October (**Fig. S5**). Such maxima in June and August might be linked to consumption of substrate/nutrient limitations on bacterial growth. In Ilasskoe Bog continuum, the number of

406 eutrophic bacteria decreased by an order of magnitude in the peatland pool and piezometer while
407 it remained constant in the stream. The number of oligotrophic bacteria (OB) increased in waters
408 of all Lake Temnoe horizons by ca. 2 orders of magnitude in August and October and 1 order of
409 magnitude in June. In contrast, the OB number did not change or slightly decreased during
410 incubations of waters from Ilasskoe Bog continuum (**Fig. S5**).
411
412 3.2.4 Trace element behavior
413 During biodegradation experiments, a number of trace metals [Group 1] demonstrated a
414 significant (X > SD, Eqn. 1) decrease in concentration across the incubation period (**Table 2**):
415 Al, Ti, Fe, Co, Cu, Ba, Nb, light REE (LREE) and Pb (as illustrated for Fe in **Fig. 5)** as well as
416 Mn, V, and La (**Figs. S6, S7 and S8**, respectively). The most significant effects were observed
417 for Fe in the 0-5 m horizon of Lake Temnoe (9 to 18 % in June, 6 to 13.5 % in August and 8 to
418 9.5 % in October) and 14% in the peatland pool of Ilasskoe Bog. Overall, for most elements
419 except Fe and Mn, this increase was less pronounced than that of DOC; maximal effects were
420 achieved for Lake Temnoe in August and October (V, Mn, Co, Cu, Ni, Nb, Hf, Pb and Th) and
421 in June (Al and Ti). These elements are typically linked to DOM and Fe and present in the form
422 of organic- and organo-mineral colloids. Second group of major and trace elements did not
423 appreciably change their concentration (< 2 % decrease): Li, B, Na, Mg, K, Ca, Si, Ge, As, Rb,
424 Sr, Mo, Sb, Mo and Ba. These elements are not linked to colloids of Fe(III) hydroxide and organic
425 matter. Finally, some elements [Group 3] exhibited unstable behavior without systematic change
426 in concentration during the exposure (X < SD, Eqns. 1-2): Cr, Zn, Cu, Sr, Cd, (Y, Zr), Cs, Tl and
427 U. These elements cannot be considered as significantly impacted by the biodegradation process
428 in Lake Temnoe water.
429 In the Ilasskoe Bog hydrological continuum, the most significant changes during
430 biodegradation were observed in the peatland pool and outlet stream. Elements strongly (> 5-10

%; X > S.D. in Eqn. 1) affected by biodegradation were organically / colloidally comlexed V, Fe, Ni, Ga, Y, LREEs and Pb.

3.3. Photodegradation of DOM

3.3.1. DOC concentration evolution

Compared to biodegradation, photodegradation demonstrated much higher values of PDOC and rates of reaction as well as higher variability among seasons and sites. In Lake Temnoe, the PDOC was the highest in June and the lowest in October (**Fig. 2 B** and **Table 2**). The maximal range of concentration change during 2-week period achieved 6-8 mg L^{-1} (**Fig. S9**) which was 10 to 20 % of the initial [DOC] values. The rates strongly decreased from May-June to the end of summer – autumn. The depth integrated (0 to 10 m) rate of DOM photodegradation in Lake Temnoe ranged from 0 in October to 0.2 mg C L^{-1} d^{-1} in June (**Table 3**; **Fig. 4 B**).

In the Ilasskoe Bog hydrological continuum during July, the photodegradation rate followed the order "outlet stream > piezometer >> peatland pool" (**Fig. 3 B**), where integral rates equaled to 0.27±0.04, 0.33±0.07, and 0±0.05 mg C L^{-1} d^{-1}, respectively (**Table 3**).

3.3.2. Optical parameters of DOM

Similar to the DOC concentration, the optical parameters of DOM more strongly evolved over the course of photodegradation compared to the biodegradation experiments. In the Temnoe Lake, the strongest decrease in $SUVA_{254}$ was observed in the waters of all horizons in June. This decrease was less pronounced in October (**Fig. S10**). The E_{254}:E_{365} ratio demonstrated a sizable increase in June, with much weaker increase in October. The E_{254}:E_{436} ratio strongly decreased with exposure time throughout all seasons (10 m depth) and only in June in the surface horizons (**Fig. S11**). An increase in the ratio E_{254}:E_{365} over the course of photodegradation corresponded to an increase in mean molecular weight of DOM. The ratios E_{365}:E_{470} and E_{470}:E_{665} decreased

456 in all experiments with the Temnoe Lake waters (**Fig. S11**), suggesting a decrease in the degree
457 of humification (Battin, 1998) and a decrease in the ratio of aromatic to aliphatic moieties.
458 The $SUVA_{254}$ in Ilasskoe Bog waters remained stable during photodegradation of stream
459 waters and piezometer and strongly decreased in the peatland pool (**Fig. S10)**. The E_{254}:E_{436} ratio
460 strongly increased in the peatland pool and exhibited a decrease in stream waters and piezometer,
461 whereas the E_{365}:E_{470} ratio systematically decreased in all photodegradation experiments with
462 the Ilasskoe Bog continuum (**Fig. S11**). Finally, the E_{470}:E_{665} ratio exhibited sizable decrease, in
463 the order 'stream >> pool ≥ piezometer'. The total spectral differences between experimental and
464 control reactors were mostly pronounced in stratified forest lake waters in June (ΔA = -0.4 to -
465 0.4) and in the bog continuum in July, where effects were strongest in the piezometer and outlet
466 stream waters (ΔA parameter as high as – 0.4 (**Fig. S12**).
467
468 3.3.3. TE in photodegradation experiments
469 The elements affected by photodegradation also formed three groups similar to those
470 impacted by biodegradation. Concentrations of Al, Fe, trivalent and tetravalent hydrolysates (Ti,
471 Ga, Zr, Y, LREE and Th) and Nb of [Group 1] significantly (> 2 %; $p < 0.05$) decreased during
472 photolysis as illustrated for Fe in **Fig. 6,** and for Ti and Zr in **Figs. S13** and **S14,** respectively.
473 The decrease of Fe was mostly pronounced in Lake Temnoe water from 10 m depth, whereas
474 that of Ti and Zr was detectable for all horizons and seasons except in October. For the Ilasskoe
475 Bog continuum, there was no systematic change in Fe concentration, whereas concentrations of
476 Ti and Zr systematically decreased over the course of sunlight exposure (**Figs. S13, S14**). Alkali
477 (Li, Rb), alkaline-earth metals (Mg, Ca, Sr, Ba), Si and oxyanions (As, Mo, Sb) of [Group 2]
478 were weakly (< 2 %) affected by photolysis. Finally, the remaining trace elements of [Group 3]
479 did not exhibit any systematic evolution of concentration during exposure to sunlight, or these
480 changes were inferior to the uncertainties of replicates (X < S.D. in Eqn. 1).

We found that, unlike for DOC, the magnitude of trace element concentration decrease during photodegradation was generally lower than that of biodegradation experiments. Overall, the strongest effects were observed for Ti (3 to 9% in Lake Temnoe; 20% in Ilasskoe Bog), Ga (6 to 14%), Zr (14-17% in Lake Temnoe), Nb (8 to 13%) and Th (8 to 19% in the Temnoe Lake and up to 50% in the Ilasskoe Bog). These effects were mostly pronounced in the Temnoe Lake in June and August and in peatland pool of the Ilasskoe Bog (July).

4. Discussion

4.1. Comparison between biodegradation and photolysis

The impact of season on the biodegradable DOC could be tested only for Lake Temnoe because it was sampled during the 3 main hydrological periods. The maximal biodegradation of the lake water was observed during autumn, when large amount of labile fresh soil OM and plant litter were delivered to the lake from the watershed via surface runoff. The water temperature seems to be of secondary importance for the intensity and rate of DOM biodegradation. This is also confirmed by lack of statistically significant (at $p < 0.05$) correlation between water temperature and BDOC parameters (overall magnitude and rate). It is worth noting that the seasonal pattern of BDOC in the humic lake quantified in this study (**Fig. 4 A**) contrasted with previous works on biodegradation of large Arctic streams and rivers whose BDOC decreased as the Arctic summer progressed (Vonk et al., 2015). Presumably, the input of fresh plant litter from the forested watershed of Lake Temnoe provided elevated biodegradation in the water column at the end of the open water season. Another reason could be due to lake overturn in October and exposure of deep, partially autochthonous, and thus biodegradable, DOM to the surface horizons. A supply of limiting nutrients (N and P) to the upper 0-10 m layer during lake overturn could also promote such biodegradation in October.

505 The highest biodegradation rates in the uppermost sections of the bog hydrological
506 continuum (piezometer, **Fig. 3 A**) are consistent with recent findings on organic-rich waters of
507 permafrost peatlands (Shirokova et al., 2019; Payandi-Rolland et al., 2020) and earlier results on
508 headwaters, small streams and soil leachates (Roehm et al., 2009; Ilina et al., 2014; Mann et al.,
509 2014, 2015; Larouche et al., 2015; Spencer et al., 2015; Vonk et al., 2015; Moody et al., 2013;
510 Pickard et al., 2017; Dean et al., 2019). This could be due to the very short water residence time
511 and freshly leached DOM in these water objects (i.e., Mann et al., 2012; Abbott et al., 2014;
512 Payandi-Rolland et al., 2020), given that bioavailable DOM components leached from plant litter
513 are rapidly utilized (Textor et al., 2018). At the same time, overly low BDOC (2-8 %) values,
514 regardless of depth and season in humic lake and across the hydrological continuum of the bog
515 (**Fig. 2 A**), are supportive of previous results for permafrost peatlands from the neighboring
516 region (Shirokova et al., 2019). A general path for DOM spectral properties modification over
517 the course of biodegradation consisted of an increase in aromaticity of DOM due to preferential
518 uptake of non-humic low molecular weight (LMW) compounds. However, this was not
519 accompanied by a sizable increase in SUVA (**Fig. S2**). Presumably, the proportion of these
520 compounds in the overall DOC level was quite low and could not impact SUVA evolution.
521 Globally, the evolution of optical ratios was consistent with bacterial consumption of aliphatic
522 LMW compounds and an increase in the overall aromaticity of DOM.
523 Concerning the seasonal variation of photodegradation in the deep humic lake, maximal
524 effects were observed in June. These effects likely occurred due to fresh terrestrial organic matter
525 that was leached from the watershed and then efficiently processed during Arctic summer. It
526 should be noted that labile phenolic, carbohydrates, N-containing bases and smaller molecular
527 weight compounds are abundant in litter leachates produced during initial decay stages (Kiikkilä
528 et al., 2011, 2012, 2013; Hensgens et al., 2021). By July, most of the biodegradable DOM was
529 already removed, and in October, the effects were much lower. Therefore, photolabile DOM is

delivered from the forested watershed to the lake essentially during surface flux, at high water flow. It is then quickly removed from the water column, which was especially seen in the 0.5 and 5 m horizons of Lake Temnoe. Although labile organic matter from litter fall was also delivered during autumn rain season, presumably, during this period, the conditions for photolysis (low temperature, short daytime period and insufficient light) were not as favorable as those in June or August.

Photodegradation of waters from the Ilasskoe Bog continuum demonstrated maximal rates in the piezometer (**Fig. 3 B**). During photolysis of humic water, a decrease in optical ratios (E_{365}:E_{470}; E_{470}:E_{665}) clearly indicated preferential degradation of humic aromatic compounds. The strong effect of photodegradation on DOM optical properties in the 650-500 nm region may be linked to decomposition of complex DOM into smaller molecules, whereas a decrease of absorbance in the 230-400 nm region **(Fig. S12**) indicates degradation of aromatic compounds, progressively increasing over insolation time. A recent study of DOM photolysis in humic-rich forested streams demonstrated that high aromatic material was photochemically converted into smaller non-fluorescent molecules (Wilske et al., 2020).

Results obtained on the more important role of photodegradation over biodegradation are generally consistent with earlier reports on the dominance of photolysis for DOM processing in Arctic waters within North America (Cory et al., 2014; Ward et al., 2017), the Canadian temperate zone (Winter et al., 2007; Porcal et al., 2013, 2014, 2015), and Swedish headwater catchments (Köhler et al., 2002). According to former results for Scandinavian surface waters, the main impact of DOM photolysis is reflected by a decrease in the proportion of aromatic (colored) DOC and a rather small (≤ 10 %) change in bulk DOC concentration (Groeneveld et al., 2016; Koehler et al., 2014), Canada (Laurion and Mladenov, 2013; Gareis and Lesack, 2018) and NW Russia (Oleinikova et al., 2017; Chupakova et al., 2018).

As a further perspective of this work, one has to consider biodegradation of photolytically altered DOM given that photo-oxidation is known to transform molecular structures into more bioavailable forms (e.g., Cory and Kling, 2018; Sulzberger et al., 2019) thereby stimulating microbial growth under sunlight, as is known for other Arctic and subarctic settings (i.e., Drozodova et al., 2020; Laurion et al., 2020).

4.2. Possible impact of microbial and photolytic processing on CO_2 emissions from water surfaces

A broad importance of DOM bio- and photodegradation dynamics is that these processes can contribute to CO_2 emissions from water surfaces thereby directly controlling the C cycling between the land and the atmopshere (Lapierre et al., 2013; Tranvik et al., 2009; Cory et al., 2014, 2018). In this study, we attempted to relate, for the first time for several diverse aquatic systems across seasons, experimentally measured rates of DOM degradation to in-situ measured CO_2 emissions. The integral rates of DOM bioprocessing in the water column of Lake Temnoe (**Table 3, Fig. 4 A**) allow quantifying the potential contribution of biodegradation to CO_2 production and emission. Assuming all biodegraded DOM is transformed into CO_2 and there is no biomass increase or sedimentation, a 1 m water layer of the lake can emit 1.7 mmol CO_2 m^{-2} d^{-1} in June and 3.3 mmol CO_2 m^{-2} d^{-1} in October. Therefore, integral flux from 10 m deep water layer amounts to 17 – 33 mmol CO_2 m^{-2} d^{-1} across the seasons. These values are comparable to typical values of CO_2 evasion from the surface of this lake during different seasons (30-70 mmol CO_2 m^{-2} d^{-1}; **Table 1 B**). For surface waters of Ilasskoe Bog, maximal CO_2 production due to DOM biomineralization alone (**Table 3**) ranged from 5.0 mmol CO_2 m^{-2} d^{-1} for the peatland pool (2 m deep) to 2.5 mmol CO_2 m^{-2} d^{-1} for the outlet stream (0.5 m deep). However, in summer, the peatland pool and stream emitted 23 and 150 mmol CO_2 m^{-2} d^{-1} (**Table 1 A**) which could not be sustained by DOM biodegradation.

579 The addition of photodegradation (assuming a photic layer depth of 3.5 m) to DOM
580 bioprocessing in the water column of the Temnoe Lake during open water season can further
581 increase potential CO_2 production in the water column. For the case of Ilasskoe Bog waters, the
582 addition of photolytic degradation increases projected CO_2 emission from the outlet stream by a
583 factor of 5, which is still below the actual CO_2 flux, whereas DOM photolysis has no impact on
584 CO_2 emissions from the peatland pool. Note that, although the depth of sunlight processing in
585 boreal waters is typically 1-0.8 m (Vähätalo et al., 2000; Koehler et al., 2014), a more recent
586 study concluded that direct photomineralization of DOM in Artic humic ponds is limited to the
587 first centimeters of the water column (Mazoyer et al., 2022). Furthermore, in typical DOM-rich
588 Arctic waters, only half of sunlight-associated DOC losses is converted into CO_2 and the rest
589 may be turned into particles through photoflocculation (e.g., Mazoyer et al., 2022). Therefore,
590 despite a faster photodegradation rate compared to biodegradation, due to the shallow photic
591 layer in humic waters, the biodegradation may provide the largest impact on CO_2 emission from
592 the water column of boreal waters.

593 At the same time, our assumption that all CO_2 in lake water is produced by bio- or
594 photodegradation of DOM might not be warranted because there are multiple sources of CO_2 in
595 the lake waters, which were not assessed in the present study. These including but not limited to:
596 particulate organic matter bio- and photodegradation, whose importance can strongly exceed that
597 of DOC (e.g., Attermeyer et al., 2018; Keskitalo et al., 2022), sediment respiration, plankton and
598 periphyton diel photosynthetic cycle, underground water discharge at the lake bottom, and
599 delivery of DOC and CO_2-rich waters via lateral surface and shallow subsurface influx. Given
600 that the contribution of each CO_2 source can vary among different water bodies and across
601 seasons, the assessment of DOM bio- and photodegradation contribution to overall CO_2 flux in
602 this study should be considered as highly conservative.
603

4.3. Impact of DOM bio- and photo transformation on trace element pattern

In this study we hypothesized the following link between DOC and TE: in humic surface waters of peatlands, most TE, which include divalent transition metals (Cu, Ni, Co, Zn, Mn), toxicants (Be, Cr, Cd, Pb), trivalent and tetravalent hydrolysates (Al, Ga, Y, REE, Ti, Zr, Hf, Th), with an exception of some alkalis and oxyanions, are strongly (> 80%) associated to DOM in the form of organic, organo-ferric and organo-aluminium colloids (Pokrovsky et al., 2012, 2016). As a result, any DOM transformation processes, be it bio- or photo-degradation, may directly affect the concentration pattern of TE. Specifically, the DOM removal via photo- or bio-degradation should change the speciation of those elements, that are strongly bound to DOM such as divalent transition metals, or incorporated into organo-mineral (Fe, Al) colloids, such as trivalent and tetravalent hydrolysates (TE^{3+}, TE^{4+}). The former might either remain in solution (during photodegradation), hence not modifying their total dissolved concentration, or being taken up by growing bacteria during bio-degradation. The latter (TE^{3+}, TE^{4+}) are capable of co-precipitating with Fe and Al hydroxides, especially during photodegradation (i.e., Kopacek et al., 2005, 2006), hence being sizably removed from the aqueous solution. From the other hand, some TE are known to be photosensitive (Mn, Fe), toxic (Al, Cu, As, Cd, Pb), or potentially limiting micronutrients (Zn, Co, Ni, Mo) for the bacteria and therefore they are capable affecting the overall rate of photo- or bio-degradation.

However, contrary to our expectations, among all major and trace elements measured in the experiments, only trivalent and tetravalent hydrolysates (TE^{3+}, TE^{4+}) were impacted by both photo- and biodegradation. It is known that these elements are essentially present in the form of large molecular size, highly polymerized and presumbaly aromatic, organo-Fe/Al colloids in humic boreal/subacrtic lakes (Pokrovsky et al., 2012, 2016), rivers (Krickov et al., 2019; Pokrovsky et al., 2010), and soil porewaters (Pokrovsky et al., 2005; Raudina et al., 2021). Therefore, insoluble TE^{3+} and TE^{4+} generally followed the removal of Fe(III) in the form of

particulate Fe hydroxides, after breaking the Fe-DOM bonds that stabilized colloidal Fe(III) hydroxides. This destabilization and Fe hydroxide particle formation is known to occur either via biodegradation (i.e., Oleinikova et al., 2018) or photolysis (Kopacek et al., 2005, 2006; Oleinikova et al., 2017; Chupakova et al., 2018). At the same time, some micronutrients (V, Mn, Co, Cu and Ba) were affected solely by biodegradation. This can reflect uptake of these metals by growing bacterial cells, as is known from laboratory experiments with pure cultures of heterotrophic bacteria (Shirokova et al., 2017a).

Note that the effects of bio- and photodegradation were more pronounced for light REE (LREE) compared to heavy REE (HREE). This result is consistent with the fact that LREE have stronger association with Fe hydroxide compared to organic complexes, as known from general chemical considerations and laboratory experiments (i.e., Bau, 1999) and evidenced in various boreal and subacrtic settings (Pokrovsky et al., 2016; Krickov et al., 2019). Given that the main effect of both photolysis and biodegradation of DOM in humic Fe(III)-rich surface waters is coagulation of dissolved Fe(III) in the form of Fe oxy(hydr)oxides, the LREE are removed from solution. This removal occurs in the form of adsorbed complexes or coprecipitated with Fe oxy(hydr)oxides, while HREE remain in the form of strong aqueous complexes.

In former studies of photo- and biodegradation of surface waters from permafrost peatlands, only a few nutrients (P, Fe, Zn and V) and insoluble low mobility trace metals (Ti, Zr, Nb and Th) demonstrated a decrease in concentration (Shirokova et al., 2019). This list of elements is generally consistent with that established in the present study of humic subarctic lakes of the non-permafrost zone, except P and Zn which did not exhibit sizable removal in our experiments. It is possible that a high proportion of low molecular weight $LMW_{< 1\ kDa}$ (and thus, potentially bioavailable) forms of macro- and micronutrients, such as P and Zn, in the permafrost ice (i.e., Kuzmina et al., 2023) can be delivered to the lake and river via suprapermafrost flow (Raudina et al., 2018, 2021). This led to elevated bioavailability of these elements in permafrost

surface waters reported in previous works, as compared to permafrost-free boreal settings of this study.

Conclusions

Seasonally resolved bio- and photo-degradability of DOM in a deep stratified lake and summer measurements from a peat bog's hydrological continuum within the boreal zone demonstrated that the subsurface and deep horizons of these stratified waters are mostly sensitive to sunlight impact, and that maximal effects of photodegradation occurred in June, during strong insolation. In contrast, the biodegradation of DOM from the humic lake was mostly pronounced during October, when fresh leachates of forest litter were exported from the watershed. Insoluble, low-mobility trace metals such as trivalent and tetravalent hydrolysates were affected by both bio- and photodegradation, as they are associated with coagulating Fe(III) oxyhydroxides.

A broad implication of obtained results is that, although DOM photodegradation rates were sizably higher compared to those of biodegradation, the rather thin photic layer in humic waters does not allow for significant contribution of photolysis in overall CO_2 emission from lake and bog surfaces. Further work is needed on biodegradation of photolytically altered DOM given that photo-oxidation is known to transform molecular structures into more bioavailable forms. The high seasonal dynamics and spatial variability in both photo- and biodegradability of DOM and related trace elements of humic surface waters in the boreal zone encountered in this study suggest the need for studying these processes during "shoulder seasons" (early spring and late autumn), the periods of maximal photo- and biodegradation, respectively. These efforts should be focused on the most dynamic components such as small streams and subsurface waters, which demonstrated the highest rates of both photo- and biodegradation.

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Authors contribution.

AVC and OP designed the study and wrote the paper; AC, NN and SB performed sampling, analysis and their interpretation; LS performed bacterial number assessment and DOC results interpretation; AVC, TV and OP provided analyses of literature data.

Competing interests.

The authors declare that they have no conflict of interest.

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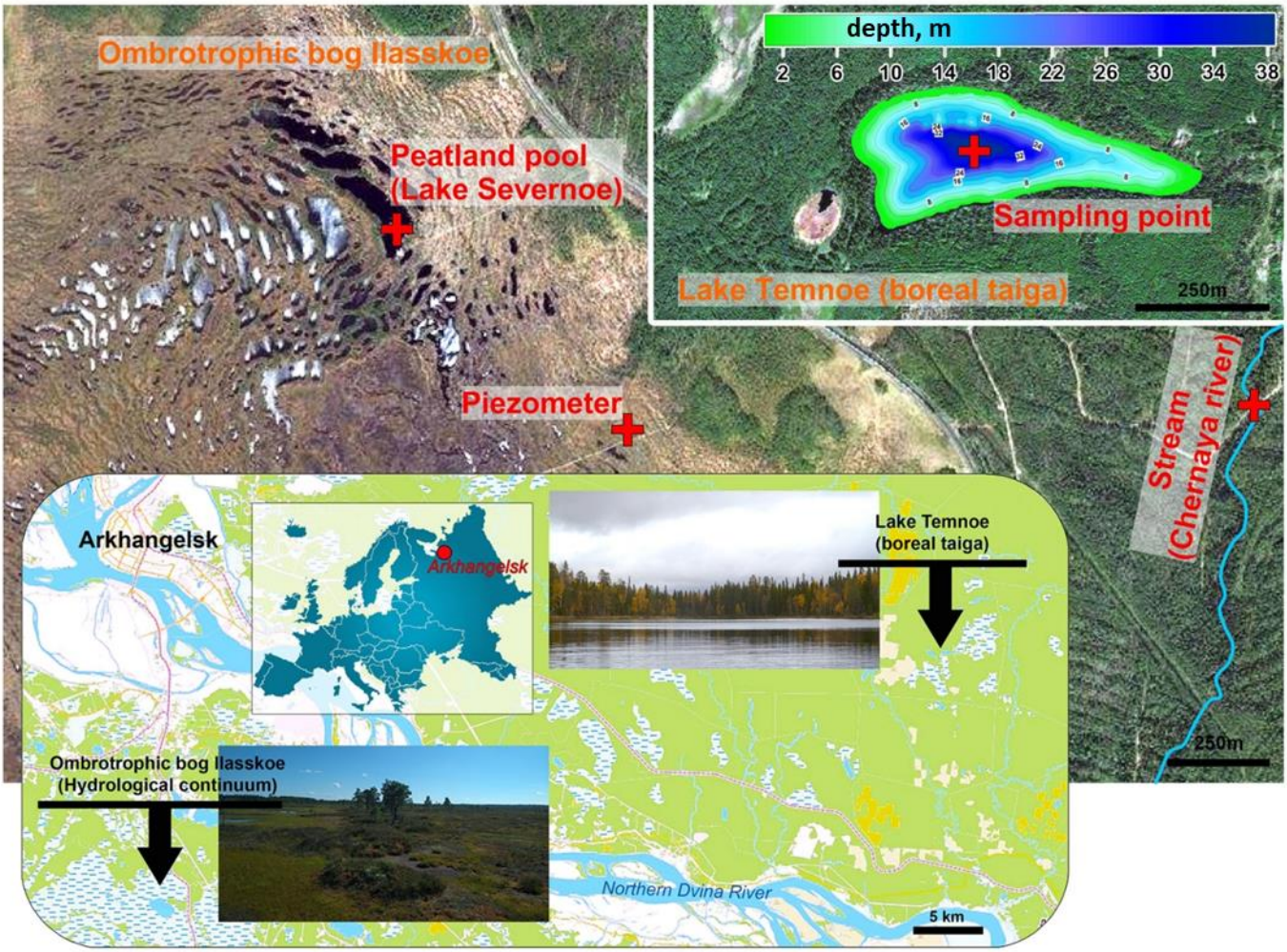

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1079 **Fig. 1.** Geographical location of studied hydrological continuum for Ilasskoe Bog waters and
1080 deep stratified Lake Temnoe in the boreal forest. Photo and map credits of Chupakov A.V.

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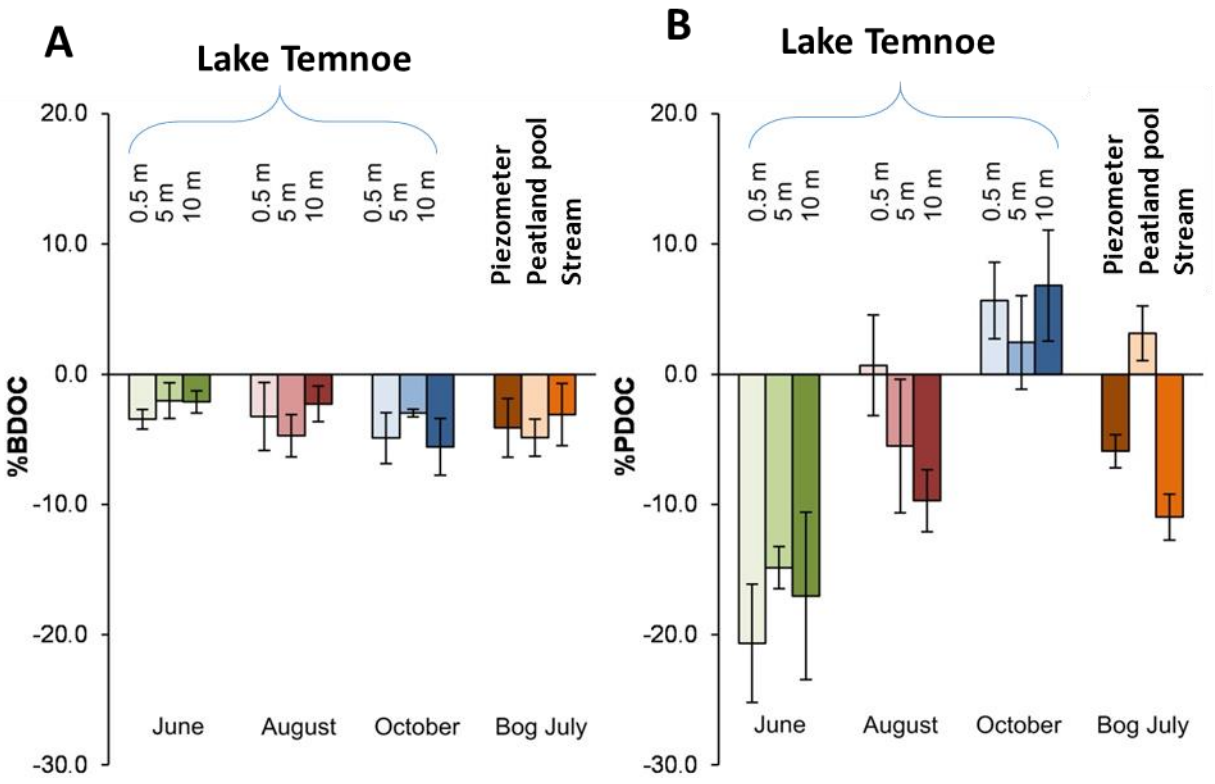

Fig. 2. Percentage of bio- (A) and photo- (B) degradable DOC presented as relative decrease in DOC concentration between the initial and final value for the Temnoe Lake (June, August and October) and Ilasskoe Bog surface waters (July). Error bars are 1 s.d. of duplicates relative to the control (see Eqn. 1-2 in the text). In accord with unified protocol of biodegradation experiments (Vonk et al., 2015), positive values signify nil photodegradation (experimental artifacts of DOC production).

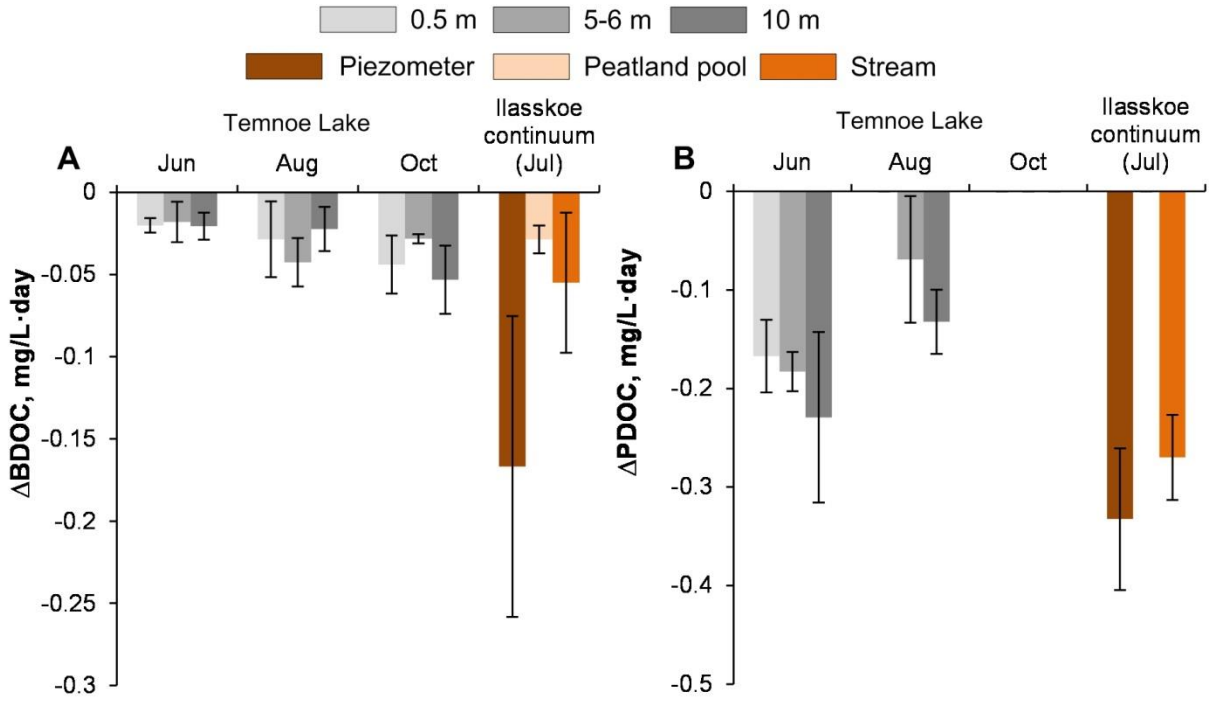

Fig. 3. Rates of DOC bio- (A) and photo- (B) degradation. The values are negative because they represent a decrease in DOC concentration over the course of the experiment.

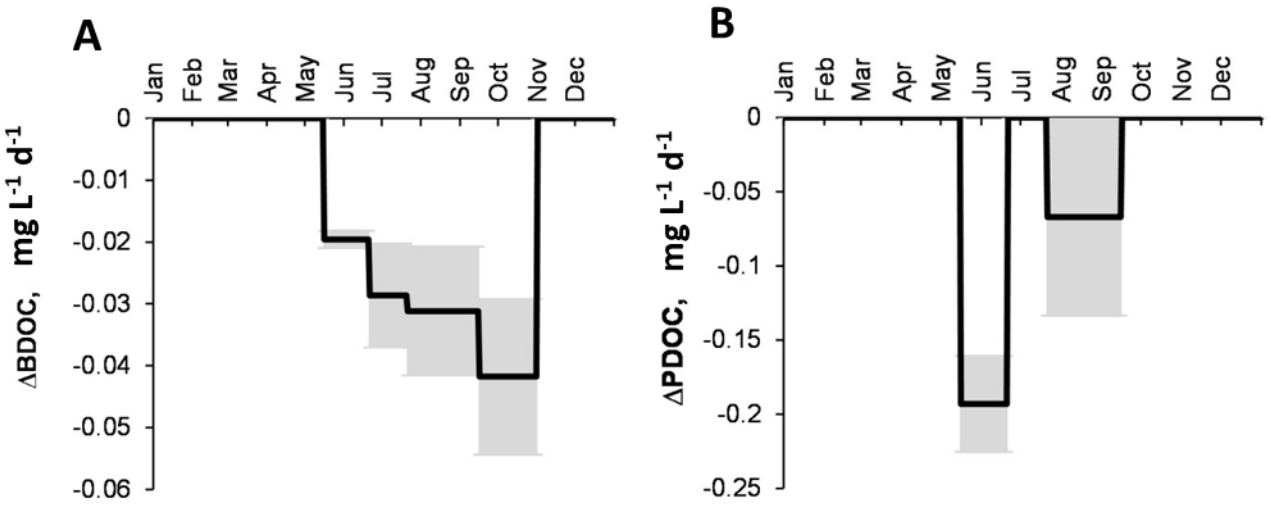

Fig. 4. Integral rates of bio- (ΔBDOC, **A**) and photo- (ΔPDOC, **B**) degradation in the 0-10 m layer of Lake Temnoe across the entire open-water period (May to October). Rate values are negative because they signify a decrease in DOC concentration. Note that there was no sampling from December to April and the photodegradation was not studied in July. Uncertainties are represented by gray shaded rectangles.

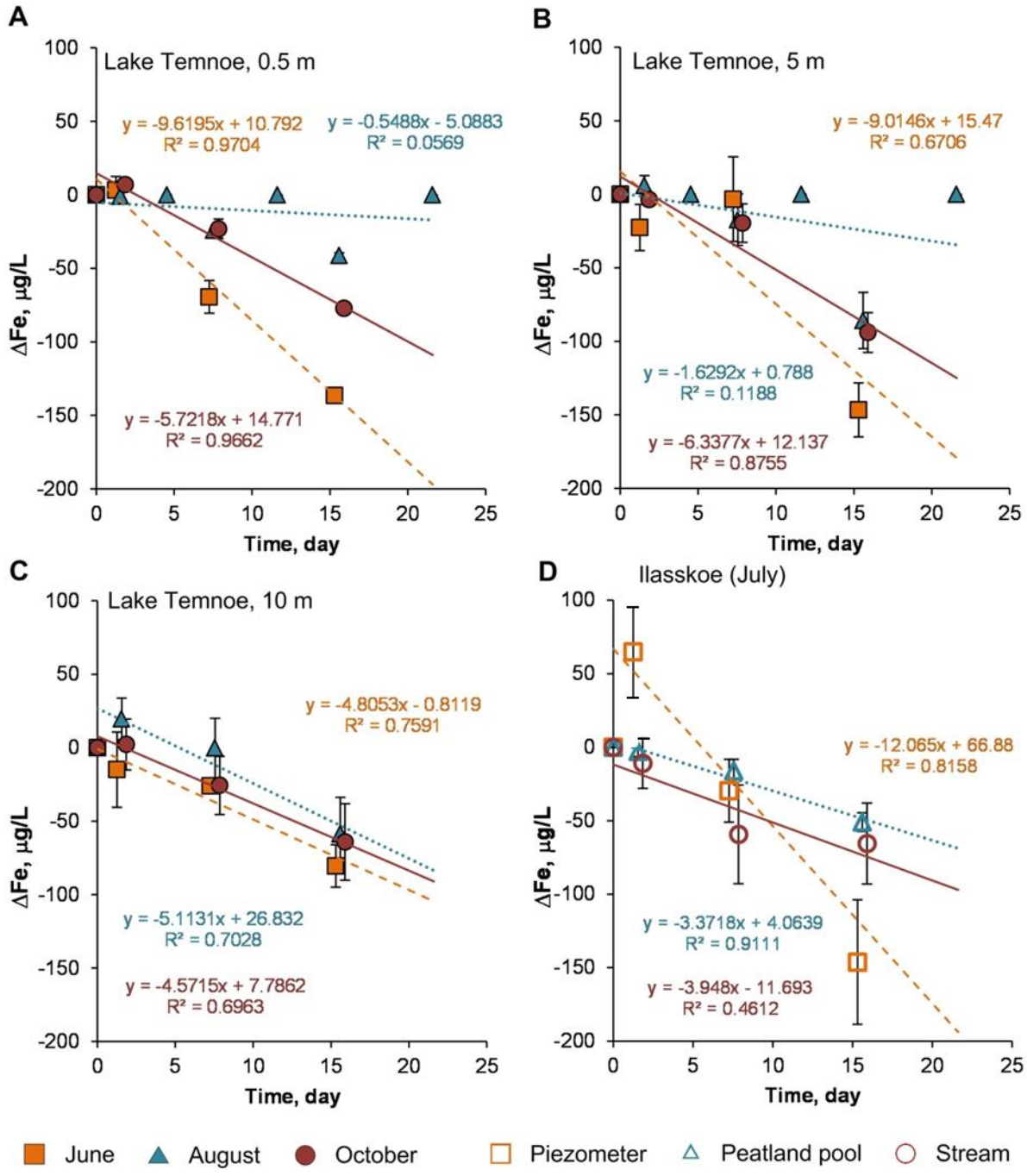

Fig. 5. Change in Fe concentration (relative to control) over time in biodegradation experiments. Error bars are 1 s.d. of duplicates. Temnoe Lake 0.5 m (A), 5 m (B) and 10 m (C) in June (squares), August (triangles) and October (circles). Ilasskoe Bog continuum in July (D) including piezometer (squares), Severnoe peatland pool (triangles) and stream Chernyi (circles).

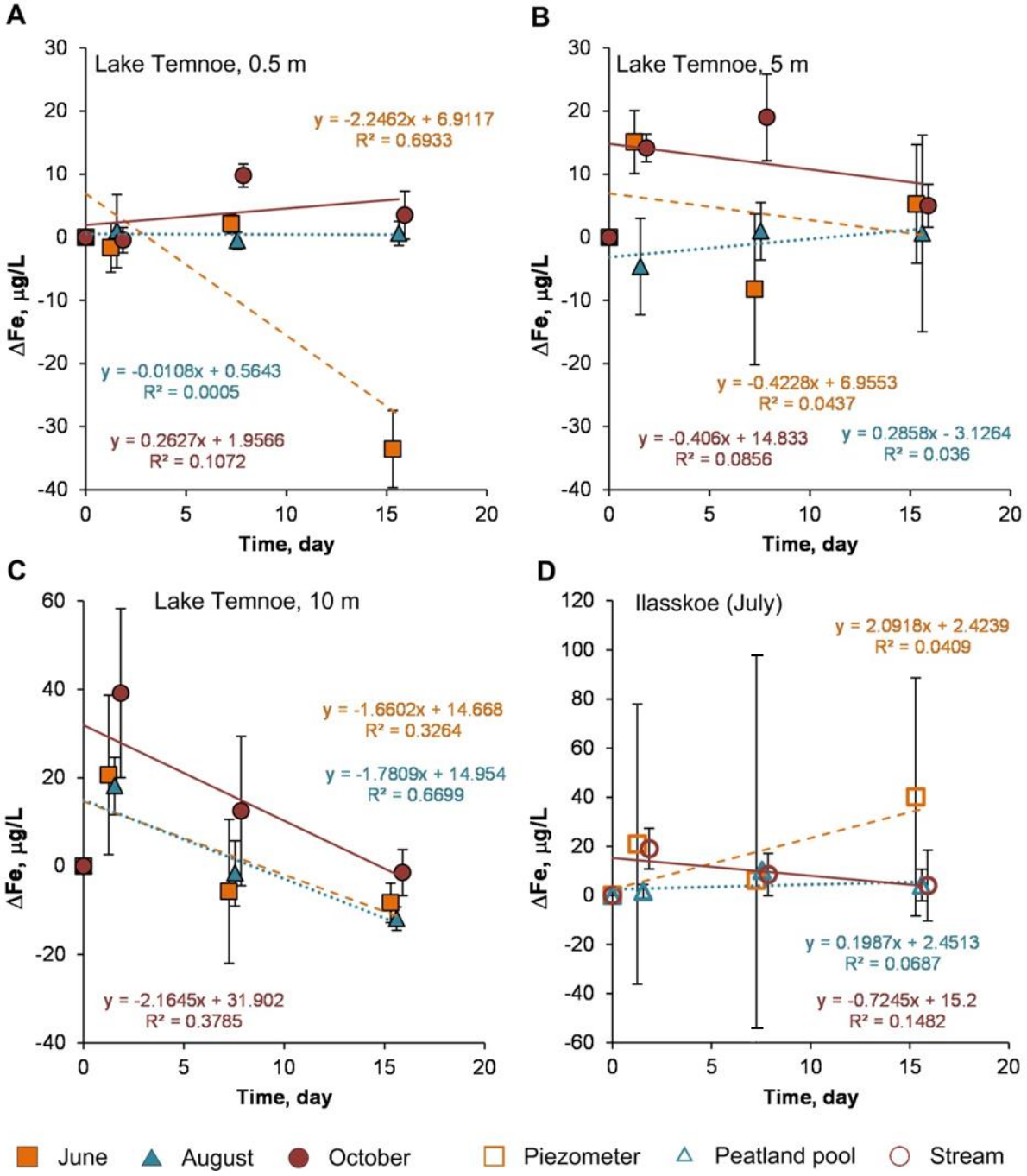

Fig. 6. Change in Fe concentration (relative to the control) over time in photo-degradation experiments. The error bars are 1 s.d. of duplicates. Lake Temnoe 0.5 m (A), 5 m (B) and 10 m (C) in June (squares), August (triangles) and October (circles). Ilasskoe continuum in July (D) includes piezometer (squares), peatland pool Severnoe (triangles) and stream Chernyi (circles)

Table 1. Landscape setting, hydrochemical characteristics and CO_2 concentration and emission flux of studied waters. S.C. is specific conductivity and EB and OB is eutrophic and oligotrophic bacteria count, respectively.

1A. Ilasskoe bog continuum in July.

	Piezometer	Lake Severnoe	Stream Chernyi
GPS coordinates	N64.328694° E40.612556°	N64.334361° E40.609667°	N64.330982° E40.653352°
Description	Shallow groundwater	Peatland pool	Outlet stream
T,°C	11.4	19.4	13
O_2, mg/L	0.6	8.6	7.5
pH	3.9	4.0	5.7
S.C., µS cm^{-1}	46	17	26
DOC, mg L^{-1}	87.6	12.7	38.4
DIC, mg L^{-1}	0.32	0.40	0.38
$SUVA_{254}$	4.13	3.80	4.85
P-PO_4, µg L^{-1}	8.6	3.0	1.7
P_{total}, µg L^{-1}	153	10	20
N-NO_3, µg L-1	111	70	98
N-NH_4, µg L^{-1}	85.4	16.1	12.6
N_{total}, µg L^{-1}	1180	222	399
Si, µg L^{-1}	1808	47	2076
CO_2, µmol/L	3360	55	318
CO_2 flux, mmol m^{-2} d^{-1}	1600	22	151
EB, CFU mL^{-1}	49360	56600	9000
OB, CFU mL^{-1}	54560	37900	21600

1B. Lake Temnoe across seasons and depths.

Month	Jun	Jun	Jun	Aug	Aug	Aug	Oct	Oct	Oct
	0.5	5	10	0.5	5	10	0.5	6	10
GPS	N64.47683° E041.74533°								
Description	Lake in the northern taiga								
T,°C	12.7	4,9	4,5	18.4	5.5	4.3	9.0	5.8	4.4
O2, mg/L	8,45	4,8	4,5	7.78	4.93	2.63	8.90	4.46	2.14
pH	5.2	5.2	5.3	6.0	5.5	5.7	5.2	5.2	5.1
S.C., µS cm^{-1}	17	17	19	17	17	19	18	18	20
DOC, mg L^{-1}	12.6	19.2	21	19	19.5	21.2	19.4	20.6	20.6
DIC, mg L^{-1}	0.55	0.53	0.49	0.70	0.71	0.66	0.67	0.67	0.64
$SUVA_{254}$	4.6	4.7	4.6	4.2	4.5	4.5	4.3	4.3	4.7
P-PO_4, µg L^{-1}	2.9	3.3	6.4	0.9	3.6	9.4	3.8	4.6	4.2
P_{total}, µg L^{-1}	19	17	19	20	16	20	18	19	20
N-NO_3, µg L-1	119	150	137	86	152	254	88	85	100
N-NH_4, µg L^{-1}	7.1	8.0	10.0	9.1	17.5	13.8	16.4	14.1	15.5
N_{total}, µg L^{-1}	305	420	408	355	315	337	425	416	396
Si, µg L^{-1}	1940	2268	2354	1183	2208	2714	2269	2380	2380
CO_2, µmol/L	99	309	329	110	256	337	223	232	253
CO_2 flux, mmol m^{-2} d^{-1}	32	-	-	46	-	-	71	-	-
EB, CFU mL^{-1}	-	36	50	259	92	270	780	220	105
OB, CFU mL^{-1}	50	570	420	-	190	-	680	150	66

Table 2. The % bio- and photodegradable solutes (mean ± s.d.) whose relative change (concentration decrease) in the course of experiment was superior to that of SD. Prefix ΔB and ΔP represents the effect of bio- and photodegradation, respectively. Duration of biodegradation and photodegradation is 21.6±0.1 and 15.6±0.1 days, respectively. W represents the probability of measurable effect, significantly different from changes in the control reactors. Only the components with W ≥ 33% are presented. Temnoe Lake is deep stratified lake in the forest. Peizometer, peatland pool and outlet stream represent the hydrological continuum of the Ilasskoe Bog.

Index	Temnoe Lake 0.5 m (Jun)	Temnoe Lake 5 m (Jun)	Temnoe Lake 10 m (Jun)	Temnoe Lake 0.5 m (Aug)	Temnoe Lake 5 m (Aug)	Temnoe Lake 10 m (Aug)	Temnoe Lake 0.5 m (Oct)	Temnoe Lake 6 m (Oct)	Temnoe Lake 10 m (Oct)	Piezo-meter (Jul)	Peatland pool (Jul)	Outlet stream (Jul)
æ, µS/cm	17	17	19	17	17	19	18	18	20	46	17	26
ΔB(æ ±SD)	-24±4	-26±7	-30±5	-23±3	-27±3	-33±16	-24±7	-23±4	-17±5	0	-18±10	-29±4
DOC, mg/L	12.6	19.2	21.0	19.0	19.5	21.2	19.4	20.6	20.6	87.6	12.7	38.4
ΔB(DOC±SD)	3.4±0.8	2.0±1.4	2.1±0.8	3.2±2.6	4.7±1.6	2.3±1.4	4.9±2.0	3.0±0.3	5.6±2.2	4.1±2.3	4.9±1.4	3.1±2.4
ΔP(DOC±SD)	20.7±4.6	14.9±1.6	17.0±6.4	0	5.5±5.1	9.7±2.4	0	0	0	5.9±1.3	0	11.0±1.8
Al, µg/L	275	298	329	254	296	335	275	288	323	276	59	388
ΔB(Al±SD)	3.5±1.4	1.8±0.9	0	2.0±1.3	0	1.4±1.5	2.0±1.9	0	0	0.9±2.2	0	1.3±1.8
ΔP(Al±SD)	1.9±1.1	2.7±0.9	3.6±1.3	0	2.5±1.3	1.7±2.0	0.7±0.9	0	0	0	0	0.8±0.9
Ti, µg/L	1.5	2.1	2.6	1.1	2.0	2.6	1.7	1.9	2.5	3.7	0.6	5.0
ΔB(Ti±SD)	-9.2±1.6	-9.9±7.4	-2.6±2.7	-4.8±3.4	-1.8±2.7	0	-3.6±1.7	-1.0±3.1	-1.0±3.9	-2.3±3.6	-2.2±1.7	-1.4±2.2
ΔP(Ti±SD)	-0.1±3	-3±3	-8±3	0±0	-9±1	-3±2	-2±4	0	0	0	-20±4	-3.3±0.5
V, µg/L	0.5	0.6	0.7	0.4	0.5	0.7	0.4	0.5	0.7	1.1	0.5	1.3
ΔB(V±SD)	-8.3±16.2	-5.4±3.2	-4.9±2.3	-6.8±7.5	-10.0±4.6	-1.7±1.6	-14.7±11	-13.9±4.3	-16.1±1.7	-3.2±2.6	-0.2±3.4	-17.9±5.0
Mn, µg/L	39	55	79	17	48	93	30	47	105	78	9	47
ΔB(Mn±SD)	0	0	-0.3±2.2	-31.8±1.3	-3.2±1.6	-0.6±2.2	-4.8±2.2	-3.2±1.7	-0.4±0.1	0	0	-1.6±2.8
Fe, µg/L	358	527	710	165	460	795	317	448	820	4402	157	1006
ΔB(Fe±SD)	-18.1±2.5	-9.1±2.6	-5.4±1.6	-13.5±1.0	-6.3±2.6	-1.4±1.9	-9.5±1.4	-7.8±1.9	-3.3±1.8	-0.8±0.8	-13.6±4.3	-4.5±2.4
ΔP(Fe±SD)	-3.9±0.6	-2.0±1.9	-4.0±1.3	0	-2.9±1.5	-0.2±0.6	-1.2±0.4	0	0	0	0	0

1171 **Table 2**, continued.

Index	Temnoe Lake 0.5 m (Jun)	Temnoe Lake 5 m (Jun)	Temnoe Lake 10 m (Jun)	Temnoe Lake 0.5 m (Aug)	Temnoe Lake 5 m (Aug)	Temnoe Lake 10 m (Aug)	Temnoe Lake 0.5 m (Oct)	Temnoe Lake 6 m (Oct)	Temnoe Lake 10 m (Oct)	Piezo-meter (Jul)	Peatland pool (Jul)	Outlet stream (Jul)
Co, µg/L	0.28	0.39	0.68	0.07	0.30	0.65	0.18	0.31	0.74	0.45	0.06	0.30
ΔB(Co±SD)	-2.2±5.1	-1.2±2.1	-3.7±4.6	-32.7±2.6	-8.1±5.6	-2.7±3.3	-11.0±4.4	-9.1±5.1	-1.6±0.4	0	0	-20.6±27.8
Cu, µg/L	0.5	0.6	0.7	0.6	0.6	0.7	0.7	0.7	0.5	1.5	0.3	0.8
ΔB(Cu±SD)	0	0	0	-14.3±1.4	-6.8±4.0	-17.9±11.0	-5.3±4.8	-4.1±8.0	-1.4±12.3	0	-7.7±9.9	0
Ga, µg/L	0.017	0.022	0.026	0.012	0.016	0.023	0.017	0.015	0.024	0.126	0.016	0.066
ΔP(Ga±SD)	-14±6	-13±5	-10±4	0	-1±8	0	-10±4	0	0	-7±5	-5±8	-6±3
Y, µg/L	0.22	0.25	0.28	0.20	0.24	0.28	0.22	0.23	0.28	0.10	0.01	0.21
ΔP(Y±SD)	-1.3±4.6	-6.7±0.9	-5.3±2.5	0	-2.3±0.7	-1.0±1.6	-1.4±0.2	0	0	0	-5.8±2.7	0
Zr, µg/L	0.4	0.4	0.5	0.4	0.5	0.5	0.4	0.4	0.5	0.3	0.1	0.4
ΔP(Zr±SD)	-15±4	-14±0	-13±2	-9±20	-17±1	-14±3	-4±4	0	0	0	-32±3	-8±1
Nb, µg/L	0.016	0.020	0.025	0.012	0.020	0.024	0.017	0.018	0.025	0.033	0.005	0.042
ΔB(Nb±SD)	-3.6±10.2	-1.7±7.0	0	-7.7±4.8	-1.1±2.6	0	-7.3±2.3	-1.5±6.3	-5.0±4.0	-2.4±1.5	0	0
ΔP(Nb±SD)	-9±3	-8±3	-9±1	-6±23	-13±2	-10±5	-8±4	0	-3±3	0	-13±10	-10±4
Ba, µg/L	4.8	5.1	5.8	4.6	5.0	5.7	4.9	4.8	5.6	54.4	1.5	56.8
ΔB(Ba±SD)	-2.2±0.7	-2.8±1.7	-1.0±2.7	0	0	0	-1.9±0.5	-1.7±3.7	-5.9±1.6	0	0	-1.3±3.4
La, µg/L	0.23	0.26	0.30	0.21	0.26	0.32	0.24	0.27	0.31	0.07	0.01	0.22
ΔB(La±SD)	-4.9±6.5	0	0	-3.9±0.9	-0.3±1.6	-2.6±1.5	-1.1±3.6	-2.4±1.2	-4.0±2.8	-0.8±10.4	-29.7±10.0	-2.0±2.8
ΔP(La±SD)	-3.8±3.6	-1.2±5.9	-2.0±2.7	0	-3.6±1.0	-3.2±1.6	-1.8±0.9	-2.6±1.0	0	0	-13.5±3.0	0
Ce, µg/L	0.58	0.65	0.71	0.50	0.62	0.78	0.59	0.63	0.78	0.21	0.03	0.56
ΔB(Ce±SD)	-5.2±4.2	0	0	-4.4±0.7	-0.1±1.4	-0.8±1.0	-0.9±2.9	-0.8±1.1	-2.2±2.2	0	-9.3±5.9	-2.0±1.9
ΔP(Ce±SD)	-4.9±1.9	-6.2±1.2	-1.1±1.7	0	-1.5±0.5	-3.2±1.7	-1.9±1.2	0	-0.047±0.46	-3.7±1.5	-2.4±1.6	-3.6±1.1

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1175 **Table 2**, continued.

Index	Temnoe Lake 0.5 m (Jun)	Temnoe Lake 5 m (Jun)	Temnoe Lake 10 m (Jun)	Temnoe Lake 0.5 m (Aug)	Temnoe Lake 5 m (Aug)	Temnoe Lake 10 m (Aug)	Temnoe Lake 0.5 m (Oct)	Temnoe Lake 6 m (Oct)	Temnoe Lake 10 m (Oct)	Piezo-meter (Jul)	Peatland pool (Jul)	Outlet stream (Jul)
Pr, µg/L	0.075	0.085	0.094	0.069	0.082	0.105	0.077	0.084	0.102	0.027	0.005	0.070
ΔB(Pr±SD)	-1.9±5.2	0	0	-4.0±1.1	0	-0.7±1.3	0	-0.9±1.6	-3.2±2.3	-3.0±1.7	-10.8±8.4	-1.7±2.3
ΔP(Pr±SD)	-3.0±3.9	-5.5±0.9	-0.4±2.9	13.7±20.8	0	-6.1±2.8	0	-1.3±1.8	0	-0.01±2.4	-16.9±3.0	-2.4±2.1
Nd, µg/L	0.33	0.34	0.39	0.29	0.33	0.42	0.33	0.32	0.41	0.11	0.02	0.27
ΔP(Nd±SD)	-7.8±2.4	-4.5±2.5	-0.8±2.1	0	0	-3.8±2.5	-2.0±3.3	0	0	-2.5±2.8	-24.7±6.2	0
Eu, µg/L	0.015	0.017	0.016	0.012	0.016	0.020	0.014	0.018	0.021	0.011	0.001	0.017
ΔP(Eu±SD)	0	-10.9±3.3	0	0	-0.8±4.7	-3.0±4.0	0	-8.7±8.4	-6.9±4.2	-23.7±8.6	-58.2±15.2	-1.0±1.0
Gd, µg/L	0.06	0.07	0.08	0.05	0.07	0.08	0.06	0.07	0.09	0.02	0.00	0.06
ΔP(Gd±SD)	-0.2±4.4	-6.5±2.9	-3.5±3.5	0	-6.0±3.2	-7.6±1.9	-2.3±2.0	0	-6.7±3.2	0	0	-3.2±3.5
Ho, µg/L	0.009	0.009	0.011	0.007	0.009	0.011	0.009	0.009	0.011	0.004	0.0004	0.009
ΔP(Ho±SD)	-1.9±1.5	-0.3±1.6	-4.6±6.5	0	-0.1±3.6	-3.4±3.2	-11.0±4.9	0	0	-21.5±11.3	0	-10.9±5.5
Er, µg/L	0.023	0.025	0.033	0.022	0.026	0.030	0.022	0.023	0.031	0.011	0.001	0.023
ΔB(Er±SD)	0	0	0	-5.2±3.0	-2.1±1.8	0	-2.1±3.5	0	-2.0±5.3	-15.6±4.9	-22.9±19.5	0
Pb, µg/L	0.23	0.24	0.23	0.16	0.23	0.39	0.28	0.28	0.32	11	0.35	0.65
ΔB(Pb±SD)	0	0	0	-21.3±2.5	-2.0±7.6	-2.4±1.6	-8.2±3.3	-7.2±9.9	-0.8±2.7	0	-17.4±1.0	-8.5±10.9
Th, µg/L	0.046	0.052	0.066	0.058	0.054	0.064	0.053	0.054	0.061	0.019	0.005	0.050
ΔP(Th±SD)	0	0	-11.6±2.6	-12.2±22.5	-7.8±3.2	-18.1±5.6	0	0	-2.0±1.9	0	-49.5±1.3	-10.6±0.8

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Table 3. Mean (±SD), depth-integrated rates of bio- and photodegradation (mg C $L^{-1}d^{-1}$)

Object	$V_{Biodegradation}$	$V_{Photodegradation}$
Lake Temnoe		
Forest Lake (Jun)	-0.02±0.0014	-0.19±0.03
Forest Lake (Aug)	-0.031±0.010	-0.067±0.066
Forest Lake (Oct)	-0.042±0.013	0
Ilasskoe Bog continuum (July)		
Piezometer water	-0.17±0.09	-0.33±0.07
Peatland pool	-0.029±0.008	0
Outlet stream (Chernyi)	-0.055±0.043	-0.27±0.043