



# Quality transformation of dissolved organic carbon during water transit through lakes: contrasting controls by photochemical and biological processes

Martin Berggren<sup>1</sup>, Marcus Klaus<sup>2</sup>, Balathandayuthabani Panneer Selvam<sup>1</sup>, Lena Ström<sup>1</sup>, Hjalmar Laudon<sup>3</sup>, Mats Jansson<sup>2</sup>, and Jan Karlsson<sup>2</sup>

<sup>1</sup>Department of Physical Geography and Ecosystem Science, Lund University, 223 62, Lund, Sweden

<sup>2</sup>Department of Ecology and Environmental Science, Umeå University, 90187, Umeå, Sweden

<sup>3</sup>Department of Forest Ecology and Management, Swedish University of Agricultural Sciences, 90183, Umeå, Sweden

**Correspondence:** Martin Berggren (martin.berggren@nateko.lu.se)

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**Abstract.** Dissolved organic carbon (DOC) may be removed, transformed, or added during water transit through lakes, resulting in changes in DOC composition and pigmentation (color). However, the process-based understanding of these changes is incomplete, especially for headwater lakes. We hypothesized that because heterotrophic bacteria preferentially consume noncolored DOC, while photochemical processing removes colored fractions, the overall changes in DOC color upon water passage through a lake depend on the relative importance of these two processes, accordingly. To test this hypothesis we combined laboratory experiments with field studies in nine boreal lakes, assessing both the relative importance of different DOC decay processes (biological or photochemical) and the loss of color during water transit time (WTT) through the lakes. We found that influence from photo-decay dominated changes in DOC quality in the epilimnia of relatively clear headwater lakes, resulting in systematic and selective net losses of colored DOC. However, in highly pigmented brown-water lakes (absorbance at 420 nm  $> 7 \text{ m}^{-1}$ ) biological processes dominated, and there was no systematic relationship between color loss and WTT. Moreover, in situ data and dark experiments supported our hypothesis on the selective microbial removal of nonpigmented DOC, mainly of low molecular weight, leading to persistent water color in these highly colored lakes. Our study shows that brown headwater lakes may not conform to the commonly reported pattern of the selective removal of colored constituents in freshwaters, as DOC can show a

sustained degree of pigmentation upon transit through these lakes.

## 1 Introduction

The color of water is a defining feature of freshwater ecosystems primarily caused by inputs of brown-pigmented dissolved organic carbon (DOC) from terrestrial runoff (Xiao et al., 2015). Recent concerns have been raised regarding widespread increases in color and DOC concentrations in the Northern Hemisphere caused by a combination of factors involving a warmer climate (Lepistö et al., 2014; Pagano et al., 2014), an intensified hydrological cycle (Weyhenmeyer et al., 2012; Fasching et al., 2016), and the release of DOC previously immobilized in soils due to acidification (Monteith et al., 2007). This rise in colored DOC, reviewed by Solomon et al. (2015), is predicted to reduce aquatic productivity (Karlsson et al., 2009), change food webs and population structures (Jansson et al., 2007), alter the stoichiometry and magnitude of bioavailable nutrients pools (Berggren et al., 2015b), and cause increased freshwater CO<sub>2</sub> outgassing (Lapierre et al., 2013). Thus, water color is key to understanding fundamental aspects of aquatic ecosystem functioning in a changing environment.

Inland waters represent a significant component in the global carbon cycle, for example by emitting greenhouse gases to the atmosphere at the rate of at least 1 or 2 Pg C per year (Raymond et al., 2013; Cole et al., 2007). A fundamen-

tally important water-column process that generates carbon dioxide (CO<sub>2</sub>) is the microbial degradation of terrestrially derived DOC (Lapierre et al., 2013; Fasching et al., 2014). Significant amounts of DOC can also be mineralized by ultraviolet (UV) sunlight in lakes (Koehler et al., 2014) and running waters (Cory et al., 2014). However, while much research attention has been drawn to CO<sub>2</sub> production from these different processes, surprisingly little is known about the relative role played by biological and photochemical processes for DOC quality transformations and, in particular, for the removal of color.

On large scales, color tends to decrease faster than DOC along the land–sea continuum (Weyhenmeyer et al., 2012), partly because noncolored DOC might be added by algae in productive waters (Creed et al., 2015), but the circumstances allowing for preferential net losses of colored DOC in unproductive lakes are unclear. Some studies have reported relative losses of colored DOC across lake basins with increasing theoretical residence times (Köhler et al., 2013; Curtis and Schindler, 1997), while other studies have found preferential loss of noncolored DOC in laboratory biodegradation experiments (Hansen et al., 2016) and in time series analyses of brown headwater lakes (Berggren et al., 2009). Although bacteria do consume colored humic substances at low rates (Tranvik, 1988), the biological degradation of DOC is unlikely a mechanism leading to selective color loss because bacteria tend to mainly consume noncolored DOC (Asmala et al., 2014; Hansen et al., 2016). An exception is the apparent preferential use of organo-ferric colloids by bacteria, whereby the removed color comes from iron, not DOC (Oleinikova et al., 2017). The UV light oxidation could theoretically explain losses of colored DOC and the production of noncolored DOC (Stubbins et al., 2010; Kellerman et al., 2014), but efficient photo-processing has been found mainly in relatively DOC poor water (Molot and Dillon, 1997) and in alkaline lakes (Reche et al., 1999) and not systematically in unproductive DOC-rich lakes (Amon and Benner, 1996; Molot and Dillon, 1997; Jonsson et al., 2001). Thus, the processing of colored DOC remains poorly understood in response to water transit through common unproductive DOC-rich headwater systems (Weyhenmeyer et al., 2014).

Most of the studies that have addressed changes in water chemistry in response to water transit times in lakes have applied fixed theoretical mean residence time values (Köhler et al., 2013; Curtis and Schindler, 1997; Weyhenmeyer et al., 2012). However, in reality water transit times through lakes and reservoirs vary several-fold over time as a result of temporal flow variations (Li et al., 2015; Rueda et al., 2006). The processing of DOC in response to the actual water transit time (WTT; average time spent by the water molecules in the lake) through a given lake has been overlooked and addressed in few studies (Berggren et al., 2009, 2010b). For this approach to be successful, the DOC needs to enter a lake in distinct pulses, each time with similar concentrations and chemical properties, such that subsequent DOC quality

changes in the lake are dependent on the WTT. Nonetheless, WTT assessment stands out as a tool that can help fill in the knowledge gap on the hydrological and biogeochemical mechanisms behind DOC quality and color change in headwater lakes.

In this study we combine WTT calculations for a range of different headwater lakes with laboratory simulations of DOC processing to determine the relative importance of different processes that remove DOC (biological or photochemical) and color over time. We chose lakes located within a 50–70 km radius around the Krycklan catchment in northern Sweden, where there is extensive research in support of the assumption that DOC export is highly episodic, with pulses that generally bring DOC of similar quantity and quality during peak flow (Laudon et al., 2011). We hypothesized that heterotrophic bacteria preferentially consume noncolored DOC fractions, resulting in small overall color loss during water retention in lakes where bio-degradation represents the dominant DOC transformation process. In contrast, the selective loss of colored DOC could be expected in lakes where the DOC transformation is dominated by photochemical processing.

## 2 Methods

### 2.1 Study design overview

Targeting the well-studied Björntjärnarna brown-water catchment in northern Sweden (Berggren et al., 2009, 2015a; Karlsson et al., 2012), a large data set for a single catchment was first compiled and analyzed. In total 260 samples were obtained over seven study years from the inlets, epilimnia, hypolimnia, and terminal outlets of two tightly connected lakes called Övre Björntjärnen and Nedre Björntjärnen. These chain lakes share 92 % of the catchment area, and their epilimnia are tightly connected by a short (70 m in length) stream making the same water pass through the two lakes in sequence. The purpose of this first analysis was to analyze one large pooled data set to maximize power, i.e., the chance of finding significant patterns in DOC quality variables with increasing WTT.

For a second part of the analysis, we collected a limited number of samples (ca. 10 per year and site) during 3–4 years from seven additional “survey lakes” along a gradient of DOC and color, increasing the total number of study lakes to nine. These data were used to increase the representativeness of the study given the variations in UV light exposure in the water column and possible differences in UV light degradation between brown-water and clear-water lakes. We selected unproductive boreal lakes ( $\text{Chl } a < 2 \mu\text{g L}^{-1}$ ) because the majority of lakes are located in the boreal region (Verpoorter et al., 2014) where nutrient concentrations are low, yet lake DOC concentrations and optical conditions vary widely (Karlsson et al., 2009). In the analyses of these survey

lakes, epilimnetic and hypolimnetic data were kept separate due to the differences in light climate with different water depths. To understand the relative role played by biological and photochemical processes in the qualitative transformations of DOC, we performed laboratory bioassay experiments on water from three of the lakes (Table 1) in which changes in optical water properties were measured in dark bacterial bioassays and under UV light irradiation, respectively. Since the two processes had systematically different impacts on DOC optical properties, we could use optical indices to determine which of the processes dominated based on how the indices changed in situ with increasing natural WTT in the lakes.

We assumed no significant variations in color induced by iron (Fe) or pH in this study. Total unfiltered Fe concentrations are on the order of  $1 \text{ mg L}^{-1}$  at the inlets, epilimnia, and outlets (personal communication, Bastviken, Linköping University; measured in the four most colored lakes, 2015). Although the abundance and speciation of Fe can affect the optical properties of freshwaters (Pullin et al., 2007), the effect on absorbance in the water should be marginal (Weishaar et al., 2003). According to Kritzbeg and Ekström (2012), the contribution from  $1 \text{ mg L}^{-1}$  of Fe to the absorption coefficient at  $420 \text{ nm}$  ( $a_{420}$ ) is  $0.8 \text{ m}^{-1}$ , corresponding to 5–10 % of the observed  $a_{420}$  in the four lakes that have been analyzed for total Fe. Regarding pH, Pace et al. (2012) showed that the absorption coefficients increase sharply beyond a pH of 7 due to changes in the three-dimensional structure of the DOC molecules. The absorption at low UV wavelengths in particular can also be enhanced by an extremely low pH ( $\sim 4$ ), potentially leading to higher photo-reactivity (Anesio and Granéli, 2003; Gennings et al., 2001), although these effects appear relatively small compared to those at high pH (Pace et al., 2012). In this study, the observed range in pH across all sites and sampling dates was 3.4–6.8 (mean = 5.4, SD = 0.6), which is below the threshold for major pH interference according to Pace et al. (2012).

## 2.2 Study site descriptions

The nine Swedish boreal lakes that were selected (5–10 m max depth) varied more than 5-fold in color and approximately 3-fold in DOC (Table 1). All lakes have previously been depth-profiled using an echo sounder. Regional mean annual temperature is  $1^\circ\text{C}$  and the average annual precipitation is 500–600 mm, of which half arrives as snow. Lake ice is generally present from late October to early May. While the lakes have small areas varying only between 0.01 and  $0.05 \text{ km}^2$ , the catchment areas vary 100-fold from 0.03 to  $3.25 \text{ km}^2$ , resulting in mean epilimnetic water transit times from 0.2–0.3 to ca 1–3 years in the lakes with the largest and the smallest watersheds, respectively (Table 1). The catchments are mainly ( $> 75\%$ ) covered by coniferous forest (*Picea abies*, *Pinus sylvestris*) and *Sphagnum*-dominated mires ( $< 25\%$ ). Forests are managed and have widely vary-

ing ages, both within and between the catchments, from 5–100 years. The location, mean optical properties, and DOC concentrations of each of the study lakes are presented in Table 1. The four clearest lakes have no permanent inlet streams, their catchments are very small (Table 1), and they are surrounded by forested hills. The five browner lakes receive hydrological input from streams and they are located in relatively flat lower parts of the catchments, with abundant peatlands in their direct surroundings.

Three of the lakes (Table 1) received inorganic N additions 2012–2014 (as part of another study) to create a slightly elevated nitrate concentration by  $0.1 \text{ mg N L}^{-1}$  (Klaus et al., 2017). This fertilization could potentially affect the DOC decay (Berggren et al., 2007). Therefore, before initiation of the lake fertilization, we tested the potential influence of N by performing 2-week in vitro DOC and color loss measurements (bioassays) on water from Nedre Björntjärnen, which has the lowest natural ratio of inorganic N to DOC among the nutrient-amended lakes and would thus be at the highest risk for bias. The bioassays were performed as described previously in detail (Berggren et al., 2009) at  $20^\circ\text{C}$  in dark conditions with ambient bacterial communities and with additions of single spikes ( $1 \text{ mg N L}^{-1}$  added) of ammonium nitrate at the beginning of the incubations using epilimnetic water obtained in winter, spring, summer, and fall. We found no N-addition effects on DOC degradation or change in water color during these bioassay experiments (Supplement Fig. S1). Moreover, none of the N-amended lakes appeared as outliers in this study. Therefore, we assume that the lake N addition had no critical impact on the results or the conclusions drawn in this work.

## 2.3 Sampling and water analysis

Sampling was carried out between 2006 and 2014 (3–7 years of data per lake). Water from epilimnion and hypolimnion (mid-depths or composite samples) was collected every 2–3 weeks throughout the ice-free seasons and occasionally under ice at the deepest point of each lake (see sample numbers in Table 1). Additional samples for the detailed analysis of the Björntjärnarna catchments were obtained on most sampling dates at the headwater inlet and outlets of the chain lakes (Björntjärnarna). Sampled water was stored in cooling boxes until processing in the lab within 2–10 h. Temperature profiles were obtained with electronic sensors at each sampling occasion (plus occasional additional dates) and used to calculate volumes above and below the thermocline depth, defined as the mid-depth of the transect where temperature changed  $< 1^\circ\text{C m}^{-1}$ . Discharge was assessed as described in the Supplement, Text S1.

In the laboratory, lake water was filtered with acid-washed  $0.7 \mu\text{m}$  glass fiber filters (Whatman GF/F). Absorbance spectra of the filtrate were measured at room temperature in 1 cm quartz cuvettes using a Jasco V-560 UV-Vis spectrophotometer. Blank values from deionized water were subtracted

**Table 1.** Characteristics and sampling details for the nine study lakes. Variables from left to right: decimal degree (DD) WGS84 latitude (Lat) and longitude (Lon) coordinates, water transit time (WTT), area of lake ( $A_{\text{lake}}$ ), mean depth of lake ( $Z_{\text{mean}}$ ), area of catchment ( $A_{\text{catchment}}$ ), ratio between the absorbance at the wavelengths of 254 and 365 nm ( $A_{254} : A_{365}$ ), carbon specific absorbance at 420 nm ( $a_{420} : \text{DOC}$ ), dissolved organic carbon (DOC), absorbance at 420 nm ( $a_{420}$ ), number of epilimnetic and hypolimnetic samples ( $n$ ), and study years. The WTT and chemical characteristics are shown as mean epilimnetic values across all sampling dates.

Site name	Lat (DD)	Lon (DD)	WTT (years)	$A_{\text{lake}}$ (km <sup>2</sup> )	$Z_{\text{mean}}$	$A_{\text{catchment}}$ (km <sup>2</sup> )	$A_{254} : A_{365}$	$a_{420} : \text{DOC}$	DOC (mg C L <sup>-1</sup> )	$a_{420}$ (m <sup>-1</sup> )	$n$ (epi/hypo)	Years (20XX)
Fisklösan <sup>1</sup>	64.150	18.800	0.94	0.017	2.1	0.089	4.55	0.32	7.7	2.5	38/28	11–14
Nästjärnen	64.160	18.777	3.13	0.010	4.2	0.034	4.50	0.35	7.6	2.7	40/30	11–14
Mångstretjärnen	64.251	18.762	1.44	0.018	5.3	0.141	4.12	0.48	11.5	5.5	41/31	11–14
Lappjärn <sup>1</sup>	64.237	18.790	0.67	0.020	2.5	0.168	4.12	0.48	13.1	6.3	41/31	11–14
Lillsjöleden <sup>3</sup>	63.845	18.616	0.19	0.008	3.8	0.254	4.10	0.49	15.7	7.6	35/33	12–14
Nedre Björntjärnen <sup>1,2</sup>	64.122	18.785	0.30	0.032	6.0	3.249	4.02	0.57	18.9	10.7	59/29	06–07, 11–14
Struptjärnen <sup>3</sup>	64.023	19.489	0.34	0.031	3.8	0.791	4.03	0.55	21.4	11.7	38/29	12–14
Övre Björntjärnen <sup>2,3</sup>	64.123	18.779	0.19	0.048	4.0	2.840	4.04	0.55	21.9	11.9	72/45	06–07, 09, 11–14
Stortjärnen	64.261	19.763	0.41	0.039	2.7	0.817	3.81	0.63	20.9	13.3	31/29	12–14

<sup>1</sup> Elevated inorganic N concentrations 2012–2014 by 0.1 mg NL<sup>-1</sup>.

<sup>2</sup> Lake included in the focal study of the Björntjärnarna catchment.

<sup>3</sup> Lake selected for laboratory incubation experiments.

from the spectra. An aliquot of 40 mL of the filtrate was acidified (50  $\mu$ L 1.2 M HCl) and stored in darkness at 6 °C until DOC analysis by high-temperature catalytic oxidation using a Hach IL 550 TOC-TN analyzer (Hach-Lange GmbH, Düsseldorf, Germany). In the Björntjärnarna chain lakes, water from 12 dates in 2009 was analyzed for organic acids, free amino acids, and simple carbohydrates using a liquid chromatography–ion spray tandem mass spectrometry (LC-MS) system. The LC-MS system consisted of a Dionex (Sunnyvale, CA, USA) ICS-2500 liquid chromatography system and an Applied Biosystems (Foster City, CA, USA) 2000 Q-trap triple quadrupole mass spectrometer. The method is described in further detail in Ström et al. (2012).

## 2.4 Water transit time assessments

The transit time of the water volume that resides in a lake at a given moment is dependent upon the retention and renewal history of that water. We assumed that WTT increases with +1 per unit of time and that it decreases in proportion to how the water volume ( $\text{Vol}_{\text{total}}$ ) is replaced by new inflowing water (flow rate; volume per unit time), which gives a change in WTT per unit time ( $d\text{WTT}/dt$ ) according to Eq. (1).

$$\frac{d\text{WTT}}{dt} = 1 - \text{WTT} \times \frac{\text{Flow rate} \times dt}{\text{Vol}_{\text{total}}} \quad (1)$$

However, this continuous function (Eq. 1) is not suited to be applied directly in this study because our data are discrete and further involve two depth strata (epilimnion and hypolimnion) with reciprocal entrainment effects due to the dislocation of the thermocline. Therefore we adapted discrete functions for the changes in epilimnetic and hypolimnetic WTT from one day ( $t$ ) to the next day ( $t+1$ ) (Berggren et al., 2010b, 2009). Using these functions (Eqs. 2–5), lake WTTs were calculated iteratively for each day in sequence from discharge (measured daily) and lake volume data (epilimnetic

and hypolimnetic; daily values obtained by linear interpolations between sampling dates). To get realistic WTT values for the first day of the study period, the iteration was initiated from a date 10 years in advance. An arbitrary WTT starting value could then be chosen without impact on the calculated WTTs of the study years. For the pre-study period, the mean seasonal mixing pattern for each lake (see Fig. S2a) was used to generate daily epilimnetic and hypolimnetic proxy volumes.

For any given day ( $t$ ), a certain volume of inflowing water ( $\text{Vol}_{\text{inflow}}$ ) was considered to mix with a certain volume of epilimnetic lake water ( $\text{Vol}_{\text{epi}}$ ; above mid-thermocline depth) and, during days with downward dislocation of the thermocline, with an additional volume of hypolimnetic water ( $\text{Vol}_{\text{hypo flow}}$ ). Thus, after the day in question ( $t+1$ ), the resulting new mean WTT equals the volume-weighted average WTT of these different volumes that mixed during day  $t$ . In addition, the transit time also changes with +1 per unit of time, as the water resides for 1 day in the lake during the day ( $t$ ). Hence the resulting WTT on day  $t+1$  was calculated according to Eq. (2) (days without downward thermocline dislocation) or Eq. (3) (days with downward thermocline dislocation).  $\text{Vol}_{\text{hypo flow}}(t)$  was given from the decrease in the hypolimnetic volume from day  $t$  to the next day ( $t+1$ ).

$$\text{WTT}_{\text{epi}}(t+1) = 1 + \frac{\text{WTT}_{\text{in}}(t) \times \text{Vol}_{\text{inflow}}(t) + \text{WTT}_{\text{epi}}(t) \times \text{Vol}_{\text{epi}}(t)}{\text{Vol}_{\text{inflow}}(t) + \text{Vol}_{\text{epi}}(t)} \quad (2)$$

$$\text{WTT}_{\text{epi}}(t+1) = 1 + \frac{\text{WTT}_{\text{in}}(t) \times \text{Vol}_{\text{inflow}}(t) + \text{WTT}_{\text{epi}}(t) \times \text{Vol}_{\text{epi}}(t) + \text{WTT}_{\text{hypo}}(t) \times \text{Vol}_{\text{hypo flow}}(t)}{\text{Vol}_{\text{inflow}}(t) + \text{Vol}_{\text{epi}}(t) + \text{Vol}_{\text{hypo flow}}(t)} \quad (3)$$

After a day with reduced thermocline depth, the new hypolimnetic WTT ( $t+1$ ) resulting from the entrainment of epilimnetic water ( $\text{Vol}_{\text{epi flow}}$ ) into the hypolimnion ( $\text{Vol}_{\text{hypo}}$ ) was calculated using Eq. (4). Again, the transit time also changes with +1, as the water resides in the lake during

the day in question. After a day without reduced thermocline depth, the  $WTT_{\text{hypo}}$  was unaffected by mixing with epilimnetic water (Eq. 5).  $Vol_{\text{epi flow}}(t)$  was given by the increase in the hypolimnetic volume from day ( $t$ ) to the next day ( $t + 1$ ).

$$WTT_{\text{hypo}}(t + 1) = 1 + \frac{WTT_{\text{epi}}(t) \times Vol_{\text{epi flow}}(t) + WTT_{\text{hypo}}(t) \times Vol_{\text{hypo}}(t)}{Vol_{\text{epi flow}}(t) + Vol_{\text{hypo}}(t)} \quad (4)$$

$$WTT_{\text{hypo}}(t + 1) = 1 + WTT_{\text{hypo}}(t) \quad (5)$$

Inflowing water from the catchment ( $Vol_{\text{inflow}}$ ) was assigned the WTT of 0, as the headwater inlet streams represent water transit times that can be considered negligible compared to the WTT in the lakes (Berggren et al., 2009). Moreover, there were no upstream lakes in the catchments in addition to the study lakes themselves, implying that the drainage represented true headwater sources directly from surrounding soils. However, in the lower chain lake (Nedre Björntjärnen), the WTT of inflowing water was considered to equal the WTT of outflowing (epilimnetic) water from the upper chain lake (Övre Björntjärnen). Similarly, we assumed that WTT of outflowing water from the lower chain lake equals the WTT for epilimnetic water in this lake.

Our assumption that all inflowing water mixed with the epilimnion (not hypolimnion) is supported by the fact that all of the lakes with permanent inlets (five of the study lakes) have inlet streams entering shallow areas of the respective lakes, i.e., without hypolimnia where the water is forced to mix with the epilimnion. Thus even if the inflowing stream water sometimes had a temperature (and thus density) similar to that of the hypolimnion, no downwelling was likely to happen.

## 2.5 Response variables

We used the decadic absorbance coefficient  $a_{420}$  ( $\text{m}^{-1}$ ) as a measure of “color”, which is conventional in water monitoring and research in the study region (Kritzberg and Ekström, 2012; Weyhenmeyer et al., 2012). In addition to DOC and the color indicator  $a_{420}$  (Weyhenmeyer et al., 2012), we analyzed two optical ratios that indicate quality changes in the dissolved organic matter in response to changing WTT. First, we used the absorbance ratio  $A_{254} : A_{365}$ , which describes a shift towards absorption in the red part of the spectrum and thus tends to be negatively related to average molecular DOC weight (Dahlén et al., 1996) and positively correlated with low-molecular-weight DOC compounds (Berggren et al., 2010a). There are many other spectral slope indices in the literature essentially providing the same information, but we chose  $A_{254} : A_{365}$  since it is a simple index that has been used previously in the study area (Berggren et al., 2007; Ågren et al., 2008a). The direction of change in this ratio is indicative of the dominant DOC transformation process:  $A_{254} : A_{365}$  increases with UV light processing (Dahlén et al., 1996), but it decreases in response to bacterial DOC processing (Berggren

et al., 2007). In agreement with the expectations based on Berggren et al. (2010a),  $A_{254} : A_{365}$  in this study was positively correlated with both absolute ( $\text{mg CL}^{-1}$ ) and relative (% of DOC) total concentrations of low-molecular-weight carbon compounds in the form of organic acids, free amino acids, and simple carbohydrates (Fig. S3).

Second, we used the ratio  $a_{420} : \text{DOC}$ . If this ratio increases with WTT, then noncolored DOC is selectively removed (or more colored DOC is added), but if  $a_{420} : \text{DOC}$  decreases, then this is either due to the selective decay of colored DOC (Weyhenmeyer et al., 2012) or the selective addition of low-pigmented DOC (Creed et al., 2015). It should be mentioned here that other indices of color per unit DOC are also common in the literature, especially specific UV light absorbance at 254 nm ( $SUVA_{254}$ ). However, in our study the overall relationship between  $SUVA_{254}$  and  $a_{420} : \text{DOC}$  was strong and linear ( $r^2 = 0.80$ ,  $n = 680$ ; all sites and sampling dates), and the two variables showed the same patterns in response to changing WTT. Therefore, to avoid presenting the same patterns twice, we do not report  $SUVA_{254}$  in this paper.

## 2.6 Laboratory experiments

We performed laboratory experiments on water from three catchments to disentangle the isolated effect on DOC quality by UV light degradation from that of microbial processing. The purpose with the experimental design was to create conditions during which either (1) photochemical reactions strongly and dominantly influenced the DOC transformation or (2) microbial degradation strongly dominated the DOC transformation. Therefore, we measured the DOC quality responses to a large light dose or a long microbial process time in the dark. However, the experiments were not designed to reflect in situ decay rates. For example, temperatures and likely also microbial communities were different in the bottle experiments compared to the field.

In the dark microbial process experiment, we used water either from the lake inlets or from the epilimnia at times when the water had only resided for a short period in the lake according to our model ( $< 1$  month on average). The unfiltered natural water samples (with ambient microbial community) were then incubated in darkness for 450 days at  $20^\circ\text{C}$  in 1 L acid-washed Duran glass bottles with a gas headspace ( $\sim 100$  mL) containing sufficient  $\text{O}_2$  to theoretically oxidize all DOC in the samples. The selected incubation time (450 days) was similar to the mean WTT in the study lakes (about 1 year on average; Table 1). Although protozoa were not removed, based on Daniel et al. (2005) it can be assumed that bacteria vastly dominate the biomass ( $\sim 90\%$ ) when dark bioassays are performed on natural humic water. We assumed that there was no nitrogen or phosphorus limitation due to the high concentrations of DOC ( $\sim 15 \text{ mg L}^{-1}$ ) known to be associated with high organic nutrient bioavailability in lakes (Soares et al., 2017) and streams (Jansson et al., 2012) of the region. Our assumption is supported by Jansson et al. (2001),

who showed that bacterial metabolism in humic lakes is generally not nutrient limited when DOC is higher than  $\sim 15 \text{ mg L}^{-1}$ .

For the light experiments, we chose a slightly different approach. While the microbial processing happens continuously in the entire water column of the lakes, UV light processing occurs only in a thin superficial layer of the lakes during daytime and ice-free conditions. This means that, although photodegradation and biodegradation can happen simultaneously, much of the DOC has likely stayed a variable and potentially long time in darkness before coming into contact with UV light (Gonsior et al., 2013). For this reason, we used water that had first been incubated with microbes in the dark (as described above) as starting material for the UV light experiments. Hence, 10 mL filtered ( $0.45 \mu\text{m}$ ) samples were incubated for 24 h in cylindrical quartz vials placed horizontally on a spinning disc (0.67 rpm) in a  $20^\circ\text{C}$  climate chamber at approximately 40 cm of distance from two xenon–sodium lamps. The UV irradiation of the different parts of the disk was within  $3.64\text{--}6.89 \text{ W m}^{-2}$  for UV-A and  $0.06\text{--}0.1 \text{ W m}^{-2}$  for UV-B according to radiation measurements (spectroradiometry; International Light Inc.). Spinning of the disk ensured an equal light dose received by all samples. Based on Bertilsson and Tranvik (2000), we used complete spectra for lamp irradiation and absorption to calculate that the samples absorbed a UV light dose of approximately  $100 \text{ MJ m}^{-3}$ , equivalent to at least 2 years of water-column-integrated in situ UV light absorption. The bacterial DOC processing during irradiation was considered negligible because the rate of DOC loss under light was 1–2 order of magnitudes higher than in dark incubations with microbial degradation only.

The dark and light experiments were performed on 14 samples from Övre Björntjärnen, Lillsjöleden, and Struptjärnen (4–5 per site from spring to fall). The DOC and absorption coefficients were analyzed before and after each experiment, and the changes from the beginning to the end of the incubation (final value minus start value) were calculated for the absorbance ratio  $A_{254} : A_{365}$  and the carbon color indicator for  $a_{420} : \text{DOC}$ . In addition, the relative change (%) in color ( $a_{420}$ ) from the beginning to the end of the incubations was calculated and used for a qualitative comparison with the loss in  $a_{420}$  during water transit through the different lakes in situ.

## 2.7 Statistics

The response of spectrophotometric variables to changing WTT was evaluated through linear mixed-effects regression (LMER) in the pooled data set from the Björntjärnarna chain lakes, where WTT impact was considered a fixed effect and “site” was added as random factor to allow for potential differences in the relationships (different intercepts only) between the monitoring stations. However, ordinary linear regression was used in the individual analyses of the survey

lakes. Temporally, all response variables showed systematic and significant ( $p < 0.05$ ) autocorrelation for a time lag of one step, i.e., the 2–3-week sampling frequency (Box–Ljung autocorrelations around 0.5; software IBM SPSS 22), but there was no systematic autocorrelation for two time steps, i.e., 4–6 weeks. In order to not grant significance too generously considering the temporal autocorrelation, we adjusted the  $\alpha$  for relationships between WTT and response variables from 0.05 to 0.01. According to standard significance tables, it takes roughly twice as many observations to obtain significance of a correlation at the 0.01 level compared to the 0.05 level, implying that an  $\alpha$  adjustment from 0.05 to 0.01 takes into account that only every second observation in the time series could be assumed to be independent. Although it would be possible to explicitly include an autocorrelation term in the LMER models, the  $\alpha$  adjustment was chosen since it could be applied in a systematic way to all regression results in the study, i.e., both to the LMER models and to the ordinary linear regression models. The 0.05 level was considered only marginally significant.

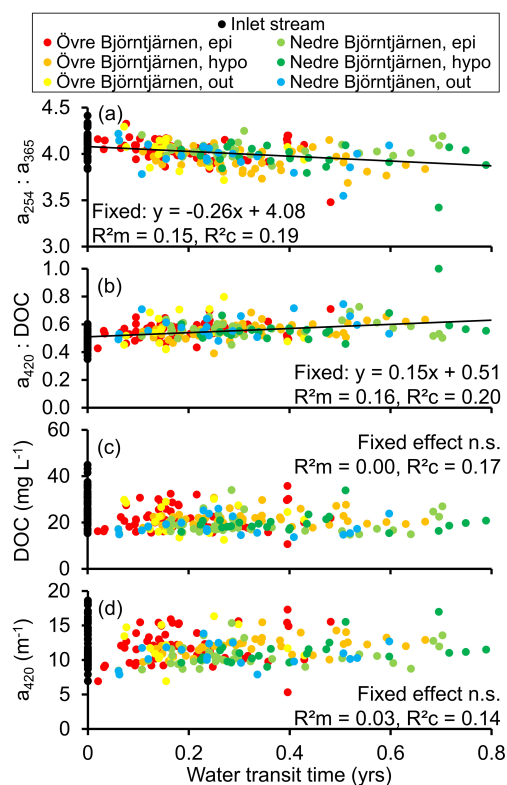
For the laboratory experiment results, we used two-tail paired  $t$  tests to test for changes between initial and ending conditions. All statistics were performed using IBM SPSS 22, except for the linear mixed-effects regressions performed using the statistical package `lm4` for *R*. To obtain conditional and marginal  $R^2$  estimates ( $R^2_c$  and  $R^2_m$ , respectively) for mixed models, the package `MuMIn` was used, while the significance of coefficients and intercepts (fixed effects) were tested with the package `lmerTest`.

## 3 Results

### 3.1 Seasonal water transit time patterns

The different study lakes showed a coherent seasonal mixing pattern, with stable thermal stratification from mid-May to mid-September between spring and fall overturns (Fig. S2). The temporal variability in WTT was partly controlled by annually recurring high-flow events that caused systematic drops in the WTT by approximately 0.2–0.5 years in spring (due to snowmelt) and 0.1–0.2 years in fall (due to rainstorms). Conversely, during low flow in winter and summer all sites showed a slow but stable increase in WTT (Fig. S2). For all catchments, 50% or more of the annual runoff was represented by discharge above the 90th percentile; i.e., a majority of the discharge happened during hydrological episodes.

In the site-by-site analysis of survey data from all nine lakes, the water transit time spanned from 0.2–3.1 years for epilimnetic samples from each respective site (mean across all sampling dates) and 0.3–3.7 years for the corresponding hypolimnetic samples. Within each lake, there was considerable variability in WTT, with on average twice as high epilimnetic WTT after the driest periods compared to the wettest

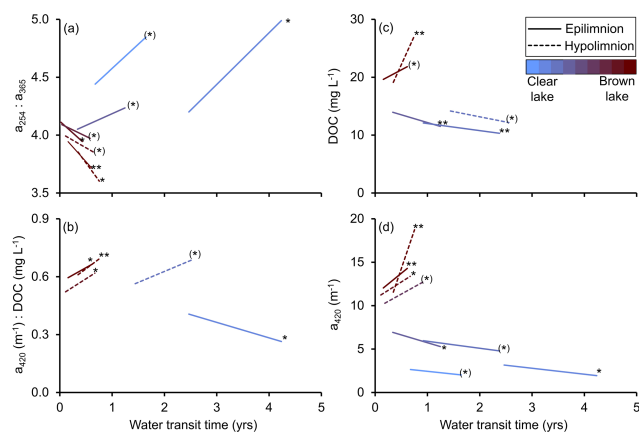


**Figure 1.** Relationships between dissolved organic carbon (DOC) properties and the water transit time in two chain lakes with 92 % shared catchment area. The y axis variables are (a) the ratio between the absorbance at the wavelengths of 254 and 365 nm, (b) the ratio between absorbance at 420 nm and DOC, (c) DOC concentration, and (d) absorbance at 420 nm ( $n = 260$ , study years 2006–2014). Solid lines are based on significant ( $p < 0.01$ ) fixed-effects coefficients and intercepts. The  $R^2m$  shows marginal  $R^2$  for fixed effects, where water transit time is the fixed effect, and the  $R^2c$  refers to conditional mixed-effects models in which site is included as a random effect (on intercept only, not slope). Inlet samples during drought (lower 5 percentiles of flow) are not included since drought inflow makes a negligible contribution to the water that resides in the lakes. In Fig. S4 all DOC property data from this figure are plotted over a calendar time line. Abbreviations: n.s. – not significant; epi – epilimnion; hypo – hypolimnion; out – outlet.

periods. The temporal span in WTT, i.e., the highest minus the lowest value for each site, varied from 0.4 to 1.8 years.

### 3.2 Björntjärnarna chain lakes

In the pooled data set from the Björntjärnarna brown-water chain lakes, we found significant relationships (linear mixed-effects regression) between spectrophotometric response variables and the WTT. The ratio  $A_{254} : A_{365}$ , which indicates the relative abundance of low-molecular-weight DOC, showed a decreasing trend (from 4.1 to 3.8) over the span from 0 to 0.8 years of WTT in this lake system ( $R^2m = 0.15$ ,  $R^2c = 0.19$ ,  $n = 260$ ; Fig. 1a). At the



**Figure 2.** Linear regression lines for significant relationships between different properties of DOC and the water transit time, shown for individual epilimnia (solid lines) and hypolimnia (dashed lines) of nine lakes in northern Sweden. The y axis variables are (a) the absorbance ratio  $A_{254} : A_{365}$ , (b) the ratio between absorbance at 420 nm and DOC concentration, (c) DOC concentration, and (d) absorbance at 420 nm. Nonsignificant regression lines are not shown. For full statistical details and data plots, see Table S1 and Fig. S5, respectively. Significance: \*  $p < 0.01$ ; \*\*  $p < 0.001$ ; (\*) marginally significant  $p < 0.05$ .

same time, the DOC became relatively more colored, as demonstrated by the significant positive trend for the ratio  $a_{420} : \text{DOC}$  ( $R^2m = 0.16$ ,  $R^2c = 0.20$ ,  $n = 260$ ; Fig. 1b). For changes in DOC (Fig. 1c) there was no significant fixed effect caused by water transit time, although the random factor (site) explained 17 % of the variance (difference between  $R^2c = 0.17$  and  $R^2m = 0.00$ ), largely due to higher DOC at the inlet than in the lakes. The overall color (absorbance at 420 nm) remained remarkably constant with increasing WTT (Fig. 1d).

### 3.3 Survey lakes

The ordinary linear regression results on data from epilimnia and hypolimnia of the nine survey lakes are detailed in Table S1, shown with complete data plots in Fig. S5, and summarized in Fig. 2. These results partly conformed to the patterns in Björntjärnarna, as three out of the nine epilimnetic sites showed significant ( $p < 0.01$ , two cases) or marginally significant ( $p < 0.05$ , one case) linear decreases in  $A_{254} : A_{365}$  with increasing WTT (Fig. 2a; Table S1). In one case this trend was significant at the 0.01 level both above and below the thermocline. In contrast, in the epilimnia of three clearer lakes, epilimnetic  $A_{254} : A_{365}$  increased with increasing WTT (Fig. 2a). Furthermore, only two of the most colored lakes showed increases in  $a_{420} : \text{DOC}$  with increasing WTT significant at the 0.01 level (Fig. 2b; Table S1), which was observed both above and below the thermocline in one case. The epilimnion of one of the lakes showed decreasing  $a_{420} : \text{DOC}$  over the WTT gradient

(Fig. 2b). Interestingly, the DOC quality in lakes with intermediate color appeared less responsive to changes in WTT than DOC quality in clearer or browner lakes, as exemplified by the complete lack of relationships between DOC properties and WTT in the intermediate Lillsjöliden (Table S1, Fig. S5).

When analyzing the nine lakes one by one, significant ( $p < 0.01$ ) losses of DOC with increasing WTT were found only in two of the lakes with intermediate color (Fig. 2c). Unexpectedly, one lake (Stortjärnen) showed a significant trend of increasing DOC with increasing WTT, indicating the release of DOC from within the lake or its benthic and/or littoral contact surfaces. In terms of the overall color of the lakes (absolute  $a_{420}$  values), significant ( $p < 0.01$ ) increases with increasing WTT were found in two hypolimnia and one epilimnion (Fig. 2d). In the epilimnetic water of the clearer lakes, the  $a_{420}$  tended to decrease, but no significant pattern of decreasing  $a_{420}$  was found for hypolimnetic sites (Fig. 2d).

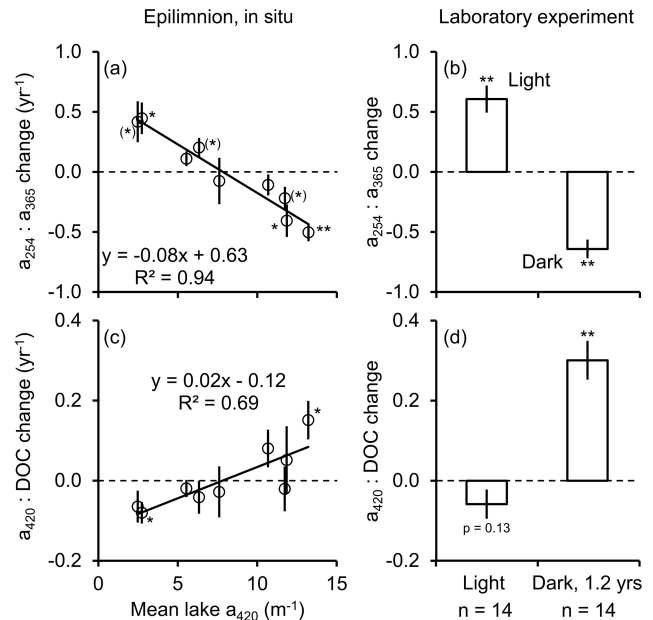
We further found that there were systematic shifts in the dynamics of both  $A_{254} : A_{365}$  and the index  $a_{420} : \text{DOC}$  along the gradient of increasing mean color of the different lakes. In fact, the rate of change in epilimnetic  $A_{254} : A_{365}$  per unit WTT was strongly negatively related to the mean  $a_{420}$  of the respective lakes ( $R^2 = 0.94$ ,  $n = 9$ ,  $p < 0.001$ ; Fig. 3a). Furthermore, the rate of change in epilimnetic  $a_{420} : \text{DOC}$  per unit WTT was positively related to the mean  $a_{420}$  ( $R^2 = 0.69$ ,  $n = 9$ ,  $p < 0.01$ ; Fig. 3b).

### 3.4 Experiments

To understand the mechanisms behind such patterns, we performed laboratory light and dark experiments. In the light treatment, the changes in the  $A_{254} : A_{365}$  ratio (significantly positive) and in  $a_{420} : \text{DOC}$  (nonsignificant slightly negative) were similar to the changes observed as functions of WTT in clear epilimnetic waters (Fig. 3c–d). In the dark bacterial bioassays, the changes in both of these ratios were significant in the opposite direction relative to the light treatment (Fig. 3c–d), i.e., similar to the changes observed in brown epilimnetic waters (Fig. 3a–b).

### 3.5 Overall color loss

Finally, we multiplied the in situ rate of epilimnetic color loss in the survey lakes (slopes  $\pm$  SE of epilimnetic  $a_{420}$  regression models; see Table S1) with the mean WTT for the respective sites (Table 1) to find out how much total change there was in water color upon transit through each lake. The change was then expressed as a proportion (%) of the mean WTT for respective lakes. This showed that losses corresponding to 19–79 % of mean  $a_{420}$  occurred in the four clearest lakes, whereas the  $a_{420}$  either showed no change or increased slightly in the five brownest lakes (Fig. 4). These changes in color upon lake transit largely overlapped the ranges of color loss shown in the light (22–51 % loss) and



**Figure 3.** Changes in DOC quality over time as indicated by the optical indices  $A_{254} : A_{365}$  (unitless) and  $a_{420} : \text{DOC}$  ( $\text{m}^{-1} : \text{mg L}^{-1}$ ), shown separately for (a–b) the lake epilimnia and (c–d) laboratory experiments. In (a–b), the y axis value of each symbol ( $\pm$  error bar) shows the linear slope ( $\pm$  standard error) of  $A_{254} : A_{365}$  or  $a_{420} : \text{DOC}$  as a function of the water transit time in a certain lake. Symbols with asterisks represent individual changes (in situ data regression slopes; see Fig. 2) that are significant. The x axis values show mean absorbance ( $a_{420}$ ) of the lakes. Panels (c–d) show changes (stop minus start values; mean  $\pm$  standard error) in  $A_{254} : A_{365}$  and  $a_{420} : \text{DOC}$  during light ( $\sim 100 \text{ MJ}$  absorbed per  $\text{m}^3$  of water) and dark (1.2 years) experiments performed on water from three of the study lakes. Bars with asterisks show changes that are significant (two-tailed paired  $t$  test comparing initial and ending conditions). Significance: \*  $p < 0.01$ ; \*\*  $p < 0.001$ ; (\*) marginally significant  $p < 0.05$ . See Table 1 for sample numbers and descriptions of each lake.

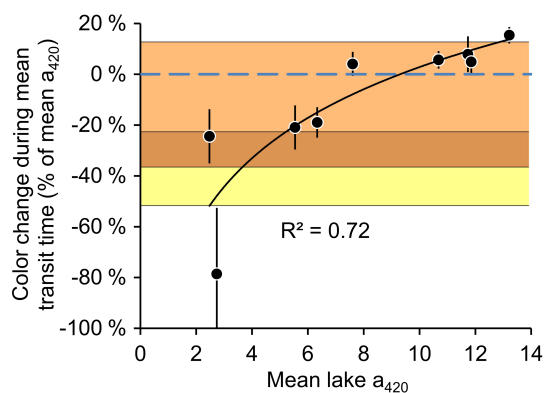
dark (13 % increase to 36 % loss) incubation experiments, respectively. The change in color upon transit through the lakes over the gradient of increasing mean  $a_{420}$  was best described (in terms of fit) by a logarithmic curve (Fig. 4).

## 4 Discussion

### 4.1 Main findings

Our results show that brown headwater lakes do not necessarily conform to the generally reported patterns of efficient and selective removal of colored constituents in freshwaters. Based on a rigorous data set from the Björntjärnarna chain lakes ( $n = 260$ ) spanning 7 years of measurements, we found that the color ( $a_{420}$ ) was sustained at a constant level over water transit times from 0 up to 0.8 years. In these lakes the major discharge pulses alone (snowmelt and storms) renew all





**Figure 4.** Percent change in epilimnetic color ( $a_{420}$ ; filled circles) during water transit through nine boreal lakes plotted over a gradient of increasing mean lake color ( $y = 0.39 \ln(x) - 0.87$ ,  $p < 0.01$ ). The change in color ( $\pm$  error bar) is based on the in situ data regression slopes ( $\pm$  SE) from Table S1 multiplied by the mean water transit time in each lake and expressed in proportion to lake mean  $a_{420}$ . The blue line denotes zero change. For the significance of the individual color loss terms, see the original epilimnetic  $a_{420}$  regression models in Table S1 or Fig. S5 in which the lakes are aligned in the same order as here, from clear to brown. For a qualitative comparison, the yellow and brown areas, respectively (overlapping in the dark brown area), indicate ranges in  $a_{420}$  change from the beginning to the end of light ( $\sim 100$  MJ absorbed per  $\text{m}^3$  of water) and dark (1.2 years of incubation) experiments performed in the laboratory.

the water at least once per year, and the processes that remove colored DOC are too slow to result in significant color loss between these discharge pulses. Thus, even if the input of humic materials from the catchment represent a relatively photoreactive DOC source (Lindell et al., 2000; Vachon et al., 2016), the photo-bleaching in Björntjärnarna apparently was not sufficient to cause a significant net loss of color. Moreover, the ratio between  $a_{420}$  and DOC increased significantly over time, likely due to the preferential microbial consumption of noncolored low-molecular-weight DOC, supporting our hypothesis.

In the multi-lake comparison our results demonstrate contrasting DOC quality dynamics for different types of lake ecosystems. We argue that changes in the DOC quality indices  $a_{420} : \text{DOC}$  and the absorption ratio  $A_{254} : A_{265}$  strongly indicate that microbial processes dominated the DOC quality transformation in the brownest lakes, while photochemical processes dominated in the clearest lakes. Epilimnetic waters of relatively clear lakes showed losses in  $a_{420}$  with WTTs, a pattern also found elsewhere in Sweden (Müller et al., 2013; Weyhenmeyer et al., 2012). Lakes of intermediate color were, however, relatively unresponsive to changes in WTT, suggesting that the DOC quality and color in these lakes would be less sensitive to hydrological events such as rainfall or drought.

## 4.2 Possible bias

A factor that could potentially bias interpretations of how DOC properties change in response to WTT is that the chemistry of source water is variable over time, leading to seasonal changes in the concentration and quality of inflowing DOC to the lakes (Ågren et al., 2008b). For example, during low flow it has been shown that headwater streams in the region can have unusually high concentrations of colored wetland-derived DOC (Laudon et al., 2011). However, in our measurements of DOC concentrations and properties in the inlet stream to the Björntjärnarna catchment (Fig. S4), we found no systematic patterns with discharge. Moreover, headwater sources in general show relatively small variability in water chemistry during episodic flow, which represents a majority of the annual DOC export from small catchments (Laudon et al., 2011) and thus also the majority of the DOC processed in the study lakes during subsequent low-flow periods. Similar to what has been reported elsewhere for headwaters (Wilson et al., 2013; Boyer et al., 1997), a majority of the discharge happened during peak flow ( $> 90$ th percentile flow rate) and the low-flow periods played a negligible role (Fig. S4). Therefore, it appears unlikely that the patterns in DOC and optical properties with increasing WTT in the lakes would be primarily driven by temporal variations in inflowing water.

Another potential bias is represented by new sources of DOC, i.e., other than catchment runoff, which can contribute to the development of lake color over time (Creed et al., 2015). For example, DOC can be added internally within lakes by release from algae, sediments, macrophytes, or littoral peats and marshes in direct contact with the lake water (Wetzel, 2001). The lakes in this study are unproductive with minor contributions from primary producers to the DOC pool, as exemplified by their negligible role in a previous organic carbon budget for Övre Björntjärnen (Karlsson et al., 2012). However, one of the study lakes (Stortjärnen) has littoral peatlands along roughly half of its shoreline, with peat virtually floating in the lake water, potentially releasing DOC directly or via groundwater to the lake. This particular lake showed significant increases in DOC and  $a_{420}$  with increasing WTT (see Table S1 statistics), possibly due to direct inputs from the littoral peat. Moreover, since peatlands in an area close to the Stortjärnen lake have been shown to have particularly high DOC concentrations during low-flow periods (Laudon et al., 2011), it is possible that Stortjärnen is a special case in which even small diffuse hydrological inputs from the peatlands surrounding the lake can be sufficient to raise the DOC concentration during low flow. Nonetheless, while it is interesting to note that DOC accumulation can overcome degradation in some small individual unproductive lakes, the results from this study are too limited to generalize such patterns.

### 4.3 Organic carbon transformation processes

To adequately understand our results it is necessary to bring the mechanisms of DOC decay in the lakes to attention. Previous studies in the region have shown that nonpigmented low-molecular-weight carbon (LMWC) is selectively used by bacteria in brown-water streams and lakes (Berggren et al., 2010b, a). In agreement with these previous results, the LMWC indicator  $A_{254} : A_{365}$  (see relationship between LMWC and  $A_{254} : A_{365}$  in Fig. S3) decreased significantly with WTT in brown-water lakes in this study, which together with increasing  $a_{420} : \text{DOC}$  in the same lakes suggests that low-pigmented fractions were consumed more (and/or produced less) compared with colored bacterial substrates. Hypothetically, the microbial processing could have increased the pigmentation of DOC by the excretion of humic-like chromophoric molecules by bacteria (Shimotori et al., 2009; Tranvik, 1993; Guillemette and del Giorgio, 2012), although this appears less likely. Regardless, our results strongly suggest that microbial processes contributed to the increase in  $a_{420} : \text{DOC}$  and the strong decrease in  $A_{254} : A_{365}$  found coherently in both our dark biological decay experiment and in the brown-water lakes in situ. The fact that these lakes showed the same in situ optical changes over time as shown in the dark bioassays indicates that dark biological degradation processes dominated the DOC transformation in the brown-water lakes, probably because of high optical density, which prevented photodegradation from most of the water column.

For the clearest lakes, the prevailing mechanism behind the DOC transformation was obviously different than that in the brownest lakes, at least in the epilimnetic waters exposed to sunlight. The clear epilimnetic lake waters showed increases in  $A_{254} : A_{365}$ , which is the expected pattern for systems in which UV light transformation dominates (Dahlén et al., 1996) and LMWC is produced by photochemical processes (Bertilsson and Tranvik, 2000). In agreement, we also found that  $A_{254} : A_{365}$  increased in the laboratory UV light experiment. Thus, the relative role that UV light processing played for the DOC quality transformations was much larger in clear lakes than in highly colored lakes. In absolute terms, however, some studies suggest that the absolute light-induced color loss is similar for brown and clear lakes given equal incoming surface irradiation, even if this loss in color is distributed in different ways over the water column (Granéli et al., 1996; Koehler et al., 2014). In our study the mean absolute  $a_{420}$  loss in the four clearest epilimnetic sites was  $1.0 \text{ m}^{-1} \text{ year}^{-1}$  (Fig. 2d). Based on Granéli et al. (1996) it can be speculated that the same processing also happened in the most colored of our lakes but that this change was too small to be distinguished from the high background  $a_{420}$  level of  $11\text{--}13 \text{ m}^{-1}$  in these lakes.

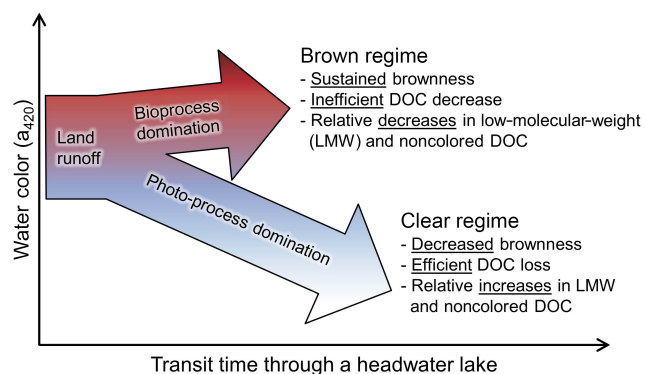
Survey lakes of intermediate color generally showed no change in the DOC quality transformation indicators with WTT (Fig. S5). Rather, the opposing influences on DOC

quality by microbial and photochemical processes appeared to have canceled each other out. Given potential positive feedbacks between these two processes on the overall DOC mineralization, for example the production of LMWC by photooxidation, which in turn fuels microbial degradation (Bertilsson and Tranvik, 2000), it would be theoretically possible that total DOC losses are largest in lakes of intermediate color. Therefore, it might not be a coincidence that the only two lakes showing significant DOC losses in our study (Mångstretjärn and Lapptjärn) had the intermediate mean color of  $5.5\text{--}6.3 \text{ m}^{-1}$  (see Fig. 2c; Table S1).

### 4.4 Comparison between temporal and spatial assessments

In spatial comparisons of lakes (or lake sub-basins) with increasing theoretical residence times, the color and DOC often decrease in a predictive way (Köhler et al., 2013; Curtis and Schindler, 1997) and colored DOC is generally lost preferentially over noncolored DOC (Weyhenmeyer et al., 2012). However, Berggren et al. (2009) demonstrated in a temporal analysis that, contrary to the expectation, the specific color of DOC can increase during water transit time through head-water lakes. The present study presents an explanation for such observations, showing that color per unit of DOC can either increase or decrease temporally in different types of lakes, likely depending on the relative importance of microbial and UV light processing, respectively. Thus, while there is an existing view that DOC transformation processes in lakes across the landscape can be understood based on spatial comparisons (Curtis and Schindler, 1997; Weyhenmeyer et al., 2012), we show here that temporal analyses are crucially needed to draw correct conclusions about the change in DOC with WTT.

Comparing the mean  $a_{420}$  and  $a_{420} : \text{DOC}$  values with mean WTT of the different lakes (Table 1; Fig. 2d), it appears that color is rapidly and preferentially lost across all lakes combined. This brings up a paradox: if it is not WTT per se that is responsible for the patterns in color and DOC properties, why do short-turnover lakes systematically have higher color and  $a_{420} : \text{DOC}$  compared with long-turnover lakes? Here we are not able to come up with a general solution, but it should be stressed that our specific study lakes have catchments of differing properties and hydrological functioning likely influencing the observed patterns. In particular, the five lakes with the shortest WTTs have relatively large catchment areas drained by streams, whereas the four lakes with the shortest WTTs lack permanent streams (see Table 1 and Text S1). This is important because lakes with inflowing streams are extensively connected to riparian soils that are key sources for DOC (Laudon et al., 2011), whereas lakes without inlets receive relatively more input from DOC-poor deeper groundwater (Tiwari et al., 2014) that has low color per unit DOC (van Verseveld et al., 2008). Moreover, the five short-WTT lakes are located in flat lower parts of the



**Figure 5.** Conceptual representation of the trajectory of color and dissolved organic carbon (DOC) characteristics in headwater lakes with increasing residence time under brown and clear lake regimes, respectively. The figure is based on the observations from this study in which DOC processing in brown lakes is characterized by microbial consumption of low-molecular-weight (LMW) and noncolored DOC. Only the clear-water lakes show efficient reductions in color and DOC over time facilitated by photochemical processing that removes colored DOC to a larger extent than microbial processes. Compare with original data from Figs. 2–4.

catchments with large direct connections to peatland areas, while the remaining lakes are surrounded by hilly forest. As peatlands release larger amounts of highly colored DOC than forests (Ågren et al., 2008b), high peatland connectivity plausibly contributed to the brown character of the short-WTT lakes. Thus, the lakes with long WTTs in this study may not be clear primarily because of their long water transit times, but because they are disconnected from the key sources of colored DOC in the catchment, i.e., peatlands and upland riparian soils.

#### 4.5 Conceptual implications

Altogether our results suggest a conceptual framework for DOC transformation (Fig. 5) with distinctly different color trajectories under “brown-water” and “clear-water” regimes. In the brown-water regime, biological processes dominate, leading to small changes in color over time. In a clear-water state, photochemical processes play a relatively much larger role, resulting in substantial decreases over time in both DOC concentrations and in color. From this proposed concept (Fig. 5), the circumstances that allow for the development of a lake into a brown-water or clear-water system can be discussed. Considering that boreal watercourses show at least 2 orders of magnitude of spatial variation in color (Lapierre et al., 2013; Temnerud et al., 2014), the color of the inflowing catchment runoff water is beyond doubt a key factor determining the regime. Additionally, the potential for regime shifts to occur in lakes that are close to the border between the clear- and brown-water regime can be speculated on, for example by rapidly increased input of unprocessed

DOC from the catchment during extreme discharge episodes as described by Raymond et al. (2016). However, shifting a brown regime into a clear regime should be relatively difficult given the inefficient color loss in brown headwater lakes (Fig. 5). This view is consistent with laboratory studies indicating that biological decay needs to proceed for a few years before substantial DOC and color exhaustion in brown water lakes (Koehler et al., 2012).

Our results do not contradict the findings from previous studies that have proposed the selective loss of pigmented DOC in freshwater networks (Ilina et al., 2014; Weyhenmeyer et al., 2012; Köhler et al., 2013). Rather, this study can help explain why previous studies have not been able to detect decreasing color with increasing water residence time in headwaters (Müller et al., 2013) where many of the lakes presumably follow the dynamics of the brown-water regime (see Fig. 5). Decreasing absorbance with increasing water residence time has mainly been possible to model at non-headwater sites with a relatively low color level (Müller et al., 2013; Weyhenmeyer et al., 2012). For such sites, a decrease in color per unit DOC could be expected from photo-processing (Molot and Dillon, 1997), iron flocculation (Weyhenmeyer et al., 2014), or DOC replenishment along the aquatic continuum, for example from algal sources that selectively add carbon with a low degree of pigmentation (Creed et al., 2015).

#### 5 Summary and conclusions

In summary, our results exemplify how individual brown-water lakes may not conform to the general reported pattern of the selective removal of colored constituents in freshwaters, but rather show a sustained level of pigmentation regardless of WTT variations. Thus change in WTT, for example due to a potentially wetter future climate, has no universal effect on lake color, at least not over the ranges in WTT that we studied. However, if combined with changes in the absorbance of catchment runoff water, an intensified hydrological cycle could possibly cause regime shifts in headwater lakes in which clear-water lakes renewed with more colored water relatively quickly will transform into brown-water systems. Conceptually, our study challenges the view of the aquatic network as a single continuum of DOC processing. In headwaters, the functioning of different aquatic networks depends on which DOC transformation process dominates.

*Data availability.* All relevant raw data are presented graphically in the Supplement to this article. These data will be made available in table format by the corresponding author upon request.

**The Supplement related to this article is available online at <https://doi.org/10.5194/bg-15-457-2018-supplement>.**

*Competing interests.* The authors declare that they have no conflict of interest.

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## References

- Amon, R. M. W. and Benner, R.: Photochemical and microbial consumption of dissolved organic carbon and dissolved oxygen in the Amazon River system, *Geochim. Cosmochim. Ac.*, 60, 1783–1792, 1996.
- Anesio, A. M. and Granéli, W.: Increased photoreactivity of DOC by acidification: Implications for the carbon cycle in humic lakes, *Limnol. Oceanogr.*, 48, 735–744, 2003.
- Asmala, E., Autio, R., Kaartokallio, H., Stedmon, C. A., and Thomas, D. N.: Processing of humic-rich riverine dissolved organic matter by estuarine bacteria: effects of predegradation and inorganic nutrients, *Aquat. Sci.*, 76, 451–463, <https://doi.org/10.1007/s00027-014-0346-7>, 2014.
- Ågren, A., Berggren, M., Laudon, H., and Jansson, M.: Terrestrial export of highly bioavailable carbon from small boreal catchments in spring floods, *Freshwater Biol.*, 53, 964–972, <https://doi.org/10.1111/j.1365-2427.2008.01955.x>, 2008a.
- Ågren, A., Buffam, I., Berggren, M., Bishop, K., Jansson, M., and Laudon, H.: Dissolved organic carbon characteristics in boreal streams in a forest-wetland gradient during the transition between winter and summer, *J. Geophys. Res.-Bio.*, 113, G03031, <https://doi.org/10.1029/2007JG000674>, 2008b.
- Berggren, M., Laudon, H., and Jansson, M.: Landscape regulation of bacterial growth efficiency in boreal freshwaters, *Global Biogeochem. Cy.*, 21, GB4002, <https://doi.org/10.1029/2006GB002844>, 2007.
- Berggren, M., Laudon, H., and Jansson, M.: Aging of allochthonous organic carbon regulates bacterial production in unproductive boreal lakes, *Limnol. Oceanogr.*, 54, 1333–1342, <https://doi.org/10.4319/lo.2009.54.4.1333>, 2009.
- Berggren, M., Laudon, H., Haei, M., Ström, L., and Jansson, M.: Efficient aquatic bacterial metabolism of dissolved low molecular weight compounds from terrestrial sources, *Isme J.*, 4, 408–416, <https://doi.org/10.1038/ismej.2009.120>, 2010a.
- Berggren, M., Ström, L., Laudon, H., Karlsson, J., Giesler, R., Bergström, A. K., and Jansson, M.: Lake secondary production fueled by rapid transfer of low molecular weight organic carbon from terrestrial sources to aquatic consumers, *Ecol. Lett.*, 13, 870–880, <https://doi.org/10.1111/j.1461-0248.2010.01483.x>, 2010b.
- Berggren, M., Bergström, A. K., and Karlsson, J.: Intraspecific autochthonous and allochthonous resource use by zooplankton in a humic lake during the transitions between winter, summer and fall, *PLoS One*, 10, e0120575, <https://doi.org/10.1371/journal.pone.0120575>, 2015a.
- Berggren, M., Sponseller, R. A., Alves Soares, A. R., and Bergström, A. K.: Toward an ecologically meaningful view of resource stoichiometry in DOM-dominated aquatic systems, *J. Plankt. Res.*, 37, 489–499, <https://doi.org/10.1093/plankt/fbv018>, 2015b.
- Bertilsson, S. and Tranvik, L. J.: Photochemical transformation of dissolved organic matter in lakes, *Limnol. Oceanogr.*, 45, 753–762, 2000.
- Boyer, E. W., Hornberger, G. M., Bencala, K. E., and Mcknight, D. M.: Response characteristics of DOC flushing in an alpine catchment, *Hydrol. Proc.*, 11, 1635–1647, 1997.
- Cole, J. J., Prairie, Y. T., Caraco, N. F., McDowell, W. H., Tranvik, L. J., Striegl, R. G., Duarte, C. M., Kortelainen, P., Downing, J. A., Middelburg, J. J., and Melack, J.: Plumbing the global carbon cycle: Integrating inland waters into the terrestrial carbon budget, *Ecosyst.*, 10, 171–184, 2007.
- Cory, R. M., Ward, C. P., Crump, B. C., and Kling, G. W.: Sunlight controls water column processing of carbon in arctic fresh waters, *Science*, 345, 925–928, <https://doi.org/10.1126/science.1253119>, 2014.
- Creed, I. F., McKnight, D. M., Pellerin, B. A., Green, M. B., Bergamaschi, B. A., Aiken, G. R., Burns, D. A., Findlay, S. E. G., Shanley, J. B., Striegl, R. G., Aulenbach, B. T., Clow, D. W., Laudon, H., McGlynn, B. L., McGuire, K. J., Smith, R. A., and Stackpoole, S. M.: The river as a chemostat: fresh perspectives on dissolved organic matter flowing down the river continuum, *Can. J. Fish. Aquat. Sci.*, 72, 1272–1285, <https://doi.org/10.1139/cjfas-2014-0400>, 2015.
- Curtis, P. J. and Schindler, D. W.: Hydrologic control of dissolved organic matter in low-order Precambrian Shield Lakes, *Biogeochemistry*, 36, 125–138, <https://doi.org/10.1023/A:1005787913638>, 1997.
- Dahlén, J., Bertilsson, S., and Pettersson, C.: Effects of UV-A irradiation on dissolved organic matter in humic surface waters, *Environ. Int.*, 22, 501–506, 1996.
- Daniel, C., Gutseit, K., Anesio, A. M., and Granéli, W.: Microbial food webs in the dark: independence of lake plankton from recent algal production, *Aquat. Microb. Ecol.*, 38, 113–123, 2005.
- Fasching, C., Behounek, B., Singer, G. A., and Battin, T. J.: Microbial degradation of terrigenous dissolved organic matter and potential consequences for carbon cycling in brown-water streams, *Sci. Rep.*, 4, 4981, <https://doi.org/10.1038/Srep04981>, 2014.
- Fasching, C., Ulseth, A. J., Schelker, J., Steniczka, G., and Battin, T. J.: Hydrology controls dissolved organic matter export and composition in an Alpine stream and its hyporheic zone, *Limnol. Oceanogr.*, 61, 558–571, <https://doi.org/10.1002/lno.10232>, 2016.

- Gennings, C., Molot, L. A., and Dillon, P. J.: Enhanced photochemical loss of organic carbon in acidic waters, *Biogeochemistry*, 52, 339–354, <https://doi.org/10.1023/a:1006499713835>, 2001.
- Gonsior, M., Schmitt-Kopplin, P., and Bastviken, D.: Depth-dependent molecular composition and photo-reactivity of dissolved organic matter in a boreal lake under winter and summer conditions, *Biogeosciences*, 10, 6945–6956, <https://doi.org/10.5194/bg-10-6945-2013>, 2013.
- Granéli, W., Lindell, M., and Tranvik, L.: Photo-oxidative production of dissolved inorganic carbon in lakes of different humic content, *Limnol. Oceanogr.*, 41, 698–706, 1996.
- Guillemette, F. and del Giorgio, P. A.: Simultaneous consumption and production of fluorescent dissolved organic matter by lake bacterioplankton, *Environ. Microbiol.*, 14, 1432–1443, <https://doi.org/10.1111/j.1462-2920.2012.02728.x>, 2012.
- Hansen, A. M., Kraus, T. E. C., Pellerin, B. A., Fleck, J. A., Downing, B. D., and Bergamaschi, B. A.: Optical properties of dissolved organic matter (DOM): Effects of biological and photolytic degradation, *Limnol. Oceanogr.*, 61, 1015–1032, <https://doi.org/10.1002/lno.10270>, 2016.
- Illina, S. M., Drozdova, O. Y., Lapitskiy, S. A., Alekhin, Y. V., Demin, V. V., Zavgorodnyaya, Y. A., Shirokova, L. S., Viers, J., and Pokrovsky, O. S.: Size fractionation and optical properties of dissolved organic matter in the continuum soil solution-bog-river and terminal lake of a boreal watershed, *Org. Geochem.*, 66, 14–24, <https://doi.org/10.1016/j.orggeochem.2013.10.008>, 2014.
- Jansson, M., Bergström, A. K., Drakare, S., and Blomqvist, P.: Nutrient limitation of bacterioplankton and phytoplankton in humic lakes in northern Sweden, *Freshwater Biol.*, 46, 653–666, 2001.
- Jansson, M., Persson, L., DeRoos, A. M., Jones, R. I., and Tranvik, L. J.: Terrestrial carbon and intraspecific size-variation shape lake ecosystems, *Trends Ecol. Evol.*, 22, 316–322, <https://doi.org/10.1016/j.tree.2007.02.015>, 2007.
- Jansson, M., Berggren, M., Laudon, H., and Jonsson, A.: Bioavailable phosphorus in humic headwater streams in boreal Sweden, *Limnol. Oceanogr.*, 57, 1161–1170, <https://doi.org/10.4319/lno.2012.57.4.1161>, 2012.
- Jonsson, A., Meili, M., Bergström, A. K., and Jansson, M.: Whole-lake mineralization of allochthonous and autochthonous organic carbon in a large humic lake (Örtrasket, N. Sweden), *Limnol. Oceanogr.*, 46, 1691–1700, 2001.
- Karlsson, J., Byström, P., Ask, J., Ask, P., Persson, L., and Jansson, M.: Light limitation of nutrient-poor lake ecosystems, *Nature*, 460, 506–509, <https://doi.org/10.1038/nature08179>, 2009.
- Karlsson, J., Berggren, M., Ask, J., Byström, P., Jonsson, A., Laudon, H., and Jansson, M.: Terrestrial organic matter support of lake food webs: Evidence from lake metabolism and stable hydrogen isotopes of consumers, *Limnol. Oceanogr.*, 57, 1042–1048, <https://doi.org/10.4319/lno.2012.57.4.1042>, 2012.
- Kellerman, A. M., Dittmar, T., Kothawala, D. N., and Tranvik, L. J.: Chemodiversity of dissolved organic matter in lakes driven by climate and hydrology, *Nat. Commun.*, 5, 3804, <https://doi.org/10.1038/Ncomms4804>, 2014.
- Klaus, M., Bergström, A.-K., Jonsson, A., Deininger, A., Geibrink, E., and Karlsson, J.: Weak response of greenhouse gas emissions to whole lake N enrichment, *Limnol. Oceanogr.*, <https://doi.org/10.1002/lno.10743>, 2017.
- Koehler, B., von Wachenfeldt, E., Kothawala, D., and Tranvik, L. J.: Reactivity continuum of dissolved organic carbon decomposition in lake water, *J. Geophys. Res.-Biogeo.*, 117, G01024, <https://doi.org/10.1029/2011jg001793>, 2012.
- Koehler, B., Landelius, T., Weyhenmeyer, G. A., Machida, N., and Tranvik, L. J.: Sunlight-induced carbon dioxide emissions from inland waters, *Global Biogeochem. Cy.*, 28, 696–711, <https://doi.org/10.1002/2014gb004850>, 2014.
- Kritzberg, E. S. and Ekström, S. M.: Increasing iron concentrations in surface waters – a factor behind brownification?, *Biogeosciences*, 9, 1465–1478, <https://doi.org/10.5194/bg-9-1465-2012>, 2012.
- Köhler, S. J., Kothawala, D., Futter, M. N., Liungman, O., and Tranvik, L.: In-lake processes offset increased terrestrial inputs of dissolved organic carbon and color to lakes, *PLoS One*, 8, e70598, <https://doi.org/10.1371/journal.pone.0070598>, 2013.
- Lapierre, J. F., Guillemette, F., Berggren, M., and del Giorgio, P. A.: Increases in terrestrially derived carbon stimulate organic carbon processing and CO<sub>2</sub> emissions in boreal aquatic ecosystems, *Nat. Commun.*, 4, 2972, <https://doi.org/10.1038/ncomms3972>, 2013.
- Laudon, H., Berggren, M., Ågren, A., Buffam, I., Bishop, K., Grabs, T., Jansson, M., and Köhler, S.: Patterns and dynamics of dissolved organic carbon (DOC) in boreal streams: The role of processes, connectivity, and scaling, *Ecosystems*, 14, 880–893, <https://doi.org/10.1007/s10021-011-9452-8>, 2011.
- Lepistö, A., Futter, M. N., and Kortelainen, P.: Almost 50 years of monitoring shows that climate, not forestry, controls long-term organic carbon fluxes in a large boreal watershed, *Glob. Change Biol.*, 20, 1225–1237, <https://doi.org/10.1111/Gcb.12491>, 2014.
- Li, Y. L., Zhang, Q., and Yao, J.: Investigation of residence and travel times in a large floodplain lake with complex lake-river interactions: Poyang Lake (China), *Water-Sui*, 7, 1991–2012, <https://doi.org/10.3390/w7051991>, 2015.
- Lindell, M. J., Granéli, H., and Bertilsson, S.: Seasonal photoreactivity of dissolved organic matter from lakes with contrasting humic content, *Can. J. Fish. Aquat. Sci.*, 57, 875–885, <https://doi.org/10.1139/cjfas-57-5-875>, 2000.
- Molot, L. A. and Dillon, P. J.: Photolytic regulation of dissolved organic carbon in northern lakes, *Global Biogeochem. Cy.*, 11, 357–365, 1997.
- Monteith, D. T., Stoddard, J. L., Evans, C. D., de Wit, H. A., Forsius, M., Hogasen, T., Wilander, A., Skjelkvale, B. L., Jeffries, D. S., Vuorenmaa, J., Keller, B., Kopacek, J., and Vesely, J.: Dissolved organic carbon trends resulting from changes in atmospheric deposition chemistry, *Nature*, 450, 537–540, <https://doi.org/10.1038/nature06316>, 2007.
- Müller, R. A., Futter, M. N., Sobek, S., Nisell, J., Bishop, K., and Weyhenmeyer, G. A.: Water renewal along the aquatic continuum offsets cumulative retention by lakes: implications for the character of organic carbon in boreal lakes, *Aquat. Sci.*, 75, 535–545, <https://doi.org/10.1007/s00027-013-0298-3>, 2013.
- Oleinikova, O. V., Shirokova, L. S., Gerard, E., Drozdova, O. Y., Lapitskiy, S. A., Bychkov, A. Y., and Pokrovsky, O. S.: Transformation of organo-ferric peat colloids by a heterotrophic bacterium, *Geochim. Cosmochim. Ac.*, 205, 313–330, <https://doi.org/10.1016/j.gca.2017.02.029>, 2017.
- Pace, M. L., Reche, I., Cole, J. J., Fernandez-Barbero, A., Mazuecos, I. P., and Prairie, Y. T.: pH change induces shifts in the size and light absorption of dissolved organic matter, *Biogeochemistry*, 108, 109–118, <https://doi.org/10.1007/s10533-011-9576-0>, 2012.

- Pagano, T., Bida, M., and Kenny, J. E.: Trends in levels of allochthonous dissolved organic carbon in natural water: a review of potential mechanisms under a changing climate, *Water-Sui.*, 6, 2862–2897, <https://doi.org/10.3390/W6102862>, 2014.
- Pullin, M. J., Anthony, C., and Maurice, P. A.: Effects of iron on the molecular weight distribution, light absorption, and fluorescence properties of natural organic matter, *Environ. Eng. Sci.*, 24, 987–997, <https://doi.org/10.1089/ees.2006.0040>, 2007.
- Raymond, P. A., Hartmann, J., Lauerwald, R., Sobek, S., McDonald, C., Hoover, M., Butman, D., Striegl, R., Mayorga, E., Humborg, C., Kortelainen, P., Durr, H., Meybeck, M., Ciais, P., and Guth, P.: Global carbon dioxide emissions from inland waters, *Nature*, 503, 355–359, <https://doi.org/10.1038/Nature12760>, 2013.
- Raymond, P. A., Saiers, J. E., and Sobczak, W. V.: Hydrological and biogeochemical controls on watershed dissolved organic matter transport: pulse-shunt concept, *Ecology*, 97, 5–16, <https://doi.org/10.1890/14-1684.1>, 2016.
- Reche, I., Pace, M. L., and Cole, J. J.: Relationship of trophic and chemical conditions to photobleaching of dissolved organic matter in lake ecosystems, *Biogeochemistry*, 44, 259–280, 1999.
- Rueda, F., Moreno-Ostos, E., and Armengol, J.: The residence time of river water in reservoirs, *Ecol. Model.*, 191, 260–274, <https://doi.org/10.1016/j.ecolmodel.2005.04.030>, 2006.
- Shimotori, K., Omori, Y., and Hama, T.: Bacterial production of marine humic-like fluorescent dissolved organic matter and its biogeochemical importance, *Aquat. Microb. Ecol.*, 58, 55–66, <https://doi.org/10.3354/ame01350>, 2009.
- Soares, A. R. A., Bergström, A.-K., Sponseller, R. A., Moberg, J. M., Giesler, R., Kritzberg, E. S., Jansson, M., and Berggren, M.: New insights on resource stoichiometry: assessing availability of carbon, nitrogen, and phosphorus to bacterioplankton, *Biogeosciences*, 14, 1527–1539, <https://doi.org/10.5194/bg-14-1527-2017>, 2017.
- Solomon, C. T., Jones, S. E., Weidel, B. C., Buffam, I., Fork, M. L., Karlsson, J., Larsen, S., Lennon, J. T., Read, J. S., Sadro, S., and Saros, J. E.: Ecosystem consequences of changing inputs of terrestrial dissolved organic matter to lakes: current knowledge and future challenges, *Ecosystems*, 18, 376–389, <https://doi.org/10.1007/s10021-015-9848-y>, 2015.
- Ström, L., Tagesson, T., Mastepanov, M., and Christensen, T. R.: Presence of *Eriophorum scheuchzeri* enhances substrate availability and methane emission in an Arctic wetland, *Soil Biol. Biochem.*, 45, 61–70, <https://doi.org/10.1016/j.soilbio.2011.09.005>, 2012.
- Stubbins, A., Spencer, R. G. M., Chen, H. M., Hatcher, P. G., Mopper, K., Hernes, P. J., Mwamba, V. L., Mangangu, A. M., Wabakanghanzi, J. N., and Six, J.: Illuminated darkness: Molecular signatures of Congo River dissolved organic matter and its photochemical alteration as revealed by ultrahigh precision mass spectrometry, *Limnol. Oceanogr.*, 55, 1467–1477, <https://doi.org/10.4319/lo.2010.55.4.1467>, 2010.
- Temnerud, J., Hytteborn, J. K., Futter, M. N., and Kohler, S. J.: Evaluating common drivers for color, iron and organic carbon in Swedish watercourses, *Ambio*, 43, 30–44, <https://doi.org/10.1007/s13280-014-0560-5>, 2014.
- Tiwari, T., Laudon, H., Beven, K., and Ågren, A. M.: Downstream changes in DOC: Inferring contributions in the face of model uncertainties, *Water Resour. Res.*, 50, 514–525, <https://doi.org/10.1002/2013wr014275>, 2014.
- Tranvik, L. J.: Availability of dissolved organic carbon for planktonic bacteria in oligotrophic lakes of differing humic content, *Microb. Ecol.*, 16, 311–322, 1988.
- Tranvik, L. J.: Microbial transformation of labile dissolved organic-matter into humic-like matter in seawater, *FEMS Microbiol. Ecol.*, 12, 177–183, 1993.
- Vachon, D., Lapierre, J.-F., and del Giorgio, P. A.: Seasonality of photochemical dissolved organic carbon mineralization and its relative contribution to pelagic CO<sub>2</sub> production in northern lakes, *J. Geophys. Res.-Biogeosci.*, 121, 2015JG003244, <https://doi.org/10.1002/2015JG003244>, 2016.
- van Verseveld, W. J., McDonnell, J. J., and Lajtha, K.: A mechanistic assessment of nutrient flushing at the catchment scale, *J. Hydrol.*, 358, 268–287, <https://doi.org/10.1016/j.jhydrol.2008.06.009>, 2008.
- Verpoorter, C., Kutser, T., Seekell, D. A., and Tranvik, L. J.: A global inventory of lakes based on high-resolution satellite imagery, *Geophys. Res. Lett.*, 41, 6396–6402, <https://doi.org/10.1002/2014gl060641>, 2014.
- Weishaar, J. L., Aiken, G. R., Bergamaschi, B. A., Fram, M. S., Fujii, R., and Mopper, K.: Evaluation of specific ultraviolet absorbance as an indicator of the chemical composition and reactivity of dissolved organic carbon, *Environ. Sci. Technol.*, 37, 4702–4708, <https://doi.org/10.1021/es030360x>, 2003.
- Wetzel, R. G.: *Limnology: Lake and river ecosystems*, 3 Edn., Academic Press, San Diego, 1006 pp., 2001.
- Weyhenmeyer, G. A., Froberg, M., Karlton, E., Khalili, M., Kothawala, D., Temnerud, J., and Tranvik, L. J.: Selective decay of terrestrial organic carbon during transport from land to sea, *Glob. Change Biol.*, 18, 349–355, <https://doi.org/10.1111/j.1365-2486.2011.02544.x>, 2012.
- Weyhenmeyer, G. A., Prairie, Y. T., and Tranvik, L. J.: Browning of boreal freshwaters coupled to carbon-iron interactions along the aquatic continuum, *PLoS One*, 9, e88104, <https://doi.org/10.1371/journal.pone.0088104>, 2014.
- Wilson, H. F., Saiers, J. E., Raymond, P. A., and Sobczak, W. V.: Hydrologic drivers and seasonality of dissolved organic carbon concentration, nitrogen content, bioavailability, and export in a forested New England stream, *Ecosystems*, 16, 604–616, <https://doi.org/10.1007/s10021-013-9635-6>, 2013.
- Xiao, Y. H., Raike, A., Hartikainen, H., and Vahatalo, A. V.: Iron as a source of color in river waters, *Sci. Tot. Environ.*, 536, 914–923, <https://doi.org/10.1016/j.scitotenv.2015.06.092>, 2015.