Water renewal along the aquatic continuum offsets cumulative retention by lakes – implications for the character of organic carbon in boreal lakes

R.A. Müller[1,*], M.N. Futter[2], S. Sobek[1], J. Nisell[2], K. Bishop[2], G.A. Weyhenmeyer[1]

[1] Department of Ecology and Genetics/Limnology, Uppsala University, Norbyvägen 18D, 752 36 Uppsala, Sweden

[2] Department of Aquatic Sciences and Assessment, Swedish University of Agricultural Sciences (SLU), Box 7050, 750 07 Uppsala, Sweden

* Corresponding author: roger.muller@ebc.uu.se, Phone: +46 (0)18 471 27 25, Fax: +46 (0)18 471 64 84

Abstract

The character of organic carbon (OC) in lake waters is strongly dependent on the time water has spent in the landscape as well as in the lake itself due to continuous biogeochemical OC transformation processes. A common view is that upstream lakes might prolong the water retention in the landscape, resulting in an altered OC character downstream. We calculated the number of lakes upstream for 24,742 Swedish lakes in seven river basins spanning from 56º to 68º N. For each of these lakes, we used a lake volume to discharge comparison on a landscape scale to account for upstream water retention by lakes ($T_{n\ tot}$). We found a surprisingly weak relationship between the number of lakes upstream and $T_{n\ tot.}$ Accordingly, we found that the coloured fraction of organic carbon was not related to lake landscape position but significantly related to $T_{n\ tot}$ when we analysed lake water chemical data from 1,559 lakes in the studied river basins (R^2 = 0.21, $p < 0.0001$). Thus, we conclude that water renewal along the aquatic continuum by lateral water inputs offsets cumulative retention by lakes. Based on our findings, we suggest integrating $T_{n\ tot}$ in studies that address lake landscape position in the boreal zone to better understand variations in the character of organic carbon across lake districts.

Keywords: lake, landscape, time, organic carbon, colour

Introduction

The rate at which lake water is being renewed is crucial to time-dependent lake processes such as nutrient budgets (Vollenweider 1976; Hecky et al. 1993), carbon processing (Algesten et al. 2004; Battin et al. 2009) and lake water clarity (Schindler et al. 1996; Weyhenmeyer et al. 2012). During the last decade, more and more aquatic studies addressed a larger landscape, where it has been suggested that water may become older and the material in the water more processed the further down the water moves in the landscape (Egeberg et al. 1999; Steinberg 2003). This is certainly true for a single molecule travelling from headwaters via a series of lakes to the sea. However, lakes that are situated at low landscape positions not only receive water that has aged in lakes upstream but also from rainfall, groundwater and lateral tributary inflow. Thus, the age of water in lakes in the landscape is a result of complex runoff and mixing processes in the catchment (Lindström et al. 1997; Canham and Pace 2009).

Studies addressing lake landscape position indicate that the presence of upstream lakes affects downstream water quality at both the temporal (Goodman et al. 2011) and the spatial (Larson et al. 2007) scale. In situations where water follows specific lake-chains from headwater lakes towards low landscape positions, material processing has been shown to occur (Soranno et al. 1999; Sadro et al. 2012). However, hydrological studies on drainage systems indicate that water residence time does not necessarily scale with catchment size, as observed for headwater systems (McGuire et al 2005; McDonnell et al. 2010) and along streams and rivers (Tokunaga 2003; Alexander et al. 2007). Alexander et al. (2007) demonstrated the important role of tributary streams for sustaining efficient water renewal throughout the entire drainage network of the northeastern United States. Martin and Soranno (2006) underpinned the relevance of such water renewal for lake-chains, as they identified stream connection as a major driver of variation in lake water chemistry and clarity across different landscape positions. The efficiency of water renewal occurring in a larger drainage network is therefore fundamental to ecosystem functioning of lakes and reservoirs.

Lake volume and water discharge determine water retention by lakes, here referred to as lake water retention time and used synonymously with water renewal time (Schindler et al. 1992; Wetzel et al. 2001) and/or residence time (Monsen et al. 2002; Algesten et al. 2004). A simple measure of water retention time for single lake water bodies can be defined as the ratio between lake volume and mean discharge through the lake outflow (T_n) and is frequently applied to indicate and compare general water exchange properties of lakes (Bolin and Rodhe 1973; Monsen et al. 2002). In the Experimental Lakes Area in Canada, comprising 58 small lakes with catchments of 0.1 to 8.4 km^2 in size, Schindler et al. (1992; 1996; 1997) have, for example, discussed OC concentrations, nutrients and water colour based on T_n. Based on empirical models, carbon sedimentation in lakes and reservoirs (Cole et al. 2007), carbon transport (Weyhenmeyer et al. 2012) and mineralization (Algesten et al. 2004) have found to be related to lake water retention times. Hence, the processing and transport of carbon in lakes is a function of time. However, estimates of water retention times are frequently applied and compared with limited knowledge of their actual distribution on a larger landscape, and the cumulative water retention by lakes remains largely unknown.

This study addresses water retention by lakes and its relevance for the character of OC. We used a dataset of 24,742 lakes (>0.01 km^2) from seven river basins in Sweden, covering a 12º latitudinal range. For each lake, we estimated two measures of water retention time. The first estimated measure (T_n) corresponded to the mean water retention time for a single lake (n), as defined above. The second estimated measure ($T_{n\ tot}$) included the sum of all lake volumes upstream of a lake (n), indicating a mean for the water retention by lakes on a landscape scale (Algesten et al. 2004). We hypothesized that lake landscape position is related to $T_{n\ tot}$ and also to the coloured fraction of OC. We further hypothesized that T_n is strongly related to the coloured fraction of OC as previously observed (Weyhenmeyer et al., 2012) but that the relationship becomes stronger when T_n is replaced by $T_{n\ tot}$. To test these hypotheses, we used lake morphometric and hydrological data, as well as lake water chemical data from the Swedish national lake monitoring program.

Materials and Methods

Datasets

Our lake dataset included 24,742 lakes larger than 0.01 km^2 from seven river basins: (1) Lagan, (2) Mälaren, (3) Dalälven, (4) Ljungan, (5) Ångermanälven, (6) Umeälven and (7) Kalixälven (Fig. 1). The lakes represent approximately 25.9 % of all lake water bodies (>0.01 km^2) in the Swedish Lake Registry (ca. 95,700 lakes) as provided by the Swedish Meteorological and Hydrological Institute (SMHI). We chose the seven river basins (Table 1) according to the following criteria: (1) estimates of long-term (1961 to 1990) mean discharge were available for the entire basin, (2) volume estimates for large lakes and reservoirs were available, (3) the river basins were preferably large in size and (4) well distributed over Sweden. For each lake (n), we estimated its volume (V_n), used the estimate of long-term mean discharge (Q_n), counted the number of lakes (>0.01 km^2) upstream, summed the lake volumes in the catchment including the volume of the lake (n) at hand ($V_{n\ tot}$) and determined our two measures of water retention time (T_n, $T_{n\ tot}$, see below).

Lake water chemistry data from the Swedish national lake monitoring program were made available by the Swedish University of Agricultural Sciences (SLU). We used water chemical data from lake water sampling campaigns run in the years 1995, 2000 and 2005 where a total of 1,559 lakes in the selected seven river basins were sampled. Some lakes were sampled several times, while others were sampled only once. We analysed the dataset for each year separately and additionally calculated a long-term lake-specific median value. Sampled lakes were well distributed across the seven river basins (Fig. 1). Sampling was conducted geographically from north to south by a helicopter. Samples were taken at a depth of 0.5 m in the central part of each lake. Sampling periods stretched from September until November, when lake waters underwent autumn water column mixing and surface water samples become more representative of a lake's water chemistry (Göransson et al. 2004). Variables considered in this study were absorbance at 420 nm, measured on filtered (0.45 μm) samples in a 5 cm quartz cuvette ($AbsF_{420}$; reported with an analytical precision of 12 % according to method SS-EN ISO 7887 v.1) and total organic carbon (TOC; reported with an analytical precision of 11 % according to method SS-EN 1484 v.1). Absorbance data were converted to Napierian absorption coefficients a_{420} [m^{-1}], Eq. 1:

$$a_{420} = \mathrm{AbsF}_{420} \times \ln(10) \times \mathrm{L}^{-1} \tag{1}$$

where $AbsF_{420}$ is the absorbance measured, ln(10) is the natural logarithm of 10 and L is the optical path-length [m], after Kirk (1994) and as recommended by Hu et al. (2002). TOC in Swedish lakes generally contains 97 ± 5 % dissolved organic carbon (DOC) (von Wachenfeldt and Tranvik 2008) and is therefore frequently used as a proxy for DOC. We calculated the ratio a_{420} [m^{-1}] over TOC [mgL^{-1}] which reflects specific absorbance at 420 nm and is thereby used as a measure of the character of OC (Wetzel 2001; Sulzberger and Durisch-Kaiser 2009).

Lake volume estimates

Information on measured lake volumes for a total of 1,263 lakes was available from the Swedish Lake Registry (2009). Though these lakes represent only a minor fraction of the lake dataset by count (5.1 %), they include the majority of large lakes. To estimate the lake volume of the remaining 23,479 lakes, we applied an existing empirically derived lake volume estimator after Sobek et al. (2011), Eq. 2 and Eq. 3:

$$\ln(\mathrm{V_{n\ unt}}) = 0.75 + 1.06 \times \ln(\mathrm{A}) + 0.056 \times \gamma_{\mathrm{max50}} \qquad (2)$$

where $\ln(V_{n\ unt})$ [-] is the natural logarithm of the lake volume estimate ($V_{n\ unt}$) [Mm^3] before correcting it for log-transformation bias, ln(A) the natural logarithm of the lake area (A) [km^2] and γ_{max50} the maximum slope [degree] within a 50 m buffer around the lake water body (n). And

$$\mathrm{V_n} = \exp\big(\ln(\mathrm{V_{n\ unt}})\big) \times \exp(0.5 \times \mathrm{s}^2) \qquad (3)$$

where V_n [Mm^3] is the lake volume estimate corrected for log-transformation bias and s^2 the residual variance of the volume estimate. Eq. 2 explained 93 % of the variability in lake volumes (V_n). The model was calibrated with known lake volumes from the entire Swedish Lake Registry (6,943 lakes) and GIS derived morphological data (γ_{max50}). The residual variance (s^2) used in Eq. 3 was 0.284 and was used to correct the volume estimates for log-transformation bias; for more detailed information on this model, we refer to Sobek et al. (2011). Published lake volumes were available for most large lakes in the selected seven river basins, but there were 49 lakes (10.04 to 60.69 km^2) beyond the size range (>10 km^2) used by Sobek et al. (2011), for which published lake volume estimates were not available. According to Sobek et al. (2011), extrapolation to lakes >10km^2 was justified, and we therefore applied the volume estimator without further calibration. Sobek et al. (2011) noted that their volume estimates performed well as long as the lake count is kept high ($n \geq 15$). While the estimated volume of an individual lake had a considerable degree of uncertainty, the relative error of cumulative lake volumes was found to decrease during error propagation with the number of lakes by $n^{(-0.5)}$, such that Eq. 2 is suitable to estimate lake volumes at a catchment or landscape scale. In our analyses, volume estimates were rounded to 10^3 m^3.

Long-term (30 years) mean discharge

We used estimates of long-term mean discharge (Qn) [m3d-1] at the outflow of each lake (n), which were generated by the HBV (Hydrologiska Byråns Vattenbalansavdelning) runoff model (Lindström et al. 1997) for the Swedish reference period 1 January 1961 to 31 December 1990. Data were available from the SMHI as a raster data file with a resolution of 50×50 m. The input data to the HBV runoff model are observations of precipitation, air temperature and estimates of evapotranspiration. For more detailed information on the HBV runoff model, we refer to Lindström et al. (1997). We tested the accuracy of Q_n estimates by calculating long-term mean runoff [mm yr^{-1}] for each lake, as the ratio between long-term mean discharge and catchment area, which we compared to the published range of long-term mean runoff (140 to 1450 mm yr^{-1}) for the respective time period. There were 213 lakes, all comparably small in size (0.01 - 0.28 km^2) which lay outside the published runoff range and were excluded, reducing an original dataset of all available 26,264 lakes to a dataset of 26,051 lakes.

Lake water retention time estimates

We used Q_n and V_n to estimate conventional water retention times (T_n) for each lake in our dataset, Eq. 4:

$$T_n = V_n \times Q_n{}^{-1} \tag{4}$$

where T_n [d], V_n [m^3] and Q_n [m^3d^{-1}] are the water retention time, lake volume and long-term mean discharge estimates for a single lake (n). In addition, we estimated the second measure of water retention time, which has earlier been used to indicate a mean for the time water had been retained in lakes within the landscape (Algesten et al. 2004), Eq. 5:

$$\mathrm{T_{n\ tot}} = \mathrm{V_{n\ tot}} \times \mathrm{Q_n}^{-1} \quad (5)$$

where $T_{n\ tot}$ [d] is the lake volume to discharge comparison between the summed lake volumes in the catchment of a lake (n), including the lake’s own volume ($V_{n\ tot}$) [m^3], and the estimate of long-term mean discharge (Q_n) [m^3d^{-1}] read at the outflow of the lake (n). $T_{n\ tot}$ assumes that runoff from lakes upstream reaches lakes downstream. Evapotranspiration as well as groundwater are both considered in the water-balance of the HBV model used to estimate Q_n (Lindström et al. 1997). The SMHI notes that evapotranspiration is effective in the southern part of Sweden, where more than 50 % of the yearly precipitation may return back to the atmosphere. Replacement of runoff along flow and a weak connectivity amongst lakes may result in an overestimation of $T_{n\ tot}$ for lakes low in the landscape. We therefore applied $T_{n\ tot}$ (1) as a conservative measure of mean water retention by lakes on a landscape scale and (2) to observe how the ratio between cumulative lake volumes ($V_{n\ tot}$) and long-term mean discharge (Q_n) changes from headwater lakes towards lakes at low landscape positions (Fig. 2). Quantifying T_n we identified water bodies (>0.01 km^2) with theoretical water retention times of a few hours to days, implying that those water bodies rather resemble rivers than lakes. Excluding the lower 5 percentile, we limited our study to water bodies with a theoretical water retention time of 4 d and greater ($T_n \geq 4$ d), leaving a final dataset of 24,742 lakes for analysis. However, we do test for the sensitivity of our results to this exclusion criterion (cut-off).

The ratio (R_v) between T_n and $T_{n\ tot}$ is equivalent to the ratio between the lake’s own volume (V_n) and the sum of the lake volumes upstream ($V_{n\ upstr}$) including V_n, Eq. 6:

$$\mathrm{R_v} = \frac{\mathrm{T_n}}{\mathrm{T_{n\ tot}}} = \frac{\mathrm{V_n}}{\mathrm{V_{n\ tot}}} = \frac{\mathrm{V_n}}{\mathrm{V_{n\ upstr}} + \mathrm{V_n}} \quad [\%] \quad (6)$$

Statistical analysis

We considered non-normality and heteroscedasticity (subgroups with unequal variability) in our data by using the non-parametrical Wilcoxon method when testing for differences between landscape positions. The strength of T_n and $T_{n\ tot}$ as predictors for a_{420}/TOC was tested by running a multiple regression model on log-transformed data. Statistical significance was assessed at $p < 0.05$. All statistical tests were carried out in JMP, version 9.0.2. (SAS Institute, Inc., 2010).

Results and Discussion

Lake water retention times in the landscape

We found that conventional water retention times (T_n) for our dataset of 24,742 lakes ranged from 4 d to 10^4 d with the upper and lower 2.5 percentiles ranging from 7 to 1,207 d. The T_n distribution had a positive skew (median = 120 d, mean = 248 d). The lower 2.5 percentile of the full T_n distribution was dominated by small lakes (< 0.1 km^2) situated within the main stems of the seven river basins. The upper 2.5 percentile included headwater lakes where long-term mean water discharges were close to zero. Such low Q_n for headwater lakes and the fact that Q_n strongly increased with an increasing number of lakes upstream (Fig. 3b) explain why a non-parametric Wilcoxon test showed significantly higher T_n values in the headwater group (median T_n = 178) compared to the groups at low landscape positions ($p < 0.0001$; Fig. 3c). Removing the cut-off for lakes with especially short water retention times (T_n < 4 d), the statistical significance of our results remained unchanged. Our findings suggest that the majority of lakes at low landscape positions experience increased water discharge, often resulting in a more efficient exchange of water, when compared to headwater lakes.

Accounting for water retention by upstream lakes using $T_{n\ tot}$ for our 24,742 lakes we found that $T_{n\ tot}$ ranged from 4 to 10^5 d with the upper and lower 2.5 percentiles ranging from 15 to 1,288 d. Like for T_n, the $T_{n\ tot}$ distribution had a positive skew (median = 178 d, mean = 326 d). In contrast to T_n however, $T_{n\ tot}$ showed lower values in the headwater group compared to the groups at low landscape positions (non-parametric Wilcoxon test: $p < 0.0001$; Fig. 3d). This pattern resulted from a strong increase in the summed lake volumes (Fig. 3a; $V_{n\ tot}$) with an increasing number of lakes upstream that exceeded the strong increase in discharge (Fig. 3b; Q_n) with an increasing number of lakes upstream. Analysing each of the seven river basins separately, the significantly increased $T_{n\ tot}$ values at low landscape positions remained similar in six of the seven river basins (Lagan, Mälaren, Dalälven, Ljungan, Ångermanälven and Umeälven; Fig. 4). Removing the cut-off for lakes with especially short water retention times ($T_n < 4$ d), the statistical significance of our results remained unchanged.

We observed that more than 75 % of the lakes at high landscape positions showed $T_{n\ tot}$ values < 365 d, implying that the water in these lake-networks is exchanged within the course of one year (Fig. 4). In boreal Scandinavia, water retention times below one year are primarily driven by strong seasonal runoff events, mainly the yearly spring runoff after snowmelt but also strong rainfalls during summer (Lindström and Bergström 2004; Ågren et al. 2010). During runoff events, water retention times may drop to a fraction of their theoretical long-term means, with strong environmental effects for respective lake-ecosystems (Meili 1992; Ågren et al. 2010). Such seasonal variations are not explicitly considered in our approach but are included in the long-term (1961-1990) mean water discharge data. If processes in single lake ecosystems are studied, then high frequency water discharge measurements are needed. Not until then underlying mechanisms of water quality variations can be understood.

In headwater lakes (no lake > 0.01 km^2 upstream) $T_{n\ tot}$ corresponds to T_n by definition. Headwater lakes made up 62 % of the lake count in our dataset. Thus, for the majority of lakes, conventional water retention time (T_n) is a suitable measure. Moving away from headwater systems, however, differences between T_n and $T_{n\ tot}$ become obvious. The strongest discrepancies between T_n and $T_{n\ tot}$, as expressed by the lowest R_v values, are reached for lakes at low landscape positions. At these landscape positions, R_v frequently showed values < 0.1 (Fig. 5a), implying that $T_{n\ tot}$ was more than 10 times larger than T_n. At low landscape positions however, we also found some lakes that showed comparatively high R_v values of ≥ 0.5. High R_v values are reached when the volume of a lake (n) is comparable to or larger than the summed lake volume upstream. This is the case for the majority of the 114 largest lakes (>10 km^2) which collectively hold 74.5 % of the total lake volume in our dataset (Table 2). Such large lakes are located at low landscape positions (circles, Fig. 5a). Consequently, the high $T_{n\ tot}$ values at low landscape positions are strongly affected by large lakes. These results and median $T_{n\ tot}$ values < 365 d for lakes upstream (see above) suggest that lakes across all landscape positions frequently receive water from lateral inputs, such as tributary or groundwater inflow and/or from precipitation. Such efficient water renewal needs to be considered by studies dealing with biochemical cycling on a landscape scale.

A comparison between the total lake volume in the seven river basins and long-term mean discharge has previously been assessed by Algesten et al. (2004) for Sweden's 21 river mouths. They found that water at the river mouths had been retained by lakes for periods between 0.5 and 13.5 years. Our own estimates at seven river mouths (Table 1) showed values between 0.4 and 3.5 years (156 to 1,265 d, respectively) and were 14.5 to 70.3 % lower than the values published earlier (ΔT; Table 1). We interpret the deviating results to differences in the assessment of lake volumes. We used newly published information on lake volumes (see methods) that covered 89.2 % of the volume held by the 114 large lakes mentioned above, as well as 73.3 % of the entire lake volume in our dataset. Indeed, 26.7 % of the total lake volume had to be estimated, but by comparing estimated lake volumes with published lake volumes (1,263 lakes), we received a slope close to 1 and a non-significant intercept ($p > 0.05$) indicating that our lake volume estimates are reliable.

The $T_{n\ tot}$ approach is based on the following two fundamental assumptions that need to be taken into consideration when $T_{n\ tot}$ is applied. (1) $T_{n\ tot}$ assumes that runoff from lakes upstream reaches lakes downstream. Replacement of the respective runoff along the aquatic continuum will result in an overestimation of $T_{n\ tot}$ for lakes at low landscape positions, suggesting that water renewal may be even more efficient than presented in this study. (2) $T_{n\ tot}$ assumes that a lake can be modelled as a well-mixed system, which neglects periods of stratification. Thus, $T_{n\ tot}$ indicates a long-term mean for water retention by lakes on a landscape scale and should not be confounded with more elaborate measurements of residence time (Bolin and Rodhe 1973) and age (Zimmerman 1988), which access the fate of solutes and particles travelling through a lake system (Monsen et al. 2002).

Lake water retention times and the character of organic carbon

In accordance with previous results (Meili 1992; Schindler et al. 1992), we found that T_n relates to the coloured fraction of OC in lakes, here defined as a_{420}/TOC. A relationship between T_n and a_{420}/TOC was significantly negative but had a low coefficient of determination (Fig. 6a; $R^2 = 0.11$, $p < 0.0001$). It has been argued that a preferential colour loss along a T_n gradient is primarily the result of solar radiation induced OC mineralization (Morris and Hargreaves 1997; Vähätalo and Wetzel 2004). Additionally, in-lake processes such as microbial degradation and flocculation with consequent sedimentation are known to result in a loss of the coloured fraction of OC (Tranvik 1998; von Wachenfeldt et al. 2008; Köhler et al. 2012). OC can also be produced in the lake, which usually is more transparent and labile than its terrestrial counterpart (Findlay and Sinsabaugh 2003; Steinberg 2003). Consequently, the presence of upstream lakes has an effect on the OC character that is found downstream, both in terms of the amount and in the light-absorbing properties (Larson et al. 2007 and Sadro et al. 2012).

Since the conventional retention time estimator T_n does not account for any water retention and carbon processing by upstream lakes, we expected an improvement in the relationship between lake water retention and a_{420}/TOC when we replaced T_n by $T_{n\ tot}$. Such an improvement was achieved for non-headwater lakes (911 lakes with $T_n \neq T_{n\ tot}$). First of all we were able to raise the coefficient of determination from 0.11 to 0.21 (Fig. 6b), and secondly we found that $T_{n\ tot}$ was the stronger predictor for a_{420}/TOC when we ran a multiple regression model with both log-transformed T_n and $T_{n\ tot}$ as explanatory variables ($t = -9.77$, $p < 0.0001$ and $t = -1.97$, $p = 0.0497$, respectively). The coefficient of determination became exceptionally high when we grouped our data and related median water retention times to median a_{420}/TOC values (Fig. 6a and 6b). An improved relation to the coloured fraction of OC by using $T_{n\ tot}$ instead of T_n supports the concept that the character of OC in boreal lakes is related to the time OC has spent in lakes within the landscape.

Replacing $T_{n\ tot}$ by lake landscape position, we found a significant ($p < 0.05$) decrease in a_{420} (Fig. 7a) and TOC (Fig. 7b) but not in a_{420}/TOC (Fig. 7c). These results suggest that there is either no preferential loss of the coloured fraction of OC across landscape positions or that highly coloured OC is supplied from lateral inflows at the same rate as it is lost by mineralization and sedimentation. Previous studies addressing lake landscape position suggested that changes in OC quantity and quality depend on hydrologic connectivity (Martin and Soranno et al. 2006) and lake-network structure (Soranno et al. 1999). Thus, measures of the character of OC, such as a_{420}/TOC, may increase or decrease when following a specific lake-chain. Moving from specific lake-chains to a large scale across lake districts, as done in our study, increases variability and may obscure patterns that have been reported from within specific lake-chains (Sadro et al. 2012). Consequently studies that address changes within specific lake-chains, e.g. the studies by Kratz et al. (1997) and Kling et al. (2000), differ from studies like ours that test differences across lake landscape positions based on data from numerous lake districts.

Seeing lakes from a large landscape perspective and assessing lakes beyond specific lake-districts become especially relevant in the carbon rich boreal zone, where river basins may hold several thousand lakes and the bulk lake area and volume is unevenly distributed (Table 2). As an integral part of the boreal landscape, lakes delay surface water runoff, giving time for carbon processing to occur, and thereby acting as regulators of carbon cycling (Cole et al. 2007; Tranvik et al. 2009). Seeing lakes from a landscape perspective may become increasingly relevant in the future, as lakes are monitored, modified and managed for such endeavours as hydropower generation (Barros et al. 2011) and multi-lake ecosystem management (Soranno et al. 2010).

Conclusions

We conclude from our results that lake landscape position does not reflect well how long the water has travelled in a boreal landscape, limiting its use in explaining water retention time dependent processes. While T_n is a useful tool to indicate lake water retention for single lake water bodies, we recommend using $T_{n\ tot}$ for large-scale approaches. Since we found a strong relationship between $T_{n\ tot}$ and the character of OC in the landscape for 1,559 lakes, we suggest that $T_{n\ tot}$ might also be a useful measure to explain variations of other nutrients in the landscape.

Acknowledgments

Many thanks go to the Swedish Meteorological and Hydrological Institute (SMHI) for providing the hydrological and geomorphological base data used in this study, as well as to the Department of Aquatic Sciences and Assessment at the Swedish University of Agricultural Sciences (SLU) for providing data from thousands of lake water samples. We thank Dolly Kothawala and Blaize Denfeld, as well as three anonymous reviewers for constructive comments. Financial support was received from the Swedish Research Council (VR), the Swedish Research Council for Environment, Agricultural Sciences and Spatial Planning (FORMAS), and the Nordic Centre of Excellence "CRAICC – Cryosphere-atmosphere interactions in a changing arctic climate" supported by NordForsk. Martyn N. Futter was funded by the MISTRA Future Forests programme. This work developed as part of the project "The colour of water – interplay with climate, and effects on drinking water supply".

References

Ågren A, Buffam I, Bishop K, Laudon H (2010) Modeling stream dissolved organic carbon concentrations during spring flood in the boreal forest: A simple empirical approach for regional predictions. Journal of Geophysical Research 115 (G1):G01012

Alexander R, Boyer E, Smith R, Schwarz G, Moore R (2007) The role of headwater streams in downstream water quality1. JAWRA Journal of the American Water Resources Association 43 (1):41-59

Algesten G, Sobek S, Bergström A-K, Ågren A, Tranvik LJ, Jansson M (2004) Role of lakes for organic carbon cycling in the boreal zone. Global Change Biology 10 (1):141-147. doi:10.1111/j.1365-2486.2003.00721.x

Barros N, Cole JJ, Tranvik LJ, Prairie YT, Bastviken D, Huszar VLM, Del Giorgio P, Roland F (2011) Carbon emission from hydroelectric reservoirs linked to reservoir age and latitude. Nature Geoscience 4 (9):593-596

Battin T, Luyssaert S, Kaplan L, Aufdenkampe A, Richter A, Tranvik LJ (2009) The boundless carbon cycle. Nature Geoscience 2 (9):598-600

Bolin B, Rodhe H (1973) A note on the concepts of age distribution and transit time in natural reservoirs. Tellus 25 (1):58-62

Canham CD, Pace ML (2009) A spatially explicit, mass-balance analysis of watershed-scale controls on lake chemistry. Real World Ecology. In: Miao S, Carstenn S, Nungesser M (eds). Springer New York, pp 209-233. doi:10.1007/978-0-387-77942-3_8

Cole J, Prairie Y, Caraco N, McDowell W, Tranvik LJ, Striegl R, Duarte C, Kortelainen P, Downing J, Middelburg J (2007) Plumbing the global carbon cycle: integrating inland waters into the terrestrial carbon budget. Ecosystems 10 (1):172-185

Egeberg PK, Eikenes M, Gjessing ET (1999) Organic nitrogen distribution in NOM size classes. Environment International 25 (2–3):225-236. doi:10.1016/s0160-4120(98)00101-9

Findlay S, Sinsabaugh R (2003) Aquatic ecosystems: interactivity of dissolved organic matter. Academic Press

Goodman KJ, Baker MA, Wurtsbaugh WA (2011) Lakes as buffers of stream dissolved organic matter (DOM) variability: Temporal patterns of DOM characteristics in mountain stream-lake systems. Journal of Geophysical Research-Biogeosciences 116. doi:10.1029/2011jg001709

Göransson E, Johnson RK, Wilander A (2004) Representativity of a mid-lake surface water chemistry sample. Environ. Monit. Assess. 95 (1), 221−238

Hecky R, Campbell P, Hendzel L (1993) The stoichiometry of carbon, nitrogen, and phosphorus in particulate matter of lakes and oceans. Limnology and Oceanography 38(4):709-724

Hu CM, Muller-Karger FE, Zepp RG (2002) Absorbance, absorption coefficient, and apparent quantum yield: A comment on common ambiguity in the use of these optical concepts. Limnology and Oceanography 47 (4):1261-1267

Kirk JT (1994) Light and photosynthesis in aquatic ecosystems. Cambridge Univ. Press.

Kling GW, Kipphut GW, Miller MM, O'Brien WJ (2000) Integration of lakes and streams in a landscape perspective: the importance of material processing on spatial patterns and temporal coherence. Freshwater Biology 43 (3):477-497. doi:10.1046/j.1365-2427.2000.00515.x

Köhler B, von Wachenfeldt E, Kothawala D, Tranvik LJ (2012) Reactivity continuum of dissolved organic carbon decomposition in lake water. Journal of Geophysical Research-Biogeosciences 117. doi:G01024

Kratz T, Webster K, Bowser C, Maguson J, Benson B (1997) The influence of landscape position on lakes in northern Wisconsin. Freshwater Biology 37 (1):209-217. doi:10.1046/j.1365-2427.1997.00149.x

Larson JH, Frost PC, Zheng ZY, Johnston CA, Bridgham SD, Lodge DM, Lamberti GA (2007) Effects of upstream lakes on dissolved organic matter in streams. Limnology and Oceanography 52 (1):60-69

Lindström G, Bergström S (2004) Runoff trends in Sweden 1807-2002. Hydrological Sciences Journal/Journal des Sciences Hydrologiques 49 (1):69-83

Lindström G, Johansson B, Persson M, Gardelin M, Bergström S (1997) Development and test of the distributed HBV-96 hydrological model. Journal of Hydrology 201 (1-4):272-288. doi:10.1016/s0022-1694(97)00041-3

Martin S, Soranno P (2006) Lake landscape position: Relationships to hydrologic connectivity and landscape features. Limnology and Oceanography 51 (2):801-814

McDonnell J, McGuire K, Aggarwal P, Beven K, Biondi D, Destouni G, Dunn S, James A, Kirchner J, Kraft P (2010) How old is streamwater? Open questions in catchment transit time conceptualization, modelling and analysis. Hydrological Processes 24 (12):1745-1754

McGuire K, McDonnell J, Weiler M, Kendall C, McGlynn B, Welker J, Seibert J (2005) The role of topography on catchment-scale water residence time. Water Resources Research 41 (5):W05002

Meili M (1992) Sources, concentrations and characteristics of organic matter in softwater lakes and streams of the Swedish forest region. Hydrobiologia 229 (1):23-41

Monsen NE, Cloern JE, Lucas LV, Monismith SG (2002) A comment on the use of water residence time, residence time, and age as transport time scales. Limnology and Oceanography 47 (5):1545-1553

Morris DP, Hargreaves BR (1997) The role of photochemical degradation of dissolved organic carbon in regulating the UV transparency of three lakes on the Pocono Plateau. Limnology and Oceanography 42 (2):239-249

Sadro S, Nelson CE, Melack JM (2012) The Influence of Landscape Position and Catchment Characteristics on Aquatic Biogeochemistry in High-Elevation Lake-Chains. Ecosystems 15 (3):363-386. doi:10.1007/s10021-011-9515-x

Schindler DW, Bayley SE, Curtis PJ, Parker BR, Stainton M, Kelly C (1992) Natural and man-caused factors affecting the abundance and cycling of dissolved organic substances in precambrian shield lakes. Hydrobiologia 229 (1):1-21

Schindler DW, Bayley SE, Parker BR, Beaty KG, Cruikshank D, Fee E, Schindler E, Stainton M (1996) The effects of climatic warming on the properties of boreal lakes and streams at the Experimental Lakes Area, northwestern Ontario. Limnology and Oceanography 41 (5):1004-1017

Schindler DW, Curtis PJ, Bayley SE, Parker BR, Beaty KG, Stainton MP (1997) Climate-induced changes in the dissolved organic carbon budgets of boreal lakes. Biogeochemistry 36 (1):9-28. doi:10.1023/a:1005792014547

Sobek S, Nisell J, Fölster J (2011) Predicting the depth and volume of lakes from map-derived parameters. Inland Waters 1 (3):177-184

Soranno PA, Webster KE, Riera JL, Kratz TK, Baron JS, Bukaveckas PA, Kling GW, White DS, Caine N, Lathrop RC, Leavitt PR (1999) Spatial variation among lakes within landscapes: Ecological organization along lake chains. Ecosystems 2 (5):395-410. doi:10.1007/s100219900089

Soranno PA, Cheruvelil KS, Webster KE, Bremigan MT, Wagner T, Stow CA (2010) Using Landscape Limnology to Classify Freshwater Ecosystems for Multi-ecosystem Management and Conservation. Bioscience 60 (6):440-454. doi:10.1525/bio.2010.60.6.8

Steinberg C (2003) Ecology of humic substances in freshwaters: determinants from geochemistry to ecological niches. Springer

Sulzberger B, Durisch-Kaiser E (2009) Chemical characterization of dissolved organic matter (DOM): A prerequisite for understanding UV-induced changes of DOM absorption properties and bioavailability. Aquatic Sciences 71 (2):104-126. doi:10.1007/s00027-008-8082-5

Tokunaga E (2003) Tiling Properties of Drainage Basins and Their Physical Bases. In: Concepts and Modelling in Geomorphology: International Perspectives, Evans IS, Dikau R, Tokunaga E, Ohmori H, and Hirano M (eds). TERRAPUB, Tokyo, Japan, pp. 147-166

Tranvik LJ (1998) Degradation of dissolved organic matter in humic waters by bacteria. In: Hessen DO, Tranvik LJ (ed) Aquatic humic substances: ecology and biogeochemistry. Ecological Studies 133. Springer, pp. 259-283

Tranvik LJ, Downing J, Cotner J, Loiselle S, Striegl R, Ballatore T, Dillon P, Finlay K, Fortino K, Knoll L (2009) Lakes and reservoirs as regulators of carbon cycling and climate. Limnology and Oceanography 54 (6):2298-2314

Vähätalo AV, Wetzel RG (2004) Photochemical and microbial decomposition of chromophoric dissolved organic matter during long (months-years) exposures. Marine Chemistry 89 (1-4):313-326. doi:10.1016/j.marchem.2004.03.010

Vollenweider RA (1976) Advances in defining critical loading levels of phosphorus in lake eutrophication. Memorie dell'Istituto Italiano di Idrobiologia Dott Marco de Marchi 33:53-83

von Wachenfeldt E, Tranvik LJ (2008) Sedimentation in boreal lakes—the role of flocculation of allochthonous dissolved organic matter in the water column. Ecosystems 11 (5):803-814

Wetzel R (2001) Limnology: lake and river ecosystems, vol 1006. Academic Press San Diego, CA

Weyhenmeyer GA, Fröberg M, Karltun E, Khalili M, Kothawala D, Temnerud J, Tranvik LJ (2012) Selective decay of terrestrial organic carbon during transport from land to sea. Global Change Biology 18 (1):349-355. doi:10.1111/j.1365-2486.2011.02544.x

Zimmerman JTF (1988) Estuarine residence times. In: Kjerfve B (ed.) Hydrodynamics of Estuaries V. 1. CRC Press, Boca Raton, pp 75-84

Table 1 Dataset by river basins; area of river basin (size), highest point (z_{max}), number of lakes larger than 0.01 km^2 (lake count), summed lake volumes ($V_{n\ tot}$), long-term (1961 to 1990) mean flow as discharge at the river mouth (Q_{sea}), and the ratio between $V_{n\ tot}$ and Q_{sea} (T_{sea}). The percentage ΔT presents the difference between T_{sea} and earlier estimates published by Algesten et al. (2004)

ID	river basin	size [km^2]	z_{max} [m a.s.l.]	lake count [-]	$V_{n\ tot}$ [Mm^3]	Q_{sea} [m^3s^{-1}]	T_{sea} [d]	ΔT [%]
1	Lagan	6,452	372	737	2,641	83	368	-
2	Mälaren	22,650	490	2,138	22,069	202	1,265	-
3	Dalälven	28,621	1,436	5,241	17,161	357	557	-51 %
4	Ljungan	12,851	1,577	2,187	5,340	150	412	-70 %
5	Ångermanälv.	31,079	1,564	5,450	29,502	520	657	-25 %
6	Umeälven	26,815	1,751	5,710	17,162	430	462	-37 %
7	Kalixälven	18,130	2,092	3,279	3,060	227	156	-15 %

Table 2 Lake count and volumes per lake size class A to E

SMHI size classes [km^2]		lake count [-, %]		lake area [km^2, %]		lake volume [Mm^3, %]	
E	0.01 – 0.1	18128	73.2 %	5,242	50.5 %	1,606	1.6 %
D	0.1 – 1	5410	21.9 %	2,943	28.3 %	5,983	6.2 %
C	1 – 10	1090	4.4 %	1,630	15.7 %	17,121	17.7 %
A – B	> 10	114	0.5 %	567	5.5 %	72,225	74.5 %
	Total	24,742	100.0 %	10,382	100.0%	96,935	100.0 %

Fig. 1 Geographic location of the seven river basins in our dataset; Lagan (1), Mälaren (2), Dalälven (3), Ljungan (4), Ångermanälven (5), Umeälven (6), Kalixälven (7). Lakes sampled by the Swedish national lake monitoring program in 1995, 2000, 2005 marked (black dots)

Fig. 2 (a) An exemplary lake (n) with three lakes upstream. The lake volume to flow comparison $T_{n\ tot}$ is indifferent of the flow and lake distribution upstream of the lake (n) at hand. The summed lake volume ($V_{n\ tot}$) includes the lake (n) at hand. (b) Following a lake series downstream, $T_{n\ tot}$ will decrease where flow as discharge (Q_n) accumulates faster than the summed lake volume ($V_{n\ tot}$) and conversely increase where $V_{n\ tot}$ accumulates faster than Q_n

Fig. 3 Distribution (median, 50 % and 90 % quantile ranges) of (a) summed lake volumes $V_{n\ tot}$ in the catchment of each lake, (b) long term (1961 – 1990) mean discharge (Q_n) at the outflow of each lake, (c) water retention time T_n for each lake. Asterisks indicate significance levels (* $p < 0.05$, ** $p < 0.001$, *** $p < 0.0001$) for a decrease in the median T_n when compared to the headwater group (hg). Crosses mark medians for lakes sampled. (d) Water retention time $T_{n\ tot}$ where the increase in median $T_{n\ tot}$ from the headwater group towards the most downstream group is marked (dark-grey). Asterisks indicate significance levels (* $p < 0.05$, ** $p < 0.001$, *** $p < 0.0001$) for an increase in the median $T_{n\ tot}$ when compared to the headwater group (hg). Crosses mark medians for lakes sampled. For all figure parts lakes grouped by no, one, 2 – 5, 6 – 10, 11 – 100 and >100 lakes (>0.01 km^2) upstream

Fig. 4 Distribution (median, 50 % and 90 % quantile ranges) of $T_{n\ tot}$ in days for each river basin, with $T_{n\ tot}$ = 365 d marked (dashed-line). Lakes grouped by no, one, 2 – 5, 6 – 10, 11 – 100 and >100 lakes (>0.01 km^2) upstream. Asterisks indicate significance levels (* $p < 0.05$, ** $p < 0.001$, *** $p < 0.0001$) for an increase in the median $T_{n\ tot}$ when compared to the headwater group (hg). For all figure parts lakes grouped by no, one, 2 – 5, 6 – 10, 11 – 100 and >100 lakes (>0.01 km^2) upstream

Fig. 5 (a) Distribution (median, 50 % and 90 % quantile ranges) for T_n versus $T_{n\ tot}$ as ratio R_v. Lakes grouped by no, one, 2 – 5, 6 – 10, 11 – 100 and >100 lakes (>0.01 km^2) upstream

(b) Direct comparison between T_n and $T_{n\ tot}$ for the entire dataset. For both figure parts R_v = 1.0, 0.5 and 0.1 marked (dashed-line), and the 114 largest lakes (>10 km^2) marked (circles)

Fig. 6 Relationship between (a) T_n and a_{420}/TOC, (b) $T_{n\ tot}$ and a_{420}/TOC, respectively for 911 non-headwater lakes (grey dots) with the regression statistics (data) and the regression curve (curved line). Shown are also median values for grouped data (black circles) with standard deviations (black bars) and the regression statistics (median)

Fig. 7 Distribution (median, 50 % and 90 % quantile ranges) of a_{420}, TOC and a_{420}/TOC for 1,559 lakes sampled. Asterisks indicate significance levels (* $p < 0.05$, ** $p < 0.001$, *** $p < 0.0001$) for a decrease in a_{420}, TOC and a_{420}/TOC when compared to the headwater group (hg). Changes in a_{420}/TOC across groups of different landscape positions were non-significant ($p > 0.05$). Lakes grouped by no, one, 2 – 5, 6 – 10, 11 – 100 and >100 lakes (>0.01 km^2) upstream

Fig. 1
Click here to download line figure: fig1_w84mm.eps

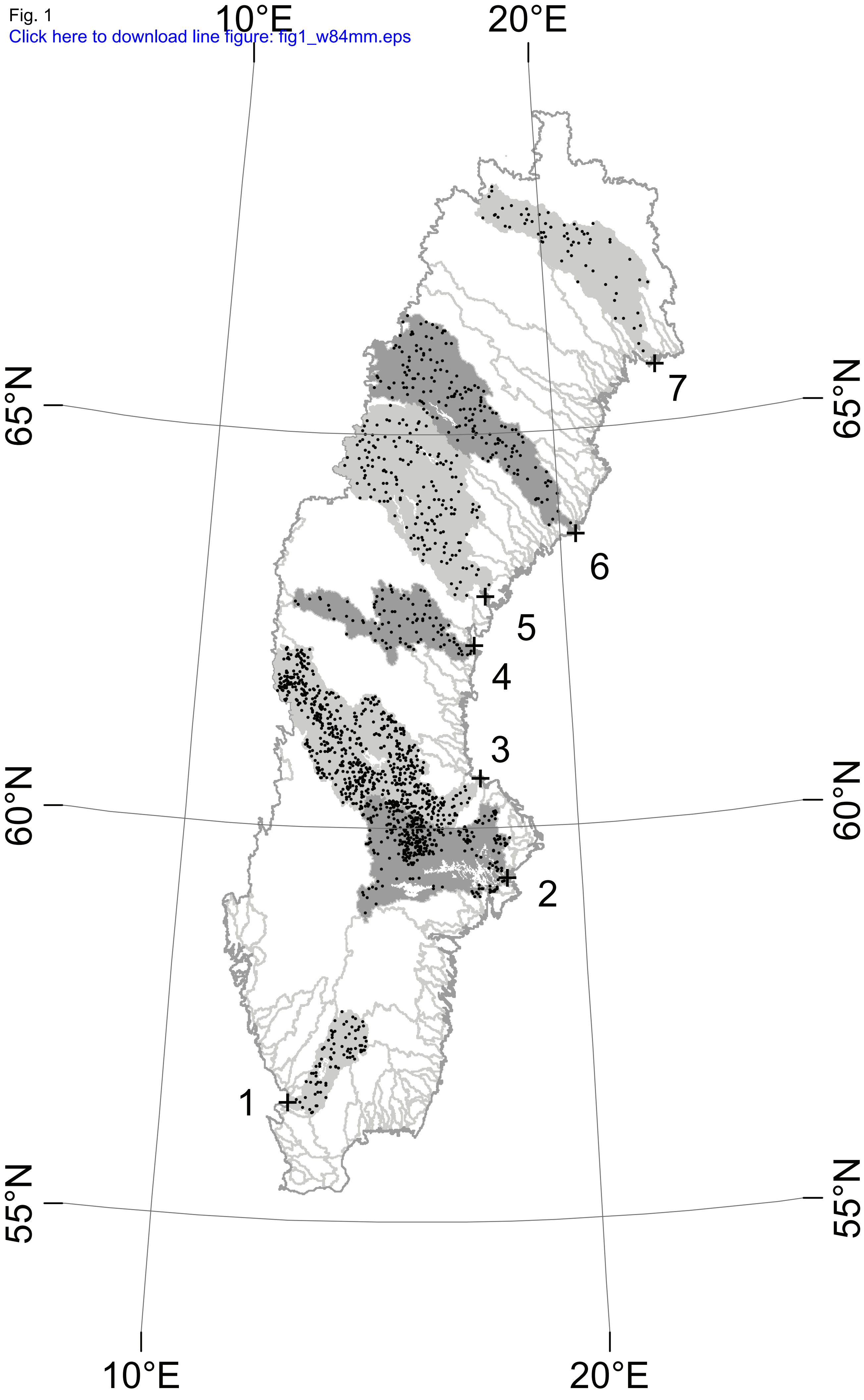

Fig. 2

Click here to download line figure: fig2_w84mm.eps

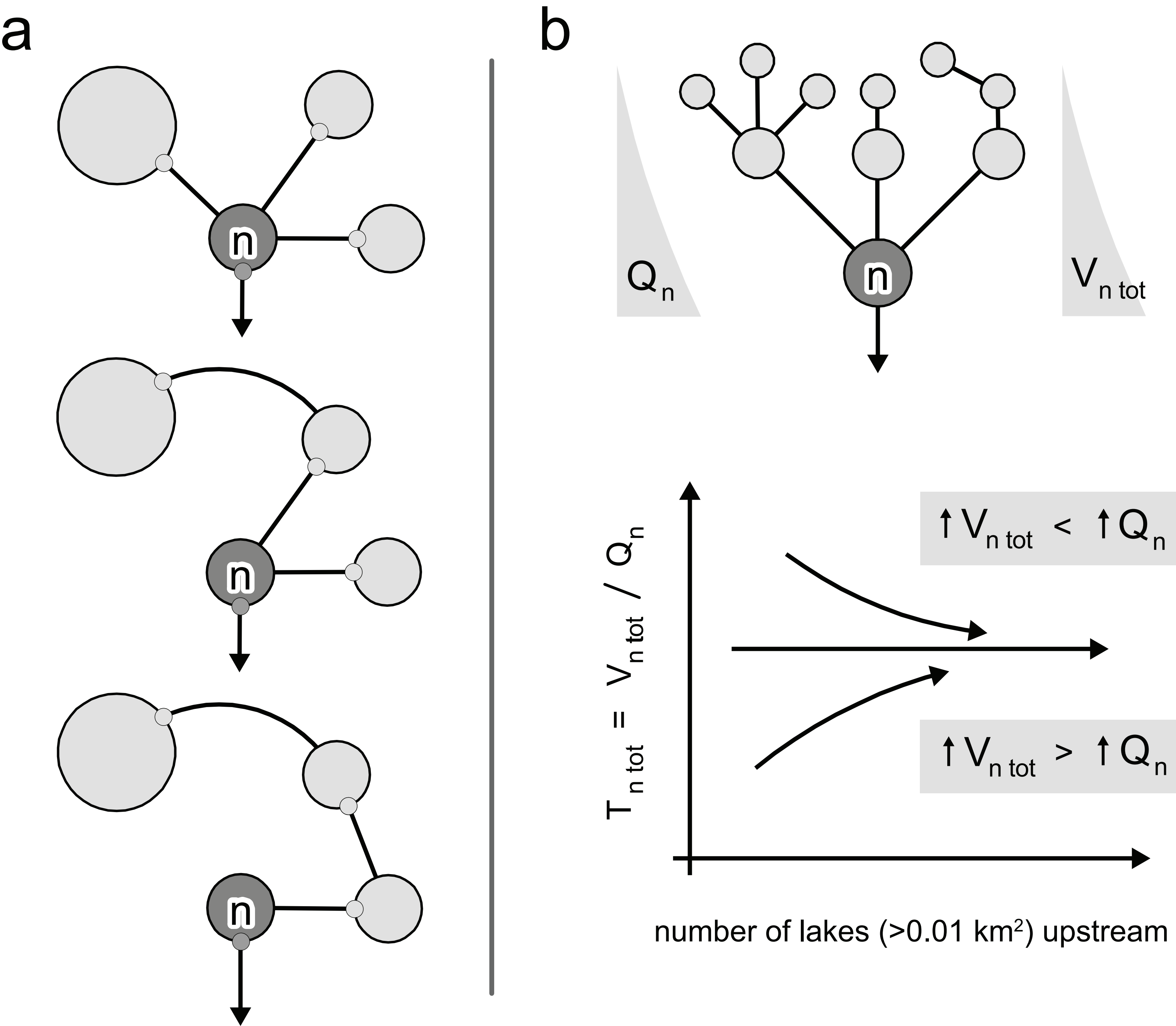

Fig. 3

Click here to download line figure: fig3_w174mm.eps

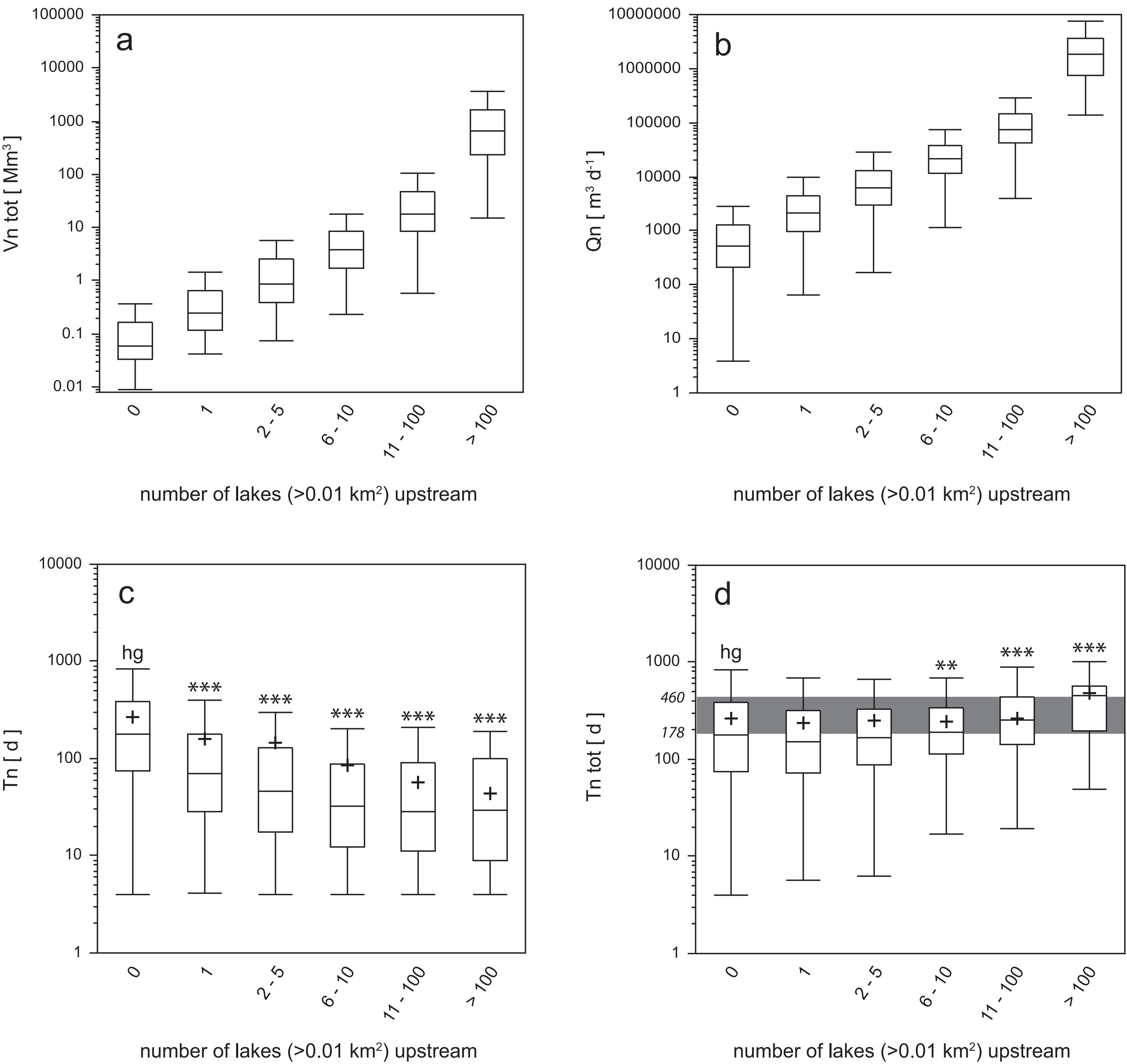

Fig. 4
Click here to download line figure: fig4_w174mm.eps

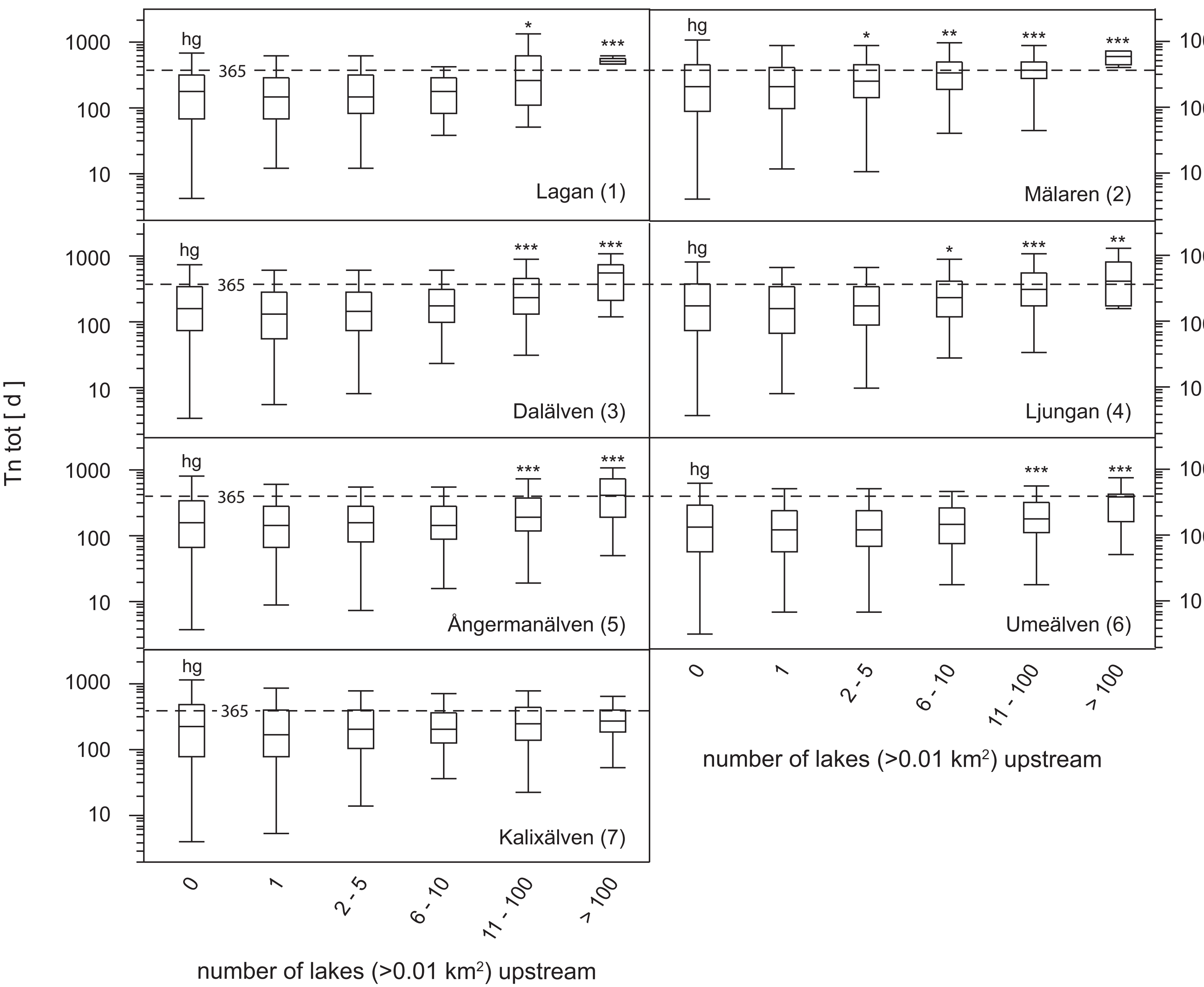

Fig. 5
Click here to download line figure: fig5_w174mm.eps

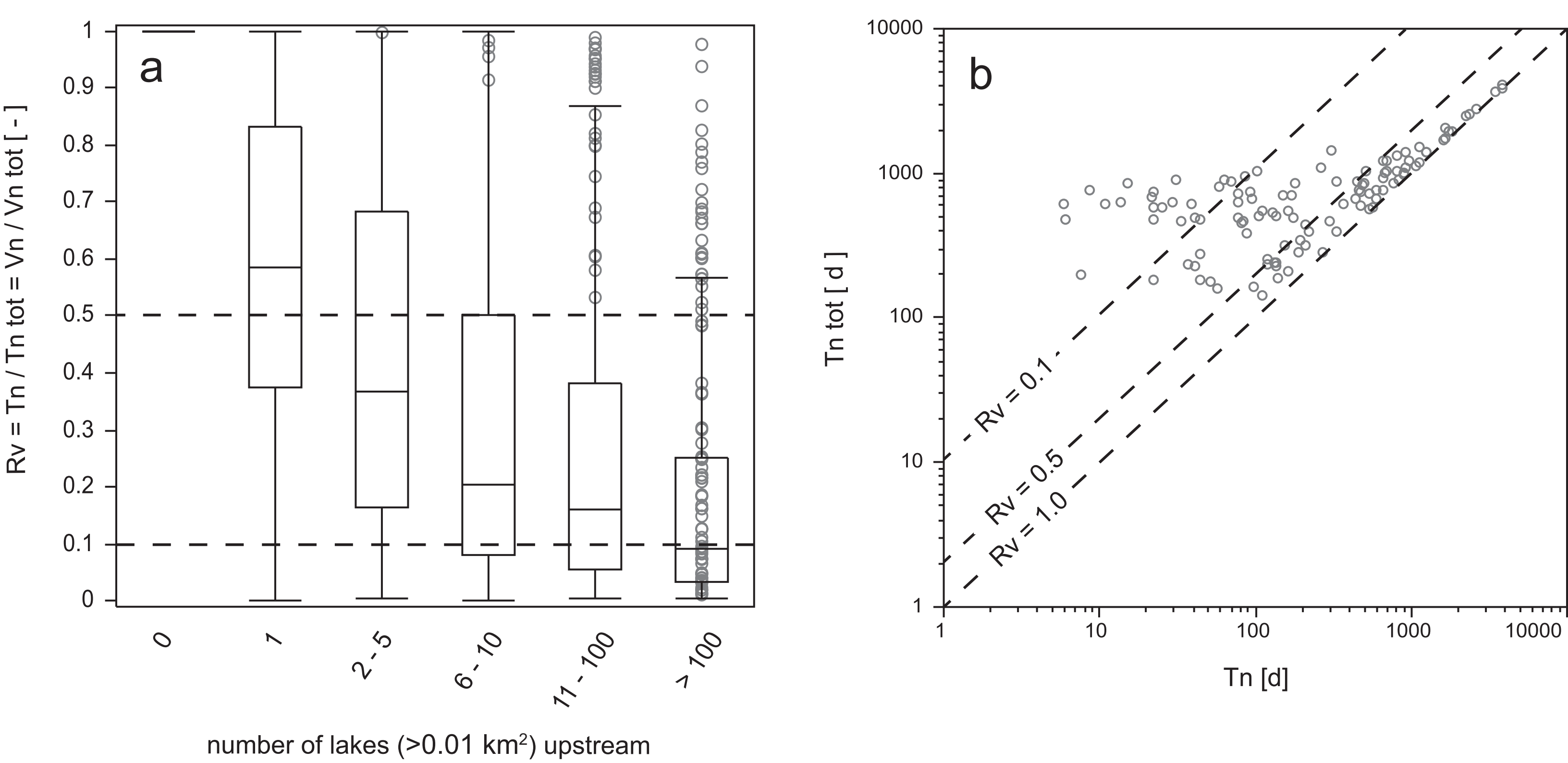

Fig. 6

Click here to download line figure: fig6_w174mm.eps

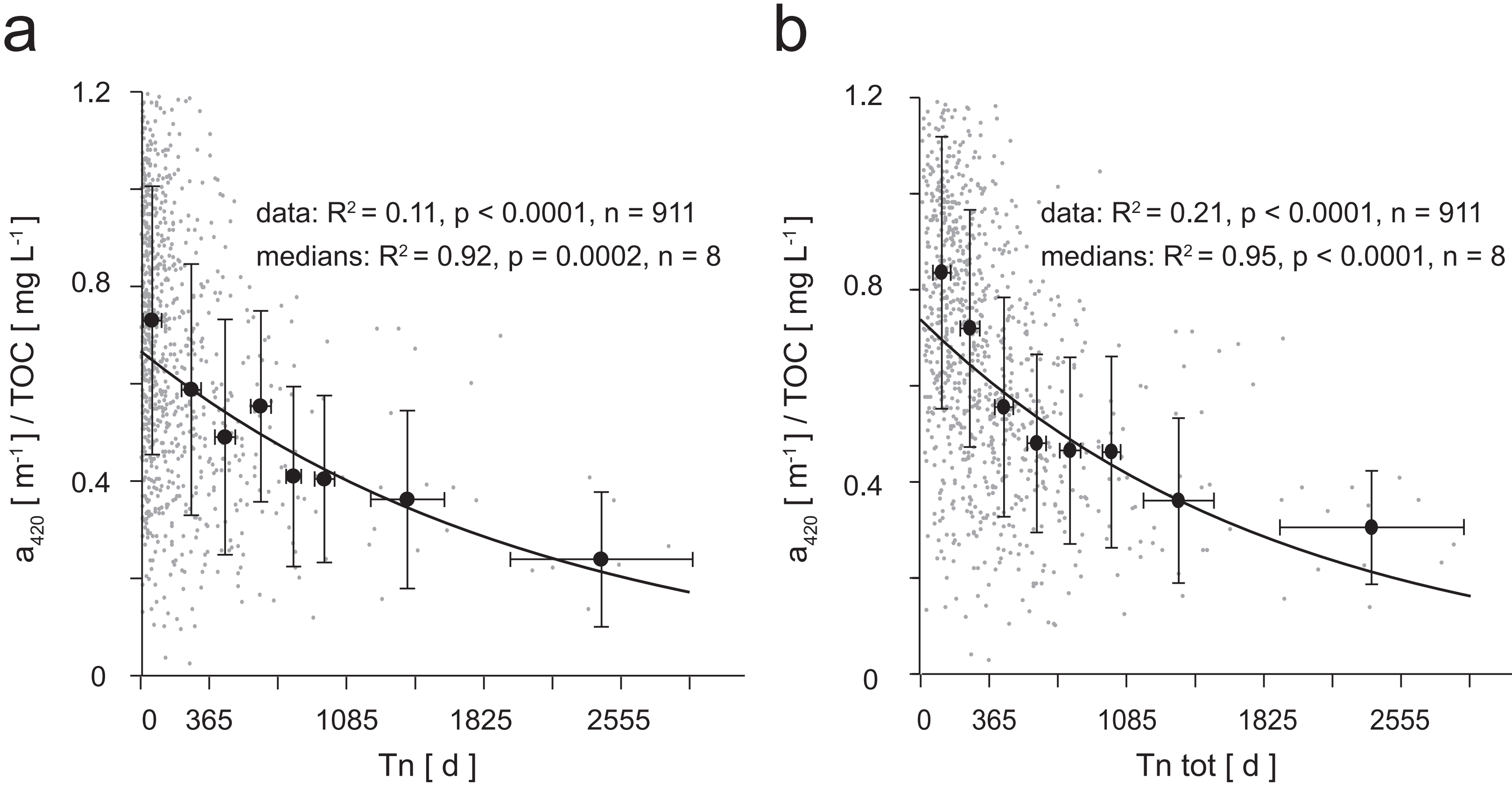

Fig. 7
Click here to download line figure: fig7_w84mm.eps