LIMNOLOGY and OCEANOGRAPHY: METHODS

ASLO

Limnol. Oceanogr.: Methods 2020

doi: 10.1002/lom3.10369

Experimental assessment and implications of long-term within-trap mineralization of seston in lake trapping studies

Alan D. Radbourne [1,2] **David B. Ryves** [2]*

[1]Centre for Ecology and Hydrology, Environment Centre Wales, Bangor, UK
[2]Geography and Environment, Loughborough University, Loughborough, UK

Abstract

Sediment trapping is a widely accepted technique in lake studies for analyzing seasonal limnological events and can provide insight into ecological succession as well as the seasonal dynamics of organic and inorganic fluxes. More recently, organic flux measurement from traps has been especially important in estimating whole-lake C sequestration as a basis for regional and global upscaling of C budgets across lake types. However, in-trap mineralization or dissolution of components of collected sediment (seston) has not been systematically examined, and thus a reliable correction factor for in-trap losses of various sedimentary fluxes (especially those involving organic carbon) is still unknown. This experimental study assesses the loss of algal biomass representative of a 6-month carousel-type (closed) sediment trap deployment in a deep, eutrophic lake under cold (~ 5°C) and anoxic ambient conditions typical of the hypolimnion in stratifying, temperate lakes. Results show a loss of organic matter (OM) at a consistent rate over 180 d, reducing the fraction of initial OM content by approximately a third after 180 d of deployment (linear regression of OM fraction loss = $-0.001864t$). The significance of these findings is demonstrated by application to published trap data; at Rostherne Mere, UK, which implies that annual OM fluxes are underestimated by 18.2% on average (range 13.7–23.2%). This highlights the far-reaching implications for lake sediment trap methodology and our understanding of seston taphonomy, suggesting a mineralization correction factor for OM should be applied to traps deployed for longer than 1 week. With loss correction factored in, this study supports the reliability of longer-term (i.e., ~ 6 months) sediment trap deployment.

Sediment traps have been widely used in lakes since the 1950s (Bloesch and Burns 1980). These simple devices for measuring downward settling flux can be utilized for high-resolution detection of short-term flux events (Flower 1991; Teranes et al. 1999; Kulbe et al. 2006) and with an array of traps at multiple depths, can provide estimates of settling velocities for mass flux events and resuspension within the water column (e.g., Charlton and Lean 1987; Bloesch 1994; Ryves et al. 2003). Analysis of trap seston has increasingly been used to measure carbon budgets, as a key method to assess the role of lakes on carbon cycling (Sobek et al. 2009; Radbourne et al. 2017). However, sediment traps can be deployed over various time frames, meaning the organic matter (OM) is stored in the traps from capture (sedimentation) to trap resetting (collection), leaving it prone to mineralization within the trap. In-trap mineralization results in a reduction of OM from capture to collection, producing an underestimation of total OM. This error increases over time, making the data less reliable and more difficult to compare between differing timescales and studies. This in turn will lead to underestimation of OM flux and accumulation rates and errors in key parameters that depend on these measurements, such as burial rate (Sobek et al. 2009; Radbourne et al. 2017) which ultimately may reduce the reliability of upscaling for regional or global parameterization of C cycling.

In response to this issue, it has been proposed that traps should be reset at regular intervals of less than 2 weeks (Bloesch and Burns 1980) to limit the perceived implications of in-trap mineralization. However, this regularity of trap recovery can be logistically difficult and financially impractical, and arguably unnecessary with the development of automated sequencing traps allowing for a consistent monitoring of settling sedimentary material (seston) over the long term.

Therefore, the development of reliable long-term trapping studies through the improved understanding of in-trap mineralization can reduce the logistical and financial issues related to short-term collection procedures. However, calculating the within-trap mineralization has been an ongoing issue that remains unresolved (Horppila and Nurminen 2005), with a

*Correspondence: d.b.ryves@lboro.ac.uk

general solution for all lake systems thought to be unfeasible due to the variety and complexity of seasonal and lake-to-lake differences (Bloesch and Burns 1980). In an attempt to limit the in-trap mineralization, previous studies have "spiked" collection bottles with preservation agents (i.e., chloroform, mercuric-chloride; Eadie et al. 1984, Meyers and Eadie 1993). However, this practice is now widely accepted to be inappropriate due to the wider environmental concern of using toxic substances. Indeed, under current legislation, this a practice not permitted by many organizations (e.g., Natural England, the public body which oversees the UK's natural environment, does not permit the use of toxic preservation additives at sites they manage, such as Rostherne Mere, where this study is based).

To improve the number, longevity and value of important sediment trap monitoring series in lakes across the world, and to ensure the data generated are comparable between studies, resolving this issue of in-trap mineralization is of great importance. Therefore, this study provides details of an experiment to replicate the mineralization loss of OM within a sediment trap over a 6-month period at Rostherne Mere (Cheshire, UK). The experimental design replicates the conditions of closed (carousel-type) trap collections in deep, stratifying lakes, using an automated sequencing trap, but the approach can be extended to other lake types to estimate a lake-specific correction factor for any in-trap taphonomic process (e.g., organic mineralization, calcite dissolution, or silica dissolution).

Materials and procedures

The study is a laboratory-based experiment, modeling the conditions of a field sedimentation study for the estimation of in-trap OM loss over time. The experimental design used replicates the hypolimnetic lake conditions at Rostherne Mere, UK, a deep ($\sim$ 30 m), temperate, highly productive lake that annually stratifies with a completely anoxic hypolimnion for $\sim$ 8 months of the year (*see* Radbourne et al. 2017 for more information about the site). Temperatures in the hypolimnion are generally $\sim$ 5°C throughout the stratified period, rising to $\sim$ 8°C on overturn in late November/early December and falling slowly over winter to $\sim$ 5°C before stratification is re-established (Radbourne et al. 2019*a*). The experiment is designed to replicate an automated sequencing trap that rotates to seal each collection bottle on a predetermined programme, such as the Technicap PPS 4/3 as currently used at Rostherne Mere at 10 m (shallow trap) and 25 m (deep trap) depths (*see* http://www.technicap.com/products/sediment-trap).

To simulate trap collection, approximately 1.25 kg of live algae was collected from the surface water of Rostherne Mere in early July 2015. The algae was collected using large sealable containers and an algae net from the side of a boat in a central lake location (at the deepest point close to sediment trap locations), representing a common assemblage for the time of year, including a mixture of macro- and microalgae such as *Aphanizomenon* spp., *Cryptomonas* spp., and *Stephanodiscus* spp. (*see* Radbourne et al. 2019b). Additionally, 20 L of epilimnion water from a water depth of 0–4 m, and 20 L of hypolimnion water from a water depth of 25 m (well below the thermocline at $\sim$ 8 m) was collected in clean high-density polyethylene (HDPE) bottles to provide a representative sample of bacterial microbes, especially anoxic microbes from the hypolimnion (Mattson and Likens 1993). All samples were transported cool, dark and sealed to the laboratory where it was stored overnight in a dark, cold room at 5°C prior to setting up the experiment.

The following day the algal collections were siphoned into a container and the macroalgae removed and placed on paper towels to remove as much of the surplus water content as possible. All weighing was carried out using a four decimal place balance. The macroalgae were manually mixed to homogenize the sample and divided into $\sim$ 23 g wet weight subsamples (mean = 22.96 g, standard deviation [SD] = 0.57 g), weighed and placed in 54 individually labeled, new 250 mL HDPE bottles that were all preweighed before adding algae. The $\sim$ 23 g "wet" macroalgae sample was used to represent a typical 2-weekly summer lake sediment trap collection, with $\sim$ 23 g of experimental wet matter equaling $\sim$ 1.9 g dry matter (the initial amount of OM for each sample at T_0). This is representative of median 2-weekly summer dry sediment mass of lake trap collections at Rostherne Mere, namely 2.6 g and 1.8 g in the shallow and deep trap, respectively (summer is June to August, shallow trap data from 2013 to 2015, deep trap from 2013 to 2014).

Next, 150 mL of epilimnion water (for detritus, zooplankton, and microalgae) and 70 mL of hypolimnion water (as an inoculum of anoxic bacteria) was added to each individual sample bottle and topped up with distilled water to the bottle brim. The epilimnion water added contains a small quantity of detritus, zooplankton, and microalgae (150 mL epilimnion water = mean = 0.0604 g, SD = 0.0045 g dry weight component) and the 20-liter container of epilimnion water was well mixed before subsampling 150 mL into each sample bottle. The hypolimnion water at this time would also contain a small additional component (e.g., senescent and dead algae) as implied by its clear appearance in the 20-liter container. However, as the 20-liter container of hypolimnion water was also settled overnight and not mixed prior to subsampling (in contrast to epilimnion water), minimal amounts of any seston it contained was included in the 70 mL added to each sample bottle, and is not quantified here. The bottles were sealed with airtight lids with no visible air bubbles and were stored in a dark and cool (5°C) location. A set of three samples (denoted $T_{0,0}$, where time = 0 d) samples and the 150 mL epilimnion water samples were frozen immediately.

Triplicate sets of the individual sample bottles (denoted T_t) were removed and frozen on a predetermined pattern at the same time of day (at 15:00 h on days t = 1, 3, 7, 14, 28, 42, 56, 70, 84, 98, 112, 126, 140, 154, 168, 182, and 196). All samples

were freeze-dried and weighed in their sample bottles, and a subsample was transferred to a crucible to be analyzed for sequential loss-on-ignition (LOI) carried out to calculate organic matter % (OM%) by weight-loss after 3 h at 550°C following standard procedures (Dean 1974). Initial sample dry weight was calculated from the measured wet weight at $T_{0,t}$ multiplied by the mean of the dry weight to wet weight ratio for the three $T_{0,0}$ samples (0.08) plus the mean 150 mL microalgae dry weight contribution (0.0604 g). Initial organic matter content for all samples frozen at time t ($OM_{0,t}$) was calculated by multiplying the mean OM% of the three $T_{0,0}$ samples by the dry weight at $T_{0,t}$.

To assess sample fractional OM loss from t = 0 at time t ($fOM_{0,t}$), first the OM at time t (OM_t) was calculated as the dry mass left at time t multiplied by the OM% measured at time t. OM loss ($fOM_{0,t}$) was finally calculated as the ratio of OM_t to $OM_{0,t}$ for each sample (expressed as a fraction). Loss of OM over time was assessed as a linear regression of triplicate set mean values to develop an experimental rate of OM loss (*see* Eq. 1).

Fractional organic carbon (OC) was calculated from OM using a lake-specific conversion factor estimated from analysis of 21 sediment samples with a range of %OM (14–64%) with total OC determined via mass-spectrometry elemental analysis. Sediment samples from a range of sources were used, with 8 from the shallow trap (10 m water depth), 8 from the deep trap (25 m water depth), and 5 from a sediment core (taken in 2011 from the deepest part of the lake at 30 m, using samples at 8, 18, 55, 83, and 110 cm depth; for more details, see Radbourne 2018).

Results

The experiment results show a loss of OM with time from deployment (Fig. 1), with the mean mass of remaining dry sample reducing from 1.9 g in T_0, to 1.7 g in T_{14}, to 1.5 g in T_{98} to 1.4 g in T_{196}, having all started from a similar ~ 23 g wet microalgae weight. The ratio weight loss combined with the trend of falling OM% with number of days (Fig. 2; $y = -0.03t + 80.78$; $r^2 = 0.42$; $p \le 0.004$), resulted in the increasing fraction of OM loss over time from the initial sample for each of the 54 bottles ($T_{0,t}$; Fig. 1). The experimental rate of OM loss (fOM, as a fraction of the original) is

$$f\mathrm{OM} = -0.001864t \left(r^2 = 0.9; p \le 0.0001\right), \qquad (1)$$

where t is the time in days (corresponding to the time the sample has been in the trap before retrieval). Note that OM will continue to be lost after retrieval until such time as the OM can be frozen or dried, though at higher temperatures and with exposure to oxygen this will be at a faster rate.

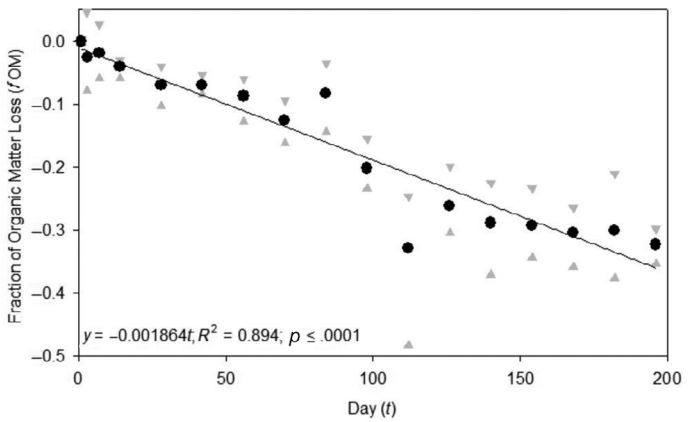

Fig. 1. Trap experiment fOM loss from set T_0. Shows triplicate set mean (black circles) with high and low range (gray triangles). Linear regression line was fit to the set mean values.

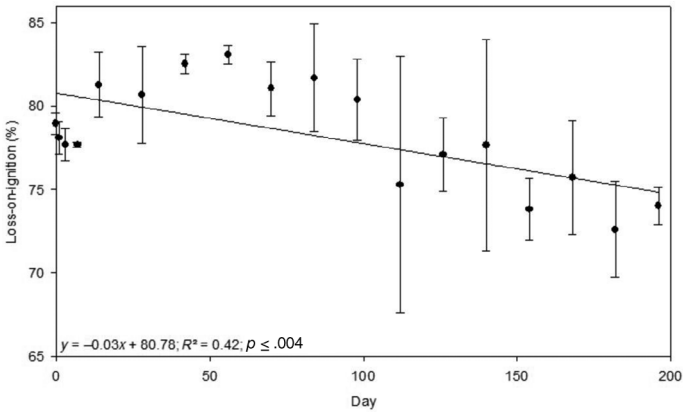

Fig. 2. Triplicate mean LOI% with SD error. The linear regression line shows a trend of decreasing LOI with increasing deployment time. Linear regression line was fit to the set mean values.

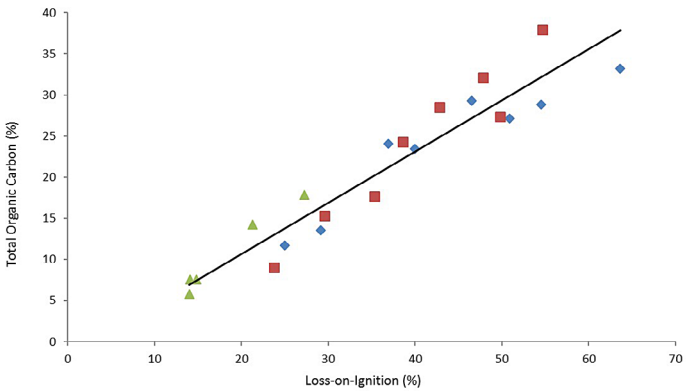

Fig. 3. Rostherne Mere specific LOI to TOC conversion factor estimated from analysis of 21 sediment samples with a range of %OM (14–64%) with total OC determined via mass-spectrometry elemental analysis. Sediment samples from a range of sources were used, including: 8 shallow trap (squares), 8 deep trap (diamonds), and 5 from a sediment core (triangles).

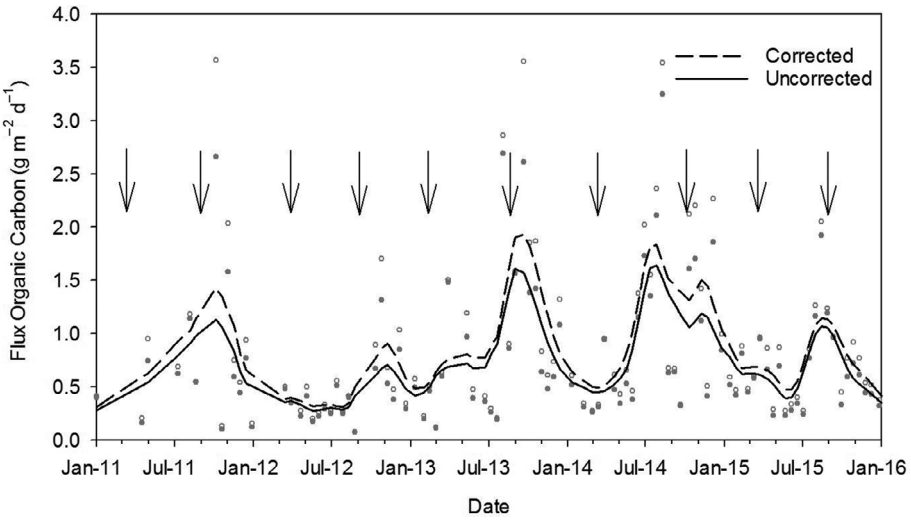

Fig. 4. Application of in-trap mineralization correction to published (uncorrected) organic carbon shallow trap sediment flux data from Radbourne et al. (2017). Lowess smoothed data (corrected = dashed line, uncorrected = solid line), point data (corrected = filled circles, uncorrected = unfilled circles), and dates of trap resetting (downward arrows).

The linear rate of OM loss was significant, with two outlying sample sets at T_{84} and T_{112} (Fig. 1). Analysis of LOI data shows fresh samples consisted mostly of OM ($\sim$ 80%; Fig. 2) as expected given the experimental design using labile macroalgae. However, some sample sets had a large variation in their LOI, specifically set 12 (T_{112}) that had one particularly low OM% content (66%; Fig. 2) that in turn resulted in an unexpectedly large OM loss (Fig. 1). The significant OM loss trend across the entire data set over 180 d enables a reliable time-correlated correction factor to be applied to estimate the original total OM flux to the traps.

Mass-spectrometry elemental analysis estimated the lake-specific total organic carbon (TOC) fraction of OM to be 0.56 (%OC = %OM * 0.56), with a strong positive relationship (Fig. 3; $y = 0.6232x - 1.785$; $r^2 = 0.9$; $p \leq 0.0001$). Sediment OM% values are lower than trap seston as expected, as more labile C will be present in seston and preferentially mineralized over time in lake sediment, and from the input to profundal lake sediments of other non-OM such as autochthonous carbonates (precipitated in spring/summer at Rostherne Mere), siliceous microfossils (diatoms), and occasional inputs of catchment minerogenic matter from inflows. The three sedimentary types (shallow trap, deep trap, and profundal lake sediment) are indistinguishable in terms of their relationship between OC% and OM%, however, suggesting they are all part of the same continuum of OC mineralization.

Table 1. Comparison of uncorrected and mineralization corrected annual mean organic carbon flux (g C m^{-2} d^{-1}) in the shallow trap (10 m) from Radbourne et al. (2017).

Year	Uncorrected	Corrected	Difference (%)
2011	0.79	0.97	23.2
2012	0.43	0.51	19.3
2013	0.90	1.06	17.9
2014	0.99	1.16	17.1
2015	0.62	0.71	13.7
Mean	0.75	0.88	18.2

Discussion

Experimental loss rate

The experimental study shows that the deployment time length of sediment traps has a significant impact on the assessment of organic seston. Shorter-term deployments (< 7 d) result in little loss of OM (less than−0.013 *f*OM), thus additional consideration is unlikely to be required. However, longer-term deployment (> 7 d) will incur increasing in-trap mineralization, with approximately −0.186 *f*OM loss in traps deployed for 100 d and −0.336 *f*OM loss in deployments of 180 d.

Often, seston loss is not accounted for in studies using sediment traps because of the difficulties of quantifying loss (Bloesch and Burns 1980; Horppila and Nurminen 2005), and is certainly not considered relative to the time samples were in traps before removal to the laboratory (where mineralization is assumed to be halted by freezing and/or drying). However, this study highlights the increasing underestimation of settling OM flux with time from capture to recovery, which needs consideration for accurate sediment trap seston analysis.

There were two outliers for the OM loss relationship (Fig. 1), set 10 (day 84, T_{84}) and set 12 (day 112, T_{112}). One of the three samples from T_{112} had a particularly high *f*OM loss ($T_{112c} = -0.484$), skewing the mean for the set beyond the experiment trend. If this outlying sample was removed, the fractional average loss of the other two samples is −0.251 *f*OM, much closer to the linear trend. The OM% for

the outlying sample was also low (66%, Fig. 2), suggesting a heightened mineralization had taken place compared to other samples. No obvious explanation is apparent, but this sample perhaps consisted of more degraded OM to begin with, and/or less organically rich algal specimens. On the other hand, set T_{84} resulted in less loss compared to the experiment trend for that time. The starting triplicate set algal weight was slightly higher than other samples (approximately +0.5 g), so potentially it is a feature of proportional action, that is, there is more OM for a limited microbial community to act on. Additionally the OM% was higher in one sample, this suggesting the reverse of the potential explanation given for set T_{112}. Otherwise, it could be an influence of a less active bacterial community and/or more recalcitrant OM, though this remains speculation. These outliers highlight the challenges (that we argue can be overcome) in confidently generating a correction method for in-trap mineralization due to the range of influences in trap sample collections.

Application to trap OM and C flux data

The conversion of OM% to TOC (Fig. 3) enables estimation of the OC loss in traps. The calibration is slightly higher at Rostherne Mere than is often used as a standard conversion (0.56, compared to 0.47 as suggested in Dean 1974). This is due to Rostherne Mere producing higher amounts of labile (algal) OC throughout the period covered by the sediment core and over the more recent monitoring campaign when traps have been installed.

Traps are an increasingly widely used method of quantifying OC cycling and burial in freshwaters (Sobek et al. 2009; Radbourne et al. 2017). Therefore, to trace accurately the fate of OC in freshwaters using sediment traps, there needs to be a consideration of in-trap loss. Application of the in-trap loss correction (Eq. 1) to published shallow trap (10 m) sediment trap data from Rostherne Mere from 2011 to 2015 (Radbourne et al. 2017) highlights the importance of developing an effective correction metric when using longer-term sediment trap deployments (Fig. 4). The correction equation was applied to the two-weekly automated sediment trap collections for total OM from 2011 to 2015, based on the mid-date of sample collection (i.e., mid-date between individual trap bottle opening to closure) to the date of trap recovery. Without correction, in-trap mineralization is estimated to create an average annual 18.2% underestimation in OM settling flux (Table 1), although this varies between 13.7% (2015) and 23.2% (2011) depending on the timing of trap deployment compared to lake production. Looking at the seasonal pattern of trap collection (Fig. 4) shows how timing of trap deployment affects estimates of OC flux for individual samples, and can change interpretations of ecosystem dynamics, both in terms of flux magnitude, but also more subtle effects on flux pattern. For example, after correction, annual maximum flux values (from individual 2-weekly periods) are remarkably similar in summer 2011, 2013, and 2014 at ~ 3.5 g C m^{-2} d^{-1}, while with uncorrected data (or indeed corrections using any constant loss factor), the highest value appears to be in summer 2014 (at ~ 3.3 g C m^{-2} d^{-1}). Maximum flux in any year (and hence inferred primary production) occurs in the same sample across all 5 yr with corrected or raw data, except for 2013, where peak flux is offset, occurring later by about 2–3 weeks (shifting from August to September 2013; Fig. 4). While these effects are subtle, they nonetheless may be significant in inferring processes driving primary production and ecosystem function at this finer temporal scale.

At an annual scale at Rostherne Mere (2011–2015; Table 1), raw trap OC flux values would underestimate OM production by 43.0 t OM yr^{-1} (uncorrected = 236.4 t OM yr^{-1}; corrected = 279.4 t OM yr^{-1}, Radbourne et al. 2017). Radbourne et al. (2017) applied an arbitrary 10% in-trap mineralization correction to all trap samples (i.e., not time-dependent), resulting in an estimate of 255.3 t OM yr^{-1}, some 24.1 t yr^{-1} less than the data corrected using Eq. 1.

In their study of 11 lakes in Québec, Ferland et al. (2014) applied a 1.67 correction factor to the sinking C trap flux based on data discussed in Bloesch and Burns (1980), who summarize past studies to highlight the lack of informative data to calculate artificial in-trap mineralization effectively. Application of the sinking OC flux data in Ferland et al. (2014) to our study findings (by removing the –0.67 *f*OM correction applied in their study and using a 37.5 d *f*OM correction from this study, –0.0699 *f*OM, i.e., the central time point for the average 75 d trap deployment) results in a 1.04 g C m^{-2} yr^{-1} loss in traps, compared to 10.3 g C m^{-2} yr^{-1} loss in traps suggested in the text. Upscaling this difference in correction factor to the entire area of the 11 lakes studied in Ferland et al. (2014; 39.04 km^2 total area) results in a ~ 360 t C yr^{-1} difference in sinking C flux to this study findings (this study = 41.96 t C yr^{-1}, original study = 402.22 t C yr^{-1}). We suggest the nontime-dependent 1.67 correction in Ferland et al. (2014) is very high, especially considering the trap deployment was only for an average of 75 d, and believe this comparison highlights the importance and applicability of this study to trap flux corrections.

Some studies do not include even an arbitrary in-trap loss correction (e.g., Lehmann et al. 2004; Urban et al. 2004). Application of our OM loss equation (Eq. 1) to the low and high mass fluxes of off-shore trap deployments in Urban et al. (2004; average 30 d deployments) suggests their study may underestimate OM fluxes by between 3.06 and 12.25 g Cm^{-2} yr^{-1}. This is a potentially significant error when upscaled to the whole lake basin and possible application beyond to regional, national, or even global OC sequestration estimates. It should be noted that the estimations for these other studies are only given as an example of impact and application. The precise figures would be different due to variances in trap type, location, and the impact of other mineralization drivers (as discussed later).

Of course, a small proportion of OM mineralized in the trap could also be lost during settling through the water column (discussed below) and substantially more following deposition

on the lake bed. However, the implications of this study in highlighting the common under- or overestimation of OM and OC flux (as measured by traps, as opposed to accumulation or burial rate measured from lake sediments) is of great importance for the future effective use of sediment trap studies, especially those involved in calculating total biomass production and OC sequestration.

Mineralization correction applicability and reliability

Equation 1 corrects for in-trap OM loss for studies conducted in similar lake conditions. We recognize that the experimental results reported here likely represent the upper end of losses experienced, as the experiment design focused on OM only, hence we have used fresh autochthonous macroalgae. The age of the algae used in the experiment (i.e., live algae from the lake surface rather than algae that has had a short time of sedimentation through the water column to the sediment trap depth; Bloesch et al. 1977, Premazzi and Marengo 1982) might on initial inspection be considered to misrepresent the sediment collected in the traps.

The OM% difference between this experiment over the 180 d (mean 78.4% OM, minimum 66.0%, maximum 83.9%) and actual Rostherne Mere summer trap data (shallow trap = mean 57.9% OM, minimum 43.3%, maximum 72.8%; deep trap = mean 54.1% OM, minimum 41.0%, maximum 64.7%) suggests a difference in OM between live (experiment) and traps of 20.5–24.3%. This difference may be partly explained by losses in OM from live to sediment trap depth at 10 m. However, it will also be a function of the non-OM component settling in the traps (which has a lower LOI%) that is not represented in the experiment design, this reducing the average trap LOI% compared to the experiment, with the OM component of the trap collections possibly behaving very much like the experimental design.

To evaluate the potential implications of this LOI difference, we reassessed the data in this study with a 22.5% reduction in experimental LOI (to represent a similar LOI as would be expected in the traps during summer), resulting in only a less than −0.01 *f*OM loss difference over a 6-month deployment compared to the current model (*f*OM loss = −0.0005 by day 10, −0.0048 by day 100, −0.0086 by day 180). We are aware that Rostherne Mere's summer OM% in traps is particularly high, with other studies reporting trap OM values $\sim$ 20% (Håkanson 1984; Niemisto et al. 2012). Adjusting model calculations with an initial OM of $\sim$ 20% for each sample to test the importance of this gave a less than −0.045 *f*OM loss difference over a 6-month deployment compared to the current model (*f*OM loss = −0.0025 by day 10, −0.0246 by day 100, −0.0443 by day 180). Overall, considering the adjustment with this OM correction is so small, the results of this study show that there is a slow early loss rate so we argue that minimal loss will have occurred before deposition in traps. Furthermore, the purpose of this study was to reduce the uncertainty of arbitrary loss estimates (rather than add another). Similarly, in vitro studies by Mattson and Likens (1993) found no difference in decomposition rate of trap seston with or without washed sand added (in effect reducing sample organic matter %). Therefore, we believe the experimental trap sediment effectively represents the dynamics of the naturally collected sediment during summer and we thus suggest that our model for *f*OM is applicable in a wide variety of lake systems across a range of seston OM%.

The significant fit of experimental OM loss supports the reliability of the correction factor to estimate the total OM flux (Fig. 1). A reliable correction factor is a great benefit to study design and management, as it will give confidence to the deployment of longer-term sediment trap studies (up to 6 months between redeployments). Reliable long-term trapping studies would reduce the logistical and financial difficulties related to short-term collection procedures, especially as collections of < 7 d would be required to avoid the potential of significant in-trap loss. This would enable more studies to take place and importantly continue the long-term development of larger and increasingly valuable data sets for monitoring aquatic ecosystem net production. We believe 6 months is a suitable period for trap deployment that would save logistical and financial pressures involved with sediment trap deployment; however, future studies might want to consider longer-term in-trap diagenetic impacts further.

Future considerations include lakes where hypolimnetic temperatures are higher, or those with different OM sources in the seston. For example, the mineralization rate will be less in systems where the relative contribution of (labile) microalgae is less; there is significant flocculation of dissolved-OM; or where particulate-OM has significant contributions from less labile C, such as from terrestrial sources. Similarly, the dominant pathway for mineralization in this experiment is through anoxia and anaerobic decomposition, as the dissolved oxygen initially contained within the sample bottles will be used up quickly (especially where seston flux is low, e.g. in oligotrophic lakes and during winter). More work is required to quantify differences relating to other lake types and catchment settings, microbial communities, redox conditions, trap types, and study locations. For example, sediment source, oxygen availability, and temperature (Jonsson et al. 2001; Sobek et al. 2009) will all influence seston mineralization rates, with higher ambient (in-trap) temperatures expected to lead to greater rates of OM mineralization. Furthermore, this experiment has only focused on the effect of in-trap mineralization on automated carousel-type sequencing traps that provide a sealed sample. In-trap mineralization of open traps (i.e., those that do not seal after collection and have open contact with ambient lake water) will be complicated by the possibility for oxygen diffusion (if in the epilimnion), bioturbation, and possible grazing impacts from detritivores, for example, all of which will likely lead to greater mineralization of OM.

Other aspects of in-trap seston taphonomy (such as calcite dissolution and diatom dissolution) can also be assessed by

the approach taken in this experiment and considered for future studies (i.e., to investigate any such taphonomic impacts on stable isotope values of calcite or diatom silica). Calcite dissolution, for example, should be enhanced by anoxia (Ohlendorf and Sturm 2001; Bluszcz et al. 2009), while diatom dissolution should be reduced (Ryves et al. 2006).

Despite the challenges with sediment trap deployment this study highlights, it does also reaffirm their importance and value as tools of in-lake assessment of ecosystem dynamics. We hope this experiment can reignite a drive toward resolving the long-standing issue of in-trap loss with sediment trap use for accurately measuring settling seston (and especially OM) flux. Resolving this issue will enable longer-term sediment trap deployments (> 6 months), thus improving the number and longevity of these important monitoring series in lakes across the world, while ensuring the data are comparable between lakes, even where traps have been deployed for differing lengths of time. Furthermore, the generation of reliable loss factors can be retroactively applied to existing sediment trap data to recalculate seston and especially OM fluxes, thus allowing a re-evaluation of published and archived trapping data sets, adding to their value.

References

Bloesch, J. 1994. A review of methods used to measure sediment resuspension. Hydrobiologia **284**: 13–18. doi:10.1007/BF00005728

Bloesch, J., P. Stadelmann, and H. Buhrer. 1977. Primary production, mineralization, and sedimentation in euphotic zone of 2 Swiss lakes. Limnol. Oceanogr. **22**: 511–526. doi:10.4319/lo.1977.22.3.0511

Bloesch, J., and N. M. Burns. 1980. A critical-review of sedimentation trap technique. Schweiz. Z. Hydrologie **42**: 15–55. doi:10.1007/BF02502505

Bluszcz, P., A. Lucke, C. Ohlendorf, and B. Zolitschka. 2009. Seasonal dynamics of stable isotopes and element ratios in authigenic calcites during their precipitation and dissolution, Sacrower See (northeastern Germany). J. Limnol. **68**: 257–273. doi:10.4081/jlimnol.2009.257

Charlton, M. N., and D. R. S. Lean. 1987. Sedimentation, resuspension, and oxygen depletion in Lake Erie (1979). J. Great Lakes Res. **13**: 709–723. doi:10.1016/S0380-1330(87)71685-2

Dean, W. E. 1974. Determination of carbonate and organic-matter in calcareous sediments and sedimentary-rocks by loss on ignition - comparison with other methods. J. Sediment. Petrol. **44**: 242–248. doi:10.1306/74D729D2-2B21-11D7-8648000102C1865D

Eadie, B. J., R. L. Chambers, W. S. Gardner, and G. L. Bell. 1984. Sediment trap studies in Lake-Michigan - resuspension and chemical fluxes in the Southern Basin. J. Great Lakes Res. **10**: 307–321. doi:10.1016/S0380-1330(84)71844-2

Ferland, M.-E., Y. T. Prairie, C. Teodoru, and P. A. del Giorgio. 2014. Linking organic carbon sedimentation, burial efficiency, and long-term accumulation in boreal lakes. J. Geophys. Res. Biogeosci. **119**: 836–847. doi:10.1002/2013JG002345

Flower, R. J. 1991. Field calibration and performance of sediment traps in a eutrophic holomictic lake. J. Paleolimnol. **5**: 175–188. doi:10.1007/BF00176876

Håkanson, L. 1984. On the relationship between lake trophic level and lake sediments. Water Res. **18**: 303–314. doi:10.1016/0043-1354(84)90104-0

Horppila, J., and L. Nurminen. 2005. Effects of calculation procedure and sampling site on trap method estimates of sediment resuspension in a shallow lake. Sedimentology **52**: 903–913. doi:10.1111/j.1365-3091.2005.00726.x

Jonsson, A., M. Meili, A. K. Bergström, and M. Jansson. 2001. Whole-lake mineralization of allochthonous and autochthonous organic carbon in a large humic lake (Ortrasket, N. Sweden). Limnol. Oceanogr. **46**: 1691–1700. doi:10.4319/lo.2001.46.7.1691

Kulbe, T., C. Ohlendorf, and M. Sturm. 2006. Lacustrine particle dynamics in high-altitude Estany Redo (Spain) - a high resolution sediment trap study. J. Limnol. **65**: 89–99. doi:10.4081/jlimnol.2006.89

Lehmann, M. F., S. M. Bernasconi, A. Barbieri, M. Simona, and J. A. Mckenzie. 2004. Interannual variation of the isotopic composition of sedimenting organic carbon and nitrogen in Lake Lugano: A long-term sediment trap study. Limnol. Oceanogr. **49**: 839–849. doi:10.4319/lo.2004.49.3.0839

Meyers, P. A., and B. J. Eadie. 1993. Sources, degradation and recycling of organic matter associated with sinking particles in Lake Michigan. Org. Geochem. **20**: 47–56. doi:10.1016/0146-6380(93)90080-U

Niemisto, J., P. Tamminen, P. Ekholm, and J. Horppila. 2012. Sediment resuspension: Rescue or downfall of a thermally stratified eutrophic lake? Hydrobiologia **686**: 267–276. doi:10.1007/s10750-012-1021-4

Ohlendorf, C., and M. Sturm. 2001. Precipitation and dissolution of calcite in a Swiss High Alpine Lake. Arct. Antarct. Alp. Res. **33**: 410–417. doi:10.1080/15230430.2001.12003449

Premazzi, G., and G. Marengo. 1982. Sedimentation-rates in a Swiss-Italian lake measured with sediment traps. Hydrobiologia **91-2**: 603–610. doi:10.1007/BF00000059

Radbourne, A. D. 2018. Disentangling the impacts of nutrient enrichment and climatic forcing as key drivers of change at Rostherne Mere. Ph.D. thesis. Loughborough Univ. https://hdl.handle.net/2134/34927

Radbourne, A. D., D. B. Ryves, N. J. Anderson, and D. R. Scott. 2017. The historical dependency of organic carbon burial efficiency. Limnol. Oceanogr. **62**: 1480–1497. doi:10.1002/lno.10512

Radbourne, A. D., J. A. Elliott, S. C. Maberly, D. B. Ryves, and N. J. Anderson. 2019a. The impacts changing nutrient load and climate on a deep, eutrophic, monomictic lake. Freshw. Biol. **64**: 1169–1182. doi:10.1111/fwb.13293

Radbourne, A. D., D. B. Ryves, G. Madgewick, and N. J. Anderson. 2019b. The influence of climate change on the restoration trajectory of a nutrient-rich deep lake. Ecosystems. **23**: 859–872. doi:10.1007/s10021-019-00442-1

Ryves, D. B., D. H. Jewson, M. Sturm, R. W. Battarbee, R. J. Flower, A. W. Mackay, and N. G. Granin. 2003. Quantitative and qualitative relationships between planktonic diatom communities and diatom assemblages in sedimenting material and surface sediments in Lake Baikal, Siberia. Limnol. Oceanogr. **48**: 1643–1661. doi:10.4319/lo.2003.48.4.1643

Ryves, D. B., R. W. Battarbee, S. Juggins, S. C. Fritz, and N. J. Anderson. 2006. Physical and chemical predictors of diatom dissolution in freshwater and saline lake sediments in North America and West Greenland. Limnol. Oceanogr. **51**: 1355–1368. doi:10.4319/lo.2006.51.3.1355

Sobek, S., E. Durisch-Kaiser, R. Zurbruegg, N. Wongfun, M. Wessels, N. Pasche, and B. Wehrli. 2009. Organic carbon burial efficiency in lake sediments controlled by oxygen exposure time and sediment source. Limnol. Oceanogr. **54**: 2243–2254. doi:10.4319/lo.2009.54.6.2243

Teranes, J. L., J. A. Mckenzie, and A. F. Lotter. 1999. Stable isotope response to lake eutrophication: Calibration of a high-resolution lacustrine sequence from Baldeggersee, Switzerland. Limnol. Oceanogr. **44**: 320–333. doi:10.4319/lo.1999.44.2.0320

Urban, N. R., X. Lu, Y. Chai, and D. S. Apul. 2004. Sediment trap studies in Lake Superior: Insights into resuspension, cross-margin transport, and carbon cycling. J. Great Lakes Res. **30**: 147–161. doi:10.1016/S0380-1330(04)70383-4

Acknowledgments

This study was supported by the Natural Environment Research Council (NERC) (grants NE/L002493/1 to DBR and ADR, NE/H011978/1 to DBR, and NE/I007261/1 Network of Sensors) and A.D.R. acknowledges the support of the research studentship award from Central England NERC Training Alliance (CENTA). Additionally, we thank Natural England and Rupert Randall for allowing regular access to Rostherne Mere.

Conflict of Interest

None declared.

Submitted 14 October 2019
Revised 07 April 2020
Accepted 03 May 2020

Associate editor: John Smol