The fluvial flux of particulate organic matter from the UK: quantifying in-stream losses and carbon sinks.

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Abstract

This study considers records of fluvial suspended sediment concentration and its organic matter content from across the United Kingdom from 1974 to 2010. Suspended sediment, mineral concentration and river flow data were used to estimate the particulate organic matter (POM) concentration and flux. Median annual POM flux from the UK was 1596 ktonnes/yr. The POM concentration significantly declined after the European Commission's Urban Wastewater Directive was adopted in 1991 although the POM flux after 1992 was significantly higher. Estimates of POM flux were compared to a range of catchment properties to estimate the flux of particulate organic carbon (POC) and particulate organic nitrogen (PON) as they entered rivers and thus estimate the net catchment losses. The total fluvial flux of N from the soil source to rivers was 2209 ktonnes N/yr with 814 ktonnes N lost at the tidal limit, and so leaving 1395 ktonnes N/yr loss to atmosphere from across UK catchments - equivalent to an N_2O flux from UK rivers of between 33 and 154 ktonnes (N_2O)/yr. The total fluvial flux of carbon from the soil source to rivers for the UK was 5020 ktonnes C/yr; the flux at the tidal limit was 1508 ktonnes C/yr, equivalent to 6.5 tonnes

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C/km^2/yr. Assuming that all the net catchment loss goes into the atmosphere, then the impact of rivers on the atmosphere is 3512 ktonnes C/yr, equivalent to 15.2 tonnes C/km^2/yr. The loss of POM from the UK suggests that soil erosion in the UK prevents soil being a net sink of CO_2 and is instead a small net source to the atmosphere.

Keywords: particulate organic carbon, POC, particulate organic nitrogen, PON, soil erosion, N_2O.

1. Introduction

Meybeck (1993) estimated the flux of carbon (dissolved organic carbon, particulate organic carbon, and dissolved inorganic carbon) from the world's rivers to the oceans was around 542 Mtonnes C/yr in proportions respectively 37:18:45, i.e. global river flux of POC is about 98 Mtonnes C/yr. Ludwig et al. (1996) used a spatially-explicit model of global fluvial carbon fluxes to suggest fluxes of 800 Mtonnes C/yr with a split of approximately 50:25:25 for DOC:POC:DIC, i.e. a global POC of 200 Mtonnes C/yr. These figures provide useful estimates of fluvial POC losses from the land to the oceans at the tidal limit, but they do not account for in-stream losses along the length of the river, between the carbon sources (e.g. soils) and the ocean. Thus, to understand total carbon losses from catchments, it is also necessary to estimate in-stream losses.

Kempe (1982, 1984) recognised that many surface freshwater bodies were saturated with respect to CO_2. Cole et al. (1994) showed, in a survey of 1835 lakes across the globe, that lakes, on average, were supersaturated with respect to the atmosphere by a factor of 3 and that, assuming this value, degassing of CO_2 from global lakes would represent an additional 140 Mtonnes C to the atmosphere each year. This excess dissolved CO_2 comes not only from the excess CO_2 present in water entering a terrestrial water body from the

relatively closed system of soil and bedrock porosity, but also from the mineralisation of DOC and POC. Thus, rivers and lakes become sources of CO_2 because water entering them carries excess dissolved CO_2. For the global scale, Cole et al. (2007) estimated that 1900 MtonnesC/yr enters rivers of which 800 Mtonnes C/yr (42% of the input) is returned to the atmosphere. Battin et al. (2009) considered the loss of DOC from rivers at a global scale and suggested 21% removal of DOC in-stream implyng that, in comparison to the values suggested by Cole et al. (2007), there must be considerable contributions from the loss of POC and DIC. The 2007 Intergovernmental Panel on Climate Change (IPCC) report included an estimate of global DOC flux from rivers (Solomon et al., 2007) but did not consider the effect of in-stream DOC losses, let alone the flux and loss of POC. In other words, the carbon fluxes to the atmosphere from in-stream losses have not yet been included in estimates of terrestrial greenhouse gas (GHG) emissions.

Analysis of the global river network's contribution of fluvial carbon to GHG emissions has relied on relatively sparse data from very few rivers. Worrall et al. (2007) used nationally-collected monitoring data for biochemical oxygen demand (BOD) as a measure of fluvial carbon turnover (where turnover is considered as the biochemical reaction of fluvial carbon to form greenhouse gases) and estimated a loss equivalent to 31% of the DOC flux across the UK fluvial network – equivalent to an additional release to the atmosphere of 1 tonne C/km^2/yr across the entire UK land surface. However, their BOD approach assumes a fixed fluvial residence time of 5 days – a long residence time for the short, relatively-unimpounded rivers of the UK. In addition, BOD is usually measured downstream in the fluvial network, away from upstream sources of more readily degraded DOC. Alternatively, Worrall et al. (2012a) modelled the DOC export from over 194 catchments across the UK, over 7 years and assessed net watershed losses through comparisons to the soil, land-use and hydro-climatic characteristics of each catchment. They found a net watershed DOC loss of up

to 78%, equivalent to between 9.0 and 12.7 tonnes C/km^2 of UK land area/yr. These figures are large when compared to other studies on single catchments. Individual studies are often based on experimental studies of DOC turnover in the dark and not daylight (Wickland et al., 2007; del Georgio and Pace, 2008) or based on samples from systems with residence times far longer than most UK rivers, e.g. lakes - Jonsson et al. (2007). Dawson et al. (2001) studied a short river reach (2 km) in a peat-covered headwater catchment and estimated DOC removal of 12-18%. Wallin et al. (2013) considered a 67 km^2 boreal catchment and found that fluvial CO_2 evasion (loss of gas from the river surface to the atmosphere) was equivalent to 53% of the fluvial carbon flux; some of this CO_2 evasion would be due to rapid loss of DOC to the atmosphere from low-order streams. The estimated net DOC watershed loss from the UK river network (Worrall et al., 2012a) represents 3% of the UK's greenhouse gas inventory (Cannell et al., 1999).

Further, the turnover of organic matter not only represents a release of carbon to the atmosphere but also a potential loss of nitrogen. Kroeze et al. (2003) found that fluvial N retention in surface waters is typically between 11 and 50% of N input and, by retention, this is the proportion of nitrogen not transferred out of the catchment via the river and includes losses to the atmosphere. Worrall et al. (2012b) compared fluxes of dissolved nitrogen species from different sized catchments and, by allowing for differences in catchment soil cover, land use and hydro-climatic properties, the net catchment loss of dissolved nitrogen was estimated as 63% of the flux entering the rivers at the soil source.

If loss of dissolved CO_2, DOC and DON have been considered, what about the turnover of POC or PON? The decline in suspended sediment export (often referred to as yield) through a catchment has often been expressed as a sediment delivery ratio (e.g. Walling, 1983). This decline in sediment yield has been associated with in-channel storage of suspended sediment (e.g. Collins and Walling, 2007) and on floodplains (e.g. Walling and

Owens, 2003). Studies have considered the concept of spiralling of organic matter (Newbold et al., 1982**)** where organic matter is cycled through the stream biota**,** but they have not considered loss by turnover (e.g. Young and Huryn, 1997, Griffiths et al., 2012). Furthermore, studies that have examined the fate of eroded soil have reported erosion as a net sink of carbon (e.g. Van Oost et al., 2007) but have not explicitly considered the possibility of CO_2 being released from the eroded soil organic matter as it is transported within the river network. Therefore, given the large potential impact that organic matter turnover in rivers has on atmospheric greenhouse gases, the aim of this study is to estimate the flux of particulate organic matter from soil source to UK rivers and from UK rivers to the tidal limit and so assess the loss of particulate carbon and nutrients in stream.

2. Methodology

The approach used to calculate fluvial fluxes for individual catchments was similar to that of Worrall et al. (2013a), and the approach used to assess net catchment losses followed that of Worrall et al. (2012a and b).

The study used data from the Harmonised Monitoring Scheme (HMS - Bellamy and Wilkinson, 2001). There are 56 HMS sites in Scotland and 214 sites in England and Wales (Figure 1, Table 1). Note that one Scottish river (River Tweed) actually is included in the NE England dataset because, although most of its catchment is in Scotland, its tidal limit is in England. HMS monitoring sites were selected for the inclusion into the original monitoring programme if they were the tidal limit of rivers with an average annual discharge greater than 2 m^3/s; in addition, any tributaries with a mean annual discharge above 2 m^3/s were also included in the original monitoring programme. These criteria provided good spatial coverage of the coast of England and Wales, but in Scotland many of the west coast rivers are too small to warrant inclusion in the HMS. No HMS data were available from Northern Ireland.

Within the database maintained as part of the HMS programme, three determinands were of particular interest to this study: suspended sediment concentration (mg/l); instantaneous river flow (m^3/s); and ash content of the suspended sediment (mg/l). From these data the suspended sediment flux was estimated and, as the ash content represents the mineral proportion of the suspended sediment (particulate mineral matter – PMM), it was possible to calculate the particulate organic matter (POM) concentration of each sample by difference. From the calculated POM concentrations and river flow data, it was then possible to calculate the POM flux.

2.1. POM concentrations

Analysis of variance (ANOVA) was used to consider all data from all sites for which the frequency of sampling was more than 12 per year. In the ANOVA three factors in relation to the concentration and percentage of mineral matter (PMM) were considered: (1) the difference between years with 37 factor levels, one for each year between 1974 and 2010 – henceforth referred to as the year factor; (2) the month of sampling with 12 factor levels, one for each calendar month – henceforth referred to as the month factor; and, (3) the differences between sampling sites – henceforth referred to as the site factor. The analysis was considered with and without covariates of suspended sediment concentration and instantaneous river discharge at the time of sampling. Both covariates were log-transformed to ensure the greatest proportion of the original variance in the dataset was explained. On the basis of the ANOVA, multiple regression was used to model PMM concentration, percentage PMM content and subsequently the POM concentration.

2.2. Flux calculation

148 Among the monitoring agencies, water quality sampling frequencies (f) vary, ranging from
149 sub-weekly to monthly or even less frequently. Annual data were rejected at any site where
150 there were fewer than 12 samples in that year with the samples in separate months (f<12); in
151 this way a range of flow conditions would be sampled. In general, 12 monthly spot samples
152 was the sampling scheme being followed within the HMS. An interpolation method was then
153 applied to calculate the flux for any site-year combination that met this criterion:
154

155 $$F_y = K\sum_{i=1}^{n_y} n_x C_i Q_i \qquad \text{(i)}$$

156 $$n_x = \frac{A_y}{n_y} \qquad \text{(ii)}$$

157

158 where: F = the annual flux at the site; C_i = the measured concentration at the site at time i;
159 Q_i= the river discharge at time i; K = a conversion factor which takes into account the units
160 used; n_y = the number of samples at the site in that year; and A_y = the number of days in that
161 year, i.e. this can vary with a leap year. This approach assumes that each sample taken at a
162 site is equally likely to be representative of an equal proportion of the year as any other
163 sample. Note that this method corresponds to "method 2" of Littlewood and Marsh (2005)
164 modified for irregular sampling.
165 When considering suspended sediment, or indeed any particulates, Webb et al. (1997)
166 considered 5 interpolation and 2 extrapolation methods and found that, for suspended
167 sediment flux estimation, extrapolation methods gave the least biased results and that bias
168 increased with decreased sample frequency. Several studies have recommended or considered
169 adaptive strategies. Kronvang and Bruhn (1996) suggested taking samples "hydrologically"
170 rather than on a regular basis and a number of studies (e.g. Cooper and Watts, 2002;

171 Skarbøvik et al., 2012) have suggested including flood samples alongside regular sampling.
172 However, the use of extrapolation and adaptive strategies is impractical when considering a
173 dataset from a monitoring scheme such as the HMS monitoring network in the UK where
174 sampling is regular rather than adaptive and often infrequent (typically monthly).

175 The quality of methods and sampling frequencies used to calculate flux need to be
176 considered in two ways. Firstly, the accuracy can be considered as the difference between the
177 true load and estimated load and represents the systematic bias. Secondly, the precision of the
178 method represents the spread of the load estimates about a central value, in other words the
179 consistency of the load estimates. In many studies that discuss uncertainty in flux estimation
180 due to changing method or sampling frequency, it is the precision that is described and not
181 the bias or accuracy. An example of this is Littlewood et al., (1998) who could only trace
182 precision with changing sampling frequency with "indicative" curves but could not discuss
183 accuracy of methods because there was no "true" value available. Johnes (2007) considered
184 17 catchments where there was daily measurement of phosphorus but had no sub-daily data
185 and chose to assume that "method 5" (Littlewood, 1995) was the true value and only
186 considered precision but not bias. The lack of a "true" value with which to compare bedevils
187 the assessment of bot accuracy and precision of changing method of sampling frequencies.
188 Cassidy and Jordan (2011), with sub-daily measurement of phosphorus, considered both bias
189 and precision in their approach and thus showed increasing bias with decreasing sampling
190 frequency, with bias of up to 60% on monthly sampling, and large uncertainty for all
191 sampling frequencies except for near continuous monitoring. Worrall et al. (2013b) showed
192 that "method 5", often quoted as the preferred interpolation method, contained a curious bias
193 because it corrected "method 2" by assuming that river discharge is normally distributed,
194 which it is not, and this led to overestimation of fluxes at high sampling frequencies ($f > 1$
195 sample per 7 days) because the method overestimated the expected value of the river

discharge, i.e. it used an arithmetic mean when other estimates of the expected value of river discharge (e.g. geometric mean) would be more appropriate. Furthermore, Worrall et al. (2013b) showed that extrapolation methods were, because of large changes in the sample set used to derive rating curve, highly erratic for low-frequency sampling leading to poor accuracy and precision.

It is clear that for the type of low-frequency data available to this study, there could be considerable sampling bias, most likely leading to underestimation. However, Worrall et al. (2103a) tackled this issue for the low-frequency data typical of the HMS by application of analysis of covariance (ANCOVA) to establish and correct for sample frequency bias: the sampling frequency for all site-year combinations was compared to a flow-weighted flux estimate (i.e. $\frac{F_y}{\sum Q}$), which is equivalent to the annual average flow-weighted concentration. In the ANCOVA, sampling frequency was considered as a factor with four levels (sampling frequency ≥ 1 per week, ≥ 2 weeks, ≥ 3 weeks, and ≥ 1 per month – note that sites with a sampling frequency of worse than 1 per month had already been removed). The annual water yield for each site-year combination ($\sum Q$) was used as the covariate. The normality of the data was tested using the Anderson-Darling test (Anderson and Darling, 1952); if the test failed at a 5% probability of the data not being normally distributed, then the data were transformed and the distribution re-tested. The Tukey test was used to assess *post hoc* differences between factor levels. If there is a significant effect due to sampling frequency at the 95% probability, then ANCOVA demonstrates that a sampling bias exists with changing sampling frequency. Where significant differences were found, a correction factor for that sampling frequency was derived relative to the other factor levels, i.e. relative to other sampling frequencies. These correction factors were then applied to the flow-weighted flux for each site-year combination to adjust the interpolation method results. This was applied in such a way as to correct all flow-weighted fluxes to the average sampling bias for the highest

221 sampling frequency; thus, all results were adjusted as though each site had been sampled sub-
222 weekly. Worrall et al. (2013a) tested this method for correcting for sampling bias by applying
223 the extrapolation method of Ferguson (1986) to calculating suspended sediment flux for the
224 two sites in the HMS dataset with the largest contrast in baseflow index within the HMS
225 catchments (BFI – Gustard et al., 1992). The extrapolation method was applied based upon
226 all the available suspended sediment concentration and flow data for that catchment and
227 given that the data were made stationary over the time series of their sampling period. For the
228 River Test (the high BFI catchment) the 10-year average suspended sediment flux the results
229 were: 4820 tonnes/yr for extrapolation method; 3179 tonnes/yr for interpolation method; and
230 4705 tonnes/yr for the corrected interpolation method, i.e. the correction method used here
231 gave a result that was 98% of that from an extrapolation method. For the River Thurso (the
232 low BFI catchment) the 10-year average suspended sediment flux the results were: 1302
233 tonnes/yr for extrapolation method; 2427 tonnes/yr for interpolation method; and 5270
234 tonnes/yr for the corrected interpolation method. The reason for the low estimate from the
235 extrapolation method in this latter case is that rating curve for this catchment shows two
236 distinct trends even once it had been made stationary, i.e. extrapolation methods can be
237 unreliable because a single relationship is assumed but not always appropriate.

238 The bias-corrected flux for each HMS site in each year was then used to calculate the
239 export rate as the flux per unit catchment area per year. The flux from the UK was then
240 calculated using an area-weighted average of export rates. For each region of the UK for
241 which POM fluxes could be estimated, an average export was calculated for each year from
242 1974 to 2010 (Figure 1, Table 1). The regions are based upon UK Environment Agency's
243 administrative areas that are bounded by watersheds. The flux from all the regions was
244 summed to give the national flux. This regional approach better represents regional
245 differences without biasing the national value due to uneven spatial distribution of available

246 records, while also using all site information to calculate national-scale flux. Errors due to
247 upscaling from catchment export estimates to the regional and national scales was estimated
248 as half the percentage difference between the values estimated from the 5^{th} and 95^{th} percentile
249 exports for each region: this gives an estimated upscaling error of ±15%. It should be noted
250 that no HMS data were available for Northern Ireland. However, the land area of Northern
251 Ireland is 13843 km^2 and so the results for Great Britain (the countries of England, Wales
252 and Scotland, i.e. the UK without Northern Ireland) could be scaled up to give an estimate of
253 the flux from the whole of the UK.

254 Where a catchment PMM or POM flux could be calculated for the period 2001 to
255 2010, the average catchment flux over those years was compared to a range of catchment
256 characteristics. The period 2001-2010 was chosen for three reasons: (1) it is the most recent
257 decade; (2) a decadal average is less likely to be distorted by particularly wet or dry years;
258 and, (3) the available land use data were collected for the middle year of this period. The land
259 use for each 1 km^2 of Great Britain (i.e. the UK minus Northern Ireland) was classified into:
260 arable, grass and urban from the June Agricultural Census for 2004 (Defra, 2005). In
261 addition, the number of cattle and sheep in each 1 km^2 were recorded within this census. To
262 provide a single measure for livestock, the equivalent sheep per hectare were calculated based
263 on published nitrogen export values of the respective livestock (Johnes et al., 1996) which
264 gives a ratio of 3.1 sheep per cow. The dominant soil-type of each 1 km^2 grid square in Great
265 Britain was classified by this study into mineral, organo-mineral and organic soils based upon
266 the classification system of Hodgson (1997), and used nationally-available data (Smith et al.,
267 2007, Lilly et al., 2009). Note that, by this definition, peat soils are a subset of organic soils.
268 The catchment area to each monitoring point was calculated from the CEH Wallingford
269 digital terrain model which has a 50 m grid interval and a 0.1 m altitude interval (Morris and
270 Flavin, 1994). The soil and land-use characteristics for each 1 km^2 were summed across each

catchment to the monitoring points with available flux information and, on each catchment, the relative proportion of different soil and land-use properties was determined in addition to a range of hydrological characteristics. For each of the catchments, for which the study could calculate a suspended sediment flux, the following hydrological characteristics were used: the base flow index, the average actual evaporation and the average annual rainfall. The hydrological characteristics for each catchment were available from the National River Flow Archive (www.ceh.ac.uk/data/nrfa/). The study did not include the average annual total river flow for each catchment within the hydrological characteristics considered. The average annual total river flow would simply be the difference between average annual rainfall and the average actual evaporation for each catchment: if total river flow were important it will be apparent from the importance of these two variables. Multiple linear regression was used to compare the average annual flux for the period 2001 to 2010 to catchment characteristics.

The multiple linear regression was performed with both explanatory variables and the response variable untransformed and log-transformed. Normality of transformed and untransformed variables was tested using the Anderson-Darling test and variables were only included in the model if they were statistically significant (probability of difference from zero at $p < 0.05$). Models were chosen both on the basis of model fit, as assessed by the correlation coefficient (R^2), and the physical interpretation of the model. Of particular interest were models which only included those soil and land-use characteristics that could be mapped across Great Britain, and models that identified a relationship between POM flux and catchment area. The latter were used because significant net watershed losses should be discernible from the relationship between total POM flux and catchment area. The best-fit significant model was obtained to judge this relationship. If the best-fit model included catchment area, the model was then recalculated excluding catchment area and the residuals of that model were compared to the catchment area. In using regression to filter the data for

296 effects other than that of catchment area, care was taken to consider information that was a
297 proxy or co-linear with catchment area, e.g. area of arable land. An analysis of residuals was
298 performed for statistically significant models, where a standardised residual (residual divided
299 by its standard deviation) greater than 2 was considered an outlier and worthy of further
300 investigation. As further analysis of fit of preferred models, the residuals after model fitting
301 were tested for normality using the Anderson-Darling test.
302
303 **3. Results**
304 *3.1. Mineral and organic matter concentrations*
305 For site-year combinations that met the criterion of f >1 sample per month, there were 35490
306 mineral concentrations from 1974 to 2010. The median PMM concentration was 7.5 mg/l
307 with a 5^{th} percentile of 1 mg/l and a 95^{th} percentile of 75 mg/l. When the percentage mineral
308 content was considered, the median percentage was 66.7% with a 5^{th} percentile of 25% and a
309 95^{th} percentile of 94%. Given the percentage mineral content, the median POM concentration
310 was 4.6 mg/l; with a 5^{th} percentile of 0.5 mg/l and a 95^{th} percentile of 23 mg/l. Assuming the
311 carbon content of particulate organic matter was between 45 and 50% by mass (Moody et al.,
312 2013), our results suggest that organic carbon content as a percentage was between 2.7% (5^{th}
313 percentile) and 38% (95% percentile) with a median value of 15.8%, which in turn yields a
314 median POC concentration of 2.2 mg C/l varying between 0.2 mg C/l (5^{th} percentile) and
315 11.5 mg C/l (95^{th} percentile). For British rivers, Hope et al. (1997) gave a "preferred" value of
316 14% organic C content, while Hillier (2001) measured the carbon content of suspended
317 sediment along the River Don in Scotland (catchment area = 1,320 km^2) and showed values
318 varied between 6.9 and 14.1%. Neal (2003) studied sediment from rivers with catchment
319 areas from 373 to 8231 km^2 and found organic carbon contents varied from 5 to 17%.

320 The ANOVA of PMM concentration data showed that all three factors and one
321 interaction were significant at $p < 0.05$ (Table 2). The most important factor was the site
322 factor (difference between catchments) and the least important was the month factor (the
323 difference between months, i.e. the seasonal cycle). However, once covariates were included,
324 differences in the suspended sediment concentrations explained most of the differences
325 between factor levels. The *post hoc* comparisons for the year and month factors showed the
326 PMM time series was dominated by an increase in PMM concentration from the early 1990s
327 (Figure 2). When covariates were included, then the apparent trend in the time series is
328 largely suppressed although *post hoc* testing shows that the years 1992 through 1995 were
329 still significantly higher than years before or after. Towards the end of the study period and
330 certainly after 1992 there is some suggestion that sampling targeted places and times of high
331 PMM concentration, so the analysis including covariates provides a more realistic picture of
332 underlying trends as the ANCOVA takes account of flow conditions. The average seasonal
333 cycle in the PMM concentrations, once covariates have been accounted for and so
334 independent of flow change, shows a clear annual minimum in January (3.8 mg/l) with a
335 maximum in July (4.4 mg/l).
336 When percentage PMM content was considered by ANOVA, all factors and one
337 interaction were found to be significant at $p < 0.05$ (Table 2). Including covariates, the
338 percentage variation explained by the covariates was smaller than that observed for the PMM
339 concentration. With the inclusion of covariates the variation due to the month factor and the
340 interaction between year and month factors were no longer significant, i.e. the covariates
341 explained the importance of this factor and interaction. The annual trend in percentage PMM
342 content shows that the proportion of PMM dramatically rose after 1991, although once
343 covariates were included, then the dramatic rise is almost reversed by 2009. The changes
344 observed in both the PMM concentration and the percentage PMM content in the first half of

345 the 1990s occur with the introduction of the Urban Waste Water Directive in 1991 (European
346 Commission, 1991).

347 The Directive required a minimum of secondary treatment for all sewage treatment
348 works with a population equivalent greater than 2000 people and tertiary treatment for works
349 with a population equivalent greater than 10000 people. Therefore, the Directive aimed to
350 remove suspended solids from sewage work outflows. If the PMM concentration and
351 percentage PMM are considered after accounting for covariates, the change in POM
352 concentration can be considered (Figure 3) which shows there was a significant change in
353 average POM concentration around 1992: prior to 1992 mean annual POM concentration was
354 4.7 mg/l, while after 1992 it was 2.8 mg/l. However, this apparent success of the Urban
355 Wastewater Directive may not be reflected in the fluxes of organic matter, as flows from
356 sewage outfalls are more likely to be dominant at lower flows.

357 The ANOVA suggests a simple multiple regression for the PMM concentration could
358 be derived:

359

360 $$ln[PMM] = \underset{(0.01)}{0.03lnflow} + \underset{(0.002)}{1.073ln[sedt]} + \underset{(0.0002)}{0.0083Year} + \underset{(0.003)}{0.023\sin\left(\frac{m\pi}{6}\right)} + \underset{(0.003)}{0.044\cos\left(\frac{m\pi}{6}\right)} - \underset{(0.5)}{17.4}$$

361

362 $R^2 = 0.91$ n= 35489 (iii)

363

364 where: [X] = concentration of X (mg/l, with PMM = PMM concentration, and [sedt] =
365 concentration of suspended sediment); flow = instantaneous river discharge (m^3/s); Year =
366 calendar year; m = month of the year with 1 = January and 12 = December. Only those
367 variables significantly different from zero at $p < 0.05$ are shown. Standard errors for each
368 coefficient are given in brackets. Equation (iii) suggests a significant increase in the PMM
369 concentration over the period of record with PMM concentration increasing on higher flows

and, not surprisingly, the PMM concentration increases as suspended sediment concentration increases.

Similarly for the percentage PMM content the best-fit equation was:

$$ln[\%PMM] = 0.014lnflow + 0.035ln[sedt] + 0.004Year + 0.01\sin\left(\frac{m\pi}{6}\right) + 0.025\cos\left(\frac{m\pi}{6}\right) - 8.2$$

(0.0007) (0.0009) (0.004) (0.001) (0.001) (0.3)

$R^2 = 0.14$ n= 35489 (iv)

The format of equation (iv) is as for equation (iii) with only variables significant at 95% probability included and standard errors given in the brackets. Equation (iv) explains only 14% of the original variance in the dataset but it still highlights that the proportion of PMM in suspended sediment concentrations increased over the study period and mineral matter was more important as both suspended sediment concentration and flow increased. The relatively poor fit of Equation (iv) is most likely because the equation makes no allowance for the differences between catchments.

For the POM concentration:

$$ln[POM] = 17.9 - 0.018lnflow - 0.794ln[sedt] - 0.009Year - 0.025\sin\left(\frac{m\pi}{6}\right) - 0.076\cos\left(\frac{m\pi}{6}\right)$$

(0.7) (0.002) (0.001) (0.0003) (0.004) (0.004)

$R^2 = 0.74$ n= 35489 (v)

Only those variables significantly different from zero at $p < 0.05$ are shown. Standard errors for each coefficient are given in brackets. Equation (v) confirms a decline in particulate organic matter concentration since 1974 ($p<0.05$) and that organic matter concentration decreases both with increasing suspended sediment concentration and with increasing river

393 flow. It might be considered that equation (v) should be similar to equation (iii) as [POM]
394 was calculated from [PMM[; however, the differences in fit between equations (iii) and (v)
395 illustrate that the suspended sediment concentration and its composition can vary
396 independently of each other.

397 Examining the individual sites means that it was possible to assess significant
398 relationships with covariates at individual sites.. For 156 sites it was possible to assess
399 significant relationships with covariates; there was a significant relationship in POM
400 concentration in 90 of them of which in only 3 was the POM concentration significantly
401 increasing with time. For 92 study catchments there was a significant relationship with river
402 flow and for all but 14 of these the POM concentration decreased with increasing flow.

403 The spatial distribution of the average POM concentrations is shown in Figure 4.

404

405 *3.2. Mineral and organic matter flux*

406 The annual suspended sediment flux could be calculated for all 270 sites in the HMS scheme
407 but out of a possible 9472 site-year combinations over the monitoring period a flux
408 calculation was possible for 6026 site –year combinations (66%) – the suspended sediment
409 concentrations and fluxes are analysed in Worrall et al. (2013a). For the mineral matter, flux
410 could be calculated for 111 sites and 2808 site-year combinations. Dividing the 2808 site-
411 year combinations there were 62 with f > 1 per week, 1164 with f = 1 sample per1-2 weeks,
412 639 with f = 1 sample per 2-3 weeks and 943 with f = 1 sample per 3 weeks - 1 month.

413 On the basis of the Anderson-Darling test, the flow-weighted annual PMM fluxes
414 were log-transformed before ANCOVA. Both the sampling frequency and the water yield
415 (defined above as the total discharge from a study catchment in a year – ΣQ) were found to
416 be significant although they collectively only explained 11.5% of the original variance as no
417 allowance was made for differences between catchments or changes over time. The *post hoc*

418 tests showed that within the sampling frequency factor there were significant differences
419 between all the levels of sample frequency. Given the *post hoc* differences, the average for
420 each class of sample frequency was compared to that for sampling frequency of less than 1
421 week to derive a correction factor such that samples from each class of sampling frequency
422 could be bias-adjusted to the equivalent flux expected from a sample frequency of less than 1
423 week. The derived correction factors were: f > 1 sample per week = 1.00, 1 week > f ≥ 2
424 weeks = 1.15, 2 > f ≤≥ 3 weeks = 1.26, and 3 weeks > f ≤≥ 1 per month = 1.31. The result
425 suggests that for sampling frequencies of 12 per year (f > 1 month) the uncorrected result
426 would be 69% of the true value. For phosphate, Cassidy and Jordan (2011) suggested for one
427 site that monthly sampling (f = 12 per year) was 40% of the true value and Worrall et al.
428 (2013b) found that for DOC monthly sampling gave a flux estimate was 48% of the true
429 value. All the site-year combinations were then corrected according to the sampling
430 frequency in each year.

431 The median POM export was 6.9 tonnes/km^2/yr, with a 5th percentile of 1.9
432 tonnes/km^2/yr and a 95% percentile as 44.4 tonnes/km^2/yr while the PMM exports have a
433 median of 16.7 tonnes /km^2/yr with a 5th percentile of 3.2 tonnes/km^2/yr and a 95th percentile
434 of 150.6 tonnes/km^2/yr. In comparison, the suspended sediment exports have a median of
435 22.2 tonnes /km^2/yr with a 5th percentile of 5.4 tonnes/km^2/yr and a 95th percentile of 107.7
436 tonnes/km^2/yr. The distribution of the POM flux as a proportion of suspended sediment flux
437 for each of the study catchments is given in Figure 5. Given the values of the carbon content
438 and C/N ratios outlined above, it is possible to estimate the range of POC flux of 3.2 tonnes
439 C/km^2/yr (with the range between the 5th and 95th percentile of 0.9 to 22.2 tonnes C/km^2/yr)
440 and for PON of 0.4 tonnes C/km^2/yr (with the range between the 5th and 95th percentile of 0.1
441 to 2.6 tonnes N/km^2/yr). For the UK, POC fluxes are commonly reported for peat-covered
442 catchments where the extent of degradation and vegetation cover control the loss of POC and

443 fluxes can be as high as 195 tonnes C/km^2/yr (Evans et al., 2006) but as low as 3.4 tonnes
444 C/km^2/yr (Worrall et al., 2011). Elsewhere in the world, Hilton et al. (2012) have proposed
445 that the steep mountain forested catchments of Taiwan are a hot-spot for POC production and
446 export to the continental shelf with average POC export of 21 tonnes C/km^2/yr for catchments
447 up to 2900 km^2. For PON, the high fluxes of particulate organic matter from organic soils is
448 reflected in PON fluxes of between 0.2 to 0.7 tonnes N/km^2/yr (Worrall et al., 2012c). Russell
449 et al. (1998) estimated PON export of between 0.06 and 0.28 tonnes N/km^2/yr for 4 UK
450 catchments up to 6,500 km^2.

451 Upscaling the POM fluxes to the national level shows the median annual UK POM
452 flux was 1596 ktonnes/yr, peaking in 1996 at 4585 ktonnes/yr with a minimum in 2003 of
453 656 ktonnes/yr; this is equivalent to an export of between 2.7 and 18.8 tonnes/km^2/yr. The
454 median flux of PMM from the UK was 8121 ktonnes/yr with a minimum of 1,543 ktonnes/yr
455 in 1978. The flux of total suspended sediment from the UK also peaked at 27,550 ktonnes/yr
456 in 1978 and had a minimum of 2,199 ktonnes/yr in 2003 (Figure 6); this is equivalent to an
457 export range of between 9.6 and 119.8 tonnes/km^2/yr. There was no significant trend with
458 time, either for the suspended sediment flux or for the POM flux and so, although
459 concentration of POM was declining (Equation (v), this has no impact on POM flux. This
460 suggests the concentration decline was predominantly at low flow where the flux is minimal,
461 indicating a decline in point, rather than diffuse sources. A one-way ANOVA was performed
462 on the national-scale POM flux where the one factor was set as before or after 1992, i.e. the
463 implementation of the Urban Wastewater Directive, with the national-scale suspended
464 sediment flux and water yield as covariates. All variables were considered both
465 untransformed and log-transformed and the result showed that there was a significant
466 difference across the 1992 boundary; the average decrease was 553 ktonnes C/yr, i.e. over the
467 18 years since the implementation of the Urban Wastewater Directive had saved; diverted as

sludge to land; or emissions to the atmosphere of 9972 ktonnes C. Note that this change is not due to a change in total river flow as that was a covariate though it might be due to change in the timing or distribution of the river flows within the year.

The proportion of the suspended sediment flux that was POM varied from 5% in 1990 to 48% in 2004 with a median value of 18%. There was a significant increase of the suspended sediment flux that is POM over time since 1974 (r^2 = 0.21, n = 36, p = 0.00) with the average annual increase in the proportion being 0.3%/yr.

Between the years 2001 and 2010 it was possible to calculate an average flux for 80 catchments for which complete land use, hydroclimatic and soil characteristics could be obtained (Figure 1), but note that during this time there was no POM concentration data measured in Scotland (excluding the 4300 km^2 of the River Tweed which is monitored as part of the NE England region). The best-fit multiple regression equation was:

$$POM_{flux} = 3827 + 6.7Orgmin + 8.1Org + 7.5Grass - 2.4Area$$

(842) (2.6) (2.2) (3.3) (1.4)

R^2 = 0.5, n=80 (vi)

where: *Orgmin* = the area of organo-mineral in the catchment (km^2); *Org* = the area of organic soils in the catchment (km^2); *Grass* = the area of grazed land within the catchment (km^2); and *Area* = the area of the catchment (km^2). Equation (vi) can be interpreted as an export coefficient model where each regression coefficient is interpreted as an export coefficient, e.g. Equation (vi) predicts that 1 km^2 of organo-mineral soil would export 6.7 ± 2.6 tonnes/km^2/yr of POM where the range denotes the coefficient's standard error. This interpretation suggests the biggest source of POM was organic soils but also suggests there is no significant flux of POM from a catchment with only arable or urban land use on mineral

soils. Equation (vi) includes a significant loss term with catchment area which implies that for every additional 1 km^2 of catchment area 2.4 tonnes/yr of POM are lost. It is this loss term with catchment area that has been used in previous studies to estimate dissolved C and N losses (Worrall et al., 2012a & b). In equation (vi) there is a non-zero y-intercept value, i.e. even at zero km^2 this equation would predict a POM flux of 3827 tonnes /yr. It is not certain what such a default flux might represent but all rivers would be expected to produce an organic particles with or without inputs from the terrestrial biosphere of the catchment.

When the export (total flux per unit area of the catchment) was considered, a significant relationship was found once two outliers were removed (the River Nant y Frendod at Llansamlet = 1540 tonnes POM/km^2; and the River Dearne at Pasture's Bridge = 626.9 tonnes POM/km^2):

$$log_{10}(POM_{export}) = 2.5 - 0.54log_{10}(Area) - 0.19log_{10}(\%Min) + 0.14log_{10}(\%Org)$$

(842) (2.6) (2.2) (3.3) (1.4)

$R^2 = 0.5$, n=80 (vii)

where: *%Min* = the percentage of the catchment that is covered by mineral soils; and *%Org* = the percentage of the catchment area that is covered by organic soils. With respect to the proportion (not percentage) of POM in the total flux the best-fit equation was:

$$log_{10}(P_{POM}) = 0.4 - 0.3log_{10}(Area)$$

(0.2) (0.06)

$R^2 = 0.24$, n=80 (viii)

514 By setting P_{POM} = 0, Equation (viii) predicts that the suspended sediment flux would be 100%
515 mineral matter for catchments smaller than 21 km^2 which is a consequence of the lack of
516 small catchments in the dataset but it also predicts the greatest decline in the proportion of
517 POM will be for catchments < 100 km^2.
518
519 **4. Discussion**
520 ***Flux of POC and PON from the UK***
521 Given that the organic carbon content of organic matter is between 45 and 50% and that the
522 average C/N ratio of suspended sediment in the UK has been found to be 8.1 ± 5.2 (n=13 –
523 Hillier (2001)) which is within the range reported by Ittekkot and Zhang (1989) for rivers
524 from across the globe of 6.2, then it is possible to estimate a time series for POC and PON
525 (Figure 7). The POC flux would now be estimated as having varied between 2431 ktonnes
526 C/yr in 1996 and 313 ktonnes C/yr in 2003. Note that when estimating budgets the study
527 provides a best estimate to 4 significant figures. The PON flux would now be estimated as
528 having varied between 24 ktonnes N/yr in 2003 and 796 konnes N/yr in 1996. Based on
529 Equation (vi) the following equations was derived:
530

$$POC_{flux} = 1722 + 3.0Orgmin + 3.6Org + 3.4Grass - 1.1Area$$

531 (379) (1.2) (1.0) (1.5) (0.6)
532 R^2 = 0.5, n=80 (ix)
533

$$PON_{flux} = 213 + 0.4Orgmin + 0.5Org + 0.4Grass - 0.13Area$$

534 (47) (0.14) (0.12) (0.18) (0.07)
535 R^2 = 0.5, n=80 (x)
536

Equations (ix and x) were used to map the export of POC and PON across Great Britain (Figures 8 and 9). The POC map of Great Britain (Figure 8) reflects the distribution of organic soils in Great Britain but the highest POC exports are predicted for the uplands of England and Wales as opposed to the Highlands of Scotland, even though the latter tends to have higher elevations, and therefore higher average slopes and higher annual rainfalls. The difference between the regions is the higher grazing intensity in the English and Welsh uplands compared to upland areas in Scotland. Using the equations calculated by Worrall et al. (2012a), it is possible to predict a map of total fluvial carbon export for the UK (Figure 8). Again this highlights the importance of the grazing on organic soils in the uplands and has exports up to 25 tonnes C/km^2/yr. Although this upper value is smaller than that reported by, for example, Evans et al., (2006), the larger reported values were for small areas of highly degraded peatlands, rather than areas typical of the English uplands. Similarly, the map of PON export was to that of fluvial export of dissolved N (Worrall et al., 2012b). PON export is much less than total fluvial N export (Figure 9): the majority of total fluvial N flux occurs from areas of organo-mineral soils under heavy grazing but does not mean that for individual N-species (e.g. nitrate) other land-uses and managements dominate. The results generated here could be used to update estimates of fluvial flux of carbon and nitrogen for the UK (Worrall et al., 2007, 2009, 2012a, b).

Fluvial flux of carbon from the UK

For POC, results suggests the mean POC flux since 1974 was 863 ktonnes C/yr a 41% increase compared to the previous estimate of 613 Ktonnes C/yr. Furthermore, Equations (ix) and (x) show that there is a loss term with increasing catchment area for both POC and PON and, thus, it is not only possible to estimate the flux of both from the UK but also to calculate the loss at source, i.e. at zero catchment area. The loss of the POC in transit through the

562 catchment can now also be estimated as 264 ktonnes C/yr, which gives an average POC flux
563 at source of 1127 ktonnes C/yr.

564 Quinton et al. (2006) suggested organic carbon losses from soil erosion in England
565 and Wales were between 200 and 760 ktonnes C/yr of which 120 and 460 ktonnes C/yr was
566 delivered to streams. Rescaling the POC flux from the soil source predicted in this study to
567 the area of England and Wales suggests that on average 720 ktonnes C/yr are delivered to
568 first-order streams, i.e. greater than that predicted by Quinton et al. (2006) but within the
569 range of their erosion predictions. In addition to other estimates of the other fluvial carbon
570 components (DOC and excess diss. CO_2 – Table 3) shows that the total flux of carbon at
571 source for the UK would be 5020 ktonnes C/yr, equivalent to 21.8 tonnes C/km^2/yr, the flux
572 at the tidal limit would be 1508 ktonnes C/ yr, equivalent to 6.5 tonnes C/km^2/yr (note that
573 due to re-assessment of rounding errors and using the most up-to-date data these values differ
574 slightly from those reported in Worrall et al, 2013a – Table 4). The difference between the
575 flux at the tidal limit and that at source is not necessarily a loss of carbon to the atmosphere
576 from UK rivers as the loss may represent changes to in-stream biological production or loss
577 to long-term storage in fluvial sediments (e.g. floodplain sedimentation).

578 If all the POC loss across the study catchments is assumed to be loss to carbon
579 turnover and subsequent release to the atmosphere without loss to any stores, then the net
580 loss of C to the atmosphere from the entire catchment area would be 3512 ktonnes C/yr,
581 equivalent to 15.2 tonnes C/km^2/yr. If all the loss to the atmosphere were as CO_2, then the
582 greenhouse gas flux would be 12882 ktonnes CO_{2eq}/yr or 56 tonnes CO_{2eq}/km^2/yr. The
583 present UK greenhouse gas inventory suggests that UK GHG flux is 553 Mtonnes CO_{2eq}/yr in
584 2011 (Salisbury et al., 2013), i.e. the estimated greenhouse gas flux from UK rivers would
585 represent an additional 2.3%.

586 The turnover of POM, or POC, within the streams of a catchment is not the whole
587 impact that POC may have on the atmosphere. As POC leaves the fluvial network at the tidal
588 limit, it will continue to be processed in estuaries and coastal waters, further contributing to
589 releases to the atmosphere, before sedimenting out and being stored in marine sediments.
590 Galy et al. (2007) report very high burial efficiencies (approx. 100%) of fluvially-derived
591 carbon in the Ganges-Brahmaputra fan, which they ascribe to rapid burial, but these
592 sediments also have remarkably small POC contents (0.6×10^{12} mol C/yr from 1×10^{9} tonnes
593 of suspended sediment, equivalent to less than 1% C content – Frances-Lenard and Derry,
594 1997) and therefore the Ganges-Brahmaputra has an export equivalent to 4.4 tonnes C/km^2/yr
595 compared to the 3.5 tonnes C/km^2/yr that the UK exports at its tidal limit. Equally, the
596 estimate of 100% burial, and therefore a large carbon sink due to the Ganges-Brahmaputra
597 fan, has neglected to account for the in-stream losses of carbon from particulates before
598 reaching the sea. For other rivers Buridge (2005) suggest a removal rate from source to ocean
599 sediment of 70% based upon a measured burial efficiency in ocean sediment of 30%. Tappin
600 et al. (2003) have reported a POC budget for the Humber estuary (17% of UK's runoff drains
601 through this estuary – residence time of 2-3 months) and found that for 3 years between 1994
602 and 1996 the flux of POC from the estuary was between 16 and 43% of the fluvial POC flux
603 into the estuary and that burial rate was 4% of inputs with the remainder (between 36 and
604 54%) of the fluvial POC flux input to the estuary being mineralised. Extrapolating for the
605 remaining UK runoff, then there is between 58 and 72% in-stream removal of POC discharge
606 from UK estuaries to the shelf seas. If there is no further mineralisation in shelf seas, this
607 suggests that a 30% burial efficiency is a conservative estimate for POC from the UK, i.e.
608 338 ktonnes C/yr.
609
610 ***Implications for soil erosion as a carbon sink***

611 At the soil source, POC flux studies such as Stallard (1998) have argued that the erosion of
612 POC from soils constitutes a carbon sink since the eroded soil organic carbon lost to POC is
613 replaced and the eroded POC itself is stored by burial. Although later studies (e.g. Van Oost
614 et al., 2007) showed that between 53 and 95% of POC was retained and buried within a
615 catchment, and only 26% of this replaced, this still suggests a net carbon sink from global soil
616 erosion of 120 Mtonnes C/yr. However, as noted by Van Oost et al. (2007), no allowance was
617 made for in-stream loss once out of the immediate catchment area, or for the burial efficiency
618 in marine waters. Van Oost et al. (2007) report between 470 and 610 Mtonnes C/yr were lost
619 due to soil erosion of which between 240 and 570 Mtonnes C/yr was retained in the
620 immediate catchment, which means between 30 and 220 Mtonnes C/yr were exported to
621 streams. On the basis of results presented here, we would argue that 70% would be removed
622 before burial, i.e. between 20 and 150 Mtonnes C/yr would be lost to the atmosphere. Given
623 that Van Oost et al. (2007) suggest that only 26% of the eroded C is replaced (60 to 270
624 Mtonnes C/yr), then the median soil erosion-driven sink of C is more like 80 Mtonnes C/yr
625 (0.08 PgC/yr), but the range of values would be from a source of 90 Mtonnes C/yr to a sink
626 of 250 Mtonnes C/yr and thus the possible range includes the possibility that soil erosion is a
627 net source of C and not a net sink.

628 The same approach can be applied to the UK (Figure 10). Quinton et al. (2006)
629 suggested that 60% of eroded soil organic carbon (SOC) was transferred into POC in
630 catchments; based on the values of POC lost at the soil source to streams from this study
631 (1127 ktonnes C/yr), this would mean 1878 ktonnes C/yr of SOC lost in soil erosion. Van
632 Oost et al. (2007) found 26% of eroded soil organic carbon was replaced, which would mean
633 that 488 ktonnes C/yr is replaced each year and stored in the soil. This would leave 751
634 ktonnes C/yr stored as part of internal redistribution that never gets to the stream network.
635 This study would suggest that 264 ktonnes C/yr is lost in the fluvial network prior to export

636 to estuaries. In calculating values in Table 3 this study assumed that all in-stream loss of POC
637 was loss to the atmosphere and not losses to increased stream biota or losses to long-term
638 sedimentary storage: this would mean that 264 ktonnes C/yr is lost to the atmosphere. Given
639 the percentage losses estimated by Tappin et al. (2003) for the Humber estuary, then from an
640 average input of 863 ktonnes C/yr as POC leaving UK rivers at the tidal limit and entering
641 UK estuaries, the estuaries would store 34 ktonnes C/yr as permanent burial, 389 ktonnes
642 C/yr would be mineralised, and 440 ktonnes C/yr would be exported to the shelf seas of the
643 UK. Given Buridge's (2005) value of 30% burial efficiency of sediment from source to burial
644 in shelf seas, this would mean that 338 ktonnes C/yr are permanently buried and would leave
645 a further 102 ktonnes C/yr to be mineralised. Using the convention that a negative value in a
646 carbon budget represents a net sink to the soil, we conclude that the carbon balance due to
647 soil erosion in the UK is between +3 and +267 ktonnes C/yr, i.e. most likely to be a source
648 rather than a sink, even if all of the net in-stream loss of POC estimated in this study were
649 losses to burial in fluvial sedimentation or changes in biological production.
650 The trade-off between replacement within catchment, transfer to the stream network
651 and then loss in stream can be simply expressed as:
652
653 $$replacement\ rate \geq delivery\ rate \times (1 - burial\ efficiency)$$ (xi)
654
655 where: *replacement rate* = the proportion of the soil erosion of soil organic carbon that is
656 replaced (e.g. 0.26 – Van Oost et al., 2007); *delivery rate* = the proportion of eroded soil
657 organic carbon delivered to the stream network (e.g. 0.6 - Quinton et al., 2006); and *burial*
658 *efficiency* = the proportion of POC flux that is buried in marine sediment (0.58 – 0.72 – this
659 study).
660

661 ***The fluvial flux of Nitrogen from the UK***

662 Similarly, the results here can be used to improve previous estimates of nitrogen flux and add
663 an estimate of in-stream PON loss (Table 4). The new estimate, based on fluxes and turnover
664 rates predicted here and those produced from Worrall et al. (2012b), is that the UK is losing
665 814 ktonnes N/yr from the river network to estuaries (equivalent to 3.5 tonnes $N/km^2/yr$),
666 while at the soil source the flux was 2209 ktonnes N/yr (equivalent to 9.6 tonnes $N/km^2/yr$).
667 Previous estimates of the net loss of dissolved nitrogen (DON, nitrate, nitrite and ammonium)
668 could not discount the fact that loss of dissolved N was due to immobilisation of N into
669 particulates. Since net PON flux is now included, and since it was assumed that there was no
670 net change in immobilisation, the net catchment loss of N would be to the atmosphere.
671 Average net loss to the atmosphere across UK catchments is estimated to be 1395 ktonnes
672 N/yr (equivalent to 5.7 tonnes $N/km^2/yr$). The overwhelming majority of this loss will be as
673 N_2 but Baulch et al. (2011**)** found a consistent N_2O yield of 0.75% across 72 watersheds in
674 the US which would mean for the UK an N_2O flux from the river network of 10.5 ktonnes
675 N/yr, equivalent to 33 ktonnes N_2O/yr. The present estimate of N_2O flux from UK rivers is 24
676 ktonnes N_2O/yr, based on IPCC guidelines that the N_2O yield would be 2.5% of leached N,
677 where leached N is calculated as 30% of applied fertiliser and manure N when runoff is
678 greater than 50% of pan evaporation (IPCC, 2007) It was not possible to use the same runoff
679 criterion for this study as pan evaporations are not known, but total dissolved N at source was
680 1996 ktonnes N/yr (the present value used is 616 ktonnes N/yr) of which 2.5% would be 49.5
681 ktonnes N/yr, equivalent to 154 ktonnes (N_2O)/yr. The greenhouse gas warming potential of
682 this would be 45892 ktonnes CO_{2eq}/yr over a 100-year window – an 8% addition to the
683 current UK greenhouse gas inventory.

684

685 ***Alternative POM loss mechansims***

686 This study assumes that the difference in POM flux with catchment size is due to loss of
687 organic matter and in Figure 10 a question mark has been left against the amount of POM
688 that is retained within the fluvial network, i.e. as permanent burial within fluvial sediments.
689 Other than due to turnover to the atmosphere, the change in POM flux across a catchment
690 could be due to fractionation during the in-catchment sedimentation processes which could
691 preferentially remove the highly organic sediment compared to more mineral-rich sediment.
692 Alternatively, the sediment sources could change through a catchment with mineral-rich
693 sources being more prevalent with increasing catchment size. Slopes lessen with increasing
694 catchment size and land use changes, but the approach here has explicitly accounted for
695 changes in suspended sediment sources with both soil and land use.

696 In the UK, mineral soils and arable land are more common near the tidal limits, and
697 UK uplands are more typically dominated by organic and organo-mineral soils on grazed
698 land but, by including all of these in the analysis, such variations have been accounted for.
699 Also, the maximum altitude of each catchment was considered, so changes in source are
700 accounted for, although it is possible sources may vary with changing slope and the absence
701 of a slope factor may explain the relatively low R^2 of equation (iv). Alternatively, the POM
702 could enter permanent burial within the fluvial network. Walling et al. (2002) noted that in-
703 channel storage was between 2 and 5% of the catchment outlet flux and, further, Walling et
704 al. (1999) estimated the rate of contemporary overbank sedimentation was between 39 and
705 40% of catchment outlet flux. However, overbank sedimentation does not necessarily
706 represent a permanent burial of carbon and POM stored in floodplain sediment might still be
707 lost to the atmosphere. Erkens (2009) gives long-term, Holocene accumulation rates of total
708 sediment in the Rhine floodplain as 27% of the upstream input, but this was not a measure of
709 the organic carbon storage. Hoffman et al. (2009) suggest that the long-term storage of
710 carbon on the Rhine floodplain is equivalent to the downstream flux of POC at the catchment

711 outlet. In contrast, Gomez et al. (2003) have found only 4% POC storage in a New Zealand
712 floodplain. Given the nature of the monitoring used in this study, the likelihood that times of
713 overbank sedimentation have been captured is very small and as such the loss POM from the
714 fluvial network estimated here is unlikely to include overbank sedimentation and more likely
715 to represent the in-channel sedimentation and turnover to the atmosphere. Therefore, this
716 study has left the value of in-stream mineralisation versus long-term fluvial basin storage as
717 unknown and has taken the precautionary and conservative approach of leaving the value of
718 POC loss to mineralisation as that estimated in this study, i.e. 264 ktonnes C/yr.

719 The decline in suspended sediment export (often referred to as sediment yield in the
720 literature) through a catchment has often been expressed as a sediment delivery ratio (e.g.
721 Walling, 1983). Most contemporary sediment budgets do not include a component of carbon
722 turnover; for example, of the 11 sediment budgets given in Walling and Collins (2008), not
723 one includes a component of loss to the atmosphere. Given the average sediment delivery
724 ratio for the UK is 10%, and suspended sediment flux at the tidal limit as calculated by
725 Worrall et al. (2013a), then the amount of suspended sediment that would have to be stored
726 would be of the order of 90%, or 84 Mtonnes/yr. This decline in sediment yield has also been
727 associated with storage of suspended sediment in channel and on to floodplains - Collins and
728 Walling (2007) measured this for a UK stream as between 18 and 57% of the suspended flux
729 at the catchment outlet for two lowland streams in the UK but noted that most of this storage
730 was transient. Of course, each alternative mechanism for the reduction in POM flux with
731 catchment requires that POM is preferentially removed, but low-density POM may be
732 preferentially lost in low-slope catchments and it is known that organic particles may be
733 preferentially eroded from soils (Quinton et al., 2006). Furthermore, if fractionation of POM
734 into long-term fluvial sediments is invoked to provide an alternative to explain loss of POM
735 across catchments as turnover to the atmosphere, then it must also be assumed that, once

stored in channel or overbank sediments, it is not turned over to the atmosphere there or conversely stimulates additional carbon storage from additional biomass in those stores. A number of laboratory studies have considered the turnover of POC in streams and Moody et al. (2013) found between 38 and 87% removal of peat-derived POC over a 10-day period, i.e. a measured removal rate larger than estimated from this study.

5. Conclusions

This study has estimated that :

i) The median POC content of suspended sediment at the tidal limit between 1974 and 2010 was 15.8%, giving a median concentration of 2.2 mg C/l ;

ii) The median POC flux at the tidal limit (between 1974 to 2010) was 3.3 tonnes C/km^2/yr. The estimated average POC flux from the UK at the tidal limit was between 312 and 2178 ktonnes C/yr, with the proportion of POC to suspended sediment flux ranging from 2.4 to 22.8%;

iii) Between1974 and 2010 there was no significant trend in the POC flux but the proportion of the suspended sediment flux that is POC has significantly increased by 0.15% per year;

iv) Despite a lack of significant long-term trend in concentration or flux of POM, there was a significant decline in concentration and flux of POM after implementation of the Urban Wastewater Directive in 1992;

v) The study has shown net watershed losses equivalent to loss of 1.1 tonnes C/km^2/yr for POC and equivalent to 0.22 tonnes N/km^2/yr for PON;

vi) This study estimates that for total nitrogen species UK rivers are gaining 2209 ktonnes N/yr (9.6 tonnes N/km^2/yr) from the terrestrial biosphere but are losing 63% of this nitrogen by the tidal limit.

vii) The total flux of carbon to UK rivers from the terrestrial biosphere is estimated to be 5020 Ktonnes C/yr (21.8 tonnes C/km^2/yr) and the net catchment loss is 70%.

viii) The inclusion of the net catchment loss of POM means that it is unlikely that soil erosion leads to a net carbon accumulation; soil is likely to be a net source rather than a net sink of carbon therefore.

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969 Figure 1. Location of monitoring points for which: a) suspended sediment export could be
970 calculated for the period 1974-2010; and b) Location of monitoring points for which a PMM
971 and POM export could be calculated for the period 2001-2010. Both maps are separated by
972 the regions used for area-weighted averaging of fluxes.
973
974 Figure 2. The annual least mean square PMM and POM concentrations from 1974 to 2010
975 with and without consideration of the covariates.
976
977 Figure 3. The annual least mean square PMM proportions from 1974 to 2010 with and
978 without consideration of the covariates.
979
980 Figure 4. The distribution of the average POM concentration for the country over the period
981 1974 - 2010.
982
983 Figure 5. The distribution of the proportion of POM flux within the suspended sediment flux
984 across the country for the years 2001 to 2010.
985
986 Figure 6. The time series of the national flux of POM in comparison to the national flux of
987 suspended sediment.
988
989 Figure 7. The time series of the national flux of POC and PON in comparison to the national
990 flux of POM.
991
992 Figure 8. The projected map of: a) POC export at source; and b) total fluvial export of C at
993 source.

Figure 9. The projected map of: a) PON export at source; and b) total fluvial export of N at source.

Figure 10. Schematic carbon budget for POM from the UK.

1001 Table 1. The distribution and spatial coverage of catchments where POM flux could

1002 calculated. Regions refers to those illustrated in Figure 1.

Region	No. of study catchments	Area of region (km^2)	Area of study catchments (km^2)	Percentage of total area sampled
NW England	23	14165	9139	64.5
NE England[2]	11	13322	11975	89.9
Trent Basin	15	21600	18328	84.9
Ouse Basin	13	14362	4314	30.0
East Anglia	21	26816	10613	39.6
Thames Basin	1	12900	150	1.2
SE England	6	10979	1825	16.6
Hampshire Basin	6	6422	3268	50.9
SW England	6	14298	1496	10.5
Wales	9	20779	2238	10.8
Scotland[3]	0	74087	0	0.0
N Ireland	0	13843	0	0.0
Total	111	243564	63346	26.0

1003

[2] The NE England includes 4300 km^2 of the River Tweed which is in Scotland but which has a tidal limit in England.

[3] The values for Scotland exclude 4300 km^2 of the River Tweed which is within the country of Scotland but discharges to the sea in England.

1004 Table 2. The proportion of the variance explained by factors, interactions and covariates for

1005 the ANOVA of PMM content and % PMM content.

Factor (covariate)	PMM content		%PMM content	
	Without covariates	With covariates	Without covariates	With covariates
ln(flow)		0.2		1.3
ln(sedt)		95.3		3.5
Year	11.5	0.8	19.7	5.8
Month	1.7	0.0	1.4	0
Site	63.5	2.5	54.4	19.5
Year*Month	13.8	0.1	8.1	0
Error	9.5	1.0	16.4	69.9

1006

1007

1008 Table 3. The summary of the fluvial carbon fluxes for the UK.

Carbon pathway	Flux (ktonnes C/yr)			Export (tonnes C/km^2/yr)
	Worrall et al. (2007)	Worrall et al. (2012)	This study	
POC	614	614	863	3.5
POC loss	0	0	264	1.1
DOC	856	909	909	3.7
DOC loss	403	2650	2650	10.9
Excess CO_2	598	598	598	2.5
Total loss at source	2472	4770	5020	21.8
Total loss to atmosphere	1002	3248	3512	15.2

1009

1010

1011 Table 4. The summary of the fluvial nitrogen fluxes for the UK.

Carbon pathway	Flux (ktonnes C/yr)			Export (tonnes N/km2/yr)
	Worrall et al. (2009)	Worrall et al. (2012)	This study	
PON	72	72	181	0.7
PON loss	0	0	55	0.2
DON	160	105	105	0.4
DON loss		99	99	0.4
Nitrate	517	402	402	1.6
Nitrate loss		1175	1175	4.8
Ammonia	27	120	120	0.5
Ammonia loss		65	65	0.3
Nitrite	7	7	7	0.04
Nitrite loss	0	0	0	
Total loss at source		2165	2209	9.1
Total loss to atmosphere		1421	1395	5.7

1012

1013