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# The fluvial flux of particulate organic matter from the UK: the emission factor of soil erosion.

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## Abstract

Soil erosion has been identified as a potential global carbon sink since eroded organic matter is replaced at source and eroded material is readily buried. However, this argument has relied on poor estimates of the total fate of in-transit particulates and could erroneously imply soil erosion could be encouraged to generate carbon stores. These previous estimates have not considered that organic matter can also be released to the atmosphere as a range of greenhouse gases, not only $CO_2$, but also the more powerful greenhouse gases $CH_4$ and $N_2O$. As soil carbon lost by erosion is only replaced by uptake of $CO_2$, this could represent a considerable imbalance in greenhouse gas warming potential, even if it is not significant in terms of overall carbon flux. This work therefore considers the flux of particulate organic matter through UK rivers with respect to both carbon fluxes and greenhouse gas emissions. The results show that, although emissions to the atmosphere are dominated by $CO_2$, there are also considerable fluxes of $CH_4$ and $N_2O$. The results suggest that soil erosion is a net source of greenhouse gases with median emission factors of 5.5, 4.4 and 0.3 tonnes $CO_{2eq}$/yr for 1 tonne of fluvial carbon, gross carbon erosion and gross soil erosion, respectively. This study concludes that gross soil erosion would therefore only be a net sink of both carbon and

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greenhouse gases if all the following criteria are met: the gross soil erosion rate were very low (< 91 tonnes/km$^2$/yr); the eroded carbon were completely replaced by new soil organic matter; and if less than half of the gross erosion made it into the stream network. By establishing the emission factor for soil erosion, it becomes possible to properly account for the benefits of good soil management in minimising losses of greenhouse gases to the atmosphere as a by-product of soil erosion.

**Keywords:** particulate organic carbon, particulate organic nitrogen, soil erosion, greenhouse gas emissions.

## 1. Introduction

It has been argued that the erosion of particulate organic carbon (POC) from soils constitutes a global carbon sink because the eroded soil organic carbon lost to POC is replaced whilst the eroded POC is stored by downstream burial (Stallard, 1998; Harden et al., 1999; Smith et al., 2005). On this premise, Van Oost et al. (2007) suggested global agricultural land was a net carbon sink of 120 Mtonnes C/yr, based on a soil erosion rate of between 470 and 610 Mtonnes C/yr. However, Van Oost et al. (2007) explicitly stated that their method made no allowance for in-stream loss of the POC to atmosphere once out of the immediate catchment area, or for the burial efficiency in marine waters; in effect, they assumed that, once outside of the immediate source area, the POC would be buried into a long-term store (e.g. alluvium). Van Oost et al. (2007) reported between 470 and 610 Mtonnes C/yr were lost globally due to soil erosion of which between 240 and 570 Mtonnes C/yr was retained in the immediate catchment, which meant between 30 and 220 Mtonnes C/yr were exported to streams and assumed to be buried. For the UK, Quinton et al. (2006) suggested that the amount of carbon stored due to soil erosion from agricultural land was up to 0.75 Mtonnes C/yr (3.1 tonnes

C/km$^2$/yr) based on a POC flux to the fluvial network equivalent to between 0.8 and 2.9 tonnes C/km$^2$/yr. Whether just soil erosion from agricultural land is considered or erosion of organic particles as a whole, the basic argument remains the same: while eroded carbon particles are transported to permanent burial, the lost carbon is replaced at source from the atmosphere.

The potential for soil erosion to be a net carbon sink arises from the balance between replacement of the eroded soil and the burial of the eroded material. Soil erosion can expose fresh mineral surfaces that have a greater capacity to stabilise accumulated organic carbon than the original uneroded soil surface (Quinton et al., 2010). Deposition of the eroded particles can rapidly bury organic matter and so protect it from decomposition (Berhe et al., 2007). Eroded mineral particles can adsorb and protect dissolved organic matter upon entering rivers (Aufdenkampe et al., 2011). However, these processes are countered by enhanced decomposition of organic matter as aggregates are broken up in the erosion process (Alewell et al., 2009) and by reduced fertility and primary production at the erosion site (Lal et al., 2003). Discussion has tended to focus on in-field processes (e.g. Ni et al., 2012) and not the fate of organic matter once it has entered the fluvial system. It must always be remembered that not all particular organic matter entering the fluvial network will come from soil erosion from agricultural land and may come, for example, from bank erosion or landslides. Hilton et al. (2015) estimated 15% weathering of particulate fossil carbon, not derived from soil particles, across catchments in Taiwan. Hoffmann et al. (2013a) cited the loss rates reported by Tranvik et al. (2009) in their discussion of whether soil erosion is a carbon sink but did not then use those values to conclude whether soil erosion was a C sink or not. Equally, Aufdenkampe et al. (2011) discussed the outgassing of $CO_2$ and the potential for soil erosion to sequester carbon but did not explicitly consider the turnover of POM to $CO_2$. If there is loss of the carbon to the atmosphere while the particles, whatever their source, are in transit from source to burial, then this limits the carbon store that soil erosion could represent. A knowledge of in-stream particulate organic matter (POM) turnover is thus vital to determine whether soil erosion is a net source or a net sink of carbon.

The existing approach used to consider atmospheric impacts of soil erosion is potentially misleading: it is common to consider the POC and the carbon component of eroded particles as they are transported through the catchment to shelf seas, but the transported organic carbon is only one component of the total organic matter. Naturally-occurring organic matter is generally only 50% carbon, so an assessment of only POC can underestimate the importance of organic matter because it ignores organic oxygen and nitrogen. The oxygen content of the organic matter controls its oxidation state which, in turn, controls its impact on atmospheric $CO_2$ (Worrall et al., 2013). The N content represents an important nutrient source to river biota and so can drive organic matter turnover and, perhaps of more consequence, the N could be released as the powerful greenhouse gas (GHG) $N_2O$. Not only does the organic matter consist of elements other than C that play a role in its turnover, but also the form of release, or species, can vary: nitrogen could be released as $N_2$ or $N_2O$, and the carbon can be released as either $CO_2$ or $CH_4$, the latter, like $N_2O$, being the more powerful greenhouse gas than $CO_2$ (Houghton et al. 1995).

In many studies there is a tacit assumption that a discussion of carbon sinks and sources is equivalent to discussion of the overall impact upon atmosphere (e.g. Hoffmann et al., 2013a), but the real impact on the atmosphere comes only from understanding the greenhouse gas fluxes; the carbon budget and the greenhouse gas budget are not the same thing. To understand the greenhouse gas budget, the form of release is important, not just a consideration of carbon but to understand the greenhouse gas warming potential from eroded organic matter; we must therefore distinguish between $CO_2$, $CH_4$ and $N_2O$. Unless the whole greenhouse gas budget is considered, it is possible to reach the potentially misleading conclusion that, if erosion of organic matter can represent a carbon sink, then soil erosion should be encouraged.

An alternative approach would be to consider the flux of carbon from rivers to estuaries and shelf seas rather than starting from fluxes of suspended sediment. Meybeck (1993) estimated that the carbon flux (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 DOC:POC:DIC as 37:18:45, respectively, 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 flux of 200 Mtonnes C/yr. These figures provided useful estimates of fluvial POC losses from the land to the oceans at the tidal limit, but they did not account for in-stream losses along the length of the river, between the carbon sources (e.g. soils) and the ocean. Battin et al. (2008) used a value of 180 Mtonnes C/yr for the global flux of POC from rivers to oceans based on values from Cauwet (2002). Bauer et al. (2013) suggested that the flux of DOC+POC from rivers to shelf seas (including estuaries) was 450 Mtonnes C/yr, based on values from Syvitski et al. (2005). Syvitski et al. (2005) suggested that pre-human POC fluxes were between 140 and 470 Mtonnes C/yr decreasing to between 126 and 380 Mtonnes C/yr in modern times with the difference being the role of reservoir storage outstripping the influence of increased soil erosion.

For the UK, a more detailed analysis of carbon fluxes through and from rivers has been possible. Worrall et al. (2007) used nationally-collected monitoring data to assess national-scale loss at the tidal limit, the loss from the terrestrial biosphere and the loss from rivers to the atmosphere. They found that in total 10.1 tonnes C/km$^2$/yr were lost from the terrestrial biosphere to rivers of which 2.5 tonnes C/km$^2$/yr was as POC. However, the study considered in-stream losses only for DOC and DIC but did not include POC. Worrall et al. (2014) updated the POC flux estimates and found that POC loss from terrestrial sources (all terrestrial sources, not just soil erosion) was 4.6 tonnes C/km$^2$/yr with the equivalent of 1.1 tonnes C/km$^2$/yr being lost in-stream.

The trade-off between replacement within catchment, transfer to the stream network, loss in stream, and eventual, permanent burial can be simply expressed as:

$$P_{replace} \geq P_{deliver}(1 - P_{burial}) \qquad \text{(i)}$$

where: $P_{replace}$ = the proportion of organic carbon that is replaced following erosion; $P_{deliver}$ = the proportion of eroded organic carbon delivered to the stream network; and $P_{burial}$ = the proportion of POC flux that is buried in marine sediment or other permanent burial. Note that Equation (i) applies for any eroded particulate organic carbon and not just from soils and not just from agricultural land. The term (1 – $P_{burial}$) represents the loss to the atmosphere in transit. However, this equation does not hold for greenhouse gases because replacement will be as $CO_2$ but loss within stream could be as $CO_2$, $CH_4$ or $N_2O$. Therefore, Equation (i) should be re-written as:

$$K_{CO2}P_{replace} \geq P_{deliver}(K_{CO2}P_{CO2} + K_{CH4}P_{CH4} + K_{N2O}P_{N2O}) \qquad \text{(ii)}$$

$$1 - P_{buried} = (K_{CO2}P_{CO2} + K_{CH4}P_{CH4} + K_{N2O}P_{N2O}) \qquad \text{(iii)}$$

where: $K_x$ = the greenhouse gas warming potential of x where x is $CO_2$, $CH_4$ or $N_2O$; and $P_y$ = the proportion of loss of particulate organic matter that is lost as y, with y as $CO_2$, $CH_4$ or $N_2O$.

The aim of this study is to assess the complete greenhouse gas impact of soil erosion of organic matter from erosion to burial and so derive a more complete emission factor for this process. If a realistic emission factor for soil erosion can be derived, then the benefit of preventing soil erosion can be better understood.

## 2. Approach

This study considers the fluvial system as outlined in Figure 1, whereby eroded organic matter (gross erosion) can be deposited within the terrestrial source area or delivered to the stream channel (net erosion). The organic matter removed in gross erosion can (eventually) be replaced. Replacement can be by a number of processes (as outlined above) but in terms of greenhouse gases, the carbon is drawn down from the atmosphere as $CO_2$ via photosynthesis to primary production. Primary production does not draw down carbon from the atmosphere as $CH_4$ or draw down nitrogen as $N_2O$. The amount of POM that reaches the stream network is referred to as the net erosion and will be less than the gross erosion due to the internal (on-slope) redistribution. Several studies have suggested that on-slope storage of gross erosion deposited within, for example, a field can prevent mineralisation of the deposited organic matter by a number of processes (e.g. rapid burial - Berhe et al. 2007).

On entering the stream network, POM is subject to a number of physical and biogeochemical removal processes. POM can be mineralised within the stream network to all three of the greenhouse gases ($CO_2$, $CH_4$ and $N_2O$). POM can also be permanently buried by in-channel storage or by overbank sedimentation on to floodplains. It is assumed that POM inputs to estuaries and oceans will also be prone to mineralisation to $CO_2$, $CH_4$ and $N_2O$ or to permanent burial.

The processes affecting each step presented in Figure 1 are quantified using data from the literature, previous results of the authors' own work and newly available data from UK monitoring programmes. For each important process and storage compartment, the range was estimated. It was not always possible to give a consistent measure of uncertainty for each input variable or parameter: for some measures a range was not available while for others the distribution was known such that a mean and standard deviation could be calculated. The expected value and measure of variation were detailed and the total carbon budget and total greenhouse gas warming potential were calculated as a stochastic combination of the estimated ranges. When not known, a distribution was taken to be uniform.

Wherever possible, the assessment was considered over a 100-year window which in turn meant that values of $K_x$ - the greenhouse gas warming potential (GWP) of x – in Equations (ii) and (iii) were then 3.67, 24, and 292 for $CO_2$, $CH_4$ and $N_2O$ respectively. By convention, fluxes of carbon, or greenhouse gases, are considered relative to the atmosphere and therefore sinks to land are negative.

## 3. Methods

In line with the approach above this study estimated: replacement ratio; enrichment ratio; mineralisation rate of internally-redistributed sediment; the ratio of net erosion leaving the site of erosion (e.g. an agricultural field) to the stream network and the gross erosion within the site of erosion (e.g. the field) (net to gross ratio – Quinton et al., 2006); fluvial flux of POM; C/N of POM; proportion of loss as $CH_4$ and $N_2O$; estuarine removal; shelf sea removal; and, terrestrial burial efficiency.

### *3.1. Replacement rate*

A key component of the argument that terrestrial erosion represents a net carbon sink is the extent to which the eroded carbon can be replaced. Van Oost et al. (2007) took the replacement rate to be between 0.19 and 0.58 with a preferred value of 0.26, i.e. 26% of the eroded carbon is replaced. However, several studies (e.g. Quinton et al., 2006) have assumed replacement rates as high as 1. Therefore, we assumed a range of 0.19 to 1.

### *3.2. Enrichment ratio*

Gross erosion of soil has been commonly observed to be enriched in carbon relative to the bulk soil; this fractionation is expressed as an enrichment ratio – the proportion of the carbon in the eroded particle relative to that in the bulk surface soil. Quinton et al. (2006) identified enrichment ratios as between 1.2 and 4.7 for UK soils. We have assumed that enrichment ratios observed for soil erosion in agricultural soils are true for erosion for other settings.

### *3.3. Mineralisation rate of internally redistributed sediment*

Most studies have assumed this to be zero (e.g. Van Oost et al., 2007) but we believe it is unreasonable to assume that no redistributed carbon can be mineralised. Van Hemelryck et al. (2010) measured the mineralisation rate of redistributed soil particles; applying their derived equations over a 100-year window, i.e. the same window of time for which the GWP values are derived, gives a value of 0.75. In other words, over a 100-year period, 25% of the internally redistributed will be mineralised to $CO_2$ and 75% will remain. Therefore, for this study we took the range of the proportion of internally re-distributed organic matter that is stored over a 100-year window as between 0.75 to 0.95.

### *3.3. Net to gross ratio*

Gross erosion (e.g. the erosion rate within the field) will not necessarily reach the fluvial network and can instead be deposited on-slope. Thus the net erosion (that which leaves the field and enters the channel network) is less than or equal to the gross erosion. One reason for the difference between the net and the gross erosion rates is the storage of sediment within field and before reaching the stream network. Quinton et al. (2006) reviewed net to gross ratio for UK settings and gave values of between 0.47 and 0.65, i.e. at its maximum, the net erosion was 65% of the gross erosion.

### *3.4. Fluvial flux of POM*

Worrall et al. (2014) examined all 35,490 records of POM concentration from 1974 to 2010 from all 270 catchments with a mean discharge > 2 $m^3/s$ included in the UK's Harmonised Monitoring Scheme. Applying the rule that for a flux calculation within any monitored catchment there would have to be a minimum of 12 samples per year, this gave 2,808 flux estimates from 111 sites. When only the flux estimates for sites at the tidal limit of the catchment were considered and only those with data between 2001 and 2010, then the flux of POM from 80 catchments across the UK for the years 2001 to 2010 could be calculated. In-stream losses were then estimated by comparing these catchments, and allowing for differences in hydro-climatic and land-use factors. This approach is based upon net POM fluxes from these catchments and so includes all sources of POM to the fluvial network and not just from one specific source or process. The 80 catchments covered a range of landscapes in England and Wales where the soil type varied from 0 to 100% of 3 soil classes (mineral, organo-mineral and organic soils as defined by Hodgson, 1997). Land use in the 80 catchments varied from: 0.1 to 70% arable; 2 to 78% grassland; and 0.4 to 36% urban.

In-stream losses can be due to turnover of the organic matter or burial in channel or in the floodplain. To calculate the proportion of the in-stream loss due to floodplain sedimentation, it is necessary to estimate the proportion of time that the discharge in any UK river is greater than bankfull discharge, and therefore the proportion of time in which there is flow and sediment delivery to the floodplain. Nixon (1959) found that 29 English rivers were at or exceeded their bankfull discharge between 0.1 and 2.9% of the time, i.e. overbank sedimentation would be occurring between 1 day every 3 years and 11 days per year. Days of overbank sedimentation are likely to be days of high flow in the main channel and thus days of considerable sediment flux. Therefore, using the POM flux data from Worrall et al. (2014), we assumed the highest flows each year were overbank flows and that any sediment flux they carried was lost to overbank storage. The number of days of flux lost to overbank sedimentation was varied from the lowest to the highest value as measured by Nixon (1959) (0.33 to 11 days per year) with the assumption that the first day of overbank sedimentation was the day with the highest flux of POM in that year and then, for each further day of the overbank sedimentation, it was assumed that these were days of progressively lower POM flux. It was then assumed that all the POM flux on days of overbank sedimentation was lost to overbank sedimentation. The POM flux lost each year to overbank sedimentation was expressed as a percentage of the total POM flux from the catchment for that year. Because all available comparator data were for overbank storage of suspended sediment and not POM, the calculation was also performed for suspended sediment based upon flux values given by Worrall et al. (2013b). It was assumed that POM flux lost to overbank sedimentation was permanently stored and not mineralised to the atmosphere within the time frame considered. The limitations of these assumptions are discussed later on.

Collins and Walling (2007) gave values of in-channel storage as between 18% and 57% of the outlet flux for two UK lowland streams but they noted that most of this storage was transient. Indeed, Walling et al. (2002) noted that permanent in-channel storage was only between 2 and 5% of the catchment-outlet flux of suspended sediment and, given that Worrall et al. (2014) showed that on average suspended sediment in the UK was 30% POM, then for this study it is assumed that in-channel storage of was 1 – 2% of the incoming POM flux.

### *3.5. C/N of POM*

Worrall et al. (2014) estimated POC and PON from POM given that the organic carbon content of organic matter was between 45 and 50% and that the average C/N ratio of suspended sediment in the UK was 8.1 ± 5.2 (n=13: Hillier, 2001). Here, we used POM data collected as part of the LOIS project (Neal and Davies, 2003) compared to literature values (Table 1). The LOIS project collected 2,484 samples for POM across 5 years for the Humber Basin (26,109 $km^2$; 17% of the UK catchment area). Analysis of variance (ANOVA) was used to test whether C/N of POM varied from between sampling site, sampling month and sampling year. To comply with the assumptions of ANOVA, the data were tested using the Levene and the Anderson-Darling tests and transformed as required. Differences between the levels of significant factors were tested using the *post hoc* Tukey test. Data from the LOIS project were augmented and compared with data from the literature. There was no information in the LOIS data on the concentration of particulate nitrogen or particulate carbon so we followed Worrall et al. (2014) and used a range of 45 to 50%.

### *3.6. Proportion of $CH_4$ and $N_2O$*

POM lost in the fluvial network, estuaries or shelf seas can be lost as $CO_2$, $CH_4$ or as $N_2O$. Therefore, this study examined the literature (Table 2 & 3) to find ranges of the loss of $CH_4$ as a proportion of C loss in aquatic systems and likewise the loss of $N_2O$ as a proportion of N loss.

### *3.7. Estuarine removal*

Estuaries will remove sediment and with it POM. Tappin et al. (2003) reported a POC budget for the Humber estuary (17% of UK's runoff drains through this estuary with a residence time of 2-3 months) and found that for 3 years (1994 -1996) the flux of POC from the estuary varied between 16 and 43% of the fluvial POC flux into the estuary and that burial rate was 4% of input with the remainder of the fluvial POC flux input to the estuary being mineralised (between 36 and 54%). The estimates from Tappin et al. (2003) are given without further estimates of any uncertainty and the ranges given were used with the additional caveat that there must be mass balance. The proportion of organic matter buried was taken such that mass balance was met, i.e. proportion of the incoming flux that was buried may be greater than 4% given the variation in the proportions lost to mineralisation and the exported from the estuary. That is:

$$P_{burial}^{estuary} = 1 - P_{mineral}^{estuary} - P_{transfer}^{estuary} \qquad \text{(iv)}$$

where: $P_{x}^{estuary}$ = the proportion of the x with x as buried (burial), mineralised to greenhouse gases (mineral) or transiting through the estuary to the shelf (transfer).

### *3.8. Shelf-sea processes*

Galy et al. (2007) reported very high burial efficiencies (approx. 100%) of fluvially-derived carbon in the Ganges-Brahmaputra fan, which they ascribed to rapid burial, but these sediments also have remarkably small POC contents (0.6 x $10^{12}$ mol C/yr from 1 x $10^{9}$ tonnes of suspended sediment, equivalent to less than 1% C content - Frances-Lenard and Derry, 1997), and therefore the Ganges-Brahmaputra has an export equivalent to 4.4 tonnes C/km$^{2}$/yr compared to the 3.5 tonnes C/km$^{2}$/yr that the UK exports at its tidal limit. Equally, the estimate of 100% burial, and therefore a large carbon sink due to the Ganges-Brahmaputra fan, has neglected to account for the in-stream losses of carbon from particulates before reaching the sea. For other rivers, Burdige (2005) suggested a removal rate from source to ocean sediment of 70% based upon a measured burial efficiency in ocean sediment of 30%. Burdige (2005) presented no new data but quoted data from Aller (1998) who in turn quoted Canfield (1994) who included data from Middleburg (1991) and Reimer et al. (1992), but relied mostly upon data from Canfield (1989). In this study we supplement this information further with data from Meyer et al. (2007), Weijers et al. (2009), Li et al (2013) and Hung et al. (2012). The data from these sources are for the carbon sedimentation rate but not the sedimentation rate of terrestrial organic carbon. Therefore, the following sources were used to estimate the burial rate of the terrestrial organic carbon compared to the sedimentation rate in shelf seas: Burdige (2005), Weijers et al. (2009), Li et al. (2013), Hung et al. (2012) and Meyer et al. (2007). These data were compiled to relate the carbon sedimentation rates to the burial efficiency of the terrestrials organic carbon. With respect to the UK, there are several studies of the sediment and carbon budgets of the North Sea (Brockman et al., 1990; de Hass et al., 1997a, b).

## 4. The net effect of soil erosion on GHG emissions

### *4.1. Fluvial flux of POM*

The best-fit multiple regression equation reported by Worrall et al. (2014) was:

$$POM_{flux} = \underset{(842)}{3827} + \underset{(2.6)}{6.7}Orgmin + \underset{(2.2)}{8.1}Org + \underset{(3.3)}{7.5}Grass - \underset{(1.4)}{2.4}Area \qquad \text{(v)}$$

$r^2 = 0.5$, $n=80$, $p < 0.05$

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$^2$). The values in brackets represent the standard errors in the coefficients. Equation (v) can be interpreted as an export coefficient model where each regression coefficient is interpreted as an export coefficient. Thus, Equation (v) 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 is organic soil; POC fluxes are commonly reported for peat-covered catchments where the extent of degradation and vegetation cover control the loss of POC and fluxes can be as high as 195 tonnes C/$km^2$/yr (Evans et al., 2006). Most contemporary studies of soil erosion in the UK and especially those concerned with the carbon or greenhouse gas emissions have focused upon mineral soils and arable land (e.g. Rickson, 2014), but the results from this study imply there is no significant flux of POM from a catchment with only arable or urban land use on mineral soils. Most studies have focused on the production of sediments from soil erosion as opposed to the study of POM more generally (e.g. Quinton et al., 2006); Equation (v) implies that certain land uses are not that important in the contemporary flux of POM from the UK. Equation (v) has a y-intercept value which predicts that any catchment will have a minimum export of 3817 ± 842 tonnes/yr; this may relate to erosion unrelated to land use, e.g. bank erosion. Equation (v) can be applied across the UK given knowledge of land use and soil distribution (Defra, 2005, Lilly et al., 2009) to give the flux of POM from the UK at the tidal limit: the average value for the period 2001 to 2010 was 1195 ± 308 ktonnes/yr.

Equation (v) 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. Given the loss rate, the amount of POM lost in transit through UK rivers would be 594 ± 206 ktonnes/yr which in turn means a loss of POM at the soil source of 1854 ± 238 ktonnes/yr. This gives an in-stream removal rate for POM of 33.5 ± 11.2%.

Using the estimates of days of bankfull discharge and the highest daily fluxes gives an estimate that, for one day each year, the loss to floodplains of suspended sediment flux is

2.5% of the total flux leaving the catchment and therefore the maximum percentage lost to overbank sedimentation would be 27.5% (i.e. for 11 days per year of bankfull or greater discharge). The same analysis for POM flux shows that only 0.97% of the annual flux is lost per day of overbank flow, in which case after 11 days of overbank flow only 10.9% of the POM flux would be removed. Note that the percentage losses are less for POM than for suspended sediment, i.e. POM is not fractionated into overbank storage relative to suspended sediment. In the UK case, the proportion of suspended sediment that is organic matter decreases as flow increases. Walling et al. (1999) estimated overbank sedimentation for the Yorkshire Ouse as 30% of the outlet flux (23% of influent suspended sediment flux) and as 40% of the outlet flux (29% of influent flux) for the River Tweed. Erkens (2009) gave a long-term, Holocene accumulation rate of total sediment in the Rhine floodplain as 27% of the upstream input, but this was not a measure of the organic carbon storage. Hoffmann et al. (2009) suggested that the long-term storage of carbon on the Rhine floodplain was equivalent to the downstream flux of POC at the catchment outlet. In contrast, Gomez et al. (2003) found only 4% POC storage in a New Zealand floodplain.

### *4.2. C/N of POM*

Across 5 years (1994 - 1998) and 16 sites across 13 rivers (Rivers Aire, Calder, Derwent, Don, Great Ouse, Nidd, Yorkshire Ouse, Swale, Trent, Tweed, Ure, Wear and Wharfe – Robson and Neal, 1997), the median POC/PON C/N ratio was 11.5 with an interquartile range of 9.2 to 14.3. The Anderson-Darling test showed the data were log-normally distributed and so ANOVA was performed on log-transformed data; the Box-Cox transformation removed 7 out of 2,477 data. ANOVA showed that all factors were significant. By far the most important factor was the difference between months with a minimum in C/N value in June and a maximum in February; the seasonal cycle is remarkably symmetrical. The second most important factor is the difference between years, although *post hoc* testing indicated that the main difference lay between the years 1993 (13.8 ± 1) and 1995 (10.1 ± 1). The year 1995 was a drought year in the UK (1 in 33 year drought – Worrall et al. 2008). The difference between sites explained only 4.8% of the original variance and was between only 3 out of the 16 sites Calder (Methley Bridge 12.7), Swale (Catterick Bridge 13.0) and Trent (Cromwell Lock 10.1).

The review of literature data allowed an estimation of mean and standard deviation for 47 catchments (Table 1). The geometric mean of the catchments, not including any of the UK data, was 10.3. The discharge-weighted average from the review of Ittekkot and Zhang (1989) was 10.7 and for the LOIS data the geometric mean was 11.7. Therefore, applying this to UK values of POM flux, the flux of POC at the tidal limit would be 601 ± 152 ktonnes C/yr. The POC flux at the terrestrial source would be 877 ± 102 ktonnes C/yr meaning that the carbon lost through the streams of the UK would be 290 ± 96 ktonnes C/yr. Equally, the flux of PON at the tidal limit would be 52 ± 13 ktonnes N/yr. The PON flux at the soil source would be 74 ± 14 ktonnes C/yr meaning that the nitrogen lost through the streams of the UK would be 23 ± 9 ktonnes N/yr.

### *4.3. Shelf-sea processes*

Burdige (2005) suggested a figure of 30% for burial efficiency of terrestrial organic matter into permanent burial in shelf sediments; Weijers et al. (2009), Li et al. (2013), Hung et al. (2012), and Meyer et al. (2007) had values between 30 and 79% with a median value of 59% (n=13, Figure 2). Given the median value of terrigenous input of the organic sedimentation rate means that the best-fit equation is for burial efficiency ($B_{tom}^{eff}$):

$$B_{tom}^{eff} = 49 + 5.7log_e(S_{om}) \quad n = 25, \qquad r^2 = 67.4\%, \qquad p < 0.05 \qquad \text{(vi)}$$

(3.0) (0.8)

where: $S_{om}$ = the sedimentation rate of organic matter (g C/m$^2$/yr).;

Brockman et al. (1990) estimate that the POC flux to the North Sea basin is 4 Mtonnes C/yr. Haas et al. (1997a, b) give the carbon budget for the basin itself. On the North Sea plateau the accumulation is limited by scouring currents to 100 ktonnes C/yr with most accumulation in the deep channels to the east of the sea (Norwegian channel and Skaggerak) at 1 Mtonnes C/yr and another 0.1 Mtonnes C/yr exported off the shelf into the Norwegian Sea (de Haas et al., 1997a,b): this would give a burial efficiency of 30%. de Haas et al. (1997a) suggest that 20% of the accumulated sediment is of terrestrial origin and they measured a series of sedimentation rates varying from 0.05 to 0.35 g/m$^2$/yr which, given the relationship presented in Equation (vi), gives a range of burial efficiency from 31 to 43% with a 95% confidence interval of 27 to 49%. However, for the UK, the North Sea is but one shelf sea and de Hass et al (2002) note that for the Celtic Sea the sea bed is entirely made of re-worked Pleistocene sediment which meant that there was no accumulation on the shelf but there are no measurements of the export off the shelf to the ocean. Therefore, in the case of UK, figures of burial efficiency for the North Sea must be viewed as the upper range of possible values.

### *4.4. Proportion of loss as $CH_4$*

Included in the analysis is the range of values for reservoirs (Guerin et al., 2006). Of the 12 measurements that are detailed in Table 2, the range used in this study was taken as the 5th to 95th percentile range (0.64% to 2.2%) with a median value of 0.97%.

### *4.5. Proportion of loss as $N_2O$*

The ranges reported in the literature are given in Table 3. The IPCC guidelines say that the $N_2O$ yield would be 2.5% of leached N, where leached N is calculated as 30% of applied fertiliser and manure N when runoff is greater than 50% of pan evaporation (IPCC, 2007). Clearly such an approach has no application to PON. Baulch et al. (2011**)** found a consistent $N_2O$ yield of 0.75% across 72 watersheds in the US. The present estimate of $N_2O$ flux from UK rivers is 24 ktonnes $N_2O$/yr, based on IPCC guidelines.

### *4.6. Stochastic modelling of GHG emissions factor*

The ranges of the input parameters used are given in Table 4. Given the stochastic combination of the ranges developed above, the total greenhouse gas flux due to particulate erosion, flux and burial from and in the UK leads to a median emission of 3853 ktonnes $CO_{2eq}$/yr with a 5th to 95th percentile range of 2970 to 7807 ktonnes $CO_{2eq}$/yr. The median greenhouse gas flux of the replacement of -2490 ktonnes $CO_{2eq}$/yr with a 5 to 95th percentile range of -1260 to -5822 ktonnes $CO_{2eq}$/yr, which is 56.9% of the emissions (ranging between 25.3 and 137.9%). Given the input ranges, there is only a 12.8% chance that erosion in the UK is a greenhouse gas sink. When the individual greenhouse gases are considered, then $CO_2$ represents 74% of the greenhouse gas warming potential (59 to 89% - 5th to 95th percentile); $CH_4$ represents 3.8% of the greenhouse gas warming potential (2.1 to 6.5% - 5th to 95th percentile); and $N_2O$ 22% of the greenhouse gas warming potential (5 to 37% - 5th to 95th percentile). The largest loss of greenhouse gases is from the shelf seas (48%), then estuaries (30%), with the least from the rivers (19%). The distribution of the greenhouse gas fluxes and POM fluxes are summarised in Figure 3. When overall flux of greenhouse gases is compared to flux of particulate carbon from the soil, then 1 tonne of particulate carbon entering the fluvial network gives a median emission factor of 5.5 tonnes $CO_{2eq}$/tonnes C/yr with a 3.3 to 9.9 as the 5th to 95th percentile.

For 1 tonne of C released as particulate matter to the fluvial network, then it is possible to give ranges of what this may mean in terms of gross erosion. Given the ranges discussed above, 1 tonne of fluvial organic particles will have come from 0.47 to 1 tonne of gross erosion of soil carbon, which will in turn have been replaced by between 0.19 and 1 tonne C. The emission factor of 1 tonne of gross carbon erosion is then:

$$GWP_{Oe} = GWP_{Ne} - \frac{N_{Ce}}{G_{Ce}} RMK_{CO_2} \qquad \text{(vii)}$$

where: $GWP_{Ne}$ = the emission factor of net erosion of organic particles (tonnes $CO_{2eq}$/yr); ${}^{N_e}\!/_{G_e}$ = the net to gross erosion ratio; R = replacement rate; M = mineralisation rate of internally redistributed carbon particles; $K_{CO_2}$ = the greenhouse gas warming potential of $CO_2$ (3.67). Given the ranges quoted above, the median value of the sink due to replacement and internal deposition is a sink of -1.1 tonnes $CO_{2eq}$/yr with a 5th to 95th percentile range of -0.6 to -2.3 tonnes $CO_{2eq}$/yr. Given this value for the sink due to replacement and internal redistribution, then the emission factor of 1 tonne of carbon ($GWP_{Oe}$) from gross erosion would be a source of 4.4 tonnes $CO_{2eq}$/yr with a 5th to 95th percentile range 1.6 to 8.9 tonnes $CO_{2eq}$/yr.

Many authors have noted that eroded soil is enriched in organic carbon compared to topsoil; Quinton et al. (2006) gave a range of enrichment ratios of between 1.2 and 4.7 for UK soils. The majority of mineral soils in the UK have been 2 and 4% soil organic carbon for a range of land uses (Bell et al., 2010). Assuming that the supply of POM to the fluvial network is predominantly supplied from soil erosion, then 1 tonne of gross mineral erosion represents 0.09 tonnes of C in gross erosion (with a range 0.03 to 0.16 tonnes C) meaning that the emission factor for 1 tonne of gross erosion is 0.30 (5th to 95th percentile range of 0.11 to

0.66) tonnes $CO_{2eq}$/yr. Erosion rates in the UK have been reviewed by Defra (2005b) which indicated median values of net soil loss from arable fields in England as 410 tonnes/km$^2$/yr and from English grasslands as 60 tonnes/km$^2$/yr. Boardman (2013) reviewed soil erosion in Britain and gives values as high 4500 tonnes/km$^2$/yr for a bare sandy loam compared to values as low as 30 tonnes/km$^2$/yr for clay soil under cereals, i.e. limits suggested on gross erosion estimated above are very low compared to those observed in the UK..

For 1 tonne C released to the fluvial network, the gross erosion can be calculated given the range of values of ${}^{N_e}/_{G_e}$ as discussed above; the amount of GHG this removes is 3.67R. Therefore, for gross soil erosion to be a net sink of GHG the following must be true:

$$\frac{K_{CO_2}}{GWP_{Ne}} R > \left[\frac{N_{Ce}}{G_{Ce}} + \left(1 - \frac{N_{Ce}}{G_{Ce}}\right) M\right] \qquad \text{(viii)}$$

Given the ranges used in this study, for a GHG sink this can only be achieved if $R > 0.7$ with ${}^{N_e}/_{G_e} < 0.47$.

### *4.7. Stochastic modelling of GHG emissions factor*

When just carbon is considered, and not all GHGs, there is a median emission of 1099 ktonnes C/yr with a 5th to 95th percentile range of 697 to 1575 ktonnes C/yr. The median carbon flux of the replacement is 656 ktonnes C/yr with a 5th to 95th percentile range of 343 to 1587 ktonnes C/yr, which is 63.7% of the emissions (with a 5th to 95th percentile between 34 and 140%). Given the input ranges, there is only a 17.9% chance that erosion in the UK is a carbon sink.

## 5. Discussion

Given the data used in this study, it is possible to update estimates of the GHG fluxes from UK rivers. This study has not updated the carbon budget estimates for UK rivers provided by Worrall et al. (2014) but has updated the estimates of the GWP of those fluxes by considering not just the turnover of DOC and POC to $CO_2$ but the turnover of POM and DOM to $CO_2$, $CH_4$ and $N_2O$. Table 5 updates the estimates of the GWP and, based upon median values of the ranges used above, the median value of GWP of UK rivers is 14254 ktonnes $CO_{2eq}$/yr; this equates to an additional 58 tonnes $CO_{2eq}/km^2/yr$ for the UK land area lost from the fluvial network.

This estimate of the GHG loses from the UK fluvial network has been based upon the assumption that any storage in the fluvial system is a permanent sink, i.e. there is no loss of organic matter from the in-channel or floodplain storage on the time scales of decades. Hoffman et al. (213b) have shown that over the last 7500 years the floodplains of European rivers have acted as a net carbon sink because they represent an environment which has a high preservation potential for carbon and that, when this material erodes and is replaced by primary productivity, then the floodplain is still a net sink of carbon. However, even with this mechanism a proportion of POM stored on floodplains would be mineralised and returned to the atmosphere and a further proportion of the POM stored on floodplains would be lost to the atmosphere as the stored POM is eroded and returned to the fluvial network. Therefore, even if the fate of POM within fluvial storage is a net sink, this study has made the even more conservative assumption that all of the fluvial storage is acting as a sink. Secondly, it should also be noted that Hoffman et al. (2013b) show their result for carbon and not for greenhouse gases. The relative high standing water tables of floodplains mean that the proportion of organic matter lost maybe smaller but the proportion of the deposited POM lost as the more powerful greenhouse gases ($CH_4$ and $N_2O$) maybe greater (Pinay et al., 2007). Thirdly, it should be pointed out that the proportion of the POM lost as storage in the fluvial network is almost 6 times smaller than that lost directly to the atmosphere via turnover in the river and, furthermore, the greatest proportion of the loss of POM is not in the river network but in the estuaries and shelf seas. Therefore, the estimates given here are likely to be underestimates.

The estimates made in this study show the critical importance of the interplay of the net to gross erosion and the replacement of soil organic matter. In this study we have assumed a broader range of replacement rates than assumed by Van Oost et al. (2007) but there is no information on what controls replacement rates and how this relates to gross erosion. It could be argued that the ability of a soil to replace carbon lost as erosion will be akin to its rate to turnover soil organic carbon (SOC). An arable soil in southern England has a surface SOC residence time of 22 years (Jenkinson and Raynor, 1977). If a soil has between 2 and 4% SOC, then the plough layer (depth of 20 cm) of an arable field contains between 5.2 and 10.4 kg of organic carbon /m$^2$ (assuming an average bulk surface soil density of 1300 kg/m$^3$) and the organic carbon is turning over at a rate of between 0.24 and 0.48 kg C/m$^2$/yr. Given an enrichment ratio (1.2 to 4.7) and the range in %SOC (2 to 4%) used in this study, then this turnover rate would be equivalent to the amount of carbon exported in a gross erosion of between 3.9 (5$^{th}$ to 95$^{th}$ percentile range of 0.5 to 8.6) kg C/m$^2$/yr equivalent to a median gross soil erosion of 30.8 tonnes/km$^2$/yr (5$^{th}$ to 95$^{th}$ percentile between 2.7 to 91.9 tonnes/km$^2$/yr). As noted above, grassland soils in the UK would tend to have the higher %SOC and the lower gross soil erosion rates which suggests that for many mineral soils under certain land uses (e.g. grassland), the replacement rate could be close to 1 but that the replacement rate (R) would decrease rapidly for arable fields where %SOC is naturally lower and gross soil erosion is commonly higher. It should also be repeated that the evidence for the UK is that mineral soils under arable usage are not the most important sources of POM leaving the UK.

Given the values of the emissions factors estimated above, then it is possible to re-consider the impact of soil erosion at the national and global scale. Quinton et al. (2006) estimated that between 0.2 and 0.76 Mtonnes C/yr were released by soil erosion in England and Wales (rescaled to the UK this would be 0.3 to 1.2 Mtonnes C/yr) which we can now equate to 3.9 Mtonnes $CO_{2eq}$/yr (with a 5th to 95th range of 1.0 to 8.3 Mtonnes $CO_{2eq}$/yr). When added to the river loss, it suggests that, considering the total fate of carbon from soil source to burial at sea, then the equivalent loss of greenhouse gas to the atmosphere for the UK is 17254 ktonnes $CO_{2eq}$/yr (71 tonnes $CO_{2eq}/km^2/yr$).

At a global scale, Van Oost et al. (2007) estimated that between 470 and 610 Mtonnes C/yr were eroded from agricultural land worldwide and, given the emissions factors presented above, meant that global, agricultural soil erosion was a median net source of 3.5 Gtonnes $CO_{2eq}$/yr (1.4 to 8.4). Lal (2003) estimated that global soil erosion was 75 Gtonnes/yr of which he estimated that 15-20 Gtonnes were lost to rivers and that between 4 and 6 Pg C/yr of carbon were lost to the oceans with 20% mineralisation of the carbon, suggesting that 0.8 to 1.2 Gtonnes C was released to the atmosphere each year. Lal (2003) did not consider replacement or enrichment ratio and assumed a %SOC of eroded sediment in rivers of between 2 and 3% while for the UK a median value of 16% was found (Worrall et al. (2014).

When studies argue that soil erosion can lead to carbon storage, the obvious conclusion is that soil erosion must be allowed – even encouraged - in order to store more and more carbon. Quinton et al. (2006) found that slope contouring significantly decreased, by 33%, gross soil and carbon erosion losses; other techniques were explored and, although having positive results, there were not such significant effects. Rickson (2014) reviewed 73 studies of soil erosion mitigation from the UK of which 43 quantified the effectiveness of the intervention. Rickson (2014) concluded that none of the 18 techniques were significantly different from each other and so it was impossible to select one mitigation technique over another; nevertheless, taken together, all 18 techniques decreased the soil erosion typically by 40%. Given that this study has concluded that soil erosion is likely a net source of GHGs, any decrease in soil erosion immediately represents a GHG saving. Thus, soil conservation can be shown to prevent GHG emissions to the atmosphere as well as protected terrestrial and aquatic ecosystem services.

## 5. Conclusions

This study has shown that:

i) It is unlikely that soil erosion in the UK represents a net sink of carbon, let alone of greenhouse gases;

ii) Losses of greenhouse gases to the atmosphere are dominated by $CO_2$ (74%) followed by $N_2O$ (22%) and then $CH_4$ (4%);

iii) The emission factor for 1 tonne of net carbon erosion is between 3.3 and 9.9. tonnes $CO_{2eq}$/yr;

iv) The emission factor for 1 tonne of gross carbon erosion is between 1.6 and 8.9 tonnes $CO_{2eq}$/yr;

v) The emission factor for gross soil erosion is estimated to be between 0.11 and 0.66 tonnes $CO_{2eq}$/yr for every 1 tonnes of gross erosion;

vi) Gross soil erosion can only represent a net sink of carbon and greenhouse gases in circumstances where the replacement is high ($R > 0.7$), the net to gross erosion rate is low ($<0.47$) and the gross erosion rate is very low ($< 91$ tonnes/km$^2$/yr).

Our results indicate that soil conservation measures are required to protect the atmosphere as well as land and water. Whilst it is possible in some circumstances that soil erosion can produce a net carbon sink, in most cases the effect of erosion is detrimental to the atmosphere in terms of GHG emissions. Our study finds little evidence therefore to support a more relaxed approach to soil erosion, indeed quite the converse. Thus, conserving soil organic carbon provides a more extensive range of ecosystem services than might previously have been thought, with protection not only of terrestrial, freshwater and marine ecosystems, but also of the GHG composition of the atmosphere.

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Table 1. The summary of C/N data used within this study.

| Ref. | Region | Min. | Max. |
|---|---|---|---|
| Bao et al. (2014) | Yangtze | 6.4 | 9.2 |
| Higueras et al. (2014) | Rhone | 2.8 | 14.7 |
| Bouillon (2012) | Congo | 7 | 10.9 |
| Petrone et al. (2010) | W.Australia | 2.8 | 14.7 |
| Jha and Masao (2013) | Japan | 5.3 | 17.8 |
| Guo et al. (2012) | Yukon | 19 | 35 |
| Wang et al. (2004) | Mississippi | 10 | 17 |
| Martinotti et al. (1997) | Po | 6.7 | 8.2 |
| This study | UK | 6.7 | 21.4 |

Table 2. The percentage of the C lost to the atmosphere that was lost as $CH_4$.

| Ref. | Region | %C loss as $CH_4$ |
|---|---|---|
| Striegl et al. (2012) | Yukon | 0.72 |
| Crawford et al. (2014) | Wisconsin | 0.81 |
| Crawford et al. (2013c) | Yukon | 1.1 |
| Silvennomen et al. (2008) | Finland | 1.2 |
| Guerin et al. (2008) | Reservoirs | 1.4 |

Table 3. The percentage of the C lost to the atmosphere that was lost as $CH_4$.

| Ref. | Region | Range |
|---|---|---|
| Higgins et al. (2008) | Colorado river | 0.17 to 4 |
| Beaulieu et al. (2008) | Midwest rivers | upto 20.7 |
| Silvennoinen et al. (2008) | Finland | 0.02 |
| Yan et al. (2004) | Changjiang | 0 – 0.14 |
| Baulch et al. (2011) | USA | 0.75 |

Table 4. The range of the input parameters

| Parameter | Range | Units |
|---|---|---|
| Replacement rate | 0.19 - 1 | dimensionless |
| Enrichment ratio | 1.2 - 4.7 | dimensionless |
| Mineralisation | 0.75 - 0.95 | dimensionless |
| Net to gross | | |
| POM flux at source | 887 - 1504 | Ktonnes/yr |
| Over bank sedimentation | 0.97 | % |
| Percentage of year with overbank sedimentation | 0.1 – 2.9 | % |
| In-channel storage | 2 - 5 | % of incoming POM flux |
| C/N | 9.2 – 14.3 | dimensionless |
| C in POM | 45 – 50 | % |
| Proportion of C loss as $CH_4$ | 0.64 – 2.2 | % |
| Proportion of N loss as $N_2O$ | 0.3 – 3.0 | % |
| Estuarine transit | 16 – 43 | % of incoming POM flux |
| Estuarine removal | 36 - 54 | % of incoming POM flux |
| Burial efficiency | 27 - 49 | % |

Table 5. The summary of the fluvial carbon and GHG fluxes for the UK rivers.

| Pathway | Flux (ktonnes C/yr) | Export (tonnes C/km$^2$/yr) | Flux(ktonnes $CO_{2eq}$/yr) |
|---|---|---|---|
| POM | 863 | 3.5 | |
| POM loss | 264 | 1.1 | 998 |
| DOM | 909 | 3.7 | |
| DOM loss | 2650 | 10.9 | 11062 |
| Excess $CO_2$ | 598 | 2.5 | 2194 |
| Total loss at source | 5020 | 21.8 | |
| Total loss to atmosphere | 3512 | 15.2 | 14254 |

Figure 1. Schematic diagram of the generation and fate of particulate organic matter as considered in this study.

Figure 2.The terrigenous organic matter burial efficiency compared to the sedimentation rate.

Figure 3. Summary of flux, sinks and sources of POM (values in **bold** – ktonnes/yr) and greenhouse gases (*values italics* – ktonnes $CO_{2eq}$/yr) as estimated by this study. Ranges on these are provided in the text.