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Suspended sediment regimes in contrasting reference-condition freshwater ecosystems: Implications for water quality guidelines and management

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HIGHLIGHTS

- Mean SS concentrations vary up to 9-fold in contrasting reference temperate rivers.
- Spatial variation can be predicted using an environment-specific SS prediction model.
- There can be high inter-annual variability in mean SS concentrations (up to 3-fold).
- Inter-annual variability can be predicted using a modified SS prediction model.
- Water quality guidelines should recognise spatial and temporal variations in SS.

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ABSTRACT

Suspended sediment (SS), ranging from nano-scale particles to sand-sized sediments, is one of the most common contributors to water quality impairment globally. However, there is currently little scientific evidence as to what should be regarded as an appropriate SS regime for different freshwater ecosystems. In this article, we compare the SS regimes of ten systematically-selected contrasting reference-condition temperate river ecosystems that were observed through high-resolution monitoring between 2011 and 2013. The results indicate that mean SS concentrations vary spatially, between 3 and 29 mg L^{-1}. The observed mean SS concentrations were compared to predicted mean SS concentrations based on a model developed by Bilotta et al. (2012). Predictions were in the form of probability of membership to one of the five SS concentration ranges, predicted as a function of a number of the natural environmental characteristics associated with each river's catchment. This model predicted the correct or next closest SS range for all of the sites. Mean annual SS concentrations varied temporally in each river, by up to three-fold between a relatively dry year (2011–2012) and a relatively wet year (2012–2013). This inter-annual variability could be predicted reasonably well for all the sites except the River Rother, using the model described above, but with modified input data to take into account the mean annual temperature (°C) and total annual precipitation (mm) in the year for which the mean SS prediction is to be made. The findings highlight the need for water quality guidelines for SS to recognise natural spatial and temporal variations in SS within rivers. The findings also demonstrate the importance of the temporal resolution of SS sampling in determining assessments of compliance against water quality guidelines.

1. Introduction

Managing global water resources is one of the greatest challenges of the 21st century (Garrido and Dinar, 2009; Poff, 2009; Staddon, 2012). Water is a resource that is under growing pressure as the global population rises, and the natural supply, in the form of precipitation, is becoming increasingly variable and uncertain with climate change (Giorgi et al., 2004; Räisänen et al., 2004; Trenberth et al., 2003). It is therefore essential that water resources are managed sustainably in terms of both their quantity and quality. One of the most commonly attributed causes for the impairment of water quality globally is the presence of excess suspended sediments, ranging from nano-scale particles and colloids to sand-sized sediments (Gray, 2008; Richter et al., 1997). Suspended sediments (SS) can have a range of detrimental effects on water resources, from aesthetic issues and higher costs of

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water treatment, to a decline in the fisheries resource and serious ecological degradation (Alabaster and Lloyd, 1982; Bilotta and Brazier, 2008; Cordone and Kelley, 1961; Gammon, 1970; Newcombe and MacDonald, 1991; Owens et al., 2005; Peddicord, 1979; Ryan, 1991; Wood and Armitage, 1997). Ultimately this can lead to a significant decline in the associated freshwater ecosystem services, estimated to have a global value in excess of $1.7 trillion per annum (Costanza et al., 1998).

In recognition of the potential for SS to cause aquatic degradation, and in an effort to minimise this degradation, government-led environmental organisations from around the world have established water quality guidelines and standards, which state recommended targets for SS (sometimes referred to as suspended solids, and occasionally assessed through proxy measurements such as turbidity) (Bilotta and Brazier, 2008; Collins et al., 2011). However, at present these guidelines are often blanket values that do not recognise the natural spatial and temporal variations of SS in streams/rivers, and are not well-linked to the biological/ecological impact evidence; therefore ultimately these guidelines may not reflect the specific requirements of the biological communities that they are designed to protect (Bilotta et al., 2012; Collins et al., 2011; Schwartz et al., 2008, 2011). For example, in Europe the Freshwater Fisheries Directive (78/659/EEC) (2006/44/EC) guideline for SS stated that concentrations should not exceed 25 mg L^{-1} in salmonid and cyprinid waters except in exceptional weather conditions or exceptional geographic circumstances (Bilotta and Brazier, 2008; Collins et al., 2011). This Directive has now been repealed and replaced by the Water Framework Directive (2000/60/EC) (2008/105/EC), but there are currently no new guidelines for SS under this Directive.

It is important to recognise that unlike some aquatic pollutants (e.g. pesticides, pharmaceuticals, veterinary medicines), SS is a natural component of freshwater ecosystems, critical to habitat heterogeneity and ecological functioning (Maitland, 2003; Swietlik et al., 2003; Vannote et al., 1980; Yarnell et al., 2006). There is natural spatial variation in SS conditions. For example, Bilotta et al. (2012) found background concentrations of SS varied by more than 15-fold between 42 temperate ecosystem-types that were in reference condition.[1] Furthermore, these differences could be predicted using data on a number of the natural environmental characteristics associated with each river's catchment, including metrics of climate, catchment geology and topography, and channel hydromorphology. The interpretation of these findings was that differences in the natural environment create unique SS conditions that support unique freshwater communities. If water quality managers wish to protect these unique freshwater communities through establishing and implementing water quality guidelines, then these guidelines should be environment-specific.

There are also natural temporal variations in SS conditions, which may be critical to the ontogeny of certain organisms (e.g. Maitland, 2003), and which should be expected given the temporally variable contributions of SS from channel and non-channel sources. Wood and Armitage (1997) suggest that the principal sources of SS available to a river from channel sources are: (i) river banks; (ii) mid-channel and point bars; (iii) fine bed material; (iv) natural backwaters; (v) fine particles associated with aquatic vegetation; and (vi) other biotic particles including phytoplankton and zooplankton. In addition there may be in-channel generation of SS due to the decomposition of aquatic macrophytes, biofilms and invertebrates. Benthic invertebrate faecal material has been shown to constitute a significant source of fine SS (Ladle and Griffiths, 1980; Ward et al., 1994). Finally, some aquatic organisms (invertebrates and vertebrates) may also act to re-suspend material stored in the bed and banks of a water body through burrowing, feeding, and breeding behaviours (Harvey et al., 2011; Hassan et al., 2008; Montgomery et al., 1996). The main non-channel sources of SS supplied to a river are: (i) exposed soils subject to erosion — this material is transported to the channel via surface and subsurface runoff; (ii) mass failures within the catchment, such as landslides and soil creep; (iii) litter fall, principally leaf material from vegetation adjacent to the channel; and (iv) atmospheric deposition, due to aeolian processes and precipitation (Wood and Armitage, 1997).

In catchments where these SS sources have been modified through anthropogenic activities, the resultant modified SS regimes can affect the biological community. However, at present the SS regime required to maintain or restore biological integrity in a given environment has not been defined. It is known that some aquatic organisms are sensitive to changes in SS, in particular, changes to the duration, frequency and timing that a given concentration is experienced (Diehl and Wolfe, 2010; Kerr, 1995; Newcombe and Jensen, 1996; Newcombe and MacDonald, 1991; Reid and Anderson, 1999; Waters, 1995; Yount and Niemi, 1990), which collectively are referred to as the SS regime (concentration, duration, frequency). The current water quality guidelines for SS, however, through their use of mean annual concentration values or total mean daily loads, fail to recognise the importance of the SS regime, because an observed mean concentration, even if regarded as being compliant with guidelines, could be achieved through an infinite number of concentration scenarios (from highly variable regimes with concentration extremes to relatively stable regimes with little variability about the mean), each scenario potentially having very different biological effects. There is therefore a need for more advanced water quality guidelines that can take into account these natural spatial and temporal variations, so that water quality managers can identify where and when SS pollution is taking place and to what extent. The first step to achieving this is to understand what the natural background SS regimes are in contrasting ecosystems that are in reference-condition.

The aims of this study are to (1) monitor the SS regimes of contrasting reference-condition river ecosystems and to examine the spatial and temporal variations in SS regimes, and based on these findings (2) consider the appropriateness of the current regulatory water quality guidelines and monitoring.

2. Materials and methods

2.1. Field sites

Field sites were selected from the RIVPACS IV database (May 2011 version) (River Invertebrate Prediction and Classification System — © NERC [CEH] 2006. Database rights NERC [CEH] 2006 all rights reserved). The RIVPACS IV model and database are described in details by Wright et al. (2000) and Clarke et al. (2011), and are only summarised here. The database contains invertebrate, water quality and catchment characteristics data, recorded between 1978 and 2004, from 795 streams and river sites in the UK. All of the sites are in reference condition; defined as having no, or only very minor, anthropogenic alterations to the values of the hydrochemistry and hydromorphology, with biota usually associated with such undisturbed or minimally disturbed conditions. In the development of the RIVPACS model, the sites were divided into similar biological communities, referred to as end groups, based on the invertebrate community composition. Forty-three end group communities were identified in RIVPACS IV. These end groups are a proxy for the wider ecosystems, i.e. specific invertebrate communities occur in specific environments and are associated with specific floral and faunal (vertebrate) communities. In the RIVPACS IV model these end groups were characterised, using Multiple Discriminant Analysis (MDA), in terms of the temporally invariant properties of the environments that they inhabit. The original purpose of the RIVPACS model was for use in assessments of ecological status. The model can be applied to any site wherever these environmental properties can be measured, allowing the user to make a prediction of the expected community composition. The magnitude of deviation

[1] 638 stream/river sites grouped into 42 ecosystem types/end groups based on the similarities in the invertebrate community composition.

between observed and expected community compositions at a site is used as an indication of the ecological status and therefore anthropogenic disturbance.

In this article, the authors used the RIVPACS IV database to identify ecosystems that occur in the most contrasting environments. In order to achieve this, the environmental properties of the 43 end groups were plotted for eight of the twelve environmental variables identified through MDA as being discriminant variables for invertebrate community composition. Latitude and longitude were excluded because they have no direct physical basis and are therefore likely to produce models that are only applicable locally. Alkalinity was excluded because it is a water quality parameter which can be influenced by pollution which could result in an incorrect reference condition model prediction. Air temperature range was excluded because it is highly correlated with mean annual air temperature.

The end groups that had the lowest and highest mean values for each of the remaining eight environmental variables were selected as the most contrasting ecosystem types; subsequently the field sites from each of the identified end groups were selected. This was decided by choosing the sites that had the value closest to the mean for that particular environmental variable for that end group. In a few instances, for logistical reasons (proximity/access), the sites that had the second closest value to the mean for the environmental parameter variable were chosen. Using this process, ten field sites were selected to encompass a wide range of environments, varying in their: (1) Climate — mean annual precipitation totals of 450 to 1400 mm and mean annual temperatures ranging from 8.42 to 11.33 °C, (2) Geology — varying from catchments with almost 100% hard igneous rocks to catchments with almost 100% clay or sandstone, and (3) Topography — altitudes at the river's source varying from 128 to 1216 m above sea level. The stream and river sites also vary in their morphometry with widths ranging from 0.60 to 50 m and depths ranging from 0.08 to 1.90 m (mean of 3 seasonal measurements). As such the findings and model developed from this study will be applicable to a wide range of streams and rivers located in mid-latitude zones. The approximate location of each site can be seen in Fig. 1, which also describes the range of environmental characteristics of each site.

2.2. Sampling, monitoring and laboratory analysis

Each field site was instrumented with a multi-parameter water quality probe (*Aquaread Aquaprobe*) and logger (*Aquaread Aqualogger*), which recorded turbidity — a surrogate measure of SS; every 15 min. These turbidity data were converted to estimated SS concentrations using site-specific turbidity-SS regression equations derived from:

(1) SS data measured through laboratory analysis of water samples that were collected from the sites on a 4–8-week temporal frequency, between May 2011 and May 2013. Water samples were collected manually in 1 L plastic bottles in a depth- and width-integrating manner across each river site. Samples were refrigerated at <4 °C and stored in the dark prior to analysis in order to avoid degradation. Following Clesceri et al. (1998), samples were analysed in the laboratory for concentrations of SS and volatile organic matter (VOM). This involved filtration of a known volume of sample through a dried, pre-combusted (550 °C for 30 min) and pre-weighed glass fibre filter paper (*Whatman GFF*), with a pore size of 0.70 μm. The filters and their residue were then dried at 105 °C for 60 min and re-weighed. Following this the filters and their residue were ignited in a muffle furnace at 550 °C for 30 min and then re-weighed to determine VOM (VOM equal to the dried particle mass minus the combusted particle mass). This is the standard technique for the determination of VOM (APHA, 1995; APHA, 1998).

(2) Laboratory-based turbidity-SS calibrations using SS collected from time-integrated samplers between May 2011 and May

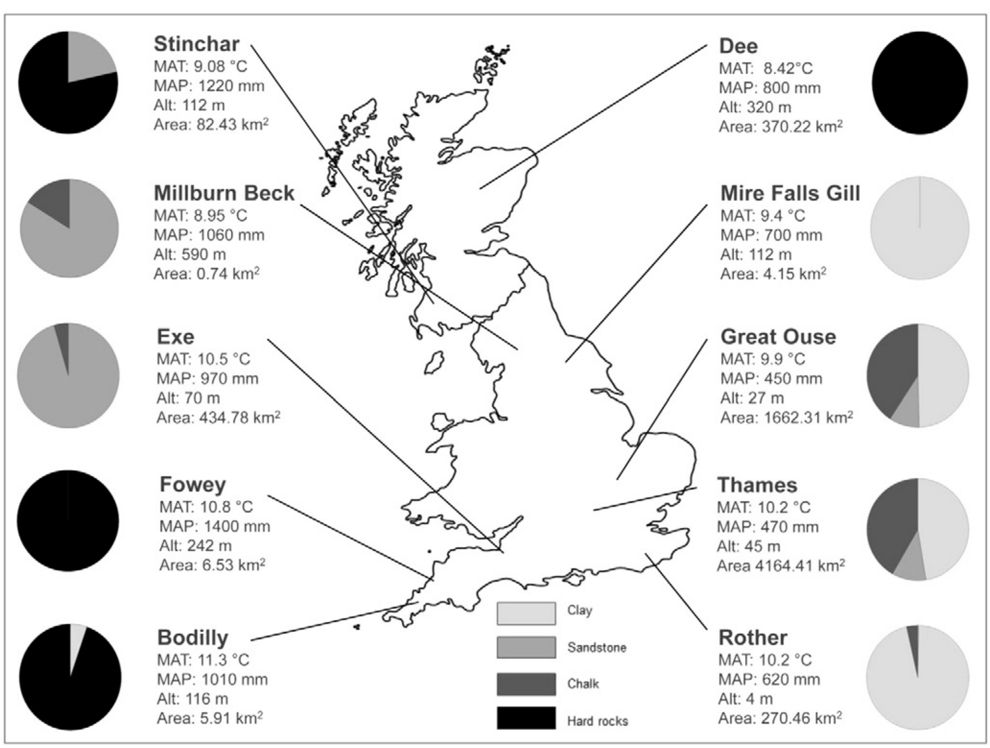

Fig. 1. Location of the ten reference-condition stream/river sites, including information on their catchment characteristics: mean annual temperature (MAT), mean annual precipitation (MAP), mean altitude of upstream catchment (Alt), and catchment area (Area). Dominant geology classes, shown in the pie-charts, represent an amalgamation of the 116 Solid Geology classes of the British Geological Survey into 8 River Habitat Survey super-classes. The environmental characteristics of each site, excluding MAP, were sourced from the RIVPACS IV (May 2011 version) database, (Clarke et al., 2011). Mean annual precipitation data were sourced from the UK Meteorological Office (http://www.metoffice.gov.uk/climatechange/science/monitoring/ukcp09/) and derived from 1 km gridded data for the monitoring location during the period 1961–1990.

2013, that were combined with filtered river water from the site to make up a wide range of known SS concentrations against which to measure turbidity. The time-integrated SS samples were collected using the time-integrating passive trap sampler method described by Phillips et al. (2000). The time-integrated SS samplers were one metre long and consisted of commercially available PE pipes with an outer diameter of 110 mm and a wall thickness of 4.2 mm. They were sealed with a plugged polythene funnel at the inlet and a cap at the outlet. A plastic tube with an inner diameter of 4 mm was passed through the funnel and the cap as inlet and outlet. The samplers were mounted parallel to the riverbed at two upright steel bars driven into the channel bed, with the inlet tube pointing upstream. The greater cross-sectional area of the main cylinder compared to that of the inlet tube reduces the flow velocity within the samplers by a factor of 600 relative to that of the ambient flow. This reduction in

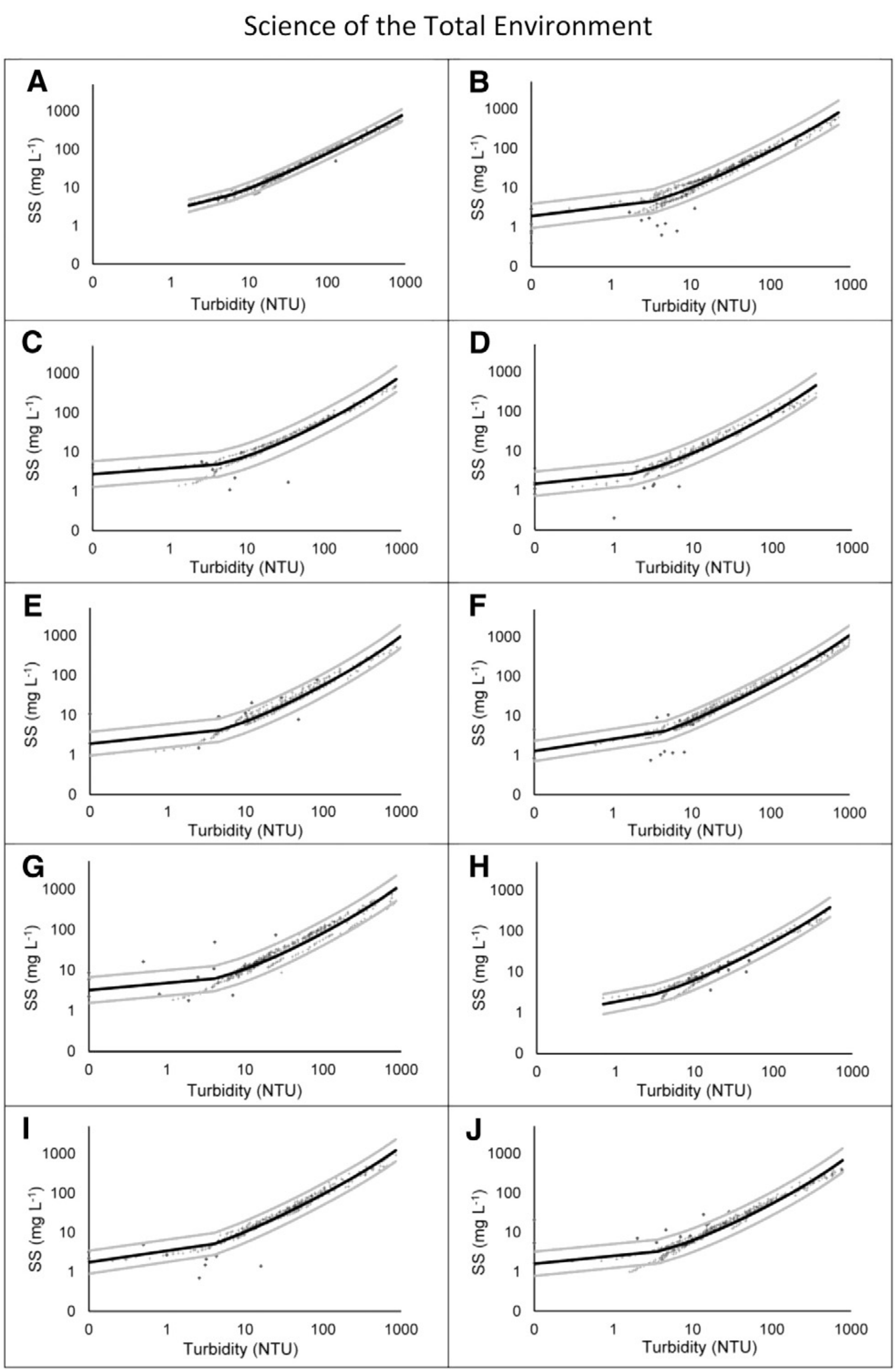

Fig. 2. Observed turbidity-SS values (grey crosses for time integrated samples; black diamonds for grab samples) and the predicted values based on regression analysis at each of the study sites (black lines represent the fitted regression value, grey lines represent the upper and lower 95% prediction intervals). Regression equations used to make predictions were carried out using quadratic equations on log-transformed turbidity (NTU) (X-axis) and SS concentration data (Y-axis). All regression equations are statistically significant (p-values < 0.001). (A) Bodilly, $R^2 = 0.98$. (B) Dee, $R^2 = 0.94$. (C) Exe, $R^2 = 0.93$. (D) Fowey, $R^2 = 0.94$. (E) Great Ouse, $R^2 = 0.95$. (F) Millburn Beck, $R^2 = 0.96$. (G) Mire Falls Gill, $R^2 = 0.93$. (H) Rother, $R^2 = 0.95$. (I) Stinchar, $R^2 = 0.96$. (J) Thames, $R^2 = 0.94$.

Table 1
Estimated mean SS concentrations with lower and upper prediction intervals (based on 95% confidence intervals derived from the turbidity-SS regression), estimated median SS, estimated minimum and maximum SS, and 5th, 30th, 50th and 90th percentiles for estimated SS (mg L^{-1}) in the ten study sites monitored between May 2011 and May 2013. n refers to the number of turbidity readings upon which the estimate is based.

Site	n	Mean	Lower–upper prediction intervals	5th Percentile	30th Percentile	50th Percentile	90th Percentile	Min	Max
				C95	C70	C50 (median)	C10		
Bodilly	59,976	29	20–42	0	11	22	66	0	357
Dee	42,052	6	3–13	0	1	3	14	0	304
Exe	57,840	18	9–39	3	3	7	35	2	591
Fowey	55,489	3	1–6	1	1	1	4	1	460
Great Ouse	51,544	29	15–57	3	3	6	79	2	857
Milburn Beck	62,157	6	3–10	1	1	3	14	1	380
Mire Falls Gill	58,169	15	8–32	3	3	8	27	2	492
Rother	43,331	11	6–19	1	3	6	25	1	407
Stinchar	54,120	15	8–28	1	1	6	32	1	701
Thames	51,862	11	6–22	2	4	7	22	1	234

flow velocity induces sedimentation of the SS as the water moves through the cylinder towards the outlet tube (Phillips et al., 2000). The time-integrated SS samplers are designed to collect a statistically representative sample under field conditions (Perks et al., 2013; Phillips et al., 2000; Wildhaber et al., 2012). The SS was extracted from these samplers on a 4–8-week temporal frequency.

The best-fitting regression equations for all of the sites are shown in Fig. 2.

2.3. Data processing and analysis

All field data were checked for consistency in turbidity signals by visual inspection as well as against discharge data gathered by the Environment Agency and the Scottish Environment Protection Agency or from the discharge estimations computed from the stage-discharge data collected during the study. In cases where the stream debris or biota covered the turbidity sensors, or if sensors did not respond to changes in stream discharge for prolonged periods of time suggesting equipment failure, the data were removed.

All statistical analysis was conducted using SPSS statistical software (IBM® SPSS® Statistics 20). Prior to the statistical analysis, the SS concentration data sets for all the ten study sites were investigated for normality and homogeneity of variance. In all cases the Kolmogorov–Smirnov test for normality ($p < 0.001$) suggests that the SS concentration data are not normally distributed. The test of homogeneity of variance (Levene Statistic; $F(9, 536530) = 7192.1$, $p = <0.001$) also indicated that the variances are significantly different. Tests of normality and of the homogeneity of variance were also carried out on the transformed SS data (common and natural logarithm, square, square root) with the same outcomes. An examination of the skewness and kurtosis values for the sites also suggests that the SS concentration data are not normally distributed. Based on those findings, non-parametric tests were applied for comparative data analysis.

3. Results

3.1. Spatial variability

Table 1 presents the summary statistics for the estimated (from herein 'estimated' refers to turbidity-derived) SS concentrations in the ten river sites during the 2011–2013 monitoring period. The raw data are provided in files S1–10 in the Supplementary online material. The estimated mean SS concentrations range from 3 mg L^{-1} (Fowey) to 29 mg L^{-1} (Bodilly and Great Ouse). A Kruskal–Wallis and post-hoc tests were conducted to determine if the differences in estimated mean SS concentrations between the sites are statistically significant. Pairwise comparisons were performed using Dunn's (1964) procedure with a Bonferroni correction for multiple comparisons. Statistical significance was accepted at the $p < 0.05$ level for the omnibus test and $p < 0.05$ for the multiple comparisons. Estimated SS concentrations were significantly different between the sites, $\chi 2\ (9) = 115315.17$, $p < 0.001$. Post-hoc analysis revealed statistically significant differences in all comparisons ($p < 0.001$), except between the following pairs of sites: the River Dee (median = 3.2 mg L^{-1}) and Milburn Beck (median = 3.0 mg L^{-1}) ($p = 0.555$); the River Rother (median = 6.1 mg L^{-1}) and River Stinchar (median = 6.2 mg L^{-1}) ($p = 0.048$); and the River Mire Falls Gill (median = 7.7 mg L^{-1}) and River Exe (median = 6.8 mg L^{-1}) ($p = 0.027$).

Table 2 presents the low-resolution, grab sample-derived mean SS concentrations observed in the ten river sites monitored between 2011 and 2013. Observed mean SS concentrations measured between May 2011 and 2013 ranged from 1 mg L^{-1} (Dee) to 17 mg L^{-1} (Bodilly and Mire Falls Gill). Table 2 also compares these observations with the historical low-resolution (monthly grab sample-derived) mean SS concentrations observed in other reference-condition (RIVPACS) sites belonging to the same ecosystem-types or 'end groups'. All of the river sites monitored between 2011 and 2013, except for the River Bodilly, had mean SS concentrations that were within the minimum and maximum mean SS concentrations observed in the sites belonging to the same end-groups. The River Bodilly's mean concentration exceeded the range of mean concentrations observed in the sites belonging to the same end-group. A comparison of the observed (grab-sample) mean SS concentration for each site against the equivalent predicted[2] SS ranges (see Fig. 3), reveals that the model predicted the correct SS range for seven of the sites, and predicted the correct or next closest SS range for the remaining three sites.

3.2. Temporal variability

Table 3 presents the estimated mean annual SS concentration observed in each river site for year one (May 2011 to May 2012) and year two (May 2012 to May 2013) of monitoring. These data are also illustrated in Fig. 4 alongside the estimated mean SS concentration for the total monitoring period. A Mann–Whitney *U* test showed that there are statistically significant differences in the distribution of SS concentrations observed in year one and year two for all the sites ($p < 0.001$ in all cases). All the river sites, except the Dee, Stinchar and Thames, experienced higher estimated mean annual SS concentrations in year two of

[2] Bilotta et al. (2012) proposed a model capable of predicting mean (monthly grab-sample-derived) SS concentrations to the correct range (6 mg L^{-1} ranges) or the next closest range (in 90% of cases), as a function of a range of site characteristics: mean annual air temperature, mean annual precipitation, mean altitude of upstream catchment, distance from source, slope to source, channel width and depth, the percentage of catchment area comprised of clay, chalk, and hard rock solid geology, and the percentage of the catchment area comprised of blown sand as the surface (drift) material. This model was used to make predictions for the sites monitored in this study.

Table 2
Comparison of the observed mean SS concentration (mg L^{-1}, based on low-resolution grab sampling) against the mean, minimum and maximum SS concentrations (mg L^{-1}) observed in the sites (n = 638) belonging to the same RIVPACS IV end groups.

River	Observed (grab) mean SS 2011–2013	Mean average SS of sites in the same end group	Min average SS of sites in the same end group	Max average SS of sites in the same end group
Bodilly	17	8	6	9
Dee	1	3	1	6
Exe	13	9	1	24
Fowey	2	4	1	10
Great Ouse	16	19	8	45
Millburn Beck	5	4	1	10
Mire Falls Gill	17	13	5	26
Rother	14	12	4	26
Stinchar	4	4	1	9
Thames	14	19	8	45

the study. This difference appears to be partially related to the mean annual discharge. The largest difference in mean annual SS concentration was observed in the River Great Ouse, which experienced a three-fold increase between year one (14 mg L^{-1}) and year two (42 mg L^{-1}) concomitant with roughly three-fold increases in mean annual discharge observed at this site from year one (6.37 m^3 s^{-1}) to year two (19.56 m^3 s^{-1}) (Table 4). The Mann–Whitney *U* test showed significant differences ($p < 0.05$ in all cases) in annual average discharge observed in year one and year two for all the sites. The relationship between estimated mean annual SS and mean annual discharge is not evident for the Thames site, which experienced remarkably similar estimated mean annual SS concentrations in year one (10.94 mg L^{-1}) and year two (10.86 mg L^{-1}), despite experiencing contrasting mean annual discharges (16.34 m^3 s^{-1} and 55.90 m^3 s^{-1}).

These temporal variations in discharge and SS are natural phenomena that shape biological communities. Therefore ideally (a) water quality guidelines should acknowledge this, and (b) the sampling frequencies used should enable accurate assessments of compliance with these guidelines. In order to assess whether the environment-specific SS prediction model developed by Bilotta et al. (2012) could predict this inter-annual variation, the model input data on mean annual temperature (1961–1990 average) and mean annual precipitation (1961–1990) was replaced with mean annual temperature (°C) and total annual precipitation (mm) measured in the year for which the prediction is to be made at UK Meteorological Office stations closest to each of the ten study sites.[3] Fig. S1a and S1b (Supplementary online material) illustrates the predicted SS ranges arising from this analysis compared to estimated mean annual SS concentrations (derived from the simulated monthly grab samples described below). As can be seen from Fig. S1a and S1b, there is good overlap between the predicted SS ranges and the simulated-grab estimated mean annual SS concentrations, for all the sites except the site on the River Rother.

In order to assess the accuracy of the sampling frequency that is currently used in routine regulatory monitoring (Bowes et al., 2009; Cooper et al., 2003), the high-resolution (15 min frequency) SS datasets were systematically subsampled, selecting one sample per month (selected at 08:00 am on the first day of each month), to calculate the low-resolution mean SS concentrations over the 2011–2013 monitoring period for each site, simulating routine regulatory water quality monitoring. Percentage errors as well as absolute differences were calculated between the high-resolution (15 min frequency) mean SS concentrations and the simulated low-resolution (monthly frequency) mean SS concentrations. The results, presented in Table 5, demonstrate that mean SS concentrations derived from monthly sampling frequencies suffered from errors of up to 53% of values derived from 15-min sampling frequencies, with an average error of 24% for all the ten sites.

4. Discussion

4.1. Spatial variability

The results presented in Table 1 show that there were over 9-fold differences in estimated mean SS concentrations between the ten sites. Two of the river sites, Bodilly and Great Ouse, had estimated mean SS concentrations (29 mg L^{-1}) that would have exceeded the Freshwater Fisheries Directive (FFD) guideline despite being in reference-condition. The implications of this finding are that the FFD's blanket water quality guideline is likely to have been inappropriate for these specific ecosystem-types as they may be non-compliant regardless of the intensity of land-use. Five of the river sites studied had estimated mean SS concentrations that were less than half of the FFD guideline, with the lowest mean SS observed in the River Fowey (3 mg L^{-1}). Although these river sites would be compliant with the FFD guideline, it is questionable whether the guideline was appropriate for them, as previous research has demonstrated that detrimental impacts can be experienced by some freshwater organisms, of all trophic levels, when exposed to concentrations below 25 mg L^{-1}. For example, Quinn et al. (1992) observed a 40% reduction in the biomass of phytoplankton exposed to SS concentrations of 10 mg L^{-1}. Lloyd (1987) observed a 3–13% reduction in the primary productivity of macrophytes and periphyton exposed to SS concentrations of 8 mg L^{-1}. Rosenberg and Wiens (1978) observed an increase in the amount of drift of benthic invertebrates exposed to SS concentrations of 8 mg L^{-1}. Robertson et al. (2007) observed increased foraging activity in Atlantic Salmon when exposed to SS concentrations of 20 mg L^{-1}. Therefore, it is suggested here that some ecosystems may require much lower guideline values in order to be effectively protected.

The degree of spatial variation in estimated mean SS concentrations (3–29 mg L^{-1}) and observed (grab sample-derived) mean SS concentrations (1–17 mg L^{-1}), supports earlier research by Bilotta et al. (2012) that analysed the mean SS concentrations derived from low-resolution (monthly grab sample) monitoring programmes in 638 reference-condition sites in the UK (grouped into 42 ecosystem-types or 'end groups'). Bilotta et al. (2012) reported that the mean SS concentrations for the 42 end groups varied between 1.7 and 26 mg L^{-1}. All of the river sites monitored between 2011 and 2013, except for the River Bodilly, had observed (grab sample-derived) mean SS concentrations that were within the minimum and maximum mean SS concentrations observed in the sites belonging to the same end-groups. Bilotta et al. (2012) proposed a model capable of predicting mean (monthly grab sample-derived) SS concentrations to the correct range (6 mg L^{-1} ranges) or the next closest range (in 90% of cases), as a function of a range of site characteristics: mean annual air temperature, mean annual precipitation, mean altitude of upstream catchment, distance from source, slope to source, channel width and depth, the percentage of

[3] Data available from: http://www.metoffice.gov.uk/public/weather/climate-historic/#?tab=climateHistoric. The contemporary meteorological data from these meteorological stations were adjusted according to percentage differences in the 1961–1990 averages between the meteorological station data and the catchment meteorological data (the latter not being available for 2011–2013).

catchment area comprised of clay, chalk, and hard rock solid geology, and the percentage of the catchment area comprised of blown sand as the surface (drift) material. This model was used to make predictions for the sites monitored in this study. A comparison of the observed (grab-sample) mean SS concentration for each site against the equivalent predicted SS ranges (see Fig. 3), reveals that the model predicted the correct SS range for the seven sites, and predicted the correct or next closest SS range for the remaining three sites.

4.2. Temporal variability

There were statistically significant differences in the distribution of SS concentrations observed in year one and year two of the study for all the sites ($p < 0.001$ in all cases). All river sites, except the Dee, Stinchar and Thames, experienced higher estimated mean annual SS concentrations in year two of the study. This difference appears to be partially related to the mean annual discharge. The largest difference

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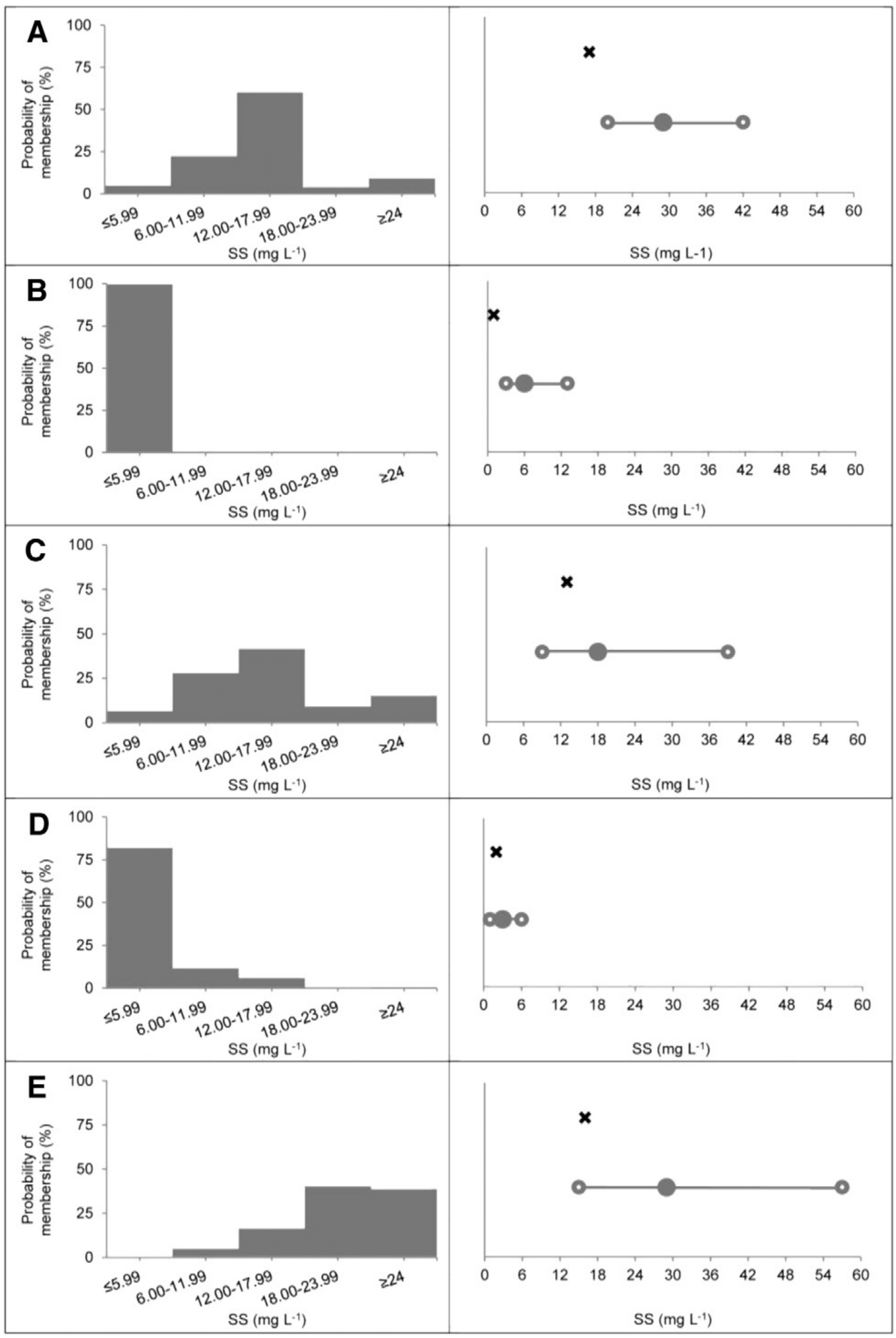

Fig. 3. A comparison of the model-predicted SS ranges (bar charts on the left panel) versus the observed (grab sample-derived; illustrated by crosses on the right panel) and estimated (turbidity-derived; illustrated by circles: a closed circle for the central estimate and open circles for the lower and upper prediction intervals) SS concentrations for 2011–2013 monitoring period. (A) Bodilly, (B) Dee, (C) Exe, (D) Fowey, (E) Great Ouse, (F) Millburn Beck, (G) Mire Falls Gill, (H) Rother, (I) Stinchar, (J) Thames.

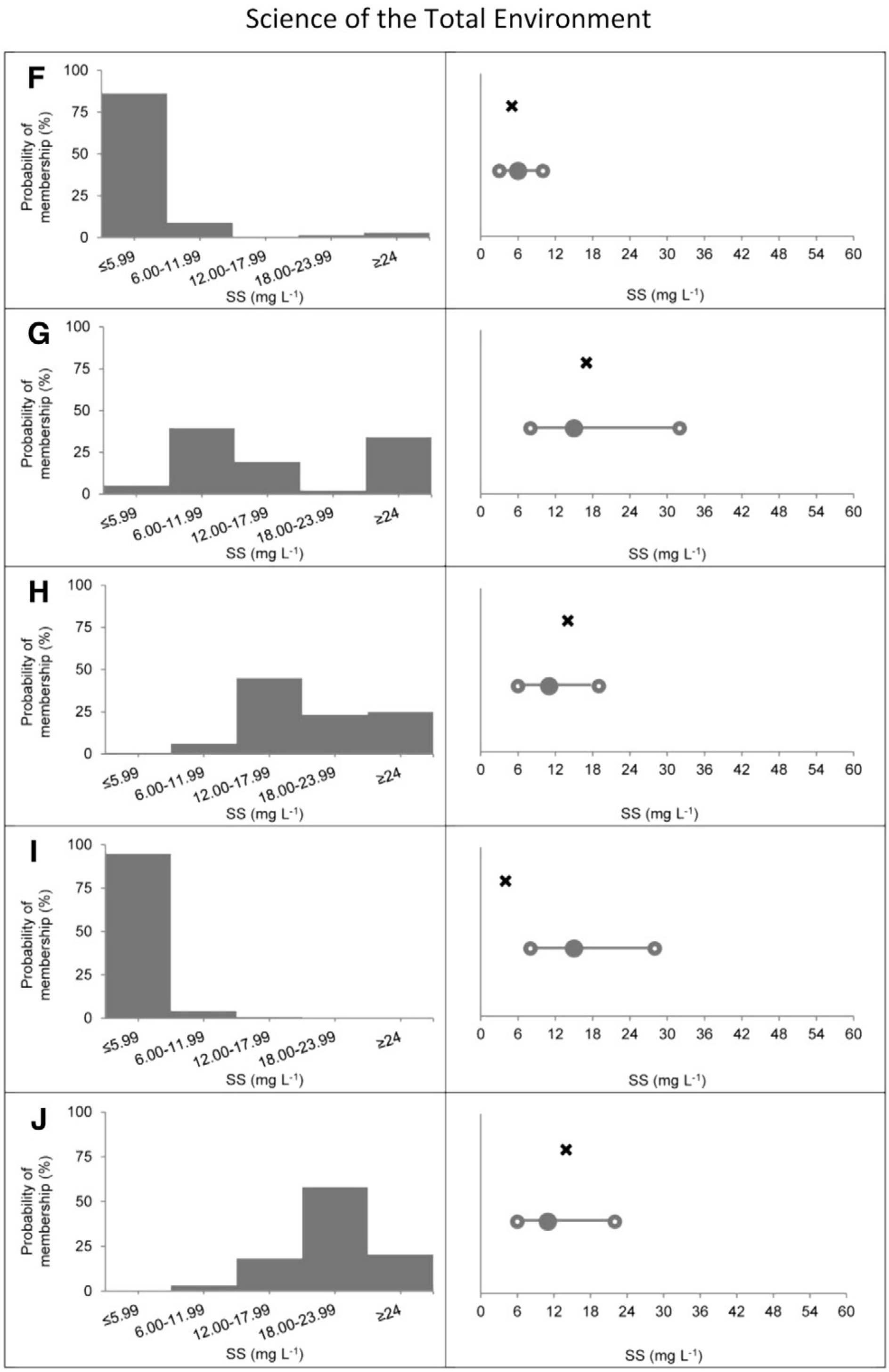

Fig. 3 (*continued*).

in mean annual SS concentration was observed in the River Great Ouse, which experienced a three-fold increase between year one (14 mg L^{-1}) and year two (42 mg L^{-1}) concomitant with roughly three-fold increases in mean annual discharge observed at this site from year one (6.37 m^3 s^{-1}) to year two (19.56 m^3 s^{-1}) (Table 4). This inter-annual variability could be predicted reasonably well (see Fig. S1a and S1b) for all the sites except the River Rother, using the environment-specific model described above, but with modified input data to take into account the mean annual temperature (°C) and total annual precipitation (mm) in the year for which the mean SS prediction was to be made.

The average discharge for the 2011–2013 monitoring period is within 20% of the historical average annual discharge for each site (where data are available). Therefore, although the individual estimated mean annual SS concentrations reported in this paper may represent observations from a drier than average year and a wetter than average year for many sites, the combined mean SS concentration for the whole monitoring period (2011–2013) may be considered to be representative of average hydrological conditions for each site and perhaps therefore average SS conditions for each site. Of course mean values can conceal significant inter-annual differences in the dynamics of both SS and discharge, distorting any relationships between these parameters. For

Table 3
Estimated mean annual SS concentrations in the ten river sites and the percentage differences between the seasons.

SS concentration (mg L^{-1})	2011–2012 Mean	2012–2013 Mean	Percentage difference between 2011–2012 and 2012–2013
Bodilly	26.70	32.60	22.1
Dee	6.53	6.33	−3.1
Exe	15.48	21.25	37.3
Fowey	2.21	3.57	61.5
Great Ouse	13.94	42.08	201.9
Milburn Beck	5.64	5.79	2.7
Mire Falls Gill	13.14	17.26	31.4
Rother	8.17	13.02	59.4
Stinchar	17.46	11.76	−32.6
Thames	10.94	10.86	−0.7

example, although the mean annual SS concentrations for the Thames were similar in year one and year two, the Mann–Whitney *U* test showed that there was a significant difference between the distribution of estimated SS concentrations across the two years.

4.3. Sampling frequency considerations

The results presented in Table 5 demonstrate that mean SS concentrations derived from monthly sampling frequencies suffer from errors of up to 53% of values derived from 15-min sampling frequencies, with an average error of 24% for all the ten sites. Similar errors have been reported by other researchers (Coynel et al., 2004; Skarbøvik et al., 2012; Thompson et al., 2014). As the errors vary in both directions there is no simple assumption that less intense sampling underestimates the 'true' values as is sometimes inferred in literature (Coynel et al., 2004; Thompson et al., 2014). Based on the results of the current study and previous research (Coynel et al., 2004; Skarbøvik et al., 2012; Thompson et al., 2014), we suggest that water quality managers should place lower confidence in the assessments of compliance with water quality guidelines, when mean annual concentrations are based on monthly-resolution sampling; higher-resolution monitoring of SS would be recommended in order to make more accurate assessments of compliance with water quality guidelines. Thompson et al. (2014) suggest that the optimum monitoring frequency for SS, in terms of accuracy and practicality (e.g. feasible using an autosampler), is likely to be achieved through a 24/7 sampling regime (i.e. a SS sample taken every 7 h to give 24 samples per week). In order to determine the accuracy of this sampling regime in the ten sites studied here, the high-resolution (15-min frequency) SS datasets were systematically subsampled (selecting one sample every 7 h) to calculate a mean SS concentration for each site based on simulated 24/7 sampling. The average error compared to the high-resolution means was just 2%. In addition to enabling a more accurate quantification of the mean SS concentration, high-resolution monitoring of SS also enables water quality managers to gain a better insight into the temporal dynamics of SS (concentration–duration–frequencies) which can be used to predict the impacts on the ecosystem. High-resolution monitoring is more likely to capture the occasional low-frequency high concentration events such as runoff from agricultural land during storms, which may otherwise be missed by low-resolution monitoring but nevertheless can have significant impacts on the aquatic biota. High-resolution monitoring may also help to identify the sources of the pollution based on temporal patterns in concentrations. It must be recognised, however, that: (1) At present there is no environment-specific model that is capable of predicting the high-resolution reference-condition dynamics of SS at a given site in order to aid assessment. At best, the environment-specific model developed by Bilotta et al. (2012) can only predict mean annual concentration ranges. The development of an environment-specific model that is capable of predicting high-resolution dynamics or alternative metrics of exposure (e.g. concentration–duration–frequencies), would require high-resolution monitoring data from studies, similar to this one, collected from a significant number of reference-condition sites. (2) If high resolution monitoring is achieved using a surrogate measure of SS (e.g. turbidity monitoring rather than measuring SS through sampling and analysis), the resultant estimated concentrations will suffer from uncertainties that arise from the turbidity-SS regression analysis. As illustrated in Fig. 3, these uncertainties can be substantial in some cases.

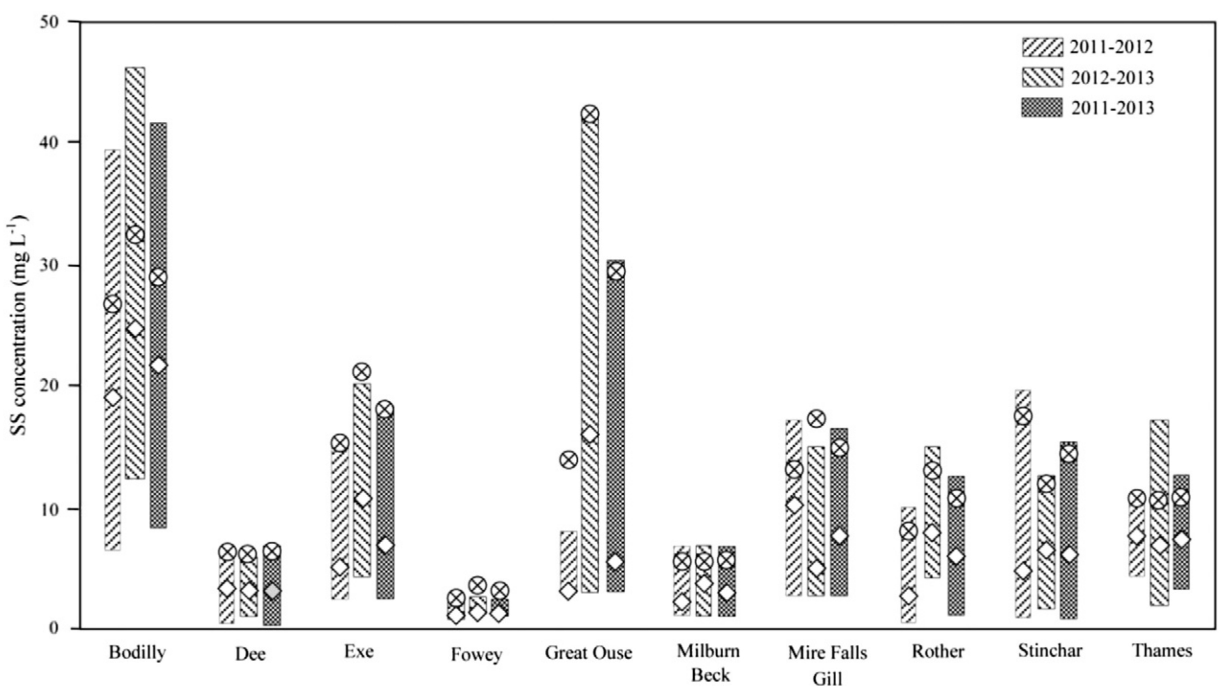

Fig. 4. Estimated (turbidity-derived) SS concentrations for the ten study sites based on high-resolution (15-min frequency) monitoring. Data displayed for 2011–2012, 2012–2013 and 2011–2013. Circle symbols represent the mean SS values. Diamond symbols represent the median SS values.

Table 4
Average river discharge for study sites for 2011–2012, 2012–2013, 2011–2013 and the long-term historical average (CEH, 2014).

River	Mean discharge ($m^3\ s^{-1}$)			Percentage difference between 2011–2012 and 2012–2013	Historical mean discharge[a] ($m^3\ s^{-1}$) and period of record (in parenthesis)
	2011–2012	2012–2013	2011–2013		
Bodilly	0.14	0.30	0.22	114.1	N/A (N/A)
Dee	17.27	13.91	15.61	−19.4	15.7 (1982–2012)
Exe	8.89	11.85	10.37	33.3	12.7 (1960–2012)
Fowey	0.20	0.34	0.28	68.2	N/A (N/A)
Great Ouse	6.37	19.56	13.18	207.0	11.5 (1972–2011)
Milburn Beck	0.10	0.14	0.12	40.9	N/A (N/A)
Mire Falls Gill	0.05	0.11	0.08	119.3	0.1 (1974–2011)
Rother	1.42	4.42	2.83	211.0	3 (1962–2012)
Stinchar	3.22	2.77	2.99	−13.9	2.6 (1973–2012)
Thames	16.34	55.90	36.08	242.0	34.5 (1938–2012)

[a] Data are area-corrected from the National River Flow Archive's supplied by the Centre for Ecology and Hydrology where available (2013).

The EU WFD recommends that sampling for the purposes of surveillance monitoring of surface waters should be conducted a minimum of four times a year, but it allows the Member States an option 'to tailor their monitoring frequencies according to the conditions and variability within their own waters' (EC, 2003). The WFD also recognises that sampling programmes will differ greatly between WQ elements, water body types, areas and countries. It also acknowledges that if the quality elements that are to be monitored are temporally variable or spatially discontinuous, more intensive sampling may be required. The observations of this study suggest that SS may be an example of one such temporally variable or spatially discontinuous water quality parameter. In this study the low-resolution sampling error of up to 53% was translated into a maximum absolute error of 8.2 mg L^{-1}, which some might consider to be relatively small. However, it is important to note that this study monitored reference-condition rivers. The same percentage error on a polluted river with higher mean annual SS concentrations would result in larger absolute errors and potentially significant uncertainties in the assessments of compliance (Skarbøvik et al., 2012).

Table 5
Estimated mean SS concentrations in the study sites based on 15 min resolution and simulated monthly resolution sampling regimes.

		Mean SS concentration (mg L^{-1})		% Errors	Absolute difference
		15 min resolution	Monthly resolution	Between mean SS concentration (mg L^{-1}) sampled every 15 min and monthly	
Bodilly	Mean	29.00	23.07	−20.5	5.9
	St. Dev.	28.17	18.48		
Dee	Mean	6.42	7.15	11.4	0.7
	St. Dev.	13.17	10.38		
Exe	Mean	18.28	19.17	4.8	0.9
	St. Dev.	40.49	38.32		
Fowey	Mean	3.00	2.99	−0.3	0.0
	St. Dev.	10.21	3.57		
Great Ouse	Mean	29.38	27.17	−7.5	2.2
	St. Dev.	61.44	39.14		
Milburn Beck	Mean	5.71	3.25	−43.0	2.5
	St. Dev.	9.20	3.78		
Mire Falls Gill	Mean	15.34	23.50	53.1	8.2
	St. Dev.	28.00	43.26		
Rother	Mean	10.85	5.66	−47.8	5.2
	St. Dev.	17.66	4.69		
Stinchar	Mean	14.56	8.34	−42.7	6.2
	St. Dev.	28.52	10.17		
Thames	Mean	10.90	12.43	14.1	1.5
	St. Dev.	15.01	11.87		

5. Conclusions

Mean SS concentrations, measured between May 2011 and May 2013, in the ten contrasting reference-condition river sites varied up to 9-fold. The degree of spatial variation in estimated mean SS concentrations (3–29 mg L^{-1}), supports earlier research by Bilotta et al. (2012) that analysed the mean SS concentrations derived from low-resolution (monthly grab sample) monitoring programmes in 638 reference-condition sites in the UK (grouped into 42 ecosystem-types or 'end groups'). Bilotta et al. (2012) reported that the mean SS concentrations for the 42 end groups varied between 1.7 and 26 mg L^{-1}. The spatial variation in mean SS could be predicted using the environment-specific SS prediction model developed by Bilotta et al. (2012); which, when applied to the ten study sites, predicted the correct or next closest SS range for all of the sites.

Analysis of temporal variation of SS showed that there are statistically significant differences in the mean annual SS concentrations observed between the two study years. The greatest difference was observed in the Great Ouse river site, which experienced a three-fold increase between year one (14 mg L^{-1}) and year two (42 mg L^{-1}). This difference corresponds well with increases in mean annual discharge observed at this site in year one (6.37 $m^3\ s^{-1}$) and two (19.56 $m^3\ s^{-1}$). These temporal variations in discharge and SS are natural phenomena that shape biological communities and therefore it is important that water quality guidelines acknowledge this. This inter-annual variability in mean annual SS concentration could be predicted reasonably well (see Fig. S1a and S1b) for all the sites except the River Rother, using the Bilotta et al. (2012) environment-specific SS prediction model, but with modified input data to take into account the mean annual temperature (°C) and total annual precipitation (mm) in the year for which the mean SS prediction is to be made.

This study assessed the accuracy of the sampling frequency that is currently used as part of routine regulatory water quality monitoring. The results demonstrate that mean SS concentrations derived from simulated monthly sampling frequencies suffered from errors of up to 53% compared to 15-min resolution monitoring, with an average error of 24% for all the ten sites. These errors vary in both directions and there is no simple correction that can be applied. Water quality managers should place lower confidence in the assessments of compliance with water quality guidelines, when mean annual concentrations are based on monthly-resolution sampling. The observations of this study suggest that SS is a temporally variable and spatially discontinuous water quality parameter that requires higher-resolution monitoring in order to make an accurate assessment of SS exposure. A 24/7 sampling regime (one sample every 7 h) may be an optimum sampling strategy for SS in terms of accuracy (mean SS concentrations calculated from this regime are, on average, within 2% of the mean SS concentrations calculated from 15-min frequency data) and practicality (samples can be collected by an automated sampler).

The authors are not suggesting that high-resolution SS monitoring should be conducted on a routine basis in all rivers in order to assess compliance with water quality guidelines. This would be prohibitively expensive. Rather, a better system for regulating SS would be to use low-resolution (lower-cost) biological monitoring to determine the ecological status of river sites on a periodic basis (as is current practice across the EU for a number of other water quality parameters) (Turley et al., 2014). At sites where the biological community composition deviates significantly from the reference-condition community composition for that ecosystem-type, high-resolution water quality monitoring could be targeted and used to confirm the likely cause of the biological deviation. If the SS regime differs significantly from that observed in equivalent ecosystems that are in reference condition, then it is likely that SS pollution is occurring and mitigation may be needed. In terms of improving the water quality guidelines for SS, the results of this study demonstrate the need for the more advanced guidelines that take into account natural spatial and temporal variations. One way to achieve this might be to develop environment-specific concentration–duration–frequency guidelines, using high-resolution data from ecosystem-scale observations in reference-condition sites, combined with existing data from microcosm to ecosystem-scale biological/ecological experiments. For the assessments of compliance against modelled guidelines, the resolution of SS monitoring at a given site, must be similar to the resolution of SS data used to develop these guidelines.

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2014.09.054.

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References

Alabaster JS, Lloyd R. Finely divided solids. In: Alabaster JS, Lloyd R, editors. Water Quality Criteria for Freshwater Fish. London: Butterworth Scientific; 1982. p. 1–20.

APHA. Standard methods for the examination of water and wastewater. Washington DC, USA: American Public Health Association/American Water Works Association/Water Environment Federation; 1995.

APHA. Standard methods for the examination of water and wastewater. Washington, DC: American Public Health Association; 1998.

Bilotta G, Brazier R. Understanding the influence of suspended solids on water quality and aquatic biota. Water Res 2008;42:2849–61.

Bilotta GS, Burnside NG, Cheek L, Dunbar MJ, Grove MK, Harrison C, et al. Developing environment-specific water quality guidelines for suspended particulate matter. Water Res 2012;46:2324–32.

Bowes MJ, Smith JT, Neal C. The value of high-resolution nutrient monitoring: a case study of the River Frome, Dorset, UK. J Hydrol 2009;378:82–96.

Clarke R, Davy-Bowker J, Dunbar M, Laize C, Scarlett P, Murphy J. Enhancement of the river invertebrate classification tool. Final Report, Project WFD119; 2011.

Clesceri LS, Greenberg AE, Eaton AD. Standard methods for the examination of water and wastewater. Washington, DC.: American Public Health Association; 1998

Collins A, Naden P, Sear D, Jones J, Foster ID, Morrow K. Sediment targets for informing river catchment management: international experience and prospects. Hydrol Process 2011;25:2112–29.

Cooper D, Naden P, Smith B, Gannon B. Siltation in rivers. 2: a minimum monitoring strategy for two cSAC rivers. Conserving Natura 2000 Rivers Conservation Techniques Series. Peterborough: English Nature; 2003.

Cordone AJ, Kelley DW. The influences of inorganic sediment on the aquatic life of streams. California Department of Fish and Game; 1961.

Costanza R, d'Arge R, De Groot R, Farber S, Grasso M, Hannon B, et al. The value of the world's ecosystem services and natural capital. Ecol Econ 1998;25:3–15.

Coynel A, Schäfer J, Hurtrez J-E, Dumas J, Etcheber H, Blanc G. Sampling frequency and accuracy of SPM flux estimates in two contrasted drainage basins. Sci Total Environ 2004;330:233–47.

Dunn OJ. Multiple comparisons using rank sums. Technometrics 1964;6:241–52.

Diehl TH, Wolfe WJ. Suspended-sediment concentration regimes for two biological reference streams in Middle Tennessee. J Am Water Res Assoc 2010;46:824–37.

EC. EC Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 of establishing a framework for community action in the field of water policy; 2003. p. 1–160.

Gammon JR. Effect of inorganic sediment on stream biota. Vol 18050 DWC 12/70. Washington, DC: USGPO, . USEPA: Water Pollution Control Research Series; 1970.

Garrido A, Dinar A. Managing water resources in a time of global change: mountains, valleys and flood plains. London: Routledge; 2009.

Giorgi F, Bi X, Pal J. Mean, interannual variability and trends in a regional climate change experiment over Europe. II: climate change scenarios (2071–2100). Climate Dynam 2004;23:839–58.

Gray NF. Drinking water quality. Cambridge University Press; 2008.

Harvey GL, Moorhouse TP, Clifford NJ, Henshaw AJ, Johnson MF, Macdonald DW, et al. Evaluating the role of invasive aquatic species as drivers of fine sediment-related river management problems: the case of the signal crayfish (*Pacifastacus leniusculus*). Prog Phys Geogr 2011;35:517–33.

Hassan MA, Gottesfeld AS, Montgomery DR, Tunnicliffe JF, Clarke GK, Wynn G, et al. Salmon-driven bed load transport and bed morphology in mountain streams. Geophys Res Lett 2008;35.

Kerr SJ. Silt, turbidity and suspended sediments in the aquatic environment: an annotated bibliography and literature review: Ontario Ministry of Natural Resources. Science and Technology Transfer Unit Technical Report TR-008; 1995.

Ladle M, Griffiths B. A study on the faeces of some chalk stream invertebrates. Hydrobiologia 1980;74:161–71.

Lloyd DS. Turbidity as a water quality standard for salmonid habitats in Alaska. N Am J Fish Manag 1987;7:34–5.

Maitland PS. Ecology of the river, brook and Sea Lamprey. Conserving Natura 2000 Rivers Ecology Series No. 5. English Nature, Peterborough. Peterborough: English Nature; 2003.

Montgomery DR, Buffington JM, Peterson NP, Schuett-Hames D, Quinn TP. Stream-bed scour, egg burial depths, and the influence of salmonid spawning on bed surface mobility and embryo survival. Can J Fish Aquat Sci 1996;53:1061–70.

Newcombe CP, Jensen JO. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. N Am J Fish Manag 1996;16:693–727.

Newcombe CP, MacDonald DD. Effects of suspended sediments on aquatic ecosystems. N Am J Fish Manag 1991;11:72–82.

Owens P, Batalla R, Collins A, Gomez B, Hicks D, Horowitz A, et al. Fine-grained sediment in river systems: environmental significance and management issues. River Res Appl 2005;21:693–717.

Peddicord R. Direct effects of suspended sediments on aquatic organisms. Abstracts of papers of the American Chemical Society. 1155 16th Street NW, Washington DC 20036: American Chemical Society; 1979. p. 38.
Perks MT, Warburton J, Bracken L. Critical assessment and validation of a time-integrating fluvial suspended sediment sampler. Hydrol Process 1979.

Perks MT, Warburton J, Bracken L. Critical assessment and validation of a time-integrating fluvial suspended sediment sampler. Hydrol Process 2013;28(17):4795–807.

Phillips J, Russell M, Walling D. Time-integrated sampling of fluvial suspended sediment: a simple methodology for small catchments. Hydrol Process 2000;14:2589–602.

Poff NL. Managing for variability to sustain freshwater ecosystems. J Water Res Plan Manag 2009;135:1–4.

Quinn JM, Davies-Colley RJ, Hickey CW, Vickers ML, Ryan PA. Effects of clay discharge on streams: 2. Benthic invertebrates. Hydrobiologia 1992;248:235–47.

Räisänen J, Hansson U, Ullerstig A, Döscher R, Graham L, Jones C, et al. European climate in the late twenty-first century: regional simulations with two driving global models and two forcing scenarios. Climate Dynam 2004;22:13–31.

Reid SM, Anderson PG. Effects of sediment released during open-cut pipeline water crossings. Can Water Res J 1999;24:235–51.

Richter BD, Braun DP, Mendelson MA, Master LL. Threats to imperiled freshwater fauna. Conserv Biol 1997;11:1081–93.
Robertson MJ, Scruton DA, Clarke KD. Seasonal effects of suspended sediment on the behaviours of juvenile Atlantic salmon. Trans Am Fish Soc 2007;136(3):822–8.
Rosenberg DM, Wiens AP. Effects of sedimentation on macrobenthic invertebrates on a northern Canadian river. Water Res 1978;12:753–63.
Ryan PA. Environmental effects of sediment on New Zealand streams: a review. N Z J Mar Freshw Res 1991;25:207–21.
Schwartz JS, Dahle M, Robinson RB. Concentration–frequency–duration curves for stream turbidity: possibilities for use assessing biological impairment. J Am Water Resour Assoc 2008;44(4):879–86.
Schwartz JS, Simon A, Klimetz L. Use of fish functional traits to associate in-stream suspended sediment transport metrics with biological impairment. Environ Monit Assess 2011;179:347–69.
Skarbøvik E, Stålnacke P, Bogen J, Bønsnes TE. Impact of sampling frequency on mean concentrations and estimated loads of suspended sediment in a Norwegian river: implications for water management. Sci Total Environ 2012;433:462–71.
Staddon C. Managing Europe's water resources: twenty-first century challenges. Ashgate Publishing, Ltd.; 2012
Swietlik W, Berry W, Gardner T, Hill B, Jha M, Kaufmann P. Developing water quality criteria for suspended and bedded sediments (SABS): potential approaches. Washington, DC.: US EPA, Office of Water; 2003
Thompson J, Cassidy R, Doody DG, Flynn R. Assessing suspended sediment dynamics in relation to ecological thresholds and sampling strategies in two Irish headwater catchments. Sci Total Environ 2014;468–469:345–57.
Trenberth KE, Dai A, Rasmussen RM, Parsons DB. The changing character of precipitation. Bull Am Meteorol Soc 2003;84.
Turley M, Bilotta GS, Brazier RE, Extence C. Evaluation of a fine sediment biomonitoring tool across a wide range of temperate rivers and streams. Freshw Biol 2014. http://dx.doi.org/10.1111/fwb.12429.
Vannote RL, Minshall GW, Cummins KW, Sedell JR, Cushing CE. The river continuum concept. Can J Fish Aquat Sci 1980;37:130–7.
Ward G, Ward A, Dahm C, Aumen N. Origin and formation of organic and inorganic particles in aquatic systems. The biology of particles in aquatic systems; 1994. p. 45–73.
Waters TF. Sediment in streams: sources, biological effects and control. Am Fish Soc Monogr, Bethesda, Maryland 1995;7:251.
Wildhaber YS, Michel C, Burkhardt-Holm P, Banninger D, Alewell C, Van Meerveld I. Measurement of spatial and temporal fine sediment dynamics in a small river. Hydrol Earth Syst Sci 2012;16:1501–15.
Wood PJ, Armitage PD. Biological effects of fine sediment in the lotic environment. Environ Manage 1997;21:203–17.
Wright JF, Sutcliffe DW, Furse MT. Assessing the biological quality of freshwaters. RIVPACS and other techniques. Ambleside, England: Freshwater Biological Association; 2000.
Yarnell SM, Mount JF, Larsen EW. The influence of relative sediment supply on riverine habitat heterogeneity. Geomorphology 2006;80:310–24.
Yount JD, Niemi GJ. Recovery of lotic communities and ecosystems from disturbance — a narrative review of case studies. Environ Manage 1990;14:547–69.