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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.