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Fine sediment transport and management

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CHAPTER 3

Fine sediment transport and management

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Background and context

Traditionally, studies of sediment transport by rivers have distinguished the coarse bedload component from the finer suspended load. The latter component is often further subdivided into a coarser fraction, designated the suspended bed material load, and a finer fraction termed the wash load (Shen, 1981).The wash load is commonly assumed to be derived from the catchment surface, to be rapidly transported through the channel system and to have limited interaction with the channel bed. As such it was generally seen by hydraulic engineers as having limited importance for river morphology and river management. By virtue of its source outside the river channel and the fact that most rivers can transport a much greater wash load than is actually transported, the wash load differs from the suspended bed material and bedload in that it is a non-capacity load that is supply controlled, rather than being controlled by the transport capacity of the river. This means that it is difficult to predict using hydraulic variables, and it was commonly excluded from theoretical treatments of sediment transport as being something that needed to be measured, should it prove important. Against this background, fine sediment transport by rivers traditionally received relatively little attention, compared with the coarser load, except where reservoir sedimentation was a potential problem or such information was used to assess rates of soil loss or land degradation (e.g., Graf, 1971; Shen, 1981).

Two developments changed this situation and directed increased attention to fine sediment transport by rivers. The first, which can be traced to the 1970s and 1980s, was the increasing recognition of the importance of fine sediment as a vector for the transfer of nutrients and contaminants through river systems (see Förstner and Muller, 1974; Golterman, 1977; Golterman *et al.*, 1983; Allan, 1986). Fine sediment particles are highly active chemically and act as a substrate for the adsorption of nutrients, particularly phosphorus (P), and many contaminants such as heavy metals, pesticides and other persistent organic pollutants (POPs). Sediment-associated transport can exert a key control on the transfer and fate of such substances within fluvial systems and an understanding of fine sediment transport and loads is an essential pre-requisite for understanding and controlling nutrient and contaminant fluxes and diffuse source

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pollution. This was well demonstrated by the pioneering work of the joint US–Canada International Commission on the Great Lakes (IJC) and its Pollution from Land Use Activities Reference Group (PLUARG) in the 1970s. This aimed to reduce eutrophication and pollution in Lake Erie and Lake Ontario and identified the need to reduce the mobilisation of sediment from agricultural land and its transport to the lakes (PLUARG, 1978).

The second development is linked to the above and reflects the growing recognition of the wider ecological importance of fine sediment in degrading aquatic and riparian ecology and habitats. This degradation is partly a response to the pollutants that are frequently associated with fine sediment, but can also reflect the physical impact of excessive amounts of fine sediment. The latter can, for example, involve reduced light transmission and smothering of the stream bed and aquatic vegetation. The silting of fish spawning gravels, which reduces the flow of water through the gravels and the supply of oxygen to the eggs (Heywood and Walling, 2007; Sear *et al.*, 2014), is another example. There are, however, many other ways in which fine sediment can impact adversely on aquatic ecology (see Chapman *et al.*, 2014; Collins *et al.*, 2011; Jones *et al.*, 2012a,2012b, 2014; Kemp *et al.*, 2011; Thompson *et al.*, 2014; Von Bertrab *et al.*, 2013; Wagenhoff *et al.*, 2013; Wood and Armitage, 1997).

The environmental problems outlined above highlight the potential role of fine sediment as a pollutant and this has been recognised in the EU within the Water Framework (European Parliament, 2000), Freshwater Fish (European Parliament, 2000) and Habitats Directives (European Council, 1992) and by the US Environmental Protection Agency (EPA) through the introduction of Total Maximum Daily Load (TMDL) standards (Hawkins, 2003). These problems have in turn directed increased attention to managing fine sediment mobilisation and transport and this has been coupled with a changing view of the significance of load magnitude. In the traditional hydraulic engineering context, linked to reservoir and channel sedimentation and land degradation, problems generally increased as sediment yields increased. In the wider ecological context, however, rivers with low sediment loads are often the most sensitive to small changes in fine sediment concentrations or load and such rivers can experience greater problems and necessitate more intensive management than those draining areas with higher sediment yields (Collins and Anthony, 2008a).

Key concepts

In seeking to develop an improved understanding of the fine sediment loads of rivers and to ultimately manage such loads, four key concepts can usefully be emphasised. These are, firstly, the non-capacity and supply-controlled nature of fine sediment transport, secondly, the significance of grain size, sediment composition and composite particles, thirdly, the importance of sediment source and finally the need to view the fine sediment load of a river as a component of the overall catchment sediment budget. These concepts will be briefly considered in turn.

Non-capacity supply controlled transport.

As indicated above, fine sediment or wash load transport differs from the transport of coarser sediment in that it cannot be treated as a capacity load. The supply is generally far more important than the transport capacity in determining the magnitude of the load. Such behaviour is clearly demonstrated by Figure 3.1. Figure 3.1a illustrates the variation of suspended sediment concentration during a sequence of storm hydrographs monitored at the outlet of the 46 km^2 catchment of the River Dart in Devon, UK. The data demonstrate that the sediment concentration and discharge peaks are out of phase and that the supply can be depleted and subsequently replenished during a sequence of events. Figure 3.1b presents the suspended sediment rating curve or plot of suspended sediment concentration versus discharge for the 262 km^2 catchment of the River Creedy at Cowley in Devon, UK. Suspended sediment concentrations can be seen to range over more than two orders of magnitude for a given water discharge or transport capacity and the sediment concentrations associated with a given discharge are significantly higher in summer than in winter and are generally higher on the rising stage than on the falling stage.

Sediment grain size, composition and composite particles

Recognition of the important role of fine sediment in the transport of nutrients and contaminants and its potential impact in degrading aquatic ecosystems has significantly expanded information requirements. In addition to information on the magnitude of fine sediment concentrations and loads, there is also a need for information on the properties, composition and structure of the sediment particles. Grain size composition exerts a key influence on sediment-associated transport, since clay- and fine silt-sized particles are generally more chemically active than larger particles (Horowitz, 1991). Likewise, the presence of organic matter, either as discrete particles, surface coatings or more complex associations with inorganic particles, can exert a key influence on the role of fine sediment as a substrate for contaminant transport. The complex nature of fine sediment transport is further emphasised by the fact that few particles will exist in isolation. Most will be transported as composite particles or flocs, comprising large numbers of smaller particles of mineral or organic matter and with highly complex structures (see Droppo, 2001; Droppo *et al.*, 2005). The individual components of flocs may be held together by several mechanisms, including electrochemical forces and sticky material and filaments associated with bacteria and extracellular polymeric substances (EPS). Figure 3.2 presents highly magnified images of several suspended sediment particles, which emphasise their complex structure. Traditional grain size analyses undertaken in the laboratory generally involve removal of organic matter and chemical and physical dispersion of the particles. The results may therefore bear little relation to the actual *in situ* or effective particle size of the particles transported by a river and any attempt to understand the hydrodynamic behaviour of suspended sediment particles must take this into account (Williams *et al.*, 2008).

The importance of sediment source

The need to understand sediment properties and the role of fine sediment in nutrient and contaminant transport necessarily directs attention to the importance of sediment source in influencing these key aspects. Source can be defined in terms of both spatial location within the upstream catchment (e.g., areas of contrasting geology or different sub-catchments) or source type, which reflect the processes responsible for sediment mobilisation and the related source areas. The latter could, for example, include

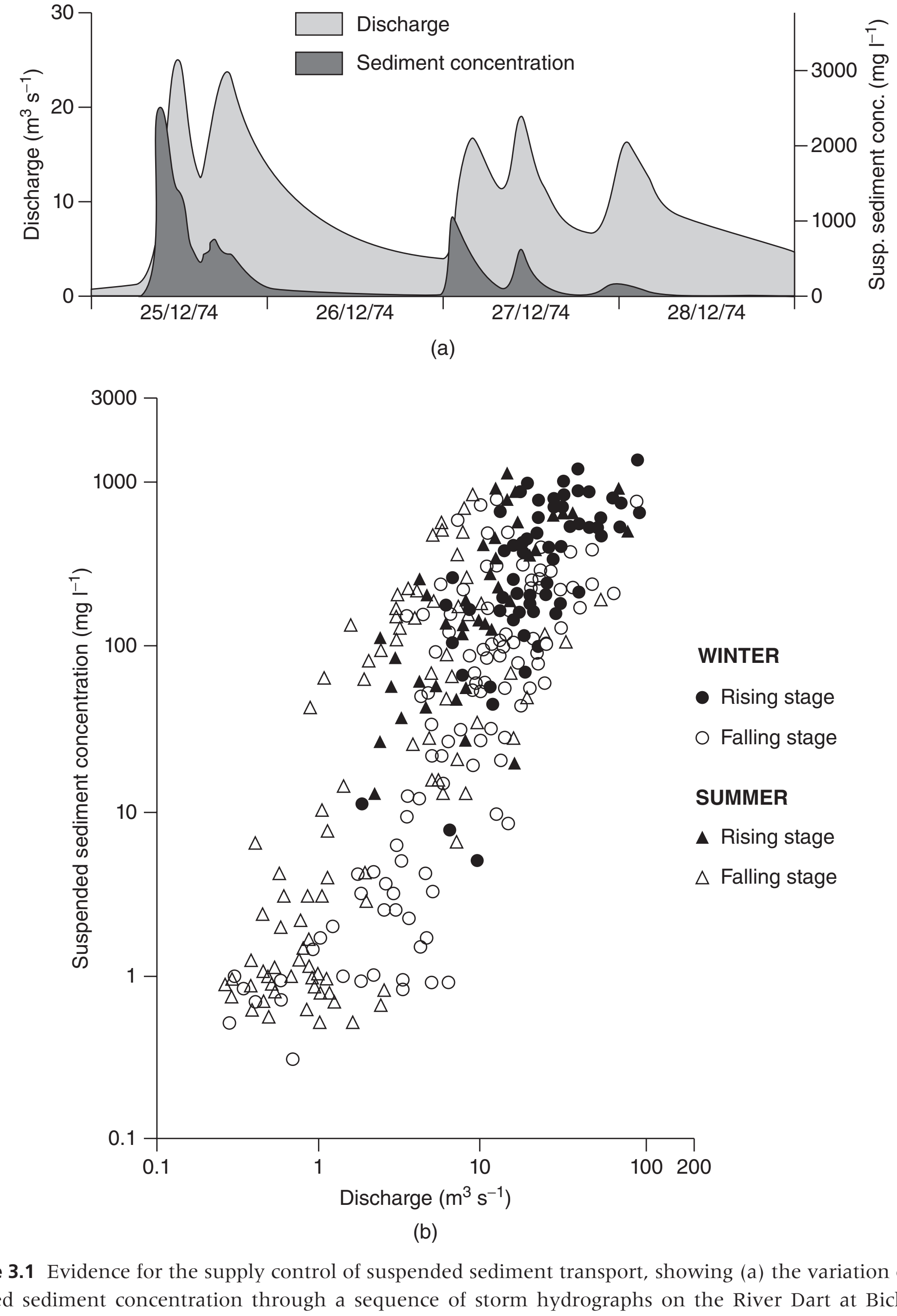

Figure 3.1 Evidence for the supply control of suspended sediment transport, showing (a) the variation of suspended sediment concentration through a sequence of storm hydrographs on the River Dart at Bickleigh, Devon, UK and (b) the relationship between suspended sediment concentraton and discharge for the River Creedy at Cowley, Devon, UK for the period 1972–74.

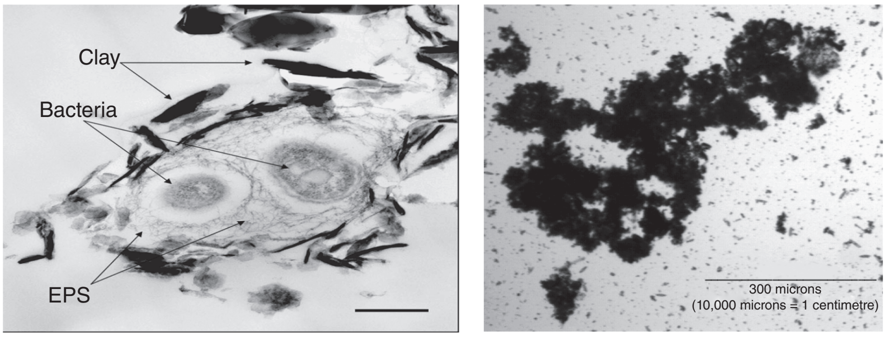

Figure 3.2 Micrographs of suspended sediment particles depicting (left) a small floc (scale bar = 0.5 μm) and (right) a group of larger flocs. (Source, Ian Droppo, Environment Canada.)

channel erosion, gully erosion and erosion of surface soils from areas under cultivation or pasture by sheet and rill erosion. In some catchments, roads and urban areas, point sources and effluent from sewage treatment can also represent important sources of fine particulates. Sediment sources can change both seasonally and between and during events and such changes can result in significant changes in sediment properties, including grain size (e.g., Ongley *et al.*, 1982). Information on the source of the sediment transported by a river is also likely to be of critical importance when developing sediment control or management strategies. To be effective and to maximise the return on expenditure, such strategies must target the most important sources (Gellis and Walling, 2011). Information on sediment source is difficult to obtain using traditional monitoring techniques, but recent advances in sediment source fingerprinting (Walling, 2013) have provided the means to obtain such information and this technique will be discussed further below.

Catchment sediment budgets

It is important to recognise that the fine sediment output from a catchment represents the result of a complex interaction of sediment mobilisation from a variety of sources within the catchment, the transfer of that sediment to and through the channel system, and the temporary and longer-term storage of the sediment as it moves through the sediment delivery continuum. This system must be understood if fine sediment transport and yields, and changes resulting from climate change or changing land use and land management, are to be successfully predicted. Much of the sediment mobilised from the upstream catchment area may not reach the catchment outlet. Equally, sediment yields could change as a result of remobilisation of stored sediment. The catchment sediment budget, as proposed by Trimble (1983) and illustrated in Figure 3.3 for the now classic example of Coon Creek, Wisconsin, USA, affords a valuable conceptual tool for representing this complex interaction of sources and sinks. In the 360 km^2 Coon Creek catchment, only ~5–7% of the sediment mobilised within the basin reached the basin outlet, with the remainder being stored within the catchment. Reduction in rates of soil loss from the agricultural areas in the catchment by about 25% after 1938, as a result of the implementation of soil conservation measures, was not reflected by reduced sediment output from the catchment. This was largely because of remobilisation of

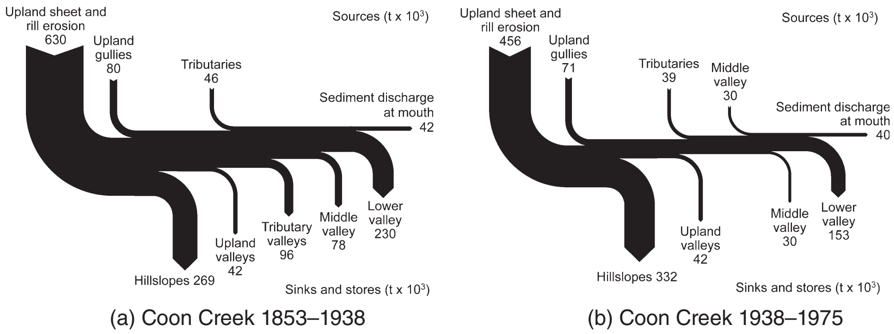

Figure 3.3 Sediment budgets for Coon Creek Wisconsin, for the periods 1853–1938 (a) and 1938–75 (b). (Based on data presented by Trimble, 1983.)

sediment from sinks in the middle valley. The sediment budget must be seen as key tool both for understanding sediment export from a catchment and, perhaps even more importantly, for supporting the design and implementation of effective sediment management programmes (Walling and Collins, 2008; Gellis and Walling, 2011).

Tools for meeting new information needs

Increased interest in the fine sediment loads of rivers has been paralleled by the development of a range of tools for meeting requirements for new information. These developments partly reflect the need to address new questions, but they are also a reflection of timely technological advances. They span improved monitoring techniques and equipment, sediment source fingerprinting, sediment tracing and modelling fine sediment yields across a range of temporal and spatial scales and for a range of purposes. These areas are reviewed below.

Monitoring techniques and equipment

The non-capacity and supply-controlled nature of fine sediment transport (e.g., Figure 3.1) means that carefully designed monitoring programmes are necessary to obtain reliable information on suspended sediment transport (Walling *et al.*, 1992). In large rivers, where discharge and sediment concentration change relatively slowly, a programme of daily sampling might be sufficient to define the record of variations in suspended sediment concentration or load. However, as the size of the catchment reduces and the response to rainfall becomes more flashy, the frequency of sampling needs to increase. Most of the suspended sediment load of a stream or river is transported during storm events and primarily by the large events. Typically, about 75% of the load is transported during about 5% of the time and it is critical that the sediment concentration record should be documented in detail during the key events. Suspended sediment samplers and sampling techniques are now well developed (see Gray *et al.*, 2008), but the need to visit the site can make it difficult to assemble a detailed record of suspended sediment concentration. The development of automatic samplers has provided a means of overcoming this problem, although problems can arise in ensuring that the sample collected is representative of the channel cross section. Such samplers can be programmed to collect suspended sediment samples when flow or concentration (turbidity) exceeds a pre-set threshold and to vary the sampling frequency according to the rate of change of flow or turbidity (e.g., Lewis, 1996). Recording turbidity meters, which now commonly employ optical backscatter (OBS) sensors, also offer a means of collecting a continuous surrogate record of suspended sediment concentration (e.g., Gray and Gartner, 2010; Schoellhamer and Wright, 2003) and are widely employed for monitoring suspended sediment transport. This approach is, however, heavily dependent on the existence of a well-defined calibration relationship between sediment concentration and turbidity and this relationship can be affected by changes in the grain size composition and colour of the sediment load (Sutherland *et al.*, 2000). The time integrating trap sampler developed at the University of Exeter (Phillips *et al.*, 2000; Russell *et al.*, 2000) is a very simple device which has met an important need for the automated collection of sizeable representative samples of suspended sediment for use in sediment fingerprinting investigations. Where large instantaneous samples of suspended sediment are required for subsequent analysis, continuous flow centrifuges have proved an effective means of dewatering and recovering the sediment (see Ongley and Blatchford, 1982).

The need for easily derived information on the grain size composition of suspended sediment samples has been addressed by the development of laboratory laser diffraction analysers. However, as indicated above, the grain size distribution measured in the laboratory may differ significantly from the *in situ* or effective distribution that exists in the river, due to the presence of flocs or composite particles, which are likely to be broken up during the laboratory measurements (Phillips and Walling, 1995). As a result, attention has been successfully directed to the *in situ* deployment of laser diffraction or scattering probes (e.g., Phillips and Walling, 1997; Gray *et al.*, 2004; Williams *et al.*, 2007). The current generation of LISST laser-based equipment developed by Sequoia Scientific specifically for river studies includes an *in situ* laser probe contained in a streamlined body (LISST-SL) and a portable battery powered streamside monitoring unit that pumps water directly from the river and which can be programmed to make measurements at intervals of between 5 minutes and 60 minutes (LISST-Streamside).

Sediment source fingerprinting

There is an increasing need for information on the source of the fine sediment transported by a river. Such information is essentially impossible to obtain using traditional monitoring techniques, but the development of sediment source fingerprinting techniques has provided a timely and effective means of meeting this need. Sediment source fingerprinting is founded on two key principles. Firstly, one or more diagnostic physical or chemical properties are used as fingerprints to discriminate the source materials associated with the potential fine sediment sources in a catchment. Secondly, comparison of the equivalent properties of the suspended sediment transported by a river with the fingerprints of the potential sources provides a means of establishing the relative contribution of the individual sources. Use of this approach can be traced back to the 1970s and the work of researchers such as Klages and Hsieh (1975), Wall and Wilding (1976) and Walling *et al.* (1979). However, the assessment of the relative importance of different sources provided by these early studies was essentially qualitative. Since then, the approach has been successfully developed and refined, with most emphasis being placed on determining the relative importance of different source types. Following Walling (2013), seven key developments which have been incorporated into current approaches, can be identified as follows:

(1) Use of multiple properties or composite fingerprints, involving a wide range of different physical and chemical properties, to strengthen the discrimination between different sources and to permit a greater number of potential sources to be identified. Sediment properties that have now been successfully used as source fingerprints include a wide range of geochemical parameters, isotopic signatures, radionuclides, sediment colour and spectral reflectance and compound specific stable isotopes (e.g., Collins *et al*. 2010a; Douglas *et al*., 2003; Gibbs, 2008; Martínez-Carreras *et al*., 2010; Tiecher *et al*., 2015; Wallbrink *et al*., 1998).

(2) Incorporation of statistical tests to confirm the ability of particular fingerprint properties to discriminate between potential sediment sources and to assist in the selection of the 'best' composite fingerprint (e.g., Collins *et al*., 2012; Juracek and Ziegler, 2009; Laceby *et al*., 2015; Motha *et al*., 2003).

(3) Use of numerical mixing (or unmixing) models to provide quantitative assessments of the relative contribution of different potential sources (e.g., Collins *et al*., 2010a; Fox and Papanicolaou 2008; Haddachi *et al*., 2014; Lamba *et al*., 2015; Lin *et al*., 2015; Nosrati *et al*., 2014; Palmer and Douglas, 2008).

(4) Use of specific size fractions to take account of contrasts in grain size composition between suspended sediment and catchment source materials, testing fingerprint properties for conservative behaviour and incorporation of grain size and organic matter enrichment/depletion effects into the mixing models used for source apportionment (e.g., Collins *et al*., 1998, 2013a,b; Motha *et al*., 2003, Russell *et al*., 2001).

(5) Extension of the approach to consider a wider range of 'targets', in addition to samples of suspended sediment. These include surrogates for suspended sediment, such as floodplain surface scrapes and fine sediment deposits from river channels (e.g., Collins *et al*., 2010a), particular 'problem sediments', such as interstitial fine sediment recovered from fish spawning gravels (e.g., Walling *et al*., 2003) and recent fine sediment deposits from lakes and estuaries (e.g., Gibbs, 2008; Haiyan, 2015). In some studies attention has focused on the source of the organic material associated with the sediment (Collins *et al*., 2013c, 2014).

(6) Extension of the approach to incorporate a temporal dimension and to document changes in sediment source through time. Such work has included both 'before and after' studies in experimental catchments where sediment control measures and changes in land management have been implemented (e.g. Merten *et al*., 2010) and use of sediment cores collected from lakes and river floodplains to reconstruct longer-term changes in sediment source (e.g., Foster and Walling, 1994; Pittam *et al*., 2009; Collins *et al*., 2010b).

(7) Taking account of the uncertainty associated with source apportionment procedures. Incorporation of Monte Carlo procedures and Bayesian statistics into the mixing models used to determine the relative contributions of potential sources has permitted the uncertainty associated with source characterisation and other components of the source fingerprinting approaches to be propagated through the calculations (e.g., Franks and Rowan, 2000; Collins *et al*., 2012, 2014; Laceby and Olley, 2015; Nosrati *et al*., 2014; Palmer and Douglas, 2008; Pulley *et al*., 2015).

Sediment source fingerprinting techniques have now been widely applied in Europe, North America and Australia, to support investigations of fine sediment transport by rivers and the development and implementation of sediment management and control programmes. In Australia, a number of studies have been undertaken to establish the primary sources of the fine sediment transported to the coast adjacent to the Great Barrier Reef (GBR) (e.g., Douglas *et al*., 2008; Hughes *et al*., 2009; Wilkinson *et al*., 2011). The GBR is currently under stress from terrestrially derived sediment and information on sediment source is a critical requirement for the design of catchment management programmes aimed at reducing land–sea sediment fluxes.

Tracing soil and sediment redistribution

Production of a contemporary sediment budget for a catchment, similar to that depicted in Figure 3.3, requires information on rates of gross and net soil loss from slopes and the deposition and storage of sediment as it is transported towards the stream and through the channel network. As with sediment source, such information is difficult to obtain using traditional monitoring and sediment tracing techniques have proved to be particularly useful for this purpose (Walling, 2006). Source fingerprinting techniques could be viewed as a tracing technique, but here attention will focus on the more direct use of fallout radionuclides to trace sediment movement and redistribution in catchments. This approach is founded on the existence of a number of natural and manmade radionuclides that reach the land surface as fallout, primarily as wet fallout in association with rainfall, and are rapidly and strongly fixed by the surface soil or sediment. By studying the post-fallout redistribution and fate of the selected fallout radionuclide, it is possible to obtain information on soil and sediment redistribution and, therefore, erosion and deposition rates.

The fallout radionuclide most widely used for this purpose is caesium-137 (^{137}Cs) (see IAEA, 2014; Walling, 2012; Zapata, 2002). Caesium-137 is a manmade radionuclide that was produced by the testing of thermonuclear weapons in the 1950s and early 1960s. Significant bomb-derived fallout occurred in most areas of the world during the period extending from the mid 1950s through to the 1970s, although the depositional fluxes were much greater in the northern than the southern hemisphere. In the absence of further bomb tests after the Nuclear Test Ban Treaty in 1963, fallout effectively ceased in the mid 1970s. However, in some areas of the world a further short-lived fallout input occurred in 1986 as a result of the Chernobyl accident.

Caesium-137 has a half-life of 30.2 years and much of the original fallout is likely to still remain within the upper horizons of the soils and sediments of a catchment. By investigating the current distribution of the radionuclide in the landscape, it is possible to obtain information on the net effect of soil and sediment redistribution processes operating over the past ca. ~50 years (i.e., since the main period of fallout) and thus quantify medium-term erosion and deposition rates. Mean soil redistribution rates over the past ~50 years are established by comparing the inventories measured at individual sampling points with the reference inventory for the study site, which represents the inventory found at a site which has experienced neither erosion nor deposition. Points with inventories less than the reference inventory are indicative of eroding areas, whereas those with inventories in excess of the reference value indicate deposition. The timescale will need to be modified where significant Chenobyl fallout has occurred. A range of conversion models have been developed for use in estimating erosion and deposition rates, based on the degree of departure of the measured inventory from the reference inventory (e.g., Walling and He, 1999a; Walling *et al*., 2011; Li *et al*., 2009). Using a similar approach, ^{137}Cs measurements have also been successfully used to document rates and patterns of overbank deposition on river floodplains over the past ~50 years (Golosov and Walling, 2014; Walling and He, 1997; Terry *et al*., 2002)

Although most studies employing fallout radionuclides have been based on ^{137}Cs, both excess lead-210 ($^{210}Pb_{ex}$) and beryllium-7 (^{7}Be) have also been used in a similar manner (see IAEA, 2014; Mabit *et al*., 2008, 2014; Walling, 2012). These two fallout radionuclides differ from ^{137}Cs in being of natural, geogenic and cosmogenic origin, respectively. Pb-210 has a similar half-life to ^{137}Cs (22.3 years) but that of ^{7}Be is very much shorter (53 days). By virtue of its ongoing fallout, $^{210}Pb_{ex}$ provides a means of assessing soil and sediment redistribution over periods of ~100 years, whereas ^{7}Be can be used at the timescale of individual events or a few weeks. Walling and He (1999b) report the successful use of $^{210}Pb_{ex}$ in soil erosion studies and He and Walling (1996) provide examples of its application for estimating rates of overbank sedimentation on floodplains. The use of ^{7}Be to document short-term soil redistribution rates is reported by Porto *et al*. (2014), Schuller *et al*. (2006) and Walling *et al*. (1999, 2009).

Most studies that have employed fallout radionuclides to document soil and sediment redistribution in catchments have focused on small areas such as individual fields or representative transects and have involved the collection of a substantial number of samples. Extrapolation of the results to larger areas can introduce problems due to restrictions on the number of samples that can be collected and analysed. Increased attention is therefore being directed to the problem of upscaling the approach (see Mabit *et al*., 2007; Walling *et al*., 2014). The approach recently documented by Porto *et al*. (2011) involves sampling an essentially random network of points distributed across a larger area and using the resulting information to provide a representative sample of erosion and deposition rates within the landscape of the study area. This will provide information on both the magnitude of erosion and deposition rates and the relative importance of zones experiencing erosion and deposition (see Figure 3.4).

Modelling sediment yields

There is a long tradition, particularly from engineering disciplines, of modelling the in-channel processes of scour, sediment transport and deposition in alluvial river systems, with the sediment transfer functions reflecting differing levels of complexity and corresponding data requirements. The US Bureau of Reclamation Generalized Stream Tube model for Alluvial River Simulation (GSTARS) is a well-known example of a sediment routing model used for practical engineering purposes (Yang *et al*., 1998). However, such models focus on the coarser channel-derived sediment.

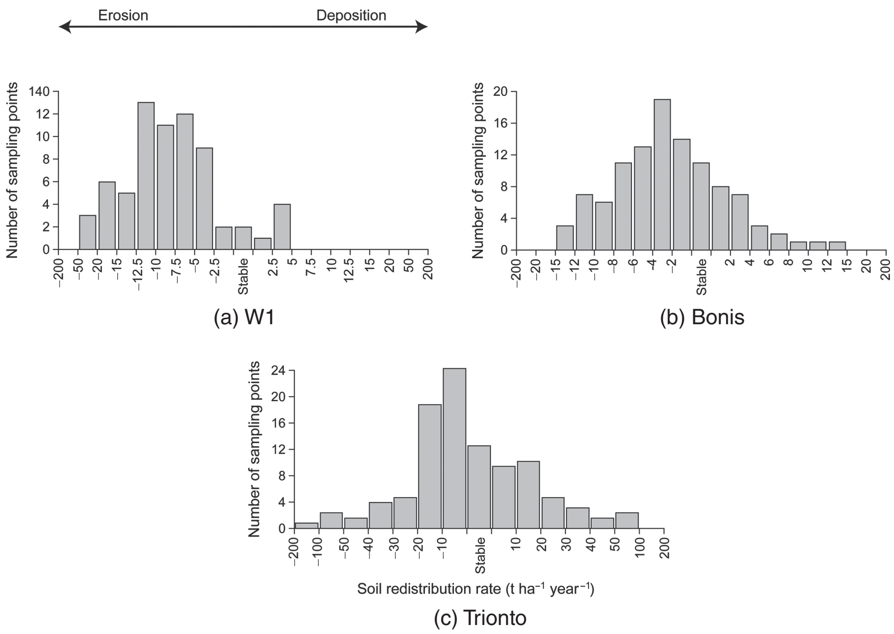

Figure 3.4 Distributions of soil redistribution rates derived from ^{137}Cs measurements in two small catchments (W1 = 0.015 km^2 and Bonis = 1.39 km^2) and an intermediate sized catchment (Trionto = 31.61 km^2) in southern Italy. Porto *et al.* 2011. Reproduced with permission from John Wiley & Sons.

The understanding and management of fine sediment problems requires models that characterise the linkages between the catchment surface and the channel network and can represent the influence of topography, soil type, land use and other factors on sediment mobilisation and delivery. The increased use of Geographical Information Systems (GIS) and digital elevation models (DEMs) has promoted the development and application of spatially distributed, process-based models of soil erosion and sediment delivery that capture many of the key controls involved. Well-known examples of such models include, amongst others, SHESED (Wicks and Bathurst, 1996), EUROSEM (Morgan *et al.*, 1998), WEPP (Nearing *et al.*, 1989) and SEDEM (Van Rompaey *et al.*, 2001). Another example from the UK is the PSYCHIC (Phosphorus and Sediment Yield Characterisation in Catchments) model (Davison *et al.*, 2008; Stromqvist *et al.*, 2008), which was designed specifically to assist catchment screening and the identification of pollution hotspots for informing mitigation planning. Its development reflects the increasing use of computer models to inform and support decisions on diffuse pollution issues and to target the implementation of abatement measures. The conceptual framework for PSYCHIC is based on the source–mobilisation–delivery transfer continuum. Mobilisation is conceptualised as initiating sediment redistribution locally at plot scale, whereas delivery represents a difference variable linking mobilisation and inputs to the river channel system. Sediment mobilisation is estimated using a modified form of the Morgan–Morgan–Finney soil erosion model (Morgan, 2001). Sediment delivery to river channels is determined by using connectivity factors based on the presence of drains predicted from the Hydrology

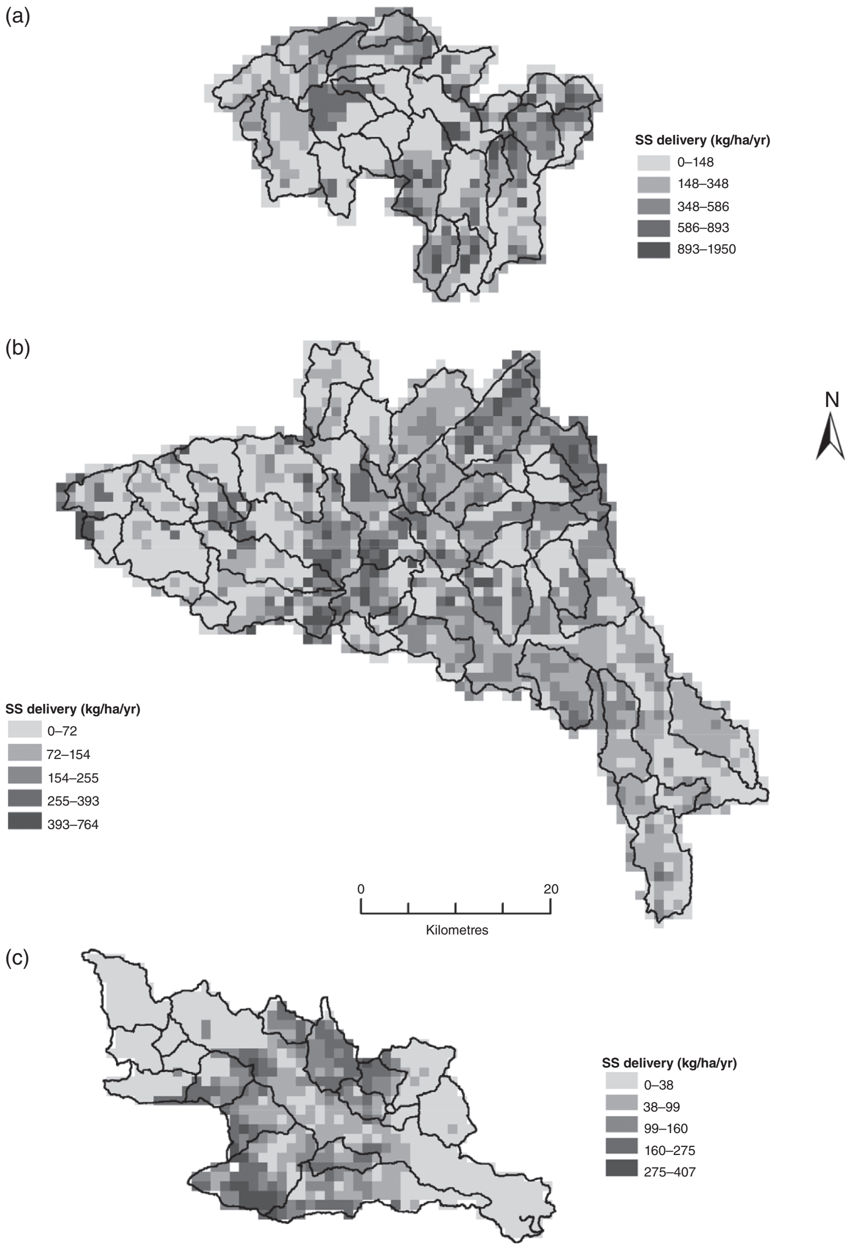

Figure 3.5 Sediment delivery to rivers predicted by PSYCHIC for the Derwent-Cocker (a), Teme (b) and Wensum (c) river catchments in England. (Based on Collins et al., 2007.)

of Soil Types (HOST) classification scheme and distance to watercourse. Figure 3.5, as an example, shows sediment delivery to streams within three contrasting river catchments in England, predicted using the PSYCHIC model. This version of the PSYCHIC model only represents sediment loss from agricultural land and does not include a channel erosion and routing function. For policy support purposes, the outputs of this model have been combined using GIS with estimates of sediment loss from additional sectors and sources, to simulate total sediment inputs to all rivers across England and Wales under current or future projected environmental conditions (see case study section and Collins *et al.*, 2009a).

Where larger river basins are involved, input data requirements and computational constraints are likely to limit the potential for applying a fully distributed and physically based approach to modelling and predicting sediment yields. In this situation, the functioning of the river basin must be simplified to incorporate the key processes and drivers of sediment yield and its area subdivided into small sub-units, which can be modelled using a lumped approach. The SedNet model, developed in Australia as a semi-lumped model for use in larger river basins (Wilkinson *et al.*, 2004, 2009), provides a good example of the potential of such models. Key features of the SedNet model are the sediment budget approach, the use of the river network to provide the basic structure and the estimation of mean annual sediment yields, rather than shorter-term yields. The network is subdivided into a series of individual links and the sediment budget is evaluated for each link, to estimate the output from the link into the next link downstream. Inputs to the link include hillslope and gully erosion from the catchment area draining to the link, bank erosion along the link and upstream inputs. Sinks within the link include overbank floodplain deposition and reservoir deposition. Within-channel storage is ignored as this is assumed to be negligible at the decadal timescale. Hillslope erosion from the catchment contributing to the link is, for example, estimated using the RUSLE model (Renard *et al.*, 1997) coupled with a sediment delivery ratio and bank erosion is modelled based on stream power and bank material properties. The model is particularly useful for management purposes, because it can provide information on the sediment yield from individual links, the contribution of each link to the sediment flux at the basin outlet and the relative importance of slope and channel (gully and bank) erosion. Such information is valuable for targeting remediation measures to reduce downstream sediment loads.

Management and policy

Since fine sediment plays a pivotal role in influencing the physical, chemical and biological integrity of aquatic ecosystems, the need to manage excess sediment stress on watercourses is integral to river catchment management and associated policy. With this recognition comes the need to assess environmental status for sediment and this, in turn, underscores the requirement for meaningful and practical sediment targets for informing compliance and gap analysis. Both water column and river substrate metrics have been proposed as river sediment targets (Collins *et al.*, 2011). Water column metrics include light penetration, turbidity, sediment concentration summary statistics and sediment regimes. Substrate metrics include embeddedness and riffle stability. However, establishing such metrics involves many problems including the uncertainty associated with toxicological dose–response experimental data. Furthermore, many of the thresholds reported in existing scientific and grey literature are based on correlative relationships that fail to capture the specific mechanisms controlling fine sediment impacts on aquatic habitats and are stationary in nature. A good example of the latter is the existing European Union Freshwater Fish Directive indicative target for annual mean suspended sediment concentration (25 mg l^{-1}) which up until 2013 was applied as a static global threshold in many Member States (Collins and Anthony, 2008b).

Against this background, the definition of meaningful fine sediment targets for informing river catchment management continues to attract debate from scientists, practitioners and policy-makers alike. The temporal windows representing the key life stages of sentinel species, such as the spawning and incubation season for salmonids, must be given greater emphasis in the identification of practical thresholds. Similarly, some consideration must be given to 'background' sediment inputs to watercourses for different physiographic settings, since no cost-effective mitigation programme should seek to address these natural levels of stress (cf. Foster *et al.*, 2011). Given the need to provide more meaningful fine sediment targets for individual contrasting catchments and to use those targets in analysing the gap between current sediment stress and good ecological condition for a range of biota, it can be argued that generic modelling toolkits capable of coupling sediment stress and its mitigation, with biotic endpoints, represent one pragmatic way forward for policy-makers working at strategic scales (Collins *et al.*, 2011). In this context, ongoing work in the UK funded by the Department for Environment, Food and Rural Affairs (Defra) is seeking to develop an integrated modelling toolkit for helping to revise fine sediment targets for individual river catchments across England and Wales. The Demonstration Test Catchment (DTC) platform (McGonigle *et al.*, 2014) established in 2009 and now in its second phase running till 2017, supported by the same body, is working to compile a robust evidence base on the impact of sediment mitigation measures from farm to catchment to national scale. Progress on these fronts is dependent on interdisciplinary working, whilst the capacity for managing excess fine sediment stress must be placed in the context of the need to maximise food production from agricultural land (Foresight, 2011; Pretty and Bharucha, 2014), which is frequently the dominant sediment source (Zhang *et al.*, 2014), for the purpose of securing food security.

Case studies

Establishing a catchment sediment budget

The use of both sediment source fingerprinting and sediment tracing techniques in tandem and in combination with information on the sediment flux at a catchment outlet provided by standard monitoring techniques can provide an effective and valuable basis for establishing a catchment sediment budget (e.g., Minella *et al.*, 2014; Walling *et al.*, 2001, 2002, 2006). Thus, for example, estimates of floodplain and channel storage can be added to the measured output flux to estimate the total sediment input to the channel system and information on the source of the sediment load can be used to estimate the primary source of this sediment input. If fallout radionuclides are used to document gross and net rates of soil loss from the slopes, comparison of these estimates with estimates of sediment input to the channel from slope sources, provides a means of obtaining a first order estimate of conveyance losses and storage associated with slope–channel transfer. This approach, coupled with additional measurements of channel storage using the approach reported by Lambert and Walling

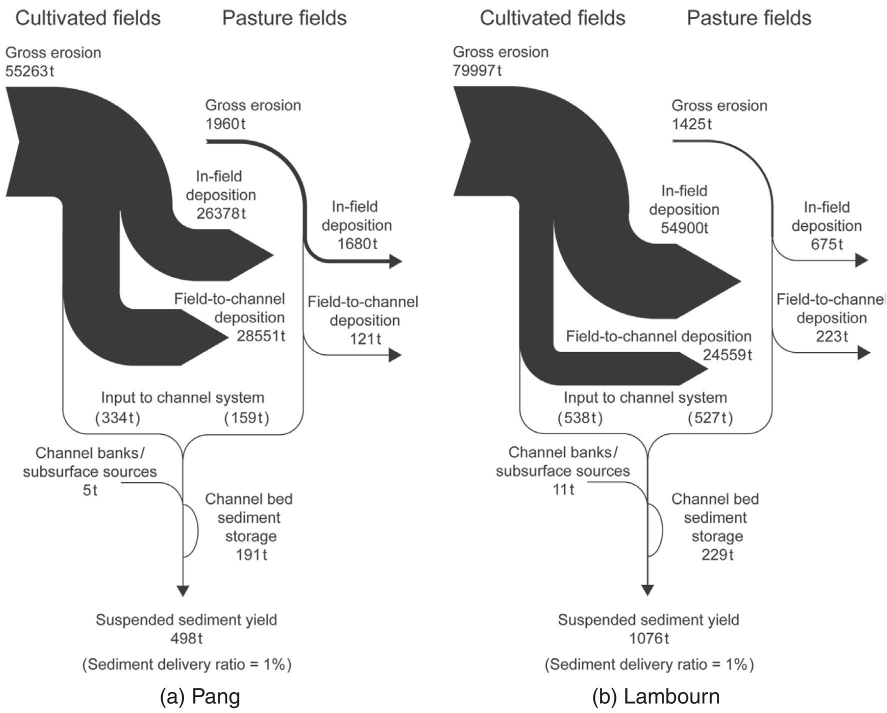

Figure 3.6 Catchment sediment budgets for (a) the Pang and (b) the Lambourn catchments in Berkshire, UK. The values indicated represent values of annual sediment flux and storage. Walling *et al*. 2006. Reproduced with permission from Elsevier.

(1988) was used by Walling *et al*. (2006) to establish tentative sediment budgets for the Pang (166 km^2) and Lambourn (234 km^2) catchments (see Figure 3.6).These two catchments, located on the chalk of southern England, formed part of the Lowland Catchment Research Programme (LOCAR) funded by the UK Natural Environment Research Council (see http://www.nerc.ac.uk/research/programmes/locar/). The location of the catchments on highly permeable strata and the resulting dominance of groundwater flow mean that storm runoff is limited and that little sediment reaches the catchment outlets. However, there is evidence of relatively high rates of sediment mobilisation and redistribution within the catchments, and their sediment budgets are dominated by slope and slope to channel sediment sinks.

Reduction of the sediment output from these catchments would clearly need to target the slopes of the cultivated areas, since these are the primary sediment source. A substantial reduction in sediment mobilisation from the cultivated slopes would, however, be required to reduce sediment output from the catchments, since only a small proportion of the soil eroded from the cultivated area reaches the channel system. However, a small increase in the conveyance loss or deposition associated with field–channel transfer could result in an appreciable reduction in the sediment input to the channel system and should thus be seen as a priority target for remedial measures. Equally, the importance of in-field and field–channel storage in reducing the sediment input to the channels means that any change in the functioning of these sinks or stores, resulting in reduced deposition or perhaps remobilisation of stored sediment, could potentially result in a major increase in the sediment outputs from the catchments in relative terms.

A national scale modelling assessment of fine sediment compliance with the EU Freshwater Fish Directive across England and Wales

A recent modelling study undertaken in the UK (Collins and Anthony, 2008b) provides a useful example of a national scale assessment of the gap between current and compliant sediment losses from the agricultural sector, based on the EU Freshwater Fish Directive (FFD) (78/659/EC) guideline standard (an annual mean concentration of 25 mg l^{-1}). The modelling methodology was founded on a statistical relationship between measured suspended sediment concentration and modelled total sediment inputs to watercourses from diffuse and point sources. Mean annual total suspended sediment loads for each Water Framework Directive (WFD) waterbody across England and Wales were estimated as the sum of the modelled individual loads for the diffuse agricultural and urban sectors, eroding channel banks and point source discharges. Diffuse agricultural sediment inputs for all rivers were calculated using the PSYCHIC process-based model (see above), which deploys 1 km^2 resolution statistical input information on a number of key environmental drivers, including climate, slope, soil types and characteristics, drainage density, land use and cropping and livestock density. National scale sediment contributions from diffuse urban sources were estimated using an Event Mean Concentration (EMC) approach based on the inter-quartile ranges of empirical data for sediment runoff from industrial areas, main roads and residential zones. The EMCs were combined with estimated mean annual runoff from urban areas derived using the Wallingford procedure (National Water Council, 1983). Corresponding total sediment inputs from eroding channel banks were estimated using a prototype national scale index based on the river regime (Gustard *et al.*, 1992), the duration of excess shear stress and channel density. Point source sediment loadings to all rivers across England and Wales were computed using a database of consented effluent discharges from sewage treatment works, but with a correction based on the relationship between measured and consented average suspended sediment concentrations.

The predicted mean annual total suspended sediment loads delivered to all rivers were coupled with corresponding flow regime distributions to estimate time-averaged suspended sediment concentrations. Structured regression modelling was used to optimise the relationship between modelled and measured time-averaged suspended sediment concentrations, for the purpose of estimating the annual mean suspended sediment concentration and the likelihood of 'good ecological status' (GES) due to sediment contributions from the agricultural sector alone (Figure 3.7). The findings suggested that on the basis of using the FFD to define GES for sediment, approximately 83% of the total catchment area across England and Wales appeared to require no further reduction in sediment loss to rivers from diffuse agricultural sources. Maps of compliance, however, will inevitably depend on the sediment thresholds used to define GES, and in recognition of the issues associated with the 'global' FFD guideline standard, alternative means of setting thresholds on a catchment-specific basis are currently

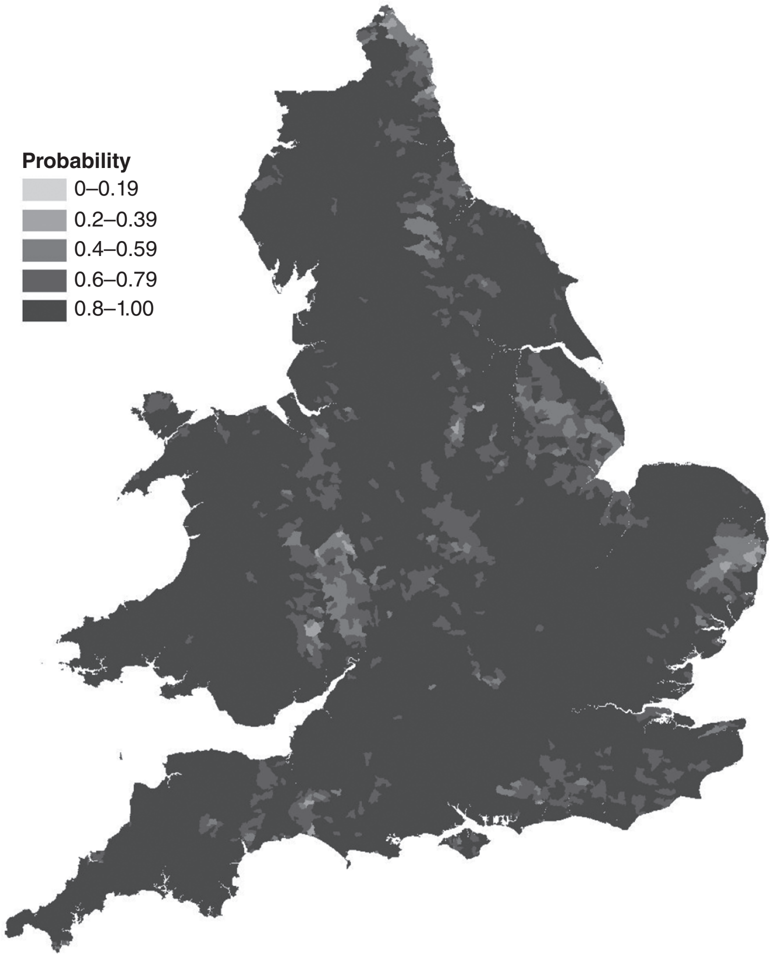

Figure 3.7 Likelihood of meeting 'good ecological status' (GES) for fine sediment across England and Wales, as defined by the EU Freshwater Fish Directive (FFD) guideline standard (Based on Collins and Anthony, 2008b.)

being investigated to inform catchment management for sediment across the UK.

Summary and the way forward

About 25 years ago the fine sediment loads of rivers were frequently seen as being of limited importance. They are now recognised as representing a key element of river behaviour with wide-ranging ecological and environmental significance and an important focus for catchment management programmes. The availability of new instrumentation to provide improved data on suspended sediment loads, the development of a range of techniques to document sediment sources and soil and sediment redistribution within catchments, as well as the development of improved catchment-based distributed models have resulted in important advances in our understanding of the fine sediment dynamics of catchments and our ability to predict their behaviour. The growing awareness of the environmental significance of fine sediment, and particularly its ecological importance, is directing increasing attention to sediment management in river catchments. The development and implementation of successful fine sediment management strategies will depend on the availability of a sound understanding of both sediment budgets and sediment-related stress and biotic impacts, as well as a reliable evidence base to support policy (cf. Collins *et al*. 2009b).

Looking to the future, there is a need to continue to improve our understanding of catchment sediment dynamics and their response to land use and climate change and our ability to model catchment behaviour. As management attracts greater attention, it is important that the available models should be capable of predicting catchment response under different management scenarios, in order to assess their likely impact and success. Sediment source tracing must be seen as providing key information for targeted management and there is a need to exploit the potential for further improvements in source discrimination, to identify source-specific inputs, and to progress its transfer from being a research tool to one that can be more easily and widely applied on a routine basis. To support policy-making it is important that further attention should be directed to establishing more meaningful sediment targets or metrics for assessing catchment compliance and this will require further research on the ecological impacts of fine sediment. In this context, attention should be directed to the relative roles of the organic and inorganic components of fine sediment loads in contributing to sediment-related stress. Developing effective strategies for controlling fine sediment loss to watercourses will require an improved empirical data base on the cost-effectiveness of mitigation options, set in the context of a competitive agricultural sector and the need to engage catchment stakeholders. In addition there is a need to develop and refine both farm-scale toolkits for guiding the selection and targeting of on-farm mitigation strategies and catchment-scale modelling frameworks for scaling up the likely benefits. The latter should incorporate the link between sediment stress and biotic impacts and thereby permit decision making to focus more directly on protecting aquatic ecosystems.

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