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SILTFLUX Literature Review

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- Office of Environmental Enforcement
- Office of Evidence and Assessment
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EPA RESEARCH PROGRAMME 2014–2020

SILTFLUX Literature Review

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EPA Research Report

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by

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DISCLAIMER

The EPA Research Programme addresses the need for research in Ireland to inform policymakers and other stakeholders on a range of questions in relation to environmental protection. These reports are intended as contributions to the necessary debate on the protection of the environment.

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Executive Summary

Sediment is a natural and dynamic component of river catchment systems, in which it is transported as bedload and/or suspended load, depending on the relationship between flow conditions, sediment supply and the structure, density, size and shape of materials. Although sediment does not feature explicitly within the Water Framework Directive (WFD), the ecological focus of the legislation with regard to surface waters means that the role of sediment as an essential component of the sustainable management of aquatic systems is recognised. Suspended sediment particles are typically < 63 μm in diameter, but can be much coarser (up to 2 mm in diameter in extreme events). The delivery of sediment to rivers is dependent on a range of factors including catchment soil type, vegetation cover, land use, hillslope hydrological processes, flow pathways, topographic setting and the presence/absence of buffer zones, such as floodplains, which can decouple hillslopes from river channels. In healthy fluvial systems, sediment provides the basis for diverse aquatic ecosystems through nutrient cycling and replenishment, as well as by forming the contributing materials from which aquatic habitats are constructed in river beds and especially in banks. Too much silt, however, can lead to the obstruction of channels, the smothering of habitats, ingress into the bed and the reduction of light penetration in the water column and at the bed, potentially leading to deoxygenation and environmental deterioration. The presence of elevated fine sediment may also degrade biological habitats through its biological oxygen demand (BOD) and the contaminants adsorbed to the sediments, and loss of habitat heterogeneity. Consequently, macroinvertebrate diversity and abundance is susceptible to change either directly, through effects on survivorship, growth, feeding, etc., or indirectly, through the alterations in habitat structure. Fish are also adversely affected by excessive sediment levels. Increased turbidity can cause gill irritation or reduce feeding activity by impairing visual range. Chemically active silt fractions (< 63 μm in diameter) can also act as important carriers of potentially hazardous nutrients and contaminants, dioxins and heavy metals. Sand deposition from the water column can "seal" the surface of valuable gravel habitats, while silt particles (< 63 μm in diameter) can infiltrate the gravel matrix; these effects can significantly reduce the permeability and porosity of spawning gravels, suppress oxygen supply rates, hinder the removal of toxins from redds and reduce egg hatching and larval survival rates. Sediment infiltration can also negatively affect microbial processes in the hyporheic zones with consequences for biodiversity and ecosystem functioning, which may impact on groundwater quality. Therefore, the effective management of aquatic systems is critically linked to understanding sediment transport and storage pathways.

While sediment is transported into river bodies through the natural processes of erosion and deposition, anthropogenic activities can also generate high sediment loadings. For example, human activities can lead to the deposition of soil as a result of the erosion of banks, due to trampling by livestock (known as "poaching"), the removal of riparian vegetation, the ploughing of land, deforestation/tree harvesting and land drainage schemes. The main route by which sediment is transported to water bodies is through drainage networks, but non-channelised surface wash flow can also be important, especially in arable contexts. Interflow and groundwater pathways can also be important in some settings.

This review summarises the key issues that affect the role of fine sediment in fluvial systems, with a focus on northern Europe, the UK and Ireland, which will be of most relevance to the SILTFLUX project. The review includes definitions of fine sediment; descriptions of typical sediment sources and delivery mechanisms; overviews of the impacts of sediment (e.g. for organisms and downstream systems); descriptions of monitoring and analytical methodologies; details of fine sediment fluxes and yields for rivers in Ireland and comparable river catchments in the UK and elsewhere in Europe; a summary of the crucial dynamics of storm-event and seasonal suspended sediment transport; details of land use and climate change effects on sediment fluxes; and a discussion of the management implications of sediment.

1 Introduction

Sediment is an integral and dynamic component of healthy fluvial systems (Yarnell *et al.*, 2006) and it plays a significant role in the geomorphological, hydrological and ecological functioning of a river (Kemp *et al.*, 2011). However, its roles as a direct pollutant and as a vector of contaminant transport are now being increasingly recognised internationally (Ballantine *et al*., 2008; Collins *et al*., 2011). Indeed, excessive sedimentation is the number-one cause of water quality violations in the USA (Downing 2005). Metal and insecticide contaminants are also leading causes of violations, and most of these contaminants are bound to suspended sediment (Downing, 2008a). The management of sediment input to rivers is now a priority in many countries. This must recognise that the impact of sediment can be both due to suspended sediment in the water column and deposited sediment on the bed of river channels, lakes and estuaries. Many authorities have suggested maximum suspended sediment concentrations (SSCs), and Table 1.1 summarises some of these initial thresholds for fish. The Irish Environmental Protection Agency (EPA) has identified deteriorating water quality as the major environmental challenge and, in general, river pollution has increased since the late 1970s (Toner *et al*., 2000). Similarly, the England Catchment Sensitive Farming Delivery Initiative recognised "40 priority catchments in April 2006 where stakeholders require assistance to improve the protection of aquatic habitats" (Collins *et al*., 2010a). Catchment Sensitive Farming officers now assess contamination impacts, including those from sediment, and support stakeholders with the implementation of good practice on farms.

The aim of this literature review is to summarise the key issues affecting the role of fine sediment in fluvial systems, with a focus on northern Europe, the UK and Ireland, which are of most relevance to the SILTFLUX project. While the review primarily focuses on suspended sediments, deposited sediments are also considered. The review includes definitions of fine sediment; descriptions of typical sediment sources and delivery mechanisms; overviews of the impacts of sediment (e.g. for organisms and downstream systems); descriptions of monitoring and analytical methodologies; details of sediment fluxes and yields for rivers in Ireland and comparable catchments in the UK and elsewhere in Europe; a summary of the crucial dynamics of storm-event and seasonal sediment transport; details of land use and climate change effects on sediment fluxes; and a discussion of the management implications of sediment. Most of the data are contained in supporting figures and tables.

There are already numerous and widely available international and European or UK-based literature reviews on suspended sediment transport, its impacts and associated pollutants, which have been published in the last 8 years and to which the reader is referred for details of specific issues (Collins and Walling, 2004; Lawler, 2005a; Owens, 2005; Owens *et al*., 2005, 2006; Walling *et al*., 2008; Gray and Gartner, 2009; Lawler *et al*., 2009; Taylor and Owens, 2009; Collins *et al*., 2011; Kemp *et al*., 2011; Owens and Xu, 2011; Vanmaercke *et al*., 2011). Therefore, reference is made to these sources as appropriate, and this review concentrates on the key issues relevant to the EPA-funded SILTFLUX research project.

Table 1.1. Initially proposed thresholds of SSCs for different effects on fish (Collins *et al*., 2012)

Thresholds (mg/L) for least, probable and definite effects			Source
Least effects: high protection, best conditions	Probable effects: moderate protection, moderate conditions	Definite effects: low protection, poor conditions	
<25	25–80	>80	EIFAC (1964)
<25	26–80	>80	Alabaster and Lloyd (1982)
<30	30–85	>83	Wilber (1983)
0	1–100	>100	DFO (1983)

In the context of this review, fine sediment (or suspended sediment) is defined operationally as sediment particles that are suspendable in the river water column (Figure 1.1 and Figure 1.2) and can ingress into gravel river beds; these particles typically range from 1 to 250 μm in diameter. For example, Walling *et al*. (2000) found that the particle diameter for > 95% of the suspended sediment load (SSL) sampled in most of the rivers in the Humber and Tweed basins in north-east England is < 63 μm (i.e. most are silt- and clay-sized particles). The median particle diameter, D_{50}, ranged from 4.1 to 13.5 μm. Clay-sized particles (< 2 μm) accounted for 15–25% of SSL. A compilation of typical particle diameter distributions and organic content for bed and suspended sediment for several rivers in England and Wales is given in Table 1.2 (Buss, 2009).

Furthermore, Droppo (2003) argued that a broadening of suspended sediment definitions, from those that reflect purely physical characteristics to those that include biological and chemical information, is now required. For example, suspended sediment can often flocculate in aquatic systems if chemical and biological conditions are favourable, and these flocs will have different hydrodynamic properties, entrainment thresholds and settling characteristics (see, for example, Williams *et al*., 2008).

The SSCs are normally measured in g/L or mg/L, while sediment fluxes are defined in, for example, kg per second (kg/s) or tonnes (t) per year. Longer term suspended sediment yields are usually normalised for catchment area to facilitate inter-basin comparisons and are presented as t/km^2 per year. Field soil erosion rates are often given on the smaller scale of kg per hectare (kg/ha) per year (or t/ha per year).

Figure 1.1. Suspended sediment in the River Alne, Warwickshire. Flow from right to left (photo: Damian Lawler).

Figure 1.2. Suspended sediment of a very different colour (and probably source) in the River Alne, Warwickshire, at the same bridge site as shown in Figure 1.1, but looking upstream. Flow towards camera (photo: Damian Lawler).

Table 1.2. Characteristics of fine sediments from selected UK rivers (source: Buss, 2009)

River	Sediment type	Organic content (%)	Particle size distribution (%)				Reference
			Sand (0.063–2 mm)	Silt (0.0004–0.062 mm)	Clay (<0.0039 mm)	Other	
Upland streams (impermeable strata)	Upper 30 cm of channel bed		23	3.5	0.6		Milan *et al*. (2000)
Small chalk streams with low rainfall			85	4.9	0.6		
Lowland limestone and sandstone streams			45	7.4	1.7		
River Test, Hampshire, England	Accumulated sediment from artificial redd	19.7 of <2 mm				10% <2 mm	Greig *et al*. (2005)
River Aran, Powys, Wales		7.5 of <2 mm				15.7% <2 mm	
River Ithon, Powys, Wales		5.3 of <2 mm				28.9% <2 mm	
River Blackwater, Hampshire, England		3.4 of <2 mm				12.2% <2 mm	
River Frome, Dorset, England	Suspended	5–60					Farr and Clarke (1984)
River Test, Hampshire, England	Suspended	25–40 during summer and autumn low flows; 15–25 during winter and spring high flows					Acornley and Sear (1999)
	Bed sediment					Summer low flows (June–September): suspended sediment (<0.25 mm) accounted for 70–90%; autumn floods (October): coarser sediment (0.25–4 mm) accounted for more	
Upper Piddle, Dorset, England	Fine bed sediment	12.2					Walling and Amos (1999)
Little Stour, Kent, England	<250 μm surficial fine sediment	13.8 (SD 4.35, $n = 51$)				Spatially and temporarily consistent ($D_{50} = 58.75$ μm; SD = 6.25 μm)	Wood and Armitage (1999)

SD, standard deviation.

2 Fine Sediment Sources, Delivery and Budgets

2.1 Sediment Sources and Delivery

2.1.1 Context

Identifying sediment sources is extremely important. Collins and Walling (2004) have constructed a typology of likely sources (Figure 2.1), and argue that "an improved understanding of catchment suspended sediment sources represents *a prerequisite* (our italics) for assisting the design and implementation of targeted management strategies for controlling off-site sediment-associated environmental problems". However, defining the provenance and types of fine sediment sources is very difficult and for many catchments these are largely unknown. Even within a given basin, dominant source locations may vary over time with different meteorological, hydrological and antecedent events. Typical fine sediment sources and how their relative dominance may change in a downstream direction in a catchment are listed in Table 2.1 for humid temperate environments. Such sources, however, must be viewed in contexts of the catchment and connectivity, as illustrated in Figures 2.2–2.4.

In addition, "information on sediment delivery to watercourses is urgently required to test and evaluate existing diffuse pollution models" (Collins *et al*., 2010a). Section 2.1.2 briefly summarises typical approaches adopted to identify or, rather, *infer* (Collins and Walling, 2004) catchment sediment sources, sediment routing and the processes that transport the sediment to the channel, that is, source-to-river connectivity.

2.1.2 Methods of identifying and locating suspended sediment sources

Several direct and indirect approaches for estimating sediment sources exist, as reviewed by Collins and Walling (2004), and a combination of methods can be most effective. Sediment "fingerprinting", through chemically "matching" likely *source* sediment with *transported* suspended sediment, has proved especially useful: its conceptual basis is shown in Figure 2.5. Multi-parameter or mixed model approaches to fingerprinting, as adopted by Walling *et al*. (2008) and Collins *et al*. (2010a) have become

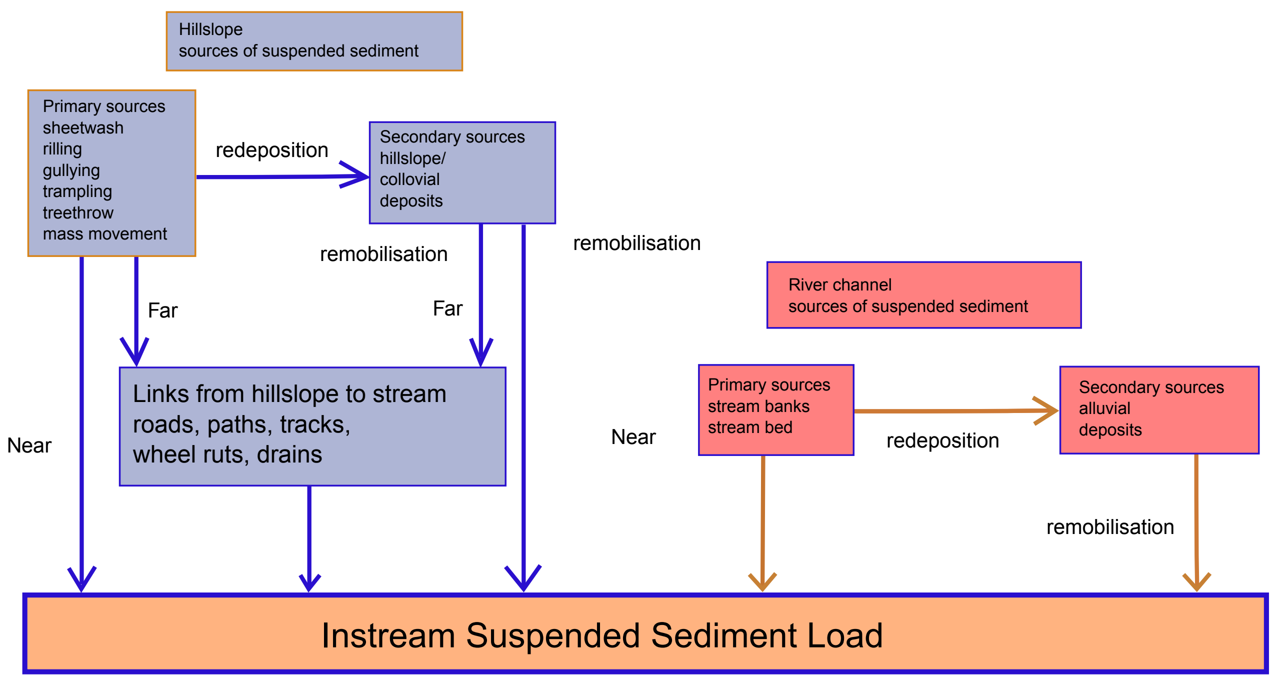


Figure 2.1. Example of an advanced classification system for potential hillslope and river channel suspended sediment sources (after Collins and Walling, 2004).

Table 2.1. Typical catchment sediment sources, and the likely variation in a downstream direction (adapted from Sear *et al.*, 2003)

Upper course	Middle course	Lower course
Rock fall	Valley side slope	Overland flow
Scree slope	Terrace slope	Tributaries
Debris flow	Soil creep	Cultivated farmland
Landslide	Floodplain erosion	Wind-blown soils
Freeze–thaw	Tributary stream	Construction sites
Sheet flow	Cultivated farmland	Urban runoff
Rills and gullies	Field drains and ditches	Gravel workings
Overgrazed, burnt or rabbit-infested areas	Urban runoff	Marine sediments (estuaries)
Ditches (forest and road)	Ditches (forest and road)	
Quarries	Mining and gravel extraction	

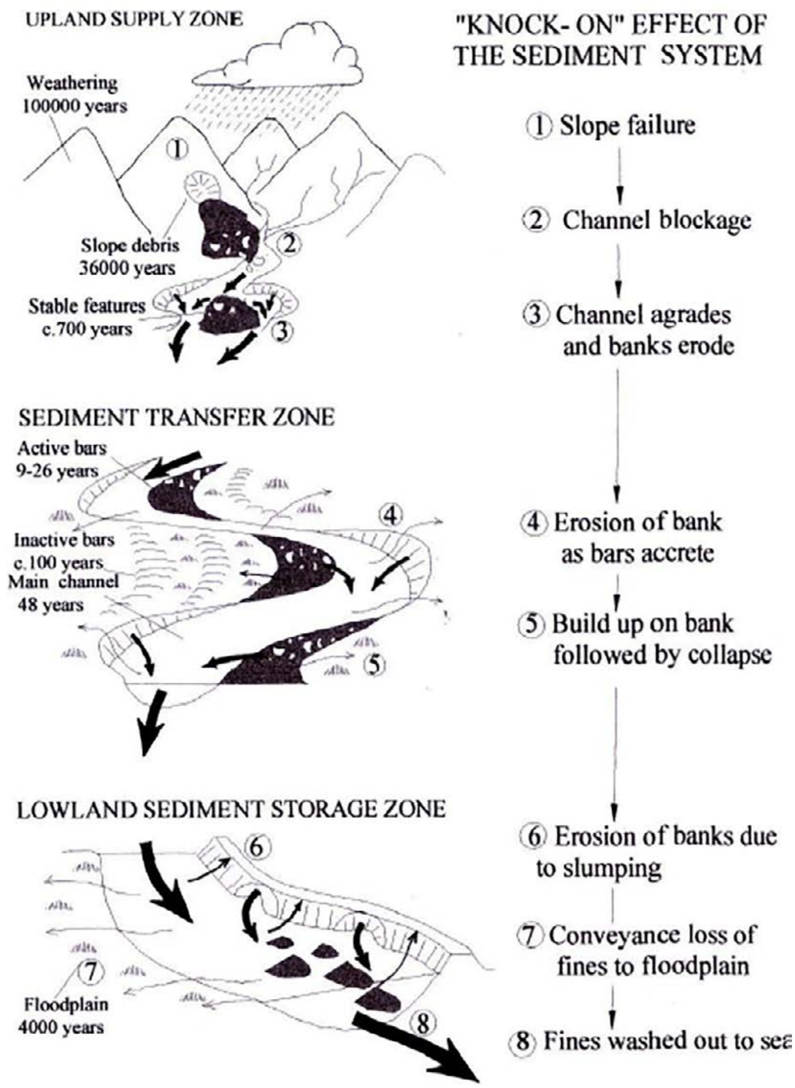


Figure 2.2. Sediment connectivity in the fluvial system (Crown copyright – Sear *et al.*, 2003).

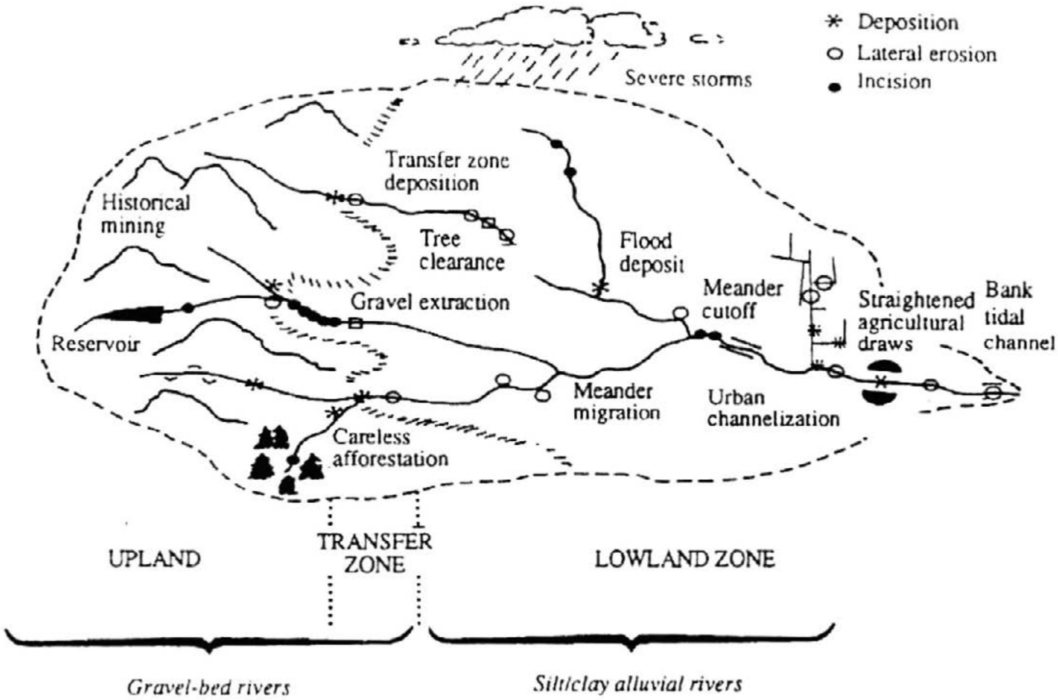


Figure 2.3. Natural and anthropogenic catchment and river processes that affect sediment dynamics (Crown copyright – Sear *et al.*, 2003).

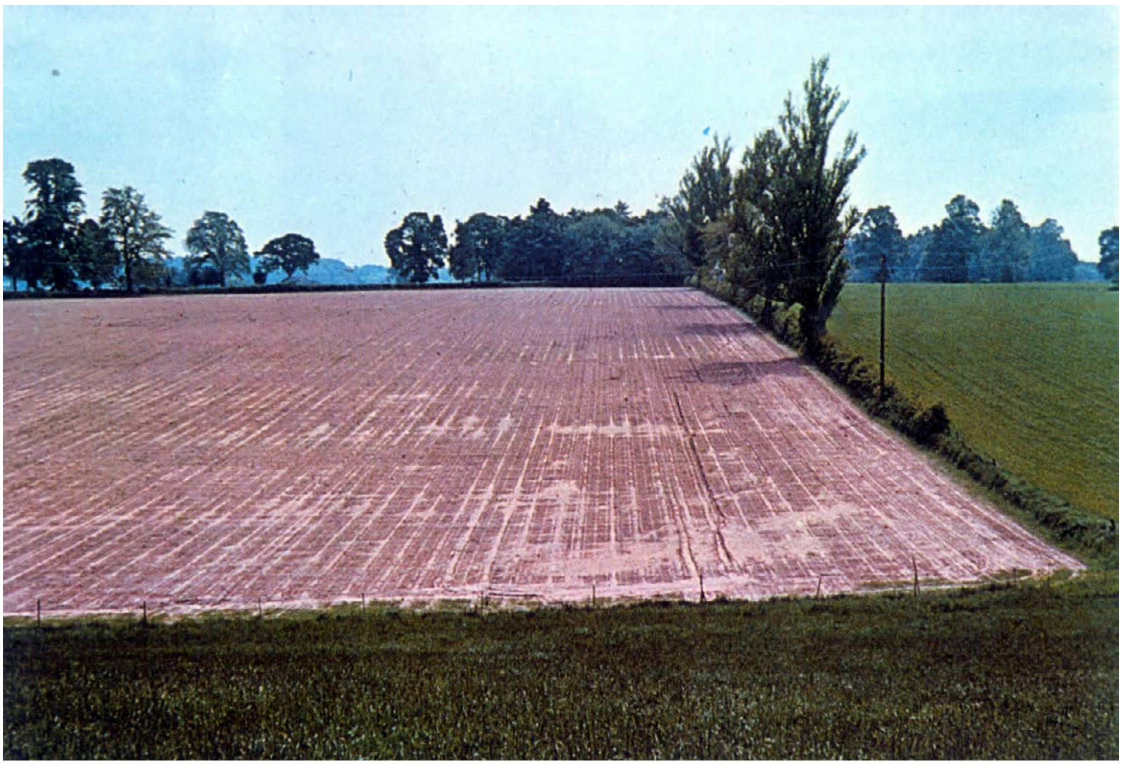


Figure 2.4. Eroding arable fields – an example of a sediment source in Shropshire, UK (photo: Damian Lawler).

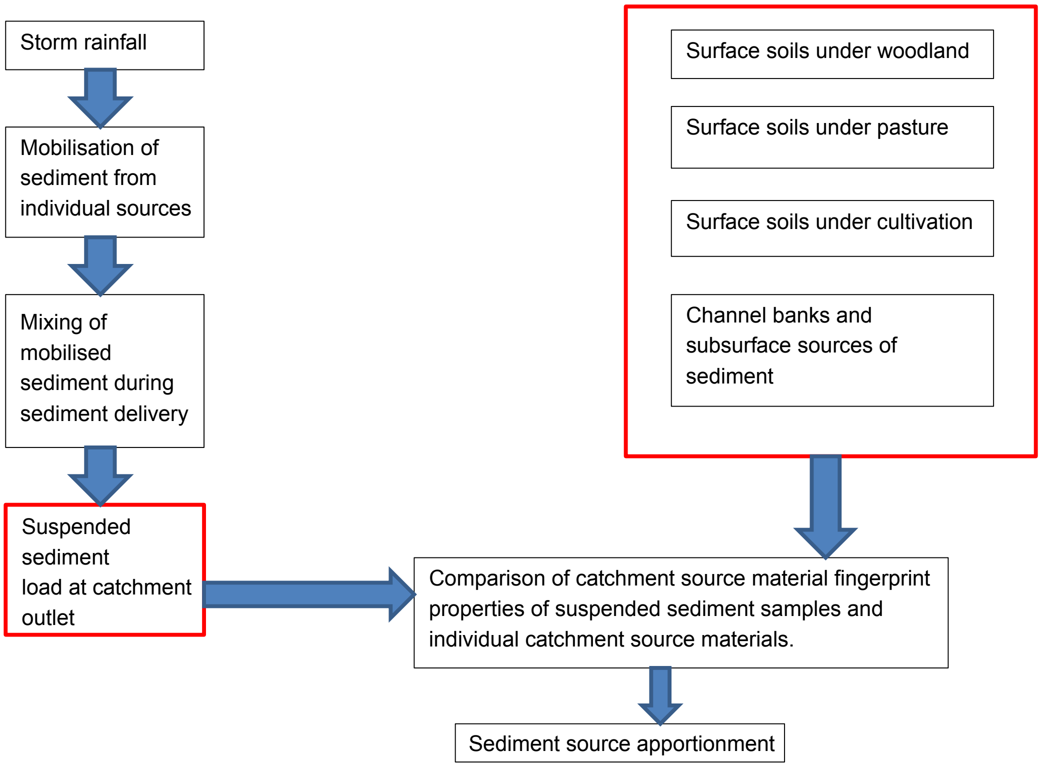


Figure 2.5. The conceptual basis of the fingerprinting technique used to establish suspended sediment sources in the PSYCHIC study (adapted from Walling *et al.*, 2008a).

popular as methods of identifying catchment sediment sources.

There are a number of models for predicting sediment sources and sediment delivery from catchments, and linking them to in-stream sediment transport. A recent promising numerical model is the dynamic, process-based INCA-Sed (the Integrated Catchment Model for Sediments) (Jarritt and Lawrence, 2006). Its conceptual basis is shown in Figure 2.6. INCA-Sed works at a daily time step, and was tested by Lazar *et al*. (2010) in the River Lugg, a tributary of the River Wye in Wales. It was found that "diffuse soil loss" was the most important sediment generation process in the Lugg, although, in this case, SSCs were relatively low and unlikely to cause significant ecological impacts. It was also tested in four catchments in Finland by Rankinen *et al*. (2010) and was found to correctly reproduce the observed spatial and temporal sediment dynamics.

2.1.3 Sediment budgets and sediment delivery ratios

Measures of soil erosion rate cannot be used to infer rates of sediment delivery to stream channels because significant deposition of sediment usually occurs (1) before it reaches the drainage network and (2) within the river channel upstream of the sediment flux monitoring station. The percentage of sediment leaving a catchment, relative to that eroded from the catchment, is called the sediment delivery ratio (SDR), defined as:

$$SDR = (100\,SSY/E) \times 100 \qquad \text{(Equation 2.1)}$$

In Equation 2.1, SSY is the fluvial suspended sediment yield per unit catchment area at some downstream river gauging point in the catchment (t/km^2 per year) and E is the spatially averaged catchment erosion rate (t/km^2 per year).

In a small catchment SDRs can be as low as 15%, and these can be even lower in larger catchments, in which numerous sediment storage opportunities exist, at between 5 and 10% (Walling, 1983). Occasionally, a "hillslope SDR" is defined, which is the percentage of sediment eroded from slopes that reaches the river.

Quantifying the relative importance of different sediment sources allows a sediment budget to be constructed. A good example is provided by Walling *et al*. (2002) for the small lowland agricultural catchments of the Rosemaund and Lower Smisby in central

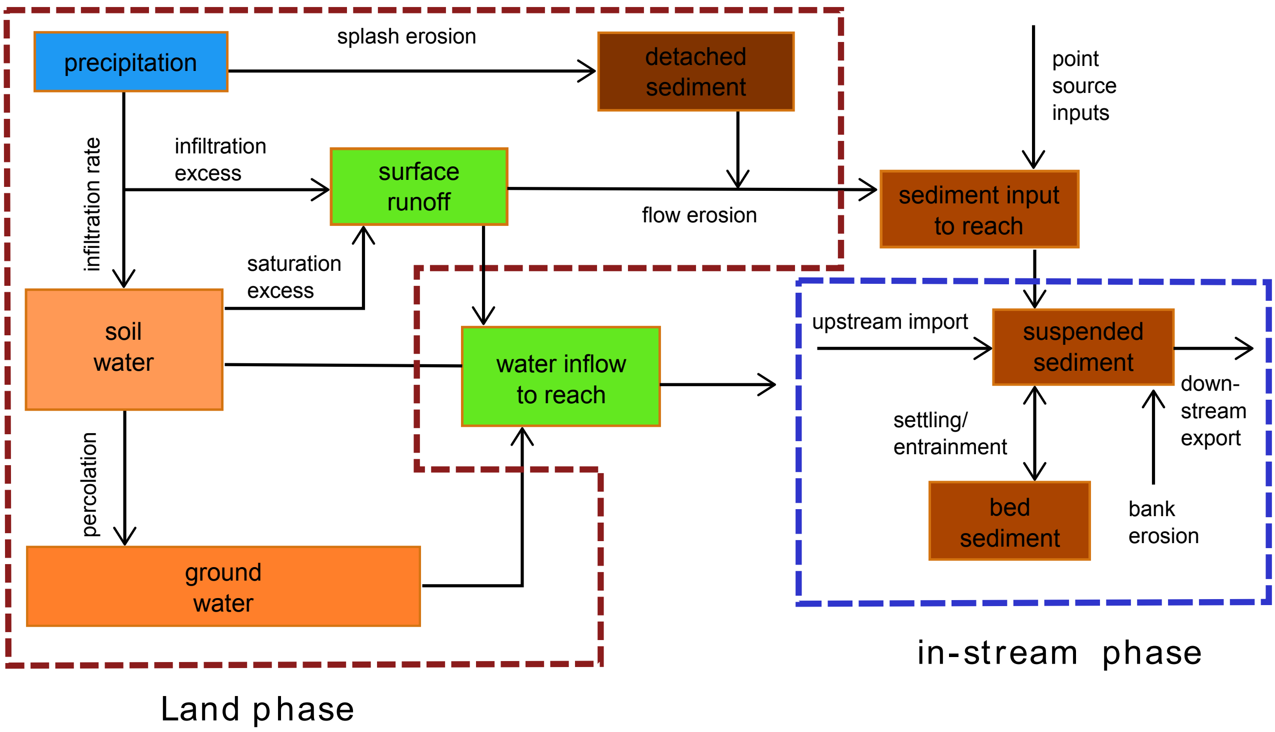


Figure 2.6. The conceptual framework that underpins the numerical INCA-Sed of Jarritt and Lawrence (2006) [after Lazar *et al.* (2010)].

England (Figure 2.7). Even in these tiny catchments (<3.6 km²), the SDRs were only 17% and 20%, respectively, suggesting that the vast majority (≥80%) of the fine sediment produced in the catchment slope headwater areas was stored at locations up-catchment of the river monitoring sites, such as in fields and the channel (Figure 2.7), although field drains carried little sediment.

Table 2.2 shows how multiple catchments can be separately assessed for their contribution to total sediment delivery, with additional evidence added for different source types (Collins *et al*., 2010a).

2.1.4 Sources and sediment pathways

Taylor and Owens (2009) compiled data that suggested that diffuse sediment from agricultural sources is responsible for 75.7% of all sediment supplied to rivers across England and Wales. Eroding channel banks (for an Irish example, see Figure 2.8) were responsible for a further 15.5% of sediment, with the remainder coming from urban sources (5.8%) and point sources (3.0%) (e.g. from wastewater treatment works). In the UK, arable fields can be key contributors of fine sediment: “sediment loss from cultivated fields has been accentuated by a number of factors including a shift towards more autumn-sown cereals which exposes bare tilled soils to the risk of erosion by winter rainfall and the expansion of maize production” (Collins *et al*., 2010a) (e.g. Figure 2.9). In the Midlands in England, Chapman *et al*. (2005) identified sub-surface drainage (macropore flow through cracked soils) as a key mechanism for transporting fine sediment from fields to streams. They also identified drains as a major mechanism for transporting fine sediment: in one of their catchments, drains contributed over half the annual sediment load.

2.1.5 Connectivity of catchments and channels

There must be good surface or sub-surface hydraulic *connections* that link eroding terrain parcels to the drainage network (i.e. strong *connectivity*) if abundant sediment is to reach the river channels (e.g. Figure 2.2). Indeed, Lane *et al*. (2007) showed, for the first time anywhere, that catchment topography and “hydrological connection discriminates both presence/ absence and abundance of juvenile brown trout populations”. This finding, based on work in the Eden catchment, Cumbria, could have wide ramifications elsewhere. For example, “if topographic control

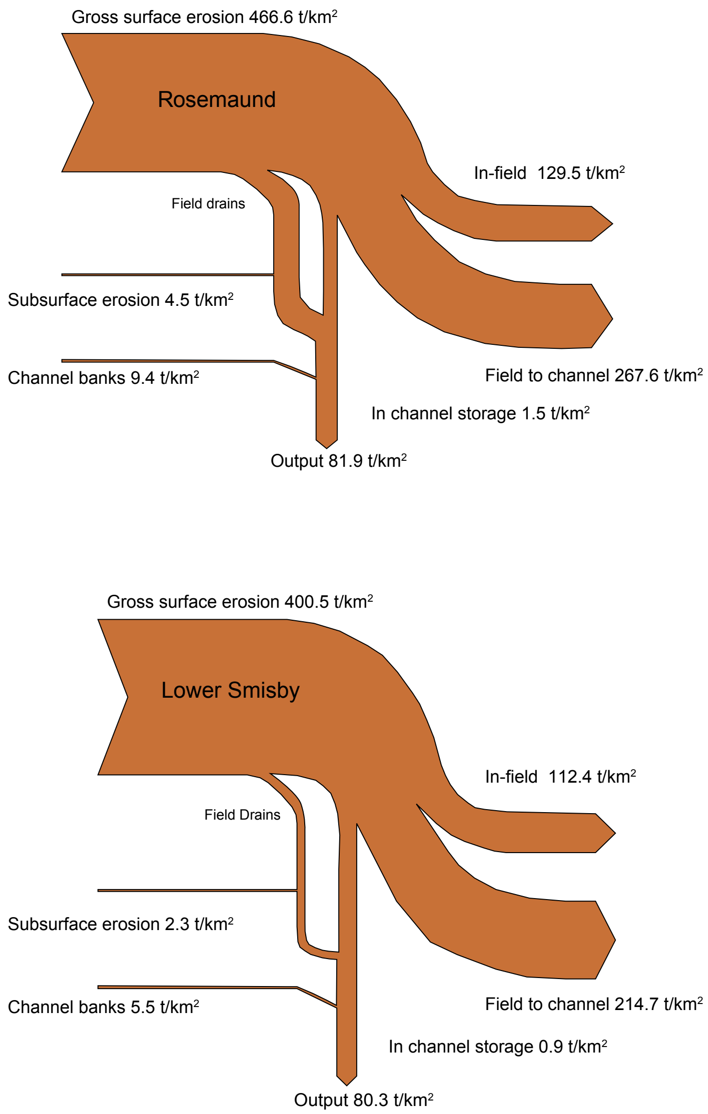


Figure 2.7. Sediment budget examples from catchments in central England (Walling *et al*., 2002).

mediates the watershed to stream linkage, land use impacts can only be appreciated with respect to their position in the landscape. This is of practical importance as locations of high connectivity should be a primary objective in targeting watershed restoration measures to where they will deliver most instream benefits" (Dugdale and Lane, 2006).

2.2 River Bank Erosion as a Suspended Sediment Source

River bank erosion events can introduce significant quantities of fine sediment into stream channels (e.g. Figure 2.8 and Figure 2.9). Indeed, Collins *et al*. (2010a) argue that "there is increasing evidence for the role of eroding channel banks as an important sediment source in fluvial systems" (based on evidence from Church and Slaymaker, 1989; Collins *et al*., 1997; Lawler *et al*., 1999; Walling and Collins, 2005; Walling *et al*., 2008). World river bank erosion rates have been tabulated by Lawler (1993) and plotted against drainage basin area, as shown in Figure 2.10 (Lawler *et al*., 1997). Rates vary widely for a given catchment area, from 0.04 m/year for small catchments (< 10 km^2) to 1 km/year for very large basins, such as the lower Mississippi (10^6 km^2) (Figure 2.10).

Bank sediment contributions can be modelled within a sediment budget framework [e.g. using the PSYCHIC model (Collins *et al*., 2007, 2009)]; however, in the field, this is challenging, even within a short river reach, as shown in Lawler *et al*. (1997) (Figure 2.11). This partly reflects the vast range of bank erosion processes possible (see below). It also reflects an

Table 2.2. Sediment sources for several south-west England catchments (delivery to watercourses in kg/ha per year), with information on source types for each catchment (adapted from Collins *et al.*, 2010a)

Sub-catchment	Sediment yield (kg/ha per year)									
	Pasture topsoils[a]		Cultivated topsoils[a]		Damaged road verges[b]		Channel banks/sub-surface sources[b]		Sewage treatment works[b]	
	200	700	200	700	200	700	200	700	200	700
Brue	169–185	591–647	428–570	1496–1995	0–2	0–7	16–24	56–84	0–2	0–7
Cary	146–162	511–567	30–42	104–146	20–24	70–84	82–90	287–315	2–6	7–21
Halse Water	94–107	327–376	369–396	1292–1386	20–24	70–84	2–6	7–21	0–4	0–14
Isle	158–173	554–607	63–88	220–308	18–26	63–91	58–62	203–217	4–8	14–28
Parrett	131–145	457–509	135–162	472–567	22–30	77–105	46–50	161–175	2–6	7–21
Tone	181–189	635–660	85–116	298–406	22–30	77–105	42–46	147–161	0–4	0–14
Upper Parrett	229–236	800–829	101–113	352–396	4–8	14–28	34–38	119–133	2–6	7–21
Yeo	33–40	114–140	207–252	723–883	54–62	189–217	56–60	196–210	2–6	7–21

[a]Calculated using the area of the sub-catchment under the land use in question based on the ADAS land use database.

[b]Calculated using the total sub-catchment areas provided by the UK Environment Agency using the *Flood Estimation Handbook*.

Figure 2.8. River bank erosion on the River Allow, Ireland, is a sediment source (photo: J.J. O'Sullivan).

Figure 2.9. Eroding river banks around a sedimentation zone on the River South Tyne, Northumberland (photo courtesy of the *Northern Echo*).

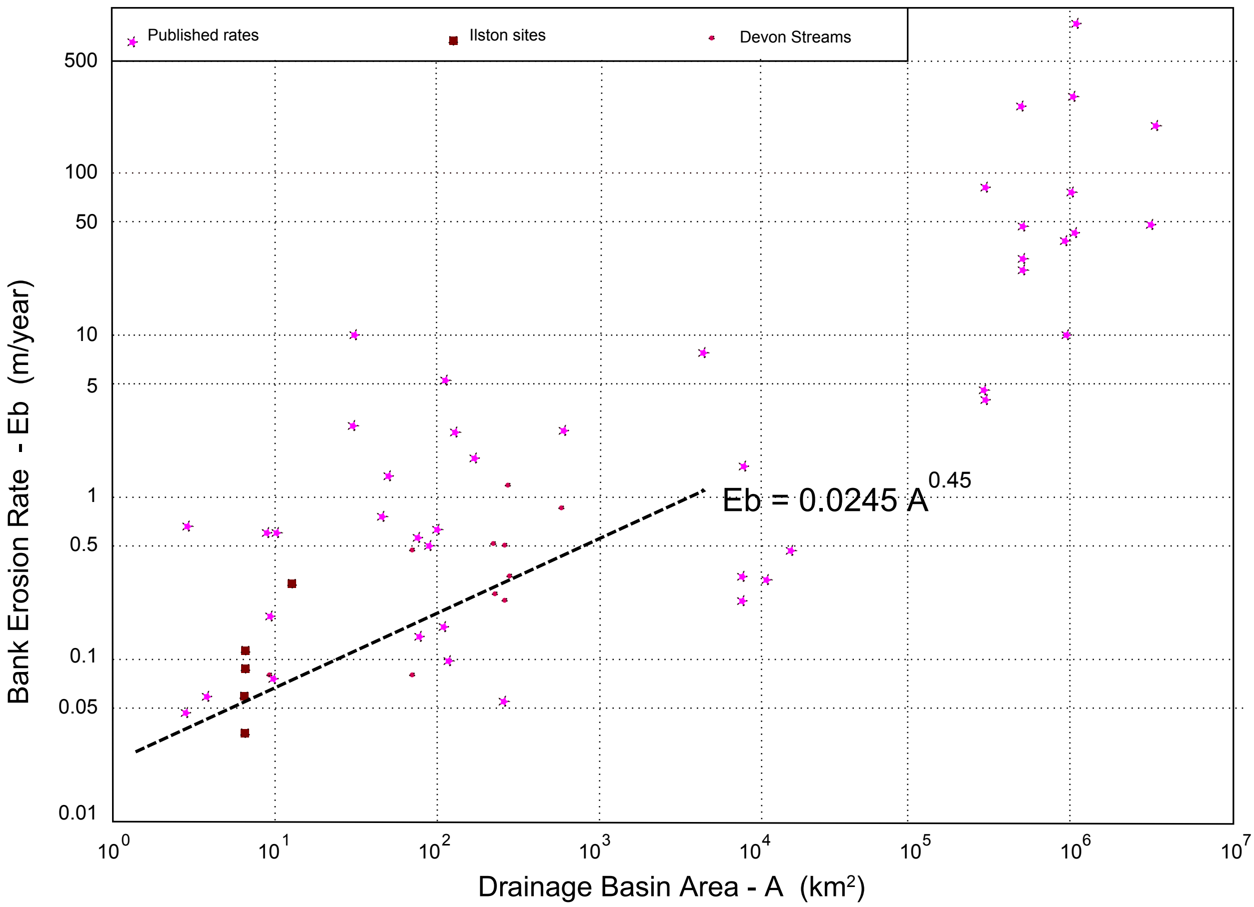


Figure 2.10. World river bank erosion rates with respect to drainage basin area (adapted from Lawler *et al.*, 1997).

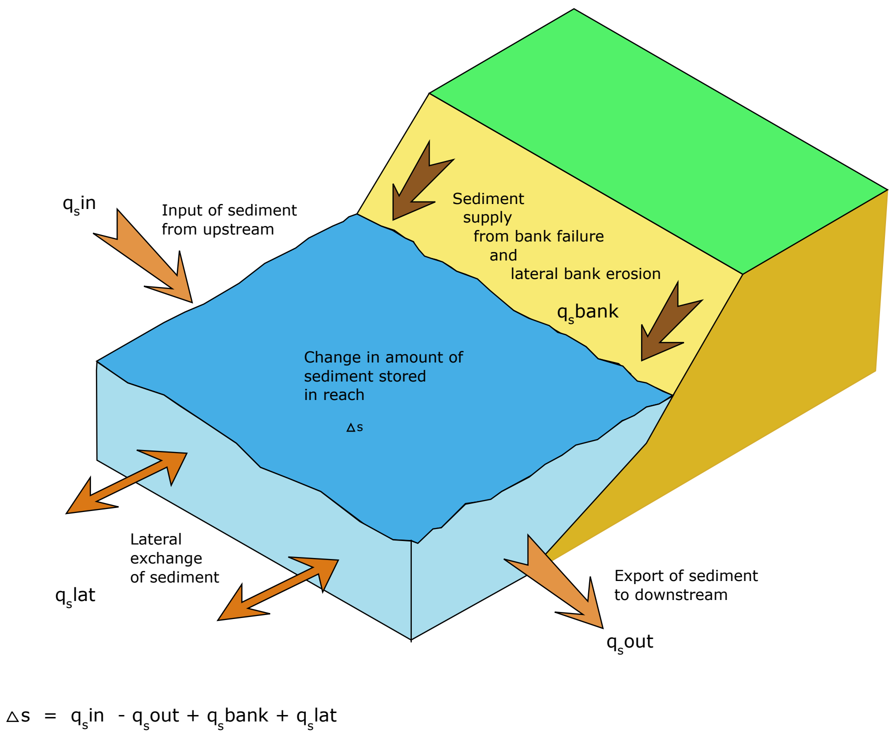


Figure 2.11. River bank erosion as a sediment source in a reach-scale budget (adapted from Thorne, 1991).

inability, until recently, to monitor, *in real time,* the dynamics of the bank erosion processes that drive the specific bank erosion events that introduce fine sediment into streams. This makes it very difficult to assess, using modelling approaches, the linkage between river bank erosion and any resultant increase in fluvial SSCs.

These difficulties have been resolved, to some extent, by the development of the photo-electronic erosion pin (PEEP) system (Lawler, 1991, 2005b, 2008), which allows the magnitude, frequency, timing and duration of bank erosion events, and thus bank sediment contributions, to be monitored automatically and continuously. Examples of bank erosion events are shown for the Upper Severn in Wales in Figure 2.12 (Lawler *et al*., 1997). From this figure, it is apparent that the upper bank failed at midday on 24 August, while the lower bank has registered deposition, upon receiving the collapsed failure block at precisely the same moment. This not only helps to diagnose the bank erosion process responsible (i.e. a geotechnical failure in the case shown in Figure 2.12), but also indicates that at least some of the sediment released from the erosion of the upper banks may be stored at the bank toe, without immediately contributing significantly to fluvial SSLs. Blocks of detached fine sediment may lie at the bank foot for several months and leak sediment more slowly into the river. Furthermore, if the bank erosion process produces

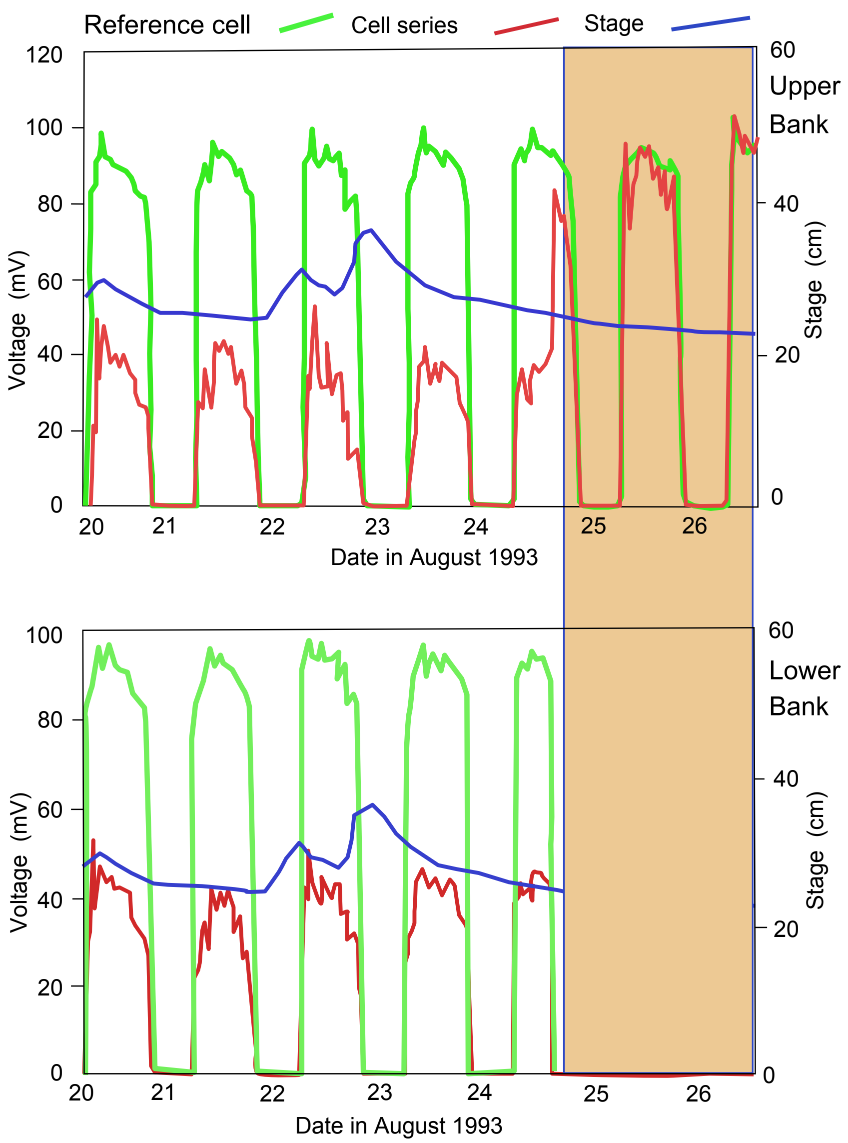


Figure 2.12. River bank erosion events detected automatically with the PEEP system on the River Severn. Note that the upper bank shows bank erosion, while, at the same time, the lower bank shows deposition; this suggests a geotechnical bank failure/block collapse process (adapted from Lawler *et al*., 1997).

fine sediment particles or aggregates, perhaps through freeze–thaw and desiccation processes followed by fluvial entrainment, then erosion processes can immediately and significantly elevate SSCs. Lawler *et al.* (1997) suggested catchment-scale differences in bank erosion process dominance, namely that freeze–thaw processes dominate in the upper reaches of a catchment, where banks are wetter and subject to more frost; fluid entrainment dominates in the middle reaches, where stream powers may peak (Barker *et al.*, 2009); and in the lower reaches, where banks are large and fine grained, geotechnical failure is the most common cause of erosion. Therefore, it is possible that, in cases where bank erosion processes are significant contributors to the sediment budget, more immediate suspended sediment responses are likely in the middle and upper reaches of catchments.

2.3 Deposition and Mobilisation

Flowing water can detach, transport and deposit sediment, whether in overland flow or in channels. Critical shear stress is an important index which characterises this process (Storm *et al.*, 1990). The shear stress required for detachment is typically much greater than required for transport: Foster (1982) reported that the critical shear stress required for the flow detachment of a certain soil was 2.9 N/m^2, while the critical shear stress required for the transport of the same soil was <0.5 N/m^2. Critical shear stress is related to soil shear strength in cohesive soils and is influenced by salinity (Kelly and Gularte, 1981). Lyle and Smerson (1965) reported that critical shear stress is related to plasticity index, percentage clay, mean particle size, dispersion ratio, vane shear strength, organic matter content, cation exchange capacity and the calcium–to–sodium ratio. Smerdon and Beasley (1961) developed regression equations relating critical shear stress to many of these soil properties. In channels, sediment can move, suspended in the turbulent water column, or can roll downstream on the bed of the channel. The latter is called "bed-load" and Yalin (1963) developed an equation to describe the transport rates. Sediment moving along the bed may form characteristic waves and these can influence the hydraulic resistance of the channel (Wang *et al.*, 2011).

The most widely used method for determining critical shear stress is the Shields diagram (Storm *et al.*, 1990). The original Shields diagram used a time and spatially averaged dimensionless shear stress and a particle Reynold's number calculated at the bed. However, it does not apply to sediment particles of low specific gravity and small diameter. Mantz (1977) extended the Shields diagram for smaller particles, but both approaches require iteration. Yalin (1977) introduced a change of variables, defining a modified Shields function, which could be used without the need for iteration.

The movement of relatively large-grained sediments tends to be associated with less frequent but more intense flows, and can be modelled using typical statistical distributions of extremes (Valyrakis *et al.*, 2011). Entrainment is influenced by the velocity profile in the boundary layer (Le Roux, 2010). Scour may occur at specific structures, such as flow deflector vanes used to improve conditions for fish (Rodrigue-Gervais *et al.*, 2010), arched culverts (Crookston and Tullis, 2011) and weirs dal (Muller *et al.*, 2011), which change the flow direction or increase its velocity.

2.4 Construction Activities

Road construction activities in the vicinity of watercourses are potential sources of sediment input, which may originate from the associated earthworks, including blasting; the pumping of water from the construction site; exposed soil banks resulting from excavations or vegetation removal; soil storage areas; or the construction of road crossings (Vice *et al.*, 1969; Barton, 1977; Beshta, 1978; Extence, 1978; Cline and Forest, 1983; Embler and Fletcher, 1983; Duck, 1985; Ellis *et al.*, 1987a; Barrett *et al.*, 1995; Maltby *et al.*, 1995; Luce and Black, 1999; Wellman *et al.*, 2000; Lane and Sheridan, 2002; Bruen *et al.*, 2006; Cerdà, 2007; Purcell *et al.*, 2012). Pipeline crossings are especially troublesome zones for fine sediment ingress into rivers and fluvial sediment impact assessment methodologies have recently been presented by Lawler and Wilkes (2015) for these types of impacts. The risk from the construction of road crossings is also high; the available data on road crossings and suspended solids (SSs) are summarised in Table 2.3. Culverting appears to pose a higher risk of sediment than the use of clear span bridges (Cocchiglia *et al.* 2012). The limited studies that relate these sediment inputs from road crossing to effects on aquatic

Table 2.3. Summary of studies that have documented an increase in SSs downstream of river crossing construction sites

Crossing type	SS concentration (mg/L) before construction/ control	SS concentration (mg/L) during construction	Note	Reference
Culvert	<5	1390	Maximum values	Barton (1977)
Pipeline	7	7620	Maximum	Tsui and McCart (1981)
Bridge foundations	3.2	15.8	Mean	Cline *et al.* (1983)
Culvert	3–17	75–81	Range	Cline *et al.* (1983)
Culvert	<30	60–130	Range	Embler and Fletcher (1983)
Unknown	35	179	Mean	Barrett *et al.* (1995)
Culvert	144	1237	Maximum values	Lane and Sheridan (2002)
Unknown	5	15	Maximum values	Chen *et al.* (2009)
Culvert	<20	70	Maximum	Purcell *et al.* (2012)

communities include Extence (1978) and Cocchiglia *et al*. (2012). Post-construction, sediment and various other pollutants have been detected in motorway drainage (Maltby *et al*., 1995, e.g. Bruen *et al*., 2006a).

2.5 External Discharges, Urban Drainage, Wastewater Treatment Plants and Farmyard Drains

Most of the sediment input to rivers from point sources is associated with local rainfall events, typically with the intense but relatively short-duration rainstorms. Such point sources include urban stormwater drainage, sewer or wastewater treatment plant (WWTP) overflows, road runoff and farmyard drainage (Table 2.4).

Sediment delivery from WWTP discharges, particularly in areas with combined sewers, and urban stormwater drainage systems is complicated by the effects of deposition within the pipe networks and subsequent mobilisation during storm flows (Walling *et al*., 2003a). Much of the sediment can be contained in settling and storage tanks on site, particularly if tanks are designed to accommodate the “first flush” load (Walling, 2006). However the organic content and nutrients in wastewaters may have a greater effect than sediment on stream ecology (Horowitz, 1991).

Sediment concentrations in urban stormwater depend on whether the urban area is residential, commercial or mixed. Median event mean concentrations tend to be in the range of 100 to 300 mg/L, but values as high as 1240 mg/L have been observed in residential areas (Van Rompaey *et al*., 2001). Total suspended solids (TSSs) are typically reduced in stormwater by detention or settling ponds (Carter and Berg, 1983; Sarangi *et al*., 2004; Richards *et al*., 2008). The process of urbanisation initially produces a large increase in sediment load as land is cleared for construction (Wolman, 1967). This sediment load will decline as more impermeable surfaces are built. However, the reduction in the sediment load in the stream may result in channel erosion, and a deepening

Table 2.4. Major point sources of sediment

Point source	Reference
Urban stormwater drainage	Carter and Berg (1983), Sarangi *et al*. (2004), Richards *et al*. (2008)
WWTP discharges	Walling *et al*. (2003a)
Road runoff	Walling (1983), Croke *et al*. (2006), Desta *et al*. (2007)
Farmyard drainage	Van Oost *et al*. (2000), Krasa *et al*. (2010)

and widening of the urban channel, if the bed material permits (Leopold, 1973). Some examples from the USA are listed by O'Driscoll (2010).

Roads are sources of sediment in runoff. In urban areas, the runoff is mainly to the urban stormwater system, but in rural areas it discharges to local streams, perhaps after some treatment in, for example, a settling/attenuation pond or wetland (Schutes *et al*., 2001). In one UK study, damaged road verges contributed up to 20% of the sediment from catchments, and this contribution seems to have increased over time (Collins *et al*., 2010b). A "first flush" effect of heavy metals is typical of runoff from busy highways (Walling, 1983), regardless of whether the road is unsealed (Croke *et al*., 2006) or sealed (Ellis *et al*., 1987b). Bruen *et al*. (2006) reviewed the typical constituents of highway runoff and their impacts, and Desta *et al*. (2007) described a study in Ireland, in which all TSS measurements in road runoff were considerably less than 1000 mg/L. In rural areas, the sediment loads in farmyard runoff can be reduced by deposition in either natural or constructed wetlands (Van Oost *et al*., 2000; Krasa *et al*., 2010).

3 Physical and Chemical Impacts of Fine River Sediments in Fluvial Systems

3.1 Importance and Processes

Despite the potential negative physical impacts of suspended sediment (the ecological impacts are discussed in Chapter 4), a little suspended sediment can limit water clarity and, therefore, the over-production of algae. Fluvial suspended sediment data can also help to identify catchment processes, such as:

1. upstream sediment mobilisation and hence data that can provide alerts to potential erosion problems, including the liberation and delivery processes (e.g. soil erosion, rilling and gullying, and landslide and debris flow significance) that route fine sediment to downstream receptors;
2. catchment erosion and sediment yields (e.g. for reservoir design purposes);
3. downstream sedimentation (e.g. in lakes, nearshore zones and harbours);
4. sediment fluxes and any associated delivery of contaminants to receptors.

However, the presence of fine sediment in catchment and fluvial systems can also have undesirable environmental impacts, because sediment can change the physical, chemical and biological properties of aquatic ecosystems. These impacts depend on the SSCs in the water column; the particle size and shape distribution; particulate behaviour and ingress into river bed gravels; sediment quality; sediment-associated contaminants; sediment fluxes; and storage characteristics/dynamics in the floodplain (Walling *et al*., 1999; Coulthard and Macklin, 2003; Walling *et al*., 2003b; Dennis *et al*., 2009), channel bed and hyporheic zone.

3.2 In-stream Processes

Once in the channel, and especially once suspended in the water column in sufficient quantities, particulate matter can make the water turbid, at both high (Figure 3.1) and low flow, especially in urban settings (Figure 3.2). This reduces the level of light that can reach the bed and affects aquatic habitats and organisms, especially predation for fish (Walling and Fang, 2003), as discussed in Chapter 4. Methods for modelling the factors that control photosynthetically active radiation in rivers, including turbidity, have recently been developed using the Benthic Light Availability Model (Julian *et al*., 2008a,b).

Sediment can also have physical abrasive “sandblasting” effects on organisms (e.g. damage to fish gills; see section 4.4) and on riverine structures, such as bridges, intakes and turbines. In addition, sediment physically interacts with the river channel

Figure 3.1. Turbid waters at high flow in the River Alne, near Little Alne, Warwickshire, UK. Flow from right to left (photo: Damian Lawler).

Figure 3.2. Turbid conditions in the urban Bournbrook stream, River Tame catchment, Birmingham (photo: Damian Lawler).

in a number of other ways (Lawler and Fairchild, 2010). "Sedimentation" (or "siltation") refers to the development of a layer of fine sediment *over* the bed surface. "Sediment infiltration" (or "colmation", as it is known in the environmental engineering literature) is the process by which fine sediment moves *into* the gravel bed structure itself (Figure 3.3). "Accumulation" is the summation of this infiltration process over time (Sear *et al*., 2008).

In the *Hyporheic Handbook*, Lawler *et al*. (2009) briefly summarise the processes involved in sediment infiltration: "in the water column, fine sediment movements are driven by two main processes: (i) gravity-driven infiltration that includes simple Stokes-type settling; and (ii) advection of fine material into the bed by fluid turbulence. All else being equal, coarser and heavier particles will drop out of suspension first, giving natural spatial and temporal size segregation in the resulting deposits. Particle shape is also a key factor, as the less spherical a particle, the slower it will settle." If fine particles flocculate they can settle in an unpredictable manner.

The hyporheic zone is the zone in which groundwater and surface water mix beneath and around the channel bed (Alley *et al*., 2002) (Figure 3.4). This zone is especially vulnerable to sediment ingress problems (see, for example, Sear *et al*., 2008; Lawler *et al*., 2009). This partly relates to a difficulty in flushing out fine sediment once it has become ingressed into the bed gravels. Data from a sample of UK stream beds,

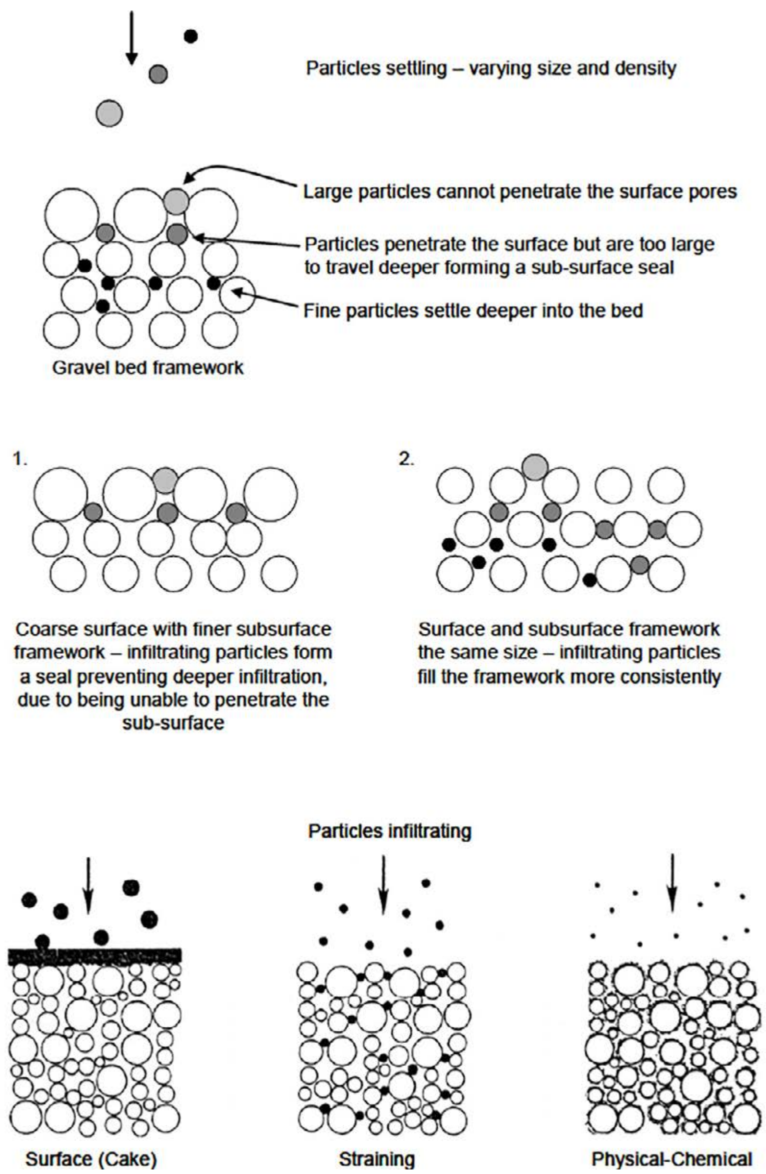


Figure 3.3. Sediment infiltration mechanisms (Sear *et al*., 2008).

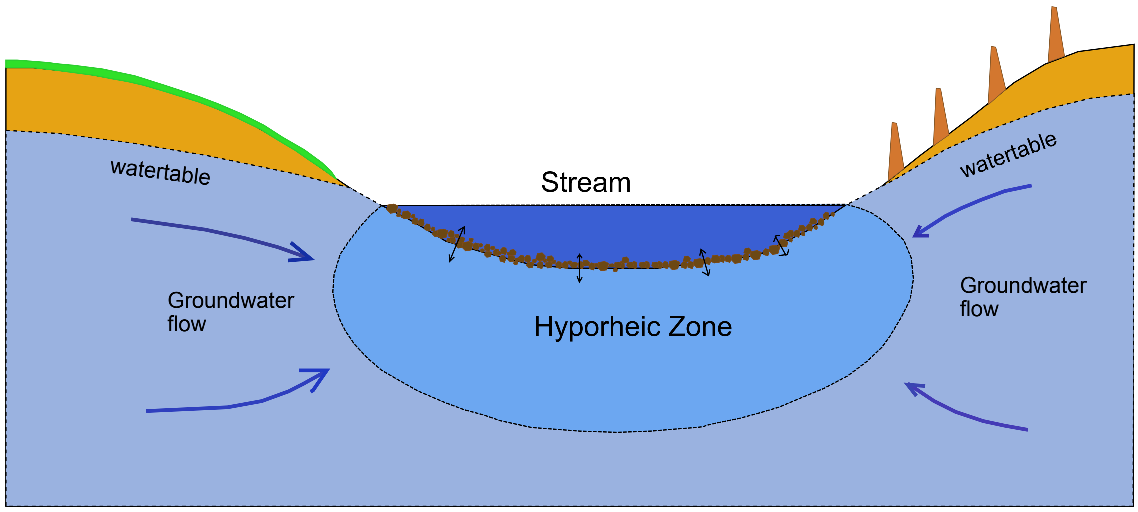


Figure 3.4. The hyporheic zone.

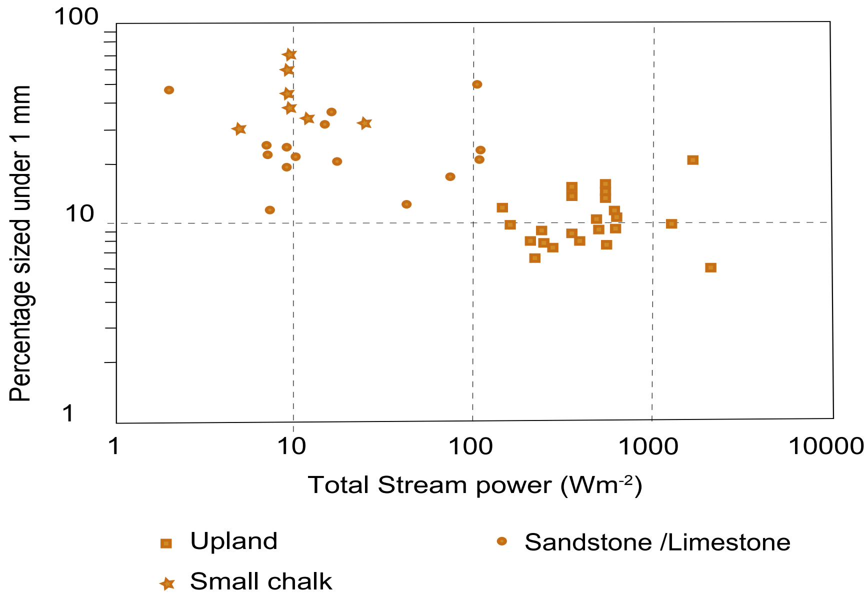


Figure 3.5. The relationship between stream power and sediment size in UK stream types: upland (Type I), small chalk (Type 2) and sandstone/limestone (Type 3) (after Milan *et al.*, 2000).

shown in Figure 3.5, suggest that there is a reasonably inverse relationship between specific stream power (a measure of the energy available per unit area of a river bed) and the percentage of sediment of < 1 mm in diameter (Milan *et al.*, 2000). If this is the case in general, it should be possible to use the high-resolution (60 m longitudinal spacing) catchment-scale quantifications of downstream changes in channel elevation, slope, bankfull discharge, and gross and specific stream power, produced for 32 UK rivers using the Combined Automated, Flood, Elevation and Stream Power (CAFES) system of Barker *et al.* (2009), as a basis to predict the fine sediment content of river beds (e.g. Figure 3.6). The resulting data could then be employed to help identify, in combination with other indices, hot-spots of ecological sensitivity.

Fine sediment in gravels can also consume oxygen and, therefore, significantly reduce organisms' oxygen supply (Greig *et al.*, 2007) (see section 4.3.1). Figure 3.7 shows that a simple doubling of sediment accumulation rate in artificial redds can reduce oxygen supply by an order of magnitude (Greig, 2004). Sedimentation also changes channel geometry, and if reductions in channel capacity are significant, then flow velocities (and shear stresses) will increase to

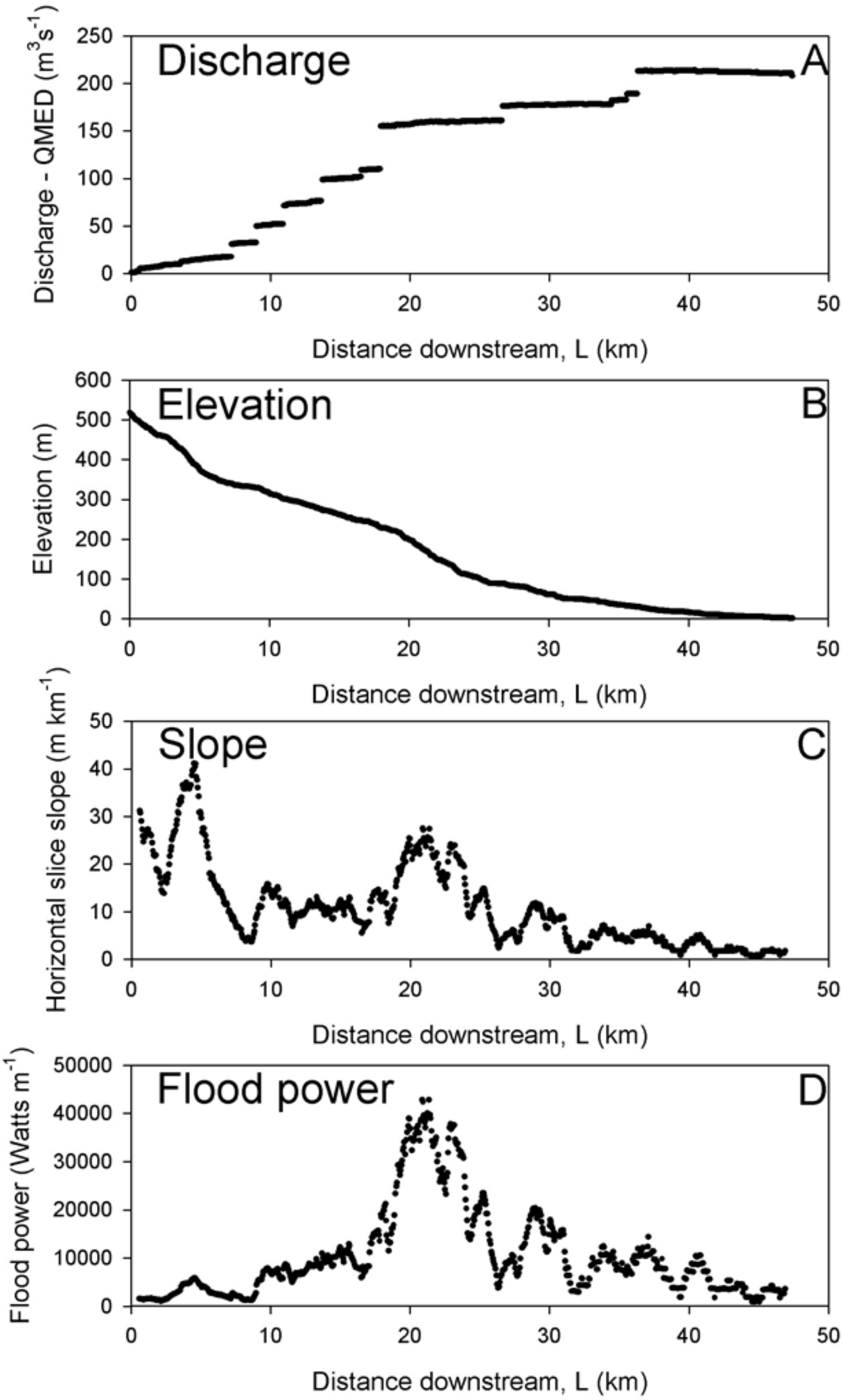


Figure 3.6. Downstream change in the hydraulic properties of the River Dart, south-west England. Sites of erosion are likely to relate to peak stream power, which in many catchments occurs in mid-basin, in which the optimum combination of water surface slope and bankfull discharge [median annual discharge (QMED), i.e. 2-year return period flow] is found, as in the River Dart in this example. The full derivation of variables was performed using the CAFES methodology (Barker *et al*., 2009). The results shown here are from Barker *et al*. (2009). Sediment deposition and associated impacts are likely at stream confluences and in areas of low stream power, at which particle settling velocities are achieved (Buss, 2009).

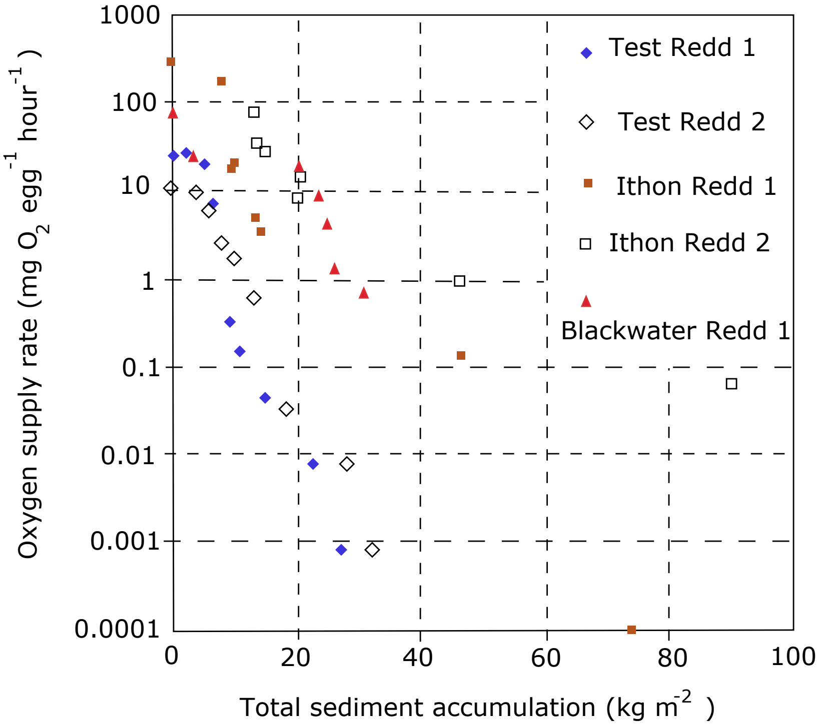


Figure 3.7. Decline in oxygen supply rate with the accumulation of fine sediment within artificial redds. These data are based on Greig, 2004.

balance the continuity equation. This can be beneficial in some cases (e.g. to flush out accumulated fine sediment), but it can also increase channel erosion and bedform instability, perhaps undesirably.

Because of these negative impacts, and those discussed in Chapter 4, organisations around the world have established limits for SSCs, turbidity and/or sediment fluxes. For example, the UK Technical Advisory Group (UKTAG, 2008) suggested that the guideline standard for annual mean SSs in freshwaters should be 25 mg/L; this is consistent with the EU Freshwater Fish Directive (Lazar *et al*., 2010). The EU Water Framework Directive (WFD) required EU Member States to achieve "good ecological status" for rivers by 2015, and this status was defined by Collins *et al*. (2007) in terms of the guideline annual average SSC of 25 mg/L of the EU Freshwater Fish Directive. Other suggested SSC thresholds are given in Table 1.1.

3.3 Sediment-associated Pollutants

The role of sediments in the transport and dispersal of river pollutants has received considerable attention since the 1970s. Gibbs (1973) and Martin and Meybeck (1979), for example, suggested that more than 90% of pollutants are transported with sediments and, more recently, Turner *et al*. (2008) showed that between 71% [copper (Cu)] and 99% [arsenic (As), lead (Pb) and zinc (Zn)] of selected heavy metals were sediment bound in waters buffered by calcareous soils, under high wash sediment loads (up to 10,000 mg/L). Significant amounts of the nutrients nitrogen and phosphorus have also been widely reported for sediment-borne fluxes in British and Irish catchments (see, for example, Foster *et al*., 1996). However, in a review of the controls on nutrient fluxes in British rivers, Walling *et al*. (2001) emphasised that the "precise magnitude of the sediment-associated component will vary from river to river in response to local conditions including relief, geology and land use, the hydro-
meteorological conditions and the relative importance of point source inputs". Similarly, at a reach scale, the morphological controls governing sediment dynamics strongly influence the transport and storage of radiothorium downstream of a mining release (Graf *et al*., 1991), while Droppo *et al*. (2011) have demonstrated that bacterially induced fine sediment flocculation can lead to an increase in the "downwards flux" of contaminated sediment, which promotes elevated pathogen distribution in bed sediments at a patch scale (see, for example, Cho *et al*., 2010).

There have been numerous reviews and case-study papers on the variety and source of sediment-associated pollutants' distribution and effects in aquatic environments; a number of these are summarised in Table 3.1. Notably, relatively few investigations have been conducted in Ireland, although Herr and Gray (1997), and a follow-up study by Gaynor and Gray (2004), did report elevated levels of sediment-associated Cu, Pb and Zn as a result of acid mine drainage in the Avoca catchment, south-east Ireland, and Regan *et al*. (2012) have estimated phosphorus exports of 0.88 to 8.8 kg/ha per year from agricultural soils in Ireland, based on total phosphorus soil values (after McGrath *et al*., 2001) and soil erosion modelling (Van Oost *et al*., 2006).

Pollutants can be introduced as particulates that move as part of the saltating load, but are more often transported in association with the "chemically active", fine (<63-μm diameter) sediment fraction, in suspension (Horowitz, 1991), through a variety of processes (adsorption, absorption and precipitation) (see Figure 3.8). Sediment-associated pollutant concentrations are therefore strongly influenced by particle size, while total pollutant fluxes are primarily governed by pollutant mobilisation, coupled with sediment supply and hysteresis dynamics (Bradley and Cox, 1990); thus, very significantly, most sediment-associated pollutant transport occurs during storm events (Horowitz *et al*., 1999; Horowitz, 2006; Edwards and Withers, 2008). The disassociation of pollutants from the dissolved,

Table 3.1. Selected examples of sediment-associated contaminants, their sources and their effects on fluvial systems (adapted from NRC, 2003; Owens *et al*., 2005; Taylor and Owens, 2009). Brief details of research conducted in Ireland are given in Italics

Pollutant	Sources and distribution	Environmental impacts	Selected examples
Metals and metalloids (Ag, Cd, Cu, Co, Cr, Hg, Ni, Pb, Sb, Sn, Tl, Zn, As)	Mining including catastrophic release, industry, acid rock/mine drainage, sewage treatment, urban runoff, pesticides	Toxicological effects Oxidisation leading to de-oxygenation Potentially long residence times (102–103 years) Diffuse source of labile contaminants and prone to physical reworking	Herr and Gray (1997)/Gaynor and Gray (2004) (*Ochre precipitates in the Avoca River*) Miller (1997), Macklin *et al*. (2003), Walling *et al*. (2003b)
Nutrients (P, N)	Agricultural and urban runoff, wastewater and sewage treatment	Eutrophication	Regan *et al*. (2012) (*Review of P exports from tillage*) Kiely (2007) (*P exports from grasslands*)
Organic compounds including pesticides, herbicides, hydrocarbons, PCBs, PAHs, dioxins	Agriculture, pest control, industry, sewage, landfill, urban runoff, forest fires, incineration	Toxic impacts, including carcinogenesis Endocrine disruption affecting sex hormones and reproductivity Environmental persistence and bioaccumulation	Thomas (1990), Potter *et al*. (1994), Warren *et al*. (2003)
Xenobiotica and antibiotics	Sewage treatment works, industry, agriculture		
Steroid hormones (e.g. androgens and oestrogens)	Sewage treatment works	Endocrine disruption	Taylor and Harrison (1999)
Radionuclides (^{137}Cs, ^{238}Pu, ^{239}Pu, ^{240}Pu, ^{235}U, ^{238}U, ^{230}Th, ^{99}Tc)	Nuclear power industry, military, geology, food irradiation		
Microbes, including pathogens (*Escherichia coli*, faecal coliforms)	Agricultural runoff, sewage, landfill	Formation of flocs leads to bed concentration of pathogens	Jamieson *et al*. (2004), Cho *et al*. (2010), Droppo *et al*. (2011)

Ag, silver; Cd, cadmium; Co, cobalt; Cr, chromium; Cs, caesium; Hg, mercury; N, nitrogen; Ni, nickel; P, phosphorus; PAH, polycyclic aromatic hydrocarbon; PCB, polychlorinated biphenyl; Pu, plutonium; Sb, antimony; Sn, tin; Tc, technetium; Th, thorium; Tl, thallium; U, uranium.

Figure 3.8. Sediment pollution event in the nearshore zone derived from erosion of a coastal catchment during an intense Mediterranean rainstorm, east-central Spain, 24 August 1997 (photo: Damian Lawler).

aqueous phase generally reduces their bioavailability and environmental impact. However, fine sediments continue to represent a risk to the environment through their ingestion and, importantly, represent a major secondary source of pollution. The diffuse nature of the sediment-associated pollutants stored in the channel, and the channel marginal and floodplain sedimentary environments, makes them more difficult to manage (Owens *et al*., 2005). Moreover, some pollutants with long residence times (up to 1000 years) can be re-introduced to surface waters, through chemical exchange and channel re-working, long after their initial release into the fluvial environment (Miller, 1997; Macklin *et al*., 2003; Dennis *et al*., 2009). In many instances, keeping rivers "clean" of fine sediment should, therefore, enhance their ability to flush through pollutants, leading to significant environmental benefits in the medium term.

3.4 Impacts on River Morphology

Form–sediment interactions can result in distinctive channel planforms (Figure 3.9) and bedforms (Table 3.2), which are largely governed by sediment calibre and supply rate, together with local flow conditions. Although sediment moves through a channel, bedform assemblages, such as riffle-pool sequences, are equilibrium forms that are morphologically stable and provide habitats for aquatic species (Brown and Brussock, 1991; EA, 2009). Sediment transfer also promotes short- and long-term storage of fine-grained sediment in a variety of alluvial depositional settings (Table 3.3). Typically, channel form–sediment dynamics are dominated by bedload sediment characteristics, but fine sediment can be an important constituent of mixed bed channels, in which it is stored in the interstices of coarser channel substrate and deposited on bar tops during waning high flows (Church, 2006). Fine sediment can build up (sometimes detrimentally) in channel pools and in the lee of in-channel obstructions, such as woody debris and man-made structures. In the upper portion of alluvial channel banks, fine sediment plays an important morphological role, because of its hydraulic and cohesive properties, and provides a store for nutrients and, in some cases, contaminants. Fine sediment predominates in the alluvial sedimentary environments of low-energy systems. Morphologically, these river systems are typically laterally stable and can display anabranching, anastomosing (*sensu* Nanson and Knighton, 1996) planforms, characterised by multiple, hydraulically independent channels separated by stable, vegetated floodplain islands. These equilibrium channel planforms are now rare, because of human disturbance and modification, but rudimentary morphological elements in the form of channel islands can still be found across north-west Europe, including in Irish catchments such as those

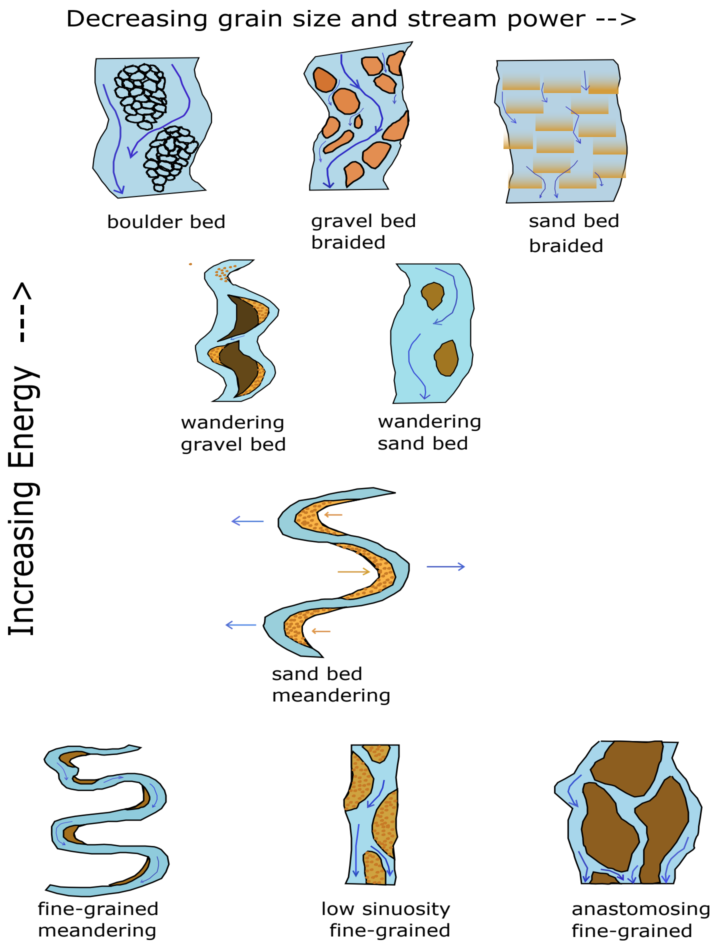


Figure 3.9. The continuum of channel planform variants of alluvial river morphology along an energy gradient is closely related to predominant sediment load and channel stability (after Brierley and Fryirs, 2005, p. 121).

of the Rivers Shannon (County Limerick) and Boyne (County Meath), and are highly developed in parts of the River Lee (County Cork).

3.5 Impacts on Riverine Structures

Suspended sediment particulates can be sharp and angular, and abrade in-stream structures such as bridges, flood defences, hydroelectric power turbines (e.g. in Iceland and Norway) and dam surfaces, particularly spillway chutes (Huang, 2006). They can also clog water intakes, creating problems in water treatment works and increasing the costs of sediment removal. Floodplain and main channel sedimentation can also become problematic. Turbulent diffusion can contribute greatly to sediment transport (Shiono, 2011), and the abrasion of coarse sediment during transport may produce fine silt (Sklar *et al*., 2006).

3.6 Downstream Effects of Siltation in Rivers, Lakes, Reservoirs and Harbours

Siltation processes can have a significant impact on fluvial systems and water resource management in river catchments, and can also account for the low SDRs reported for many river basins (see section 2.3). Fine sediments are temporarily stored in-channel, in substrate interstices and bar tails (section 3.5). Artificial channels, such as drainage ditches (also known as field drains), both capture and supply fine sediment from agricultural land (Forster and Abrahim, 1985). An increase in the drain density will improve hydrological and sediment connectivity, but can lead to elevated particulate phosphorus transfer to aquatic ecosystems (Blann *et al*., 2009) and high sediment yields.

Table 3.2. Bedform classification system (modified from Church and Jones, 1982a, and Knighton, 1998)

Bedform	Dimensions	Shape	Behaviour and occurrence
Small-scale forms	*(10^{-2}–10^{2} m)*		
(1) Sand-bed rivers			
Ripples	$\lambda < 0.6$ m; $H < 0.04$ m	Triangular profile, sharp crest perpendicular to flow direction	Generally restricted to sediment finer than 0.6 mm
Dunes	λ is 4- to 8-fold higher than flow depth; H is up to 1/3 of flow depth	Curved crests	Upstream slope may be rippled, coarse grains deposited at crest, flow separation occurs
Plane bed		Flat surface	Super critical flow, Froude No. > 1
Antidunes	H is dependent on flow depth/velocity	Sinusoidal profile	Antidunes move upstream, sediment moves downstream
(2) Gravel-bed rivers			
Pebble clusters	0.1–1 m	Linear in direction of flow	Large obstacle particle with smaller stoss particle
Transverse ribs	1–10 m	Transverse to direction of flow	Repeated ridges of coarse particles, spacing proportional to largest particle in ridge crest
Riffle/pool sequence	1–10 m	Transverse to direction of flow	Alternative deep (pool) and shallow (riffles) spaced in relation to channel width
Step/pool systems	1–10 m	Transverse to direction of flow	Star-like sequence formed in steep channels, steps formed coarse material, spacing ≈2- to 3-times the channel width
Large-scale forms	*(10^{1}–10^{3} m)*		
Bars	Length comparable to channel width	Variable	Longitudinal bars, form in channel centre and elongated in direction of flow
			Transverse bars, lobe shape with relatively steep downstream face
			Point bars, form in meandering channels because of secondary flow
			Diagonal bars, bank attached bars running obliquely across channel
			Mid-channel bars, common in braided channels

λ, wavelength; H, height.

Table 3.3. Alluvial depositional environments in which fine sediments may accumulate (modified from Church and Jones, 1982b, and Hoey, 1992)

Depositional site	Name	Characteristics
Within channel	Transient channel deposit	Temporarily stagnant bedload deposits including ripples, dunes, transverse ribs, pebble clusters and steps
	Alluvial bars	Lag deposits of coarser sediment including riffles, mid-channel bars and sedimentation zone
Channel margin	Lateral deposits	Point bars that form on the inside of meander bends
Floodplain	Vertical accretion deposits	A comparatively flat alluvial depositional landform that forms as a result of deposition of fine-grained suspended load of overbank floodwaters; provides sediment storage space for drainage basin
Piedmont	Alluvial fans	A cone-shaped depositional feature that occurs because of a sudden reduction in sediment transport capabilities owing to an abrupt change from confined to unconfined conditions, or a sudden decrease in slope; sediment grain size decreases rapidly with distance from fan apex
River mouth	Deltas	Morphological feature formed when a river enters a sea or lake and deposits its load; characteristics of sediment supply determine morphology

Fine sediments naturally accumulate in natural catchment sinks, such as lakes, flood basins and estuaries, but can be affected by changes in sediment supply. For example, Lake Tahoe, on the California/Nevada border, has traditionally attracted many tourists; however, this tourism is being threatened by an apparent increase in turbidity resulting from an increase in the sediment and nutrient supply (Langlois *et al*., 2005; Simon *et al*., 2008). In artificially constructed reservoirs and impounded rivers, the effects of siltation can also have a detrimental impact on storage capacity, particularly if trap efficiencies are underestimated; this impacts upon a variety of uses, such as flood control, domestic water supply, irrigation, recreation and fish farming. Siltation can also exert pressure on dam walls, increasing the risk of dam failure; while these events are rare, they can have catastrophic downstream effects, because of the mobilisation of trapped fine sediment (e.g. Evans, 2000, 2007).

Harbours and marinas often suffer from fine sediment accumulation, which leads to high maintenance costs for dredging and the disposal of spoil, particularly if it is contaminated (Owens, 2005). These effects can result from the downstream deposition of fine sediment (Mitchell, 2005; Pontee and Cooper, 2005), together with the re-suspension of fine sediment from tidal flats (Dobereiner and McManus, 1983).

4 Ecological Impacts of Fine River Sediments in Fluvial Systems

4.1 Introduction

Excessive sediment inputs can have detrimental effects on fluvial systems, negatively impacting biota in a variety of complex direct and indirect ways (Figure 4.1), from reducing primary productivity to altering faunal abundance, diversity and community structure. The key papers documenting such effects on aquatic biota are summarised in Table 4.1, whilst Table 4.2, Table 4.3 and Table 4.4 summarise the available studies that have considered the influence of sediment concentration and the duration of exposure.

4.2 Periphyton and Macrophytes

4.2.1 *Periphyton*

Periphyton is, to some extent, more susceptible to effects resulting from elevated sediment levels than macrophytes. Suspended and saltating solids alter light penetration within the water column, and thereby reduce primary production (Van Nieuwenhuyse and LaPerriere, 1986; Lloyd *et al*., 1987) and lead to a decrease in periphyton biomass (Davies-Colley *et al*., 1992; Quinn *et al*., 1992; Francoeur and Biggs, 2006;

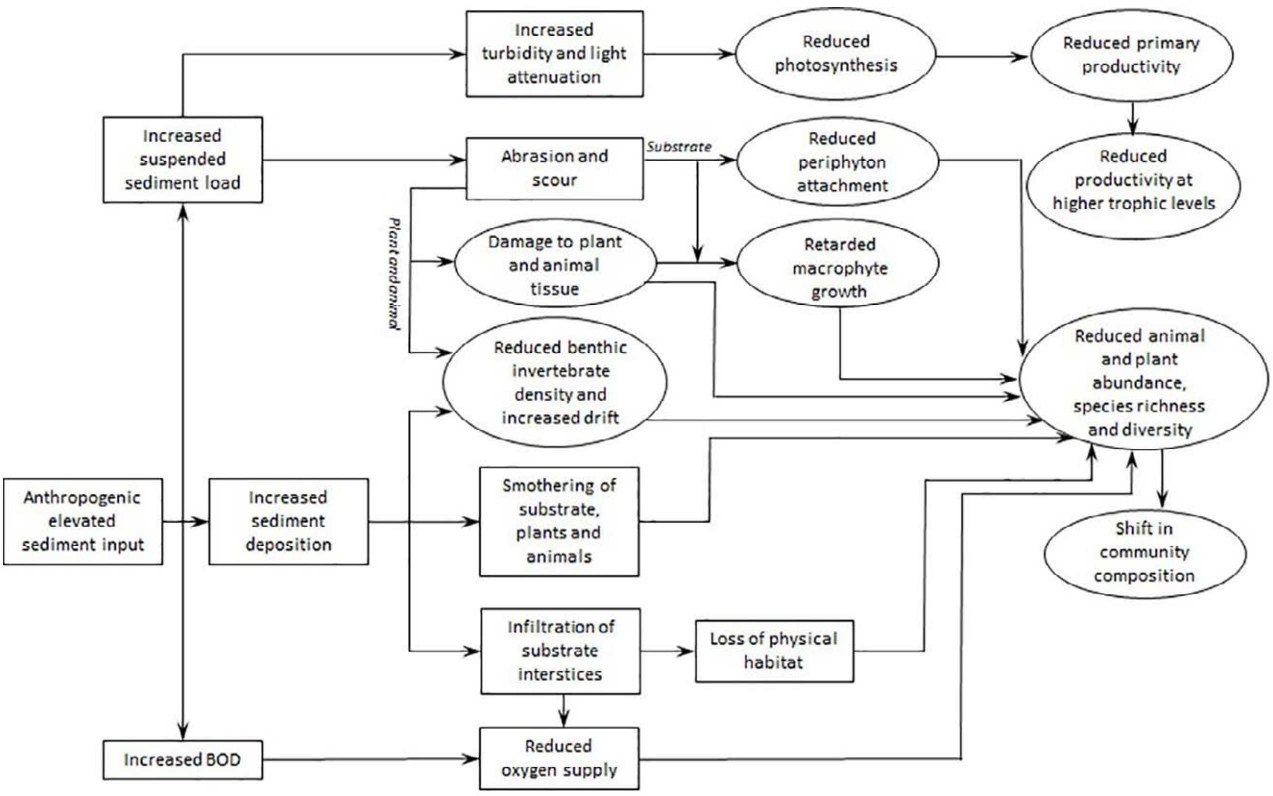


Figure 4.1. Negative impacts of anthropogenically enhanced sediment on lotic aquatic systems. Rectangles represent physicochemical effects, ovals represent direct and long-term biological and ecological responses (after Kemp *et al*., 2011). BOD, biological oxygen demand.

Table 4.1. The ecological impact and sources of suspended and deposited sediment in rivers (modified from Wood and Armitage, 1997, and Jones *et al.*, 2011)

Impact	Sediment type	Source	Reference(s)
Primary producers			
Absence of rooted vegetation	D	China clay extraction	Nuttall and Bielby (1973)[a]
Elimination of macrophytes	D	Channelisation	Brookes (1986)[a]
Reduced productivity, biomass and organic content	S and D	Placer gold mining	Van Nieuwenhuyse and LaPerriere (1986), Davies-Colley *et al.* (1992)[b,c]
Reduced species diversity and organic content	S and D	Road construction	Cline *et al.* (1982)[b]
Reduced organic content	D	Impoundment	Graham (1990)[c]
Macroinvertebrates			
Reduced abundance	D	Drought – abstraction	Wright and Berrie (1987), Wood and Petts (1994)[a]
Reduced abundance and diversity	D	China clay extraction	Nuttall (1972), Nuttall and Bielby (1973)[a]
Reduced abundance and diversity	S and D	Desilting operation	Doeg and Koehn (1994)[e]
Reduced species diversity	D	Removal of riparian vegetation	Armstrong *et al.* (2005)[e]
Reduced species density	S and D	Placer gold mining	Quinn *et al.* (1992)[c]
Reduced species density	S and D	Induced in field experiment	Angradi (1999), Zweig and Rabeni (2001)[b]
Reduced species density and diversity	D	Water filtration facility	Erman and Ligon (1988)
Reduced richness	D	Agriculture	Lemly (1982)[b]
Reduced taxon and EPT richness	D	Induced in field experiment	Matthaei *et al.* (2006)[c]
Altered composition and reduced EPT richness (patch scale)	D	Bank erosion	Larsen *et al.* (2009)[a]
Reduced EPT richness	D	Agricultural	Kaller and Hartman (2004)[b]
Altered functional composition/reduced abundance of filterers	D	Induced in field experiment	Bo *et al.* (2007)[f]
Impaired filter-feeding and reduced metabolic rate of mussels	S	Induced in experiment	Aldridge *et al.* (1987)[b]
Altered community structure/reduced density/ increased drift	S and D	Induced in experiment	Rosenberg and Wiens (1978); Larsen and Ormerod (2010)[d,a]
Altered community structure	D	Road construction	Extence (1978)[a]
Altered community structure	S and D	Agriculture	Richards *et al.* (1993)[a]
Decreased density of functional feeding groups and increased density of gathers.	D	Natural	Rabení *et al.* (2005)[b]
Altered richness, abundance, community composition, trait diversity and trait composition	D	Induced in field experiment	Larsen *et al.* (2011)[a]
Altered species/EPT density, richness and abundance	D	Altered land use	Niyogi *et al.* (2007)[c]
Reduced egg hatching	D	Induced in experiment	Kefford *et al.* (2010)[e]
Increased drift	D and Sa	Induced in field experiment	Culp *et al.* (1986)
Increased *Baetis* mayflies drift	D		Gibbins *et al.* (2005)[a]

Table 4.1. Continued

Impact	Sediment type	Source	Reference(s)
Fish			
Reduced abundance	D and S	Desilting operation	Doeg and Koehn (1994)[e]
Reduced abundance	D	Agriculture	Berkman and Rabeni (1987)[b]
Decline in salmonid spawning habitat quality	D	Natural	Carling and McCahon (1987)[b]
Decline in salmonid spawning habitat quality	D	Coal mining	Turnpenny and Williams (1980), Lisle (1989)[a,b]
Decline in salmonid spawning habitat quality	D	Impoundment	Sear (1993)[a]
Reduced survival of salmonid eggs	D	Water filtration facility	Erman and Ligon (1988)[b]
Reduced survival of salmonid eggs	D	Agriculture	Soulsby *et al.* (2001)[a]
Reduced survival of salmonid embryos	D	Laboratory and field experiments	Greig *et al.* (2005)[a]
Reduced alevin survival	D	Laboratory and field experiments	O'Connor and Andrew (1998)[g]
Reduced survival to pre-eyed, eyed and hatched stages	D	Induced in field experiment	Julien and Bergeron (2006)[d]
Increased mortality of sac fry	D	Placer gold mining	Reynolds *et al.* (1989)[d]
Decreased growth and survival of juveniles	D	Induced in field experiment	Suttle *et al.* (2004)[b]
Reduced length and mass	D	Induced in field experiment	Shaw and Richardson (2001)[d]

[a]Country of study: UK.

[b]Country of study: USA.

[c]Country of study: New Zealand.

[d]Country of study: Canada.

[e]Country of study: Australia.

[f]Country of study: Italy.

[g]Country of study: Ireland.

D, deposited sediment; D and Sa, deposited and saltating sediment; EPT, index named after the orders Ephemeroptera, Plecoptera and Trichoptera; S, suspended sediment.

Table 4.2. The effects of varying the concentrations of and the duration of exposure to suspended sediment on periphyton and macrophytes (from Bilotta and Brazier, 2008)

Organism	SSC (mg/L)	Exposure time (hours)	Effect on organism	Reference
Macrophytes and algae	8	–	3–13% reduction in primary production	Lloyd *et al.* (1987)[b]
	40	–	13–50% reduction in primary production	Lloyd *et al.* (1987)[b]
	200	–	50% reduction in primary production	Van Nieuwenhuyse and LaPerriere (1986)[b]
	2100	–	No primary production	Van Nieuwenhuyse and LaPerriere (1986)[b]
Phytoplankton	10	1344	40% reduction in algal biomass	Quinn *et al.* (1992)[c]
Aquatic moss	100	504	Extensive abrasion of leaves	Lewis (1973)[a]
	500	168	Severe abrasion of leaves	Lewis (1973)[a]
Periphyton	100	–	Enhanced growth and filament length (low-flow velocities)	Birkett *et al.* (2007)[b]
	200	–	Significant reduction in biomass and filament length	Birkett *et al.* (2007)[b]
	0–6500	–	Abrasive damage and reduced biomass	Francoeur and Biggs (2006)[c]

[a]Country of study: UK.

[b]Country of study: USA.

[c]Country of study: New Zealand.

Table 4.3. The effects of varying the concentrations of and the duration of exposure to suspended sediment on macroinvertebrates (from Bilotta and Brazier, 2008, Collins *et al*., 2011, Jones *et al*., 2011)

Organism	SSC	Exposure time (hours)	Effect on organism	Reference
Cladocera and Copepoda	300–500 mg/L	72	Clogging of gills and gut	Alabaster and Lloyd (1982)[g]
Diptera	> 50 mg/L	–	Feeding inhibition	Kurtak (1978), Gaugler and Molloy (1980)[b]
Plecoptera/Trichoptera	1.5 mg/L	336	Feeding inhibition	Hornig and Brusven (1986)[b]
Bivalvia	600 mg/L	Intermittent exposure	Feeding inhibition/reduced metabolism	Aldridge *et al*. (1987)[b]
Cladocera	82–392 mg/L	72	Reduced reproduction and survival rates	Robertson (1957)[b]
Chironomids	300 mg/L	2016	90% reduction in population size	Gray and Ward (1982)[b]
Benthic invertebrates	62 mg/L	2400	77% reduction in population size	Wagener and LaPerriere (1985)[b]
Benthic invertebrates	743 mg/L	2400	85% reduction in population size	Wagener and LaPerriere (1985)[b]
Invertebrates	Pulses	456	Reduced abundance and richness	Shaw and Richardson (2001)[d]
Invertebrates	550–700 kg/50 m stream reach	840	Reduced richness, increased drift	Matthaei *et al*. (2006)[c]
Invertebrates	0.6–1.8 kg/m^2	456	Reduced overall abundance and trait diversity	Larsen *et al*. (2011)[a]
Ephemeroptera, Plecoptera, Trichoptera	7.3–12.3%	–	Reduced relative abundance	Bryce *et al*. (2010)[b]
Stream invertebrates	130 mg/L	8760	40% reduction in species diversity	Nuttall and Bielby (1973)[a]
Benthic invertebrates	8 mg/L	2.5	Increased rate of drift	Rosenberg and Wiens (1978)[d]
Invertebrates	8–177 mg/L	1344	26% reduction in invertebrate drift	Quinn *et al*. (1992)[c]
Macroinvertebrates	133 mg/L	1.5	Drift increased (by 7-fold)	Doeg and Milledge (1991)[e]
Ephemeroptera	2680 mg/L	–	Increased drift	Ciborowski *et al*. (1977)[d]
Amphipoda Trichoptera	> 2000 mg/L	Varying exposure times	Drift and survival unaffected	Molinos and Donohue (2009)[f]
Mayfly (leptophlebiid)	1000 NTU	336	Mortality unaffected	Suren *et al*. (2005)[c]
Odonata	1000–1500 NTU	1	Reduced feeding efficiency	Kefford *et al*. (2010)[e]
	1000 NTU	1	Increased survival	Kefford *et al*. (2010)[e]

[a]Country of study: UK.
[b]Country of study: USA.
[c]Country of study: New Zealand.
[d]Country of study: Canada.
[e]Country of study: Australia.
[f]Country of study: Ireland.
[g]Country of study: Germany.
NTU, nephelometric turbidity units.

Table 4.4. The effects of varying the concentrations of and the duration of exposure to sediment on fish (from Bilotta and Brazier, 2008, Collins *et al*., 2011)

Organism	Sediment concentration (mg/L)	Exposure time (hours)	Effect on organism	Reference
Atlantic salmon	20	–	Increased foraging activity	Robertson *et al*., 2007[c]
Atlantic salmon	60–180	–	Avoidance behaviour/increased foraging activity	Robertson *et al*., 2007[c]
Chinook salmon	488	96	50% mortality of smolts	Stober *et al*., 1981[b]
Chinook salmon	207,000	1	100% mortality of juveniles	Newcomb and Flagg, 1983[b]
Sockeye and Coho salmon	800–47,000	–	80% reduction in successful egg fertilisation at sediment concentration of > 9000 mg/L	Galbraith *et al*., 2006b[c]
Coho salmon	2000–3000	192	Reduced feeding efficiency and immunity	Redding *et al*., 1987[b]
Coho salmon	400,000	96	Physical damage to gills; stress response	Lake and Hinch, 1999b[c]
Rainbow trout	47	1152	10% mortality of incubating eggs	Slaney *et al*., 1977[c]
Rainbow trout	Pulses	456	Reduced growth	Shaw and Richardson, 2001[c]
Brown trout	5838	8670	85% reduction in population size	Herbert and Merkins, 1961[a]
Arctic grayling	25	24	6% mortality of sac fry	Reynolds *et al*., 1989[c]
Arctic grayling	65	24	15% mortality of sac fry	Reynolds *et al*., 1989[c]
Arctic grayling	185	72	41% mortality of sac fry	Reynolds *et al*., 1989[c]

[a]Country of study: UK.
[b]Country of study: USA.
[c]Country of study: Canada.

Birkett *et al*., 2007), species diversity and organic content (Cline *et al*., 1982; Graham, 1990). In addition, an increase in flow and sediment levels can cause an increase in abrasion (Lewis, 1973; Francoeur and Biggs, 2006) and scouring (Steinman and McIntire, 1990); this reduces the attractiveness of periphyton to algal grazers (Graham, 1990).

4.2.2 Macrophytes

Macrophytes are easily abraded by suspended sediments, but their presence can actually enhance the deposition and accretion of fine sediments (Wood and Armitage, 1997), which may ultimately lead to their elimination if substantial sediment accumulation occurs (Nuttall and Bielby, 1973). Macrophyte beds can reduce flow rates, thereby leading to an increase in sediment accumulation and a decrease in turbidity; this, in turn, leads to an increase in light availability and enhanced macrophyte growth (Madsen *et al*., 2001). The seasonal growth of in-stream and marginal macrophytes also leads to increases in channel roughness and water depth (Hearne and Armitage, 1993), thereby creating habitat diversity (Armitage, 1995). However, any decrease in macrophyte abundance can destabilise sediments leading to sediment resuspension (Madsen *et al*., 2001).

4.3 Macroinvertebrates

Macroinvertebrates have the capacity to withstand occasional increases in deposited, saltating and suspended sediment levels (Ryan, 1991). However, increases in the levels of anthropogenically derived sediment reduce species abundance and diversity, and alter community structure. Moreover, as macroinvertebrates are a significant food source for fish, any changes in species abundance, diversity or quality will directly impact on fish populations. The various mechanisms of sediment impact on macroinvertebrates are illustrated in Figure 4.2.

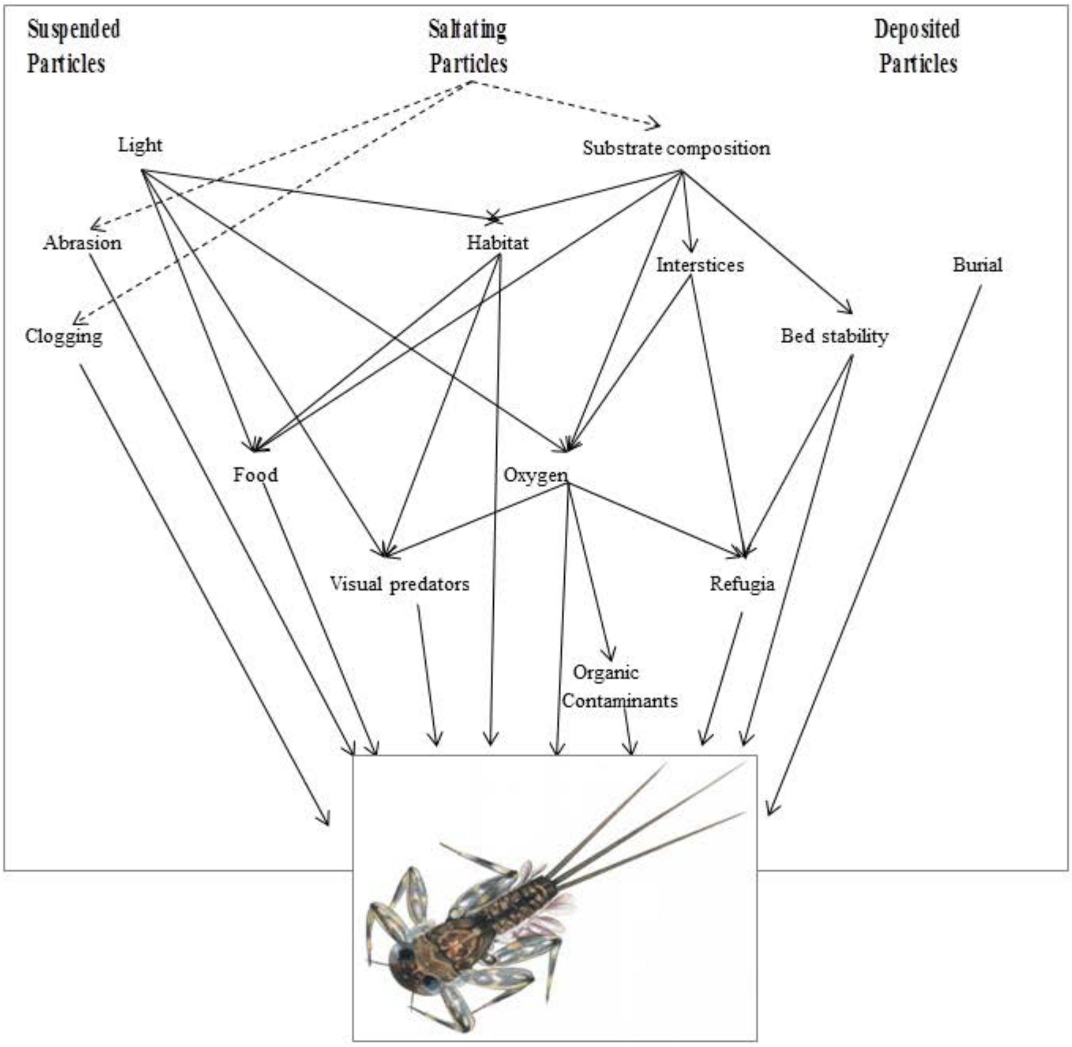


Figure 4.2. Schematic showing the mechanisms by which macroinvertebrates are affected (directly and indirectly) by suspended, deposited and saltating sediment particles (modified from Jones *et al*., 2011).

4.3.1 Mechanisms by which deposited sediment affect macroinvertebrates

Studies have shown that deposited sediment can affect invertebrates as a result of burial (Wood *et al*., 2001, 2005), altered benthic substrate composition, interstical infilling, bed instability (Kaufmann *et al*., 2009), reduced refugia availability (Lancaster and Hildrew, 1993) and oxygen depletion (Swan and Palmer, 2000; Stead *et al*., 2004), which restricts the depth to which some invertebrates can penetrate deposited sediment. Sedentary species, such as the freshwater pearl mussel *Margaritifera margaritifera* (L), are particularly susceptible to deposited sediment accumulation that limits interstitial sediment oxygen supply (Geist and Auerswald, 2007; Osterling *et al*., 2008). Kaller and Hartman (2004) suggested a threshold level for EPT diversity in excess of 0.8–0.9% fine sediment (< 0.25 mm) substrate accumulation.

4.3.2 Mechanisms by which suspended sediment affect macroinvertebrates

Suspended sediment affects macroinvertebrates by clogging (Alabaster and Lloyd, 1982) and abrasion of feeding mechanisms (Kurtak, 1978; Gaugler and Molloy, 1980; Hornig and Brusven, 1986; Aldridge *et al*., 1987); reduced feeding leads to reduced reproduction (Robertson, 1957), population size (Gray and Ward, 1982; Wagener and LaPerriere, 1985), and species abundance, richness (Shaw and Richardson, 2001) and diversity (Nuttall and Bielby, 1973). Suspended sediment reduces light penetration, thereby affecting invertebrate food sources, habitats and visual predators. Severe sediment loading may increase mortality rates (Hynes, 1970).

Macroinvertebrates can utilise behavioural responses to protect delicate structures (Kurtak, 1978); for example, they may retract filter combs or switch food source [e.g. *Brachycentrus* (Trichoptera) switches from trapping food particles with filtering limbs (Gallepp, 1974) to grazing under conditions of high SSCs (Voelz and Ward, 1992)]. However, these behavioural responses may have implications for feeding and growth rates.

4.3.3 Combined effects of suspended and deposited sediment on macroinvertebrates

Invertebrate drift is a natural process in fluvial systems, which expedites the movement of invertebrates among patches. Whilst moving particles may contribute to the dislodgement of invertebrates (Culp *et al.*, 1983), the behavioural responses exhibited by some species suggest that increased drift may be used by invertebrates to evade the negative impacts (e.g. burial and clogging) of suspended, saltating or deposited particles (Ciborowski *et al.*, 1977; Culp *et al.*, 1986; Gibbins *et al.*, 2007, Molinos and Donohue, 2009). An increase in invertebrate drift can reduce the benthic density and richness (Culp *et al.*, 1983; Doeg and Milledge, 1991; Suren and Jowett, 2001; Larsen and Ormerod, 2010) of some sediment-sensitive taxa, while the abundance of sediment-tolerant taxa may be unaffected or, indeed, may increase (Kreutzweiser *et al.*, 2005).

4.4 Fish

Extensive research has been undertaken into the impact of fine sediment on fish (Turnpenny and Williams, 1980; Berkman and Rabeni, 1987; Carling and McCahon, 1987; Erman and Ligon, 1988; Lisle, 1989; Reynolds *et al.*, 1989; Sear, 1993; Doeg and Koehn, 1994; O'Connor and Andrew, 1998; Shaw and Richardson, 2001; Soulsby *et al.*, 2001; Suttle *et al.*, 2004; Greig *et al.*, 2005; Julien and Bergeron, 2006), which reflects their economic importance (Ryan, 1991). Most fish can tolerate short-term naturally occurring increases in sediment levels; however, some species, such as salmonids, are highly sensitivity to sediment (Waters, 1995). The main mechanisms by which sediment affects fish are described by Kemp *et al.* (2011).

4.4.1 Effects of deposited sediment on salmonids

A decline in the quality of spawning habitats because of deposited fine sediment infiltration (Turnpenny and Williams, 1980; Carling and McCahon, 1987; Sear, 1993) has repercussions for egg survival and subsequent fish recruitment (Wood and Armitage, 1997). The deposition of fine sediment within redds, in the areas in which they infiltrate the gravel voids, reduces gravel permeability and porosity, and thereby adversely affects egg and alevin survival through diminished oxygen supply (Turnpenny and Williams, 1980; Erman and Ligon, 1988; Reynolds *et al.*, 1989; O'Connor and Andrew, 1998; Argent and Flebbe, 1999; Shaw and Richardson, 2001; Soulsby *et al.*, 2001; Armstrong *et al.*, 2003; Greig *et al.*, 2005; Julien and Bergeron, 2006; Heywood and Walling, 2007). The presence of sediment-associated oxygen-consuming material (Greig *et al.*, 2005; Dumas *et al.*, 2007) and/or the inadequate removal of metabolic waste from incubating eggs can also be an issue. Conversely, coarser particles, which form a seal in the upper redd layer, can reduce fry emergence rates (Crisp, 1993). Fry and older fish may be affected by habitat alterations, such as a reduction/loss of juvenile rearing habitats and reduced water depths within pool areas, which are essential for adult salmonids, as a result of infilling due to the deposition of fine sediment (Waters, 1995).

4.4.2 Effects of suspended sediment on salmonids

The effects of suspended sediment on salmonids can be categorised as behavioural, sub-lethal or lethal, depending on the SSC and the exposure time. Behavioural responses range from an alarm reaction, the abandonment of cover to the avoidance of sediment fluxes (Servizi and Martens, 1991). Sub-lethal effects are more wide ranging and include a reduction in the tolerance to toxicants (Lloyd *et al.*, 1987), a decrease in disease resistance (Redding *et al.*, 1987), damaged gills (Herbert and Merkins, 1961; Redding *et al.*, 1987), the interruption of gas exchange and osmoregulation (Bruton, 1985; Waters, 1995), the disruption of development (Suttle *et al.*,

2004) and a delay in growth (Shaw and Richardson, 2001; Sutherland and Meyer, 2007). Lethal effects due to long-term exposure to high SSCs can lead to a population-level response resulting in an increase in mortality rates (Stober *et al*., 1981) and the loss of species from affected reaches (Birtwell *et al*., 1984).

4.4.3 Sediment characteristics that influence ecological effects

Interactions between sediment sources and associated contaminants, the sediment concentration, the duration of exposure to sediment, and sediment particle size and chemical composition, along with the sensitivity of the particular taxa, certain life-history traits and species life-stage are all important factors that determine the ecological effects of sediment (Swietlik *et al*., 2003; Bilotta and Brazier, 2008; Collins *et al*., 2011).

4.4.4 Particle size and geochemical composition

Sediment geochemistry, which is influenced by soil type and catchment geology, and its physical characteristics, including the size, shape and angularity of particles, will also influence ecological impacts. For a given turbulence condition and sediment type with identical particle densities and shapes, smaller particles usually stay in suspension for longer than larger particles (Schindl *et al*., 2005). Under low-flow conditions, fine particles remain close to or at the water column surface (Schindl *et al*., 2005); this affects the respiratory and feeding mechanisms of invertebrates and fish. Coarse particles settle out of the water column (Schindl *et al*., 2005), which potentially affects salmonid redds and invertebrate habitats, as previously described (Greig *et al*., 2005).

4.4.5 Concentration and duration of exposure

Whilst extremely low concentrations of sediment can negatively affect aquatic species, the duration of exposure (Tables 4.2–4.4) is also a critical factor (Newcombe and McDonald, 1991; Newcombe and Jensen, 1996; Swietlik *et al*., 2003; Cocchiglia *et al*., 2012b). Short-term exposure may result in ephemeral effects, whereas long-term exposure will result in sustained impacts on biota (Collins *et al*., 2011). A meta-analysis study by Newcombe and McDonald (1991) found that the ranked response of aquatic biota was poorly correlated with SSC and more strongly correlated with sediment intensity (the product of sediment concentration and the duration of exposure). This suggests that both the concentration and the duration of exposure need to be assessed in order to predict the effects of sediment on aquatic biota. Studies on the effects of sediment on fish have shown increases in foraging activity and avoidance behaviour (Robertson *et al*., 2007), gill damage (Lake and Hinch, 1999), effects on the mortality of eggs, fry and smolts (Slaney *et al*., 1977; Stober *et al*., 1981; Reynolds *et al*., 1989), and reductions in feeding efficiency (Redding *et al*., 1987), egg fertilisation (Galbraith *et al*., 2006) and growth rates (Shaw and Richardson, 2001) depending on sediment concentration and the duration of exposure.

4.4.6 Sediment and associated contaminants

As previously stated, fine sediment is a natural and vital component of aquatic ecosystems (Wood and Armitage, 1997; Owens *et al*., 2005) that ensures habitat heterogeneity and ecosystem functioning (Yarnell *et al*., 2006). However, anthropogenically derived fine sediment fluxes can alter temporal and spatial fluvial sediment dynamics significantly, and increase the delivery and transport of sorbed contaminants, including nutrients, heavy metals, pesticides and pathogens, with the <63-μm sediment fraction to aquatic systems (Salomons and Forstner, 1984; Stone and Droppo, 1994; Owens *et al*., 2001, 2005).

The concomitant effects of the sedimentation and nutrient enrichment that are associated with managed and cultivated catchments may result in additive, antagonistic or synergistic biological responses as a result of complex interactions between these stressors (Lemly, 1982; Townsend *et al*., 2008; Matthaei *et al*., 2010). Some studies suggest that temporal patterns of sediment and nutrient disturbances drive ecosystem response to multiple stressors (Molinos and Donohue, 2010), whist others have shown that localised catchment modifications due to “poaching” and/or bank erosion may result in reach- and patch-scale ecological impacts (Larsen *et al*., 2009).

5 Measuring and Monitoring Suspended Sediment Concentrations and Loads

5.1 Introduction

It has long been known that it is difficult to estimate fluvial SSLs ("silt fluxes") accurately and precisely (see, for example, Walling and Webb, 1981a; Ferguson, 1987). At the simplest level, it requires the determination of river discharge (Q) and *spatially* and *temporally* representative SSCs. Thus, the instantaneous SSL in kg/s, is given by:

$$SSL = Q \times SSC \qquad \text{(Equation 5.1)}$$

In Equation 5.1, Q is the water discharge in m^3/s and SSC is the instantaneous suspended sediment concentration in g/L (i.e. kg/m^3).

The estimation of sediment fluxes over a given period requires the integration of this simple equation over time, and this is discussed in Section 5.9. River discharge measurement techniques are well reviewed elsewhere (see, for example, Hershey, 1999). However, the robust estimation of fluvial SSCs is far from simple. Therefore, this section focuses on the methodologies, techniques and instrumentation available for the estimation of SSC in rivers.

Ideally, the best way to obtain the most accurate estimates of SSC is to sample river waters by hand at very high resolution in (1) *time*, in order to capture variations related to hydrograph events (e.g. take samples at 15-min intervals before, during and after storms); and (2) *space*, that is, take samples from water surface to river bed, in order to embrace the entire water column, and across the full channel width to account for any cross-sectional variation in SSC, which is especially pronounced in large rivers with low turbulence and coarse suspended sediment particle size distribution. Repeated field sampling missions should then be followed by the laborious processing of thousands of water samples in a laboratory to determine SSCs. However, although this might be possible for short-term studies, for long-term investigations of fluxes, the resources required for this highly labour-intensive approach would be impracticable. Thus, various compromises have to be made; these problems, solutions and alternative approaches and methodologies are outlined in this section, although all approaches benefit from an initial reconnaissance study, as summarised below.

5.1.1 *The value of reconnaissance suspended sediment data*

Manually collected, reconnaissance SSC or turbidity data – especially from repeat "storm-chasing" runs during elevated SSCs – can give vital information, at an early stage, on key indicators of the sediment transport system, and can help with planning a more spatially and temporally intensive sampling and monitoring campaign; this is vital for the accurate and precise estimation of suspended sediment fluxes. Key indicators include:

1. approximate fluvial SSCs;
2. suspended sediment particle size distribution (this can give an indication of the sediment sources, and help to predict likely cross-sectional variations in SSC); along with information on fluvial SSCs, this information can help to guide the selection of the most appropriate turbidity meters, which can be geared towards specific ranges of SSCs and particle size distributions (e.g. via their beam intensity or the wavelength of the instrument);
3. suspended sediment *composition*, which can indicate likely sediment sources;
4. the extent of any width or depth variation in SSC across the gauging cross-section;
5. catchment-scale and downstream changes in SSC, which can provide clues on the locations of sediment sources, storage and conveyance losses, and can inform decision-making with regard to the choice of sites for monitoring stations;
6. Q data; if such data are available or estimable, preliminary instantaneous SSL data can be obtained using Equation 5.1 and used as guidance for later turbidity meter and automatic water sampler installation, siting and intake, and instrument head locations (e.g. for channel-edge sampling).

Reconnaissance runs throughout the catchment more widely (not just of the stream network itself) during rainstorms can also help to identify key sediment sources and delivery processes.

5.2 Manual Sampling

5.2.1 Cross-sectional variations in SSC

It has long been known that significant cross-sectional variations in SSCs exist (e.g. Guy and Norman, 1970; Horowitz *et al*., 1989): SSCs generally increase towards the bed, especially for coarser particles, and towards the channel centre (Figure 5.1 and Figure 5.2). Figure 5.3 shows data supporting this, and although datasets are rare, channel centre SSCs can be twice that of those at the channel edge in some large rivers (Nordin and Richardson, 1971). Acoustic Doppler current profiler (ADCP) data have recently helped to define spatial patterns in cross-sections and, for flux investigations, SSC could ideally be representative of the whole cross-section, but this is rarely possible to achieve with high temporal resolution monitoring. Normally, just a single, fixed or floating, intake or monitoring point is placed at the channel edge. Therefore, especially in large rivers of low turbulence and/or with coarse suspended sediment in which strong lateral and vertical SSC gradients develop, the following three additional procedures can be introduced to help attain the most accurate and representative values of suspended sediment flux:

1. Attempts should be made to achieve a spatially representative position for the intake or monitoring point (e.g. not in a turbid "hot-spot").

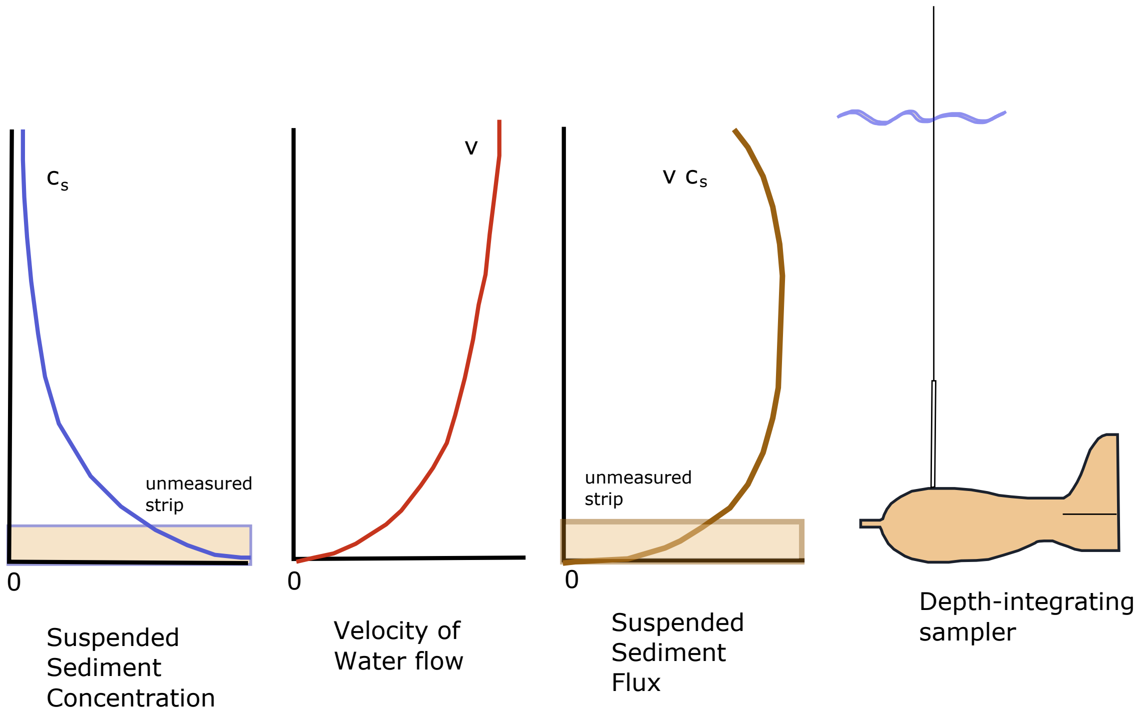


Figure 5.1. Schematic of SSC cross-sectional variations.

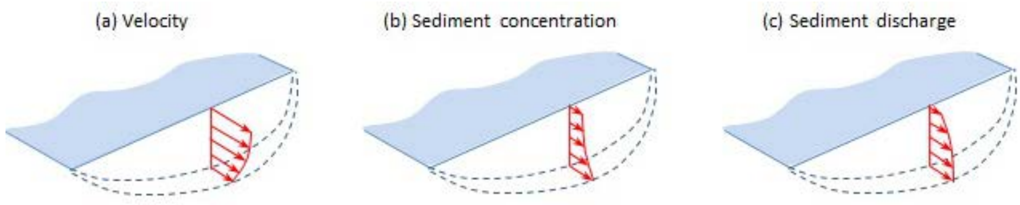


Figure 5.2. Schematic cross-sectional variation in flow velocity, SSC and sediment flux.

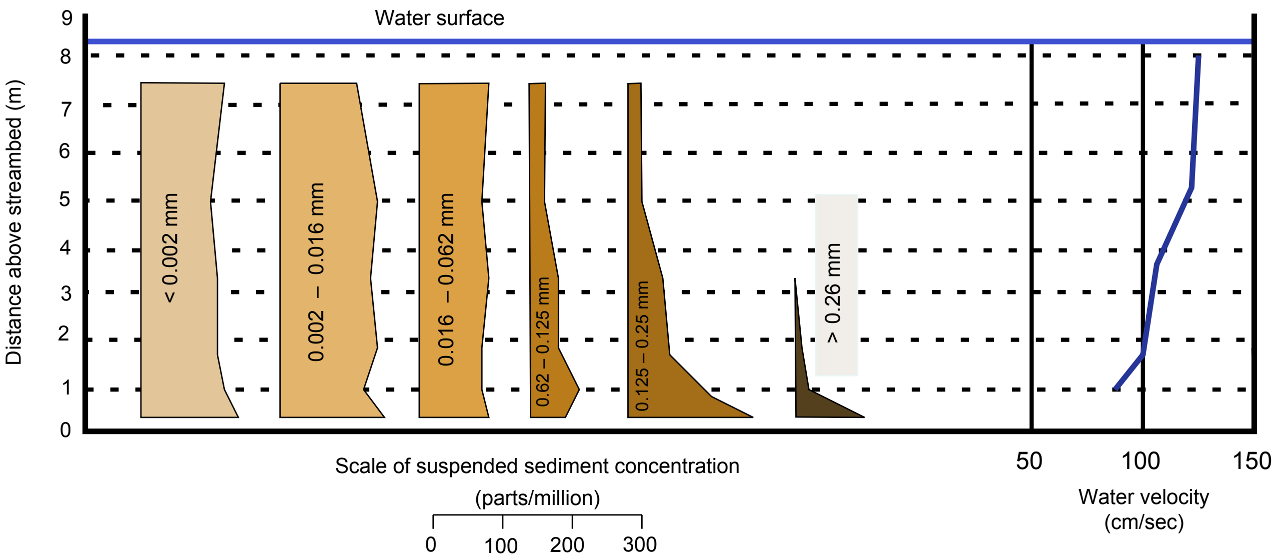


Figure 5.3. Vertical distribution of concentration of various particle sizes in a stream section (after Nordin and Richardson, 1971).

2. SSCs at this single point should be compared with those from occasional depth-integrated manual sampling points across the channel width.
3. If the single-point concentration value tends to significantly over- or underestimate cross-sectional average concentrations, a suitable correction/calibration factor should be determined and applied.

5.2.2. *Sampling to detect cross-sectional variation in suspended sediment*

For very turbulent flows, or for fine suspended sediment size distributions, lateral and vertical variability in SSC will normally be less notable than cross-sectional variation. Sediment mixing is also likely to vary with stage for a given cross-section, although few datasets demonstrate this. To check for sediment mixing, spatial sampling should be done with specially designed point- or depth-integrated isokinetic samplers (Guy and Norman, 1970; Davis, 2005 in Sabol *et al.*, 2010), as pioneered by the United States Geological Survey (USGS).

Isokinetic sampling

The key feature of a USGS isokinetic sampler is that the bottle is allowed to fill through only a carefully machined and chamfered 6-mm special nozzle that, if pointed upstream, allows water to flow into the sampler at the same velocity (±10%) as water passing around the sampler. This is called "isokinetic sampling" and is very important for representative suspended particulate sampling.

Isokinetic sampling is described in Glickman (2000) as "any technique for collecting airborne (waterborne) particulate matter in which the collector is so designed that the air stream (water stream) entering it has a velocity equal to that of the air (water) passing around and outside the collector. The advantage of isokinetic sampling consists in its freedom from the uncertainties due to selective collection of the larger, less easily deflected particulates. In principle, an isokinetic sampling device has a collection efficiency of unity for all sizes of particulates in the sampled air or water".

At each vertical, such samplers need to be lowered at an even rate to the bed and raised again to the surface. This allows sampling of most of the water column, except the lowest few centimetres (e.g. the lowest 9 cm for the US DH-48 sampler). Transits through the water column are repeated, if necessary, at the same vertical until the bottle is approximately 70% full, thus avoiding overfilling, which can bias the sample. This is called the "equal transit rate" method (Guy and Norman, 1970; Edwards and Glysson, 1999; Gray and Gartner, 2009).

This vertical sampling procedure is then repeated at "no less than 3 and usually 5 to 10 verticals in a given cross section" (Dunne and Leopold, 1978) to capture the lateral variations in SSC (in a similar way as is done for stream gauging using impeller or electromagnetic current meters). Point samplers can be used to sample at only given depths for each vertical.

In shallow rivers, depth-integrating samplers can be fitted to a wading rod (e.g. US DH-48 sampler developed by the USGS) and hand-lowered to the bed by a fieldworker in waders or a drysuit. In deep rivers, depth-integrated sampling will need to be carried out from boats or at bridges by deploying the sampler on a long rod or cable.

Samples are transferred into plastic sample bottles, sealed and labelled, and transported back to the laboratory for analysis of SSC. To do this, samples of known volume are simply filtered normally through 0.45-μm filter membranes. Vacuum filtration can be used if clogging of the filter paper/membrane occurs. SSC can be calculated as follows:

$$SSC = wt/vol \qquad \text{(Equation 5.2)}$$

In Equation 5.2, “wt” is the dry weight of sediment (in g) and “vol” is the water sample volume (in L). (Note that organic material may need to be removed to determine the minerogenic component of the suspended material.)

5.2.3 Time-integrating samplers

To collect larger samples of suspended sediment for chemical analysis (e.g. for source tracing), time-integrating samplers are available. These are tubes with intake and outflow nozzles at the upstream and downstream ends, respectively. These samplers can be left in the flow for some time to collect sediment, which settles in the sampler (Figure 5.4). A useful video showing the use of these samplers is available on YouTube (http://www.youtube.com/watch?v=PnSm4hNAJ4Q).

Infiltration baskets can also be inserted into river beds to measure the accumulation of fine sediment. The mass and composition of the fine sediment can be quantified upon removal of the basket. An example from Heywood and Walling (2007) is shown in Figure 5.5.

5.3 Acoustic Doppler Current Profiler Method

The principles of the ADCP (essentially acoustic backscatter) method, which is used to estimate SSCs (as a spin-off from the normal usage of ADCP for velocity profiling), are outlined by Gray and Gartner (2009). Wall *et al.* (2006) describe the typical procedures used to estimate fluvial and tidal SSCs from ADCP measurements. They stress that, although ADCPs were not designed for SSC measurements, they can be useful, but only after considerable investment in time and appropriate software to obtain a suitable transfer function between the acoustic signal and SSC (see also Filizola and Guyot, 2004). A particular strength of the technique is its ability to produce a “snapshot” of cross-sectional variation in SSC, especially for larger rivers.

5.4 Automatic Sampling for Suspended Sediment Concentration Time Series

Many authors have argued that suspended sediment data were traditionally collected using a range of manual sampling devices and that “the associated logistical problems and financial constraints mean that most manual sampling strategies fail to coincide with the main periods of sediment transport, i.e., flood events” (Collins and Walling, 2004). Such sampling programmes, if used alone, significantly underestimate *actual* suspended sediment fluxes and yields. Thus, although manual sampling can provide very useful data on spatial variations in SSC across, and between, single cross-sections and catchments, it is rarely sufficient to provide the necessary high-resolution time series of SSC, which are *absolutely crucial* for accurate and precise estimates of sediment fluxes and SSC.

The deployment of automatic water samplers (AWSs) is one way to address this need to sample at high temporal resolution. AWSs can be placed next to rivers in waterproof and vandal-proof housings, and set up to automatically extract water samples at, for example, 30-minute, 60-minute or 3-hour intervals (depending on the aims and resources of the study, and the flashiness of the river flow regime) and store the water samples in a crate on site. Crates are replaced at intervals, and the samples are retrieved and transported to the laboratory for processing and filtration. AWSs can be seen in action on YouTube (at: http://www.youtube.com/watch?v=kQl1qk7z8do).

However, the deployment of “passive” AWSs is likely to generate numerous samples at low flow and low SSC,

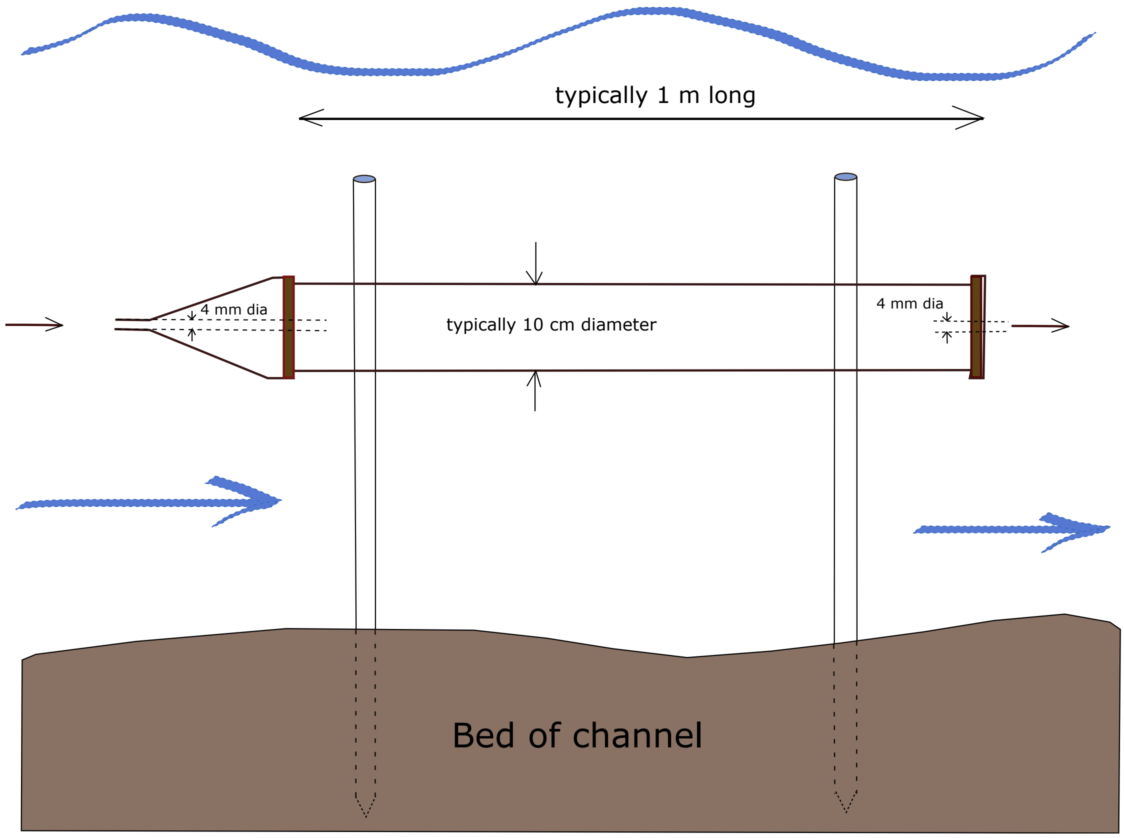


Figure 5.4. Time-integrating suspended sediment sampler for collecting large amounts of suspended sediment for composition analysis (after Collins *et al*., 2010).

Figure 5.5. Infiltration basket for capturing fine sediment in gravel river beds (photo: Liz Conroy).

and very few samples at high flow when suspended sediment fluxes peak. In many catchments, 95% of the sediment flux occurs only 10% of the time, normally during high-flow events. Therefore, it is important to use active, *programmable* AWSs (such as the ISCO, American Sigma or EPIC models), which can, if desired, lie dormant during periods of low flow/turbidity when sediment transport is minimal, and activate only if linked stage and/or turbidity triggers are exceeded. Such event-actuated sampling avoids the unnecessary use of valuable crated bottles, and saves them for the vital samples taken during key sediment transport events, which are fundamental to accurate sediment flux estimation. Of course, for completeness, some low-turbidity samples are desirable.

The intakes for AWSs are normally fixed at a single point in the cross-section near the bank (below the water level at all times, but above the level at which fine sediment clogging or damage by bedload transport may occur); however, floating intakes can also be used for sampling at a consistent proportional depth.

Although AWSs can provide strong datasets on SSCs and therefore SSLs, they have three disadvantages: (1) a relatively low frequency of sampling, typically limited by crates of no more than 24 bottles; (2) the lack of available capacity for sampling *sequences* of floods, when the bottles have already been filled by sampling from the initial event; and (3) the generatation of large volumes of samples that have to be processed and filtered in the laboratory before estimates of SSC can be obtained. Turbidity metres can ease these problems considerably, and are discussed below.

5.5 Turbidimetric Instrumentation

5.5.1 Introduction

Turbidity "is an expression of the optical property of a medium which causes light to be scattered and absorbed rather than transmitted in straight lines through the sample" (American Public Health Association *et al.*, 1995). Turbidimetry is a surrogate method for estimating SSC. However, the field is hampered by a lack of consistency with regard to units, sensors and calibration techniques, and there are many papers on these issues (e.g. Lawler, 2005a; Lawler, 2015). Turbidity used to be monitored manually by lowering a Secchi disk into the water, and simply noting the depth at which the disk markings became invisible to an observer on the surface (usually in a boat); although this is a subjective method, it provided the crucial long-running dataset (Figure 5.6) that was used to address the well-publicised problems of increasing turbidity over the last few decades in Lake Tahoe on the California/Nevada border (Jassby *et al.*, 1999), which led to legal disputes with loggers in an attempt to prevent the loss of amenity resources in a key tourist area.

However, as reviewed by Gray and Gartner (2009) and Lewis (1996), many relevant and more modern techniques for measuring turbidity are now available.

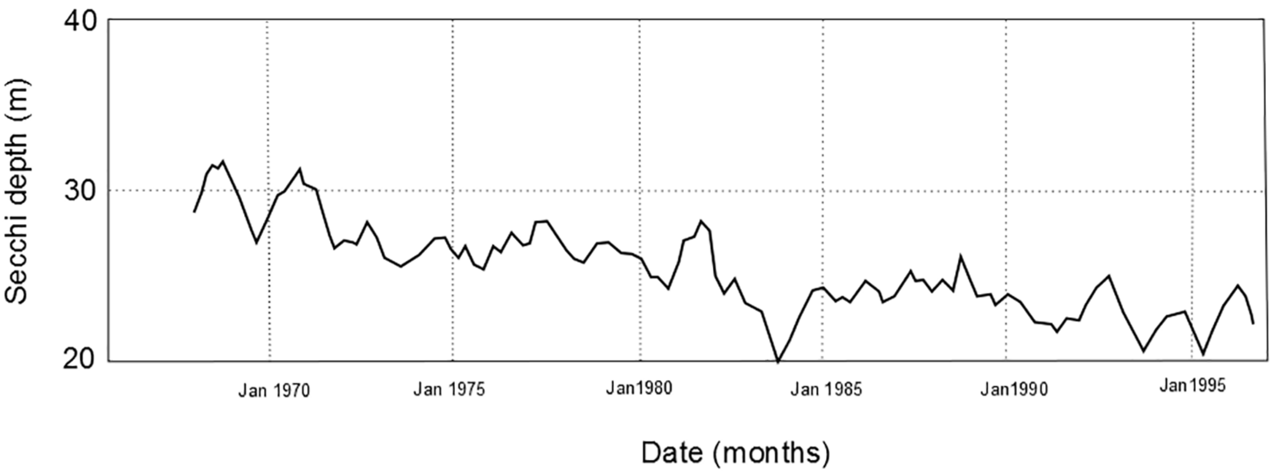


Figure 5.6. Declining water clarity in Lake Tahoe, measured using the Secchi disk. The clarity decrease (turbidity increase) is thought to have been driven by fine sediment discharge from the surrounding lake catchment as a result of, for example, logging activities (after Jassby *et al.*, 1999).

The units of measurement are typically FTU (formazin turbidity units) or NTU (nephelometric turbidity units). One of the longest running investigations of turbidity was carried out in the USA by Jones and Schilling (2011); these authors demonstrated that sediment load decreased during the 20th century and recommended focusing management on reducing mobilisation.

5.5.2 The turbidimetry principle

The principle of turbidimetry is essentially simple: a beam, normally in the infra-red (IR) spectrum, of constant and known initial intensity, is emitted from a very high specification light-emitting diode (the kind used on aircraft control panels), and transmitted to an appropriate IR sensor across a known gap through which river water is allowed to pass. Particles within the gap, such as fine sediment, scatter or absorb some of the photons in the beam. This attenuates the beam intensity, as recorded by the sensor in the instrument. This reduction in the IR light recorded by the sensor is directly related to the concentration of suspended matter in the light path.

It is well known, however, that transmittance and scatter are functions not just of SSC, but also the "number, size, colour, index of refraction, and shape of suspended particles" (Gray and Gartner, 2009). Particle size distribution is a key issue: a given mass (or concentration) of *fine* sediment will cause greater light scattering (and therefore beam attenuation) than the same mass of *coarser sediment*. Fine sediment has a greater specific surface area than the same mass of coarse sediment, and the particle specific surface area and roughness largely control the light scattering ability of sediment (Ward and Chikwanha 1980; Lawler, 2005a). Indeed, data from Ward and Chikwanha (1980) show that, for the *same* SSC (1000 mg/L), particles in the 12- to 18-μm diameter range absorb an *order of magnitude* more light than particles in the 30- to 50-μm diameter range. Thus, if the particle size distribution of the SSL varies in space and time, this will affect the applicability of the calibration relationship.

5.5.3 Turbidity meters

Turbidity meters can be used as hand-held portable devices for "snapshot" measurements and vital, early field reconnaissance. However, their real value lies in the *automated* monitoring of SSC. Because of typically rapidly changing SSCs – capturing which is beyond the capability of hand-sampling or even AWSs – much contemporary research on suspended sediment dynamics and fluxes is based on the automatic and high-resolution monitoring of turbidity as a surrogate variable for SSC (see, for example, Lawler, 2005a; Walling, 1974).

The great advantage of measuring water turbidity is that it can now be monitored automatically using electronic turbidity meters, such as the Partech IR40 and IR15 instruments, which are connected to data loggers. Data loggers (e.g. those manufactured by Campbell Scientific Ltd) can be programmed to simply scan and store turbidity data at high resolution (e.g. 15-minute frequency), or to store, if desired, the *average* of several scans if logger memory capacity is limited. Hence, it can much more usefully capture *changing* suspended sediment transport, including peaks, troughs and flat-lining in the SSC signal. These attributes are essential for the robust quantification of sediment concentrations and hence fluxes; in addition, the time-dependent sediment behaviour with respect to other events can aid understanding of sediment dynamics and sources.

Turbidity meters can be mounted directly in the river (normally at the channel edge), although they can be vulnerable to damage from large floating objects, clogging by organic and other debris, or bio-fouling of the optics (e.g. with algae). Therefore, such instruments require periodic (e.g. weekly) cleaning in order to preserve the integrity of measurements. Some turbidity meter models are self-cleaning, with a motorised optics wiper system powered by an on-site battery or solar panel. To obviate some of the difficulties of in-stream deployment, turbidity instrumentation can instead be installed in bankside housings away from potential sources of mechanical damage and vegetation fouling. In such cases, river water can be automatically withdrawn by pump or vacuum and directed to the instrument in this "safe house" (e.g. England and Wales Environment Agency automatic water quality monitoring stations; Lawler *et al*., 2006).

Turbidity meters work especially well at sites in which little seasonal or storm-event change in suspended sediment composition takes place; this should be assessed in any detailed investigation. However, any uncertainties regarding the role of such confounding variables locked into the turbidimetry principle are usually outweighed by the huge advantages of the continuous high-resolution datasets obtained.

Turbidity meters can also be deployed in the tidal zone (often a zone of repeated sediment deposition and re-erosion) to monitor the magnitude and migration of the estuarine turbidity maximum which, in turn, can partly be a function of fluvial sediment flux (e.g. Mitchell *et al*., 2003).

5.5.4 Laboratory and field calibration of turbidimetric instrumentation

Data are often reported directly in FTU or NTU. However, for many geomorphological or sediment transport investigations, there is a need to report results in terms of SSLs or fluxes. This entails calibration, which can be difficult. Generic calibration factors between turbidity and SSC do exist, and can be provided by instrument manufacturers. However, it is preferable to derive a *site-specific* calibration factor for turbidity based on *known* SSCs, because turbidity–SSC relationships typically vary widely from river to river, station to station and over time, largely in response to changes in the *composition* of the SSL (see, for example, the field calibrations illustrated in Figure 5.7 and Figure 5.8).

There are a number of possible approaches to calibration. Laboratory procedures to derive calibrations are given in Lawler and Brown (1992) and Old *et al*. (2003).

Conventional calibrations used to be made with formazin, which is a polymer that was developed in 1926. Although formazin is considered a carcinogen, it is still used as a primary standard today (Sadar, 1998), and can be used for calibration as an alternative to sediment from the river or catchment being monitored. An effective method is simply to match up SSCs from samples withdrawn by on-site AWSs with synchronous, usually automated, turbidity meter measurements made as close as possible to the sampler intake point. If this is not possible, then large quantities of turbid water can be taken from the river for laboratory calibration purposes. Another option is to mix up a stock solution of turbid water using fine sediment (e.g. in the 1- to 250-μm diameter range) derived from likely, *suspendable* sediment sources upstream of the intended monitoring site (e.g. from river banks, exposed bars, fine bed sediment in pools, field poaching sites, farm tracks or hillslope gullies). The first step of calibration involves measuring the turbidity of a stock solution with the maximum expected SSC (e.g. 1000 mg/L). This stock is then sequentially diluted by adding distilled water in known volumes to produce a range of turbidity values (e.g. 20 values) until the lowest turbidity value is reached. Associated turbidity meter readings are taken for each known SSC. These high-to-low turbidity measurements minimise noise in the calibration by avoiding the need to add fresh sediment – possibly of a different particle size distribution – to the solution.

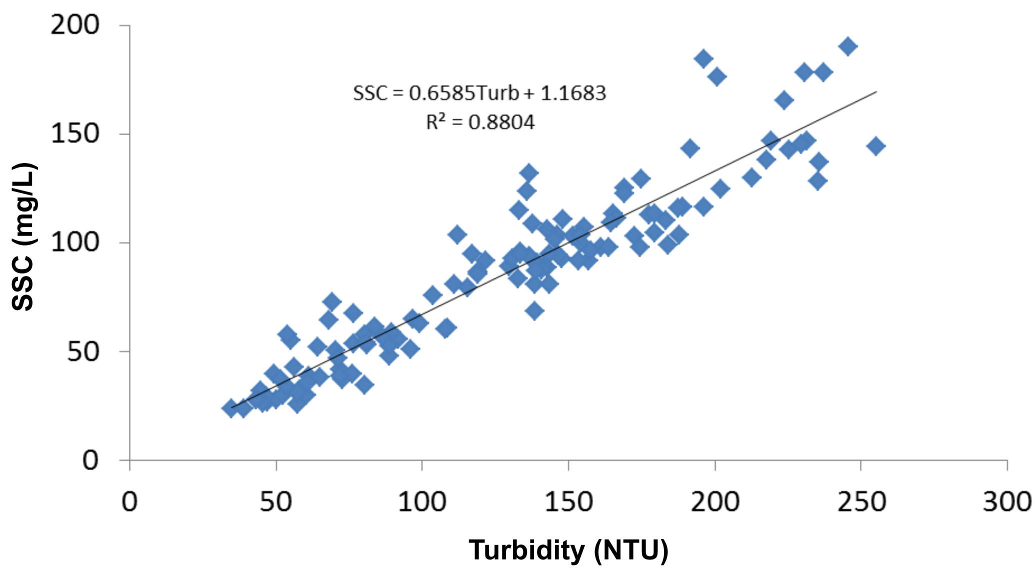


Figure 5.7. Turbidity versus SSC: calibration for the urbanised area at James Bridge, River Tame, Birmingham, UK (Lawler *et al*., 2006).

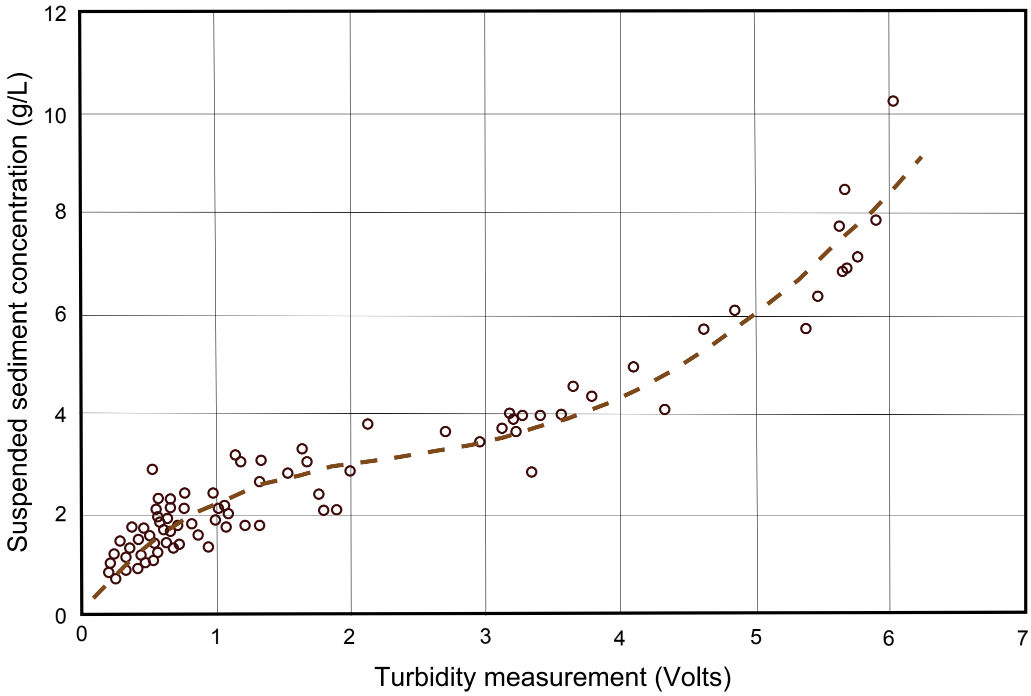


Figure 5.8. Turbidity versus SSC: calibration for the large Skaftá river, south Iceland (Old *et al.*, 2005a).

Manufacturers of turbidity instrumentation (e.g. Hach or Partech) can supply generic calibration factors, stock calibration solutions and gel slides which have a fixed assemblage of particles embedded within them, for a range of sediment concentrations. Such gel slides can be placed between the turbidity meter beam source and the sensor.

5.6 Optical Backscatter Sensor Instrumentation

The optical backscatter sensor (OBS) and nephelometer are similar to turbidity meters except that these instruments directly measure the amount of light or IR *scattered* by particles in the water column, rather than beam *attenuation* itself. For OBS instruments, backscatter is measured at 180° to the incident beam, that is, OBS instruments measure the IR radiation that is reflected back to the emitting sensor from particles approximately 0.5–30 cm from the front of the device, and outputs can be monitored automatically to create a useful time series of SSCs (e.g. Ruhl *et al.*, 2001).

As for turbidity meters, the particle size distribution of suspended sediment influences OBS instrument turbidity measurements. For example, "one gram of silt, with a grain size of 10 microns, suspended in a litre of water (SSC = 1000 mg l^{-1}) might produce an OBS signal of 1 Volt; whereas a gram of sand with a grain size of 100 microns would produce only a 0.25-Volt signal, with other factors such as shape and mineral composition being the same" (Downing, 2008b). This is illustrated for a range of particle size distributions in Figure 5.9 and Figure 5.10.

A typical OBS field monitoring procedure is helpfully described by Chappell *et al.* (2011, p. 94) as follows:

> At this station an OBS-3+SB-2.5-T4 turbidity probe (D & Instruments / Campbell Scientific Inc., Logan, USA) mounted on a self-cleaning Hydro-Wiper device (Zebra-Tech Ltd, Nelson, New Zealand) was attached to a protective steel manifold. A CR1000 data-logger (Campbell Scientific Inc., Logan, USA) was used to monitor turbidity every 30 seconds (on both 0–2000 and 0–4000 Nephelometric Turbidity Units or NTU ranges), with the average stored every 1 minute. Averaging and storage was changed to every 10 minutes after the first month of monitoring. A relationship between turbidity (NTU) and suspended sediment concentration (mg l^{-1}) was established using water samples collected from the Ramu River over the ranges of 37 to 810 NTU and 74 to 2676 mg l^{-1}.

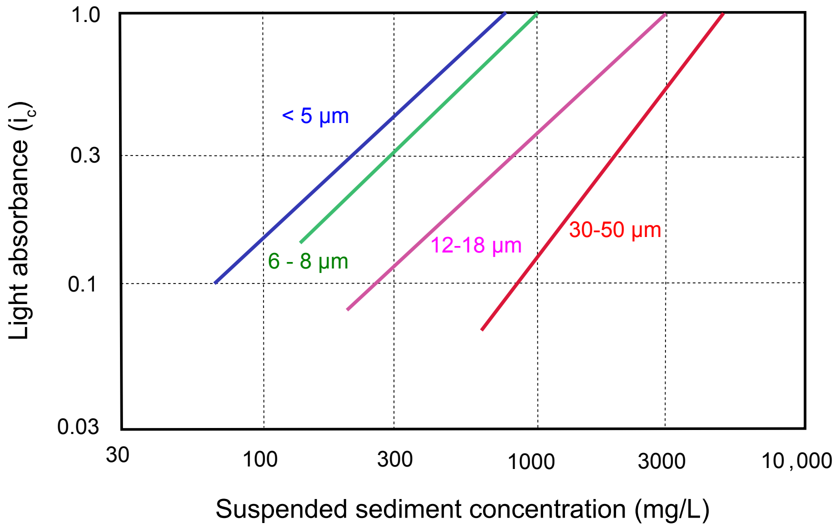


Figure 5.9. Dependence of light absorbance on sediment particle size (after Lawler, 2005a).

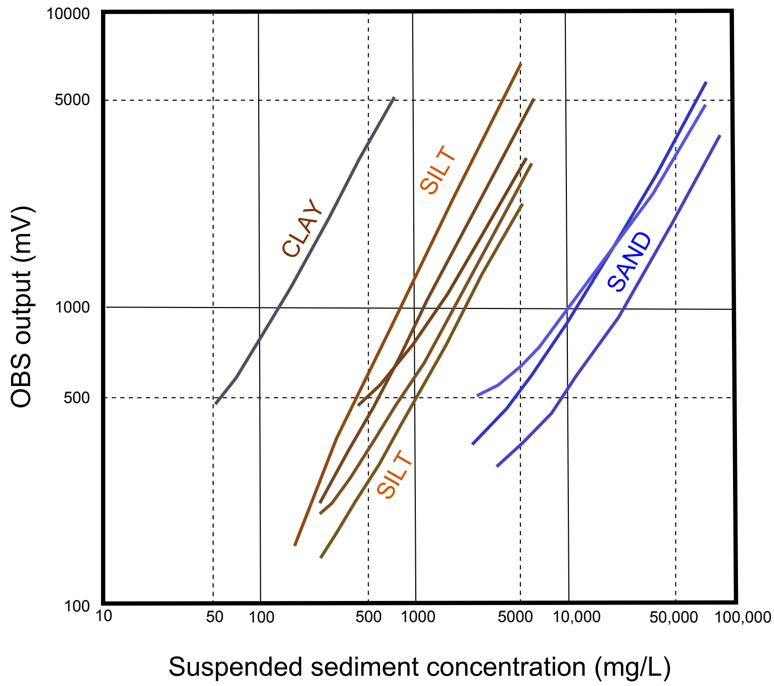


Figure 5.10. Effect of particle size on OBS response (Downing, 2008b).

5.7 Laser *In Situ* Scattering and Transmissometry

Laser *in situ* scattering and transmissometry (LISST) instrumentation also provides key information on sediment transport (e.g. Melis *et al*., 2003; Old *et al*., 2006; Gray and Gartner, 2009). LISST instruments are widely used by the USGS (e.g. Gray and Gartner, 2009) and are increasingly being used in Europe (e.g. Thonon and Van der Perk, 2003). In particular, LISST sensors usefully provide particle size distribution information for the suspended sediment in 32 logarithmically spaced size classes (Thonon and Van der Perk, 2003), within a total range of 1.5–250 µm (which is a typical fluvial suspended sediment range). Particle size distribution information may be used to pinpoint sediment source types or locations and pathways, and hence identify the key contributors to and explanations for any SSL problems in upstream catchments.

Different LISST instrument models are available for different contexts (e.g. oceanic or fluvial) and various size classes. For example, the Sequoia model (see http://www.sequoiasci.com/) is a "streamlined isokinetic version of the LISST-100X developed for river sediment monitoring. The instrument senses the river current velocity using a pitot tube and adjusts (sampler) intake velocity to match. The river water is thus pumped through the instrument at the same speed as the water flowing around the instrument, making the sampling isokinetic"; this helps to ensure representative sampling. Data are transmitted from the submersible sensor head to a data logger on the bank.

As with turbidity meters, LISST optics should be cleaned at approximately weekly intervals to remove biological fouling, algal growth etc., and more frequently under some conditions (e.g. tidal rivers).

5.8 Remote Sensing of Suspended Sediment Concentration

It is also possible to obtain useful data on the spatial variation of SSCs of surface and near-surface waters using remote sensing imagery from appropriate satellite, helicopter or fixed-wing airborne platforms (see, for example, Ruhl *et al*., 2001). Advanced very high resolution radiometers (AVHRR) from satellites have proved very useful in this regard (e.g. San Francisco Bay; see Ruhl *et al*., 2001). Such imagery is especially helpful for large-scale water bodies, rivers and estuaries because it can show, for example, turbidity hot spots and high-turbidity plume dynamics, especially in a relative sense. The imagery and AVHRR reflectance has to be calibrated against actual SSC values obtained using other methods in order to deliver usable data on SSCs (see, for example, Ruhl *et al*., 2001). This is done by synchronous sampling or monitoring at the same place and time of the image capture.

5.9 Estimation of Suspended Sediment Loads

5.9.1 Introduction

As shown in Equation 5.1, SSLs are derived from the product of SSC and Q. Although apparently simple, the calculation of fluvial sediment fluxes is fraught with difficulty, and the process is often referred to as an "estimation". Suspended sediment flux estimates need to maximise accuracy (i.e. proximity to the *actual* value) and precision (i.e. high repeatability). Problems include the following:

1. The many non-linearities in the sediment transport system, especially with regard to the relationship between Q and SSC (Figure 5.11), which are often power functions (e.g. Figure 5.12), are problematic.
2. There are problems related to the temporal variability of river flows and the variable SSC for a given flow. Sediment rating curves show that SSC can vary by 2 to 3 orders of magnitude for a given flow [as in Figure 5.11 from Walling and Webb (1981a)]. This is quite different from a stage-discharge rating equation in which, in the absence of geometrical changes of the channel at the gauging station, a given stage is always associated with the same discharge (notwithstanding rising and falling limb water surface slope differences as the flood wave passes).
3. Most suspended sediment transport occurs over a very limited time. For example, Walling *et al*. (1992) found, for the River Exe at Thorverton in south-west England for the 1978–1980 period, that 50% of the SSL was present just 1% of the time, and 90% of the total SSL was present only 5% of the time. It is crucial, therefore, to sample or monitor during such key periods. Typically, this leads to huge scattering with regard to Q–SSC relationships (sediment rating curves).
4. There are characteristically log–linear relationships between Q and SSC.
5. Hysteresis effects drive different SSCs for a given flow (during storm events and different seasons).
6. Cross-sectional SSC variations may complicate single-point (e.g. channel-edge) sampling or monitoring, etc.
7. Changing sediment particle size distribution effects, during and between storm events and seasons, can degrade simple relationships between turbidity and SSC.

Ensuring that sampling/monitoring is quasi-continuous is key to estimating SSLs. Walling and Webb's (1981a) pioneering study demonstrated the importance of this by using a re-sampling approach on data from the River Creedy in Devon, England. They first computed "true" SSLs from a master high-resolution dataset, obtained from samples collected at 60-minute intervals. They then compared these values with those

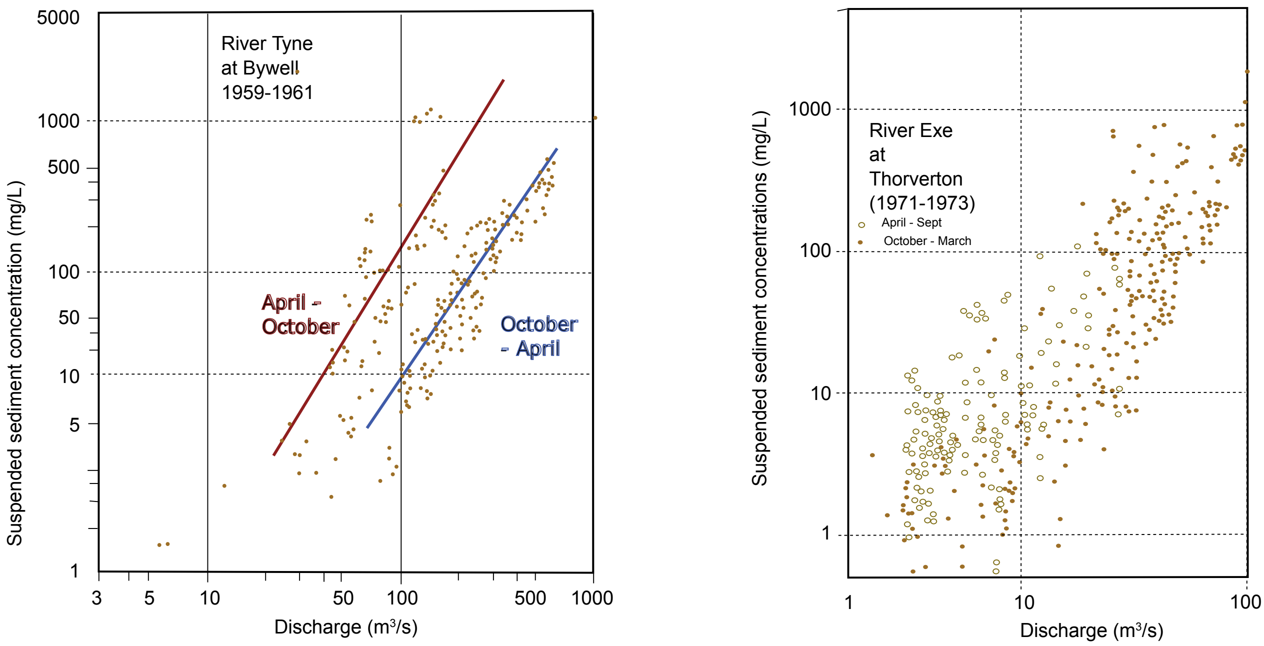


Figure 5.11. Examples of SSC–Q relationships for two British rivers (after Walling and Webb, 1981b).

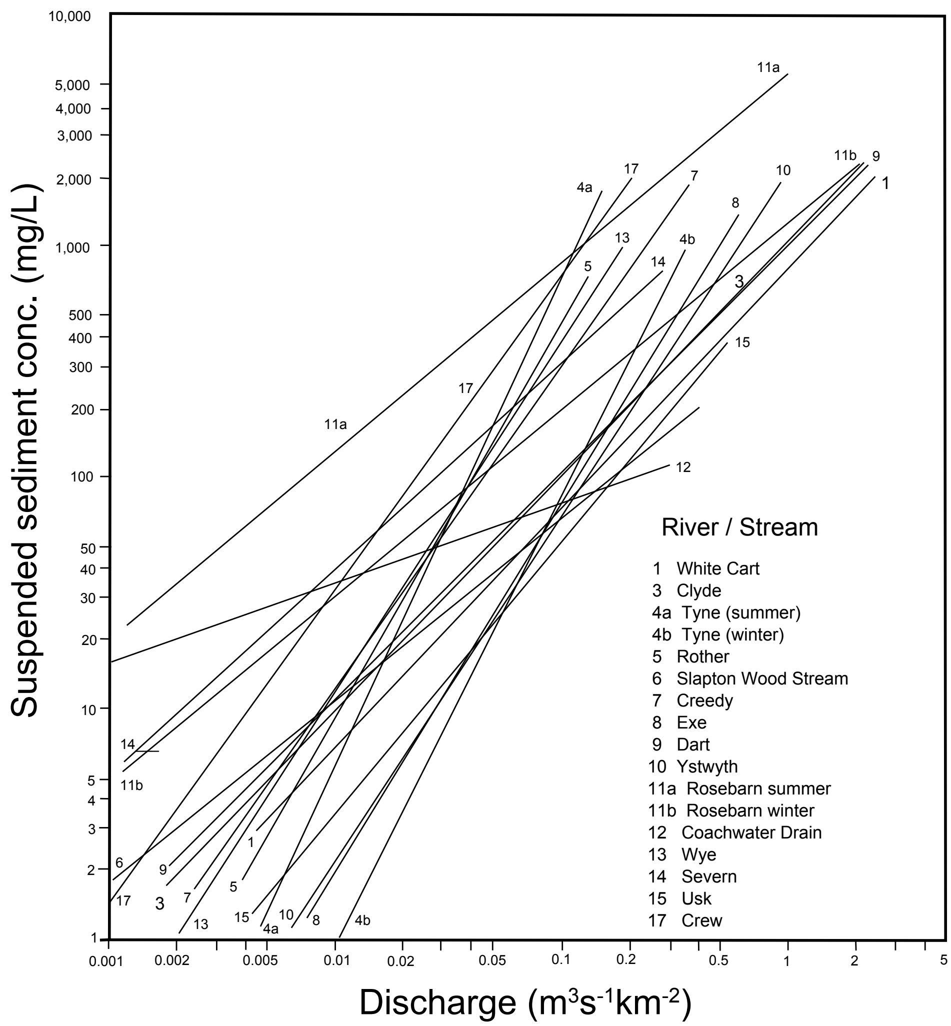


Figure 5.12. Relationships between suspended sediment and area-weighted Q for several named British rivers (after Walling and Webb, 1981b).

obtained from sampling at successively longer time intervals (e.g. daily, weekly, 2-weekly intervals) to mimic a low-resolution field investigation. Such studies demonstrate that very large load errors can arise with low temporal resolution sampling (see, for example, Walling and Webb, 1981a, 1985).

Several methods have been evaluated rigorously by Walling and Webb (1981a, 1985), and further examinations of the statistical methodologies involved have been published by Ferguson (1987) and Clarke (1990a, b). Phillips *et al.* (1999) evaluated 22 SSL estimation equations for catchments in northern England.

However, the problems associated with SSL estimation persist and more recent examinations include those by Webb *et al*. (1997), who investigated British rivers; Phillips *et al*. (1999), who examined rivers in the UK and showed that errors are magnified in larger catchments; Horowitz (2003), who, on the basis of sediment rating curves for US and European rivers, recommended that sampling is done on a hydrological, rather than a calendar, basis to minimise errors; Crowder *et al*. (2007), who studied SSL–Q relationships for US Midwest basins; and Brown *et al*. (2010), who used multiple load estimation approaches for basins in the UK, Germany and the Netherlands. As Moatar and Meybeck (2005) argued, some of these issues also apply to *nutrient* load estimation: "concentration data are still commonly the limiting factor on the quality of river flux estimates".

5.9.2 High-resolution monitoring method: interpolation approaches

The most effective and accurate method of deriving total SSLs is field intensive and involves automated high-resolution, quasi-continuous sampling or monitoring, at sampling intervals that are short enough to ensure that SSC or flow does not change significantly (i.e. it is "safe" to interpolate Q and SSC between sample values). This interval is often set at between 10 and 30 minutes (commonly 15 minutes) in small- to medium-scale catchments in which flows and suspended sediment fluxes can change quickly (e.g. drainage basin area of < 1000 km^2). Examples of such high-resolution studies in the UK include those of Lawler (2005b, 2006) and Old *et al*. (2003, 2006), and an example of such a study in Iceland is provided by Lawler *et al*. (2003). The monitoring period should also be sufficiently long to be representative of longer term sediment transport conditions, and include a range of hydrological and meteorological events, especially storm sequences during which most sediment transport occurs.

Total suspended sediment load, ($L_{tot,}$) is then evaluated as the flux integral over a given period, as in Ferguson (1987):

$$L_{tot} = \sum_{i=1}^{T/\delta T} Ci.Qi.\delta t \qquad \text{(Equation 5.3)}$$

In Equation 5.3, in SI unit terms, L_{tot} is the total SSL over time (in kg/s), T is the time (in seconds) over which loads are calculated, δt is the fixed sampling or monitoring interval used (in seconds), C_i is the instantaneous SSC (in g/L or kg/m^3) at time i, and Q_i is the water discharge at time i (in m^3/second).

A number of variants of Equation 5.3 exist, depending on data (Q and SSC) availability and national conventions (see, for example, Horowitz, 2003).

5.9.3 Sediment rating curve method: extrapolation techniques

If automatic, continuous monitoring of turbidity is not practicable, then a reasonable, but less accurate and precise, alternative involves the use of sediment rating curves. These have been exhaustively discussed by numerous investigators from several disciplines (e.g. Walling and Webb, 1981a; Ferguson, 1986, 1987; Horowitz, 2008), so only a brief discussion is given here. Based on a limited period of sampling, a relationship is obtained between Q and SSC. The most common form is a log–log power function of the form:

$$\text{SSC} = aQ^b \qquad \text{(Equation 5.4)}$$

In Equation 5.4, a and b are coefficients determined empirically through regression.

The sediment rating relationship is then applied to a high-resolution discharge time series to produce values for the SSLs transported over a given period. Separate sediment rating curves are sometimes used to cater for (1) lower or higher flows; (2) any significant differences between rising and falling limbs of a hydrograph; or (3) seasonal variations in SSC for a given flow (see Figure 5.11). However, the sediment rating curve method has attracted controversy, partly because the relationship between Q and SSC is normally characterised by huge scattering and uncertainty (see, for example, Figure 5.11), and the SSC values used are, of course, *estimates*. Ferguson (1987) recommends the use of a statistical correction factor if log–log sediment rating curves are used.

Figure 5.11 (Walling and Webb, 1981b) shows a cluster of log–log regression lines for 17 rivers (many in western or upland Britain), which demonstrates that for discharges normalised for catchment areas ranging from 0.001 to 5 m^3/second per km^2, SSCs varied from 1 to 5000 mg/L. The value of exponent *b* (from Equation 5.4) for British rivers ranged from 0.3 to 2.0, and typically, was approximately 1.2. It was tentatively suggested by Walling and Webb (1981a) that lowland clay catchments have high *a* values and low *b* values and, conversely, upland catchments with resistant lithology are characterised by low *a* values and high *b* values.

6 Suspended Sediment Concentrations, Fluxes and Yields

6.1 Introduction

Material is delivered from catchments to downstream river reaches, floodplains, reservoirs, lakes, and tidal zones and seas in three main forms: suspended sediment, bedload and dissolved load (solutes). Exceptionally high SSCs can result. For example, a concentration of 500,000 mg/L was observed in the Rio Puerco, New Mexico, USA, by Dunne and Leopold (1978) (i.e. the sample was half sediment by weight). High SSCs, well above 2000 mg/L, have also been sampled in glacial rivers (e.g. Lawler *et al*., 2003) and a Mediterranean river (see Turner *et al*. 2008), but in humid temperate environments, concentrations tend to be lower, often < 1000 mg/L. A range of SSC values with respect to discharge is shown for several UK rivers in Figures 5.11–5.12.

In terms of total river loads, suspended sediment is often the most important flux, although its importance varies widely across the globe and over time. Typically, in humid temperate environments, the percentages of different sediment outputs are as follows: 50% suspended sediment; 45% solutes; and 5% bedload (Walling and Webb, 1981a). Coarse bed material may move on only a few occasions per annum in such catchments. However, in mountain and, especially, glacial rivers, bedload fluxes can account for 50% of total output. In these types of rivers, high and variable discharges and stream powers, and an almost limitless supply of loose, coarse gravels in the subglacial and extraglacial zones and braid plains, combine to allow the drainage system to access and transport large volumes of bedload material downstream. However, in these cold environments, in which chemical reaction rates are low, and runoff is rapid (reducing solute acquisition opportunities), dissolved loads may be as low as 5% of total sediment output.

This chapter summarises some of the available suspended sediment transport data for Irish rivers, and for similar environments in the UK and northern Europe.

6.2 Ireland

6.2.1 Existing research on suspended sediment release to Irish waters

Only a limited number of sediment flux studies have been undertaken in Ireland compared with other Eurpoean countries (Vanmaercke *et al*., 2011), and no comprehensive countrywide Irish study has yet been undertaken, unlike in, for example, the UK (e.g. Cooper *et al*., 2008; Walling *et al*., 2008b). The implementation of the EU WFD (2000/60/EC) has been the principal driver for suspended sediment research in Ireland. The integrated approach, which links water quality to catchment management, that is required according to this legislation has resulted in an increase in the number of studies investigating sediment levels in Irish catchments. Relevant studies are shown in Figure 6.1. All studies, apart from the “Eutrophication from Agricultural Sources” and “Pathways” projects, are ongoing.

Measured Irish sediment yields range from 2.1 to 48.4 t/km^2 per year (Table 6.1) and are relatively low in comparison with those reported for other European countries (Vanmaercke *et al*., 2011). Data related to lake sediments confirm these low sediment levels. As an example, Jordan *et al*. (2002), in a study of Friary Lough (in Northern Ireland), reported reasonably constant yields of approximately 10 t/km^2 per year. Sediment yields throughout the 20th century of between 14 and 58 t/km^2 per year were estimated by Huang and O’Connell (2000) in a study of Ballydoo Lough (Connemara).

The annual mean SSCs reported as part of the Agricultural Catchments Programme (ACP) (Sherriff *et al*., 2015) are lower than the annual mean Freshwater Fish Directive guideline value of 25 mg/L. Furthermore, as reported by Sherriff *et al*. (2015) and Thompson *et al*. (2014), event-based exceedances in Irish catchments are typically limited to short durations. These, however, can be associated with significant

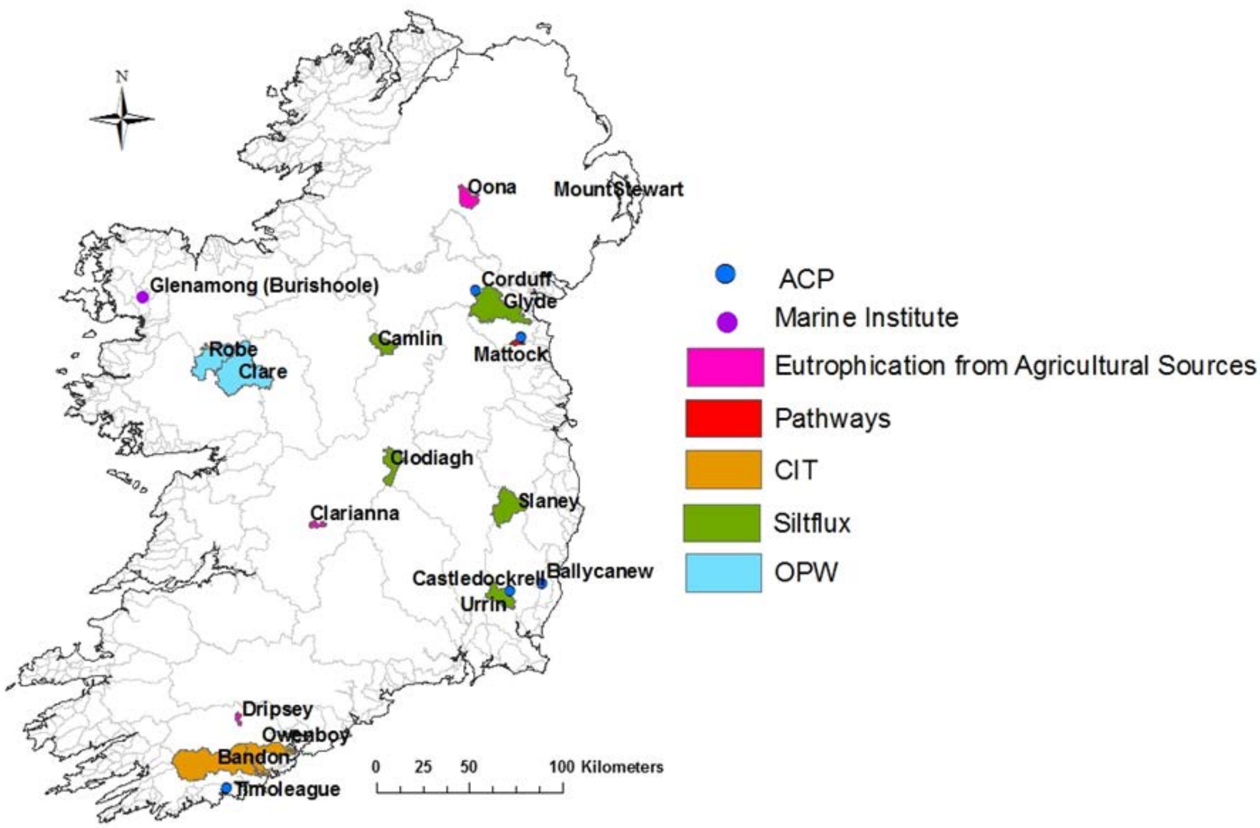


Figure 6.1. Catchments that have been studied in recent sediment-related Irish studies. "Eutrophication from Agricultural Sources" refers to EPA project 2000-LS-2.1.1a-M1; "Pathways" refers to EPA project 2007-WQ-CD-1-S1. ACP, Agricultural Catchments Programme; CIT, Cork Institute of Technology; OPW, Office of Public Works data.

Table 6.1. Summary of Irish sediment yields reported in the scientific literature

Catchment	Area (km^2)	Sediment yield (t/km^2 per year)	Reference
Corduff	3.34	6.07–22.28	Sherriff *et al*. (2015)
Mount Stewart	7.52	6.7	Thompson *et al*. (2014)
Timoleague	7.9	3.95–14.92	Sherriff *et al*. (2015)
Dunleer	9.4	13.5–41.8	Melland *et al*. (2012), Sherriff *et al*. (2015)
Castledockrell	11	2.11–23.1	Melland *et al*. (2012), Sherriff *et al*. (2015)
Ballycanew	11.4	6.65–48.39	Sherriff *et al*. (2015)
Dripsey	15.24	9.8–16.1	Kiely *et al*. (2007)
Glenamong	17.91	16	May *et al*. (2005)
Mattock	20.96	44	Thompson *et al*. (2014)
Clarianna	29.8	8.5	Kiely *et al*. (2007)
The Oona	84.5	29–41	Kiely *et al*. (2007)
Owenabue	103	25.6	Harrington and Harrington (2013)
Bandon	424	14.2	Harrington and Harrington (2013)

sediment loadings and can present issues during sensitive periods of potential salmon migration and spawning. These patterns of sediment mobilisation are consistent with those reported by Harrington and Harrington (2013), obtained from studies of the Bandon and Owenabue Rivers in south-west Ireland. For example, Harrington and Harrington (2010) estimated that 28% of the annual sediment flux of the

Owenabue River was transported over only 5% of the time.

As expected, peak SSCs for agricultural catchments are typically reported during storm events. For example, Harrington and Harrington (2013) reported maximum SSCs of between 837 mg/L and 979 mg/L in their study catchments, which are characterised predominantly by pasture land use. Peak SSCs for the ACP grassland catchments reported by Sherriff *et al*. (2015) range from 419 to 1020 mg/L. These values are comparable to those reported for catchments primarily used for arable farming, which have a higher potential for soil loss from increased tillage operations that expose bare soils. For example, Sherriff *et al*. (2015) reported peak SSCs of 221 to 2141 mg/L for this land use type.

However, there is an increasing recognition that sediment loadings from pasture lands can be increased by poor animal husbandry and land management. This includes, but is not limited to, overgrazing and soil poaching by livestock, particularly around gateways and at drinking points on land or adjacent to rivers. These factors have been shown to adversely affect Irish freshwater systems (Conroy *et al*., 2016). Bank erosion caused by livestock poaching is also problematic in Irish catchments and is one of the factors that contributed to elevated sediment loadings in the Bush catchment, Northern Ireland (Evans *et al*., 2006).

Moreover, the erosion of upland peatland because of overgrazing by sheep can contribute to sediment loadings (Allot *et al*., 2005; May *et al*., 2005). This is a particular problem in upland peat catchments in Ireland. Significantly, sediment from these catchments is characterised by high levels of particulate organic carbon (Ryder *et al*., 2014), which can be more damaging to the ecological status of receiving waters than other types of sediment.

An increase in suspended sediment loadings can also occur during forestry clearfelling (Allott *et al*., 2005, Rodgers *et al*., 2011) and windrowing (Clarke *et al*., 2015) operations. Increases in peak SSC, from 88 mg/L before windrowing to 502 mg/L during windrowing operations, were reported by Clarke *et al*. (2015) and illustrate that forestry operations can produce SSC levels that are comparable to the upper range of those for agricultural catchments.

Although suspended sediment yields in Ireland are relatively low in comparison with those reported for other European countries (Vanmaercke *et al*., 2011), Irish catchments are often characterised by the presence of sensitive habitats, including Atlantic salmon spawning grounds and freshwater pearl mussel beds (National Parks and Wildlife Service, 2008), and, therefore, the ecological impacts of elevated sediment levels are particularly significant. Ireland must, therefore, develop adequate regulations for sediment control. Data from Irish sediment research (Figure 6.1) continue to contribute to the characterisation and understanding of the processes that influence the mobilisation and delivery of fine sediments from source to receptor. This knowledge, combined with sediment yield data from Irish catchments, will be essential for the establishment of standards in a regulatory framework for sediment control.

6.3 Britain and Northern Europe

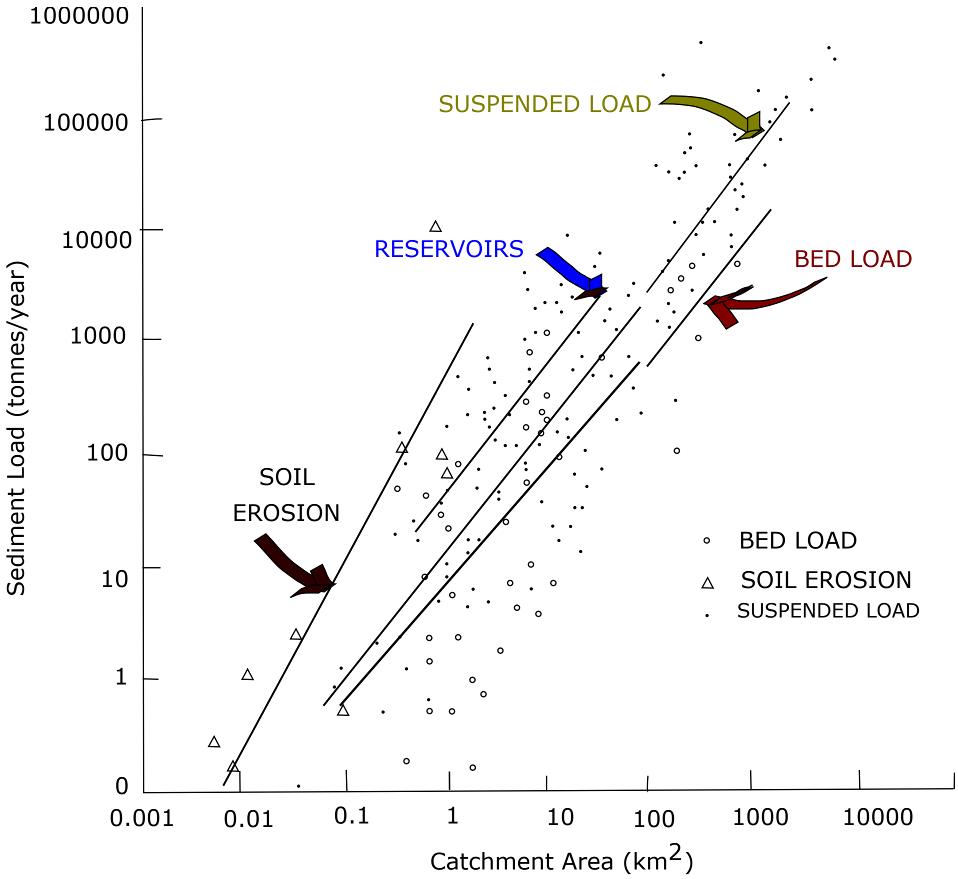
Relationships between suspended sediment yields, bedload and catchment areas for British rivers were derived by Sear *et al*. (2003) and are shown in Figure 6.2. SSLs, in t/year, tend to be higher than bedload fluxes, and are linked to the area of the catchment (A), in km^2, as shown in Equations 6.1 (for small "sediment source" catchments of < 100 km^2) and 6.2 (for large catchments of < 100 km^2):

$$SSL = 11.64A^{1.16}\ (n = 60,\ r^2 = 0.63) \qquad \text{(Equation 6.1)}$$

$$SSL = 31.04A^{1.04}\ (n = 44,\ r^2 = 0.48) \qquad \text{(Equation 6.2)}$$

Note that the exponent for large catchments (i.e. 1.04) is lower than that for smaller catchments (i.e. 1.16); this probably reflects the declining erosion (sediment production) and increasing sediment storage opportunities in a downstream direction.

Cooper *et al*. (2008) and Collins *et al*. (2009) have produced some very useful suspended sediment yield data, which map to over 100 British catchments in south-western England, southern and mid-Wales, the Midlands and the Pennines [including many data from the Natural Environment Research Council's Land Ocean Interaction Study (NERC LOIS) programme], and southern Scotland. This compilation, based on lake sediment accumulation data and fluvial suspended sediment transport estimates, shows that suspended sediment yields vary from 2 to 160 t/km^2



From catchments with Area < 100 km^2

Bed	$L = 5.85\ A^{1.08}$	($n=33$, $r^2 = 0.31$)
Suspended	$L = 11.64\ A^{1.16}$	($n=60$, $r^2 = 0.63$)

From catchments with Area > 100 km^2

Bed	$L = 2.50\ A^{1.16}$	($n=7$, $r^2 = 0.41$)
Suspended	$L = 31.04\ A^{1.04}$	($n=44$, $r^2 = 0.48$)

Figure 6.2. Observed sediment yield (bedload and suspended load) data as a function of catchment area for UK rivers (after Sear *et al.*, 2003).

per year, with the highest yields observed in northern Britain (in the Pennines and the Lake District) and in northern and mid-Wales .

Cooper *et al.* (2008) attempted to link suspended sediment yields to catchment characteristics and three different typologies (the Walling system, the WFD system and a new typology created by the authors of the study) within a combined geographical information system and digital elevation model framework. Table 6.2 shows the relationship between suspended sediment yields and the Walling typology (Walling *et al.*, 2008b): the highest suspended sediment yields were found for upland, low-impact catchments (types 1 and 2; see Table 6.2). For the WFD typology, 16 catchment types (of the 44 types in the WFD definition) were represented in the British dataset. The highest yields tended to be generated from mid-altitude, extra-small and small catchments (Table 6.3).

The new typology developed by Cooper *et al.* (2008) is shown in Table 6.4; from this table, it is clear that

Table 6.2. The Walling catchment typology: links to suspended sediment yield (adapted from Cooper *et al.*, 2008).

Type	Altitude	Impact	Size	Yield (t/km^2 per year)						
				< 3	3–7	7–23	23–56	56–90	> 90	Total
1	Mid	Low	XS	1	2	2	2	5	9	21
2	Mid	Low	S				1		2	3
3	Mid	Agric	XS		1		3			4
4	Mid	Agric	S				3			3
6	Low	Low	XS			1				1
7	Low	Agric	XS				7	3		10
8	Low	Agric	S	3	2	6	7	1		19
9	Low	Agric	M	3	4	9	7	2	2	27
10	Low	Agric	L		1	4	1	3		9
11	Low	Urban	XS			1				1
12	Low	Urban	S			1	1	1		3

Impact is characterised as "low", "agricultural" or "urban". The replacement of "geology" by "impact" in the typology of Walling broadly replaces "siliceous" and "organic" with "low impact", and "calcareous" with "agricultural", with the "urban" classification being new. The catchments are classified according to mean altitude (low: < 200 m; mid: 200–600 m; high: > 600 m) and size (XS: < 10 km^2; S: 10–100 km^2; M: 100–1000 km^2; L: 1000–10,000 km^2; XL: > 10,000 km^2).

Table 6.3. The WFD catchment typology: links to suspended sediment yield (UKTAG, 2008). This typology is based on three catchment characteristics (altitude, size and geometry) (adapted from Cooper *et al.*, 2008)

Type	Altitude	Size	Geology	Yield (t/km^2 per year)						Total
				< 3	3–7	7–23	23–56	56–90	> 90	
1	Low	S	SI			2	1	1		4
2	Low	S	CA	3	2	4	8			17
4	Low	M	SI			1	1	1		3
5	Low	M	CA	3	4	3	2		1	13
8	Low	L	CA		1	4	1			6
10	Mid	S	SI				3		2	5
11	Mid	S	CA		1		1	1		3
13	Mid	M	SI			4	2	1		7
14	Mid	M	CA				2		1	3
16	Mid	L	SI			1		2		3
17	Mid	L	CA					1		1
37	Low	XS	SI				2			2
38	Mid	XS	SI	1	2	1	4	2	6	16
40	Low	XS	CA			3	2	3		8
41	Mid	XS	CA				2	2		4
44	Mid	XS	OR			1	1	1	3	6

The catchments are classified according to mean altitude (low: < 200 m; mid: 200–600 m; high: > 600 m), size (XS: < 10 km^2; S: 10–100 km^2; M: 100–1000 km^2; L: 1000–10,000 km^2; XL: > 10,000 km^2) and dominant geology. Geology is classified by the British Geological Survey as SI, CA, OR or SA (Kinniburgh and Newell, 2003). The SI and CA classes relate to solid geology, while the OR and SA classes are based on near-surface characteristics.

CA, calcareous; OR, organic; SA, saline; SI, siliceous.

Table 6.4. The new "Natural England" typology: links to catchment suspended sediment yield (adapted from Cooper *et al*., 2008)

Altitude	Permeable/ impermeable	Geology	Index	Yield (t/km^2 per year)						Total
				< 3	3–7	7–23	23–56	56–90	> 90	
Low	Permeable	Chalk	LPC	6	7	3				16
Low	Permeable	Other	LPO		1	16	19	5	2	43
Low	Impermeable	Other	LIO			3	5	8	1	17
Low	Impermeable	Peat	LIP				1		1	2
High	Permeable	Other	HPO	1			1			2
High	Impermeable	Other	HIO		2	1	2			5
High	Impermeable	Peat	HIP			1	4	2	9	16

The first letter in the index classification refers to altitude with H and L referring to high (> 330 m) and low (< 330 m) altitudes, respectively. The middle letter relates to the Standard Percentage Runoff with P and I referring to permeable and impermeable catchments, respectively. The third letter refers to the Hydrology of Soil Types soil classification, with C, P and O describing chalk, peat and other soil classes, respectively. HIP, high impermeable peat; LIP, low impermeable peat.

the two highest classes of suspended sediment yields are delivered from catchments classed as "hip" (high impermeable peat; 16 catchments) and "lip" (low impermeable peat; two catchments). The authors also use this new typology to estimate suspended sediment yields across England and Wales from catchment typology information generated by national databases of catchment characteristics on a 1-km^2 grid scale (Figure 6.3).

However, British suspended sediment yields are low by world standards and compared with mountain and glacial catchments in northern Europe. For example, Lawler (2003) showed that the approximately 100-km^2 glacierised basin of Jökulsá á Sólheimasandi in southern Iceland delivers suspended sediment yields of 10,000 t/km^2 per year, and suspended sediment yields for glacial catchments of similar orders of magnitude have been analysed by Gurnell *et al*. (1996). However, the relatively low sediment fluxes of well-vegetated humid temperate environments can still lead to significant negative impacts on fluvial hydroecology (see Chapter 4).

An excellent compilation and analysis of sediment yield data for the whole of Europe, including Ireland, has recently been published by Vanmaercke *et al*. (2011). This was based on a dataset of over 29,000 catchment-years and 1794 sites (507 reservoirs and 1287 gauging stations), for drainage basin areas ranging from 0.01 to 1,360,000 km^2. They found that the lowest sediment yields were in the flat areas of western, northern and central Europe: these catchments accounted for half of the low sediment yields (< 40 t/km^2 per year) in the database, and approximately 80% of the sediment yields of < 200 t/km^2 per year.

Vanmaercke *et al*. (2011) also found a statistically significant ($p < 0.05$) inverse relationship (as others have in smaller-scale studies) between catchment area and sediment yield. This relationship emerged for most climatic zones, including zones classified as "Atlantic", according to their classification, which are of most relevance to Irish environments. This suggests that sediment storage opportunities increase downstream, as steep topography gives way to relatively low and flat landscapes where sediments can be deposited at slope bases, in floodplains or on channel bars and beds. Generally, they found stronger relationships between climate and sediment yield, than between topographic indices and sediment yield.

In a survey of downstream water clarity in three unregulated catchments in New Zealand and Wisconsin, USA, Julian *et al*. (2008a) found that turbidity increased (and clarity decreased) along the channel length to a zone approximately 70% along the continuum; after this point (i.e. further downstream), turbidity declined slightly (Figure 6.3). This is consistent with the mid-basin peak in-stream power modelled by Lawler (1992) and validated by Barker *et al*. (2009), thought to result from enhanced bed instability and river bank erosion.

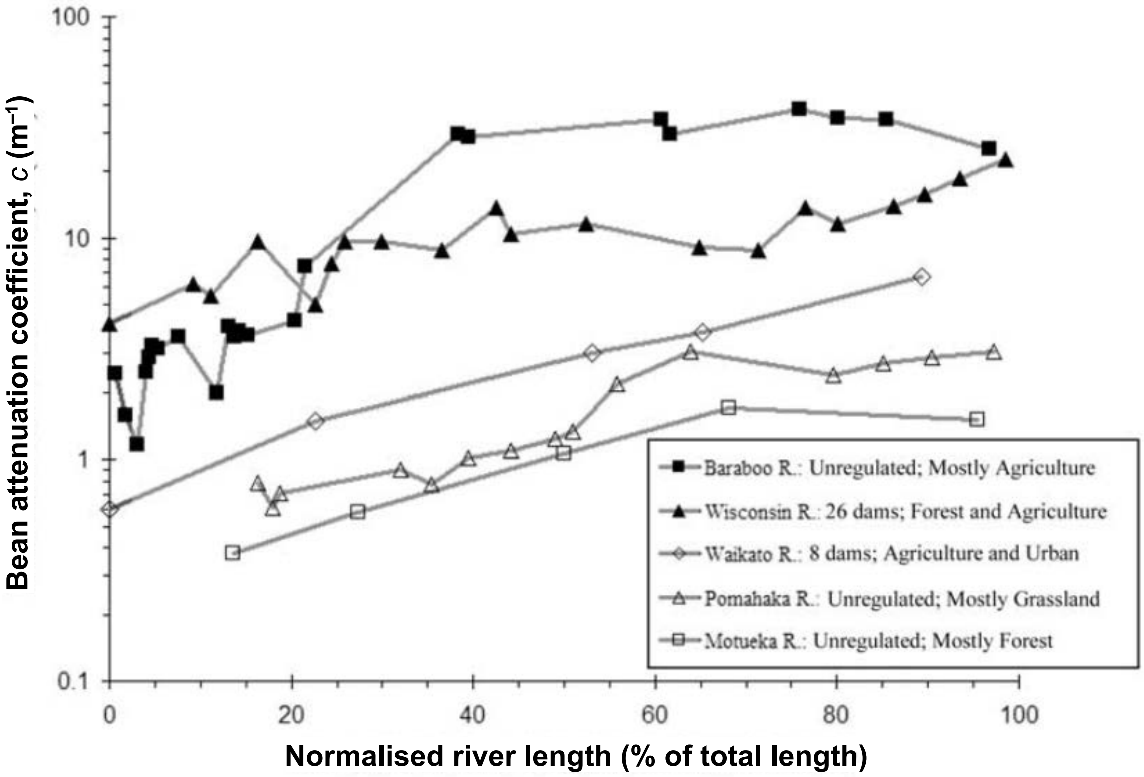


Figure 6.3. Downstream changes in optical water quality in six rivers in Wisconsin (USA) and New Zealand (Julian *et al.*, 2008a).

7 Storm-Event and Seasonal Suspended Sediment Dynamics

7.1 Storm-Event Suspended Sediment Dynamics

Suspended sediment yields are useful, but unless attempts are made to understand the sources, delivery, processes and dynamics of sediment transfer from sources to rivers, then management initiatives aimed at sediment control could be less targeted; it may be more difficult to engage potentially sceptical stakeholders and farmers, and such initiatives may be problematic to implement and possibly fund. A consideration of sediment dynamics is thus crucial for the understanding of the sediment system, and thereby informing any management solutions. This section summarises the typical dynamics of sediment responses in river systems.

Although significant suspended sediment pulsing can occur independently of Q variation in some catchments (e.g. glacierised basins; see Lawler and Brown, 1992; Lawler, 2005b), numerous empirical studies have shown that, in humid temperate environments, most suspended sediment is transported during storm events (e.g. Walling, 1974; Walling and Webb 1981b; Old *et al*., 2003, 2006; Owens, 2009). Indeed, in many catchments, most suspended sediment flux is transported in just a few events, mainly during and soon after rainstorms. For example, Old *et al*. (2003) showed that 40% of annual SSL in the Bradford Beck urban stream is transported during only approximately 1% of the year, and Horowitz (2008) reported that >94% of the fluvial fine sediment transported in Atlanta, USA, occurs during storm events occupying <20% of the time. It is, therefore, *vital* that, in any suspended sediment investigation, maximum resources are devoted to high temporal resolution monitoring and/or sampling of SSCs during such high-flow (or other turbidity-generating) events.

7.2 Hysteresis

One of the most common sediment-related responses to occur during storm events, especially in small catchments, is the rapid rise in SSCs during the rising limb of the hydrograph, then a peak ahead of the flow maximum, before a quick decline. This is the classic response described by the "first-flush model" (Figure 7.1 and Figure 7.2). If SSC is plotted against synchronous discharge, a classic *positive* (i.e. clockwise) hysteresis relationship often emerges, as is shown by, for example, Walling and Webb (1981a) and Walling (1997).

7.2.1 Intra-storm hysteresis effects

A common attribute of catchment sediment systems is positive hysteresis, by which the maximum SSC precedes the flow peak (e.g. Stubblefield *et al*., 2007). Thus, SSCs are already *falling* by the time of the flow peak, suggesting that channel hydraulics are not the only variables that influence suspended sediment transport.

Walling (1974), and many others since, have argued that this positive hysteresis is driven by the *early* flushing of readily accessible, but finite, supplies of loose erodible fine sediment in the catchment. This applies particularly to supplies from river bank erosion, which can be more pronounced early in the storm, when rising flood waters "attack" bank materials that have previously been weakened by weathering and preconditioning processes, such as freeze–thaw action (Hill, 1973, study in Northern Ireland; Lawler *et al*., 1997; Lawler, 2005) (see Chapter 2). Loose, highly erodible, exposed sediment sources are often widely available on catchment surfaces (e.g. from arable fields and poaching sites), and these supplies can be relatively quickly entrained during the start-up phase of the storm. However, these supplies are also quickly depleted – a concept known as "sediment exhaustion" – leaving little available fine sediment for transport in the later stages of the hydrograph.

Positive hysteresis may also arise because of "baseflow dilution", that is, waters arriving later in the storm via sub-surface baseflow routes can be sediment poor because the potential for erosion and entrainment of fine sediment is limited by low through-flow velocities.

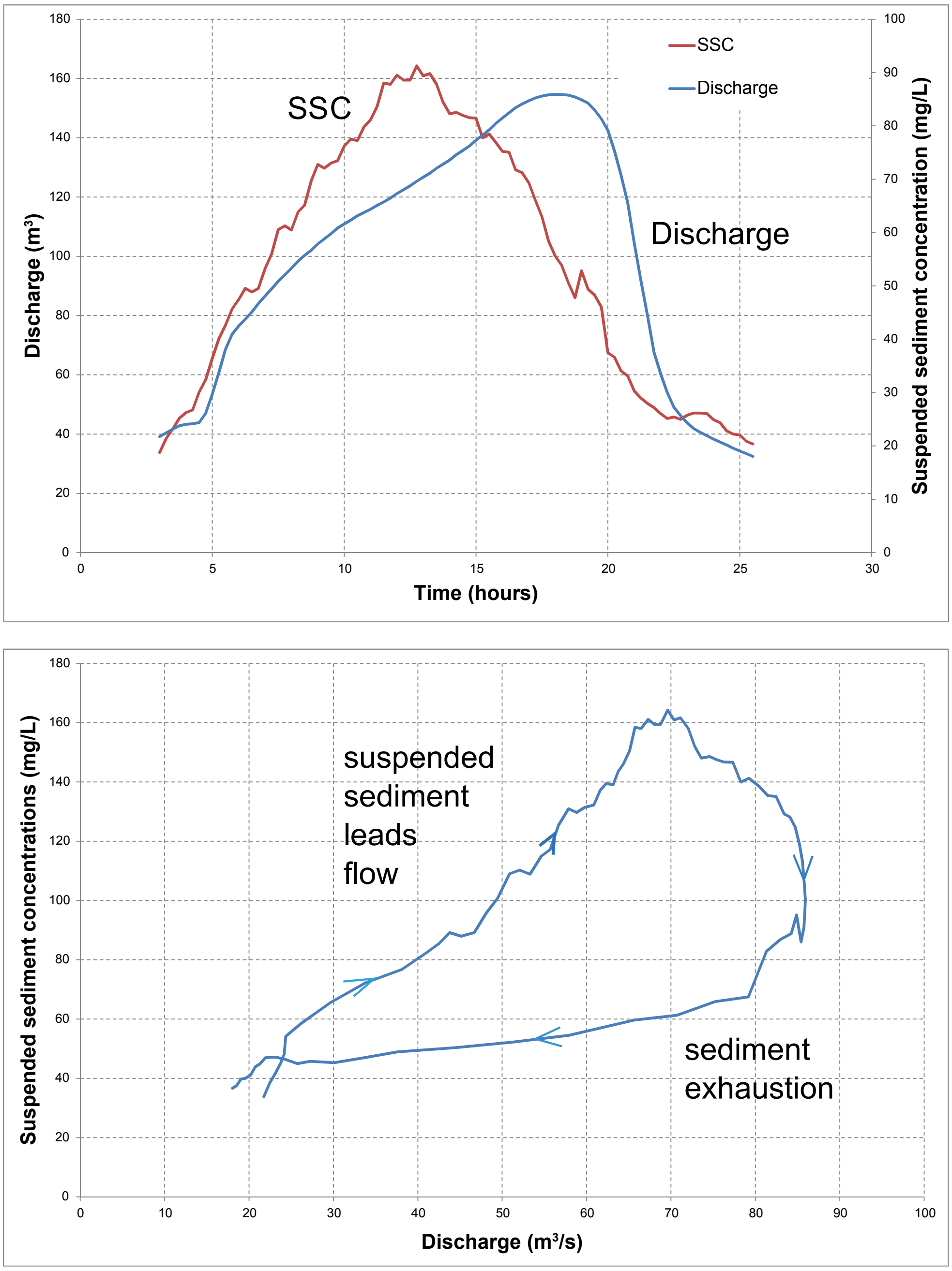


Figure 7.1. SSC response dynamics: SSC leading the flow (top panel) and the classic positive hysteresis and first-flush model of sediment dynamics (bottom panel) (after Lawler *et al.*, 2006).

This first-flush concept is now enshrined in numerical sediment transport models and those for other contaminants [e.g. the Storm Water Management Model of the United States Environmental Protection Agency (USEPA); see Akan and Houhhtalen, 2003]. Such models allow for the accumulation of sediments and/or contaminants between storm events; these supplies are then very quickly entrained by rainfall or river flows early in the storm event, leaving few supplies for the recessional limb of the hydrograph.

7.2.2 Inter-storm hysteresis effects

Positive hysteresis is also evident on weekly timescales if multiple storm events occur. For example, Figure 7.2 shows that for a sequence of storm discharge rises, successive SSC peaks tend to progressively decline, despite flow maxima that are of similar magnitudes. This can also lead to multiple clockwise hysteresis loops.

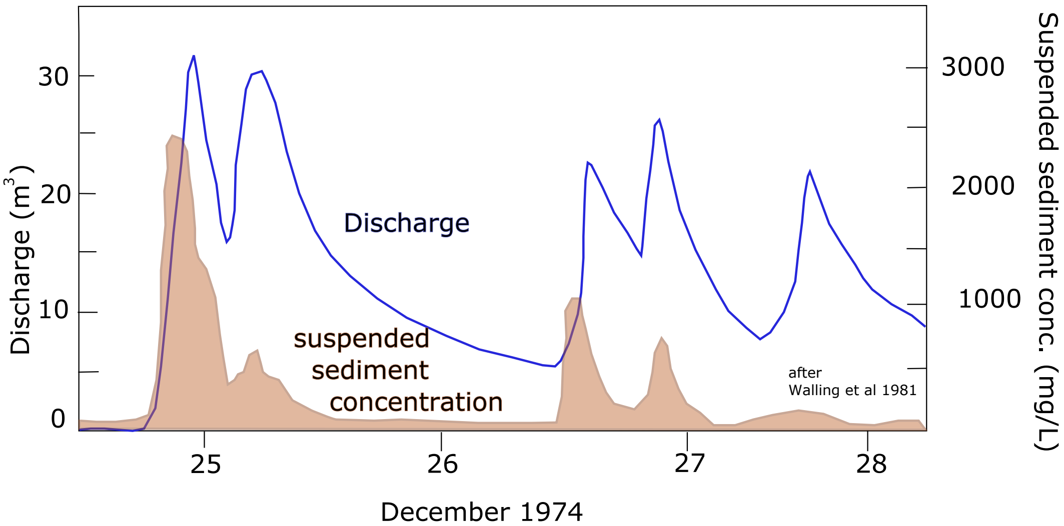


Figure 7.2. Classic suspended sediment dynamics in response to storm-event discharge changes on the River Dart, south-west England (after Walling and Webb, 1981a).

7.2.3 Negative, anticlockwise hysteresis

It is important to stress that the first-flush model, although commonly applicable, is not entirely ubiquitous. For example, studies by Heidel (1956) and Asselman *et al*. (2003) on the Rhine, and Lawler *et al*. (2006) on the urbanised River Tame, have shown that negative, anticlockwise hysteresis, in which peak SSCs occur just *after* the flow maximum (e.g. Figure 7.3), can also be a typical response. For the Tame, although some clockwise hysteresis responses emerged, anticlockwise hysteresis was by far the most common response (Figure 7.4). Lawler *et al*. (2006) quantified the magnitude and direction of hysteresis for these events with a simple hysteresis index, and suggested a range of hypotheses to account for the anticlockwise hysteretic behaviour, including:

1. sediment-wave versus water-wave translation differences (Heidel, 1956; Marcus, 1989; Bull, 1997; Knight, 2005), i.e. wave celerity phenomena;
2. distal sediment sources that take a long time to arrive in the channel, as Heidel (1956) observed;
3. biofilm break-up late in the hydrograph, releasing trapped sediment;
4. combined sewer overflows, which spill late in storm events.

One possibility is the lack of baseflow dilution in urban catchments, because the necessary basin water storage for strong baseflows to occur on recessional hydrograph limbs is largely restricted by highly impervious urban surfaces, such as concrete, glass, steel, tarmac, brick and stone.

Thus, hysteresis, both positive and negative, is very common in fluvial sediment transport systems. One implication of hysteresis is that it makes *predictions* of SSCs from Q using a rating curve approach very difficult (see Section 5.9).

7.3 Seasonal Changes in Suspended Sediment Fluxes

For humid temperate environments, there have been surprisingly few studies devoted to seasonal changes in SSCs or fluxes. This is in contrast to the numerous investigations of seasonal changes in glacierised catchments, in which the high flows of the summer-melt season are matched by very high SSCs and loads (e.g. Lawler *et al*., 2003; Old *et al*., 2005; Stott *et al*., 2009).

Exceptions include the work of the Exeter group led by Walling (Walling, 1974, 1988; Walling and Webb, 1981a) on Devon rivers. These studies showed that, perhaps surprisingly, summer SSC values tend to be higher than winter values (e.g. Figure 5.11). This was thought to be because the inter-arrival time for summer storms is usually much longer than that for winter storms, allowing a longer period for the preparation of fine sediment supplies ready for removal from catchment and river bank surfaces and delivery to the river mainstem. Such preparation processes may

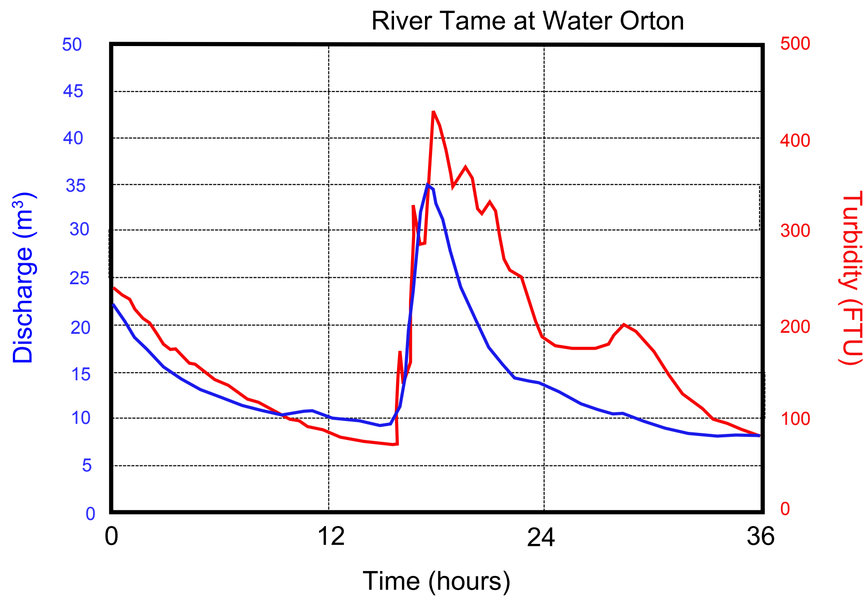


Figure 7.3. The typical suspended sediment dynamic response in the urbanised River Tame catchment (Birmingham, UK) is negative, anticlockwise hysteresis, in which peak SSCs occur just after the flow maximum (after Lawler *et al*., 2006).

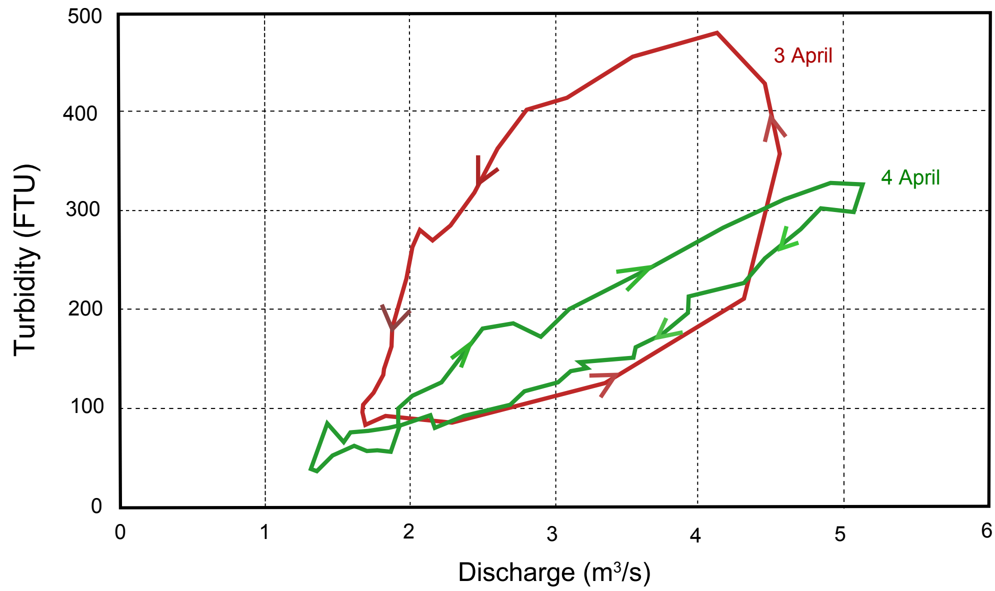


Figure 7.4. Clockwise hysteresis and anticlockwise hysteresis (the most common loops in the Q–turbidity relationship) for the River Tame, Birmingham (after Lawler *et al*., 2006).

include desiccation-related cracking of field soil and, especially, bank materials (see, for example, Lawler *et al*., 1997). This may be exacerbated by a tendency for intense convective storms and storms with greater raindrop diameters to strike in summer. Seasonal suspended sediment hysteresis was also noted for mountain streams that drain to Lake Tahoe (California–Nevada border, USA) by Stubblefield *et al*. (2007).

Although winter suspended sediment *concentrations* may be lower, suspended sediment *yields* may be higher, given the higher river discharges and the tendency for rainstorms to fall on bare arable fields from which soil particulates can be entrained more easily. Kemp *et al*. (2011) show a useful example of this for agricultural catchments, and with respect to seasonally changing cultivation patterns and risk periods for fish at different life stages.

7.4 Longer Term Changes

Changes in suspended sediment yields over decadal and century timescales have been considered, but less intensively. For example, Walling *et al*. (2003) studied sediment cores from lakes and reservoirs in order to reconstruct suspended sediment yields over the previous 150 years for the Ouse and Tweed river catchments in northern England. The sediment yields for the Ouse generally increased after 1963, which reflects the intensification of agriculture at this time. For the Tweed, suspended sediment yields were higher in the earlier period of the study, which the authors ascribed to afforestation and switches from pasture to arable land use in the 1940s and 1950s. Climate apparently had little effect on either catchment. Walling and Fang (2003) analysed data from a selection of the world's rivers and reported a decreasing trend for sediment concentrations; they concluded that reservoir sedimentation played the largest role in controlling sediment flux to oceans.

Lawler *et al*. (2003) found that sediment yields in three Icelandic glacierised basins were amongst the highest in the world, at approximately 10,000 t/km^2 per year. However, in general, these yields decreased during the 1973–1992 period examined, and, although *meltwater* season sediment transport *increased*, this appeared to be more than offset by *decreased* river flows and sediment transport in spring and autumn, leading to a net reduction in suspended sediment yields. Walling (2005) identified a trend for decreasing suspended sediment yields for world river basins, which was ascribed to the impact of dams and reservoirs on sediment storage. Horowitz (2010) documented a decline in suspended sediment fluxes in the Mississippi River from 1981 to 2007, and suggested that this resulted from a reduction in sediment supply (rather than discharge) associated with the emplacement of artificial structures, but also improved land management. Conversely, fine sediment generation appears to be increasing in some catchments, including that of Lake Tahoe (Jassby *et al*., 1999).

8 Effects of Land Use and Climate Change on Sediment Fluxes

8.1 Introduction

The sediment yield of a basin depends on numerous factors including rainfall, storm runoff, basin relief, size and morphology, tectonics, bedrock lithology and structure, topography, soil type, land cover/use and climate (Milliman and Syvitski, 1992; Gordeev *et al*., 1996; Inman and Jenkins, 1999; Farnsworth and Milliman, 2003; Molina *et al*., 2008). This chapter looks at the impact of climate change and land use on sediment fluxes. Land use and climate change (along with runoff) are the only factors that influence sediment yield that are subject to change over relatively short periods. It is therefore important to understand and be able to predict the impacts of these factors on sediment yields.

Every catchment's sediment yield will change uniquely in response to alterations in climate and land use. This often leads to difficulties with regard to making general statements on how a particular change in a certain characteristic of a catchment will affect that catchment's sediment yield. As noted above, land use and climate change are only two of a number of variables that influence the sediment yield of a basin. There are numerous examples in the literature of increasing and decreasing sediment yields of different catchments undergoing the same change in land use (e.g. Bakker *et al*., 2008; Hunter and Walton, 2008). Some basins are better able to cope with changes by remobilising stored sediment in times of deficit (Walling, 1999), while some intensive land use changes may occur in areas of poor connectivity with the river channel, and therefore will not affect the sediment yield of the basin at all. It is important to keep this in mind when trying to identify the likely effects of land use and climate changes on sediment yield.

To accurately determine why sediment fluxes change, it is necessary to have a detailed understanding of past and present information on sediment fluxes, land uses and climatic conditions (Houben *et al*., 2006).

8.2 Climate Change with Particular Reference to Ireland

Over the course of the 20th century, the global-average surface temperature increased by 0.6°C (±0.2°C) (Betts, 2002; McElwain and Sweeney, 2003) and overland precipitation increased by 1% (Kiely, 1999). Global mean temperatures are predicted to have increased by between 1.4°C and 5.8°C by the end of the 21st century, with a corresponding increase in global mean precipitation of 2.4% per 1°C increase in temperature (Betts, 2002; Jasper *et al*., 2004; Fealy and Bates, 2009).

Enhanced hydrological cycles (leading to increases in the rates of precipitation and the rates of evaporation and runoff) are expected for parts of the northern latitudes, including western Europe (Kiely, 1999). In recent decades, there has been an increase in mean precipitation, as well as more frequent heavy precipitation events over the majority of land in northern Europe (Steele-Dunne *et al*., 2008).

While it is uncertain whether or not the trend in Irish temperatures over the previous century matches those identified by the Intergovernmental Panel on Climate Change, the Irish climate, in general, is following similar projections to those predicted by global climate models (McElwain and Sweeney, 2003; Fealy and Bates, 2009).

Ireland and Great Britain have experienced an enhancement of hydrological cycles and a stronger seasonality than before, in precipitation and streamflow, is becoming evident (Kiely, 1999; Betts, 2002); furthermore, even more apparent seasonality is predicted for the 21st century (Arnell and Reynard, 1996; Pilling and Jones, 2002; Fealy and Bates, 2009). February, March and October are the wettest months and appear to be getting wetter, while May, August and September tend to be the driest (McElwain and Sweeney, 2003), although extreme events of higher magnitudes are occurring more frequently in September and October (Kiely, 1999). Winter precipitation in Ireland is expected to increase by between 6% and 13% over the next 40 years, and summer precipitation is expected to remain unchanged or decrease by up to 8% (Betts, 2002).

There is some debate regarding the possible future spatial variations in precipitation in Ireland (Fealy and Bates, 2009), but Kiely (1999) noted that there was a significant increase in precipitation on the west coast of Ireland since 1975, with little or no increase in the east. Charlton *et al*. (2006) predicted a decrease in runoff in the east and south-east, and stated that an increase in the north-west is likely.

With regard to the possible future combination of temporal and spatial variations in Irish precipitation, McElwain and Sweeney (2003) and Charlton *et al*. (2006) have acknowledged that there are likely to be increases in the north-west in winter and decreases in the south-east in summer.

Observational records suggest that both the frequency of occurrence and the intensity of extreme events is increasing (Fealy and Bates, 2009). Kiely (1999) provided evidence that the number of extreme events in Ireland has increased since 1975 (although this probably has more to do with the natural climate variability/influence of the North Atlantic Oscillation rather than global climate change). Murphy and Charlton (2008) and Fealy and Murphy (2009) predicted that the frequency of extreme events (of both high and low magnitudes) will increase over the course of the 21st century.

8.2.1 Effects of climate change on sediment fluxes

More intense rainfall patterns are expected to result in increased runoff, which, in turn, will transport more non-point source runoff into waterways (Kiely, 1999; Dornblaser and Striegl, 2009) resulting in an altered nutrient balance and greater levels of turbidity (Penck *et al*., 2009). More importantly, most sediment is delivered to water bodies during low-frequency storm events; therefore, an increase in the frequency of events is likely to lead to an increase in sediment concentrations/loads (Hunter and Walton, 2008; Scheurer *et al*., 2009).

Thodsen *et al*. (2008) discussed *the secondary* effects that climate will have on sediment yields. An increase in temperature will change the growing and sowing seasons for many crops, in turn changing the temporal and total erodibility of the soil in which they grow. Growing periods for river macrophytes and plants will be extended and this will alter the sediment transport dynamics within the river channel.

The majority of stream flow/runoff values, and hence sediment yields, predicted for the 21st century were based on the assumption that catchment characteristics will remain constant. This assumption is unrealistic as land uses may, and inevitably will, change in the future; such changes will alter the runoff behaviour in catchments and the associated sediment delivery to water bodies.

8.3 Land Use

Moderate land use can double or triple the sediment yields of a basin, while intensive land use can increase yields by an order of magnitude (Saunders and Young, 1983).

Erosion processes create the supply of sediment, while factors and processes such as basin slope, runoff, and hillslope/river channel linkage (Foulds and Macklin, 2006) influence the transport of this sediment. Agricultural land use changes, changes in forest management practices and climate change have led to an increase in erosion rates (Scheurer *et al*., 2009). Global soil erosion potential is estimated to have increased by 17% during the 20th century because of the development of arable land (Yang *et al*., 2003).

The amount of sediment that actually reaches the river channel, relative to the amount available for transport, is denoted by the SDR (Walling, 1997; Asselman *et al*., 2003); this value is unique for each catchment. Some land conditions are more susceptible to erosion than others, while some are better than others at transporting sediment. Changes in land use/cover alter the erosion and transport processes of a catchment; this, in turn, will alter the SDR and, ultimately, the sediment yield. Table 8.1 shows some of the different effects that land cover changes can have on the hydrological characteristics of a catchment.

Table 8.1. Some effects of land cover changes on catchment characteristics (modified from Bronstert *et al.*, 1999)

Change of land cover	Affected catchment characteristics
Urbanisation (increase of impermeable areas)	No soil storage
	Accelerated runoff concentration
Deforestation	Less canopy storage and root depth
	Less litter storage
	Reduced infiltration capacity
Forest damage (by acid rain)	Less canopy storage
Consolidation of farmland (re-allocation and rationalisation)	Less soil and canopy storage
	Reduced surface storage
	Accelerated runoff concentration
Intensive agricultural usage (soil compaction, use of heavy machinery)	Reduced infiltration capacity
	Reduced ponding storage
	Accelerated runoff concentration
Change in vegetation composition (triggered by climate change or by anthropogenic activities)	Changed canopy storage and root depth
	Changed soil parameters, e.g. infiltration capacity

8.3.1 Dominant global agricultural land uses/ covers and influences

Agriculture can have a major impact on sediment fluxes in a basin. The clearance of protective vegetative land cover, for example by the use of machinery on fields (Scheurer *et al*., 2009) or the degradation of land by animal pastures, leaves the soil exposed to erosion and creates drainage pathways over the land, allowing for more rapid and larger volumes of sediment transport. Agricultural practices can affect a landscape's water retention potential and its infiltration capacity (Bronstert *et al*., 1999).

Ward *et al*. (2009) found that in the Meuse catchment (Western Europe), the primary factor controlling long-term changes in sediment yield was the conversion of forests to arable land. Furthermore, the expansion and intensification of agriculture, which consumed and decreased basin buffer strips, is thought to have increased fine sediment loads in rivers in Alpine regions (Scheurer *et al*., 2009).

It is assumed that changes in agricultural management and precipitation patterns have increased the sediment yield in the Danube catchment by 30–50% (Scheurer *et al*., 2009), while the sediment dynamics of the Waipaoa River in New Zealand were severely affected by the large-scale clearance of forest and the subsequent conversion of the land to pasture during the late 19th and early 20th centuries (Gomez *et al*., 2007). Bakker *et al*. (2008) reported substantial decreases in sediment export to rivers in Europe because of the conversion of marginal agricultural arable land to forests, grasslands and scrublands.

8.3.2 Urbanisation

Urbanisation within a basin will induce a hydrological change, as natural land is replaced by impermeable or hard cover layers; this will increase runoff and alter the flood peak of the basin. These changes have implications for the sediment yield of the basin: soil previously available for erosion may be concealed, thus reducing erosion processes and leaving less sediment available for transport.

A study carried out by Inman and Jenkins (1999) identified that the extensive hard cover of the streets and river channels of the Los Angeles urban area has significantly reduced the sediment yield. Urbanisation of a basin in California led to a 20-fold decrease in the SSC, with respect to discharge, in the river by a non-linear dilution process due to changes in the hydrological regime (Warrick and Rubin, 2007).

However, large increases in sediment yields and concentrations can occur in basins undergoing urbanisation, as a result of construction activity (Chen, 1974; Walling, 1974; Warnock and Lagoke, 1974). Chen (1974) found that erosion rates were over 50-times higher in urbanising basins compared with natural basins in the eastern USA. SSLs in Canadian streams were three- to five-times higher in basins undergoing urbanisation than in rural basins in neighbouring streams of approximately equal size (Warnock and Lagoke, 1974). Walling (1974) found that sediment loads and concentrations in a basin near Exeter, UK, increased by 5- to 10-fold and five-fold, respectively, during construction in an area comprising 25% of the basin.

8.3.3 Forestry

Forested catchments have the best natural water storage capacity and lowest runoff yields of all catchment types (Bronstert *et al*., 1999). Sediment is often "trapped" (deposited) in forests, breaking the link between flow paths and the river channel (Van Rompaey *et al*., 2002). The unique features of forest cover, such as root systems, litter layers and canopies, are much more effective than most agricultural crops at protecting soil from water erosion (Chang *et al*., 1982).

Piégay *et al*. (2004) noted that afforestation appears to reduce sediment supply. During a study in France, Liébault *et al*. (2005) found that an unvegetated basin frequently had SSCs higher than 300 g/L, whereas a forested basin of the same size had a maximum SSC of 35 g/L.

The sediment yield of a catchment can increase significantly (by up to 1300%) with increasing rainfall after deforestation (Foulds and Macklin, 2006). As with urbanisation, deforestation results in a change in the hydrologic response of a catchment (Bronstert *et al*., 1999). Deforestation can leave land susceptible to erosion, landslides and mudslides, which can cause further elevations of SSCs by, in some cases, up to nine times that of the undisturbed levels (Fredriksen, 1970).

The sediment yield rates of a basin in New Zealand, which was tracked, harvested and burnt, were eightfold higher than the yield rate of a nearby forested control basin (O'Loughlin *et al*., 1980). The first flush of sediment from a forested basin that had undergone patch-cutting and road construction had a peak SSC that was 250-fold higher than would have been expected from the basin if it had remained undisturbed, and sediment concentrations consistently remained above pre-deforestation levels (Fredriksen, 1970). In Ireland, conifer tree harvesting and windrowing, in preparation for replanting, resulted in elevated episodic inputs of sediment to watercourses that exceeded water quality standards, with the largest releases near the end of the operations (Clarke *et al*., 2015; Kelly-Quinn *et al*., 2016).

Aside from the direct effects of deforestation, it has been shown that the release of suspended sediment into streams because of secondary sources, such as forest roads, tracks and landings, resulted in the most widespread water quality problems in basins in the USA in circa 1970 (USEPA, 1975; Anderson *et al*., 1976; Swanston and Swanson, 1976). It should be noted, however, that Lees *et al*. (1997) found that sediment yields can increase significantly immediately after afforestation and claim that other UK studies have shown order-of-magnitude increases in sediment yield relative to pre-afforestation levels; however, these instances are rare and probably involve a spatial element.

Table 8.2. Total net rainfall, runoff and soil loss resulting from 30 storms between 28 May 1980 and 27 February 1981 in Nacogdoches, Texas (modified from Chang *et al*., 1982)

Forest condition	Net rainfall (mm)	Runoff (mm)	Soil loss (kg/ha)		
			SUS	DEP	Total
Undisturbed forest	359.6	7.4	4.2	6.5	10.7
Thinned forest	377.8	17.0	11.6	5.5	17.1
Clearcut (a)	430.4	41.7	42.5	113.3	155.8
Clearcut chopped	459.4	81.4	75.8	189.2	265.0
Clearcut KG bladed	459.4	137.3	2201.6	1260.0	3461.6
Clearcut cultivated	459.4	119.0	1009.4	2414.0	3423.4

DEP, deposited sediment; SUS, suspended sediment.

Table 8.2 shows the effects of various destructive forest practices on runoff and soil loss. These data prove that the sediment supply and transport levels increase with an increasing intensity of deforestation.

8.3.4 Other land uses and issues: fire, mining and population changes

The land uses discussed so far are the most common and general in global terms. However, there are many other events and types of land use that can change the land. For example, fire, grassland, shrubland, mining and changes in population can dramatically alter the sediment yield of a basin.

An increase in the frequency of fires will leave more forest soil susceptible to erosion (Dornblaser and Striegl, 2009). Soil exposed by fire can become a source of sediment, particularly coarse sediment (Fredriksen, 1970), as the filtering/trapping action of the original vegetation is negated. Warrick and Rubin (2007) noted that sediment discharge and SSCs increased as a result of an increase in the frequency of upland wildfires in California, while Peart *et al*. (2009) showed that hill fires increase storm-period SSCs.

Fiener and Auerswald (2006) highlighted the potential of grassed waterways with regard to reducing sediment delivery: grassed waterways led to a 93% reduction in sediment delivery from an experimental plot in Germany. However, strong gully formation can occur in unmanaged grasslands (Bakker *et al*., 2008), which are more sparsely covered than managed grasslands.

The widespread conversion of arable land to shrubland in Amendoeira, Portugal, during the 1960s resulted in large decreases in sediment export (Bakker *et al*., 2008). In a study by Molina *et al*. (2008), it was found that surface vegetation exerted a first-order control on sediment yield. O'Loughlin *et al*. (1980) discussed the benefits of a riparian strip with regard to reducing sediment export to river channels.

Piégay *et al*. (2004) and Walling (2006) referred to the negative effects that mining can have on sediment yields in a basin, the latter acknowledging it as a key driver of increased sediment loads. Lewin and Macklin (1987) coined the phrase "active transformation" to reflect the morphological impacts of large-scale sediment releases as a result of mining, although this primarily relates to coarse (rather than fine) sediment loads.

Jinze (1991) and Walling (1997) noted the historical impacts of rising populations on sediment loads. For example, Jinze (1991) showed that population increases have, historically, been related to sediment loads, but that under managed conditions this is not the case.

9 Management Implications

Management measures should seek to achieve some combination of (1) a reduction in the sediment load that enters or is carried by a river; and (2) a reduction of the impact of that sediment. It is useful to classify management measures according to the physical process they seek to address. The main physical processes that contribute to SSCs are detachment, mobilisation, advection/dispersion and deposition. Management measures that influence these processes will affect the amount of sediment in rivers.

9.1 Reducing Sediment Load

9.1.1 Rural areas

Management measures seek to (1) reduce the amount of sediment detached from parent material; (2) reduce the mobilisation of such sediment; and (3) prevent the transport of any mobilised sediment into a watercourse, as outlined below:

1. Sediment is detached from its parent material by any combination of (1) the impact of falling raindrops, (2) erosion by flowing water (Ballantine *et al*., 2009) and (3) bank collapse:

   (a) Raindrop impact has been studied by Sharma *et al*. (1993), Salles *et al*. (2000), Thompson *et al*. (2001), Gao *et al*. (2005) and Kinnell (2011). It may be influenced by wind speed and direction (Erpul *et al*., 2004). The effects of raindrop impact are reduced by cover crops or mulching (Meyer *et al*., 1970; Jennings and Jarrett, 1984; Albaladejo Montoro *et al*., 2000; Grismer, 2007). Lal (1984) has reviewed mulching in the tropics. A special case is the use of greenwaste mulch to control the amount of sediment in surface runoff from landfill sites, although it increased dissolved oxygen concentration (Brodie and Misra, 2009). The effects of raindrop impact are reduced if infiltration occurs (Walker *et al*., 2007).

   (b) Erosion by flowing water can be reduced in a field by regrading slopes by, for example, contour terracing, in order to reduce flow velocities. Bank erosion in channels has been studied by Odgaard, (1984), Green *et al*. (1999), Riedel *et al*. (2005) and Kessler *et al*. (2012); in the study by Kessler *et al*. (2012), lidar techniques were used. Erosion can be influenced by sub-surface flows on hillslopes (Fox and Wilson, 2010) and piping (Hagerty and Spoor, 1989). Van Eps *et al*. (2004) developed a bank erosion hazard index based mainly on bank angle, root depth and bank material. However, Segura and Booth (2010) also demonstrated relationships between erosion and the geomorphic setting and degree of urbanisation. Bank erosion can be reduced by restoring the natural character of the stream (Chen *et al*., 2005); by installing special guide vanes (Bhuiyan *et al*., 2010); by bank-side vegetation (see, for example, Abam, 1993), particularly on bends (Hagerty and Spoor, 1989); by rock armouring (Bogen and Bösnes, 2004); and by using flow redirection structures (Yescas *et al*., 2011) and structures made from large woody debris (Shields Jr *et al*., 2004).

2. Mobilisation normally occurs as a result of turbulent flowing water; the turbulence is related to the depth and velocity of flow. Sediment mobilisation is often associated with the mobilisation of phosphorus (Ballantine *et al*., 2009). The process was modelled (using the PSYCHIC model) by Davison *et al*. (2008) and evaluated by Stromqvist *et al*. (2008). A special case of mobilisation is the disturbance of settled sediments; this can have chemical as well as physical effects (Eggleton and Thomas, 2004).

3. The sediment in flowing water can be prevented from reaching a stream by (1) disconnecting the drainage from the stream, (2) retarding the flow so

that sediment settles or (3) storing the water for a certain period, allowing the sediment to settle. Riparian buffer strips are a common measure used to prevent sediment in overland flow from reaching a stream channel. Wenger (1999) recommended a minimum width of 30 m for these buffer strips, mainly to ensure a healthy biota, but acknowledged that wider buffer strips may be necessary on steeper slopes. Detention pond removal efficiencies are in the range of 50–90% (Hartigan, 1988; Schueler *et al.*, 1992).

Measures for reducing the mobilisation of sediments from agriculture and the delivery of sediments to watercourses are summarised in Table 9.1 and Table 9.2. The performances, in terms of sediment reduction efficiency, of selected measures are shown in Table 9.3.

9.1.2 Urban areas

Some common measures for reducing sediment export from urban areas to watercourses are listed in Table 9.4. Street sweeping is performed in most urban areas and, depending on the method used, can be effective at removing particles as small as 10 µm in diameter (CASQA, 2003). With regard to efficiency, street sweeping measures can remove up to 70% of total solids (Schilling, 2005). Hallock (2007) reviewed performance data from a number of studies and summarised the sediment removal efficiencies of some commonly adopted measures (Table 9.5). However, quantifying sediment loads can be difficult because of the possibility of deposition and re-suspension of sediment in pipe systems. Rabinovich and Kalman (2011) reviewed the factors that can influence particle transport in pipes.

Table 9.1. Reducing mobilisation of sediment from agricultural activities

Measure	Reference
Cover crops	Krutz *et al.* (2009)
Crop residue management or straw mulching	Ritter and Shirmohammadi (2001)
Straw bales or wattles, which can also be used as sediment traps	Baxter (2008)
Contour terracing	Al Ali *et al.* (2008)
Careful design of farm and forest roads including choice of location far from riparian zone	Anderson *et al.* (2011)

Table 9.2. Reducing delivery of mobilised sediment to watercourse

Measure	Reference
Buffer strip	Dillaha and Inamdar (1997), Wenger (1999), Owens *et al.* (2007), Grace III and Davis (2010)
Sediment trap (in channel)	Konwinski (1978)
Wetland (natural or constructed)	Zierholz *et al.* (2001)
Vegetated filter strip	Abu-Zreig *et al.* (2004), Verstraeten *et al.* (2006)
Check dams	Shieh *et al.* (2007), Boix-Fayos *et al.* (2008)
Silt fences	Barrett and Malina (2004)
Streambank protection	Van Eps *et al.* (2004), Simon *et al.* (2008)

Table 9.3. Some estimated sediment reduction efficiencies (Evans and Corradini, 2001)

Measure	Estimated efficiency (%)
Crop residue management	64
Vegetated buffer strips	58
Cover crops	15
Terraces and diversions	71
Streambank protection	76

Table 9.4. Reducing sediment export from urban areas to watercourses

Measure	Reference
Sediment traps	Verstraeten *et al*, (2006)
Swales	Shaw and Kuo (2001), Barrett and Malina (2004), Barrett *et al.* (2004)
Retention/detention ponds	Hartigan (1988), Schueler *et al.* (1992)
Street sweeping	CASQA (2003), Schilling (2005)
Curb inlet filters	Smith *et al.* (2007)

Table 9.5. Estimates of reduction efficiencies of best management practices for urban sediment (Hallock, 2007)

Measure	Average efficiency (%)	No. of studies
Catch basin	52	2
Centrifugal separator	12	2
Constructed wetlands	72	13
Dry ponds, vegetated	69	7
Filter strips	64	7
Porous pavement	71	8
Infiltration basins	85	4
Street sweeping	74	2
Wet ponds	56	31

9.2 Monitoring the Effectiveness of Measures

Muleta (2010) used field measurements, obtained using an extensive range of measures, to calibrate a model, based on the Soil Water Assessment Tool (SWAT), that could be used to predict the effectiveness of such measures over a wider spatial area. Similar modelling was done by Bracmort *et al*. (2004, 2006). Sediment source tracking can be used to evaluate the effectiveness of measures (Minella *et al*., 2008). However, the results of specific field investigations can depend on the spatial scale of the study (e.g. Delmas *et al*., 2012). Barrett and Malina (2004) reviewed the effectiveness of silt fence measures.

9.3 The Use of Modelling for the Design and Evaluation of Measures

In the USA, the KINEROS and SWAT models have been used to simulate the effectiveness of soil conservation measures (Arabi *et al*., 2007; Van Liew *et al*., 2007). In Europe, the EUROSEM–GRIDSEM modelling system was developed for a similar purpose (Morgan *et al*., 1994; Botterweg *et al*., 1998; Morgan *et al*., 1998; Kinnell, 1999). It was initially evaluated for European catchments (Folly *et al*., 1999), but has since been more widely used, for example in China (Cai *et al*., 2005), Kenya (Mati *et al*., 2006), Costa Rica, Nicaragua and Mexico (Veihe and Al, 2001). It has also been compared with both KINEROS and SWAT (Smith *et al*., 1995).

10 Standards and Targets

Table 1.1, at the beginning of this review, lists some of the initial sediment limits, which were based on varying ecological impacts. Collins *et al*. (2011) listed international sediment targets for river catchment management, while APEM (2007) and Cooper *et al*. (2008) discussed UK requirements. A special report for the Canadian Council of Ministers gives standards for Canada (CCME, 2001). Foster *et al*. (2011) used palaeolimnological analyses to estimate sedimentation rates in water bodies prior to agricultural intensification in the mid-20th century to assist with setting sediment standards. Rose *et al*. (2011) addressed the wider picture for European lakes since AD 1850.

Table 10.1 shows proposed target and critical suspended sediment yields for various catchment types in England and Wales, while Table 10.2 demonstrates the diversity of the approaches taken and the turbidity limits set by a sample of regulations related to different regions. Note, for instance, that Australia and New Zealand differentiate between catchments on the basis of altitude. In addition, New Zealand now has targets for deposited sediment (Clapcott *et al*., 2011).

Table 10.1. Proposed target and critical suspended sediment yields for various catchment types in England and Wales (Cooper *et al*., 2008; Collins *et al*., 2011)

Catchment type	Target suspended sediment yield (t/km^2 per year) (lower quartile)	Critical suspended sediment yield (t/km^2 per year) (upper quartile)
High impermeable peat	50	> 150
Low impermeable peat	No data	No data
Low impermeable (non-peat, non-chalk)	40	> 70
Low impermeable (non-peat, non-chalk)	20	> 50
High impermeable (non-peat, non-chalk)	10	> 20
High permeable (non-peat, non-chalk)	No data	No data
Low chalk	2	> 5

Table 10.2. Examples of standards/regulations for various countries

Directive/Regulation	Country/state/region	SSs	Turbidity
Freshwater Fish Directive (78/659/EEC and 2004/44/EC)	European Union	≤25 mg/L (apart from exceptional conditions e.g. floods and droughts)	
Canadian Environmental Quality Guidelines for Protection of Freshwater Aquatic Life (CCME, 2007)	Canada	Low flow:	
		<25 mg/L above background (<24 hours exposure)	
		<5 mg/L above background (>24 hours but <30 days exposure)	
		High flow:	
		<25 mg/L above background when backgrounds are >25 but <250 mg/L	
		<10% above background when background is >250 mg/L	
Alaska Water Quality Standards (ADEC, 2016)	Alaska (USA)	The percentage accumulation of fine sediment between 0.1 and 4.0 mm in the gravel bed of waters used by anadromous or resident fish for spawning may not be increased more than 5% by weight above natural conditions and must not exceed a maximum of 30% by weight (grain size accumulation graph)	May not exceed 25 NTU above natural
		In all other surface waters no sediment loads (suspended or deposited) that can cause adverse effects on aquatic animal or plant life, or their reproduction or habitat may be present	
National Recommended Water Quality Criteria (USEPA, 2007)	USA	Suspended and settleable solids should not reduce the depth of the compensation point for photosynthetic activity by >10% from the seasonally established norm for aquatic life	
Oregon Department of Water Quality Standards (Oregon DEQ, 2005)	Oregon (USA)		≤10% cumulative increase relative to upstream control
Australia and New Zealand Guidelines for Freshwater and Marine Water Quality (ANZECC, 2000)	Australia:		
	South-eastern upland (>150- to <1500-m altitude)		2–25 NTU
	South-eastern lowland (<150-m altitude)		6–50 NTU
	South western upland and lowland		10–20 NTU
	Tropical upland and lowland		2–15 NTU
	South central upland and lowland		1–50 NTU
	New Zealand:		
	Upland (>150- to <1500-m altitude)		4.1 NTU
	Lowland (<150-m altitude)		5.6 NTU

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Abbreviations

ACP	Agricultural Catchments Programme
ADCP	Acoustic Doppler current profiler
As	Arsenic
AVHRR	Advanced very high resolution radiometers
AWS	Automatic water sampler
CAFES	Combined Automated, Flood, Elevation and Stream Power (system)
Cu	Copper
D_{50}	Median particle diameter
EPA	Environmental Protection Agency
FTU	Formazin turbidity units
INCA-Sed	Integrated Catchment Model for Sediments
IR	Infra-red
LISST	Laser *in situ* scattering and transmissometry
NTU	Nephelometric turbidity units
OBS	Optical backscatter sensor
Pb	Lead
PEEP	Photo-electronic erosion pin
Q	River discharge
SDR	Sediment delivery ratio
SS	Suspended solid
SSC	Suspended sediment concentration
SSL	Suspended sediment load
SSY	Suspended sediment yield
SWAT	Soil Water Assessment Tool
TSS	Total suspended solid
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
WFD	Water Framework Directive
WWTP	Wastewater treatment plant
Zn	Zinc

AN GHNÍOMHAIREACHT UM CHAOMHNÚ COMHSHAOIL

Tá an Ghníomhaireacht um Chaomhnú Comhshaoil (GCC) freagrach as an gcomhshaol a chaomhnú agus a fheabhsú mar shócmhainn luachmhar do mhuintir na hÉireann. Táimid tiomanta do dhaoine agus don chomhshaol a chosaint ó éifeachtaí díobhálacha na radaíochta agus an truaillithe.

Is féidir obair na Gníomhaireachta a roinnt ina trí phríomhréimse:

Rialú: *Déanaimid córais éifeachtacha rialaithe agus comhlíonta comhshaoil a chur i bhfeidhm chun torthaí maithe comhshaoil a sholáthar agus chun díriú orthu siúd nach gcloíonn leis na córais sin.*

Eolas: *Soláthraímid sonraí, faisnéis agus measúnú comhshaoil atá ar ardchaighdeán, spriocdhírithe agus tráthúil chun bonn eolais a chur faoin gcinnteoireacht ar gach leibhéal.*

Tacaíocht: *Bímid ag saothrú i gcomhar le grúpaí eile chun tacú le comhshaol atá glan, táirgiúil agus cosanta go maith, agus le hiompar a chuirfidh le comhshaol inbhuanaithe.*

Ár bhFreagrachtaí

Ceadúnú

Déanaimid na gníomhaíochtaí seo a leanas a rialú ionas nach ndéanann siad dochar do shláinte an phobail ná don chomhshaol:

- saoráidí dramhaíola *(m.sh. láithreáin líonta talún, loisceoirí, stáisiúin aistrithe dramhaíola)*;
- gníomhaíochtaí tionsclaíocha ar scála mór *(m.sh. déantúsaíocht cógaisíochta, déantúsaíocht stroighne, stáisiúin chumhachta)*;
- an diantalmhaíocht *(m.sh. muca, éanlaith)*;
- úsáid shrianta agus scaoileadh rialaithe Orgánach Géinmhodhnaithe *(OGM)*;
- foinsí radaíochta ianúcháin *(m.sh. trealamh x-gha agus radaiteiripe, foinsí tionsclaíocha);*
- áiseanna móra stórála peitril;
- scardadh dramhuisce;
- gníomhaíochtaí dumpála ar farraige.

Forfheidhmiú Náisiúnta i leith Cúrsaí Comhshaoil

- Clár náisiúnta iniúchtaí agus cigireachtaí a dhéanamh gach bliain ar shaoráidí a bhfuil ceadúnas ón nGníomhaireacht acu.
- Maoirseacht a dhéanamh ar fhreagrachtaí cosanta comhshaoil na n-údarás áitiúil.
- Caighdeán an uisce óil, arna sholáthar ag soláthraithe uisce phoiblí, a mhaoirsiú.
- Obair le húdaráis áitiúla agus le gníomhaireachtaí eile chun dul i ngleic le coireanna comhshaoil trí chomhordú a dhéanamh ar líonra forfheidhmiúcháin náisiúnta, trí dhíriú ar chiontóirí, agus trí mhaoirsiú a dhéanamh ar leasúchán.
- Cur i bhfeidhm rialachán ar nós na Rialachán um Dhramhthrealamh Leictreach agus Leictreonach (DTLL), um Shrian ar Shubstaintí Guaiseacha agus na Rialachán um rialú ar shubstaintí a ídíonn an ciseal ózóin.
- An dlí a chur orthu siúd a bhriseann dlí an chomhshaoil agus a dhéanann dochar don chomhshaol.

Bainistíocht Uisce

- Monatóireacht agus tuairisciú a dhéanamh ar cháilíocht aibhneacha, lochanna, uiscí idirchriosacha agus cósta na hÉireann, agus screamhuiscí; leibhéil uisce agus sruthanna aibhneacha a thomhas.
- Comhordú náisiúnta agus maoirsiú a dhéanamh ar an gCreat-Treoir Uisce.
- Monatóireacht agus tuairisciú a dhéanamh ar Cháilíocht an Uisce Snámha.

Monatóireacht, Anailís agus Tuairisciú ar an gComhshaol

- Monatóireacht a dhéanamh ar cháilíocht an aeir agus Treoir an AE maidir le hAer Glan don Eoraip (CAFÉ) a chur chun feidhme.
- Tuairisciú neamhspleách le cabhrú le cinnteoireacht an rialtais náisiúnta agus na n-údarás áitiúil *(m.sh. tuairisciú tréimhsiúil ar staid Chomhshaol na hÉireann agus Tuarascálacha ar Tháscairí)*.

Rialú Astaíochtaí na nGás Ceaptha Teasa in Éirinn

- Fardail agus réamh-mheastacháin na hÉireann maidir le gáis cheaptha teasa a ullmhú.
- An Treoir maidir le Trádáil Astaíochtaí a chur chun feidhme i gcomhair breis agus 100 de na táirgeoirí dé-ocsaíde carbóin is mó in Éirinn.

Taighde agus Forbairt Comhshaoil

- Taighde comhshaoil a chistiú chun brúnna a shainaithint, bonn eolais a chur faoi bheartais, agus réitigh a sholáthar i réimsí na haeráide, an uisce agus na hinbhuanaitheachta.

Measúnacht Straitéiseach Timpeallachta

- Measúnacht a dhéanamh ar thionchar pleananna agus clár beartaithe ar an gcomhshaol in Éirinn *(m.sh. mórphleananna forbartha)*.

Cosaint Raideolaíoch

- Monatóireacht a dhéanamh ar leibhéil radaíochta, measúnacht a dhéanamh ar nochtadh mhuintir na hÉireann don radaíocht ianúcháin.
- Cabhrú le pleananna náisiúnta a fhorbairt le haghaidh éigeandálaí ag eascairt as taismí núicléacha.
- Monatóireacht a dhéanamh ar fhorbairtí thar lear a bhaineann le saoráidí núicléacha agus leis an tsábháilteacht raideolaíochta.
- Sainseirbhísí cosanta ar an radaíocht a sholáthar, nó maoirsiú a dhéanamh ar sholáthar na seirbhísí sin.

Treoir, Faisnéis Inrochtana agus Oideachas

- Comhairle agus treoir a chur ar fáil d'earnáil na tionsclaíochta agus don phobal maidir le hábhair a bhaineann le caomhnú an chomhshaoil agus leis an gcosaint raideolaíoch.
- Faisnéis thráthúil ar an gcomhshaol ar a bhfuil fáil éasca a chur ar fáil chun rannpháirtíocht an phobail a spreagadh sa chinnteoireacht i ndáil leis an gcomhshaol (*m.sh. Timpeall an Tí, léarscáileanna radóin).*
- Comhairle a chur ar fáil don Rialtas maidir le hábhair a bhaineann leis an tsábháilteacht raideolaíoch agus le cúrsaí práinnfhreagartha.
- Plean Náisiúnta Bainistíochta Dramhaíola Guaisí a fhorbairt chun dramhaíl ghuaiseach a chosc agus a bhainistiú.

Múscailt Feasachta agus Athrú Iompraíochta

- Feasacht chomhshaoil níos fearr a ghiniúint agus dul i bhfeidhm ar athrú iompraíochta dearfach trí thacú le gnóthais, le pobail agus le teaghlaigh a bheith níos éifeachtúla ar acmhainní.
- Tástáil le haghaidh radóin a chur chun cinn i dtithe agus in ionaid oibre, agus gníomhartha leasúcháin a spreagadh nuair is gá.

Bainistíocht agus struchtúr na Gníomhaireachta um Chaomhnú Comhshaoil

Tá an ghníomhaíocht á bainistiú ag Bord lánaimseartha, ar a bhfuil Ard-Stiúrthóir agus cúigear Stiúrthóirí. Déantar an obair ar fud cúig cinn d'Oifigí:

- An Oifig um Inmharthanacht Comhshaoil
- An Oifig Forfheidhmithe i leith cúrsaí Comhshaoil
- An Oifig um Fianaise is Measúnú
- An Oifig um Cosaint Raideolaíoch
- An Oifig Cumarsáide agus Seirbhísí Corparáideacha

Tá Coiste Comhairleach ag an nGníomhaireacht le cabhrú léi. Tá dáréag comhaltaí air agus tagann siad le chéile go rialta le plé a dhéanamh ar ábhair imní agus le comhairle a chur ar an mBord.

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SILTFLUX Literature Review

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epaResearch
Climate - Water - Sustainability
Identifying pressures • Informing policy • Developing solutions

Identifying Pressures

Fine sediment delivery to rivers is increasingly recognised internationally as a substantial water quality and hydro-ecological problem. The SILTFLUX project aimed to improve knowledge of fine sediment delivery as a pollution pressure in Irish rivers. The project has studied sediment flux dynamics with respect to key flow events and their actual and potential ecological impacts in different Irish river systems that are subject to variable land-use pressures. The SILTFLUX Literature Review distils the current knowledge of these effects and of methodologies for reducing their impacts in conditions typical of Ireland.

Informing Policy

The SILTFLUX project will help inform environmental management and policy in the setting of standards for suspended sediment fluxes and concentrations appropriate for the protection of sensitive catchments in Ireland. The SILTFLUX Literature Review synthesises the considerable international debate that has surrounded the basis for establishing such standards, particularly on the issues of (i) whether they should be based on sediment loads, suspended sediment concentrations, deposited sediment or all three, and (ii) how such standards can account for the biological impacts of both transported and deposited sediment. Addressing these issues in a framework that is easily and reliably measurable and which lends itself to monitoring on a national scale remains a challenge.

Developing Solutions

The SILTFLUX project has also identified the benefits of suspended sediment reduction possible from a broad range of measures and land management practices, to support the development of mitigation policies. A review of the published literature and existing measured data was undertaken to establish an initial "state of the art" position. The SILTFLUX Literature Review was international in scope but included key foci on information that was relevant to the conditions and pressures experienced in Irish catchments. Existing sediment datasets from the literature, as well as the projects own data, were compiled in a database that identifies the range of fluxes to be expected in Irish river systems.

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