Quantifying Fine-Grained Sediment Storage in Lowland Headwater Streams of the East Midlands

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Abstract

Fine-grained (≤2mm) sedimentation of streams can cause detrimental impacts on ecological quality. However, there is currently little information on the quantity and spatial variability of fine-grained sediment (FGS) stored in river channels. As a result, there are few baseline data available to evaluate the success of any measures designed to reduce FGS supply, and by extension FGS storage (e.g. catchment land management). There is, therefore, a need for a reliable in-stream monitoring technique to quantify FGS storage. This research, which was based in the East Midlands, developed and employed a field-based methodology for determining FGS storage in stream channels. The method used a combination of sampling a given volume of bed material using a McNeil corer, and a resuspension technique using a streambed shear stress achieved with a mixing paddle attached to a cordless electric drill. FGS storage was evaluated in riffles and pools of Stonton Brook ($42km^2$), Eye Brook ($61km^2$) and the Upper Welland ($53km^2$). FGS was found to be dominated by sand-sized particles. The mean average at-a-site storage in the Upper Welland, the Eye Brook and Stonton Brook were 4977 ± 511 g m^{-2} cm^{-1}, 5710 ± 437 g m^{-2} cm^{-1} and 4626 ± 342 g m^{-2} cm^{-1}, respectively. FGS storage in pools exceeded that of riffles. Surficial remobilisable storage of fines was also higher in pools than riffles. The organic matter content of the FGS was low, and showed little variation between pools and riffles. The baseline data set collected could be used in the future to evaluate the success or otherwise of catchment land management interventions in reducing the quantity of fine sediment stored in the streams investigated.

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A final thank you is to my family and friends for their continued support and encouragement, especially my housemate Nicola who had to deal with countless samples and pieces of muddy equipment decorating the dining room for a year!

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List of abbreviations

BOD – Biochemical Oxygen Demand

CEH – Centre for Ecology and Hydrology

CSF – Catchment Sensitive Farming

DEFRA – The Department for the Environment, Food and Rural Affairs

DO – Dissolved oxygen

DSEH – Differential sediment entrainment threshold

EA – Environment Agency

EC – European Commission

EU – European Union

FGS – Fine-grained sediment

GES – Good ecological status

GSD – Grain size distribution

GWCT – Game and Wildlife Conservation Trust

LOI – Loss on Ignition

LWD – Large woody debris

masl – metres above sea level

NIEA – Northern Ireland Environment Agency

OM – Organic matter

RO – Reverse osmosis

SDR – Sediment delivery ratio

SEM – Standard error of the mean

SOD – Sediment oxygen demand

SS – Suspended sediment

SS_c – Suspended sediment concentration

STAR – Standardisation of River Classifications project

SWBs – Surface water bodies

UKTAG – United Kingdom Target Action Group

WFD – Water Framework Directive

WFF – Water Friendly Farming

Chapter 1 – Introduction

1.1 Overview

Rivers play an important role in landscapes, transporting water and sediment from catchment hillslopes to the sea. Anthropogenic activities (e.g. deforestation, construction and agriculture) can modify sediment dynamics, and the intensification of catchment land use has resulted in many river systems experiencing an increased supply of fine-grained sediment (FGS) (Owens *et al.*, 2005; Cooper *et al.,* 2008; Jones *et al.*, 2012b). Although FGS is an integral part of the functioning of river systems, fine sediment levels in many river systems now exceed natural conditions (Kaller and Hartman, 2004), and 'sediment problems' exist (Walling *et al.*, 2006) for the overall water quality and ecological quality of affected waterbodies. The impacts of FGS include changes in organism abundance, habitat quality and quantity, and burial, abrasion and scour (Wood and Armitage, 1997; Acornley and Sear, 1999; Petticrew *et al.*, 2007; Navratil *et al.*, 2010; Descloux *et al.*, 2013; Von Bertrab *et al.*, 2013).

The European Union's (EU) Water Framework Directive (WFD) aims to ensure that all water bodies in the EU meet good ecological and chemical status (Hering *et al.*, 2010). Where FGS deposition is considered to be a driver for the deviation of ecological quality from the reference state, methods to reduce sediment supply may be implemented. Alongside in-channel management and stream restoration, improving catchment land management practices have been proposed as ways of reducing FGS inputs to surface waters, and thence improving the ecological status of water bodies (Soulsby *et al.*, 2001). Monitoring schemes have also been set up to classify the status of surface water bodies (SWBs) and to evaluate the impact of any catchment management strategies that have been implemented (European Commission [EC], 2015).

Although significant monitoring of suspended sediment (SS) dynamics has occurred, under the WFD, FGS is neither a priority substance, nor a specific pollutant, therefore, recommendations for environmental standards have not been fully developed (Department for Environment, Food and Rural Affairs [DEFRA], 2014). The assignment of management targets is complicated by the complex, highly variable relationship between sediment delivery and ecological impact (Walling *et al.,* 2007). Furthermore, most focus to date has been directed on determining sediment yields at catchment outlets. However, ecological effects are probably more related to the net deposition of sediment rather than simply its transport through the catchment. Hence, FGS storage is likely to be a key player in determining ecological quality, making quantifying it of importance. Monitoring the success of catchment management strategies should also include estimates of FGS storage, with changes in storage representing changes in inputs and outputs to/from the system (Young *et al.*, 1991; Sutherland, 1998; Descloux *et al.*, 2013; Duerdoth *et al.*, 2015). Such monitoring should examine storage at local scales because this is the scale of the habitat for many riverine taxa. However, there is currently little information on the quantity and spatial variability of FGS stored in river channels (Petticrew *et al.*, 2007). As a result, there are few baseline data available to evaluate the success of land management schemes. Quantifying FGS storage is therefore necessary.

1.2 Research aims and objectives

A reliable methodology and a set of baseline data is required to determine the ecological quality of surface water bodies and the effect of catchment management initiatives on FGS storage. The aim of this research, therefore, is to employ a method for quantifying the spatial variation of FGS storage within and between river catchments in order to provide information on FGS storage in lowland environments, and to provide critical baseline data needed to evaluate the success or otherwise of catchment land management interventions in reducing the quantity of fine sediment in river systems. This will allow changes in the amount, and characteristics, of FGS stored in river systems to be determined in the future. The method will also allow the spatial variation of sediment storage in riffle-pool sequences within and between catchments to be quantified.

Several research objectives have been identified. The main objectives of this research are:

Objective 1: Quantify FGS storage per unit bed area, and as a proportion of total sediment, in riffles and pools of the study reaches.

Objective 2: Determine the proportion of FGS which could be mobilised when the stream bed is disturbed.

Objective 3: Determine the particle size distribution and organic matter content of the FGS stored in pools and riffles.

Objective 4: Collect a robust set of baseline data which could be used in future years to determine if there have been any long-term changes in FGS storage due to catchment land management initiatives.

1.3 Structure and content

This thesis is structured into seven chapters. Chapter 2 reviews the literature on FGS and provides the rationale for the research aims and objectives. Chapter 3 outlines the study sites and provides an overview of the methodologies used in the collection and processing of data. Results are presented in Chapters 4 and 5 and discussed in Chapter 6 before finally drawing a conclusion to the work in Chapter 7.

Chapter 2 – Literature review

2.1 Introduction

As outlined in Chapter 1, FGS plays an important role in river systems, and is critical for their ecological functioning. This chapter will explore the role that FGS plays in river systems. A definition of FGS will be provided, and sources of FGS and anthropogenic influences on FGS load will be considered. FGS dynamics in the fluvial system will be explored. As a result of FGS storage being a major environmental concern, a particular focus will be placed on FGS storage. It will also look at approaches to managing FGS problems, as well as methodologies for monitoring and quantifying FGS in river channels.

2.2 Definitions of fine-grained sediment

FGS has been defined in various ways, with size definitions ranging from the very coarse sand fraction (≤2mm) to the coarse silt fraction (≤0.063mm) (Table 2.1) (Wood and Armitage, 1997; Gibson *et al.*, 2009; Bryce *et al.*, 2010; Fox *et al.*, 2010; Grabowski *et al.*, 2012; Evans and Wilcox, 2014). The sand fraction has important ecological and geomorphological implications in river systems, since it fills interstitial pore spaces, thereby modifying river flow and transport dynamics. Studies that ignore sand have therefore found to be limited (Petticrew *et al.*, 2007). It is also important to consider particles ≤0.063mm in FGS studies, since this size fraction acts as a pollutant vector. Nutrients and contaminants commonly sorb to silt and clay particles, meaning silt and clay particles are both ecologically and environmentally significant (Fox *et al.*, 2010). Similarly, organic matter (OM) is a key contributor to the FGS load in some river systems, and also has important implications for benthic communities (Wood and Armitage, 1997), making it an important component of FGS. Therefore, in this study, FGS is defined as sediment ≤2mm, incorporating inorganic sand, silt and clay particles, as well as organic particles (Jones *et al.*, 2014).

As a result of grain sizing techniques not maintaining sediment structure (Droppo *et al.*, 1998), this definition of FGS is notwithstanding the fact that cohesive fluvial suspended sediment commonly travels as flocculated particles, which form as particles collide in transport. Flocs are common in river systems (Syvitski, 2007).

Inorganic matter, biota and bioorganic matter, water and pore spaces can aggregate to form dynamic flocs (Droppo, 2001). If collision stresses are less than the shear strength of a particle, collision will initiate flocculation. If collision stresses exceed the shear stress of the particles, disaggregation will occur, with the number of particles deflocculating dependent upon the strength of the collision stresses (McAnally and Mehta, 2002). Flocculation and deflocculation can modify the settling velocity of particles (Kranck, 1975), and cause deposition to occur, impacting water quality (Droppo, 2001; McAnally and Mehta, 2002; Bilotta and Brazier, 2008; Thompson and Wohl, 2009). As a result, grain size distributions cannot always be directly related to transport dynamics (Kranck, 1975; Droppo *et al.,* 1998).

Table 2.1. Particle size classifications. **Source:** Blott and Pye (2001).

Size term		Class size (mm)	Upper size limit (phi)
	Boulder	D≥64.00	-6
	Gravel	2.00≤D<64.00	-1
Sand	*Very coarse*	1.00≤D<2.00	0
	Coarse	0.50≤D<1.00	1
	Medium	0.25≤D<0.50	2
	Fine	0.125≤D<0.25	3
	Very fine	0.063≤D<0.125	4
Silt	*Very coarse*	0.031≤D<0.063	5
	Coarse	0.016≤D<0.031	6
	Medium	0.008≤D<0.016	7
	Fine	0.004≤D<0.008	8
	Very fine	0.002≤D<0.004	9
	Clay	D<0.002	N/A

2.3 Fine-grained sediment in lowland agricultural catchments

Many channels adjust to a steady, equilibrium state, whereby there is a balance between the sediment supplied to the system and the ability of the flow to transport the sediment; form and process are thus balanced (Gregory and Walling, 1973; Brandt, 2000). FGS dynamics play a role in adjustment to equilibrium conditions. This section will explore FGS dynamics in river systems, including sediment supply, entrainment, transport, deposition and storage.

2.3.1 Fine-grained sediment supply in lowland agricultural catchments

Sources of FGS to river channels are numerous. As Wood and Armitage (1997) suggest, sources can be autochthonous or allochthonous. That is, sediment can originate from within the river channel, or outside of it. Channel banks, bars subject to erosion, interstitial fines and surficial deposits are autochthonous sediment sources. Fines trapped in aquatic vegetation e.g. macrophyte stands, and biotic particles e.g. phytoplankton and zooplankton, are also autochthonous sources. Kronvang *et al*. (2013) suggest that autochthonous fines can contribute significantly to FGS yields. Allochthonous sources include exposed soils, mass failures e.g. landslides and soil creep, litter fall from riparian vegetation, atmospheric deposition due to precipitation, aeolian processes and anthropogenic activities (Wood and Armitage, 1997; Helfield and Naiman, 2001). Active transport pathways are required to deliver allochthonous FGS to the river channel; without active pathways, the sediment will not be delivered to the channel. Allocthonous fines are generally transported to the river channel diffusely, however they can also transported to the river system through point source mechanisms, e.g. field drains (Buendia *et al.*, 2013b).

General controversy exists over the relative importance of different FGS sources to total FGS inputs (Russell *et al.*, 2001). Inputs of FGS from point source pollution are easy to identify, however, diffuse inputs – which are thought to be a key contributor to the sediment problem of streams – are more difficult to track (Collins *et al.*, 2009). Sediment tracing methods, e.g. sediment fingerprinting, are commonly used in an attempt to identify the sources of FGS in the catchment (Walling, 2005; Collins and Walling, 2007b; Minella *et al.*, 2008; Minella *et al.*, 2009). The technique has proved particularly useful for determining the contribution of different sources to the FGS load, and thus the importance of the source in contributing to FGS problems (Collins *et al.*, 2013). Results of such studies have shown that anthropogenic activity, e.g. deforestation, agriculture, construction and quarrying, has a key influence on FGS dynamics and has generally resulted in increased FGS supply to river channels (Milan *et al.*, 2000; Marzin *et al.*, 2012).

Paleolimnology has been used to reconstruct historic sediment yields (Foster *et al.*, 2011), and, where records of land use are kept, sediment yield data can be related to them (David *et al.*, 1998). Modern rates of erosion from cultivated lands are higher than historic levels, and are thought to be up to an order of magnitude higher than erosion levels from undisturbed conditions. Erosion rates on cultivated land can range from 0.01-0.30 kg m^{-2} a^{-1}, whereas natural soil erosion rates range from 0.01-0.05 kg m^{-2} a^{-1} (Walling, 1995). As a result, sediment delivery from catchments impacted by anthropogenic activities has increased over time (Owens *et al.*, 2005; Larsen *et al.*, 2009), and exceeds that of unimpacted catchments (Bennion and Battarbee, 2007). Kaller and Hartman (2004) suggest that the threshold level of fine sediment accumulation –the level beyond which the river system becomes impaired – has been exceeded in many lotic environments. Land use practices threaten the ecological integrity of freshwater habitats (Galbraith *et al.*, 2006).

Agricultural expansion and intensification are thought to be key anthropogenic pressures contributing to the increased trends of run-off, erosion and FGS delivery to river channels (Cooper *et al.,* 2008; Wagenhoff *et al.*, 2011; Jones *et al.*, 2012b). In the UK, such changes in land use have largely been driven by government policies (Evans, 2006). Subsidies in the 1800s, for example, encouraged the installation of field drains, which control the water level in soils, and help enhance soil productivity. They are also responsible for the transport of FGS. When the subsidies ceased in the mid-1980s, construction of field drains reduced, however field drains are still thought to be responsible for up to 55% of the suspended load in lowland catchments (Russell *et al*., 2001). Joining the EU in 1973 resulted in further changes to farming patterns, and subsequent increases in sediment yields (Evans, 2006). Overgrazing of pasture land associated with increasing livestock densities has also generated reduced infiltration and increased soil erosion (McGinty *et al.*, 1979). Livestock can poach channel banks, releasing FGS to channels (Walling and Amos, 1999). Walling *et al.* (2003b) and Greig *et al.* (2005) also note the importance of increased erosion and sediment yields attributable to the conversion of permanent pasture to arable land during World War II.

River regulation programmes have further modified fine sediment dynamics in streams and contributed to the fine sediment problem of river systems. Stream regulation often results in reduced peak discharges, preventing fines from being flushed from the bed, thereby increasing storage longevity. Regulation and other anthropogenic activities are thought to have resulted in FGS dynamics being disrupted (Diplas and Parker, 1992; Sear, 1992).

The effects of anthropogenic activity on FGS are not constant over time and space; anthropogenic activity impacts catchments differently. David *et al*. (1998) were able to identify sources of FGS in lake sediment records in Leicestershire and, contrary to expectations, found that quarrying, road construction and agricultural intensification had little impact on sediment yields. They attributed this to the low drainage density of the catchment. Walling and Fang (2003) and Walling (2008) also note that anthropogenic activity has, in some catchments, reduced FGS loads. Soil and water conservation, land management practices, extraction of sand for construction, and reservoir and dam construction have attenuated fine sediment supply and decreased fluxes of FGS. Reservoirs trap up to 30% of the global sediment flux to oceans (Owens *et al.*, 2005), and prevent it being transported downstream. This can have the effect of creating "hungry water" (Kondolf, 1997: 535). As inputs of sediment to the river system are reduced, the excess energy the stream possesses is used to move sediment off the stream bed (Collier *et al.*, 1997; Hazel *et al.*, 2006). If the sediment supplied to the stream is continuously less than the transport capacity of the stream, the bed will coarsen to a point beyond which no more sediment transport can occur (Kondolf, 1997).

2.3.2. Sediment transport

Criteria for sediment transport must be met in order for sediment to be transported through the system. Transport criteria are generally defined in terms of entrainment thresholds, and thresholds of entrainment must be exceeded in order for a particle to be transported (Newson, 1992). Entrainment thresholds are often defined in terms of a critical dimensionless shear stress (θ_c) (Equation 2.1; Figure 2.1) or a critical velocity (Figure 2.2). If the dimensionless shear force exceeds the critical value, entrainment will result. Fine particles require a high θ_c to be entrained in the flow as a result of the cohesive nature of FGS. Turbulent bursts can cause instantaneous increases in uplift which are large enough to entrain a sediment particle that would otherwise remain on the stream bed. The forces required to lift a particle into suspension are greater than those required for the particle to remain in suspension, therefore turbulence can increase suspension and suspended sediment transport.

$$\theta_c = \frac{\tau_c}{gD(\rho_s - \rho)} \qquad (2.1)$$

where θ_c is the critical dimensionless shear stress, τ_c is the critical shear stress and $\tau_c = \rho gRS$, ρ is fluid density (1000 kg m^{-3}), g is gravitational acceleration (9.8 m s^{-2}), R is the hydraulic radius ($R = \frac{A}{P}$; A = cross sectional area [m^2], P = wetted perimeter [m]), S is channel slope, D is grain diameter (usually D_{50}) and ρ_s is particle density.

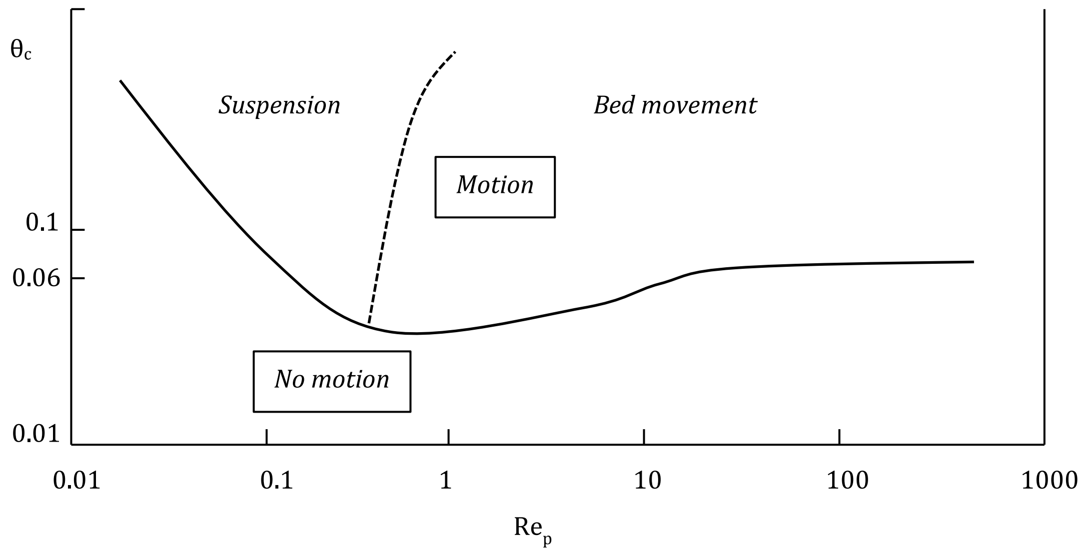


Figure 2.1. Shields' entrainment function. Dimensionless critical shear stress (θ_c) is related to particle Reynolds number (Re_p). Modified from Knighton (1998: 110).

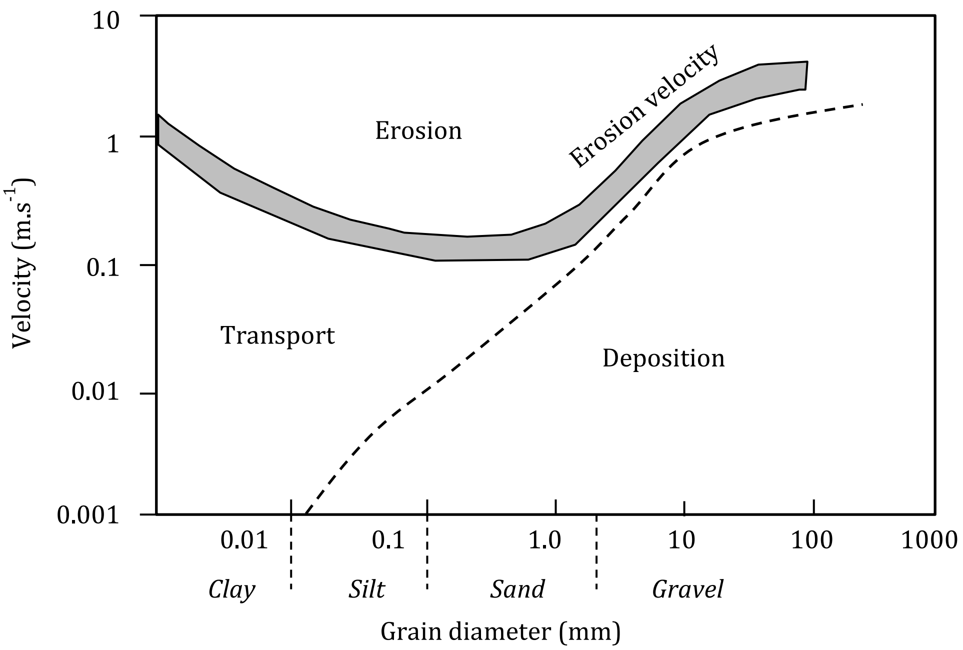


Figure 2.2. Erosion, transportation and deposition criteria, as defined in terms of threshold velocity. As flow velocity increases, transport is more likely. As a result of the cohesive nature of FGS, coarse and fine particles have similar erosion thresholds. **Source:** Hjulström (1935).

Once sediment is entrained in the flow, it will be transported through the river system, and contribute to the sediment yield of a catchment, with the sediment yield representing the total amount of sediment passing the catchment outlet over a given time period, per unit area (t km^{-2} yr^{-1}) (Kusimi *et al*., 2014). Sediment yield information can be used to estimate sediment delivery ratios (SDR), and by extension storage. The SDR equals the ratio between sediment yield and gross erosion (Walling, 1983).

Various modes of motion for sediment transport exist, including traction, saltation and suspension (Figure 2.3). FGS is most commonly suspended in the flow, but coarser fractions can saltate along the stream bed. These modes of transport can be separated into two components: bed load and the suspended load. FGS transported as part of the suspended load is generally ≤0.5mm (medium and fine sand, silt, and clay particles), whereas the bed load is most commonly 0.5mm to 2mm in diameter (coarse sand) (Knighton, 1998; Breugem, 2012).

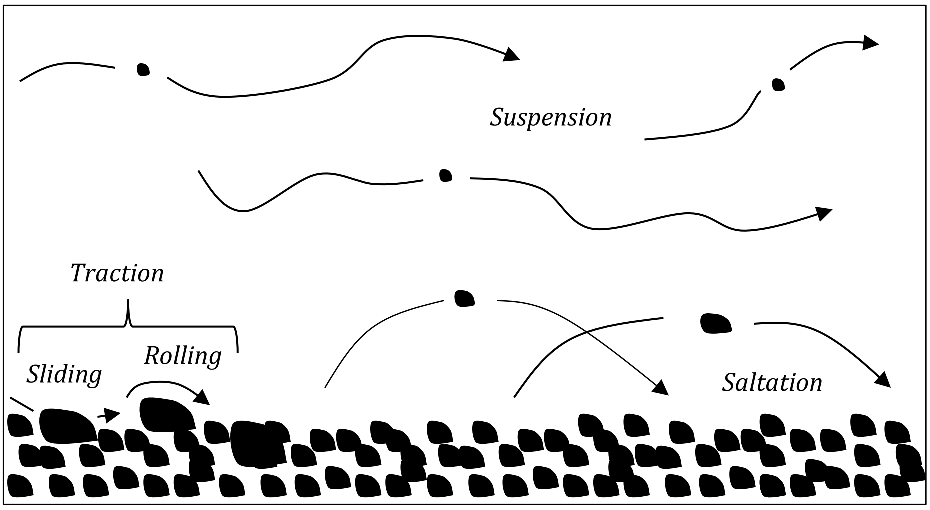


Figure 2.3. Modes of sediment transport in the river channel.

FGS stored on the channel bed is the primary source of sediment for bed load transport. Stream power determines the ability of the flow to transport the bed load (Equation 2.2), and as a result, transport rates of FGS transported as part of the bed load are often assumed to be capacity limited (Knighton, 1998). As discharge increases, the amount of sediment transported also increases. If flow increases enough, particles can be carried higher and further, and eventually enter suspension (Robert, 2003). If sediment supplied to the channel is less than the transport capacity of the stream, scouring of the bed is likely to occur (Bunte and Abt, 2001).

$$\Omega = \rho g Q S \qquad (2.2)$$

where Ω is stream power per unit channel length, Q is discharge ($m^3\ s^{-1}$) and S is channel slope.

The suspended load accounts for the largest component of sediment transport in most rivers (Reid and Frostick, 1994; Phillips and Walling, 1995; Skalak and Pizzuto, 2010). SS can be sourced from the wash load, or suspended bed material load. The wash load originates from slope run-off. It is generally already entrained in the flow when it is delivered to the river channel, and, as a result, is readily transported, even at low discharges (Richards, 1982). Once particles are being transported in suspension, it is thought that they travel at approximately the same speed as the flow velocity (Breugem, 2012) and rarely settle out of transport (Pritchard, 2006). SS transport can be modelled using a sediment rating curve, which describes the relationship between suspended sediment concentration (SS_c) and river discharge (Q) (Figure 2.4). Unlike bed load transport, the rate of transport of SS is determined by the rate of supply, rather than the transport capacity of the flow.

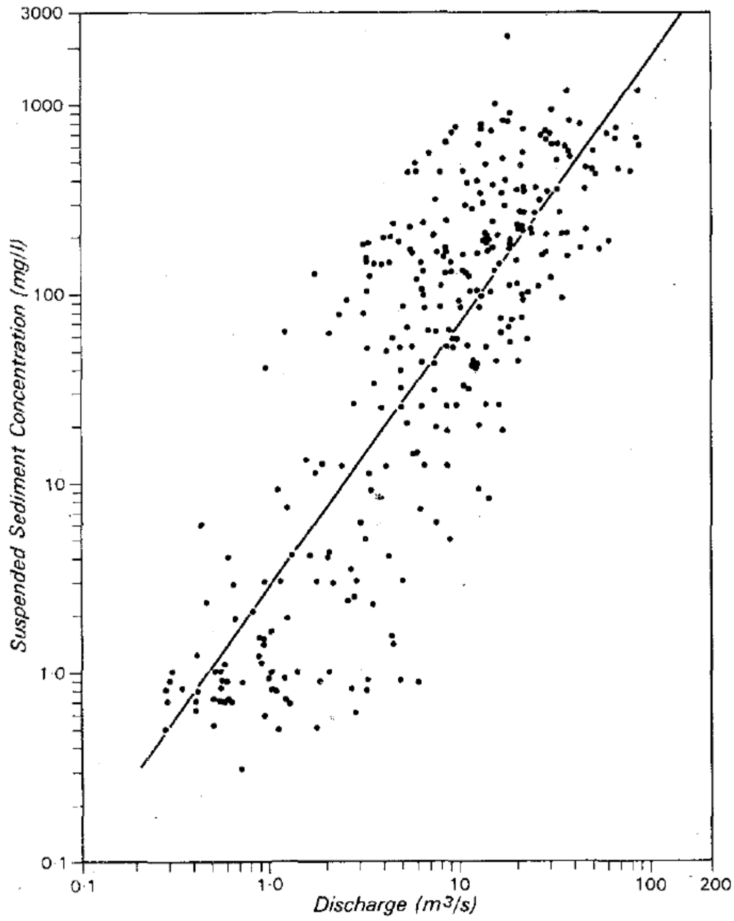


Figure 2.4. Sediment rating relationship, as derived from discharge and SS_c measurements. **Source:** Walling (1977: 534).

However, it has been suggested that SS does not behave as is traditionally thought, and instead of remaining entrained in the flow, is repeatedly deposited and resuspended (Parsons *et al.*, 2015), travelling in a series of hops, similar to saltating particles (Figure 2.5) (van Rijn, 1984; Graf and Altinakar, 1998). The distinction between saltating and suspended particles is therefore somewhat unclear (Nickling and McKenna Neuman, 2008), with differences between the two modes of transport thought to be related to the distance the particle travels before retouching the stream bed (van Rijn, 1984). As a result of the differences in distinguishing between suspension and saltation, it has been suggested that FGS transport should be treated, and modelled, as a continuum, rather than being split into separate processes (Parsons *et al.*, 2015).

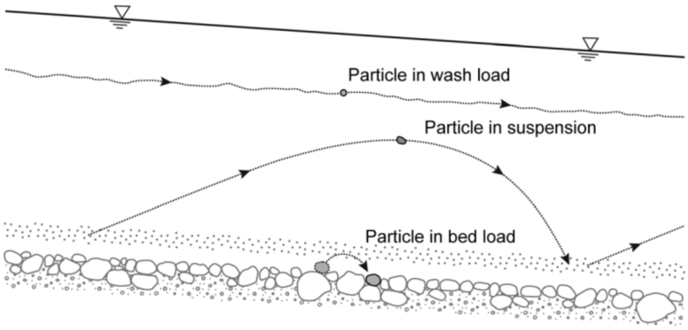


Figure 2.5. Modes of sediment transport according to Parsons *et al*. (2015).

2.3.3. Sediment deposition

Deposition of FGS occurs when the instantaneous vertical velocity of the flow falls below that of the fall velocity of a particle (Richards, 1982), with the fall velocity of a particle defined as the maximum velocity that a particle can attain when falling freely in a fluid. The fall velocity is determined by the balance between downward acting weight forces and upward acting fluid drag forces (Robert, 2003). Coarser particles have higher fall velocities, and will deposit at lower flow velocities than fine particles. As Figure 2.2 shows, clay particles are transported, even at very low flow velocities.

Stokes Law and the Impact Law define the fall velocities for sediment particles (Equation 2.3 and 2.4; Figure 2.6) (Kranck, 1980). The Reynolds number impacts the operation of Stokes Law. It is thought that as the Reynolds number increases, the Stokes equation becomes less appropriate for describing particle settling velocity. When the particle Reynolds number indicates that flow is laminar, Stokes Law is appropriate for estimating fall velocity. If flow is turbulent, the Impact Law is more appropriate. In general, silt-sized particles obey Stokes Law, whilst the settling velocity of sands and gravels can be described with the Impact Law (Allen, 1994).

Stokes Law: $\omega \approx 9000D^2$ (2.3)

Impact Law: $\omega = 33\sqrt{D}$ (2.4)

where ω is the fall velocity

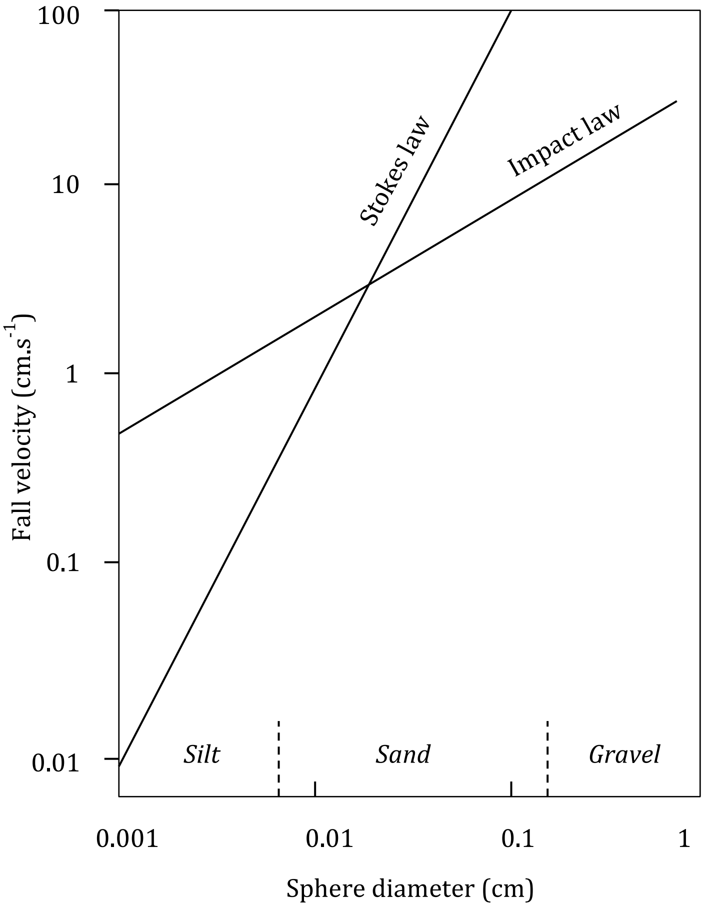


Figure 2.6. Fall velocities for quartz spheres at 20°C. Adapted from Richards (1982: 77).

Flow turbulence and the drag coefficient of the particle impact fall velocities, and influence whether a particle will deposit on the stream bed, or remain in suspension (Sear *et al.,* 2008). Secondary flows, generated by turbulence, can inhibit deposition processes and prevent settling occurring (Bai *et al.*, 2013). Quantitative estimates of deposition can be made based on flow characteristics (Droppo *et al.*, 2015). Furthermore, settling velocity equations assume that particles are spherical in shape. If particles are non-spherical – which is a common characteristic of individual grains (McAnally and Mehta, 2002) – the drag coefficient of the flow will increase, and the settling velocity of the particle will be reduced.

Flocculation and deflocculation can also result in the modification of the settling velocity of particles (Kranck, 1975), encouraging sediment deposition and impacting water quality (Droppo, 2001; McAnally and Mehta, 2002; Bilotta and Brazier, 2008). Flocculation ensues that particles that would otherwise remain in suspension and be transported through the system are deposited on the stream bed (Thompson and Wohl, 2009). Deposition dynamics cannot therefore always be directly related to grain size distributions (Kranck, 1975).

Sediment deposited on the stream bed can commonly be divided into two layers: a surface and an underlying layer. The surface layer represents an ephemeral deposit of FGS commonly 1 to 5mm thick which is readily available for resuspension (Lambert and Walling, 1988; Navratil *et al.*, 2010). Surficial deposits are often eroded in autumn and winter as macrophyte cover decreases and discharge increases above the threshold level for erosion (Sand-Jensen, 1998). The underlying layer is composed of framework and matrix material. The framework material is the coarse sediment stored on the streambed making the pores which the finer, matrix material infiltrates into. Matrix fines commonly originate from the suspended load (Frostick *et al.*, 1984). FGS can also deposit into the bed matrix with the bed load as the stream bed scours and fills during high discharges (Lisle, 1989). Forces acting within the water column, and those acting on the sediment bed, determine whether deposited fines will remain on the bed surface, infiltrate into the subsurface layer, or be remobilised into suspension (Sear *et al.,* 2008).

2.3.4. Sediment infiltration

Hyporheic exchanges result in surficial FGS being redistributed into pore spaces in the matrix material (Frostick *et al.*, 1984; Ren and Packman, 2007). FGS can infiltrate into the matrix material in two main ways: unimpeded static percolation and fine bridging (Figure 2.7). In unimpeded static percolation, deposited FGS falls between bed clasts and continues to do so until a physical barrier (normally the bedrock) prevents further infiltration. During fine bridging, interstitial deposits form a thin layer in shallow gravel pores, preventing FGS from infiltrating further (Gibson *et al.*, 2009; Huston and Fox, 2015).

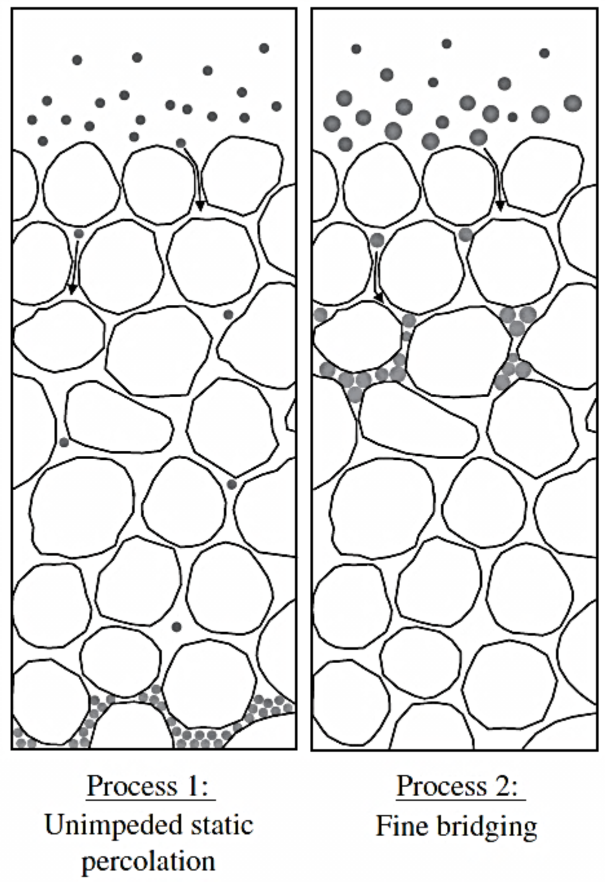


Figure 2.7. Schematic of the processes dominating the infiltration of FGS into a gravel framework. **Source:** Gibson *et al.* (2009: 662).

FGS is commonly reported to only infiltrate to depths of 1 to 10cm (Lambert and Walling, 1988; Collins and Walling, 2007b; Collins *et al.*, 2013). However, numerous studies investigating FGS deposition and storage have sampled to depths of greater than 10cm, suggesting FGS infiltrates to deeper than 10cm. Milan and Large (2014) and Acornley and Sear (1999) sampled to 15cm. Levasseur *et al.* (2006) and Petticrew *et al.* (2007) sampled to 20cm, and St-Hilaire *et al.* (2005) collected samples from the top 25cm of the stream bed.

Fine bridging is generally thought to dominate infiltration processes. However, numerous factors determine the depth and type of infiltration that will occur, including FGS size characteristics (Frostick *et al.*, 1984), framework material pore size (Sear *et al.,* 2008), the framework to matrix material diameter (Diplas and Parker, 1992; Huston and Fox, 2015), the shape and packing of the framework material (Lisle, 1989), the residence time of fines on the stream bed (Evans and Wilcox, 2014) and flow character (Frostick *et al.*, 1984). Probability of infiltration increases as grain size decreases. If FGS particles are smaller than the interstitial pore diameter, fines will fill the matrix from the bottom up. If particle and pore diameters are similar, fines will likely create an impermeable seal on the surface and prevent infiltration from occurring (Bunte and Abt, 2001). Sear *et al*. (2008) suggest that the pore size of the framework material is equivalent to approximately $0.4D_{50}$, where D_{50} is median particle size; particles smaller than $0.4D_{50}$ should, theoretically, pass freely into the matrix.

2.3.5. Sediment storage

As the SDR concept suggests, not all sediment delivered to a river channel will be transported out the system (Walling, 1983). FGS storage is an important component of sediment dynamics, representing the balance between inputs to and outputs from the river system, i.e. the sediment mass balance concept (Figure 2.8). If a river system is in equilibrium, inputs and outputs will balance, and storage will be constant. A change in storage represents a shift in river conditions; inputs or outputs have changed. The stream will adjust to a new steady state, and storage will equilibrate to a new level (Figure 2.9).

Figure 2.8. The sediment mass balance concept. Bed storage represents a balance between inputs to, and outputs from, a river system.

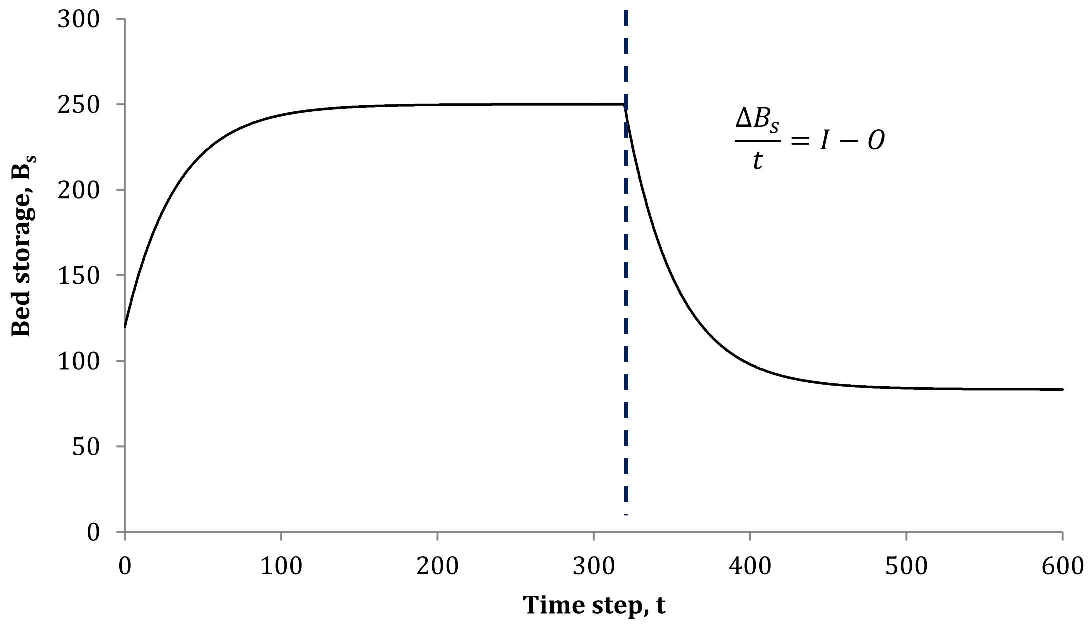


Figure 2.9. Theoretical change in bed storage (B_s) over time (t). Change in storage over time is equal to inputs to the system minus outputs from the system. B_s increases until a steady-state level is reached (B_s = 250). At t=320, inputs to the system decrease and B_s adjusts to a new steady-state level (B_s = 83), and anticipated ecological recovery.

Storage within and between channels is non-uniform (Table 2.2). Headwater streams are key contributors to storage. In a study by Marttila and Kløve (2014), 90% of FGS storage was in the headwater reaches of the study catchment. Quantifying storage in headwater streams is therefore important. Storage is also locally variable within catchments, and areas of preferential accumulation are found on the stream bed (Rathburn and Wohl, 2003). Duerdoth *et al.* (2015) describe FGS storage in river channels as patchy, attributing the spatial variation in storage to heterogeneous hydrological and sedimentological interactions. Low velocity, backwater areas are particularly prone to sediment accumulation and as a result, channel margins, areas downstream of obstructions and macrophyte stands are areas of preferential accumulation and storage of FGS (Wohl and Rathburn, 2003; Milan and Large, 2014). Presence of obstructions in the stream channel, e.g. debris dams and large woody debris (LWD), can create recirculation currents, which also encourage deposition and storage (Figure 2.10) (Bunte and Abt, 2001).

Table 2.2. Values of channel bed storage of FGS reported in the literature. Individual values represent mean average FGS storage.

Author	Definition of fines	River	Catchment area (km^2)	FGS Storage ($g\ m^{-2}\ cm^{-1}$)
Heppell *et al.* (2009)	<2mm	Frome	437	2320 – 13360
		Piddle	183	180 – 4700
Walling and Amos (1999)	<2mm	Upper Piddle	63.5	10 – 1500
Collins *et al.* (2005)	<63μm	Frome	437	184
		Piddle	183	316
		Upper Tern	230	478
		Pang	166	213
		Lambourn	234	251
Owens *et al.* (1999)	<150μm	Tweed	4390	56
Walling et al. (1998)	<150μm	Ouse	3315	34 – 1848 324
Lambert and Walling (1988)		Exe	1500	40
Walling *et al.* (2003c)	<63μm	Swale	1363	8 – 68
		Aire	1932	22 – 116
		Calder	930	21 – 290
Marttila and Kløve (2014)	*Not defined*	Sanginjoki, Finland	400	24 – 1340 305
Duerdoth *et al.* (2015)	<2mm	Various catchments in England		1 - 6000

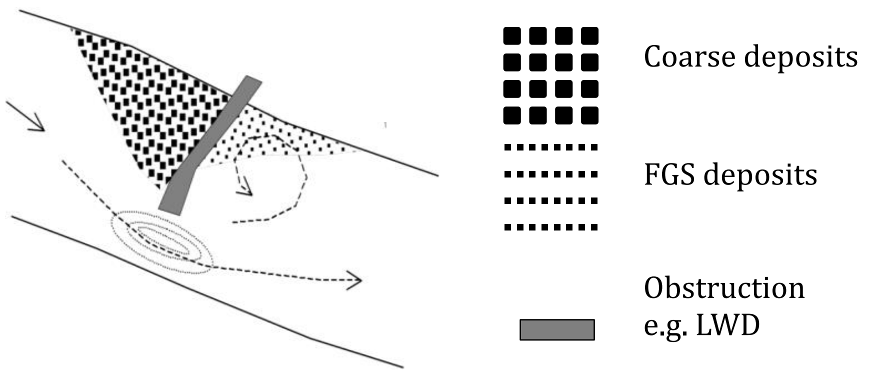


Figure 2.10. Presence of an obstruction in the stream channel has encouraged deposition of coarse sediment upstream of the blockage and FGS deposition in lateral recirculation eddies downstream of the obstruction. **Source:** Bunte and Abt (2001: 125)

The morphology of the stream bed also plays a key role in determining sediment storage. The adjustment between form and process in alluvial channels results in the adjustment of bed morphology, and the generation of bedforms (Clifford and Richards, 1992; Knighton, 1998). Montgomery and Buffington (1997: 597) classified the bed morphology of upland alluvial rivers into five distinct morphologies: cascade, step-pool, plane bed, riffle-pool and dune ripple. Riffle-pool sequences are also common in sinuous single-thread channels with low to moderate slopes (<2%) (Bunte and Abt, 2001). Riffles and pools are inundated, macroscale morphological units (Figure 2.11) (Carling and Orr, 2000; Bunte and Abt, 2001) representing large scale roughness elements on the stream bed which dissipate stream energy (Richards, 1982). They have distinct hydrological and sediment sorting characteristics (Robert, 2003), resulting in them representing distinct, ecological niches (Milan, 2013), and thus important habitats in stream beds.

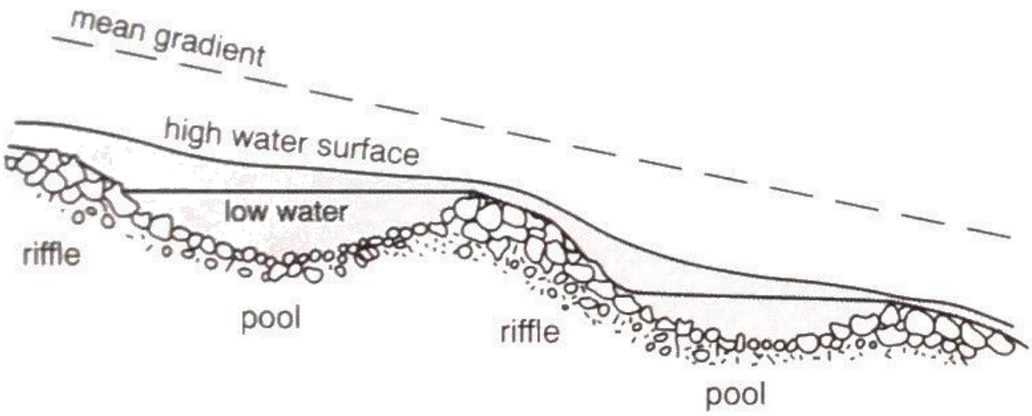


Figure 2.11. Longitudinal view of the riffle-pool sequence. **Source:** Knighton (1998: 194).

Riffles are topographic highs associated with shallow, fast, turbulent flows which winnow fines from the surface. The surface of riffles is typically coarse-grained, though sandy sediment can fill interstitial pore spaces (Robert, 2003). Clifford and Richards (1992) suggest that riffles can be thought of as giant cluster bedforms with structural elements within them, e.g. imbricated particles (Sear, 1996), making them generally more stable than pools (Bunte and Abt, 2001). Conversely, pools are topographic lows associated with deeper, slower flows and a gentler water surface slope; they have a near horizontal water surface during low discharges (Lisle and Hilton, 1992). Flow through pools is less turbulent, limiting the development of structural elements (Sear, 1996). The bed material of pools is typically finer than riffles (Robert, 2003). The difference in grain size characteristics of riffles and pools is evidenced by pools having a lower roughness coefficient than riffles; Manning's n values are thought to be approximately 0.05 in typical lowland pools and 0.12 in riffles (Richards, 1978).

The contrasting sediment transport and deposition dynamics over riffles and pools results in preferential pool filling and riffle scour, during normal flows (Robert, 2003). The fines winnowed from the surface of riffles are likely to be deposited in pools, as the flow velocity in pools is no longer high enough to sustain transport (Richards, 1982). As a result, pools represent areas of preferential accumulation with high concentrations of mobile sediment (Lisle and Hilton, 1992). When quantifying FGS storage, it could be expected that FGS storage in pools will exceed that in riffles. It could also be expected that, over time, pool filling would result in the structure of riffle-pool sequences being destroyed.

Keller's velocity reversal hypothesis is a widely accepted mechanism explaining the maintenance of riffle-pool sequences (Figure 2.12) (Clifford and Richards, 1992; Robert, 2003; Harrison and Keller, 2007). During flood flows, the rate of change of velocity in pools exceeds that in riffles, such that the velocity of water flowing through pools exceeds the velocity of water flowing over riffles. Fines have high transport velocities, therefore are generally rapidly flushed from the system; sediment is scoured out of the pool, transported, and deposited on the riffle. During waning flows, fines are selectively transported. Transport will continue in areas of high shear stress (e.g. riffles) and deposition will occur in areas of low boundary shear stress (Lisle and Hilton, 1992). The velocity reversal hypothesis makes several assumptions, including that the reversal occurs for all cross-sections in the riffle-pool unit. Although near-bed velocity at the pool midpoint is found to be higher than at the riffle crest during reversal discharges, this is not enough evidence for reversals occurring across the entire cross-section of the channel (Clifford and Richards, 1992). An alternative method which has been proposed for riffle-pool maintenance is the differential sediment entrainment hypothesis

(DSEH). Pools have loose packing in comparison to riffle sediment, and thus particles in pools have lower pivoting angles and greater exposure to transport than riffle sediments. Entrainment thresholds for transport in riffles are high. It has been suggested that these sedimentological contrasts are enough to maintain the morphology of riffles and pools and allow scour and degradation to occur, even during low flows; a velocity reversal is not required (Robert, 2003; Hodge *et al.*, 2013).

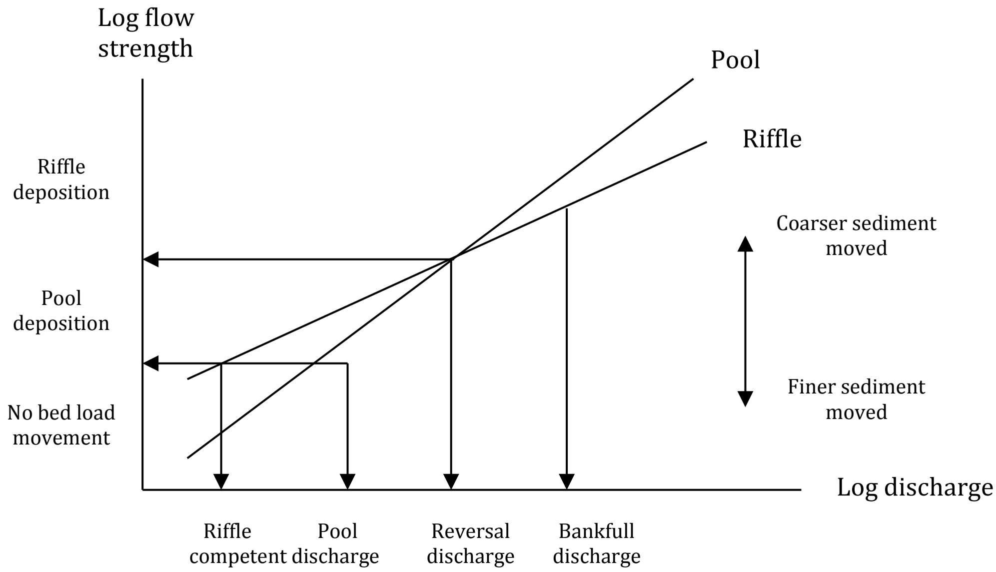


Figure 2.12. Schematic representation of flow-sediment interactions in riffles and pools. At reversal discharge, flow strength in pools exceeds that in riffles, and scour in pools occurs. **Source:** Clifford and Richards (1992: 45)

As a result of riffles and pools being readily identifiable geomorphic features, they are ideal sampling sites because consistent, reproducible sampling is possible between sites (Schuett-Hames *et al.,* 1996). Collecting samples from riffles and pools also helps to minimise inter-habitat differences in FGS storage (Buendia *et al.*, 2013a), which could arise if samples were collected from random locations within the channel. Although sediment largely influences aquatic flora and fauna, stream organisms can also influence sediment retention dynamics (Jones *et al.*, 2014). Repeat monitoring of pools can provide information on temporal changes in the quantity of FGS transported because, whilst FGS may not always be visible on riffles, it would be expected in pools (Bunte and Abt, 2001). Similarly, sampling riffles can provide an indication of whether sediment problems exist in the river system; even in catchments unimpacted by land use change, pools are likely to act as stores of fines (Swanston, 1991; Kaller and Hartman, 2004). Riffles and pools also represent important habitats in stream bed environments due to their physical diversity (Schuett-Hames *et al*., 1996; Robert, 2003). Riffles are particularly important habitats, offering high production of benthic invertebrates and periphyton (Graham, 1990). Knowledge of the FGS content of riffles and pools is therefore important for determining the overall ecological quality of the stream bed. As a result of the above factors, sites with riffle-pool sequences were selected for sampling.

2.4 Impacts of fine-grained sediment on river systems

FGS is an intrinsic part of fluvial systems and without it, river systems will not be able to function (Owens *et al.*, 2005; Turley *et al.*, 2014). Increases in FGS can therefore be beneficial to river ecology. Notwithstanding the importance of FGS in lotic ecosystems, it is the case that, in many river systems, threshold levels of FGS are exceeded. Enhanced FGS can have detrimental and deleterious physical and chemical impacts on river systems, the impacts of which are both well understood and well documented (Wood and Armitage, 1997; Petticrew *et al.*, 2007; Acornley and Sear, 1999; Navratil *et al.*, 2010; Descloux *et al.*, 2013; Von Bertrab *et al.*, 2013). FGS is often viewed as a diffuse source pollutant in freshwater systems (Walling *et al.,* 2007; Minella *et al.*, 2008). This section will explore the impacts of increased FGS loads on lotic environments.

Due to the spatial and temporal variability of FGS dynamics in streams, the impacts of FGS on river systems are not only numerous, but also spatially and temporally variable. Impacts are context dependent (Jones *et al*., 2014) and catchment characteristics, including relief, soil type and climate, are important factors influencing the extent of the impacts of FGS on river systems. Sediment size and quality are also important considerations for determining the impact of excess FGS loads (Cooper *et al.,* 2008). Recovery from the effects of FGS can occur naturally, with the speed of recovery being controlled by the nature of the impact, as well as the survival rate of the impacted species (Wood and Armitage, 1997).

2.4.1 Benefits of fine-grained sediment

Increased FGS concentrations in stream channels can result in an increase in the abundance of some aquatic taxa (Logan, 2007). Although species with unfavoured life-history traits (e.g. gill respiration) are excluded as FGS levels increase, organisms with favoured life-history traits (e.g. tegumental respiration) benefit and can experience an increase in abundance (Larsen *et al.*, 2011). An increase in the OM content of streams – which is common with increasing levels of FGS in agricultural catchments (Greig *et al.*, 2005) – can also benefit some species (Jackson *et al.*, 2007; Jones *et al.*, 2012b); increases in the OM content of sediment is commonly coupled with an increase in invertebrate communities, as the availability of food sources increase (Jones *et al.*, 2012a).

The hyporheic zone is a potential source for nitrates and soluble reactive phosphates. Hyporheic zone interactions can therefore benefit stream productivity, and increase diatom growth (Maazouzi *et al*., 2013; Jones *et al*., 2014). Furthermore, increased sediment loadings can result in increased presence of macrophyte stands, which can be beneficial for reducing nutrient concentrations as macrophyte stands act as temporary nutrient sinks during periods of growth (Clarke and Wharton, 2001). The increase in habitat availability for macrophyte-dwelling invertebrate can result in a subsequent increase in the population of these species (Jones *et al.*, 2012), with population increases further aided by a decline in predation associated with increased egg mortality and subsequent reduced fish species populations (Jones *et al.*, 2012).

2.4.2. Detrimental biological and ecological impacts of fine-grained sediment

Excess FGS is the cause of physiochemical, biological, and ecological impairment in aquatic ecosystems (Figure 2.13). The detrimental impacts of FGS can be grouped into several categories.

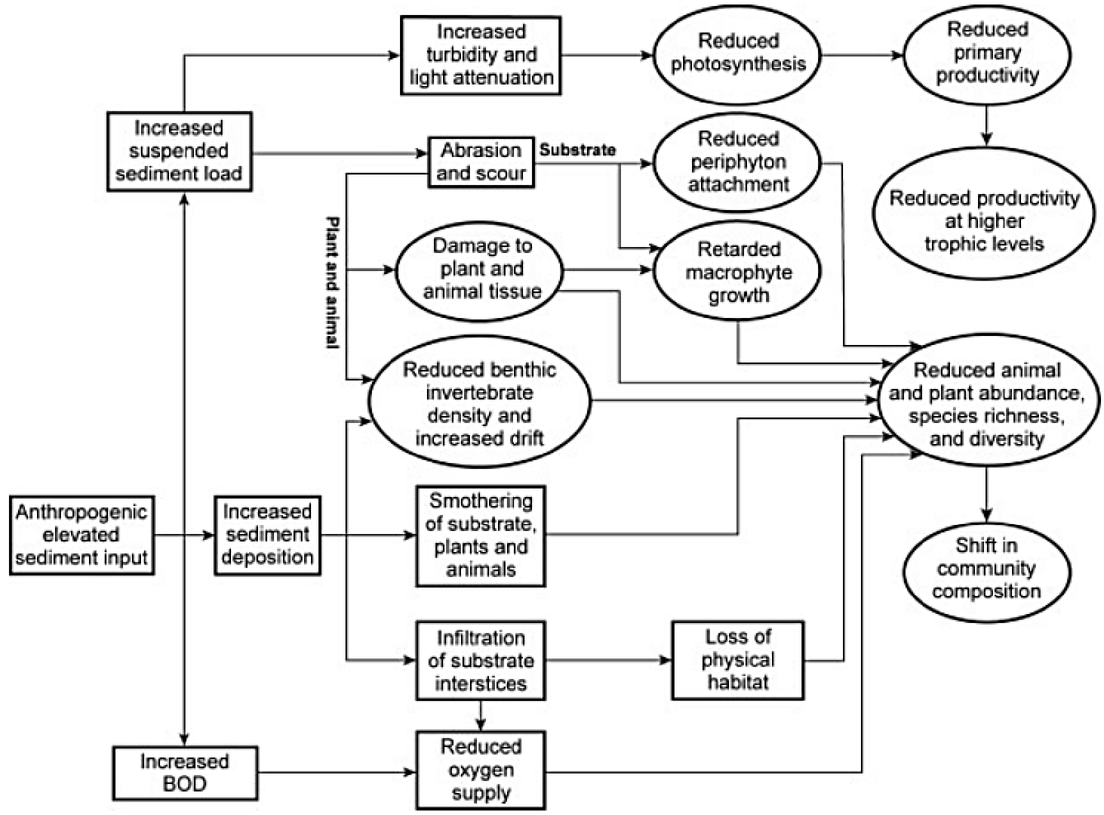


Figure 2.13. The negative effects of increased sediment input to aquatic ecosystems caused by anthropogenic activity. Rectangles represent the physiochemical effects on the ecosystem, and ovals the biological and ecological ecosystem response. **Source:** Kemp *et al.* (2011: 1801).

1) ***Burial:*** Many creatures are adapted to live in specific habitat conditions, including those which are prone to FGS deposition (Bryce *et al.*, 2010). The ability to excavate from deposited sediments is one such example of adaptation. However, if sediment accretion rates exceed excavation rates, burial and physical entrapment will result, increasing mortality rates as individuals are unable to access food and oxygen (Jones *et al.*, 2012b). Sediment accumulation could reach the point beyond which sediment-sensitive assemblages are no longer sustainable (Bryce *et al.*, 2010). Burial can also inhibit alevins from emerging after hatching, reducing survival rates of spawning species (Kondolf, 2000). Similarly, coarse sands deposited on the surface of the stream bed can form a surface seal which can also prevent alevins from emerging (Acornley and Sear, 1999).

2) ***Clogging:*** FGS in suspension can impact aquatic species. If the SS_c is high, fish gills can be clogged (Wood and Armitage, 1997). Silt and clay particles can have particularly detrimental impacts on organisms, and although creatures are adapted to deal with clogging, if the SS_c is especially high, energy expended removing particles clogging gills can exceed energy obtained from feeding. If loads are especially high, feeding can cease (Jones *et al.*, 2012b). Clogging can also reduce the permeability of the bed substrate, impacting dissolved oxygen (DO) concentrations (Rowe *et al.,* 2003).

3) ***Abrasion and scour:*** Particles suspended in the flow can abrade invertebrates, the effects becoming more pronounced as the SS_c increases. Coarse sand moving as part of the bed load can also abrade aquatic fauna on the streambed (Jones *et al.*, 2012b), and dislodge and damage diatoms attached to the bed substrate (Jones *et al.*, 2014). Damage to macrophyte stands can occur as FGS particles abrade their leaves and stems (Wood and Armitage, 1997).

4) ***Oxygen concentration:*** Deposition and infiltration of FGS into the streambed can modify interstitial flows and reduce oxygen percolation (Jones *et al.*, 2012b). This can be particularly detrimental for spawning species as reduced oxygen concentrations in spawning nests (redds) leads to reduced egg survival (Soulsby *et al.*, 2001). Oxygen flow into redds can also be reduced if clay particles are deposited post-redd creation, with flocculation resulting in the formation of a sediment seal around the incubating embryos (Greig *et al*., 2005).

   Reductions in oxygen concentrations can also be attributed to the deposition of OM. Although OM is generally less dense than water and would thus be expected to float, flocculation with FGS can cause OM to settle and deposit onto the stream bed (Petticrew and Arocena, 2003; Owens *et al*., 2005), influencing the sediment oxygen demand (SOD) of the stored FGS. Microbial activity to remove organic fine sediments also affects the DO concentration of a water body. Microbial activity increases biochemical oxygen demand (BOD) and reduces the oxygen supply available to invertebrates (Petticrew *et al.*, 2007; Jones *et al.*, 2012b). Animal carcasses, e.g. of fish, are also a source of OM to river systems as they decay (Helfield and Naiman, 2001). In a study by Wold and Hershey (1999), microbial activity, periphyton biomass and nitrate concentrations were found to correlate with carcass decomposition. Enhanced OM concentrations can enhance bacterial and algal growth, which in turn affects sediment storage and retention (Barko and Smart, 1983).

5) ***Substrate composition:*** The average grain size of the bed material substrate can be reduced as FGS infiltrates into the framework material. Invertebrates have specific requirements for bed substrate composition and will avoid patches that fail to meet requirements for establishment. Community composition and distribution can thus be modified. Filling of interstices with FGS can also prevent invertebrates moving between pore spaces and habitat patches, further modifying invertebrate distributions (Jones *et al.*, 2012b).

6) ***Habitat quantity and quality:*** As substrate composition changes, a change in habitat quantity and quality will occur. Gibson *et al.* (2009) note the importance of habitat quality, suggesting that invertebrates and macroinvertebrates are dependent on the presence of habitable substrate conditions in order to establish. High FGS loads generally result in a reduction in habitat availability for macroinvetebrate communities (Richards and Bacon, 1994), leading to species drift and decreased species diversity (Larsen and Ormerod, 2010).

Habitat modification due to FGS infiltration is of particular importance for spawning species, and FGS infiltration has been suggested as the primary mechanism for spawning gravel obstruction. Siltation at spawning sites is particularly problematic (Walling, 2006) because, whilst the bed material requirements for spawning species vary with life stage (e.g. spawning, incubation, emergence), the presence of FGS in interstices reduces habitat quantity and quality for spawning species (Lisle, 1989; Kondolf, 2000). Furthermore, spawning is common in the shallow zone (8 – 30cm) of rivers, which are particularly sensitive to changes in FGS loads (Milan *et al.*, 2000). A reduction in habitat quantity and quality can also modify the migration patterns of spawning fish, and increase species mortality (McNeil and Ahnell, 1964; Wood and Armitage, 1997).

Presence of macrophyte stands on the stream bed can influence sediment dynamics and modify habitat quality and quantity. Sparse stands have a negligible impact on sediment deposition and remobilization. Dense stands can however reduce water velocity, encouraging sediment deposition and a reduction in habitat quality of quantity (Jones *et al.*, 2012a). In-stream vegetation cover varies seasonally, thus the quantity of sediment retained by them, and the potential implications on habitat quality and quantity, also vary temporally (Cotton *et al.*, 2006).

7) ***Food quantity and quality:*** Deposited and suspended sediments can modify food web interactions. Fine sediment in the water column increases turbidity, limiting light penetration, and reducing primary productivity (Wood and Armitage, 1997). In extreme cases, light penetration can be reduced so much that macrophytes cannot establish (Jones *et al.*, 2012a). Modification of primary productivity will impact food chains. Maazouzi *et al.* (2013) note the importance of FGS deposition on primary productivity, suggesting that if hyporheic interactions result in the stream bed acting as a source of nutrients, productivity and nutrient cycling will be limited. As well as turbidity limiting primary productivity, it can also negatively impact upon visual feeders (Wood and Armitage, 1997; Jones *et al.*, 2012a), modifying predator-prey interactions and ultimately the food web.

The nutritional quality of periphyton can also be reduced by deposition of FGS as the proportion of inorganic sediment on the stream bed surface increases. Jones *et al.* (2012a) suggest that this will be particularly detrimental for scraping invertebrates, e.g. snails. Periphyton can also encourage FGS retention as they act as a 'sticky' surface for siltation (Graham, 1990). This can reduce food availability as the periphyton is covered, and unavailable to predators. It is important to note that the impact of FGS deposition on reducing OM available to organisms is dependent on the ratio of periphyton growth to siltation; if siltation rates exceed growth rates, the river system will be susceptible to reductions in the proportion of OM. Furthermore, reduced egg survival attributed to FGS deposition and reduced oxygenation can reduce populations of spawning creatures, further modifying predator-prey and food web interactions (Wood and Armitage, 1997).

2.4.3 Geomorphological and hydrological impacts of fine-grained sediment

Modification of FGS loads in streams can also cause geomorphological and hydrological changes (Tena *et al.*, 2012). As Owens *et al.* (2005) suggest, increased FGS supply can reduce channel capacity, increase sedimentation on floodplains, and modify channel morphology and river behaviour. Changes in channel capacity have important implications for flood risk. Sidorchuk and Golosov (2003) observed a reduction in channel length and stream order on the Russian Plain. Sedimentation of stream channels was associated with conversion to cultivated land, and caused upper reaches to aggrade. Watercourses are sensitive to changes in land use and subsequent changes in sediment supply.

2.4.4 Fine-grained sediment as a pollutant vector

FGS can be seen as a pollutant vector responsible for the transfer of nutrients and contaminants (Russell *et al.*, 2001; Walling, 2005; Walling *et al.,* 2007; Fox *et al.*, 2010) making FGS not only ecologically important, but also environmentally important. That is, FGS is important in terms of both aquatic biota and contaminant concentrations. Increasing nutrient concentrations can lead to eutrophication (Owens *et al.*, 2005). Furthermore, sediment <63μm often bonds with toxic substances (Fox *et al.*, 2010), which can impact diatom assemblages (Jones *et al.*, 2014), as well as potentially exceed water quality guideline concentrations. Deposition of contaminants in the stream channel and on floodplains can create a "chemical time bomb" (Owens *et al*., 2005: 699), which can be reactivated and transferred downstream as sediment stored on the stream bed is remobilised (Walling *et al*., 2006). The low gradient of lowland watersheds acts to promote sediment storage, making these environments particularly vulnerable to high contaminant concentrations which often exceed concentration guidelines (Owens *et al.*, 2005).

2.5 Management issues

As a result of the well-documented environmental impacts of anthropogenic activity on water quality, various obligations – subnational, national and international (Collins *et al*., 2011) – have been developed to drive management of water quality issues (Collins and McGonigle, 2008). This section will explore water quality legislation, as well as approaches to catchment management and quantifying FGS in river systems.

2.5.1 Water Quality Legislation

Numerous directives have been put in place for managing water quality, with focus over time shifting from public health protection and environmental protection to sustainability and integrated management (Kallis and Butler, 2001). EU Directives include the Shellfish Directive (79/923/EEC), Nitrates Directive (91/676/EEC), and the Freshwater Fish Directive (2006/44/EEC) (Collins and McGonigle, 2008). As Bennion and Battarbee (2007) suggest, in order for management to be effective, it should be integrated. The WFD (2000/60/EC), adopted in October 2000, represents a novel, integrated approach to water policy (Petersen *et al.*, 2009), and is the current over-arching water quality policy for EU Member states (Collins *et al.*, 2011).

The WFD aims to prevent further deviation in water quality, as well as provide protection for water bodies and encourage sustainable usage of them (Foster *et al.*, 2011). A further aim is to enhance the status of SWBs so that they attain Good Ecological Status (GES) by 2027 (Hering *et al.*, 2010). SWBs are assigned a quality status based on the difference from reference conditions, with reference conditions in Europe representing a water system with minimal anthropogenic impairment (Hübener *et al.*, 2015). If the water quality meets the undisturbed reference conditions – as determined through the Ecological Quality Ratio – a high status is granted (Hatton-Ellis, 2008). A good status represents low deviation from reference conditions and other status classifications (moderate, poor and bad) represent increasing degrees of deviation from reference conditions (Hübener *et al.*, 2015). Classification is based on numerous factors, including biological, chemical and hydromorphological elements (Figure 2.14) with the lowest classed classification for an individual element designating the overall surface water status.

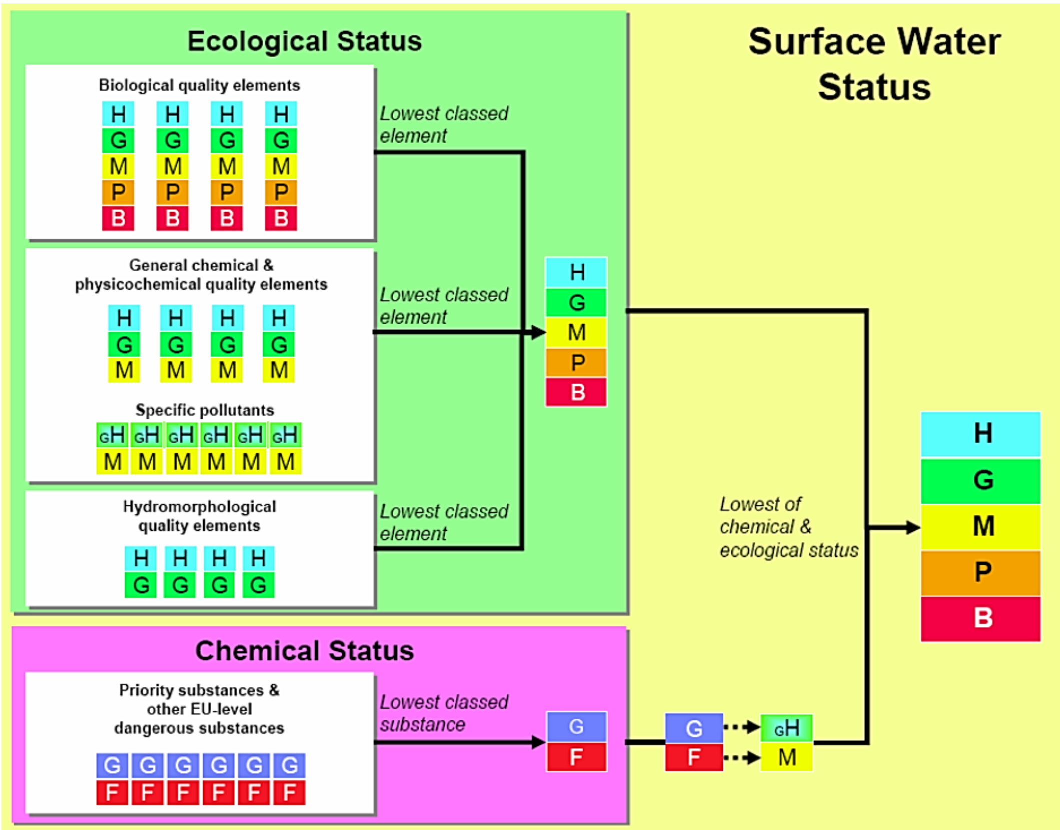


Figure 2.14. Schematic representation of the surface water status classification. Surface waters are classified with the lowest status of the individual elements. Key: H = high; G = good; gH = good or better (normally treated as high for calculating); M = moderate; P = poor; B = bad; F = failing to achieve good chemical status. **Source:** Northern Ireland Environment Agency (NIEA) (2009: 3).

Use of reference conditions for status classification has however been criticised. Determining reference conditions (i.e. separating anthropogenic activity from natural variability), and the accuracy and precision of sampling strategies are of particular concern (Bouleau and Pont, 2015; Hübener *et al.*, 2015; Skeffington *et al.*, 2015). Bouleau (2008) criticises the WFD, suggesting that achieving GES is merely an overambitious ecological dream. Lassaletta *et al*. (2010) also note issues with the definition of SWBs since headwater streams, due to their small size, are often not classified as SWBs, despite their vulnerability to anthropogenic activity, importance in FGS storage, and influence on downstream water quality.

River Basin Management Plans are a requirement for the WFD and have been developed to describe water quality conditions, to identify targets for districts, and outline methods to implement management (Bennion and Battarbee, 2007; Collins and McGonigle, 2008). Focus is placed on priority substances and specific pollutants, with the hope that improvements to water quality will result if these are managed (DEFRA, 2014). Use of river basins as a management scale represents a unique approach to management since management is traditionally based on political and administrative, rather than hydrological boundaries (Cabezas, 2012; Hüesker and Moss, 2015).

2.5.2 Sediment targets

The WFD identifies annual average and maximum allowable concentrations for priority substances and specific pollutants. It is thought that reducing the concentrations of these substances will help reduce water pollution and enhance the status of SWBs, helping achieve the aim of GES by 2027. However, FGS is neither a priority substance nor a specific pollutant, therefore recommendations for environmental standards have not been fully developed (DEFRA, 2014). The importance of FGS on water quality is well recognised, and as Cooper *et al*. (2008) highlight, SS is a pollutant for which critical values need defining if requirements of the WFD are to be met. The successful implementation of the WFD requires management of identified water quality pressures (Collins and Anthony, 2008). The Freshwater Fish Directive, which was repealed in 2013, had a guideline mean annual average SS_c of $25mg.l^{-1}$. This target has been adopted to represent GES for SS in river systems (Collins and Anthony, 2008). Numerous complex factors determine the influence of SS on water quality. Determining the concentration of fines which are detrimental to ecological quality is also difficult to achieve because numerous factors influence ecological quality, not just sediment (Milan *et al.*, 2000). SS_{cs} of 4 – 330,000 mg l^{-1} have been reported as having a negative effect for aquatic species (Collins *et al.*, 2011). Use of a single SS_c target is therefore debated. As a result, the UK Technical Advisory Group (UKTAG) – set up to provide advice on the EU's WFD – has recognised that the 25 mg l^{-1} mean annual average SS_c is not sufficient and that new standards need developing. The need to include standards for deposited sediment, as well as to develop a standard assessment procedure for measuring suspended and deposited sediment has also been recognised (APEM, 2007; Von Bertrab *et al.*, 2013). Although sediment quality guidelines have been developed in some countries (Owens *et al.*, 2005), there is a general absence of sediment targets for European countries. As a result, much potential exists for developing sediment targets (Walling *et al.,* 2007).

In recognition of the complexity of the relationship between sediment and water quality, alternative targets for FGS have been proposed (Bilotta and Brazier, 2008; Jones *et al.*, 2012b). These include the use of water column metrics (e.g. turbidity and light penetration), substrate metrics (e.g. embededness and riffle stability) and substrate flow parameters (e.g. DO concentration) (Rowe *et al*., 2003; Collins *et al*., 2011). Collins *et al*. (2011) propose a target suspended sediment yield of 40 t km^{-2} yr^{-1} for lowland impermeable soils. Rowe *et al.* (2003) suggest that the proportion of sediment ≤0.85mm stored on the stream bed should not exceed 10%.

Importantly, in order to avoid problems associated with single target values, targets should be catchment specific. General targets may be unrealistic for a specific catchment; quantifying the modern background sediment delivery to rivers could provide a method for defining the "maximum ceiling of sediment reduction" (Collins *et al.,* 2012: 128). Rowe *et al*. (2003) suggest the use of site-specific targets, noting that, if the required data is available, nothing precludes their development. Use of the WFD's river typology classification could help determine sediment targets for rivers based on their catchment area, dominant geology and altitude. Scope exists for improving typology classifications through including factors relating to anthropogenic activity (Cooper *et al.,* 2008). Targets should also be modified as new information becomes available.

2.5.3 Approaches to management

Regardless of the sediment target chosen, in order to reduce the impacts of FGS in streams and improve the status of SWBs, management of FGS is required. The realisation that management should be targeted and appropriate to the problem in hand has led to the development of best-management practices (Wood and Armitage, 1999). However, as Soulsby *et al*. (2001) suggest, deciding FGS management approaches is difficult because the problems associated with fines are numerous. Approaches to management are largely in-channel or land management based. Traditionally, river channel management projects focused on hard engineering approaches which created static, stable stream banks. Static banks are, however, unusual, with active banks helping to maintain the natural functioning of river systems (Florsheim *et al.*, 2008). Focus is now placed on ensuring management is sustainable (Nakamura *et al.*, 2006).

Soulsby *et al*. (2001) stress the importance of managing whole catchments rather than individual reaches. Catchment land management practices are therefore commonly used to improve water quality, with their aim being to mitigate the problem at the source and reduce FGS supply to river catchments. Catchment land management strategies can effectively disconnect sediment sources from transport pathways, encouraging sediment attenuation on the land, rather than in the river channel (Perks *et al.*, 2015). This can have the effect of creating "hungry water" (see section 2.3.1) and initiating a decrease in in-channel sediment storage.

Further support for catchment-based management is provided by the sediment mass balance concept (Figure 2.9; Figure 2.10). If agricultural practices are the source of inputs to a river catchment, and improving catchment land management practices reduces the supply of FGS to the river channels, a decrease in inputs will occur. If outputs exceed inputs, storage will decrease. Changing land use has the potential to reduce the flux of sediment to watercourses, thereby facilitating a process of passive stream restoration whereby water quality and riverine ecology gradually improve as FGS stored within channel systems is evacuated downstream by sediment transport. Minella *et al*. (2009) highlight this in their study of sediment yields in southern Brazil, and suggested that if inputs to a system are reduced, the contribution from autochthonous sources to sediment yield is likely to increase, and thus in-channel storage decreases.

Examples of catchment-based management include delaying wheelings in fields sown with winter crops, cover cropping, installing buffer strips and use of natural and artificial wetlands to trap sediment (Collins and Davison, 2009; Newman *et al.,* 2015). Countryside stewardship programmes which engage stakeholders have been developed to encourage the adoption of these best management practices (Kleinman *et al.*, 2015), e.g. Catchment Sensitive Farming (CSF). The CSF project, launched in 2006, is the primary agri-environment scheme used to target agriculture-caused diffuse pollution in the UK. Developed to help catchments at risk of non-compliance with the WFD (Glavan *et al.*, 2012), it assists stakeholders with tackling the cause of diffuse pollution. It initially targeted 40 priority catchments, although the scheme was expanded to include a further 28 areas (Collins *et al.*, 2013). The CSF aims to engage stakeholders in the priority catchments, offer free training and advice, and provide grants for installation of infrastructure which will improve environmental performance (Natural England, 2015). Education and engagement of stakeholders is often considered necessary for successful catchment management (Collins and McGonigle, 2008).

Although the implementation of catchment land management strategies is primarily to reduce the flux of sediment to watercourses and improve water quality status, they can also be beneficial for agricultural productivity, e.g. by reducing top soil losses and soil fertility (Collins *et al.*, 2007). As Collins and McGonigle (2008) suggest, "win-win" management is beneficial. However, in order to ensure the success of management strategies, it is important to ensure that catchment scale factors, including land-use and geology, and ecological information is considered when devising catchment management. Without such consideration, management and restoration schemes are not likely to be optimal or sustainable (Wood and Armitage, 1999; Milan *et al.*, 2000). Management should also be targeted to the problem in the catchment in order to ensure that it is appropriate, and ecologically feasible (Deasy *et al.*, 2009).

2.5.4 Catchment management evaluation

Catchment land management initiatives are widespread. It is important to monitor changes in water quality to determine the success, or otherwise, of such schemes. Monitoring is important from both ecological and economic perspectives. Approaches to managing geomorphological problems also benefit from reliable and efficient monitoring, and subsequent quantification and modelling of in-stream parameters (Wilcock, 2001). In order to fully monitor the effectiveness of mitigation measures and assess the status and water quality of aquatic habitats, knowledge of sediment quantity and quality is imperative (Bronsdon and Naden, 2000).

Monitoring at the catchment scale has occurred under the WFD in order to classify the status of the water body (EC, 2015). The COMMPS project helped identify priority substances which require monitoring for status classification (Brils, 2005). Several research projects conducted in catchments which are representative of many areas of lowland England, e.g. the Demonstration Test Catchments project and the European Standardisation of River Classifications project (STAR), have also identified guidelines, suggested approaches for monitoring FGS at the catchment scale, and assessed the cost-effectiveness of mitigation measures (Johnson *et al.*, 2007; Harris, 2015).

Despite the recognised importance of monitoring FGS, the effectiveness of catchment land management strategies at reducing FGS supply and improving ecological quality is rarely evaluated. Furthermore, it is generally difficult to evaluate the success, or otherwise, of land management initiatives. The Water Friendly Farming (WFF) project, launched in 2012 by the Game and Wildlife Conservation Trust (GWCT), has been designed to assess the effectiveness of catchment-based mitigation measures that are being inputted to reduce the impact of rural land use of SWBs (Biggs *et al.*, 2014). The WFF project also aims to demonstrate the management approaches that are successful in reducing diffuse source pollution and improving water quality (GWCT, 2012). Land use in Stonton Brook and the Eye Brook – tributaries of the Welland River – is being actively managed with interceptor wetlands, farmyard measures, grass buffer strips, and streamside fencing. Additional habitat protection and creation measures are also being used in Stonton Brook to improve freshwater biodiversity. Barkby Brook, located in the Soar Basin, is being used as a control catchment (Biggs *et al*., 2014).

2.5.5 Quantifying fine-grained sediment in river channels

There are numerous techniques available for quantifying FGS (Kondolf, 2000), with methodologies well-documented in literature. Approaches include estimating sediment yield, quantifying suspension, deposition and storage, and using proxy measures (McNeil and Ahnell, 1964; Lambert and Walling, 1988; Acornley and Sear, 1999; Kondolf, 2000; Wilcock, 2001; Clarke *et al.,* 2003; Extence *et al.*, 2013; Heng and Suetsugi, 2014; Turley *et al.*, 2014; Duerdoth *et al.*, 2015). Sediment in suspension has important ecological impacts, therefore a focus has been placed on monitoring it. It is also thought that quantities of SS are correlated to sediment storage on the stream bed (Acornley and Sear, 1999). SS_c measurements currently dominate FGS monitoring (Jones *et al.*, 2012b). This focus has been supported by the existence of the 25 mg l^{-1} annual mean SS_c guideline (Collins and Anthony, 2008). Extensive sampling networks are required to account for spatial variability in SS_c. Achieving a representative average SS_c is therefore both time and labour intensive (Clarke and Scruton, 1997). In recognition of the limitations of measuring SS_c, measurement of suspended sediment yield is an increasingly common approach to monitoring FGS in streams. As the ultimate aim of catchment management initiatives is to reduce downstream sediment fluxes, monitoring at the catchment outlet is potentially advantageous (Minella *et al.*, 2008). As with SS_c, sediment yields vary temporally, therefore high frequency, long-term monitoring is necessary to accurately predict sediment yields (Walling, 1983).

Although SS has ecological implications, invertebrates are also severely impacted by the deposition of FGS. Reducing the SS_c, turbidity and light penetration will not improve conditions for invertebrates if FGS deposition is still occurring and growth is being limited by stream bed conditions. It has also been suggested that suspended fines are commonly transported out the system and not stored

(Pizzuto, 2014); measuring SS_c will provide little information on the overall habitat quality. Furthermore, catchment scale monitoring is complex, with sediment yield at the catchment outlet being impacted by numerous factors, including in-channel sources (Minella *et al.*, 2009); when using suspended sediment yields as a measure of management success, reductions in these due to successful land use management may be masked by mobilisation of, and therefore a subsequent reduction in, stored sediment. Deposited sediment is also a key component of the sediment budget and is important for contaminant studies (Owens *et al.*, 2001); knowing the quantity of sediment on the stream bed is necessary for determining management strategies (Forstner and Owens, 2007). Storage can be equivalent to several years' sediment yield (Walling, 1983); measuring sediment yield could underestimate the true FGS level (Walling *et al.*, 1998). It is therefore arguably better to monitor stream bed metrics which investigate sediment storage than sediment in suspension.

Measuring storage, rather than SS_c can provide a solution to the need for the long-term monitoring necessary for fully understanding sediment dynamics, as well as provide information necessary for supporting catchment land management decisions (Lambert and Walling, 1988; Young *et al.*, 1991; Walling and Fang, 2003; Collins and Walling, 2007b; Duerdoth *et al.*, 2015). Assuming that storage is representative of equilibrium conditions, long-term monitoring will determine if a significant shift in FGS storage has occurred. Quantitative measurements of sedimentation are also useful for determining potential biotic impacts (Hedrick *et al.*, 2013). Monitoring programmes should therefore include storage, and will be ineffective if they do not. Therefore, in this study, FGS will be quantified in terms of FGS storage in river channels.

One of the simplest methods of measuring sediment storage on the stream bed is Lambert and Walling's (1988) disturbance technique. A cylinder is pushed into the stream bed, isolating the bed material from flow. A known surface area of the stream bed can be manually disturbed, a water sample collected, and sediment storage estimated. The water column can also be disturbed providing an estimate of surficial storage. Disturbance can be to a given depth, providing a measure of matrix storage (Owens *et al.*, 1999; Walling and Amos, 1999; Navratil *et al.*, 2010). The method is quick, simple and low-cost to perform. The method also requires little auxiliary equipment making it suitable for use in remote areas. Although this is a validated method for quantifying FGS storage (Duerdoth *et al.*, 2015), limitations are associated with it, including the assumption that the sample collected is representative of bed storage, problems with replicating disturbance across sites and between users, and difficulty disturbing to depths of >5cm (Walling *et al.*, 1998; Navratil *et al.*, 2010; Duerdoth *et al.*, 2015). It is also not possible to estimate sediment proportions with this method.

An alternative to the disturbance technique is collection of bulk samples, e.g. a scoop sample. Scoop samples can be used to monitor FGS along stream reaches, and, it is thought that by increasing the number of particles collected in the sample, the precision of the results are likely to increase (Bunte and Abt, 2001; St-Hilaire *et al.*, 2005). A shovel is inserted vertically to a given depth, levered until it is parallel with the surface and the sediment on it removed from the streambed and analysed under laboratory conditions (Grost *et al.*, 1991). Collecting multiple replicates at each site can allow average sediment deposition to be estimated, as well as calculation of the proportion of fines and the amount of fines stored on the stream bed. Particle size distributions can also be generated, allowing the character of the FGS to be determined.

However, scoop samples are particularly prone to the loss of fines, especially when the shovel is removed from the stream bed. Excavated core samplers, e.g. the McNeil core sampler (McNeil and Ahnell, 1964), have been developed to reduce the loss of fines (Schuett-Hames *et al.,* 1996). Bulk core samples were found to most frequently approximate the true substrate composition when compared with a shovel sample and freeze-core sample (Kondolf, 2000). A hollow-core tube is worked into the stream bed to a desired depth, and the sediment excavated by hand into a retaining basin (Watschke and McMahon, 2005). The sample can then be analysed under laboratory conditions. Use of a McNeil core sampler with a test substrate showed that this method is accurate for determining the stream bed composition (Young *et al.*, 1991). This method also has the advantage of being portable and easy to use (Watschke and McMahon, 2005), although samples can be heavy (Hedrick *et al.*, 2013).

2.6 Summary

As the above discussion has shown, FGS and water quality are important issues from both ecological and geomorphological perspectives, and FGS plays an important role in river systems. Anthropogenic activity, especially agricultural expansion and intensification, has, through generating increased FGS supply, disrupted natural FGS dynamics. Although, the effects of anthropogenic activity are not constant over time and space, it is generally thought that FGS supply (and by extension FGS storage) in catchments impacted by anthropogenic activity exceeds that of unimpacted catchments. Increased FGS supply can be beneficial, but in general, increased FGS supply generates sediment problems in river systems as threshold levels of storage are exceeded. Biological, ecological, geomorphological, hydrological and chemical impacts are often observable in river systems with elevated FGS levels.

Despite recognition of the numerous problems associated with FGS, several knowledge gaps remain in this field, providing the rationale for the research objectives of this project:

- Much focus has been placed on measuring sediment in suspension and sediment yields. However, measuring storage has been suggested to be a better measure of FGS, with storage in river systems representative of the long-term deposition of sediment, which is largely a reflection of sediment supply and catchment land use. Knowledge of the quantity of FGS stored in the river channel is therefore useful for determining the success, or otherwise, of catchment management initiatives (Objective 1).

- Knowledge on the quantity of FGS stored on the stream bed is also useful for assessing potential ecological impacts. Distinguishing the available sediment from the total FGS store will allow the quantity of sediment which is likely to be dynamically exchanged between the stream bed and the water column and potentially cause ecological effects to be determined. Quantifying the FGS that is remobilised when the water column is disturbed would allow this to be achieved (Objective 2).

- Although FGS dynamics in river systems are generally well understood, it is the case that a greater understanding of FGS storage is needed to fully assess the ecological impacts of FGS, and to evaluate the success of land use management practices. Information on the quality of the FGS sediment in the river channel would allow the success of catchment land use management practices to be fully evaluated (Objective 3).

- Water quality legislation has been developed to drive management of water quality issues. In order to achieve GES in SWBs by 2027, focus has been placed on reducing FGS supply to river channels, and by extension, in-channel storage. However as FGS is neither a priority substance nor a specific pollutant, standards for quantifying FGS in river channels remain undefined. As a result, there is a lack of information on the quantity and spatial variability of FGS stored in river channels. There is therefore much potential to collect a baseline data set to evaluate the success of any measures designed to reduce FGS storage in river channels (Objective 4).

Chapter 3 – Materials and Methods

3.1 Study Sites

Research was carried out in the upper reaches of the Welland River basin (catchment area: 1680km^2) (Figure 3.1). The source of the Welland is in Sibbertoft, Leicestershire, from where it follows a predominantly north-easterly course to its mouth in The Wash. Data collection was focussed in two relatively homogenous stream catchments: Stonton Brook (catchment area: 42km^2) and Eye Brook (catchment area: 61km^2). The Stonton Brook and Eye Brook catchments are the experimental catchments in the WFF project making it important to collect data on FGS storage in these catchments. The collected data will allow the success of the catchment management to be evaluated. Data was also collected from the headwaters of the Welland River (catchment area: 53km^2). The study catchments are representative of agricultural lowland England, making the results of this study of value because they can be applied elsewhere (Angradi, 1999; Walling *et al.*, 2006). Headwater streams also influence sediment dynamics in the main river channel (Muskatirovic, 2008), making studying them important.

Table 3.1. GPS co-ordinates of the field sites sampled in a) the Upper Welland, b) Stonton Brook and c) Eye Brook.

a)	Site	Latitude	Longitude
Upper Welland	W.1	52° 28.171' N	000° 59.680' W
	W.2	52° 28.257' N	000° 58.552' W

b)	Site	Latitude	Longitude
Stonton Brook	S.1	52° 36.225’ N	000° 54.767’ W
	S.2	52° 36.206’ N	000° 54.796’ W
	S.3	52° 36.216’ N	000° 54.761’ W
	S.4	52° 36.168' N	000° 54.694' W
	S.5	52° 35.278' N	000° 54.028' W
	S.6	52° 35.247' N	000° 54.091’ W
	S.7	52° 32.843' N	000° 54.999' W
	S.8	52° 32.819' N	000° 55.034' W

c)	Site	Latitude	Longitude
Eye Brook	E.1	52° 37.848’ N	000° 54.333’ W
	E.2	52° 37.840' N	000° 54.314' W
	E.3	52° 37.435' N	000° 53.280' W
	E.4	52° 36.762' N	000° 51.788' W
	E.5	52° 36.793’ N	000° 51.829’ W
	E.6	52° 35.993' N	000° 50.315' W
	E.7	52° 35.813' N	000° 47.583' W
	E.8	52° 35.015' N	000° 46.280’ W

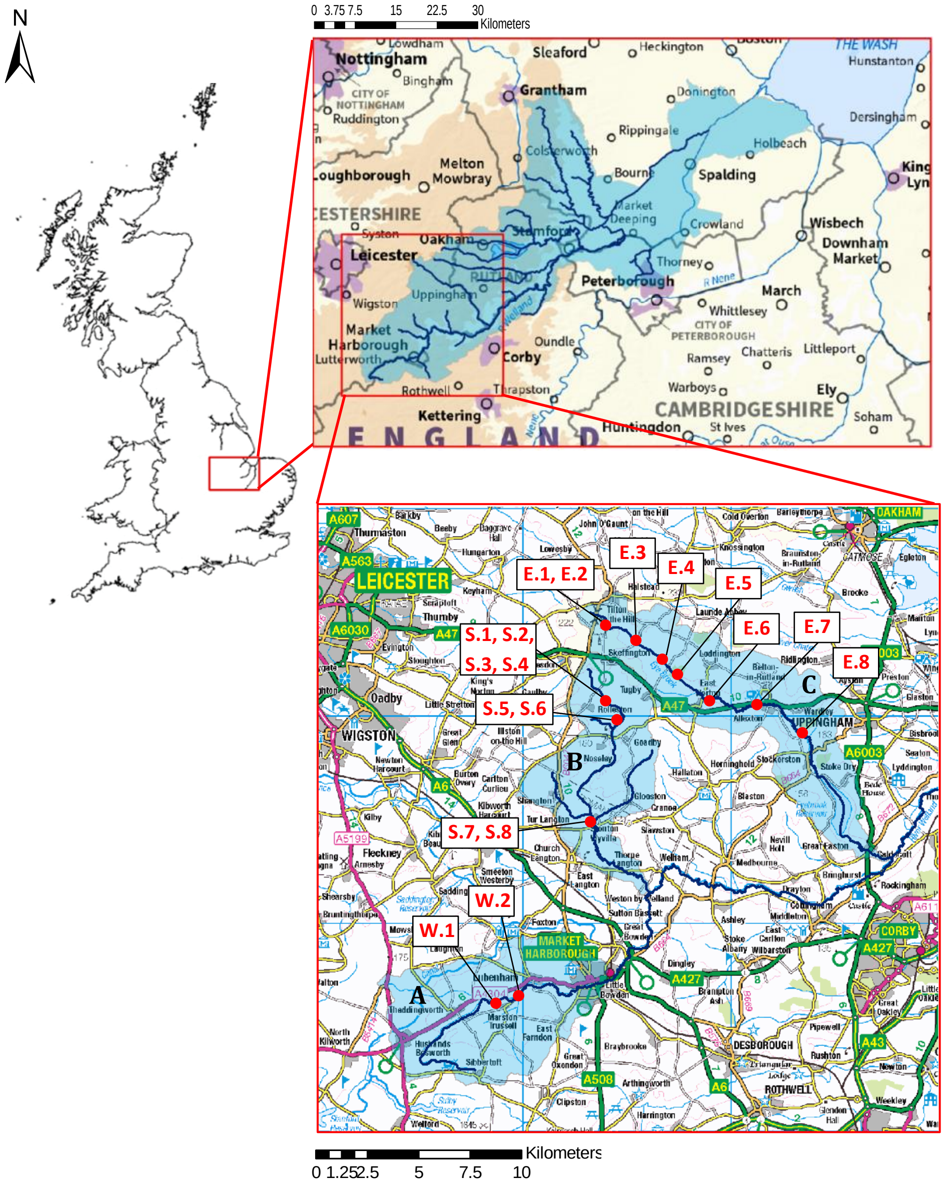


Figure 3.1. Location map showing the study catchments and field site locations. A = Upper Welland; B = Stonton Brook; C = Eye Brook. Labels represent the location of the study sites. GPS co-ordinates are shown in Table 3.1. **Source:** Environment Agency (EA) (2014) and EDINA (2015).

3.1.1 Catchment Characteristics

The soils in the study catchments are predominantly poorly drained, medium to heavy soils (Figure 3.2). Drainage is generally impeded and groundwater is naturally high. There are small areas of freely draining soils at the eastern edge of the Eye Brook catchment, as well as in the north-western edge of Stonton Brook's catchment (UKSO, 2015). Fertility of the soil is medium to high, making the land suitable for arable and grassland usage. The bedrock of the region is Charmouth Mudstone (part of the Blue Lias formation), which originates from the Sinemurian (199 to 190 Ma) and Pliensbachian ages (190 to 182 Ma) of the Early Jurassic period (British Geological Survey, 2015).

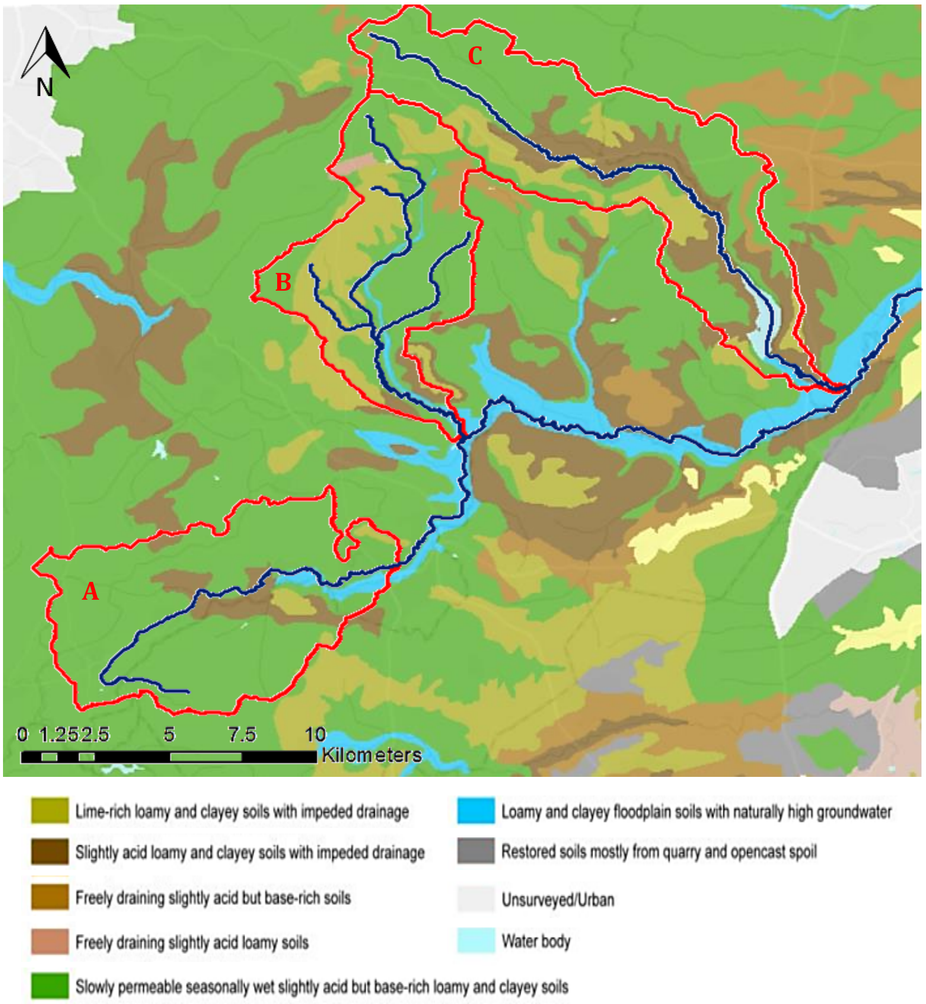


Figure 3.2. Soil map of the study catchments. A = Upper Welland; B = Stonton Brook; C = Eye Brook. **Source:** UKSO (2015).

Elevation in the study catchments ranges from 195 metres above sea level (masl) at the source of the Eye Brook to 42masl in the lower reaches of the catchments (Figure 3.3a). The elevation at the source of Stonton Brook and the Welland River are 167masl and 157masl respectively. Steep slopes (>5°) are common in Stonton and Eye Brook (Figure 3.3b), which, coupled with the impervious geology of the region, result in high overland flow rates. Slopes in the Upper Welland catchment are gentler, typically less than 5°.

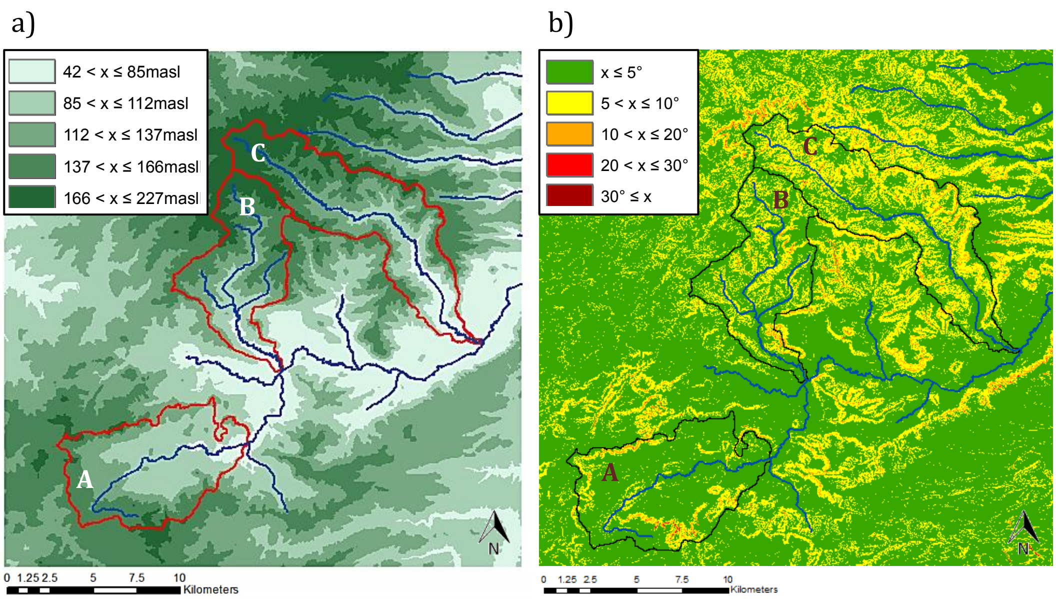


Figure 3.3. a) Shaded relief map of the study catchments b) Map of catchment slope. A= Upper Welland; B = Stonton Brook; C = Eye Brook. **Source:** EDINA (2015).

Average annual precipitation varies across the catchments, ranging from 700mm in the upper reaches of the Eye Brook to 575mm in areas of lower relief (Figure 3.4) (CEH, 2015). Precipitation is generally low intensity. 130-140 days each year have precipitation totalling 1-10 mm d^{-1} and 20-25 days have rainfall totals greater than 10 mm d^{-1}. The mean January and July temperatures are 3.5°C and 15.5°C, respectively. The annual average temperature for the Welland basin is 9.5°C. The region receives between 1400 and 1500 hours of sunshine a year (58-62 days). Snow can be found lying on the ground for, on average, between 10 and 20 days, and a ground frost is thought to occur, on average, for 100-125 days a year (MetOffice, 2015).

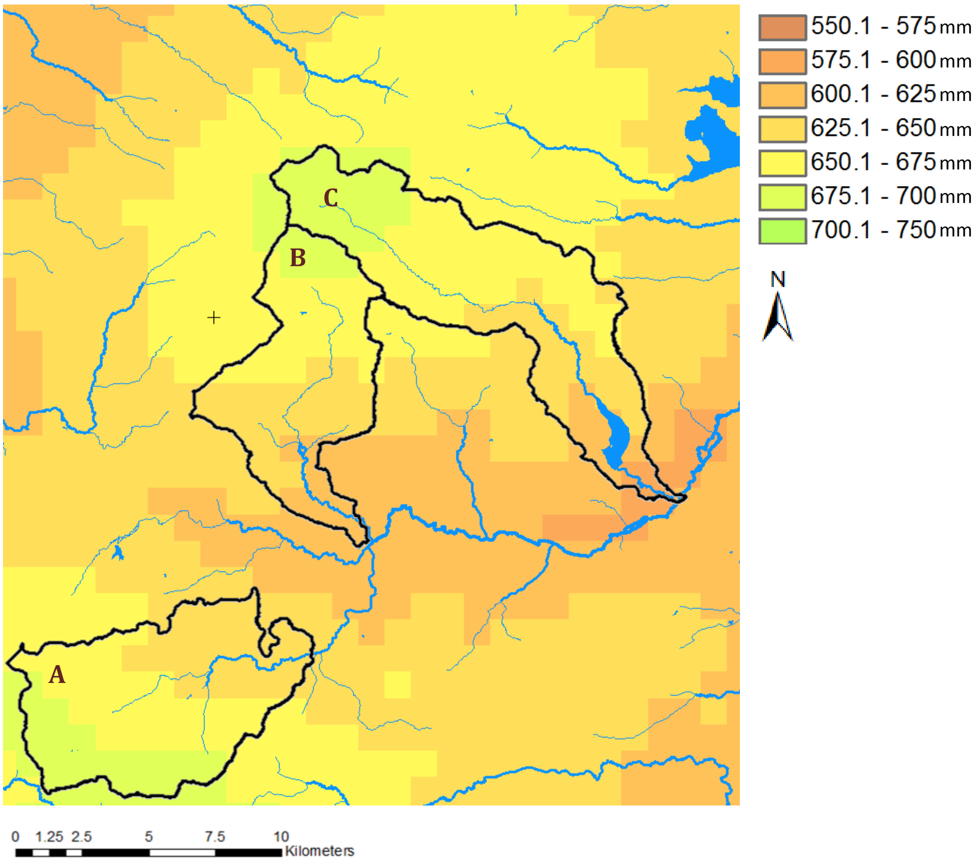


Figure 3.4. Average annual precipitation. A = Upper Welland; B = Stonton Brook; C = Eye Brook. **Source:** Centre for Ecology and Hydrology (CEH) (2015).

The study catchments are predominantly rural. Land use is dominated by arable land (55%), with 33% pasture, 7% urban and 3% forests (Figures 3.5 and 3.6). The remaining land use is non-agricultural green spaces and surface water. The town of Market Harborough (population = 23,000) lies at the eastern edge of the Upper Welland and is the largest town in any of the study catchments. Grazing of pasture means bank poaching by sheep and cattle is evident at many sites in the study catchments (Figure 3.7). Winter cereals (sown late October and November) are common in the region. In these crops, bare soils are exposed to precipitation when erodibility and erosivity, and thus potential erosion rates, are at their highest. Figures 3.6b) and 3.6c) show evidence of bare soils exposed during the winter. Fine seedbeds are also a common crop (sown August to October) (Finnie and Blackman, 2010). These limit surface storage capacity and increase surface runoff.

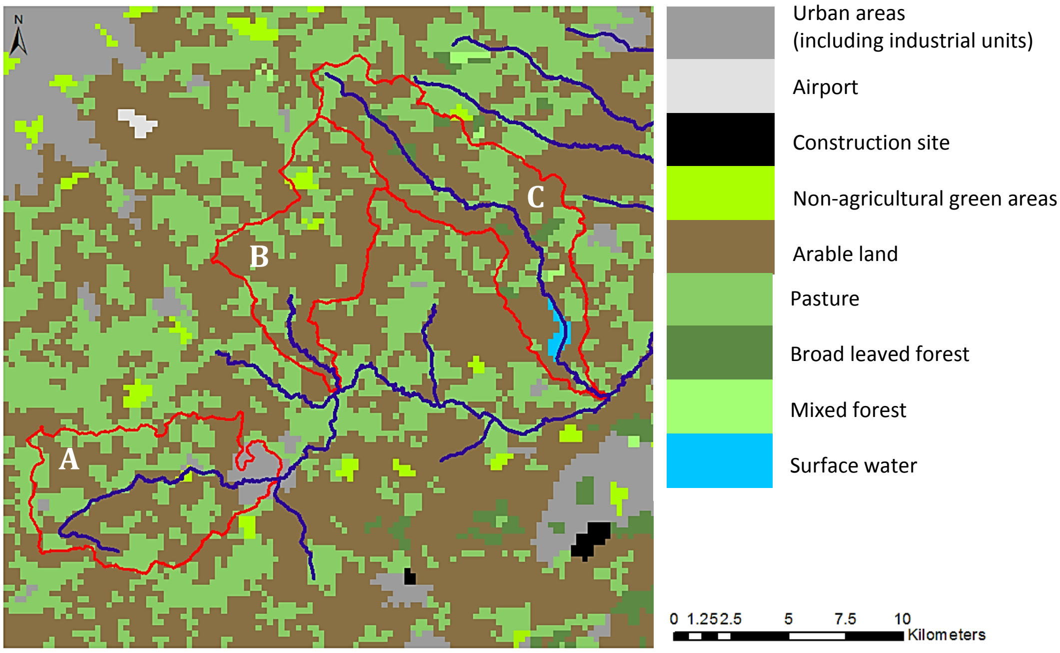


Figure 3.5. 1km resolution land cover map. The catchments are predominantly rural. Arable land accounts for 55% of land cover and pasture 33%. A = Upper Welland; B = Stonton Brook; C = Eye Brook. **Source:** European Environment Agency (2015).

Figure 3.6. a) Pasture land neighbouring the headwaters of Stonton Brook (March 2015), b) Arable land neighbouring Stonton Brook (February 2015), c) Arable land surrounding the Upper Welland (December 2014) and d) Woodland along the banks of the Eye Brook (March 2015). Photos: author's own.

a) b)

Figure 3.7. a) Poaching on the banks of Eye Brook at East Norton. b) A defence installed on Stonton Brook, near Welham, to reduce bank poaching. Photos: author's own.

3.1.2 Stream Characteristics

The Upper Welland, Eye Brook and Stonton Brook are low order (first, second and third order) alluvial streams, hereafter referred to as headwater streams. Naturally meandering planforms are present along the course of the channels. Riffle-pool sequences are a characteristic feature of the study streams, and display the typical characteristics described in section 2.3.5 (Figure 3.8; Figure 3.9). Despite steep hillslope gradients in the upper reaches of the streams, channel gradients are generally shallow (Table 3.2). The stream channels are largely unmodified, although where they pass through residential areas lining of channel walls, weir installation and adjustment of bed substrate is evident.

Figure 3.8. Typical reaches of a) the Upper Welland, b) Stonton Brook and c) Eye Brook. Photos: author's own.

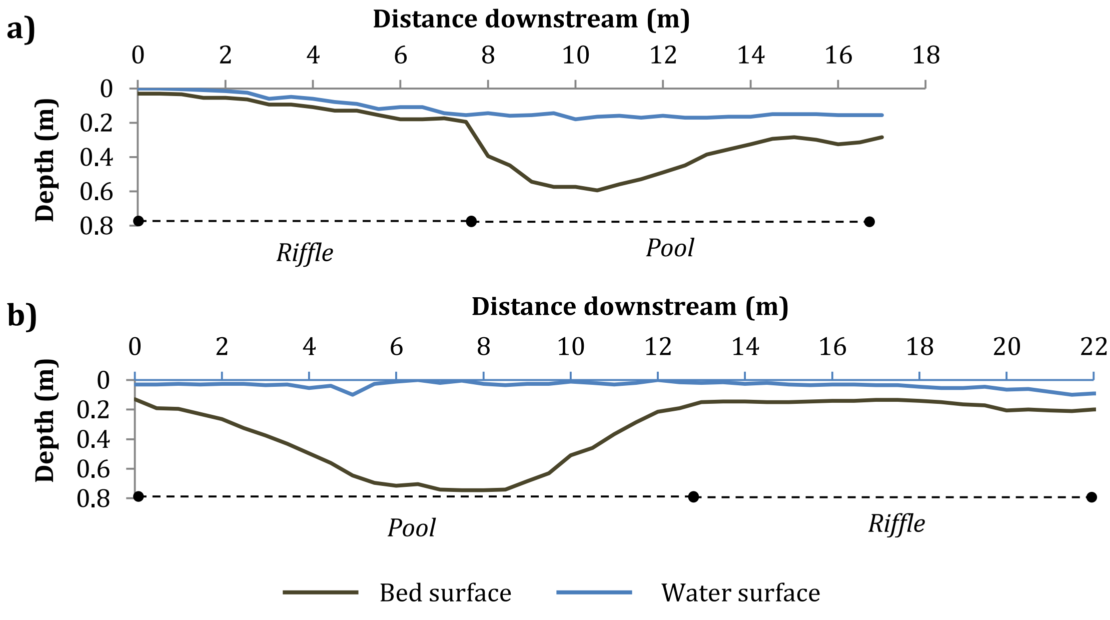


Figure 3.9. Stream bed profiles for a) site W.1 and b) E.8 showing the riffle-pool topography (as described in section 2.3.5) on the stream bed of the study catchments.

Table 3.2. Summary data describing variation in bed surface and water surface slope. Reach slope represents the average bed slope across the riffle-pool sequence.

	Bed surface slope (%)			**Water surface slope (%)**
	Riffle	**Pool**	**Reach**	
Minimum	0.00	0.08	0.20	0.20
Maximum	5.50	3.60	2.20	1.70
Mean	1.37±0.33	0.87±0.19	1.03±0.13	0.78±0.11

Mean annual flows in the catchments are low. The mean annual flow for the Welland at Ashley (catchment area = 251km^2) is 1.424 m^3 s^{-1} (CEH, 2015). Mean annual flow south of Eye Brook reservoir is 0.231 m^3 s^{-1}. As a result of the impermeable geology dominating the catchments, the streams respond rapidly to precipitation. Flows are characteristically flashy (Figure 3.10a). At the southern end of the Eye Brook, flow is constrained by Eye Brook reservoir, resulting in flow having little annual variation (Figure 3.10b).

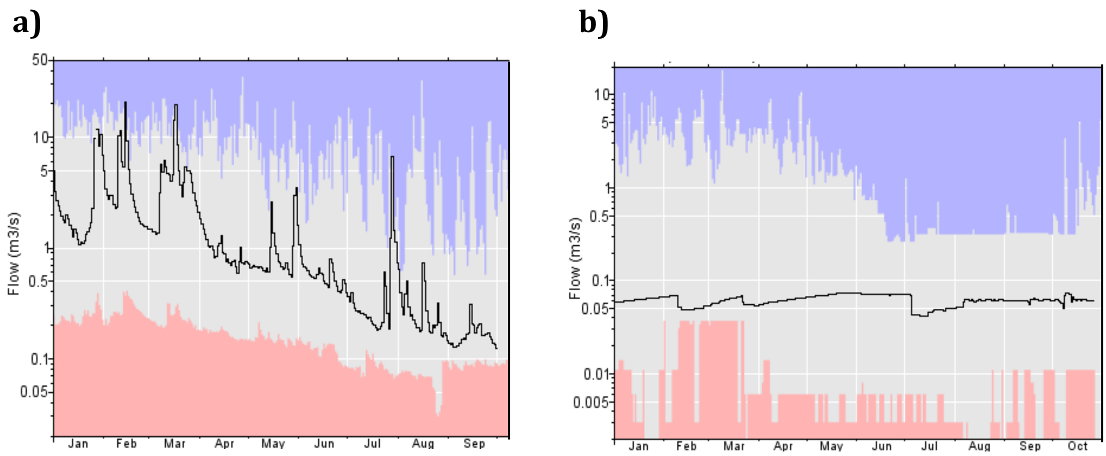


Figure 3.10. Daily flow hydrograph for a) the Welland at Ashley and b) Eye Brook downstream of Eye Brook Reservoir. Red and blue envelopes represent the lowest and highest flows on each day. Flows in the Welland are flashy. Flows in the Eye Brook are constrained by Eye Brook reservoir. **Source:** CEH (2015).

During normal discharge conditions, the study streams are shallow. Average flow depth was 0.15m. Average flow depth over riffles and pools was 0.05m and 0.22m, respectively. Maximum recorded flow depth was 0.74m (pool at E.8). At high discharges, flow depths of over 1m were recorded in some pools. Channel width was generally narrow. Width in the upper reaches was 0.6m. The widest section of channel was the riffle at site W.1 (5.1m). Pools are generally wider than riffles, with average widths of 1.9m and 2.5m respectively. The average channel width was 2.2m. Flow expansions are common downstream of bridges (e.g. where Stonton Brook is crossed by Palmers Lane near Goadby) and at fords (e.g. the Eye Brook at Thorpe Langton).

The ecological status of the water bodies in the study catchments has been monitored under the WFD. Survey results suggest that, in 2014, the studied water bodies had bad or poor ecological quality (Table 3.3). This is attributable to the low fish, macrophyte and diatom status of the catchments, and their high phosphate concentrations (Finnie and Blackman, 2010). The fish status was classified as poor, bad and moderate for the Upper Welland, Stonton Brook and Eye Brook respectively in 2014 (EA, 2014). As Moore (2012) suggests, the Welland River and its tributaries should support the majority of England's freshwater fish species, including spawning species. Fish stocks, particularly upstream of Market Harborough, are however very poor. Macrophyte and phytobenthos status classification is also moderate or poor (EA, 2014). Furthermore, the quality of the Eye Brook has deteriorated from good quality in 2010 to poor quality in 2014. Although the Upper Welland saw an improvement in ecological quality between 2010 and 2012, it has subsequently deteriorated. The study catchments have also been identified as being at risk from diffuse source sediment input (Figure 3.11). Visual inspection of riffles and pools supported this, and indicated that the study catchments have been impacted by FGS and areas of FGS accumulation were visible on the stream bed.

Table 3.3. Ecological status for the study catchments, as measured for the WFD in 2010, 2011, 2012, 2013 and 2014. **Source:** EA (2014).

	Ecological status				
	2010	**2011**	**2012**	**2013**	**2014**
Upper Welland	Poor	Moderate	Moderate	Bad	Bad
Stonton Brook	Moderate	Bad	Bad	Bad	Bad
Eye Brook	Good	Good	Moderate	Moderate	Poor

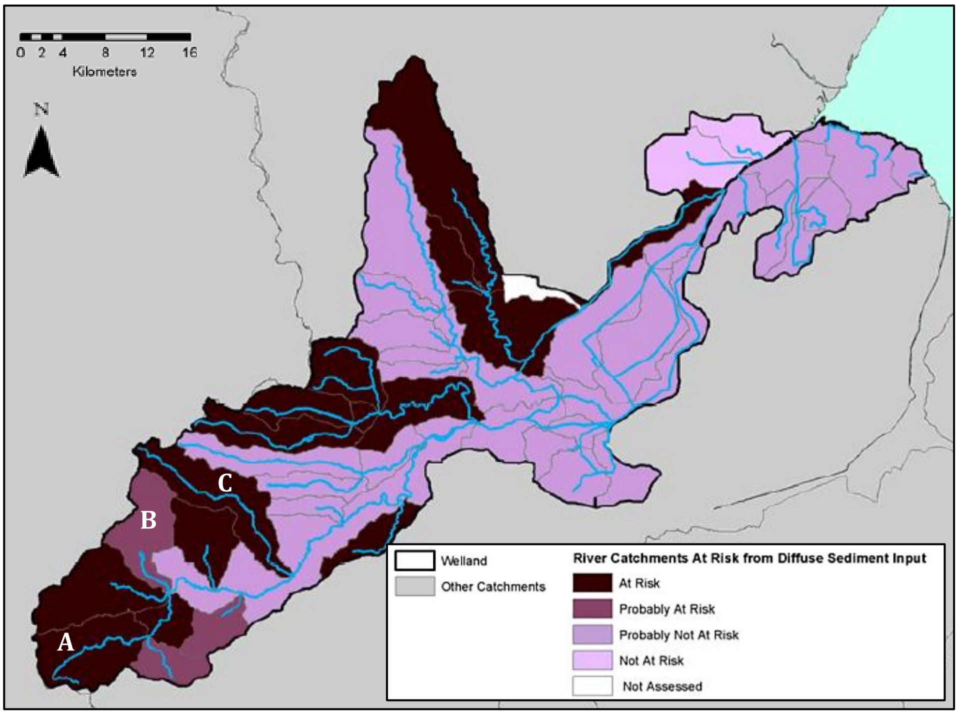


Figure 3.11. Catchments at risk from diffuse source sediment. A = Upper Welland; B = Stonton Brook; C = Eye Brook. **Source:** Finnie and Blackman (2010: 31).

3.2 Methodology 1: Estimates of fine-grained sediment storage

Reconnaissance surveys were carried out to determine riffle-pool sequences which were suitable for sampling stream bed material. Feasibility of the sites was also considered (e.g. access) when conducting the reconnaissance surveys (Schuett-Hames *et al.,* 1999). Sites that represented local rather than characteristic conditions were avoided, e.g. sites with concrete lined channels, artificial bed substrate material, weirs and visible bank poaching were not selected. In total, eight pool-riffle sequences in the Eye Brook, eight in Stonton Brook and two in the Upper Welland were identified as suitable for collecting stream bed samples. Samples were collected from November 2014 to March 2015 (Table 3.4). In total, 98 core samples were collected across the sites. 50 samples were collected from riffles and 48 from pools. The remobilisable fraction of FGS was also estimated at these sites. The number of samples collected from each riffle and pool ranged from one to seven. On average, three samples were collected from each bedform.

Table 3.4. Sampling dates for the study sites.

Upper Welland		Stonton Brook		Eye Brook	
Site	**Date**	**Site**	**Date**	**Site**	**Date**
W.1	5/12/14	S.1	6/3/15	E.1	2/3/15
W.2	5/12/14	S.2	24/2/15	E.2	11/11/15
		S.3	6/3/15	E.3	11/11/15
		S.4	9/12/15	E.4	4/11/15
		S.5	20/3/15	E.5	6/3/15
		S.6	20/3/15	E.6	8/12/15
		S.7	8/1/15	E.7	29/1/15
		S.8	8/1/15	E.8	2/3/15

No standard method has been developed to quantify the FGS stored on the stream bed (Lambert and Walling, 1988; Von Bertrab *et al.*, 2013). Various methods are available for quantifying FGS storage. The selected method should be the one which is most suitable for the aims of the research, rather than the one which is most accurate (Kondolf, 2000). Cost, labour requirements and device portability are also important considerations when selecting which sampling method to use. Monitoring should take a multi-scaled, holistic approach (Rickson, 2006; Walling and Collins, 2008). Furthermore, the chosen methodology should be precise (Duerdoth *et al.*, 2015), and care must be taken to minimise potential bias when collecting samples (Evans and Wilcox, 2014).

A review of the literature suggested that bulk core samplers, e.g. the McNeil core sampler, are the most accurate device for sampling the stream bed substrate (Grost *et al*., 1991). Subsurface matrix sediments, which can be sampled accurately with the McNeil corer, play an important role in water quality, making quantifying them of importance (Kondolf, 2000). Furthermore, the WFD places a focus on the ecological quality of water. Fish stocks are a key aspect of ecological quality. Sediment storage should be monitored to depths at which spawning species are found, which is achievable with a McNeil core sampler. Samples of stream bed sediment were therefore collected with a McNeil core sampler (McNeil and Ahnell, 1964; Grost *et al.*, 1991; Watschke and McMahon, 2005; Hedrick *et al.*, 2013).

Samples were processed using laboratory-based analyses in order to determine the amount, and proportion, of fine-grained (≤2mm) sediment stored on the channel bed at each study site.

3.2.1 Field methodology

Prior to collecting a McNeil core sample, a 250ml water sample was collected to determine the suspended sediment concentration of riffles and pools (SS_cpre). This allows for correction of sediment already in suspension (Duerdoth *et al.*, 2015).

A McNeil core sampler (height 24cm, surface area 0.0182m^2) (Figure 3.12) was inserted into the stream bed until the collection cylinder rested on the bed surface. The sediment stored in the coring cylinder was manually excavated to the stop ring, and stored in the collection basin. If bedrock material was reached before the stop ring, no further sediment was removed, and the depth of excavation was recorded.

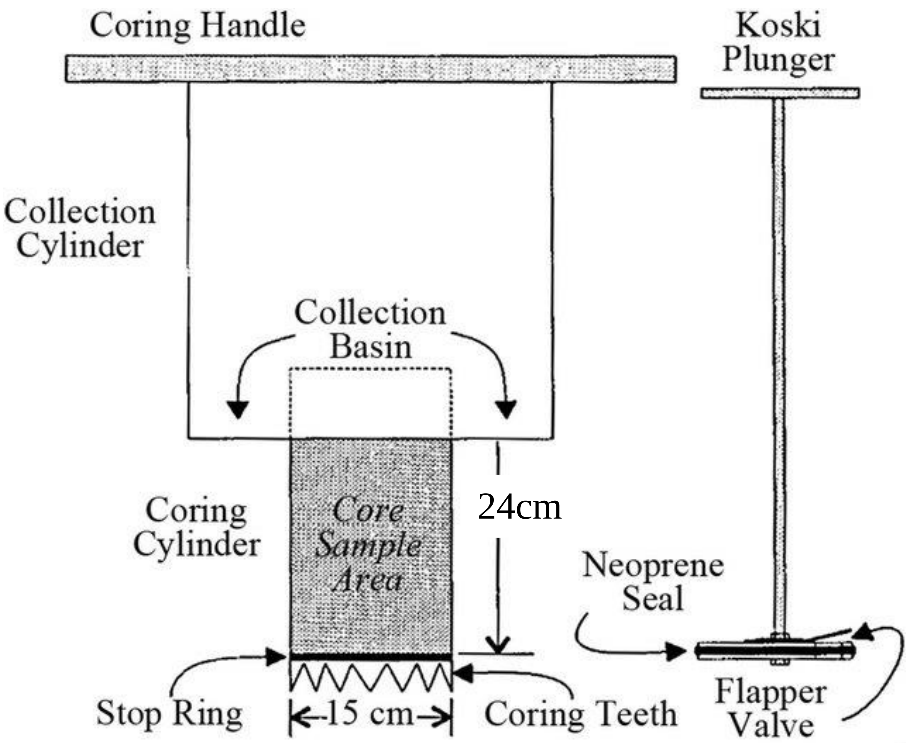


Figure 3.12. Schematic of the McNeil core sampler used to sample the stream bed substrate. **Source:** Hedrick *et al*. (2013: 94).

Once all sediment was removed from the coring cylinder, a Koski plunger was inserted into the coring cylinder. This removed excess water from the sample, whilst retaining the FGS that was in suspension in the coring cylinder. It is assumed that the loss of fines due to sediment remaining in suspension in the coring cylinder once the plunger was removed was negligible (Platts *et al.,* 1983). Figure 3.13 shows the corer in-situ with fine sediment trapped on the Koski plunger. Care was taken to ensure that any sediment attached to the hands and arms of the excavator was rinsed off back into the sample.

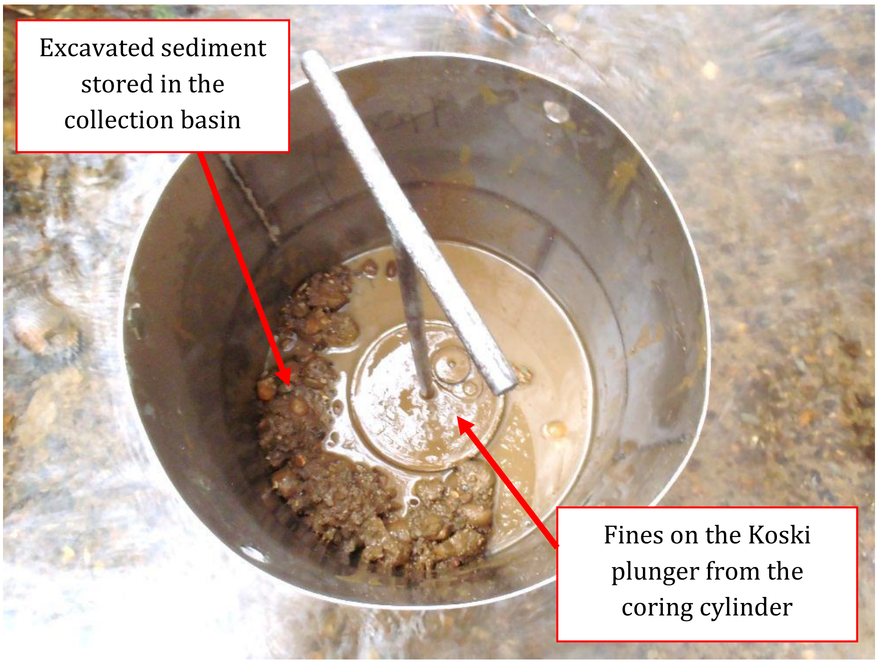


Figure 3.13. The McNeil core sampler *in-situ* with the Koski plunger inserted to minimise loss of fines from the sample. Photo: author's own.

Once the Koski plunger was in place, the corer was removed from the streambed and the sample (B) decanted into buckets. The settled sediment from the collection basin (B_1) was put into one bucket. The water and sediment in suspension in the collection basin (B_2) was put into a second bucket. The collection cylinder was then rinsed, and the rinse water added to sample B_2. B_2 was left to settle for 5 minutes to separate the coarse and fine fractions, after which, the settled fraction of B_2 was transferred to B_1. The volume of water in B_2 was recorded and the water thoroughly agitated before collecting a 250ml aliquot. B_2 was then discarded. B_1 was stored in a lidded bucket at 5°C prior to analysis.

It is important to collect replicate samples at each sampling site in order to account for spatial variability in FGS storage, and ensure that results are representative of FGS storage (Peterson and Quinn, 1996; Walling *et al.*, 2003a; Collins and Walling, 2007a). Owens *et al.* (1999) collected three samples from each site to account for spatial variability in storage. Collins and Walling (2007b) and Ballantine *et al*. (2009) collected one sample from the thalweg and one sample from near the channel bank, and suggested that the average of the two would provide a meaningful estimate of FGS storage. In this study, multiple samples were collected from each bedform in order to provide a representative estimate of FGS storage in riffles and pools (three on average).

During the sampling procedure, several considerations were taken into account to ensure that the methodology used was robust. Samples were collected in an upstream direction to prevent disturbing the fines stored on the stream bed (Gillette *et al.*, 2004). If the sampler could not be fully inserted into the stream bed (e.g. because the bed material was too coarse), the depth to which the corer was inserted was recorded. Recording the sample depth was important to allow results to be depth-normalised and compared to other similar studies. If an obstruction in the bed meant that a sample depth of at least 10cm could not be collected, the sampler was moved to a new upstream location on the same bedform. It was important to move the sampler at least 1m upstream of the previous site (Schuett-Hames *et al.,* 1999) because fines settled on the bed surface could have been disturbed when the corer was being inserted. If there were no other suitable locations on the bedform, samples were instead collected from the adjacent upstream bedform. All samples were collected from the channel mid-point. However, in the case of pools, it was sometimes necessary to sample closer to the channel margins because the depth of the retaining cylinder meant that sampling was restricted to water depths less than 40cm. Walling *et al.* (1998) suggest that this is an appropriate solution where flow depth would otherwise prevent sampling.

3.2.2 Laboratory Methodology

Two main analytical sieving methodologies can be used to collect grain size information: volumetric and gravimetric procedures (Rex and Carmichael, 2002). Volumetric procedures involve wet sieving, and are based on the volume of water displaced by sediment (displacement volume), providing a measurement of sediment volume in millimetres. Gravimetric analysis is thought of as a 'dry' method. Samples are oven dried prior to analysis, and the mass of sediment determined post sieving. It is thought that gravimetric analysis provides more precise and accurate data compared to the results obtained from volumetric analysis. Gravimetric analysis was therefore used to determine the stream bed sediment composition.

Sediment samples B_1 were oven dried at 110°C for 24 hours. As per the methodology in Ramos (1996), drying pans were pre-weighed. After the drying time elapsed, samples were removed from the oven, cooled and weighed. Samples were then returned to the oven for a further hour, and the cooling and weighing process repeated. If the weights differed by greater than 5%, drying continued. When the mass remained constant between dryings (< 5% change in weight), the sample was ready for analysis.

Prior to sieving, the oven-dried samples were ground using a pestle and mortar to disaggregate them. This is because sample handling and storage can change the grain size distribution of the sample (Phillips and Walling, 1995; Syvitski, 2007). Disaggregating samples which were not previously aggregated ensures analyses are conducted on samples which have not been modified by their handling and storage.

A known quantity of sediment was passed through a mechanically shaken sieve tower to separate the samples into three size fractions: D>2mm, $2 \geq D > 1$mm and D≤1mm. The mass of sediment contained in each size fraction was recorded as M_2, M_{f1} and M_{f2}, respectively. Care was taken to ensure that all particles were removed from the sieve tray before sieving another sample. Sieve pans were weighed prior to sieving to minimise losses associated with transferring sediment between containers. A selection of samples were reweighed to ensure analytical and quality assurance. If the weight difference was greater than 5%, all samples were reweighed (Rex and Carmichael, 2002).

The mass of fines contained in B_2 was determined using the vacuum filtration methodology outlined in Bartram and Balance (1996). 47mm diameter cellulose nitrate gridded membrane filters with 0.45μm pores were used on a reusable filter holder with receiver. Filters were pre-cleaned with distilled water. As Eaton *et al.* (1969) suggest, leachable material can be contained on filters, with the amount varying between filters. Pre-cleaning helps reduce variability between filters and reduce sources of error. Filters were then oven dried at 105°C for 2 hours, cooled in a desiccator, weighed and returned to the oven for half an hour. The cooling process was repeated. If the change in mass was less than 0.5mg, the filters were ready to use, otherwise they were returned to the oven until a constant weight was achieved. Eaton *et al*. (1969) identify other potential sources of error when using membrane filters for gravimetric analysis. These include mass change caused by static charge on the filter paper, and uptake of atmospheric moisture by the filter. Care was taken to minimise these. Filters were handled with smooth-faced forceps to reduce the influence of static charge. The filters were also stored in a desiccator prior to weighing to help prevent absorption of atmospheric moisture.

The 250ml aliquots collected from B_2 were thoroughly agitated to resuspend any sediment that had settled out during storage. A known volume of the agitated sample was then pumped through the pre-cleaned, pre-weighed filter. Bartram and Balance (1996) suggest filtering 200ml of sample in order for a weight change to be detected between pre- and post-filtration. The water samples collected from the corer had visually high sediment concentrations, therefore a weight change could be detected with a smaller volume of sample, e.g. Engelbrecht and McKinney (1956) used sample sizes of 25-50ml in their study of suspended solids, supporting the use of smaller samples than Bartram and Balance suggest. The volume of water filtered varied between samples, ranging between 20ml and 50ml of water.

In order to minimise loss of fines, once the sample had passed through the filter, the filter holder was rinsed with a small volume of distilled water. The pump was left on for two minutes more to ensure the distilled water filtered through to the receiver. This was a necessary procedure because the high sediment concentrations meant some fines settled onto the filter holder during the filtration process. On occasions, the loss of sediment to the filter holder was so high that a second filter paper was used with distilled water to collect any sediment trapped on the holder. The mass of the sediment contained on the two filters was combined to provide an overall sediment concentration for that sample. Filter holders were rinsed prior to filtering another sample.

The filters were oven dried using the procedure outlined above, and their mass recorded. The mass of sediment contained in B_2 could then be estimated using Equations 3.1 and 3.2.

$$M_{f3} = C \times V \qquad (3.1)$$

$$C = \frac{W_s}{V_1} \times 1000 \qquad (3.2)$$

where M_{f3} is the mass of FGS contained in the sample B_2 (g), C is the sediment concentration (g l^{-1}), as determined by vacuum filtration, V is the volume of water (l) in the bucket containing sample B_2, $W_s = W_2 - W_1$, W_1 is the filter mass pre-filtration (g), W_2 is the filter mass post filtration (g), and V_1 is the volume of water filtered (ml).

The amount, and proportion, of FGS stored on the channel bed at each study site was estimated using Equation 3.3 and 3.4.

$$S_f = \frac{M_1}{A \times d} \qquad (3.3)$$

$$P_f = \frac{M_1}{M_1 + M_2} \times 100 \qquad (3.4)$$

where S_f is the storage of FGS per unit area per unit (g m^{-2} cm^{-1}) (hereafter referred to as FGS storage), A is the surface area of the inner cylinder (m^2) and d is the excavation depth (cm). $M_1 = M_{f1} + M_{f2} + M_{f3}$. M_{f1} is the mass of fine-grained sediment 2≥D>1mm in sample B_1, M_{f2} is the mass of fine-grained sediment ≤1mm (g) in B_1, M_{f3} is the mass of FGS in sample B_2 and M_2 is the mass of coarse-grained sediment (D>2mm) collected in sample B_1 (g). P_f is the proportion of stream bed material that is FGS (%).

3.2.3 Data Analysis

Mean average FGS storage was calculated for each bedform. This allowed variability of storage within a bedform to be accounted for, and reduces the potential for reporting unrepresentative storage values (Owens *et al.*, 1999). At-a-site storage was also calculated as a simple average to represent the mean reach average storage (i.e. average storage in both riffles and pools at a site). All mean values are reported with ± the standard error of the mean (SEM).

Statistical techniques were used to identify trends in the amount of and proportions of FGS stored on the stream bed. An Anderson-Darling test revealed that some data was not normally distributed (Figure 3.14), therefore non-parametric statistics were used for statistical analyses. The Mann-Whitney U test was used to determine whether the medians of two independent samples were significantly different (Ashcroft and Pereira, 2002). The Wilcoxon Signed-Rank test was used for matched pairs. A 95% confidence interval was used to determine whether the null hypothesis, H_0 (the tested data come from the same group), or alternate hypothesis, H_a (the tested data come from two separate groups), should be accepted. If $p<0.05$, H_0 was rejected in favour of H_a, suggesting that differences are statistically significant. Statistical analyses were carried out for data collected from Stonton Brook and Eye Brook. As a result of the lack of samples collected in the Upper Welland, statistical analyses were not conducted on samples collected from this catchment.

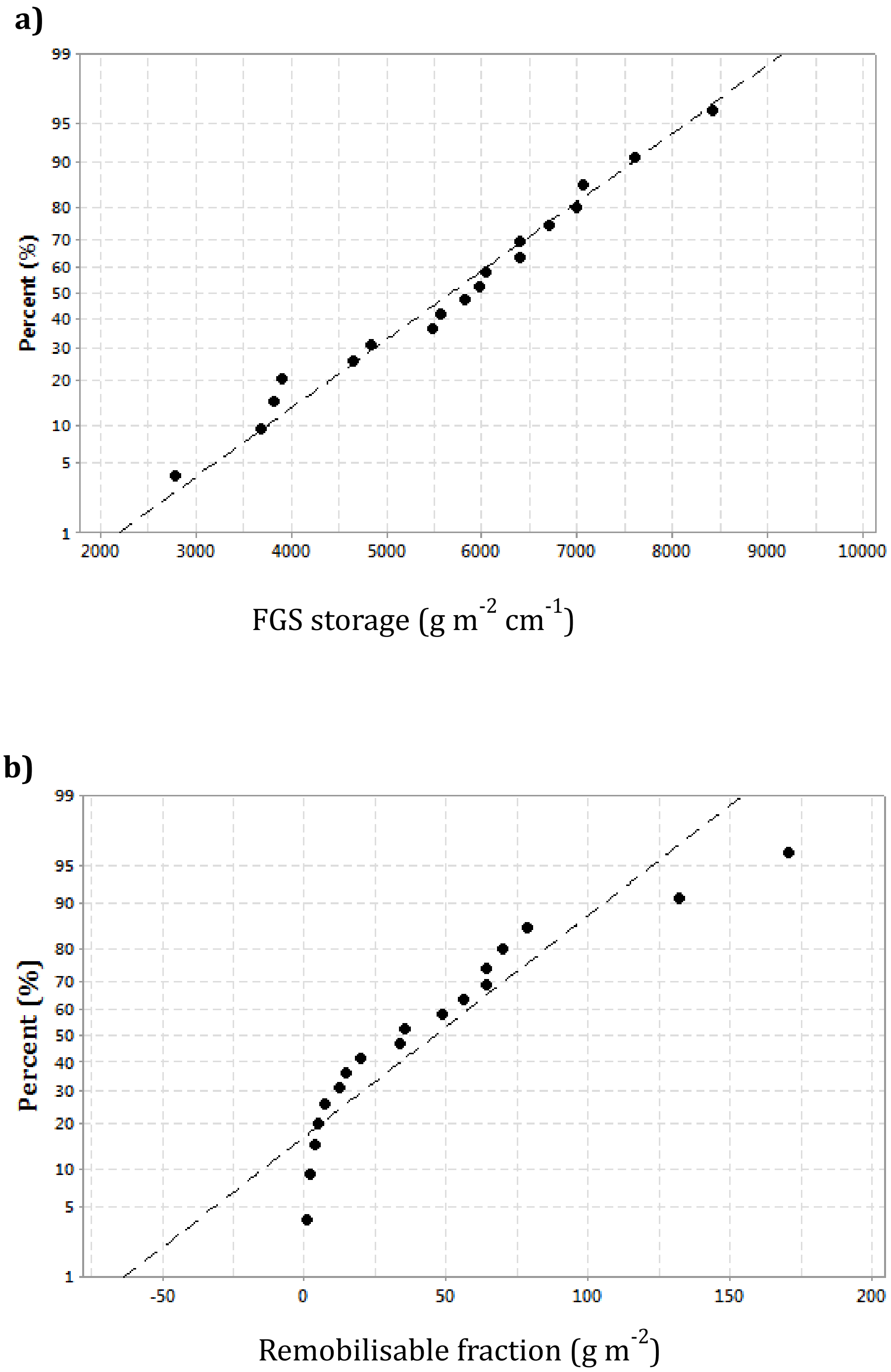


Figure 3.14. Example probability plot of a) FGS storage in pools and b) surficial storage in riffles. FGS storage in pools displays a normal distribution ($p = 0.849$). Surficial storage in riffles is non-normal ($p = 0.018$). According to the Anderson-Darling test, if $p>0.05$, the data is normally distributed.

3.3 Methodology 2: Remobilisable Fines

SS is a key pathway in the transport and fate of sediment. Knowledge of the remobilisable fraction stored on the stream bed is therefore important (Phillips and Walling, 1995). A modified version of Lambert and Walling's (1988) disturbance technique was used to determine the amount of FGS stored on the stream bed surface that is remobilisable under a given shear stress.

3.3.1 Field Methodology

An open-ended cylinder (surface area $0.26m^2$) was inserted into the stream bed until the water in the cylinder was sealed from the flow (Figure 3.15). Care was taken to minimise the disturbance of fines. If the flow could not be sealed, the barrel was relocated to another location on the bedform and reinserted. Unless the cylinder is watertight, some dilution of the sediment concentration will occur (Lambert and Walling, 1988); this is particularly problematic in shallow, fast-flowing areas, e.g. riffles.

Figure 3.15. The barrel in-situ for sampling surficial storage of FGS. The water within the barrel is still, demonstrating that flow has been isolated. Photo: author's own.

Once the barrel was on the stream bed, water depth was measured at four locations within the barrel. Water volume in the barrel was calculated as a product of mean depth within the barrel and cross-sectional area (Equation 3.5).

$$V = (A \times \bar{d}) \times 1000 \qquad (3.5)$$

where V is the water volume in litres, A is the surface area of the barrel (m^2) and $\bar{d}$ is mean flow depth (m), as estimated from the mean of the four measurements taken within the cylinder.

The water column was disturbed using a plaster mixing paddle and a cordless drill. Automating the disturbance helps prevent problems with ensuring the force used to disturb the stream bed is constant between sampling locations and bed materials, thereby helping to reduce operator error. The water column was agitated for 60 seconds. During disturbance the paddle was kept at a constant depth above the stream bed and rotated in a clockwise direction around the circumference of the barrel. In sites where the bed material protruded above the bed, the paddle was raised above it to prevent contact with the bed material. After one minute of disturbance, a 250ml depth integrated aliquot of water was collected from the barrel. Integrating the sample ensures that the whole vertical concentration gradient of SS_c is sampled, thus ensuring a representative sample.

Surficial storage of remobilisable FGS was sampled at the same locations as FGS storage was sampled with the McNeil core sampler. This ensured that the sample of surface material was collected from areas of similar flow conditions, and thus also similar deposition and storage conditions. Figure 3.16 shows an example of the distribution of the samples collected at one of the field sites. Furthermore, as was the case when collecting core samples, the site was approached in an upstream direction to prevent the bed material being disturbed.

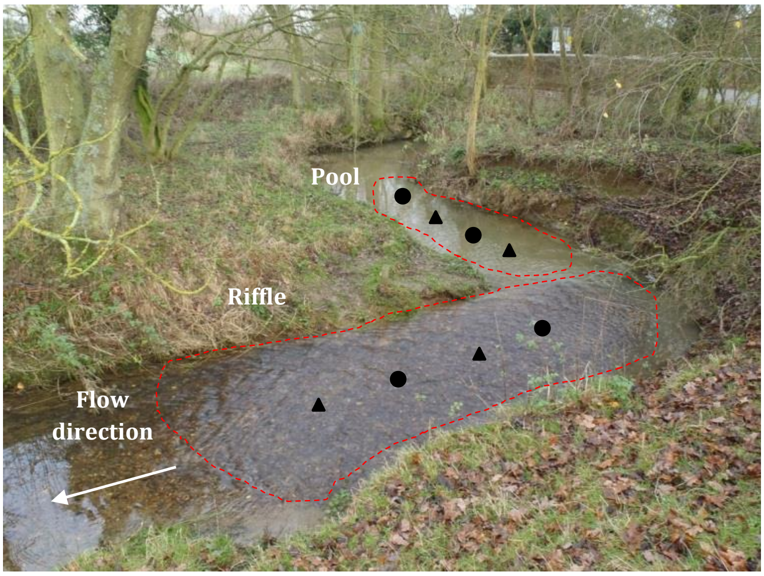


Figure 3.16. Distribution of samples along the pool and riffle at site W.2. ● = McNeil core samples ▲ = remobilisation of surficial fines. Photo: author's own.

3.3.2 Laboratory Methodology

The SS_c of the water samples collected after manual disturbance ($SS_c post$) was determined gravimetrically using the methodology outlined in Section 3.2.2b). 200ml of water sample was filtered to determine $SS_c pre$ and $SS_c post$. The amount of FGS released per unit surface area of the channel bed ($B_r(t)$; g m^{-2}) when manually disturbed was estimated using Equation 3.6 (Walling *et al*., 2003c).

$$B_r(t) = \frac{SS_c \times V}{A} \tag{3.6}$$

where $SS_c = SS_c pre - SS_c post$ (g l^{-1}), V is the volume of water in the sampling cylinder (l) and A is the surface area of the barrel (m^2).

3.3.3 Data Analysis

The statistical techniques used in section 3.2.3 were used to analyse the results on the amount of FGS released per unit surface area of the channel bed upon disturbance.

3.4 Methodology 3: Grain size distribution

3.4.1 Laboratory Methodology

Grain size is an important consideration when assessing the ecological impacts of FGS (Church *et al.,* 1987; Bilotta and Brazier, 2008; Cooper *et al.,* 2008). Laser granulometry was therefore used to determine the grain size distribution (GSD) of the FGS contained in riffles and pools. Numerous methods are available to determine GSDs. These include sieving, standard sedimentation methods e.g. the pipette procedure, and more recently developed methods, including optical and electrical sensing and laser diffraction techniques (laser granulometry) (Loizeau *et al.*, 1994; Konert and Vandenberghe, 1997; Chappell, 1998; Di Stefano *et al.*, 2010). Grain size is defined differently for each method. The sieving method defines grain size in terms of the width of the particle, as determined by the length of the side of a square hole through which the particle can just pass. The pipette method defines the diameter of a particle as equal to that of the diameter of a sphere with the same settling velocity (Konert and Vandenberghe, 1997). Particle diameters provided by laser granulometry are based on the scattering of light, and are inversely proportional to the angle of diffraction, i.e. an increased angle of diffraction represents a decreased particle size (Loizeau *et al.*, 1994). Laser granulometry assumes that all particles are spherical. A mean diameter based on the cross-sectional area of the particles is given (Di Stefano *et al.*, 2010). All particle sizing methods make various assumptions, and arguably none give true results (Konert and Vandenberghe, 1997). Due to the different assumptions each method makes, the results yielded from measurements of the same sample will not be identical (Beuselinck *et al.*, 1998).

Use of laser diffraction to provide a GSD for sediment samples is a disputed method. Traditional methods of GSD determination are, as a result, still used. However, these methods are slow to perform, require a large amount of sample (at least 10g) to carry out the procedure, and are subject to operator error (Beuselinck *et al*., 1998; Di Stefano *et al*., 2010). By comparison, 100-200mg of sample is required for laser diffraction of samples containing high proportions of silt and clay. Approximately 5g is required for coarser, sandy samples (Konert and Vandenberghe, 1997). Laser diffraction also offers rapid analysis and can provide wide GSDs in a single analysis (Beuselinck *et al*., 1998; Di Stefano *et al*., 2010). As a result of the advantages of laser granulometry compared to traditional methods, this method was selected for the analysis of grain size.

Laser granulometry can be used on samples containing particles up to 2mm (2000μm) in diameter. However, a pilot study showed that results of consecutive repetitions of a sample were not very reproducible when the sample contained particles up to 2000μm (Figure 3.17a). The reproducibility of samples are improved when the sample is re-sieved to 1mm (1000μm), removing the very coarse sand fraction (Figure 3.17b). Beuselinck *et al*. (1998) observed similar issues with reproducibility, suggesting that the reproducibility of results is generally greater for laser diffraction compared to traditional methods, except when the sample contains very coarse sand.

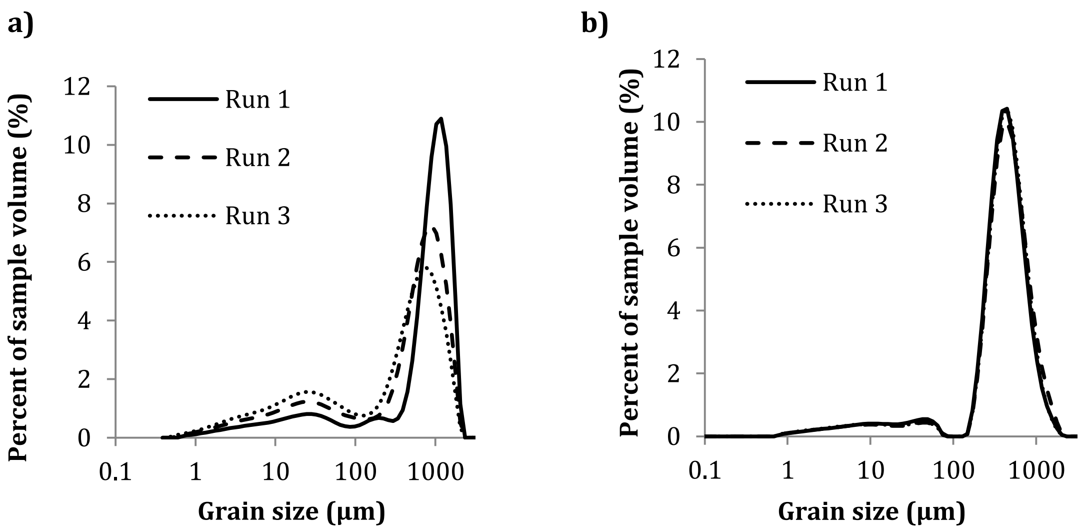


Figure 3.17. Consecutive repetitions of sediment sample W.1 (riffle) sieved to a) ≤2mm and b) ≤1mm. Reproducibility improves when sediment 2≥D>1mm is removed.

One subsample was analysed for each riffle and pool at a field site. Chappell (1998) suggests that subsampling can affect the reproducibility of results. Therefore care was taken to ensure that subsamples were representative of the whole sediment sample. A riffle box was used to collect each subsample. Sediment samples were then added to Reverse Osmosis (RO) water in a Malvern Hydro 2000 unit until laser obscuration was between 5 and 15%. The sample was subjected to ultrasonic dispersion immediately prior to analysis. Although there are other more effective methods of dispersal, e.g. Calgon, ultrasonic dispersion is a favoured approach because it helps to reduce sample preparation time.

Three consecutive reruns of each subsample were performed, allowing instrument and site variability to be accounted for. As a result of analysing only one subsample, inter-sample variability was not accounted for. However, a pilot study suggested inter-sample variability was low (Figure 3.18), supporting the chosen sampling methodology.

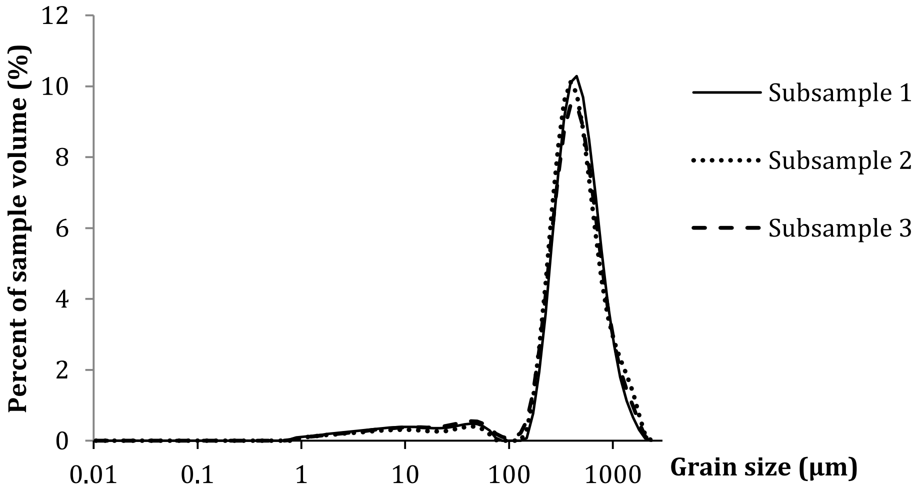


Figure 3.18. GSD results for three subsamples of sample W.1 (riffle). Inter-sample variability is low. Percentage values represent a mean average of three consecutive measurements for each subsample.

The measurement cell, pump and suspension system were manually flushed out with RO water between measurements to ensure that all sediment was removed from the machine and to minimise the risk of contaminating subsequent samples. Improper rinsing means results of the next sample may be contaminated (Chappell, 1998).

3.4.2 Data Analysis

The parameters used to describe the character of the sediment samples were calculated using equations from Folk and Ward (1957) and the computer program GRADISTAT (Blott and Pye, 2001). El-Sayed and Mostafa (2014) support the use of Folk and Ward's equations, suggesting that they are insensitive to GSDs which contain a large range of particle sizes in the tails of the distribution. Sambrook Smith *et al.'s* (1997) bimodality index was used to determine whether the GSDs are bimodal. Grain sizes were provided in μm. Results were log-transformed to phi units (Φ) prior to analysis (Equation 3.7).

$$\Phi = -log_2 D \qquad (3.7)$$

where *D* represents grain size in mm

Bimodality: There is little utility in describing a sediment sample with median grain size if the GSD is bimodal. The degree of bimodality, therefore, was estimated using Equation 3.8 (Sambrook Smith *et al.*, 1997: 1180).

$$B^* = |\Phi_2 - \Phi_1|(\frac{F_c}{F_f}) \qquad (3.8)$$

where B^* is the degree of bimodality, Φ_1 and Φ_2 are the coarse and fine modes, F_c is proportion in the coarse mode and F_f is the proportion in the fine mode. B^* values over 1.5 are bimodal. B^* values less than 1.5 represent a unimodal distribution.

Mean Grain Size: Mean grain size ($\bar{\Phi}$) for the samples was determined using Equation 3.9 (Folk and Ward, 1957).

$$\bar{\Phi} = \frac{\Phi_{16}+\Phi_{84}+\Phi_{50}}{3} \qquad (3.9)$$

where Φ_{84} is the grain size of the 84th percentile, Φ_{16} is the grain size of the 16th percentile, and Φ_{50} is the median grain size.

Sorting: The standard deviation (σ), or sorting, of the sample was estimated using Equation 3.10. This includes the tails of the sample in the calculation, and therefore gives an adequate representation of the sorting of bimodal distributions. Table 3.5 provides a qualitative description of sorting for different values of σ.

$$\sigma = \frac{\Phi_{84}-\Phi_{16}}{4} + \frac{\Phi_{95}-\Phi_{5}}{6.6} \qquad (3.10)$$

where Φ_{95} is the grain size of the 95th percentile and Φ_5 is the grain size of the 5th percentile.

Table 3.5. Logarithmic graphical measures of sorting (Folk and Ward, 1957).

σ	Description
σ < 0.35	Very well sorted
0.35 ≤ σ < 0.50	Well sorted
0.50 ≤ σ < 1.00	Moderately sorted
1.00 ≤ σ < 2.00	Poorly sorted
2.00 ≤ σ < 4.00	Very poorly sorted
4.00 ≤ σ	Extremely poorly sorted

Skewness: Equation 3.11 can be used to describe the asymmetry, or skewness (S_k), of the overall grain size distribution.

$$S_K = \frac{\Phi_{16}+\Phi_{84}-2\Phi_{50}}{2(\Phi_{84}-\Phi_{16})} + \frac{\Phi_{5}+\Phi_{95}-2\Phi_{50}}{2(\Phi_{95}-\Phi_{5})} \qquad (3.11)$$

Skewness values can range from +1.00 to -1.00, with a value of 0.00 representing perfect symmetry. Table 3.6 provides a description of the graphical measures of skewness for different values of *Sk*. A positive skew represents a distribution with a tail extending towards the fine end of the GSD, and a negative skew the coarse fraction.

Table 3.6. Logarithmic graphical measures of skewness (Folk and Ward, 1957).

S_k	**Description**
$-1.00 \leq S_k < -0.30$	Very negatively skewed
$-0.30 \leq S_k < -0.10$	Negatively skewed
$-0.10 \leq S_k < +0.10$	Symmetrical
$+0.10 \leq S_k < +0.30$	Positively skewed
$+0.30 \leq S_k < +1.00$	Very positively skewed

Kurtosis: Kurtosis (K) provides a description of the ratio of the sorting in the extremes of the distribution to the sorting in the central part of the distribution (Equation 3.12).

$$K = \frac{\Phi_{95} - \Phi_5}{2.44(\Phi_{75} - \Phi_{25})} \qquad (3.12)$$

The minimum K value possible is 0.41, and, although there is no upper limit, results for natural sediments are generally less than 8.00 (Folk and Ward, 1957). Table 3.7 provides a description of K values, with leptokurtic representing an excessively peaked distribution (i.e. the sediment is better sorted in the centre of the distribution than at its tails) and platykurtic a deficiently peaked distribution. A mesokurtic distribution represents a normal distribution with no excessive or deficient peak.

Table 3.7. Logarithmic graphical measures of Kurtosis (Folk and Ward, 1957).

K	**Description**
$K < 0.67$	Very platykurtic
$0.67 \leq K < 0.90$	Platykurtic
$0.90 \leq K < 1.11$	Mesokurtic
$1.11 \leq K < 1.50$	Leptokurtic
$1.50 \leq K < 3.00$	Very leptokurtic
$3.00 \leq K$	Extremely leptokurtic

3.5 Methodology 4: Organic matter content

The OM content of sediments affects macrophyte growth, which in turn affects sediment storage and retention (Barko and Smart, 1983). OM content also plays an important role in influencing the SOD and hence the DO content of both the sediment and the water column. It is therefore important to consider the OM fraction of FGS storage.

3.5.1 Laboratory Methodology

The OM content of the FGS was determined through loss on ignition (LOI), and estimated using Equation 3.13.

$$OM = \frac{M_{LOI1} - M_{LOI2}}{M_{LOI1}} \times 100 \qquad (3.13)$$

where *OM* is the organic matter content of the sample (%), M_{LOI1} is mass prior to combustion (g) and M_{LOI2} is mass after combustion(g).

LOI is a widely used, standard method for estimating OM content (Heiri *et al.*, 2001). The technique was used only on the fine-grained fraction for several reasons. Firstly, this size fraction is thought to contain most of the organic content (Sutherland, 1998). This is because in headwater streams many invertebrate species are specialised for breaking down coarse OM into fine OM. Invertebrates in small streams are especially able to process coarse OM into fine OM (Bilby and Likens, 1980). The bulk of the samples collected with the McNeil core sampler also make it impractical to determine the OM content of the whole sample, and taking a reliable subsample from an entire GSD is difficult (Napier, 1993). Furthermore, this matches with the operational definition of OM for soil scientists which states that "soil organic matter includes only those organic materials that accompany soil particles through a 2-mm sieve" (Sutherland, 1998: 157).

550°C is a much reported figure for use with LOI studies. However, numerous other temperatures have been reported. These range from 375°C to 800°C (Bisutti *et al.*, 2004). 550°C has been suggested as a potentially inappropriate temperature for determining the OM content of clay-rich soils (John, 2004; Grove and Bilotta, 2014). This is due to mineral dewatering. Rather than OM combusting, hygroscopic and intercrystlalline water is burnt off. However, lowering the ignition temperature could result in combustion not fully completing (John, 2004). A pilot study to determine the GSD suggested that samples are dominated by sand and silt-sized particles, and as a result, 550°C was deemed an appropriate LOI temperature for this study.

The recommended combustion time for LOI also varies between studies. Carling (1983) recommends one hour in a muffle furnace. Acornley and Sear (1999) suggest five hours is appropriate. Heiri *et al.* (2001) recommend four hours, suggesting that the combustion reaction plateaus after an initially high weight loss in the first two hours. John (2004) highlights the need to conduct a pilot study to determine the procedures which best match the sediment. The results of a pilot study conducted to determine the length of ignition showed that the combustion reaction plateaued after five hours of burning (Figure 3.19). Therefore, five hours was deemed an appropriate combustion time for this study.

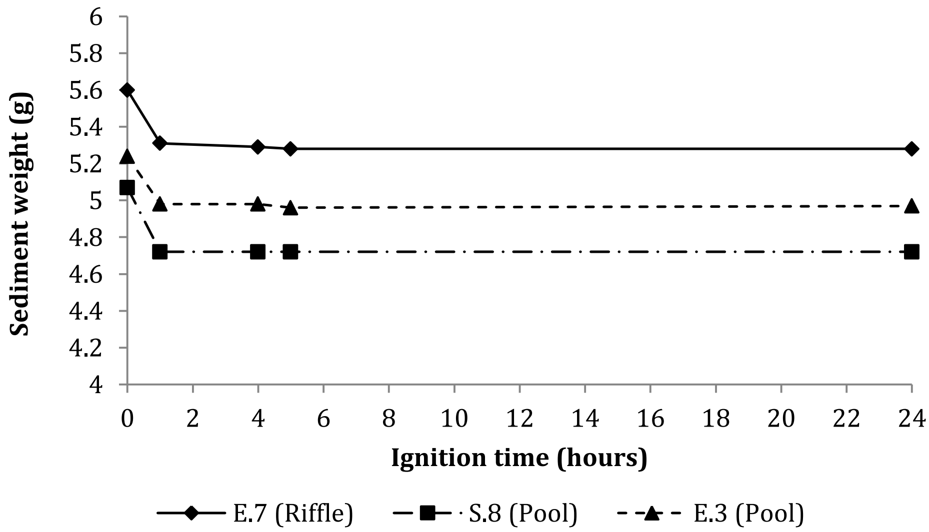


Figure 3.19. Results of the pilot study conducted to determine LOI duration. Sample weight remained constant after five hours of ignition.

Three subsamples of approximately 5g were tested for each sample. This helped to ensure results are representative of the OM content of the sample (Napier, 1993) and that there was no bias in the selected subsamples. Subsamples were taken using a riffle box. LOI was conducted at 550°C for five hours. Samples were cooled in a desiccator before weighing to prevent absorption of atmospheric moisture during the cooling phase.

3.5.2 Data Analysis

The statistical techniques used in section 3.2.3 were used to analyse the results obtained from LOI.

Chapter 4 – Spatial Variation in Fine-Grained Sediment Storage

4.1 Introduction

The quantity of sediment stored on a stream bed is a key determinant of the quality of the stream environment and its associated ecological status. This chapter presents results of the quantity of FGS stored on the stream beds of Stonton Brook, Eye Brook and the Upper Welland.

This chapter is split into four sections. In the first section, summary statistics describing the quantity of FGS stored in the study catchments are presented. These are then compared to results from similar studies, in order to provide context.

In the second section, the differences in the amount of FGS stored on the stream bed are analysed. Analyses were conducted at a multi-scale level, allowing a thorough, holistic analysis of FGS storage (Rickson, 2006; Walling and Collins, 2008). The difference in FGS storage between riffles and pools was investigated, as well as differences within and between catchments.

The third section looks at differences in the bulk density of the stream bed sediment, and the implications that these differences have for estimating the proportions of FGS stored on the stream bed. As with the previous section, analyses were multi-scaled and considered differences between riffles and pools and inter- and intra-catchment variations.

The fourth section considers the amount of sediment which is remobilisable when the water column is disturbed and, thus, how much is likely to be available for transport during flood events. Differences in the remobilisable fraction between riffles and pools, as well as within and between the study catchments were estimated.

4.2 Fine-grained sediment storage

4.2.1 Fine-grained sediment storage in UK rivers

The range and mean values of FGS storage measured in the study catchments is shown in Table 4.1. The data are broken down into different size fractions (≤2mm, ≤150μm and ≤63μm). Although FGS storage in the study catchments is within the range of FGS storage reported in other UK rivers, average storage is generally higher than elsewhere (Table 4.1). Values display similar variability to results reported in other catchments. As Table 2.2 suggests, other studies have reported storage of 10 – 13360 g m^{-2} cm^{-1} for FGS less than 2mm. Values ranging from 34 to 1848 g m^{-2} cm^{-1} and 8 to 478 g m^{-2} cm^{-1} have been reported for sediment ≤150μm and ≤63μm respectively.

Table 4.1. The range of at-a-site channel bed storage, and the mean average at-a-site storage in the study catchments for sediment a) ≤2mm, b) ≤150μm and c) ≤63μm.

a)

Stream	FGS Storage (g m^{-2} cm^{-1})	
	Range	Mean
Upper Welland	4254 – 5700	4977 ±511
Stonton Brook	2841 – 6115	4626 ±342
Eye Brook	4181 – 8560	5710 ±437

b)

Stream	FGS Storage (g m^{-2} cm^{-1})	
	Range	Mean
Upper Welland	456 – 1027	742 ±202
Stonton Brook	666 – 2141	1536 ±181
Eye Brook	1148 – 2293	1712 ±128

c)

Stream	FGS Storage (g m^{-2} cm^{-1})	
	Range	Mean
Upper Welland	75 – 396	236 ±113
Stonton Brook	301 – 1573	924 ±159
Eye Brook	359 – 911	590 ±60

4.2.2 Average Storage

The average amount of FGS stored in pools (5678 ±343 g m^{-2} cm^{-1}) is higher than the amount of FGS stored in riffles (4616 ±370 g m^{-2} cm^{-1}) (Figure 4.1a). The lowest FGS storage was recorded in a riffle (2002 g m^{-2} cm^{-1}). Minimum FGS storage in a pool was 2779 g m^{-2} cm^{-1}. The highest recorded FGS storage was at a riffle site, and was 8702 g m^{-2} cm^{-1}. Maximum pool storage was 8417 g m^{-2} cm^{-1} (Figure 4.1b). Differences in the amount of FGS stored in riffles and pools are statistically significant according to the Mann-Whitney U Test ($p = 0.034$). Differences in storage between riffle-pool pairs are also statistically significant according to the Wilcoxon Signed-Rank Test ($p = 0.014$).

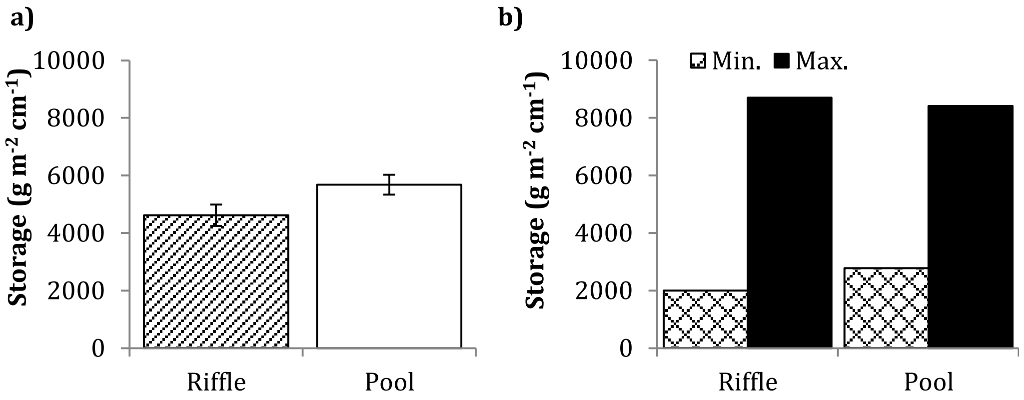


Figure 4.1. a) Average FGS storage in riffles and pools of the study catchments. b) Minimum and maximum FGS storage in riffles and pools of the study catchments. Error bars represent SEM.

4.2.3 Inter-catchment variability

Average FGS storage appears to be higher in Eye Brook (5710 ±437 g m^{-2} cm^{-1}) than Stonton Brook (4626 ±342 g m^{-2} cm^{-1}) (Figure 4.2). Average FGS storage in riffles was also greater in Eye Brook (5145 ±606 g m^{-2} cm^{-1}) than Stonton Brook (4041 ±497 g m^{-2} cm^{-1}). Average FGS storage in pools of Eye Brook and Stonton Brook were 6274 ±491 g m^{-2} cm^{-1} and 5212 ±474 g m^{-2} cm^{-1}, respectively. However, none of these differences between catchments were statistically significant ($p>0.05$).

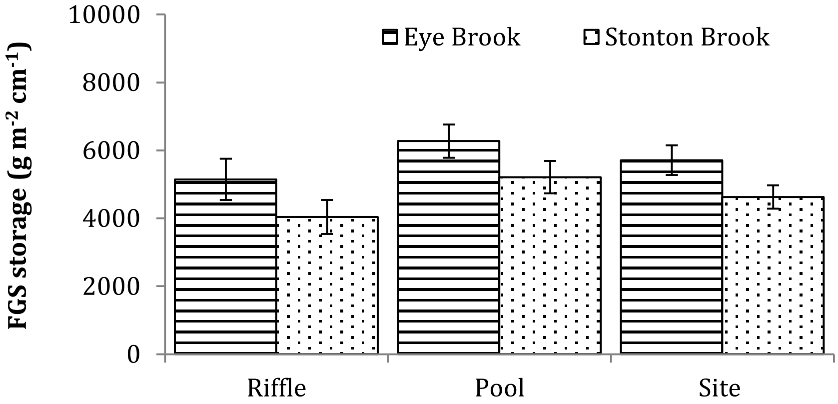


Figure 4.2. Mean average FGS storage in riffles, pools and at-a-site in the Eye Brook and Stonton Brook catchments. Differences between catchments are not statistically significant. Error bars represent SEM.

4.2.4 Intra-catchment variability

Differences in FGS storage within catchments is shown in Figure 4.3. FGS storage is higher in pools in Eye Brook than in riffles. At 13 out of 18 sites, FGS storage in pools was higher than in riffles (five out of eight sites in the Eye Brook, seven out of eight in Stonton Brook and one out of two in the Upper Welland). Storage is higher in riffles than at pools for one site in the Upper Welland, one in Stonton Brook and three in Eye Brook. Differences in storage between riffles and pools within catchments are not statistically significant according to the Mann-Whitney U test ($p>0.05$).

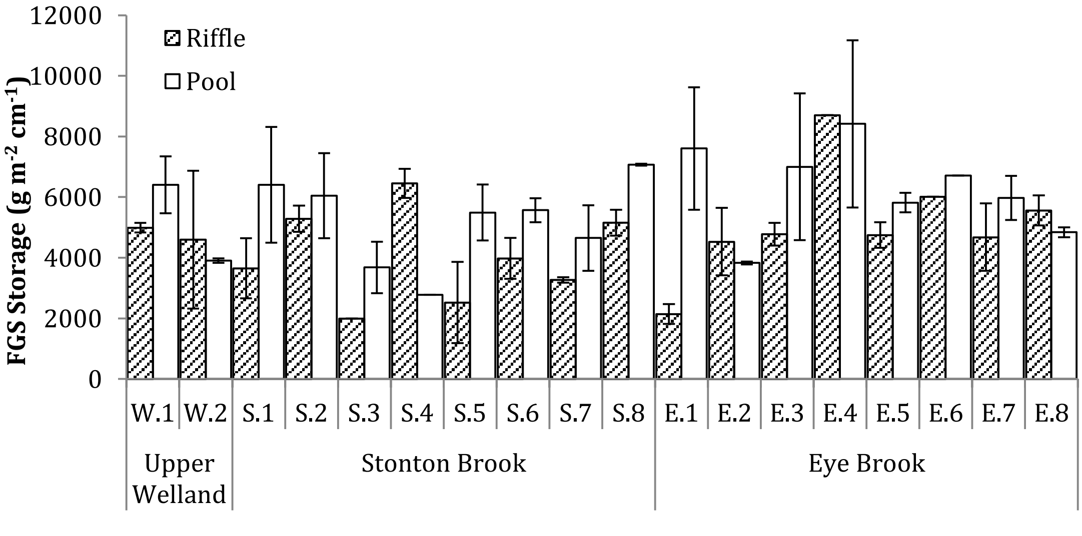


Figure 4.3. FGS storage at riffle and pool sites in the study catchments. FGS storage is greater in pools than riffles. Error bars represent SEM.

An assessment was made of possible relationships between the amount of FGS stored at individual sites and the characteristics of the contributing catchments (e.g. upstream catchment area and channel length). Channel length and catchment area are strongly correlated ($p<0.01$) (Figure 4.4), and drainage density ranged from 0.77 to 1.48 km^{-1} (average = 1.17±0.043 km^{-1}). At-a-site FGS storage and FGS storage in riffles and pools was not correlated with upstream channel length or catchment area ($p>0.05$) (Figure 4.5). This suggests that an increase in the total load of FGS generated in the contributing catchment area is accompanied by an equivalent increase in transport capacity (due to an increase in discharge).

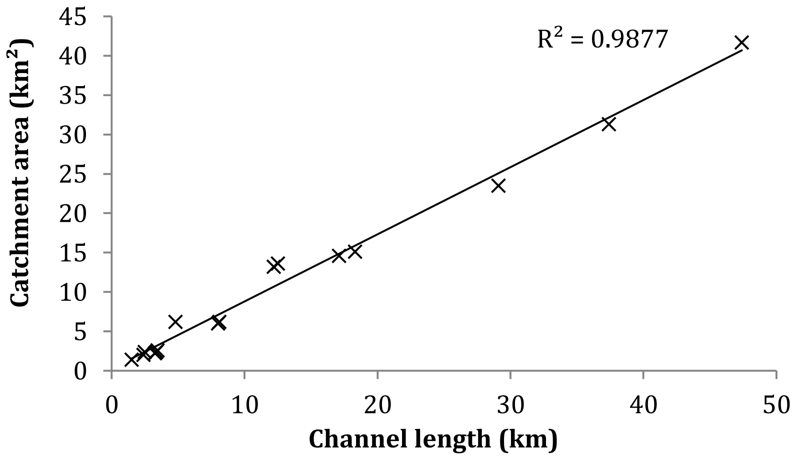


Figure 4.4. Channel length versus catchment area. Channel length and catchment area are correlated.

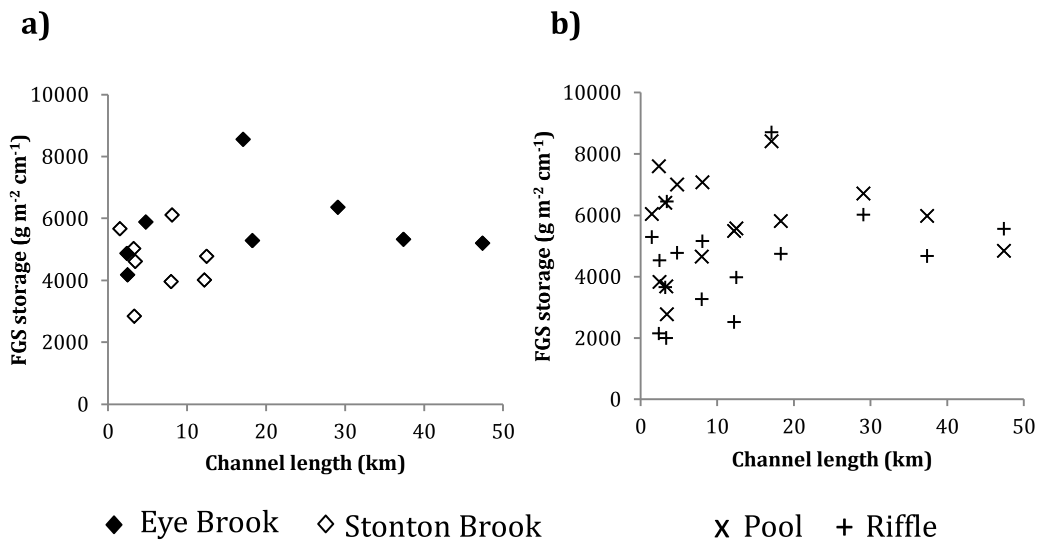


Figure 4.5. Downstream variation in FGS storage a) at-a-site and b) at individual pools and riffles.

4.3 Proportion of fine-grained sediment

Although knowledge of the total amount of FGS stored is useful, it provides little context on the extent of the FGS problem. Information on percent fines is useful for determining the potential ecological implications of FGS loads.

4.3.1 Fine-grained sediment storage and percent fines

As FGS storage increases, the proportion of the stream bed that is FGS (percent fines ≤2mm) also increases ($p<0.01$) (Figure 4.6). However, there are some anomalies to this trend. An increase in FGS storage is not always associated with a higher percent of fines. Notable outliers are sites S.6 and E.5 (Table 4.2).

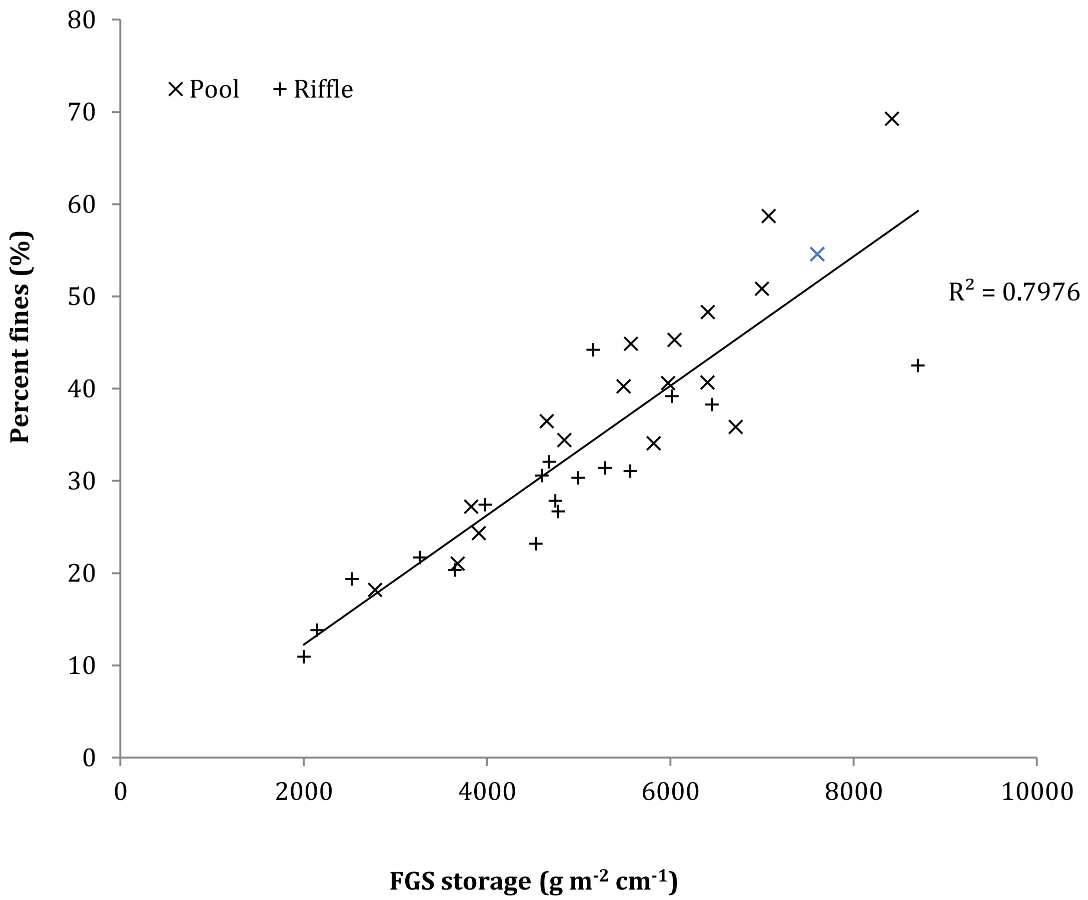


Figure 4.6. Relationship between FGS storage and percent fines for riffles and pools in the study catchments.

Table 4.2. Comparison of FGS storage and percent fines. Increases in FGS storage are not always associated with an increase in percent fines. Notable outliers in riffles include E.3, E.7, E.8 and S.3. Notable outliers in pools include E.6 and S.6.

Field site	FGS storage in riffles				Field site	FGS storage in pools			
	$g\ m^{-2}\ cm^{-1}$	Rank	%	Rank		$g\ m^{-2}\ cm^{-1}$	Rank	%	Rank
S.3	2002	*1*	11	*1*	**S.4**	2779	*1*	18	*1*
E.1	2146	*2*	14	*2*	**S.3**	3680	*2*	21	*2*
S.5	2528	*3*	19	*3*	**E.2**	3828	*3*	27	*4*
S.7	3268	*4*	22	*5*	**W.2**	3909	*4*	24	*3*
S.1	3649	*5*	20	*4*	**S.7**	4650	*5*	36	*7*
S.6	3979	*6*	27	*7*	**E.8**	4842	*6*	34	*5*
E.2	4533	*7*	23	*6*	**S.5**	5490	*7*	40	*9*
W.2	4598	*8*	31	*11*	**S.6**	5571	*8*	45	*12*
E.7	4679	*9*	32	*14*	**E.5**	5818	*9*	34	*6*
E.5	4745	*10*	28	*9*	**E.7**	5975	*10*	41	*10*
E.3	4777	*11*	27	*7*	**S.2**	6047	*11*	45	*12*
W.1	4995	*12*	30	*10*	**W.1**	6405	*12*	41	*10*
S.8	5157	*13*	44	*18*	**S.1**	6408	*13*	48	*14*
S.2	5287	*14*	31	*11*	**E.6**	6712	*14*	36	*7*
E.8	5564	*15*	31	*11*	**E.3**	6999	*15*	51	*15*
E.6	6019	*16*	39	*16*	**S.8**	7072	*16*	59	*17*
S.4	6455	*17*	38	*15*	**E.1**	7603	*17*	55	*16*
E.4	8702	*18*	42	*17*	**E.4**	8417	*18*	69	*18*

4.3.2 Average Storage

As with the amount of FGS stored, the proportion of bed sediment that is fine is generally greater in pools than riffles ($p = 0.0062$) (Figure 4.7). Of the pool and riffle pairs examined, 15 out of 18 had a higher fraction of fines in sediment sampled from pools than riffles. Percent fines values range from 11% to 69% (a range of 58%). The average percent fines in riffles is 28±2%. The average percent fines in pools is 40±3%.

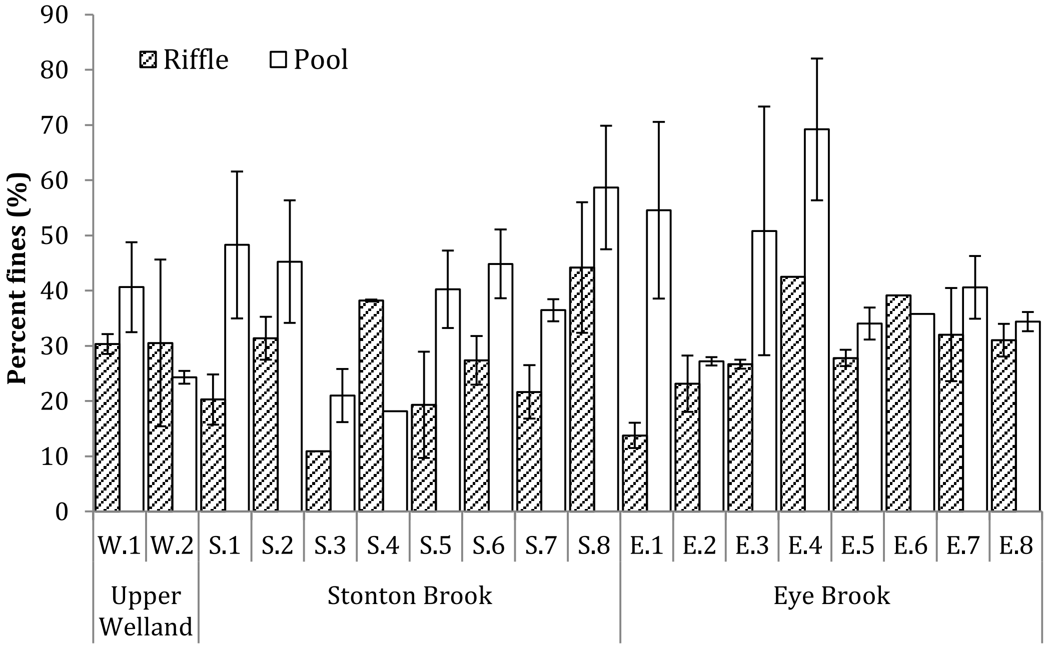


Figure 4.7. Proportion of sediment stored on the stream bed at riffle and pool sites that is ≤2mm. Error bars represent SEM.

4.3.3 Inter-catchment variability

The average percent fines was 36±3% in riffles sampled from the Eye Brook, 33±3% in Stonton Brook and 31±3% in the Upper Welland. The average percent fines in pools is 43±5% in the Eye Brook and 39±5% in the Stonton Brook. The average percent fines in the Upper Welland for pools is 32±6%. Differences in the fraction of fines stored in Stonton Brook and Eye Brook were not statistically significant ($p>0.05$), according to the Mann-Whitney U test.

4.3.4 Intra-catchment variability

The Wilcoxon signed-rank test showed that there was no significant difference between the percent fines in riffles and pools in Stonton Brook ($p = 0.066$), i.e. in individual reaches there is no difference between the pool and the riffle. However, the Mann-Whitney U test suggested there was a statistically significant difference in the percent fines in riffles and pools overall ($p = 0.042$).

4.3.5 Bulk density and the proportion of fine-grained sediment

Sediment bulk density is, on average, higher in riffles than pools (Figure 4.8). For 14 out of 18 sites, sediment bulk density was higher in riffles than pools. Average bulk density in riffles is 1.64±0.050 g cm^{-3} and in pools it was 1.44±0.044 g cm^{-3}. Bulk density is highest in Eye Brook, and lowest in the Upper Welland. Average bulk densities for the bed material of the Upper Welland, Stonton Brook and Eye Brook are 1.44±0.087 g cm^{-3}, 1.48±0.053 g cm^{-3} and 1.63±0.055 g cm^{-3}, respectively. A Mann-Whitney U test suggests that differences in bulk densities between catchments are significant ($p = 0.0062$). The paired Wilcoxon signed-rank test showed that bulk density differences between riffles and pools in the Eye Brook is also statistically significant ($p = 0.045$). However, the difference is not statistically significant when unpaired ($p = 0.270$). The opposite is true in Stonton Brook (paired: $p = 0.101$; unpaired: $p = 0.027$).

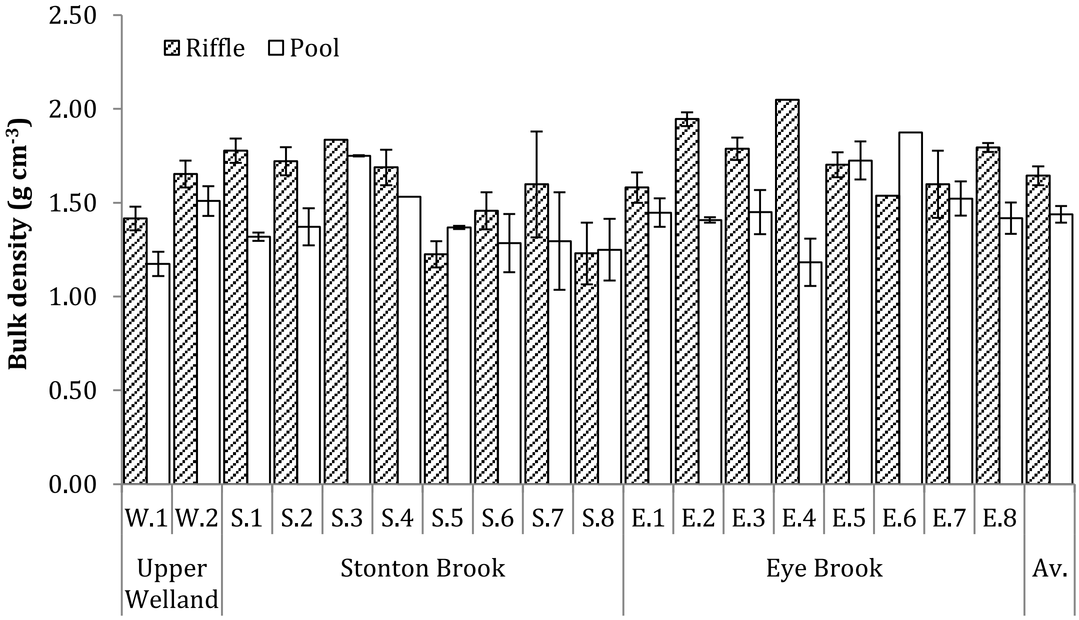


Figure 4.8. Bulk density of the bed material at riffle and pool sites. Bulk density is greater in riffles than pools. Error bars represent SEM.

Although there was no relationship between bulk density and FGS storage according to the Mann Whitney U-test ($p = 0.53$) (Figure 4.9a), there was a significant inverse relationship between bulk density and percent fines ($p = 0.003$)

(Figure 4.9b). It is thought that differences in the bulk density of stream bed material results in FGS accounting for different proportions of the bed material.

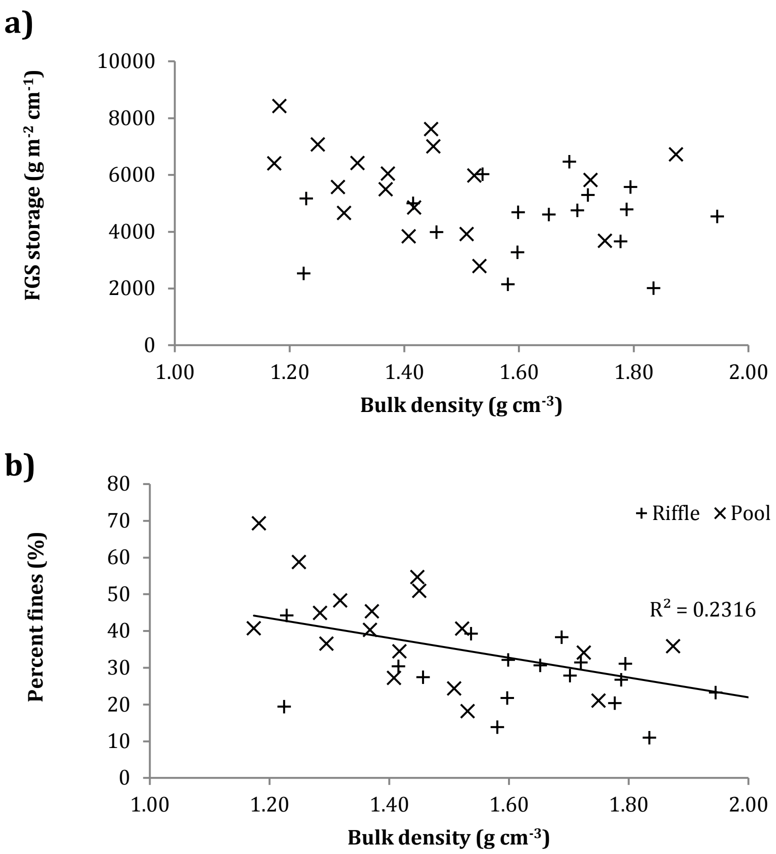


Figure 4.9. The relationship between average bulk density and a) FGS storage and b) percent fines in samples collected from riffles and pools.

4.4 Surficial storage of fine-grained sediment

Estimates of FGS storage collected using the McNeil corer include sediment settled on the bed surface that could be remobilised into suspension during high flows. Results from the water column disturbance test (section 3.3) provided estimates of the remobilisable fraction of FGS storage.

4.4.1 Surficial storage in UK rivers

The remobilisable fraction of surficial FGS storage in the study catchments is greater in pools than in riffles. Surficial storage of remobilisable fines in the study catchments is lower than levels of surficial storage reported in other UK rivers

(Table 4.3). Results collected in the study catchments display similar levels of variability to those observed elsewhere.

Table 4.3. a) Estimates of surficial storage of remobilisable FGS in the study catchments. b) Values of surficial storage reported in the literature. Individual values represent the mean. Error values represent the SEM.

a)

	FGS Storage (g m^{-2})			
Stream	**Riffle**		**Pool**	
	Range	**Mean**	**Range**	**Mean**
Upper Welland	20 – 56	38±13	234 – 499	367±94
Stonton Brook	5 – 170	67±16	64 – 630	244±68
Eye Brook	1 – 132	26 ±15	58 – 546	213±61

b)

Author	**Definition of fines**	**River**	**Catchment area (km^2)**	**FGS Storage (g m^{-2})**
Owens *et al*. (1999)	<150µm	Tweed	4390	60 – 780 410
Walling *et al*. (1998)	<150µm	Ouse	3315	50 – 4970
		Nidd	516	600 – 6250
		Swale	1446	70 – 3480
		Wharfe	818	410 – 1980
Duerdoth *et al*. (2015)	<2mm	Various catchments in England		2 - 20000

4.4.2 Average Storage

The amount of fines remobilised when the water column was disturbed was higher in pools (mean = 244±43 g m^{-2}) than in riffles (mean = 45±11 g m^{-2}) by approximately an order of magnitude (Figure 4.10a). The lowest surficial storage was recorded in a riffle at just 1 g m^{-2} (Figure 4.10b). The minimum surficial storage in a pool was 58 g m^{-2}. The differences in the amount of remobilisable FGS stored on the stream bed surface in riffles and pools was statistically significant, according to the Mann-Whitney U test ($p < 0.001$).

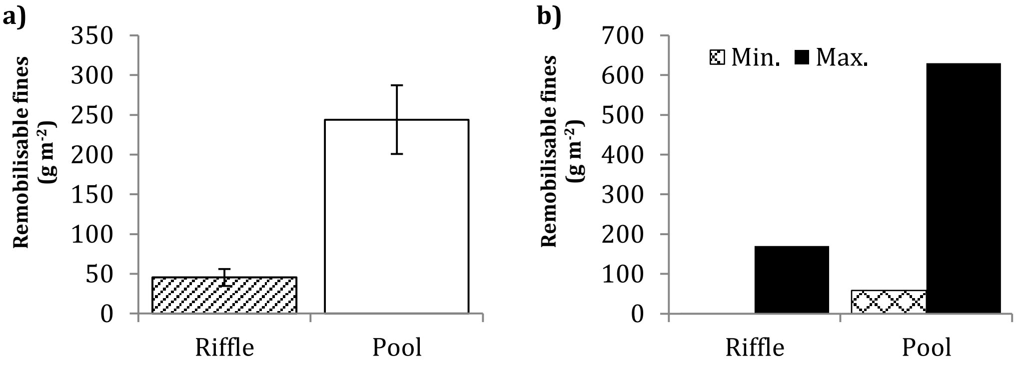


Figure 4.10. a) Average surficial remobilisable FGS storage in riffles and pools of the study catchments. b) Minimum and maximum surficial fine-grained sediment storage in riffles and pools of the study catchments. Minimum riffle storage was 1 g m^{-2}. Error bars represent SEM.

4.4.3 Inter-catchment variability

The average storage of remobilisable FGS, broken down by catchment and morphology, is shown in Figure 4.11. In general, the amount of remobilisable material generated in the Stonton brook (156±39 g m^{-2}) was higher than in the Eye Brook (119±35 g m^{-2}). This was also the case for riffle and pool units. Average surficial storage for riffles in the Eye Brook and Stonton Brook was 26±15 g m^{-2} and 67±16 g m^{-2}, respectively. This difference is statistically significant ($p = 0.041$), according to the Mann-Whitney U test. Average surficial FGS storage in pools was also higher in the Stonton Brook (244±68 g m^{-2}) compared with the Eye Brook (213±61 g m^{-2}), but this was not statistically significant, according to the Mann-Whitney U test ($p = 0.9581$).

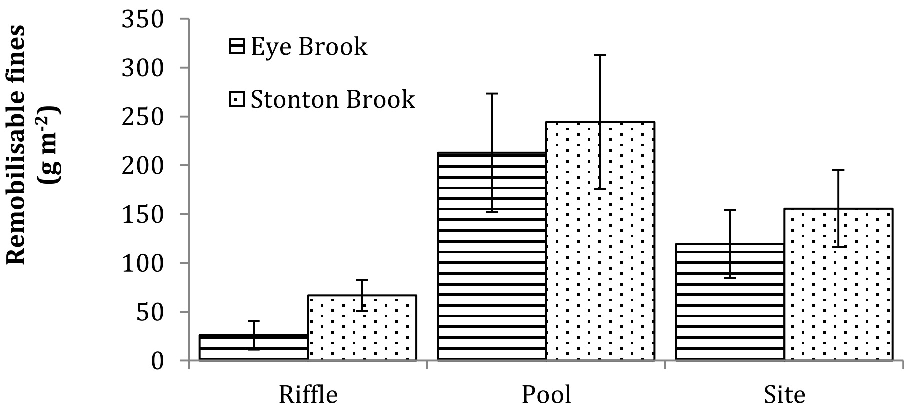


Figure 4.11. Mean average surficial storage in riffles, pools and at-a-site in the Eye Brook and Stonton Brook catchments. Error bars represent SEM.

4.4.4 Intra-catchment variability

Remobilisable sediment storage also varies within catchments. Surficial storage is generally higher in pools than in riffles (Figure 4.11), but it is also generally higher in Stonton Brook compared with Eye Brook. In Stonton Brook, seven out of eight pools had a higher amount of remobilisable fines than riffles. In the Upper Welland, pools at both sites had much more remobilisable fines compared to riffles. Differences in surficial storage between riffles and pools within catchments are statistically significant ($p<0.05$ for both the Mann-Whitney U test and Wilcoxon signed-rank test [unpaired and paired tests]).

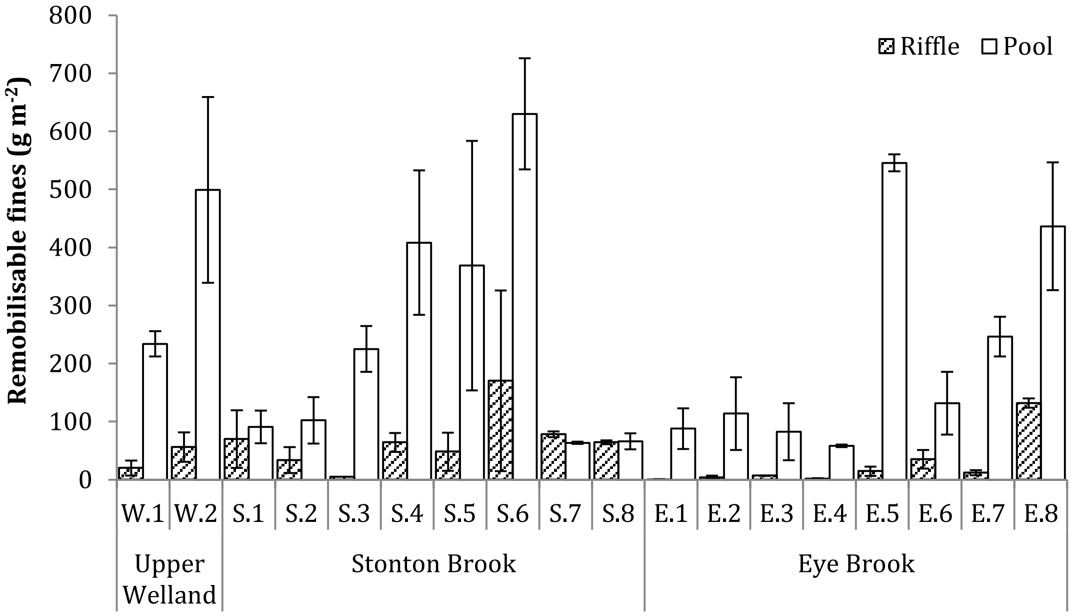


Figure 4.12. Surficial storage of remobilisable FGS at individual riffle and pool sites. Surficial storage of fines is greater in pools than riffles in all sites except S.7. Error bars represent SEM.

Remobilisable FGS in riffles and pools of Stonton Brook is not correlated with upstream channel length ($p>0.05$) (Figure 4.13a), but in the Eye Brook there did appear to be a significant relationship with the amount of remobilisable fines per unit area and the upstream catchment area, as represented by channel length ($p<0.05$) (Figure 4.13b). The increase in surficial storage with distance downstream is polynomial in pools, with a rapid increase in remobilisable fines with increasing distance downstream beyond about 30km.

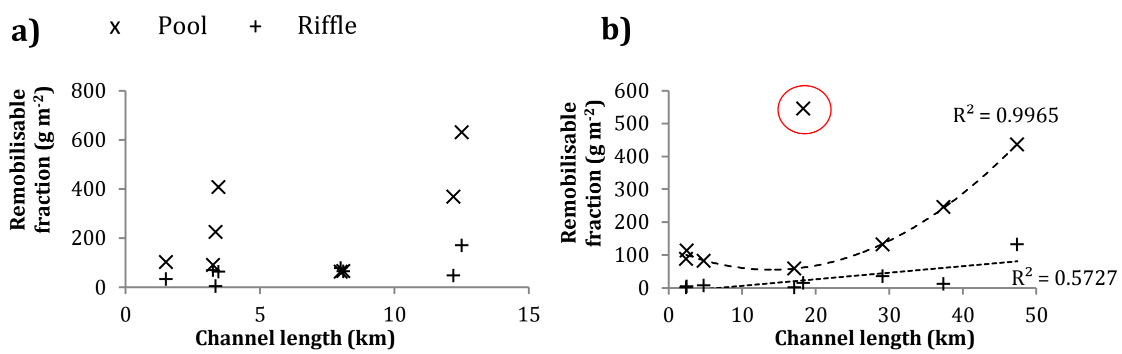


Figure 4.13. Downstream variation in FGS storage in a) Stonton Brook and b) the Eye Brook. The dashed lines show the best fit relationships for Eye Brook, which was linear in the case of riffles and polynomial in the case of pools. The outlier highlighted in the Eye Brook was ignored.

4.4.5 Surficial storage as a proportion of total FGS storage

In general, the remobilisable sediment represents a relatively small proportion of total FGS storage (Figure 4.14). On average, surficial storage represents a greater proportion of total FGS storage in pools (average = 5±0.9%) than riffles (average = 2±0.6%), although the fraction of remobilisable fines was slightly higher for riffles than pools in four out of 18 sites (all in the Stonton Brook). Surficial remobilisable storage was 0.5±0.3% in the Eye Brook and 3±1% in the Stonton Brook. This difference was statistically significant, according to the Mann-Whitney U test ($p = 0.018$). Although pools in Stonton Brook (5±1%) also have a greater proportion of surficial fines than in the Eye Brook (4±1%), this difference was not statistically significant ($p = 0.189$). These results suggest that FGS is less mobile in riffles than pools, and that interstitial deposition is greater in riffles of the Eye Brook compared to riffles in the Stonton Brook.

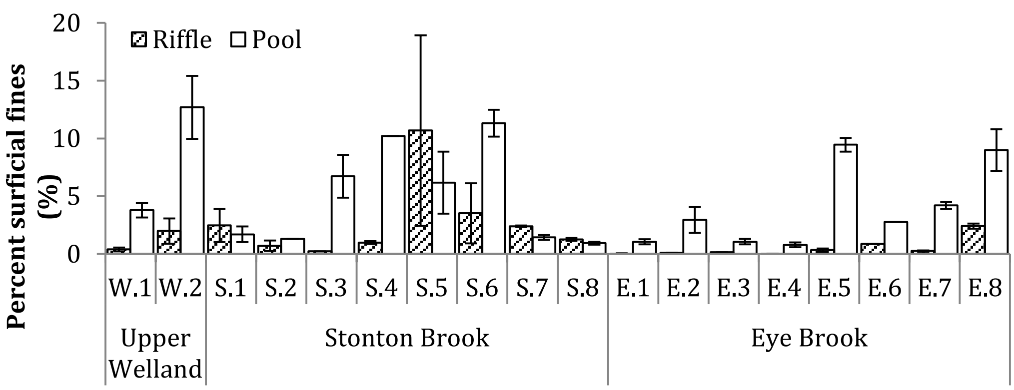


Figure 4.14. Surficial storage of FGS at riffle and pool sites as a percentage of total FGS storage. Error bars represent SEM.

Chapter 5 – Fine-grained Sediment Character

5.1 Introduction

The quality of, as well as the quantity of sediment stored on the stream bed can play an important role in determining the quality of a river system and its associated ecology. This chapter therefore investigates the character of the FGS sampled from the stream bed.

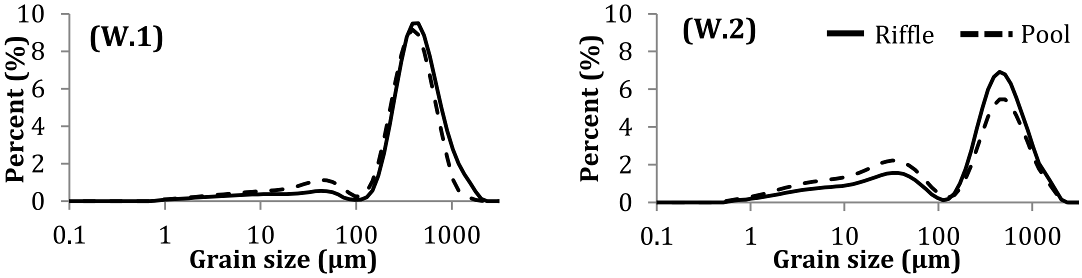
This chapter is split into two sections. In the first section, grain size characteristics of the FGS stored in the study catchments are presented. Grain size can affect the sorption capacity of the sediment (in terms of both cation exchange and by hydrophobic interactions by way of increased OM content), and thus their likelihood of being associated with nutrients and contaminants. The OM content of FGS has important implications for ecological quality, particularly in terms of the DO content (Petticrew and Arocena, 2003). Therefore, in the second section, the OM content of the FGS is described and analysed. Analysis is multi-scaled. Differences in the OM content in riffles and pools, and differences within and between catchments are presented.

5.2. Grain Size Distribution

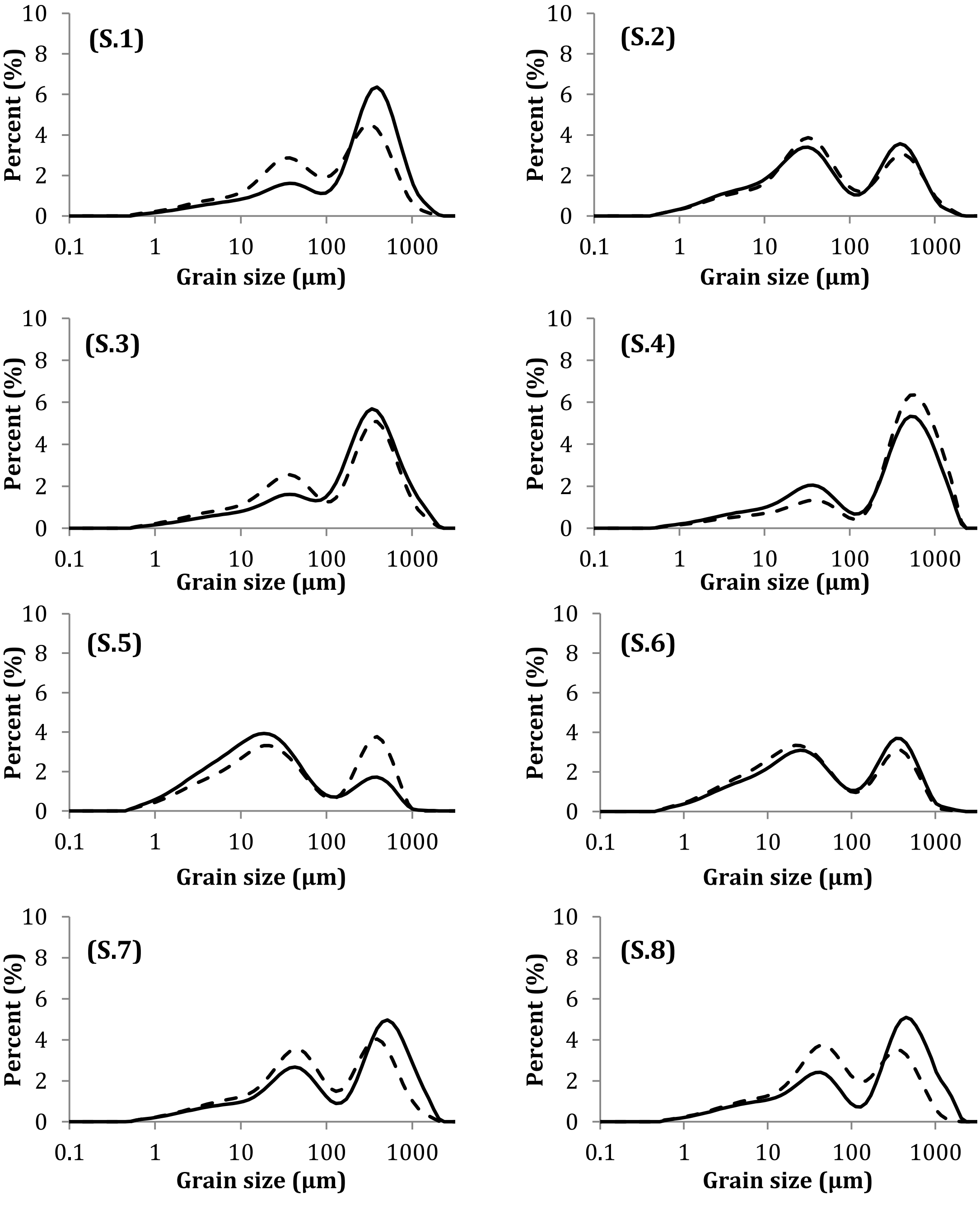
5.2.1 Index of bimodality

GSDs of the McNeil core samples collected from riffles and pools at the 18 sites are shown in Figure 5.1. All sample sites have two distinct peaks in percent by volume values, representing bimodal GSDs, although the bimodality index for some suggests they are unimodal (Table 5.1). For the bimodal distributions, the peak modes are dominated by medium sand and very coarse silt (100 to 1000μm). All bimodal distributions in the Eye Brook have a coarse peak mode. The riffle at site W.2 also has a peak mode at the coarse end of its distribution. The peak mode of the bimodal distributions in the Stonton Brook is in the coarser fraction for four pools and four riffles.

a) Upper Welland



b) Stonton Brook



c) Eye Brook

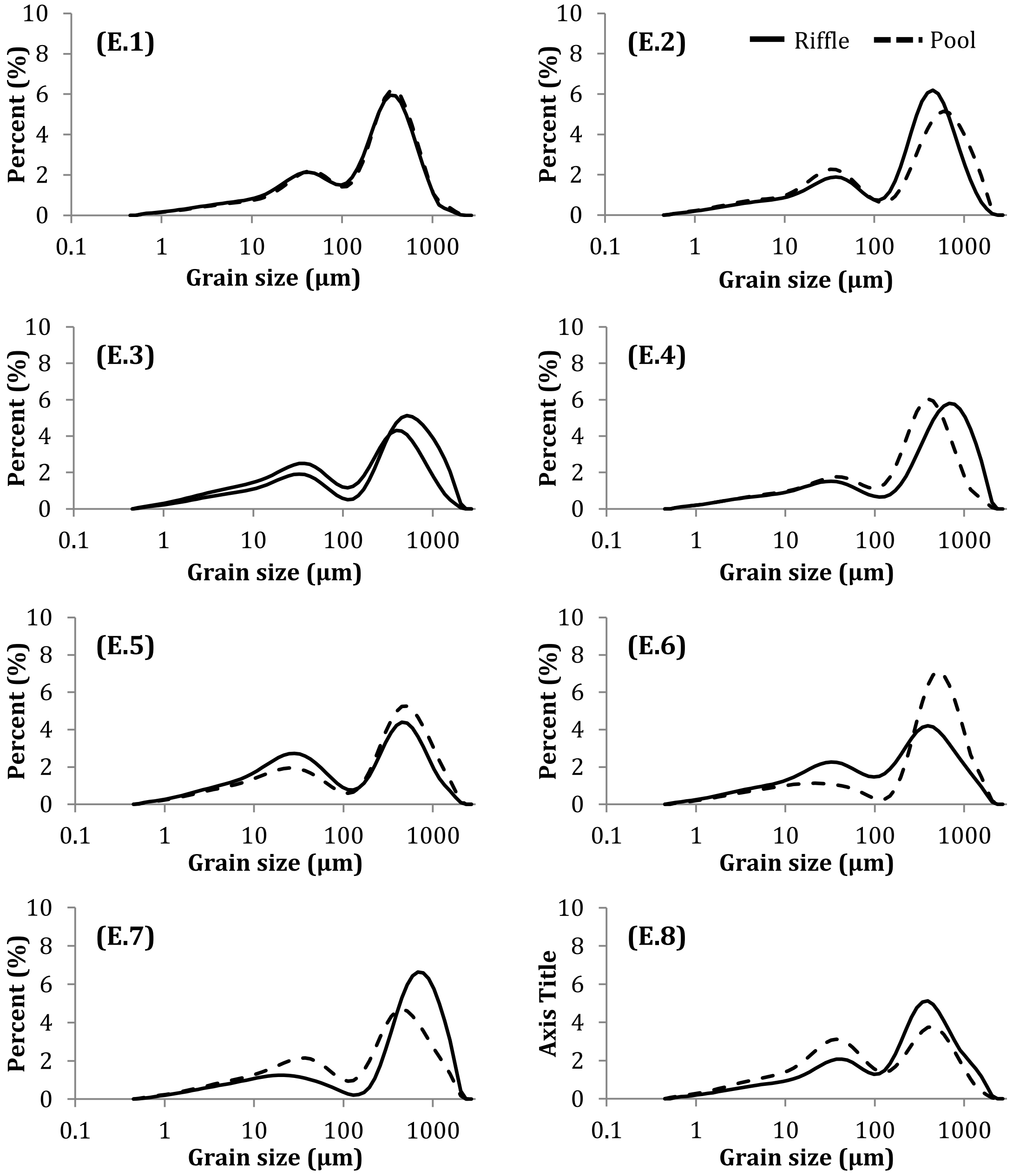


Figure 5.1. Grain-size distributions for the FGS stored on the stream bed of riffles and pools in a) the Upper Welland, b) Stonton Brook and c) Eye Brook. Percent values are percent by volume. Dashed lines denote pools, solid lines denote riffles.

Table 5.1. Modal grain sizes and bimodality index values for riffles and pools at the study site. Bold modal grain sizes represent the peak mode. B* values >1.5 are considered bimodal distributions.

Field site		Riffle				Pool			
		Modal grain size (Φ)		B*	Bimodal/ Unimodal	Modal grain size (Φ)		B*	Bimodal/ Unimodal
		Coarse	Fine			Coarse	Fine		
Upper Welland	W.1	**1.06**	-	-	-	**1.26**	-	-	-
	W.2	**1.06**	4.85	1.54	Bimodal	**1.06**	4.85	0.86	Unimodal
Stonton Brook	S.1	**1.26**	4.65	0.88	Unimodal	**1.46**	4.65	1.98	Bimodal
	S.2	**1.26**	4.85	3.41	Bimodal	1.26	**4.85**	4.59	Bimodal
	S.3	**1.46**	4.65	0.91	Unimodal	**1.26**	4.65	1.75	Bimodal
	S.4	**0.86**	4.65	1.43	Unimodal	**0.66**	4.65	0.85	Unimodal
	S.5	1.26	**5.64**	10.52	Bimodal	**1.26**	5.45	3.81	Bimodal
	S.6	**1.46**	5.25	3.15	Bimodal	1.46	**5.45**	4.29	Bimodal
	S.7	**0.86**	4.45	1.94	Bimodal	**1.26**	4.45	2.02	Bimodal
	S.8	**1.06**	4.45	1.57	Bimodal	1.46	**4.45**	3.20	Bimodal
Eye Brook	E.1	**1.46**	4.45	1.05	Unimodal	**1.26**	4.45	1.18	Unimodal
	E.2	**1.06**	4.65	1.07	Unimodal	**0.66**	4.85	1.87	Bimodal
	E.3	**0.86**	4.85	1.49	Unimodal	**1.26**	4.65	1.93	Bimodal
	E.4	**0.46**	4.85	1.16	Unimodal	**1.26**	4.65	1.00	Unimodal
	E.5	**1.06**	5.05	2.48	Bimodal	**0.86**	5.25	1.61	Bimodal
	E.6	**1.26**	4.85	1.97	Bimodal	**0.86**	5.45	0.72	Unimodal
	E.7	**0.46**	5.44	0.93	Unimodal	**1.06**	4.65	1.60	Bimodal
	E.8	**1.26**	4.45	1.34	Unimodal	**1.06**	4.65	2.92	Bimodal

5.2.2 Mean grain size

Describing bimodal distributions with the median grain size is of little value (Sambrook Smith *et al.*, 1997) because the D_{50} will often represent a size class with little sediment in it, especially if it falls between two modes. Mean grain size ($\overline{D}$), therefore, provides a better summary statistic for the FGS. Figure 5.2a shows the mean grain size for the study sites. In general, the mean grain size of the FGS was coarser in riffles than pools (11/18 sites) (Figure 5.2b). This is also reflected in the average mean grain size for all pools ($\overline{D}$=129±15μm; fine sand) and riffles ($\overline{D}$=162±20μm; fine sand). This difference is statistically significant, according to the Mann-Whitney U test (p = 0.0332). Apparent differences in mean grain size were also observed between catchments (Figure 5.2c). Mean average $\overline{D}$ was 250±56μm (fine sand), 109±16μm (very fine sand) and 156±11μm (fine sand) in the Upper Welland, Stonton Brook and the Eye Brook respectively. Differences in mean grain sizes of the Eye Brook and Stonton Brook are statistically significant according to the Mann-Whitney U test (p = 0.0332).

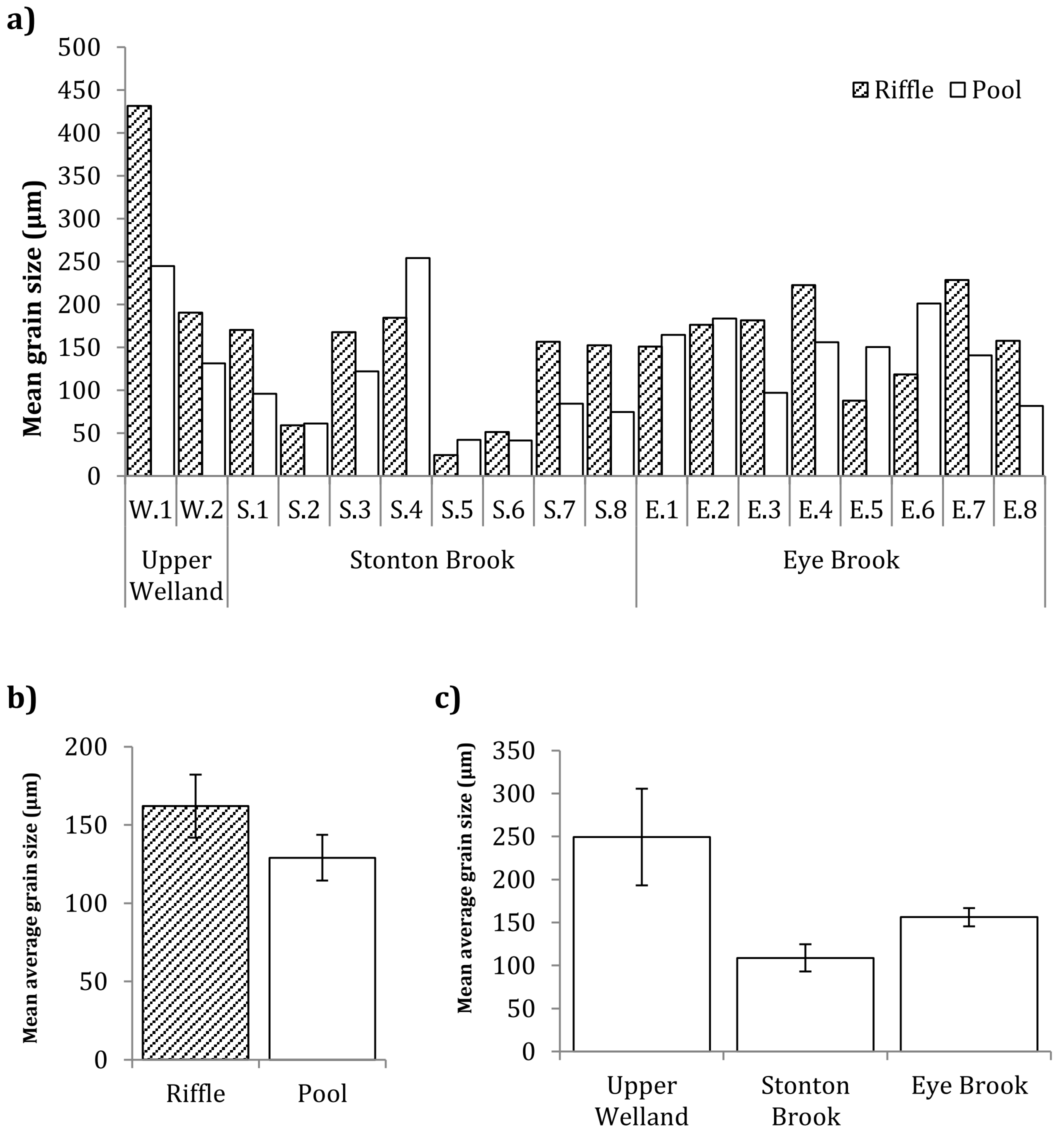


Figure 5.2. a) Mean grain size ($\overline{D}$) for riffles and pools of the study catchment, as estimated using Folk and Ward's equation. b) Mean average $\overline{D}$ for riffles and pools. c) Mean average $\overline{D}$ for the study catchments. Error bars represent SEM.

5.2.3 Fine-grained sediment composition

The FGS sampled from both riffles and pools in the study catchments tended to be dominated by sand particles (Figure 5.3a), with an average percent sand in riffles of 65±4% and 61±3% in pools. Mean average percent silt and clay is 33±3% and 2±0.2% in riffles and 36±3% and 2±0.2% in pools. The sand content in FGS ranges from 23% to 90% in riffles (Figure 5.3b), and 37% to 84% in pools (Figure 5.3c). The silt content ranges from 9 – 71% and 15 – 58% in FGS of riffles and pools respectively. Clay content ranges from 1 – 5% in riffle FGS and 1 to 4% in pool FGS. Differences in FGS content of riffles and pools is not statistically significant, according to the Mann Whitney U-test ($p>0.05$).

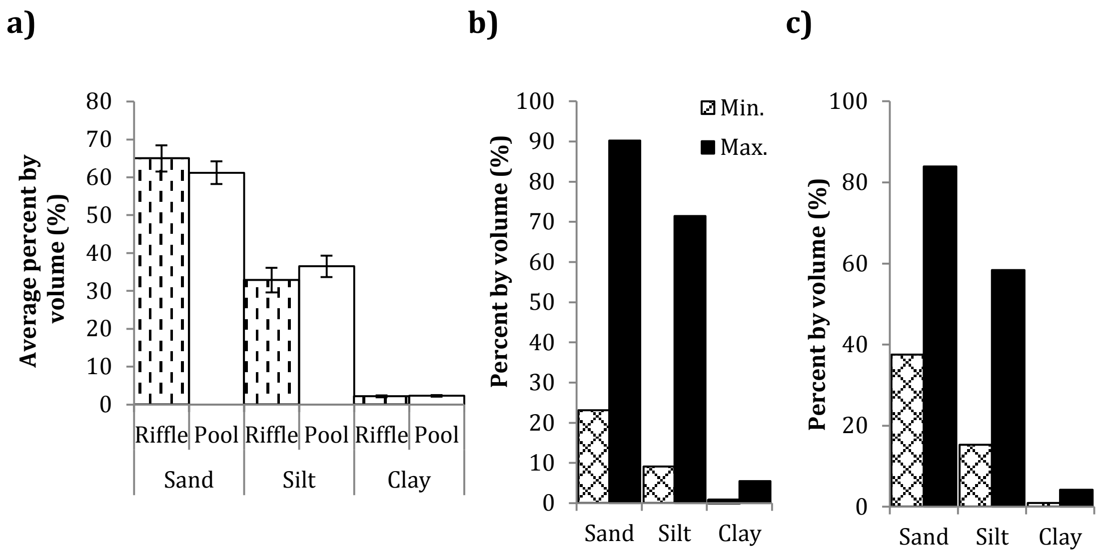


Figure 5.3. a) Average percent by volume of sand, silt and clay particles in FGS stored in riffles and pools. Error bars represent SEM. b) Minimum and maximum percent by volume of sand, silt and clay particles found in FGS in riffles. c) Minimum and maximum percent by volume of sand, silt and clay particles found in FGS in pools.

Samples collected from the Upper Welland sites were slightly coarser than those collected elsewhere (Figure 5.4) and had, on average, a higher content of sand (82±6%) than samples from the Stonton Brook (58±6%) and the Eye Brook (68±2%). The Stonton Brook had, on average, more silt (40±6%) in its FGS than the Eye Brook (17±6%) and the Upper Welland (30±2%). Samples from the Stonton Brook also had the highest content of clay in its FGS (3±0.5%). Differences in the character of the FGS of Stonton Brook and the Eye Brook was not statistically significant, according to the Mann-Whitney U test (p>0.05).

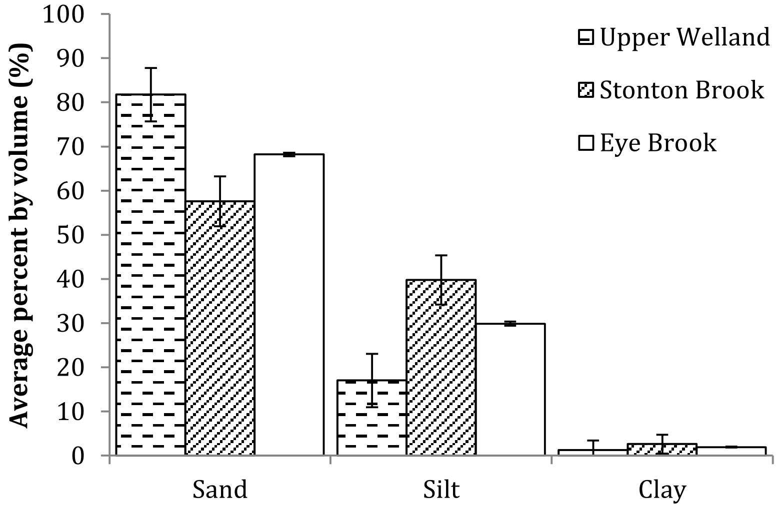
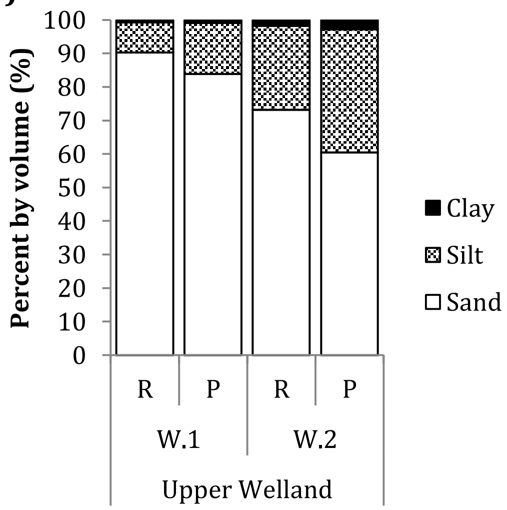


Figure 5.4. Average percent by volume of sand silt and clay in the McNeil core samples collected from the study catchments. Error bars represent SEM.

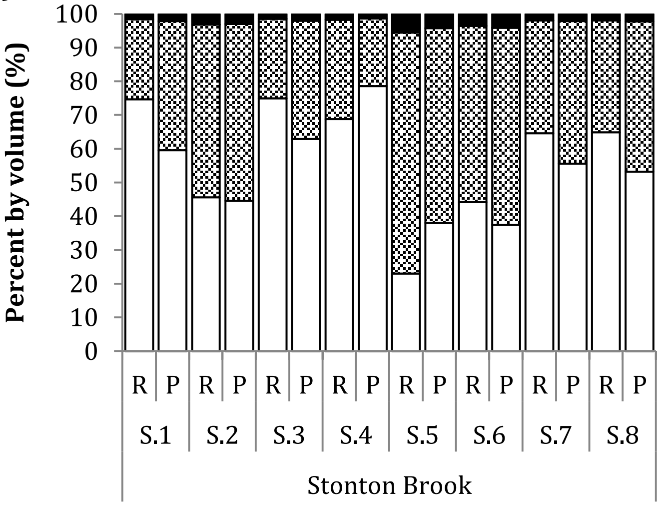
FGS composition also appears to vary within individual catchments. FGS is sandier in riffles compared to pools in all three catchments, and the pools tend to contain more silt (Figure 5.5). The clay content of riffles and pools is similar in all three catchments, however the observed differences in FGS composition within catchments were not statistically significant according to the Mann-Whitney U test (p > 0.05).

a)



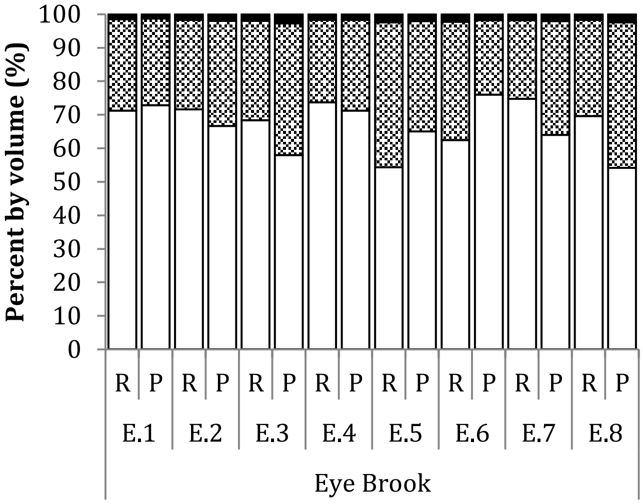
	Mean average percent by volume	
	Riffle	**Pool**
Sand	82 ±6%	72 ±8%
Silt	17 ±6%	26 ±8%
Clay	1 ±0.4%	2 ±0.7%

b)



	Mean average percent by volume	
	Riffle	**Pool**
Sand	57 ±6%	54 ±5%
Silt	40 ±6%	43 ±4%
Clay	3 ±0.5%	3 ±0.3%

c)



	Mean average percent by volume	
	Riffle	**Pool**
Sand	68 ±2%	66 ±2%
Silt	30 ±2%	32 ±2%
Clay	2 ±0.1%	2 ±0.1%

Figure 5.5. Percent by volume of clay, silt and sand in FGS of riffles and pools in a) the Upper Welland, b) Stonton Brook and c) Eye Brook. R = riffle; P = pool.

5.2.4 Skewness

The majority of FGS samples were fine, or negatively, skewed (i.e. the tail of the distribution extends towards the finer size fractions) (Figure 5.6). All samples in the Upper Welland were very fine skewed, and the majority of samples in the Eye Brook and Stonton Brook were also fine skewed. Six samples were symmetrical (E.8 pool, S.2 pool, S.2 riffle, S.6 riffle, S.7 pool and S.8 pool). Coarsely skewed samples were only found in the Stonton Brook (riffle S.5 and pools S.5 and S.6).

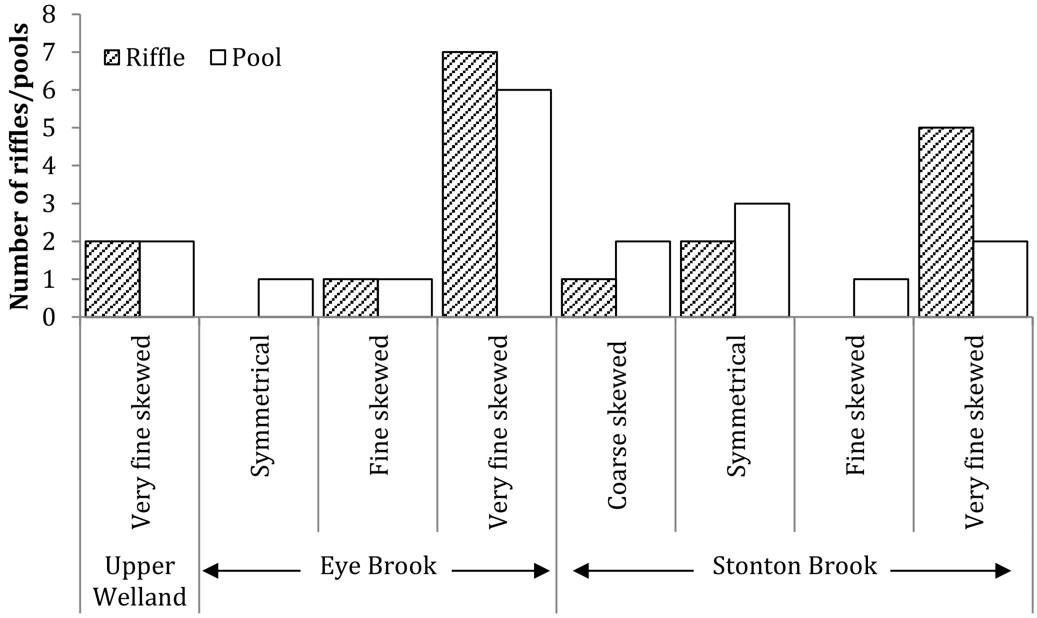


Figure 5.6. Frequency distribution of sample skewness for pools and riffles in the study catchments.

For bimodal distributions, skewness was inversely correlated with mean grain size (Figure 5.7). As mean grain size coarsens (i.e. decreases), the sample becomes increasingly positively skewed (Folk and Ward, 1957). Skewness also appears to be influenced by whether the modal diameter is in the coarse or fine fraction of the GSD (Figure 5.8). Samples with a peak mode at the coarser end of the distribution tend to display a positive skew. Samples with a peak mode at the fine end of the sample tend to be negatively skewed. Symmetrical distributions have peak modes at the coarse and fine end of the distribution.

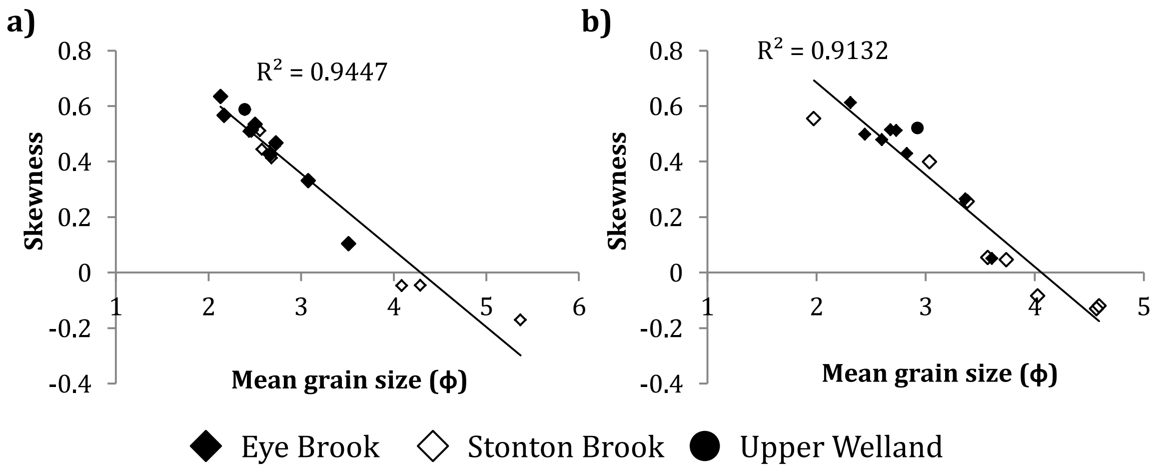


Figure 5.7. Skewness versus mean grain size for a) riffles and b) pools.

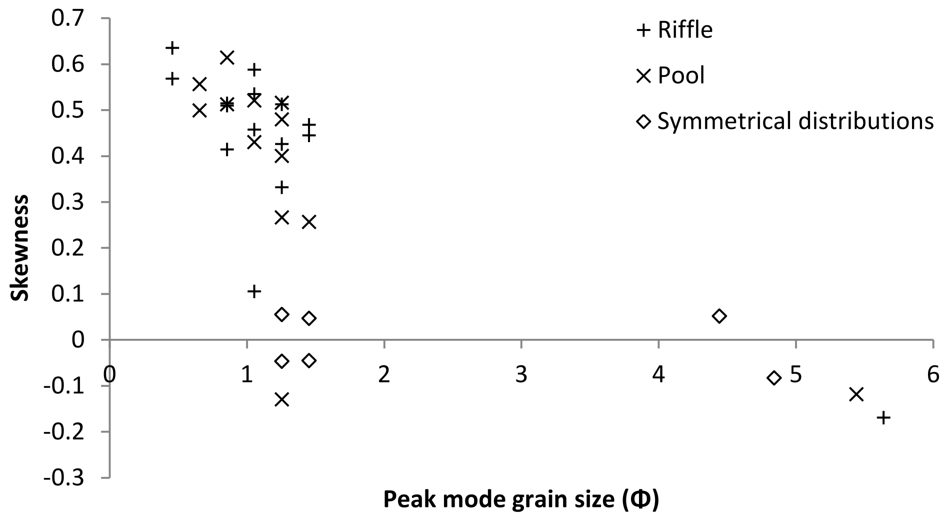


Figure 5.8. Peak mode grain size versus skewness for riffles, pools and symmetrical distributions. Positively skewed samples tend to have the peak mode grain size at the coarse end of the GSD.

5.2.5 Kurtosis

Most samples (24/36) were platykurtic. Eight samples were mesokurtic, two were leptokurtic and two were very leptokurtic (Figure 5.9). Folk and Ward (1957) suggest that platykurtic distributions are common when the modes of a bimodal distribution account for unequal proportions of the sample. Modal values rarely account for similar proportions of the GSD (Figure 5.10), explaining the observed kurtosis values.

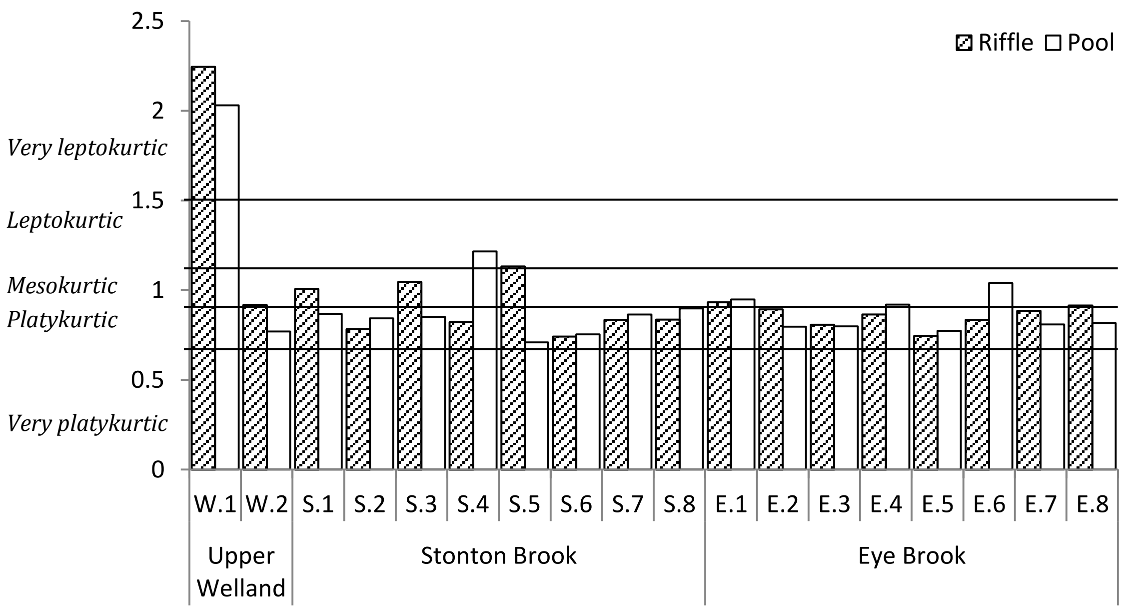


Figure 5.9. Kurtosis values of riffle and pool sites in the study catchments.

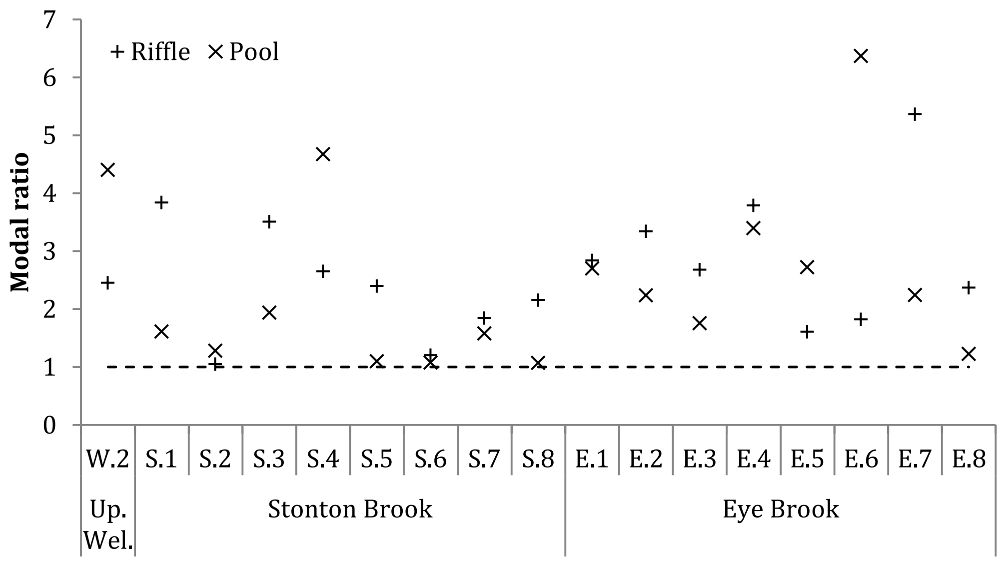


Figure 5.10. Ratio of the relative frequency of peak modal sediment diameter to the relative frequency of the second modal sediment diameter. A ratio of 1 indicates the two modes account for similar proportions of the GSD. Up. Wel. = Upper Welland.

5.2.6 Sorting

FGS sorting in the sediment sampled from the McNeil corer at most of the sites was generally poor, or very poor (Figure 5.11). Poor sorting is expected in bimodal sediment distributions by definition.

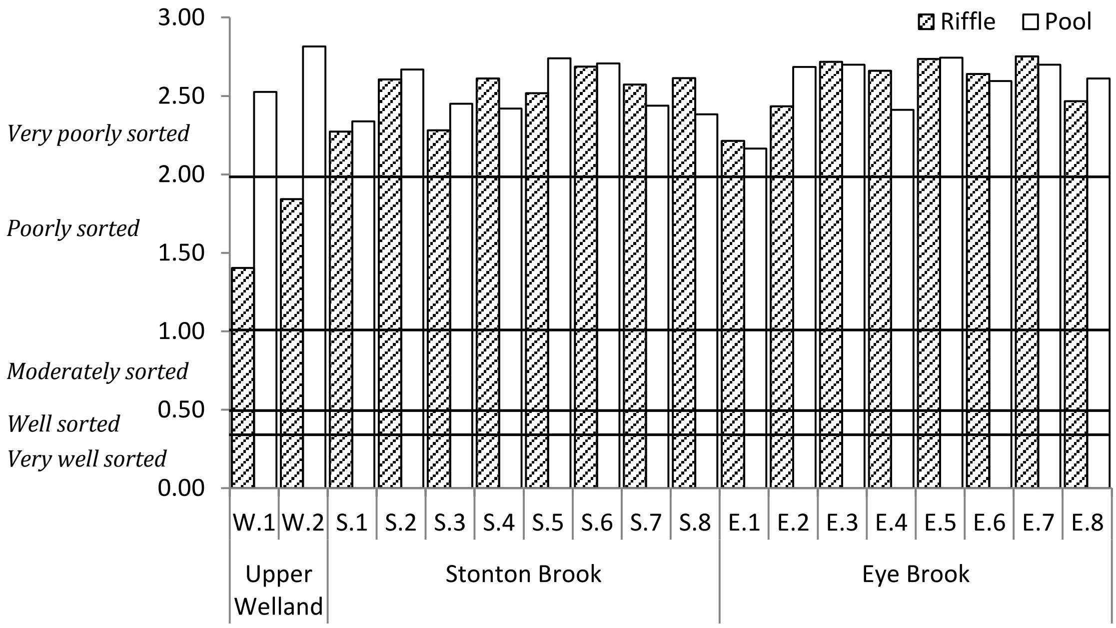


Figure 5.11. Sorting values for the FGS in riffles and pools. Sediment is poorly sorted. See section 3.4.2 for sorting equation.

5.3. Organic Matter Content

5.3.1 Data Comparisons

The OM content of the FGS sampled from the study catchments is shown in Table 5.2a. The OM content of the sediments was in the range 3.17 to 8.66% by mass. This range is lower than average values reported in similar studies elsewhere. The average OM content of FGS in the Eye Brook and the Upper Welland was 5%, and 6% in the Stonton Brook. OM content values reported in the literature range from 4 to 47%, with means of 9% and 17% reported (Table 5.2b).

Table 5.2. a) Measured OM content of the FGS stored in the study catchments, as determined by LOI. b) Values of OM content of FGS documented in the literature. Individual values represent the mean.

a)

Stream	Catchment area (km^2)	OM content (%)	
		Range	Mean
Upper Welland	53	3.39 – 6.28	5.02 ±0.60
Stonton Brook	42	3.17 – 8.06	5.59 ±0.35
Eye Brook	61	3.58 – 7.29	5.40 ±0.29

b)

Author	River	Catchment area (km^2)	OM content (%)
Owens *et al.* (1999)	Tweed	4390	4 – 13 9
Walling *et al.* (1998)	Wharfe	815	15 – 22 17
	Ouse	3315	2 – 16 9
Marttila and Kløve (2014)	Sanginjoki, Finland	400	15 – 47
Carling and Reader (1982)	Egglesthorpe Beck	12	7 – 13
	Carl Beck	5	9 – 14
Acornley and Sear (1999)	River Test	1250	15 – 40

5.3.2 Average OM content

The OM content of FGS in riffles and pools is shown in Figure 5.12a. For both pools and riffles, the average OM content was 5±0.3%. The lowest OM content was recorded in a riffle (3%) (Figure 5.12b), and the minimum OM content in a pool was 4%.

a) **b)**

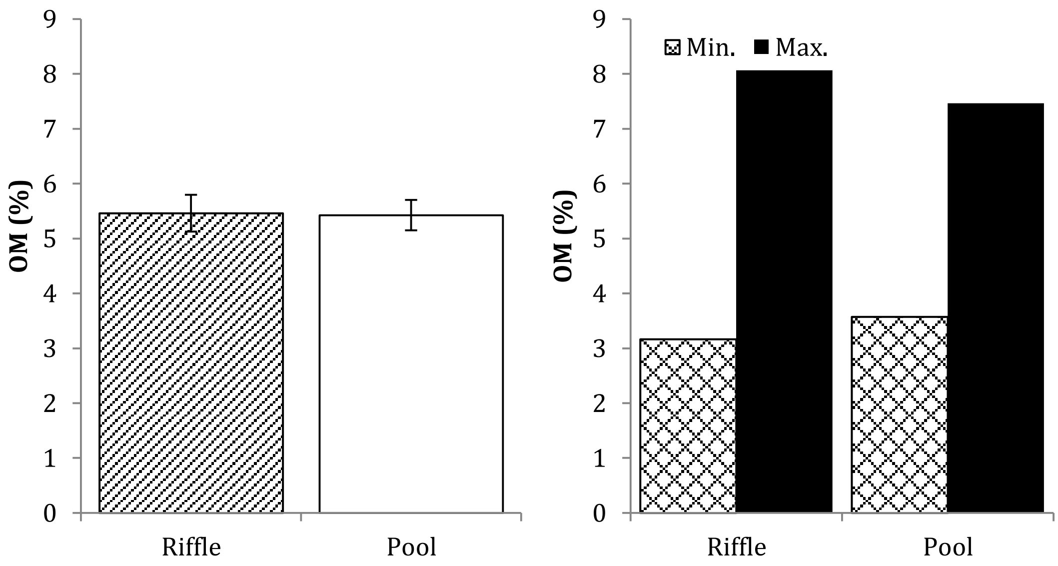


Figure 5.12. a) Average OM content of FGS in riffles and pools of the study catchments. Error bars represent SEM. b) Minimum and maximum OM content in riffles and pools of the study catchments.

5.3.3 Inter- and intra-catchment variability in OM content

The OM content of the sediment sampled from riffles was similar between the Eye Brook (5±0.4%) and the Stonton Brook (6±0.4%) (Figure 5.13a). The average OM content of pools is 5±0.4% in both the Stonton and Eye Brook. OM content displays some variability within catchments (Figure 5.13b), but there is little systematic difference between pools and riffles. Differences in the OM content within and between catchments was not statistically significant, according to the Mann-Whitney U test ($p>0.05$).

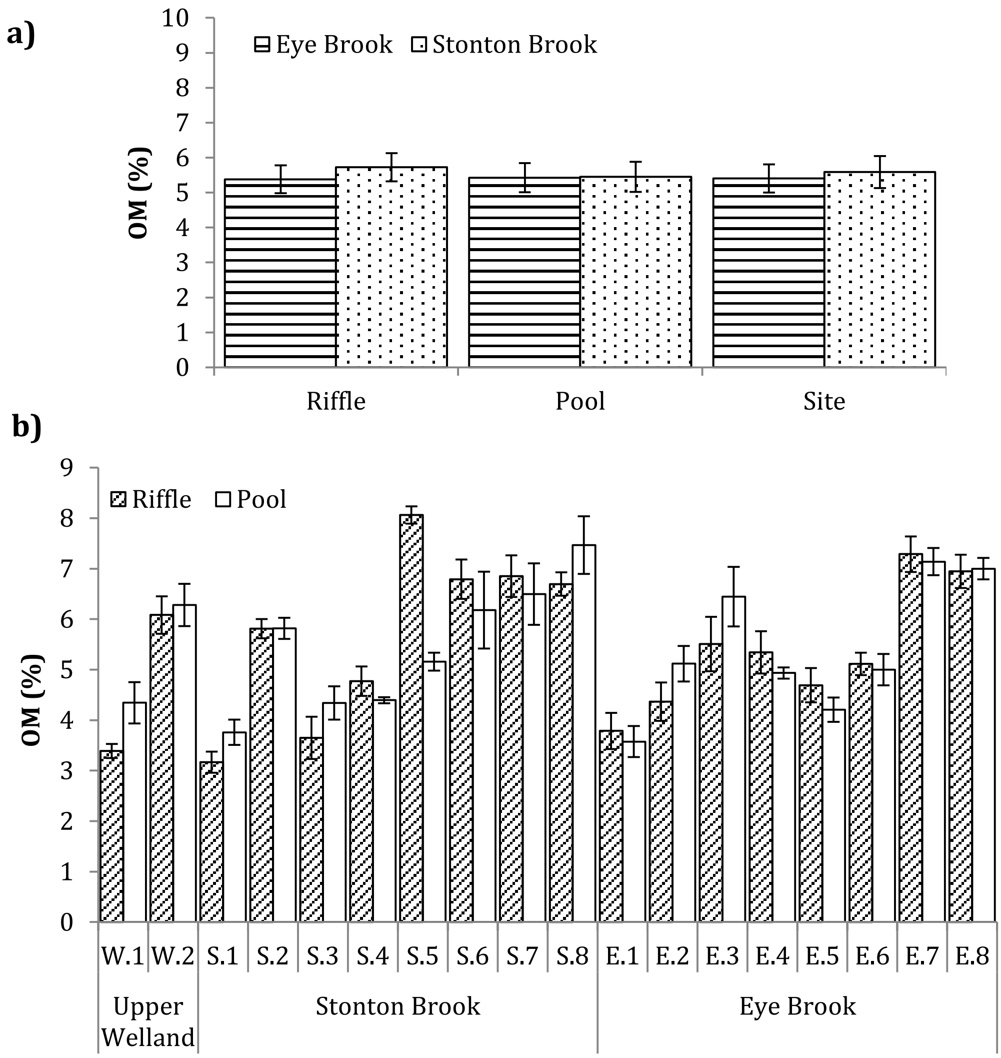


Figure 5.13. a) Average OM content of FGS in riffles, pools and at-a-site in the Eye Brook and Stonton Brook catchments. b) OM content of FGS in riffles and pools. Error bars represent SEM.

Chapter 6 – Discussion

6.1 Fine-grained sediment storage in UK rivers

The total amount of depth-normalised FGS storage in the study catchments appears to be higher than that reported by others for UK catchments (e.g. Owens *et al*., 1999; Walling *et al*., 2003c; Collins *et al*., 2005). Sediment transfer can be less efficient in headwater streams, such as those sampled in this study, than main channels as thresholds for entrainment and transport are less likely to be exceeded in headwater streams. This means that headwater reaches can be particularly prone to accumulating sediment (Marttila and Kløve, 2014). Differences in storage between catchments could be due to differences in the method employed to quantify the sediment sampled and real differences between sites. Flow characteristics, and thus transport dynamics, will be very different in permeable and impermeable catchments. Impermeable catchments tend to display flashy hydrographs which quickly recede. Sear *et al*. (2008) suggest that flashy flows generate poorly sorted sediment, as the rapid recession of flow results in the deposition of a wide range of particle sizes. This may explain the fact that the FGS in the study catchments is generally poorly sorted. Infiltration into interstices, and sheltering from further transport, are likely if residence times are high. Higher levels of FGS storage could therefore be expected in impermeable catchments compared to catchments with higher base flow indices (Acornley and Sear, 1999; Evans and Wilcox, 2014).

All studies reported in the literature used Lambert and Walling's disturbance technique. Although a validated technique, this methodology is limited by difficulties in replicating disturbance between sites and difficulty disturbing the bed sediment (Duerdoth *et al*., 2015). This may result in underestimation of the total amount of FGS stored on the stream bed. Collecting bulk samples using the method reported in this study from permeable catchments would reveal whether differences in storage are attributable to the method used or catchment characteristics.

6.2 Fine-grained sediment storage

6.2.1 Riffle and pool storage

FGS storage was, on average, observed to be higher in pools than in riffles, as was the amount of remobilisable surficial storage per unit area. These results support visual examinations of the stream bed surface, which suggested that, prior to disturbance, pools contained more surficial fines than riffles (Figure 6.1). As Kaller and Hartman (2004) and Lisle and Hilton (1992) suggest, FGS dynamics in streams mean this pattern of storage would be expected. Pools tend to be areas of preferential accumulation of FGS, such that even in streams unimpacted by land use change, pools are likely to act as stores of fines, especially mobile fines. The results also suggest that FGS storage occurs both on the bed surface and in the bed matrix, with storage in the matrix exceeding surficial storage; values of storage obtained with the McNeil corer are higher than those obtained when the water column was disturbed. This pattern of storage was also observed by Walling *et al.* (1998). A greater amount of material was remobilised when the bed was agitated compared to when just the water column was agitated, suggesting that the remobilisable fraction of FGS storage is restricted to the near surface layer, with much of the matrix material remaining undisturbed during events with low and moderate shear stresses.

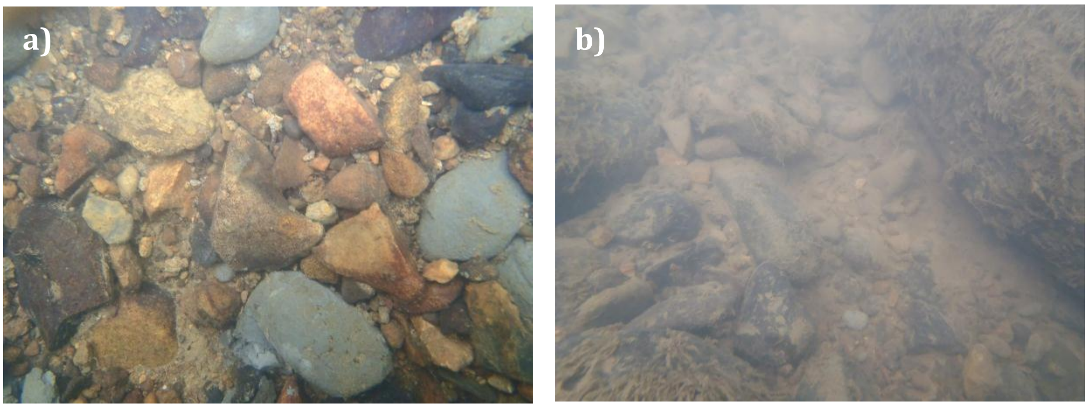


Figure 6.1. Surficial gravels at site E.4 in Eye Brook. a) Riffle gravels show little evidence of surficial deposits of FGS. b) FGS has accumulated on the bed surface of the pool. Photo: author's own.

At sites W.2, S.4, E.2, E.4 and E.8 the quantity of FGS sampled from riffles exceeded that of pools (Figure 4.3). The proportion of fines in riffles also exceeds that of pools at sites W.2, S.4 and E.6 (Figure 4.7). Site S.7 generated higher amounts of remobilisable surficial fines from its riffles than its pools (Figure 4.12). Under normal sediment loads, riffles would be expected to have low levels of FGS storage (Kaller and Hartman, 2004). The patterns of storage observed in the study catchments therefore indicate that there may be sediment problems along some reaches in the study catchment. This could have also been due to a short term derivation from the normal situation (due to natural variability) but, equally, could represent an accumulation of fines in a location at which this would not obviously take place. Such a deviation may result from a change in the balance between inputs and outputs from the reach in question due, for example, to an increase in the rate of sediment transfer from the contributing catchment area which is not matched by an increase in sediment transport capacity.

6.2.2 Inter-catchment variability

There does not appear to be significant variability in FGS storage between the catchments studied. Catchment and channel characteristics are similar in the Eye Brook, the Stonton Brook and the Upper Welland so the main factors governing FGS dynamics are also likely to be similar. The observed pattern of FGS storage could therefore be expected.

6.2.3 Intra-catchment variability

Observed sediment storage varied within catchments in terms of pools versus riffles, but also in terms pool versus pool and riffle versus riffle. The range in at-a-site storage observed in the Eye Brook was 4379 g m^{-2} cm^{-1} compared with 3274 g m^{-2} cm^{-1} in the Stonton Brook and 1447 g m^{-2} cm^{-1} in the Upper Welland. Intra-catchment variability could be explained by catchment and channel characteristics. As Richards (1982) suggests, aggradation typically occurs until the channel slope is steep enough to transport the supplied sediment. It could, therefore, be expected that as bed slope increases there would be an initial increase in storage, followed by a levelling off (or even a decrease) once the slope is steep enough to transport the sediment coming in. At the local scale, storage is not correlated with bed slope (Figure 6.2). Similarly, despite the often cited link between agricultural practices and FGS supply, land use and FGS storage are not correlated (Figure 6.3). The lack of correlation between storage, catchment characteristics and channel characteristics in the study catchments suggests that FGS dynamics are complex and do not always conform to expected patterns.

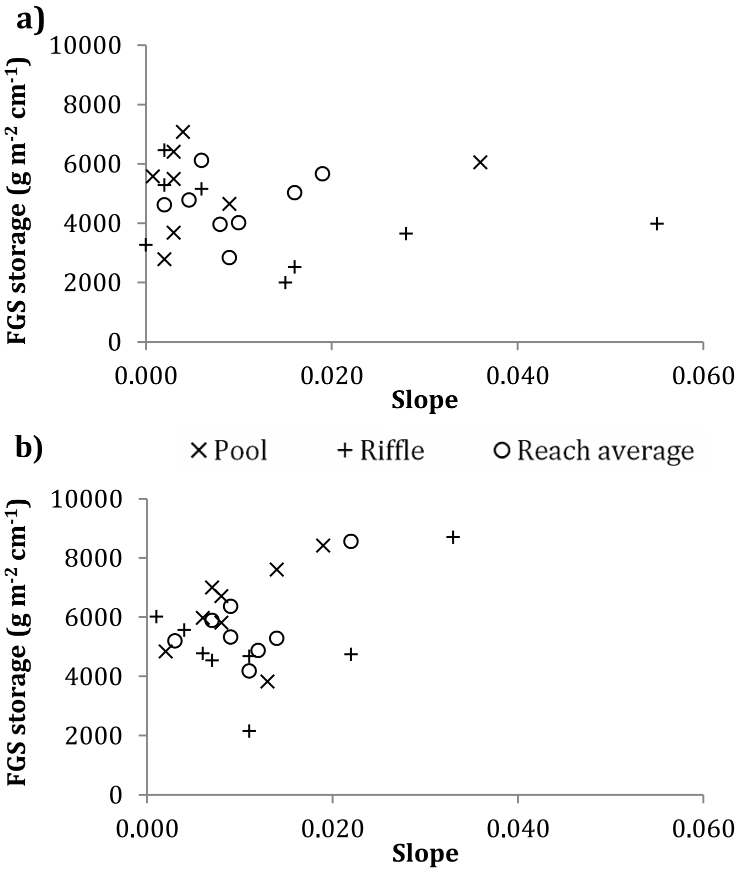


Figure 6.2. Local bed slope versus FGS in pools, riffles and reach average storage for a) Stonton Brook and b) Eye Brook. Slope and storage are not correlated ($p>0.05$).

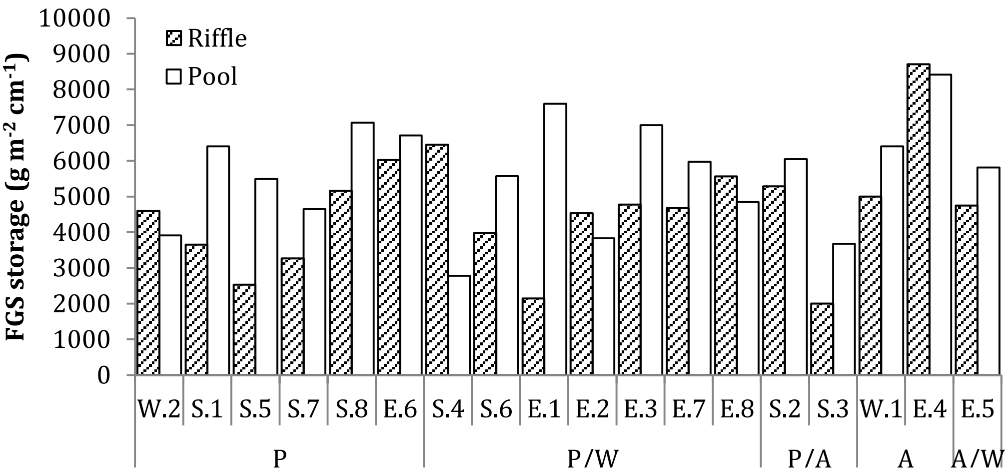


Figure 6.3. Land use versus FGS storage. Land use and FGS storage are not correlated. P = pasture; P/W = pasture and woodland; P/A = pasture and arable; A = arable; A/W = arable and woodland.

Field sites were selected to avoid locations which displayed local (e.g. poaching by animals) rather than characteristic channel conditions. However, as a full walk of the channel network was not possible, it could have been the case that there were local features or phenomenon operating upstream of a field site which could be generating locally elevated FGS input and hence storage, e.g. a field drain or channel poaching. Poaching was evident downstream of site S.5, as Figure 6.4a shows. This could have generated higher average storage at S.6 (4775 g m^{-2} cm^{-1}) compared to site S.5 (4009 g m^{-2} cm^{-1}). Field drains are thought to contribute to FGS loads (Evans, 2006) and could be observed downstream of site S.1 (Figure 6.4b).

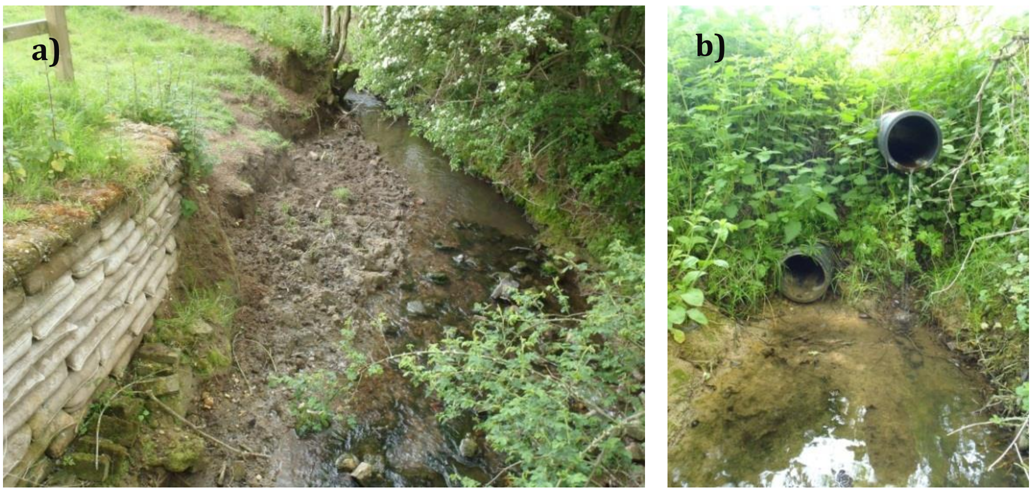


Figure 6.4. a) Poaching evident downstream of S.5. b) Field drain downstream of S.1. Photo: author's own.

Walling *et al*. (1998) examined sediment storage in the River Ouse and Wharfe, Yorkshire, using Lambert and Walling's disturbance technique. They reported an increase in apparent FGS storage downstream, and suggested that the highest FGS storage is found in lower reaches. As catchment area increases, total sediment load increases, even if the SDR decreases. If the increase in inputs to a reach exceeds the sediment transport capacity, sediment will accumulate, therefore this pattern of storage could be expected. They also suggested that the amount of sediment stored at a site is proportional to the basin size, with geomorphological processes being largely-scale dependent. As Figure 4.5 shows, storage and upstream channel length were not correlated in this study. However, the study catchments are much smaller in area compared to catchments used in other studies (e.g. Lambert and Walling, 1988; Walling and Amos, 1999; Collins *et al*., 2005; Heppell *et al*., 2009). The increases in channel length and catchment area across the study sites may not be great enough for a significant difference in inputs, outputs, and thus storage to be observed.

Variability within catchments has been reported in the results of some similar studies (e.g. Owens *et al*., 1999; Collins and Walling, 2007a; Collins and Walling, 2007b). The range in storage reported by these other lowland river studies was generally smaller than the range observed here, e.g. Collins and Walling (2007b) observed storage ranges in the Tern, Pang and Lambourn of 186 g m^{-2} cm^{-1}, 73 g m^{-2} cm^{-1} and 40 g m^{-2} cm^{-1}, respectively. However, in other studies, intra-catchment variability is of a similar magnitude to that observed here. Walling and Amos (1999) observed a storage range of 1600 g m^{-2} cm^{-1} in the Piddle catchment (a chalk stream in Dorset). The intra-catchment variability in FGS storage observed in the study catchments described here is, therefore, not unusual. Variation in storage appears more significant between catchments with different characteristics, rather than within or between catchments displaying similar characteristics.

6.2.4 Depth of infiltration

At some sites, the bed substrate was shallow and bed rock material was reached before sediment had been excavated to 24cm. As a result, some sampling depths were shallow. Sample depth varied from 2cm to 24cm. Normalising by depth allowed the FGS storage to be compared between sites. It is commonly suggested that sediment infiltrates to depths of 1-10cm (e.g. Lambert and Walling, 1988; Collins and Walling, 2007b; Collins *et al.*, 2013). As Figure 6.5 shows, there is no correlation between sample depth and FGS storage in riffles ($p = 0.11$) and pools ($p = 0.81$). If infiltration in the study catchments did not extend below 10cm depth, sampling beyond 10cm should result in decreasing amounts of FGS; the weight of FGS collected in the sample is divided by a greater depth than it occupies in the stream bed. FGS in the study catchments could therefore be thought to infiltrate to depths of greater than 10cm.

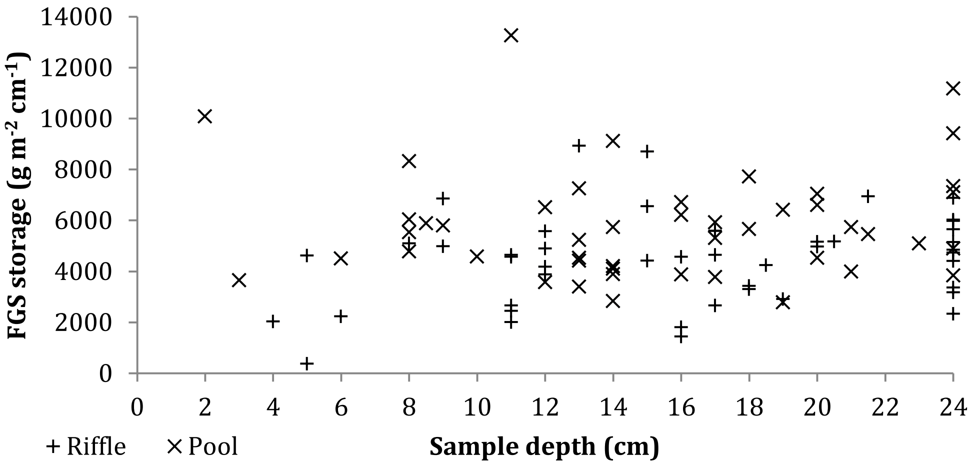


Figure 6.5. Sample depth versus FGS storage. Sample depth and storage are not correlated.

6.3 Fine-grained sediment character

6.3.1 Sediment composition

The clay content of pools and riffles was similar (Figure 5.5). Flow conditions in pools are likely to be more conducive to the deposition of clay sized particles, particularly in terms of hydrograph recession and baseflow. However, clay remobilisation may be inhibited in riffles due to the protection afforded by larger particles which provide interstitial pore spaces where small particles can be protected from entrainment. Therefore, the observed character of the FGS could be expected.

6.3.2 Grain size distribution

Samples were sieved to 1000μm prior to being put through the laser sizer in order to reduce machine error. Despite this, as the results show, particles with a diameter of greater than 1000μm (maximum = 2187μm) were recorded in GSDs (Figure 5.1). At some sites, sediment greater than 1000μm in diameter accounted for as little as 0.06% of the sample (e.g. pool at S.5), but at others, it made up 20% of the GSD (e.g. riffle at E.7). On average, sediment >1000μm accounted for 6±0.8% of the GSD. As Konert and Vandenberghe (1997) suggest, there are often discrepancies in grain sizes present between different methodologies. They suggest that when sizing particles >63μm with a laser sizer, results tend to be generally coarser than results for the same sample obtained by sieving. The results from this study support this theory.

Variability in GSDs persisted, despite sieving to 1000μm (Figure 6.6). Chappell (1998) observed similar variability in the coarse fraction of sediment samples from south-West Niger (>500μm), and described the variation of both reruns and subsamples as erratic. Loizeau *et al*. (1994) also observed similar reproducibility issues with laser granulometry, describing reproducibility as satisfactory.

6.3.3 Particle shape

Laser granulometry and sieving define particle diameter differently. Laser granulometry defines particle size as the mean diameter of the particle, whereas sieving provides information on particle width. A coarser mean diameter than particle width suggests that not all axes of the particle have equal length. From these results, it could be inferred that particles in the sample may be non-spherical. McAnnally and Mehta (2002) observed similar sediment characteristics, suggesting that although flocced particles are generally spherical, the individual grains in the flocs are often not spherical.

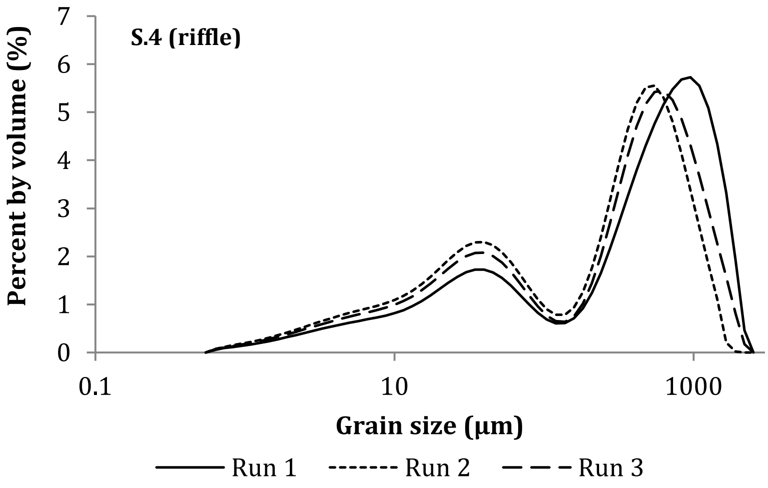


Figure 6.6. Variability in GSDs for three re-runs of the same sample in the laser granulometer. The sample is from the site at S.4. Mean grain size: 1 = 233.4μm; 2 = 139.2μm; 3 = 171.8μm. Variability in GSDs is attributable to machine error.

6.3.4 GSD and sediment transport

The GSD of bed sediment is, in part, a reflection of the processes governing sediment transport and can be used to help predict sediment transport rates. Houssais and Lajeunesse (2012) suggest that although wide GSDs can complicate predicting transport rates, knowledge of the GSD of stream bed sediment is useful for predicting sediment dynamics and the potential ecological effects of sedimentation. As section 2.3.2 suggests, coarse sand is likely to be transported as part of the bed load, whereas sediment ≤0.5mm is likely to be transported in the suspended load.

6.3.5 GSD and infiltration

The GSD has been suggested as a potential limitation on sediment infiltration depths. If FGS particles are smaller than interstitial pore diameters, fines will infiltrate into the matrix material. If particle and pore diameters are similar, fines will likely create an impermeable seal on the surface and prevent infiltration from occurring. Probability of infiltration has been suggested to increase as grain size decreases (Bunte and Abt, 2001). Results of this study suggest that grain size does not limit infiltration. Difference in FGS storage in riffles of Stonton Brook and Eye Brook are not statistically significant (Figure 4.2). However, the Eye Brook has, on average a lower proportion of FGS storage on the surface of riffles compared to Stonton Brook (Figure 4.11), suggesting that interstitial infiltration is lower in Eye Brook. Grain size information reveals that fines are, on average, slightly coarser in the Eye Brook (Figure 5.2). As section 2.3.4 suggests, grain size can influence the depth of infiltration. The coarser diameter of the FGS in the Eye Brook is however not likely to be limiting infiltration into the matrix material of the Eye Brook.

6.3.6 Fine-grained sediment sources

In principle, FGS characteristics can be used to infer the source of stored sediment, e.g. through sediment fingerprinting (Walling *et al.*, 2003a). They can also be used to infer the dominant transport processes. If clay dominates the GSD, the suspended load will be the main contributor to interstitial fines. Conversely, the presence of sand in the sediment matrix suggests that fines originate predominantly from the bed load (Frostick *et al.*, 1984). The bedrock of the study region is Charmouth Mudstone. This is a clay member of the Blue Lias formation, and is characterised by mudstone layers weathering to clay at the surface. In-channel sources of FGS, especially those originating from the channel banks could, therefore, be expected to be dominated by clay-sized particles. Since the clay content of the samples was generally low (ranging from 1 to 5% of the FGS) (Figure 5.3), interstitial fines in the study catchments are likely to originate from the bed load rather than the suspended load. The paucity of a clay sized fraction in the stored FGS suggests that suspended fines are transported out the system and not stored (Pizzuto, 2014).

It should be reiterated at this stage that laser sizing often underestimates the clay fraction compared to other grain sizing procedures, e.g. the pipette and sieve-hydrometer methodologies (Loizeau *et al.*, 1994; Konert and Vandenberghe, 1997;

Di Stefano *et al.*, 2010). Inferences of the source of FGS should therefore be interpreted with care.

6.3.7 Organic Matter Content

The OM content of the FGS in the study catchments was generally lower than those reported in other lowland UK river catchments. This reflects a balance between OM being introduced into the stream system (allochthonous and autochthonous) and losses due to degradation and advective removal. As Bilby and Likens (1980) highlight, headwater streams often contain many invertebrate taxa which are specialised in breaking down coarse OM. Small streams are especially able to process coarse OM into fine OM which can then be more easily degraded by microorganisms, or armoured in storm events as particlulate or dissolved OM which can be resuspended and transported out the catchment. A low OM content could therefore be expected in the FGS of headwater streams.

6.4 Limitations of research

The developed methodology was one which produced relatively consistent results, suggesting that it may be reliable for sampling stream beds. Despite this, it is important to consider the potential limitations with the research methods employed.

6.4.1 Sample size

One potential limitation in this study is sample size. Samples were collected from eight sites in the Stonton Brook and eight sites in the Eye Brook. As a result of site accessibility, sites were not distributed evenly down the channel (Figure 3.1). Statistical comparisons were also only possible between Stonton Brook and the Eye Brook because of the small data set collected in the Upper Welland (two field sites). The strength of statistical conclusions could be improved if samples were collected from more study sites. Increasing the number of samples collected at each bedform could also improve the accuracy and representativeness of the data set. As sample size increases, the estimate of the mean is also likely to improve.

That said, sample sizes in other studies are similar to the samples size of this study. For example, Collins and Walling (2007b) collected samples from 16 sites in the Pang, Lambourn and Tern catchments and Owens *et al*. (1999) sampled 10 sites in the Tweed catchment.

6.4.2 Remobilisable fines

The method developed for assessing the amount of remobilisable fines per unit area of bed was a novel extension of other disturbance-based techniques. It has the advantage of reproducing the type of disturbance exerted by flowing water in storm events. In addition, the use of a motorised paddle allows a relatively reproducible rotational shear stress to be applied at all sites.

However, there remain a number of potential issues with this system. When inserting the barrel into the stream bed prior to disturbing the water column, some fine sediment entered suspension. These fines could have been transported away in the flow before the stream bed was sealed from flow, representing a potential source of error. Furthermore, in causing sediment suspension, the arrangement of particles on the stream bed could have been modified. This could have modified sediment dynamics, and caused a potential over estimation of remobilisable surface storage. Some fines could have originated from within the interstices, and entered suspension when the barrel was worked into the stream bed. Fines that entered suspension were allowed to settle out before the water column was disturbed and the sample collected, helping to reduce potential error.

Similarly, through using a motorised paddle, the rotational shear stress exerted by the paddle resulted in shear stresses being exerted in all directions (Figure 6.7). Although natural flows exert a set of shear stresses due to turbulence, these are predominantly in a downstream direction. The disturbance technique may therefore potentially release what would be otherwise protected FGS into suspension.

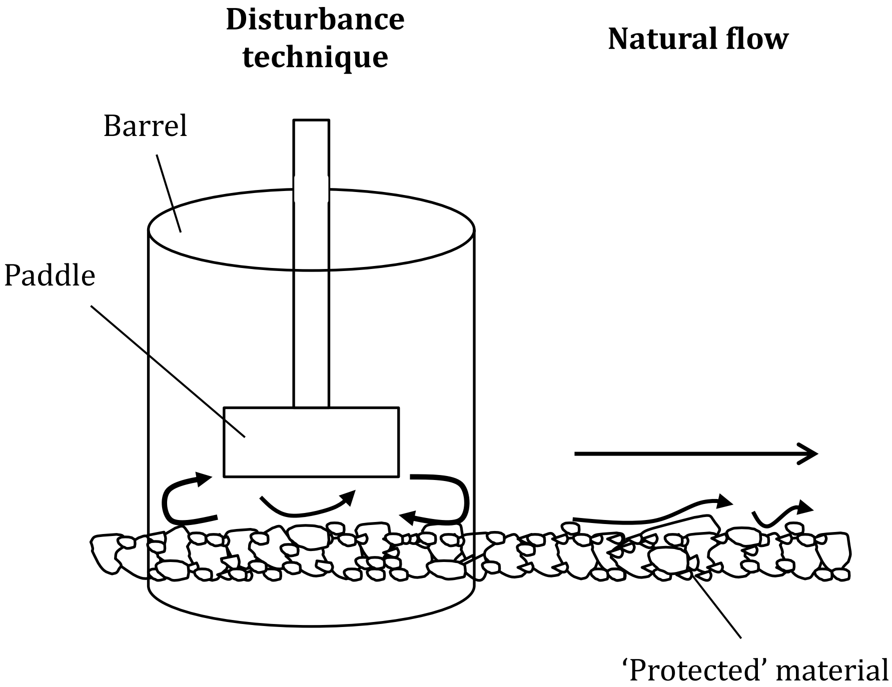


Figure 6.7. The shear stresses exerted by the disturbance technique are different to those experienced in natural flows. Arrows represent the direction of the shear stresses.

At some sites, dilution may have occurred. Although the majority of sites had the stream bed completely sealed from flow, in some riffles, coarse particles present on the stream bed meant that it was not possible to insert the barrel completely into the stream bed to a depth sufficient enough to seal the flow. Fines lifted into suspension during disturbance may have been taken out of the barrel by the moving flow, resulting in an under estimation of the quantity of remobilisable FGS stored on the stream bed surface.

6.4.3 Seasonality of sampling

Lambert and Walling (1988) suggest that estimates of storage are not dependent on seasonality, and that storage estimates provide long-term measures of the net balance between sedimentation and removal. However, it could be the case that the seasonality of sampling affects the estimates of storage obtained. Sediment dynamics are influenced by sediment supply and flow conditions. Supply is likely to be high during winter months, with sources of sediment and transport pathways both active. River discharge, and thus sediment transport, are also likely to be at their highest during winter months. In summer, both flow and sediment supply are generally lower resulting in low sediment inputs, but also low rates of removal. Systematic differences in sediment storage between winter and summer should, therefore, be minimal but could occur if the relationship between sediment input and river discharge (i.e. transport potential) is non-linear. Repeating the study during summer months would allow conclusions to be drawn as to whether FGS storage is temporally variable over short time periods.

6.4.4 Comparability of the data set

Kondolf (2000) highlighted that reporting summary statistics prevents comparisons being made between data sets unless the same statistics are used in each study. Although comparisons were made with summary statistics reported in the literature, it is the case that comparisons had to be made between data sets collected from catchments with fundamentally different characteristics, including catchment area and bed rock permeability. Comparisons were also made with results obtained with different methodologies. The conclusions drawn from this data set could have been improved if the data set was able to be compared to data sets collected from catchments with similar characteristics.

6.5 Limitations of the discipline

No standard methodology has been developed to quantify the FGS stored on the stream bed (Lambert and Walling, 1988; Von Bertrab *et al*., 2013). As a result, numerous techniques are employed to quantify FGS in river channels, making comparisons between studies difficult to achieve. It is difficult to determine whether FGS storage differs as a result of true differences between sites, or as a result of the method used to quantify storage. This is further compounded by the fact that Lambert and Walling's disturbance technique, despite proving a popular method for quantifying FGS storage, does not suggest a standard disturbance depth. Thus the depth used in studies applying this technique – if reported – is rarely consistent between studies, with depths of 0 cm, 5 cm and 10 cm often used. FGS has also been defined in various ways, as Table 2.2 and Table 4.3 show, reducing the ease of comparability. Truncation of data sets is required to ensure that storage comparisons are made between data sets measuring storage of the same nature. Similarly, although in this study, the GSD was determined with OM contained in the sample, some studies remove OM before performing grain size analyses. However, in other studies, information on whether the OM content is included in the GSD or has been removed is absent, making GSD comparisons difficult to achieve. In order for FGS studies to be of more use, and to allow developments in this field of research to occur, standardisation of definitions and procedures should occur.

6.6 Implications of this research

The results from this research are intended to represent a baseline data set which could be used to evaluate the success, or otherwise, of catchment land management interventions in reducing the quantity of fine sediment input to river systems, thereby facilitating a process of passive stream restoration whereby water quality and riverine ecology gradually improve as FGS stored within the channel system is evacuated downstream by sediment transport. The data set can be used to identify whether statistically significant shifts in in-channel FGS storage have occurred within these catchments which might be considered as a criterion of success for catchment land management. If data collected from repeat studies yield average values of storage which fall outside of the 95% confidence interval (±1.96 SEM), a statistically significant change in FGS storage can be considered to have occurred. Otherwise, differences in storage can be attributed to natural variability (sampling error), rather than catchment management. A statistical test (e.g. a Mann-Whitney U test) would determine whether differences in FGS storage in repeat data sets are statistically significant.

The data collected in this study have also helped increase understanding of FGS storage, and increased the volume of information available on the quantity and spatial variability of fine sediment stored in river channels. Such knowledge is important for assessing the ecological impacts of FGS and developing appropriate FGS targets. The data were collected from catchments which are representative of many areas of lowland England, and therefore provide useful context for fine sediment studies conducted elsewhere. This data set could be used alongside water quality data and ecological surveys to determine whether any reductions to FGS levels which can be achieved actually improve ecological quality and, hence contribute to achieving the aims of the WFD.

A further implication of this research is the support it provides for quantifying FGS storage in the stream bed substrate with a bulk corer, rather than disturbing the water column, or measuring suspended sediment concentrations. Fines infiltrate to depths greater than 10cm, where disturbance (Lambert and Walling, 1988) is unlikely to release the subsurface sediments, potentially underestimating FGS storage. Sediment in gravels can also be remobilised and flushed in flood flows. It is therefore important to measure the total FGS storage in the stream bed as this is likely to change after flood flows (Petticrew *et al.*, 2007). Sampling the stream bed substrate with a McNeil corer is better placed to allow the success, or otherwise, of catchment land management interventions in reducing the total quantity of fine sediment in river systems to be successfully evaluated.

In addition, the application of water-column based remobilization technique using a reproducible shear stress, such as that described in this thesis allows the 'available' FGS to be distinguished from the total FGS store. It is this available sediment which dynamically exchanges between the bed and the water column in low to moderate flood events and which can potentially cause ecological effects, such as smothering coarse gravel substrates (used for fish spawning gravels, for example).

Chapter 7 – Conclusions

The amount and character of the FGS stored on the stream bed of riffles and pools of headwater tributaries in the Upper Welland River basin was quantified using an excavated core sampler, achieving Objective 1. In addition, the mass of remobilisable fines per unit area of bed was determined using a novel disturbance technique in which a reproducible rotational shear stress is applied to the water column in a closed chamber inserted into the stream bed. In doing so, Objective 2 was met. Notwithstanding the potential limitations of the chosen methodology, the results of this study have shown that FGS storage is spatially variable within and between catchments, highlighting the importance of conducting catchment scale investigations. Through work to achieve research Objective 3, the particle size distribution and OM content of the FGS was determined. FGS is sand dominated in the Upper Welland and the Eye Brook. Sand also dominates FGS composition in Stonton Brook, although some sites are comparatively siltier. FGS storage was generally higher in pools than in riffles, and a small proportion of total FGS is stored as remobilisable surficial fines. Surficial storage is significantly higher in pools than in riffles. FGS storage was higher in Eye Brook than Stonton Brook, and the mean grain size of the FGS stored in Eye Brook was coarser than Stonton Brook. Results suggest that interstitial infiltration of FGS may be lower in the Stonton Brook than in the Eye Brook, with grain size not acting as a factor limiting interstitial infiltration. Bulk density is higher in riffles than pools, and accounts for the mismatch between the amounts of FGS stored on the stream bed, and the proportion of the stream bed that is FGS. The OM content of the sampled FGS is generally low, suggesting that FGS does not represent a significant store of organic carbon, and is likely to have less significant implications for ecological quality than in catchments with higher OM contents.

There is much potential to address the prevailing limitations of this research, and for widening the scope for research in this field. This research has focused on quantifying FGS storage in riffle-pool sequences of headwater streams. Through expanding the data set and increasing the number of field sites and the number of catchments sampled, a more thorough evaluation of FGS storage in riffles and pools would be facilitated. Samples of FGS storage could also be collected from additional locations in the channel (e.g. glides and flow expansions). The introduction of a temporal aspect to the study also has the potential to improve understanding of the variation of FGS storage. Future research could also seek to investigate the direct ecological benefits of reducing FGS storage in river systems, allowing determination of whether reducing FGS storage in river systems will help to meet the WFD objective of good ecological status by 2027.

The results of this study have helped increase understanding of the spatial variability of FGS storage in river channels. In doing so, this study has met the requirements of Objective 4 and provided critical baseline data needed to evaluate long-term changes in sediment storage and water quality (due, for example, to catchment management), and, by extension, the success or otherwise of catchment management. This research has important implications for catchment management strategies, FGS target development, and surface water quality.

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