A REVIEW OF LITERATURE ON COHESIVE SEDIMENT TRANSPORT PROCESSES, METHODOLOGICAL FRAMEWORKS AND MANAGEMENT STRATEGIES IN RELATION TO THE NORTH SASKATCHEWAN RIVER

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1.0 Introduction

Knowledge of the source, transport, fate and effect of sediment-associated contaminants in rivers is fundamental to understanding and managing anthropogenic impacts on water quality and ecosystem health. The North Saskatchewan River (NSR) receives inputs of sediments and associated contaminants from multiple sources that include municipal and industrial wastewater discharges, storm and combined sewer discharges, tributary inputs, diffuse overland sources, direct erosion from banks and riparian areas and the erosion of river bed deposits. However, the chemical, physical and biological characteristics of these sediment-associated contaminants have not been fully elucidated and additional information is required to assess their provenance, storage (short and long term) and transport (fluxes) in the NSR.

As a first step towards improving knowledge of the fate of suspended solids and associated contaminants in the North Saskatchewan River (i.e. transport, deposition, resuspension and changes in the nature and composition of suspended matter), this report presents a review of literature on 1) the nature, source, transport and fate of cohesive sediment and associated contaminants in rivers 2) the effect of land-use change on the source and fate of sediment-associated contaminants and 3) an assessment of various frameworks and methodologies used to evaluate the provenance and fate of sediments in river environments. Because most contaminants of concern in aquatic systems are bound to and primarily transported in association with cohesive sediment (inorganic and organic particulate matter <63 µm), this report specifically reviews literature on the source, transport and fate of cohesive sediment and associated contaminants and standard methodological frameworks used to evaluate the provenance and fate of cohesive sediments in river environments.

The work presented herein comprises a Phase 1 Study (Literature Review) to provide an assessment of the current state of knowledge regarding the nature of sediment inputs, transport and contaminant relationships in rivers with the intention of using this review to outline a scope and methodology to address key knowledge gaps. Such information is necessary to develop strategic and cost-effective monitoring programs for management of the North Saskatchewan watershed.

Specific objectives of the study are to review and compile current knowledge regarding:

1) the nature, source, transport and fate of cohesive sediment and associated contaminants in rivers
2) the effect of land-use change on the source and fate of sediment-associated contaminants
3) an assessment of frameworks and methodologies used to evaluate the provenance and fate of sediments in river environments and
4) make recommendations for future work to evaluate the provenance, transport, fate and effect of cohesive sediment in the NSR
2.0 Nature, source, transport and fate of cohesive sediment and associated contaminants in rivers

2.1 Nature and origin of fine sediment

Sediments in aquatic systems are classified in two broad categories based on grain size distribution; coarse (cohesionless) and fine (cohesive). The first category refers to sediments ranging in size from fine sand to coarse gravel; whereas the second includes silt and clay fractions. In river flows, coarse sediments in transport are generally represented in appreciable quantities in the bed and their transport rates are a function of flow conditions. In contrast, cohesive sediment is typically found in small quantities in proportion to their total load and initially it was thought that cohesive sediment transport rates were unrelated to flow parameters and depended mainly on supply rates (Einstein, 1950; Einstein et al., 1940). More recent research on cohesive sediment transport has shown that cohesive sediments can interact with gravel beds in streams and their transport characteristics do depend on flow dynamics (Krishnappan and Engel, 1997; Krishnappan and Marsalek, 2005). Accordingly, an understanding of key hydrological processes and factors affecting land use change that control the spatial and temporal variability in cohesive sediment supply to and transport in rivers is key to developing appropriate management responses.

The major difference between coarse and fine sediments is related to the mutual interaction between particles in the aquatic environment (Partheneiades, 2009). With the exception of mechanical interaction in highly dense suspensions, coarse grain particles behave independently of each other. However, cohesive sediments are controlled by a set of attractive and repulsive forces that act both on their surfaces and within their mass. These forces result from factors such as sediment mineralogy, adsorption of ions on the particle surfaces and biological properties of the particles. The biological properties include benthic bacteria, microalgae and macrofauna that secrete polymers (extracellular polymeric substances; EPS) that bind mineral grains (biostabilization) together in a mucilaginous matrix (Dade et al., 1990; Paterson, 1997). Under certain conditions in the flow field when the attractive forces exceed the repulsive forces, colliding particles “stick” together to form flocs that can have settling velocities that are much higher than the individual properties (Droppo et al 2004). This process is known as flocculation and in flocculated cohesive sediment suspensions the settling unit is a “floc” rather than an individual particle (Droppo, 2001). Upon settling, flocs become the building blocks of fine-grained deposits referred to as surficial fine-grained laminae (SFGL) (Droppo and Stone, 1994). These cohesive deposits contribute to both external and internal colmation (retention of particles in stream beds) that occurs at the ground water - surface water interface (Brunke, 1999) and represent and important but transient in-channel sink for sediment associated contaminants (Stone and Droppo, 1994).
A growing body of data on the magnitudes of suspended sediment yields of the world’s rivers and their spatial variation in response to environmental and climatic controls has resulted from the recent expansion of river monitoring programs (Milliman and Meade, 1983). However, much less is known about the particle size characteristics of suspended solids in rivers primarily because the cost and logistical constraints for their determination at large spatial and temporal scales is often prohibitive and few attempts have been made to compile existing particle size data. Walling and Morehead (1987) examined the spatial and temporal variability in particle size characteristics of suspended sediment in the River Exe (England) and reported that the highly variable nature of sediment size is related to a range of factors that include climate, geology, vegetation type, basin scale, sediment erosion and delivery processes as well as land use (Walling and Morehead, 1989). In an assessment of particle size characteristics of suspended solids in southern Ontario rivers tributary to the Great Lakes, Stone and Saunderson (1992) reported that the < 63 µm fraction accounted for 60 to 99% of the calculated seasonal suspended sediment load and that the fine grained fractions tended to dominate during fall and summer months while coarser materials dominated in the spring and winter. A subsequent study of regional patterns of sediment yield in the Laurentian Great lakes basin reported that the highest sediment yields coincide with agricultural and industrial land use activities in basins with predominantly fine-grained parent materials of glacial fluvial origin (Stone and Saunderson, 1996).

In a study of the effect of large scale land disturbance (wildfire) on sediment transport, Silins et al (2008) described the initial magnitude of post-fire changes in watershed scale sediment production (concentration and specific yield) after a wildfire in the high water yielding headwaters of Alberta’s southern Rocky Mountains and reported the incremental effects of post-fire land management intervention using salvage logging. They reported that total suspended sediment (TSS) concentration and yield were strongly affected by disturbance and that the magnitude of these effects varied with time after the wildfire (Fig.1). The largest differences in sediment production between undisturbed and disturbed landscapes were observed during the snowmelt freshet and periodic stormflows. A comparison of the relationship between mean TSS concentration and discharge between the unburned, burned and post-fire salvage logged watersheds provides additional insight into sediment production in the watershed groups across a range of streamflows (Fig. 2). While greater TSS concentrations were observed at higher streamflows in all watershed groups, there were significant differences in the slopes and intercepts of the discharge and TSS relationship between burned and unburned basins for the initial two years (2004 and 2005) after the fire.

Analysis of the grain size characteristics of suspended solids shows that the morphology and the transport characteristics of suspended solids in streams draining burned watersheds (Fig 3) have very differently settling behavior (Fig 4) than reference unburned stream sediment (Stone et al., submitted).
These analyses generally indicated that mass settling rates associated with the unburned eroded sediments decreased more rapidly those associated with the burned eroded sediments, thereby implying that the unburned eroded sediment settling velocities were higher than those of the burned eroded sediments.

In an examination of the particle size distribution of suspended solids from fire impacted watersheds during various parts of the flow regime (spring melt, storm flow and base flow), the predominant size range of suspended solids was < 100 μm (Stone, unpublished data) and that during low flow conditions, these fine grained materials settle on the stream bed (Fig. 5) and form significant deposits of cohesive sediment. Currently, there is a paucity of suspended solids particle size data for many rivers in Alberta and further monitoring of sediment size and mass transport during a range of flow events is warranted to provide sufficient data to model the source, storage, transport and fate of sediment associated contaminants.

Figure 1: Box-and-whisker plots of A) annual total suspended solids (TSS) concentration (mg l\(^{-1}\)), and B) annual TSS specific yield (kg ha l\(^{-1}\) d l\(^{-1}\)) for unburned, burned, and salvage logged watersheds from 2004 to 2007 (Silins et al., 2008)
Figure 2: Relationship between mean total suspended sediment (TSS) concentration (mg l$^{-1}$) and instantaneous discharge (mm d$^{-1}$) for unburned, burned, and salvage logged watersheds from 2004 to 2007. (Silins et al., 2008)

Figure 3: Suspended solids in reference streams (left) and streams draining burned landscapes (right) (Photomicrographs by M. Stone)
Figure 4: Still water settling rates of cohesive sediment deposits collected from streams draining burned and unburned watersheds. (Stone et al., submitted to Water Research).
Cohesive sediments are a concern in aquatic ecosystems because their transport and fate are closely linked with a wide range of water quality problems (Horowitz, 1991; 1999). Most pollutants of concern, such as heavy metals, pesticides and phosphorus preferentially bind to cohesive sediments and the fate and effect of these pollutants is ultimately linked to the transport, storage and fate of the cohesive materials. This section of the report reviews literature that examines the geochemical and contaminant properties of cohesive sediment.

The importance of sediment associated contaminant flux in rivers has been widely reported (e.g. Horowitz, 1999; Walling and Woodward, 1993; Logan, 1995) and particle size distribution of cohesive
sediment has been shown to be an important factor influencing its transport and fate. Several studies have shown that pollutant concentrations of sediment tend to increase with decreasing particle size because the surface area per unit mass increases rapidly with decreasing particle size (Logan, 1999). Particle size also determines the flux of contaminants in rivers because the particle size distribution of suspended solids varies with stream flow. Accordingly, the quantification of sediment-associated contaminant transport requires information regarding particle size distribution as well as the pollutant concentration on sediment. In laboratory experiments conducted to examine the adsorption of inorganic phosphate on separated particle size fractions of two river bed sediments and a reference material (pure silica sand), Stone and Mudroch (1989) reported that chemical composition and mineralogy were parameters controlling phosphate sorption of sediment particles < 13 µm. Stone and English (1993) examined the geochemistry, phosphorus speciation and mass transport of fine grained sediment in two Lake Erie tributaries. The non-apatite inorganic (NAIP) and organic (OP) concentrations of sediment associated phosphorus were found to increase with decreasing grain size and that the < 8 µm fraction was the most significant for the potential release of bioavailable P form (NAIP) into the water column. Stone et al. (1991) developed a sediment-associated phosphate transport model to evaluate and quantify the kinetic control of phosphate by size fractions of river sediment. Simulation analyses of the model provided a quantitative assessment of the release rates of sediment bound phosphate for use in current lake eutrophication models. In a study of metals (Pb, Cu, Zn) in size fractionated river bed sediment, Stone and Droppo (1996) demonstrated that the concentration and potential bioavailability of these metals increased with decreasing grain size. They estimated that over 80% of the sediment associated metal yield in the < 63 µm fraction was associated with solids < 31 µm.

Stone and Haight (2000) used a field-based water elutriation system to investigate the occurrence and distribution of dioxins and furans in separated size fractions of suspended sediment during a spring storm event in Canagagigue Creek near Elmira, Ontario. Several furan and dioxin compounds including 2,3,7,8 – T₄CDD and 2,3,7,8 – T₄CDF were detected in suspended solids at comparable levels to those reported in previous investigations of creek bottom and floodplain sediments. However, in contrast to several other studies that report increasing contaminant concentrations with decreasing grain size, Stone and Haight (2000) reported no significant relationship between grain size and organic contaminants in suspended solids. They attributed this to the highly flocculated and bio-stabilized nature of river bottom sediment that was eroded from the study site which ultimately obscured the expected grain size - contaminant relationship.
2.3 Biological effects of fine sediment in rivers

The delivery of excessive levels of suspended solids into aquatic systems can have significant deleterious impacts on its physical, chemical and biological properties and the magnitude of the effect will depend upon the suspended solids concentration, grain size distribution, duration of exposure, chemical composition but will also vary within and between organisms and environments (Bilotta and Brazier, 2008). As a result, the impacts of high sediment loads on aquatic biota are difficult to generalize, quantify and compare. However the general consequences of these impacts have been reported in the literature. Fine sediments suspended in the water column increase turbidity, lower light penetration and reduce primary productivity which subsequently impact food chain dynamics (Van Nieuwenhuyse and LaPeriere, 1986). Sedimentation modifies the substrate by altering its surface conditions (Graham, 1990) and the volume of fine sediment in the hyporheic zone (Richards and Bacon, 1994). Excess deposition of fine-sediment reduces spawning habitat by clogging potential spawning gravels (Lisle and Lewis, 1992) and creates a smoother bed (Diplas and Parker, 1992). The decrease in bed roughness reduces habitat complexity and availability but also increases water velocity and the need for shelter from the water current (Richardson and Jowett, 2002). In extreme cases (such as landscape scale disturbance by wildfire i.e. Silins et al., 2008), fine sediment can cover an entire river bed thus changing the channel morphology (Doeg and Koehn, 1994), clogging the interstitial voids between substrate clasts, increasing the invertebrate drift and reducing available habitat for benthic organisms (Petts 1984).

The effects of excess sediment on salmonoids have been studied. Collins et al. (2008) report that high suspended solids concentrations impede the movement of fish and alter fish migration. Increased turbidity alters foraging behavior and reduces territoriality (Berg, 1992) as well as decreases habitat connectivity (Cohen, 1995) and heterogeneity (Boles, 1981). Grieg et al (2005) examined the impact of fine sediment transport on the quality of salmonid spawning and incubation habitats. Their study demonstrated that because of fine sediment intrusion to coarse river bed deposits, incubation success is inhibited by 1) the impact of fine sediment accumulation on gravel permeability and, subsequently, the rate of passage of oxygenated water through the incubation environment 2) reduced intra-gravel O₂ concentrations that occur when O₂ consuming material infiltrates spawning and incubation gravels and 3) the impact of fine particles (clay) on the exchange of O₂ across the egg membrane. They concluded that only a small proportion of the total suspended sediment load infiltrates spawning and incubation gravels but it has significant effects on salmonid spawning and incubation habitats. Accordingly, this observation casts some doubt regarding the ability of current catchment-based land use management strategies to adequately reduce fine sediment inputs. Given the above information, there is a critical need to recognize and identify and further study the physical, chemical and biological effects of fine sediment deposition and their impact on biota before mitigation measures can be effectively implemented. Wood and
Armitage (1997) argue that by considering the river system more holistically (Fig. 6), the generation and transfer of fine sediment to the stream and its transport deposition and storage in the channel can be better elucidated.

Figure 6: Overview of fine sediment sources, transport, storage and impacts in the lotic system (Wood and Armitage, 1997)
2.4. Cohesive sediment transport

2.4.1 Flocculation

The transport of cohesive sediment in aquatic systems is characterized by interactions among fine-grained sediment particles that cause flocs to form (Droppo, 2001). Flocs have relatively low densities, large pore spaces and reactive surfaces that remove contaminants from the water column. Flocculation is an important mechanism for particle removal in streams, lakes and oceans that alters the hydrodynamic characteristics of solids by changing the density, porosity, settling velocity and surface area. Numerical models designed to simulate contaminant transport, fate and bioaccumulation in aquatic environments have begun to include a cohesive sediment transport component (Willis and Krishnappan, 2004) but a better understanding of cohesive sediment transport processes (erosion, deposition, flocculation) is required to improve model predictions. The following sections review selected recent studies that have advanced knowledge regarding the nature and transport of cohesive sediment in aquatic systems.

Cohesive materials represent variable proportions of the total annual sediment flux in many Canadian rivers (Stone and Saunderson, 1992; Droppo et al., 1998; Krishnappan, 2001) and urban storm water (Droppo et al., 2002). The morphology and settling characteristics of these materials vary in response to physical, chemical and biological attributes of individual rivers and sediment sources (Petticrew and Droppo, 2000). Cohesive suspended sediment is commonly transported in fluvial systems in a flocculated form and larger flocs do not typically settle within the Stokes’ region of Reynolds numbers (Droppo et al., 2000). There are dramatic differences between natural flocs and their disaggregated inorganic constituents (Fig. 7) and the resulting grain size distributions of these solids (Fig. 8). These two figures demonstrate the difference between the actual or effective particle size distribution of suspended solids and the ultimate grain size distribution generally reported in traditional laboratory techniques.

Figure 7: Examples of flocs in (a) natural flocculated form and (b) disaggregated inorganic constituent form (Droppo, 2004)
In a study of river and lake sediment, Droppo et al. (1999) reported that only flocs < 100 μm (equivalent spherical diameter) settled within the Stokes' region (Re < 0.2). The densities of these flocculated materials ranged from 1 to 1.4 g cm\(^{-3}\) but the majority of flocs had densities of less than 1.1 g cm\(^{-3}\). Floc porosity increases with floc size and low floc densities are caused by the entrapment of water in the pore spaces of flocs (Droppo et al., 2000). The relationship between floc density and fall velocity as a function of floc size is shown in Fig. 9. Flocs consist of a complex matrix of microbial communities, organic particles, inorganic particles, inter-floc pore spaces, and interstitial water (Fig. 10) and association of waterborne pathogens with flocs has significant implications for water quality regulatory affairs (Droppo et al., 2009). The flocculation mechanism is dependent upon a number of factors including particle mineralogy, the electrochemical nature of the flowing medium, biological factors such as bacteria and other organic material and hydrodynamic properties of the flow field.
Advances in understanding the structural components of a floc and its individual properties have led to the development of a conceptual model that links both the structural and behavioral components of flocs (Droppo, 2001). The model redefines the traditional view of suspended solids as discrete particles, to a collection of compositionally diverse flocculated particles that behave as individual micro-ecosystems with complex physical, chemical and biological behaviors (Fig. 11).

Fractal dimensions have been used to quantify the morphology of particle populations formed in different fluid mechanical (Jiang and Logan, 1991; Logan and Kilps, 1995), stream (De Boer, 1997) and marine environments (Logan and Wilkinson, 1990). Fractal dimensions reflect the nature of particles and their mechanism of formation and particle properties such as settling velocity and density are a function of their fractal dimensions. De Boer and Stone (1999) examined the fractal dimensions of suspended solids in streams to compare settling and filtration sampling techniques for particle size analysis. Systematic differences between the two methods were observed but the filtration method was more sensitive to indicating differences within and between the sites in two basins. De Boer et al. (2000) investigated the fractal dimensions of individual flocs and floc populations of suspended solids collected during snowmelt in southern Ontario streams with contrasting riparian buffer zones. Fractal dimensions of both individual flocs and floc populations provided similar information about temporal changes in sediment source contributions and about the contrasting effectiveness of the riparian buffer zones in the two basins.
Stone and Krishnappan (2003) examined the fractal dimensions of particle populations of cohesive river sediment in a rotating circular flume and used image analysis to evaluate the structure and size distribution of flocs formed during the deposition process at four conditions of steady state flow. As shear stress increased from 0.058 to 0.121 Pa, particle boundaries became more convoluted and shape irregularity of larger particles increased compared to the smaller ones (Fig. 12). Micro-flocs were the building blocks of the larger flocs suspended in the water column and the stability of larger flocs was a function of the shear stress at steady state. Stone and Krishnappan (2005) determined the fractal dimensions of particle populations of cohesive sediment during settling experiments in an annular flume with different initial conditions at a constant bed shear stress. The study demonstrated that the ratio of initial and steady state sediment concentration for both runs was 0.54 and is a function of bed shear and not the initial sediment concentration. Fractal dimensions (D, D1, D2) were not significantly different for the two experimental runs at steady state (t = 300 minutes).
2.5 Cohesive Sediment Transport

Numerical models have been developed to predict the transport and fate of sediment and contaminants but they require information on the transport characteristics of sediments as input parameters (Hayter and Mehta, 1986; Mehta, 1989; Willis and Krishnappan, 2004; Krishnappan, 1991, 1997, 2000; Partheniades, 2009). For cohesive sediment, variables such as erosion rate and critical shear stress for erosion and deposition must be determined by direct measurement. Accordingly, rather than focusing on the development of models and related equations of sediment transport models, the following section reviews recent laboratory and field investigations that have advanced knowledge of cohesive sediment transport.

2.5.1 Laboratory Studies

Flocculation of cohesive materials in the water column and settling of flocs on the river bed result in the formation of surficial fine grained laminae [SFGL] (Droppo and Stone, 1994) that represents a significant potential sink for contaminants bound to cohesive sediment (Stone and Droppo, 1994). Several recent studies have advanced knowledge regarding processes that govern the formation and erodibility of SFGL. For example, Droppo et al. (2001) conducted experiments in an annular flume using commercially available kaolinite clay and contaminated bed sediment from Hamilton Harbour to assess the effect of depositional history on the stability of contaminated bed sediment. Their results demonstrate that bed strength (erodibility) is dependent on both the degree of bio-stabilization and the flow conditions under which the bed is deposited. In a related study, Lau and Droppo (2000) report that the critical shear stress for beds deposited under shear was up to eight times larger than for beds deposited under quiescent
conditions. In a series of sequential erosion/deposition experiments, Lau et al. (2001) demonstrated the effects of depositional history on sediment erosion and showed how the rate of erosion and the amount of sediment eroded for a given level of shear stress are a function of the structure of the bed and flocs that formed it. This research demonstrates that layers of sediment deposited with different depositional history will not have the same shear strength and therefore similar flow conditions will not necessarily produce the same erosion rates.

Using a rotating circular flume, Krishnappan and Marsalek (2002a) measured the transport characteristics of cohesive sediment deposits in an on-stream storm water management pond. The critical shear stress for deposition (0.050 Nm⁻²) and erosion (0.12 Nm⁻²) of pond sediment were determined and used to develop empirical relationships to estimate sediment deposition and erosion as a function of shear stress. The used the data to develop a new model to predict transport characteristics of sediment from an on-stream storm water management pond (Krishnappan and Marsalek, 2002b). Skafel and Krishnappan (1999) investigated the depositional characteristics of mud from Port Stanley harbor using a rotating annular flume and showed that duration of sample storage, presence of bacteria and textural composition of the sediment affected depositional behavior.

Millburn and Krishnappan (2003) carried out an intensive field program before river-ice break up and conducted controlled experiments in a rotating annular flume to determine the critical shear stress for erosion and deposition of Hay River sediment. They proposed a modeling strategy for analyzing the under-ice transport of cohesive sediments in the Hay River. Krishnappan (2000) developed a new algorithm for the transport of fine sediments in the Athabasca River based on laboratory experiments in a rotating circular flume. Accounting for differences in the critical conditions for erosion and depositional processes of fine sediment, the algorithm was incorporated into and improved the performance of the contaminant transport model (WASP5). These algorithms have been used to improve accuracy of the models RIVFLOC and FINSED that were initially developed by Krishnappan (1991, 1997).

Stone et al., (Submitted) used an annular flume to determine erosion characteristics and bed stability of fine sediment deposits in streams that drain wildfire-affected landscapes. Biofilms were grown in the flume on cohesive stream bed sediments collected from a wild-fire affected stream (Lynx Creek) and a reference undisturbed stream (Castle River) in southern Alberta, Canada. Factors examined that influence sediment erosion, settling and bed stability included applied shear stress, geochemical and physical properties of the sediment, floc structural characteristics and biofilm growth period (2, 7, 14 days). Erosion characteristics and sediment properties were strongly influenced by wildfire, consolidation period and biostabilization of the bed. The fire-modified Lynx Creek sediment was found to be more resistant to erosion than the reference unburned Castle River sediment. The measured critical shear stresses for erosion were 1.6 and 1.8 times higher for the burn-associated sediment after 7 and 14 days of
consolidation (Fig 13). Erosion depths significantly decreased with bed age; however, these depths were significantly greater as a result of wildfire-associated biostabilization. Results of their study suggest that biogenic sediment stabilization will have a significant effect on the rates and magnitudes of sediment erosion and associated contaminant transport in wildfire-affected streams. Once the critical shear stress for erosion is exceeded, the eroded cohesive materials which have low settling velocities will remain in the water column and be transported downstream. Any attempt to model sediment transport from wildfire-affected streams will require consideration of how biofilms influence sediment erodibility.

Figure 13: The effect of biofilm formation on the transport properties of reference unburned sediment (left - Castle River) and burned sediment (Right Lynx Creek) as a function of bed age and shear stress (Stone et al., submitted to Water Research).

2.5.2 Field Studies:

A series of field studies have been conducted with portable flumes to determine the in situ transport properties of cohesive sediment in lakes and rivers. Droppo and Amos (2001) used an in situ annular flume to assess the effect of shear stress on the structure and stability of bottom sediments in Hamilton Harbour. They used sediment erosion and floc characterization data to propose a general three layer model that describes the formation, structure and stability fine sediment deposits. The model depicts a surface organic floc layer (Layer 1) that compresses within a collapse zone (Layer 2) to form a consolidated bed (Layer 3) shown in Fig. 14. They showed that the structure of eroded materials evolved from low-density flocs from the fine-grained surface layer to dense aggregates of the consolidated bed. In a related study, Amos et al. (2003) compared three methods to estimate the threshold shear stress ($\tau_c$) of lakebed sediment using the benthic flume Sea Carousel. The method, extrapolates a regression of suspended sediment concentration and fluid transmitted shear stress, is recommended for evaluation of
the erosion threshold conditions.

Figure 14: General three-layer model of cohesive bed sediment illustrating A) the transformation of SFGL into a collapse zone and finally into a consolidated bed and B) conceptualization of an erosion sequence for the three layer model (Droppo and Amos, 2001).

Krishnappan (2000) used a submersible laser particle-size analyzer to show that suspended solids in the Fraser River downstream of a pulp mill outfall were transported as flocs and that fibrous organic material in the effluent promoted flocculation of inorganic solids suspended in the water column. He demonstrated that flocculation of suspended solids by pulp mill effluent increased the deposition rate of sediment in the river. In an evaluation of the structure and composition of suspended sediments upstream and downstream of a point source of pulp mill effluent discharged into the Fraser River, no significant differences in the morphological characteristics (fractal dimensions) and only a very small increase in the $d_{84}$ of the floc populations downstream of the effluent were found (Petticrew and Bickert, 1998; Bickert, 1999). The significant seasonal variation in floc size far exceeded the near-field effects of the pulp mill effluent and no significant differences were observed between the upstream and downstream sites with regard to the total amount, size and deposition rate of fine sediment collected in gravel traps. While this result differs from the findings of Krishnappan (2000), the differences can be explained by the use of different sampling techniques and by the scale at which the effect of the effluent was evaluated. Krishnappan (2000) tracked the plume downstream using an in-situ laser particle analyzer, whereas Bickert (1999) used field collection and laboratory microscopy and sampled at stationary sites approximately 300 and 600 m downstream of the effluent pipe.
2.6 Entrapment of fines in coarse sediment beds

The entrapment of cohesive sediment in coarse bed substrate influences both the quality of the river habitat and the roughness characteristics of the river bed. Excessive inputs of fine sediment to streams can fill in the interstitial void space and ultimately changes the grain size distribution of the material forming the bed layer which can have an effect on the ecology of the hyporheic zone. The entrapment of fines in coarse gravel beds has been studied extensively in flumes but to a much lesser extent in the field. Packman et al. (2000) studied coarse gravel fine sediment interactions in a straight flume and developed a bed-form induced advective pumping model to calculate the colloid exchange between the water column and the hyporheic zone. In this model, the dynamic head variation along the stream bed resulting from bed forms was considered to be the driving force for subsurface flow in the hyporheic zone. They demonstrated that the net exchange of fine sediment from the stream into the hyporheic zone occurs because of the settling velocity of the sediment particles and the sediment immobilization in the bed due to the filtration effect.

Using a rotating circular flume, Krishnappan and Engel (2006) studied fine sediment entrapment under plane bed conditions. The flume was covered with different sizes of coarse sediment and pre-mixed slurry of kaolin was injected into the flume. The concentration of fine sediment in suspension was monitored as a function of time to evaluate entrapment. The results of the entrapment experiments were analyzed using a mathematical model (Krishnappan and Marsalek, 2002) for characterizing fine sediment transport in a rotating annular flume and the data were used to introduce a boundary condition into the model to simulate fine sediment entrapment in coarse gravel substrate. Further research (flume studies) is required to determine the entrapment coefficient for various bed material types and bulk hydraulic parameters and field studies are required to validate the model under a range of field conditions.

Lambert and Walling (1988) developed a technique for investigating fine sediment storage in gravel bed river channels and documented its application to sediment bed storage in the River Exe. They estimated that the total amount of fine sediment stored within a 35 km reach of the river ranged from a minimum of 100 t to a maximum of 400 t and determined that the mass of stored fine sediment in the gravel bed was relatively low in comparison to the annual suspended sediment output from the basin. While they discounted the channel storage in gravel beds as an important sediment source during runoff, they did not preclude the possibility of a quasi steady state where a large proportion of the suspended sediment transported by individual storm events is deposited on the recessional limb of the hydrograph then remobilized during a subsequent event. Accordingly, under such conditions, the transient storage of fine sediment may represent a more significant proportion of the sediment load.
3.0 The effect of land-use change on the source and fate of sediment-associated contaminants

Cohesive sediments are typically found in small quantities in proportion to their total load and their transport rates appear to be unrelated to flow parameters and depend on supply rates. Accordingly, an understanding of key hydrological processes and factors affecting land use change that influence the spatial and temporal variability in cohesive sediment supply to and transport in rivers is central to developing an appropriate management response. This section reviews some of the literature that describes the effect of land-use change on the source and fate of sediment-associated contaminants in river systems, with a particular focus on cohesive sediment.

3.1 Sources of fine sediment

The two main sources of sediment available to the river are 1) channel sources (mainly derived from the stream bed and banks and tributaries and 2) non-channel sources within the watershed (i.e. soil erosion and mass wasting processes). While the supply of sediment from channel sources is strongly related to discharge and bed stability, the supply of non-channel sediment is highly variable and depends upon the mode of production and transport to the stream (Waters, 1995; Collins and Walling, 2007a, b). The main sources of cohesive sediment available to the stream from channel sources are: river banks subject to erosion under high shear; mid-channel and point bars subject to erosion; fine bed material stored in the interstitial matrices or channel deposits (i.e. SFGL); flood plains and fine particles trapped within aquatic macrophyte stands. The main non-channel sediment sources include: eroded soils (sheet, rill and gully erosion); mass wasting processes; increased sediment delivery due to agriculture and urbanization. Processes that control the inputs of non-channel sediment are controlled primarily by land use, geology, soil type, climate, vegetation (Walling 1983; Ritter et al., 2002). The influence of these factors will vary depending upon the time of year and the magnitude, frequency and duration of precipitation events. In the following section, selected studies describing the effects of agriculture, forestry, urbanization and flow regulation on availability, transport and effects of fine sediment are presented.

3.1.2 Agricultural

The delivery of sediment from uplands to streams by overland erosion is controlled by geology, soil type, vegetation, slope and landuse. It is widely known that agriculture represents a significant sediment source to streams in most watersheds (Waters, 1995). The contribution of sediment from agricultural lands depends upon agricultural practices such as row crop cultivation on floodplains and livestock grazing in riparian zones (Armour et al. 1991). In a study of sediment sources in the Frome catchment (UK), Collins and Walling (2006) reported that the mean relative contributions from areas of
woodland, pasture and cultivation, and from channel banks/subsurface sources ranged between 1 ± 1%–6 ± 2%, 10 ± 2%–42 ± 2%, 44 ± 4%–81 ± 2% and 7 ± 2%–19 ± 4%, respectively. The corresponding mean relative contributions in the Piddle study area were 1 ± 1%–11 ± 4% (woodland), 10 ± 2%–28 ± 4% (pasture), 44 ± 2%–80 ± 2% (cultivated) and 7 ± 2%–21 ± 2% (channel banks and subsurface sources). The study demonstrates that the deposition of fine sediment in the stream has important implications for stream ecology and identifies the need for establishing a scientific basis for assisting the targeting of sediment management and control policies.

The contribution of agriculture to non-point pollution of watercourses has been the focus of increasing attention in recent years. A number of well-documented environmental problems in aquatic ecosystems have been attributed to the excessive loading of fine sediment and associated nutrient loadings. For example, nutrient enrichment and eutrophication encourage the loss of species diversity and community structure and increased fine sediment fluxes impact detrimentally upon macrophyte communities, fish populations and invertebrate biodiversity. Elevated fine sediment and associated nutrient levels in aquatic ecosystems frequently originate from farmed land and, thus, represent the off-site products of the delivery of pollutants from agricultural diffuse sources in river basins (Collins et al., 1997; Clarke and Wharton, 2001; Carpenter et al., 1998; Acornley and Sear, 1999; Scullion, 1983).

Subsurface drainage (tile drains) is a common agricultural water management practice in areas with seasonally perched water tables or shallow groundwater. The extent of agricultural tile drains in many agricultural regions has caused concern regarding its potential undesirable effects on surface and subsurface water quality. Tile drainage and related agricultural runoff are sources of bacteria (Panti et al. 1984), contaminants (Richards and Baker, 1993) and suspended solids (Culley et al., 1983). However, the role of tile drains as a fine sediment source represents a relatively unknown component of sediment budgets and little is known about the particle properties and fluvial transport characteristics of tile sediment in relation to varying soil texture, land use and moisture conditions. Stone and Krishnappan (2002) examined the effects of irrigation on tile sediment transport in a headwater stream near Kintore Ontario. Estimates of sediment loading from tile drains reported in the literature range from 1 to 220 kg ha\(^{-1}\) but the sediment yield measured from a controlled irrigation event was 4.6 kg ha\(^{-1}\). They reported that the Kintore tile sediments were predominantly fine-grained (\(d_{50} \approx 5\ \mu m\)). During the irrigation event, tile effluent represented 66% of the mass of suspended solids measured in the stream downstream of the tile outflow. Flow in the study reach was modelled with MOBEd to determine the bed shear stress and relate the size characteristics and degree of flocculation to previously conducted flume studies (Stone and Krishnappan, 1997).
3.1.3 Forestry and related landscape disturbance

It is widely recognized that vegetation change in forested watersheds, either through natural vegetation change processes (i.e. wildfire, pine beetle infestation, drought) or human induced (prescribed burns, logging, recreation) has significantly impacted a range of natural resources as well as source water quality and quantity. The frequency and severity of large-scale natural disturbances such as wildfire in forested regions of North America has significantly increased in recent decades, primarily due to warmer spring and summer temperatures and drought (Westerling et al., 2006). Because of the severity and magnitude of wildfire related landscape disturbances, sediment fluxes (Silins et al., 2008, 2009; Blake et al., 2005; Moody and Martin, 2001) are modified at rates and magnitudes that cause profound and often irreversible changes in river system function (DeBano et al., 1998; Bladon et al., 2008). From a geomorphic perspective, vegetation changes such as wildfires lower erosion thresholds in watersheds causing increased runoff and erosion to occur. The magnitude of wildfire disturbance depends on factors such as the sensitivity of the watershed to erosion, precipitation regime, geological and vegetation characteristics, the severity and areal extent and frequencies of burns. Wildfires change the infiltration and runoff properties of hill slopes by altering the soil properties (Moody & Martin, 2001) which ultimately influences the sediment and nutrient flux in watersheds (Blake et al., 2005; 2007; 2009). Accordingly, literature on the response of forested watersheds to wildfire has focused on understanding 1) hill slope hydrology and river flow and 2) soil erosion and sediment associated nutrient/contaminant transfer.

Knowledge of the type and quality of eroded materials supplied to and transported by rivers (e.g. trace elements, nutrients, organic matter content, particle size distribution) is critical from a modelling perspective to quantify wildfire impacts on aquatic ecology and water quality. Accordingly, there has been an increasing focus on evaluating the physical and chemical properties of burned soils (fire modified aggregates) and related particulate matter in streams at the reach and hill slope scale (Blake et al., 2005; Petticrew et al., 2006). While the onsite effects of wildfires are known, the downstream effects of fire modified aggregates and their impact on water quality are less well understood. There is a knowledge gap regarding the physical properties (particle morphology, density, porosity, settling velocity, particle strength) of fire modified aggregates in streams draining forested catchments at the regional scale. Using an annular flume to characterize the transport characteristics of cohesive sediment collected from streams draining burned and unburned landscapes, Stone et al (submitted) reported there was ~ 2 fold increase in the critical shear stress for erosion for the burn-associated sediment which was attributed to the formation of biofilms (Fig.15). They also reported that median settling velocity and porosity of the burn-associated sediment was lower than cohesive materials collected from unburned streams. Improved knowledge of these sediment properties and their subsequent impact on the erosion, transport and deposition behavior of
fire modified aggregates in streams is central to modelling post fire sediment storage and redistribution dynamics so that appropriate source water protection and management protocols can be developed for use by utility managers and other relevant stakeholders (e.g. regulators, consulting engineers).

Figure 15: Formation of biofilms in wildfire impacted streams (right) compared to streams in reference (unburned) watersheds (Photographs: Dr. U Silins).

Few studies have been conducted to determine the impact of large scale land disturbance (i.e. wildfire) on sediment concentrations in on the eastern slopes of the Rocky Mountains. After the 2003 Lost Creek fire which burned 21 000 ha of forested land in the Oldman River basin (Alberta), Silins et al. (2008; 2009) monitored concentrations of suspended solids in watersheds with varying degrees of natural and man-made land disturbance (burned, post-fire salvage logged, unburned) for three years to assess sediment concentrations and production (export and yield) over a range of flow regimes (spring melt, baseflow and stormflow). They reported that suspended sediment concentrations were 6-times higher in burned watersheds and 11-times higher in post-fire salvage logged watersheds, than in unburned watersheds. Sediment availability was greater in both burned and post-fire salvage logged watersheds but varied with flow condition; particularly during the snowmelt freshet and stormflow. In burned watersheds, sediment yield was 5-times higher during snowmelt and 13-times higher during stormflow than in unburned watersheds. Post-fire salvage logging produced much greater impacts than wildfire alone, with mean sediment yield 19-times higher during snowmelt and 9-times higher during stormflow compared to unburned watersheds.

3.1.4 Urbanization

It is well known that urbanization affects the amount and timing of sediment delivery to river channels and can also profoundly influence the water quality of receiving water bodies. In an early study
of the effects of construction on fluvial sediment in urban and suburban areas of Maryland, Wolman and Schick (1967) reported that sediment concentrations from areas undergoing construction ranged from 3000 to over 150,000 ppm, whereas in natural or agricultural catchments the highest comparative concentration was 2000 ppm. In terms of annual values, yields from construction areas range from several thousand to a maximum of 140,000 t/m$^2$/yr (i.e., up to 55,000 t/km$^2$/yr) from a small area. Total yield declines with increasing drainage area as a result of dilution from waters draining urban and other land not actually under construction. Observations demonstrate that sediment storage occurs on construction sites as well as in valley bottomlands. Actual yields from a given unit surface may be even larger than those derived from measurements in streams. Data on erosion from road cuts in Georgia, when converted to soil loss per area, result in sediment yields similar to those from building sites: 50,000—150,000 t/m$^2$/yr (20,000–50,000 t/km$^2$/yr), and local measurements indicate depths of erosion on road cuts of 0.1–0.2 ft (3–6 cm) over time intervals of generally less than one year.

According to Droppo et al. (2002), the urban continuum as it applies to sediments and associated contaminants, represents an area over/through which sediments are conveyed from a sediment source to a receiving water body. This concept comprises 4 linked compartments/pathways (Fig. 16). Accordingly, as it moves through the urban continuum sediment undergoes a variety of physical, chemical and biological transformations that ultimately influence its environmental fate and effect. Hydraulic sorting of particles in urban runoff results in a winnowing of the fines which tends to enhance contaminant transfer (Fig 17). In a study of trace metal composition and speciation of street sediment in Sault Ste Marie, Stone and Marsalek (1996) reported that metal levels of urban street sediment exceeded the lowest effect levels specified in Ontario Provincial Sediment Guidelines for metals and that a significant fraction of these metals were potentially bioavailable. Elevated levels of Cr, Cu, Zn, Pb and Mn in the exchangeable and soluble phases suggest that the sediment associated metals when mobilized from the streets of Sault Ste Marie during runoff and snowmelt would adversely impact water quality in receiving waters. Lévesque and De Boer (2000) investigated the trace element chemistry of surficial fine-grained laminae in the South Saskatchewan River. Concentrations of Cu, Zn, Cd, Pb and U in cohesive sediment deposits downstream of the City of Saskatoon were significantly greater than upstream samples collected on days when the flow velocities were low. These studies demonstrate the significance of urban centers as sources of fine sediment and associated contaminants that are transferred to aquatic environments where their can adversely impact biota.
3.1.5 Flow Regulation and impoundments

The construction of dams for water supply, hydro electric power generation and flood control have dramatically altered the sediment flux in many rivers of the world. It has been estimated that > 40% of the water discharge and world’s rivers is controlled by impoundments and that as much as 25% of the sediment flux from the land to oceans is trapped by dams (Vorosmarty et al. 1997). Meade and Parker (1985) report that over a 25 year period, the Missouri River sediment input to the Mississippi River was reduced by 25% of its former value. Depending upon the nature and quantity of non-channel sediment sources delivered to river channels (Walling 1983; Ritter et al., 2002), the magnitude and spatial variability of sediment and associated contaminant storage in reservoirs will reflect both the contaminant
content and the amount of fine sediment involved. Ziegler and Nisbet (1995) predicted patterns of cohesive sediment erosion and deposition in Watts Bar Reservoir (Tennessee) using the SEDZL model. The model showed that 76% of the incoming sediments for the 30-year period of record would be trapped in the reservoir, which was in agreement with the observed trapping efficiency of 72±6% (95% confidence interval). Van Metre and Mahler (2004) used sediment coring techniques to demonstrate the impacts of influent stream water quality on that of reservoirs. There is increasing concern that reservoirs can alter downstream ecosystems (Stanford and Ward, 1979; Petts, 1984) by modifying the downstream flux of water, sediment and water temperature and creating barriers to upstream-downstream movement of organisms and nutrients (Poff and Hart, 2002). These fundamental alterations to the abiotic environment have significant downstream ecological consequences (Pizzuto, 2002).

As part of the Southern Rockies Watershed Project, Dr. Stone and colleagues are currently participating in a study to evaluate the effect of wildfire on the downstream propagation of sediment associated contaminants in the Oldman River basin. Preliminary results indicated that significant quantities of fine sediment have been deposited in the Oldman reservoir (Fig. 18). This material is predominantly fine grained (<30 µm) and has elevated concentrations of phosphorus and some metals (Stone unpublished data). Fig 17 shows that sediment in the Oldman reservoir is being directly impacted by source materials primarily generated by the Lost Creek wildfire. In addition to wildfire, it has been demonstrated that agricultural activities on range lands can have a significant impact on the bacterial composition of sediment in reservoirs (Stephenson and Rychert, 1982) which can lead to taste and odor problems in drinking water and negatively influence the health of livestock.
Approximate layers sectioned in the Oldman reservoir sample (D3).

Figure 18: The effects of wildfire on sediment deposition in the Oldman Reservoir (Photo by M. Stone)
4.0 Frameworks and methodologies used to evaluate the provenance and fate of sediment in river environments

4.1 Introduction

Growing recognition of the important role of fine sediment in the transfer and fate of nutrients and contaminants through aquatic systems (e.g. Balades et al., 1984; Boxall and Maltby, 1995; Meharg et al., 1999; Owens et al., 2001; Kronvang et al., 2003; House, 2003; Warren et al., 2003; Collins et al., 2005; Owens et al., 2005; Carter et al., 2006; Horowitz et al., 2007), and in the degradation of aquatic habitats, including fish spawning gravels (e.g. Iwamoto et al., 1978; Crisp, 1993; Alonso et al., 1996; Newcombe and Jensen, 1996; Acornley and Sear, 1999; Suttle et al., 2004) and macrophyte beds (Best et al., 2001), has emphasized its wider environmental and ecological significance as a diffuse source pollutant. Effective sediment control strategies are therefore a frequent requirement in catchment management plans, in order to reduce the associated problems.

Although there is a need to develop generic sediment management frameworks that can be used in any catchment, it is important to remember that each catchment is inherently different and unique. Consequently, different objectives, pressures, impacts and mitigation measures will need to be considered in individual catchments and even at different sites within a given catchment. Due to such complexity and inevitable budget constraints, no reasonable monitoring programme can sufficiently address all aspects of potential interest in all catchments. As a result, sediment management frameworks must be designed to identify, the most important issues requiring assessment, their uncertainties, and the kind of decisions they will inform. The challenge is therefore to develop, refine and use frameworks and methodologies that are sufficiently generic to be relevant to most catchments, but which can be tailored to specific circumstances.

In the above context, it is possible to define some key attributes that would typically need to be included in any generic sediment management framework:

- A conceptual model of the river basin sediment system to allow identification of all possible sources, pathways and receptors/impacts
- A spatially explicit approach, allowing identification of key sources, impact regions and transfer pathways both within and between them
- The ability to consider multiple cross-sector sediment sources with differing temporal and physical-chemical characteristics and connectivity with channel systems
- The ability to define multiple cross-sector endpoints or management targets reflecting a number of criteria including impacts on key indicator ecological species recognised by water policy or legislation and social acceptability
The ability to test the sensitivity of the approach to data uncertainties and/or data/knowledge gaps
The ability to identify key information gaps to focus future monitoring and data collection
The ability to represent the impacts of mitigation measures or changing environmental conditions (e.g. climate, land use)

The following sections detail some key frameworks and methodologies appropriate for supporting improved sediment and associated contaminant management at catchment scale. In short, the frameworks comprise those incorporating sources, pathways and receptors/fate/impacts (DPSIR, relative risk model, sediment budgets) and those detailing sediment sources alone. Available techniques and methodologies for applying these frameworks of contrasting complexity are discussed.

4.2 The DPSIR framework

The complexity of the sediment problem at catchment scale and the numerous challenges associated with policy implementation demand that the entire catchment system be framed within a common conceptual framework. A pragmatic policy framework is required that recognizes the environmental significance of sediment and associated contaminants, taking account of multiple sources and impacts, within an ecological and socio-economic context. For ease of comparability, the framework should be common to, and readily applicable in, all catchments, although the detail of the individual components will recognise the catchment-specific nature of the sediment problem.

The DPSIR (Driver-Pressure-State-Impact-Response) framework provides a pragmatic holistic systems approach based around a chain of cause and effects (EEA, 1999, 2000). It represents an extension to the PSR framework developed by the Organisation for Economic Cooperation and Development (OECD, 1993). As a framework, DPSIR highlights the complex linkages between pressures and impacts and stakeholder response to these. The framework has recently proved useful within the context of soil erosion by water by helping to structure the interpretation of policy-relevant indicators of soil loss across Europe (Duwel and Utterman, 1999; Gobin et al., 2004; Davies and Rees, 2004). The DPSIR approach has also been used to structure an investigation of the linkages between socio-economic drivers and nutrient and contaminant fluxes in the Humber catchment, UK (Cave et al., 2003), to provide an historical perspective on heavy metal contamination in the Seine River basin, France (Meybeck et al., 2007) and to provide a framework for a GIS-based screening tool at European scale (Guipponi and Vladimirova, 2006). Other applications of the DPSIR approach in pollution studies include the work by Elliot (2003).

By using indicators to characterize each component in the chain of human-environment interaction for a given pollutant such as sediment, DPSIR simplifies the interpretation and synthesis of information in order that it is most useful for stakeholders. The approach also provides transparency to
the decision-making process in relation to formulating a catchment management plan. The DPSIR approach complies with the planning and management processes of current water policy e.g. the EU Water Framework Directive (European Commission, 2003) and provides simplicity. The DPSIR framework does not concentrate upon a detailed analysis of the environmental problem, but rather provides information to support and clarify the reasoning behind manager’s decisions. Above all, the DPSIR approach facilitates the collation and synthesis of socio-economic and environmental data to support decision-makers in their quest for integrated and sustainable management strategies. As a result, the DPSIR approach is most readily applied in those catchments where substantial volumes of empirical data already exist for the various components of the catchment sediment pressure-impact system.

4.3 The relative risk framework

Clearly, when management decisions are being made at drainage basin or larger scales, addressing the effects of multiple stressors on multiple receptors in various habitats, it becomes increasingly difficult to determine what issues might be dominant. There have to date been very few environmental management strategies based upon catchment or watershed-scale considerations, but it is impossible to assess realistically the relative impacts of various activities and potential mitigation actions unless they are examined at this scale. Unlike management carried out at a specific site, catchment or drainage basin scale management must address a complex mix of risk scales, sources, drivers and receptors. Given the realities of limited resources for targeting and implementing mitigation options, it is clear that any expectation of complete risk characterisation and/or removal will not be met. Rather, risk mitigation at the larger (e.g. regional) scale (and, realistically, even at all but the most simple of sites) seeks to provide maximum net risk reduction within the region with the resources available (Landis, 2005). Successfully achieving such goals will require meaningful conceptual models, but will also require a clear understanding and definition of net risk within a catchment or region. Whilst there are well-established methods for assessing ecological or human health risk at a specific site or region, if only one type of pressure is considered at a time, it remains difficult to inform decisions effectively which have to address cumulative or net risk to various receptors at various scales. One promising approach to addressing such complexity is the relative risk model (RRM), which develops regional-scale risk assessments. Regional scale risk assessment using the RRM is defined by (Landis, 2005) as:
A risk assessment (which) deals at a spatial scale that contains multiple habitats with multiple sources of multiple stressors affecting multiple endpoints, and the characteristics of the landscape affect the risk estimate. Although there may only be one stressor of concern to the decision maker, at a regional scale the other stressors acting upon the assessment endpoints are to be considered.

Figure 18: The relative risk model (RRM) approach (Landis, 2005).

This approach is designed to assess the probability of impacts of a variety of potential stressors in a spatially explicit manner within a dynamic landscape, and is thus ideal as a starting point for developing frameworks for assessing and managing the disparate impacts of sediment within catchments. The model uses data on land use, hydrology, types of contamination, the distribution of assessment endpoints and the history of disturbance to derive measures of risk gradients within systems being managed. The calculation uses a process of setting ranks for a broad range of sources, stressors and habitats with filters to characterize exposure and effects to look at combined risk at large geographic scales (see Figure 18). It provides conceptual and graphical tools to summarize and communicate cumulative risk as a function of source, stressor, assessment endpoint or site. The RRM approach has recently been applied to support strategic sediment management in England.

Landis (2005) describes ten steps for using the RRM to carry out a regional risk assessment. These can be linked to steps in a standard ecological risk assessment (ERA) framework but, as ERA examines the effect of a single stressor on a single site, they are not identical.

1. List the important management goals for the region. What do you care about and where?
This requires a definition of what is to be protected. Once management objectives are identified, they must be translated into quantitative measures that will be used to define those goals in the context of a RRA. Risks to these objectives can then be quantified and mitigation measures can be evaluated in the context of these goals and measures. Management objectives are, in part, defined by statutory requirements, but they must also take into account the viewpoints of disparate groups of stakeholders within a catchment. Management goals may include ecological, recreational, regulatory and/or socioeconomic objectives. They can be objectives such as safe and palatable drinking water, a sustainable recreational and commercial fishery, navigable waters, floodplain management, etc. At times, two desired goals may appear to be in conflict, but goals addressed should be as inclusive as possible. Ultimately, RRA may show the goals to be compatible, or may provide information such that decision makers can make informed trade-offs between goals.

2. Make a map. Include potential sources and habitats relevant to the management goals.
   An important part of a catchment-scale or regional risk model is that risk is spatially distributed in a dynamic system, and that the potential links between risk sources and assessment endpoints must be addressed in a spatially explicit way. The RRM provides a framework for mapping multiple stressors and multiple risk pathways in a transparent manner.

3. Break the map into regions based on a combination of management goals, sources and habitats/target zones.

4. Make a conceptual model that links sources of stressors to the receptors and to the assessment endpoints.
   This step will be the one that frequently requires the most conceptual development. In Phase 1 of many sediment management projects, the objective is to identify expected sediment-specific risk sources, and the models, tools and indicators that can be used to rank them in a spatially explicit manner. In subsequent phases, the generic framework can be tailored to specific catchments with objectives and conditions that are representative of regional pressures and impacts. In all applications of the RRM, the pathway between source and habitat/target is implicit and is not characterized to a great extent. However, we know for sediment, that sources may not impact on all sites downstream. Whilst a distributed modelling approach is often sought, the reality is that there are frequently never enough spatial or temporal data to drive such models. Under such circumstances, other methods must be used to elucidate the upstream-downstream (source-habitat) linkages. Such approaches might be based on landscape analysis, stream power, stream
flashiness, critical shear stress values, flow-duration curves, sediment exceedance curves, river morphology or sediment budget approaches. The complexity of approach used will almost certainly be constrained by data availability for the catchment or region in question.

5. Decide on a ranking scheme to allow the calculation of relative risk to the assessment endpoints. This task requires an explicit definition of the indicators, models, tools and measures that will be used to define links in the conceptual model, and how they will be quantified and ranked. This should be based on best available knowledge or expert judgement as well as data.

6. Calculate the relative risks.

7. Evaluate uncertainty and sensitivity of the relative risks.
   There will be uncertainty in both the data used to quantify risks and as a result of differences of opinion between stakeholders over management endpoints. A set of scenarios reflecting such uncertainty can be defined and tested to illustrate the possible range of outcomes. This information can then be used both in setting management priorities and defining monitoring requirements to reduce uncertainty.

8. Generate testable hypotheses for future field and laboratory investigations to reduce uncertainties and to confirm the risk rankings.

9. Test the hypotheses listed in Step 8.
   If the model predicts high risks as a function of source, site, etc., available datasets can be examined to assess whether there is a weight of evidence to confirm risks.

10. Communicate the results in a fashion that portrays the relative risk and uncertainty in a response to the management goals.
    Both application of the framework and any output from it must be transparent and comprehensible to the wide range of catchment stakeholders likely to be involved. Because sediment is a cross-sectoral issue, results must be framed in language that is relevant to each sector.

4.5 The sediment budget framework

An alternative to the DPSIR approach is to focus on the sediment continuum per se as opposed to any resulting impacts such as those on aquatic ecology or related socio-economic circumstances and
factors. The precise link between upstream erosion and sediment mobilisation and downstream sediment yield and contaminant transfer involves many uncertainties, due to sediment retention and both short- and longer-term storage at intermediate locations, such as the foot of slopes and the river channel and its floodplain. The proportion of the sediment mobilised within a catchment that is intercepted and stored during transfer or delivery through the catchment will frequently exceed the proportion exported (Slaymaker, 1982; Phillips, 1987; Trimble, 1995). From a management perspective, it is therefore essential to consider the sediment system in its entirety, as opposed to focussing on measuring the downstream fluxes. The sediment budget concept provides an effective basis for representing the key components of the sediment delivery system within a catchment and for assembling the necessary data to elucidate, understand and predict catchment sediment delivery (Swanson et al., 1982; Reid and Dunne, 1996; Renwick et al., 2005; Owens, 2005; Rommens et al., 2006). However, despite its clear utility for catchment management and in the design and implementation of measures for mitigating sediment-related diffuse source pollution, few practitioners are familiar with the concept, the various approaches that can be used to construct a sediment budget or indeed examples of their application (Reid and Trustrum, 2002). There is substantial scope for policy makers and catchment managers to make greater use of the sediment budget concept as a practical framework for targeting mitigation strategies (e.g. Wilkinson et al., 2005). Equally, the budgeting approach offers a useful means of improving our understanding of catchment response to different land use scenarios and management programmes, as well as longer-term climate change (Walling, 1995; Summer et al., 1996; Wasson et al., 1998).

4.6 The sediment budget concept

As a concept, the sediment budget approach was first applied over 40 years ago (e.g. Rapp, 1960) and is still being applied (e.g. Wilkinson et al., 2005; Collins and Anthony, 2008a,b; Walling and Collins, 2008). It is readily applicable at the catchment scale, which is now widely adopted as the most appropriate spatial unit for characterising and managing diffuse source sediment problems. Based on a mass balance of sources, sinks and outputs, the sediment budget of a catchment provides an effective means of addressing the need for a holistic understanding of the interaction and linkages between sediment mobilization, transport, storage and yield (Dietrich and Dunne, 1978; Slaymaker, 2003). The utility of the concept in relation to catchment management lies in the identification of the key sources, stores and transfer pathways and thus the sensitivity of a catchment to perturbations in either intrinsic or extrinsic controls. The sediment delivery ratio, which expresses the ratio of the sediment output or sediment yield from the catchment to the total sediment mobilization within the catchment, provides a valuable measure of the importance of storage and thus of the overall catchment response. What is now widely recognised to be a classic example of the value of applying a sediment budget approach in a
management context is provided by the work of Trimble (1983) in the 360 km² Coon Creek catchment in Wisconsin, USA. This work provides a clear example of how the sediment budget approach can help support understanding of catchment response to mitigation programs. Sediment budgets for this catchment were established for two periods, 1853-1938 and 1938-75 using a range of morphological, and sedimentological evidence, coupled with available prediction techniques (see Figure 19). During the first period, poor land management caused severe soil erosion, although, as the budget indicates, a major proportion of the mobilised sediment was stored within the basin and only ca. 5% of the mobilised sediment was transported out of the basin. During the subsequent period 1938-75, the widespread application of soil conservation measures resulted in soil erosion rates being reduced by ca. 25%. However, the sediment yield at the catchment outlet remained essentially the same, due to the low proportion of the sediment mobilized by soil erosion previously reaching the catchment outlet and the remobilisation of sediment stored in the tributary and upper main valleys during the second period. Thus although the application of soil conservation measures considerably reduced on-site problems of soil degradation, they had negligible impact on downstream sediment loads and associated water quality problems. From the latter perspective, the implementation of improved land management and soil conservation measures within the catchment could be seen as having been of very limited benefit.

Walling (2006) has highlighted a paradox associated with the application of the sediment budget approach, whereby, despite its obvious utility, it has proved difficult to assemble the necessary information to establish a reliable sediment budget for a catchment and the approach has therefore not been widely applied to support catchment management. In the case of the sediment budget for Coon Creek presented above, the aim was to highlight key differences between the two periods and the budget was based primarily on broadscale generalisations of the catchment behaviour, rather than measurements of the processes involved. In most studies aimed at establishing the contemporary sediment budget of a catchment, with a view to assessing the efficacy of potential mitigation measures in reducing downstream sediment fluxes, there will be a need for more detailed information on the sources, sinks and fluxes involved. Sediment mobilization, transport and storage are characterized by appreciable spatial and temporal variability (Walling, 1998) and it is necessary to take account of this variability when constructing a sediment budget.
There is no widely accepted or generally applicable procedure for establishing a comprehensive sediment budget for a catchment, because it has proved difficult to adapt traditional measurement techniques to address the spatial and temporal variability associated with sediment mobilisation and transfer processes at the catchment scale. Traditional techniques, including the use of erosion pins, profilometers and photogrammetry to document erosion rates, and the use of sediment traps or post-event surveys to document sediment storage, possess many logistical and operational limitations as well cost constraints (Collins & Walling, 2004). The potential for coupling recent advances in sediment tracing with more traditional monitoring techniques has, however, provided new opportunities to assemble the information required for sediment budget construction (Walling, 2003, 2004, 2006; Walling et al., 2001, 2006).

Figure 19: Sediment budgets for Coon Creek, USA (after Trimble, 1983).
Examples of sediment budgets and their utility for sediment management

The sediment budget provides a sensitive indicator of the sediment response of a catchment (Phillips, 1991; Nagle et al., 1999; Walling, 1999) and an effective means of targeting mitigation measures to optimise their effectiveness in reducing downstream sediment flux and assessing their likely impact on that flux. To demonstrate this utility, it is useful to present several examples of sediment budgets and their potential use in supporting catchment management.

Walling et al. (2001) used an integrated approach to establish the sediment budget for the 63 km² upper Kaleya catchment in southern Zambia (Figure 20). The budget indicated that areas of communal cultivation (76%) and bush grazing (16%) were the most important sediment sources within the catchment, although a significant proportion of this sediment was deposited during transfer to the river channel. Sediment storage in local reservoirs and on the floodplain bordering the main river channel was also shown to be appreciable. As a result, the overall sediment delivery ratio was estimated to be only 9%, indicating that implementation of mitigation strategies to control soil erosion and sediment mobilization within the catchment would not necessarily result in a major reduction in the sediment flux at the catchment outlet and that there would be a need for careful management of the main sediment stores, to reduce the risk of remobilisation of sediment from these depositional zones. In addition, the sediment budget highlighted the need for any sediment control strategy to include provision for the protection of eroding channel banks and gullies, since these contributed ca. 17% of the annual suspended sediment load measured at the catchment outlet and this source is directly connected to the channel system, with little or no opportunity for depositional losses during transfer from the source to the channel system.
Walling et al. (2002) used a similar integrated approach to assemble the information required to establish the sediment budgets for the Rosemaund (1.5 km$^2$) and Smisby (3.6 km$^2$) catchments in central England (Figure 21). The overall sediment delivery ratios for these catchments were higher than those of the upper Kaleya catchment at 17% and 20%, respectively, indicating that the sediment yields of these smaller catchments are likely to be more sensitive to land use change or sediment mitigation programmes. A significant proportion of sediment mobilised on local fields is, nevertheless, subsequently stored either within the fields or between the fields and the river channel. Failure to recognise these stores and the potential for remobilisation could result in an over-optimistic assessment of the scope for reducing the downstream sediment fluxes. Equally, the sediment budgets of these two catchments highlighted the need to manage sediment transfers via artificial drains, which serve to increase connectivity of the catchment surface to the watercourses and promote the rapid delivery of sediment to the catchment outlets during storm events. The management of sediment delivery via under-drainage can be difficult and poses problems for by-passing certain mitigation measures including riparian buffer strips (Collins et al., 2009).
The sediment budgets for a number of small and medium-sized river basins on the Russian Plain were reported by Golosov et al. (1992). In this case, the information used to construct the budgets included both field monitoring and sediment tracing as well as the use of prediction equations to estimate rates of soil loss. The budgets established for four of the catchments, which are presented in Figure 22, emphasise the wide range of sediment delivery ratios (0-89%) associated with this landscape. Accordingly, whereas it could be expected that changes in land use and erosion rates would not be readily reflected by the sediment flux at the outlet of the Veduga Creek catchment, the suspended sediment yield of KiJuchi Creek could be expected to be much more sensitive to such changes. The results from these four catchments emphasise the need to recognise the potential variability of catchment sediment budgets within even a relatively small region and thus variation in the effectiveness of improved management and mitigation measures in reducing the sediment fluxes at their outlets.
Figure 21: Sediment budgets for the Belmont and Lower Smisby catchments, England (Walling et al., 2002).
Figure 22: Sediment budgets for various sub-catchments on the Russian Plain (Golosov et al., 1992).

The sediment budgets for a number of small and medium-sized river basins on the Russian Plain were reported by Golosov et al. (1992). In this case, the information used to construct the budgets included both field monitoring and sediment tracing as well as the use of prediction equations to estimate rates of soil loss. The budgets established for four of the catchments, which are presented in Figure 22, emphasise the wide range of sediment delivery ratios (0-89%) associated with this landscape. Accordingly, whereas it could be expected that changes in land use and erosion rates would not be readily reflected by the sediment flux at the outlet of the Veduga Creek catchment, the suspended sediment yield of Kijuchi Creek could be expected to be much more sensitive to such changes. The results from these four catchments emphasise the need to recognise the potential variability of catchment sediment budgets within even a relatively small region and thus variation in the effectiveness of improved management and mitigation measures in reducing the sediment fluxes at their outlets.

As a final example, Figure 23 presents the sediment budgets established by Walling et al. (2006) for the Pang (166 km²) and Lambourn (234 km²) catchments in southern England using a combination of...
fallout radionuclide measurements, sediment source fingerprinting, and measurements of sediment flux at the catchment outlet and fine sediment storage on the channel bed. Both catchments are underlain by chalk, and the very low sediment yields are consistent with the low sediment yields commonly associated with chalk catchments. However, the sediment budgets emphasise that, although the sediment yields of these two catchments are very low, substantial amounts of sediment are mobilised by erosion within the catchments, but only a very small proportion of this reaches the catchment outlets. The budgets underscore the importance of cultivated fields as a sediment source and again highlight the importance of conveyance losses during the transfer of sediment from the slopes to the channel system, with 51% of the sediment mobilised from cultivated fields in the Pang catchment and 31% in the Lambourn catchment being sequestered between the fields and the river channel. The measurements of storage of fine sediment on the channel bed also emphasise the importance of this temporary store in the transfer of sediment through the channel system to the catchment outlet. The estimates of the magnitude of this store represent the mean storage on the channel bed during the year and the values indicated are of the order of 20% (Lambourn) and 40% (Pang) of the annual sediment flux at the catchment outlets. Since there will be frequent remobilisation and replenishment of this store, a substantial proportion of the sediment flux at the catchment outlet will have passed through this temporary store. Overall, the low sediment delivery ratios (ca. 1%) for both catchments indicate that their downstream sediment fluxes are unlikely to be sensitive to mitigation strategies targeting soil erosion and sediment mobilisation in fields. If, however, land use change or climate change was to cause a reduction in the in-field or field-to-river conveyance losses, appreciable additional quantities of sediment could be delivered to the watercourses, causing harm to the unique aquatic habitats supported by these chalk systems with their abundance of benthic invertebrates including nymphs, caddis larvae, stoneflies and white-clawed crayfish. Under current environmental conditions, eroding channel banks and subsurface sources are of minimal importance as a sediment source and do not require targeted mitigation in the Pang and Lambourn catchments.
Figure 23: Sediment budgets for the Pang and Lambourn catchments, southern England (Walling et al., 2006).

The above examples serve to emphasise that it is important for policy makers and catchment managers to recognise that many rivers appear to be characterised by a lack of sensitivity to land use change and mitigation programmes (Walling, 1999). This reflects, at least in part, the sediment delivery ratio, in that those rivers with a low sediment delivery ratio are likely to exhibit an enhanced buffering capacity and delayed response to improved management. The potential for remobilisation of stored sediment to offset or at least delay the downstream effects of a reduction in sediment mobilisation from the primary sediment sources within a catchment must also be recognised, introducing the need to raise awareness amongst stakeholders that any projected catchment response to sediment mitigation programmes could be delayed, at least in the short-term (Collins and McGonigle, 2008).

The importance of fine sediment as a diffuse source pollutant means that there is increasing recognition of the need to include provision for sediment control within catchment management strategies. However, it is important that the design of sediment control strategies should be founded on a holistic understanding of the sediment dynamics of the catchment concerned. A sediment budget fulfils
that need by providing key information on the sources, sinks and transfers involved. Focusing attention on an individual component of the sediment delivery system, without appropriate understanding of the overall sediment budget, may result in an incorrect assessment of the potential benefits of sediment mitigation programmes. It is therefore suggested that the sediment budget concept should be more widely adopted and utilised as a practical framework to support the design and implementation of sediment control programmes aimed at reducing diffuse source pollution by fine sediment and associated contaminants. Assembling the information required to establish a sediment budget is, however, likely to prove a demanding task in terms of the resources required, and may as a result constrain the use of the approach.

**Sediment sourcing frameworks**

In the absence of resources to construct the full catchment sediment budget, attention can focus on key sediment and associated contaminant sources. The adoption of a sourcing framework for data collection and interpretation provides a pragmatic basis for focusing on the key component of the catchment system that is most responsive to management and mitigation. The principal sources of the suspended sediment fluxes from many river basins, or sediment pressures more generally, have not been documented. It is, however, increasingly recognized that reliable information on the nature and relative significance of catchment sediment sources represents an important requirement. For instance, such information is necessary for establishing catchment sediment budgets (Reid and Dunne, 1996; Walling et al., 2001), for assisting the interpretation and modelling of suspended sediment yields (Dedkov and Moszherin, 1992; Summer et al., 1996) and for elucidating the importance of secondary sediment sources associated with the remobilization of sediment stored in depositional sinks (Phillips, 1993). Equally, an improved understanding of catchment sediment sources represents an essential prerequisite for assisting the design and implementation of targeted management strategies for controlling off-site sediment-associated environmental problems (United States Environmental Protection Agency, 1999; Collins et al., 2001a).

Despite the dearth of information on the provenance of fluvial suspended sediment loads or sediment pressures more generally, considerable progress has been made in terms of conceptualizing and understanding the key controls of erosivity and erodibility governing sediment mobilization in drainage basins (Morgan, 1995). More specifically, the potential for different portions of a river basin to contribute to the downstream suspended sediment flux is controlled by a complex interplay of factors of strength, morphological, locational and filter resistance (cf. Brunsden, 1993; Burt, 2001). The former two factors control sediment mobilization in situ and the latter govern the delivery of mobilized particles to the river channel. Whilst it is over simplistic to discuss these factors of resistance individually, their importance
with respect to the provenance of fluvial suspended sediment loads and pressures more generally can be usefully demonstrated in this manner. A classic example of the influence of strength resistance on sediment origin is provided by the recent shift in the UK from spring- to autumn-sown cereal crops. Autumn sowing renders bare rolled soils susceptible to erosion during winter rains by lowering strength resistance and is thereby partly responsible for an increase in the proportion of suspended sediment loads in the UK reported to be originating from cultivated fields (Boardman, 1990). Because locational resistance is dependent upon the juxtaposition of suspended sediment sources and the river, eroding channel banks may contribute significantly to suspended sediment loads (Collins and Walling, 2004). Filter resistance is controlled by the density of roads, paths, tracks or field drains and by the occurrence of hedges or buffer strips and is especially important in governing the delivery of sediment from distal sources to the stream network (Wemple et al., 1996; Laubel et al., 1999; Ziegler et al., 2000a,b, 2004; Wemple et al., 2001; Croke et al., 2005). Road-to-stream linkages lower filter resistance by enhancing drainage density and can be generated by gully growth between road drain outlets and river channels or direct river crossings by paved and unpaved routes (Croke and Mockler, 2001; La Marche and Lettenmaier, 2001). Equally, the reduced filter resistance caused by the low density of field boundaries or hedges in open uplands mean that such areas are frequently characterized by more efficient sediment delivery to river channels. Tectonic activity can also influence the magnitude of filter resistance. In tectonically active areas, drainage densities are typically higher, causing a reduction in filter resistance and an increase in slope–channel coupling. Alternatively, slopes and stream networks are frequently decoupled in tectonically stable areas, thereby increasing filter resistance to sediment delivery (Harvey, 1994). Under the latter circumstances, although soil erosion may be severe, the source of fluvial suspended sediment may be reworked material deposited in alluvial stores (Fryirs and Brierley, 1999).

Clearly, however, these factors of resistance typically interact. For example, even where a sediment source is characterized by low strength and morphological resistance to erosion, the sediment released may not contribute significantly to the suspended sediment load measured downstream if locational and filter resistance are high.

Given the catchment-specific nature of the complex interplay of the principal factors controlling sediment and associated contaminant mobilization and delivery to river channels, it is not surprising that existing studies of the origin of fluvial sediment report contrasting results. In many instances, the erosion of surface soils has been identified as the primary source of suspended sediment flux, reflecting the detrimental environmental impact of various land management practices including, amongst others, commercial forestry (Stott, 1986), grazing (Evans, 1997) and winter cereal production (Evans, 1990). Alternatively, research has also demonstrated that in some cases, channel bank erosion can be an important, if not dominant, source of suspended sediment loads (Duijsings, 1987; Church and Slaymaker,
Key catchment attributes including the percentage coverage of agricultural versus urban or industrial land use clearly play an important role in determining the significance of different types of sediment and associated contaminant sources.

**Key problems for sediment sourcing frameworks**

Assembling meaningful information on the primary sources of sediment pressures in river catchments is highly problematic. Although many difficulties and operational problems are specific to particular measurement and monitoring procedures (see following discussion), two principal constraints are common to most of the methods employed. First, suspended sediment sources are spatially and temporally variable in response to the complex interactions between the major factors governing sediment mobilization and delivery. It is therefore necessary to undertake measurements at a range of spatial locations and over different temporal scales in order to obtain representative data. The need to address potential spatiotemporal variations in the relative contributions from individual sediment sources inevitably introduces logistical, practical and sampling constraints. Secondly, the costs of employing many of the available methods constrain the spatial coverage and temporal duration of monitoring programmes and therefore further compound the problems of obtaining representative data. The problems of representativeness and cost combine to compromise the reliability of the information reported by many studies (Stocking, 1987; Collins and Walling, 2004).

**Examples of sediment sourcing frameworks**

Following visual appraisal of field evidence for sediment mobilization and delivery, potential sources of sediment pressures in a catchment should be grouped into categories in order to organize and rationalize data collection and interpretation (Figure 24). Because suspended sediment sources are predominantly diffuse in nature, it is inappropriate to adopt the point and nonpoint source categories traditionally used in water quality studies (Collins and Walling, 2004). A number of classification schemes can be employed depending upon whether information on the *types* or *spatial location* of individual sediment sources is required (Figure 25). Different types of suspended sediment sources can be readily classified in terms of hillslopes and river channels, or the surface and subsurface portions of a catchment, whilst spatial provenance can be easily categorized on the basis of individual tributary subcatchments or geological units. The precise components of these principal sediment source categories will inevitably depend upon the key land cover attributes and major erosive processes in the study catchment and any measurement or monitoring programme should be directed accordingly. Owing to the likelihood of sediment deposition, storage and remobilization, potential sediment source types can be further subdivided into primary and secondary categories and these can be either proximal or distal to river
channels (Figure 26). In contrast, spatial provenance groupings combine primary and secondary, as well as proximal and distal sediment sources. Selection of the most appropriate classification scheme is commonly governed by study catchment size and key data requirements. In smaller (~ <50 km²) catchments, where the number and spatial complexity of sediment sources can be expected to be lower, it is frequently most meaningful to investigate sediment provenance in terms of individual source types. Alternatively, in larger drainage basins, where the opposite is true, it is frequently more practical and meaningful to address the spatial location of sediment sources. The intended use of the sediment provenance data should, nevertheless, be carefully considered. In the context of catchment management, information on individual source types is generally more appropriate for devising sediment control strategies. Assembling such information for larger drainage basins remains, however, a difficult task and a compromise between data requirements and practical considerations may therefore be necessary. Similarly, the inclusion of all potential sediment sources in measurement programmes is frequently impractical and constrained by available resources. Consequently, most sediment sourcing investigations target the primary sources of sediment transport and delivery to receiving watercourses.

Figure 24: A logical framework for documenting catchment sediment sources (Collins and Walling, 2004).
Figure 24: An elementary classification of catchment sediment sources (Collins and Walling, 2004).

Figure 26: An example of an advance classification scheme for catchment slope and channel sediment sources (Collins and Walling, 2004).

Methodologies

*Sediment budget construction*

The demand for more holistic information to underpin sediment management and control strategies has strengthened the need for new approaches to assembling the data required to construct reliable catchment sediment budgets. One important development in this context has been the use of
fallout radionuclides ($^{137}$Cs, unsupported $^{210}$Pb, $^7$Be) as sediment tracers (Walling, 2004). Because such fallout radionuclides are commonly rapidly and strongly adsorbed by soil particles upon reaching the catchment surface as fallout, their subsequent redistribution proves a means of tracing sediment mobilization, transfer and deposition (Ritchie and McHenry, 1990; Zapata, 2002). Assessment of the post-fallout redistribution of the radionuclides offers a basis for documenting time-integrated rates and patterns of sediment redistribution and storage within the catchment system. The majority of studies employing fallout radionuclides to trace sediment mobilization and delivery have been based upon measurements of caesium-137 ($^{137}$Cs) activities and inventories. Caesium-137 is an artificial fallout radionuclide produced by the testing of thermonuclear weapons, during the period extending from the mid-1950’s to the early 1960’s. Upon release into the stratosphere, $^{137}$Cs was distributed globally and deposited as fallout, with the magnitude of the latter reflecting annual precipitation and location with respect to weapons testing (Walling, 2002). Although $^{137}$Cs fallout declined to near zero by the late 1970s, its relatively long half-life (30.2 years) ensures that detectable amounts remain in soils and sediment 40 years after the main period of fallout in the late 1950s and the 1960’s. By establishing the current distribution of $^{137}$Cs in the landscape, it is possible to assess soil and sediment redistribution during the period elapsed since the time of the main period of fallout and the time of sampling. Collection of soil cores, measuring their $^{137}$Cs content using gamma spectrometry and comparing their inventories with the corresponding estimate for an undisturbed reference site experiencing neither erosion nor deposition, using appropriate empirical relationships or theoretical conversion models (e.g. Walling and He, 1999a), provides a means of documenting medium-term (i.e. ca. 50-year) rates of gross erosion, deposition and net soil loss (e.g. Walling et al., 1999a; Collins et al., 2001). Although simple to use, empirical relationships (e.g., Bajracharya et al., 1998) are constrained by a number of limitations on account of being derived from erosion plots that may not be wholly representative of catchment conditions. The theoretical approaches currently available comprise the proportional, profile distribution and mass balance models. Early versions of the proportional model (e.g., Martz and De Jong, 1991) are over-simplistic and fail to take into account various factors including the effects of the removal of freshly deposited fallout before its incorporation into the tillage horizon. Similarly, early versions of the profile distribution model (e.g., Zhang et al., 1990) are hampered by a number of constraints including the failure to take account of the time-dependent nature of $^{137}$Cs fallout and the potential for its redistribution following deposition as fallout. Mass balance models provide a more rigorous accounting procedure (e.g., Yang et al., 2000) by addressing the principal limitations of the proportional model. Walling and He (1999a) proposed improved versions of the main types of $^{137}$Cs-based erosion models. Similarly, the $^{137}$Cs inventories measured in floodplain cores can be compared with those of a nearby reference site to estimate mean annual sedimentation rates (Walling and He, 1997).
A number of key advantages are associated with the use of the $^{137}$Cs technique to assess soil redistribution rates. These include the provision of retrospective medium-term (~40 years) information on erosion rates and patterns that encompasses the sum of all erosive processes and that avoids the need for longer term monitoring programmes. The $^{137}$Cs technique can be applied in a wide range of environments and at different spatial scales (Wicherek and Bernard, 1995; Nagle et al., 2000; Collins et al., 2001b), typically involves only a single site visit thereby overcoming the sampling constraints of traditional monitoring methods and involves only minimal site disturbance. The soil redistribution estimates derived for individual sampling points can be extrapolated to larger areas and $^{137}$Cs analysis is nondestructive. The principal limitations of the $^{137}$Cs approach include the costs of analytical equipment and the difficulties experienced in interpreting medium term estimates of average soil redistribution rates in the absence of complementary information on land use patterns and intensity. Application of the approach is problematic in heavily gullied landscapes and in semi-arid areas (Chappell, 1999). Methodological uncertainties are associated with selecting the optimum coring strategy necessary for characterizing the variability of $^{137}$Cs inventories in reference locations (Sutherland, 1996) and on hillslopes (Sutherland, 1994). The global pattern of bomb-derived fallout, which results in reduced inventories in some parts of the world, hampers $^{137}$Cs measurements on samples collected in equatorial regions and the Southern Hemisphere. Existing calibration models require validation (Porto et al., 2001).

The potential utility of unsupported $^{210}$Pb measurements in soil redistribution studies has been explored far less (Walling and He, 1999b; Walling et al., 2003a; Porto et al., 2009; Kato et al., 2010). Unsupported $^{210}$Pb measurements afford an alternative to using the $^{137}$Cs technique in those parts of the world where low inventories pose measurement problems and provide a means of assembling retrospective longer term (~100 years) information. Analysis of $^{137}$Cs and $^{210}$Pb can be undertaken simultaneously and the two radionuclides can be employed conjunctively to provide a more detailed erosion history. Caesium-137 measurements are more sensitive to erosive processes at the time of peak atomic weapons fallout, whereas unsupported $^{210}$Pb measurements are more sensitive to present day erosive agents since the fallout of this specific radionuclide is constant through time. Existing calibration models for unsupported $^{210}$Pb measurements do, however, require further refinement and validation.

By virtue of its short half-life, $^{7}$Be provides an opportunity for investigating soil erosion rates and patterns at the event scale (Walling et al., 1999a; Mabit et al., 2008). Beryllium-7 measurements therefore address the problems associated with the derivation of short-term erosion estimates using either $^{137}$Cs or unsupported $^{210}$Pb data and permit the interpretation of erosion in relation to short-lived land use or hydrometeorological conditions. It is, however, important to recognize that the use of $^{7}$Be measurements is best suited to situations where significant erosion events are separated by ~5 months in order to minimize the effect of previous erosion, otherwise it is necessary to take account of the temporal pattern
of fallout and the sequence of erosion events (Walling et al., 1999a). Sepulveda et al. (2008) reported the use of the $^7$Be approach to document soil erosion rates associated with a short period of intensive rainfall in south-central Chile. Palinkas et al. (2005) used $^7$Be measurements to identify both event and season sediment deposition in the River Po delta, Italy. Feng et al. (1999) used $^7$Be as a tracer to investigate the transport and sources of particulate-associated contaminants in the Hudson River estuary.

Much of the existing work based on fallout radionuclide measurements has been associated with soil erosion studies and has therefore involved the exclusive assessment of particular sediment sources without consideration of their connectivity with river channels. However, radionuclide measurements can be used to assist the construction of catchment suspended sediment budgets and therefore offer a unique opportunity for assembling the information necessary for linking sediment sources and stores with estimates of sediment flux (Loughran et al., 1992; Owens et al., 1997; Walling et al., 2001, 2002, 2006). Alternatively, carefully designed soil coring programmes extending from hillslope summit to river channel can provide a basis for estimating the net contributions of sediment from different areas or land use to river channels.

Another tracing technique capable of providing useful information to assist in constructing catchment sediment budgets is the fingerprinting approach. Sediment source fingerprinting can generate valuable information on the relative importance of individual potential sources contributing to the downstream suspended sediment flux of a river. Such information is clearly of considerable value both for providing information on the linkages between upstream sediment sources and downstream sediment yield required to construct a sediment budget and more directly for targeting sediment control measures and thus optimising the effectiveness of such work in reducing downstream sediment fluxes. Sediment fingerprinting is discussed in more detail in the section covering sediment sourcing methodologies.

In the absence of a well-defined single procedure for assembling the information required to establish a sediment budget, ongoing work in the UK has focused on developing and testing an integrated approach, which combines a number of complementary techniques, including both sediment tracing and more traditional monitoring. These techniques include the use of fallout radionuclides to estimate soil redistribution and floodplain deposition rates, sediment fingerprinting to establish sediment sources, more traditional sampling techniques to document storage of fine sediment on the channel bed and continuous monitoring using turbidity sensors to quantify the suspended sediment flux at the catchment outlet (Walling and Collins, 2000; Walling et al., 2001, 2002, 2006).
Sediment sourcing methodologies

Existing approaches to assembling data on catchment sediment sources comprise two basic categories (Loughran and Campbell, 1995; Collins and Walling, 2004). The indirect approach to sediment source assessment is founded upon the use of a range of methods to measure or evaluate sediment mobilization in situ. By virtue of being primarily developed in association with erosion rather than sourcing studies, these methods take no account of the uncertainties in linking potential suspended sediment sources to the river channel. Sediment sourcing studies are, by definition, concerned with the major sources contributing to sediment pressures in river channels including downstream sediment flux as opposed to those portions of a catchment experiencing erosion per se. Whilst the latter, sensu stricto, represent sediment sources, the former can only be inferred using the indirect approach, unless the linkages between erosion, transport, deposition and sediment yields are quantified. Numerous uncertainties are associated with these linkages because of the sediment delivery problem (Walling, 1983) and it is therefore essential that the findings provided by the indirect approach to sediment source assessment are interpreted on the basis of complementary information on the remaining components of the sediment delivery system. By contrast, the direct approach to sediment source assessment attempts to link sources to the stream channel using alternative means and thereby avoids inference or the need to supplement estimates of sediment mobilization in situ with information on the catchment sediment budget.

Indirect approaches to establishing catchment sediment sources

Estimates of erosion rates collected using the following techniques must be interpreted on the basis of complementary information on sediment routing and sediment yield in order to provide a reliable means of apportioning the relative contributions from a number of individual sources to the sediment pressures (e.g. fluxes) measured downstream. Where erosion measurements do not encompass all potential sediment sources, the lumped contribution from those portions of a catchment not included in a monitoring programme can be estimated indirectly by calculating the difference between the contribution from monitored sources and the measured sediment pressure or load.

a) Mapping

Mapping represents an important traditional method of recording information on suspended sediment provenance (Skrodzki, 1972; Lao and Coote, 1993). Sediment source maps can be used to provide semi-quantitative assessments of sediment origin when produced sequentially and are particularly useful for relating information on the spatial distribution of erosion to corresponding information on physiographic, ecological and anthropogenic controls (Morgan, 1995). Maps have been employed to
record the occurrence of particular types of erosion including rilling (Hasholt and Hansen, 1993), gullying (Dollar and Rowntree, 1995; Figure 27) and channel bank degradation (Hooke and Redmond, 1989; Thorne et al., 1993; Lawler et al., 1997), as well as the percentage of bare ground (Kirkbridge and Reeves, 1993), the number of fields evidencing erosion (Boardman, 1990), or the extent of tree root exposure (Carrara and Carroll, 1979).

Advanced mapping procedures have been proposed by a number of workers. For example, Williams and Morgan (1976) described a geomorphological mapping system for recording information on the types and distribution of erosion and for interpreting the resulting maps in terms of a number of catchment attributes including erosivity, runoff and land use. Mapping has also been used in conjunction with the ‘pedogenic baseline approach’ to evaluate soil erosion (Yanda, 2000) or landslide hazard (Guzzetti et al., 1999) on the basis of soil profiles. Amongst the principal disadvantages of using mapping as a tool for assessing sediment provenance are subjectivity, the need for cartographic skills, difficulties in interpreting whether eroded surfaces are contemporary or historical, and the time-consuming nature of map production.

![Gullying map in the Bell River catchment, South Africa](from Collins and Walling, 2004)

**b) Surveying**

The use of surveying techniques to evaluate sediment mobilization is founded upon the establishment of a datum relative to which erosion or deposition can be measured.

**(i) Profilometers.**

One of the most commonly used devices for assessing surface advance or retreat is the profilometer, comprising a frame mounted on permanent benchmarks and with a series of vertical rods that can be
lowered to the ground surface (Sirvent et al., 1997). The length of the rods can be repeatedly measured in the field or determined from sequential photographs as a means of assessing erosion or aggradation. Examples of profilometers include the apparatus reported by Toy (1983), which takes three measurements of the designated surface and the devices described by Lam (1977) and McCool et al. (1981), which take more measurements and comprise a more stable platform. A lightweight version, termed the ‘erosion bridge’, has recently been described by Shakesby (1993). The principal advantages of employing profilometers to measure sediment mobilization include the minimal disturbance of the catchment surface and the negligible interference with the processes of erosion or deposition (Campbell, 1981). Potential disturbance of the benchmarks by erosion, cultivation or vandalism, the cumbersome nature of the available devices and the associated operator requirements are amongst the principal disadvantages. Difficulties are frequently experienced in the use of these devices to measure soil loss in stony locations or areas where a deep litter layer exists (Shakesby, 1993). The spatial coverage of a profilometer is minimal, although a series of frames can be used to establish a slope monitoring network (Lam, 1977).

(ii) Erosion pins.

The insertion of rods or nails into the surface of slopes and channel banks can provide a datum against which erosion or deposition can be manually assessed on the basis of the length of pin exposed or movement of a washer placed on the pin. Pin readings can be taken with either vernier or digital calipers. A number of recommendations for the use of manual erosion pins are provided by Haigh (1977). These emphasize that pins should be non-rustable, deployed in clusters and measured at least every six months. Although cheap and relatively simple to use, a range of problems are encountered with the deployment of erosion pins (Lawler, 1986). Couper et al. (2002) summarize the major difficulties. First, the reliability of pin readings can be compromised by pin movements associated with operator, animal or tillage disturbance, frost heave, pin loss in non-cohesive materials or other disturbances in dynamic environments (cf. Thorne, 1981; Lawler, 1993). Secondly, measurement errors can be caused by changes in the elevation of the slope or channel bank occurring independently of erosion or deposition, e.g., as a result of swelling or shrinkage. Thirdly, pin insertion can interfere with erosion processes by reinforcing soil peds or cause deposition by intercepting material moving downslope. Fourthly, vandalism can result in measurement errors or the loss of pins. Erosion pins are inappropriate for detecting micro-changes in surface elevation in situations where banks are prone to mass failure, or where retreat rates between site visits exceed pin length (Lawler et al., 1997). The spatial resolution of pin measurements is restricted to the points under investigation. Despite these problems, erosion pins have been used to monitor a variety of catchment sediment sources including soil erosion by sheetwash (Haigh, 1977) or gullying (Oostwoud Wijdenes and Bryan, 2001) and channel bank retreat (Lawler, 1993; Bull et al., 1995; Lawler et al., 1997).
The latter, in particular, has been the focus of a great deal of attention using manual erosion pins. For instance, Stott (1999) deployed conventional pins to measure main channel, tributary and forest ditch bank erosion rates in the Nant Tanllwyth catchment, UK and reported an increase in sediment supply from these sources in association with clear felling. Alternatively, a study on the River Arrow, UK, by Couper and Maddock (2001) identified the importance of sub-aerial weathering of river banks during low flow periods. Negative pin readings, however, can pose an important problem for the monitoring of bank erosion rates and reflect a number of factors including deposition during high flows, soil fall from the upper to lower bank and expansion of bank surfaces resulting from temperature or moisture fluctuations. Couper et al. (2002) report a useful examination of these factors on the Afon Trannon, Nant Tanllwyth and River Arrow, UK, concluding that no single factor accounts for negative readings and that studies employing pins should indicate how such readings are treated in the derivation of mean bank erosion rates. An important development in the use of erosion pins to study channel bank suspended sediment sources is the Photo-Electronic Erosion Pin (PEEP) monitoring system (Lawler, 1991). This instrument comprises a row of photovoltaic solar cells in a sensor that, once inserted in the bank surface and connected to a datalogger, can be used to record quasi-continuous information on the magnitude, frequency and timing of erosion or deposition, on the basis of peaks, ramparts and troughs in the voltage record. The deployment of the PEEP system has greatly improved the temporal resolution of information on channel bank erosion. For example, Lawler et al. (1997) demonstrated that bank erosion on the Upper Severn, UK, occurred during a series of large discrete events rather than as a slow continuous process and that bank erosion may continue after the flood peak because of the draw-down effect caused by the removal of lateral support for bank faces in association with receding water levels. Bull (1997), using PEEP s and estimates of suspended sediment flux in the same study area, estimated that bank erosion contributed 38% and 64% of the suspended sediment load at the monthly and event timescales, respectively (Figures 27 and 28). PEEP s have also proved useful in examining the relationship between spatial variations in bank erosion rates and downstream changes in channel hydraulics or bank material properties (Lawler et al., 1999).

(iii) Cross-profiling.

The accurate surveying of cross-profiles has been widely used to determine the volumes of sediment exported from rills and gullies (Evans, 1993; Fryirs and Brierley, 1999; Steegen et al., 2000). This approach does, however, frequently underestimate gross erosion rates by failing to take account of soil loss by sheetwash on the intervening areas of catchment slopes and the reliable identification of pre-erosion surfaces can prove difficult. Repeat cross-section surveys have also been commonly exploited as a means of determining rates of channel bank erosion (Lawler, 1993). Such procedures typically involve
the establishment of monumented cross-sections, which are repeatedly surveyed using a variety of means including inclinometers (Kesel and Baumann, 1981) or scour chains (Battala et al., 1995). Superimposition of the cross-profiles recorded at different times allows the retreat of the entire designated river bank to be estimated. In a recent example, Springer et al. (2001) reported the use of cross-profiling to measure channel erosion rates in two mountainous drainage basins in Virginia, USA.

Figure 27: Monthly suspended sediment yields from eroding channel banks in the upper River Severn catchment, UK (after Bull (1997) and from Collins and Walling, 2004).
Figure 28: Event suspended sediment yields from eroding channel banks in the upper River Severn catchment, UK (after Bull (1997) and from Collins and Walling, 2004).

(iv) GPS.

Geomorphologists are increasingly exploiting the potential for using Global Positioning System (GPS) techniques in monitoring landscape systems (e.g., Gili et al., 2000; Malet et al., 2002). GPS techniques lend themselves to periodic or continuous monitoring and can be used to detect short-duration change more readily than alternative survey. Furthermore, GPS techniques can provide centimetric precision, are nondependent upon direct lines of sight between measurement points and can be employed in a wide range of weather and light conditions. Assuming a reliable power supply, GPS equipment requires less maintenance than conventional geodetic equipment and the processing of data can be conveniently undertaken by non-specialists using commercially available computer software. It is, nevertheless, important to note that GPS techniques are constrained by a number of problems. For example, it is necessary to ensure that the sky is visible in all directions in order to guarantee receipt of the signals emitted by at least four satellites and the baseline of measurement must be <5 km in order to maximize accuracy (Malet et al., 2002). The identification of fixed survey points in highly unstable areas can pose
additional problems, whilst antennas must be inserted within 10 cm of the ground surface in order to limit problems of wind turbulence. Despite such limitations, GPS offers a convenient and increasingly affordable means of assembling quasi-real time information. Although to date, deployment has focused on the monitoring of landslides, volcanoes and glaciers, alternative applications to the study of catchment sediment sources are likely to prove equally successful because GPS techniques provide both a complement and alternative to conventional survey.

(v) Miscellaneous.
Some studies have combined surveyed estimates of differences in height between ground protected by shrub canopies and that of the surroundings, with tree ring information to assess sediment loss from different portions of river catchments (e.g., Stromquist, 1981). So-called dendrochronological methods (Alestalo, 1971) have also been employed to quantify sediment mobilization from gullies (Vandekerckhove et al., 2001) and mass movement (Fantucci, 1999; Lang et al., 1999). Conventional surveying has been used in reconnaissance surveys to provide information on the length, width and depth of erosion features. Grieve et al. (1995) used surveying to evaluate the extent of sheetwash in upland Scotland, UK, whilst more recently, McHugh et al. (2002) reported the use of conventional surveying to estimate upland soil erosion in the UK at established field sites located on an orthogonal grid.

(c) Photogrammetry.
The use of photographs in fluvial geomorphology has traditionally involved the interpretation of sequential air photos to provide qualitative information (e.g., Werrity and Ferguson, 1980). Modern methods of photogrammetry advance photographic interpretation by virtue of extracting quantitative information on the landforms under investigation by terrestrial or aerial means. Choosing between the latter is heavily dependent upon scale and financial considerations. Photogrammetry affords a number of advantages compared with alternative techniques (Chandler, 1999). Photographs can be used for detecting morphological change at the micro-, meso- and macro-scales, record the spatial relationship of landforms and provide three-dimensional information that can be used to construct Digital Terrain Models (DTMs), as well as supplementary details useful for interpreting erosion rates or patterns, e.g., vegetation cover. The collection of photographs requires minimal landform disturbance, reduces the need for alternative expensive fieldwork and provides a means of archiving information. Current quantitative procedures based on the analytical or mathematical approach, pose fewer optical and mechanical limitations compared with the traditional analogue method and produce digital output. The software required for the digital processing of photographs is now available at commercially competitive rates. Among the main disadvantages of photogrammetry are the high cost of phototheodolites, difficulties in locating stable
camera stations for repeat photography and problems caused by poor light incidence or obstacles between camera and subject (Barker et al., 1997; Chandler, 1999). Photographs represent incidental measurements, which may not take appropriate account of the frequency of observation necessary for reliably monitoring a given subject and require careful calibration. The grain size resolution of the camera film employed can preclude the accurate assessment of erosion rates.

Both aerial and terrestrial photogrammetry has been used to monitor a range of fluvial suspended sediment sources including eroding channel banks (Painter et al., 1974; Bathurst et al., 1986; Barker et al., 1997). These studies demonstrate the utility of photogrammetry for documenting the spatial variability of bank erosion rates on account of photos retaining spatial rather than point-specific information. Photogrammetry has also been employed to monitor landslides (Matthews and Clayton, 1986) and gully formation (Nachtergaele and Poesen, 1999). Despite the associated costs, aerial photogrammetry has proved useful for undertaking reconnaissance surveys of soil erosion at a range of spatial and temporal scales (Whiting et al., 1987; Vandaele et al., 1996).

(d) Erosion plots.

Erosion plots, which provide a simple and widely used means of obtaining data on soil erosion rates, can be conveniently divided into bounded and unbounded categories (Loughran, 1989). Bounded plots comprise small areas of demarcated hillslope from which runoff and sediment are collected over a storm event or alternative temporal basis (United States Department of Agriculture, 1979). Numerous examples of the use of bounded plots are reported in the literature. For instance, Thomas et al. (1981) used bounded plots to investigate sediment mobilization from areas supporting different land use in a 11.3 km\(^2\) catchment in Kenya, whilst Lewis and Nyamulinda (1996) and Vacca et al. (2000) describe similar studies in Rwanda and Sardinia, respectively. A number of potential problems and limitations are associated with the use of bounded plots. First, bounded plots are partially closed systems and are therefore not wholly representative of natural conditions, especially because of so-called ‘boundary effects’. Monitoring programmes employing bounded plots need to be of long duration and based on numerous plots, in order to improve the reliability of erosion data (Elwell, 1990). However, the erosion estimates provided by essentially uniform plots during the same rainstorm events are frequently characterized by appreciable differences in magnitude (Wendt et al., 1986). Non-standardization of plot design and measurement period mean that it is frequently impossible to undertake meaningful comparisons of the results of independent studies. Significant errors are commonly associated with sediment concentration estimates for plots, because of sedimentation in runoff collection tanks or unrepresentative sampling by automatic equipment (Lang, 1992; Morgan, 1995). Plots typically overestimate erosion rates by failing to encompass major catchment sediment stores. Extrapolation of
erosion estimates directly from plot to basin scale therefore involves many problems and uncertainties (Roels, 1985; Evans, 1990, 1993, 1995; Brown and Schneider, 1999). Unbounded plots are most commonly represented by Gerlach troughs and comprise metal gutters installed perpendicular to the slope axis to collect runoff and sediment (Mutchler et al., 1988). These installations avoid ‘boundary effects’ and are cheap and simple to use. It is, however, difficult to determine accurately the contributing area, although topographic survey or the confirmation of runoff paths using dyed water has proved useful in this respect (Loughran, 1989). Numerous troughs are required to ensure representative soil erosion estimates (Roels and Jonker, 1983; Roels, 1985; Evans, 1995). Unbounded plots have been used to estimate soil erosion rates in many environments, including the tropics (Brown and Schneider, 1999). Megahan et al. (2001) report the use of 75 unbounded plots to estimate sediment mobilization from cutslopes in the aftermath of forest road construction in Idaho, USA.

(e) Suspended sediment flux monitoring.

An alternative approach, which has proved particularly convenient for helping to document spatial sediment sources in larger drainage basins, is the monitoring of suspended sediment fluxes from individual tributary sub-catchments. Comparison of the latter estimates with the total sediment flux at the basin outlet provides a means of evaluating the relative contributions from individual spatial sources represented by the sub-catchments. Monitoring programmes aimed at assembling suspended sediment flux estimates are dependent upon the collection of discharge and sediment concentration data. The latter information has traditionally been collected using a range of manual sampling devices (World Meteorological Organisation, 1989; Walling and Collins, 2000), but the associated logistical problems and financial constraints mean that most manual sampling strategies fail to coincide with the main periods of sediment transport, i.e., flood events. Although automatic sampling equipment has helped to address this problem, automated sample collection programmes continue to fall short of providing a practical means of assembling continuous information on suspended sediment concentration. In the absence of a detailed temporal record of suspended sediment concentration, sediment loads can be estimated using a range of conventional load calculation procedures that interpolate or extrapolate the available data (Phillips et al., 1999). It is, nevertheless, important to note that these methods are confounded by more general problems of accuracy and precision and it is now generally accepted that sediment load estimation in anything other than large catchments requires short-duration sampling intervals (Olive and Rieger, 1988; Walling and Collins, 2000). The collection of high frequency sediment concentration data has been greatly assisted by the development of commercially available optical turbidity sensors. However, these devices must be used in conjunction with a number of supportive procedures including regular
calibration, lens cleaning and the development of a rating relationship for converting turbidity readings to actual suspended sediment concentration (Walling and Collins, 2000). The recent development of self-cleaning probes has reduced the lens cleaning problem. Although not without problems (Gippel, 1995; Lawler et al., 2006), the use of turbidity sensors offers a more economical means of assembling reliable information on suspended sediment fluxes because the errors are less than those associated with regular but infrequent water sampling programmes. A classic example of the estimation of suspended sediment fluxes from tributary sub-catchments is provided by the Exe basin study, UK (Walling and Webb, 1987; Figure 13). Likewise, Wass and Leeks (1999) used turbidity sensors to estimate sediment fluxes from ten major tributaries of the River Humber, UK, and Rondeau et al. (2000) reported a similar study on the St Lawrence River, Canada. Sutherland and Bryan (1990) describe the use of this approach to calculate sediment contributions from the different portions of a small catchment in Kenya. The use of sediment flux estimates to represent the relative contributions from individual spatial sources in larger drainage basins avoids spatial sampling constraints. Many of the uncertainties associated with sediment routing are avoided. But, this approach does not readily lend itself to interpreting the relative significance of key sediment mobilization processes or land management practices, except in situations where these factors are tributary-specific.

Figure 29: Estimate typical contributions of suspended sediment from tributary sub-catchments in the River Exe catchment, UK (after Walling and Webb (1987) and from Collins and Walling, 2004).
Remote sensing.

The use of remote sensing as a monitoring tool in geomorphology is underpinned by a number of assumptions (Milton et al., 1995). These include, first, that the geomorphic processes of interest produce detectable changes in the spatial or temporal pattern of electromagnetic radiation and, secondly, that any geometric distortions arising from the sensor can be discriminated from real changes in landscape features. Remote sensing instruments are carried by aircraft or satellites. Monitoring geomorphic systems with these sensors is founded on a process of geomorphic registration. The transformation of the image geometry to fit a map projection is normally accomplished by matching ground control points on the image and on a map of an appropriate scale of the corresponding area. The pixel values in the corrected image are interpolated using a procedure known as re-sampling. Re-sampling typically involves the choice between convolution, cubic, bilinear or nearest-neighbour algorithms and these need to be complemented by human interpretation by trained personnel (Duggin and Robinove, 1990). The nearest-neighbour method is recommended for quantitative analysis because this interpolation procedure leaves digital information unchanged. The geometric correction of two images taken at different times and coregistered using the above procedure provides a basis for monitoring change. Remote sensing affords a useful tool for visualizing and analysing geomorphic systems over numerous temporal and spatial scales. Satellite systems are most suited to larger-scale surveys and provide the only practical means of assembling multiregional or global information (Donoghue, 1999). Airborne sensors are best suited to monitoring smaller areas and to responding to specific geomorphic events and therefore offer improved temporal flexibility (Donoghue, 2000). Potential problems include costs, practical difficulties arising from cloud cover, flight path courses or timings and unfavourable geometry and the contrasting results yielded by different algorithms for detecting change (Wilson, 1994; Lyon et al., 1998). The mathematical transformation of airborne images is more difficult owing to the corrections required for aircraft altitude and stability (Wilson, 1994). Training is required in the interpretation and analysis of spectral images. Recent developments in synthetic aperture radar are, however, helping to avoid problems associated with weather conditions and the spatial resolution of remote sensing imagery continues to be improved (e.g., 65 cm for Quickbird, compared with 15 m for Landsat ETMþ and 5 m for SPOT-5 HRV).

Both airborne and satellite remote sensing platforms have been deployed to investigate suspended sediment mobilization in river basins. With sensors detecting electromagnetic radiation at an increasing range of wavelengths and at a wider variety of spatial, temporal and spectral scales (Curran et al., 1998), the use of remote sensing for such purposes is likely to increase. Bryant and Gilvear (1999) report the use of multispectral airborne images to quantify geomorphic changes associated with an extreme flood event on the River Tay, UK. Bocco et al. (2001) describe the use of Landsat images to evaluate land degradation arising from deforestation in Michoacan State, Mexico. Alternatively, Islam et al. (2001)
report the use of Landsat TM data to investigate seasonal variations in the relative suspended sediment loads of individual tributaries in the Ganges-Brahmaputra system. More recently, the spatial distribution of erosion and deposition in four small catchments in Australia has been estimated on the basis of gamma radiometric data (K, Th, U) recorded using airborne remote sensing surveys (Pickup and Marks, 2000). This approach is, nevertheless, confounded by a number of problems; K, Th and U are the only geogenic radioisotopes with gamma ray emissions of sufficient energy to be detected by this procedure and gamma ray signals are attenuated by dense forest and soil or surface water. Improved procedures are required to differentiate between gamma ray patterns caused by erosion and geological variability.

**Direct approaches to establishing catchment sediment sources**

A range of techniques provide a direct means of documenting catchment sediment sources by virtue of taking account of both sediment mobilization and delivery. These procedures avoid the need for inference or supportive information on wider aspects of the catchment budget including sediment routing and yield (Collins and Walling, 2004).

**a) Erosion vulnerability indices.** A variety of procedures available for evaluating the vulnerability of land to erosion and the efficiency of sediment delivery afford a direct means of assessing catchment sediment sources. These procedures have commonly been developed to address erosion problems in drainage basins in the USA, where environmental problems concerning either forestry or fishery issues are reported. An example is provided by the Timber, Fish and Wildlife (TFW) index, which comprises rankings for the susceptibility of different areas of a catchment to erosion and the probability of mobilized sediment impacting upon fishery resources (Washington Forest Practices Board, 1994). Erosion vulnerability indices can be used to assist priority setting for sediment management strategies and afford valuable guidance for erosion measurement programmes. The wider applicability of many existing indices is, however, constrained by their catchment-specific nature, whilst the accuracy of surrogate measures of erosion is questionable. Application of erosion vulnerability indices is not cost-effective in larger catchments.

**b) Sedigraphs and hysteretic loops.** The characteristics of storm period hysteretic rating loops for suspended sediment concentration–discharge relationships have been tested as a basis for evaluating the provenance of fluvial suspended sediment. Anticlockwise loops have, for example, been interpreted in terms of sediment supply from channel sources (Klein, 1984). Clockwise hysteresis has also been attributed to the re-suspension of sediment from channel sources during the rising limb of storm
hydrographs (Bogen, 1980), but alternative explanations have been advocated including the exhaustion of sediment supply from surface (Doty and Carter, 1965) or subsurface (Carling, 1983) sources and reduced detachment of surface soil particles owing to the cessation of effective rainfall (Novotny, 1980). It is therefore possible to explain similar hysteretic loops in terms of sediment supply from either surface or channel sources and additional information is clearly necessary for verifying sediment provenance (Peart and Walling, 1988). In a recent study of suspended sediment sources in the Peijbaye River basin, Costa Rica, Jansson (2002) confirmed the problems associated with using hysteretic loops to infer sediment provenance and contended that the sedigraph-rainfall method, whereby the timing of rainfall and sediment concentration peaks is combined with information on storm water travel times, represents a valuable alternative method.

c) Sediment source fingerprinting

Due to the many problems and uncertainties associated with indirect approaches to documenting catchment sediment sources, the fingerprinting approach has attracted increasing attention as a reliable alternative direct means of assembling the information required. Sediment source fingerprinting is founded upon the link between the geochemical properties of suspended sediment and those of its sources. Assuming potential sediment sources can be readily distinguished on the basis of their constituent properties or ‘fingerprints’, the provenance of the sediment can be established using a comparison of its properties with those of the individual potential sources.

The discrimination of individual potential sediment sources using the fingerprinting approach has traditionally involved a wide range of fingerprint properties (Walling and Collins, 2000; Collins and Walling, 2002, 2004). The choice of properties has, to some extent, typically reflected access to the necessary laboratory analytical equipment as well as previous experience. Some studies have used mineral-magnetism to identify sediment sources on account that mineral-magnetic measurements are simple, cheap, rapid and non-destructive (Walling et al., 1979; Caitcheon, 1993; Slattery et al., 1995; Walden et al., 1997; Caitchen, 1998; Lees, 1999; Dearing, 2000). Mineral-magnetic measurements provide a basis for discriminating catchment sediment sources because the non-directional magnetic properties of individual sources such as topsoil and subsoil can differ due to their inherent iron mineralogy and granulometry. Alternatively, some source fingerprinting investigations have used mineralogy or colour as a means of distinguishing potential sediment sources in catchments with heterogeneous geology and pedology (Wall and Wilding, 1976; Wall et al., 1978; Wood, 1978; Grimshaw and Lewin, 1980; Garrad and Hey, 1989; Woodward et al., 1992; Peart, 1993). In other cases, sediment geochemistry (Lewin and Wolfenden, 1978; Jones et al., 1991), environmental radionuclides (Walling
and Woodward, 1992; Olley et al., 1993; He and Owens, 1995; Wallbrink et al., 1998), organic constituents (Peck, 1973; Brown, 1985; Hasholt, 1988; Oldfield and Clark, 1990; Peart, 1995), stable isotopic properties (Salomans, 1975; Douglas et al., 1995, 2003) or particle size measurements (Fenn and Gomez, 1989; Stone and Saunderson, 1992; Kurashige and Fusejima, 1997; Hillier, 2001) have been used to discriminate individual sediment sources.

Due to the frequent need to distinguish several potential sediment sources, it is now widely accepted that the quest for a single diagnostic property is inappropriate on account of the problem of spurious source-sediment matches (Collins and Walling, 2002). In consequence, most recent source fingerprinting studies have used so-called ‘composite fingerprints’ comprising a range of different diagnostic properties (Collins and Walling 2002, 2004; Nelson and Booth, 2002; Evans et al., 2006). Composite fingerprints comprise individual properties influenced by differing environmental controls and which thereby improve source discrimination by affording a substantial degree of independence. Such fingerprints can represent several diagnostic properties from either a particular property subset e.g. several radiometric (He and Owens, 1995; Foster et al., 1998), mineral-magnetic (Oldfield and Clark, 1990) or geochemical (Kelley and Nater, 2000; Collins and Walling, 2002) properties, or a combination of geochemical, radiometric and organic constituents (Walling et al., 1993; Foster and Charlesworth, 1994; Collins et al., 2001; Collins and Walling, 2002). In order to satisfy dimensionality, the number of fingerprint properties should exceed the number of potential sediment sources being discriminated (Foster and Lees, 2000; Collins and Walling, 2004).

Sediment source fingerprinting assumes that the selected fingerprint properties are readily transported and deposited in association with suspended sediment and that selective erosion and sediment delivery processes do not transform the properties (via enrichment, depletion, dilution) beyond what can be corrected for using appropriate procedures. Composite fingerprints should be identified using statistical verification (Collins et al., 1997a, 2000; Collins and Walling, 2002). Many investigations using the fingerprinting approach have used a simple qualitative comparison between the fingerprint properties of different potential sources and sediment samples as a means of elucidating sediment provenance (Peart, 1993; Walling and Kane, 1984; Walling and Amos 1999). But, in order to provide more useful quantitative information on sediment contributions from individual sources, composite fingerprints are now generally used in conjunction with a multivariate numerical mixing model (Walling et al., 1993; Collins et al., 1997a, 2001; Wallbrink et al., 2003; Krause et al., 2003; Motha et al., 2004). Sediment mixing models can be based on linear programming (Yu and Oldfield, 1989, 1993; Caitcheon, 1993, 1998) or optimisation algorithms (Collins et al., 1997a; Walling et al., 1999c; Owens et al., 2000; Owens and Walling, 2002; Walling, 2005; Martinez-Carreras et al., 2010; Collins et al., 2010a).
Application of the sediment fingerprinting approach to document catchment sediment sources necessitates collection of representative samples of individual potential sediment sources. The latter can be defined in a variety of ways. In some investigations, especially those in large-scale river drainage basins, it has proved most meaningful to investigate the spatial provenance of suspended sediment sources, defined in terms of individual tributary sub-catchments (Collins et al., 1996; Walling et al., 1999; Collins et al., 2010a) or discrete geological zones (Collins et al., 1998; Walling et al., 1999; Owens et al., 2000; Bottrill et al., 2000). In smaller catchments, it is commonly more appropriate to characterise sediment provenance in terms of individual source types comprising either surface and subsurface categories (Peart and Walling, 1986, 1988; Walling et al., 2003b) or surface soils supporting different land use and eroding channel banks (Slattery et al., 1995; Collins et al., 1997a,b; Walling and Woodward, 1995; Walling et al., 1999c; Collins et al., 2000, 2001; Russell et al., 2001; Krause et al., 2003; Motha et al., 2004; Walling and Collins, 2005; Walling, 2005; Figure 30). Carter et al. (2003) recently used the approach to investigate suspended sediment source types (topsoils supporting woodland, pasture and cultivation, channel banks, road dust and solids from sewage treatment works) in an urban river system in northern England. Relative sediment inputs from STWs versus more diffuse sources (agricultural land, road verges, channel banks) have also recently been investigated by Collins et al., 2010a (Figure 31). Collins et al., (2010b) recently used the approach to apportion sediment inputs from grass road verges damaged and eroded by vehicle and livestock movements relative to alternative agricultural topsoil and channel bank sources in a lowland permeable catchment in southern England. The findings suggested that damaged road verges can represent a significant sediment source in sub-catchments with dense road networks and widespread opportunity for poaching and vehicle damage of the associated grass verges (Figure 32). Sediment source fingerprinting affords a convenient basis for investigating spatial provenance and source type in an integrated manner (Walling and Woodward, 1995; Collins et al., 1997b; Collins et al., 2010a). As well as documenting contemporary suspended sediment sources, the fingerprinting approach provides a unique means of reconstructing longer-term sediment provenance and thus for examining linkages between soil erosion patterns and land use change (De Boer, 1997; Collins et al., 1997c; Owens et al., 1999; Huang and O’Connell, 2000; Collins et al., 2010b) or the occurrence of extreme flood events (Collins et al., 1997d). The approach has recently been used to examine the contribution of channel bed sediment remobilisation to suspended sediment flux at the outlets of lowland groundwater-fed catchments in the UK (Collins and Walling, 2006).
Figure 30: Inter-storm (a), intra-storm (b) and seasonal (c) variations in suspended sediment source types for the upper Kaleya River catchment, Zambia (after Collins et al., 2001 and from Collins and Walling, 2004).
Figure 31: Relative sediment source proportions for a number of sub-catchments comprising the Somerset Levels, southwest England (Collins et al., 2010a). Estimates based on floodplain sediment samples.
Figure 32: Relative contributions from individual source types to time-integrated suspended sediment samples collected in the Sem sub-catchment of the River Avon basin, southern England (Collins et al., 2010b)

In tandem with the adoption of statistical and numerical data processing techniques for sediment fingerprinting, other important developments are associated with the use of various corrections and weightings during sediment source ascription. Since the properties of soil and sediment samples are strongly controlled by particle size composition and organic matter content (Horowitz and Elrick, 1987; Horowitz, 1991), it is necessary to correct for differences in these characteristics. The selectivity of
sediment delivery processes means that sediment samples are typically enriched in fines and organic matter content relative to the individual contributing source areas of the catchment. Approaches to correct for contrasts in particle size and organic matter content have varied in complexity. The most basic approach has been to restrict laboratory analyses to only the <63 μm (<0.063 mm) fraction of source and sediment samples, thereby ensuring a focus upon the dominant size class of suspended sediment (Motha et al., 2002). Given that the composition of the <63 μm fraction tends to differ between the samples collected to characterise sources (Figure 33), fingerprint property concentrations measured on this size fraction have been corrected using specific surface area information (Collins et al., 1997a, 1998; Gruszowski et al., 2003). Specific surface area provides a useful surrogate for particle size and is governed by the entire composition of a given size fraction. More complex approaches to correcting for particle size composition have been based on detailed information on the precise relationship between grain size composition and the concentrations of individual fingerprint properties (He & Walling, 1996; Russell et al., 2001). These approaches avoid the assumption that there is a consistent linear relationship between concentration and particle size composition for all properties. But, the identification of correction factors for individual properties requires substantial investment in laboratory resources. Alternatively, some researchers have adjusted the fingerprint property concentration data for source materials using information on the grain size characteristics of sediment and the concentration information for different size fractions of the source samples. Under these circumstances, source material fingerprint property concentrations are adjusted to reflect the same particle size composition as that measured for sediment (Slattery et al., 1995; Motha et al., 2002). Less attention has been directed towards correcting for contrasts in the organic matter content of samples. Organic matter content adjustments typically rely upon a simple ratio between the organic carbon content of source material and sediment samples (Collins et al., 1997a, 1998), or the adjustment of source material fingerprint property concentrations to reflect a similar organic matter content to that measured for sediment (Motha et al., 2002). Correcting for organic matter content frequently reduces the errors associated with numerical sediment source ascription (e.g. Walling et al., 2003), although the risk of double correction, in tandem with the use of a particle size correction factor should be carefully explored during each fingerprinting study.
In addition to corrections for particle size and organic matter content, the varying levels of precision of laboratory analyses for individual sediment properties has also been taken into account in order to ensure that greater emphasis is placed on those properties affording the greatest precision (Collins et al., 1997a, 1998). Alternatively, the recent work of Collins et al. (2010a) has explored the use of within-source variability weightings for individual tracer properties as a means of constraining the uncertainty ranges computed for source proportions using mixing models. Such weightings reflect a number of sources of uncertainty including those associated with source material and sediment sampling, as well as with property analyses in the laboratory.

Since sediment source fingerprinting typically relies on the collection of relatively few source and sediment samples, there is a need to take explicit account of the uncertainty associated with source proportions generated using sediment mixing models. As a result, uncertainty analysis is now incorporated into the quantitative source apportionment procedure, using a selection of Bayesian statistics and Monte Carlo routines (Rowan et al., 2000; Small et al., 2002; Motha et al., 2004; Douglas et al., 2003; Collins & Walling, 2007 a, b; Wilkinson et al., 2009; Collins et al., 2010a, b). Such uncertainty analysis is increasingly taking account of the uncertainty arising from sediment as well as source material sampling using a combination of parametric and robust statistics combined with both conventional random and Latin Hypercube sampling during repeat model iterations.

Because sediment fingerprinting studies are now based on composite signatures, bulk sediment samples (ca. 100 g) need to be collected. The highly episodic nature of sediment transport means that ca. 90% of the annual suspended sediment load is commonly transported within only ca. 10% or less of the time (Walling and Webb, 1987). Accordingly, bulk suspended sediment samples have traditionally been collected during storm events, when suspended sediment transport is most active, using either a
submersible pump powered by a portable generator or a mobile continuous-flow centrifuge or time-integrating isokinetic traps (Phillips et al., 2000; Walling et al., 2006, 2008). In situations where representative information on recent sediment provenance is required quickly, it is more practical and meaningful to use floodplain surface sediment samples as end members in the fingerprinting exercise (Collins et al., 2010a). Floodplain sedimentation occurs during the higher magnitude flow events accounting for the greatest proportion of annual suspended sediment transport. Sampling floodplain surface sediment thereby provides a basis for the paragmatic retrieval of samples of temporally-integrated sediment characterising major recent periods of sediment delivery. Where river channel sedimentation represents a significant sediment stressor, the source fingerprinting approach can be used to apportion the key sources of channel bed sediment (Collins and Walling, 2007a, b). Under such circumstances, representative samples of fine sediment silting the river channel substrate are commonly collected using a simple remobilisation technique (Collins and Walling, 2007c).

In response to the above sediment sampling issues, recent fingerprinting studies (e.g. Walling et al., 2006, 2008; Collins et al., 2010b) have adopted simple inexpensive time-integrating devices that deploy the principles of sedimentation (Phillips et al., 2000). The samplers operate in situ without the need for power and collect composite samples of sediment continuously during the period of operation. Natural variations in sediment properties during either individual storm events or a series of floods are therefore captured, whilst the devices provide a means of collecting sufficient sample mass for laboratory analyses. Deployment of the time-integrating devices negates the need for quick response site visits during storm events and therefore ensures that sediment flux is continuously sampled. The samplers (Figure 34) comprise a PVC pipe (98 mm internal diameter, 1 m length) with two end caps containing a central inlet / outlet pipe (4 mm internal diameter). During insertion, the sampler is filled with clean native water and submerged in alignment with water flow. The sampler is typically secured to dexion or kee-klamp uprights, which are either driven into the river bed or concreted into heavy blocks. Following submersion, water flow enters the sampler via the inlet tube and upon progressing into the main chamber, represented by the PVC pipe, its velocity is reduced in excess of 600, thereby encouraging sedimentation. Because most fine-grained sediment is transported in the form of aggregates, the sedimentation within the sampler collects a heterogeneous mix of the primary particle sizes comprising the local absolute grain size distribution for suspended sediment. Settling velocities for aggregates exceed those for individual primary particles, thereby ensuring that sediment deposition within the sampler occurs more readily than predicted by Stokes Law.
Figure 34: Schematic of the time-integrating suspended sediment sampler

**Modelling catchment sediment sources**

A range of increasingly complex models can be used to simulate sediment mobilization and delivery rates. Traditionally, simplistic empirical relationships for predicting soil erosion rates, e.g., Universal Soil Loss Equation/Revised Universal Soil Loss Equation (USLE/RUSLE) (Renard et al., 1991) could be used in conjunction with estimated sediment delivery ratios and sediment yield data as a means of determining channel bank erosion by difference (Peart and Walling, 1988). Spatially distributed models of soil erosion and sediment delivery are, however, now increasingly used in response to the emergence of more sophisticated technology and computational (e.g., cellular automata, neural network, genetic algorithm) procedures (Brooks and McDonnell, 2000). Distributed models can be based on either a semi-empirical approach involving the combined use of the USLE/RUSLE and distributed information on drainage networks, routing factors or sediment delivery ratios (Bradbury et al., 1993; Fraser et al., 1996; Pilotti and Bacchi, 1997; Van Rompaey et al., 2001) or complex physically based process descriptions, e.g., Water Erosion Prediction Project (WEPP) (Nearing et al., 1989), European Soil Erosion Model (EUROSEM) (Morgan et al., 1994), Système Hydrologique Europeen (SHE/SHESED) (Bathurst et al., 1995), Limburg Soil Erosion Model (LISEM) (De Roo et al., 1996) and Ephemeral Gully Erosion Model (EGEM) (Nachtergaele et al., 2001). Although useful for simulating sediment mobilization and delivery, the output from distributed models frequently represents integrated response and must therefore be further scrutinized and processed in order to assess the relative importance of individual sediment sources. Existing models frequently incorporate only a selection of potential sediment sources or even an individual source. The availability of reliable information on the spatial and temporal variability of sediment mobilization and delivery constrains the meaningful evaluation and validation of distributed models. Existing work has thus largely focused on ensuring that distributed models are theoretically
acceptable as opposed to consistent with field data (Beven, 2002). Highly parameterized distributed models therefore remain largely untested. The use of the most commonly available information, i.e., sediment yield data, permits validation of integrated basin response but precludes verification of internal performance. It is obviously possible to simulate basin output in terms of incorrect compensating internal behaviour. There are, nevertheless, a few examples of validating distributed predictions of soil redistribution using $^{137}$Cs measurements (e.g. Chappell, 1996; He and Walling, 1998). A number of more general problems are associated with distributed modelling per se (Brooks and McDonnell, 2000; Beven, 2002; Beven and Feyen, 2002). The accurate representation of land surfaces continues to pose problems, but remains critical because variations in elevation are important in governing sediment mobilization and redistribution. Numerous uncertainties exist in assuming that Digital Elevation Model (DEM)-based hydrological modelling is reliable, because DEMs comprise artefacts, which must be carefully scrutinized (Wise, 2000). Improved process representation and inclusion are required. Specific areas warranting attention include the effects of scale and uncertainties in accounting for the spatial variability of precipitation, the representativeness of runoff flow paths or the occurrence of ephemeral gully erosion (Garen et al., 1999). Existing models need to include more small-scale processes, especially whilst predicting change over a larger temporal or spatial resolution. Parameter and predictive uncertainty introduce equifinality (Beven, 2002), which must be explicitly recognized. Despite such problems, spatially distributed models of catchment erosion and sediment delivery afford a useful means of predicting the erosive behaviour of river basins under changing environmental or management scenarios.
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