The impact of excess fine sediment on invertebrates and fish in riverine systems

Literature Review

I. Introduction

At a global scale, suspended solid (SS) concentrations in many rivers have dramatically changed in recent years (Walling, 2006). Existing evidence indicates that natural sediment loadings have been substantially exceeded in many catchments in the UK, particularly since World War II (Evans, 2006). Sediment loadings delivered to watercourses originate from a number of upstream primary and secondary sediment sources, the main anthropogenic activities increasing sediment supply to watercourses include:

- Changes in agricultural practices; for example, increased areas of arable cultivation, leading to greater areas of bare exposed soil susceptible to erosion by winter rainfall (Greig, *et al*., 2005), and mechanised farm practises which compact the soil and increase runoff and soil erosion (McMellin *et al*., 2002: Bilotta, *et al*., 2007). For instance, sediment-fingerprinting research indicated 61% of the sediment load of the River Tweed in Scotland was derived from arable and pasture top soils (Owen *et al*., 2000).
- Intensification of agricultural practices; for example, increasing stock density (Heaney *et al*., 2001) and multiple cropping on arable land.
- Increased bank erosion due to loss of natural hydrology.

Erosion processes and sediment delivery form an integral part of aquatic systems, influencing their geomorphology, habitat distribution and water quality. Aquatic communities are adapted, and hence able to cope with, natural 'baseline' sediment inputs. Healthy freshwater ecosystems require sediment inputs to maintain habitat and nutrient fluxes, but excessive loading can have catastrophic effects on river ecosystem function. The main direct physical effects are reduction in habitat availability and modification of habitat biogeochemical conditions through reduction of oxygen and increased concentrations of toxic compounds (Kemp *et al*. 2011; Jones *et al*. 2012). Sediment suspended in the water column can also cause sublethal effects from turbidity and direct physical damage, particularly to fish species (Wilber & Clarke, 2001).

Sediment in the water column can be measured in three main ways 1) turbidity; the optical (light scattering) property of the water, 2) total SS; direct measurement of particulate weight present in a volume of water and 3) water clarity; also an optical property of water. Deposited sediment can also be measured using sediment traps. Despite this, there is a distinct lack of SS monitoring data from around the UK, mainly due to cost implications but also a historic perception that other water quality parameters were of greater importance.

II. Effects on Invertebrates

The negative impacts of high and persistent sediment loads on invertebrate assemblages and abundances are well documented with Ephemeroptera, Plecoptera, Trichoptera (EPT) taxa exhibiting the greatest negative response to increased sediment. Sediment can trigger invertebrate decline in various ways including; scour damage, burial of heavy or immobile species, the clogging of gills or feeding structures, and reduction in interstitial habitat and primary production (Newcombe and MacDonald, 1991) (Fig. 1). Much of the recent literature has explored the effects of excessive sediment deposition on invertebrates from physical and biological perspectives, looking beyond the focus on reduced total abundance and taxonomic richness found in older papers (Richards & Bacon, 1994; Shaw & Richardson, 2001; Zweig & Rabeni, 2001). Changes in invertebrate life has trophic implications for fish species, particularly juvenile salmonids, so understanding and controlling the impacts of sedimentation is crucial to maintain ecological fitness of river systems.

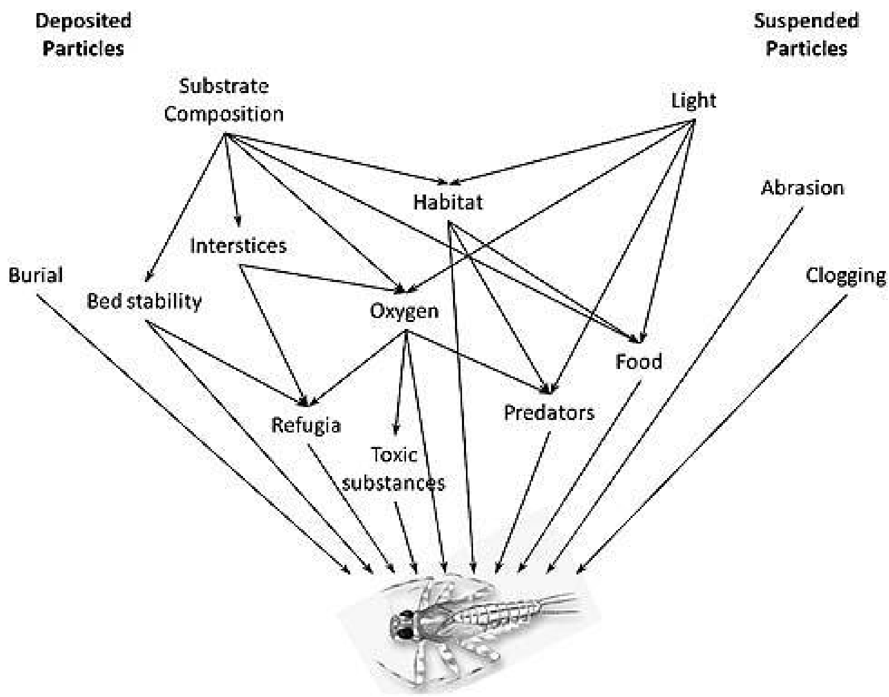

Figure 1: *Summary diagram illustrating the direct and indirect mechanisms by which fine sediments impact upon macro-invertebrates (Jones et al. 2012)*

<u>*Biological changes*</u>

A study by Buendia *et al*. (2013) assessed evidence of patterns in assemblage structure and functional traits of benthic invertebrates in response to excessive fine sediment deposition. Invertebrate groupings in high sediment areas were only a subset of the groups found in locations with minimal fine sediment and these invertebrates had biological traits that favoured resistance and resilience to fine sediment, such as shorter life cycles and smaller body sizes. Descloux *et al*. (2014) similarly examined changes in biological characteristics caused by colmation in benthic and hyporheic invertebrate assemblages. In the benthic zone colmation significantly modified eight invertebrate trait profiles and in the hyporheic zone it significantly modified five. As found in the previous paper, trait selection in benthic invertebrates leaned more towards resistance or resilience capacities of species and features related to physiological and trophic functions. As only morphological traits were modified in hyporheic zone invertebrates, it was concluded that colmation is biologically more selective on benthic than hyporheic assemblages as benthic invertebrates have less adaptations to cope with excessive sedimentation.

<u>*Behavioural/Habitat changes*</u>

Vadher *et al*. (2015) demonstrated via lab experiments that the addition of fine sediment inhibited movement of *Gammarus pulex* into sediment refuges during exposure to water column disturbances. In the control 93%

of the individuals moved into subsurface sediments when the water level was reduced. This was reduced to 0% when as little as 20% fine sediment was added. This physical barrier means larger macroinvertebrates cannot protect themselves from stressors such as floods and streambed drying. Reduced availability of refuges caused by fine sediment can also increase the density of invertebrate drift, as invertebrates have nowhere to hide and become forced to leave the river substrate to prevent smothering. Benthic invertebrate drift rates have shown to increase in SS concentration as little as 8mg/l (Rosenberg and Wiens, 1978) and population size has been shown to reduce by 77% when exposed to 62 mg/l of SS for 100 days (Wagener and LaPerriere, 1985). A 40% reduction in stream invertebrate species diversity has been recorded in response to prolonged SS concentrations of 133 mg/l over the period of a year (Nuttall and Bielby, 1973). Molinos & Donohue (2009) examined the response of 4 common macroinvertebrate taxa to different doses and exposures of sedimentation. They found that invertebrate drift was strongly affected by exposure time and all taxa exhibited statistically significant responses within the first day under all exposure concentrations. Larsen & Ormerod (2010) similarly found that even very small, short-term increases in fine sediment reduce benthic density by promoting invertebrate drift. When sediment was added drift density overall increased by 45% with the strongest contributors being mayflies (*Baetis rhodani*, *B. muticus* and *Ecdyonurus* spp.), helodid beetles and simuliid/chironomid dipterans.

III. Effects on Fish

Excessive fine sediment, in suspension or deposited, can have damaging impacts on all life stages of fish, particularly salmonids. This has been worsened for salmonids by a shift in the timings of arable cultivation in the UK, from spring to autumn sown cereals, which now coincides with their egg incubation times (Collins and Walling, 2007; Collins *et al.,* 2008). The effect on ecosystems will, however, depend on several key factors, including: the concentration of fine sediment in suspension; the duration of exposure to fine sediment; and the sediment chemical composition and particle size (Bilotta and Brazier, 2008). This makes determining the impact of SS on fauna and flora difficult to generalise, quantify and compare. However, generic consequences of increased SS concentrations in the water column for fish can include:

Mortality

The relationship between SS concentration and direct mortality is highly complex. The effect of sediment on fish will vary depending on life stage, time of year, size of fish, composition of sediment and the availability of off-channel habitat (Bash *et al*., 2001), as well as the exposure magnitude, duration and frequency (Servizi and Martens, 1992). For example, in a review of published critical SS concentration thresholds based on dose-response experiments examining impaired growth, reduced feeding and mortality, Berry *et al*. (2003) reported ranges of 27-80,000 mg l^{-1} for mollusca and 4-330,000 mg l^{-1} for various fish species. Such ranges in the severity of effect of SS concentration are a function of associated stressors including sediment particle size, species life history stage, temperature, the presence of sediment-associated contaminants and sediment load duration (Swietlik *et al*., 2003). Due to the complex interaction of such stressors, it is unlikely that a comprehensive list of genus-based critical SS concentration targets can be developed in the short-term (USEPA, 2003).

Reduction in suitable spawning habitat and declines in egg/early life stage success

Effects of excessive deposition of fine sediment on salmonid spawning success and egg survival have been well documented over the years. It has been proved in a vast plethora of literature that infiltration of fine sediment limits success of eggs hatching through the reduction of gravel permeability and oxygen availability (Ingendahl, 2001; Greig *et al*. 2007; Schindler Wildhaber *et al*. 2014). Salmonid eggs (as well as many cyprinid fish and lamprey eggs) require a well-oxygenated environment during the embryonic development stage, so eggs are laid in permeable gravel beds with interstitial pore spaces, which allow the passage of oxygenated water. Excess fine sediment in the water, when deposited, can clog these interstitial pores, obstructing the circulation of oxygenated water, which reduces egg survival (Carling, 1984; Magee *et al*. 1996). Salmonid egg mortality of between 98-100% has been recorded within spawning gravels experiencing high siltation loads (Turnpenny and Williams, 1980). The effect is particularly damaging when sediments contain a high organic component, as its subsequent decomposition also reduces oxygen from the water (Rubin, 1998). Excess deposited sediment can also reduce interstitial and hyporheic (region beneath streambed) flow velocities (Chapman, 1988; Acornley and Sear, 1999). This decreases the natural flushing process, which removes the harmful metabolic waste excreted by the embryos (Burkhalter and Kaya, 1975). SS can also be damaging to fish species, such as perch and roach, depositing eggs on macrophytes, as silt particles can adhere to the eggs and reduce oxygen and carbon dioxide exchange (Stuart, 1953).

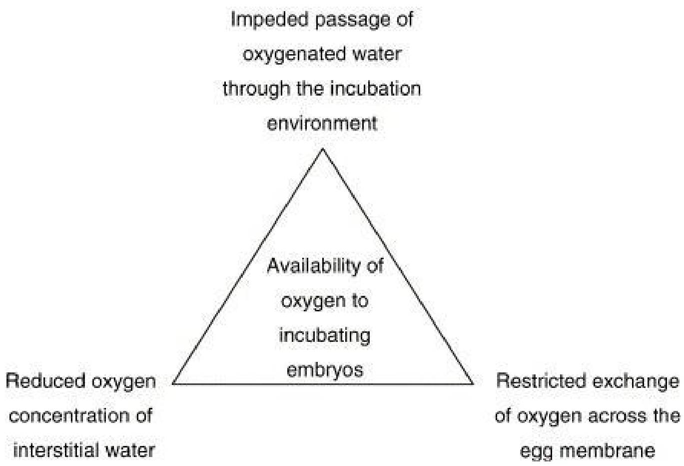

Figure 2: *The three specific mechanisms by which fine sediment accumulation restricts O2 availability to incubating salmonid embryos (Greig et al. 2005)*

Fine sediment can also exert sub-lethal effects on fish fry including: delaying emergence by trapping fry in interstitial pores (Phillips *et al*., 1975); and premature hatching of smaller and poorer quality fry, due to exposure to low dissolved oxygen concentrations (Alderdice *et al*., 1958: Mason, 1969). Researchers have found a relationship between fine sediment (less than 0.850 mm) and fry survival, with decreasing survival of up to 3.4% for each 1% increase in fine sediment (Cederholm *et al*., 1981). Louhi *et al*. (2011) showed that brown trout (*Salmo trutta*) fry experienced predator-masking effects associated with high sedimentation. Control fish tended to postpone fry emergence when exposed to predator odour, whereas fish in the high-sedimentation treatment showed no response to predators. This indicates that in high sediment environments predation is a bigger risk to fry survival. The study also confirmed that in high sediment environments trout fry were emerging earlier with larger yolk sacs. As these fry emerged prematurely to escape the suboptimal conditions associated with increased sedimentation it is highly probable that they will be poor swimmers and thus more vulnerable to predation and downstream displacement than fully developed fry. Bowerman *et al*. (2014) found the same phenomenon occurred in bull trout (*Salvelinus confluentus*) fry; they too were emerging from the incubation environment at much earlier development stages when sediment percentages were higher. This work also highlighted later life implications for premature fry that do survive to adulthood, as early emergence can lead to fitness reduction in later life stages. The deposition of sediment on the riverbed also changes and degrades physical habitat for bottom dwelling fish and fry, leading to lower fry and parr density (Lisle and Lewis, 1992). The sediment fills in the spaces between substrate particles, creating a smoother riverbed (Diplas and Parker, 1992). This, not only reduces the available habitat complexity and availability, but also increases water velocity and the need for shelter from the water current (Richardson and Jowett, 2002). Sedimentation also decreases habitat connectivity (Cohen, 1995) and heterogeneity (Boles, 1981).

Gill irritation/trauma

Fish gills are delicate and easily damaged by abrasive sediment particles. Fish species have been found with increasing levels of deformities, eroded fins, lesions, tumours, gill flaring and 'coughing', all related to increasing SS in the water column (Berg, 1982; Schleiger, 2000).

Altered blood physiology

Research has show increases in plasma glucose (Servizi and Martens, 1987), blood sugar levels (Servizi and Martens, 1992) and plasma cortisol (Redding *et al*., 1987) in fish species exposed to fine sediment. These are all indicators of stress, and can impact physiological systems, reduce growth and decrease immunological competence against other stressors, such as disease. In a study by Sutherland & Meyer (2007) all three life stages of the spotfin chub (*Erimonax monachus*) and whitetail shiner (*Cyprinella galactura*) showed increases in immunoreactive corticosteroid (stress) levels when exposed to moderate sediment concentrations. This study showed that fish species experience physiological stress as a result of suspended sediment regardless of their life stage. Respiratory impairment caused by sediment in the gills was suggested to be the main factor instigating the stress response in the spotfin chubs. Stress to salmonids can also affect the smoltication process, leading to decreased osmoregulatory ability, impaired migrations and a reduction in early marine survival (Wedemeyer and McLeay, 1981).

Altered movement/swimming performance

Migrating fish species, such as salmonids, are commonly known to migrate through high SS concentrations in estuaries. However, like other containments, exposure time is a key element in the impact of SS as well as concentration (Newcombe and MacDonald, 1991). This means high exposure rates to sediment loads can halt fish migration, particularly upstream. Fish are known to exhibit avoidance reactions and move away from the vicinity of adverse sediment conditions, if refuge conditions are present (Sigler *et al*., 1984; Bash *et al*., 2001). The effects of suspended sediment on swimming ability on juvenile brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*) were explored by Berli *et al*. (2014). Both species experienced a decrease in swimming performance as turbidity increased, but rainbow trout were impaired to a greater extent. This was attributed to impairment in the ability of the fish to utilise anaerobic metabolic pathways in high sediment environments. The authors concluded that the ability of salmonids to maintain swimming performance is hindered when fish are exposed to environmentally relevant, suspended sediment-generated turbidities.

Changed foraging behaviour and reduced territoriality

Turbidity can reduce feeding rates, and affect prey selection and prey abundance. This is particularly significant for visual feeders, such as salmonids, where SS can reduce the effectiveness of them obtaining food (Berg, 1982). However, research also suggests the turbid-clear water interface may sometimes assist feeding, by offering concealment and protection within the turbid water (Scullion and Edwards, 1980). Robertson *et al*. (2007) showed that during autumn initial introduction of sediment increased foraging activity in juvenile Atlantic salmon (*Salmo salar*). It was suggested that the small increase in turbidity provided visual isolation from predators. As sediment levels were raised to over 180 mg/L this foraging activity declined along with a rapid reduction in territorial behaviour. Such responses to increased water turbidity have been shown to instigate increased emigration from preferred habitats as excess sediment makes habitats unsuitable. Although salmon were less responsive to increases in suspended sediment levels in colder water temperatures (winter months) this work highlighted the importance of not assuming salmon are more sediment tolerant at these times. Pulses of turbid water have also been shown to breakdown normal social organisation and territoriality, which can decrease feeding rates and may affect overall growth rates (Berg, 1982).

IV. Ecosystem-scale effects

Community homogenisation

Sedimentation can affect aquatic biota at both a population and community level, and result in homogenisation. This means regionally distinct faunas may be replaced with few invasive and disturbance tolerant species (Walters *et al*., 2003). This could be a serious threat to biodiversity, both now and in the future, by reducing species' resilience to climate change. Sedimentation can also increase the susceptibility of invasive species such as Canadian pondweed and the common carp, which potentially have major disruptive effects on aquatic ecosystems.

Transfer of pollutants

Fine sediment exerts an important control on the transfer and fate of a wide range of agricultural and industrial contaminants (Warren *et al*., 2003; Collins *et al*., 2005). Sediment therefore represents an important vector for contaminants such as phosphorus (Haygarth *et al*, 2005); heavy metals (Neal *et al*., 1999) and organic pollutants like sheep dip substances (Long *et al*., 1998). These associated pollutants can alter species assemblages by poisoning the water system, and accelerating eutrophication. The capacity of fine sediment to bind contaminants can also lead to an increase in the resident times of the pollutants in aquatic systems (Foster and Charlesworth, 1996), thereby increasing exposure times and the opportunity for pollutant remobilisation.

Reduced primary productivity

Suspended solids reduce water clarity and increase turbidity, exerting a negative effect upon primary production. Research suggests in subarctic Alaskan steams concentrations of SS as little as 8mg/l can reduce primary production by 3-13% (Lloyd, 1987), and above 2100 mg/l no primary production can occur (Van Nieuwenhuyse and LaPerriere, 1986).

Depressed oxygen levels in the water

Suspended solids can contribute towards raising the Biological Oxygen Demand (BOD; Petts *et al*., 2002), and hence lowered oxygen levels potentially to stressful or lethal levels for vulnerable species and life stages.

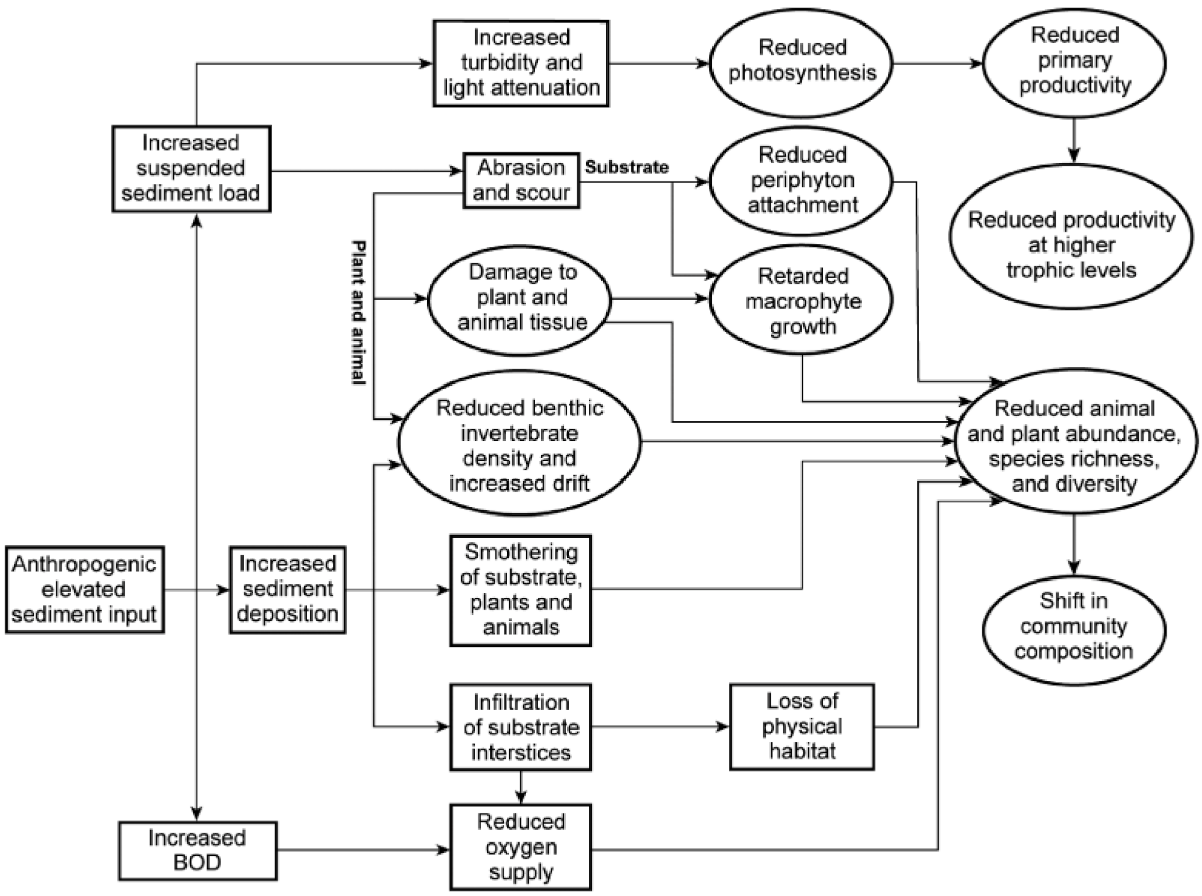

Figure 3: *Effects of high sediment loads on aquatic ecosystems. Rectangles = Physicochemical effects. Ovals = Direct/long-term biological and ecological responses. (Kemp et al. 2011)*

V. Current Management

Until it was revoked, the Freshwater Fish Directive (FFD) stipulated that suspended solid concentrations should not exceed an annual mean of 25mg/l. However, this standard was not imperative and the recommended figure was simply a guideline. In 2009 the FFD and Shellfish Waters Directive were both revoked, with their key requirements being transferred to the Water Framework Directive (WFD). In 2010, UKTAG published a note detailing recommendations that should be considered in the transfer, including an environmentally protective solids standard. Application of a standard was deemed necessary to protect shellfish against potential smothering, release of sediment associated pollutants and dissolved oxygen sags related particularly to dredging activity. It was also recommended that suspended solids continue to be monitored due to their ecological significance (UKTAG, 2010). Since the repeal of both Directives, the initial FFD guideline target was lost and no standards for suspended solids under the Water Framework Directive currently exist (Cascade Consulting, Thames Water Utilities, 2011).

In the scientific community attempts have been made to identify target values for both deposited fine sediment and sediment loading, yet the relationship between deposited fine sediment and agricultural sediment pressure is still poorly understood. Collins *et al.* (2007) used a structured modelling methodology to predict the impact of projected structural evolution in agriculture (land use change) and the uptake of sediment mitigation methods due to programmes like the England Catchment Sensitive Farming Delivery Initiative (ECSFDI) on annual mean suspended sediment concentrations across England and Wales by 2015. This work suggested that structural and mitigation work could potentially reduce the national sediment loss from the agricultural sector by 9% by 2015.

Collins and Anthony (2008a) also modelled catchment compliance across England and Wales using the previous FFD guideline standard. The study provided the first national scale assessment of sediment sources for England and Wales under current environmental conditions (year 2000), suggesting that source contributions are in the order: agricultural sector (1929 kt = 76%) eroding channel banks (394 kt = 15%), diffuse urban sources (147 kt = 6%) and point source discharges (76 kt = 3%). A structured regression model was used to convert the predicted total sediment loadings into time-averaged suspended sediment concentrations at national scale. The findings suggested that approximately 83% of the total catchment area of England and Wales appears to require no further reductions in sediment loss to rivers from diffuse agricultural sources for the purpose of meeting 'Good Ecological Status' (GES) as defined by the Water Framework Directive (WFD). It is important to note, however, that the use of the FFD sediment threshold failed to identify catchments across England and Wales where the detrimental impacts of sediment are widely reported e.g. the chalk catchments of southern and eastern England. Chalk catchments are particularly vulnerable to sedimentation due to the lack of any significant flushing effect owing to their baseflow-dominated hydrology.

Naden *et al.* (2016) took these concepts further and analysed instantaneous measurements of deposited fine sediment in 230 agricultural streams across England and Wales in relation to 20 potential explanatory catchment and channel variables. Two main practical recommendations were made from this work regarding fine sediment load targets: 1) the ability of streams to transport/retain fine sediment needs to be taken into account, 2) where agricultural mitigation measures are implemented to reduce delivery of sediment, river management to mobilise/remove fines may also be needed in order to effect an improvement in ecological status in cases where streams are already saturated with fines and unlikely to self-cleanse.

There are, however, serious concerns regarding the use of a single global threshold concentration for suspended sediment. This is because of the large variability of effects caused by sediment, the diversity of habitat and conditions within catchments, the existence of sub-threshold effects on both fish and their supporting ecosystems, and the failure of an annual mean to capture the highly episodic nature of sediment pressures which are focused during flood events. To address this alternative sediment targets were proposed for England and Wales using an alternative sediment target scheme (Cooper *et al*., 2008). This was based on nationally extrapolated suspended sediment yields and uses the lower quartile of the measured ranges for catchment types to represent potential targets and the upper quartiles as critical thresholds. These tentative targets were intended for use in the identification of thresholds from a local perspective. Collins and Anthony (2008b) used a structured modelling framework taking explicit account of sediment sources derived from different societal sectors to assess catchment compliance at national scale across England and Wales using these alternative sediment targets. This work successfully identified catchments where negative sediment impacts on fish are being reported.

The use of sediment yields to represent sediment targets is undermined by a number of problems. Since suspended sediment fluxes represent the aggregate of sediment delivery, their utility is best found in helping to define overall catchment response to environmental pressures as opposed to ecological impacts. Reliable coupling of sediment loadings to ecological impacts requires understanding of additional metrics such as sediment deposition and flushing and sediment grain size characteristics. It also important to highlight that all modelling data is based by common consensus on inadequate knowledge of all pathways and adequate monitoring data assumptions and therefore should not be used in isolation, but as part of an integrated modelling and monitoring approach, to help mange uncertainty and ground truth results. Anecdotal evidence from stakeholders on the impacts of fine sediments upon ecosystem can also provide important insight, and therefore should not be ignored.

VI. Concluding Remarks

Understanding and monitoring sedimentation pressure is key to ensuring the delicate ecological mechanisms of riverine ecosystems are preserved. Natural sediment pressures within river systems vary dramatically depending on catchment topography, geology, vegetation, local climate and land use (Hicks and Griffiths, 1992). It is now accepted that excess sediment can cause deterioration in water quality and aquatic biodiversity (Collins *et al*., 2008). The evidence here highlights the threats our aquatic fauna and flora face because of excess fine sediment pressures. The WFD objective of GES cannot be achieved without addressing this important pressure. Given the problems associated with using the FFD threshold or the alternative sediment yield based target scheme, urgent action is required to identify more meaningful revised sediment targets for England and Wales. Revised targets must take more explicit account of the impacts of sediment on aquatic ecology and should be developed in a catchment-specific manner (Collins and

McGonigle, 2008). A generic measurement of sedimentation is not reliable; therefore management should focus on the river basin scale to ensure source control, taking more account of observed impacts rather than modelled inputs. Preventing further damage to river habitats and associated species requires catchment-scale, holistic management, involving the cooperation and regulation of all land users. Managing excess sediment requires prevention and restoration measures, all of which require sound understanding of the key sources (Collins and Walling, 2004) and appropriate monitoring to gauge catchment compliance against revised and improved catchment-specific sediment targets. In order for sediment management to progress in England and Wales, better-informed sediment targets, and replicable monitoring methods are urgently required for compliance testing.

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VII. References

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