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Anthropogenic sediment traps and network dislocation in a lowland UK river

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AUTHOR CONTRIBUTIONS

Author	a	b	c	d	e	f	g	h	i
IDLF	√	√	√	√			√	√	√
JB	√	√	√	√			√	√	√
JLE	√		√	√	√				√
RC-P			√	√					√
ANV			√	√					√
SW			√	√	√				√
ALC		√	√	√	√				√
CM		√	√	√	√				√

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Data will be uploaded to the University of Northampton Data Repository and a link provided if accepted for publication or it can be requested directly from the corresponding author.

Anthropogenic sediment traps and network dislocation in a lowland UK river

ABSTRACT: Farm ponds, reservoirs and in-stream weirs exist in most lowland UK river catchments and often dominate over natural features such as lakes, wetlands, floodplains and debris dams. Artificial structures have served multiple purposes, including provision of power for historic flour milling and iron ore crushing and provision of water for medieval fishponds, canals, crop irrigation and potable supply. Although unintentional, they can significantly affect longitudinal connectivity, including sediment delivery pathways, through river catchments.

We report results from three spatially nested case studies that were undertaken in the Rother catchment ranging in scale from small farm ponds of a few square metres in area, to larger in-stream weirs and reservoirs (locally called ponds). Reservoirs typically trap sediment, decreasing sediment availability downstream, while inducing valley sediment accumulation upstream. We focus on the quantity and particle size characteristics of sediment trapped behind these structures compared to catchment soils and to sediments that are transported through, and deposited in, 'natural' gravel-bed reaches.

At all scales our results demonstrate that sediment trapping and release is particle size specific. Fine to coarse sands (125 µm to 2 mm diameter) and coarser sediments are retained behind structures at all scales while silts and clays (< 63 µm diameter) and organic matter are generally depleted in the stored sediment. Even though 75% of the

surveyed reservoirs have very low estimated trap efficiencies (<5%) , they slowly fill over time with sediment.

An important management question relates to the likely benefits of impoundment, structure or sediment removal, and whether fine (here defined as <63 μm or coarser (>63 μm) sediment is a priority for management.

Key Words: Absolute Particle Size, Farm Ponds, Connectivity, Instream Weirs, Reservoirs, Sediment Management

Introduction

Since the quantitative revolution in British Geomorphology in the 1960s and 70s, fluvial geomorphologists have attempted to quantify the impact of environmental change, including human impacts throughout the Holocene, on fluvial forms and processes at a range of timescales (e.g. Gregory, 1976; 1983). Early studies, including those undertaken or prompted by Ken Gregory for example, often built on concepts and approaches developed in North America (e.g., Leopold et al., 1964; Schumm, 1977) and used instrumented small catchments and / or explored a range of factors altering hydrological processes downstream of urban areas and reservoirs and their consequent impacts on hydraulic geometry, water quality, sediment transport and stream ecology (e.g., Walling & Gregory, 1970; Gregory & Walling, 1973; Gregory, 1978; Gregory & Park 1974; Petts, 1984). While the word 'connectivity' was not used explicitly in these early publications, it was abundantly clear that the concept of 'connectivity' existed in several contexts including in conjunction with sediment delivery (and the sediment delivery ratio expressing the balance between erosion and sediment yield) and in the explicit search for relationships between process and form. The idea of disconnecting parts of the fluvial system was also explicit, for example, in relation to large scale structures such as reservoirs and smaller scale features such as natural debris dams which interrupted the delivery of sediment and carbon downstream and, in the context of debris dams, provided a range of habitats within which carbon was processed and subsequently released for further processing downstream (e.g., Gurnell & Gregory 1988; Gurnell *et al*., 1995; 2000). Here it is noteworthy that Ferguson (1981) described sediment delivery as a 'jerky conveyor belt', again emphasising the variable temporal nature of sediment (dis)connectivity.

While we now understand that contemporary studies of fluvial geomorphology interact with, and have often borrowed from, a range of disciplines including the geological and engineering sciences, they also link directly to river ecology, catchment process studies and with climatology which has been recognised as a key driver of change (e.g. Petts, 1995; Figure 1). While the 1960s and 1970s laid the foundation for measuring and modelling fluvial processes, Gregory *et al*. (2002) and Gurnell (2018) have traced subsequent developments in Physical Geography suggesting that the discipline has become more professional, more focused on human / environment interactions and, importantly, more interdisciplinary.

Human modification of longitudinal connectivity in most British river catchments occurs at a range of temporal and spatial scales. On occasions, structural connectivity may be boosted by the presence of roads, tracks, culverts and pipes or drains (e.g., Zhang *et al*., 2016; Boardman *et al*., 2019) but can also be disrupted and delayed by the construction of hedgerows, small edge of field detention ponds, instream weirs ranging in age from years to centuries and by the presence of major reservoirs (e.g., Ward, 1981; Rickson, 2014; Brown et al., 2018). Of particular significance for UK rivers is their dramatic alteration for a range of purposes since medieval times for reasons that include communication, settlement, resource exploitation and defence (Lewin, 2010) all of which would have significantly modified longitudinal and lateral connectivity within river catchments. Wohl *et al*. (2019) further emphasise that the dominant role of human impacts in river catchments is to increase storage (increase dis-connectivity) but that connectivity is complex and time dependent (Wohl *et al*., 2019; Figure 1).

Detention ponds in the Rother catchment are usually located either at the end of ditches or in slope foot locations intercepting channelled flow resulting from the development of rills and gullies, hence our contention that their function is to disrupt longitudinal connectivity. In general, detention ponds have also been perceived as cost-effective mitigation options by the UK Environment Agency for almost a decade (EA, 2012).

These structural landscape elements alter the fluxes of water and sediment, especially during storm runoff events, demonstrating the impact of form, or structure, on hydrological processes. The majority of British rivers carry predominantly silt and clay-sized particles in suspension despite the coarser particle size distribution of local parent soils because particle size transport on hillslopes, and delivery to rivers and streams is selective, with a general increase in the clay (<2 μm diameter) fraction and a depletion in the proportion of coarse silt and fine sand (Walling, 1990; Walling *et al*., 2000; Kiani-Harchegani *et al*., 2018) although the latter show there is some dispute as to whether rill or inter-rill flows are able to transport sand most efficiently. Evans (1990a) notes that the amount of soil lost from fields by rill erosion is generally higher on soils with higher clay contents.

Very fine sediment (<63 μm diameter), comprising silts and clays, is generally more chemically active than larger sand-sized particles (e.g., Horowitz, 1991) and is frequently associated with undesirable contaminant transport including heavy metals, pesticides, some nutrients and fallout radionuclides (Walling & Foster, 2016). By contrast, microplastics in UK rivers are more usually associated with the coarser (> 63 μm – 2 mm diameter) fraction and understanding the fate of sediment of this size fraction will clearly help define the transport and fate of microplastics as these emerge as a new pollution threat (Hurley *et al*., 2018; Woodward *et al*., 2021). Most fine sediment is not transported as individual particles but as composite particles or flocs (Woodward &

Walling, 2007; Walling & Collins, 2016; Krishnappan *et al*., 2020; Upadhayay *et al*., 2021) but measuring the size of composite particles during transport was not undertaken in this project. Because fine sediment plays a pivotal role in influencing the physical, chemical and biological integrity of aquatic ecosystems there is an urgent need to manage excess fine sediment (Woodward & Foster, 1997; Collins & Zhang, 2016; Boardman & Foster, 2020; Evans, 2017). Setting informed targets for sediment concentration, load and or particle size is fraught with difficulty (Collins & Anthony, 2008; Collins *et al*., 2011; Foster *et al*., 2011; Foster & Greenwood, 2016) as background sediment inputs are required in natural systems to maintain essential nutrient supply and benthic habitat diversity. It could be the case that setting a target for zero sediment inputs to rivers would have equally detrimental impacts to that of excess sediment supply.

A recent systematic review of the relationships between invertebrates, macroinvertebrates and sediment particle size metrics analysed independent citations from two sources, Scopus and Academic Search Complete (McKenzie, 2020). Thirty (23%) of the 131 reviewed articles did not specify the particle size being considered in relation to ecological response which raises fundamental questions about how we define 'fine sediment' and what particle size fraction poses the greatest environmental threat to water quality and / or stream ecology.

Ecologically, fine sediment has been defined as particles less than 2 mm diameter (Wood & Armitage, 1997). Particles less than 2 mm can have detrimental effects on habitat quality and river ecosystem functioning (Wood & Armitage, 1997; Navel *et al*., 2010; Mathers *et al*., 2017). Defining all particles less than 2 mm as fine sediment could be considered as too broad as particles nearer 2 mm may result in different ecological impacts compared to particles of less than 63 µm Ecosystem impacts are associated with coarser particles (e.g., Jones *et al*., 2012; Burdon *et al*., 2013; Vadher *et al*., 2015; Mathers and Wood, 2016; Vadher *et al*., 2018)) whereas smaller particles may pose a more physiological threat to organisms (e.g., McKenzie *et al*., 2020; Franssen *et al*., 2014; Bašic *et al*., 2017). . Colmation, the process by which fine sediment infiltrates coarser river gravels, has also been perceived as a threat to benthic organisms as it reduces the movement of oxygenated water through substrate gravels (e.g., Descloux *et al*., 2013; Harper *et al*., 2017; Wharton *et al*., 2017) Identifying which particle size fraction poses the greatest threat to ecosystem functioning is essential for effective management of river catchments yet often remains poorly defined.

The overall aim of this paper is to explore the impact of the disruption to longitudinal connectivity caused by artificial structures in different parts of the Rother catchment, UK, focusing on particle size. This is delivered through the following four objectives.

1 To compare the absolute particle size distribution and characteristics of local source soils and sediment transported by the Rother.

2 To characterise the effects of small edge-of-field sediment traps on longitudinal connectivity.

3 To compare the absolute particle size characteristics of sediment transported by the river with that stored in 'natural' gravel-bed reaches at different time periods and with sediment stored behind in-stream weirs.

4 To analyse the upstream impacts of dam construction on valley sedimentation since the onset of ^{137}Cs fallout in 1954 and on sediment delivery downstream.

The Rother Catchment

The River Rother (Figure 1) has a catchment area of ~350 km^2. It has two major left bank sub-catchments (the Hammer ~ 25 km^2 and the Lod ~54 km^2 streams). Right bank tributary catchments are generally much smaller and flow shorter distances to the main river. Most right bank tributaries also have their origins at the foot of a chalk escarpment. The main Rother is 52 km long to its confluence with the River Arun and altitudes in the catchment range from ~240 m to ~0.4 m asl.

Average annual (1881-2016) rainfall (~863 mm) comes from the long-term record at Petworth Park. Highest mean monthly totals are recorded in December (102 mm) and November (100 mm). Occasional extreme daily rainfalls occur; the highest of which for the 20th Century of over 100 mm was recorded in 1945. Burt *et al*. (2016), using the same record for the period 1907-2014, showed statistically significant declines in average summer rainfall, in the number of summer rain days and in the annual number of rain days. The daily amount of rain per rain day in the autumn data set showed a statistically significant increase. This may be significant as cereal crops shifted from a dominant Spring sowing to Autumn sowing in the UK in the 1970s and peaked from the mid-1980s (Robinson & Sutherland, 2002), thereby exposing bare soils to increased rainfall intensities.

Flow data are from the National River Flow Archive (NRFA, n.d.). The flow gauging station at the most downstream weir at Hardham (Figure 1) records a mean annual discharge of 4.97 $m^3 s^{-1}$, but peak flow measurement is affected by backwater ponding upstream from the confluence of the Rother and Arun and is inaccurate. Discharge recording at the gauging weir immediately downstream of the Mid Rother sampling station at Iping (Figure 1) has been operational since 1967. (Mean, Q95 and Q5 discharges for the long-term record at Iping are 2.381, 0.708 and 6.46 $m^3 s^{-1}$ respectively). Iping recorded its highest peak flow of 125 $m^3 s^{-1}$ on 24th December 2013. This flood occurred at approximately the same time as the dam wall at Lurgashall Mill pond on the River Lod also failed. A similar failure of the dam occurred at Lurgashall in 1968. This earlier failure coincided with the second highest recorded peak (flow of 114.7 $m^3 s^{-1}$) at Iping on the 16th September 1968 (although the peak flow at Iping was not directly related to the dam failure on the Lod). Flow in the Rother is strongly seasonal, but the significant groundwater inputs to the river from the local Chalk aquifer in summer are reduced significantly due to abstraction for irrigation and potable water supply from the river.

The area is underlain by Cretaceous age rocks (140 – 66 my). The Lower and Upper Greensand which outcrop on either side of the river comprise sandstones, mudstone and siltstones while chalk outcrops on the drainage divides (Robinson, 2013). The Greensand produces well-draining, sandy loam soils with their fine-grained sand particles <2 mm diameter (Robinson, 2013). These sandy loam soils (e.g. Frilford, Fyfield and Shirrell Heath soil Associations) dominate the central part of the catchment and are at moderate to high risk of water erosion as are channel banks which also contain significant amounts of reworked sand in the alluvial deposits (Evans 1990b; Boardman *et al.,* 2009). Chalk soils dominate at the drainage divides, and between here and the main Rother is a band of silty clay soils developed on the Gault Clay (e.g., Denchworth Association) that are mapped as being at low risk of water erosion (Evans, 1990b).

Key land use changes in the Rother catchment between the 1930s and 2010 were recently highlighted by Foster *et al.* (2019) showing a decrease in the area of grassland (49% to 36%) and an increase in the area of arable land (13-27%). Woodland cover increased from 20% to 30% over the same timescale. The Hammer and Lod sub-catchments have significantly more woodland (42.4% and 41.8%, respectively) and less arable land (22.1 and 18.6%, respectively) and grassland (29% and 33.7%, respectively) than the whole Rother catchment. On the erosion-vulnerable Greensand soils, land use is dominated by arable crops including cereals, maize, potatoes, oil seed rape, salad and vegetables.

These fields are also very often well connected to the river system (Boardman *et al*., 2019).

Field Sampling

Field sampling required collection of potential source soil and channel bank samples to compare with actively transported fluvial sediments and with deposits accumulating in traps, and behind in-stream weirs and dams and extending as much as 2 km up-valley from the dam at Lurgashall Mill Pond. The following sections detail the sampling methods.

Several soil associations were sampled as part of a research project focusing on sediment source tracing undertaken by Evans (2019) and supplemented by additional sampling undertaken in the autumn of 2019. Sampling involved taking ~5 at 5 cm diameter shallow (5 cm deep) bulk density cores or five at 5 cm depth samples with a stainless-steel trowel from an area enclosed within ~25 m radius of a randomly selected sampling point (Collins *et al*., 2012) at least 100 m from the mapped boundary of a specific soil association. Channel bank material was collected from the upper and lower banks along the Rother and were composites from at least five locations at each site.

Two small sediment traps located southwest of Petworth (Figure 1) and installed to reduce sediment delivery to the Rother from locally erodible fields are shown in Figures 2A & B. Local fields are underlain by the erodible sandy Fyfield 2 soil association (see below). Samples were collected opportunistically in January and February 2014 from the Crows Hole (Figure 2A) and Three Gates (Figure 2B) traps. For the former, samples of the contributing field were taken using bulk density cores as described above at two sampling locations (5 samples bulked from each) while four bulk density samples were also recovered from the trap after it had dried out in late February 2014. At Three Gates, an opportunity arose during a field visit in early February 2014 to collect a sample of water from the ditch and from the sediment trap overflow for further analysis. These samples were collected towards the end of the storm on the 5^{th} February after the rills and gullies had already formed in the contributing fields but where water was still being delivered by inter-rill and rill runoff to the ditch as it continued to rain (Figure 2C). While only 15 mm fell at Petworth on the 5^{th} February, the previous two months had seen four daily rainfalls in excess of 30 mm. 76 mm fell on Dec 12^{th} 2013, and was probably responsible for initiating rills and gullies, with a further 36 mm, 40 mm and 41 mm recorded at Petworth on January 6^{th} and 12^{th} and February 2^{nd} 2014 respectively. The outflow on the 5^{th} February was sampled using a wide necked 1 litre bottle placed under the trap outflow pipe as it entered the ditch (Figure 2B). A similar container was used to sample the upper 10-15 cm of water (ca. 30 cm deep) flowing in the ditch to avoid collecting material that could have been moving as bedload. Both samples provided information on sediment concentrations by filtering through Whatman GF-C filter papers (pore size 1.2 μm). Samples from both traps were also subjected to absolute particle size and gamma-emitting radionuclide analyses as described below.

Time-integrating tube samplers, constructed to the design of Phillips *et al.* (2000), were deployed on the main Rother stream at three locations marked on Figure 1. Choice of sampling site was made difficult by access constraints and by the number of weirs located along the main river that might impede free movement of sediment. The most upstream sampling point is ~ 500 m downstream of an Environment Agency river gauging station but the weir design was unlikely to have an impact on the transport of sediment downstream. The most downstream sampling location was approximately midway between two widely spaced weirs and is again unlikely to have been influenced by either weir. However, there is some evidence that the weir downstream of the Mid Rother sampling point may have induced sand accumulation upstream and bed disturbance experiments were not carried out at this site (see below). In all cases, samplers were fixed with their inlet tubes at least 20 cm above the river bed to avoid collecting bedload. On no occasion were the tubes more than ~ 1/3 full of sediment at the time of emptying.

Tube samplers were emptied at approximately two-month intervals between January 2015 and May 2016 and were used to provide a representative suspended sediment sample under a range of flow conditions and to trap a sufficiently representative range of particle sizes for fine sediment investigation (Russell *et al.*, 2000). While Smith & Owens (2014) have suggested that some very small particles may not be trapped if they are not transported as aggregates, analysis of sediments collected from the Hammer stream suggests that this is not a significant issue in this catchment (Foster *et al*., 2019).

The riverbed disturbance method described by Lambert and Walling (1988), and evaluated in detail by Duerdoth *et al*. (2015) and McKenzie *et al*. (2021), was used to estimate the quantity of sediment stored on and within the upper ~5 cm of the riverbed gravels at the most upstream and downstream river sampling locations on the Rother (Figure 1). The method was not used at the Mid Rother site downstream of the Hammer stream confluence as sand often accumulated upstream of the weir marked on Figure 1 which would have provided an unrepresentative sample of the amount and calibre of sediment stored within the gravel substrate. Disturbance experiments were conducted when tube samplers were removed for emptying and each experiment was repeated three times (at the thalweg and 2 marginal sites) within each river reach. Approximately 2.5 litres of water were collected at each of the three disturbance sites and these samples were bulked for further analysis. Samples from the tube samplers and bulked disturbance experiments were allowed to settle for 24 hr at room temperature before syphoning off excess liquid and drying the sediment in an oven at 40^{o} C.

From a boat, 14 transects were surveyed for a distance of ~ 1 km upstream from Stedham Mill weir (Figure 2D) in September and October 2020. Water depth was measured using a Secchi disk while the depth of sand accumulation was estimated using a sharp-pointed ranging pole that was manually driven into the sediment from the anchored boat until resistance prevented further penetration. This provided a minimum estimate of sand accumulation as it was not always possible to determine whether the underlying river gravels had been reached. The difference between water depth and ranging pole penetration depth provided an estimate of the depth of sand accumulation. At the central point of each of the 14 transects an Ekman grab sampler was used to recover a surface sediment sample. These samples were subjected to absolute particle size analysis by dry sieving and the <125 μm fraction was analysed for the fallout nuclides ^{7}Be and $^{210}Pb_{ex}$ to calculate residence times and the amount of new sediment added to the point of deposition using the method of Matisoff *et al*. (2005: see below).

Lurgashall Mill Pond was cored in 2014 after the water level had drawn down due to a major flood and dam breach. Samples were recovered using a 30 cm long Russian corer (see Foster *et al*., 2019 for sampling methods). In addition, four percussion cores (coring locations 1-4, Figure 3) and one Russian core of sediment (coring location 5) were taken along a floodplain transect (Figure 3) extending approximately 2 km upstream of the pond to estimate the depth and approximate amount of sediment accumulating in the pond and valley since 1954; dated using a ^{137}Cs chronology (Walling & Foster, 2016).

Laboratory Methods & Residence Time Calculations

Low temperature loss on ignition (LOI) followed the method of Heiri *et al*. (2001). Pre-dried samples of ~ 5 g in mass were ignited in a Carbolite Muffle furnace at 450 °C for four hours, cooled in a desiccator and reweighed. Organic matter was measured as it is often associated with undesirable contaminants such as heavy metals (Horowitz, 1991) and, as it is usually of lower density than minerogenic sediment, was likely to have passed through traps, weirs and ponds rather than being stored.

Absolute particle size analysis was undertaken both by mechanical sieving at 1 ø intervals (samples upstream of Stedham weir) and by laser granulometry (catchment soils / source sediments, river, pond and floodplain sediments) using a Malvern Mastersizer with a Hydro 2000 dispersion unit. In both cases, the protocol established by Collins *et al*., (2010) and Blott *et al*., (2004) were followed. Absolute particle size was measured as there is no direct way of comparing soil and channel bank particle size characteristics with the aggregates actively transported by the river. Organic matter was removed by initially immersing the sample in ~ 10 ml of 30% hydrogen peroxide overnight and subsequently heating to 70 °C in an oven until no further CO_2 was effervesced. Once the samples were cool, 3 ml of sodium hexametaphosphate (3 %) was added to aid dispersion with samples left to stand for 2 minutes before analysis (Gray *et al*., 2010). The samples were then exposed to ultrasonic dispersal for 30 seconds in the Hydro 2000 unit before measurement.

A number of geochemical and radionuclide signatures were measured using ICP-OES and gamma spectrometry respectively following the methods described by Pulley *et al*. (2015) and Walling & Foster (2016). For the geochemical analysis, 0.5 g of dried sample was added to tetraflouromethacrylate (TFM) pressure vessels with 5 ml of aqua regia (70:30: 70% analytical reagent grade nitric acid: 30% analytical reagent grade hydrochloric acid) and microwaved in a CEM Mars 6 digestion unit for 45 minutes while slowly ramping up the temperature. After cooling, samples were made up to 50 ml with RO water before analysis (Chen *et al.* 1999; Collins *et al.* 2013). Radionuclides were typically measured over a two-day period in Ortec hyper-pure Ge well detectors cooled to liquid N temperatures. Samples were packed into 11 mm diameter high density PTFE sample tubes to a depth of 40 mm to match the Ge crystal well geometry, sealed with a turnover seal and paraffin wax, and left for 21 days to equilibrate before measurement (Walling & Foster, 2016). ^{7}Be activities were back calculated to the date of sampling from the date samples were removed from the detector using its known half-life.

Absolute particle size analysis of most soils, floodplain, pond and river samples showed a strong bimodal distribution (see below) and geochemical and radionuclide analysis of tube sampler and disturbance experiment samples were performed on two separate absolute particle size groups; 125-2000 µm (subsequently referred to as sand) and <38 µm (referred to as fine silt and clay).

Several methods are available to calculate residence times or the amount of new fine sediment added to riverbeds (Collins *et al*., 2020). Some methods require the use of a range of sediment fingerprints on source materials that were not available as part of this study. In consequence, the method described in detail by Matisoff *et al*. (2005) and applied in several recent studies (e.g., Wilson *et al*., 2007; Karwan *et al*., 2018) was used here. The method of Matisoff *et al*. (2005) assumes that while ^{7}Be and ^{210}Pb$_{ex}$ activities in atmospheric fallout vary significantly through time, their ratios remain fairly constant. Baskaran *et al*. (1997), for example, estimated that the range of ratio values for individual rainfall events could vary by as much as 2.2 to 32.6 with a mean of 14.4; a value close to that of the global average ratio of 16 (Matisoff *et al*., 2005). While the two fallout radionuclides are delivered to the ground surface at a reasonably constant ratio, the very short half-life radionuclide, ^{7}Be, decays at a much faster rate than ^{210}Pb$_{ex}$, thereby reducing the ratio over time. Unfortunately, while the method provides an estimate of sediment age, the ratio can also reduce as ^{7}Be depleted sediment from other sources (e.g., rill and gully erosion or resuspended old riverbed sediment) is added to the sediment newly deposited on the bed. The calculations made by Matisoff *et al*. (2005) and presented here, therefore use both assumptions to calculate a residence time and the percentage of new sediment added to the deposit as both assumptions are equally plausible and cannot be differentiated.

Results

The results from these analyses first explore the absolute particle size characteristics of the major catchment soil associations and data from the three tube samplers located on the mainstream of the Rother. We then examine the impact of small sediment traps on the absolute particle size distribution of stored and transported sediment and explore the implications of these impacts on contaminant transport. We then compare the characteristics of fine sediment transported by the Rother with that of sediment stored in natural gravel bed reaches of the river and with that of sediment accumulating upstream of a weir. Finally, we report the results from analysing rates of sedimentation upstream of the dam at Lurgashall Mill Pond.

Particle Size Characteristics of Soils and Transported Sediment

Figure 4A and Table 1 give absolute particle size and LOI data for four of the major soil associations in the catchment derived from the Greensand lithology (Frilford, Fyfield 2 and Shirrell Heath 2 associations) and the Gault Clay (Denchworth association). The Denchworth association clearly contains significantly less sand than the other 3 associations and has a higher organic matter content. The sandy soils located either side of the main Rother are usually well connected to the river (Boardman *et al*., 2019) by roads, tracks, ditches and culverts. While the percentage of sand drops on average between the catchment soils and the tube samples from the river (Figure 4B & Table 2), all three sites consistently contain

Soil Association	Lithology	% clay	% silt	% sand	LOI (%)
Denchworth (n = 3)	GC	8.9	71.9	19.1	7.3
Frilford (n = 3)	LG	4.4	20.4	75.3	3.6
Fyfield 2 (n = 3)	LG	6.4	48.6	45.0	2.4
Shirrell Heath 2 (n = 3)	LG	4.7	31.7	63.6	2.0

GC Gault Clay LG Lower Greensand

Table 1 Sand (>63 μm), silt (2-63 μm) and clay (<2 μm) percentages and loss on ignition data for four common soil associations found in the Rother catchment

Particle	Stodham (n = 9)		Mid Rother (n = 9)		Shopham (n = 9)	
Size	Mean	St Dev'	Mean	St Dev'	Mean	St Dev'
% Clay	13.3	2.6	11.8	4.3	15.6	3.6
% Silt	72.9	6.5	55.9	18.3	67.2	13.0
% Sand	13.8	8.6	32.3	22.4	17.2	15.3

Table 2 Average and standard deviation of sand (>63 μm), silt (2-63 μm) and clay (< 2 μm) percentages in sediment trapped in tube samplers on 9 bi-monthly occasions at three sites on the Rother arranged in order from upstream (Stodham) to downstream (Shopham)

large amounts of sand averaged over the 9 sampling occasions with the mid-Rother site, immediately downstream of its confluence with the Hammer stream.

Edge of Field Sediment Traps

Sediment traps significantly alter the amount and absolute particle size characteristics of sediment reaching the Rother although their impact is only likely to be detectable locally as the total area drained through traps is significantly less than 1% of the total catchment area and they rapidly fill with sediment during individual storm runoff events; thereby reducing their trap efficiency until emptied. Comparison between the local Fyfield 2 association soil and the sediment retained in the Three Gates Trap show a mean LOI of 3.2 (+/- 0.2%) for the field and 0.7 (+/- 0.1%) for the trap. Figure 5A compares the average absolute particle size distribution using laser granulometry for the same samples. Sand, silt and clay percentages for the source field and trap, along with LOI data, are given in Table 3. Clearly the trap is efficient at removing a significant amount of sand (~ twice the amount stored in the trap compared with the source field), but a large proportion of the silt and clay, along with the lower density organic matter, is not retained in the trap (Boardman & Foster, 2020). This raises questions regarding the most appropriate disposal method for sediment excavated from the trap once it has filled to the overspill (capacity ~ 115 t dry sediment).

The sediment concentration in the sample collected from the ditch was 3112.5 mg L^{-1} while the concentration in the trap overflow sampled at the same time was 662.6 mg L^{-1}, a fivefold decrease. Enough sediment was available for the analysis of gamma emitting radionuclides and absolute particle size analysis but not for LOI. Absolute sand, silt and clay percentages are given in Table 3 and the distributions are given in

Site (n)	Field (n = 4)		Trap (n = 8)		Ditch (n = 1)		Overflow (n = 1)	
	Mean	St Dev'	Mean	St Dev'	Mean	St Dev'	Mean	St Dev'
% Clay	4.3	0.1	1.7	0.3	6.7	N/A	14.0	N/A
% Silt	54.7	2.5	14.2	2.7	54.1	N/A	82.5	N/A
% Sand	41.0	5.1	84.1	2.9	39.2	N/A	3.5	N/A
% LOI	3.2	0.2	0.7	0.1	N/A	N/A	N/A	N/A

Table 3 Summary statistics for Crows Hole and Three Gates sediment samples

Figure 5B. While the absolute particle size distribution of the overflow is dominated by silt and clay, the sediment moving in the ditch is very similar to that of the field with a very small decrease in sand content and a very small increase in clay suggesting that there was no significant selectivity in absolute particle size transport from the field to the ditch at the time the sample was taken.

The activity of fallout and geogenic radionuclide activities are shown in Figures 6A and 6B. The short-lived isotope ^{7}Be (~ 50 day half-life) is not detectable in the field or the trap samples. ^{7}Be will usually only be detectable in the upper few mm of catchment soils (e.g., Walling & Woodward, 1992; Chapman *et al*., 2005; Walling & Foster, 2016) and the coarse resolution of the sampling (to 5 cm depth) probably means activities are unlikely to reach limits of detection (~ 6 Bq kg^{-1} for a two day count). Most fallout nuclides are adsorbed or co-precipitated on the outer coating of silt and clay-sized sediment and would not be expected to bind strongly to the coarser sand contained in the trap. The longer half-life nuclides $^{210}Pb_{ex}$ (half-life ~ 22 yr) and ^{137}Cs (half-life of ~ 30 yr) are of similar activity in both field and ditch samples probably because of their similar absolute particle size characteristics. Both show enhanced activity relative to the trap samples but have much lower activities than the very fine trap overflow sediment. While there are significant differences in activity between the fallout nuclides in different samples, the geogenic nuclides, which will form part of the primary and secondary mineral structure, show much smaller differences between the four sample types as shown in Figure 6B. That ^{7}Be and ^{137}Cs have high activities in the trap overflow sample suggests that the sediment is probably derived from the upper one or two mm of the soil profile and was probably delivered to the trap by inter-rill erosion (c.f., Walling & Woodward, 1992).

Instream Storage of Fine Sediment

At the upstream and downstream sampling points on the Rother, riverbed disturbance experiments were conducted to establish similarities or differences between the absolute particle size distributions of sediment stored in the gravel bed compared to that transported by the river over the same sampling period. Two examples are given in Figure 7. Figure 7A shows samples collected at the end of one of the wettest periods during the sampling programme, whereas Figure 7B shows samples collected over one of the driest periods. The tube samplers at Shopham and Stodham in February contained significant amounts of sand but far less than that stored and resuspended from the riverbed (Table 4). In August

Site	Method	Month	% Clay	% Silt	% Sand
Stodham	Disturbance	February	2.4	14.8	82.8
	Tube		6.7	36.9	56.4
Stodham	Disturbance	August	5.0	32.1	63.0
	Tube		16.7	77.5	5.8
Shopham	Disturbance	February	2.1	16.6	81.3
	Tube		9.4	63.2	27.4
Shopham	Disturbance	August	5.6	43.5	50.9
	Tube		11.6	85.0	3.4

Table 4 Comparison of tube sampler and bed disturbance particle size classes for February and August 2015

there was an even bigger difference between the percentage of sand retained in the tube samplers and that resuspended from the riverbed. As the previous two months were one of the driest, little sand would have been transported by the river and at both sites there was an increase in the amount of silt and clay caught in the tube samplers. Irrespective of the time of sampling, the percentage of sand resuspended from the riverbed exceeded 50 per cent. Fractionation of the tube sampler sediment into a silt and clay and a fine sand fraction showed statistically significant differences (2 tailed Student t test, $p < 0.05$, $n=7$) for all mineral magnetic signatures, two fallout radionuclides (Figure 8A; note ^{7}Be was not measured on these samples) and a number of chemical elements more likely to be associated with surface adsorption / co-precipitation than part of the mineral structure (Figure 8B). In all cases, concentrations / activities were higher in the finer fraction as would be expected.

Sand accumulation was measured over a ~1 km reach of river upstream of Stedham Mill Weir. From the 14 river cross sections surveyed, the average width of the river was 14.01 m and from 55 individual sand depth measurements throughout the reach, the average depth of sand was 0.453 m (SD 0.229 m). This provided an estimate of sand volume of 6,350 +/- 3,200 m^3. Assuming a dry bulk density of ~ 1.3 t m^3 gives a sediment mass of 8,255 +/- 4,167 t.

The 14 samples collected using the Eckman grab at Stedham Mill Weir were subjected to absolute particle size analysis by dry sieving and a selection of results are shown in Figure 9 while summary statistics are given in Table 5. Unlike the gravel bed river sections, there was less than 2% of very fine sand, silt and clay (<125 μm) on average stored in the riverbed; in significant contrast to the amount of the same size fraction transported by the river or stored in the natural bed gravels. LOI based on the <2mm fraction of the 14 samples analysed for absolute particle size produced a mean of 1.5 +/- 0.8 %, similar to that of the sediment stored in the edge of field sediment traps.

Measurement of ^{7}Be and $^{210}Pb_{ex}$ was performed on 11 of these samples, where sufficient of the <125 um fraction material was available, to calculate residence times or the proportion of recent sediment added to the riverbed following the simple approach of Matisoff *et al*. (2005) detailed in the methods section. Ideally, the $^{7}Be/^{210}Pb_{ex}$ signature in sediment should be compared to that of atmospheric fallout. As we currently have no measured $^{7}Be/^{210}Pb_{ex}$ fallout data in the area, we used

Size Range	< 125 µm	125-250 µm	250-500 µm	500 µm – 1 mm	1-2 mm	2-4 mm	4-8 mm
Mean	1.5	4.7	36.0	36.3	12.4	6.0	3.1
St Dev'	1.1	2.8	17.6	13.9	10.5	8.0	4.5

LOI on < 2mm sand fraction (%)

Mean	1.5
StDev	0.8

Table 5 Mean and Standard deviation of percentages of sediment in particle size classes from 14 samples collected from the river bed upstream of Stedham Mill Weir and loss on ignition for the < 2mm fraction

the average global ratio (16) reported by Matisoff *et al*. (2005). On the basis of these calculations, the average number of days stored was 209 +/-64 and the amount of new sediment added to the bed (as a percentage of the <125 µm fraction already there) was estimated to be 11% (+/- 11%). As $^{7}Be/^{210}Pb_{ex}$ data were not available for sediment resuspended from the gravel bed, we are currently unable to compare these estimates with the gravel bed sections of the river.

Sedimentation Upstream of Lurgashall Mill Pond

^{137}Cs profiles for the 5 floodplain and Lurgashall Mill Pond sediment cores are shown in Figure 10. While the base of the ^{137}Cs profile is at ~ 1.4 m in the Mill Pond core, a total of ~ 5.5 m of reservoir sediment was recovered from the same sampling location shown in Figure 3. As the Pond is estimated to be ~ 400 years old, sediment accumulation is ~13.4 cm per decade on average at this sampling point. Assuming the base of the ^{137}Cs profile approximates the year 1954, sediment accumulation between 1954 and the date of the dam breach in 2013 approached 23 cm per decade despite trap efficiencies currently estimated to be significantly less than 5% for this and two of the other surveyed Medieval ponds (Foster *et al*., 2019). This is likely to be a minimum estimate as no account could be taken of the possible loss of sediment from the pond during and after the 1968 dam breach until its repair.

Lurgashall Mill Pond, along with two other Medieval Ponds analysed in the Lod and Hammer catchments, trap a range of absolute particle sizes with only the very finest sediment escaping through the outflow (Foster *et al.*, 2019). However, recovery of the sand fraction at ~ 700 m downstream of the Hammer pond (Foster *et al*., 2019), and the presence of significant amounts of sand in a tube sampler located just upstream of the confluence of the Lod with the Rother, suggest that local sediment sources can rapidly replace the sand trapped in the ponds (Foster *et al*., 2019).

The total mass of sediment stored in Lurgashall Mill Pond and the upstream valley was estimated using the ^{137}Cs cores as representative of the areas upstream and downstream of the coring location and multiplying the depth to ^{137}Cs by average floodplain width and mid-point distance between sampling locations. Mass was estimated using the average dry bulk density of the floodplain cores above the first occurrence of ^{137}Cs. While crude, the analysis provides a first approximation of the impact of this Pond on sediment storage upstream of the ~8 m high dam wall. Our estimate suggests floodplain and lake storage of between 75,000 and 90,000 t over 60 years; equivalent to an average of between 1,250 and 1,500 t yr^{-1}. Upstream of Lurgashall Mill Pond, the catchment has an area of ~ 31.6 km^2. This gives an estimate of annual storage equivalent to a sediment yield of between 40 and 47 t km^{-2} yr^{-1} for the 60 years between the first ^{137}Cs fallout in 1954 and the dam breach in late 2013. Monitoring of suspended sediment yield for the River Rother was last undertaken between 1985 and 1995 based on rating curve estimates (Sear, 1996). These estimates suggest yields at the time of between 19 and 30 t km^{-2} yr^{-1}. While the Lod is a much smaller catchment than the Rother, the river is also protected by a pond in its headwaters (Furnace Pond), although it has a very low trap efficiency at the present time (Foster *et al*., 2019). The Lod catchment also has much less arable land and a lower proportion of erosion-vulnerable soils than the Rother as a whole, so the high reconstructed yield based on the amount of sediment deposited was unexpected.

Discussion and Conclusion

Small ponds, weirs and reservoirs have complex impacts on downstream sediment delivery. While edge of field traps probably reduce sediment concentration while their storage capacity remains high, they preferentially trap carbon-depleted sand and allow most silt, clay, surface bound contaminants and organic matter to pass through. The remaining sediment is problematic as to maintain trap efficiency, the traps need regular emptying but returning sand directly to the field would likely increase erosion risk and deplete existing soil organic matter and nutrient levels. A very high sediment concentration was recorded in the single ditch flow sample of over 3000 mg L^{-1} despite being sampled towards the end of only a 15 mm daily rainfall event. However, 193 mm had been received in four high intensity (>30 mm) daily events since 12th December 2013 suggesting that sediment delivery from fields was not supply limited.

In-stream weirs are also effective sand traps but they retain very little silt and clay. Removal of weirs, without first removing the sand, would likely impact on downstream river gravels and reduce habitat diversity, especially in a river like the Rother which has 15 weirs along its main channel where sand would likely remain trapped. What to do with thousands of tonnes of sand stored behind weirs, if dredged, leads to similar problems with the sand retained in edge of field traps. To date we have observed sediment being returned directly to fields from edge of field traps but have no information on potential alternative commercial exploitability of the trapped material. Smothering what was probably originally a gravel bed in the Rother with fine sediment (Sear, 1996) has resulted in local anglers reporting poor fishing potential upstream of Stedham Mill weir. These reports may be attributed to the effect that sand

and fine sediment deposition can have on the ecological function of spawning gravel beds. The impact of sand and fine sediment deposition on and in gravel beds include reduced flow of oxygenated water to embryos, reduced metabolic waste removal and consequently a reduced potential for fry emergence (e.g., Kemp *et al*., 2011; Sear *et al*., 2016; Bašic *et al*., 2017).

An important debate in recent literature concerns the potential benefits of removing of in-stream weirs and larger dams. Large dam removal is much more common in North America, and recent research there suggested that few studies have investigated the impacts of large dam removal and that, in some cases, there is significant conflict between proponents of river restoration and local communities who value the benefits of the impoundment (e.g., Foley *et al*., Magilligan *et al*., 2017). The original medieval construction of the weir at Stedham was to provide waterpower for milling; a function that is clearly no longer required. As with many weirs and other in-stream structures in the UK, the potential benefits and disbenefits of weir removal are under consideration (e.g., Rickard *et al*., 2003; de Leinez, 2008) but to date no known definitive guidance exists to help plan the process of weir removal whilst minimising environmental disturbance. This potential trend towards the removal of in-stream structures may begin to reverse some of the river modifications that began in medieval times (Lewin, 2010) and reverse the human impacts highlighted by Wohl *et al*., (2019) which tend to increase dis-connectivity in fluvial systems.

The impact of reservoirs changes significantly through time. Post construction they will have high trap efficiencies and significantly reduce the amount of sediment transported downstream. Rates of sedimentation in many of the Medieval structures in the Rother catchment is much higher than those reported in other areas of the UK, including Inholms Pond in the headwaters of the Hammer stream, which has reconstructed erosion rates under forest at least 4 to 7 times higher than background rates reported elsewhere in the UK (Foster *et al*., 2011; 2019) probably because of the presence of erodible soils in the catchment.

Unusually in the UK, we were able to detect a significant adjustment to the valley long profile due to sedimentation for up to 2 km upstream of Lurgashall Mill Pond. This has led to a major reduction in annual sediment yield downstream equivalent to 30-50 t km^{-2} yr^{-1} from the contributing catchment for the ~60-year period between 1954 and late 2013. Removing only 1 m of sediment from the pond alone would cost in the region of £600,000. A recent survey also suggests the ponds trap efficiency is significantly less than 5%, but even dredging to the depths of the 1954 onset of ^{137}Cs fallout would only increase trap efficiency to ~ 35% (Foster *et al*., 2019). Removing the dam wall would likely pose serious long-term implications for sediment transfer downstream by exposing stored sediment to erosion and remobilisation, and by reconnecting the upstream catchment to the downstream river channel (c.f. Boardman & Foster, 2011).

There is a lack of consistency in the published literature about what constitutes fine sediment. Indeed, 23% of the publications analysed by McKenzie (2019) failed to specify the size or size range of sediment being defined as 'fine' in a systematic review of 131 papers focusing on the relationship between benthic invertebrates and fine sediment. Lack of such information in published papers reduces their value to the wider scientific community. Importantly, acceptance of such a wide definition is also unhelpful as, physically and chemically, this range of absolute particle sizes behaves in different ways, poses different ecosystem threats and is likely to derive from different catchment sources (Sear *et al*., 2016; Evans 2019). Silt and clay-sized particles in the Rother catchment are often dominated by secondary minerals with an affinity for nutrients and contaminants which may pose a pollution risk (Boardman & Foster, 2020), whereas sand-sized material is often dominated by unreactive or mostly un-weathered primary silicate minerals with very different surface chemical properties. Regardless, increased retention of interstitial fines will generate negative consequences for aquatic biology dependent on benthic habitats for critical life stages through a range of indirect impacts on parameters such as dissolved oxygen availability or exchange and removal rates for metabolic waste (Greig *et al*., 2005; Von Bertrab *et al*., 2013; Murphy *et al*., 2015). In these situations, the natural river gravel begins to approach fine sediment saturation. That fine sand or silt and clay pose different risks to habitat, but equally, interact in critical processes for interstitial sediment ingress, retention and possible exfiltration, needs to be given greater recognition in catchment sediment management.

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Data Availability

Data are available from the corresponding author.

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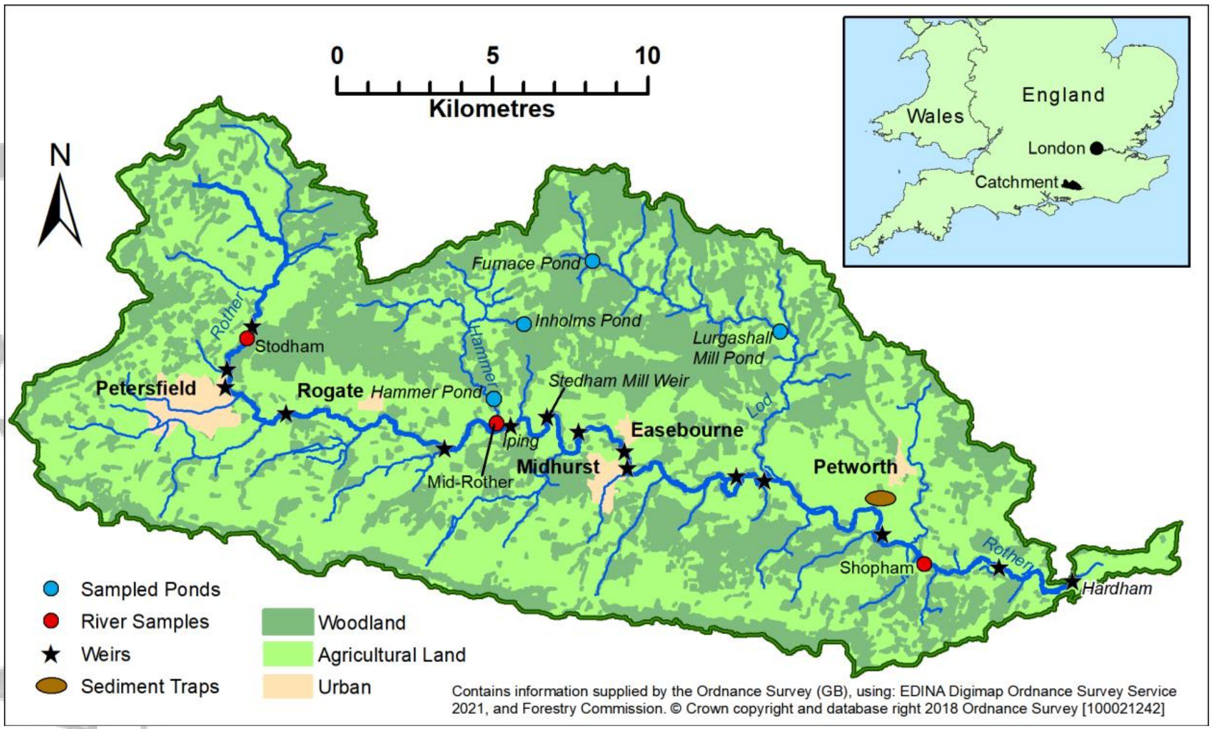

Figure 1 Location and key monitoring and sampling sites in the Rother Catchment

Figure 2 The two main sediment traps (see Figure 1 for locations) at Crows Hole Barn (A) (Photographed on 5th February 2014; the sediment trap is 16 m long by 6 m wide and collects up to ~ 1 m of sediment to the overspill at the far end in photo) and Three Gates (B) (Photographed on 7th February 2014 to show overflow pipe; the trap is of similar area to Crows Hole but is almost 2 m deep with a central overflow pipe which connects to an adjacent ditch flowing across the floodplain for ~ 50 m directly to the river Rother). A rill feeding the ditch connected to the Three Gates sediment trap (C). Note the coarse flint gravel deposited at the bottom right of 2C. Photograph taken just before collecting sediment samples from the ditch and trap overflow at around 1 pm on 5th February 2014. (D) The Stedham Weir photographed from downstream.

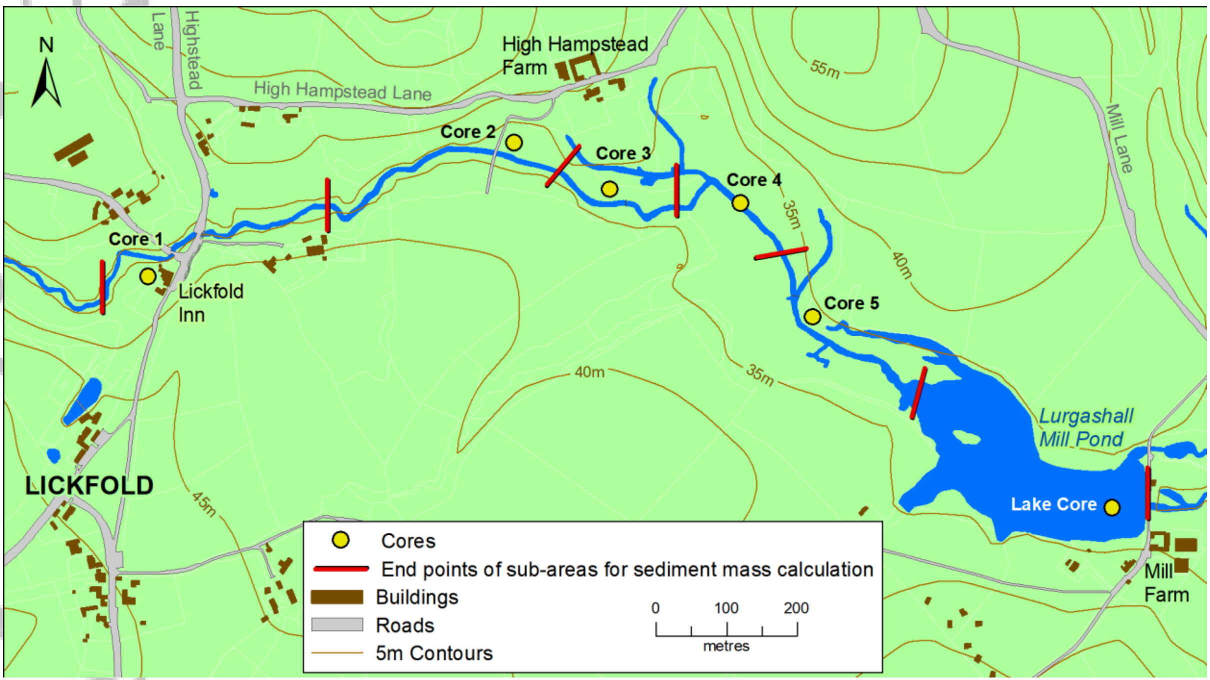

Figure 3 Location of Floodplain and Lake Cores at Lurgashall Mill Pond (see Figure 1 for location in the Rother catchment).

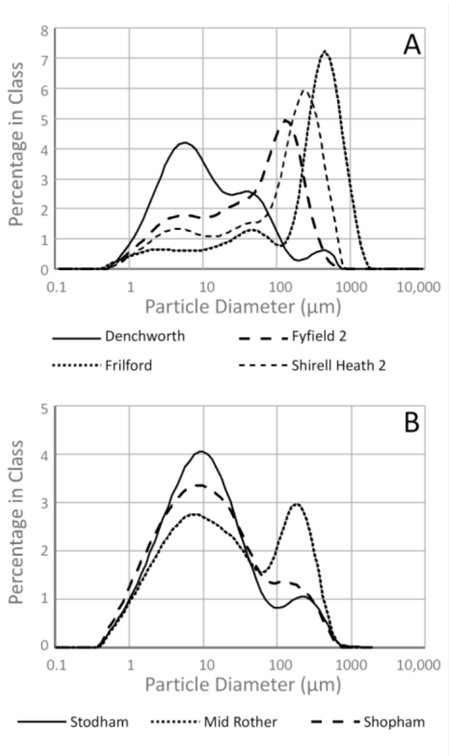

Figure 4 Particle Size Distribution of 4 major soil associations in the Rother Catchment (A) and average particle size distribution of sediment collected in the tube samplers located on Figure 1 in the River Rother at the most upstream monitoring point (Stodham), in the Mid Rother downstream of its confluence with the Hammer stream and at the most downstream sampling location (Shopham), (B) (Summary statistics are given in Tables 1 and 2)

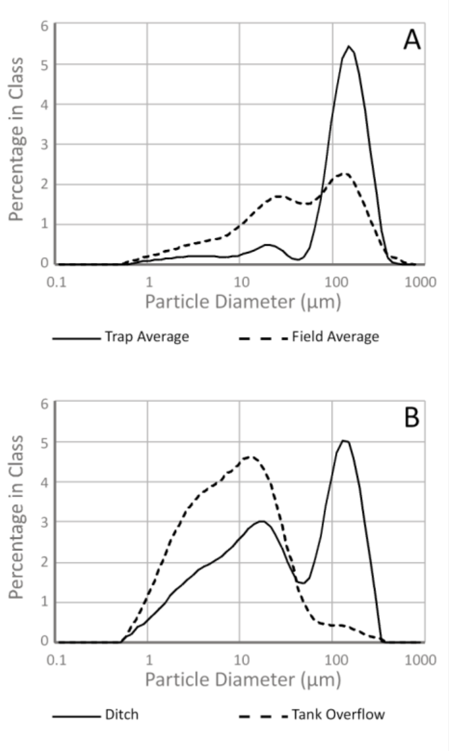

Figure 5 Comparison between average particle size characteristics of sediment collected by the Crows Hole sediment trap with that of adjacent field soils (A) and between sediment in the ditch and trap overflow at the Three Gates sediment trap (B). Summary statistics are given in Table 3.

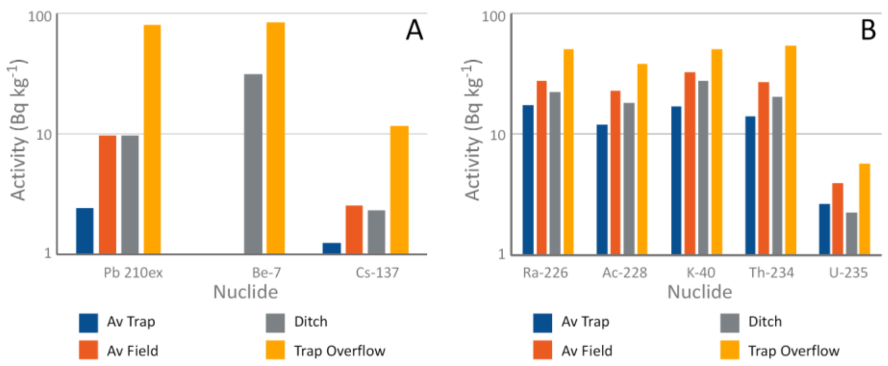

Figure 6 Comparison between fallout (A) and geogenic (B) radionuclide activities of local field soils and sediment collected at the Crows Hole and Three Gates sediment traps (note log10 Y axes on both plots)

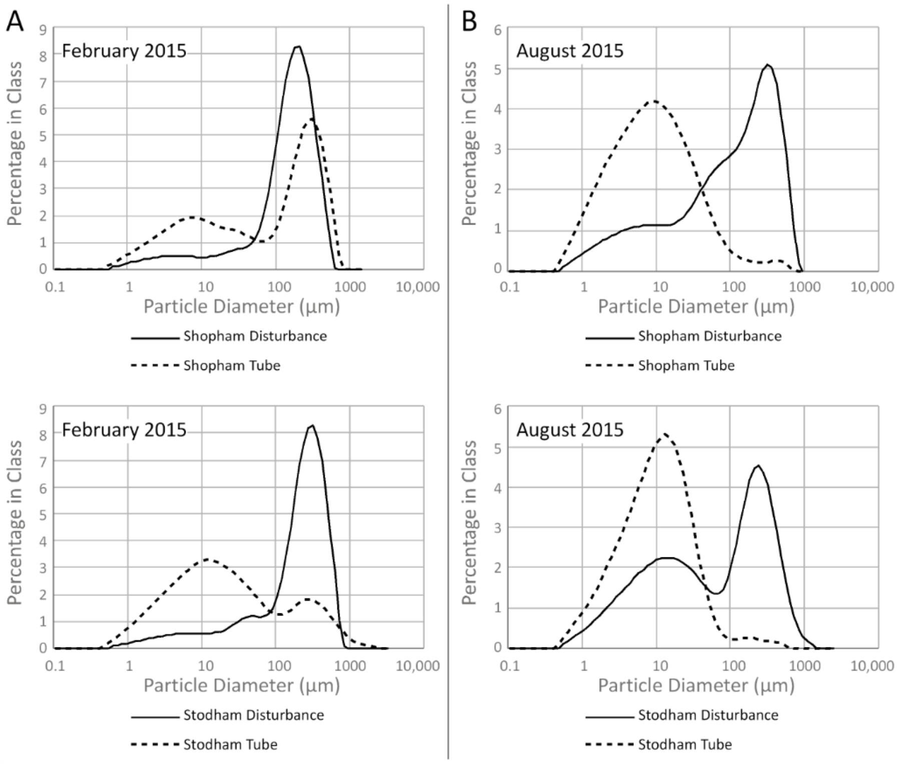

Figure 7 Comparison between the particle size distributions from bed disturbance samples and sediment collected in tube samplers at Stodham and Shopham for February (A) and August (B) 2015.

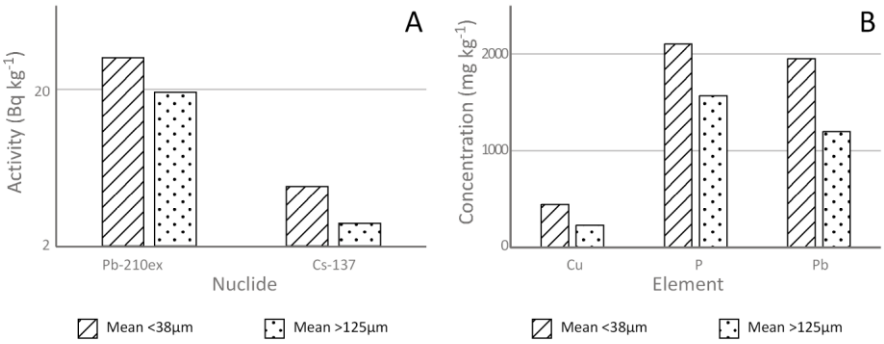

Figure 8 Average (n=7) fallout radionuclide activities (A) and selected heavy metal and phosphorus concentrations (B) in 2 particle size fractions separated from the Mid Rother tube sampler (note log10 Y axis on A only). Samples collected at bi-monthly intervals (insufficient coarse sediment available in 2 sampling months). (All differences are statistically significant at <0.05 probability using Student t test).

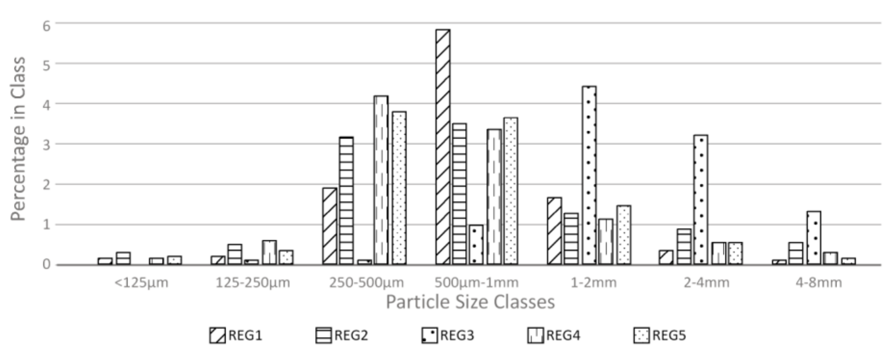

Figure 9 Absolute particle size distribution of sediment trapped behind Stedham Mill Weir in 5 of the 14 samples analysed (summary statistics for 14 samples collected are given in Table 5).

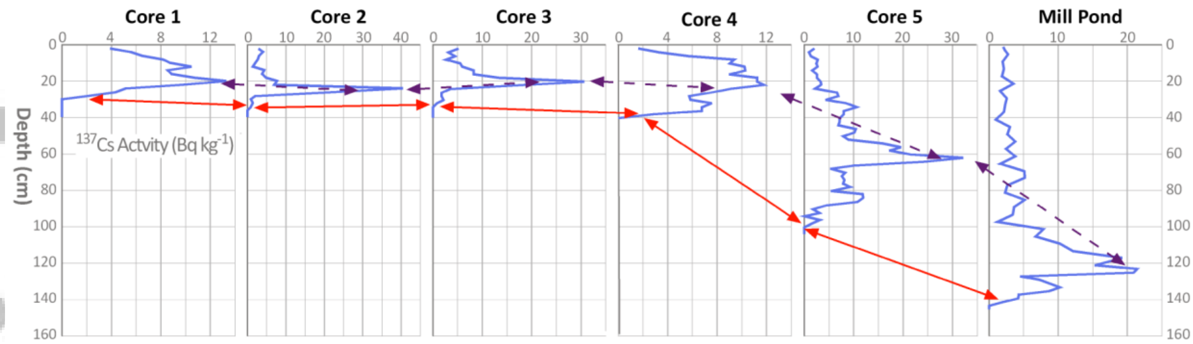

Figure 10 ^{137}Cs profiles in the 5 floodplain cores and in the main core from Lurgashall Mill Pond Arrows indicate time- synchronous horizons with years. (Lower red arrows 1954, upper purple arrows 1963 weapons ^{137}Cs fallout peak).

Graphical Abstract

A typical edge of field sediment trap with central overspill (above) and comparison between typical trap and field soil particle size (A in diagram right) and between ditch suspended sediment and trap overflow suspended sediment (B in diagram right). Note the dominance of sand retained by the trap.