The scale problem in tackling diffuse water pollution from agriculture : insights from the Avon Demonstration Test Catchment programme in England

BIDDULPH, M. <http://orcid.org/0000-0002-2953-3287>, COLLINS, A.L. <http://orcid.org/0000-0001-8790-8473>, FOSTER, I.D.L. <http://orcid.org/0000-0001-7148-7619> and HOLMES, Naomi <http://orcid.org/0000-0002-0665-3518>

Scale problems in tackling diffuse agricultural pollution

The scale problem in tackling diffuse water pollution from agriculture: insights from the Avon Demonstration Test Catchment programme in England

Biddulph, M.[1]*, Collins, A.L.[2], Foster, I.D.L.[1,3], Holmes, N.[4]

[1]Faculty of Art, Science and Technology, University of Northampton, NN2 6JD, UK
[2]Sustainable Agriculture Sciences Department, Rothamsted Research, North Wyke, Okehampton, Devon EX20 2SB, UK
[3]Geography Department, Rhodes University, Grahamstown 6140, Eastern Cape, South Africa.
[4]Department of the Natural and Built Environment, Sheffield Hallam University, City Campus, Howard Street, Sheffield S1 1WB, UK

**Correspondence to: Matilda Biddulph, Faculty of Art, Science and Technology, University of Northampton, NN2 6JD, UK. Email: mattie.biddulph@gmail.com.*

Abstract

Mitigation of diffuse water pollution from agriculture (DWPA) is of concern in the UK, so that freshwater quality can be improved in line with environmental objectives. Targeted on-farm mitigation is necessary for controlling sources of pollution to rivers; a positive impact must also be delivered at the sub-catchment and catchment scales before good ecological status can be achieved. A farm on the River Sem in the Hampshire Avon Demonstration Test Catchment (DTC) was selected for monitoring due to its degraded farmyard, track and drainage ditch, which was targeted by the DTC programme for improvement using a treatment-train of interventions. The river was monitored before and after, upstream and downstream, of the potential sources of pollution and subsequent mitigation, both locally at farm-scale, and downstream at the sub-catchment scale. Sediment was obtained from the riverbed using a conventional disturbance technique, and source samples were collected from across the sub-catchment. Samples were analysed for geochemistry, mineral magnetism, and environmental radionuclide activity using the <63m fraction, before sediment source fingerprinting was conducted to apportion sources. Source tracing revealed that, although the degraded farm track was experiencing channelized flow and erosion in the pre-mitigation period, it was not a major sediment source even at farm scale. Repeat source apportionment during the pre- and post-mitigation periods showed that the targeted treatment-train did not result in statistically significant decreases in predicted contributions from the farm track sources at either scale. Sediment sources must be determined at a range of spatial scales to support effective mitigation.

Keywords: Mitigation; Agriculture; Connectivity; Water quality; Sediment fingerprinting; Diffuse water pollution

Introduction

Mitigation of diffuse water pollution from agriculture (DWPA) is of primary concern in the UK, due to policy objectives to improve water quality and requirements to achieve ‘good ecological status’ of freshwaters under the EU Water Framework Directive (WFD; European Parliament, 2000; 2000/60/EC). Agricultural land covers approximately 70% of England and Wales (McGonigle *et al.*, 2012) and, with a growing population and increasing demands for food production, the intensity of agricultural practice has increased, leading to enhanced connectivity between the landscape and rivers and resulting in elevated losses of sediment and associated contaminants such as phosphorus and nitrogen (Johnes *et al.*, 2007; Foster *et al.*, 2011; Collins and Zhang, 2016).

Targeted, farm-scale mitigation is necessary, to control pollutant sources and prevent delivery of excess sediment and associated contaminants (Ockenden *et al.*, 2012). Mitigation can involve changes to farm management, such as the timings of fertiliser spreading and over-winter housing of livestock, but also can involve improvements to farm infrastructure, such as roofing farm yards, clean and dirty water separation, resurfacing farm tracks, maintaining drainage ditches and increasing the length and impermeability of hedgerows and riparian vegetation (e.g. Cuttle *et al.*, 2007; 2016).

However, farm-scale improvements to water quality through targeted mitigation of DWPA also need to deliver a positive impact at sub-catchment and catchment scales before good ecological status can be achieved at the compliance reporting scales (e.g. WFD waterbodies) used for current policy delivery and assessment. It is important, therefore, that on-farm mitigation is effective enough to show an impact further downstream. Here, there are many common challenges for the signal-to-noise effect, i.e. isolating the impact of the targeted intervention from background variability in hydroclimatology, water quality and sediment transport as landscape scale increases. Issues include targeting the most important on-farm pollutant sources and delivery pathways, the density of the on-farm measures across different landscape scales, the contribution of agricultural inputs to the water quality problem in the context of non-agricultural sources, including urban areas and domestic septic tanks, changing hydrological/biogeochemical process domains, and the maintenance of measures following implementation.

A challenge for managing DWPA concerns delivering robust empirical evidence on the efficacy of on-farm interventions at landscape scale (Lloyd *et al.*, 2014). There is a lack of such evidence in the current literature (McGonigle *et al.*, 2014), yet it is essential for keeping major stakeholders, including farmers, engaged in the direction of travel for environmental improvement. Here, lags in the response of conventional water quality data to targeted intervention (e.g. Boesch *et al.*, 2001; Wang *et al.*, 2002; 2016; McDowell *et al.*, 2003) pose a challenge for stakeholder engagement, since those lags can be up to decadal in duration, especially in the case of diffuse nutrient and sediment pollution. In this context, a toolkit of monitoring methods is required to ensure that empirical data streams, with more sensitivity to targeted intervention, are collected. Against this background, sediment source fingerprinting is a useful tool for identifying the major sources of sediment and associated contaminants across scales (e.g. Collins *et al.*, 1997; 2010a; Walling *et al.*, 2006; Pulley *et al.*, 2015; Walling and Foster, 2016; Collins *et al.*, 2017), as well as assessing the effectiveness of mitigation measures at farm and sub-catchment scales by quantifying the source contribution before and after mitigation (e.g. Collins *et al.*, 2010b).

In England, the DTC programme was established in December 2009 to test the efficacy of targeted on-farm interventions for water quality control at multiple (i.e. farm to landscape to catchment to national) scales (McGonigle *et al.*, 2014). This programme is founded on testing on-farm interventions using a comparison of control and manipulated areas within a before-after-control-impact (BACI) experimental design and seeks to employ a toolkit of monitoring methods (e.g. Outram *et al.*, 2014; Lloyd *et al.*, 2016), rather than conventional water quality monitoring alone. More specifically, in the Hampshire Avon DTC, work as part of a PhD programme assessed the efficacy of targeted intervention at measure to landscape scales to provide valuable insight into the challenges of delivering improvements in water quality across these scales.

Study area

The Hampshire Avon DTC drains an area of 1700 km^2, rising in Pewsey, Wiltshire and flowing south into the English Channel in Christchurch, Dorset (Figure 1). The River Avon and its tributaries are a Special Area of Conservation (SAC) and a priority catchment as part of the Catchment Sensitive Farming (CSF) programme for helping to deliver WFD environmental objectives. The headwaters of the River Sem (~5km^2), representing the Priors Farm sub-catchment, were used for the study reported here since this area was identified as suffering from DWPA at the start of the DTC programme. This sub-catchment is underlain almost entirely by the Kimmeridge clay (Jurassic) formation, has slowly permeable soils (Wickham and Denchworth soil series) prone to seasonal waterlogging and is characterised by very little topographical variation and flashy hydrology (Allen *et al*., 2014). Annual average rainfall is ~863mm. Land use is dominated by dairy farming and low intensity mixed livestock grazing (91% of the sub-catchment area).

On-farm mitigation implemented by the DTC programme

The headwaters of the River Sem flow through a dairy farm (Hays Farm), before continuing downstream to a neighbouring lowland grazing farm (Figure 2). Catchment walkover surveys at the start of the DTC programme identified a degraded farmyard (clean and dirty water separation and lack of roofing issues) and a track linking that farmyard to the stream on Hays Farm. The degraded farm track was producing and delivering sediment and associated contaminants down slope towards a drainage ditch connected to the river, as well as off a bridge crossing into the river directly (Figures 3 & 4). Targeted intervention was implemented between June and July 2013 whereby a pollution control cascade comprising the farmyard and track linking the yard to the stream was funded by the DTC programme. Work involved resurfacing the steepest (upper) part of the farm track (FTU; Figure 4) and digging a swale to one side, which was connected to a retention pond at the foot of the slope (Figure 3). The drainage ditch running beside the lower part of the degraded farm track (FTL; Figure 3) was also dredged (Figure 5), to improve storage capacity and help reduce delivery of sediment and associated contaminants to the stream. DTC funding was not sufficient to re-surface and improve FTL substantially, although the surface was rolled to remove any major erosion channels. The banks of the drainage ditch were allowed to re-vegetate naturally to trap runoff and sediment from the track, encourage uptake of contaminants and increase flow retention (Figure 6). V-notch weirs were also installed in the drainage ditch to further increase flow retention (Figure 5). It should also be noted that the channel banks of the River Sem through this site are steep and prone to fluvial scour during flashy runoff that characterises this sub-catchment. In 2012, before the study began, the channel banks were re-profiled and fencing was installed along either side to prevent poaching from cattle and to allow the development of a vegetated buffer. As this intervention was implemented before research began, it was not possible to analyse the differences in sediment contribution between pre- and post-mitigation, however the change in overall contribution over time could still be examined.

Methods

Field work

The impact of the targeted on-farm interventions at Hays Farm in the headwaters of the River Sem was monitored following the BACI approach (e.g. Stewart-Oaten *et al.*, 1986; Roley *et al.*, 2012). To assess the impact of the on-farm interventions, fine-grained sediment (<63μm) stored on the riverbed was collected at sampling locations upstream (A) and downstream of the bridge crossing (B) and ditch (C) confluence, as well as further downstream at the sub-catchment outlet (D) used by this study (Figure 2). Bed sediment disturbance is commonly used to provide sediment samples for the analysis of sediment properties and provenance (Lambert and Walling, 1988; Duerdoth *et al.*, 2016; Naden *et al.*, 2016), and was one of the methods employed in this study. A hard plastic stilling well, 70 cm in height and 50 cm in diameter, was pushed firmly into the riverbed until a seal was created within the well. The depth of the water was measured, then the water and top ~5 cm of the riverbed substrate was manually agitated for around one minute with a wooden pole until the stored sediment was suspended in the water (e.g. Walling *et al*., 2003). Five 500 ml polyethylene bottles, secured together in a line, were then immediately plunged into the agitated water and filled. The disturbance measurements were repeated in three areas at each monitoring location, to achieve a spatial representation of sediment stored within the reach (e.g. Walling *et al*., 1998). The three repeat areas were selected to represent the erosional and depositional areas at the sampling location; measurements were not repeated in the exact same positions each month, due to constraints with creating a seal and the need for an adequate flow depth for water sampling but recent tests of this method have underscored its reliability even in the context of such factors (Duerdoth *et al*., 2015). Bed sediment disturbance was undertaken monthly between January 2013 and April 2014, and thereafter every other month until March 2015. To assess the impact of the targeted on-farm intervention on the river before and after mitigation, data were grouped into pre-mitigation (January to June 2013), and post-mitigation (November 2013 to March 2015) periods. The intervening period of July to October 2013 encompassed the on-farm works to deliver the treatment-train.

Sediment source sampling was conducted to determine the provenance of the in-stream sediment. Source samples were collected from eroding channel banks, damaged road verges, topsoil sources (e.g. poached pasture soils), and Hays Farm track sources (upper pre-mitigation, upper post-mitigation, and lower track). These potential sources were identified using topographic maps and walkover surveys of the sub-catchment to identify areas of potential connectivity with the river. Samples were obtained by collecting surface scrapes to approximately 2 cm depth (e.g. Collins *et al.*, 2012), to collect material likely to be mobilised by water (Gruszowski *et al*., 2003; Walling *et al*., 2008; Collins *et al*., 2010a). Channel bank samples were collected from the entire bank profile (e.g. Collins *et al.*, 2010a), and from the upstream and downstream extent of the River Sem sub-catchment, excluding the drainage ditch on Hays Farm. Samples of each source were collected from across the entire sub-catchment to ensure a full spatial representation of the potential sources, and were collected in three sampling campaigns in December 2012, June 2014 and February 2015. The numbers of samples collected to characterise each sediment source are shown in Table 1. Sediment source fingerprinting was used to determine sediment provenance, at the farm scale, upstream (A; Figure 2) and downstream (B and C; Figure 2) of the targeted interventions, and at the sub-catchment scale further downstream (D; Figure 2).

Laboratory methods

In the laboratory, all samples were dried at 40°C, disaggregated with a pestle and mortar, and sieved to <63μm, the size fraction primarily associated with higher concentrations of pollutants (Horowitz, 1991). The samples were weighed for mass before and after sieving, then the <63μm fraction was analysed for several fingerprint properties. Firstly, geochemistry, using an ICP-MS after acid (*aqua regia*) digestion following the methods from Pulley *et al.* (2015); ~0.8g of sample sediment was digested in 10ml of *aqua regia* at 180°C for 45 minutes in a CEM Mars 6 microwave digestion unit, before being measured using a Thermo Scientific iCAP 6500 Duo View inductively coupled plasma optical emission spectrometer for Al, B, Ba, Ca, Cr, Cu, Fe, K, Li, Mg, Mn, Mo, Ni, P, Pb, S, Sr, Ti, V, Y, Zn and Zr. Secondly, mineral magnetism was determined using ~10g samples of sediment packed in 10ml sample pots to a depth of 2cm. Low frequency susceptibility (χ_{lf}), saturation isothermal remanent magnetisation (1 T) (SIRM), soft isothermal remanent magnetisation (-100 mT) (IRM-100), and hard isothermal remanent magnetisation (HIRM), were measured following the procedures in Foster *et al.* (2008). Thirdly, environmental radionuclide activity was measured using ~3g of sample sediment, packed to a depth of 4cm in a PTFE sample pot and sealed with a turnover cap and paraffin wax. All samples were left to equilibriate for a minimum of 21 days to allow for in-growth of ^{226}Ra. Sediment samples were then measured for a minimum of 24 hours (86,400s) using Ortec EG&G hyper-pure Ge γ detectors, corrected for detector efficiency, background interference, sample mass, specific surface area of the sediment, and storage time. Activities of ^{137}Cs, ^{210}Pb, ^{7}Be, ^{226}Ra, ^{228}Ac, ^{40}K, ^{234}Th, ^{235}U, and ^{212}Th were then determined from analysis of the resulting spectra as described by Foster *et al.* (2007).

Data analysis

Composite fingerprints using geochemistry, mineral magnetism and environmental radionuclides, were determined using a 2-stage statistical procedure (Collins *et al.*,1997), comprising a Kruskal-Wallis *H*-test and discriminant function analysis (DFA), to test the ability of the fingerprints to discriminate between the individual potential sediment sources identified in the sub-catchment. This method has been used extensively in previous fingerprinting studies (e.g. Collins *et al.*, 1997; 2010a, b; Walling *et al.*, 2006; Pulley *et al.*, 2015). Three composite fingerprints (based on (i) sediment geochemistry, (ii) mineral magnetism (iii) fallout and geogenic radionuclides) were used in a multivariate un-mixing model (e.g. Pulley, 2014) to estimate the relative contributions of the sediment sources. Composite signatures help avoid spurious source-sediment matches and different composite signatures permit the use of properties responding to differing environmental controls, thereby providing a basis for more robust conclusions to be drawn on sediment source apportionment. The un-mixing model was constrained so that individual source contributions could only lie between 0 and 100% Source apportionment uncertainty was determined using Monte Carlo analysis, which ran 3000 iterations for each sediment sample using the median ± one median absolute deviation (MAD) of each fingerprint property for each potential source group. Goodness-of-fit (GOF) between the source-weighted predicted and measured sediment sample fingerprint property concentrations was used to assess the reliability of the un-mixing model predictions.

Any model iteration with a GOF below 80% was deemed potentially unreliable and was therefore not used for further analysis (e.g. Pulley *et al.*, 2015). Further detailed discussion of the sediment fingerprinting methodology and modelling used here can be found in Collins *et al*. (2017). For this specific study, Kruskal-Wallis *H* tests were used to test for statistically significant differences in the overall contribution of sediment sources between the farm scale (site C) and sub-catchment scale (site D), to highlight any contrasts in mitigation effectiveness as scale increases. As the constraints of this study did not allow for equal timescales for pre- and post-mitigation, additional statistical tests were conducted to compare January to March of both the pre- and post-mitigation periods to account for potential seasonal differences in sediment mobilisation and delivery from the sources under scrutiny.

Results

Figure 7 shows the range in the averaged median predicted contributions from the individual sediment sources in the River Sem sub-catchment for the pre- and post-mitigation monitoring periods. These ranges reflect the un-mixing model predictions for the individual sampling dates comprising each time-period (i.e. pre- or post-mitigation). Table 2 presents the corresponding overall averaged median source contributions at each bed sediment sampling site, again for the pre- and post-mitigation periods. The data show that, pre-mitigation, the major predicted source contribution was from eroding channel banks, with an overall averaged median at A of 91%, at B of 91%, at C of 88% and further downstream at the sub-catchment scale at D of 75% (see Figures 2 & 3 for locations of these bed sediment sampling sites). Post-mitigation, the predicted contribution from eroding channel banks remained high at 80% A, 81% B, 84% C, and a statistically significant decrease at D to 65% ($p= 0.05$; Table 2). Predicted contributions from eroding topsoil sources were far lower at the farm scale. In the pre-mitigation period there was an overall averaged median predicted contribution from topsoils of 7% to A, 6% to B and 8% to C, but a statistically significant increase to 20% at D at the sub-catchment scale ($p= 0.00$; Table 2). In the post-mitigation period, the corresponding overall averaged median predicted contribution to A was 17%, but only 5% at B and 4% at C, with a statistically significant increase to 30% at D at the sub-catchment scale ($p= 0.04$; Table 2). Corresponding predicted contributions from damaged road verges were far lower, not exceeding 3% in either the pre- or post-mitigation periods at any site (Table 2). Table 2 shows there was a relatively low contribution from the farm track sources (FTUO, FTL, FTUN) at both the farm and sub-catchment scales. In the pre-mitigation period, the overall averaged median predicted contribution to A from the upper farm track (FTUO) was 1%, at B 3%, at C 2%, and at D at the sub-catchment scale 0%. The corresponding contributions from the lower farm track (FTL) were predicted at 1% for A, 0% for B, 1% at C and 4% at D (Table 2). In the post-mitigation period, the overall averaged median predicted contribution from the upper farm track (FTUO) to A was 0%, at B 14%, at C 12% and at D a statistically significant decrease to 2% at the sub-catchment scale ($p= 0.00$; Table 2). From the lower farm track (FTL), there was an overall averaged median predicted contribution of 0% to all sites during the post-mitigation period. There was no predicted contribution from the new, resurfaced, upper farm track (FTUN) to any site during the pre- or post-mitigation periods (Table 2). To account for differences in timescale between the pre- (6 months) and post-mitigation (17 months) periods, a subset of months was compared. This subset comprised January to March 2013 in the pre-mitigation period, and January to March 2014 in the post-mitigation period (Table 2).

In the pre-mitigation period, the overall averaged median contribution from eroding channel banks decreased from 89% to 75% between sites C and D with a corresponding decrease from 3% to 0% for FTUO. In contrast, the predicted contribution from topsoils increased from 8% to 24% (Table 2). Similarly, in the post-mitigation period, the overall averaged median contribution from eroding channel banks decreased from 87% to 78% and from 9% to 0% for FTUO, whereas the corresponding contribution from topsoils increased from 4% to 18%. These winter season results, in terms of the scaling of source contributions, are consistent with those shown by the entire dataset.

Discussion

Sediment source fingerprinting identified eroding channel banks as an important source of fine-grained sediment at the farm scale during the pre-mitigation period. The Jurassic clay geology supports steep well-defined channel banks that are prone to both fluvial scour during the flashy runoff experienced in this impermeable sub-catchment and additional erosion resulting from livestock trampling and poaching. Evidence of the latter was detected during the walkover surveys at the start of the DTC programme. Discussions between DTC scientists and the farmer at Hays Farm resulted in channel bank re-fencing to address the river bank poaching issue; which was co-funded by the farmer and the CSF initiative. In conjunction with this fencing work, channel re-profiling was undertaken in October 2012. These works pre-dated the monitoring for this research, as well as the DTC funded treatment-train, implemented to address the degraded upper farm track and drainage ditch, so could not be analysed using the BACI approach, but could still be analysed for change over time. As a result of this re-profiling, the banks were steep, up to 2m in height, and were bare of vegetation, leaving them vulnerable to erosion and collapse (Figure 8). The risk of sediment mobilisation from the re-profiled channel margins was confirmed by additional DTC work using hysteretic loops to infer pollutant sources and pathways in the study area (Lloyd *et al.*, 2016). In this case, the prevalence of clockwise hysteretic loops suggested an important source of fine sediment juxtaposed to the river channel and walkover surveys confirmed that the re-profiled banks represented the most extensive potential source of this nature. Bank erosion contributions decreased between the pre- and post-mitigation periods at farm scale. The pre-mitigation period (Jan – June 2013) experienced 83% of the long-term (1961-1990) monthly average rainfall, whereas the post-mitigation period experienced 94%, but with individual months including February 2014 (235%), April 2014 (128%) and May 2014 (183%) receiving well above the LTA. Against the expectation that fluvial scour and bank erosion would be higher during wetter periods, the reduction in bank erosion contributions between the two periods suggest that the bank fencing intervention was preventing further bank instability. Scaling up from farm to sub-catchment scale, the source tracing data for both the pre- and post-mitigation periods suggested there was a decrease in the relative contributions from eroding channel banks, as the importance of other sources became greater, but that they remained high. The continued high contribution from eroding channel banks at the landscape scale, means that other farmers downstream of Hays Farm also need to consider the potential for bank fencing and cattle exclusion from the riparian zone in order to reduce bank contributions further.

Eroding topsoils were shown not to be an important source of fine-grained sediment by the fingerprinting work at the farm scale. However, scaling up from farm to sub-catchment scale, the source tracing data for both the pre- and post-mitigation periods exhibited a statistically significant increase in the relative contribution from eroding topsoils. This is consistent with the area of topsoils at risk of erosion and delivery to the river channel increasing with scale across this agricultural landscape. The study sub-catchment is heavily under-drained, which has been shown in previous studies to deliver significant quantities of mobilised topsoil to rivers (e.g. Chapman *et al*., 2001; Foster *et al*., 2003; McDowell and Wilcock, 2004; Bilotta *et al*., 2008; Zhang *et al*., 2016). Several areas of heavily poached soils were also noted during walkover surveys and some of these were directly connected to the river channel either due to proximity or as a result of surface runoff pathways, thereby also increasing the signal from eroding pasture topsoils as scale increases from farm to sub-catchment level. In the context of the results for eroding channel banks discussed above, the source tracing data clearly suggested the reduced importance of channel bank sources and a corresponding increased importance of topsoil sources with increasing scale. This has important implications for targeting of future on-farm interventions for diffuse pollution control as interventions need to reflect the dominance of specific sources at different scales.

Damaged road verges were not important sources of the fine-grained sediment at either farm or sub-catchment scale. This reflects the limited extent of the road network in this headwater sub-catchment used by the DTC programme. Previous work, however, has shown that this source type becomes more important locally as scale increases beyond the headwater study area used here in conjunction with the length of road margins and the concomitant risk of their degradation increasing (Collins *et al.*, 2010c). This reiterates the importance of implementing farm scale mitigation in the context of the larger sub-catchment scale.

The results from this research showed that the targeted treatment-train mitigation on Hays Farm did not result in significant decreases in predicted contributions from the farm track sources directly downstream of the bridge crossing and drainage ditch (sites B and C; Figure 2). Furthermore, there was a negative impact between the pre- and post-mitigation period from upper farm track sources at the farm and sub-catchment scale. The relative contribution from the lower farm track declined from pre- to post-mitigation, suggesting that either the routing of runoff from the upper farm track into the swale together with the minor works on the lower part of the track were preventing erosion, or that the drainage ditch and re-established riparian vegetation was trapping sediment mobilised from this specific source. The overall low relative contribution of this source highlights the importance of appropriate monitoring and informed decision making when implementing mitigation in a catchment to target multiple sources.

The results reported here are highly relevant to the use of treatment-trains for mitigating DWPA. Such approaches are increasingly encouraged by policy initiatives and on-farm advice programmes in that they technically help deliver multiple lines of defence against water pollution. However, the evidence at different scales presented herein underscores the need for a dual approach using treatment-trains. One approach needs to target obvious pollutant delivery pathways such as the example targeted in this study linking a polluting farmyard to the stream system, whereas the other approach needs to take due account of pollutant source and process domains across a range of scales, designing cascades or trains of measures on that basis. In the case study used in this paper, there is clear evidence of increasing sediment inputs from eroding pasture topsoils with increasing spatial scale, meaning that an appropriate treatment-train approach targeting the most common configurations of risk in the landscape needs to be rolled out on multiple farms throughout the sub-catchment. On the basis of field observations from walkover surveys, the latter will need to combine grassland compaction management and grazing management during wet weather/winter, with feeder ring management and maintenance of buffer strips. The latter intervention will also assist in managing bank erosion associated with cattle poaching which was observed below the headwater study farm which implemented bank fencing works.

Conclusions

The findings of this study underscore that it is vital that the major sources of sediment are identified at a variety of spatial scales within any given landscape prioritised for mitigation of DWPA, so that interventions can be targeted correctly. Failure to consider sediment sources and process domains across a range of spatial scales, from individual farms to landscape scale, is likely to reduce the efficacy of the on-farm interventions, especially at those scales currently used for water quality compliance reporting. This highlights the potential benefits of collaboration between farmers, coordinating multiple farm scale interventions within a sub-catchment to ensure overall improvement at increasing landscape scales. It also underscores the need for on-farm pollution management advice delivered to any individual holding within a landscape to be placed carefully in the context of the scaling issues highlighted herein. Farm advisors therefore need to be equipped with tools and information for such considerations and to be trained accordingly, to help deliver maximum impact for environmental sustainability. The pre- and post-mitigation source tracing data for farm track sources highlight the risk of contributions at both farm and landscape scale being elevated as a consequence of on-farm remedial works, at least in the short-term (1 to 2 years) during and immediately after implementation. Longer-term studies are clearly required to convince farmers that such deviations in the outcomes arising from targeted interventions are indeed short-term and must therefore be placed in a longer-term management perspective. Longer term studies would also enable short term variability in weather and climate to be evaluated in relation to changing sediment sources independent of the applied mitigation. This is important since hydroclimatic variability has the potential to govern mitigation impacts meaning that monitoring programmes must span the range of hydroclimatic variation to deliver robust assessments.

Acknowledgements

The co-funding provided for a PhD studentship (MD) by the Department of Environment, Food and Rural Affairs (Defra project WQ0225; awarded to ALC) and the University of Northampton (awarded to IDLF) is gratefully acknowledged. On-farm interventions were funded by Defra project WQ0225. Access provided by local landowners is also gratefully acknowledged. No conflict of interest is declared for this paper. Rothamsted Research receives strategic funding from the UK Biotechnology and Biological Sciences Research Council (BBSRC).

References

Allen DJ, Darling WG, Davies J, Newell AJ, Gooddy DC, Collins AL. 2014. Groundwater conceptual models: implications for evaluating diffuse pollution mitigation measures. *Quarterly Journal of Engineering Geology and Hydrogeology* **47**: 65-80.

Bilotta GS, Brazier RE, Haygarth PM, Macleod CJA, Butler P, Granger S, Krueger T, Freer J, Quinton J. 2008. Rethinking the contribution of drained and undrained grasslands to sediment related water quality problems. *Journal of Environmental Quality* **37(3):** 906-914.

Boesch DF, Brinsfield RB, Magnien RE. 2001. Chesapeake Bay eutrophication: scientific understanding, ecosystem restoration and challenges from agriculture. *Journal of Environmental Quality* **30**: 303-320.

Chapman AS, Foster IDL, Lees JA, Hodgkinson RA, Jackson RH. 2001. Particulate phosphorus transport by sub-surface drainage from agricultural land in the UK. Environmental significance at the catchment and national scales. *Science of the Total Environment* **266**: 95-102.

Collins, AL and Zhang, Y. 2016. Exceedance of modern ‘background’ fine-grained sediment delivery to rivers due to current agricultural land use and uptake of water pollution mitigation options across England and Wales. *Environmental Science and Policy* **61**: 61-73.

Collins AL, Walling DE, Leeks GJL. 1997. Fingerprinting the origin of fluvial suspended sediment in larger river basins: combining assessment of spatial provenance and source type. *Geografiska Annaler* **79A**: 239-254.

Collins AL, Walling DE, Webb L, King P. 2010a. Apportioning catchment scale sediment sources using a modified composite fingerprinting technique incorporating property weightings and prior information. *Geoderma* **155**: 249-261.

Collins AL, Walling DE, McMellin GK, Zhang Y, Gray J, McGonigle D, Cherrington R. 2010b. A preliminary investigation of the efficacy of riparian fencing schemes for reducing contributions from eroding channel banks to the siltation of salmonid spawning gravels across the south west UK. *Journal of Environmental Management* **91**: 1341-1349.

Collins AL, Walling DE, Stroud RW, Robson M, Peet LM. 2010c. Assessing damaged road verges as a suspended sediment source in the Hampshire Avon catchment, southern United Kingdom. *Hydrological Processes* **24**: 1106-1122.

Collins AL, Zhang Y, Walling DE, Grenfell SE, Smith P, Grischeff J, Locke A, Sweetapple A, Brogden D. 2012. Quantifying fine-grained sediment sources in the River Axe catchment, southwest England: application of a Monte Carlo numerical modelling framework incorporating local and genetic algorithm optimisation. *Hydrological Processes* **26**: 1962-1983.

Collins AL, Pulley S, Foster IDL, Gellis A, Porto P, Horowitz AJ. (2017). Sediment source fingerprinting as an aid to catchment management: a review of the current state of knowledge and a methodological decision-tree for end-users. *Journal of Environmental Management* **194**: 86-108.

Collins AL and Zhang Y. 2016. Exceedance of modern 'background' fine-grained sediment delivery to rivers due to current agricultural land use and uptake of water pollution mitigation options across England and Wales. *Environmental Science and Policy* **61:** 61-73.

Cuttle SP, Haygarth PM, Chadwick DR, Newell-Price P, Harris D, Shepherd MA, Chambers BJ, Humphrey R. 2007. An Inventory of Measures to Control Diffuse Water Pollution from Agriculture (DWPA). User manual. Report under Defra project ES0203.

Cuttle SP, Newell-Price Harris D, Chadwick DR, Shepherd MA, Anthony SG, Macleod CJA, Haygarth PM, Chambers BJ. 2016. A method-centric 'User Manual' for the mitigation of diffuse water pollution from agriculture. *Soil Use and Management* **32(S1)**: 162–171.

Duerdoth CP, Arnold A, Murphy JF, Naden PS, Scarlett P, Collins AL, Sear DA, Jones JI. 2015. Assessment of a rapid method for quantitative reach-scale estimates of deposited fine sediment in rivers. *Geomorphology* **230**: 37-50.

European Parliament 2000. Establishing a framework for community action in the field of water policy. Directive EC/2000/60, EU, Brussels.

Foster IDL, Chapman AS, Hodgkinson RM, Jones AR, Lees JA, Turner SE, Scott M. 2003. Changing suspended sediment and particulate phosphorus loads and pathways in underdrained lowland agricultural catchments; Herefordshire and Worcestershire, UK. *Hydrobiologia* **494**: 119-126.

Foster IDL, Boardman J, Keay-Bright J. 2007. Sediment tracing and environmental history for two small catchments, Karoo Uplands, South Africa. *Geomorphology* **90(1-2)**: 126-143.

Foster IDL, Oldfield F, Flower RJ, Keatings K. 2008. Mineral magnetic signatures in a long core from Lake Qarun, Middle Egypt. *Journal of Paleolimnology* **40(3):** 835-849.

Foster IDL, Collins AL, Naden PS, Sear DA, Jones JI, Zhang Y. 2011. The potential for paleolimnology to determine historic sediment delivery to rivers. *Journal of Palaeolimnology* **45**: 287-306.

Gruszowski KE, Foster IDL, Lees JA, Charlesworth SM. 2003 Sediment sources and transport pathways in a rural catchment, Herefordshire, UK. *Hydrological Processes* **17**: 2665-2681.

Horowitz AJ. 1991. *A primer on sediment-trace element chemistry*. Michigan: Lewis Publishers.

Johnes PJ, Foy R, Butterfield D, Haygarth PM. 2007. Land use for Good Ecological Status: an evaluation of scenarios for water bodies in England and Wales. *Soil Use and Management* **23(1)**: 176–194.

Lambert CP and Walling DE. 1988. Measurement of channel storage of suspended sediment in a gravel-bed river. *Catena* **15:** 65–80.

Lloyd CEM, Freer JE, Collins AL, Johnes PJ, Jones JI. 2014. Methods for detecting change in hydrochemical time series in response to target pollutant mitigation in river catchments. *Journal of Hydrology* **514**: 297-312.

Lloyd CEM, Freer JE, Johnes PJ, Collins AL. 2016. Using hysteresis analysis of high-resolution water quality monitoring data, including uncertainty, to infer controls on nutrient and sediment transfer in catchments. *Science of the Total Environment* **543**: 388-404.

McDowell RW and Wilcock RJ. 2004. Particulate phosphorus transport within stream flow of an agricultural catchment. *Journal of Environmental Quality* **33:** 2111-2121.

McDowell RW, Sharpley AN, Folmar G. 2003. Modification of phosphorus export from an eastern USA catchment by fluvial sediment and phosphorus inputs. *Agriculture, Ecosystems and Environment* **99**: 187-199.

McGonigle DF, Harris RC, McCamphill C, Kirk S, Dils R, MacDonald J, Bailey S. 2012. Towards a more strategic approach to research to support catchment-based policy approaches to mitigate agricultural water pollution: a UK case-study. *Environmental Science and Policy* **24:** 4-14.

McGonigle DF, Burke SP, Collins AL, Gartner R, Haft MR, Harris RC, Haygarth PM, Hedges MC, Hiscock KM, Lovett AA. 2014. Developing Demonstration Test Catchments as a platform for transdisciplinary land management research in England Wales. *Environmental Science: Processes and Impacts* **16**: 1618-1628.

Naden PS, Murphy JS, Old GH, Newman J, Scarlett P, Harman M, Duerdoth CP, Hawczak A, Pretty JL, Arnold A, Laize C, Hornby DD, Collins AL, Sear DA, Jones JI. 2016. Understanding the controls on deposited fine sediment in the streams of agricultural catchments. *Science of the Total Environment* **347**: 366-381.

Ockenden MC, Deasy C, Quinton JN, Bailey AP, Surridge B, Stoate C. 2012. Evaluation of field wetlands for mitigation of diffuse pollution from agriculture: sediment retention, cost and effectiveness. *Environmental Science and Policy* **24:** 110-119.

Outram FN, Lloyd CEM, Jonczyk J,' Benskin C McW H, Grant F, Perks MT, Deasy C, Burke SP, Collins AL, Freer J, Haygarth PM, Hiscock KM, Johnes' PJ, Lovett AA. 2014. High frequency monitoring of nitrogen and phosphorus response in three rural catchments to the end of the 2011-2012 drought in England. *Hydrology and Earth System Sciences* **18**: 3429-3488.

Pulley (2014) *Exploring fine sediment dynamics and the uncertainties associated with sediment fingerprinting in the Nene river basin, UK.* Doctoral thesis. The University of Northampton.

Pulley S, Foster I, Atunes P. 2015. The uncertainties associated with sediment fingerprinting suspended and recently deposited fluvial sediment in the Nene river basin. *Geomorphology* **228**: 303-319.

Roley SS, Tank JL, Stephen ML, Johnson LT, Beaulieu JJ, Witter JD. 2012. Floodplain restoration enhances denitrification and reach-scale nitrogen removal in an agricultural stream. *Ecological Applications* **22(1)**: 281-297.

Stewart-Oaten A, Murdoch WW, Parker KR. 1986. Environmental impact assessment: "Pseudoreplication" in time? *Ecology* **67(4)**: 929-940.

Walling DE, Owens PN, Leeks GJL. 1998. The role of channel and floodplain storage in the suspended sediment budget of the River Ouse, Yorkshire, UK. *Geomorphology* **22**: 225-242.

Walling DE, Owens PN, Carter J, Leeks GJL, Lewis S, Meharg AA, Wright J. 2003. Storage of sediment-associated nutrients and contaminants in river channel and floodplain systems. *Applied Geochemistry* **18(2):** 195-220.

Walling DE, Collins AL, Jones PA, Leeks GJL, Old G. 2006. Establishing the fine-grained sediment budgets of the Pang and Lambourn LOCAR study catchments. *Journal of Hydrology* **330**: 126-141.

Walling DE, Collins AL, Stroud R. 2008. Tracing suspended sediment and particulate phosphorus sources in catchments. *Journal of Hydrology* **350**: 274-289.

Walling DE, Foster I. 2016. Using environmental radionuclides, mineral magnetism and sediment geochemistry for tracing and dating fine sediments. In: Kondolf G.M. & Piegay H. (Eds.) *Tools in Fluvial Geomorphology*. Chichester: Wiley. doi: 10.1002/9781118648551.ch9

Wang L, Lyons J, Kanehl P. 2002. Effects of watershed best management practices on habitat and fish in Wisconsin streams. *Journal of the American Water Resources Association* **38**: 663-680.

Wang L, Stuart ME, Lewis MA, Ward RS, Skirvin D, Naden PS, Collins AL, Ascott MJ. 2016. The changing trend in nitrate concentrations in major aquifers due to historical nitrate loading from agricultural land across England and Wales from 1925 to 2150. *Science of the Total Environment* **542**: 694-705.

Zhang Y, Collins AL, Hodgkinson RA. 2016. Use of farm survey returns from the Demonstration Test Catchments to update modelled predictions of sediment and total phosphorus loadings from subsurface drains across England and Wales. *Soil Use and Management* **32(1)**: 127-137.

Tables

Table 1. Numbers of samples collected from each source in the River Sem (Priors) sub-catchment.

Potential Source	Priors Sub-catchment
Channel Banks (CB)	36
Pasture Topsoils (TS)	33
Damaged Road Verges (DRV)	16
Farm Track Upper; Old (FTUO)	10
Farm Track Upper; New (FTUN)	17
Farm Track Lower (FTL)	18

Table 2. Overall averaged median contributions from potential sediment sources to A (upstream of bridge crossing), B (downstream of bridge crossing) and C (downstream of drainage ditch) (farm scale) and D (sub-catchment outlet) in the pre- and post-mitigation periods. Kruskal-Wallis *H*-tests were used to test for statistically significant differences in predicted contributions between C (farm scale) and D (sub-catchment scale). P values <0.05 were deemed statistically significant and are highlighted in green for a decrease and red for an increase (* = <0.05; ** = <0.001).

Source		*Pre-mitigation (%)*	*Post-mitigation (%)*	*Pre-mitigation Jan-Mar (%)*	*Post-mitigation Jan-Mar (%)*
CB	A	91	80	93	81
	B	**91**	**81**	**94**	**69**
	C	88	84*	89	87*
	D	**75**	**65**	**75**	**78**
TS	A	7	17	7	14
	B	**6**	**5**	**6**	**3**
	C	8**	4*	8	4*
	D	**20**	**30**	**24**	**18**
DRV	A	0	3	1	4
	B	**0**	**0**	**0**	**0**
	C	1	0**	0	0
	D	**1**	**3**	**1**	**4**
FTUO	A	1	0	0	0
	B	**3**	**14**	**0**	**19**
	C	2	12**	3*	9*
	D	**0**	**2**	**0**	**0**
FTL	A	1	0	0	1
	B	**0**	**0**	**0**	**0**
	C	1	0	0	0
	D	**4**	**0**	**0**	**0**
FTUN	-	0	0		