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SPECIAL ISSUE PAPER

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

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Abstract

Mitigation of diffuse water pollution from agriculture is of concern in the United Kingdom, 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 subcatchment and catchment scales before good ecological status can be achieved. A farm on the River Sem in the Hampshire Avon Demonstration Test Catchment was selected for monitoring due to its degraded farmyard, track, and drainage ditch, which was targeted by the Demonstration Test Catchment 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 subcatchment scale. Sediment was obtained from the riverbed using a conventional disturbance technique, and source samples were collected from across the subcatchment. Samples were analysed for geochemistry, mineral magnetism, and environmental radionuclide activity using the <63-μm 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

agriculture, connectivity, diffuse water pollution, mitigation, sediment fingerprinting, water quality

1 | INTRODUCTION

Mitigation of diffuse water pollution from agriculture (DWPA) is of primary concern in the United Kingdom, 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 (Collins & Zhang, 2016; Foster et al., 2011; Johnes, Foy, Butterfield, & Haygarth, 2007).

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 fertilizer spreading and over-winter housing of livestock, but can also involve improvements to farm infrastructure, such as roofing farm yards, clean and dirty water

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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 subcatchment 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, that is, 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 nonagricultural 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, Freer, Collins, Johnes, & Jones, 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, Brinsfield, & Magnien, 2001; McDowell, Sharpley, & Folmar, 2003; Wang et al., 2016; Wang, Lyons, & Kanehl, 2002) pose a challenge for stakeholder engagement, because 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., 2017; Collins, Walling, & Leeks, 1997; Collins, Walling, Webb, & King, 2010; Pulley, Foster, & Atunes, 2015; Walling, Collins, Jones, Leeks, & Old, 2006; Walling & Foster, 2016), as well as assessing the effectiveness of mitigation measures at farm and subcatchment scales by quantifying the source contribution before and after mitigation (e.g., Collins, Walling, McMellin, et al., 2010).

In England, the Demonstration Test Catchment (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., Lloyd, Freer, Johnes, & Collins, 2016; Outram et al., 2014), 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.

1.1 | Study area

The Hampshire Avon DTC drains an area of 1,700 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 and a priority catchment as part of the

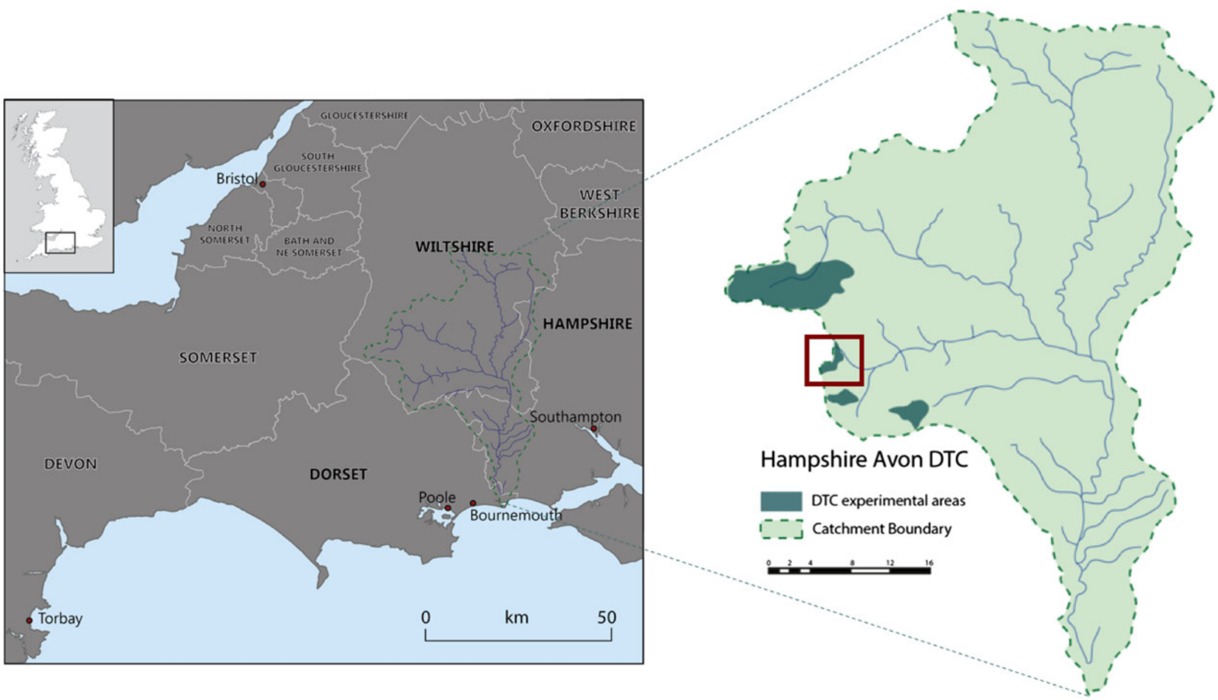

FIGURE 1 Hampshire Avon catchment and Demonstration Test Catchment (DTC) subcatchments. Red box: Priors DTC subcatchment (River Sem) [Colour figure can be viewed at wileyonlinelibrary.com]

catchment-sensitive farming programme for helping to deliver WFD environmental objectives. The headwaters of the River Sem (~5 km^2), representing the Priors Farm subcatchment, were used for the study reported here because this area was identified as suffering from DWPA at the start of the DTC programme. This subcatchment 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 characterized by very little topographical variation and flashy hydrology (Allen et al., 2014). Annual average rainfall is ~863 mm. Land use is dominated by dairy farming and low intensity mixed livestock grazing (91% of the subcatchment area).

1.2 | 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 and 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 resurface and improve FTL substantially, although the surface was rolled to remove any major erosion channels. The banks of the drainage ditch were allowed to revegetate naturally to trap run-off 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 run-off that characterizes this subcatchment. 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.

2 | METHODS

2.1 | 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., Roley et al., 2012; Stewart-Oaten, Murdoch, & Parker, 1986). To assess the impact of the on-farm interventions, fine-grained sediment (<63 μm) 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 subcatchment 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 (Duerdoth et al., 2015; Lambert & Walling, 1988; 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

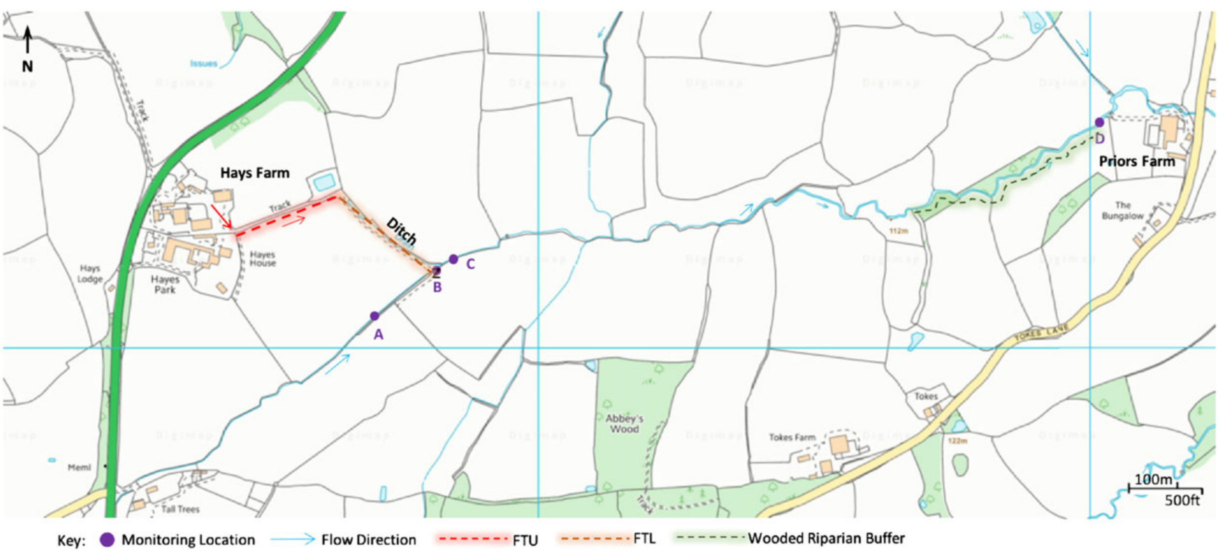

FIGURE 2 Map showing the River Sem flowing from Hays Farm to the downstream site on Priors Farm in the River Sem subcatchment. On Hays Farm, the farm track (upper farm track [FTU] and lower farm track [FTL]) and ditch are labelled, with red arrows showing the direction of run-off from the farmyard, see Figure 3 for schematic diagram of mitigation measures at this site. Map taken from the Edina Digimap© Ordnance Survey data 2007 (2016) [Colour figure can be viewed at wileyonlinelibrary.com]

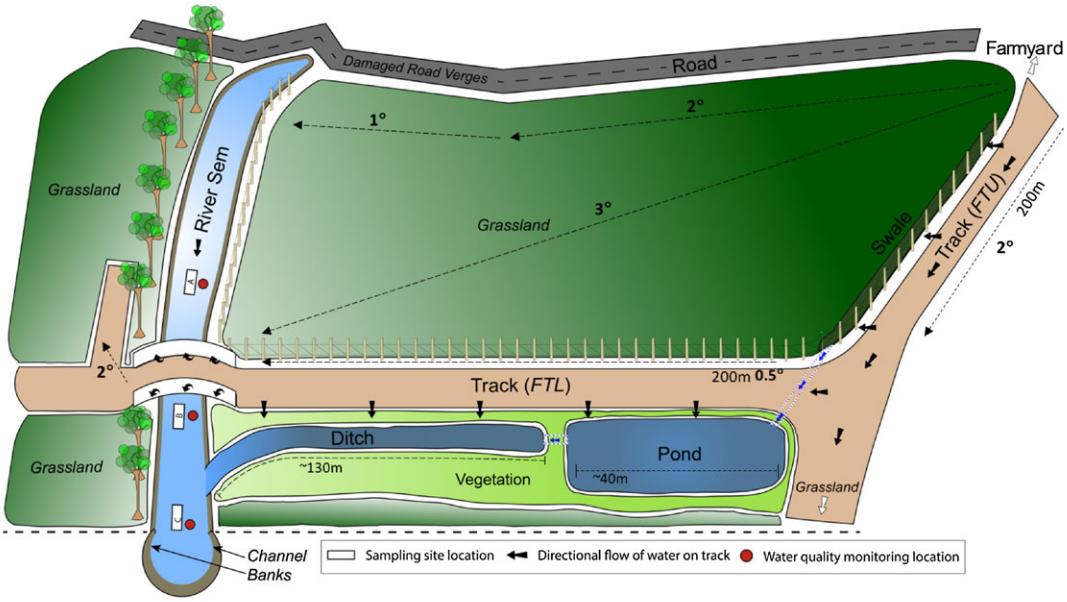

FIGURE 3 Targeted mitigation at Hays Farm implemented by the Demonstration Test Catchment programme and the sampling locations. Not to scale. FTU, upper farm track; FTL, lower farm track [Colour figure can be viewed at wileyonlinelibrary.com]

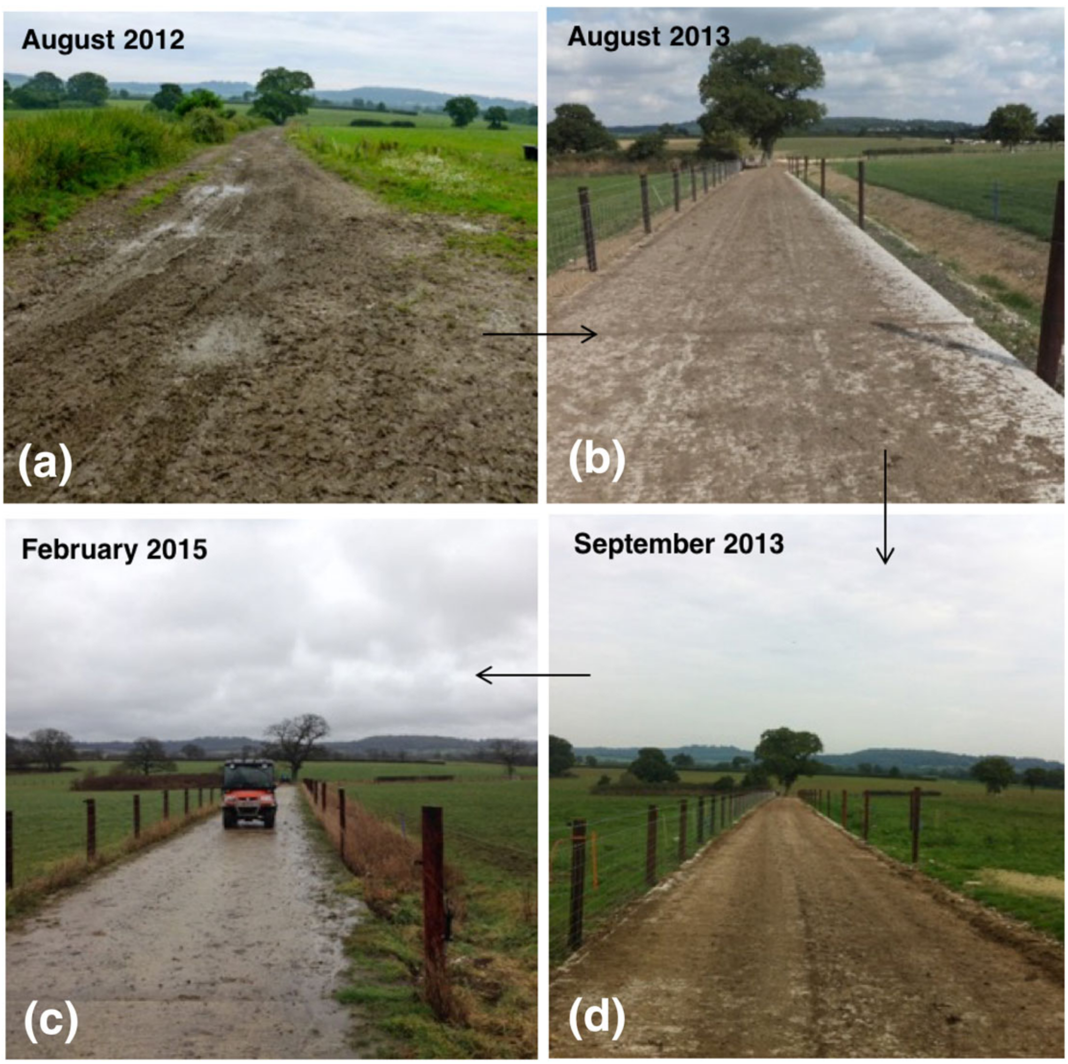

FIGURE 4 Photographs of the upper farm track at Hays Farm. (a) Standing from the farmyard looking downslope pre-mitigation and (b–d) post-mitigation [Colour figure can be viewed at wileyonlinelibrary.com]

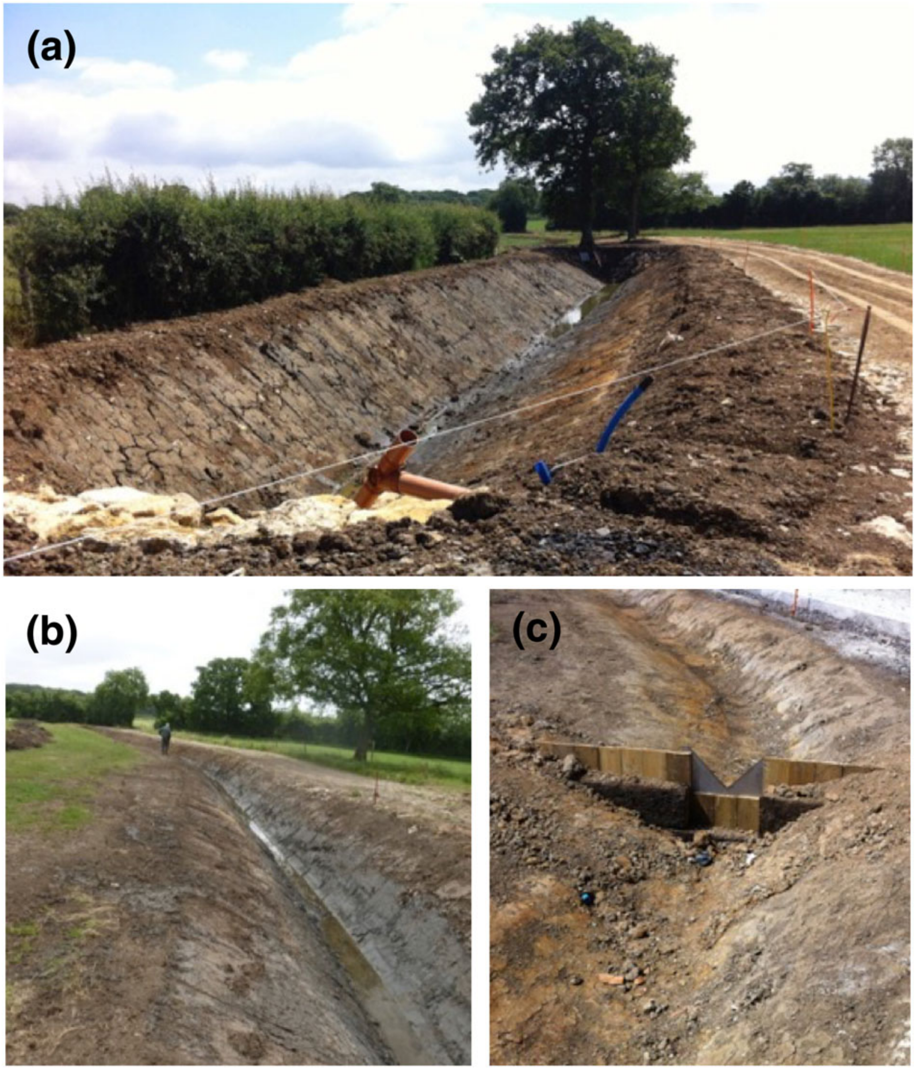

FIGURE 5 Photographs of (a) the setting pond, (b) newly dredged drainage ditch running beside lower farm track leading towards River Sem, and (c) V-notch weirs installed along the drainage ditch [Colour figure can be viewed at wileyonlinelibrary.com]

FIGURE 6 View of the drainage ditch on Hays Farm looking from the River Sem to upper farm scale: (a) Post-dredging in July 2013 and (b) dense vegetation and fencing in December 2015 [Colour figure can be viewed at wileyonlinelibrary.com]

substrate was manually agitated for around 1 min 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, Owens, & Leeks, 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 walk-over surveys of the subcatchment 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 mobilized by water (Collins, Walling, Webb, et al., 2010; Gruszowski, Foster, Lees, & Charlesworth, 2003; Walling, Collins, & Stroud, 2008). Channel bank samples were collected from the entire bank profile (e.g., Collins, Walling, Webb, et al., 2010) and from the upstream and downstream extent of the River Sem subcatchment, excluding the drainage ditch on Hays Farm. Samples of each source were collected from across the entire subcatchment 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 characterize 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 subcatchment scale further downstream (D; Figure 2).

TABLE 1 Number of samples collected from each source in the River Sem (Priors) subcatchment

Potential source	Priors subcatchment
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

2.2 | Laboratory methods

In the laboratory, all samples were dried at 40 °C, disaggregated with a pestle and mortar, and sieved to <63 μm, the size fraction primarily associated with higher concentrations of pollutants (Horowitz, 1991). The samples were weighed for mass before and after sieving, and then the <63-μm fraction was analysed for several fingerprint properties. First, geochemistry, using inductively coupled plasma mass spectrometry (ICP-MS) after acid (*aqua regia*) digestion following the methods from Pulley et al. (2015); ~0.8 g of sample sediment was digested in 10 ml of *aqua regia* at 180 °C for 45 min 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. Second, mineral magnetism was determined using ~10g samples of sediment packed in 10ml sample pots to a depth of 2 cm. Low frequency susceptibility (χ_{lf}), saturation isothermal remanent magnetization (1 T), soft isothermal remanent magnetization (−100 mT; IRM-100), and hard isothermal remanent magnetization were measured following the procedures in Foster, Oldfield, Flower, and Keatings (2008). Third, environmental radionuclide activity was measured using ~3 g of sample sediment, packed to a depth of 4 cm in a polytetrafluoroethylene sample pot, and sealed with a turnover cap and paraffin wax. All samples were left to equilibrate for a minimum of 21 days to allow for in-growth of ^{226}Ra. Sediment samples were then measured for a minimum of 24 hr (86,400 s) using Ortec EG&G hyperpure 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, Boardman, and Keay-Bright, 2007.

2.3 | Data analysis

Composite fingerprints using geochemistry, mineral magnetism, and environmental radionuclides were determined using a two-stage statistical procedure (Collins et al., 1997), comprising a Kruskal–Wallis *H* test and discriminant function analysis, to test the ability of the fingerprints to discriminate between the individual potential sediment sources identified in the subcatchment. This method has been used extensively in previous fingerprinting studies (e.g., Collins et al., 1997; Collins, Walling, McMellin, et al., 2010; Collins, Walling, Webb, et al., 2010; Pulley et al., 2015; Walling et al., 2006). Three composite fingerprints (based on [a] sediment geochemistry, [b] mineral magnetism, and [c] fallout and geogenic radionuclides) were used in a multivariate unmixing 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 unmixing 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

3,000 iterations for each sediment sample using the median ± one median absolute deviation of each fingerprint property for each potential source group. Goodness-of-fit between the source-weighted predicted and measured sediment sample fingerprint property concentrations was used to assess the reliability of the unmixing model predictions. Any model iteration with a goodness-of-fit 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 subcatchment 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 mobilization and delivery from the sources under scrutiny.

3 | RESULTS

Figure 7 shows the range in the averaged median predicted contributions from the individual sediment sources in the River Sem subcatchment for the pre- and post-mitigation monitoring periods. These ranges reflect the unmixing 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 subcatchment scale at D of 75% (see Figures 2 and 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 = .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 subcatchment scale (p = .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 subcatchment scale (p = .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 that there was a relatively low contribution from the farm track sources (FTUO, FTL, and FTUN) at both the farm and subcatchment 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 subcatchment scale 0%. The corresponding contributions from the 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 subcatchment scale (p = .00; Table 2). From the 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 FTU 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.

4 | 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 run-off experienced in this impermeable subcatchment 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 catchment-sensitive farming initiative. In conjunction with this fencing work, channel re-profiling was undertaken in October 2012. These works predated the monitoring for this research, as well as the DTC funded treatment train, implemented to address the degraded FTU 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 2 m in height, and were bare of vegetation, leaving them vulnerable to erosion and collapse (Figure 8). The risk of sediment mobilization 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 (January–June 2013) experienced 83% of the long-term (1961–1990) monthly average rainfall, whereas the post-mitigation period

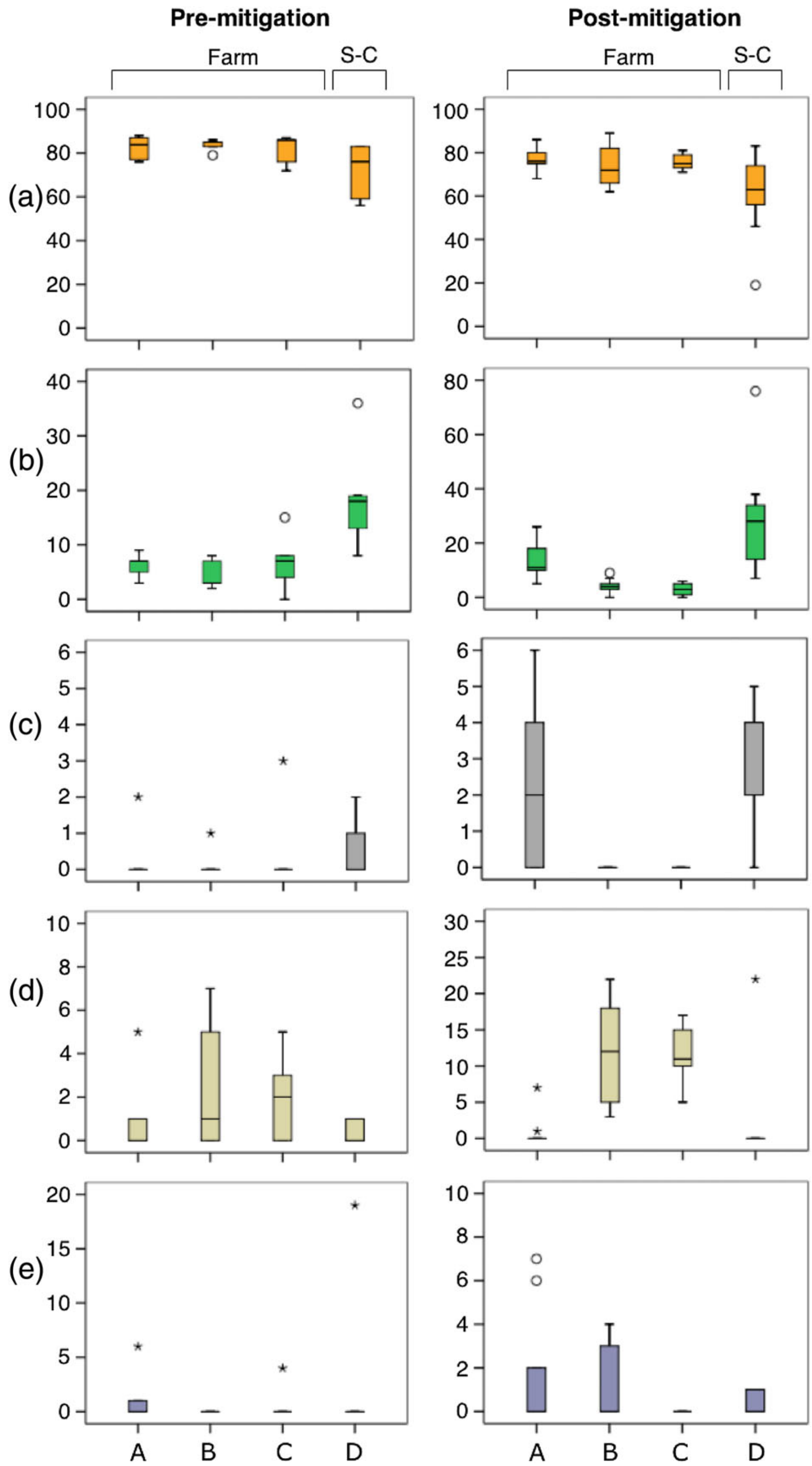

FIGURE 7 Overall averaged median predicted sediment source contribution in the pre- and post-mitigation periods. (a) Channel banks, (b) topsoil sources, (c) damaged road verges, (d) upper farm track, and (e) lower farm track. (Farm scale sites are A = upstream of bridge crossing, B = downstream of bridge crossing, and C = downstream of ditch. Subcatchment (S-C) scale is Site D) [Colour figure can be viewed at wileyonlinelibrary.com]

experienced 94%, but with individual months including February 2014 (235%), April 2014 (128%), and May 2014 (183%) receiving well above the long term average (LTA). Against the expectation that fluvial scour and bank erosion would be higher during wetter periods, the reduction

TABLE 2 Overall averaged median contributions from potential sediment sources to A (upstream of bridge crossing), B (downstream of bridge crossing), C (downstream of drainage ditch; farm scale), and D (subcatchment outlet) in the pre- and post-mitigation periods

Source		Pre-mitigation (%)	Post-mitigation (%)	Pre-mitigation January–March (%)	Post-mitigation January–March (%)
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	0	0

Note. Kruskal-Wallis *H* tests were used to test for statistically significant differences in predicted contributions between C (farm scale) and D (subcatchment scale). *p* values <.05 were deemed statistically significant and are highlighted in green for a decrease and red for an increase. CB = Channel banks; TS = pasture topsoils; DRV = damaged road verges; FTUO = old, upper farm track; FTL = lower farm track; FTUN = new, resurfaced, upper farm track.

*$p < .05$.

**$p < .001$.

in bank erosion contributions between the two periods suggests that the bank fencing intervention was preventing further bank instability. Scaling up from farm to subcatchment scale, the source tracing data for both the pre- and post-mitigation periods suggested that 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 subcatchment 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 subcatchment is heavily underdrained, which has been shown in previous studies to deliver significant quantities of mobilized topsoil to rivers (e.g., Bilotta et al., 2008; Chapman, Foster, Lees, Hodgkinson, & Jackson, 2001; Foster et al., 2003; McDowell & Wilcock, 2004; Zhang, Collins, & Hodgkinson, 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 run-off pathways, thereby also increasing the signal from eroding pasture topsoils as scale increases from farm to subcatchment 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 subcatchment scale. This reflects the limited extent of the road network in this headwater subcatchment 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, Walling, Stroud, Robson, & Peet, 2010). This reiterates the importance of implementing farm-scale mitigation in the context of the larger subcatchment 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 periods from FTU sources at the farm and subcatchment scale. The relative contribution from the FTL declined from pre- to post-mitigation, suggesting that either the routing of run-off from the FTU into the swale together with the minor works on the lower part of the track was preventing erosion or that the drainage ditch and re-established riparian vegetation was trapping sediment mobilized 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

FIGURE 8 Channel bank at Site B in the River Sem at Hays Farm in (a) December 2014 and (b) March 2015 [Colour figure can be viewed at wileyonlinelibrary.com]

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 subcatchment. 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 that was observed below the headwater study farm that implemented bank fencing works.

5 | 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 prioritized 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 subcatchment 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 because hydro-climatic variability has the potential to govern mitigation impacts meaning that monitoring programmes must span the range of hydro-climatic variation to deliver robust assessments.

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