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Keeping agricultural soil out of rivers: Evidence of sediment and nutrient accumulation within field wetlands in the UK

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Intensification of agriculture has resulted in increased soil degradation and erosion, with associated pollution of surface waters. Small field wetlands, constructed along runoff pathways, offer one option for slowing down and storing runoff in order to allow more time for sedimentation and for nutrients to be taken up by plants or micro-organisms. This paper describes research to provide quantitative evidence for the effectiveness of small field wetlands in the UK landscape. Ten wetlands were built on four farms in Cumbria and Leicestershire, UK. Annual surveys of sediment and nutrient accumulation in 2010, 2011 and 2012 indicated that most sediment was trapped at a sandy site (70 tonnes over 3 years), compared to a silty site (40 tonnes over 3 years) and a clay site (2 tonnes over 3 years). The timing of rainfall was more important than total annual rainfall for sediment accumulation, with most sediment transported in a few intense rainfall events, especially when these coincided with bare soil or poor crop cover. Nutrient concentration within sediments was inversely related to median particle size, but the total mass of nutrients trapped was dependent on the total mass of sediment trapped. Ratios of nutrient elements in the wetland sediments were consistent between sites, despite different catchment characteristics across the individual wetlands. The nutrient value of sediment collected from the wetlands was similar to that of soil in the surrounding fields; dredged sediment was considered to have value as soil replacement but not as fertiliser. Overall, small field wetlands can make a valuable contribution to keeping soil out of rivers.

1. Introduction

Soil underpins life on our planet, supporting food production, sustaining biodiversity and storing carbon. However, pressures such as intensification of agriculture, with associated pollution and soil compaction, have resulted in widespread soil degradation and erosion. In the UK, the costs of soil degradation and its resulting contribution to flooding and diffuse pollution have been quantified in monetary terms as £0.2 billion–£0.3 billion per year (DEFRA, 2009). In some parts of the world, soil loss is up to 100 times faster than the rate of soil production (Banwart, 2011). Soil preservation in the landscape is increasingly recognised as a vital step towards sustainable agriculture, along with sustainable use of nutrients. The inter-linked nitrogen and phosphorus cycles have been identified as one of nine important biophysical systems for the planet (Rockstrom et al., 2009), with operating thresholds within which humanity should strive to live; crossing these planetary thresholds could lead to unacceptable and possibly irreversible environmental change. Agriculture is one of the main sectors contributing to sediment and nutrient pollution of freshwaters, particularly in the form of diffuse pollution (Ulen et al., 2007; Wood et al., 2005). This non-point pollution is difficult to measure and mitigate because of its distributed nature. However, the resulting reduction in surface water quality and its effect on ecological status contribute to the failure of many European surface waters to achieve the objectives of the Water Framework Directive (Howarth, 2011).

Land management policy is starting to encourage and reward best management practices applied by farmers, but widespread adoption is necessary in order to see catchment-scale improvement of water quality (Collins and McGonigle, 2008; Winter et al., 2011). There is consensus among farmers and land managers that

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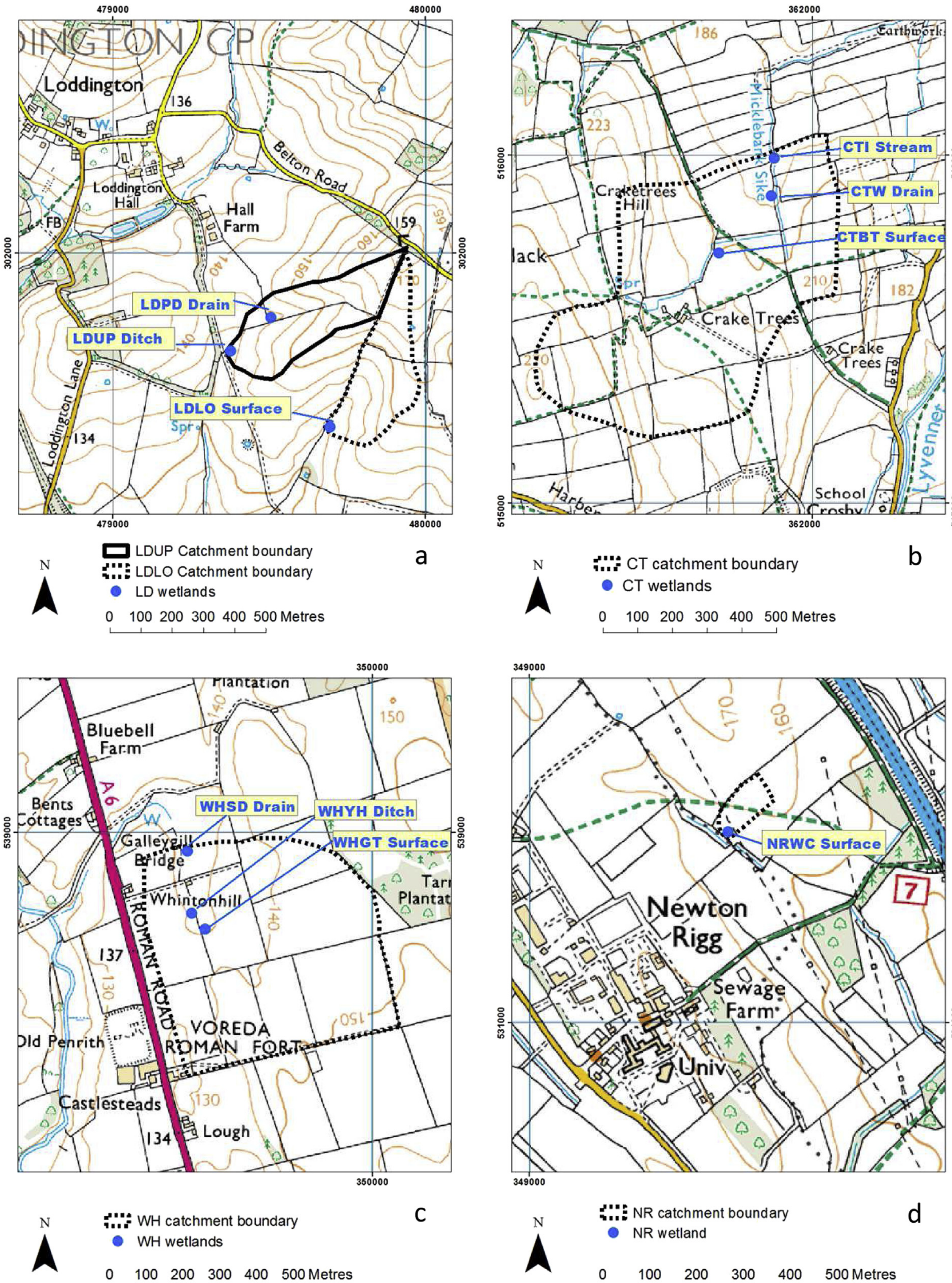

Fig. 1. Field wetland locations and contributing catchment boundaries: a) Loddington, Leicestershire, b) Crake Trees Manor, Cumbria, c) Whinton Hill, Cumbria, d) Newton Rigg, Cumbria © Crown Copyright/database right 2013. An Ordnance Survey/EDINA supplied service.

Table 1
Summary of field wetlands. The wetland dimensions are measured at the base of the ponds.

Site	Soil type	Name	Contrib. area (ha)	Dimensions (m) $l \times w \times d$	Runoff source	Land use	Monitoring period
Loddington, Leicestershire (SK 792021)	Clay	LDUP ditch	10	$15 \times 5 \times 0.5 + 5 \times 7 \times 0.5$[a]	Ditch	Arable	August 2008–December 2012
	Clay	LDPD drain	4	$2 \times 2 \times 1.5 + 8 \times 2 \times 0.5$	Surface runoff & drain	Arable	November 2009–December 2012
	Clay	LDLO surface	9	$11 \times 2 \times 0.5$	Surface Runoff	Arable	January 2010–December 2012
Crake Trees Manor, Cumbria (NY 616156)	Silty loam	CTBT surface	20	$17 \times 6 \times 0.5 + 17 \times 6 \times 0.5$	Surface runoff	2009 Grass, 2010 Arable	October 2009–December 2012
	Silty loam	CTW drain	10	$17 \times 3 \times 0.5$	Surface runoff & drain	Grass	November 2009–December 2012
	Silty loam	CTI stream	50	$25 \times 5 \times 0.5$	Stream	2009 Grass, 2010 Arable	September 2010–December 2012
Whinton Hill, Cumbria (NY 495388)	Sand	WHSD drain	30	$8 \times 8 \times 1.5 + 32 \times 8 \times 0.5$	Drain	Grass	October 2009–December 2012
	Sand	WHYH ditch	20	$17 \times 3 \times 0.5$	Ditch	Arable	August 2010–December 2012
	Sand	WHGT surface	1.5	$4 \times 1 \times 0.5 + 3.5 \times 1 \times 0.5$	Surface runoff	Arable	November 2010–December 2012
Newton Rigg, Cumbria (NY 494313)	Silt	NRWC surface	1	$2.5 \times 1 \times 1.5 + 2.5 \times 1 \times 0.5$	Surface runoff	Arable	January 2011–September 2012

[a] This wetland was shaped to fit in a field corner. The aim was to have a wetland width-to-length ratio of greater than 1:4 to increase the effectiveness of the wetland area. In this wetland, wooden boards were used to increase the flow path length.

soil erosion and diffuse pollution should be tackled primarily at source through soil management and targeted nutrient application. Plot-scale studies of in-field measures, such as minimum tillage, contour cultivation and tramline disruption, have shown reductions in runoff and associated pollutant transfer (Deasy et al., 2009). However, Deasy et al. (2009) also demonstrated large variations in the effectiveness of these measures, with certain practices being successful in some conditions but not in others. Even if sufficient financial incentives were available to encourage widespread adoption of in-field measures, there are circumstances under which they would not always work well or would be impossible to implement. In such circumstances, edge-of field measures can provide additional or backup protection of receiving waters. Constructed field wetlands are one such edge-of-field option available to farmers for reducing the loss of sediment and nutrients from the landscape, in addition to providing other ecosystem services (Vymazal, 2011). They are now used for diffuse pollution mitigation in temperate environments worldwide (e.g. Braskerud, 2002; Braskerud et al., 2005; Díaz et al., 2012; Johannesson et al., 2011; Millhollon et al., 2009; Raisin et al., 1997). Field wetlands are not yet widely used in the UK, partly because of the perception that they need to be large to be effective. Although Millhollon et al. (2009) recommended that wetlands should occupy around 2% of the catchment area, Braskerud (2002) showed that smaller wetlands, occupying only 0.03–0.4% of the catchment, could be effective when strategically located in 1st or 2nd order catchments. If these smaller wetlands could be located in relatively unproductive areas of land in the UK then they might provide a useful mitigation option for diffuse pollution. This paper describes research to measure the effectiveness of small wetlands (0.025–0.1% of catchment area) in the intensively farmed UK landscape. The aims of the research were 1) to adapt effective Norwegian designs (Braskerud, 2002) for the UK landscape and 2) to build and test the effectiveness of these small field wetlands in three agricultural settings, including quantification of sediment and nutrients that accumulated within the wetlands and the nutrient value of the sediment if it were returned to the fields. Ockenden et al. (2012) reported initial results related to two years of sediment accumulation at the trial wetlands. This paper extends the sediment accumulation data to three years, alongside considering total phosphorus (TP), total nitrogen (TN) and total carbon (TC) accumulation over the same period, and Olsen P concentrations within sediments collected in 2011, as a surrogate for potential agronomic value of the accumulated sediments.

2. Methods

2.1. Study sites

Ten field wetlands were built at four sites in the UK between 2008 and 2010 (Fig. 1). The wetlands were unlined ponds, excavated along runoff pathways, often in relatively unproductive field corners or in naturally wet hillslope hollows. The wetlands were one of three designs: a single shallow cell, paired shallow cells or paired deep and shallow cells. All maintained a width to length ratio of at least 1:5 as recommended by Norwegian designs, but occupied less than 0.1% of the catchment area. The sites covered a range of soil types and climatic conditions, and represented a mixture of grassland, arable and mixed farming systems.

Details of the wetlands, including construction costs are described by Ockenden et al. (2012), and only summary characteristics of the sites are repeated here (Table 1). Each site had an ARG100 tipping bucket rain gauge (Campbell Scientific Ltd., Shepshed, UK) logging to a Campbell Scientific CR800 or CR1000 data logger.

2.2. Sediment accumulation

The mass of sediment that accumulated in each wetland was estimated on an annual basis. After construction, the wetlands were surveyed using a Trimble S6 Total Station (Trimble Navigation Ltd., Sunnyvale, California, USA). In August 2010, the depth of accumulated sediment at the bed of each wetland (for the six wetlands established in 2009) was measured at nine locations in each cell and sediment samples were collected for determination of bulk density. The total mass of accumulated sediment was estimated as the product of the wetland cell area, the average depth of sediment in each cell and the average bulk density of sediment in that cell. Sediment traps were used in 2010/11 and 2011/12 in an attempt to improve accuracy when accretion rates were small. The traps, constructed of squares of artificial grass matting (0.25 m × 0.25 m) in a wire frame base, were placed manually at nine positions on the base of each cell (for all ten wetlands), when wetlands were dry or water levels were low. In August 2011 and August 2012, these sediment traps were lifted, where possible, and dried to determine the mass of sediment trapped. The rubberised matting base did not allow sediment to pass through and the artificial grass held the sediment in place to minimise loss from the sides of the mats as they were lifted. The artificial grass showed no

Table 2
Annual rainfall at each site (1 October – 30 September), and annual sediment accumulation in each wetland, 2009–2012.

Site	Name	Rainfall 2009/10 mm	Rainfall 2010/11 mm	Rainfall 2011/12 mm	Contrib. area (ha)	Method	Sediment accumulation 2009/10 t yr^{-1}	Sediment accumulation 2010/11 t yr^{-1}	Sediment accumulation 2011/12 t yr^{-1}
Loddington	LDUP ditch	680	460	790	10	Mat[a]	0.6	0.09	0.2
						Mat[a]	0.1	0.02	0.04
	LDPD drain				4	Mat[a]	0.09	0.03	0.03
						Mat[a]	0.2	0.06	0.03
	LDLO surface				9	Mat[a]	0.05	0.06	0.05
Crake Trees Manor	CTBT surface	980	1260	1350	20	Mat/survey[b]	0.2	3	10
						Mat/survey[b]	0.2	0.9	1
	CTW drain				10	Mat/survey[b]	4	0.2	0.4
	CTI stream				50	Survey	Not built	14	6
Whinton Hill	WHSD drain	690	830	1090	30	Survey	23	3	3
						Survey	3	11	−1
	WHYH ditch				20	Survey	Not built	16	2
	WHGT surface				1.5	Survey	Not built	5	0.2
						Survey		4	0.2
Newton Rigg	NRWC surface	N/A	830	1140	1	N/A	Not built	N/A	0.02

All wetlands were surveyed in 2010.
[a] Sediment mat used in 2011 and 2012.
[b] Sediment mat used in 2011, survey used in 2012.

deterioration from being underwater. New mats were placed for the following year. Where there was too much sediment to lift the sediment traps, the wetlands were re-surveyed using a Trimble S6 Total Station, to determine the depth of sediment accumulation, and sediment samples were collected to determine bulk density and nutrient concentrations. For the field wetlands which were surveyed, the mass of sediment that accumulated in each season was calculated as the total mass of accumulated sediment minus the mass that had accumulated in previous seasons.

Particle size distributions of the sediments were measured with a Malvern MasterSizer 2000 (Malvern Instruments Ltd., Malvern, Worcestershire, UK), after removal of organic matter (Gale and Hoare, 1991).

2.3. Nutrient accumulation

Nine sediment samples were collected from each cell of each wetland during the annual surveys. The sediment samples were oven dried at 105 °C (air dried for Olsen P), lightly ground and sieved to pass a 1 mm mesh. TP was determined by wet oxidation digestion (Rowland and Grimshaw, 1985) followed by colorimetric analysis (Murphy and Riley, 1962), while TN and TC were determined by elemental analysis (CEH Lancaster, 2013). Olsen P, a measure of the plant-available P, was determined by extraction with sodium bicarbonate (pH 8.5) followed by colorimetric analysis (Olsen et al., 1954). Annual nutrient accumulation in each wetland was estimated as the annual mass of sediment that accumulated in each cell of the wetland multiplied by the mean concentration of each nutrient element per kilogram of sediment in that cell. The ratios C:N:P were calculated to investigate geochemical similarities and differences between the sediments that accumulated within the individual wetlands, and to gain insights into possible sources of these sediments. The ratios were compared to the Redfield ratio (atomic ratio 106:16:1 for C:N:P), the stoichiometric ratio found in marine plankton biomass (Redfield, 1958). Wetland sediments composed solely of planktonic material produced within the wetland water column, without post-depositional changes in stoichiometry, would be supported by C:N:P = 106:16:1, the Redfield ratio. Molar data were calculated by converting the concentration of each nutrient element from mg kg^{-1} to mmol kg^{-1} by dividing by the atomic mass of that element, i.e. TC (mmol kg^{-1}) = TC (mg kg^{-1})/12; TN (mmol kg^{-1}) = TN (mg kg^{-1})/14; TP (mmol kg^{-1}) = TP (mg kg^{-1})/31.

3. Results

3.1. Sediment

The annual mass of sediment that accumulated for the years 2009/10, 2010/11 and 2011/12 is reported in Table 2, along with the annual recorded rainfall at each site for the same period. The largest mass of sediment accumulated at Whinton Hill (sandy soil), with a combined total of approximately 70 tonnes of sediment at the three wetlands at this site over 3 years (mean average accumulation rate 0.8 t ha^{-1} yr^{-1}). A smaller mass of sediment accumulated at Crake Trees Manor (silty soil) with a total of 40 tonnes of sediment over 3 years (average 0.3 t ha^{-1} yr^{-1}); the lowest mass of sediment accumulated at Loddington, with approximately 2 tonnes over 3 years (0.04 t ha^{-1} yr^{-1}). The recorded rainfall (1 October – 30 September) shows considerable inter-annual variability and differences between sites, with lower rainfall totals at Loddington than the Cumbrian sites. 2010/11 was the driest of the three years at Loddington (460 mm rainfall) though not in Cumbria, and 2011/12 was the wettest year at all sites.

3.2. Nutrients

Table 3 shows the mean annual TP, TN and TC accumulation in each field wetland, along with the range of accumulation rates (kg ha^{-1} yr^{-1}) calculated from the mass of nutrient that accumulated each year. Full details of the mass that accumulated in each year, along with the measured mean and standard deviation of the concentration of TP, TN and TC in the sediments is provided as supplementary information. The largest mass of TP accumulated at Whinton Hill (approximately 100 kg over 3 years), followed by Crake Trees Manor (30 kg over 3 years), and Loddington (3 kg over 3 years). TN and TC showed a similar pattern to TP, with the largest mass of TN and TC accumulating at Whinton Hill (300 kg TN over 3 years; 4000 kg TC over 3 years) and the lowest mass at Loddington (7 kg TN; 60 kg TC over 3 years). Accumulation rates for TN were 0.02–0.3 kg ha^{-1} yr^{-1} at Loddington and 0.5–7 kg ha^{-1} yr^{-1} at Whinton Hill. Accumulation rates for TC were approximately tenfold those for TN.

Sediment samples collected from WHYH Ditch in 2011 were taken from bands across the wetland, with samples in each band being at the same distance from the inlet. Particle size analysis of the 2011 sediments from WHYH Ditch showed a significant trend of

Table 3
Mean annual Total phosphorus (TP), Total nitrogen (TN) and Total carbon (TC) accumulation in each wetland; accumulation rate ranges for TP, TN and TC, calculated from the annual mass of nutrient that accumulated.

TP mean annual accumulation kg yr^{-1}		TP accumulation rate kg ha^{-1} yr^{-1}	TN mean annual accumulation kg yr^{-1}	TN accumulation rate kg ha^{-1} yr^{-1}	TC mean annual accumulation kg yr^{-1}	TC accumulation rate kg ha^{-1} yr^{-1}
LDUP ditch	0.57	0.03–0.1	1.6	0.1–0.3	13	1–2
LDPD drain	0.17	0.03–0.1	0.5	0.1–0.2	3.7	1–2
LDLO surface	0.08	0.006–0.01	0.3	0.02–0.04	2.7	0.1–0.4
CTBT surface	4.7	0.01–0.5	14	0.03–2	140	0.3–15
CTW drain	1.0	0.04–0.2	3.7	0.2–0.6	50	2–10
CTI stream	6.5	0.2–0.3	25	0.4–0.6	300	4–8
WHSD drain	21	0.5–2	56	0.6–7	820	9–100
WHYH ditch	18	0.3–1.5	55	0.5–5	650	10–60
WHGT surface	2.2	0.3–3	5.5	0.7–7	55	7–70
NRWC surface	0.03	0.03	0.1	0.1	1	1

decreasing particle size with location relative to the wetland inlet (Fig. 2), (for D50, Spearman rank correlation $\rho = -0.87$, $p < 0.01$).

Fig. 3a shows TP concentration and median particle size for all the sediments collected in 2011. The decrease in TP concentration with increasing median particle size was described by a power law. The highest concentrations of TP were recorded in the sediments at WHSD Drain and WHYH Ditch wetlands (up to 4000 mg kg^{-1} TP), which are connected in series along a ditch with a known wastewater input. The lowest concentrations of TP in 2011 were recorded in the sediments at WHGT Surface (220 mg kg^{-1}), a wetland which captures surface runoff from a sandy soil. The sediments from WHSD Drain and WHYH Ditch also showed the greatest variability, with the highest standard deviations and coefficients of variation (standard deviation divided by mean). Fig. 3a indicates that each site showed an increase in TP concentration with decreasing median particle size, with the highly polluted Whinton Hill site having higher concentrations of P for the same median particle size compared to other sites. Both TN (Fig. 3b) and TC (Fig. 3c) showed similar trends, with increased nutrient concentration for smaller median particle size (with power law relationships shown), but with enriched concentrations in the sediments from Whinton Hill. Fig. 3 also indicates that in wetlands made up of paired ponds, the median particle size in the first pond of the pair was generally larger than in the second pond, as expected given initial, rapid sedimentation of larger particles, and hence nutrient concentrations were generally lower in the first pond of the pair.

Molar ratios of C to N to P for sediments collected in 2011 are reported in Fig. 4. Positive, linear relationships were observed between N and P, C and N and C and P concentrations within the sediments, although sediments from WHSD appear to diverge from a linear C to N relationship at higher concentrations of C and N, in which C was enriched relative to N. Average C:N:P for the sediments collected in 2011 was 97:8:1, compared to the Redfield ratio of 106:16:1 that is also reported in Fig. 4.

The plant-available P (Olsen P) in the sediments collected in 2011 is reported in Table 4. The mean value of Olsen P ranged from 17 to 62 mg kg^{-1}. This is equivalent to a soil P index in the range 1–3 (taking into account the density of the sediment in each wetland for conversion to mg l^{-1}).

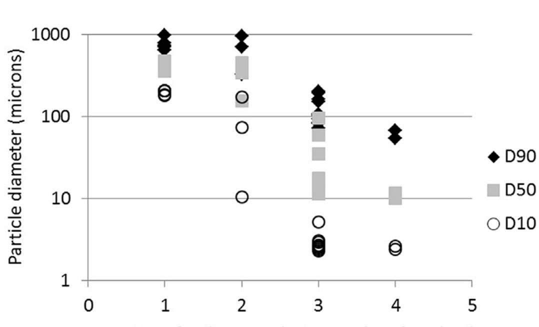

Fig. 2. Location of sediment samples collected from WHYH Ditch wetland in 2011 against particle size percentiles. Distance from wetland inlet increases with location number (0 = wetland inlet, 5 = wetland outlet; not to scale). D10 = 10% of sample with smaller diameter; D50 = median particle size; D90 = 90% of sample with smaller diameter.

4. Discussion

The sediment accumulation rates reflect differences in the erodibility of the soil across the individual wetland sites, with greater soil erosion at the sandy site (Whinton Hill) than at the clay site (Loddington). Lower annual rainfall totals at Loddington compared to the Cumbrian sites were also a contributing factor, resulting in less sediment mobilisation and therefore a smaller mass of sediment available for accumulations within the wetlands. However, the importance of the timing of the rainfall is demonstrated by the sediment accumulation rates in the third year of the reported dataset. Although the hydrological year 2012 was the wettest year across all sites, much of the rain fell from April–September 2012 when there was good crop cover in the arable fields, resulting in little sediment mobilisation or transport. At Whinton Hill, Cumbria, the mass of sediment trapped in 2009/10 (26 t) and in 2010/11 (39 t) was in both years largely due to bursting of a field drain higher up the catchment, with subsequent rill erosion in the sandy soil (Ockenden et al., 2012). The burst drain did not occur in 2011/12 in spite of the large yearly rainfall total, probably because of the timing of the rainfall when there was full crop cover (oil-seed rape in 2012), meaning that runoff was reduced to within the capacity of the field drain. In fact, continual heavy rain in autumn 2012, after the monitoring period had finished, resulted in the field drain bursting again, but this time runoff was channelled in the direction of the tramlines in the field, which bypassed WHGT Surface wetland. Sediment was deposited further down the field or entered the wetland-ditch system further downstream. This highlights the importance of locating wetlands in areas of runoff convergence.

At Crake Trees Manor, Cumbria, the large increase in sediment that accumulated in the surface runoff-fed wetland CTBT Surface between the year 2009/10 and the year 2010/11 was attributed to the change in landuse in the field draining into the wetland, from grass in 2009/10 to arable in 2010/11 (Ockenden et al., 2012). This field remained in arable in 2011/12 and the sediment that accumulated in the wetland in 2012 was largely due to a few big storms in May/June 2012 (e.g. 52 mm in 24 h on 9/10 May 2012) after which a new large mound of sediment was observed in the wetland. The importance of edge-of-field wetland features for capturing sediment is highlighted by these instances of rain occurring at critical times in the farming calendar. Even where

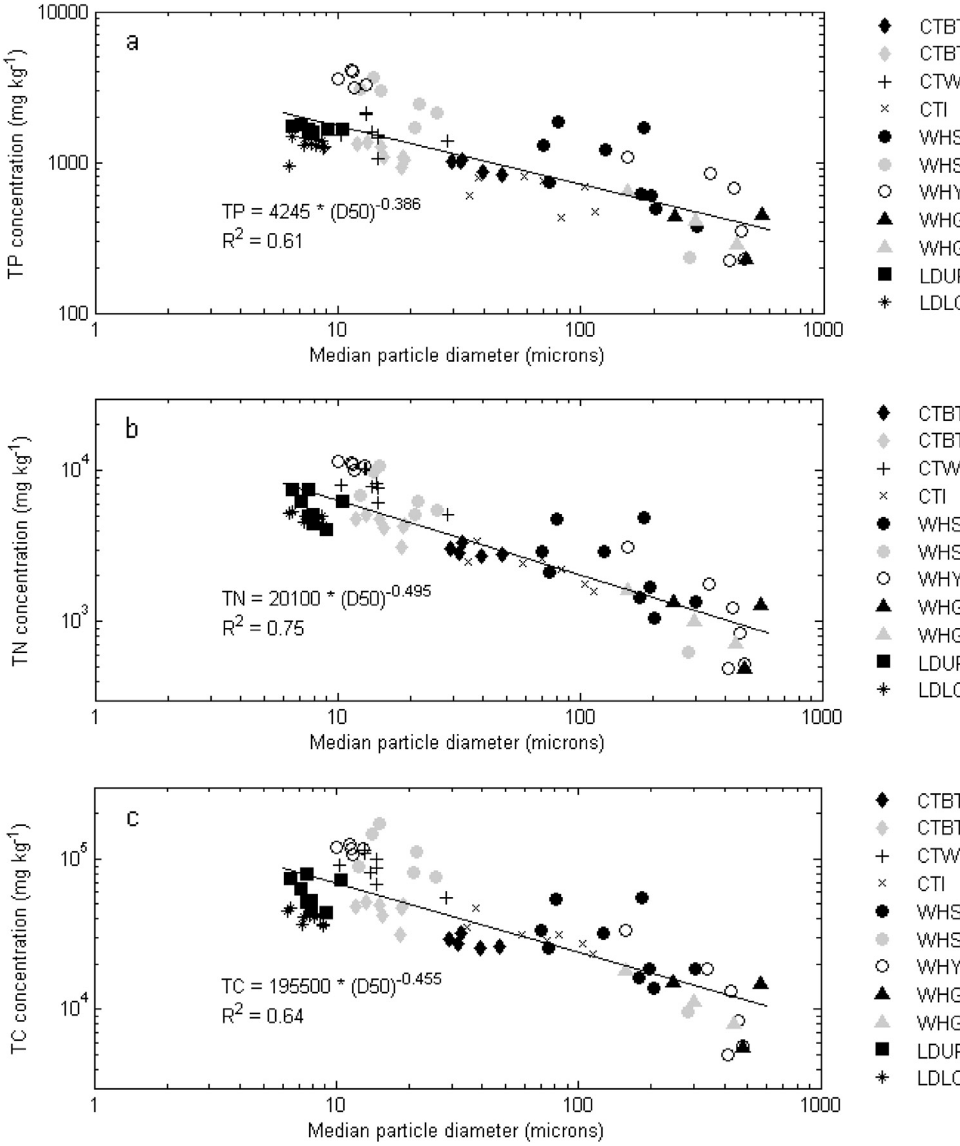

Fig. 3. Median particle diameter against concentration of a) Total phosphorus (TP) b) Total nitrogen (TN) and c) Total carbon (TC) in sediment collected from field wetlands in 2011 and a regression line (power law) based on the data from all wetlands. Key to wetlands as in Table 3; suffix 1 and 2 in wetland name relates to the first or second cell of a paired wetland.

efforts have been made to improve soil structure and reduce runoff at source, there is still a need for backup measures such as field wetlands.

The total mass of sediment that accumulated at the Whinton Hill site (69.4 t in 3 years) equated to an average accumulation rate of 0.8 t ha^{-1} yr^{-1}, compared to 0.4 t ha^{-1} yr^{-1} at Crake Trees and 0.03 t ha^{-1} yr^{-1} at Loddington. These rates are similar in magnitude to erosion rates measured by Withers et al. (2007) for plot scale studies (sediment erosion of 0.04–0.8 t ha^{-1}) and to predicted suspended sediment yields of 0.4–0.7 t ha^{-1} yr^{-1} for lowland (<200 m) wet catchments and 0.2–0.5 t ha^{-1} yr^{-1} for lowland dry catchments in the UK (Collins and Anthony, 2008). This suggests that the small field wetlands may be trapping a substantial proportion of the sediment load, particularly for wetlands located in catchments with sandy soils.

The nutrient accumulation rates in the wetlands were dominated by the sediment accumulation rates rather than by the concentrations of nutrient in the sediment. Nutrient concentrations were generally higher in sediments with a low D50 (high percentage of clay and silt), as expected with smaller particles offering a larger surface area to volume ratio, and therefore more potential binding sites for the nutrients per unit mass of sediment. However, the variation of mean nutrient concentration between sites was less than an order of magnitude (Supplementary Information), whereas the mass of sediment that accumulated varied over two orders of magnitude (Table 2). Phosphorus accumulation rates of 0.3–3 kg ha^{-1} yr^{-1} (sandy site), 0.01–0.5 kg ha^{-1} yr^{-1} (silty site) and 0.006–0.1 kg ha^{-1} yr^{-1} (clay site) were comparable with a total phosphorus accumulation rate of 0.22 kg ha^{-1} yr^{-1} reported for a small (0.3% of catchment area) Swedish wetland (Kynkaanniemi et al., 2013). A higher accumulation rate of 2.8 kg ha^{-1} yr^{-1} was reported for a larger wetland (2% of catchment area) in Sweden (Johannesson et al., 2011). The accumulation rates in our research compare favourably with total phosphorus export rates reported for UK agricultural catchments: 0.02–0.9 kg ha^{-1} yr^{-1} for catchments around Hereford (Jarvie et al., 2010), 1.2 kg ha^{-1} yr^{-1} for the Taw catchment in Devon (Wood et al., 2005) and a UK average of 0.5 kg ha^{-1} yr^{-1} (Ulen et al., 2007). These

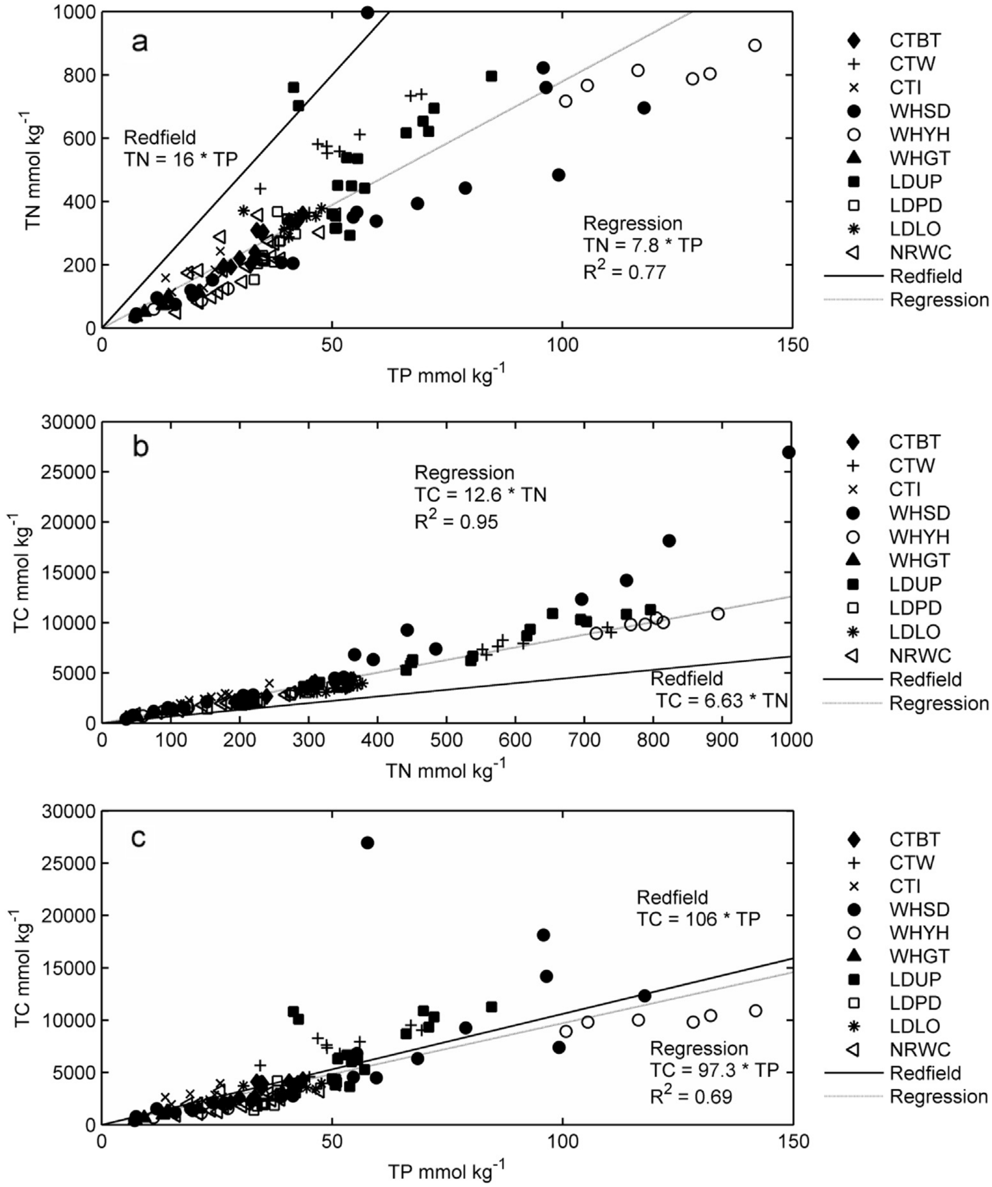

Fig. 4. Molar nutrient concentrations in 2011 sediments a) TN against TP; b) TC against TP; c) TC against TN. Also shown is the Redfield ratio (molar ratio = 106:16:1 for TC:TN:TP) and a regression line describing the observed molar ratio for each nutrient pair for the data from all wetlands, excluding WHSD Drain. Key to wetlands as in Table 3; suffix 1 and 2 in wetland name relates to the first or second cell of a paired wetland.

export rates provide useful context for the nutrient accumulation rates in the wetlands, because difficulties with monitoring equipment at the field wetlands prevented calculation of changes in nutrient loads through the wetlands.

The carbon that accumulated within the wetlands is not insignificant, with annual totals ranging from 26 kg C at the Loddington site to 2800 kg C at Whinton Hill, equating to between 0.1 and 100 kg C ha^{-1} yr^{-1}. These accumulation rates are of a similar magnitude to the 76–312 kg ha^{-1} given for C mobilisation by soil erosion based on ten years of data collected on a Sandy Loam soil under arable agriculture in Bedfordshire, UK (Quinton et al., 2006). As there are estimated to be 1,157,000 ponds hydrologically connected with an inlet in England and Wales (Williams et al., 2010), and assuming a conservative C accumulation rate of 25 kg C yr^{-1}, we estimate an accumulation rate of 2.9 Gt C yr^{-1} for English and Welsh ponds which would account for between 1 and 3.5% of the estimated sediment-associated C deposited in English and Welsh terrestrial landscapes (Quinton et al., 2006). If the assumed accumulation rate was increased towards the rate measured at Crake Trees and Whinton Hill, this would result in an order of magnitude increase in the C accumulation rate for hydrologically connected ponds across England and Wales. However, it is important to note that not all the accumulated sediment within the wetlands will necessarily be derived from the surrounding catchment, with in-wetland primary production likely to account for some of the particulate material accumulating on the bed of a wetland. In addition, the longer-term fate of sediment following accumulation within the wetland remains uncertain; some may be stored indefinitely, other material may be oxidised and lost to the atmosphere, remobilised during high flow events through the wetlands, or dredged and put back on the land. The long term fate of sediment and associated nutrients that accumulate within these type of wetland systems remain important subjects for future research.

The large variance in the nutrient concentrations observed in WHYH Ditch and WHSD Drain wetlands can be attributed to a

Table 4
Olsen P concentration in wetland sediments collected in 2011. Missing wetlands did not have sufficient sediment for analysis or were not analysed.

Site	Wetland	Cell	Olsen P (mg kg^{-1})	
			Mean	SD
Loddington	LDPD drain	1	19	(2)
Crake Trees	CTBT surface	1	17	(2)
	CTBT surface	2	25	(5)
	CTW drain		56	(17)
	CTI stream		17	(5)
Whinton Hill	WHSD drain	1	34	(14)
	WHSD drain	2	62	(21)
	WHYH ditch		51	(37)
Whinton Hill soil from fields surrounding wetlands[a]			52	(16)

[a] From Bellwood (2012).

gradient in median particle size within these wetland (Fig. 2 for WHYH Ditch), and specifically to an inverse power law relationship between nutrient concentration and particle sizes (Fig. 3). The more rapid settling of larger particles was most obvious at the Whinton Hill site, where the particle size distribution through the wetland spanned three orders of magnitude. However, the effect was also identifiable to a lesser extent at the other sites in paired wetlands, where the median particle size was generally larger (and associated nutrient concentration generally lower) in the first pond of the pair.

Molar C:N:P in sediments collected from across the ten wetlands were similar considering the extent to which catchment landuse, soil type and climate varied between the individual systems. Average C:N:P in sediments from these ten wetlands (97:8:1) was not consistent with the Redfield ratio (106:16:1) for marine plankton, being slightly enriched with P relative to C, and moderately depleted in N relative to both C and P, compared to the Redfield ratio. Therefore, molar ratios in these sediments do not support in-wetland phytoplankton production, death and accumulation as the source of particulate material on the bed of the ten wetlands. Instead, our data may indicate an external source of sediment to the wetlands. For example, the C:N:P data that we report lie between the molar ratios reported for soil microbial biomass and soil total C, N and P pools in a global synthesis by Cleveland and Liptzin (2007). Alternatively, our data may indicate post-depositional changes in the stoichiometry of the sediments, and particularly a mechanism through which N is removed from the sediments, such as ammonia volatilisation or denitrification. However, further research would be required to constrain these possible explanations.

For Whinton Hill, where the plant-available P concentration in soil within fields surrounding the wetland was determined (Bellwood, 2012), the Olsen P concentration in the wetland sediments was similar to the Olsen P of the surrounding fields, suggesting that there would be little fertiliser value in sediment dredged from the wetlands and spread back on the fields given current fertiliser application practice within the catchment. However, the return of the sediment as valuable source of topsoil remains a worthwhile activity to help to maintain soil reserves within the landscape.

When compared to typical fertiliser application rates of 30 kg phosphate ha^{-1} and 140 kg N ha^{-1} for cropped land, the nutrient accumulation rates at Crake Trees account for up to approximately 1% of the nutrients applied. Although accumulation rates at Whinton Hill were higher, these are partly due to the nutrient content of wastewater input to this wetland, which cannot be separated from the agricultural input using annual sediment surveys as reported in this paper.

5. Conclusions

Soil preservation and management is an important part of sustainable agriculture, and high environmental, economic and social costs are associated with soil degradation. Where soil erosion occurs in the agricultural landscape, small field wetlands can offer positive benefits in terms of accumulation of sediment and associated nutrients, thereby offering a potential reduction in the loads and concentrations of these pollutants that reach surface waters. Sediment accumulation rates were greatest in wetlands located in catchments with a sandy soil (average accumulation rate 0.8 t ha^{-1} yr^{-1} over 3 years) and lowest in wetlands on a clay soil (average 0.04 t ha^{-1} yr^{-1} over 3 years). Most of the sediment that accumulated in the wetlands was likely to be transported during a few large rainfall events, particularly when these coincided with times of bare soil or poor crop cover. However, careful siting of the wetlands was shown to be important in order to maximise their potential to intercept runoff and associated sediment and nutrient transport.

At each of the three main wetland sites, nutrient concentrations within the sediments were inversely related to median particle size, but nutrient concentrations across all sites varied over only one order of magnitude. In contrast, sediment accumulation rates varied over two orders of magnitude and therefore nutrient accumulation rates largely reflected sediment accumulation rates. Olsen P concentration in the wetland sediments was similar to Olsen P concentration in the fields surrounding the wetlands, suggesting that whilst the wetlands could be dredged and sediment returned to surrounding fields, which would return valuable soil to the landscape, this would provide little additional nutrient value given current fertiliser application practice in the catchments.

Overall, field wetlands represent a promising option that may contribute to mitigating the loss of sediment and associated nutrients from agricultural land. They offer a permanent backup when other, perhaps more desirable and sustainable, but often more difficult, soil management options do not prove possible to implement or have only limited effectiveness.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jenvman.2014.01.015.

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