Mitigating river sediment enrichment through the construction of roadside wetlands

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Abstract

Metalled roads have been shown to act as a major pathway for land-to-river sediment transfer, but there currently exists limited research into mitigation solutions to tackle this pollution source. The aim of this study was to assess the effectiveness of three roadside constructed wetlands, installed in September 2016, at reducing sediment enrichment in a tributary of the River Wensum, UK. Two wetland designs were trialled (linear and 'U-shaped'), both of which act as settling ponds to encourage entrained sediment to fall out of suspension and allow cleaner water to discharge into the river. Wetland efficiency was monitored through automated, high-resolution (30 min) turbidity probes installed upstream and downstream of the wetlands, providing a near-continuous record of river turbidity before (October 2011 – August 2016) and after (November 2016 – February 2018) installation. This was supplemented by lower resolution monitoring of the wetland inflows and outflows, as well as an assessment of sediment and nutrient accumulation rates within the linear wetland. Results revealed median river sediment concentrations decreased up to 14% after wetland construction and sediment load decreased by up to 82%, although this was largely driven by low river discharge post-installation. Median sediment concentrations discharging from the linear wetland (7.2 mg L^{-1}) were higher than the U-shaped wetland (3.9 mg L^{-1}), confirming that a longer flow pathway through wetlands can improve sediment retention efficiency. After 12 months of operation, the linear wetland had retained 7,253 kg (305 kg ha^{-1} y^{-1}) of sediment, 11.6 kg (0.5 kg ha^{-1} y^{-1}) of total phosphorus, 29.7 kg (1.3 kg ha^{-1} y^{-1}) of total nitrogen and 400 kg (17 kg ha^{-1} y^{-1}) of organic carbon. This translates into mitigated pollutant damage costs of £392 for sediment, £148 for phosphorus and £13 for nitrogen, thus giving a combined total mitigated damage cost of £553 y^{-1}. With the linear wetland costing £3,411 to install and £145 – 182 y^{-1} to maintain, this roadside constructed wetland has an estimated payback time 8 years, making it a cost-effective pollution mitigation measure for tackling sediment-enriched road runoff that could be widely adopted at the catchment-scale.

Keywords: Swale; sediment trap; settling pond; sustainable urban drainage; river; sediment fingerprinting.

1. Introduction

Intensification of agriculture and extensive urbanisation have resulted in widespread sediment enrichment of environmentally sensitive freshwater environments (Cordell et al., 2009; Quinton et al., 2010; Wilkinson, 2005). River systems affected by sustained high sediment concentrations experience an array of detrimental impacts which threaten sustainable ecosystem functioning. Elevated concentrations of fine clay and silt sized (<63 µm) fractions increase water turbidity, restricting light penetration to underwater plants and thereby lowering rates of photosynthesis and dissolved oxygen concentrations. Sediments smother gravel salmonid spawning grounds and benthic habitats, reduce oxygen circulation through the streambed, clog fish gills and abrasively scour macrophytes, periphyton and small invertebrates (Acornley and Sear, 1999; Bilotta and Brazier, 2008; Hilton et al., 2006).

Sediment is also a major vector for the transport of nutrients and other potentially toxic pollutants due to its high surface area providing ample opportunity for the sorption of dissolved constituents (Cooper et al., 2015b; Evans et al., 2004; House et al., 1995; Russell et al., 1998). In fact, it has been found that up to 90% of riverine total phosphorus (TP) load is transported in association with the fine grained sediment in rural catchments in the United Kingdom (Bowes et al., 2003; He et al., 1995). This means nutrient-rich sediment plays an important role in the development of eutrophic conditions, fuelling blooms of phytoplankton and neuro-toxin secreting cyanobacteria colonies, which can dramatically lower species diversity and lead to a fundamental breakdown of aquatic ecosystems (Smith et al., 1999; Withers and Jarvie, 2008). Ultimately, the degree of environmental degradation caused by elevated sediment concentrations is highly variable and known to be a function of sediment concentration, chemical composition, particle size, duration of exposure, species sensitivity and the seasonal timing of enrichment (Bilotta and Brazier, 2008; Bilotta et al., 2012).

Alongside ecological concerns there are also economic impacts to consider, with high rates of sedimentation reducing navigability, enhancing flood risk, increasing dredging requirements, increasing water treatment costs and reducing the lifetimes of dams and reservoirs (Owens et al., 2010; Posthumus et al., 2015; Pretty et al., 2003). Consequently, under national and international legislation, such as the US Clean Water Act (1972) and the EU Water Framework Directive (2000/60/EC), governments have an obligation to ensure that waterbodies achieve good ecological and chemical status. Some legislation, such as the EU Freshwater Fisheries Directive (78/659/EEC; 2006/44/EC), set a guideline standard of 25 mg L^{-1} of sediment in waters suitable for salmonid and cyprinid fish populations during normal flow conditions. Unfortunately, many fluvial systems across Europe are at risk of failing to achieve this recommended standard in water quality due to excessively high sediment ingress from the eroding terrestrial environment (European Environment Agency, 2015). Mitigation measures are therefore required to help reduce the amount of land-to-river sediment transfer if water quality is to be improved.

The River Wensum, UK, is one such river which experiences excessive sediment loading. In order to determine the provenance of this sediment, sediment fingerprinting was employed on the Blackwater Drain tributary of the River Wensum between 2012 and 2015 to derive high-temporal resolution sediment source apportionment estimates throughout the progression of 14 storm events (Cooper et al., 2015a). The results identified road verges and arable topsoil as major contributors of suspended sediment during heavy precipitation events, whilst subsurface sources (e.g. river channel banks and agricultural field drains) dominated sediment supply under baseflow conditions. Furthermore, catchment walkover surveys revealed soil from damaged road verges, field entrances and areas of concrete hardstanding is washed down metalled roads during rainfall events and into roadside ditches where it discharges directly into the river at sediment concentrations of up to 1,500 mg L^{-1} (Cooper et al., 2015a). Other studies in the UK have reported similar findings on the impact of metalled road networks (Collins et al., 2010; Collins et al., 2013).

In order to tackle the problem, in October 2016 three constructed wetlands (also known as sediment traps, swales or settling ponds) were installed near a road bridge crossing the Blackwater Drain to capture sediment-laden road runoff before it enters the river channel. Constructed wetlands are structural mitigation measures designed to intercept surface runoff by diverting the flow into a static body of water which has insufficient kinetic energy to keep the sediment in the runoff entrained (Kadlec et al., 2000; Ockenden et al., 2012). The sediment thus settles to the bottom of the wetland from where it can later be dredged out and put back on the land, whilst the cleaner, lower turbidity water can either be discharged off the surface of the wetland into a neighbouring watercourse (i.e. an open system) or simply allowed to infiltrate down into the soil (i.e. a closed system).

Constructed wetlands are generally considered to be a secondary mitigation measure to capture eroded soil after primary mitigation measures, such as cover crops (Cooper et al., 2017; Dabney et al., 2001) and reduced tillage (Deasy et al., 2009; Deasy et al., 2010; Stevens et al., 2009), have failed to retain the soil on the land. Vegetated constructed wetlands also act as biofilters as plants remove nitrogen (N) and phosphorus (P) from the water column and thereby help to mitigate eutrophication risk (Braskerud et al., 2005; Díaz et al., 2012; Fisher and Acreman, 2004), whilst they can also provide other ecosystem services such as habitat provision and flood alleviation (Verhoeven et al., 2006). There have been numerous studies on the effectiveness of 'edge-of-field' and 'after-field' constructed wetlands (Barber and Quinn, 2012; Dabney et al., 2006; Ockenden et al., 2014), with sediment removal/retention efficiencies of 30-80% (Braskerud, 2001), 54-85% (Fiener et al., 2005) and 31-96% (Díaz et al., 2012) being reported. Furthermore, a review of constructed wetlands reported average sediment, P and N retention rates in agricultural catchments of 69%, 35% and 29%, respectively (Stevens and Quinton, 2009).

However, despite this previous research, a paucity of scientific studies on 'roadside' constructed wetlands means the efficacy of these pollution mitigation measures is poorly understood, with limited evidence available to demonstrate quantitatively that these features can significantly improve downstream river water quality. The aim of this study was to assess the effectiveness of the three roadside constructed wetlands on the Blackwater Drain at reducing sediment enrichment during the first 16 months of operation. Specifically, we address the following objectives:

(i) To quantify the downstream impact of the constructed wetlands upon river turbidity and sediment loads within the Blackwater Drain;

(ii) To determine areal sediment and nutrient accumulation rates within the wetlands after 12 months of operation;

(iii) To evaluate the economic performance of the wetlands through a cost-benefit analysis to determine the feasibility of wider deployment as a catchment-based pollution mitigation measure.

2. Material and Methods

2.1 Study Location

The River Wensum is a 78 km length, lowland, calcareous river in eastern England which drains an area of 660 km^2 and has a mean annual discharge of 4.1 m^3 s^{-1} near its outlet (CEH, 2017). The Wensum is designated a Site of Special Scientific Interest (SSSI) and European Special Area of Conservation (SAC) due to the diversity of its internationally important calcareous flora and invertebrate fauna (Sear et al., 2006). However, the ecological condition of the river is in decline, with 99.4% of the protected habitat considered to be in an unfavourable or deteriorating state due, primarily, to excessive sediment and nutrient loadings from agriculture and sewage treatment works (Evans, 2012; Grieve et al., 2002; Sear et al., 2006).

This study focuses upon the 19.7 km^2 Blackwater Drain sub-catchment of the Wensum, which represents the area intensively monitored as part of the UK government-funded River Wensum Demonstration Test Catchment (DTC) research platform (**Figure 1**). The DTC is evaluating the extent to which on-farm mitigation measures can cost-effectively reduce the impact of agricultural pollution on river ecology whilst maintaining food production capacity (McGonigle et al., 2014). The Blackwater Drain at site E has a median discharge of 0.049 m^3 s^{-1}, ranging from a minimum of 0.002 m^3 s^{-1} during summer low flows to a maximum of 0.965 m^3 s^{-1} during winter storm events. The gentle (slopes < 1°) and low-lying (~40 m above sea level) topography is ideally suited to intensive arable agriculture which dominates the land use here (74%), alongside other small areas of improved grassland (14%), mixed woodland (11%) and rural settlements (1%). Surface soils are predominantly clay loam to sandy clay loam (0–0.5 m depth) developed on Quaternary deposits of chalky, flint-rich boulder clays and glaciofluvial and glaciolacustrine sands and gravels (0.5–20 m). The bedrock is Cretaceous White Chalk at a depth of ~20 m (Hiscock et al., 1996; Lewis, 2014). The site experiences a temperate maritime climate, with a mean annual temperature of 10.2 °C and a mean annual precipitation total of 674 mm (1981–2010; Met Office, 2017). During the six years of monitoring reported here, annual precipitation totals were 833 mm (2012), 588 mm (2013), 753 mm (2014), 679 mm (2015), 717 mm (2016) and 685 mm (2017). Precipitation intensities ranged from 0.8 mm h^{-1} up to 53.6 mm h^{-1} during the largest summer storm events, with a mean intensity of 1.6 mm h^{-1}.

2.2 Constructed Wetland Design

For this scheme, two roadside constructed wetland designs were trialled, both of which act as settling ponds to encourage the entrained sediment to settle out of suspension and allow cleaner water to discharge into the river (**Figure 2**). The first consists of two (CW1, CW2) larger 'U-shaped' constructions (*ca*. 50 m length, 7 m wide, 2 m depth) which increase water transit time through the wetland, dissipating kinetic energy and thus, in theory, initiating greater sedimentation rates. These U-shaped wetlands also contain two short sections (3 – 4 m length) at the entry point and U-bend that are 1 m deeper than the rest of the wetland (i.e. 3 m deep) to create pools for enhancing settling. The second design (CW3) is a smaller linear pond (*ca*. 30 m length, 4 m width, 1.5 m depth) which is shallowest at the side closest to the road and 1 m deeper (i.e. 2.5 m deep) along the opposite side to promote enhanced settling in the deeper pool. The bottom of all three wetlands intercept the water table, such that they fill with a standing body of groundwater to depths of up to 1 m in the deepest sections. The maximum water level within the wetlands is determined by the position of the outflow pipes, which in both the linear and U-shaped wetlands restricts water depths to ~1.5 m in the deepest sections.

Constructed wetlands CW1 and CW3 share the same catchment area, draining 23.75 ha of the road network and neighbouring arable fields, whilst CW2 drains an area of 3.79 ha, as determined from interrogation of a 2 m resolution digital terrain model (**Figure 1**). Collectively, the wetlands drain an area of 27.54 ha, which represents 5% of the 538 ha river catchment area draining down to monitoring site A. However, due to the positioning of the road storm drains, the vast majority of the runoff from the road is first directed into the linear CW3 wetland and only enters into CW1 if the former wetland overflows back onto the road. CW3 has therefore captured the majority of the road runoff and sediment (*c*. >70%) since installation and thus the sediment accumulation rates discussed below relate solely to CW3, whilst CW1 monitoring is omitted at present.

Vegetation within all three wetlands was allowed to establish naturally with no planting of submergent or emergent macrophytes, although the exposed soil on the upper banks of the wetlands was seeded in spring 2017 with a herbaceous wildflower mix to encourage pollinating insects.

2.3 Riverine Monitoring: High-resolution

To monitor the effectiveness of the constructed wetlands at mitigating fluvial sediment enrichment, automated, high-resolution (30 min) YSI optical turbidity probes were installed within three bankside monitoring stations located 360 m upstream (site M) and 690 m and 1300 m downstream (site A and site E, respectively) of the wetlands. This yielded a near-continuous record of river turbidity (NTU) for a period of 58 months prior to wetland installation (October 2011 – August 2016) and 16 months after installation (November 2016 – February 2018). These turbidity measurements were then calibrated against suspended particulate matter concentrations (SPM) by ordinary least squares regression using between 93 and 299 river water grab samples previously collected at each site under a range of high- and low-flow conditions between May 2012 and March 2014 (**Figure S1**) (Cooper et al., 2016).

Both the high-resolution turbidity and SPM time-series were smoothed with 49 point (24 hour), first order Savitzky-Golay filters (Savitzky and Golay, 1964) for plotting to remove spurious isolated turbidity peaks which were present throughout much of the turbidity record. This random high-frequency 'noise' in turbidity datasets has been observed in other water quality monitoring studies (Navratil et al., 2011; Sherriff et al., 2015) and is linked to the temporary biofouling of the turbidity probe and debris interference around the sensor by leaves and air bubbles.

SPM loads were calculated from estimated SPM concentrations using stage-discharge rating curves constructed from manual flow-gauging measurements made under a wide range of flow conditions ($0.002 – 0.543$ m^3 s^{-1}) at each monitoring site (**Figure S2**). Calculated percentage changes in sediment and flow dynamics for downstream sites A and E are reported after subtraction of the percentage change recorded at the upstream site M, thus accounting for the inherent background variability within the river system.

At the site E monitoring station, 30-min resolution measurements were also made of total phosphorus (Hach Lange Sigmatax SC combined with Phosphax Sigma) and nitrate-N (Hach Lange Nitratax SC optical probe) concentrations.

2.4 Wetland Monitoring: Low-resolution

After wetland construction, water samples were collected from the outflows of CW2 (n = 15) and CW3 (n = 15) at approximately weekly intervals between November 2016 and March 2017 in 1 L polypropylene bottles. These were supplemented with water samples collected from river monitoring sites M (n = 24), A (n = 21) and E (n = 21) during the same time period. In addition to this post-installation sampling, weekly-to-monthly sampling was also conducted at sites M (n = 125), A (n = 183) and E (n = 183) in the 5 years (October 2011 – August 2016) prior to wetland installation to provide background measurements. All water samples were returned to the laboratory in cool boxes and analysed within 48 hours. SPM concentrations were determined gravimetrically after filtration through pre-weighed 0.45 μm filters and oven dried at 105°C for 2 h. Total phosphorus (TP) concentrations were determined colorimetrically (molybdate) using a Skalar SAN++ continuous flow analyser with an accuracy of <9 μg L^{-1}. Nitrate (NO_3-N) concentrations were determined by ion chromatography using a Dionex ICS-2000 with an accuracy of <0.2 mg L^{-1}.

In addition to the water sampling, 500 mL sediment samples were collected at approximately monthly intervals between March and September 2017 from both the inlet and outlet of CW2 (n = 16) and CW3 (n = 16), as well as from immediately upstream (n = 8) and downstream (n = 8) of the wetlands within the river channel itself. On return to the laboratory samples were oven dried at 60°C for 24 h, lightly disaggregated with a pestle and mortar and sieved to 1.7 mm. TP and total nitrogen (TN) were then extracted from the sediments following the methods of Aspila et al. (1976) and Wheatley et al. (1989), respectively, prior to analysis of the extract with a Skalar SAN++ continuous flow analyser for TP and a Dionex ICS-2000 for TN. Organic carbon contents were determined for two sediment size fractions (<2 mm and <63 μm) via loss-on-ignition (LOI) at 450°C for 8 h, with organic carbon (OC) taken to be 58% of the LOI (Broadbent, 1953). Lastly, a 1 g aliquot of each sediment sample was analysed in a Malvern Mastersizer 2000 particle size analyser to determine the grain size distribution.

2.5 Wetland Accumulation Rates

The sediment accumulation rate for CW3 was derived in November 2017, 12 months after wetland installation. Wet sediment volume (m^3) was calculated by dividing the length of the wetland into 10 cross-sections at 3 m intervals and then dividing these into five subsections by making four equally spaced measurements across each of the 10 cross-sections (i.e. 40 measuring points in total). At each point, sediment depth was measured using a metre rule and the average depth of sediment between measuring points was used as the depth of sediment for that subsection. The sum of all subsections gave the total volume of wet sediment accumulated in the first 12 months of operation. The dry mass of sediment was then calculated by collecting 500 mL of wet sediment from the centre of each of the ten cross-sections and weighing to establish the wet sediment density. These samples were dried at 100°C for 24 h and reweighed to calculate the percentage moisture content and dry mass of sediment. This dry sediment mass was then multiplied by the mean concentrations of TP, TN and OC within the sediment to determine the mass of phosphorus, nitrogen and organic carbon retained.

2.6 Sediment Fingerprinting

To assess whether installing constructed wetlands had reduced the contribution of road runoff-derived material to overall fluvial sediment load, the sediment fingerprinting procedure described in Cooper et al. (2015a) was rerun in 2017. To summarise, three potential sediment source areas were identified across the 5.4 km^2 section of the Blackwater sub-catchment draining down to monitoring site A below the wetlands. These were eroding arable topsoil, damaged road verges and a combined river channel bank and agricultural field drain 'subsurface' source. From each source area, 10 soil/sediment samples were collected, wet sieved to <63 µm to extract the fine clay-silt fraction and transferred onto quartz fibre filter papers. For the target riverine sediment, an automatic ISCO water sampler (Teledyne ISCO, Lincoln, NE) located at the site A monitoring station was programmed to collect a 1 L river water sample every 60–90 min for 24–36 h during four heavy precipitation events (>10 mm rainfall) between December 2016 and May 2017. The samples were then vacuum filtered onto quartz fibre filter papers to extract the SPM. Both source and target filter papers were then analysed by X-ray fluorescence spectroscopy (XRFS) to determine the geochemistry (wt. %) following the method of Cooper et al. (2014b). In total, concentrations of eight major elements (Al, Ca, Ce, Fe, K, Mg, Na, Ti) were determined and selected as fingerprints for use in the mixing model. Prior to running the model, the geometry of the source geochemistry mixing space was examined via a principal component analysis to ensure efficient differentiation. The sediment fingerprinting mixing model used was the empirical Bayes version presented in Cooper et al. (2014a). The model is solved as a mass balance, whereby the concentration of each fingerprint in the target riverine sediment (Y) is obtained from the concentration of each fingerprint in each potential sediment source area (S) multiplied by the proportional sediment contribution (P) derived from that source. This can be summarised by the following likelihood function:

(1) $\mathrm{L}(S, P \mid Y)$

2.7 Economic Damage Costs

To provide an economic basis for implementing sediment and nutrient pollution mitigation measures across river catchments (e.g. Pretty et al., 2000; Pretty et al., 2003), an economic estimation of pollution damage costs was calculated for wetland CW3. The total dry masses of sediment, TP and TN captured in CW3 during the first 12 months of operation were translated into economic damage costs by multiplying by the 2014 pollutant prices set by the UK government (DEFRA). These pollutant prices account for remediating the ecological impacts of the pollutants (e.g. tackling eutrophication from N and P), making water drinkable (e.g. cost for water companies to remove N) and the cost of keeping rivers navigable (e.g. dredging costs to remove excess sediment). The pollutant prices used were £0.054 kg^{-1} (range = £0.047 – 0.061 kg^{-1}) for sediment, £12.79 kg^{-1} (range = £2.77 – 22.66 kg^{-1}) for TP and £0.43 kg^{-1} (range = £0.24 – 0.62 kg^{-1}) for TN, as per the DTC project (McGonigle et al., 2014).

3. Results and Discussion

3.1 Riverine Impacts

Riverine SPM concentrations recorded at site M (upstream) and sites A and E (downstream) displayed considerable variability over the six year monitoring period, with concentrations ranging from <1 mg L^{-1} up to 771 mg L^{-1} across all sites, whilst turbidity ranged from 0.9 to 451 NTU (**Figure 3**). However, the monitoring results reveal a complex picture of wetland performance due largely to the dry conditions experienced post-installation during winter (74% of average rainfall) and spring (89% of average rainfall) 2017 when the river almost dried up at sites M and A (discharge = <1 L s^{-1}).

Median SPM concentrations at site M were significantly ($p < 0.01$) higher after wetland installation (12.4 mg L^{-1}) than before (9.8 mg L^{-1}), with this 26.2% increase thought to be driven by the very low flow conditions during spring/summer 2017 which concentrated the particulate material being transported (**Table 1**). Consequently, significantly ($p < 0.01$) higher SPM concentrations were also recorded downstream at sites A (pre = 12.1 mg L^{-1}; post = 13.6 mg L^{-1}) and E (pre = 6.4 mg L^{-1}; post = 7.9 mg L^{-1}) post-wetland installation, which would initially suggest poor sediment mitigation performance of the wetlands. However, after correcting for this 'background' increase recorded at site M, concentrations actually significantly ($p < 0.05$) decreased by 13.9% and 4.1% at sites A and E, respectively, after the wetlands were constructed. Even larger decreases in SPM load of 81.5% and 78.4% ($p < 0.05$) were observed post-installation at sites A and E, respectively, although this was largely driven by 55.9% and 51.3% declines in river discharge during the November 2016 – February 2018 period. Overall, sediment concentrations exceeded the 25 mg L^{-1} guideline value 11% and 9% of the time at sites A and E, respectively, after wetland installation, compared to 9% and 5% previously, thus there was no improvement in water quality with regard to meeting WFD directive targets during the first 16 months of operation.

In contrast to previous studies (e.g. Fisher and Acreman, 2004), median TP concentrations downstream at site E also changed very little following the installation of the wetland (-1.5%), although TP loads were reduced by 50% due to the lower flow conditions. Conversely, median NO_3-N concentrations actually increased significantly ($p < 0.05$) by 14.5% downstream of the wetlands, although without the benefit of nitrate monitoring upstream of the wetlands it is difficult to determine whether this increase was due to the impact of nutrient release from the wetland or elevated N inputs from elsewhere in the catchment.

3.2 Sediment Source Apportionment

Sediment fingerprinting conducted after wetland installation revealed an overall decrease in sediment contributions from road verges in the Blackwater Drain downstream of the wetlands, thus confirming these mitigation features were successfully capturing and retaining road runoff material (**Table 2**). During the 14 storm events monitored prior to wetland installation (2012 – 2015), mean sediment contributions were 25.7% from road verges, 49.1% from subsurface areas and 23.2% from arable topsoil. During the four storm events monitored post-installation, mean road verge contributions reduced to 9.6%, with a further 53.3% from subsurface areas and 24.3% from topsoil. This represents a 16.1% reduction in road verge material entering the river since the wetlands were constructed, albeit within a wide range of uncertainty (95% credible interval = 0.0 – 60.4%). Such wide uncertainty is typical of sediment fingerprinting studies using this type of Bayesian end-member mixing models (Cooper and Krueger, 2017).

3.3 Wetland Pollutant Discharge

Sediment and nutrient concentrations discharging from the wetlands are shown in **Figure 4**, alongside the low-resolution grab sampling results for the three river sites. Median SPM discharge concentrations were higher from the linear CW3 (7.2 mg L^{-1}) than the U-shaped CW2 (3.9 mg L^{-1}) wetland, supporting the hypothesis that the longer flow path of the U-shaped design increases sediment settling rates, although this difference was not significant ($p = 0.269$). Sediment concentrations discharging from the linear wetland were also greater than the median concentrations observed instream at sites M (2.2 mg L^{-1}) and A (4.7 mg L^{-1}), indicating that CW3 was acting to increase sediment concentrations within the river, albeit below the EU WFD standard.

With respect to nutrients, median TP concentrations were significantly ($p < 0.01$) higher in the linear wetland discharge (91 µ L^{-1}) than the U-shaped wetland (19 µ L^{-1}) and were 2-4 times higher than the TP concentrations observed in the river (22–52 µ L^{-1}). This indicates that CW3 was acting as a net source of TP into the Blackwater Drain, supporting the findings of previous studies which have also reported increases in P export from wetlands due to the decomposition of biological material within the wetland itself (Díaz et al., 2012; Johannesson et al., 2011). This is a particular problem where vegetation management is not conducted and where algal blooms can occur readily, as was the case with CW3, leading to an accumulation of organic matter and nutrients within the wetland. Additionally, P bound to the sediment deposited within the wetland can dissolve into the overlying water column and be discharged into the river channel rather than being captured and retained.

On the other hand, median nitrate concentrations were lower in the wetland discharges (2.0 – 3.1 mg N L^{-1}) than in the neighbouring river (6.0 – 7.2 mg N L^{-1}), thus confirming that the wetlands were not acting as a source of N enrichment and emphasising that most nitrate input into the catchment is via fertiliser leaching/runoff from arable fields rather than from the road network. Denitrification could also be occurring within the wetlands to reduce nitrate concentrations, principally where anoxic conditions develop within the deposited sediment.

3.4 Wetland Pollutant Retention

After the first 12 months of operation (November 2016 – November 2017), wetland CW3 had retained 7,253 kg of sediment, 11.6 kg of TP, 29.7 kg of TN and 400 kg of organic carbon **(Table 3)**. For a catchment area of 23.75 ha, this equates to retention rates of 305 kg ha^{-1} y^{-1} for sediment, 0.5 kg ha^{-1} y^{-1} for TP, 1.3 kg ha^{-1} y^{-1} for TN and 17 kg ha^{-1} y^{-1} for organic carbon. This compares with accumulation rates of 40–800 kg ha^{-1} y^{-1} for sediment, 0.006 – 3 kg ha^{-1} y^{-1} for TP, 0.02 – 7 kg ha^{-1} y^{-1} for TN and 0.1

– 100 kg ha^{-1} y^{-1} for total carbon, reported previously for edge-of-field wetlands in the UK (Ockenden et al., 2012; Ockenden et al., 2014).

The mean particle size of the retained sediment decreased across the length of the wetlands, with coarser sand and silt being deposit at the wetland inlets (CW2 = 292 μm; CW3 = 670 μm) and finer silt and clay near the outlets (CW2 = 196 μm; CW3 = 315 μm) (**Figure 5**). This demonstrates that larger particulates readily dropped out of suspension upon entry into the wetland. The finer particle size at the outlet of CW2 relative to CW3 could potentially be explained by the longer flow path of the U-shaped wetland allowing increased time for sediment settling. However, the particle size at the wetland inflow was also substantially lower in CW2 and this is likely to have been the dominant influence on outlet particle size here, with visual observations indicating that a greater volume of coarser sandy material was moving northwards down the road network and entering CW3. For both CW2 and CW3, the mean particle size near the wetland outlet was smaller than the mean particle size in the river just downstream (447 μm) and thus both would be a local net source of fine sediment at the outlet location should this material be entrained out of the wetlands and into the river during storm event flushing.

Significant ($p < 0.01$) non-linear negative correlations were found between the mean particle size and both the TP concentration (r = -0.706) and organic carbon content (r = -0.695) of the river and retained wetland sediments. This association, which has also been reported elsewhere (Ockenden et al., 2014), indicates that the finer silt and clay deposited near the wetland outlets is more nutrient rich due to the sorption of P onto metal oxyhydroxides (Cooper et al., 2015b) and thus this sediment has increased risk of generating eutrophic conditions. The higher organic carbon content means this finer material also carries greater risk of causing enhanced microbial decomposition leading to elevated biological oxygen demand and the development of anoxic conditions within the wetland.

3.5 Wetland Maintenance

The flushing of stored sediment from the wetlands into the river channel during heavy precipitation events will ultimately limit their efficacy as a pollution mitigation feature (Barber and Quinn, 2012). To overcome this, retained sediment will need to be periodically dredged out of the wetlands and redistributed on the neighbouring arable land. The frequency at which this maintenance needs to be carried out will depend upon the rate of sediment accumulation, which in part will be dependent upon wetland size, with larger features requiring less frequent dredging. The Broads Authority have estimated average dredging costs of £12-15 m^{-3} for watercourses in eastern England (Environment Agency, 2015) and thus whilst the smaller, linear CW3 wetland had lower design and construction costs than the two larger U-shaped ponds, this will in-part be offset by higher maintenance costs incurred from more frequent dredging. To date, no dredging has been carried out in any of the three wetlands, but it is envisaged that CW3, which has the largest catchment area and smallest wetland volume, will require sediment removal within 2–3 years of operation based on current accumulation rates of 12.15 m^3 y^{-1}.

Dredging this sediment would thus incur maintenance costs of approximately £145 – 182 y^{-1}. Additionally, it is expected that the performance of sediment and nutrient retention will be further improved once vegetation establishes itself within the wetlands. The vegetation will absorb nutrients and act to stabilise the currently exposed banks of the wetlands, thus reducing the risk of erosion and will also increase resistance to water flow, thus reducing kinetic energy and promoting increased sedimentation (Braskerud, 2001).

3.6 Economic Performance

Using the UK government's 2014 pollutant prices, the damage costs mitigated by pollutant retention within CW3 during the first 12 months of operation were £392 (range = £340 – 442) for sediment, £148 (range = £32 – 263) for TP and £13 (range = £7 – 18) for TN (**Table 3**). This gives a combined total mitigated damage cost for CW3 of £553 (range = £380 – 724) per year. With CW3 costing £3,411 to install (£1,400 for design; £2,011 for construction) and having annual maintenance costs of £145 – 182, this mitigated damage cost means an estimated payback time of 5 – 17 years, with a best estimate of 8 years. This makes the linear wetland an affordable and cost-effective pollution mitigation measure for sediment and nutrients running off metalled roads.

The other two U-shaped wetlands had higher design (£2,800 per wetland) and construction (£4,034 per wetland) costs due to the more complex engineering and larger excavation, potentially making them a less affordable option for wider catchment-scale deployment. However, calculation of pollutant retention in CW1 and CW2 would need to be conducted before it is possible to make an assessment of the cost-effectiveness of these U-shaped wetlands and their potential scalability across catchments. As a guide, Ockenden et al. (2012) reported general construction costs of £280 – £3,100 for wetlands with areas of between 5 and 320 m^2.

4. Conclusions

This study provides the first quantitative evidence of the effectiveness of constructed wetlands at mitigating fluvial sediment enrichment from road runoff in the UK. The results presented here demonstrate that diverting surface runoff from metalled roads into roadside wetlands can prevent large volumes of sediment, nutrients and organic matter from entering the river network and thus can minimise many of the detrimental impacts of water pollution which threaten sustainable ecosystem functioning. With dense road networks covering many developed countries, the problem of sediment-laden road runoff discharging into ditch, stream and river channels is a widespread issue that will require a catchment-based approach. The retention performance and relative simplicity of the linear wetland trialled here has demonstrated that it can provide a relatively cost-effective solution to mitigate road runoff pollution if deployed widely at many of the main road-river crossing throughout a river catchment. Further research is clearly required to determine whether the U-shaped wetlands offer similar potential and to assess how both types of wetland perform as aquatic vegetation and microbial communities establish themselves over the next 4 – 5 years. However, these early results offer a promising solution to tackling surface runoff pollution from roads, particularly in agricultural areas.

Acknowledgements

The River Wensum Demonstration Test Catchment project is funded by the Department for Environment, Food and Rural Affairs (Defra) under projects WQ0212 and LM0304. The constructed wetlands were designed by Matt Philpott of the Norfolk Rivers Internal Drainage Board (IDB) and constructed by Aylsham Plant Hire. Capital funding came from an EC LIFE+ WaterLIFE grant with funding from Coca-Cola. The project was facilitated by Neil Punchard of the Broadland Catchment Partnership and Jonah Tosney of Norfolk Rivers Trust. We are grateful to Salle Farms Co. for hosting monitoring equipment and granting site access. Gillia Sünnenberg is thanked for assistance with GIS mapping. We thank the three anonymous reviewers for their constructive comments which improved an earlier version of this manuscript.

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Tables

Table 1: Summary results from the high-resolution (30 min) water quality monitoring at sites M, A and E for the period February 2013 to August 2016 (pre-wetland installation) and November 2016 to February 2018 (post-wetland installation). Values presented as medians with one standard deviation in parentheses. Percentage change in discharge, turbidity and SPM for downstream sites A and E is reported after subtraction of the percentage change at upstream site M.

Site	Installation Stage	Discharge (L s^{-1})	Turbidity (NTU)	SPM (mg L^{-1})	SPM Load (kg h^{-1})	TP (µg L^{-1})	TP Load (g h^{-1})	Nitrate (mg N L^{-1})	Nitrate Load (kg N h^{-1})
M	Pre	3.23 (26.28)	7.2 (15.4)	9.8 (18.9)	0.11 (3.64)	-		-	
	Post	3.64 (17.32)	9.3 (16.4)	12.4 (20.1)	0.19 (5.98)	-		-	
	Change (%)	**+12.7**	**+29.2**	**+26.2**	**+72.7**	-		-	
A	Pre	11.89 (38.27)	7.1 (13.9)	12.1 (18.9)	0.50 (7.10)	-		-	
	Post	6.75 (35.84)	8.2 (11.7)	13.6 (15.8)	0.43 (7.60)	-		-	
	Change (%) after M	**-55.9**	**-13.7**	**-13.9**	**-81.5**	-		-	
E	Pre	56.80 (61.84)	3.6 (11.5)	6.4 (20.6)	1.04 (20.96)	67 (47)	14 (41)	5.5 (2.2)	1.1 (2.4)
	Post	34.90 (78.27)	4.4 (9.4)	7.9 (16.9)	0.92 (25.20)	66 (28)	7 (21)	6.3 (3.4)	0.8 (3.7)
	Change (%) after M	**-51.3**	**-6.9**	**-4.1**	**-78.4**	**-1.5**	**-50.0**	**+14.5**	**-27.3**

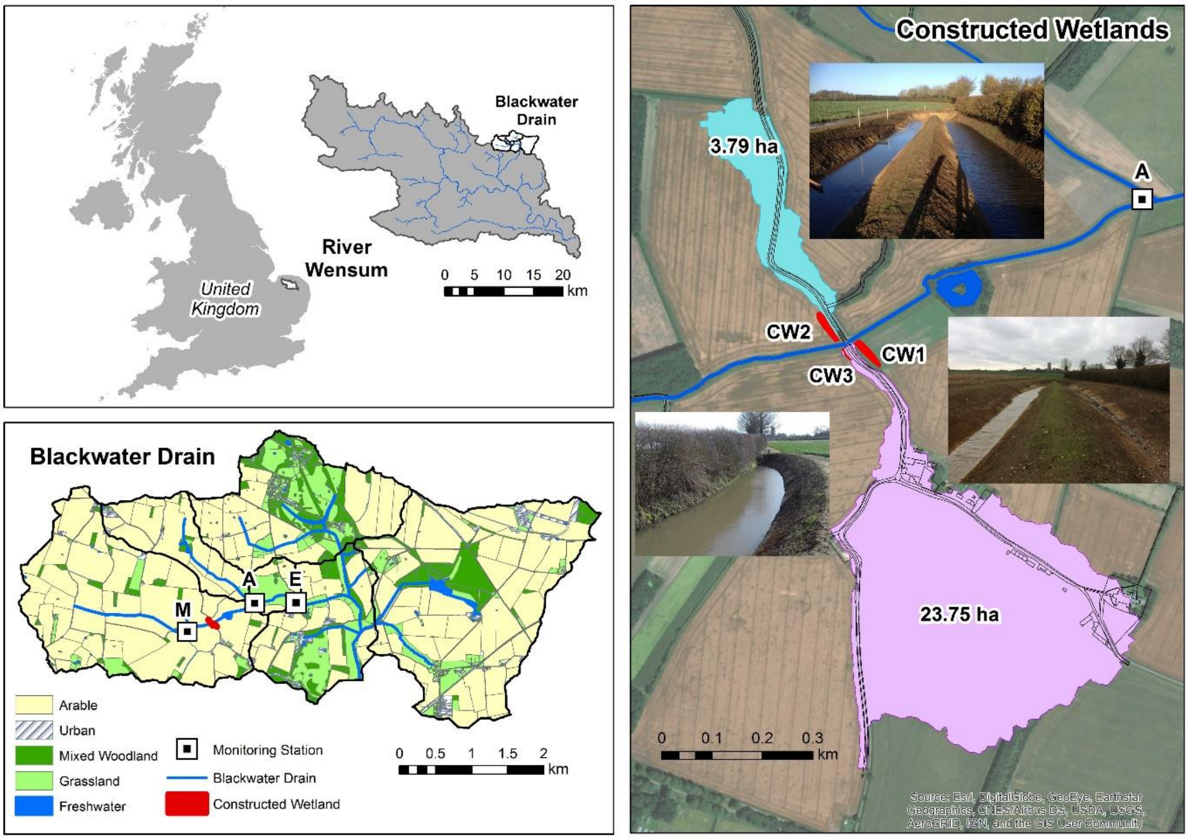
Table 2: Sediment source contributions apportioned by sediment fingerprinting downstream of the constructed wetlands at site A during storm events before and after wetland installation. Values presented as the mean 50th percentile and the mean 95% credible intervals in parentheses. Note: total of all 50th percentile source contributions will not necessarily sum to 100% due to skewed posterior distributions.

	***n* storm events**	***n* samples**	**Source contribution (%)**		
			Subsurface	**Road verge**	**Topsoil**
Pre-installation	14	254	49.1 (30.0 –68.9)	25.7 (9.1 – 50.1)	23.2 (5.9 – 47.4)
Post-installation	4	66	53.3 (34.9 – 80.5)	9.6 (0.0 – 60.4)	24.3 (0.0 – 59.8)

Table 3: Wetland CW3 retention rates and economic damage costs for the first 12 months of operation (November 2016 – November 2017). Values in parentheses represent the 'low' and 'high' pollutant prices assigned by the UK government.

Parameter	**Retention (kg)**	**Retention rate (kg ha^{-1} y^{-1})**	**Pollutant price (£ kg^{-1})**	**Mitigated damage cost (£)**
Sediment	7,253	305	0.054 (0.047 – 0.061)	391.66 (340.89 – 442.43)
Total phosphorus	11.6	0.5	12.79 (2.77 – 22.66)	148.36 (32.13 – 262.86)
Total nitrogen	29.7	1.3	0.43 (0.24 – 0.62)	12.77 (7.13 – 18.41)
Organic carbon	400	17	-	-
			Total mitigated damage cost	552.79 (380.15 – 723.73)
			Cost of wetland CW3	3,411
			Annual maintenance cost	145 – 182
			Payback time	5 – 17 years

591 **Figures**



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593 **Figure 1:** Location of the roadside constructed wetlands and their catchment areas within the
594 Blackwater Drain sub-catchment of the River Wensum, Norfolk, UK.

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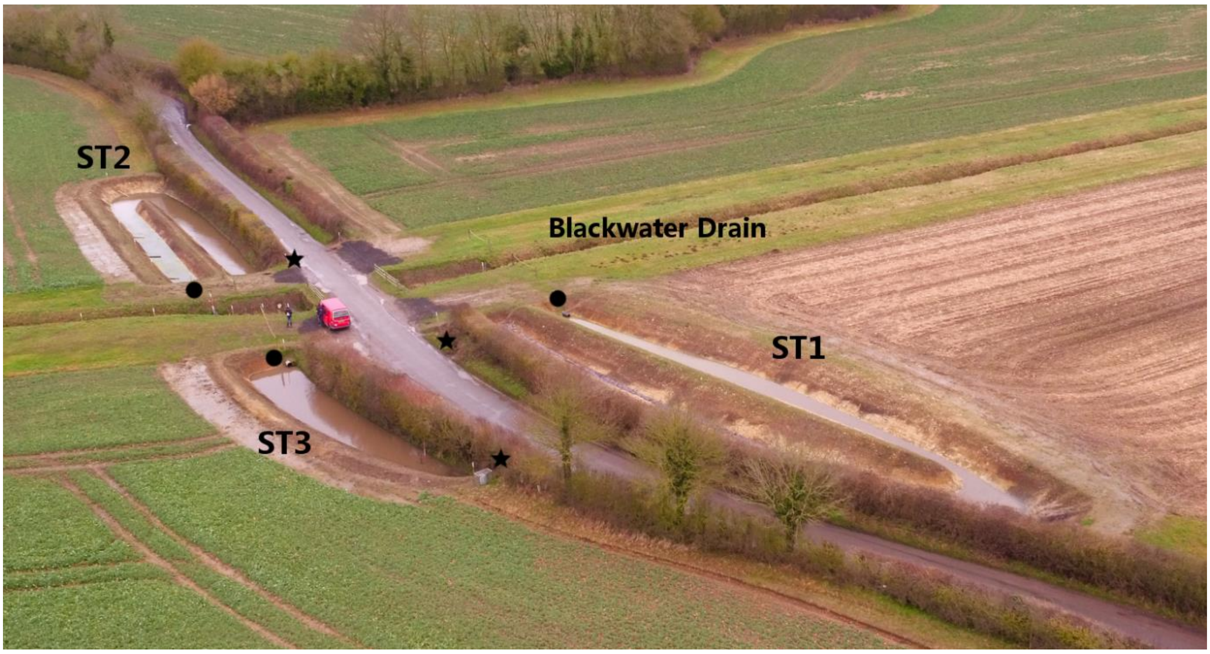

Figure 2: Aerial-drone photograph looking north-east of the three roadside constructed wetlands captured in February 2017. Black stars and circles denote the inlet and outlet pipes for the wetlands, respectively. For location see Figure 1.

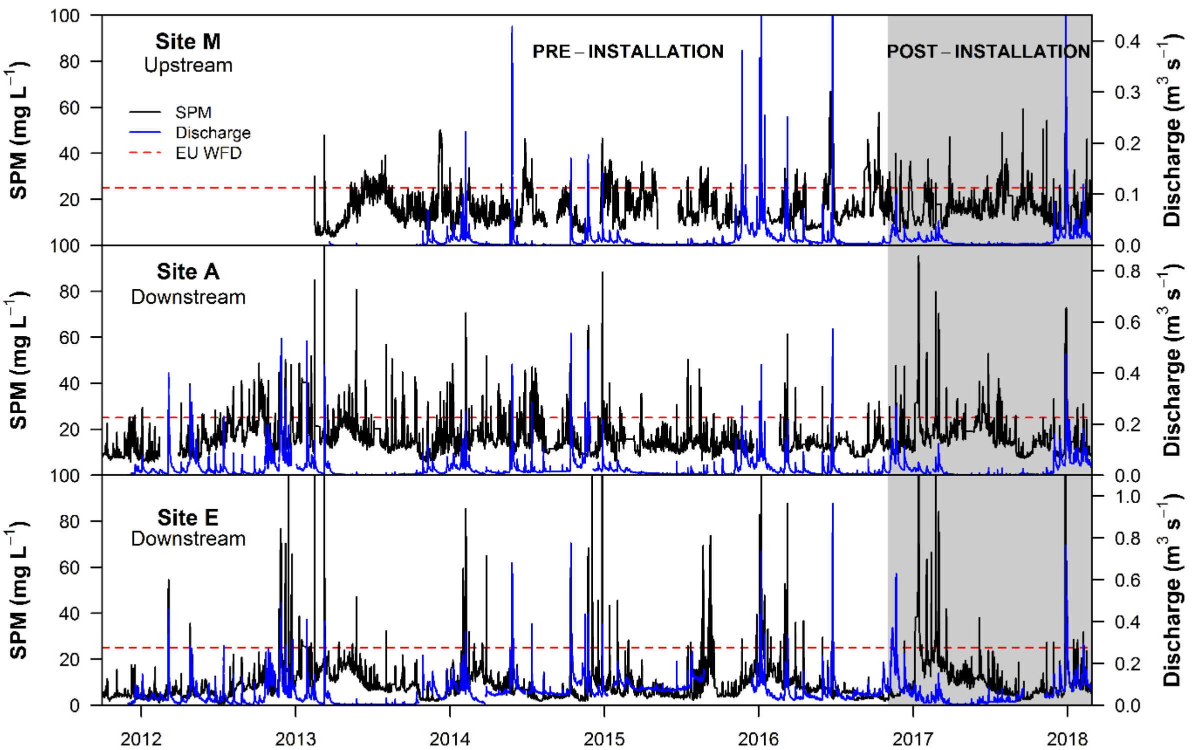

Figure 3: Suspended particulate matter (SPM) concentrations and river discharge recorded at 30-min resolution at monitoring sites M, A and E between October 2011 and August 2016 (pre-wetland installation) and November 2016 and February 2018 (post-wetland installation).

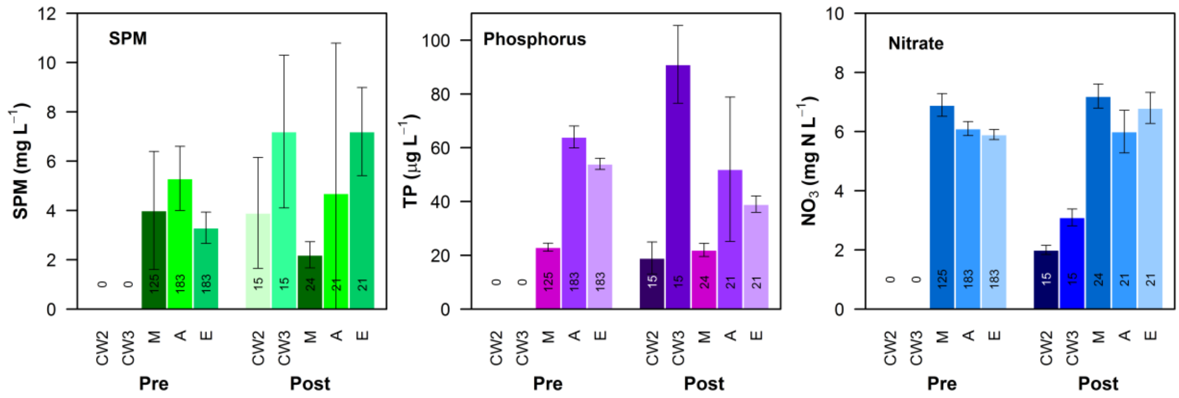

Figure 4: Median sediment, phosphorus and nitrate concentrations recorded at the wetland outflows and within the Blackwater Drain before (October 2011 – August 2016) and after (November 2016 – March 2018) wetland installation. Error bars represent one standard error; figures on bars represent the number of samples.

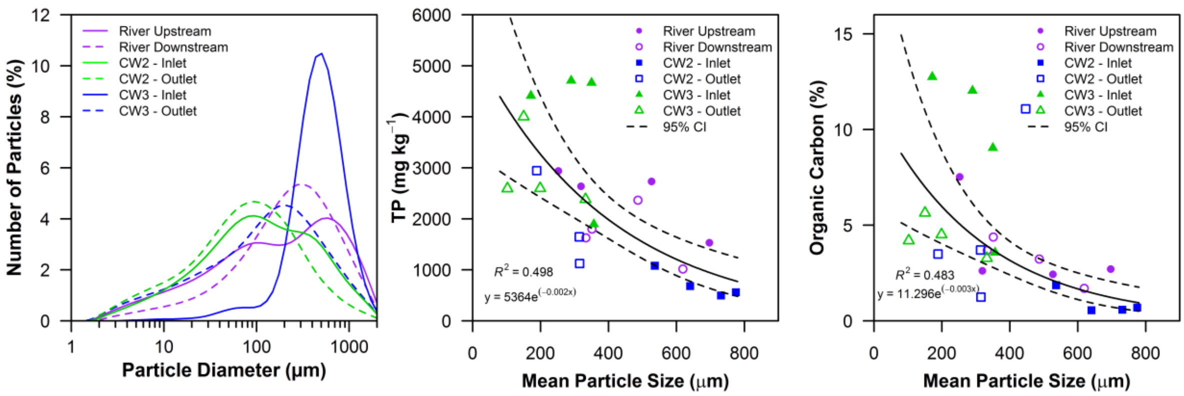

Figure 5: (*left*) average particle size distribution of sediment collected monthly between June and September 2017 from the wetland inlets, wetland outlets and the river upstream and downstream of the wetlands; (*centre*) relationship between sediment particle size and sediment TP concentration; (*right*) relationship between sediment particle size and sediment organic carbon content.