Characterising the geomorphological and physicochemical effects of water injection dredging on estuarine systems.

Andrew Pledger[1,2], Matthew Johnson[3], Philip Brewin[4], John Phillips[5], Sarah L. Martin[1], Dapeng yu[1].

[1] Department of Geography, Loughborough University, Loughborough, Leicestershire, LE11 3TU, UK.

[2] AP Environmental Solutions, 15 Hillside, Sawston, Cambridge, CB22 3BL.

[3] School of Geography, University of Nottingham, Nottingham, NG7 2RD, UK.

[4] Somerset Drainage Boards Consortium, Bradbury House, 33-34 Market Street, Highbridge, Somerset, TA9 3BW, UK.

[5] Environment Agency, Rivers House, East Quay, Bridgwater, Somerset, TA6 4YS, UK.

Corresponding author: Dr Andrew Pledger; gyagp@lboro.ac.uk

[1] Department of Geography, Loughborough University, Loughborough, Leicestershire, LE11 3TU, UK.

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Abstract

Dredging is a globally important aquatic system management activity, used for navigation improvement, contamination removal, aggregate production and/or flood risk mitigation. Despite widespread application, understanding of the environmental effects of some dredging types remains limited. Field campaigns in 2016 and 2017 in the River Parrett estuary, UK, therefore investigated the geomorphic and physicochemical effects of Water Injection Dredging (WID), a poorly studied hydrodynamic dredging technology. WID, applied to restore channel capacity for the maintenance of flood water conveyance in the tidal River Parrett, influenced surface elevations but not grain-size characteristics of dredged bed sediments. Topographic alterations due to the 2016 WID operation were short-lived, lasting less than 10 months, although benefits of the 2017 WID operation, in terms of volumetric change, outlasted the ≈ 12-month study period. Dredging had a significant impact on water physicochemistry (pH, dissolved oxygen, total suspended solids and turbidity) when comparing pre- and during-dredging conditions within the dredge reach, although time-series analysis found dredging effects were comparable in magnitude to tidal effects for some parameters. WID is typically targeted at the thalweg and not the banks, rendering the geomorphic signature of the method different to those of other, often more invasive dredging technologies (e.g. mechanical dredging methods). Further, thalweg not bankside dredging may have potential positive ecological implications, particularly where the majority of biomass is located within the channel margins, as in the tidal River Parrett. Collectively, data suggest WID can be an effective method for sediment dispersal within tidal systems although regular application may be required to maintain cross sectional areas, particularly where management precedes periods of low flows and/or high rates of sediment accumulation. In future, more work is required to better understand both the physical and ecological implications of WID as a flood risk management tool in estuaries and rivers.

1.0 Introduction

Fine sediment deposition in aquatic systems can have significant detrimental impacts on the physical environment, ecosystem health and human life. Subaqueous mining or dredging (dredging hereafter) is an engineering practice used to maintain channel capacity, improve navigation potential (van Maren *et al.*, 2015; Wenger et al., 2017, Wu et al., 2018), remove contaminants (Bormans et al., 2016; Gustavson et al., 2008; Chen et al., 2018) and/or reduce flood risk (Gob et al., 2005) across the world.

Numerous dredging technologies exist, the main categories being mechanical, hydraulic and hydrodynamic types, which are ubiquitously applied globally (e.g. China, Luo et al. 2007; Jing *et al.*, 2013; Wu et al., 2016, 2018; Egypt, Ismail and Samuel, 2011; France, Petit et al., 1996; Bravard et al., 1997; Gaillot and Piégay, 1999; Liébault and Piégay, 2001; Germany, Reich, 1994; Italy, Rinaldi and Simon, 1998; Surian, 1999; Rinaldi, 2003; Surian and Rinaldi, 2003; Poland, Wyzga, 1993; Lach and Wyzga, 2002; United Kingdom, Sear and Archer, 1998; United States of America, Kondolf, 1994, Pinter et al., 2004). Mechanical methods typically involve the removal of sediments using heavy construction equipment, including bucket excavators. Mechanical technologies can be either vessel- or land-based with the latter often associated with bankside habitat degradation (Brooker, 1985). Hydraulic technologies utilise centrifugal pumps and pipelines to raise fine-grained sediments from the bed (Vivian et al., 2012). The financial cost of extracted sediment or “spoil” relocation or disposal can be significant (Inman, 1976), rendering both mechanical and hydraulic methods infeasible in some places. Hydrodynamic methods involve sediment displacement through water jetting and unlike mechanical and hydraulic methods, the emphasis is on the downflow displacement, not extraction of sediments, with potential efficiency and cost benefits (Wang et al., 2012).

The environmental effects of dredging will vary between sites and as functions of operation extent and persistence (Kondolf, 1994). Further, different dredging technologies utilise different extraction/disturbance mechanisms and will therefore elicit different environmental impacts (Wang et al., 2012), although knowledge of the physical effects of some types and technologies remains rudimentary. Removal of instream sediments through mechanical dredging can have profound impacts on channel morphology, potentially leading to bed degradation (Lagasse and Winkley, 1980; Pinter et al., 2004), water level reductions (Kornis and Laczay, 1988; Ellery and McCarthy, 1998), channel instability, removal of gravel armouring and/or increased channel instability and erosion (Lagasse, 1975), via knickpoint migration (Scott, 1973; Stevens et al., 1990) or incision downstream of the excavated pit. Alternatively, hydraulic technologies are typically applied where high production rates are required (Wenger et al., 2017), and their impacts can be significant, particularly where dredging is spatially and temporally persistent (e.g. Luo et al., 2007). Some hydraulic and mechanical methods may either directly (e.g. via turbation in the direct vicinity of the dredging vessel; Collins, 1995; Mikkelsen and Pejrup, 2000; Smith and Fredrichs, 2011) or indirectly (e.g. through channel deepening and so, tidal amplification; van Maren et al., 2015) cause sediment resuspension. Activities may also release contaminants (Goosens and Zwolsman, 1996, Van Maren et al. 2015), particularly particulate matter and pore water from the bed which may be rich in trace metals (Van Den Berg *et al*., 2001). Reduced sediment flux through aggregate extraction can have important implications for sediment budgets (Collins and Dunne, 1989; Kondolf and Swanson, 1993) and so, erosion rates (Simeoni and Corbau, 2009).

In contrast to the vast literature on mechanical and hydraulic dredging, there is currently little published information on the use of hydrodynamic dredging methods as techniques for subaqueous sediment management. Gravel jetting has been shown to influence surface but not sub-surface size distributions, resulting in coarser and better-sorted surficial sediments with potential benefits for shallow-, not deep-spawning fishes (Basic et al., 2017). Water injection dredging (WID hereafter) is an emerging and relatively novel (Spencer et al., 2006) hydrodynamic dredging technology first developed in the Netherlands in the mid-1980s and brought about by rising costs of spoil disposal (Spencer et al., 2006). The method utilises vessel-mounted pumps that inject high volumes of low-pressure water into channel sediments via a horizontal spraybar located beneath the water surface (Scuria-Fontana, 1994). Hydraulic agitation reduces bed sediment cohesion and resistance to entrainment, resulting in the mobilisation of fine-grained particles (sands and silts) that are transported downstream through persistent jetting under ambient flows (Scuria-Fontana, 1994). The majority of particles are transported within the lower third of the profile and as a fluid mud layer, that varies in thickness – from a few centimetres to a few decimetres – with the thickness of the density current much smaller than the overlying water column (Winterwerp, 2002).

Given the nature of hydrodynamic methods and the corresponding sediment dispersal mechanisms, there is significant potential for fine sediment and trace metal resuspension effects, particularly where contaminant-rich sediments are dredged. Some work has considered the potential physicochemical effects of WID on contaminant release and so, water quality, with potential implications for aquatic biota (Spencer et al., 2005). Sediment cores collected from a proposed WID site in South East England were analysed for contaminants and biota were found at risk if native sediments were resuspended, due to elevated porewater ammonia concentrations and bed sediment toxicity (Spencer et al., 2006). WID-induced release of sediment-bound contaminants into the water column and dispersal of contaminated sediments over large spatial scales remain significant environmental concerns (Sullivan, 2000; Ospar Commission, 2004; Spencer et al., 2005), with more work required to assess the net effect of the method.

To our knowledge, no other published studies exist on the geomorphic and/or physiochemical impacts of WID. Therefore, the primary objective of this in-situ study was to determine the nature and persistence of WID effects on estuarine channel morphology at the reach scale. WID was selected as an emerging yet poorly studied hydrodynamic dredging method and utilised in the tidal River Parrett (see subsection 2.1) to restore channel capacity for the maintenance of flood water conveyance and in an effort to find a sustainable method of sediment management. These avenues of inquiry are important in the contexts of assessing method efficacy and the intentional and unintentional impacts of the technique, which remain poorly understood. Additionally, there remains significant uncertainty over contemporary rates of sedimentation and how these vary pre- and post-dredging within the tidal River Parrett, hence this study will also generate baseline data for future management. The work had three principle aims:

1. To investigate the influence of WID on channel (bathymetry and bed sediment grain-size) and flow (bathymetry)_conditions within the dredged reach to assess dredge efficacy and so, suitability.

2. To assess the temporal persistence of changes in channel condition caused by WID.

3. To quantify the channel and physicochemical impacts of WID on downstream localities through processes of downstream sediment dispersion.

2.0 Materials, methods and data analysis

2.1 Study sites, dredging history and sampling overview

The River Parrett is in South West England (Figure 1a) and is characterised by gravel and sand, fine gravel and coarse sand and silt bed typologies in its upper, lower and tidal reaches, respectively. The river is 59 km long and drains a catchment of 1,700 km^2 (Figure 1b), before discharging into Bridgwater Bay via the Somerset Levels (the Levels hereafter). The Levels are a coastal floodplain and wetland area of Somerset located between the Mendips and Blackdown Hills. The area is low lying and geologically complex, with Triassic formations underlying protrusions of younger lithologies, including upper lias sand, carboniferous limestone and marl (Hardy, 1999). The catchment includes wetlands of international importance (e.g. SPA and Ramsar; Natural England, 2013) and nationally significant floodplain grazing marshes (e.g. SSSI; Natural England, 2013) and there are important differences in land use between the wider catchment and the Levels. Arable and large-scale (often zero graze) dairy farming are prevalent across areas surrounding the Levels. Dairy farming has declined across the peat moors but is still important along the periphery of the Levels. Beef farming is now the main form of grazing in the wetter areas of the catchment and whilst Withy Growing there has declined, it remains an important land use on some Moors.

The sediment regime of the upper and middle reaches of the River Parrett is characterised by a significant sand and gravel bed load together with a fine-grained suspended sediment load derived mainly from agricultural land. The estuarine reach of the Parrett is extensive extending through the Levels and some 18 km inland from the Severn Estuary. The reach has a sediment regime characterised in a downstream direction increasingly by a high concentration of marine derived suspended load of silt and clay with a coarser sand and fine gravel bed load (Ambios, 2017; Partrac, 2009).

The River Parrett is heavily modified in its lower reach as it passes through the Levels. The river is constrained by artificial levees for flood relief purposes and highly affected by tidal processes, with a regular supply of marine sediment delivered from Bridgwater Bay, in the upper Severn Estuary, during penetration of tidal flows upstream. In addition, spillways and sluice gates are located across the Levels and used to divert high flows out of the fluvial Parrett and into a bypass channel (the River Sowy) or onto the inland Moors for short-term storage. High rates of sedimentation within the river’s lower estuarine reaches reduce the capacity of the channel and its ability to convey fluvial flood flows from upstream leading to an increased risk of flooding of low-lying land across the Levels. Historically, the impacts of sedimentation on flood water conveyance have been mitigated through human intervention, particularly via mechanical dredging activities. Dredging of the riverbed and banks has predominantly been achieved using boat-mounted or land-based bucket excavators and spoil was used to develop levees and/or improve agricultural land. These methods present a number of challenges as they tend be expensive and disruptive operations. Further, where the majority of biomass is located within the margins, as in the River Parrett, land-based excavation of sediments and associated bankside degradation can have detrimental ecological effects, reducing for example, faunal abundance and diversity (Brooker, 1985). The frequency by which the method needs to be applied to mitigate high rates of sediment accretion is high, prompting questions over the suitability of dredging by extraction for maintaining flood flow conveyance through the tidal reaches of the River Parrett (HR Wallingford, 2016). Thus, in the winter months of 2016 and 2017, the Somerset Rivers Authority and Somerset Drainage Boards Consortium undertook two hydrodynamic dredging trials using WID and two environmental monitoring campaigns spanned these periods to gain better understanding of the impacts of these operations on river geomorphology and water physicochemistry.

Environmental monitoring was conducted between 20th October 2016 and 24th October 2018 (Figure 2) on a ≈ 7.7 km reach of the River Parrett, located between the M5 road bridge and the village of Burrowbridge (experimental reach hereafter; Figure 1c). Dredging occurred 15th November - 16th December 2016 (WID trial 1; WID 1 hereafter) and 3rd December - 9th December 2017 (WID trial 2; WID 2 hereafter) along a ≈ 1.7 and ≈ 5.0 km reach, respectively, and sampling occurred before, during and after these periods (dredging periods hereafter) at a variety of locations within the experimental reach (details are provided herein; Figure 1c) where morphological, sedimentological and/or physicochemical changes were anticipated.

Differences in dredging period were due to differences in campaign objectives and associated environmental monitoring programs. Specifically, Bathymetric and fixed-depth and depth-integrated water physicochemistry sampling was performed to assess effects and longevity of effects of WID 1 and additional measures of bathymetry and bed sediment grain-size were made to gauge long- and short-term effects of WID 2. Additional geomorphological, hydraulic and/or ecological monitoring occurred during both dredging campaigns but the collected data are not considered here.

2.2 Dredging activities

Dredging was completed by Van Ord UK Ltd. using the WID vessel Borr (S1). An articulating spraybar mounted to the stern of the vessel was lowered below the dredge and fed river water via a low pressure (~1 bar) pump. The spray bar was positioned horizontal to and immediately above the bed, with the spray nozzles pointing downwards. Water released from the nozzles agitated bed sediments, reducing bed sediment cohesion and resistance to entrainment and fine sediments mobilised via hydraulic action were transported downstream by the flow. WID was concentrated in a 6m-wide swathe along the dredge reach in the channel centre within the thalweg zone. WID only occurred on ebbing high spring tides in an up-estuary direction and ceased when low river levels prevented safe navigation. Multiple passes of the dredge reach were completed during successive tides over several days within each campaign (Figure 2) to achieve a specific design profile.

2.3 Characterising and analysing differences in channel bathymetry

How much and where material is displaced by WID and where displaced sediments are deposited and for how long, represent important avenues of inquiry in the context of determining the efficacy, sustainability and overall suitability of the method for sediment management in estuarine systems. Thus, the reach between the M5 bridge and Burrowbridge (Figure 1c) was surveyed on five occasions to assess impacts and longevity of impacts of the two dredging campaigns on channel bathymetry (Aims 1 and 2). Specifically, sampling occurred November 1st 2016 (pre-WID 1), February 14th 2017 (post-WID 1a), October 18th 2017 (post-WID 1b and pre-WID 2), January 8th 2018 (post-WID 2a) and October 24th 2018 (post-WID 2b; Figure 2).

Data were collected using a GeoSwath 4R interferometric sonar, mounted on a boat in conjunction with a Real-Time Kinematic (RTK) GNSS positioning system and motion and head referencing unit. The unit sampled at 500 Hz and swathe width was set at 10-15 m, allowing for interrogation of the bed up to the waterline. Typical water depths during sampling were 0.5 - 3.0 m and the bed immediately beneath the transducer head represented a blindspot for the unit. Thus, on each sampling occasion, the boat made two passes of the survey reach – one in the upstream and one in the downstream direction – to ensure full lateral coverage. Sampling always occurred under spring high tides and depth measurements were corrected by an offset of -0.413 m because the transducer was mounted below the RTK and water surface. The Real-Time Kinematic (RTK) GNSS reduced sensor heights into Ordnance Survey levels, ensuring data comparability. Point values of locations and depths (scans hereafter; XYZ.txt format) were exported from Kongsberg Geoacoustics Ltd GS4 software and saved to external storage media for ex-situ analysis.

On each occasion the surveyed reach included both a dredged section of channel (dredge reach hereafter; ≈ 1.7 and ≈ 5.0 km during WID 1 and WID 2, respectively) and a section of river located downstream and between the dredge reach and M5 bridge (downstream reach hereafter; ≈ 8.0 and ≈ 4.2 km during WID 1 and WID 2, respectively; Figure 1c).

Topographic parameters were extracted from DEMs constructed using scans of the surveyed reach. Scans corresponded to the central region of the channel and a proportion of each of the relatively stable and un-dredged bank regions and consisted of approximately 491,000 irregularly spaced x, y and z coordinates with an average x-y spacing of 0.5 m. The point clouds were then converted into raster DEMs (x-y spacing = 0.5 m) using a kriging interpolation algorithm in ArcGIS © v.10.4.1 (Environmental Systems Research Institute, Redlands, CA, USA). Both the dredge and downstream reaches were digitised (polygon) and the areas extracted in ArcGIS for independent analyses (S2).

Topographic changes (Δd) to the bed surface were quantified by creating digital elevation models of difference (DoDs hereafter) from the surface DEMs of the dredge and downstream reaches, before and after dredging (Figure 3). The GNSS positioning system had an accuracy of 0.05 m in the vertical plane whereas the interferometric device had an accuracy of approximately 0.1 m. Thus, an error factor of ± 0.30 m was applied during DoD analysis, due to there being two scans per DoD and so, twice the potential for error. Differences in channel topography that exceeded ± 0.30 m were considered a result of dredging or natural erosional/depositional process and not a result of equipment errors. Bed change was partitioned into 11 bins corresponding to differences in elevation in 1 m increments above and below the error factor.

Volumetric change of bed sediment through time as a function of dredging and/or natural erosional and depositional processes was also assessed. Data were extracted from each of the dredge reach and downstream reach DoDs using the surface volume (3D Analyst) tool in ArcGIS. During analysis, plane height was set to ± 0.30 m and volumes above and below this threshold were extracted and used to calculate net volumetric change. Negative and positive values imply volumetric sediment losses (net erosion) and gains (net deposition), respectively.

The standard deviation of surface elevations was used as a measure of bed surface roughness (Aberle & Smart, 2003) and was compared before and after the dredging periods for both the dredge reach and downstream reach. Other bed elevation parameters including mean, maximum and minimum elevations were extracted from DEMs and compared in the same way.

Finally, the effects of dredging on channel cross-section properties and so, river flow characteristics (Aim 1) were assessed at 6 locations along each of the dredge and downstream reaches (WID 1 and WID 2). In ArcGIS, cross sections were digitised (polyline) at the up and downstream limits of and then at regular intervals along dredge and downstream reaches. The number of cross sections did not differ between reaches and indeed, dredging programmes, but the spacing between cross-sections did due to differences in reach lengths. Three subsequent analyses were performed. First, in ArcGIS, 3D analyst interpolate line and point profile tools were used to extract profile elevation data from each of the DEMs. The length of each parabola (channel perimeter length hereafter; proxy for cross-section capacity) was then approximated using the trapezoidal rule. Second, volumetric change of bed sediment through time was assessed for each of the cross sections. Digitised areas corresponding to cross sections were extracted from DEMs in ArcGIS before the cut fill (3D Analysis) tool was used to determine the total volume of sediment eroded/deposited per profile. Third, to investigate the effect of WID on riverflow characteristics, the total volume of water passing through each profile under a simulated flow condition was determined. Prior to analysis, DEMs maintaining identical aerial extents but different elevations to their equivalent cross section DEMs were created using the Raster Calculator in ArcGIS. Importantly, elevations of the newly created DEMs were 6 m, slightly above the highest recorded cross section elevation (5.91 m). The Cut Fill tool was then used to determine the volume of water contained within the surveyed channel under simulated flow conditions. Each analysis gave 6 values per occasion and dredging campaign for each of the dredged and downstream reaches (n = 72 total). Summary statistics were used during all analyses.

2.4 Characterising and analysing differences in bed material grain-size distributions

Sieve and particle laser size analysis of sediments recovered from the dredge reach and up and downstream localities were used to quantify the initial impact of WID 2 on bed sediment grain-size distributions (Aim 1). This was an important test as WID induced changes in bed sediment composition will likely have implications for particle entrainment thresholds and so, transport fluxes. Benthic sediment surveys were undertaken at five sites (Figure 1c) and on two and one occasion(s), before and after dredging, respectively (Figure 2b). Sites were located within (dredged; n = 3) and downstream (downstream; n = 2) of the dredge reach and samples were collected on two occasions before (14th and 15th November 2017) and on three occasions after (26th December 2017, 9th February 2018 and 18th May 2018) dredging. At each site and on each occasion, an Ekman grab sample was collected from the unvegetated inter-tidal sediment surface on the left bank, right bank and channel centre using a boat at low tide. Samples were returned to the laboratory and processed to remove and determine the prevalence of particles > 1 mm, prior to laser sizing. Specifically, individual samples (n = 3 per site and per occasion) were dried and disaggregated using a pestle and mortar before being sieved to 2 mm using sieve stacks and a shaker. A ≈ 4.5 g subsample was then taken from the pan material of individual samples and used during further grain size analysis.

Grain size analyses were performed on the subsamples using a Beckman-Coulter LS230 laser particle size analyser with polarisation intensity differential scattering (PIDS). Particles across a range of 0.04 to 2000 μm can be measured across 116 class intervals with the system. The Fraunhofer theory was selected to calculate grain-sizes, from the intensity of the diffracted light, since the samples contained limited material finer than 10 µm (Beckman-Coulter, 1994; de Boer *et al.*, 1987). Prior to grain size analysis, organic matter within samples was removed by loss on ignition (LOI; CEN, 2007). To ensure complete disaggregation, preliminary tests illustrated samples required agitating within the fluid module for 120 minutes (determined by monitoring run time effect on mean, median, mode, D_{10} and D_{90} percentile values; data not shown) until a negligible variability was reached. After this time the particle size distribution was recorded over 10 runs lasting 60 seconds. Output data included statistical summary parameters, which mathematically described the particle size distribution across the 10 runs, including mean, mode, standard deviation, variance, skewness (degree of symmetry), kurtosis (degree of peakedness) and cumulative percentile values (D_{10}, D_{25}, D_{50}, D_{75}, D_{90}; the particle size at which a specified percentage of the particles are finer), and statistics were calculated geometrically based on a log-normal distribution (Beckman-Coulter, 1994) using equations 1-5 (S2). Sample means were calculated from the ten measurements per sample to provide 6 pre- and 3 post-dredging data points per site, which were used during statistical analyses. Comparisons of pre- vs post-dredging data for each of the sites were made using normality tests (Shapiro-Wilk test) followed by either Oneway ANOVA or the Kruskal-Wallis test, as appropriate.

2.5 Characterising and analysing differences in water physicochemistry

Time series and depth-integrated water physicochemistry data were collected during WID 1 and were used to address Aim 3. SI 6600 V2 Sondes were installed at the downstream end of the dredge reach at Westonzoyland and upstream of the dredge reach at Burrowbridge (Figure 1c) on the 20th October 2016 and were retrieved 16th January 2017 (Figure 2a). Thus, the time period included 20, 23 and 41 days of pre-, during- and post-dredging data. Probes monitored conditions at approximately 1 m depth and measured optical back scatter (turbidity), conductivity (µS l^{-1}) and dissolved oxygen (mg l^{-1}, %) every 15 minutes. Stage data for the river Parrett were recorded within the dredge reach at Northmoor pumping station, and provided by the EA.

Fixed-depth sampling prohibits assessment of water physicochemistry through the water column. Thus, and in addition, a Partech 740 turbidity meter and YSI ProDSS sonde, measuring conductivity (used to calculate salinity), pH, turbidity and dissolved oxygen were attached to an Owen tube and used to measure characteristics of the water column before and during WID (Figure 2a). Specifically, 21 profiles were collected within the dredge reach pre-dredging and 39 profiles were collected downstream of the active dredging vessel. Profiles consisted of approximately 114 measurements that were approximately equally distributed throughout the water column. Water samples for calibrating the deployed optical turbidity sensors (to convert from nephelometric turbidity units (NTU) to total suspended solids(TSS)) were also collected with the Owen tube.

Summary statistics were used to analyse time series data. Pre- and during dredging differences in water physicochemistry were tested using normality tests (Shapiro-Wilk test) followed by either Oneway ANOVA or the Kruskal-Wallis test, as appropriate.

3.0 Results

3.1 The effect of WID on channel bathymetry (aim 1)

The majority (70%) of the surface area of the dredge reach was shown to be modified (i.e. elevation change > ± 0.3 m) following both WID 1 and WID 2 (Figures 4a and 5a). Within the modified areas (i.e. elevation change < -0.3 and > 0.3 m), the majority of disturbance (82 and 83%, respectively) fell within the $-1 < \Delta d \leq -0.3$ m category. Reductions in maximum bed elevation (-13 %) and bed surface roughness (-15 %) and an increase in minimum bed elevation (+296 %) were detected following WID 1 (Table 1). Following WID 2, reductions in mean (-14 %), maximum (-7 %) and minimum (-56 %) bed elevations and an increase in bed surface roughness (+25 %) were recorded (Table 1). Volumetric changes after WID 1 and WID 2 were -6,955 and -17,565 m^3 sediment, respectively (Figure 6a and b). In terms of channel cross sections, mean channel perimeter length increased from 14.84 ± 1.34 m^3 to 15.36 ± 1.33 m^3 (+ 3.56 ± 0.01%) and from 14.74 ± 3.31 m^3 to 15.08 ± 3.20 m^3 (+ 2.51 ± 0.02 %) following WID 1 and 2, respectively (S3, S4). Mean volumetric changes subsequent to WID 1 and 2 were -3.96 ± 1.77 m^3 and -3.64 ± 2.29 m^3, respectively (S5), and mean water volumes increased from 21.99 ± 2.55 m^3 to 25.88 ± 4.00 m^3 (+ 17.40 ± 0.07 %) following WID 1 and from 23.92 ± 5.10 m^3 to 27.56 ± 5.90 m^3 (+ 15.37 ± 0.09 %) after WID 2 (S4).A smaller proportion of the total surface area of the downstream reach was changed following WID 1 and WID 2 in comparison to the upstream dredged reach (38% and 48%, respectively; Figures 4c and 5c). Similarly, volumes of losses of sediment were also much reduced (-2,689 and -2,469 m^3 sediment, respectively; Figure 6a and b) in comparison to upstream dredged reaches, and erosion was entirely caused by natural fluvial scour over the winter period rather than including the effects of dredging. After WID 1, bed surface roughness values increased by 96 % (from 0.46 to 0.90) but decreased by 6% (from 0.95 to 0.89) after WID 2 (Table 1). With regard channel cross sections, mean channel perimeter lengths increased from 15.66 ± 1.69 m^3 to 15.89 ± 1.73 m^3 (+ 1.48 ± 0.01 %) and from 14.54 ± 1.15 m^3 to 14.66 ± 1.22 m^3 (+ 0.80 ± 0.02 %) following WID 1 and 2, respectively (S3, S4). Mean volumetric changes subsequent to WID 1 and 2 were -2.45 ± 1.28 m^3 and -2.91 ± 2.73 m^3, respectively (S5), whereas mean water volumes increased from 27.37 ± 4.23 m^3 to 29.82 ± 4.08 m^3 (+ 9.33 ± 0.06 %) following WID 1 and from 26.03 ± 5.35 m^3 to 28.94 ± 3.06 m^3 (+ 13.91 ± 0.18 %) after WID 2 (S4).During periods of recovery (WID 1: February 14th – October 18th 2017; WID 2: January 18th – October 24th 2018) between

WID 1 and WID 2 (period 1) and following WID 2 (period 2), large proportions of the riverbed were modified in both the dredge and downstream reaches (Figures 4 and 5). The majority of change (dredge reach 65%, downstream 67%) during period 1 fell within the $0.3 < \Delta d \leq 1$ m elevation change category whereas during period 2, dominant elevation change categories were $1 < \Delta d \leq 2$ m (dredge reach 46%) and $-0.3 < \Delta d \leq 0.3$ m (downstream 54%). During period 1, mean bed elevation increased from 2.90 to 3.68 m (as did maximum and minimum values; Table 1) in the dredge reach. During period 2, mean and minimum bed elevations increased whereas maximum values decreased (Table 1). Mean channel perimeter length decreased from 15.36 ± 1.33 m^3 to 15.20 ± 1.30 m^3 (-1.06 ± 0.01 %) and from 15.08 ± 3.20 m^3 to 14.56 ± 3.14 m^3 (-3.42 ± -0.02 %) during periods 1 and 2, respectively (S3, S4). Mean volumetric changes during periods 1 and 2 were 6.28 ± 1.43 m^3 and 0.76 ± 3.09 m^3, respectively (S5). Mean water volumes subsequently decreased from 25.88 ± 4.00 m^3 to 19.65 ± 2.85 m^3 (-23.95 ± 0.03 %) following period 1 and from 27.56 ± 5.90 m^3 to 26.80 ± 7.96 m^3 (-3.81 ± 0.10%) after period 2 (S4).

The downstream reach also saw mean, maximum and minimum elevations increase during period 1 and channel roughness decreased in the dredge reach from 0.71 to 0.65 but increased from 0.90 to 0.93 in the downstream reach. During period 2, reductions in mean, maximum but not minimum bed elevation values were detected for the downstream reach (Table 1). Volumetric changes of 11,411 and 13,324 and 7,564 and -2,421 m^3 sediment were recorded for dredge and downstream reaches following periods 1 and 2, respectively (Figure 6). Mean channel perimeter length decreased from 15.89 ± 1.73 m^3 to 15.78 ± 1.62 m^3 (-0.61 ± 0.02 %) and from 14.66 ± 1.22 m^3 to 13.88 ± 1.27 m^3 (-5.34 ± 0.02 %) during periods 1 and 2, respectively (S3, S4). Mean volumetric changes during periods 1 and 2 were 6.40 ± 2.52 m^3 and -1.08 ± 0.97 m^3, respectively (S5). Mean water volumes subsequently changed from 29.82

± 4.08 m^3 to 23.42 ± 5.40 m^3 (-22.07 ± 0.10 %) following period 1 and from 28.94 ± 3.06 m^3 to 30.01 ± 3.95 m^3 (+ 3.49 ± 0.03 %) after period 2 (S4).

3.2 The effect of WID on bed material grain-size distributions (aim 2)

Dredged and downstream sites did not record significant changes in any of the measures grain-size parameters before and after dredging (Oneway ANOVA or Kruskall Wallace test, α = 0.05; Table 2).

3.3 The effect of WID on water physicochemistry (aim 3)

After initial installation of the static sondes one of the probes became detached 7th - 11th November 2016 and was recovered a short distance downstream. The effect of this was that five days of pre-dredging data were lost at the upstream sampling location. The time-series indicated that turbidity levels were substantially elevated in the dredge reach in comparison to upstream of the dredge during WID 1 (Figure 7a and b). However, because dredging occurred during periods of high tide (Figure 7c), the impact of the dredge could not be isolated from elevated turbidity associated with tidal effects. Indeed, the elevation in turbidity during high tide and dredging in mid-November was of similar magnitude to during high tide with no dredging in mid-December (Figure 7). Dissolved oxygen levels were generally higher at the dredge site than the upstream site but dropped during dredging to well below those of the upstream site (Figure 8a). However, oxygen levels returned rapidly to pre-dredge levels, usually within an hour of dredging being ceased, and also dropped during other periods of high tide. Conductivity is consistently higher at the dredge site and rises substantially during dredging (Figure 8b). Unlike other measures, similar peaks are not observed during later high tides without dredging.

Dredging was associated with statistically significant increases in TSS (Kruskal-Wallis: H_1 = 12.32; $P < 0.001$) and Turbidity (Kruskal-Wallis: H_1 = 12.21; $P < 0.001$) and statistically significant decreases in pH (Kruskal-Wallis: H_1 = 17.97; $P < 0.001$) and dissolved oxygen (Kruskal-Wallis: H_1 = 5.66; $P = 0.017$) relative to pre-dredging conditions (Figure 9). Salinity data were similar before and during dredging.

4.0 Discussion

WID 2 reduced the elevation of the bed surface, as indicated by reductions in mean, maximum and minimum bed surface elevations (Table 1), and topographic alterations corresponded with a volumetric change of -17,565 m^3 sediment (Figure 6). The extent and nature of disturbance was consistent between dredging operations. Specifically, similar proportions of dredge reaches, by area, were disturbed during dredging operations and in both cases, the majority of topographic alterations involved a 0.3 - 1 m reduction in sediment surface height, suggesting dredge-induced turbation was largely restricted to the uppermost meter of sediment. That the majority of topographic alterations involved surficial sediment rearrangement represents a significant finding given concentrations of contaminants in some systems are elevated in the upper layer (dozens of centimeters) of sediments (Liu et al. 2016). Thus, application of WID in systems where high concentrations of contaminants are detected within surface sediments requires careful consideration of the specific risks associated with potential contaminants.

Visual assessments of DEMs (Figure 3; S3) suggest WID was consistently applied within the dredge reach, creating a relatively homogeneous bed with topographic alterations concentrated around the channel centre and not the margins. By contrast, extraction operations typically target much smaller spatial areas, often over longer time scales, resulting in aggregate shoal reduction and formation and/or widening of pits and troughs, respectively (Wu et al., 2016).

Findings demonstrate that the WID method was effective in removing the required volume of sediment from the ecologically-poor thalweg without directly impacting the relatively ecologically rich inter-tidal bank face, which is a significant disadvantage of some extraction methods. Some localised loss of inter-tidal bank material and bank slumping was noted following both WID trials (Ambios, 2017) where the toe of the bank had effectively been over-steepened by the deeper thalweg zone created by WID. This was an anticipated outcome and supports one of the aims of the WID trials – that is, the required loss of bank volume is achieved through enhanced fluvial scour processes rather than a more damaging direct impact on habitat and species through physical excavation.

Both dredging operations were associated with significant reductions in sediment volume along the dredge reach (Figure 6; S3 and S5), increasing channel perimeter length and channel capacity (S4) and so, potentially reducing flood risk locally. However, following WID 2, an increase in bed surface roughness (+ 25%) was detected which might typically be associated with increased wetted perimeters and flow retardation and so, a theoretical potential to reduce flow conveyance. However, in this case, it is far more likely that detected increases in the standard deviation of surface elevations reflect a deepening of the channel and therefore, an increase in the range of bed elevations rather than net roughness.

Temporal persistence of dredging effects within the dredge reach varied between dredging campaigns. Changes in topography were relatively short-lived following WID 1, with significant sediment accretion detected between February and November 2017, as evidenced by increases in surface elevations (Table 1) and an 11,412 m^3 increase in sediment volume, which far exceeded initial reductions due to dredging (- 6,955 m^3 sediment; Figure 6). By contrast, topographic change due to WID 2 was longer-lasting – with increases in mean and minimum bed elevations corresponding with a 7,564 m^3 increase in sediment volume. Importantly, and unlike WID 1, losses due to WID 2 exceeded rates of sedimentation during the period of channel recovery. Thus, WID 2 continued to provide benefit following the study period, whereas benefit of WID 1 was limited to the first 10 months post-dredging. In line with previous studies (e.g. Partrac, 2009), differences in the persistence of WID effects were almost certainly caused by differences in fluvial flows and marine sediment delivery between years/ recovery periods. Indeed, gauge data (S6) indicate higher fluvial flows during the second recovery period than the first likely resulting in decreased deposition, increased scour and observed differences in the duration of dredging effects between the two campaigns.

It is reasonable to assume that bed disturbance through WID may have influenced natural sediment transport processes in at least three ways. First, particles displaced during WID may be deposited on the bed surface in positions of relative instability, making them more susceptible to entrainment, particularly during ebbing spring tides or under high fluvial flows. Second, turbation of surface sediments during WID might result in the loss of stabilising bed sediment structure and/or disturbance of the coarser surface armour layer. Modification of sediments in this manner, generating weakly structured sediments and/or substrates lacking a coarse surface layer, may promote sediment transport and erosion by reducing critical shear values required for particle entrainment. Third, channel deepening in turbid estuaries like the River Parrett estuary has been found to exacerbate natural rates of sedimentation (e.g. van Maren et al., 2015), potentially through reduced mean channel and nearbed flow velocities and so, shear stresses post-dredging. Thus, one might expect an increase in sedimentation potential and reduced scour following WID, relative to natural levels. Further, it is reasonable to assume reduced basal shear stress may have implications for several other natural processes, including tidal propagation and tidal flats inundation, although these avenues of enquiry were beyond the scope of this study.

The benefits of WID over other dredging methods include reduced perceived environmental effects, no dredge spoil disposal and relatively low time costs (Scuria-Fontana, 1994). WID has the potential to deliver maintenance outcomes in a more environmentally sustainable manner and it is more cost effective compared to bankside excavation and land disposal. Costs can be further reduced through recognition that natural fluvial scour can in some years be more effective in channel maintenance than dredging – thus to reduce costs as much as possible, dredging programmes should be evidence-based and adapt to inter-annual variation in accretion rates. Another advantage of WID over extraction methods is that dredged sediment remains within the system. This is particularly important in closed and/or low suspended sediment systems where sediment depletion may have significant long-term implications e.g. inter-tidal habitat degradation and/or loss.

Findings demonstrate WID was effective in removing large quantities of fine sediment from dredge reaches. However, detected rates of sedimentation post-dredging indicate changes were short-lived and regular repeat dredging is required to maintain channel size, and flood flow conveyance, in the River Parrett. It is reasonable to assume a need for river sediment and/or water management will change through time for at least three reasons. First, future temperature increases as a function of climate change will likely impact on river flows, reducing baseflow conditions in summer resulting in increased fine sediment accumulation and so, channel aggradation. Second, precipitation patterns may change with the expectation being that in a warmer climate, heavy rainfall will increase with fewer more intense rainfall events, which will have implications for the frequency and magnitude of flood events. Third, future sea level rise is predicted (Natural England, 2013) and expected to cause marine transgression, characterised by the upstream migration of the normal tidal limit and marine-influenced sedimentary environments (Pye and Blott, 2014). In the context of the River Parrett estuary it is reasonable to assume river base levels may be raised, decreasing stream power in the lower reaches and so, increasing sedimentation there and penetration of tidal flows upstream. The potential for a tidal barrier to mitigate against flood risk caused by extreme tidal surges is currently being investigated for Parrett estuary downstream of Bridgwater. Such a scheme could have uniuntentional implications for sediment dynamics, particularly during spring high tides when the barrier is operational to reduce flood risk, and when peaks in upstream marine sediment transport are expected. The presence of this artificial barrier may influence the process of natural adjustment to sea level rise (Pye and Blott, 2014), potentially resulting in reduced and increased aggradation of intertidal and/or subtidal deposits on the landward and seaward sides of the barrier, respectively. Long-term management strategies will therefore need to account for and be sensitive to both increased rates of anthropogenic changes in sediment and flow regimes and in addition, sedimentation and increased risk of high intensity rainfall and thus, elevated river flows. Furthermore, cost-effective and environmentally sustainable catchment-wide approaches to land management are required to increase field soil-water retention and, reduce fluvial sediment loadings in rivers where this represents a significant source of fine-grained sediment.

Surprisingly, dredged and downstream sites (Figure 1c) did not record significant changes in grain-size parameters before and after dredging (Table 2). This suggests WID mobilised the full range of particle sizes, rather than preferentially sorting sediments within the dredge reach, generating a coarse lag there and causing fining downstream. By contrast, other hydromorphological methods such as gravel jetting (see Basic et al., 2017) have been shown to modify the surface but not sub-surface size distribution of sediments, though substrates were coarser in nature and maintained a broad range of size classes.

Sediments mobilised from the dredge reach during both campaigns had a negligible impact on bed surface elevations downstream. Within downstream reaches, small proportions of scanned surfaces corresponded to elevation gain categories ($\Delta d > \pm 0.3$; 4 and 3%) and net volume change was negative (-2,689 and -2,469 m^3 sediment), following WID 1 and WID 2, respectively. Findings imply erosional processes were dominant and deposition negligible within downstream reaches during the periods between sampling. However, sampling frequency was insufficient to capture any short-term impacts (i.e. in the days and weeks after WID) post-dredging but before winter fluvial flood flows, which may have led to an underestimation of dredging effects downstream. Further, a lack of deposition within downstream monitored reaches prompts questions around the fate of sediments displaced from the dredge reach during WID. It is reasonable to assume these may have been deposited beyond the survey reach within Bridgwater Bay, potentially influencing estuarine sediment dynamics and/or hydraulics there.

The nature of disturbance during recovery periods varied between locations, with dredge reaches recording greater proportions of scanned surfaces within higher positive elevation change categories than downstream reaches (Figures 4 and 5). For example, during the post-WID 2 recovery period, 271% more surface elevations were recorded in the $1 < \Delta d \leq 2$ m disturbance category for the dredge reach relative to the downstream reach. This is strong evidence of the dredge reach being taken out of 'regime' (where sediments are in equilibrium with average energy conditions) and placed in a state of change post-dredging (Ambios, 2017). Specifically, the increased cross-sectional dimensions as a function of dredging will have reduced mean flow velocities and so increased potential rates of sedimentation within the dredged reach but not the downstream reach. A similar but less marked trend was observed during the recovery period following WID 1.

In terms of water physicochemisty, turbidity peaks were relatively unaffected by dredging and comparable in magnitude to pre- and post-dredge high tide peaks (Figure 7). Turbidity was higher at the upstream site during dredging than during subsequent high tides without dredging, indicating that sediment was transported upstream by the tide in greater concentrations in conjunction with dredging than without. In addition, the conductivity was substantially higher during dredging than during high tides alone (Figure 8b). Collectively, this evidence may indicate that bed materials are rendered less stable by dredging and are washed higher into the water column, allowing them to be moved relatively more easily and further upstream than would be expected in the absence of dredging. However, it is clear that the physiochemical impacts of dredging were similar to high spring tides in the study reach and that penetration of the tide is highly variable and strongly influenced by fluvial flows. Dissolved oxygen levels dropped during WID, but these declines were similar in magnitude to those associated with high sping tides and were of short duration (< 1 hour; Figure 8a).

At the ecosystem level, a primary concern is water quality and dredging activities capable of detrimentally influencing water physicochemistry may have negative consequences for resident biota. Although significant changes in pH, dissolved oxygen, turbidity and SSC were detected during dredging (Figure 9), it is reasonable to assume the dredging program had a negligible impact on in situ populations for at least three reasons. First, changes in water physichochemistry parameters were short-lived, with individual events lasting less than 60 minutes. Environmental impact is at some level proportional to exposure time which was short, especially for highly mobile species like fish that could evade a perceived danger. Second, water physicochemistry was found highly variable within the experimental reach due to tidal influence. Thus, species naturally present are likely to be resilient to rapid changes in water physichochemistry and to sub-optimal conditions, as observed during dredging and under high tides without dredging. Third, dredging operations were scheduled around migrations of potentially sensitive migratory species such as Atlantic Salmon *Salmo salar* and eels *Anguilla anguilla*. Appropriate planning can help mitigate impacts on ecological communities. Thus, WID can have considerable physicochemical effects although the ecological consequences of these and other dredging related factors (e.g. vessel noise) for biota and fisheries are poorly understood.

5.0 Conclusion

The present study begins to fill the knowledge gap regarding the environmental impacts and sustainability of WID, an increasingly used yet understudied dredging technology. We found strong evidence of a dredging effect, although the effects of WID could not be isolated from those of other processes/factors that affect tidal and indeed, heavily modified systems such as the River Parrett estuary. The paper revealed WID is an effective sediment removal technique for the surveyed reach although geomorphic effects of the method can be short-lived, lasting less than 10 months, meaning regular repeat dredging may be necessary to maintain channel geometries. The study also indicates that the method failed to measurably change bed sediment grain-size distributions and given the highly tidal nature and associated high suspended sediment concentrations typical of the study reach, effects of WID relative to the tide on some water physicochemistry parameters were negligible, although in other less naturally turbid tidal waters the environmental impacts may be more significant. Importantly, the biogenic signature of WID was uniquely different to that of other, better-studied dredging technologies, including mechanical and hydraulic methods, with channel deepening targeted at the thalweg. This paper indicates an urgent need to better understand the geomorphological, physicochemical and ecological effects of WID across a range of environmental conditions and management legacies – from un-dredged to regularly dredged systems. Investigations of the long-term geomorphological and ecological responses to continued WID and differences in effects when applying WID in recently dredged and undredged reaches are required. Until this knowledge is developed, the ability of river managers to fully understand the potential environmental and operational ‘costs’ and ‘benefits’ of WID will remain highly constrained.

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References

Aberle, J., Smart, G.M., 2003. The influence of roughness structure on flow resistance on steep slopes. *Journal of Hydraulic Research*, **41** (3), 259–269. https://doi.org/10.1080/00221680309499971

Ambios, 2017. Dredging Trials Monitoring Programme November - December 2016. Technical report to the Somerset Drainage Boards Consortium.

Basic, T., Britton, R.J., Rice, S.P., Pledger, A.G., 2017. Impacts of gravel jetting on the composition of fish spawning substrates: Implications for river restoration and fisheries management. *Ecological Engineering*, **107**, 71-81.

Beckman-Coulter, 1994. Coulter LS Series: Product Manual. Beckman Coulter, Miami, FL, 1-345.

Bormans, M., Marsalek, B., Jancula, D., 2016. Controlling internal phosphorous loadings in lakes by physical methods to reduce cyanobacterial blooms: a review. *Aquatic Ecology*, **50**, 407-422.

Bravard, J.P., Amoros, C., Pautou, G., Bornette, G., Bournaud, M., Creuze des Chatelliers, M., Gibert, J., Peiry, J.L., Perrin, J.F., Tachet, H., 1997. River incision in south-east France: morphological phenomena and ecological effects. *Regulated River Research and Management*, **113**, 75-90.

Brooker, M.P., 1985. The Ecological Effects of Channelization. *Geographical Journal*, **151** (1), 63-69.

Chen, M., Cui, J., Lin, J., Ding, S., Gong, M., Ren, M., Tsang, W.C.W., 2018. Successful control of internal phosphorous loading after sediment dredging for 6 years: A field assessment using high-resolution sampling techniques. *Science of the Total Environment*, **616-617**, 927-936.

CEN, 2007. Characterization of Waste – Determination of Loss on Ignition in Waste, Sludge and Sediments. CEN, Brussels.

Collins, B., Dunne, T., 1989. Gravel transport, gravel harvesting, and channel-bed degradation in rivers draining the Southern Olympic Mountains, Washington, USA. *Environmental Geology and Water Science*, **13** (3), 213-224.

Collins, M.A.,1995. Dredging-Induced Near-Field Resuspended Sediment Concentrations and Source Strengths, Miscellaneous Paper D-95-2, US Army Engineer Waterways Experiment Station.

de Boer, G.B., de Weerd, C., Thoenes, D., Goossens, H.W., 1987. Laser diffraction spectrometry: Fraunhofer versus Mie scattering. *Particle Characterization*, **4**, 14–19.

Ellery, W.N., McCarthy, T.S., 1998. Environmental change over two decades since dredging and excavation of the lower Boro River, Okavango Delta, Botswana. *Journal of Biogeography*, **25**, 361-378.

Gaillot, S., Piégay, H., 1999. Impact of gravel mining on stream channel and coastal sediment supply: example of the Calvi bay in Corsica (France). *Journal of Coastal Research*, **15** (3), 774-788.

Gob, F., Houbrechts, G., Hiver, J.M., Pettit, F., 2005. River dredging, channel dynamics and bedload transport in an incised meandering river (the River Semois, Belgium). *River Research and Applications*, **21**, 791-804.

Goossens, H., Zwolsman, J.J.G., 1996. An evaluation of the behaviour of pollutants during dredging activities. *Terra et Aqua*, **62**, 20-28.

Gustavson, K.E., Burton, G.A., Francingues, N.R., Reible, D.D., Vorhees, D.J., Wolfe, J.R., 2008. Evaluating the effectiveness of contaminated-sediment dredging. *Environmental Science and Technology*, **42**, 5042-5047.

Hardy, P., 1999. The Geology of Somerset. Ex Libris Press, 45. ISBN: 0948578424.

HR Wallingford, 2016. Opportunities for further dredging in Somerset. Part 1 – River Brue and tidal sections of the Rivers Parrett and Tone.

Inman, D. L., 1976. Man’s impact on the California coastal zone. Summary report to California Department of Navigation and Ocean Development, Sacramento.

Jing, L.D., Wu, C.X., Liu, J.T., Wang, H.G., Ao, H.Y., 2013. The effects of dredging on nitrogen balance in sediment-water microcosms and implications to dredging projects. *Ecological Engineering*, **52**, 167-174.

Ismail, S.S., Samuel, M.G., 2011. Response of River Nile dredging on water levels. Fifteenth International Water Technology Conference, IWTC-15. Egypt, Alexandria.

Kondolf, G.M., 1994. Geomorphic and environmental effects of instream gravel mining. *Landscape and urban planning*, **28** (2-3), 225-243.

Kondolf, G.M., Swanson, M.L., 1993. Channel adjustments to reservoir construction and gravel extraction along Stony Creek, California. *Environmental Geology and Water Science*, **211**, 256-269.

Kornis, S., Laczay, I.A., 1988. Effects of extensive dredging on the river regime. In: White, W.R. (Ed.), International Conference on River Regime, Wiley, London, 385-394.

Lach, J., Wyzga, B., 2002. Channel incision and flow increase of the upper Wisloka River, southern Poland, subsequent to the reafforestation of its catchment. *Earth Surface Processes and Landforms*, **27**, 445-462.

Lagasse, P.F., 1975. Interaction of River Hydraulics and Morphology with Riverine Dredging Operations. Unpublished PhD Dissertation. Colorado State University: Fort Collins, CO.

Lagasse, P.F., Winkley, B.R., 1980. Impact of gravel mining on river system stability. *Journal of the Waterway Port Coastal Ocean Division*, **106** (3), 389-402.

Liu, C., Zhong, J., Wang, J., Zhang, L., Fan, C., 2016. Fifteen-year study of environmental dredging effect on variation of nitrogen and phosphorus exchange across the sediment-water interface of an urban lake. *Environmental Pollution*, **219**, 639-648.

Liebault, F., Piégay, H., 2001. Assessment of channel changes due to long term bedload supply decrease: example of the Roubion river, France. *Geomorphology*, **36**, 167-186.

Luo, X.-L., Zeng, E.Y., Ji, R.-Y., Wang, C.-P., 2007. Effects of in-channel sand excavation on the hydrology of the Pearl River Delta, China. *Journal of Hydrology*, **243**, 230-239.

Mikkelsen, O.A., Pejrup, M., 2000. Insitu particle size spectra and density of particle aggregates in a dredging plume. *Marine Geology*, **170** (3), 443-459.

Natural England, 2013. National Character Area Profile: 142: Somerset Levels and Moors. http://publications.naturalengland.org.uk/publication/12320274 Last accessed: 10/06/2019.

Ospar Commission, 2004. Environmental impacts to marine species and habitats of dredging for navigational purposes. Biodiversity Series. ISBN 1-904426-50-6.

Partrac, 2009. Sediment Budget Report (River Parrett/Tone). Technical report for the Environment Agency, Partrac Ltd, Glasgow.

Petit F, Poinsart D, Bravard JP., 1996. Channel incision, gravel mining and bed-load transport in the Rhone river upstream to Lyon, France (Canal Miribel). *Catena*, **26**: 209–226.

Pinter, N., Miller, K., Wlosinki, J.H., Van Der Ploeg, R.R., 2004. Recurrent shoaling and channel dredging, Middle and Upper Mississippi River, USA. *Journal of Hydrology*, **290** (3-4), 275-296. https://doi.org/10.1016/j.jhydrol.2003.12.021.

Pye, K., Blott, S.J., 2014. The geomorphology of UK estuaries: The role of geological controls, antecedent conditions and human activities. *Estuarine, Coastal and Shelf Science*, **150**, 196-214.

Reich, M., 1994. Les impacts de l'incision des riviéres des Alpes bavaroises sur les communaute´s terrestres de lit majeur. *Revue de Géographie de Lyon*, **69**, 25-30.

Rinaldi, M., 2003. Recent channel adjustments in alluvial rivers of Tuscany, Central Italy. *Earth Surface Processes and Landforms*, **28**, 587-608.

Rinaldi, M., Simon, A., 1998. Bed-level adjustments in the Arno River, central Italy. *Geomorphology*, **22**, 57-71.

Scott, K.M., 1973. Scour and fill in Tujunga Wash – a fanhead valley in urban southern California – 1969. US Geological Survey Professional Paper 732-B.

Scuria-Fontana, C., 1994. Faster and cheaper water-injection dredging. Mechanical Engineering-CIME, 22. Academic OneFile, accessed 8 May 2019.

Sear, D.C., Archer, D., 1998. Effects of gravel extraction on stability of gravel-beds rivers: the Wooler Water, Northumberland, UK. In Gravel-bed Rivers in the Environment, Klingeman, P.C., Bescta, R.L., Komar, P.D., Bradley, J.N. (eds). Water Resources Publications: Highlands Ranch, USA; 415–432.

Simeoni, U., Corbau, C., 2009. A review of the Delta Po evolution (Italy) related to climatic changes and human impacts. *Geomorphology*, **107**, 64–71.

Smith, J.E., Friedrichs, C., 2011. Size and settling velocities of cohesive flocs and suspended sediment aggregates in a trailing suction hopper dredge plume. *Continental Shelf Research*, **31**, 50–S63.

Spencer, K.L., Dewhurst, R.E., Penna, P., 2005. Potential impacts of water injection dredging on water quality and ecotoxicity in limehouse Basin, River Thames, SE England, UK. *Chemosphere*, **63** (3), 509-521.

Spencer, K.L., MacLeod, C.L., Tuckett, A., Johnson, S.M., 2006. Source and distribution of trace metals in the Medway and Swale estuaries, Kent, UK. *Marine Pollution Bulletin*, **52**, 214-238.

Stevens, M.A., Urbonas, B., Tucker, L.S., 1990. Public-private cooperation protects river. APWA Reporter September: 25-27.

Sullivan, N., 2000. The use of agitation dredging, water injection dredging and sidecasting: result of a survey of ports in England and Wales. *Terra et Aqua*, **78**, 11-20.

Surian, N., 1999. Channel changes due to river regulation: the case of the Piave River, Italy. *Earth Surface Processes and Landforms*, **24**, 1135-1151.

Surian, N., Rinaldi, M., 2003. Morphological response to river engineering and management in alluvial channels in Italy. *Geomorphology*, **50**, 307-326.

Van Den Berg, G.A., Meijers, G.G.A., Van Der Heijdt, L.M., Zwolsman, J.J.G., 2001. Dredging-related mobilisation of trace metals: a case study in the Netherlands. *Water resources*, **35** (8), 1979-1986.

Van Maren, D.S., van Kessel, T., Cronin, K., Sittoni, L., 2015. The impact of channel deepening and dredging on estuarine sediment concentration. *Continental Shelf Research*, **95**, 1-15.

Vivian, C., Birchenough, A., Burt, N., Bolam, S., Foden, D., Edwards, R., Warr, K., Bastreri, D., Howe, L., 2012. Literature Review of Dredging Activities: Impacts, Monitoring and Mitigation. Cefas technical report.

Wang, X.D., Liu, X.D., Lu, J., 2012. Urban River Pollution Control and Remediation. *Procedia Environmental Sciences*, **13**, 1856-1862.

Wenger, A.S., Harvey, E., Wilson, S., Rawson, C., Newman, S.J., Clarke, D., Saunders, B.J., Browne, N., Travers, M.J., Mcilwain, J.L., Erftemeijer, P.L.A., Hobbs, J-P. A., Mclean, D., Depcynski, M., Evans., R., 2017. A critical analysis of the direct effects of dredging on fish. *Fish and Fisheries*, **19**, 967-985.

Winterwerp, J.C., Wang, Z.B., van Khester, J.A.Th.M., Verweij, J.F., 2002. Far-field impact of water injection dredging in the Crouch River. *Proceedings of the Institution of Civil Engineers Water and Maritime Engineering 154*, **4**, 185-296.

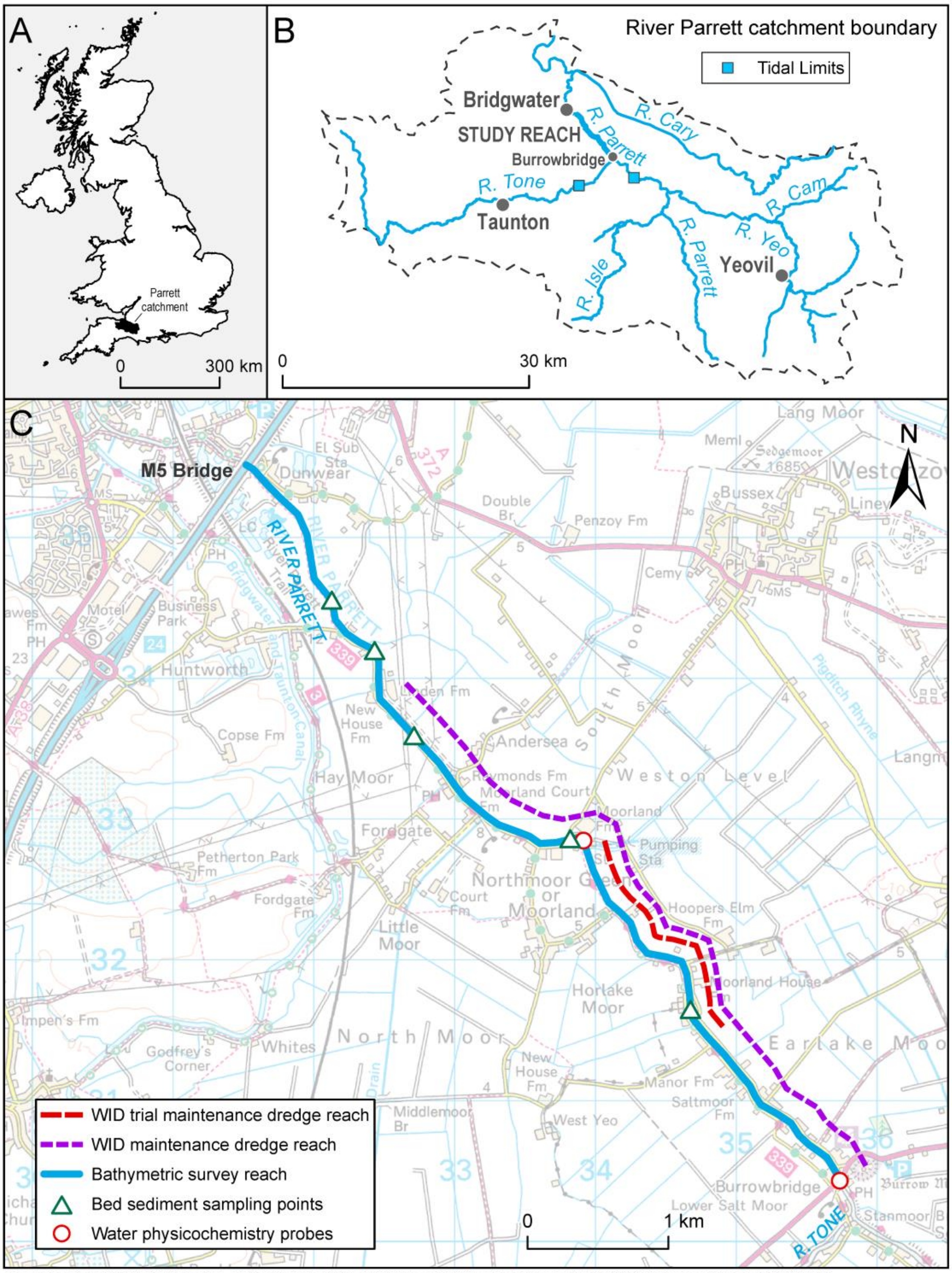
Wu, Z.Y., Saito, Y., Zhao, D.N., Zhou, J.Q., Cao, Z.Y., Li, S.J., Shang, J.H., Liang, Y.Y., 2016 Impact of human activities on subaqueous topographic change in Lingding Bay of the Pearl River estuary, China, during 1955-2013. *Scientific Reports*, **6**, 1-10.

Wu, Z., Milliman, J.D., Zhao, D., Cao, Z., Zhou, J., Zhou, C., 2018. Geomorphic changes in the lower pearl River Delta, 1850-2015, largely due to human activity. *Geomorphology*, 314, 42-54.

Wyzga, B., 1993. River response to channel regulation: case study of the Raba River, Carpathians, Poland. Earth Surface Processes and Landforms, 18, 541–556.

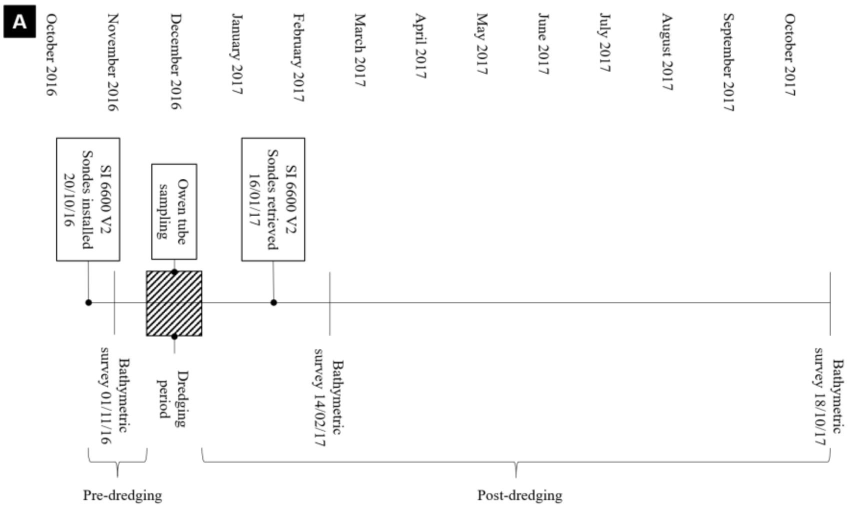
Figures

Figure 1: (A) Location and (B) catchment of the River Parrett and (C) details of the experimental reach between the M5 bridge and Burrowbridge, where bathymetric, bed sediment and water physicochemistry sampling occurred. The experimental reach in Figure 1C is highlighted in Figure 1B.

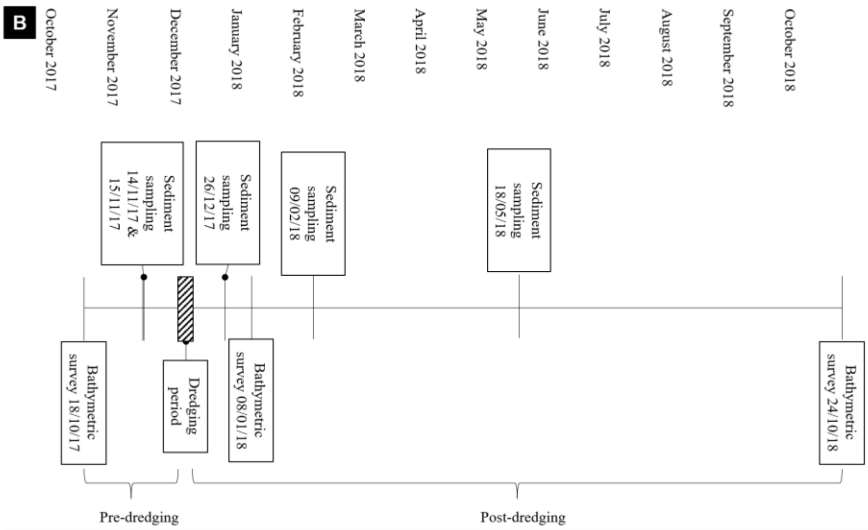


Created using ESRI ArcGIS Pro 2.2, British National Grid. © Crown copyright and database rights 2019 Ordnance Survey (100025252).

Figure 2: Schematic diagrams presenting the environmental monitoring and dredging programmes associated with A WID 1 and B WID 2. Timings and durations of dredging appear in tabular form.



Date	Start time	WID duration (mins)
15/11/2016	09:00	45
16/11/2016	09:45	75
17/11/2016	10:40	45
	12:40	30
18/11/2016	11:00	120
	22:40	30
30/11/2016	08:45	270
13/12/2016	20:20	160
14/12/2016	07:45	195
	19:00	270
15/12/2016	06:30	240
	12:00	60
	19:00	300
16/12/2016	11:10	70



Date	Start time	WID duration (mins)
03/12/2017	18:27	120
04/12/2017	06:49	180
	19:14	245
05/12/2017	07:34	285
	20:01	270
06/12/2017	08:20	270
	20:49	225
07/12/2017	09:08	225
	21:38	240
08/12/2017	09:58	180
	22:29	225

Figure 3: Examples of DEMs collected within and downstream of the dredge reach before and after (A) WID 1 and (B) WID 2, and the resultant DoDs (digital elevation models of difference). Dredging occurred between November 2016 and February 2017 in Figure 3A, and between November 2017 and January 2018 in Figure 3B.

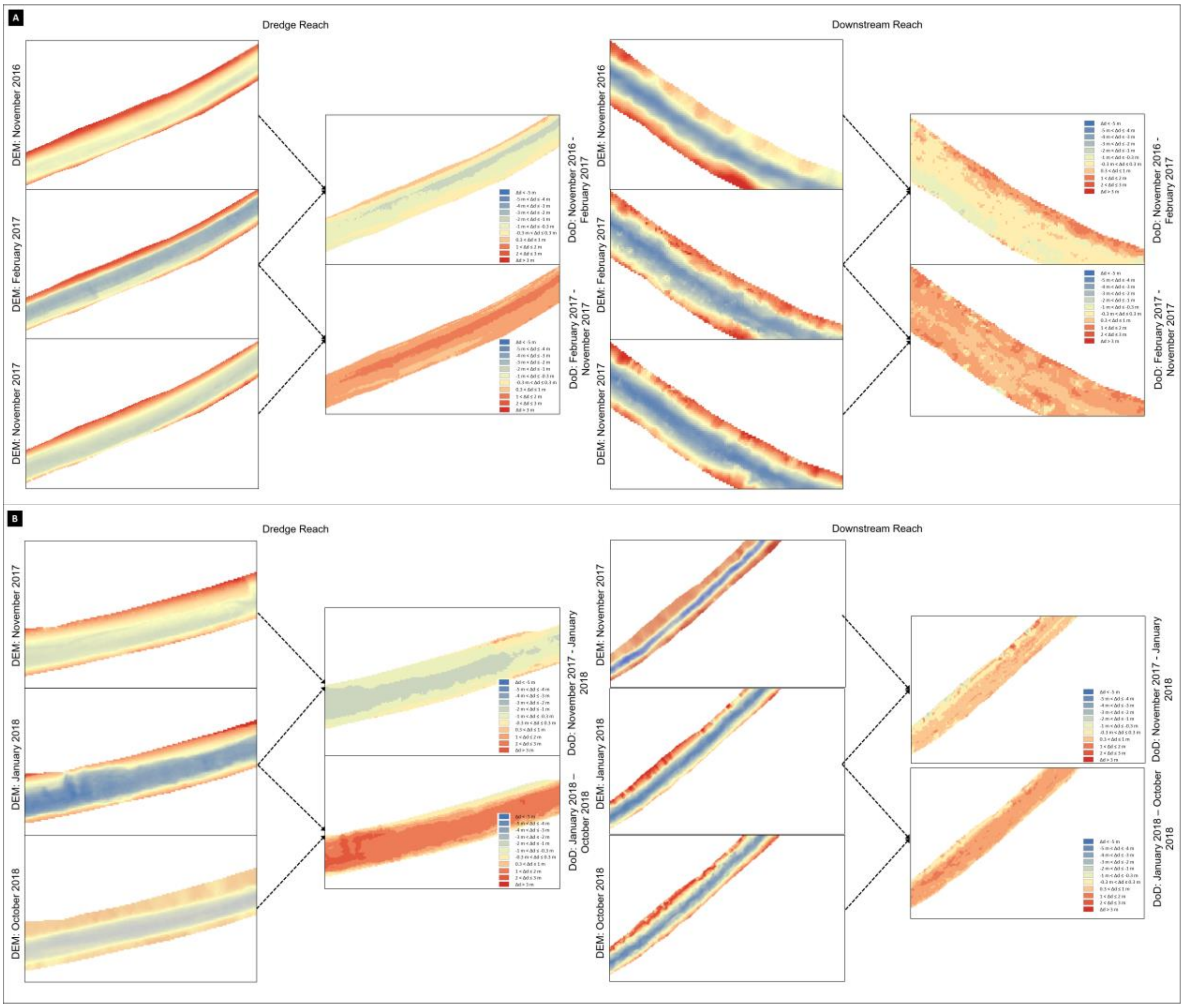


Figure 4: Surface elevation change as a percentage of the DEM surface area before and after WID 1, within (A - B) and downstream (C - D) of the dredge reach. A and C correspond to pre- vs immediately post-dredging comparisons whilst B and D highlight elevation change during the period of channel recovery. Presented are discrete data.

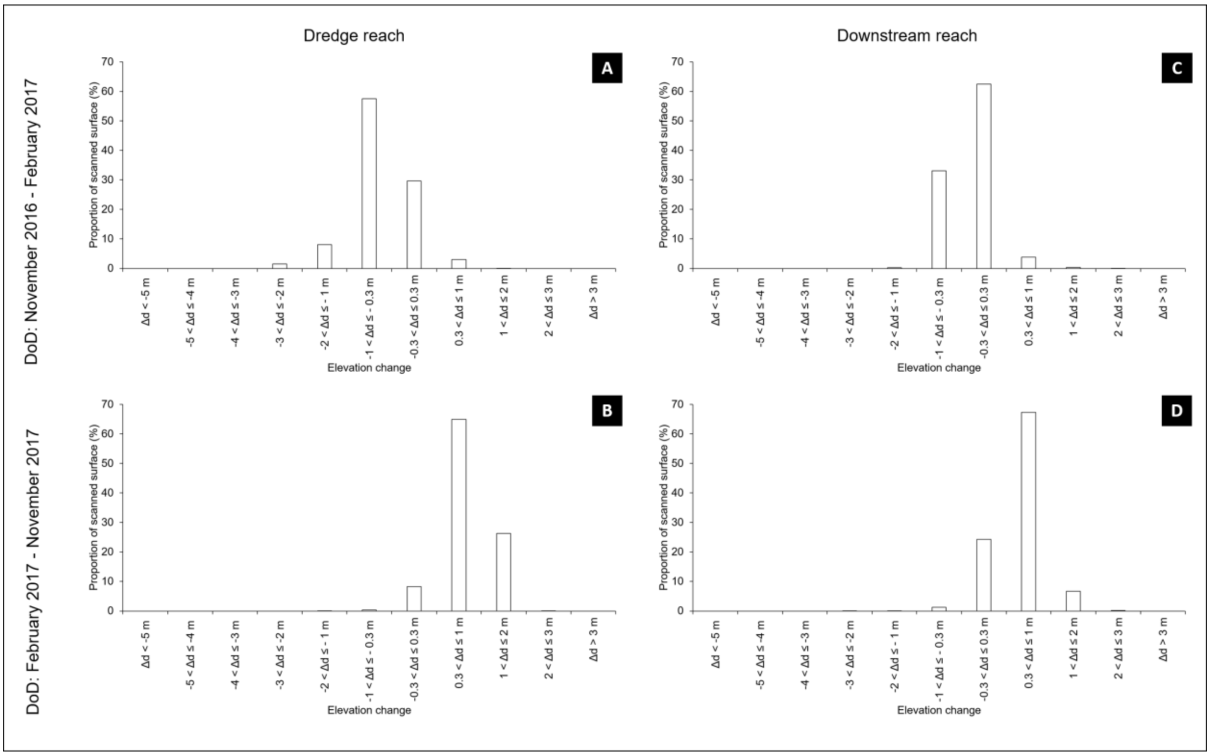


Figure 5: Surface elevation change as a percentage of the DEM surface area before and after WID 2, within (A - B) and downstream (C - D) of the dredge reach. A and C correspond to pre- vs immediately post-dredging comparisons whilst B and D highlight elevation change during the period of channel recovery. Presented are discrete data.

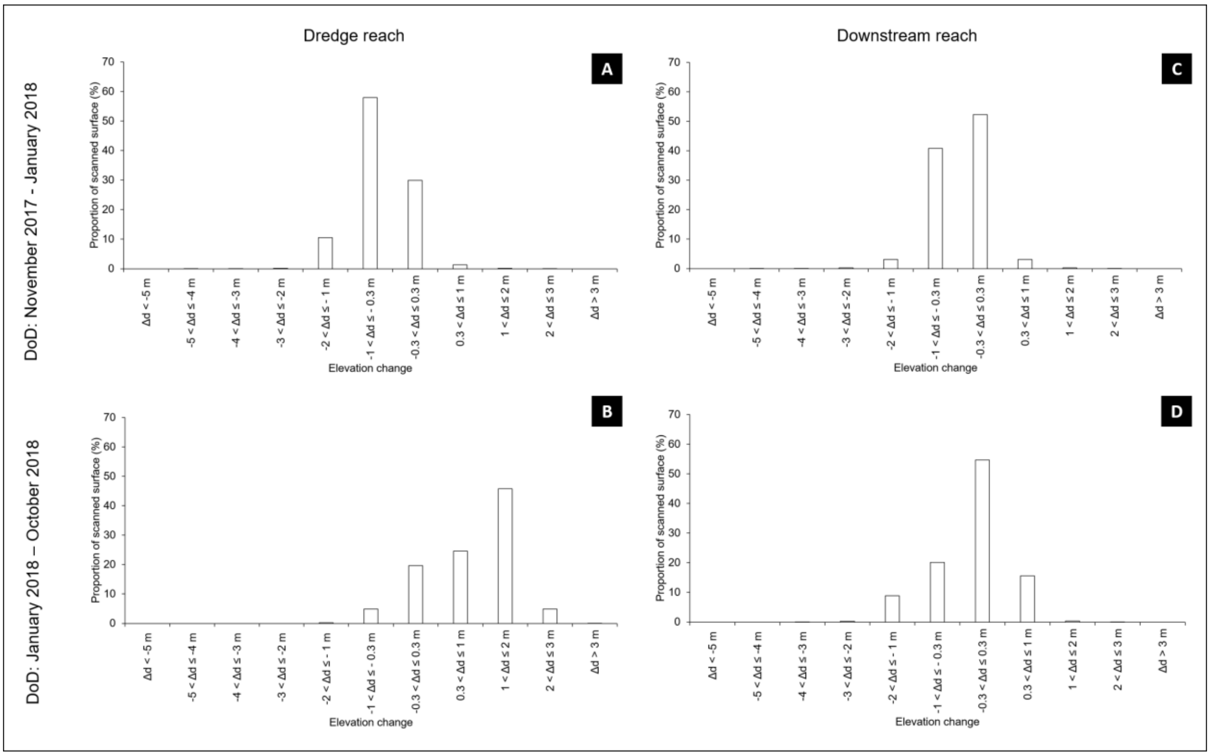


Figure 6: Volumetric change of bed sediment through time within the dredge reach and downstream of the dredge reach following (A) WID 1 and (B) WID 2. Presented are discrete data.

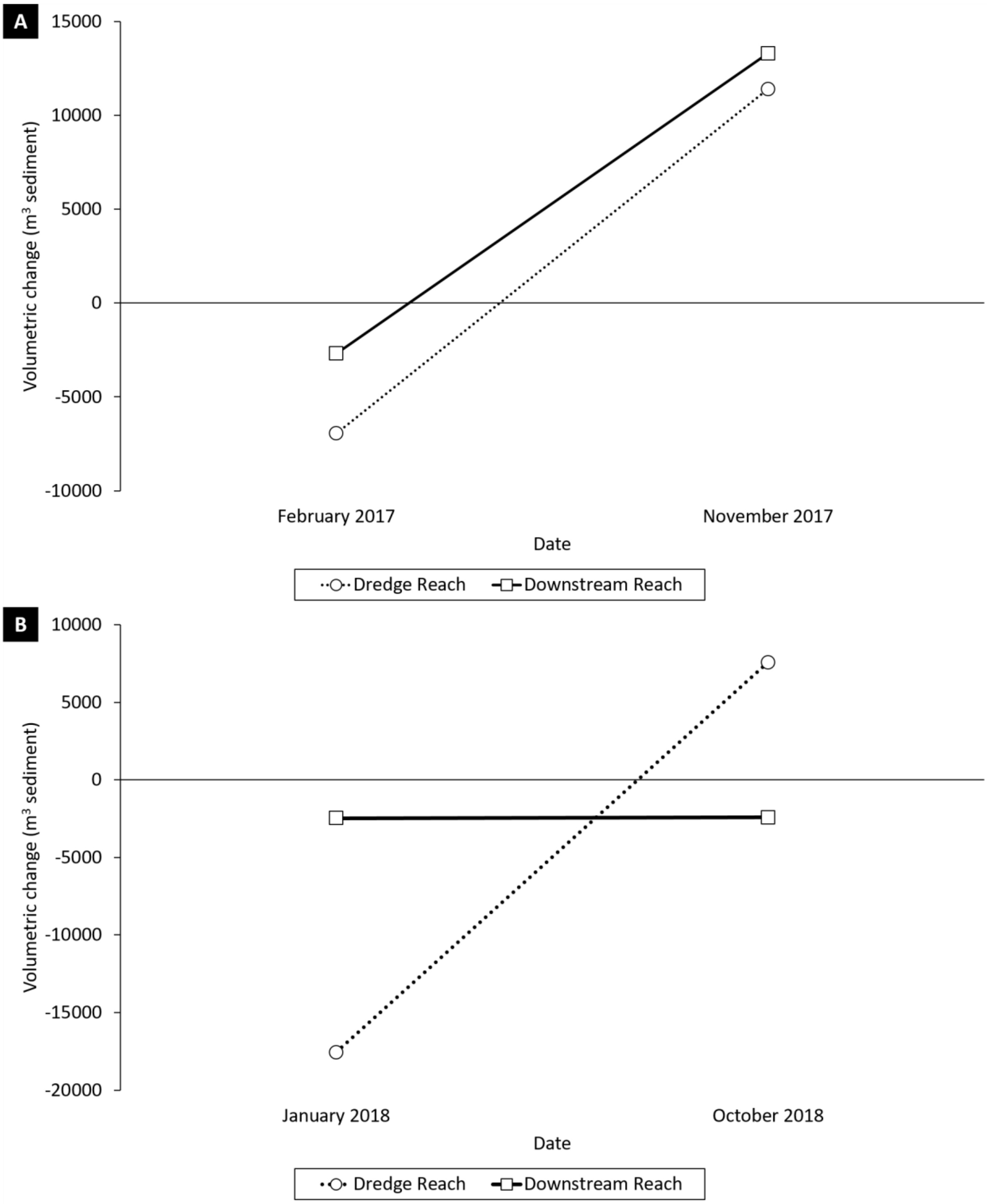


Figure 7: Turbidity recorded between 20th October 2016 and 16th January 2017 (WID 1), with grey regions indicating periods of dredging. Figure 7A shows the difference in turbidity between sites recorded up and downstream of the dredge reach. Turbidity traces are shown in Figure 7B, with the upstream site shown in red and the downstream in black. In Figure 7C, the trace represents water depth and highlights tidal effects on turbidity.

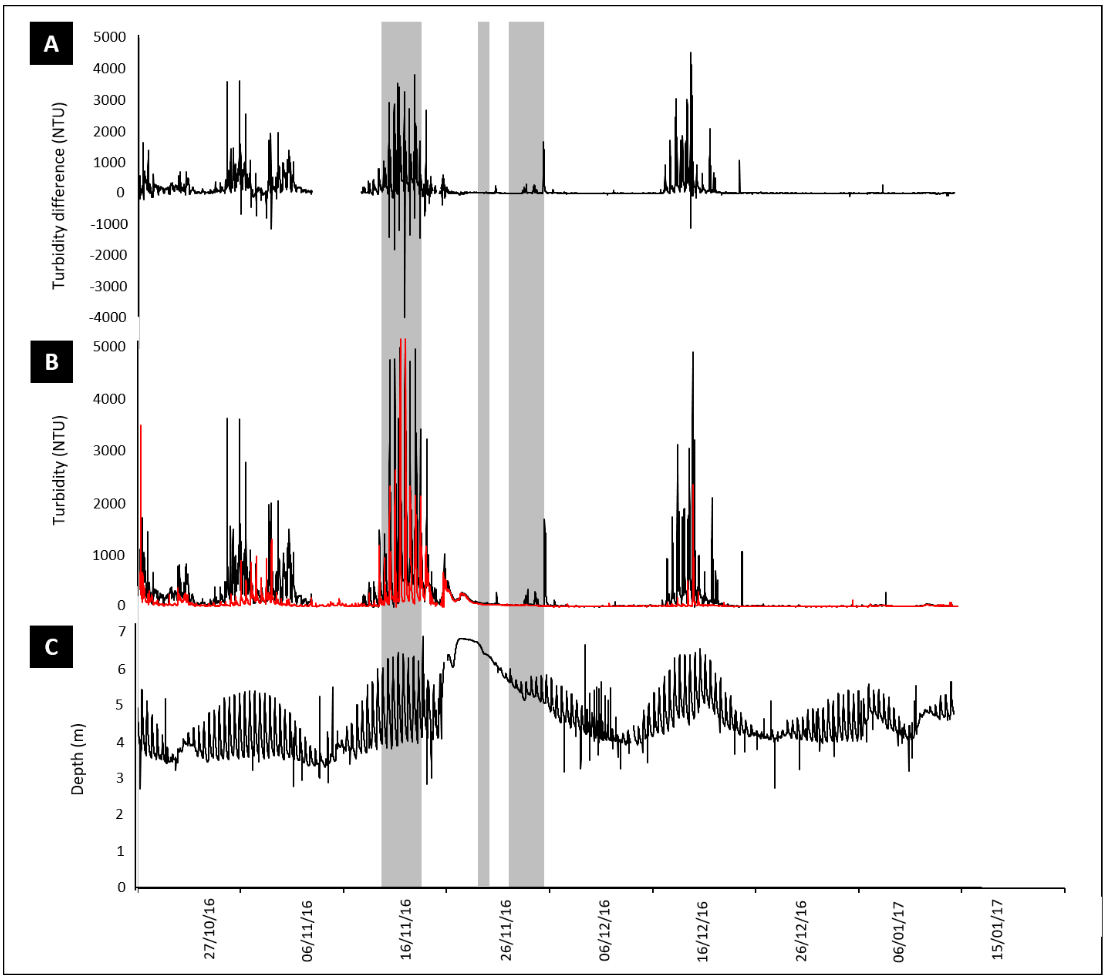


Figure 8: Water physiochemistry between 20th October 2016 and 16th January 2017 (WID 1) with dredging periods shown in grey. Figure 8A indicates the difference in oxygen concentration (%) between up and downstream sites and Figure 8B shows difference in conductivity.

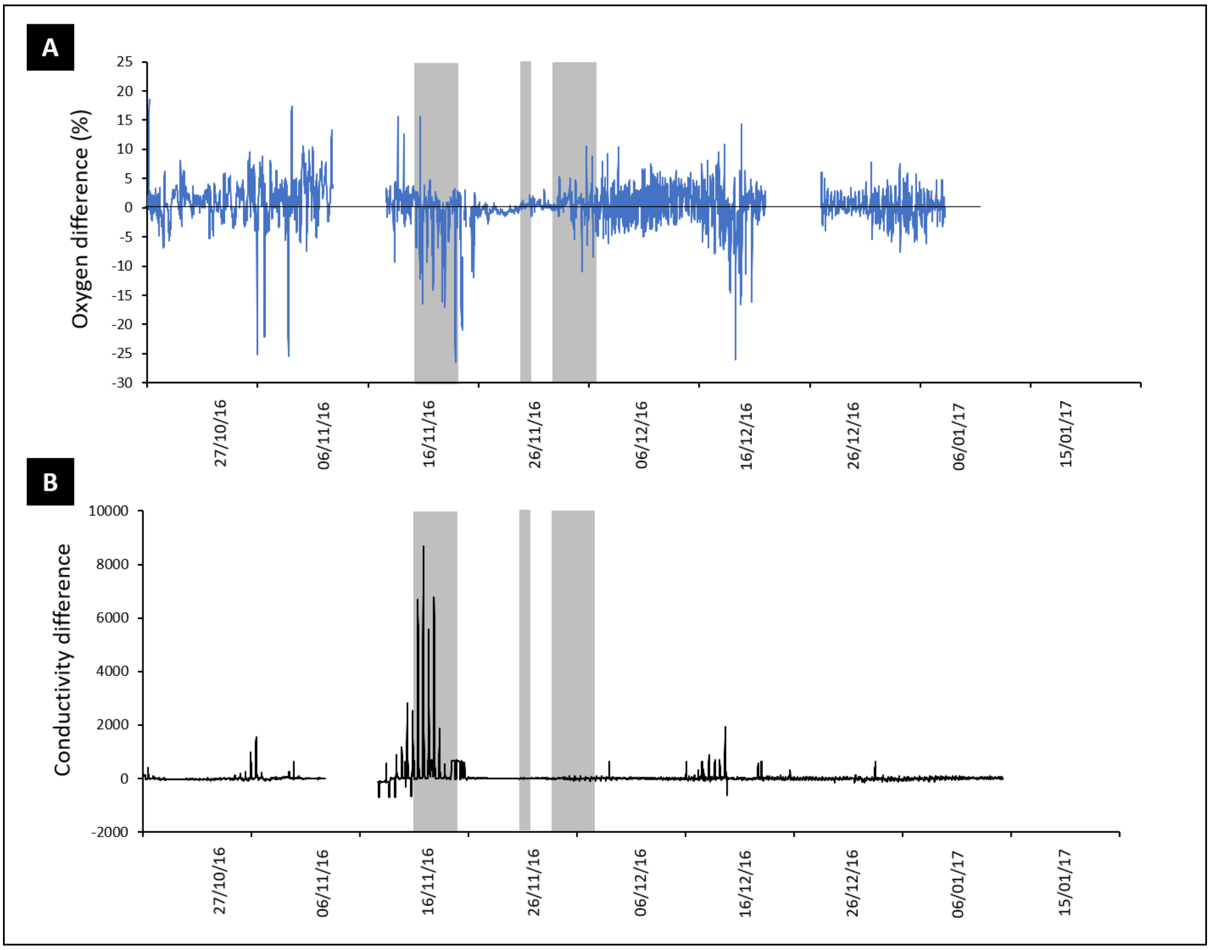
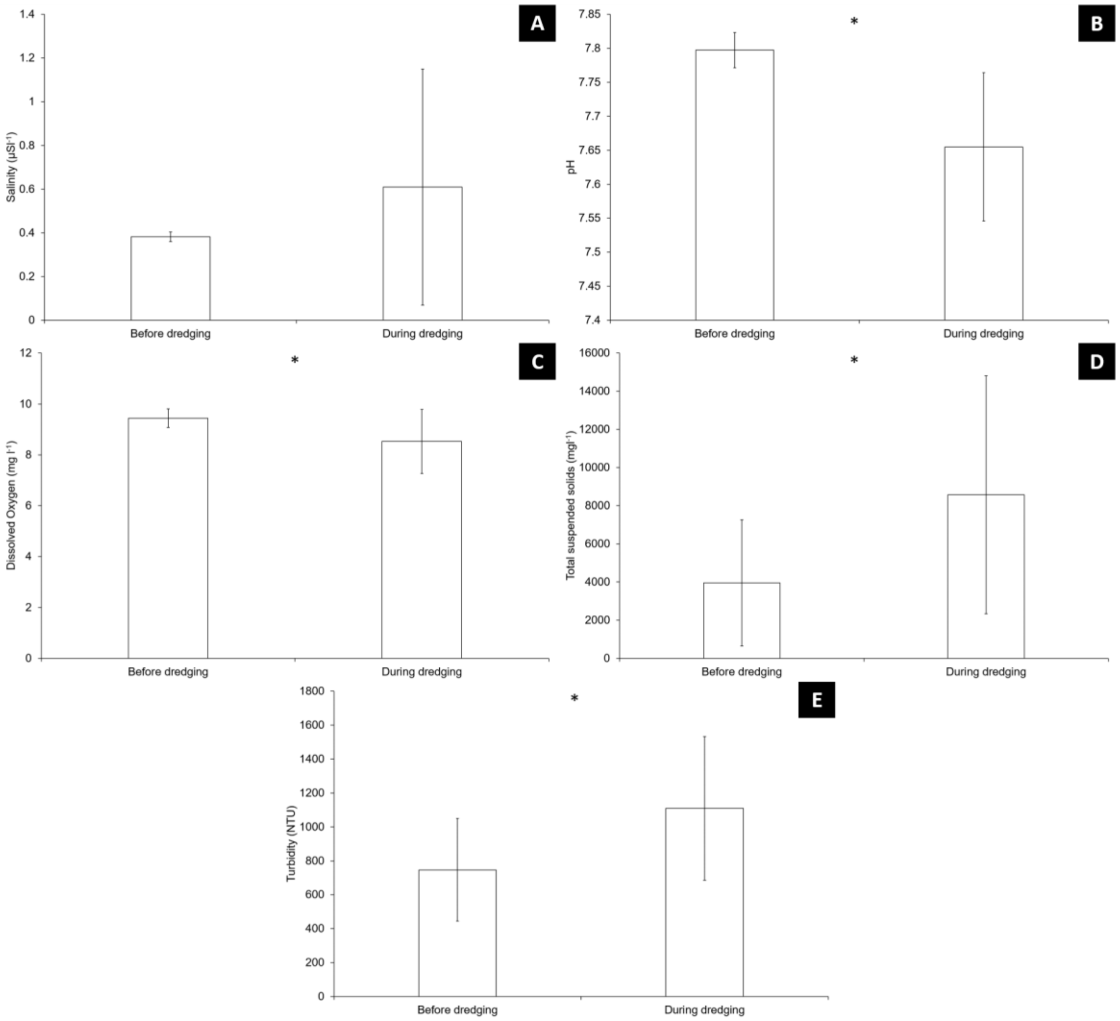


Figure 9: Water physicochemistry within the dredge reach before and during WID 1. Data derive from depth integrated sampling. Values represent means (±SD) and an asterisk above bars indicates the difference between pre- and during dredging values is significant (α = 0.05).



Tables

Table 1: Mean, maximum and minimum bed surface elevations and standard deviations of bed surface elevations through time within and downstream of the dredge reach, in response to WID 1 and WID 2.

	Reach	Date	Mean (masl)	Maximum (masl)	Minimum (masl)	Standard Deviation (masl)
WID 1	Dredge	16/11/2016	2.90	5.68	0.59	0.84
	Dredge	14/02/2017	2.90	4.95	1.75	0.71
	Dredge	18/10/2017	3.68	6.22	2.38	0.65
	Downstream	16/11/2016	3.41	5.26	2.34	0.46
	Downstream	14/02/2017	2.70	5.28	0.41	0.90
	Downstream	18/10/2017	3.20	6.55	0.73	0.93
WID 2	Dredge	18/10/2017	3.59	6.47	1.51	0.64
	Dredge	08/01/2018	3.09	6.00	0.82	0.80
	Dredge	24/10/2018	3.34	5.11	1.46	0.46
	Downstream	18/10/2017	2.76	5.84	0.73	0.95
	Downstream	08/01/2018	2.47	6.10	0.19	0.89
	Downstream	24/10/2018	2.33	3.49	0.79	0.50

Table 2: D_{10}, D_{25}, D_{50}, D_{75}, D_{90} and mean, mode, skewness and kurtosis metrics for pre- and post-WID 2 conditions at individual sites within dredge (n = 3) and downstream (n = 2) reaches, and details of statistical tests comparing differences between the two. Values represent pre- and post-dredging site means ± SD (n = 6 and n = 3, respectively).

Reach	Pre-dredging	Post-dredging	Test	Test result	P value	Pre-dredging	Post-dredging	Test	Test result	P value	Pre-dredging	Post-dredging	Test	Test result	P value
Dredge	21.575 ± 21.171	9.667 ± 1.891	Kruskall Wallace	2.400	0.121	35.083 ± 20.513	24.633 ± 3.010	Kruskall Wallace	0.824	0.364	49.583 ± 21.335	39.500 ± 3.470	Kruskall Wallace	0.600	0.439
Dredge	10.036 ± 2.985	9.613 ± 1.994	One-way Anova	0.048	0.833	24.225 ± 4.246	25.300 ± 2.987	One-way Anova	0.150	0.710	42.007 ± 5.080	41.633 ± 3.232	One-way Anova	0.013	0.912
Dredge	21.813 ± 24.160	13.193 ± 2.790	Kruskall Wallace	0.000	1.000	37.700 ± 23.149	30.367 ± 1.877	Kruskall Wallace	0.067	0.796	57.333 ± 22.246	50.333 ± 3.062	Kruskall Wallace	0.067	0.796
Downstream	11.247 ± 4.174	30.170 ± 34.740	One-way Anova	2.005	0.200	27.500 ± 8.143	4.567 ± 33.72	One-way Anova	1.564	0.251	48.150 ± 9.791	62.267 ± 34.439	One-way Anova	0.978	0.356
Downstream	14.488 ± 4.113	14.933 ± 0.757	Kruskall Wallace	0.067	0.795	35.500 ± 7.158	30.933 ± 1.069	Kruskall Wallace	2.400	0.121	59.717 ± 10.347	49.633 ± 3.754	Kruskall Wallace	2.400	0.121

	D75 (mm)					D90 (mm)					Mean (mm)				
Reach	Pre-dredging	Post-dredging	Test	Test result	P value	Pre-dredging	Post-dredging	Test	Test result	P value	Pre-dredging	Post-dredging	Test	Test result	P value
Dredge	66.933 ± 21.101	55.967 ± 6.007	Kruskall Wallace	1.067	0.302	90.750 ± 16.374	6.867 ± 14.27	One-way Anova	1.544	0.254	44.125 ± 22.754	31.710 ± 4.125	Kruskall Wallace	1.667	0.197
Dredge	78.590 ± 40.366	60.400 ± 4.757	Kruskall Wallace	0.267	0.606	164.730 ± 151.92	33.067 ± 8.956	Kruskall Wallace	1.667	0.197	38.185 ± 11.443	33.180 ± 3.571	One-way Anova	0.516	0.496
Dredge	81.467 ± 20.018	76.500 ± 5.797	Kruskall Wallace	0.067	0.796	112.083 ± 14.800	11.333 ± 9.01	One-way Anova	0.006	0.939	49.490 ± 24.723	41.677 ± 3.007	Kruskall Wallace	0.267	0.606
Downstream	75.283 ± 12.551	83.467 ± 33.604	One-way Anova	0.308	0.596	110.900 ± 18.545	10.633 ± 25.85	One-way Anova	0.000	0.986	39.118 ± 8.695	56.360 ± 36.684	One-way Anova	1.356	0.282
Downstream	89.933 ± 16.817	74.100 ± 5.730	Kruskall Wallace	1.667	0.197	122.233 ± 23.233	06.000 ± 3.46	One-way Anova	1.355	0.283	47.758 ± 8.927	41.853 ± 1.746	Kruskall Wallace	1.667	0.197

	Mode (mm)					Skewness					Kurtosis				
Reach	Pre-dredging	Post-dredging	Test	Test result	P value	Pre-dredging	Post-dredging	Test	Test result	P value	Pre-dredging	Post-dredging	Test	Test result	P value
Dredge	53.870 ± 25.831	44.400 ± 2.356	Kruskall Wallace	0.000	1.000	-1.529 ± 0.615	-1.760 ± 0.165	One-way Anova	0.386	0.554	3.852 ± 2.474	3.670 ± 0.459	One-way Anova	0.003	0.957
Dredge	45.080 ± 1.666	47.250 ± 2.581	Kruskall Wallace	2.000	0.157	-1.151 ± 0.579	-1.718 ± 0.103	One-way Anova	2.653	0.147	2.301 ± 1.437	3.502 ± 0.540	One-way Anova	1.852	0.216
Dredge	64.668 ± 25.740	55.300 ± 5.152	Kruskall Wallace	0.073	0.786	-1.455 ± 0.512	-1.715 ± 0.127	One-way Anova	0.703	0.430	2.929 ± 1.957	4.004 ± 0.912	One-way Anova	0.778	0.407
Downstream	55.878 ± 10.184	69.803 ± 41.172	One-way Anova	0.694	0.432	-1.558 ± 0.281	-1.141 ± 0.623	Kruskall Wallace	1.667	0.197	3.152 ± 1.393	1.888 ± 2.094	One-way Anova	1.210	0.308
Downstream	69.592 ± 13.034	53.81 ± 7.474	Kruskall Wallace	3.379	0.066	-1.831 ± 0.218	-1.807 ± 0.038	One-way Anova	0.033	0.862	4.223 ± 1.187	4.657 ± 0.319	One-way Anova	0.363	0.566

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