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The effects of water injection dredging on low-salinity estuarine ecosystems: Implications for fish and macroinvertebrate communities

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A B S T R A C T

Subaqueous dredging is a management activity undertaken globally to improve navigation, remove contaminants, mitigate flood risk and/or generate aggregate. Water Injection Dredging (WID) is a hydrodynamic technique involving the turbation and downstream displacement of fine sediments using vessel-mounted water jets. Despite the technique being widely applied internationally, the environmental and ecological effects of WID are poorly understood. For the first time, this study used a Before-After-Control-Impact (BACI) experimental design to assess the effects of WID on water physicochemistry, and macroinvertebrate and fish communities within a 5.7 km-long reach of tidal river. WID targeted the central channel (thalweg) to avoid disturbance of the channel margins and banks. Mean but not peak turbidity levels were substantially elevated, and dissolved oxygen levels were reduced during periods of WID, although effects were relatively short-lived (≈3 h on average). Dredging resulted in significant reductions in benthic macroinvertebrate community abundance (particularly taxa that burrow into fine sediments), taxonomic richness and diversity. In contrast, minor changes were detected in marginal macroinvertebrate communities within and downstream of the dredged reach following WID. Reductions in fish taxonomic richness and diversity were recorded downstream of the dredged reach most likely due to behavioural avoidance of the sediment plume. No visibly stressed or dead fish were sampled during dredging. Results suggest that mobile organisms and marginal communities were largely unaffected by thalweg WID and that the technique represents a more ecologically sensitive alternative to traditional channel margin mechanical dredging techniques.

1. Introduction

Subaqueous dredging is an internationally ubiquitous engineering practice (Pledger et al., 2020) used to mitigate flood risk (Gob et al., 2005), remove contaminants (Bormans et al., 2016; Gustavson et al., 2008; Chen et al., 2018), generate aggregates and/or improve navigation (Van Maren et al., 2015; Wenger et al., 2017; Wu et al., 2018). Several dredging technologies exist (mechanical, hydraulic and hydrodynamic), with each involving a sediment extraction/dispersal, transport and/or disposal stage (see Manap and Voulvoulis, 2016). Mechanical methods extract sediments for relocation and disposal using heavy machinery (Blazquez et al., 2001) whilst hydraulic techniques involve the extraction, transportation and disposal of benthic sediments using centrifugal pumps (Vivian et al., 2012). Hydrodynamic methods differ and involve the turbation, not extraction, of sediments using water jets, with mobilised particles transported and deposited downstream by the flow (Pledger et al., 2020).

The environmental and ecological effects of mechanical and hydraulic dredging techniques are relatively well known. Dredging alters the physical characteristics of waterbodies, changing bed morphology (Kondolf, 1994) and hydraulic conditions (Kornis and Laczay, 1988; Ellery and McCarthy, 1998), which can modify ecological communities in a variety of ways. Increased water depth and associated reduced light penetration may limit macrophyte and algae biomass and richness

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(Rivier and Seguier, 1985), with implications for nutrient dynamics (Freedman et al., 2013). Removal of aquatic habitat structure, such as macrophytes, large wood and coarse sediments (e.g. cobbles and boulders) during dredging, and the resulting homogenisation of instream habitats (Kanehl and Lyons, 1992; Kondolf, 1997; Gob et al., 2005), may result in degraded habitat quality and increased risk of organism entrainment (Drabble, 2012; Reine et al., 1998; Armstrong et al., 1982) and/or predation (Kanehl and Lyons, 1992). Through entrainment and/or other mechanisms, fish (e.g. Freedman et al., 2013; Swales, 1982) and macroinvertebrate (e.g. Brooker, 1985; Szymelfenig et al., 2006; Meng et al., 2018) populations may be modified due to reductions in abundance and/or species richness.

Dredging may also cause channel instability through the removal of gravel armouring and/or knickpoint migration (Scott, 1973; Stevens et al., 1990), with the downstream transport and deposition of fine sediments causing habitat degradation and the loss of marginal and/or shallow water habitats (Kanehl and Lyons, 1992; Rinaldi et al., 2005). Where fish spawning gravels are dredged, habitat loss and degradation through increased sedimentation (Newcombe and Macdonald, 1991; Newcombe and Jensen, 1996) can have detrimental impacts on emergence and/or recruitment (Harvey, 1986; Fudge et al., 2008; Basic et al., 2018). Macroinvertebrate trophic guild structures may also be influenced by changes in substrate composition, with reductions in collector-gatherers and algal grazers corresponding with increases in burrowing detritus-shredding taxa and decomposers, resulting in changes in predator foraging efficiency and behaviour (Brown et al., 1998; Rempel and Church, 2009; Freedman et al., 2013).

At the ecosystem level, fine sediment suspension is a primary concern and dredging operations can either intentionally (e.g. during hydrodynamic dredging; Pledger et al., 2020; Winterwerp et al., 2002) or unintentionally (e.g. through bed disturbance in the vicinity of mechanical and/or hydraulic dredgers; Collins, 1995; Mikkelsen and Pejrup, 2000; Smith and Friedrichs, 2011) cause sediment resuspension. Suspended sediment may inhibit fish behaviours, including foraging (Utne-Palm, 2002) and avoidance and alarm behaviours (Servizi and Martens, 1992; Shaw and Richardson, 2001; Sweka and Hartman, 2001), and impact fish physiology and functioning (Wenger et al., 2017). Dredging may also release contaminants (Goossens and Zwolsman, 1996; Lewis et al., 2001; Spencer et al., 2005), particularly particulate matter and pore water from the bed, which may be rich in heavy metals (van den Berg et al., 2001). Exposure to contaminants released during dredging may result in behavioural changes, physical damage, physiological and sublethal impacts and/or mortality in fish (Johnson et al., 2014; Wenger et al., 2017). Further, the sound generated by dredging operations may cause short- and long-term hearing losses in fishes and/ or modify behaviours. For example, noise caused through dredging activities may mask natural sounds used by juvenile fish to detect predators (Simpson et al., 2015).

Extracted sediment or "spoil" may be discarded or retained for use in land reclamation, restoration or development projects (Wenger et al., 2017), or sold commercially as aggregate. Depositing sediment above the water line, sometimes in mounds on the bankside, can have negative consequences for biota. For example, a range of organisms including macrophytes, amphibians (Grygoruk et al., 2015), fish (Aldridge, 2000; Grygoruk et al., 2015) and macroinvertebrates (Aldridge, 2000; Grygoruk et al., 2015; Killeen et al., 1998) can be removed and deposited in spoil heaps during extraction activities resulting in decreased abundances and loss of species unable to migrate back into the water (Szymelfenig et al., 2006).

While there is data available regarding the environmental impacts of traditional mechanical and hydraulic dredging techniques, there is limited information on those utilising sediment release mechanisms. One such technique is Water Injection Dredging (WID hereafter), a form of hydrodynamic dredging (van Raalte and Bray, 1999) where fine sediments are mobilised using vessel-mounted low-pressure water jets and transported downstream under ambient flows. WID avoids the need for spoil disposal, maintains the sediment balance within the system and is cheaper than dredging by extraction. The effects of this method on aquatic geomorphology and physicochemistry are partially understood (see Pledger et al., 2020), although to our knowledge, the ecological implications of the technique remain unquantified. Consequently, to develop an understanding of the effects of this technique on estuarine ecology, this paper assesses faunal community responses to changes in environmental conditions associated with WID spatially and temporally within the River Parrett estuary, South West England. The objectives of the study were to quantify the temporal responses of water physicochemistry, and macroinvertebrate (marginal and benthic) and fish communities to WID. To achieve this, we characterised: (1) water physicochemistry before, during and after dredging; (2) benthic macroinvertebrate communities from the channel centre and margins, and fish community characteristics pre- and post-dredging at sites subject to dredging, as well as sites upstream and downstream of the operations and; (3) fish mortality and health during WID operations.

2. Materials and methods

2.1. Study system

Research was conducted on a 5.7 km tidal stretch of the River Parrett, which flows for 59 km through the counties of Dorset and Somerset in Southwest England (Fig. 1A). The River Parrett has a catchment area of 1700 km^2 (Fig. 1B) draining the Quantock, Blackdown and Mendip Hills before entering Bridgwater Bay (Environment Agency, 2009). The lower River Parrett has a legacy of dredging, and is constrained by artificial levees, primarily for flood defence purposes. The need for management is largely a function of the river's tidal nature for some 18 km upstream of the River Severn Estuary, with significant quantities of sand, silt and clay delivered from the tidal environment on high spring tides, impacting turbidity (Pledger et al., 2020) and resulting in increased sedimentation and aggradation. The upper reaches of the Parrett estuary, including between Burrowbridge and Westonzoyland pumping station, lie above the normal zone of saline water intrusion (Ambios, 2017), with salinity increasing with decreasing distance from Bridgwater Bay. Historically, the effects of fine sediment accumulation on flood water conveyance have been mitigated through dredging – using either mechanical bank- or vessel-based technologies such as excavators with buckets or draglines. These methods can be ecologically damaging (Brooker, 1985), particularly to bankside habitats and need to be undertaken frequently to mitigate high rates of sediment accumulation (Pledger et al., 2020), prompting questions over the suitability of dredging by extraction for the River Parrett estuary (HR Wallingford, 2016). Thus, in the winter of 2017 the Somerset Rivers Authority and Somerset Drainage Boards Consortium commissioned a hydrodynamic dredging trial using WID, an understudied dredging technology, in an effort to find an ecologically sensitive alternative to excavation dredging.

2.2. Dredging activities

Dredging was completed 3rd–9th December 2017 by Van Ord UK Ltd using Borr, a 18.73 × 5.32 m inland water vessel capable of dredging to a maximum depth of 14.00 m. This dredging program and vessel are described in detail by Pledger et al. (2020). During the operation, WID was targeted at the channel centre in a 6 m swathe along a 5.7 km stretch of the River Parrett (Fig. 1C). Dredging occurred on each ebb tide during the 7-day dredging period (dredging period hereafter) and on each occasion, dredging ceased when river levels prevented safe navigation. A total of 41 h and 5 min of dredging were completed during the campaign. The environmental and biological monitoring campaign spanned this period to gain a better understanding of the effects of the operation on water physicochemistry, and macroinvertebrate and fish community characteristics.

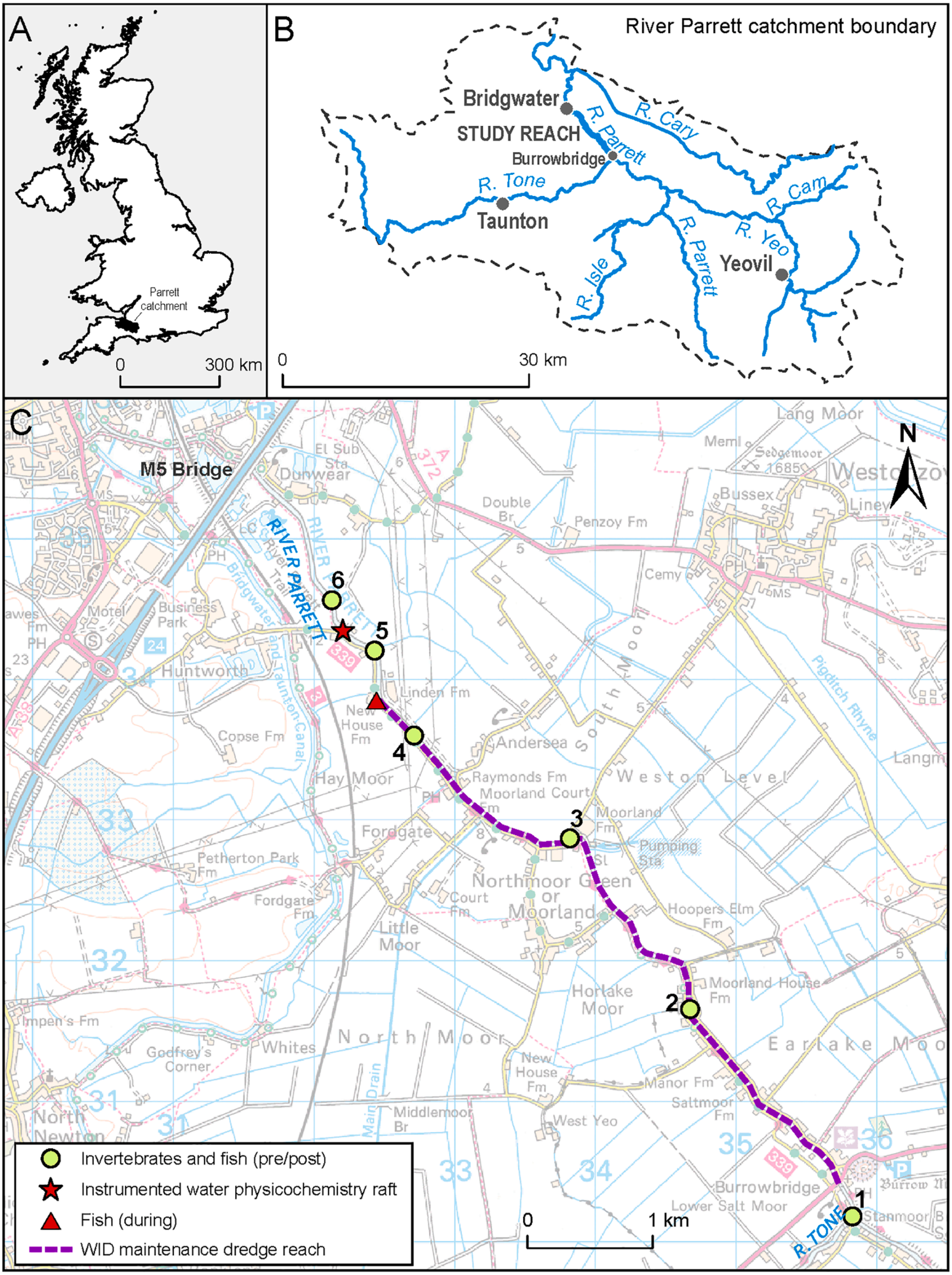

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Fig. 1. (A) Location and (B) catchment of the River Parrett and (C) details of the experimental reaches where ecological and water physicochemistry monitoring took place. The entire experimental stretch in Fig. 1C is highlighted bold in Fig. 1B. .
Adapted from Pledger et al. (2020)

2.3. Characterising differences in water physichochemistry associated with WID activities

Time series of water physicochemistry were collected during the WID maintenance dredge. A raft fitted with a SI 6600V2 Sonde (water physicochemistry raft hereafter) was anchored at the downstream end of the dredge reach at Westonzoyland (Fig. 1c). The water physicochemistry raft was installed 30th Nov 2017 and retrieved 10th Dec 2017. Thus, the time period included approximately 4, 5 and 1.25 days of pre-, during- and post-dredging campaign data, and tidal intensity varied between days. The probe measured turbidity (NTU) and dissolved oxygen (mg l^{-1}, %) every 15 min at approximately 1 m depth. The strong tidal nature of the study site and need to avoid debris accumulation around any equipment in the channel precluded the installation of the probe on a permanent basis or closer to the bed, and should be considered when interpreting the results. Depth data for the River Parrett were recorded and provided by the Somerset Drainage Board Consortium via their Timeview DBi telemetry database.

2.4. Characterising differences in benthic and marginal macroinvertebrate community characteristics associated with WID activities

To assess dredging effects on macroinvertebrate community characteristics, a Before-After-Control-Impact (BACI) experimental design was utilised. Study sites were selected based on their proximity to the dredge reach (Fig. 1C). Sampling occurred at an upstream control site (site 1; upstream hereafter), at three sites within the dredged reach (sites 2–4; dredged hereafter) and at two sites downstream of the dredged reach (sites 5–6; downstream hereafter). The control was located between Burrowbridge and the Parrett/Tone confluence, with the confluence and changing characteristics upstream rendering further control sites inappropriate. Each site was sampled on two occasions before, one occasion immediately after, and approximately three and five months after dredging. At each site and on each occasion, benthic macroinvertebrate communities were characterised using three Ekman grab samples (≈0.15 × 0.15 × 0.15 m) from the unvegetated inter-tidal sediment surface of i) left bank, ii) right bank, and iii) channel centre using a boat at low tide. In addition, a single 3-minute sweep sample was collected from the submerged vegetated left bank at each site and on each occasion, to characterise any dredging operation effects on marginal macroinvertebrate communities. Each sampling location was submerged under low water conditions and marked with a wooden stake, allowing for repeated sampling in the same area to provide consistent comparison over time. Macroinvertebrate samples (Ekman and sweep) were preserved in 4% aqueous formaldehyde solution in the field for subsequent processing.

In the laboratory, macroinvertebrates derived from Ekman (benthic macroinvertebrates hereafter) and sweep (marginal macroinvertebrates hereafter) samples were passed through a 1 mm mesh sieve and identified to the lowest practical taxonomic level-typically species or genus level with the exception of some taxa which were identified to family level (Palaemonidae, Sphaeriidae, Zonitidae, Baetidae cf., *Cleon* sp., and some Diptera families) and Oligochaeta (order) which were recorded as such.

2.5. Characterising differences in fish community characteristics, mortality and health associated with WID activities

Dredging effects on fish community characteristics were assessed within the BACI design at an upstream control site (site 1; upstream), a site within the dredged reach (site 3; dredged) and a site downstream of the dredged reach (site 6; downstream), on multiple occasions before and after dredging. Each site was sampled at the same time intervals as macroinvertebrates (two occasions before, one occasion immediately after and three and five months after dredging). At each site and on each occasion, a 50 × 5 m seine net was used to sample both slack tides which marked the transitions between spring flood and ebb tides and spring ebb and flood tides. Collected fish samples were held in separate tanks of oxygenated river water before processing when individuals were speciated.

To quantify effects of WID on fish, assessments of mortality and health were completed at a single site downstream of the dredge reach during WID (Fig. 1). Sampling was achieved via pelagic trawling (13 × 2 m net) from a boat and all captured fish were recovered in tanks of aerated water before being speciated and examined for signs of respiratory stress (gulping at the surface), damage and mortality. Dredging typically occurred twice daily, and sampling occurred on one of these occasions per day throughout the dredging program (providing temporal replication as spatial replication was not feasible due to the time required to undertake trawls and ensure fish welfare). Recovered fish were returned to the river channel upstream of the dredge site.

3. Data analysis

3.1. Analysing differences in water physicochemistry associated with WID activities

Pre-, during- and post-dredging mean and peak values were calculated/extracted from turbidity and dissolved oxygen time series data and used to investigate effects of WID on water physicochemistry.

3.2. Management of ecological data

The data analysis approach employed was consistent between faunal groups (see Sections 3.3 and 3.4), although there were some minor differences in data management prior to analysis. For macroinvertebrates, data from Ekman grab samples from each site were pooled (left and right bank and channel centre samples) for each occasion to provide a single sample unit; although this reduced the total sample size it provided a direct comparison with the single channel margin sample and removed issues associated with pseudo-replication. For fish (seine), ebb and flood samples were also combined for each site to form a single sample unit for analysis for each occasion. Total abundance, taxonomic richness, Shannon-Wiener diversity index (species diversity, S1; Shannon and Wiener, 1949) and Berger Parker dominance index (species evenness, S1; Berger and Parker, 1970) (metrics hereafter) values were calculated for each organism unit (fish and benthic and marginal macroinvertebrates) for use in subsequent statistical analyses (section 3.3). Metrics were calculated using Species Diversity and Richness IV software (Pisces Conservation, 1998).

3.3. Analysing differences in macroinvertebrate and fish community characteristics associated with WID

Univariate general linear models (GLM) were used to identify how metrics of macroinvertebrate (benthic and marginal) and fish communities responded to dredging in locations within and downstream of the dredged reach relative to a control. During analysis, "treatment" was specified as a fixed factor comprising three groups; 1) a "control" group comprising the combined pre-dredging data for all sites and the post-dredging data for Site 1; 2) a "dredged" group comprising post-dredging data for the dredged site(s) (benthic and marginal macroinvertebrates, sites 2–4; fish, site 3); and 3) a "downstream" group which included post-dredging data for the downstream site(s) (benthic and marginal macroinvertebrates, sites 5–6; fish, site 6). This provided 15, 9 and 6 replicates per metric for the "control", "dredged" and "downstream" benthic and marginal macroinvertebrate treatment groups, and 9, 3 and 3 replicates per metric for the "control", "dredged" and "downstream" fish treatment groups, respectively. Where significant effects were observed, differences between treatments were examined using Fisher's least significant difference post hoc tests.

Where spatial replication permitted (benthic and marginal macroinvertebrate, but not fish) and significant differences in dredged and/or downstream communities were detected due to dredging, GLMs were used to determine the persistence of dredging effects. Within the model, time/treatment was specified as a fixed factor comprising four groups; 1) the "control" group described above, and 2) "immediately after", 3) "three months after" and 4) "five months after" groups. This yielded 15, 3, 3 and 3 replicates respectively for the dredged communities and 15, 2, 2 and 2 replicates for the downstream communities. Differences between time/treatment were tested using Fisher's least significant difference post hoc tests. All univariate statistical tests were performed in IBM SPSS v23.0 (IBM Corp, 2015).

3.4. Analysing effects of WID activities on community composition of macroinvertebrate and fish communities

Differences in community composition between upstream, dredged and downstream locations were examined via a Principal Coordinate Analyses (PCoA) using the Bray-Curtis dissimilarity index and 'cmdscale' function in the 'vegan' package (Oksanen et al., 2015). Statistical differences in community composition associated with the additive explanatory factors of site, treatment (control, dredged and downstream as per 3.3) and time (occasions 1–5) were assessed via a permutational multivariate analysis of variance (PERMANOVA) using the 'adonis' function in 'vegan'. To examine where differences in community composition of the treatment groups occurred, pairwise comparisons of differences were performed using the 'pairwise.adonis' function with Bonferroni corrections applied (Arbizu, 2019). To assess if community heterogeneity varied across treatment groups, homogeneity of multivariate dispersions among assemblages were examined using the 'betadisper' function and tested for statistical differences via Tukey post hoc tests. Taxa affected by dredging activities at the dredged and downstream sites for benthic communities were identified via Similarity Percentage (SIMPER) with data coded as pre-dredging (occasions 1–2) and post dredging (occasions 3–5). Abundances were square root transformed prior to SIMPER analyses. SIMPER was conducted in PRIMER V7 (PRIMER-E Ltd, Plymouth, UK; Clarke and Gorley, 2015) and all multivariate analyses in the R environment (version 3.6.0; R Development Core Team, 2019).

3.5. Analysing differences in fish mortality and health associated with WID

Total numbers of live and dead fish caught (by species) during the dredging operation were used to investigate effects of WID on fish mortality. Numbers of fish displaying physical damage or symptoms of respiratory distress were used as measures of fish health.

4. Results

4.1. The effect of WID on water physicochemistry

Turbidity peaks were relatively unaffected by dredging and were comparable in magnitude to pre-dredge hightide peaks (Fig. 2A). However, mean turbidity was elevated during the dredging period (pre-dredging = 91 NTU; during dredging = 635 NTU; 697% increase) and did not return to pre-dredging levels during the sampling period (approximately 1.25 days after dredging). As dredging occurred during periods of high tide and because tidal strength varied through time (Fig. 2C), the effect of the dredging operations could not be separated from elevated turbidity associated with tidal effects. Thus, there is no clear trend indicating dredging elevated turbidity over tidal fine sediment dynamics. Dissolved oxygen levels dropped substantially during and immediately after dredging, although effects were relatively short-lived (≈3 h on average). Unlike turbidity, dissolved oxygen (negative) peaks did not occur during high tides without dredging, indicating a WID effect. Oxygen levels declined and remained consistently low throughout the dredging period, relative to pre-dredging conditions (before = 11.45 mg/l; during = 10.14 mg/l; 11% decrease). It should be noted that sampling occurred near the water surface and will almost certainly have underestimated WID effects on turbidity and oxygen concentration. Although the results probably underestimated the effect of dredging activities on dissolved oxygen within the water column (below the surface layers where the probe was located), the consistent reduction compared to non-dredging periods clearly demonstrates a WID effect.

4.2. The effect of WID on benthic and marginal macroinvertebrate community characteristics

During the pre-dredging surveys, the total abundance of macroinvertebrates from benthic (Ekman) and marginal (sweep) samples were 384 and 956, representing 10 and 15 species, respectively. Benthic communities were dominated by Oligochaeta, Chironomidae, Sphaeriidae and *Potamopyrgus antipodarum*, which collectively comprised ≈92% of communities. *Gammarus zaddachi* were ubiquitous within marginal samples and comprised ≈96% of populations.

Dredging resulted in significant differences in benthic macroinvertebrate communities within and downstream of the dredged reach relative to the control group. Community abundance varied between treatment groups (GLM $F_{2,29}$ = 16.234, P = <0.001; Fig. 3A), with dredged and downstream means being quantitatively similar but significantly lower than the control mean. Taxonomic richness varied between treatment groups (GLM $F_{2,29}$ = 7.939, P = 0.002; Fig. 3B), with dredged and downstream values being significantly lower than control. Significant differences were observed between groups for the Shannon-Wiener index (GLM $F_{2,29}$ = 5.948, P = 0.007; Fig. 3C), with dredged and downstream samples being significantly different to those from the control group; dredged and downstream groups were not significantly different. Significant differences between groups were detected for the Berger-Parker dominance index (GLM $F_{2,29}$ = 4.163, P = 0.027; Fig. 3D), with control and dredged samples being significantly different, but downstream and control, and downstream and dredged not displaying any difference.

Within the dredged reach, community abundance (GLM $F_{3,23}$ = 25.357, P < 0.001; Fig. 4A) and taxonomic richness (GLM $F_{3,23}$ = 7.619, P = 0.001; Fig. 4B) had recovered to control levels after five months, whereas Shannon-Weiner (GLM $F_{3,23}$ = 7.313, P = 0.002; Fig. 4C) and Berger-Parker (GLM $F_{3,23}$ = 3.3595, P = 0.032; Fig. 4D) took three months to recover. Downstream communities followed a similar pattern with abundance (GLM $F_{3,20}$ = 14.396, P < 0.001; Fig. 4E), taxonomic richness (GLM $F_{3,20}$ = 4.160, P = 0.022; Fig. 4F) and Shannon-Weiner (GLM $F_{3,20}$ = 3.902, P = 0.027; Fig. 4G) being comparable to control samples after five months.

Differences in macroinvertebrate community composition were observed and were associated with treatment (control, dredged, downstream) but not sample site or time (Table 1). Principal Coordinate Analysis (PCoA) highlighted clear differences in macroinvertebrate communities between the control group and those affected by dredging (dredged and downstream) on axis 1 (Fig. 5a). These differences were statistically significant (pairwise PERMANOVA both P = 0.003; Table 2). Dredged assemblages were more heterogeneous (average multivariate dispersion distance = 0.423) than control assemblages (average distance = 0.255), with downstream assemblages (average distance = 0.349) also displaying higher values than control assemblages, although these differences were not significant (P > 0.05; Tables 3 and 4). SIMPER indicated that reductions in three Mollusca taxa (*Potamopyrgus antipodarum*, Sphaeriidae and *Radix balthica*), Chironomidae and Oligochaeta were responsible for differences between pre- and post-dredging assemblages for the dredged and downstream groups (Table 5; average pre vs post dissimilarity of communities 62.89% and 53.66%, respectively). Of particular note, *P. antipodarum* appeared to be severely affected with no individuals recorded within or downstream of

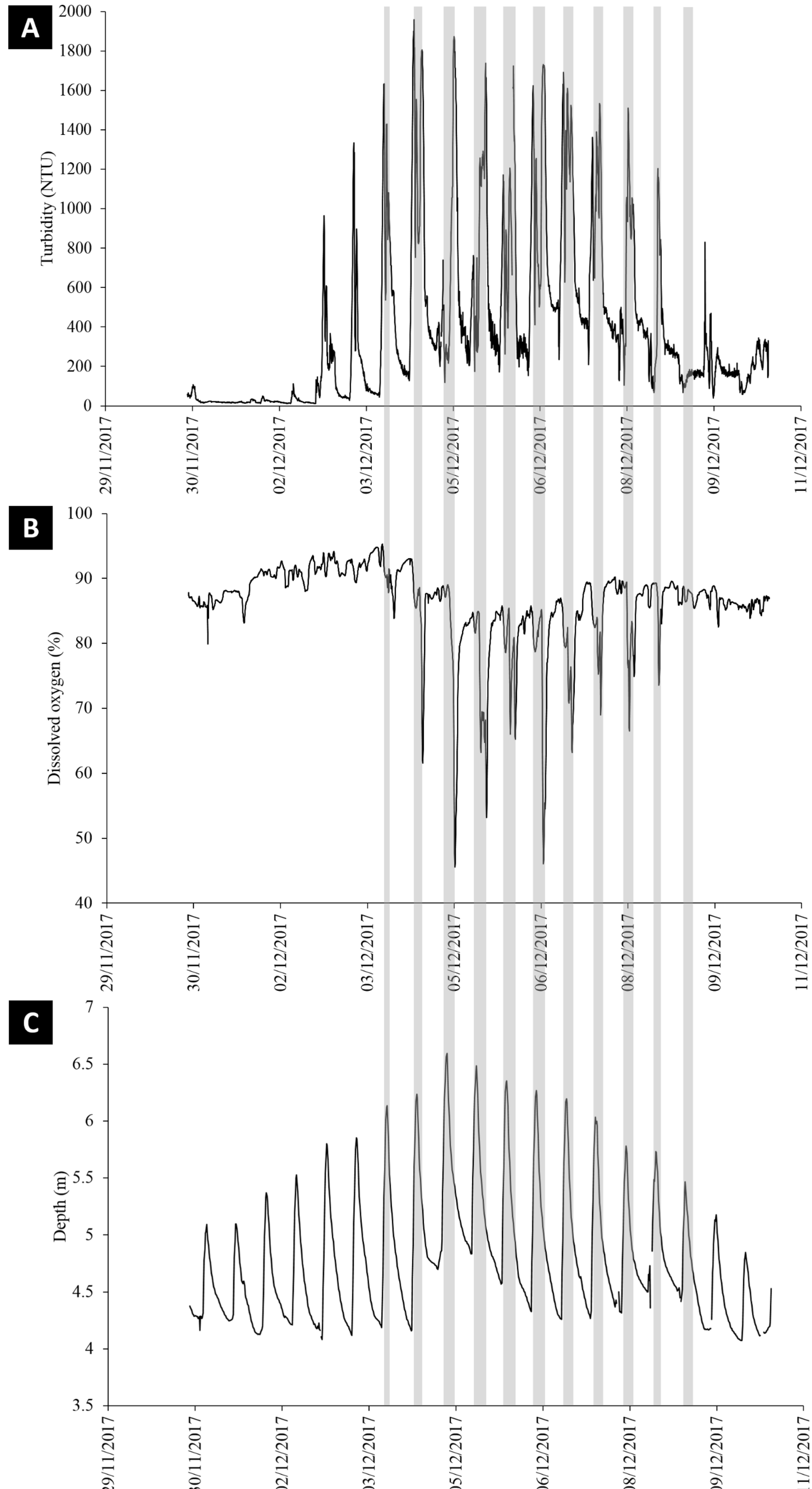

Fig. 2. (A) Turbidity, (B) dissolved oxygen and (C) depth recorded between 30th November 2017 and 10th December 2017, with grey regions indicating periods of dredging. Data in (C) derive from TimeviewDbi (Somerset DBC, 2017).

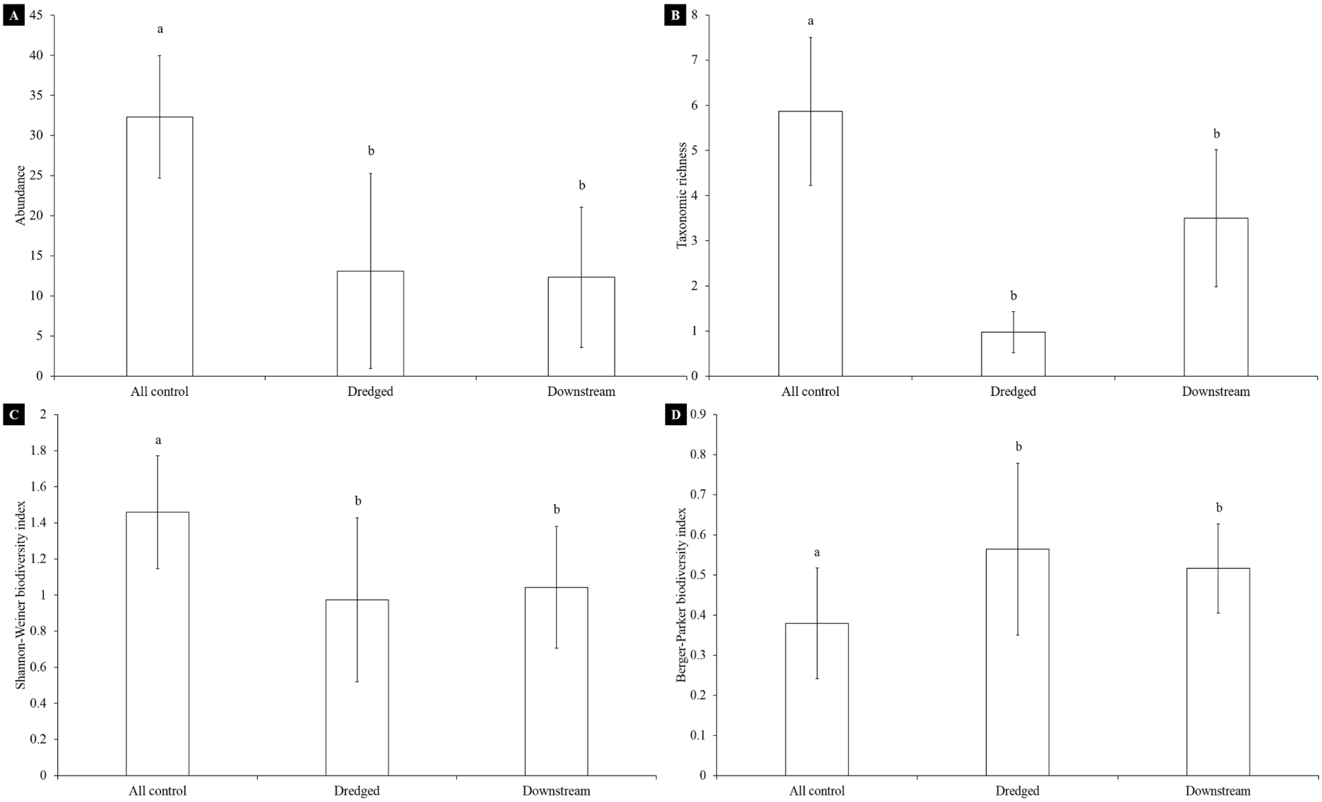

Fig. 3. Benthic macroinvertebrate, **A** abundance, **B** taxonomic richness, **C** Shannon-Wiener biodiversity index and **D** Berger-Parker biodiversity index data per treatment group. Presented are treatment means (control, n = 15; dredged, n = 9; downstream, n = 6; ±STDEV) and letters above bars indicate no statistical differences.

the dredged reach (sites 2–6) post-dredging. When temporal recovery was visually examined, benthic community composition in the dredged and downstream groups had recovered and mirrored that seen in the control group 5 months post dredging (Fig. 5a).

Marginal macroinvertebrate communities were negligibly affected by dredging, with no significant differences in metrics detected between groups (control, dredged and downstream) for abundance (GLM $F_{2,29}$ = 1.124, P = 0.340; Fig. 6A), taxonomic richness (GLM $F_{2,29}$ = 1.036, P = 0.369; Fig. 6B) or Shannon-Wiener diversity (GLM $F_{2,29}$ = 1.998, P = 0.155; Fig. 6C). Significant differences were observed between treatment groups for the Berger Parker dominance index (GLM $F_{2,29}$ = 5.011, P = 0.082; Fig. 6D), with control and downstream groups being similar and dredged being statistically different to both downstream and control groups. Dredged site Berger-Parker data were temporally variable (GLM $F_{3,23}$ = 5.544, P = 0.006; Fig. 7), with no significant difference detected between the control and immediately after groups; but significant differences between the three months after and control and five months after and control groups were recorded.

Differences in marginal macroinvertebrate community composition were associated with treatment (control, dredged and downstream) and time but not site, although site explained the greatest amount of variance (Table 1). Principal Coordinate Analysis (PCoA) indicated that marginal assemblages displayed a similar pattern to benthic communities with axis 1 highlighting an effect of dredging on marginal communities (Fig. 5b). Control communities were significantly different to dredged site communities (Table 2) and demonstrated greater heterogeneity (average distance = 0.610) than dredged (average distance = 0.393) and downstream communities (average distance = 0.495), although differences were not statistically significant (Tables 3 and 4). When temporal recovery was visually examined, marginal community composition in the dredged and downstream groups showed some evidence that community composition had recovered at some sites 3–5 months post-dredging, whilst some sites still displayed altered composition compared to the control group 5-months post dredging (Fig. 5b).

4.3. The effect of WID on fish community characteristics, mortality and health

A total of 215 fish comprising 10 species were recorded pre-dredging and communities were dominated by Thin-lipped Grey Mullet (*Chelon ramada* – 41), Roach (*Rutilus rutilus* – 45), Gudgeon (*Gobio gobio* – 48) and Chub (*Squalius cephalus* – 37) which comprised 80% of the total abundance. Dredging corresponded with differences in fish communities within and downstream of the dredge reach, relative to the control. Taxonomic richness ($F_{2,14}$ = 4.152, P = 0.043; Fig. 8B) and the Shannon-Weiner index ($F_{2,14}$ = 4.435, P = 0.036; Fig. 8C) varied between groups, with differences detected between control and downstream but not control and dredged or dredged and downstream groups. Berger-Parker varied between treatment groups (GLM $F_{2,14}$ = 4.4611, P = 0.033; Fig. 8D), with dredged and downstream samples being quantitatively similar but significantly lower than the control group. No significant differences were recorded between treatment groups for abundance (GLM $F_{2,14}$ = 3.228, P = 0.076; Fig. 8A).

Differences in fish community composition were detected as functions of treatment and site but not time, with site explaining the greatest amount of variation (Table 1). Principal Coordinate Analysis (PCoA) indicated no clear differences between control and dredged communities (Fig. 5c), however downstream and control communities were statistically different (Table 2). Downstream communities demonstrated a higher degree of clustering relative to control and dredged communities (average distance = 0.230; Table 3), but differences between these were not significant (Table 4).

A total of 236 fish (*Chelon ramada*: abundance = 91; *Gobio gobio*: abundance = 16; Sea bass *Dicentrarchus labrax*: abundance = 9; *Squalius*

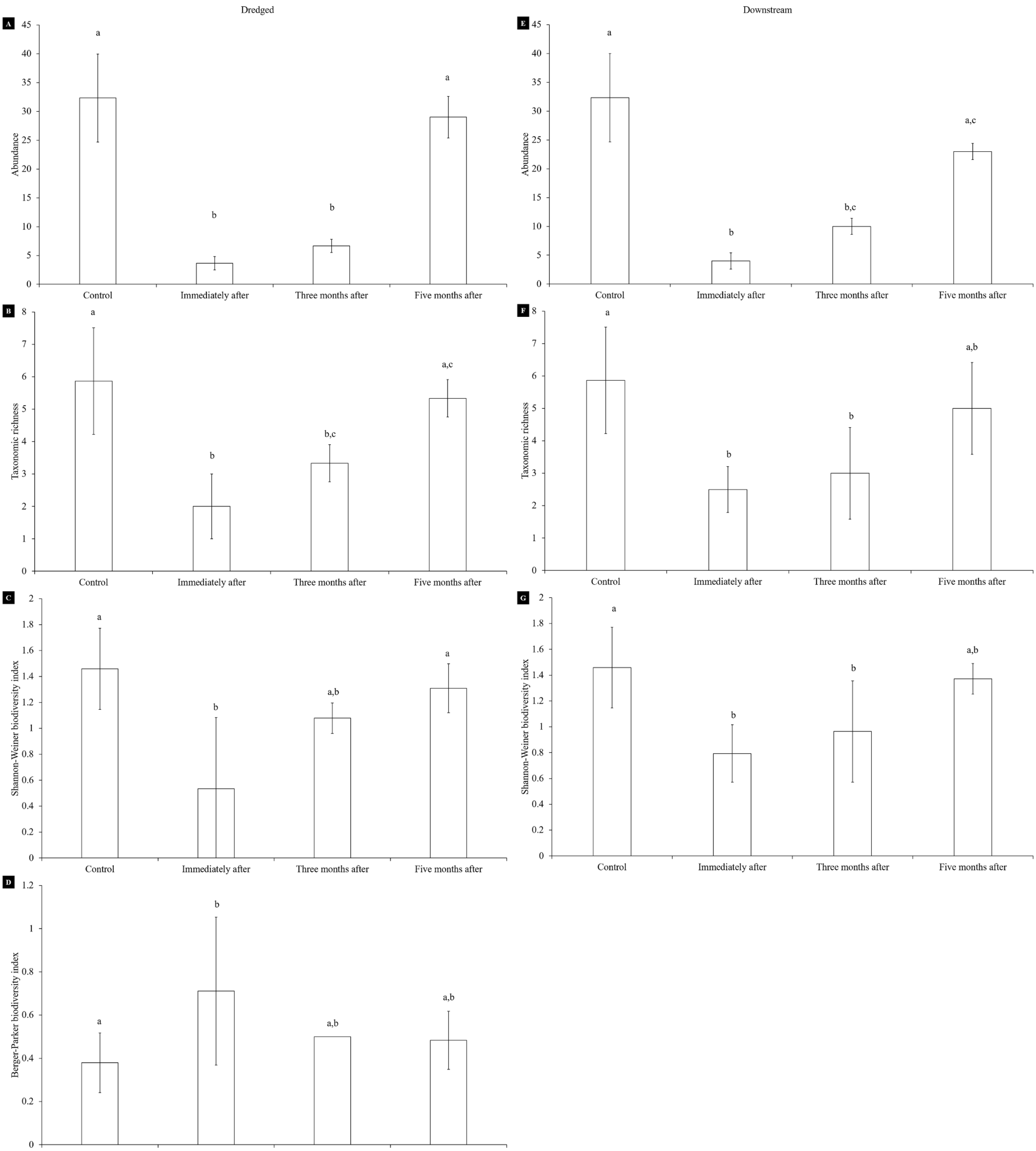

Fig. 4. Benthic macroinvertebrate abundance, taxonomic richness, Shannon-Wiener biodiversity index and/or Berger-Parker biodiversity index data per treatment/time group for dredged (A-D) and downstream (E-G) sites. Presented are treatment means (dredged: control, n = 15; immediately after, n = 3; three months after, n = 3; 5 months after, n = 3; downstream: control, n = 15; immediately after, n = 2; three months after, n = 2; 5 months after, n = 2) and letters above bars indicate no statistical differences.

cephalus: abundance = 2) were captured via pelagic trawling during the dredging operation. All captured fish were alive and showed no obvious signs of dredging induced symptoms including respiratory distress (e.g., gulping at the surface). The caudal fins of 7 *Chelon ramada*, representing 8.26% of the total catch, were either split or torn but did not appear to influence locomotive abilities of fish. No other signs of fish damage were observed.

5. Discussion

5.1. The effect of WID activities on water physicochemistry and potential implications for estuarine ecology

Maximum turbidity peaks were relatively unaffected by WID given the strong tidal influence on turbidity. However, mean turbidity levels

Table 1
Summary of PERMANOVA output indicating macroinvertebrate and fish responses to independent factors (site, treatment and time). Significant results (p < 0.05) are emboldened.

	F	r^2	*p*
Benthic macroinvertebrates			
Site	1.91	0.051	0.115
Treatment	7.12	0.191	**0.001**
Time	2.29	0.061	0.071
Marginal macroinvertebrate			
Site	1.32	0.38	0.212
Treatment	3.99	0.12	**0.002**
Time	3.34	0.10	**0.003**
Fish			
Site	7.62	0.34	**0.001**
Treatment	2.75	0.12	**0.024**
Time	0.77	0.03	0.608

Table 2
Summary of pairwise PERMANOVA tests by treatment group (control, dredged, downstream) on fish and macroinvertebrate communities. Significant results (p < 0.05) are emboldened.

	F	r^2	*p*
Benthic macroinvertebrates			
Control vs Dredged	10.35	0.32	**0.003**
Control vs Downstream	7.82	0.29	**0.003**
Dredged vs Downstream	0.34	0.03	1.000
Marginal Macroinvertebrate			
Control vs Dredged	3.87	0.15	**0.009**
Control vs Downstream	2.45	0.11	0.102
Dredged vs Downstream	0.93	0.07	1.000
Fish			
Control vs Dredged	1.87	0.16	0.237
Control vs Downstream	6.04	0.38	**0.009**
Dredged vs Downstream	4.82	0.55	0.300

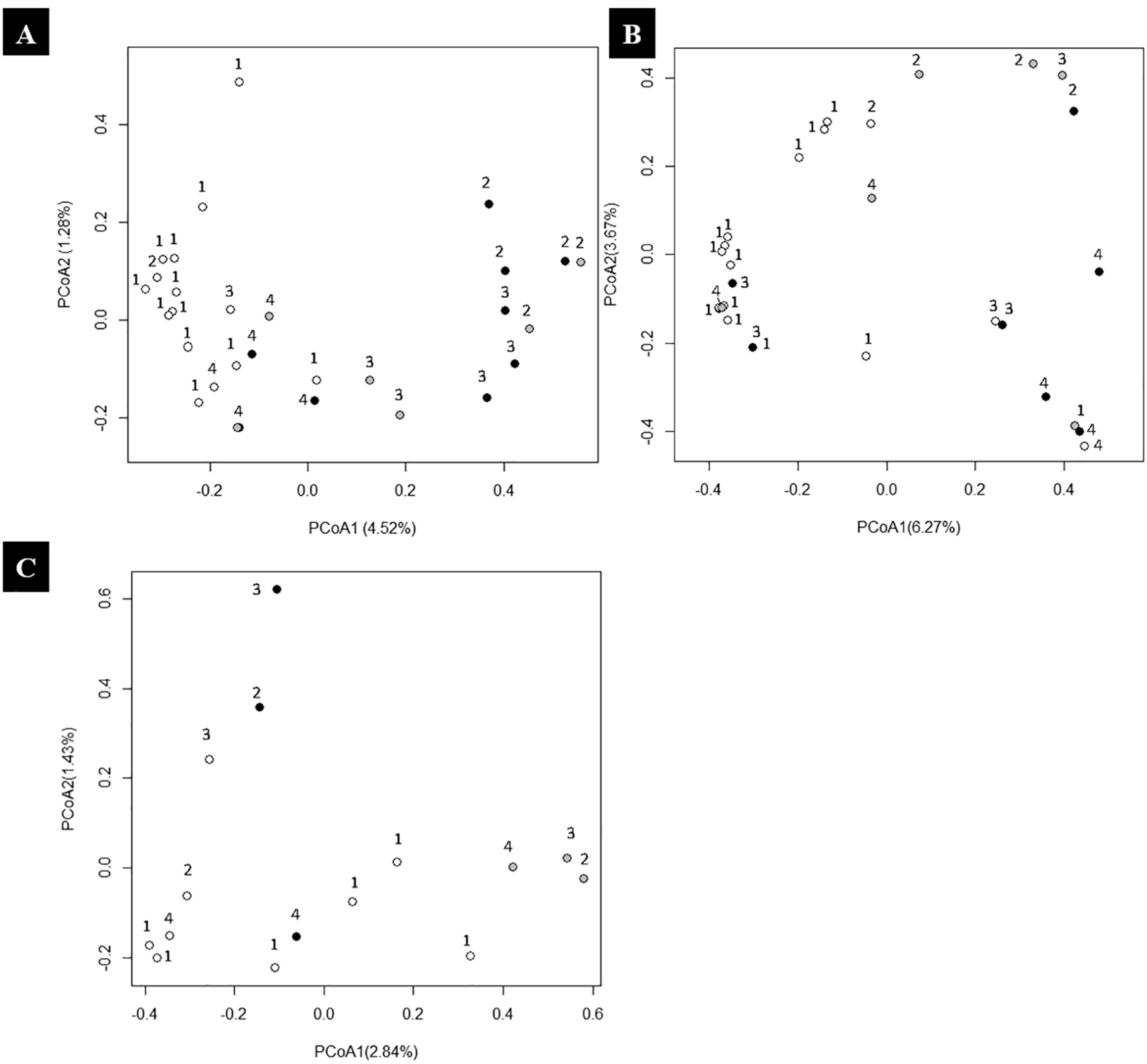

Fig. 5. Principal Coordinates Analysis (PCoA) of **A** benthic macroinvertebrate, **B** marginal macroinvertebrate and **C** fish communities by treatment (control, dredged, downstream) using Bray-Curtis similarities coefficients. Open = control, black = dredged and, grey = downstream. Numbers represent temporality (1 = pre-dredging; 2 = immediately after; 3 = three months after; 4 = 5 months after).

Table 3
Summary of multivariate dispersion distance by treatment (control, dredged, downstream) for benthic and marginal macroinvertebrate and fish communities.

Organism	Control	Dredged	Downstream
Benthic macroinvertebrates	0.255	0.423	0.349
Marginal macroinvertebrates	0.393	0.610	0.495
Fish	0.432	0.432	0.230

Table 4
Summary of pairwise multivariate dispersion distance tests by treatment (upstream, dredged, downstream) on fish and macroinvertebrate communities. Significant results ($p < 0.05$) are emboldened.

	p
Benthic macroinvertebrates	
Control vs Dredged	**0.011**
Control vs Downstream	0.284
Dredged vs Downstream	0.521
Marginal macroinvertebrates	
Control vs Dredged	0.075
Control vs Downstream	0.622
Dredged vs Downstream	0.603
Fish	
Control vs Dredged	0.999
Control vs Downstream	0.107
Dredged vs Downstream	0.207

were elevated during the dredging- vs pre-dredging period and did not return to pre-dredging levels 1.25 days after dredging had ceased. The ecological effects of elevated turbidity and/or fine sediment deposition are well documented (e.g. Quinn et al., 1992; Richards and Bacon, 1994). It is possible that WID may have induced symptoms in macroinvertebrates and fish that we did not detect. For example, in terms of macroinvertebrates, increased physical clogging (McKenzie et al., 2020), loss of food (e.g. periphyton; Shaw and Richardson, 2001) and decreased respiratory abilities may have occurred.

Sediment suspended from the bed during WID may also have inhibited fish foraging (Utne-Palm, 2002) and predation (Reid et al., 1999; Fiksen et al., 2002; de Robertis et al., 2003), and may increase food detectivity in some instances (Utne-Palm, 1999; Wenger et al., 2014). Increased turbidity may also cause significant light attenuation and reduce growth and feeding in fish (Gard, 2002), and change natural avoidance and alarm behaviours (Newcombe and MacDonald, 1991; Rowe and Dean, 1998; Sweka and Hartman, 2001), impacting visual acuity (Jones et al., 2015; Vogel and Beauchamp, 1999) and reducing the reactive distances of fish (Barrett et al., 1992; Sweka and Hartman, 2003; Zamor and Grossman, 2007). Increased suspended solids may also have impaired respiratory function in fish by damaging gill tissues, reducing gill efficiency (Appleby and Scarratt, 1989; Wenger et al., 2017), and expose fish to contaminants released during dredging that may result in behavioural changes (Collin and Hart, 2015), physical damage, physiological and sublethal impacts and/or mortality (Johnson et al., 2014; Wenger et al., 2017). However, in the case of the River Parrett, turbidity levels are naturally high without dredging (Fig. 2), particularly during spring high tides, so resident fish will be acclimatised to these conditions to some degree.

Dissolved oxygen levels declined substantially during and immediately after dredging, with levels quickly recovering post-dredging. Unlike turbidity, dissolved oxygen (negative) peaks did not occur during high tides without dredging, suggesting a WID effect. Levels dropped and remained consistently low throughout the dredge period, relative to pre-dredging conditions. It is unlikely the short-term reductions in mean oxygen concentrations will have had significant detrimental impacts on ecological communities, as evidenced by the relatively modest changes in fish populations during this study. Further, it is reasonable to assume that macroinvertebrates may have drifted downstream (Calpez et al., 2017) and fish moved within the channel (Burleson et al., 2001) – probably vertically within the water column – to avoid potentially harmful conditions, which we assume were more extreme at the bed than near the surface (where water physicochemistry measurements were made). However, future research should seek to quantify physical and chemical changes throughout the water column.

5.2. *The effect of WID on benthic and marginal macroinvertebrate communities*

Benthic macroinvertebrate communities of the River Parrett were not abundant or diverse prior to WID due to the strong tidal influence and relatively homogeneous habitat present – a relic of historic river management. Despite this, WID led to significant reductions in abundance (−26%), taxonomic richness (−83%) and Shannon-Wiener diversity (−33%) and a significant increase in Berger-Parker dominance (+49%) within the dredge reach relative to the control group (Fig. 3). WID disturbed the upper metre of riverbed sediment (Pledger et al., 2020) and in doing so, caused significant reductions in burrowing taxa (e.g. *Potamopyrgus antipodarum*, *Sphaeriidae* and Chironomidae) which drove changes in dredge reach community composition. Macroinvertebrates were not removed from the channel as in extraction activities (e.g. Aldridge, 2000; Grygoruk et al., 2015; Killeen et al., 1998) so whilst it is reasonable to assume some will have been injured or died, many will have been entrained and drifted downstream. Thus, unlike mechanical extraction activities where sediment and biota are removed and deposited above the waterline, there is a high probability of survival of macroinvertebrates associated with WID activities.

The benthic macroinvertebrate communities downstream of the dredged reach were found to differ relative to the control group. Macroinvertebrate abundances (−24%) and taxonomic richness (−40%) were significantly lower and caused a significant reduction and increase in Shannon-Wiener and Berger-Parker biodiversity indices, respectively. Whilst trends in dredged and downstream populations were similar it is likely the mechanisms of disturbance and therefore, population change will have varied between locations. Unlike the dredged reach, where direct entrainment of macroinvertebrates probably occurred, changes in downstream communities were most likely a function of increased macroinvertebrate drift, driven by changes to physicochemical conditions (e.g. reduced dissolved oxygen levels and elevated average suspended sediment concentrations; Ciborowski et al., 1977; Doeg and Milledge, 1991) and resultant increases in saltation, scour and abrasion

Table 5
Summary of taxa driving differences in dredged and downstream benthic macroinvertebrate populations as determined by SIMPER. Total change in abundance following dredging indicated in parentheses (±).

Dredged sites (2–4)			Downstream sites (5–6)		
Species	Average before	Average after	Species	Average before	Average after
Potamopyrgus antipodarum (–)	2.26	0.00	*Potamopyrgus antipodarum* (–)	1.77	0.00
Sphaeriidae (–)	2.6	0.58	Sphaeriidae (–)	2.01	0.71
Chironomidae (–)	2.68	1.46	Oligochaeta (–)	3.03	2.16
Oligochaeta (–)	3.23	1.93	Chironomidae (–)	1.66	1.64
Lymnaea peregra (–)	1.33	0.11			

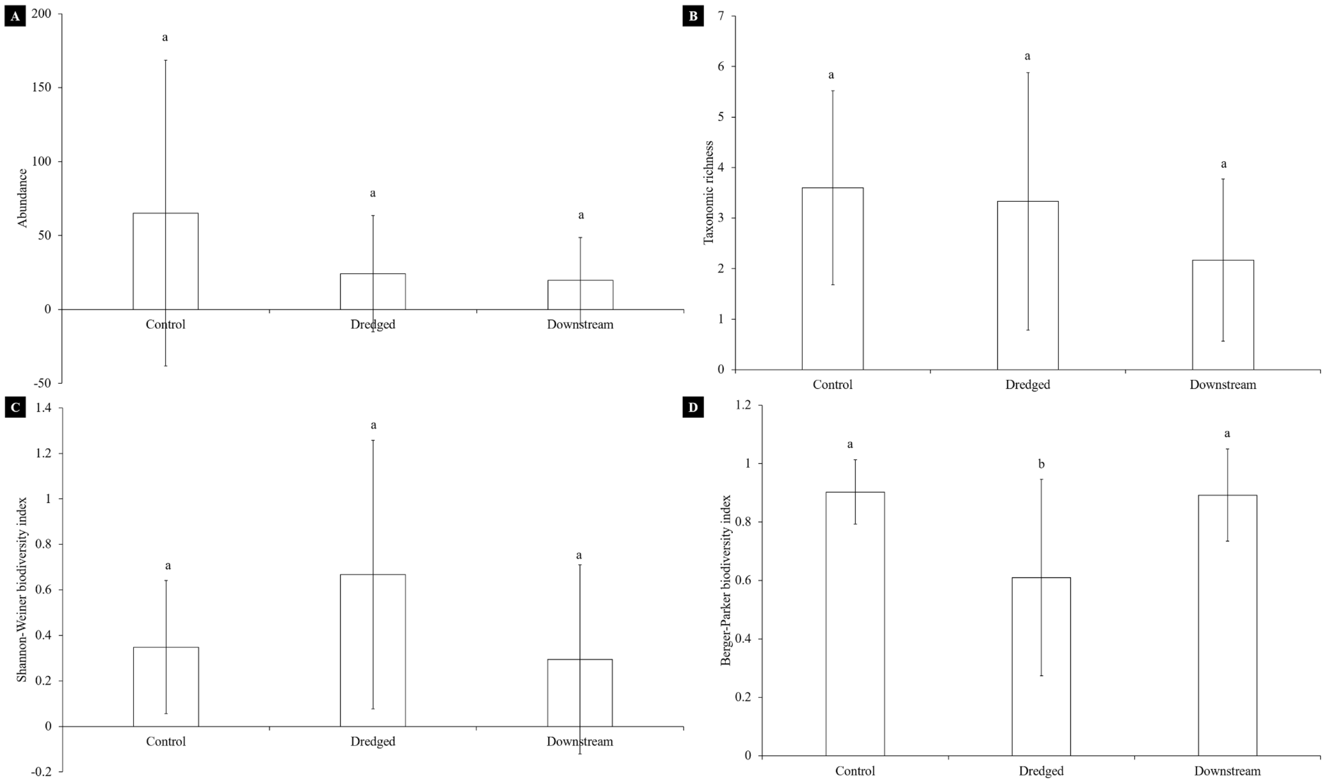

Fig. 6. Marginal macroinvertebrate **A** Abundance, **B** Taxonomic richness, **C** Shannon-Wiener biodiversity index and **D** Berger-Parker biodiversity index data per treatment group. Presented are treatment means (control, n = 15; dredged, n = 9; downstream, n = 6; ±STDEV) and letters above bars indicate no statistical differences.

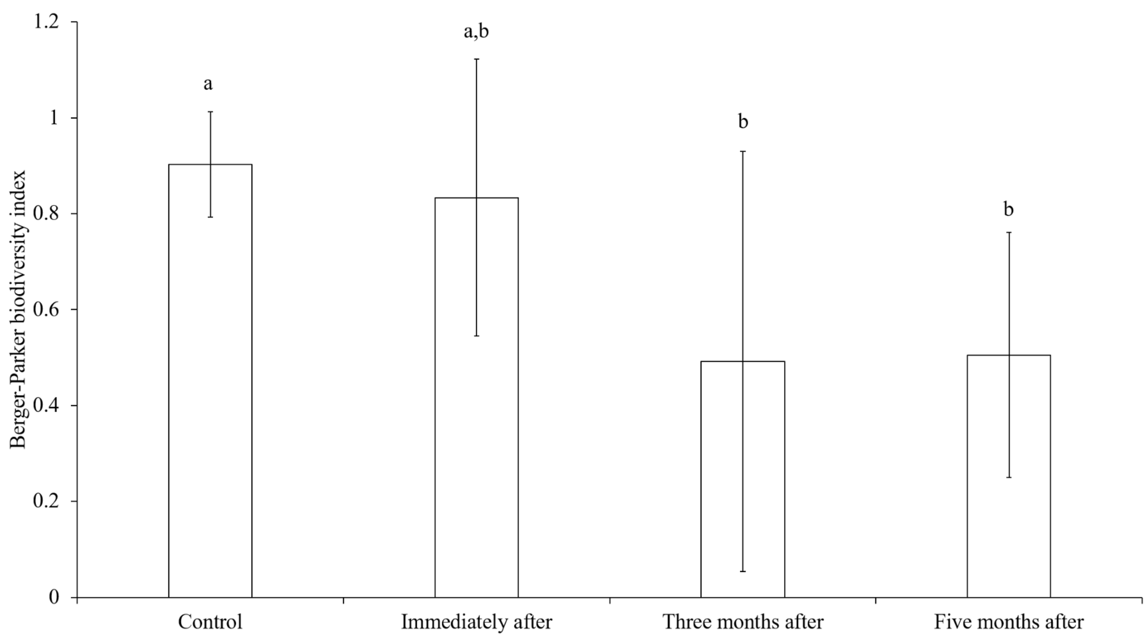

Fig. 7. Marginal macroinvertebrate Berger-Parker biodiversity index data per treatment/time group for dredged sites. Presented are treatment means (control, n = 15; immediately after, n = 3; three months after, n = 3; 5 months after, n = 3) and letters above bars indicate no statistical differences.

(Culp et al., 1986).

Marginal communities were more abundant and diverse relative to benthic communities prior to WID, due to increased availability of more heterogeneous habitats, and therefore by targeting WID at the channel thalweg and avoiding ecologically rich bankside habitats, the effects of management on macroinvertebrate populations may be minimised. In this study marginal communities were largely unaffected by WID in both dredged and downstream locations, with just a significant reduction in dredged relative to control populations detected. The management practice of targeting less ecologically sensitive habitats was therefore successful in terms of marginal macroinvertebrate communities, mirroring results from previous work investigating mechanical dredging

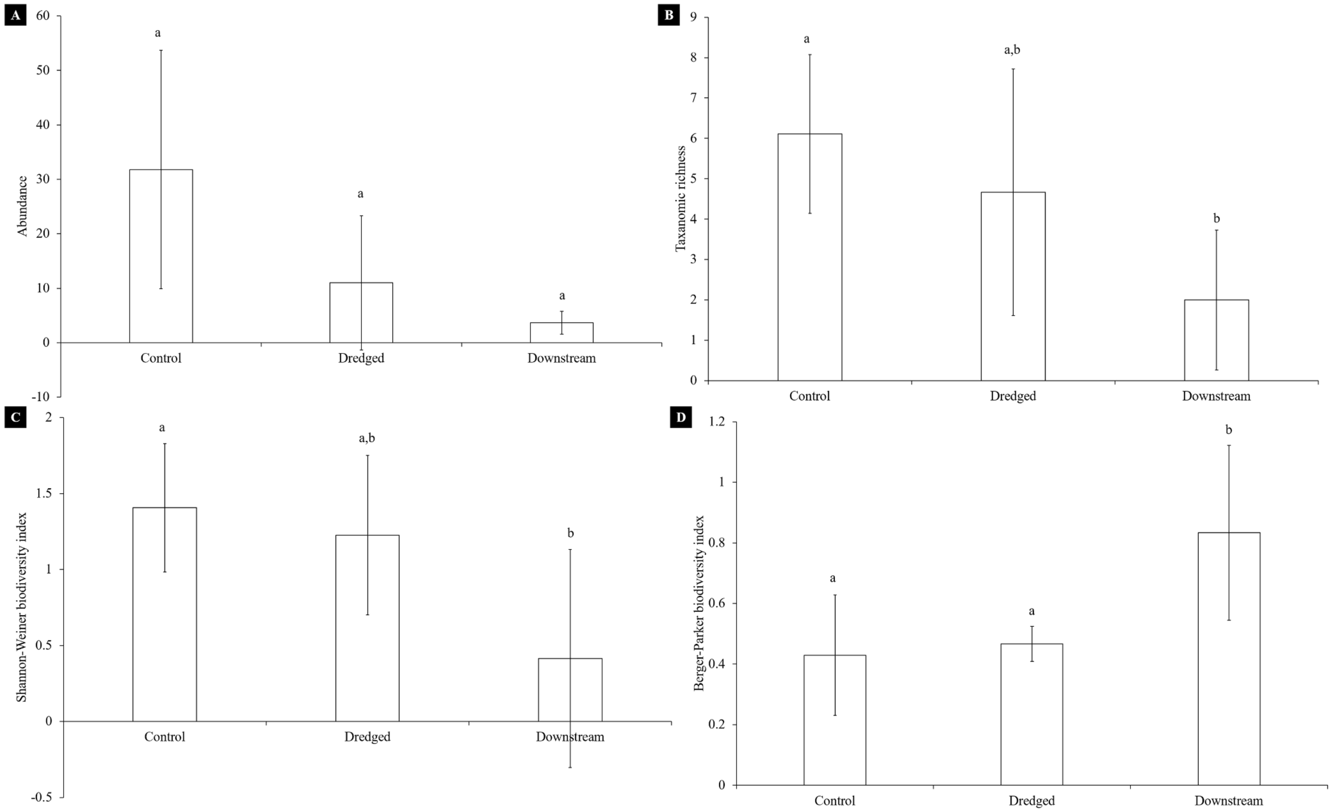

Fig. 8. Fish **A** Abundance, **B** Taxonomic richness, **C** Shannon Wiener biodiversity index and **D** Berger-Parker biodiversity index data per treatment group. Presented are treatment means (control, n = 9; dredged, n = 3; downstream, n = 3; ±STDEV) and letters above bars indicate no statistical differences.

(Aldridge, 2000).

Benthic and marginal communities had fully recovered after 5 months, with some metrics being quantitatively similar to the control within 3 month and benthic community composition also demonstrating recovery. Interestingly, and in terms of marginal communities, there was no significant difference in the Berger Parker diversity index between the control and immediately after groups, but there was for the control and three months after and control and five months after groups. Moreover, some sites demonstrated altered marginal community composition three-five months post-dredging. Together, this could suggest no initial impact of dredging, and a potential change in the ecological community following the first post-dredging sampling occasion. Collectively however, results suggest that any effects of WID were relatively short-lived.

5.3. The effects of WID on fish community characteristics, health and mortality

Fish populations were relatively unaffected by dredging, with only communities in the downstream reach being significantly affected relative to the control data. Given their highly mobile nature, it is reasonable to assume that fish were able to avoid the perceived threat of the dredging operation (Wenger et al., 2017) and migrate either up or downstream. As a result, reductions in taxonomic richness which drove changes in Shannon-Weiner and Berger-Parker diversity indices were likely due to behavioural avoidance of the sediment plume/elevated turbidity (Collin and Hart, 2015; Newcombe and Jensen, 1996) and the potential impacts of sub-optimal water physicochemistry (see Section 5.1), with some species potentially more capable of avoiding the perceived threat.

Based on available evidence collected within this study, WID had a negligible effect on fish health and mortality. However, it should be noted that we were limited to sampling the surface layers of the water column meaning potential effects on fish mortality and the wider ecosystem – particularly at greater depths within the water column – remained unquantified.

5.4. Management implications

Five environmental management strategies to mitigate ecological risk during WID campaigns can be derived from this and earlier research. First, application of WID in systems where high concentrations of contaminants have been detected within surface sediments (where WID is targeted; Pledger et al., 2020) should be avoided. This is to prevent contaminant dispersal and potential physical and/or ecological effects, including ingestion of metals by benthic fauna which is considered the primary cause of bioaccumulation of metals in food chains (Bordajandi et al., 2003). Second, greater understanding of a system's ecological resistance and ability to recover from anthropogenic stressors is required to inform WID and other dredging approaches. Community recovery rate varies between systems and dredging operations employed (Harrison et al., 1964; Pfitzenmeyer, 1970; Stickney, 1973) and is likely to be quickest where highly mobile organisms are abundant, facilitating escape during management and/or rapid recolonization post-dredging (Aldridge, 2000). Further, in areas subject to highly dynamic hydrological and sedimentological conditions, such as estuaries, benthic populations may be adapted and/or accustomed to these conditions, rendering the extent and persistence of dredging effects limited (Díaz, 1994). The rates of recolonization and growth of all species and the interactions between these and their environment should always be considered during the planning of dredging operations. Third, due to high sediment production rates during WID (e.g. 17,565 m^3 sediment displaced from the dredge reach in 2017; Pledger et al., 2020), the spatial extent of impact should be considered during program development. Ecological assessments must identify the potential risks to ecological communities both within and downstream of the dredged site

– not least because sediment plumes generated during subaqueous dredging can extend several kilometres from dredged sites, with the magnitude of the effect varying as functions of operation duration and site substrate and flow characteristics (Evans et al., 2012; Fisher et al., 2015). Fourth, while the relatively impoverished benthic community was affected during this study, the marginal community (where over 95% of the biodiversity and abundance of macroinvertebrates was located) was relatively unaffected by WID. This is an important finding and suggests that selective conservation of key habitats during dredging programmes may be an effective management strategy. Other studies have also reported the diversity and richness of macroinvertebrate and plant species can be locally increased by selective dredging of small sections of channel (Bracewell et al., 2019), leaving some sections unaltered to promote recolonization. Further, a "rotation model" is suggested whenever practicable, which would involve periodic dredging of short reaches at non-contiguous sites with dredging operations separated by significant periods of time (Buczyński et al., 2016). This approach may help secure high species diversity by preserving heterogeneous habitats at different stages of succession (Buczyński et al., 2016) and has been successfully utilised in the management of peat bog pools and ditches (Wildermuth, 2001; Buczyński, 2015). Thus, reach scale habitat removal (including channel and marginal habitats) should be avoided where practicable– ensuring the conservation of important ecological communities and/or isolated habitats from which recolonisation of dredged sections can occur. Fifth, in freshwater environments, such as locks and marinas, unaffected by tidal processes and rapid changes in water physicochemistry, the ecological impacts of WID may be profound. It is reasonable to assume the legacy of dredging may also be important, with "pioneer" dredging of undisturbed areas potentially more ecologically damaging than "maintenance" dredging operations, as biota are less accustomed to adapting to frequent, significant and/or rapid changes in physicochemistry through anthropogenic activities. Thus, diligence is required where sensitive species to ensure environmental conditions – including dissolved oxygen and turbidity levels – are maintained to support potentially sensitive floral and faunal communities.

6. Conclusion

For the first time, an in-situ experiment has quantified the effects of WID on water physicochemistry, and macroinvertebrate and fish communities within a tidal river. Turbidity peaks were relatively unaffected by WID and comparable in magnitude to pre-dredge hightide peaks. Despite this, mean turbidity was elevated throughout and 1.25 days following the dredging period. Negative peaks in dissolved oxygen levels occurred during and immediately after WID and were not detected during high tides without WID, which provides strong evidence of a dredging effect. WID corresponded with significant effects for benthic communities in the dredged and downstream environments, and minor changes in marginal macroinvertebrate communities in the dredged reach. WID effects on fish communities were limited to reductions in taxonomic richness and diversity downstream of the dredged reach. Based on available evidence, thalweg WID provides an ecologically sensitive alternative to traditional channel margin mechanical dredging methods. However, our abilities to sample fish and water physicochemistry during dredging were highly constrained as the sediment plume and drifting debris mobilised during WID limited sampling to the upper layers of the water column. Sampling the entire profile during dredging would almost certainly have resulted in a greater observed effect on water physicochemistry and may have revealed further effects on fish community characteristics, mortality and health.

Future research should investigate 3 avenues. First, there is a need to quantify the impacts of different dredging technologies, including hydrodynamic, mechanical and hydraulic methods, on a broad range of biota whilst utilising appropriate experimental designs that include adequate replication and post-management monitoring. Second, it is imperative we investigate the environmental tolerances of biota across the full range of life stages, under both field and laboratory conditions. This research would generate important benchmark data to assess whether environmental change resulting from dredging operations may have ecological consequences. This could be used to limit ecological impacts by ensuring environmental parameters do not exceed key thresholds. An existing lack of organism- and method-specific knowledge makes the selection of appropriate dredging techniques difficult and risks significant ecological damage. Third, cost-benefit analyses of flood alleviation projects are required that consider a range of freshwater, intertidal and marine environments and biota, and the associated resource requirements and environmental impact implications of these. This information will allow system managers and regulatory bodies to select the most appropriate management techniques for individual waterbodies.

CRediT authorship contribution statement

Andrew G. Pledger: Conceptualization, Methodology, Formal analysis, Investigation, Writing - original draft, Writing - review & editing, Visualization, Supervision, Project administration, Funding acquisition. **Philip Brewin:** Writing - original draft, Writing - review & editing, Project administration, Funding acquisition. **Kate L. Mathers:** Writing - original draft, Writing - review & editing, Formal analysis, Visualization. **John Phillips:** Writing - original draft, Writing - review & editing, Conceptualization. **Paul J. Wood:** Conceptualization, Methodology, Formal analysis, Investigation, Writing - original draft, Writing - review & editing, Writing - original draft, Writing - review & editing, Funding acquisition. **Dapeng Yu:** Writing - original draft, Writing - review & editing, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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