Detecting the impacts of harbour construction on a seagrass habitat and its subsequent recovery

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Abstract:

Managing coastal development requires a set of tools to adequately detect ecosystem and water column degradation, but it also demands tools to detect any post-disturbance improvement. Structural seagrass indicators (such as shoot density or cover) are often used to detect or assess disturbances, but while they may be very sensitive to the impact itself, it is unclear if those indicators on their own can effectively reflect recovery at time scales relevant to managers. We used the construction of a harbour affecting a nearby *Posidonia oceanica* seagrass community to test the ability of a set of indicators (structural and others) to detect alterations and to evaluate their sensitivity to recovery of environmental quality after harbour construction was complete and the disturbance ceased. We used a Beyond Before After Control Impact (BBACI) design to evaluate effects on one impacted and three control meadows where we used structural, morphological, community and physiological indicators (26 in total) to asses disturbance impacts. Additionally, we measured some of the potential environmental factors that could be altered during and after the construction of the harbour and are critical to the survival of the seagrass meadow (light, sediment organic matter, sediment accrual).

Harbour construction caused a clear increase in sediment organic matter and in sediment deposition rates, especially fine sand. Light availability was also reduced due to suspended sediments. Sediment and light conditions returned to normal levels 5 and 15 months after the construction began. As expected, seagrass structural indicators responded unequivocally to these environmental changes, with clear reductions in shoot density. Additionally, reduced light conditions quickly resulted in a decline in carbohydrate content in affected meadows. Unexpectedly, we also recorded a significant increase in metal content in plant tissues. No response was detected in the physiological indicators related to eutrophication (e.g. N and P content in tissues) and in morphological (shoot biomass) and community (epiphyte biomass) indicators. More than three years after the completion of the harbour, structural indicators did not show any sign of recovery. In contrast, physiological indicators, mainly heavy metal and carbohydrates content, were much better in detecting the improvement of the environmental conditions over the fairly short period of this study. These results indicate that while structural indicators are critical to evaluate the immediate effect of disturbances and the recovery on impacted systems, specific physiological indicators may be much better suited to determining the timing of environmental quality recovery. The design of impact and monitoring protocols in the wake of coastal developmental projects need to consider the differential effectiveness and time-response of measured indicators carefully.

Keywords: indicator, disturbance, *P. oceanica*, physiological indicators, time-response.

Introduction:

Coastal zones around the world have been and are still facing intensive development that includes the construction of marine infrastructures such as harbours and breakwaters (Short & Wyllie-Eciieverria 1996; Waycott *et al.* 2009). These large, physical structures modify the interface between the sea and the land, can destroy valuable marine habitats (Inglis & Lincoln-Smith. 1995) and alter currents and sediment dynamics (Morales, Borrego & Ballesta 2004). In addition, the process of construction itself also produces several associated effects that may have extended areas of influence. Specifically, the construction of harbours has been associated with increases in the fine sediment fraction and in water turbidity (Erftemeijer & Robin Lewis III 2006) and with changes in current dynamics that can affect sediment deposition (Morales, Borrego & Ballesta 2004; Anfuso, Pranzini & Vitale 2011) among other effects. Detecting the appearance of ecological impacts, assessing their consequences and understanding the time course for natural conditions to re-establish following cessation of the disturbance are some of the main challenges for environmental management in the coastal zone. Indicators are among the most important tools used by managers to detect changes in ecosystems due to anthropogenic impacts or improvements due to successful management actions (Heink & Kowarik 2010). Their present-day importance is reflected by the huge effort devoted to develop a large array of indicators in many different environments, from forest ecosystems (Brooks *et al.* 1998) to freshwater (Harig & Bain 1998; Munné & Prat 2009) and coastal water marine systems (Carignan & Villard 2002; Ballesteros *et al.* 2007; Martínez-Crego *et al.* 2008). According to Heink and Kowarik (2010), an indicator in ecology and environmental planning is defined as something used to depict or evaluate environmental conditions or changes or to set environmental goals, where this *something* can be either a component or a measure of environmentally relevant phenomena. For the present work, we use the term "indicator" only in the second sense, that is, a measure of environmentally relevant phenomena.

The rate at which the indicators respond to degradation and improvement in physical environmental conditions is therefore, a key aspect for their use and interpretation, yet it is often overlooked (Donangelo *et al.* 2010). Indeed, most of them have been validated only to track trajectories of ecosystem degradation. Few have proven successful in tracking recovery, since recovery is often more protracted and, in many cases, may follow complex, non-linear trajectories (Scheffer *et al.* 2001; Carstensen *et al.* 2011). This is especially true for ecosystems with slow-growing species, where recovery processes are typically slow, often occurring over significantly longer time periods than standard monitoring programs are funded for. The failure to detect recovery in these systems may result in the erroneous conclusion that disturbance has persisted or that remedial actions were inadequate, both of which may have important consequences for long-term management.

Seagrass meadows are one of the dominant ecosystems in shallow coastal marine waters over the world with important contributions to their goods and services (Cullen-Unsworth & Unsworth 2013). Additionally, seagrass ecosystems are extremely sensitive to changes in water quality and to other human induced disturbances (Short & Wyllie-Echevarria 1996; Krause-Jensen, Greve & Nielsen 2005; Lopez y Royo *et al.* 2010). As a result, seagrass ecosystems have been used in many monitoring programs (Marbà *et al.* 2012) to obtain reliable indicators. Among them, structural ones are the most widely used because of their ease of measurement and their clear links to ecosystem structure and services. Likewise, morphological parameters have been used worldwide as a good measure of plant health and stress (Marbà *et al.* 2012). Finally, physiological indicators are increasingly being used in monitoring programs as they are reported to be efficient tools for early detecting of anthropogenic disturbances (Martinez-Crego et al. 2008).

Posidonia oceanica (L.) Delile is the most important and widespread seagrass in the Mediterranean sea, where it forms extensive meadows from the surface down to 40 m depth (Bouduresque *et al.* 2006). *P. oceanica* is a foundation species (sensu Dayton & Hessler 1972) that performs important ecological functions in the ecosystem but is also extremely sensitive to changes in environmental conditions. This makes *P. oceanica* one of the species from which the largest number of indicators have been described so far (Montefalcone 2009). In particular, with a set of structural, physiological, morphological and community indicators, this plant has been observed to effectively detect changes in light availability, sediment characteristics and increases in organic matter – the most frequent physical changes associated with coastal development (Ruiz & Romero 2001; Ruiz & Romero 2003; Erftemeijer & Robin Lewis III 2006; Frederiksen *et al.* 2006; Pérez *et al.* 2007; Serrano, Mateo & Renom 2011). Of these, physiological indicators are well known to have driver-specific responses and this specificity has been used as a tool to identify the causal factors behind deterioration in the ecosystem or in water quality (Martínez-Crego *et al.* 2008). Nevertheless, there is still little known about the rate of response of these indicators to improvements in the physical environment once the disturbance has ceased (i.e. how they track recovery). As already stated, the inability to track the timing and form of response to improved environmental conditions can lead to erroneous management decisions, with potentially negative economic, social and ecological consequences. In this context, we examined the response of a range of indicators within a *P. oceanica* seagrass ecosystem to the construction of a harbour (discrete disturbance) in NW Catalonia, Spain. Our main objective was to test the ability of 26 commonly used indicators to detect alterations during the construction of the harbour and their sensitivity to potential recovery in environmental conditions over three years after the construction had been completed.

2. Materials and methods:

2.1. Study design

The study was designed to detect the impacts of a harbour construction on a nearby *Posidonia oceanica* meadow and its potential recovery when the construction had been completed. We employed a Beyond-BACI design (BBACI, Underwood (1992), measuring responses from a *P. oceanica* meadow close to an expanding harbour ('impact' location) and at three distant (non-impacted) meadows before, during and after harbour construction ceased, see (Table 1). At each location we measured 26 commonly used seagrass indicators to test their ability to track the time course of recovery. In parallel, we also measured the main environmental drivers associated with the ecological impacts of harbour construction: water transparency, sediment deposition and sediment grain size, produced during and after the construction (Erftemeijer & Robin Lewis III 2006).

2.2. Study area and harbour construction

The study area is situated in the NE coast of Spain between two localities, Blanes and Lloret de Mar, both with intense tourism development. Blanes had one of the biggest harbours in the area, with a mooring capacity for 59 fishing vessels and 684 recreational boats. In March of 2010 (Table 1) construction began to add a new external breakwater to the harbour. This meant the occupation of 42.037 m^2 of sea surface, dredging 40,000 m^3 of sediment from the seafloor and using several tonnes of sand and stones to stabilize the new structure. During this period, which lasted from March to July 2010

(Table 1), a superficial and mid water net barrier was placed to limit the spread of fine sediment released from dredging activities and sand addition to nearby waters. After this period, some minor works continued, but dredging activities were less intense.

The area is characterized by clear coastal waters, and the coastline has numerous *P. oceanica* meadows reported to be in good ecological condition (Romero *et al.* 2010). We selected a *P. oceanica* meadow situated between 9 and 14 meters depth (density of 290 shoots m^{-2}, SE=28) on a rocky substrate and at 160 m (at its closest point) from the old breakwaters (Fig. 1, 41°40'19"N 2°48'04"E) as the potentially impacted site. The foundations of the new breakwaters were built only 20 m from the meadow (at its closest point). Additionally, three control sites were selected north of the construction area due to the absence of *P. oceanica* meadows further south. The first control site was situated close to Mar i Murtra garden in Sa Forcanera beach (41°40'31"N 2°48'19"E), the second in Fenals beach (41°41'19"N 2°50'7"E) and the third was situated next to Cala Canyelles (41°42'4"N, 2°53'21"E, Fig.1).

2.3. Sampling design and data acquisition

Driver measurements and seagrass sampling was carried out at the impact and control meadows before the disturbance started (February 2010), during the disturbance (March 2010 to early August 2010) and after the disturbance (late August 2010 to August 2013; see Table 1) following a Beyond BACI design. All meadows were sampled between 13 and 15 m depths to minimize bathymetric variability.

2.3.1. Monitoring environmental drivers

To assess light availability, irradiance (photosynthetic active radiation, 400-700 nm) was measured in situ as photosynthetic photon flux density (PPFD, µmol quanta m^{-2} s^{-1}), using Apogee PAR QSO-Sun 2.5v light sensors connected to HOBO u12-013 data loggers that recorded at 10-minute intervals. One light sensor was placed in each meadow just above canopy level for one month when the harbour was being expanded (April 2010) and 10 months after the disturbance (April 2011). As some fouling appeared on the sensors at the end of the recording period, and in order to prevent its shading effects, we only used the first 15 days of data when the sensors were totally clean.

To assess the importance of the sedimentation processes over the meadow, sediment deposition rates (g m^{-2} day^{-1}) were measured using six cylindrical sediment traps (16 cm height and 4.5 cm diameter) installed in groups of three, in two independent tripods just above the canopy level at each site, similarly to those used by Gacia, Granata and Duarte (1999). Sediment traps were installed for one month in all sites with light sensors only while dredging activities were occurring (in April 2010).

To follow changes in sediment granulometry and organic matter content in the seagrass meadow, three random samples of surface sediments were taken at each site using 50 ml cups. Samples were taken during the disturbance (April 2010) and at a single time after construction work ceased (July 2010). Sediment composition was analysed with an optical particle analyser Mastersizer 2000. Organic matter was determined as the difference in the weight of the sediments before and after drying in a muffle furnace for 5 hours at 500ºC.

2.3.2. Monitoring seagrass indicators

We chose 26 seagrass indicators commonly used in ecological assessments (Marbà *et al.* 2012) that have known functional associations with a wide range of coastal disturbances: morphological and structural indicators (shoot density, cover, number of leaves, leaf length, leaf area and % of leaf necrotic tissue), physiological indicators related to changes in light availability ($\delta^{13}C$, sucrose, starch and total carbohydrate content), physiological indicators related to nutrient variations and eutrophic conditions (C, N, P, $\delta^{15}N$, $\delta^{34}S$), physiological indicators linked to metal availability (Fe, Pb, Cd, Mn, Ni, Cu, Zn) and community indicators (leaf epiphyte biomass). All physiological parameters were analysed in rhizomes; in addition, we analysed C, N, P, $\delta^{15}N$, $\delta^{13}C$ from the leaves. For most variables, we collected samples before harbour construction began, during the disturbance and at several intervals after the cessation of dredging activities, depending on the variable (see Table 1). Sampling consisted of collecting ten seagrass shoots, randomly chosen in each site and at each sampling period. Seagrass shoot density was estimated at each site and sampling time in three 40x40 cm fixed quadrats in each station. Sampling was performed four times (once before the disturbance, and three times after the disturbance, Table 1). Shoot density counts during the disturbance (March 2010, April 2010 and July 2010) were not possible due to high sedimentation values, with underwater visibility close to 0 m during that period.

2.4. Laboratory analysis

From the 10 shoots collected for each station and sampling time, 5 shoots (one replicate each) were firstly used to measure morphological indicators (number of leaves, leaf length, leaf area and % of leaf necrotic tissue) and epiphyte biomass. Then, all 10 shoots were pooled together and randomly grouped in pairs. The first 1-1.5 cm of the rhizomes and leaf number two (without epiphytes) of each pair were separated, dried and ground to a fine powder, resulting in 5 replicates per site and time. Laboratory analyses were performed to measure 14 biochemical or physiological indicators: Carbon and nutrient content (C, N, P in leaves and rhizomes), stable isotopic composition ($\delta^{15}N$, $\delta^{13}C$ in leaves and rhizomes, $\delta^{34}S$ in rhizomes), metal content (Fe, Pb, Cd, Mn, Ni, Cu, Zn in rhizomes), sucrose, starch and total carbohydrate content (in rhizomes). Processing and analysis of samples was carried out according to methods detailed in Romero et al. (2007) and Martínez-Crego *et al.* (2008) at Centre d'Estudis Avançats de Blanes (CEAB-CSIC), Scientific and Technical Services of the University of Barcelona (SCT-UB), Servicios de Apoio a la Investigación (SAI) of University of la Coruña, and Millbuck labs (United Kingdom).

2.5. Data analysis

Asymmetrical analyses of variance were used to examine temporal differences between the potentially impacted meadow and the average of control meadows. The mechanics and the logical structure of these analyses are fully explained in Underwood (1991); (1992; 1993; 1994). Data of the different environmental drivers and indicators were compared between periods (Before/After or During/After) and meadows (Impacted/Controls).

In order to determine the time of response and recovery of the different indicators BBACI analyses were carried out comparing values before to each of the sampling times during and after the disturbance (0-1, 0-2, 0-5, 0-15 months, Table 1, Table 2). If the BxC test was not significant (Marked with * in table 2), the impact was tested with an *F*-ratio Mean Square BxI/Mean Square Residual. Otherwise impact was tested with an *F*-ratio Mean Square Before vs After x Impact vs Controls (BxI)/ Mean Square Before vs After x Among Controls (BxC). The same method was used to analyse changes in shoot density with time (0-8, 0-15 and 0-38 months, Table 1, Table 2).

Differences in sedimentation rates between impacted and control sites were tested using one-way ANOVA. Changes in light availability, sediment grain composition and percentage of organic matter in sediments during and after the disturbance were also tested using an asymmetrical analysis of the variance (Table1).

Data were examined for normality with the Shapiro-Wilks test and homogeneity of variances was tested with Bartlett's test. The assumptions of normality and homogeneity of variances between samples was not met for all variables and tests. However, analyses of variance are robust with respect to these problems, particularly in large designs (Underwood 1981). All statistics were performed in R software (R Development Core Team 2012).

3. Results:

3.1. Responses of environmental drivers

We detected an 88.4% reduction in light availability in the impacted meadow relative to the controls ($p=0.046$; Table S2, Fig. 2). Some seasonal variability not directly related to the construction was also recorded in light levels at both impacted and control meadows, and was probably caused by storms during the course of the study (Fig. 2). In addition, fine materials in suspension increased sedimentation rates over 14 times compared to controls 2 months after the impact began, from 33 to 480 g m^{-2} day^{-1} ($p<0.01$, Table S3, Fig. 3). As a result, the impacted meadow was buried under 5 to 10 cm of fine sediments that produced a significant change in granulometry ($p<0.01$,Table S4, Fig. 4), noticeably decreasing grain size, and a ca. two-fold increase in the organic matter content of sediments ($p<0.01$, Table S5, Fig. 5). Dredging activities and sand additions were completed 5 months after the works began, and natural hydrodynamics washed out the fine sediment. Light availability was found to have recovered 15 months after commencement (Fig.2), but light availability probably recovered sooner, since granulometry (Fig.4) and organic matter in surface sediments (Fig. 5) recovered to control levels five months after the construction began.

3.2. Responses of indicators to disturbance:

Shoot density and indicators directly related to light availability (starch, sucrose and total carbohydrates) and metal pollution (Ni, Fe, Mn, Pb, Cd) responded most to the disturbance (Table 2). Ni and starch content in the rhizomes showed significant differences after just one month, being the first indicators to detect the impact of the harbour construction. After two months, sucrose, starch and total carbohydrates were significantly lower in the impacted site compared to control sites (Fig. 6A,B), which started to increase their concentrations following the seasonal cycle of carbohydrate production (Fig. 6-B). Also two months after the start of the disturbance, Fe, Mn and Pb in the rhizomes showed significant differences from 'before' values (Table 2). Five months after the start of the disturbance, starch remained significantly lower at the impact site. Sucrose and total carbohydrates were also lower at the impact site, as seen in Table 2 and Fig. 6-A, although these differences were not significant due to the high variability in controls. After five months, the effects of the disturbance were also evident in Fe, Mn, Pb, (Table 2, Fig. 6-C), which continued to record increasing values of these metals. Also, five months after the construction began, Cd levels increased for the first time with respect to the controls. See all results of Beyond BACI variance analysis of indicators in Table (S6).

Shoot density was not measured during the disturbance due to low visibility in the area (see methods). The first values of shoot density were obtained 8 months later when the disturbance had already ceased. Shoot density had declined by 50 % at the impacted site 8 months after the disturbance (Fig. 6-E, Table 2, Table S7).

3.3. Recovery time of indicators values:

The time taken for the indicators to return to pre-disturbance values varied depending on the indicators considered. Sucrose and total carbohydrates started recovering right after the cessation of the disturbance while other physiological indicators (starch, Mn, Cd, Pb, Fe) recovered fully after 10 months. In contrast, shoot density did not show any sign of recovery at any sampling time, until our last sampling event, 32 months after the cessation of the disturbance (Fig. 6, Table 2, Table S7).

4. Discussion

As expected, harbour works produced a pulsed disturbance that increased water turbidity, reduced light availability and covered the meadow close to the harbour with fine sediment for approximately 5 months. After that period, environmental conditions recovered, i.e. water clarity was restored and the fine sediment that covered the meadow disappeared, probably washed away by hydrodynamics. The disturbance was sufficiently intense to halve shoot density in the impacted meadow within 8 months. Physiological indicators were highly sensitive to the disturbance and changes in indicators related to light availability (starch, sucrose and total carbohydrates) and metallic pollution (Ni, Fe, Mn, Pb, Cd) were detected just two months after construction began. In contrast, morphological indicators (% of leaf necrotic tissue, number, length and leaf area), epiphyte biomass and nutrient contents and isotopic signatures (%N, %C, $\delta^{13}C$, δ^{15} N, %P, $\delta^{34}S$) did not appear to be affected by the harbour construction. Physiological indicators began an immediate recovery after the cessation of the work and 10 months later all these indicators had returned to pre-disturbance levels. In contrast, the structural indicator (shoot density) did not show any recovery even 38 months after the disturbance.

4.1. Drivers of disturbance and impact on the ecosystem

The breakwater construction had important detrimental effects on the surrounding environment. Sedimentation rates at the impact site during the disturbance period were extremely high, i.e. 14 times higher than observed at our control sites, where rates matched natural deposition rates previously observed in the same area by Gacia, Granata and Duarte (1999). Deposited sediment was mainly composed of fine particles that spread over the meadow, and, as documented in other instances, the sediment retention net used to prevent the spread of sediment was not fully effective (Erftemeijer & Robin Lewis III 2006). Sediment in suspension caused a reduction in light availability comparable to the effect of using an 80-90% shading mesh (Ruiz & Romero 2001; Mackey, Collier & Lavery 2007). Moreover, the content of organic matter in the sediment of the impacted meadow doubled during the disturbance, probably due to the effect of the settlement of fine particles. These reductions in light availability may be the main cause of the recorded decline in shoot density as has been found in previous studies (Ruiz & Romero 2001; Serrano, Mateo & Renom 2011). However, it is plausible that sediment deposition, increases in organic matter, or their interactions, also contributed to the mortality or reduced production of shoots observed in the ecosystem; each of these three drivers were individually above thresholds known to cause shoot mortality (Manzanera, Pérez & Romero 1998; Ruiz & Romero 2003; Erftemeijer & Robin Lewis III 2006; Pérez *et al.* 2007; Serrano, Mateo & Renom 2011). Taken together, our results clearly indicate that the physical disturbance caused by the construction of the harbour was too close to the protected *P. oceanica* meadow (Habitat Directive 92/43/CEE), causing important damage to the ecosystem.

4.2. Responses of indicators to disturbance

Of the 26 seagrass indicators we examined, none of the morphological (number of leaves, leaf length, leaf area and % of leaf necrotic tissue) or community indicators (epiphyte biomass) analysed responded to the harbour construction. The physiological indicators related to nutrients (%N, %P, $\delta^{15}N$, %C, $\delta^{13}C$) did not show any influence either. This suggests that sediment dredging and material addition did not spread nutrients into the system. Additionally, despite the observed increase in organic matter that has been linked to an increased sulphur production and changes in $\delta^{34}S$ (Oakes & Connolly 2004; Frederiksen *et al.* 2006; Frederiksen *et al.* 2008), we did not detect a response in $\delta^{34}S$. The lack of $\delta^{34}S$ response can be due to the high abundance of Fe and Mn in the environment that can competitively inhibit sulphate reduction processes and, consequently, H_2S production (Myers & Nealson 1988; Thamdrup, Fossing & Jørgensen 1994; Frederiksen *et al.* 2006). As pointed out above, shoot density, that appeared to be mainly responding to a reduction in light availability, halved 8 months after the harbour works began although, it is highly probable that it had reduced as early as five months after the disturbance, in accordance with time course of shoot mortality previously reported (Serrano, Mateo & Renom 2011).

Physiological indicators related to carbon storage (starch, sucrose and total carbohydrates) were the first to respond to the disturbance. It took just one month for starch, and two months for sucrose and total carbohydrates to respond to light reduction. The harbour works took place between the end of winter and at the beginning of the summer season, matching the plants' seasonal peaks in growth, photosynthetic rates and carbon storage in the rhizomes (Alcoverro, Duarte & Romero 1995; Alcoverro, Manzanera & Romero 2001; Serrano, Mateo & Renom 2011). This is the period of the year when light deprivation becomes critical because plants are exhausting their reserves (Alcoverro, Manzanera & Romero 2001). Therefore, while plants from control sites were increasing their photosynthetic rates and carbohydrate reserves, plants in the impacted site could not photosynthesise and consequently could not restore their carbohydrate reserves after winter. Conversely, no effects were observed in $\delta^{13}C$ content in leaves and rhizomes, which contrasts with the results of some shading experiments where this indicator responded relatively quickly to light reduction (Serrano, Mateo & Renom 2011).

The metal content of tissues also responded quickly to the disturbance, confirming the response times found in recent studies by Richir *et al.* (2013), and also suggest that metals were remobilized during the harbour construction. Fe and Mn accumulated in the rhizomes of the plant within 2 months of the commencement of the construction and continued to increase during the disturbance period. In 5 months, their concentrations were 3 times greater than observed in the controls and 3 to 4 times higher than observed at any site of available monitoring programs (Romero *et al.* 2010). Pb and Cd levels also increased 2 and 5 months after the start of the disturbance. In contrast, concentrations of Ni in the rhizomes decreased more than the controls during the disturbance probably due to the antagonistic uptake interactions with Fe, Mn and other metallic elements (Richir *et al.* 2013).

4.3. Recovery time of indicators

More or less ten months after the construction ceased, all physiological indicators affected by the coastal development returned to pre-disturbance levels. As has been documented elsewhere, carbohydrate reserves in plant tissues responded positively to improvements in environmental quality (Longstaff *et al.* 1999; Ruiz & Romero 2001; Invers *et al.* 2004; Pérez *et al.* 2007; McMahon, Lavery & Mulligan 2011). In contrast, little is known on the accumulation of metals in seagrasses (but see Richir et al. 2013). Our results show that processes regulating fluxes of metallic elements (Ni, Fe, Mn, Pb, Cd) in plant rhizomes can be fast and dynamic and can be sensitive indicators to detect degradation and improvement of water quality conditions. The fact that metallic trace elements are highly dynamic in plants is especially relevant for managers when assessing the results of seagrass and water quality monitoring programs. Indeed, it suggests that the presence of these elements in seagrass rhizomes, at least in the younger parts of the plant, may reflect the presence of these metals in the surrounding environment, and is not the consequence of historical accumulation of metals in the plants.

Shoot density did not recover to pre-disturbance levels 38 months after the construction ceased. Recuperation of *P. oceanica* structural indicators such as shoot density or cover requires longer time frames to respond due to the very slow rhizome elongation rates for this species, reported to be only of 2– 4 cm year^{-1} (Duarte 1991; Marbà & Duarte 1998) and the low shoot recruitment observed (Marbà & Duarte 1998). In fact, very few studies have reported the recuperation of structural indicators except over very long periods of time (Badalamenti *et al.* (2011). Full recovery, if possible, is thought to require decades (Duarte 2002; González-Correa *et al.* 2005).

5. Summary and conclusions

The Blanes harbour breakwater was built with apparently adequate mitigation measures including sediment retention nets designed to reduce the impacts of dredging and construction on the threatened *Posidonia oceanica* seagrass meadows. Despite this, the activity resulted in an intense, pulsed disturbance, significantly reducing water quality (light levels and sediment deposition), and causing a dramatic structural decline in adjacent seagrass meadows. Three years after the harbour construction, shoot densities at meadows 20 m away from the site had still not shown signs of recovery. Unsurprisingly developmental activities of this nature, particularly so close to *P. oceanica* meadows, can be catastrophic for this legally protected ecosystem. From a management perspective, we observed that not all common seagrass indicators responded specifically to this type of disturbance, and even from those that responded, very few were able to detect ecosystem recovery when the disturbance had ceased. Only some indicators such as shoot density, carbohydrate content and metal-related measures responded as early indicators to this type of disturbance. These indicators are useful in determining the potential mechanisms of post-disturbance habitat responses, and suggest that the plant degradation we recorded after harbour construction was linked to deposition caused by sediment movement, reduced light levels, and metal contamination. Structural indicators such as shoot density showed no response at management time scales (3 years), and while this may be linked to the slow growth rate of the seagrass, it may reflect a potential post-disturbance ecosystem shift. In contrast, physiological indicators (light and metal-related) were much more sensitive to changes in environmental quality and returned to a pre-disturbance state within 3 years. Although improvement of water quality does not represent ecosystem recovery, it can help evaluate the effectiveness of mitigation and remedial actions, critical for coastal management and planning.

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Table footnotes

Table 1: Sampling time of environmental drivers and seagrass indicators. Before (February 2010), During (March, April 2010) and After the disturbance (July 2010, October 2010, May 2011, July 2013)

Table 2. Results from BBACI analysis. Time before was compared against 1, 2, 5 and 15 months after the beginning of the disturbance. If BxC was not significant (marked with *), impact was tested with the *F*-ratio Mean Square BxI/Mean Square Residual. Otherwise impact was tested with the *F*-ratio Mean Square Before vs After x Impact vs Controls (BxI)/ Mean Square Before vs After x Among Controls (BxC). When significant, arrows indicate increase or decrease of different indicators. Dashes indicate that no differences were found. Blank spaces mean no measurements taken.

Figure footnotes

Fig.1. Map of the study area showing the impacted site and the three control sites (C1, C2, C3).

Fig.2. Daily maximum irradiance measured at canopy level at the four sites (impact site with black circles and 3 controls) during the disturbance (upper panel, April 2010) and after the disturbance (bottom panel, April 2011).

Fig.3. Average sediment deposition during the disturbance in the impact and control sites. Due to the low variability we grouped the three control sites together.

Fig.4. Surface sediment granulometry in the impacted and control sites. Each bar indicates the % in volume of the different grain size ranges two months after the

disturbance (left panel) and 8 months after the disturbance (right panel). Black bars represent the impacted site and white bars the controls.

Fig.5. Average organic matter content in surface sediments in the impacted and control sites. Low variability within controls allowed grouping the three control sites.

Fig.6. Indicators response in the impacted and control sites. Boxplots of the impacted site and control sites (grouped) for shoot density, total carbohydrates and Fe indicators before (time=0) and at different times after the disturbance (time=1, 2, 5, 8, 15, 38). The grey shadow indicates the duration of the impact.

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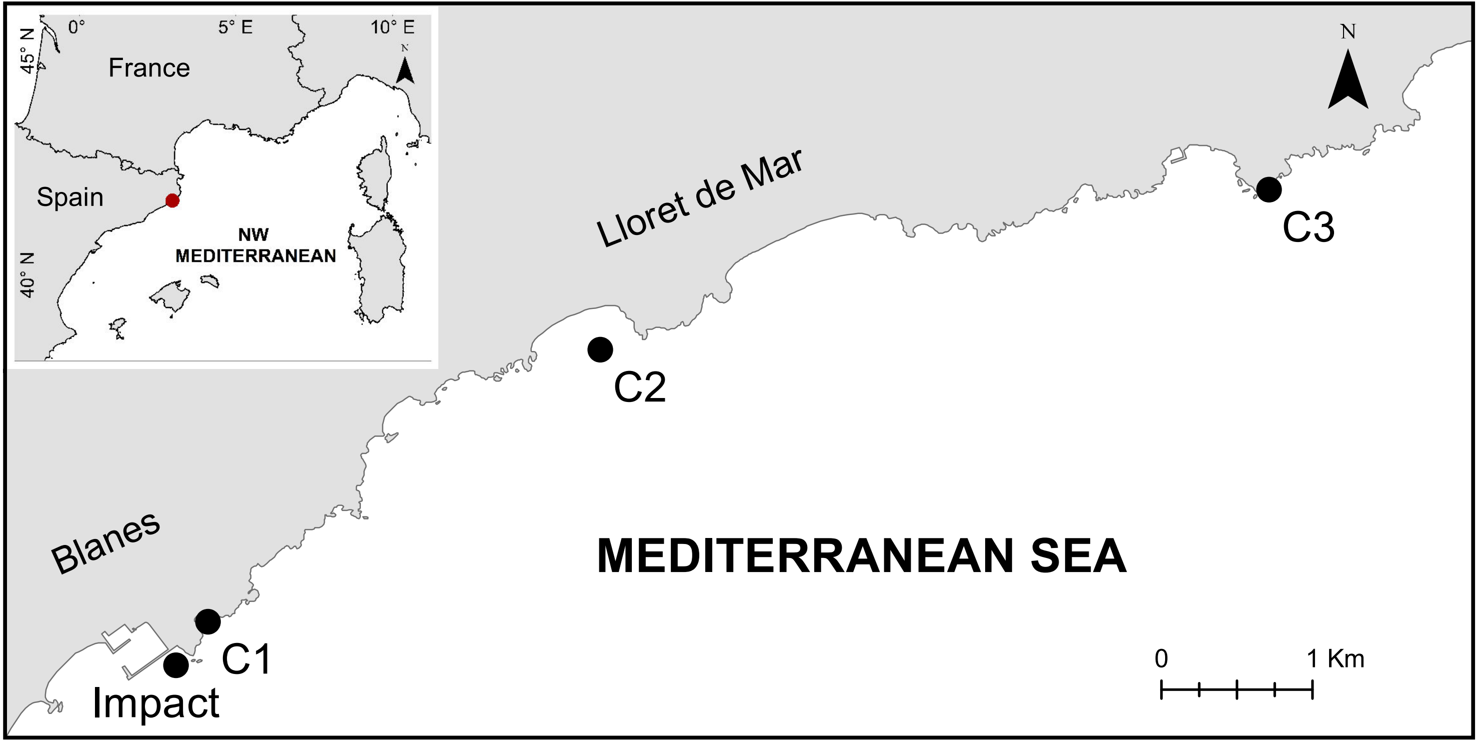
	Before	During				After			
Time from beginning of the works (months)	0	1	2	3	4	5	8	15	38
Drivers									
Light sensor			x					x	
Sediment traps			x						
Surface sediment		x				x			
Seagrass									
Shoot samples	x	x	x			x		x	
Structure measures	x						x		x

Table 1: Sampling time of environmental drivers and seagrass indicators. Before (February 2010), During (March, April 2010) and After the disturbance (July 2010, October 2010, May 2011, July 2013).

		During disturbance		After disturbance		
Group	Parameters	0-1	0-2	0-5	0-15	0-38
Morphology	Leaf length	-	-	-	-	
	Number of leaf	-	-	-	-	
	% Leaf necrosis	-	-	-	-	
	Leaf area	-	-	-	-	
Community	Epiphyte biomass	-	-	-	-	
Physiology	%N	-	-	-	P<0.05*	
	%C	-	-	-	-	
	δ^{15} N	-	-	-	-	
	δ^{13}C	-	-	-	-	
	%N Leaf	-	-	P<0.05*	-	
	%C Leaf	-	-	-	-	
	δ^{15} N Leaf	-	-	-	-	
	δ^{13}C Leaf	-	-	-	-	
	δ^{34}S	-	-	-	-	
	Fe	-	↑P<0.05	↑P<0.05	-	
	P	-	-	-	-	
	Pb	-	↑P<0.05*	↑P<0.05*	-	
	Cd	-	-	↑P<0.05*	-	
	Mn	-	↑P<0.05	↑P<0.05	-	
	Ni	↑P<0.05	-	-	-	
	Cu	-	-	-	-	
	Zn	-	-	-	-	
	Starch	↓P<0.05*	↓P<0.05	↓P<0.05	-	
	Sucrose	-	↓P<0.05	-	-	
	Total Carbohydrates	-	↓P<0.05	-	-	
				0-8	0-15	0-38
Structure	Shoot density			↓P<0.05	↓P<0.05	↓P<0.05

Table 2. Results from BACI analysis. Time before was compared against 1, 2, 5 and 15 months after the beginning of the disturbance. If BxC was not significant (marked with *), impact was tested with the *F*-ratio Mean Square BxI/Mean Square Residual. Otherwise impact was tested with the *F*-ratio Mean Square Before vs After x Impact vs Controls (BxI)/ Mean Square Before vs After x Among Controls (BxC). When significant, arrows indicate increase or decrease of different parameters. Dashes indicate that no differences were found. Blank spaces mean no measurements taken.


0°
5° E
10° E
45° N
40° N
France
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NW MEDITERRANEAN
N
Lloret de Mar
Blanes
C1
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C3
Impact
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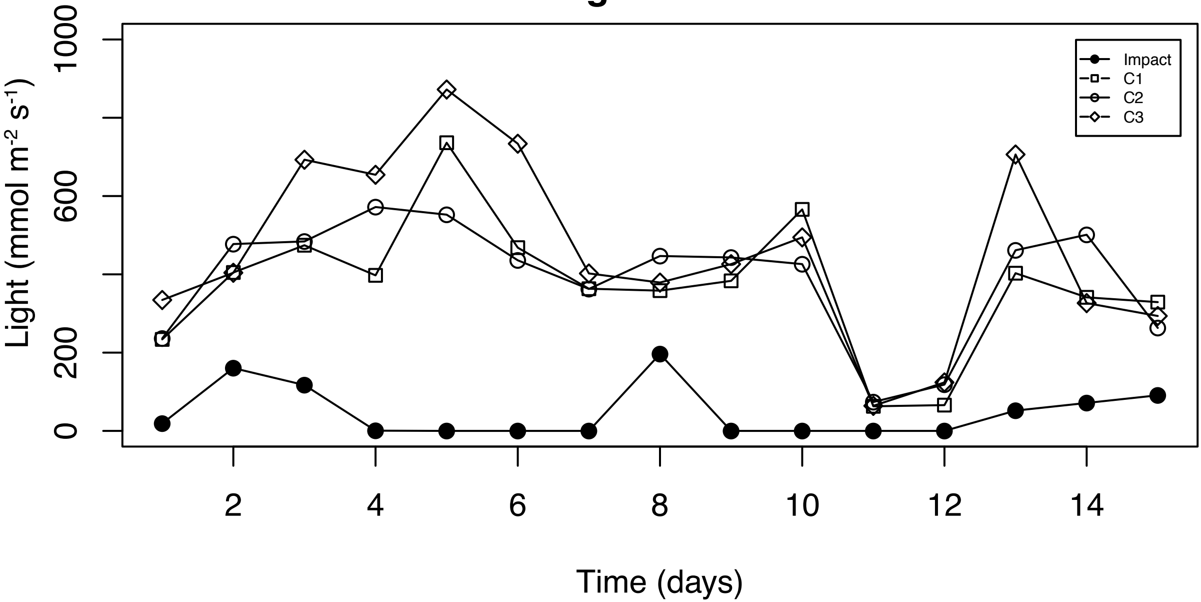
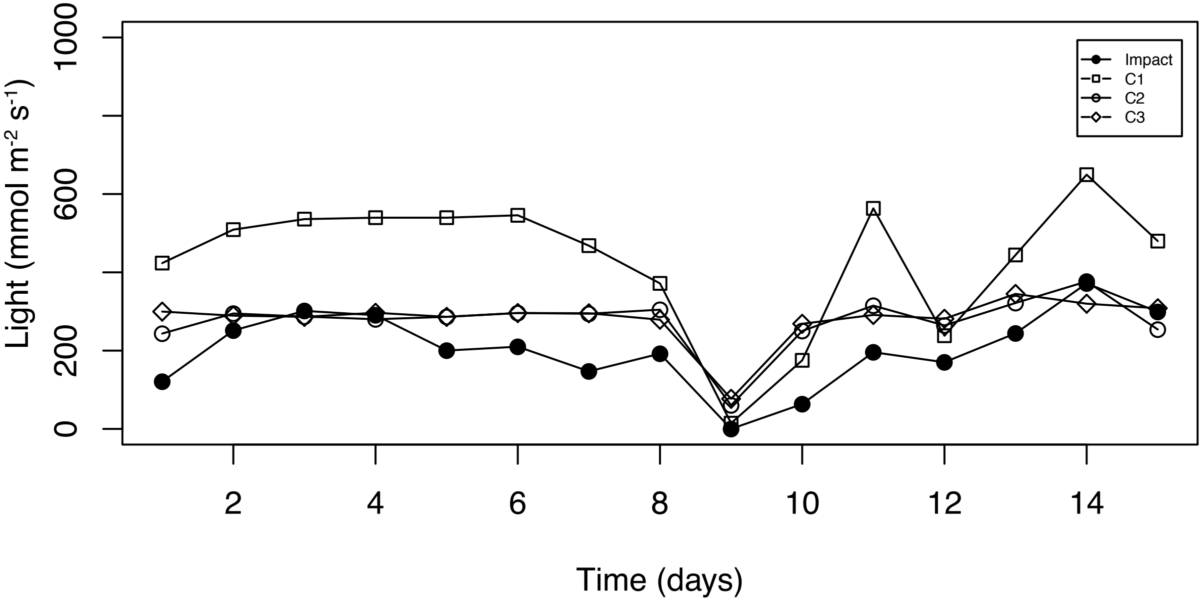
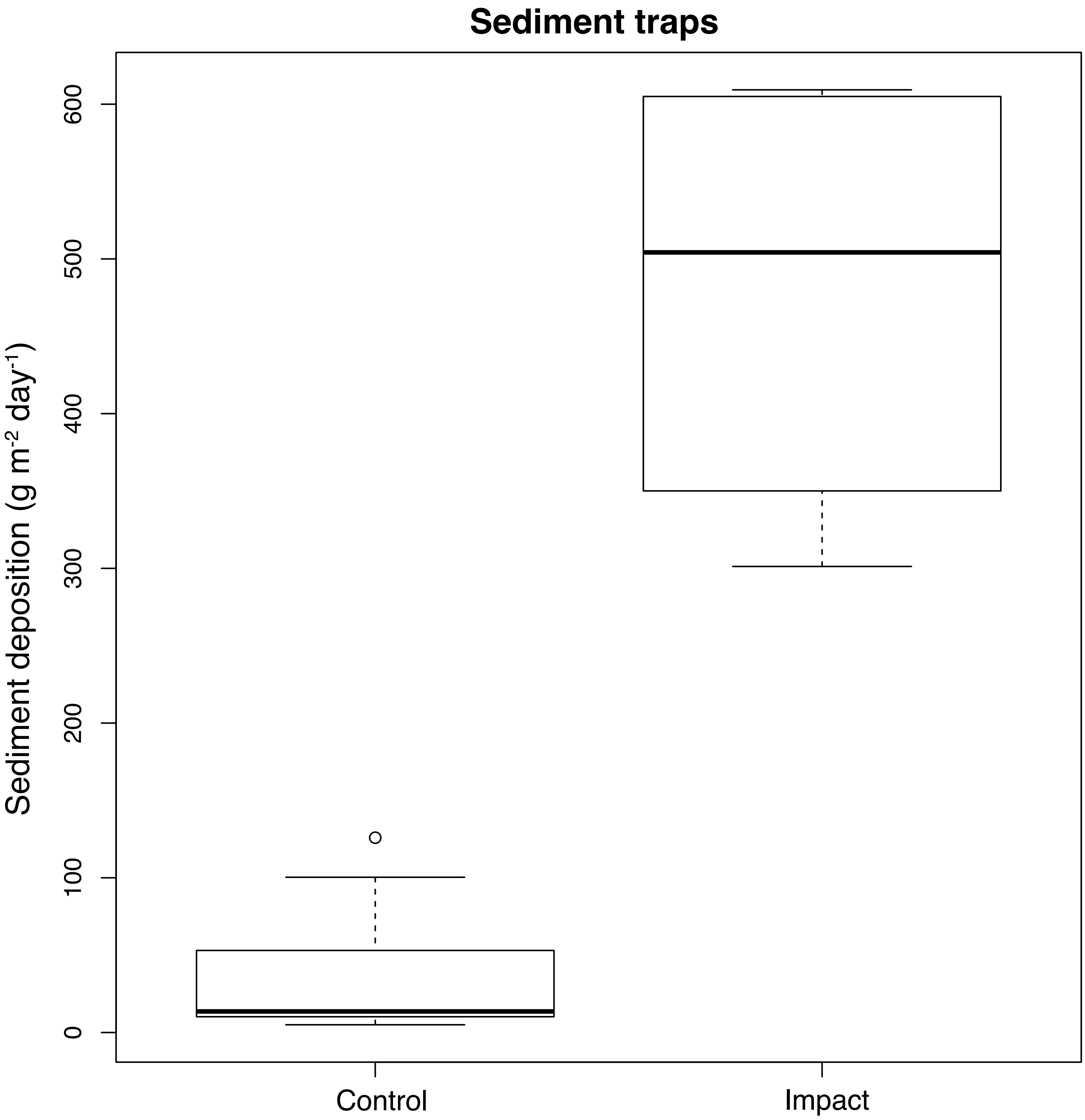
0
1 Km

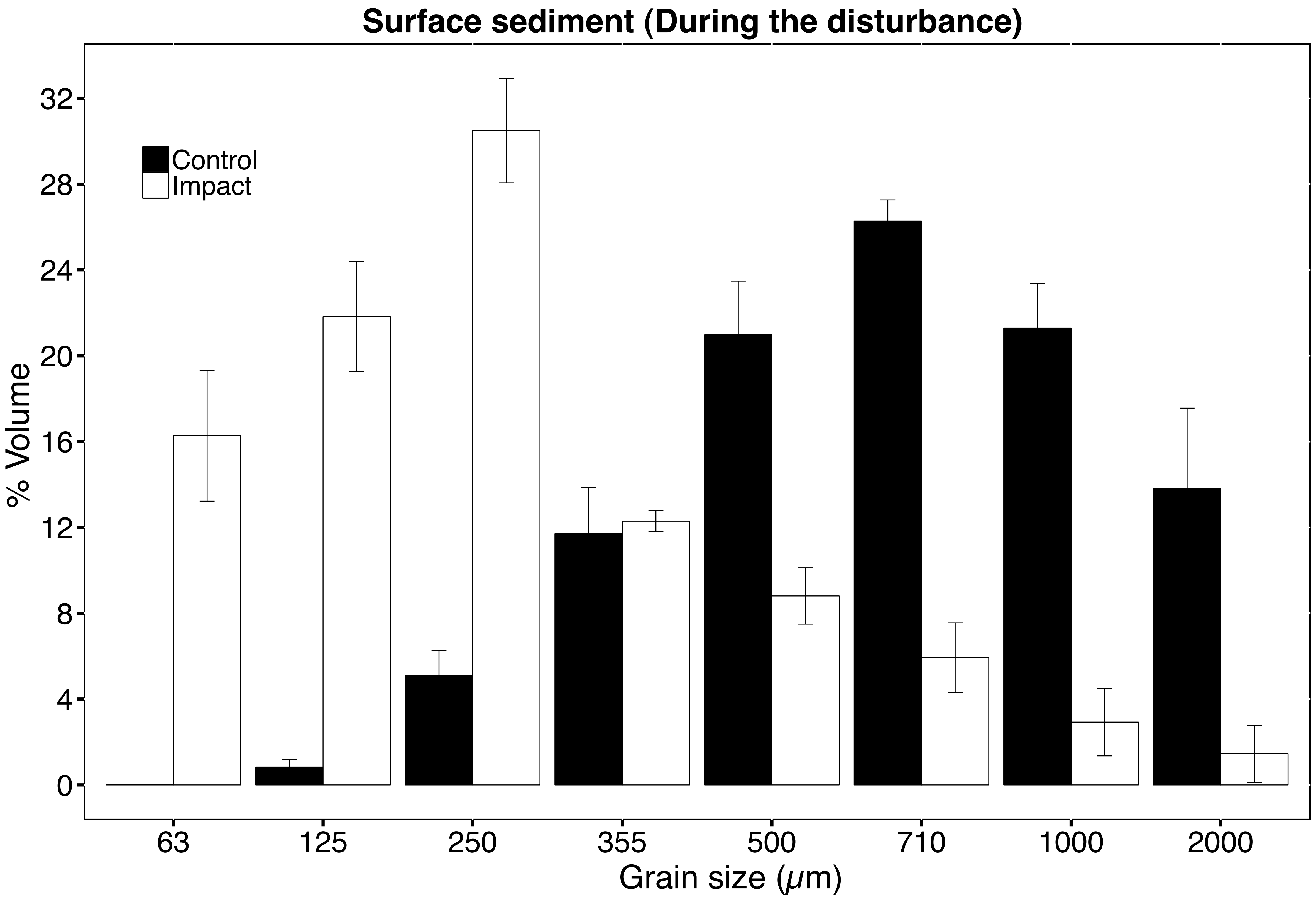


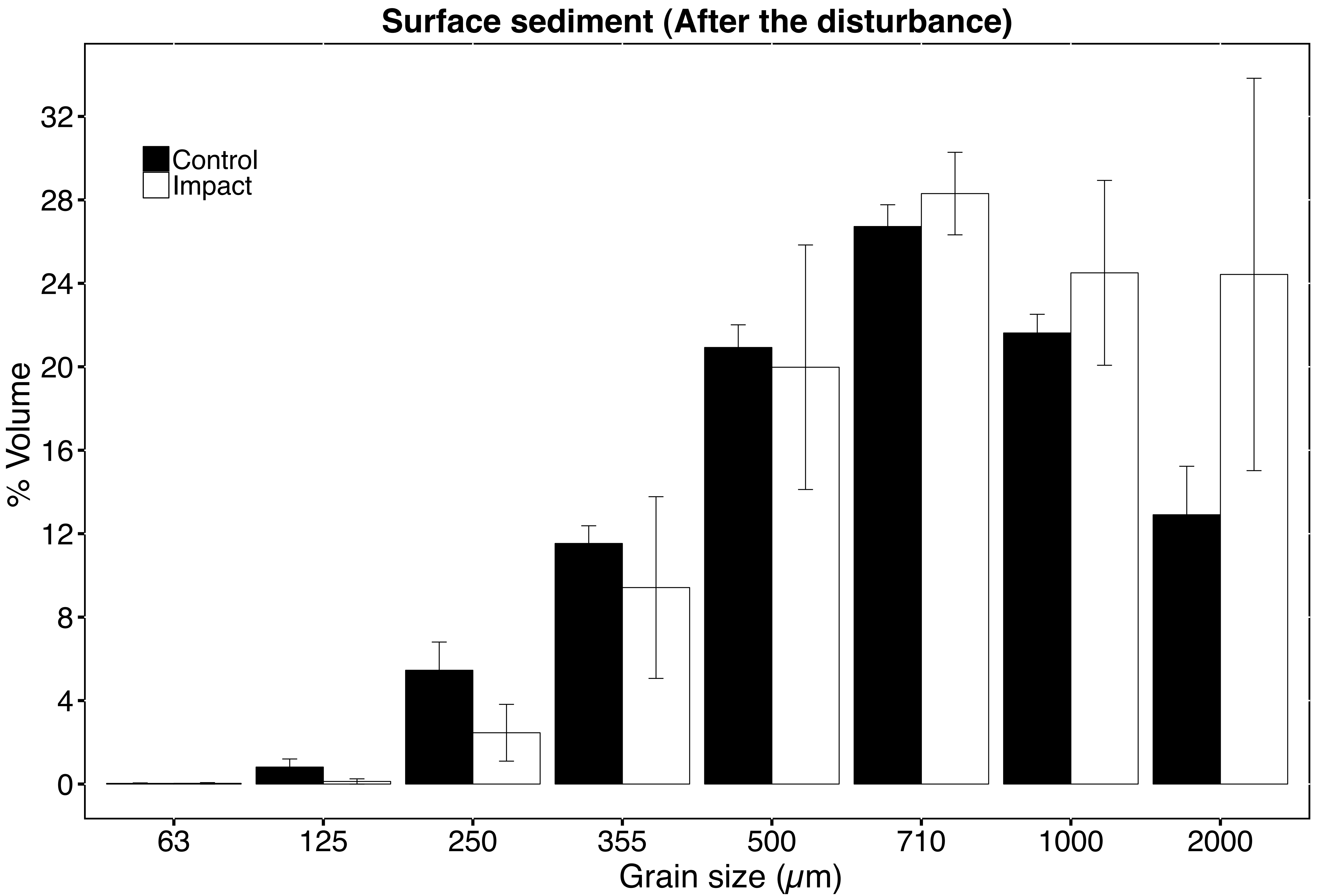
During disturbance

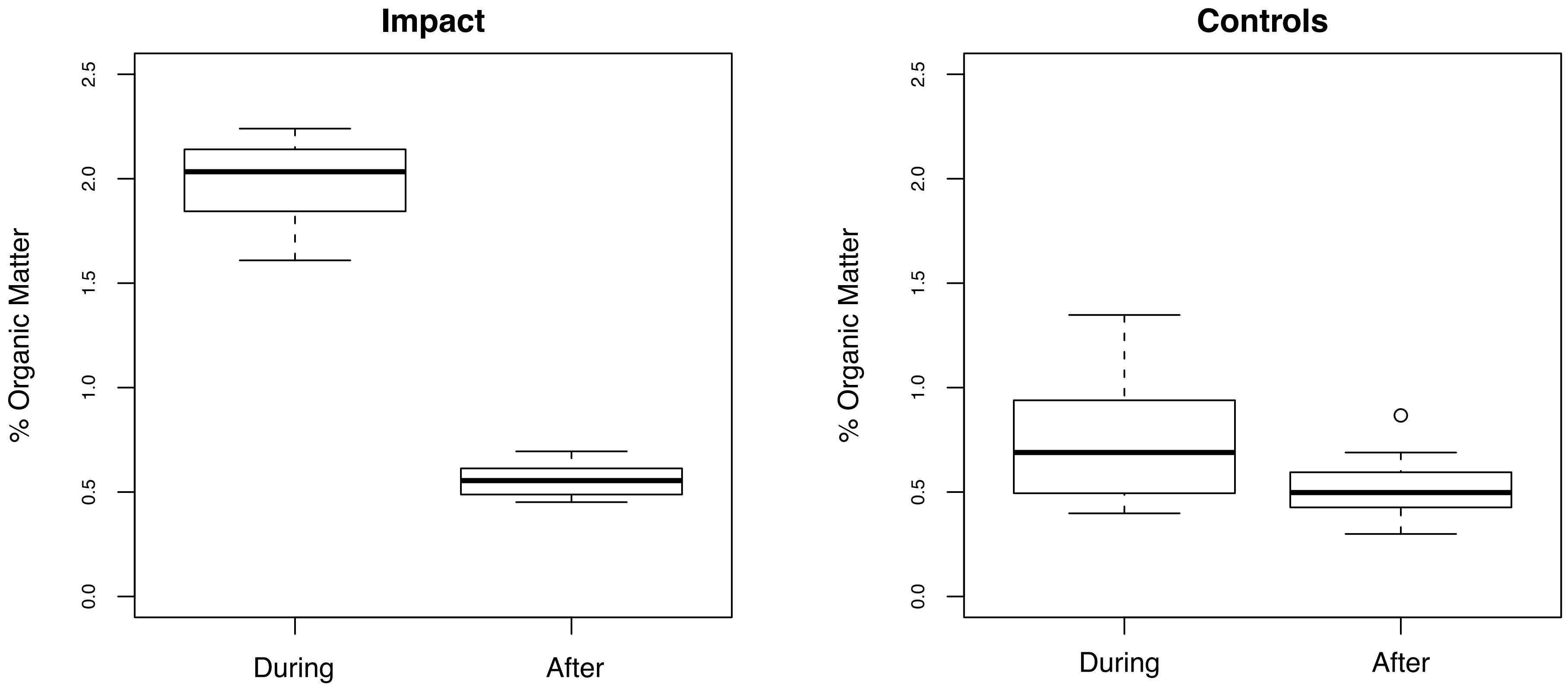


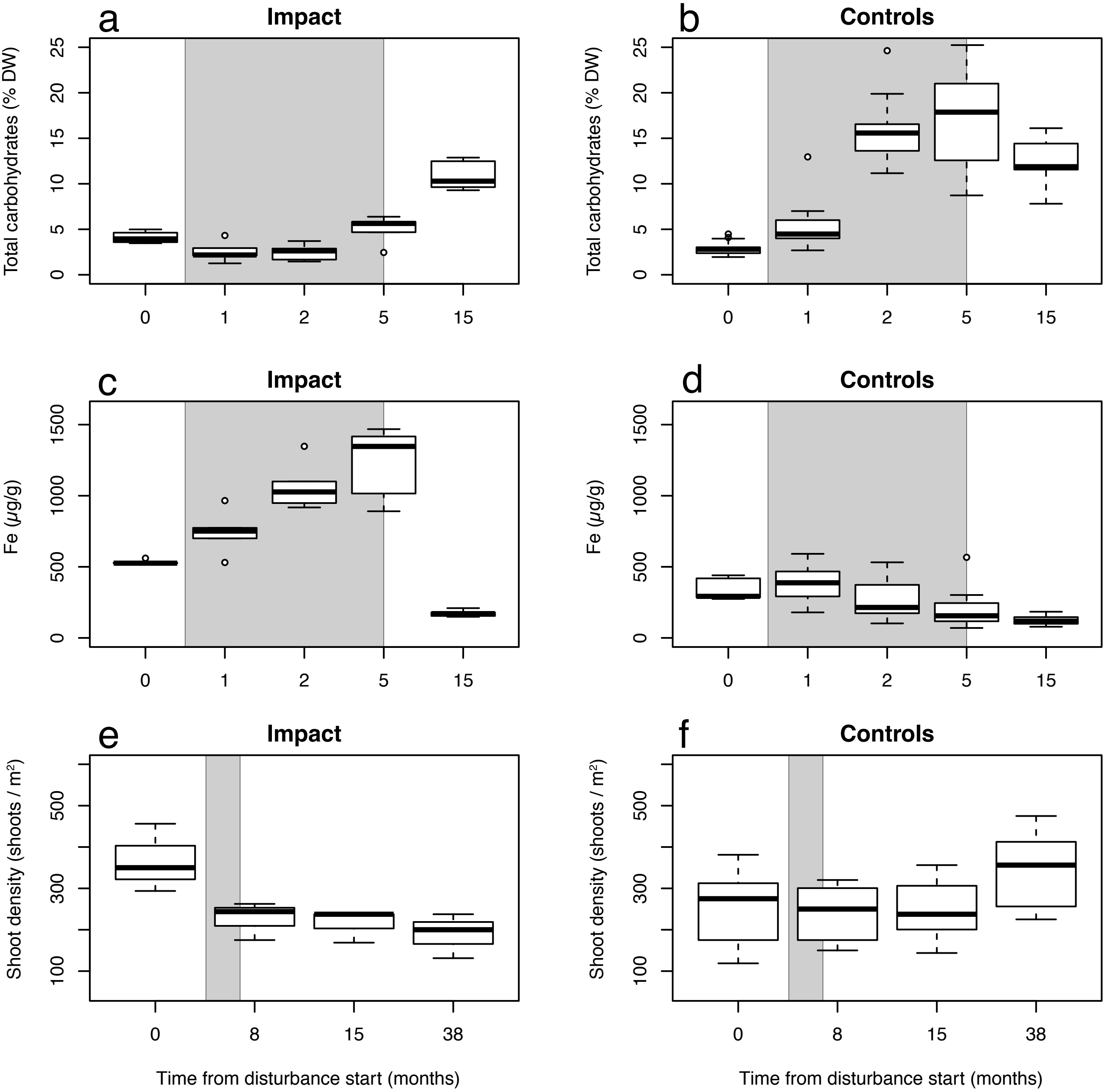
After disturbance


Sediment traps
Sediment deposition (g m-2 day-1)
0
100
200
300
400
500
600
Control
Impact


Surface sediment (During the disturbance)
Control
Impact
% Volume
0
4
8
12
16
20
24
28
32
63
125
250
355
500
710
1000
2000
Grain size (μm)


Surface sediment (After the disturbance)
Control
Impact
% Volume
0
4
8
12
16
20
24
28
32
63
125
250
355
500
710
1000
2000
Grain size (μm)


Impact
Controls
% Organic Matter
0.0
0.5
1.0
1.5
2.0
2.5
During
After


a
Impact
b
Controls
Total carbohydrates (% DW)
0
5
10
15
20
25
0
1
2
5
15
c
Impact
d
Controls
Fe (μg/g)
0
500
1000
1500
0
1
2
5
15
e
Impact
f
Controls
Shoot density (shoots / m²)
100
300
500
0
8
15
38
Time from disturbance start (months)
Time from disturbance start (months)