Natural England Commissioned Report NECR110

# Is ‘minimising the footprint’ an effective intervention to maximise the recovery of intertidal sediments from disturbance?

## Phase 1: Literature review

First published 01 March 2013

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# Foreword

Natural England commission a range of reports from external contractors to provide evidence and advice to assist us in delivering our duties. The views in this report are those of the authors and do not necessarily represent those of Natural England.

## Background

Activity on intertidal soft sediments can cause damage or disturbance of the habitat from abrasion, penetration and removal. For example, vehicles used to reach cockle beds for fishing or access to structures such as piers for maintenance works.

Natural England's advice in these situations has been based around sensitivities of habitats assessed on the MarLIN website **www.marlin.ac.uk/**. Generally, the advice has been that developers and developments should minimise the scale and extent, of the impact within the environment. This approach of 'minimising the footprint' generally leads to a more intense disturbance over a relatively smaller area. Recent anecdotal evidence suggests that recovery is quicker, and more complete, if the disturbance covers a wider area, but at a lower intensity.

This work was commissioned to review existing information on recovery of intertidal soft sediment habitats from different levels and intensities of disturbance.

The results will be used by Natural England to give more informed advice on appropriate mitigation in a range of intertidal sediment environments and should contribute to future work to:

- Carry out experimental work to test predictions of recovery from two types of disturbance: High intensity, small footprint; and Low intensity, large footprint.
- Develop a tool to allow developers, stakeholders and Natural England to assess the likely recovery of an intertidal sediment environment to different intensities of disturbance and therefore be able to quickly and efficiently provide appropriate advice.

This report should be cited as:

MAZIK, M. & SMYTH, K. 2013. *Is 'minimising the footprint' an effective intervention to maximise the recovery of intertidal sediments from disturbance? Phase 1: Literature review*. Natural England Commissioned Reports, Number 110.

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**Keywords** - Marine, sediment, intertidal, recovery, disturbance, advice

**Further information**

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ISSN 2040-5545

## TABLE OF CONTENTS

## 1. SUMMARY

Intertidal sediment habitats and the communities they support can be subject to significant physical disturbance resulting from both natural processes and anthropogenic activities (e.g. intertidal fishing activities, construction work and recreational activities). Whilst natural physical disturbance plays a key role in the structuring of soft sediment communities, the need for economic development means that coastal and intertidal areas are under increasing pressure of disturbance from human activities. The ecological and socio-economic importance of intertidal areas necessitates effective management of these pressures to enable economic development without compromising their integrity.

Current advice by Natural England is to minimise the impacts of anthropogenic physical disturbance by ensuring that disturbance is localised, accepting that disturbance intensity may be high. That is, a high level of impact over a small spatial area. Whilst there is evidence to suggest that soft sediment habitats and communities will recover following physical disturbance, the processes and timescale for recovery are not well understood, particularly in relation to different disturbance types, scales and intensities. It is hypothesised that the recovery potential of intertidal soft sediment habitats may be increased by increasing the area of impact, thus reducing its intensity, i.e. a low level of impact over a wide spatial area. This study aimed to review the evidence for recovery following different types of physical disturbance in different soft sediment habitats.

In general, there was strong evidence to suggest that intertidal soft sediment habitats and their communities have the potential to recover following physical disturbance of varying types and intensities and over varying spatial scales. However, there was a high degree of variability in the definition of recovery between studies and very few studies addressed the potential for recovery of ecological function. Indeed, whilst several methods for determining ecological function exist, many of them are only indicative (few provide a quantitative measure of function) and there is currently no clear guidance on what functions should be measured.

With respect to macrofaunal communities, the most widely used interpretation of recovery appears to be comparable species richness, abundance and, in some studies, community structure to pre-disturbance or reference levels. This generally appears to be achievable within 6-12 months (although may be longer where particularly large scale disturbance has taken place). The fastest recovery times were recorded when recovery was allowed to begin coincidentally with the summer larval recruitment period. However, if population structure and biomass are considered, recovery time can be up to five years, or greater. It is of note that this is an important aspect of ecological functioning. Meiofaunal communities generally recover rapidly (based on the definition of recovery provided by individual authors), within hours or days. In contrast, seagrass recovery times of between 2 and >18 years have been reported with poor dispersal ability being a key factor in recovery time. Where permanent habitat changes have taken place (e.g. changes in elevation, sediment composition or hydrodynamic regime), recovery to a pre-disturbance or reference state may be unrealistic.

The evidence for recovery following physical disturbance is variable due to:

- Definition of recovery and the physical/biological parameters used to measure it.

- The state of the community when the processes of recruitment and colonisation leading to recovery began (total or partial defaunation). This relates to the scale and

intensity of disturbance but also to the longevity, fragility and tolerance of the species present.

- Proximity to undisturbed habitat and a source of adult colonists.

- Degree of physical modification to the sediment (changes in elevation, particle size, redox gradients, porosity).

- The level and nature of disturbance experienced by the reference community. That is, how representative is the reference community of undisturbed conditions.

- The biological feature and the parameters being monitored. In the context of this study, biological features include meiofauna, macrofauna and seagrass.

- Differences in the scale and source of the disturbance. Recovery times appear to increase with the spatial scale of disturbance although the relationship between scale and recovery is unclear due to variation in the parameters being measured and the point at which recovery is considered to be complete.

- Temporal variability in community. Some studies showed convergence of community structure and then divergence several months later and variability in the reference community has also been found to imply recovery. However, few studies have compared temporal patterns in recovering and reference communities, or convergence/divergence between the two after the point at which recovery is considered to be complete.

- The monitoring timescale and frequency which influence the recorded recovery time in that frequent sampling will enable detection of recovery as it happens. In contrast, annual monitoring programmes may only enable detection of recovery several months after it has occurred, potentially giving a false indication of recovery time.

- The time taken to achieve recovery is habitat and disturbance type (and a combination of both) specific and differences in these parameters between studies mean that the effects of spatial scale and intensity cannot be separated from those associated with habitat, disturbance and experimental design parameters.

Overall, there was a broad indication that widespread impacts over large areas and/or high intensity disturbance will lead to prolonged recovery times. However, the variability in the definition of recovery between studies, together with variability between habitats, disturbance types and experimental design, makes it difficult to draw any firm conclusions about the relationship between disturbance scale and intensity and recovery. Furthermore, the timescale of many of the studies reviewed was insufficient to confirm recovery with many studies reporting recovery times of >9 or >12 months (i.e. greater than the duration of the study). Finally, very few studies have replicated the scale and intensity of disturbance associated with construction work where sediment excavation or smothering to depths of tens of centimetres can occur over large areas. Due to the nature of such activities, it might be difficult to minimise the impacts by spreading the disturbance out.

## 2. INTRODUCTION

Intertidal soft sediment habitats and the communities they support are subject to continual physical disturbance due to natural influences acting at scales ranging from a few centimetres (e.g. infaunal bioturbation, Snelgrove & Butman, 1994) to kilometres (e.g. storms Zajac & Whitlatch, 2003). Soft sediments are relatively easily disturbed due to their physical nature (e.g. low degree of consolidation, cohesion and high porosity and permeability) (Zajac et al., 1998). In ecological terms, disturbance is defined as ‘an event initiating species population change due to mortality or removal and/or a change in the resource base of the community’ (Zajac & Whitlatch, 1982). Sousa (1985, in Barnes & Conlan, 2007) expanded on this and described disturbance to biological systems as ‘a discrete, punctuated killing, displacement, or damaging of one or more individuals (or colonies) that directly or indirectly creates an opportunity for new individuals (or colonies) to become established’. The impact of a disturbance event and subsequent recovery (return to a pre-disturbance state) is dependent upon the resilience of the system. That is, the amount of disturbance which a system can absorb and still remain in the same condition (Holling, 1973, in Lee et al., 2011; Peterson et al., 1998) or how quickly the variables return to equilibrium following perturbation (Pimm, 1984). Resilience, in turn, is therefore related to the spatial scale and intensity of the disturbance.

Natural physical disturbance plays a key role in controlling soft sediment community structure on a spatial and temporal scale (Grassle & Sanders, 1973; Hall, 1994; Schratzberger, 2009). The intermediate disturbance hypothesis (Connell, 1978; Dial & Roughgarden, 1998) states that maximum species diversity occurs when the level of disturbance is intermediate in frequency and implies a predictable relationship between the frequency and intensity of disturbance and the resultant species richness (Dernie et al., 2003a). At low levels of disturbance, organisms with high competitive abilities will dominate, preventing the existence of opportunistic species (Dial & Roughgarden, 1998). This dominance is reduced or prevented under conditions of intermediate disturbance, thus enabling maximum species diversity by enabling the coexistence of competitive and opportunistic species. However, high levels (intense) of disturbance may lead to the loss of all species and the spatial scale to which this hypothesis can be applied is not known (Thrush & Dayton, 2002).

Although natural disturbance events can have large repercussions, anthropogenic disturbance due to industrial and recreational activity adds significant pressure on these habitats and the recovery process is poorly understood. Anthropogenic disturbance in intertidal areas results from fishing activities (e.g. cockle harvesting), dredging, beach recharge, construction work and recreational activities. Such activities result (to varying degrees) in sediment disturbance, removal, smothering, compaction, liquefaction and changes in sediment transport dynamics over various spatial and temporal scales. The impacts of disturbance on benthic communities include complete defaunation, due to mortality, displacement and migration, and changes to community structure and function. Indirect effects include increased susceptibility to disease (e.g. wasting disease in *Zostera*), parasites and predation.

Intertidal sedimentary habitats and their communities are vital to ecological functioning: the direct and indirect regulation of ecological processes by the component species within an ecosystem (Peterson et al., 1998). In physico-chemical terms, such processes may include biogeochemical cycling), the degradation of organic material and contaminants, sediment disturbance, resuspension and transport (Thrush & Dayton, 2002). In biological terms, ecological processes may be regulated through productivity and trophic interactions (Paterson et al., 2011). These functions are the basis of vital ecosystem services and societal benefits (Costanza et al., 1997) such as food provision, coastal defence and nutrient/carbon sequestration (Thrush & Dayton, 2002) and following anthropogenic impacts, it is important that ecological function is restored.

Since coastal economic development is of paramount importance, increased pressure from developers on intertidal habitats, especially in estuaries, is inevitable. Understanding the recovery process of benthic habitats and communities is fundamental to the effective management and mitigation of impacts. The current lack of understanding, together with legislative constraints, hinders the commissioning of proposed developments (Jackson, 2011).

## 2.1. Aims

The rate of recovery of community structure and function, following disturbance, relies initially on intensity, spatial scale and degree of habitat damage (Bell & Devlin, 1983; Thrush et al., 1996) and is dependent upon physical recovery of the habitat (Dernie et al., 2003a). The subsequent success of organism immigration and recruitment is determined by (a) proximity of a community with a similar structure to that of the original community (Santos & Simon, 1980) and (b) mobility, dispersal and timing of recruitment of benthic organisms in relation to the season in which recovery can start (Beukema et al., 1999; Herkul et al., 2011). These factors strongly influence the trajectories (sequence and rate) of recovery, as will successive competitive interactions, modification of the sediment by organisms, their resistance to change and the resilience of the system as a whole (Elliott et al., 2007; Borja et al, 2010). Hence, recovery potential and rate could be greatly improved by carefully timing activities that cause disturbance. Similarly, current mitigation involves minimising the spatial scale of impacts, often resulting in localised high level disturbance, yet evidence (Dernie et al., 2003a; Gray & Elliott, 2009) suggests a higher recovery potential is associated with more widespread, low-level disturbance.

Natural England currently advises developers to minimise the spatial scale of their impacts 'minimising the footprint', accepting that this usually leads to a high intensity of disturbance in a limited area. This study will provide a review of existing literature on the recovery of soft sediment habitats and communities following physical disturbance. It will identify the effects of different combinations of pressure, intensity, scale and sediment type on recovery and use these to make assumptions about recovery from other pressures and activities where there may be knowledge gaps.  This will provide the basis for wider experimental studies to test the hypothesis that low intensity disturbance over a wider area may enable intertidal sediment communities to recover more rapidly than under the present approach of minimising the footprint.

The review was therefore targeted at the question:

**Is 'minimising the footprint' an effective intervention to maximise the recovery of intertidal sediments from disturbance?**

## 2.2. Approach and sources of literature.

Literature databases accessed include: ISI Web of Science, Science Direct/SciVerse, Scopus, JSTOR, BIOME, ETHoS, www.findathesis.com, Scirus and PubMed.  Further searches were carried out using Google Scholar and internet searches, Conservation Agency, EA, Defra, Cefas and MarLIN online catalogues and discussions with contacts within these organisations. A significant amount of literature is also held in IECS (EndNote Libraries).

Articles were assessed for inclusion or exclusion as follows (based on principles established in CEBC, 2009):

- Obviously irrelevant titles were rejected. Broadly relevant titles were considered but rejected if the article did not contain relevant information.
- Articles which did not contain useful figures/tables/data were rejected.
- Articles which did not directly relate to the question were rejected. Those which were indirectly related (e.g. wider ecology of recruitment dynamics, dispersal mechanisms, subtidal sediments) were included if they met all other criteria.
- Studies providing a clear quantitative pattern of recovery and rate of recovery were included. Conceptual and descriptive studies were excluded although the literature cited was considered.
- Efforts were made to avoid biased and emotive articles; and studies that did not have a high degree of objectivity. Emphasis was, therefore, placed on articles containing primary details of experimental work and data with an experimental design to enable objective hypothesis testing, rather than reviews.
- Articles must have clearly stated aims, objectives and hypotheses to be included.
- Articles were excluded if the experimental design (replication, control and impact treatments, repeatability), methods and data treatment were inappropriate/illogical.
- Studies which were not genuinely informative were excluded.
- Priority was be given to peer reviewed scientific literature and Government Agency reports (written by qualified, practising scientists) but others were considered provided that they met the above criteria for objectivity, experimental design etc.
- Most of the accepted articles were from peer reviewed literature with experimental design of similar, suitable, quality. Some information was included from ‘grey’ and un-published literature but experimental design in many of these was compromised by time and resources and many were reviews of primary evidence included elsewhere.

### 2.2.1. USE OF ECOLOGICAL TERMS.

The terms recovery, colonisation/re-colonisation and recruitment are commonly used in studies of individual species, populations and communities yet they are often used interchangeably. For the purpose of this study, these terms have been used as used by the author of a specific paper but, in general, **recruitment** and **colonisation** are considered to refer to the successful settlement of larval recruits and adult migration into newly available habitat, respectively. **Recovery** is facilitated by recruitment and colonisation, which may be described as ‘rate’ processes, and is considered to be a successional end point. Whilst use of the term recovery is highly variable, it is generally considered to be the return to a pre-disturbance state or reference condition and may refer to any of the parameters describing a species, population or community.

# 3. CAUSES AND IMPACTS OF PHYSICAL DISTURBANCE ON INTERTIDAL SEDIMENT HABITATS AND COMMUNITIES.

This section summarises the causes and impacts of physical disturbance on intertidal sediment habitats. It does not attempt to exhaustively review all the literature on this topic as that is not the aim of the review. The focus of this review is on recovery, however, the following section provides useful context to the review of recovery in section 4.

## 3.1. Naturally induced physical disturbance.

A comprehensive review of the types of disturbance affecting soft sediment communities is given by Probert (1982) and covers natural physical and biologically induced disturbance and anthropogenic disturbance. On a large scale (km), the most frequent and regular form of natural physical disturbance to which intertidal organisms are exposed is the diurnal tidal cycle where organisms are influenced by wave action, tidal current, periods of immersion and emersion and salinity fluctuations (Thistle, 1981; Filho et al., 2006). During tidal inundation, there is potential for the resuspension and re-distribution of sediment and organisms, influencing the composition and structure of sediment habitats and communities. During storms, the potential for sediment transport and deposition increases with extreme events potentially leading to large scale alteration of the physical and biological characteristics of a habitat (Posey et al., 1996). Furthermore, during stormy periods, flood waters and run-off from the land may introduce significant amounts of terrigenous (terrestrial) sediments to intertidal habitats creating unfavourable conditions for many benthic species (Cummings et al., 2009). A reduction in salinity may also result from prolonged periods of high rainfall, particularly in upper estuarine areas. Other large scale forms of physical disturbance include ice scour and ash deposition following volcanic activity (Barnes & Conlan, 2007). In temperate regions (particularly the UK), these forms of disturbance are unlikely to occur but their impacts on sedimentary environments are similar to some forms of anthropogenically induced disturbance, particularly in the case of ice scour. Finally, sea level rise alters the tidal flooding regime and in embanked areas, may lead to changes in erosion/deposition dynamics and beach profile.

On a small scale, bioturbation by infaunal and epifaunal organisms represents a continual source of physical disturbance. Benthic (sea-bed) organisms may be classed as 'ecosystem engineers' in that their activity has a profound effect on their environment (Snelgrove & Butman, 1994). Such effects result from burrow and tube construction, irrigation, feeding (deposit and suspension), defecation and movement within the sediment (Kristensen et al., 2012). The resultant creation of pits, mounds and voids within the sediment leads to changes in sediment stability, porosity, grain size distribution, chemical flux, organic matter decomposition (Mazik et al., 2008). Nutrient and organic fluxes are drivers of microbial processes and primary productivity (e.g., Thrush & Dayton, 2002), factors which are also related to sediment grain size, and bioturbation is known to affect the settlement and survival of benthic organisms (Snelgrove & Butman, 1994).

## 3.2. Anthropogenic physical disturbance.

Anthropogenic disturbance can occur at a range of scales and intensities from localised low level trampling associated with recreational activities to more widespread sediment compaction, sedimentation and sediment excavation. Anthropogenic activities can be broadly divided into those associated with recreation, fishing, construction work, coastal defence and dredging/dredge disposal:

- **Recreational activities** including sunbathing, walking, horse riding and collection of animals result in varying degrees of trampling and digging, depending upon the number of people and the frequency with which they use an area (Keough & Quinn, 1998; Rossi et al., 2007). More significant disturbance may be caused by the use of quadbikes and other off-road vehicles (quad bikes, motorcycles, all terrain vehicles, and 4x4 vehicles) in intertidal areas (Schlacher & Morrison, 2008; Tyler-Walters & Arnold, 2008; McKnight, 2010) and by boat mooring which cause sediment scouring as chains from mooring buoys move with the tide (Herbert et al., 2009).

- **Intertidal fishing** activities, including shellfish collection (Allen, 1995; Thrush et al., 1995; Hall & Harding, 1997; Bell & Walker 2006) and bait digging (McLusky et al., 1983; Carvahlo et al in press), range in intensity from single collectors accessing the shore by foot to the widespread use of tractor towed dredges and hydraulic suction dredgers. In areas of intensive shellfish collection, heavy plant may also be used in intertidal areas (pers. obv).

- **Coastal defence** work may include habitat creation schemes (e.g. French, 2006), the construction and maintenance or upgrade of existing sea walls or groynes (Airoldi et al., 2005) and beach nourishment schemes (Peterson & Bishop 2005; Speybroeck et al., 2006; Bolam et al., 2003). **Construction work** in coastal areas includes (for example) pipeline and offshore energy cable installation, port/marina construction and extension work, artificial reef construction, the installation of renewable energy devices and barrages and landclaim (e.g. Moreira, 1988; Lewis 2002; 2003; Airoldi & Bulleri 2011). This involves the use of heavy machinery and has the potential to cause significant compaction, liquefaction, excavation, smothering and churning of sediments.

- **Dredging and dredge disposal,** involving the removal or redistribution and emplacement of sediment, respectively, are predominantly subtidal activities but may occur sufficiently close to the coast to impact upon intertidal areas.

- **Military activities** (bombing ranges)

## 3.3. Impacts of physical disturbance.

The impacts of physical disturbance are generally greatest in areas of low natural disturbance where the organisms are less well adapted to withstand physical stress. The impacts of physical disturbance lead to sediment compaction (Tyler-Walters & Arnold, 2008) or liquefaction/fluidisation (Hall & Harding, 1997), removal of surface sediment layers (e.g. during storms, Zajac & Whitlatch, 2003) sediment homogenisation (Thrush & Dayton, 2002) and direct smothering as a result of excavation work (Bolam & Whomersley, 2003). In turn, these impacts modify the local hydrodynamic regime leading to changes in erosion and deposition dynamics and changes in the structural properties of the sediment (particle size distribution, organic content, consolidation, porosity and cohesion). Hydrodynamic modification also arises from the presence of engineering structures. Similar modification of the sediment, at a smaller scale, is also induced by biological activity. Indirect effects of physical disturbance include the release of sediment nutrients and contaminants sequestered at depth within the sediment. Whilst the effects of contaminants on biota are beyond the scope of this study, it is worth noting that recovery times could be increased by chemical changes in the sediment.

The impacts of physical disturbance on soft sediment communities are reviewed in detail by Hall (1994). Briefly, they include complete or partial defaunation resulting from organism death, uprooting of plants, organism damage, emigration and increased susceptibility to predation, disease and parasites. For surviving organisms, there are increased energetic costs associated with physiological responses to the new conditions, reconstructing collapsed burrow structures or migrating to alternative habitats which may ultimately impact upon growth and reproductive output depending on the life history of the species and the timing of the disturbance. The loss or reduced performance of species leads to changes in population and community structure which then has wider implications for predators (Lewis et al., 2003) and ultimately for ecosystem function.

# 4. EVIDENCE FOR RECOVERY OF INTERTIDAL SEDIMENT HABITATS AND COMMUNITIES FOLLOWING DISTURBANCE.

The literature reviewed in this study covers a wide variety of physical disturbance types, intensities and impacts ranging from small scale, localised (centimetres to meters in area and cm in depth) disturbance caused by, for example, bioturbation and trampling to large scale (km in area and tens of centimetres in depth) disturbance caused by coastal construction work and flooding events. The examples provided are predominantly from intertidal and shallow subtidal areas although some offshore examples have been used where considered appropriate due to similarities in the recovery mechanisms. Additionally, the review includes studies of experimentally induced disturbance and monitoring studies following actual anthropogenic disturbance such as beach recharge, bait digging and pipeline construction. The impacts from this range of disturbance events include temporary displacement of organisms through to complete defaunation at different spatial scales.

Whilst bioturbation is not classed anthropogenically induced disturbance, it represents small scale low impact physical disturbance from which recovery can be rapid. Similarly, flood events are natural (although anthropogenic activities can increase the potential for and the impacts of flooding on coastal environments) but they represent hydrodynamic impacts and changes in water quality (particularly turbidity and the introduction of terrigenous sediments to intertidal areas) which may also be induced by, for example, coastal construction work. Finally, the impacts of hypoxia have been included since the recolonisation mechanisms (but perhaps not the recovery trajectories) of areas recovering from hypoxia are similar to those in areas recovering from physical disturbance. Furthermore, the presence of construction materials and equipment (e.g. boards to protect the sediment surface) on intertidal sediments may induce anoxia over prolonged periods.

The evidence for recovery following different physical disturbance types in different habitats is summarised in Tables 1-5 and described in detail in sections 4.1-4.3. These sections are divided into meiofauna (although the literature was limited), macrofauna and segrasses and, where a large number of studies were found (predominantly relating to macrofauna), is further divided into anthropogenic disturbance, experimentally simulated anthropogenic disturbance and biological disturbance.

## 4.1. Meiofauna

This section examines studies where recovery of meiofauna has been the main focus and where the sampling design and techniques (core size, separation techniques) have specifically targeted meiofaunal organisms. The number of studies relating specifically to meiofauna is limited and most relate to small-scale (<5 $m^2$) biological (bioturbation) or experimental (e.g. trampling) disturbance of the sediment.

**A summary of documented recovery times for meiofauna is given in Table 1.**

Recovery of meiofaunal communities appears to be rapid, occurring within hours or days (e.g. Sherman & Coull, 1980; Chandler & Fleeger, 1983; Cross & Curran, 2004; Johnson et al., 2007). However, recovery time is dependent upon the timing and the frequency of the disturbance. For example, Cross & Curran (2004) recorded meiofaunal recovery within 48 hours of disturbance by feeding rays (feeding pit formation) in July but following recruitment of juvenile fish and crustaceans the following August, recovery took 72-168 hours. The longer recovery time was attributed to continuous feeding (and therefore continuous disturbance) by new recruits and adult rays. Similarly rapid meiofaunal recovery was recorded by Johnson et al. (2007) following trampling of intertidal muddy sediments in relation to bait collection (crab tiling). Species richness, abundance and community structure had recovered to control (undisturbed) levels over 12-36 hours and it was suggested that low level disturbance such as trampling may cause meiofaunal organisms to migrate deeper into the sediment and that vertical migration to the surface may occur shortly after the disturbance has ceased. If this hypothesis is true, such rapid recovery might not be expected in areas subjected to more severe disturbance such as sediment removal and compaction.

Recovery trajectories may differ according to the prevailing environmental conditions. For example, Schratzberger et al. (2006) monitored the development of meiofaunal communities following the emplacement of dredged material in four different areas. All four areas received dredged material (fine grained muddy sediments) from the same source yet considerable differences in the nematode species, and therefore functional groups, which colonised were found. Whilst the import of organisms with the dredged material could not be ruled out, differences in species composition were attributed to differences in wave disturbance (exposure), elevation and consolidation with spatial variation in species composition reflecting the morphological and functional traits necessary to successfully colonise the various available habitats. Furthermore, the rate of colonisation (expressed as the time taken to achieve a three-fold increase in abundance) in exposed sites was slower than that in more sheltered sites with a difference of approximately 5 months.

### 4.1.1. SUMMARY

In general, based on the limited number of studies in Table 1, meiofaunal recovery occurs within hours or days. Recovery would appear to be driven by movement of individuals into the disturbed area from adjacent areas, facilitated by natural process, at least for small scale disturbance. The exception to this is Thrush et al. (1996) who only studied the larger meiofauna, together with the macrofauna and no distinction was made between the two groups.

**Table 1. Evidence for recovery of meiofaunal communities. > refers to recovery times beyond the duration of the experiment or monitoring period.**

| Reference | Organisms | Sediment type | Disturbance type | Disturbance area/patch size | Recovery time | Definition of recovery |
|---|---|---|---|---|---|---|
| Cross & Curran, 2004 | Meiofauna<br>USA | Intertidal sand | Biological<br>Feeding pits created by rays | 2086-3027 $cm^2$<br>8847-16,253 $cm^3$ | 72-168 h | Equivalent abundance of total meiofauna and key meiofaunal groups (Nematodes, Copepods etc) |
| Thrush et al. 1996 | Macrofauna and large meiofauna (250 µm sieve)<br>New Zealand | Intertidal fine/very fine sand | Experimental defaunation (induced anoxia) on troughs and ridges.<br>No change to physical structure of the sediment | 3.24 $m^2$<br>0.81 $m^2$<br>0.203 $m^2$ | >9 months*<br>>9 months*<br>>9 months* | Return to ambient conditions in terms of species present, mean abundance and community structure |
| Johnson et al. 2007 | Meiofauna<br>UK | Intertidal mud | Trampling associated with crab tiling (trampled 6 times over 2 weeks) | 12-36h | 144 h | Return to undisturbed (control plot) community structure |

*No distinction was made between macrofaunal and meiofaunal recovery, hence the comparatively long recovery time.

## 4.2. Macrofauna

A large number of studies relating to macrofaunal recovery following a wide range of physical disturbance types were found. Therefore, the present section has been divided into:

- Studies where recovery following large-scale anthropogenic disturbance has been monitored. This includes construction work, fishing activities, trampling, moorings and beach recharge works (Section 4.2.1).
- Studies where the impacts of the above activities have been experimentally created in order to provide an indication of the likely recovery times following such impacts (Section 4.2.2).
- Studies representing biological disturbance (e.g. bioturbation) where both natural and experimentally induced disturbance have been examined (Section 4.2.3).

### 4.2.1. STUDIES OF ANTHROPOGENIC DISTURBANCE

**A summary of documented recovery times for macrofauna following anthropogenic disturbance is given in Table 2.**

#### 4.2.1.1. CONSTRUCTION WORK

Coastal construction work, such as pipeline and cable installation, coastal defence work and the construction or extension of ports and marinas can involve significant (in terms of spatial area and depth) excavation and emplacement of sediment. Access to intertidal areas by construction traffic and workers causes trampling, churning and compaction of sediments. Hence, this form of physical disturbance can have more significant impacts than those associated with natural disturbance, recreational activities or intertidal fishing activities.

A pipeline construction scheme on the southern Irish coast resulted in complete defaunation of sediments within the immediate vicinity of the construction work (Lewis et al., 2002; 2003). The spatial extent and nature of this disturbance is significantly greater than that created in most of the studies cited in this report, being 360 m in length, 50 m wide and 3 m deep. Lewis et al. (2002) monitored the recovery of key invertebrate species (*Hediste diversicolor, Scrobicularia plana* and *Tubifex* spp.) for 6 months following disturbance and found that whilst the abundance of *Hediste diversicolor* recovered reasonably quickly (comparable abundance to pre-disturbance abundance after 6 months), *S. plana* had not colonised the site at all. Rapid recovery of *H. diversicolor* was also noted by Bolam et al. (2004) and was attributed to the opportunistic lifestyle of this species. Further monitoring (Lewis et al., 2003) revealed that recovery in terms of abundance of these species had occurred 1 year post disturbance. However, overall community structure was not determined.

#### 4.2.1.2. BEACH RECHARGE AND BENEFICIAL USE SCHEMES

Beach recharge (or nourishment) involves the mechanical or hydraulic placement of sediment onto the shore and is now a commonly used method of protecting eroding beaches (Speybroeck et al., 2006). It also provides a solution to the disposal of dredged material, creating an opportunity for the beneficial use of previously unwanted material (Bolam & Whomersley, 2003). Beach nourishment and beneficial use schemes have been successful in the creation of naturally functioning intertidal habitats and the practise is largely considered to be environmentally beneficial (Speybroeck et al., 2006). However, the deposition of large amounts of sediments in intertidal areas results in the smothering of existing communities and recovery is not always successful, often due to differences in the physical structure of the habitat (e.g. Peterson et al., 2006). Such differences may include sediment grain size distribution, organic content, beach profile and elevation (Bolam & Whomersley, 2003; Bolam et al., 2010).

Bolam & Whomersley (2003) studied macrofaunal recolonisation following the deposition of 49-57 cm of sediment on the intertidal marshes of the Crouch estuary in SE England. Whilst species richness, diversity and abundance were all representative of reference areas after 3 months, community structure was not, even after 18 months, and this was attributed to the comparatively high elevation of the re-charge sites (3.7-4.1 m compared to 3.6-3.7 m in the reference area). Similar findings are documented by Bolam et al. (2006), 42 months after dredged material emplacement. It was argued that reference areas of similar elevation did not exist (Bolam & Whomersley, 2003) and this study, together with Bolam et al. (2006) highlights the importance of defining appropriate criteria for recovery, assuming sufficient knowledge exists. The lack of convergence between impacted/recovering and reference areas should not necessarily be interpreted as an adverse impact or failure to recover (Bolam et al., 2006). That is, where spatial variation in habitat parameters would prevent the convergence of community structure between reference and recovering areas, an alternative assessment of ecological quality may be more appropriate. However, in such situations, the long term maintenance of the new community should be considered since in high elevation areas where high levels of accretion are allowed to occur (e.g. the presence of features to retain accreted sediments), the long term existence of typical soft sediment communities may not be feasible (Mazik et al., 2010).

Bolam et al. (2010) highlighted the importance of physical conditions in determining the rate of recovery. In a comparison of macrofaunal colonisation rates of 4 areas influenced by dredged material emplacement, delayed community development was recorded from high elevation, sheltered areas compared to lower elevation areas with a difference of approximately 12 months. A similar response by meiofaunal invertebrates was noted by Schratzberger et al. (2006).

#### 4.2.1.3. INTERTIDAL FISHING AND BAIT DIGGING

Bait collection from the intertidal zone causes varying degrees of physical damage and disturbance, as well as altering the community structure through removal of organisms. Two of the most common intertidal fishing activities are harvesting bivalves (e.g. *Cerastoderma edule, Mercenaria mercenaria*) and the lugworm (*Arenicola marina*). The disturbance caused by clam digging substantially affects both the target species and other benthic fauna, decreasing community richness and increasing community patchiness (Griffiths et al., 2006). *Arenicola marina* may be a relatively sensitive species, not only in terms of physical disturbance, but also in other respects. For example, mass mortalities (up to 99%) reported by Olive & Cadnam (1990) in South Wales were tentatively attributed to an algal bloom.

Hand collection methods for bait and bivalves are generally considered to have less impact on the environment than mechanical harvesting techniques, however, the intensity of the hand collection method itself may have an effect on the recovery time. Collection of *A. marina* for bait, by digging holes 20-30 cm deep and 40-50 cm in diameter, leaves marked differences in the number of worms inhabiting the area, with infilling of basins and recovery of the abundance taking 24-45 days. Disappearance of mounds (created by excavation) and subsequent recovery of the population can take 4 months. However, re-filling excavated trenches enabled recovery after only 22 days (McLusky et al 1983). Patch size must also be considered in assessing the affects of disturbance and the potential for recovery. Hand raking for cockles caused a threefold increase in damage to undersized cockles when compared to unraked control sites. Recovery of species richness and abundance after disturbance occurred after 56 days in small 6m$^2$ plots, however in this timeframe, large disturbed plots (36m$^2$) did not show recovery (Kaiser et al., 2001).

Mechanised methods appear be have longer term impacts on the sediment, possibly because the disturbance is more intense, often deeper, and also tends to cover a larger area. English Nature (1992) experimentally dredged an area of Langstone harbour (UK) to study the effects of fishing for the clam *Mercenaria mercenaria* on the benthic macrofaunal communities. Although the study period was only eight days, there was no evidence of recovery of species richness or abundance in this timeframe. Hall & Harding (1997) reported recovery of macrobenthic communities within 56 days once dredging had ceased (hydraulic suction dredging and tractor dredging) although species richness was consistently lower in areas impacted by suction dredging. However, it should be noted that the recovering communities in the experimentally dredged areas were being compared to communities which had previously been dredged. Due to a ban, no hydraulic suction dredging had been carried out for 4.5 months and local residents stated that tractor dredging had not been seen for at least 2.5 months prior to the experiment. However, compared to the timescale of recovery documented by some authors (e.g. Beukema et al, 1999; Dernie et al., 2003b; Herbert et. al., 2009) these reference areas can be considered to be recently disturbed. A similar study commissioned by MAFF (1996) in the intertidal River Exe showed that, after the winter harvest, hand raked areas experienced a 50% reduction of invertebrate species and numbers and suction harvested areas experienced a 90% reduction. In both cases, recovery of species richness and abundance took 8 months and trenches caused by the suction took 3-4 months to infill. A similar recovery time was found when tractor towed cockle harvesters were used experimentally in Burry Inlet, UK. Large (20m x 30m) plots were dredged and at the end of the 6 month trial, cockle populations were still significantly reduced. Only at the end of this 6 month period did the other benthic invertebrate species show signs of recovery in terms of comparable species richness and abundance to reference areas (Rostron 1993).

In a long term study (1975 to 1994) in tidal flats of Wadden Sea, mechanical harvesting of lugworms (*Arenicola marina*), caused strong reductions in the abundance and biomass of benthic species in the dredge gullies. Recovery of long lived species such as *Mya arenaria* took around 5 years from the cessation of lugworm harvesting. Lugworm populations recovered after at least three years. In general the loss of biomass of benthic species was due to the loss of the largest individuals (Beukema 1995). Recovery of lugworms from high mortalities, possibly caused by an algal bloom, was observed within 30 days, presumably due to immigration by adults from nearby areas (Olive & Cadnam 1990). Such immigration was also reported by Flach & Beukema (1994).

Disturbance by bait pumping has been shown to cause a population decline in ghost shrimp (*Trypaea australienses*) populations, with no recovery (based on burrow counts) within three months. This reduction in population was believed to be due to the bait pumping technique causing a reduction in sediment porosity and a change in redox conditions which created an unfavourable habitat for the shrimp (Contessa & Bird 2004). In addition to long lived species taking longer to recover (Beukema 1995), infauna which have tube based life cycle (e.g. the polychaete *Manayunkia aestuarina*), are also slower to recover in an area that is commercially harvested (Bonsdorff, 1983). Bonsdorff (1983) suggested that whilst species composition and abundance may recover relatively quickly, it may take several years for stabilisation to occur (Bonsdorff, 1983).

The method and intensity of the harvesting technique may not be the only factor affecting recovery. Allen (1995) summarised the work of several authors who monitored recovery of infauna following mechanical cockle harvesting (hydraulic suction dredging). In general, significant adverse effects were immediately noted but, in naturally disturbed areas (due to tidal currents, wave action and severe weather events), recovery was noted within two weeks of dredging and it was assumed that benthic communities in such areas were robust to physical disturbance. The converse was observed by Rostron (1993), in the Burry Inlet where evidence of impact on the benthic community from cockle dredging was still detectable 3-6 months post dredging, despite storm action.

#### 4.2.1.4. MOORINGS

There is a large amount of literature on the impacts of boat moorings and propeller scars on sedimentary environments, particularly in relation to seagrass beds (e.g. Uhrin & Holmquist, 2003; Eriksson et al, 2004). However, the effects on benthic invertebrates and their recovery has been less well studied. Herbert et al (2009) monitored recovery of macrofaunal communities following removal of mooring buoys and found that, whilst there was community development in the disturbed areas, convergence between the disturbed and control community had not been achieved after 15 months (the duration of the study). The presence of mooring buoys results in the formation of depressions caused by scouring as the chains move according to wind driven and tidal currents. Whilst these depressions filled over time, the composition of the sediment differed to that of the surrounding sediment, being composed of coarser particles and shell fragments. Considering this, Herbert et al. (2009) questioned whether or not complete recovery would be achievable in the long term. Similar findings were also recorded by Kenny & Rees (1996) who found that organism abundance recovered after 2 years following dredging but that biomass and community structure did not. Furthermore, dominant species were quick to recolonise but the rarer species were not and, based on the ratio between biomass and abundance, it would appear that the organisms were considerably smaller than those in the pre-dredging and reference communities. These findings were explained by a reduction in sediment stability, caused by dredging, and subsequent sediment transport during winter storms.

#### 4.2.1.5. Trampling/vehicular access

Trampling by humans and animals results in mortality of infaunal organisms, either directly due to crushing, or indirectly due to sediment compaction, burrow collapse, burial and anoxia (Johnson et al., 2007; Rossi et al, 2007, Tyler-Walters et al., 2008). Rossi et al. (2007) provides one of the few examples of macrobenthic recovery following trampling on intertidal mudflats. Whilst negative effects were found on adults of the bivalves *Cerastoderma edule* and *Macoma balthica,* juveniles of *M. balthica* seemed to benefit from the absence of negative adult-juvenile interactions. Whilst these authors did not make a direct assessment of recovery, their findings imply that colonisation by juveniles could be rapid during the growing season but that the positive effect on recruitment and colonisation by *M. balthica* might lead to a dominance of this species and an overall functional change in the long term. Ensuring that high intensity and frequent trampling in small, localised areas does not occur may be effective mitigation against this.

Whilst Tyler-Walters & Arnold (2008) reported that vehicle tracks were visible on the mudflats for 6 months after disturbance, studies of macrofaunal recovery in relation to this form of disturbance were not found.

**Table 2. Evidence for recovery of macrofaunal communities following anthropogenic disturbance. > refers to recovery times beyond the duration of the experiment or monitoring period.**

| Reference | Organisms | Sediment type | Disturbance type | Disturbance area/patch size | Recovery time | Definition of recovery |
|---|---|---|---|---|---|---|
| Lewis et al. 2002; 2003 | Macrofauna Ireland | Intertidal mud and fine sand. | Pipeline construction | 360 x 50 m<br>3 m deep | 1 year | Return to pre-disturbance abundances for key species. |
| Peterson et al., 2006 | Macrofauna USA | Intertidal sand | Beach nourishment | 10.8 km shoreline receiving 100-176 $m^3$ sediment | >7 months<br>Density of some species recovered but not all. | Equivalent abundance of key species to control. |
| Bolam & Whomersley, 2003 | Macrofauna UK | Intertidal mud | Beach nourishment (beneficial use) | Section of marsh. Deposition of 49-57 cm sediment | **3 months**<br>Species richness, diversity, abundance<br>**>18 months**<br>Community structure | Comparable species richness, abundance, diversity and community structure to reference areas. |
| Bolam et al., 2010 | Macrofauna UK | Intertidal mud | Beach nourishment (beneficial use) | 100s of meters | **12 months**<br>Low elevation areas<br>**24 months**<br>High elevation, sheltered areas | Comparison species richness, abundance and community structure between 4 recharge areas with differing habitat characteristics |
| Herbert et al., 2009 | Macrofauna UK | Intertidal mud | Scouring by anchor chains. | **Large scale**<br>6 m radius around mooring. Approx 3% of the mudflat. | >1.5 years | Comparable species richness, abundance, biomass and community structure to undisturbed areas.<br>Comparable abundance of key prey species. |
| Kenny & Rees, 1996 | Macrofauna UK | Subtidal coarse sand and gravel | Dredging | 500 x 270 m | >2 years | Comparable species richness, abundance, biomass and community structure to reference and pre-dredge sites. |

**Table 2 cont.**

| Reference | Organisms | Sediment type | Disturbance type | Disturbance area/patch size | Recovery time | Definition of recovery |
|---|---|---|---|---|---|---|
| Hall & Harding, 1997 | Macrofauna UK | Intertidal muddy sand / sandy mud | Cockle harvesting<br>Tractor dredging | 15 x 15 m<br>30 x 30 m<br>45 x 45 m | 56 days | Comparable community structure to surrounding (un-dredged) area. |
| | | | Suction dredging | 1.4 m wide bands, 20 mins | | |
| Bolam et al., 2004. | Macrofauna UK | Intertidal estuarine mudflats | Dredged material deposition (experimental) with different sand and organic (Org.) content. | 1 $m^2$ plots | | Comparable species richness, abundance and community structure to reference area. |
| | | | | Low Org./low sand | 12 months | |
| | | | | High Org. | >12 months | |
| | | | | High sand | 12 months | |
| | | | | High Org/high sand | >12 months | |
| McLusky et al 1983 | *Arenicola marina*, UK | Intertidal mudflat | Bait digging by:<br>a. Basin digging/mound creation | **Medium - large**<br>10 x 10 m | Basins = 24 to 45 days<br>Mounds = 4 months | Recovery when counts of *Arenicola* casts resembled that of the surrounding un-dug area |
| | | | | 20 x 20 m | Infill trenches = 22 days | |
| English Nature 1992 | Macrofauna, UK | Soft muddy gravel | Clam/oyster dredging | 0.064$m^2$ cores to 15 cm deep within a dredge channel (no size of dredge or area dredged given) | >8 days | Comparison to undredged control site nearby in terms of species abundance and evenness |

**Table 2 cont.**

| Reference | Organisms | Sediment type | Disturbance type | Disturbance area/patch size | Recovery time | Definition of recovery |
|---|---|---|---|---|---|---|
| MAFF 1996 | Macrofauna, UK | Intertidal R. Exe, Devon, UK | Clam cultivation:<br>-Netting sediment during cultivation<br>-Rake harvesting<br>-Suction harvesting | 10m x 1.5m | Hand raking – species and numbers recovered in ≈ 8 mo<br>Suction harvesting – trenches filled in 3-4 mo, spp recovery in ≈ 8 mo | Comparison of species number and abundances to control plots 50m away |
| Boyd et al. 2004 | Subtidal benthic fauna | Subtidal dredge sites of East Coast UK (gravelly) | Dredging<br>16-35 m deep<br>High and low intensity sites | 1.35 – 3.1 km$^2$ | Dredge tracks still visible after 3-10 years depending on intensity of dredging<br>Low intensity areas recovered fauna after 6-7 years. High intensity did not recover in this timeframe. | Comparison to reference sites nearby (not called control site due to lack of information on pre-dredge status of area) considered to be representative of local area in terms of structure of macrobenthic communities. |

**Table 2 cont.**

| **Reference** | **Organisms** | **Sediment type** | **Disturbance type** | **Disturbance area/patch size** | **Recovery time** | **Definition of recovery** |
|---|---|---|---|---|---|---|
| Baukham 1998 | *Cerastoderma edule,* UK | Intertidal flat, Dyfi cockle fishery, Wales | Cockle harvesting by raking (hand collection) | 5m$^2$ plots with only the central 3m$^2$ of each plot disturbed to provide a buffer zone around the disturbance.<br>Cockles collected and riddled<br>Benthic macroinvertebrates obtained through 3 10.4mm diam cores to 15cm depth per plot. | 16 days for species counts to return to control levels<br>8 days for sediment appearance to resemble that of control plots | Comparison of cockle numbers per plot to untreated control plots |
| Kaiser et al. 2001 | Macrofauna, Dee estuary, UK | Silty sand | Raking for cockles to 10 cm depth | Small 6m$^2$ patches<br>Large 36m$^2$ patches<br>Control sites 6m$^2$ | Small plots recovered after 56 days<br>Large plots had not recovered after 56 days | Comparison of number of individuals, number of taxa to untreated control plots nearby |

**Table 2 cont.**

| Reference | Organisms | Sediment type | Disturbance type | Disturbance area/patch size | Recovery time | Definition of recovery |
|---|---|---|---|---|---|---|
| Rostron 1998 | Macrofauna, Burry Inlet, UK | Muddy/sandy tidal flat | Tractor towed cockle dredge | | 3 weeks for sediment composition to recover<br>6 months for species population numbers to recover<br>Cockle populations reduced and did not recover in 6 months | Differences in species populations between control and treated plots |
| Contessa &Bird 2004 | *Trypaea australiensis*, Melbourne, Australia | Intertidal sand and mud flats | Standard commercial bait pump down to 50 cm depth | 3m x 3m plots | No recovery of shrimp populations within the three months of the experiment | Comparison of burrow counts to control plots. Control plots were bait pumped but shrimp not harvested as in the test plots and instead were allowed to reburrow. |

### 4.2.2. STUDIES REPRESENTING ANTHROPOGENIC DISTURBANCE

Many studies of the impacts of physical disturbance on macrofaunal communities, and their subsequent recovery, have focused on experimentally induced disturbance to represent the impacts of anthropogenic disturbance. In general, the scale of disturbance in these experimental studies is considerably smaller than that associated with actual anthropogenic disturbance (particularly construction work and beach re-charge) and, therefore, these studies have been considered separately.

**Recovery following experimental disturbance is summarised in Table 3.**

#### 4.2.2.1 Bait digging/intertidal fishing

Dernie et al. (2003b) experimentally created disturbance on intertidal sandflats at a scale relevant to various anthropogenic disturbances in intertidal areas (bait digging, intertidal fishing activities). Pits (2x2 m in area) were dug to depths of 10 or 20 cm to represent high and low intensity disturbance, resulting in almost complete defaunation of the sediment. Recovery times (disturbed communities converging with ambient communities) were 64 and 208 days (approximately 2-7 months) for low and high intensity disturbance, respectively and, shortly after disturbance, were strongly influenced by pit depth and the depth of water in the pit. Over time, the importance of this parameter decreased with the importance of other sediment parameters (silt/clay, proportion of fine sands and organic content) increasing and becoming more representative of structuring forces in ambient communities.

#### 4.2.2.2. Beach recharge

Whilst Dernie et al. (2003a,b) did not find a direct relationship between sediment properties and macrofaunal recovery (at least not in the early stages of recovery), other authors have found recovery to be dependent upon the degree of change to the sediment structure (i.e. percentage of silt/clay and organic matter). Bolam et al. (2004) experimentally simulated (in the field) the effect of dredged material emplacement (beach recharge) by manipulating defaunated sediments to produce sediments of varying organic (increase from 0.9 to 2.8%) and fine sand content (increase from 12 to 47%). Macrofaunal sampling was carried out after 1 week and after 1, 3, 6 and 12 months. The addition of organic material resulted in reductions in redox potential (increase in anoxia) to depths of -4 cm, an increase in sediment shear strength and an increase in sediment drying (cracking) which led to a reduction in the rate of macrofaunal recovery (based on community structure) in organically enriched sediments. Macrofaunal recovery had not occurred in sediments treated with organic matter after 12 months whereas recovery (convergence with reference conditions) was recorded at the end of the 12 month sampling period in sediments treated with sand only. Recolonisation patterns, although gradual, were found to be species specific with abundances of the polychaete *Hediste diversicolor* and the gastropod *Hydrobia ulvae* recovering to ambient levels within one week. This was explained by the life history of these species which allows rapid colonisation and enables their survival in stressful environments such as estuaries.

#### 4.2.2.3. Anoxia

Thrush et al. (1996) defaunated sediments of various patch sizes (0.203-3.24 $m^2$) by inducing anoxia. Whilst this type of disturbance does not cause physical alteration of the sediment, this study demonstrates the effect of disturbance scale, sediment stability and environmental heterogeneity on recolonisation processes. Following initial disturbance, there was no evidence of preferential exploitation by opportunistic species (e.g. due to organic enrichment resulting from dead and decaying organisms). After more than 9 months, there were still significant differences in species richness, abundance and community structure between experimental and ambient sediments. Community development followed a similar trajectory in all treatments (different spatial scales of disturbance) but had not converged at the end of the 9 month study period. In contrast, other authors have reported recovery after a few months following small-scale disturbance (e.g. Zajac & Whitlatch, 2003).

In contrast to the small-scale study carried out by Thrush et al. (1996), Beukema et al. (1999) defaunated sediments by experimentally inducing anoxia over large areas (120 $m^2$), which were considered more representative of the scale of anthropogenic disturbance events. As is the case in most studies, recovery of species richness and total abundance was rapid with recovery of these parameters being apparent after 6 months if the recovery period began in spring/summer and 1 year if recovery began in autumn/winter. However, biomass and population structure (individual body size and age class) took significantly longer to recover. For example, only 5 and 40% of the biomass had recovered following winter and summer recovery, respectively and complete recovery of the biomass took 3 or more summers. Similarly, the communities in defaunated areas were still characterised by a lack of large (older) individuals of long lived bivalve species such as *Mya* and *Macoma* after 4 years of study.

Rapid recovery of species abundance and community structure following anoxia induced defaunation was also recorded by Zajac & Whitlatch (2003). However, these authors also pointed out that recovery of population structure had not been achieved by the end of the 5 month study period and that convergence between the recovering and ambient communities had occurred during a period of seasonal species abundance decline towards low winter levels. Hence, studying the recovery of this community at a different time of year may have indicated longer recovery times. Smith and Brumsickle (1989) and Thrush et al. (1996) also reported a lag in population-level recovery with differences between disturbed and undisturbed populations being detectable after 41and 235 days (the duration of the study), respectively. Due to the duration of these studies, it is not possible to assess when population recovery might have been detected but according to Beukema et al. (1999), this may take several years for large, long lived species.

Certain meiobenthic and small macrobenthic species preferentially inhabit the burrows of larger organisms, such as the polychaete *Arenicola marina* (Reise, 1983; Beukema et al., 1999). An indirect implication of not restoring population structure (i.e. a community dominated by juveniles and small individuals) is the lack of potential for recovery of such species.

Zajac & Whitlatch (2003) highlighted the importance of variation between ecologists in their interpretation of successional endpoints, such as recovery, stating that recovery of species abundance and community structure may not mean that the population structure of the component species has recovered. This would then imply that ecological function had not necessarily been restored (Peterson et al., 1998).

#### 4.2.2.4 Terrigenous sediment

Thrush et al. (2003) and Cummings et al. (2009) both highlight the impact of terrigenous (terrestrial) sediment deposition in the intertidal zone. Terrigenous sediments originate from rainfall events, riverine inputs/runoff or directly from landslides and can take a significant length of time to disperse (Cummings et al., 2009). Thrush et al. (2003) refer to rainfall events in New Zealand which have led to the deposition of 5-10 cm of sediment in the intertidal zone over an area of tens to thousands of square meters which can result in mass mortality of benthic infauna (Cummings et al., 2009). Whilst such large scale smothering may be a relatively rare phenomenon, Cummings et al. (2009) found that terrigenous sediment layers >2mm thick were sufficient to cause mass mortality, from which recovery can take years, but that thinner layers could also cause a decrease in macrofaunal abundance. Input of terrigenous sediments to intertidal areas is exacerbated by urbanisation (Thrush et al., 2003) but significant inputs may potentially arise from coastal defence work, habitat creation schemes where earth banks are breached and as a result of vehicular access and trampling during construction work.

Thrush et al. (2003) carried out experimental introduction of terrigenous sediments at different shore heights in 3 locations of an intertidal sandflat in New Zealand, using 3 m diameter plots with 3 cm depth of introduced sediment. Hydrodynamic conditions, such as wave action, current velocity and tidal immersion/emersion time, were found to play a key role in the rate of recovery of both the sediments and, subsequently, the macrofauna. This was also reported in a similar study by Norkko et al (2002). Furthermore, the recovery rate of surface (top 2 cm) macrofauna was considerably faster than that of deeper burrowing species (2-15 cm). This was considered important to the understanding of the recovery process indicating that the surface layers of the sediment were generally inhabited by larvae with high dispersal ability (via hydrodynamic processes) and high potential for colonisation. In contrast, adults inhabit the deeper layers and movement is largely restricted to burrowing and crawling and is energetically expensive. Separating the two layers accounts, to some extent, for the different stages of community development. Recovery in some areas appeared to be facilitated by the presence of large bioturbating species (e.g. crabs) which effectively mix the sediments and enhance sediment transport. However, in areas of high hydrodynamic energy, bioturbation may also have inhibited colonisation in the surface layers by enhancing sediment transport.

Whilst the physico-chemical properties of the sediment recovered after approximately 50 days (with considerable spatial variation), the macrofauna took significantly longer. Based on multivariate analysis (convergence between experimental and reference areas in terms of community structure), 3 of the six sites had not recovered after 212 days (the duration of the experiment) and 4 of the six sites had not recovered in univariate terms (equivalent values of species richness and abundance in experimental and reference areas). It is of note that the sedimentation did not result in complete defaunation; had it done so, recovery times may have been even longer. In a similar experiment, Norkko et al. (2002) found that, regardless of the depth of the deposited sediment, macrofaunal abundance was reduced by 90% after 10 days and that complete recovery of deeper burrowing species did not occur after 412 days (14 months). Following a meta analysis, Thrush et al. (2003) found that slow recovery following the introduction of terrigenous sediments was consistent between studies. They also highlighted that sedimentation events may become increasingly frequent due to land use changes (e.g. Coastal development work) and climatic variability associated with global warming.

**Table 3. Evidence for recovery of macrofaunal communities following experimental disturbance representing anthropogenic disturbance . > refers to recovery times beyond the duration of the experiment or monitoring period.**

| Reference | Organisms | Sediment type | Disturbance type | Disturbance area/patch size | Recovery time | Definition of recovery |
|---|---|---|---|---|---|---|
| Zajac, 2004 | Macrofauna UK | Intertidal mud | Experimental (pit-mound system representing bioturbation) | 1400 cm$^2$ (maximum) | **2 weeks**: sediment characteristics<br>**3-4 weeks:** macrofauna | Return to ambient conditions in terms of species present and mean abundance |
| Dernie 2003b | Macrofauna UK | Intertidal sand (fine sand with shell debris, 1% silt). | Experimental defaunation to a depth of:<br>10 cm (low intensity)<br>20 cm (high intensity | 4 m$^2$ plots | 64 days<br>208 days | Return to ambient conditions in terms of species present, mean abundance and community structure |
| Thrush et al. 1996 | Macrofauna and large meiofauna (250 µm sieve) New Zealand | Intertidal fine/very fine sand | Experimental defaunation (induced anoxia) on troughs and ridges.<br>No change to physical structure of the sediment | 3.24 m$^2$<br>0.81 m$^2$<br>0.203 m$^2$ | >9 months<br>>9 months<br>>9 months | Return to ambient conditions in terms of species present, mean abundance and community structure |
| Beukema et al. 1999 | Macrofauna Wadden Sea | Intertidal. Stable fine sand (2-4% silt) | Experimentally induced anoxia (defaunation) | 120 m$^2$ | 4.5 years | Return to ambient abundance, species richness, biomass and population structure |
| Herbert et al., 2009 | Macrofauna UK | Intertidal mud | Scouring by anchor chains. | 6 m radius around mooring. Approx 3% of the mudflat. | >1.5 years | Comparable species richness, abundance, biomass and community structure to undisturbed areas. Comparable abundance of key prey species. |

**Table 3 cont.**

| Reference | Organisms | Sediment type | Disturbance type | Disturbance area/patch size | Recovery time | Definition of recovery |
|---|---|---|---|---|---|---|
| Thrush et al., 2003 | Macrofauna<br>New Zealand | Intertidal<br>Fine-medium sand | Experimental deposition of terrestrial sediment | 3 m diameter<br>3cm depth of sediment deposition | 50 days<br>>212 days | Sediment properties<br>Comparable community structure to surrounding area. |
| Bolam et al., 2004. | Macrofauna<br>UK | Intertidal estuarine mudflats | Dredged material deposition (experimental) with different sand and organic (Org.) content. | 1 m$^2$ plots<br>Low Org./low sand<br>High Org.<br>High sand<br>High Org/high sand | 12 months<br>>12 months<br>12 months<br>>12 months | Comparable species richness, abundance and community structure to reference area. |
| Zajac & Whitlatch, 2003 | Macrofauna<br>USA | Intertidal medium sand. Occasional clam digging | Experimental defaunation (sediment dug out to 15 cm depth and replaced with defaunated sediment | 1 m$^2$ plot<br>Sediment grain size<br>Species abundance<br>Community structure<br>Population structure | 2 months<br>2.5.months<br>4-5 months<br>>5 months | Physical recovery of the habitat (sediment grain size).<br>Comparable species abundance, community structure and population structure to ambient sediment. |
| Herkül et al 2011 | Macrophyte and benthic macrofaunal communities Baltic Sea | Soft bottom, shallow water | Mimicking ice scour or severe storm in shallow (1 m depth) water | Removal of upper 3 cm of sediment from 1m$^2$ plots | Disturbance in spring had recovered by autumn (≈ 6 months) | Comparable to community of control plots in surrounding area – based on species composition, abundance and biomass |
| Bell & Devlin 1983 | Macrofauna | Subtidal silty-sand<br>Old Tampa Bay, Florida | Stirring sediment with wire brush to depth of 20 cm | 8 cm diam cores, 20 cm deep | 7.5 h | Comparison of number of individuals and number of species to those in control sites |

### 4.2.3. STUDIES REPRESENTING BIOLOGICALLY INDUCED PHYSICAL DISTURBANCE

Zajac (2004) conducted a review of macrofaunal recovery in response to small scale biogenic disturbance (natural and experimentally induced or created) of intertidal and subtidal soft sediments in the form of feeding pits and mounds resulting from infaunal burrowing and feeding activity. Documented recovery times were variable but generally decreased with increasing area of disturbance and increasing density and longevity of pits and mounds (Table 4). For example, in intertidal mud, macrofaunal recovery took between 40 and >76 days in a 0.4 $m^2$ area (with 8-50% coverage of pits and mounds) compared to 21 days in an area of <0.04 $m^2$ with 10% coverage of pits. Numbers of the polychaete *Streblospio shrubsolii* (adults and juveniles) were lower than ambient in the pits but the number of adults was greater than ambient for the mounds. Similarly, low abundances of oligochaetes, *Hediste diversicolor* and *Manayunkia aestuarina* were found in pits although all species recovered to ambient density after 4 weeks.

Following small scale experimental disturbance (pits 3-6 cm deep; 10-12 cm diameter) of sandy sediments, Savidge & Taghon (1988) found that recovery (in terms of species present and abundance) was quicker in depressions than in defaunated sediments with ambient topography and attributed this to passive advection. It was suggested that shear stress would be reduced in the pits, facilitating the deposition of sediments and organisms, and that the pits would provide some protection from resuspension.

### 4.2.4. SUMMARY

Based on the studies in Tables 2-4, the timescale for macrofaunal recovery ranges from a few days to over 4 years although the majority of studies declare recovery after a few months. However, the duration of many studies is 1 year or less and by the end of the study period, recovery has often not been achieved. In these cases, the time to reach recovery has been documented as, for example, > 9months or >12 months. Recovery following very small scale disturbance (e.g. bioturbation features, which in one study actually facilitated recovery) is rapid and occurs within a few days. In general, the time taken to achieve recovery increases with increasing spatial scale, intensity and, importantly, is dependent upon the definition of recovery. That is, the biological parameters measured to assess recovery. The fastest recovery times are documented in studies where simple measures of species richness and abundance have been recorded. Recovery time increases in studies where community structure has been examined and, particularly, in studies where biomass and population structure have been assessed. The time taken to achieve recovery is habitat specific and appears to be prolonged following large-scale anthropogenic disturbance compared to small-scale, experimentally induced disturbance. Taken together, general principles around recovery can be derived from the current evidence base. However, it is important for anyone undertaking an assessment of impacts, and likely recovery, to consider the primary literature in relation to the specifics of each case.

**Table 4. Evidence for recovery macrofaunal communities following biologically induced disturbance.**

| Reference | Organisms | Sediment type | Disturbance type | Disturbance area/patch size | Recovery time | Definition of recovery |
|---|---|---|---|---|---|---|
| Cross & Curran, 2004<br>**Details in section 3.2.2.** | Meiofauna<br>USA | Intertidal sand | Biological<br>Feeding pits created by rays | 2086-3027 $cm^2$<br>8847-16,253 $cm^3$ | 72-168 h | Equivalent abundance of total meiofauna and key meiofaunal groups (Nematodes, Copepods etc) |
| Zajac, 2004 | Macrofauna<br>UK | Intertidal mud | Experimental (pit-mound system representing bioturbation) | 1400 $cm^2$ (maximum) | **2 weeks**: sediment characteristics<br>**3-4 weeks:** macrofauna | Return to ambient conditions in terms of species present and mean abundance |

## 4.3. Seagrasses

In comparison to the macrobenthos, the number of studies relating to recovery of seagrasses following physical disturbance is limited. The majority of studies focus on eutrophication. This section includes those which have examined the effects of increased turbidity and sediment deposition following large-scale flooding events and storms, shellfish harvesting, beach re-charge and experimental removal of *Zostera* shoots.

**Recovery of seagrasses is summarised in Table 5**

Disturbance in seagrass beds can occur at a localised scale (grazing, trampling, localised discharges, moorings) or can occur at much larger scales as a result of flooding and climatic events (Campbell & McKenzie, 2004). Eutrophication is considered to be a major cause of seagrass loss but physical disturbance resulting in excessive sediment deposition or erosion is also known cause widespread loss (Cabaço et al, 2008). Large scale changes to sediment dynamics may result from natural events such as storms but also result from coastal construction work, beach re-charge and dredging activities. Land use changes in coastal areas have exacerbated problems such as increased sediment and nutrient load (Thrush et al., 2003) which in turn, exacerbate the impacts of high turbidity, sediment dynamics and eutrophication on seagrasses (Campbell & McKenzie, 2004). Cabaço et al. (2008) stated that whilst natural disturbance processes, such as storms or dune migration, allow seagrass species to recover and/or adapt to sediment deposition and erosion dynamics, most human activities resulting in sediment deposition cause permanent changes to the sediment and permanent negative effects on seagrass.

Campbell & McKenzie (2004) used a combination of aerial, boat-based and foot surveys to map the distribution of seagrass beds (*Zostera capricorni*) on the intertidal mud and sand flats of the Great Sandy Strait, Queensland, Australia, following a large scale flood event. Seagrasses are known to be sensitive to increased sediment and organic loads associated with floods which reduce light and increase organic matter and bacterial oxygen demand in the sediment (Campbell & McKenzie, 2004) which, in this study, led to the disappearance of approximately 90% of the intertidal seagrass from the area. Recovery was assessed in terms of spatial extent and percentage cover of *Z. capricorni* and was found to be related to nutrient concentration and sediment load in the water column and to the organic content and particle size properties of the sediment. Full recovery to pre-flood abundance was recorded after 2 years in one area with seedling growth beginning 18 months after seagrass loss and full recovery taking a further 6-9 months. It should be noted that this study was carried out in a sub-tropical habitat and in temperate areas, seasonality could strongly influence recovery.

Considerably longer recovery times have also been recorded. For example, Neckles et al. (2005) found that the biomass of *Zostera marina* in areas influenced by mussel harvesting (dredging) areas was only 1-61% of that in undisturbed areas a year after dredging had ceased and that substantial differences were still apparent after 7 years. They predicted that complete recovery in intensively dredged areas would take, on average, over 10 years but could take up to 20 years if the conditions were not suitable for seagrass growth (e.g. high turbidity, high nutrient concentrations, inappropriate substratum).

Valdemarsen et al. (2011) found that seagrass recovery times could be protracted as a result of bioturbation and found that densities of the polychaete *Arenicola marina* of 5-10 ind. $m^{-2}$ could negatively impact upon seagrass colonisation and growth. During a laboratory mesocosm study, Valdemarsen et al. (2011) found that 95% of seeds and 75% of seedlings were buried below the critical depth for survival and growth after 1-2 months of bioturbation by *A. marina.* Therefore, even though water quality may be suitable for seagrass growth, in terms of suspended solids and nutrient concentration, the potential for seagrass recolonisation in disturbed areas may be reduced in the presence of bioturbating organisms.

Negative effects on subtidal seagrass beds have been documented following beach recharge as a result of sediment movement and subsequent burial of seagrasses. Furthermore, variability in the definition and interpretation of recovery leads to variation in the documented recovery time. For example, eighteen years after damage to *Posidonia oceanica* meadows caused by beach recharge, González-Correa et al (2008) reported a reduction in leaf production (horizontal rhizomes, 21%), total net rhizome production (45%) and starch concentration in impacted compared to unimpacted sites (25%). The reduction in non-structural starch in the rhizomes indicates mobilisation of carbon to meet the plants demands in the absence of light (Cabaço &Santos. (2007). Whilst total shoot cover (a parameter commonly measured in other studies) did not differ between impacted and non-impacted areas, the percentage of live shoots was considerably lower in impacted areas. Overall, these authors concluded that natural recovery rate of *P. oceanica* had been reduced by 45% and was attributed to high levels of silt/clay, organic matter and sulphide in the sediment. Hence, full recovery of the *P. oceanica* meadow had been prevented due to unfavourable sediment conditions caused by the beach recharge scheme.

Recovery of seagrasses (specifically *Zostera marina*) appears to be dependent on the size of the disturbed area and the degree to which shoot removal has taken place. Boese et al., (2009) experimentally removed *Z. marina* shoots from 4 $m^2$ areas and reported recovery (equivalent vegetative, reproductive and seedling shoot density to undisturbed areas) after 24 months. However, in the centre of the plots, recovery took 30 months with completely unvegetated areas still remaining after 24 months. It was concluded that recolonisation was exclusively due to rhizome growth from adjacent vegetated areas. This was supported by the poor survival of transplanted seedlings. Therefore, in areas of extensive disturbance where complete removal of shoots has taken place, recovery times are likely to be significantly longer than the 30 months documented by Boese et al., (2009). Furthermore, in areas where the widespread total loss of *Z. marina* has occurred, full recovery may be unlikely (Giesen *et al,* 1990).

### 4.3.1. MICROALGAE

Sediment inhabiting diatoms are an important component of intertidal estuarine mudflat communities (Admiraal, 1984) and are considered to be the most important primary producer in intertidal mudflats (Blanchard *et al.*, 2000). The microphytobenthos not only supplies the benthic food web with organic matter but also plays a key role in the biostabilisation of sediments (Blanchard *et al.*, 2000). Diatom growth rates are generally high and recovery potential following loss is thought to be high (Holt et al., 1995). It is of note that high microalgal biomass is associated with fine grained, organic rich sediments and therefore, recovery potential may be reduced by forms of disturbance which result in coarsening of the sediment.

### 4.3.2. SUMMARY

The potential for recovery of seagrasses appears to be poor compared to that of the meio- and macrobenthos. Seagrasses are highly susceptible to changes in nutrient status, turbidity and physical removal and recovery rates range from 2 years to over 7 years with model predictions (where recovery did not occur within the timescale of the study) of 6-18 years. Functional attributes, such as leaf and rhizome production and starch concentration can take more than 18 years and if the physical structure of the sediment is permanently changed, recovery may never be achieved.

**Table 5. Evidence for recovery of seagrasses following physical disturbance. > refers to recovery times beyond the duration of the experiment or monitoring period.**

| Reference | Organisms | Sediment type | Disturbance type | Disturbance area/patch size | Recovery time | Definition of recovery |
|---|---|---|---|---|---|---|
| Hammerstrom et al, 2007 | Seagrass USA | Subtidal seagrass bed (1.5 m deep at high water) | Sediment excavation to represent propeller scars | **Scale**<br>50x150 cm<br>10 cm deep<br>20 cm deep<br>30 cm deep, filled with pea rock | 2 yr (seagrass)<br>5 yr (macroalgae). | Comparable shoot density and macroalgal cover to undisturbed plots. |
| Campbell & McKenzie, 2004. | Seagrass meadows. Australia | Seagrass on intertidal mud and sand | Flooding | Meadow scale ~100-3000 ha | 24-31 months | Comparable abundance of seagrass (spatial extent and density) to pre-flood conditions |
| Neckles et al., 2005 | Seagrass meadows. USA | *Z. marina* on intertidal/shallow subtidal mud and clay | Mussel harvesting (dredging/dragging)) | 3-32 ha | >7 years<br>Model predictions of 6-20 years | Comparable shoot density and total biomass to pre-dredged conditions. |
| González-Correa et al., 2008 | Seagrass meadows. Spain | *P. oceanica* in shallow subtidal sand and mud. | Beach re-charge | Meadow scale<br>Sampling using 40x40 cm quadrats | >18 years | Comparable leaf production, rhizome recruitment and starch concentration to unimpacted beds. |
| Boese et al., 2009 | Seagrass USA | Intertidal *Z. marina* | Experimental shoot removal | 4 $m^2$<br>Plot edges<br>Plot centre | 24 months<br>30 months | Comparable number of vegetative, reproductive and seedling shoots to undisturbed plots. |

# 5. FACTORS INFLUENCING RECOVERY RATE

## 5.1. Physical recovery of the habitat and habitat type

Physical disturbance not only impacts upon sediment dwelling biota but also the physical structure of the sediment in terms of changes to particle size distribution, organic content, porosity, redox profiles, compaction and erosion potential. The effects of sediment characteristics on the distribution of benthic invertebrates have been extensively studied and were reviewed in detail by Snelgrove & Butman (1994). Whilst generalisations of distinct associations between animals and specific sediment types have been made (often based on particle size), the occurrence of a species in a particular habitat is due to the complex interaction between particle size, organic and microbial content, hydrodynamic and chemical conditions and biological interactions (e.g. predation, competition, bioturbation). Therefore, in order for biological recovery to occur, the physical condition of the habitat must first recover. This may simply mean recovery of grain size distribution and porosity or it may mean the infilling of water logged pits, the erosion and redistribution of sediment mounds and the subsequent recovery of grain size distribution, sediment consolidation, organic and microbial content and the development of chemical gradients (e.g. redox profiles).

Zajac & Whitlatch (2003) found similarity of sediment grain size distributions between disturbed and non-disturbed areas after 2.5 months following defaunation of sediments by anoxia. However, it should be noted that whilst these authors removed ambient sediments and replaced them with defaunated sediment, anoxia induced defaunation does not represent the same degree of physical modification as sediment churning, smothering or digging of pits and may therefore be an underestimation of recovery time in some circumstances. Peterson et al. (2006) reported that a significant change in grain size distribution and beach profile (steepening) were still evident 7 months after beach nourishment and that these factors were responsible for the lack of invertebrate colonisation and the subsequent reduction in the number of birds feeding in the area. However, in the early stages of recovery, sediment grain size may not be a good predictor of biological recovery rates. For example, Dernie et al (2003a) found that sediment grain size was only related to macrofaunal community structure in the later stages of recovery once disturbance features such as pits had disappeared.

Dernie et al. (2003a) hypothesized that the recovery of infaunal communities following physical disturbance would be dependent upon habitat type (specifically the gradient between clean sand and mud) and that the relationship between recovery of the physical sediment parameters and biological recovery would be predictive. They proposed that recovery of physical habitat parameters could be used as predictors for macrofaunal community recovery although standard parameters used to describe sediment structure, such as particle size distribution and percentage silt, clay and organic content were not related to temporal variability in community structure. However, the infilling rate of experimentally created pits (10 and 20 cm deep) was found to be a reasonable predictor ($r^2 = 0.65$) of macrofaunal recovery in terms of organism abundance. Clear differences were found between the infilling rates of different sediment types. In clean sandy sediments, all pits had infilled after 105 days (3.5 months) whereas, in muddy sands and muds, depressions and pools of water were still evident after 213 days (approximately 7 months). Similarly, the rate of macrofaunal recovery was greater in clean sandy sediments. However, the rate of infilling did not necessarily show a linear relationship with particle size or percentage of silt/clay. That is, infilling was slower in muddy sands than in muds and was explained by differences in the cohesive properties, sediment water content and physic-chemical factors related to increased slumping compared to sandier sediments.

Dernie et al. (2003a) suggested that infilling rate may represent a combination of factors indicative of the nature of the environment. That is, physical biological relationships are more complex than those that can be described by single parameters. Furthermore, the sediment sampling technique needs to be considered, particularly in relation to depth of sampling. Newly accreted sediments are likely to be surficial and changes in particle size distribution are unlikely to be detectable if sampling is carried out to depths of a few centimetres. Similarly, larger burrowing organisms will be influenced by deeper sediments.

At a smaller scale (up to 1400 $cm^2$ and 5 cm deep), naturally and artificially created pit and mound systems in muddy sediments (representing biogenic features resulting from feeding and bioturbation) were found to reduce rapidly with pits decreasing (due to infilling) from a few cm to a few mm in depth within 2 weeks. Significant slumping (and an increase in area occupied) and erosion of mounds occurred within 1 week (Zajac, 2004). Savidge & Taghon (1988) reported extremely rapid recovery as a result of stormy weather which caused slumping and infilling of pits (depth of 3-6 cm; diameter of 10-12 cm) by 50% within 24 hours. The difference in the rate of physical recovery between these studies can be attributed to differences in disturbance area and depth and to differences in hydrodynamic conditions.

In contrast, Tyler-Walters & Arnold reported that vehicle tracks were still clearly visible on mudflats for 6 months after initial disturbance and Thrush et al. (1996) attributed unusually long infaunal recovery times (>9 months) to sediment instability caused by the removal of polychaete tube mats. They proposed that large-scale disturbance which removed species with a role in maintaining habitat structure and sediment stability may result in prolonged recovery times. Hence, the timescale for recovery of habitat structure is variable according to sediment type, the prevailing environmental conditions such as wave exposure and storm frequency and the degree to which species with a functional role in sediment stabilisation or disturbance are removed.

It is generally accepted that recovery time is longer in muddy sediments than in sands, partly because organisms inhabiting sands are adapted to a higher degree of natural disturbance than those inhabiting more sheltered muds. Zajac & Whitlatch (2003) also stated that the potential for organism transport was much greater in sandy habitats and Lundquist et al. (2006) found the potential for organism transport and deposition to be greatest during periods of wind induced waves. Schratzberger & Warwick (1999) found that nematodes in sandy habitats were less impacted by physical disturbance than those in muddy habitats and proposed that meiofaunal assemblages in the two environments were controlled by different mechanisms. Hydrodynamic conditions were thought to be important in unstable sands whereas biological interaction was thought to be more important in sheltered muddy environments.

Whilst this generalisation may apply to broad habitat classes and sediment types, Dernie et al. (2003a) found that muddy habitats recovered (physically and biologically) more quickly than muddy sands and concluded that, along a narrow spectrum of sediment types, habitat and community recovery was related to physical properties of the sediment such as cohesion, porosity and stability. Furthermore, Dernie et al (2003b) suggested that recovery rate could be influenced by the degree of shelter and reported surprisingly slow recovery rates in sheltered sandy sediments despite the lack of mud. Further evidence against this generalisation is provided by Peterson et al. (2006) who found that beach nourishment (in a high energy environment) significantly, negatively, impacted upon the benthic community which, in turn, significantly reduced the value of the habitat for foraging birds. The lack of recovery was explained by a steepening of the shore profile and the change in sediment composition, principally the median particle size and the shell content. This study demonstrates that recovery potential is strongly influenced by the type and scale of disturbance, the impact it has on the physical habitat and the duration of the physical modifications.

### 5.1.1. SUMMARY

In summary, the distribution of benthic species (including invertebrates, seagrass species and microalgae) is strongly related to a combination of physical factors including sediment particle size, consolidation, stability and cohesion, organic and microbial content, hydrodynamic and chemical conditions and biological interactions (e.g. predation, competition, bioturbation). In order to achieve recovery (defined by most studies as the return to pre-disturbance species richness, abundance, diversity and, in some cases, population and community structure), the combination of physical properties present before the disturbance must be restored. This may not be possible where species with a key role in maintaining the environment (e.g. tubiculous polychaetes which, at high densities, may enhance sediment stability) do not recover. It is also important to recognise that the scale and type of disturbance will influence the potential for recovery with widespread sediment churning, smothering or extensive digging leading to more severe and persistent modifications than, for example, anoxia. Indeed, large scale disturbances often result from entirely different processes or activities than smaller-scale disturbances. Finally, the impact of a particular physical disturbance and the potential for and rate of recovery are highly habitat specific. Generally held beliefs, such as muddy habitats are slowest to recover, may not always be true. As stated before it is important to consider the evidence relevant to each case.

## 5.2. Life history and recruitment

Recruitment is fundamental to community structure (and recovery following disturbance) since it is the foundation for all subsequent community interactions and therefore strongly influences adult population dynamics (Woodin et al., 1995). Successful recruitment depends on larval transport and dispersal ability, larval settlement and larval retention. Following settlement, mortality (due to physical and biological processes see section 4.1 and 4.3) further influences recruitment (Woodin et al., 1995). Most faunal bed species characteristically produce large numbers of larvae and so have relatively good recruitment rates. They therefore have high potential to recolonise defaunated habitats (Holt et al., 1995). In contrast, seagrass species have poor dispersal and recovery ability (Gonzålez-Correa et al., 2008). The mobile nature of many infaunal species also allows for rapid recolonisation (Bell and Devlin 1983). Colonisation of defaunated or disturbed sediments results from planktonic dispersal (where meiofauna and macrofaunal-larvae are deposited via flow) and settlement of larvae and/or post settlement movement by juveniles or adults (Santos & Simon, 1980; Smith & Brumsickle, 1989; Lundquist et al., 2006). Post settlement movement of juveniles and some adults can occur passively as a result of sediment resuspension and deposition resulting from tidal current and wave action and as a result of active migration through burrowing, crawling or swimming (Santos & Simon, 1980; Lundquist et al., 2006). Additionally, vertical migration may occur following smothering, depending on the depth of sediment (Bolam & Whomersley, 2003).

Settlement and successful recruitment of planktonic larvae will depend upon the suitability of the habitat together with the timing of reproductive events in relation to the disturbance, the availability of larvae and hydrodynamic conditions (Filho et al., 2006). For example, Pickett (1973) suggested that prolonged dredging may impair settlement of cockle spat, thus negatively impacting on the cockle population and benthic community as a whole. Santos & Simon (1980) also stated that the ability of larvae to select appropriate substrata and post settlement predation were important factors in determining successful recruitment. Beukema et al. (1999) reported higher macrofaunal densities in sediments recovering from anoxia compared to the surrounding, undisturbed, sediments and attributed this to the lack of adults and large organisms which may inhibit recruitment through bioturbation and predation. Rossi et al. (2007) reported a similar ‘overshoot’ due to the low level of bioturbation.

In cases where defaunation has occurred over a large area, Filho et al. (2006) proposed that colonisation via the water column would play an important role in the recovery process due to the potential for rapid transport of organisms over large distances. Filho et al. (2006) demonstrated that, in small defaunated areas (15 cm diameter), the structure of an assemblage could be restored within a month as a result of this mode of colonisation. However, they also found that colonisation mode was taxon specific and also age specific with, for example, colonisation via the water column being important for juvenile bivalves whilst lateral colonisation (crawling) was more important for larger polychaetes. This taxon specificity was also noted by Santos & Simon (1980) who found that polychaetes and molluscs colonised sediments predominantly as newly metamporphosed larvae whereas gastropods, cumaceans and flatworms were initially present as adults. Hence, disturbance area is likely to strongly influence the colonisation potential of species with low dispersal abilities which do not colonise via the water column.

The time at which post-disturbance recovery is allowed to begin appears to be fundamental to the rate of recovery. With the exception of those involving large scale anthropogenic disturbance, most studies document recovery of macrobenthic species composition and abundance (sometimes community structure) within 6-12 months (e.g., Hall & Harding, 1997; Bolam & Whomersley, 2003; Dernie, 2003b; Thrush et al., 2003; Zajac & Whitlatch, 2003; Zajac, 2004; Herkül et al., 2011). Recovery times following anthropogenic disturbance are generally longer, partly because of the higher degree of disturbance, in terms of spatial area and depth, but also because of the timing and duration of the monitoring programme. However, Beukema et al. (1999) reported rapid recovery (6 months) following a summer disturbance but that recovery time doubled if recovery began in the winter. That is, faster recovery occurred when recovery began during the summer recruitment period. Similarly, in shallow areas of the Baltic Sea, physical disturbance in the spring had an immediate effect on the community, but in summer there was no immediate effect. This was due to drifting algal mats in the summer helping to quickly reintroduce the species that had been lost as a result of the disturbance (Herkül et al 2011).

### 5.2.1. RECRUITMENT CUES

Given the variability of sediments (caused by erosion/deposition, oxygenation, bioturbation, inter- and intra-specific interactions) and the susceptibility of juveniles of infauna to such variability, Woodin et al. (1995) hypothesised that selection should favour recruits with the ability to distinguish between disturbed and undisturbed sediments and those able to actively avoid unfavourable sediments. Santos & Simon (1980) also stated that successful recruitment was dependent on the ability of larvae to select appropriate substrata.

Woodin et al. (1995) studied the ability of post larval juveniles of three species of infauna (*Nereis vexillosa, Arenicola marina* (Polycheata) and *Mercenaria mercenaria* (Bivalvia)) to discriminate between recently disturbed and undisturbed sediment. Disturbance was in the form of simulated erosion (sediment removal or deposition), fresh faeces, burrow tailings and feeding tracks. All three species showed a reluctance to burrow in sediments disturbed by erosion (interpreted as rejection and an ability to distinguish between disturbed and undisturbed sediment) and, to a lesser extent, in sediments influenced by burrow tailings, feeding tracks and fresh faecal matter. Organisms which did not readily burrow were entrained by currents. This ability to distinguish between favourable and unfavourable sediments could have significant implications for recovery rates in areas influenced by anthropogenic disturbance.

### 5.2.2. SUMMARY

In summary, recruitment, recolonisation and recovery are dependent upon the dispersal mechanism of the species in question. Species which disperse passively, via the water column, are likely to recolonise quickly, particularly if recovery begins during the summer recruitment period. Adults and species which disperse by crawling and burrowing may be slow to recolonise and recovery rates are likely to decrease with increasing spatial scale. This has been documented by Beukema et al. (1999) for macrofaunal organisms, with colonisation by adults and large individuals of long-lived species generally taking 4-5 years. Seagrasses have particularly poor dispersal abilities and therefore comparatively long recovery periods. Favourable sediment conditions appear to be an important influence over larval recruitment but are also likely to strongly influence colonisation by adults.

## 5.3. Biological interaction

Many of the studies reviewed here have involved the study of recovery following complete or almost complete defaunation or, the presence of a surviving community has not been assessed yet the impact of sediment-dwelling organisms on their environment and on each other is well known (Snelgrove & Butman, 1994). An organism may become dislodged by the burrowing, ploughing and feeding activity of another leading to increased exposure to waves, tidal currents and predators or burial (Zajac, 2004). Organisms in suspension may be vulnerable to ingestion by suspension feeding organisms and new recruits may simply not be able to attach themselves to the substratum. Furthermore, bioturbating organisms alter the sediment structure (e.g. settlement of fine particles and organic matter) which may make the sediment more or less favourable to colonising species (Volkenborn et al., 2009). In contrast, some species (e.g. tubiculous polychaetes) may stabilise sediments making them more suitable for colonisation by certain species (Eckman et al., 1981). Additionally, commensalistic interactions may occur between certain organisms (Reise, 1983; Beukema et al., 1999). For example, certain meiobenthic and small macrobenthic species preferentially inhabit the burrows of larger organisms, such as the polychaete *Arenicola marina* (Beukema et al., 1999). An indirect implication of not restoring population structure (i.e. a community dominated by juveniles and small individuals) is the lack of potential for recovery of such species.

Cockles (for example) are classed as bioturbators and cause a significant amount of sediment destabilisation and resuspension (Ciutat et al.,2007). In addition, their regular shaking and rapid valve adductions disturb the sediment to a distance of approximately 0.5 cm around their shell (Flach, 1996; Ciutat et al.,2007). Flach (1996) reported negative effects at low cockle densities of 125-250 / $m^2$ and noted a reduction in the abundance of other species of up to 50% at densities of 500 / $m^2$. Volkenborn et al. (2009) found that the removal of the polychaete *Arenicola marina* from sediment plots enabled the colonisation of sessile, tubiculous polychaetes such as *Lanice conchilega* which were previously absent or present only at low abundances. Furthermore, this then encouraged the settlement of juvenile bivalve species. Similar positive effects on larval settlement were noted by Beukema et al. (1999) and Rossi et al. (2007) as a result of the weakening of adult-juvenile interactions following disturbance by anoxia and human trampling, respectively. Valdemarsen et al. (2011) reported bioturbation by *Arenicola* to significantly increase recovery times for seagrass beds due to burial of seed and seedlings.

Given their significance, biological interactions must be considered if a strategy of low intensity disturbance over a wider area is to be proposed as mitigation against the long term impacts of anthropogenic physical disturbance. The importance of bioturbation in influencing recovery dynamics is likely to be greater in sheltered habitats where the effects are less likely to be over-ridden by frequent hydrodynamic disturbance (Probert, 1984).

### 5.4. Spatial scale and habitat fragmentation

Whilst there is a high degree of variability between studies in terms of timing of disturbance and recovery, definition of recovery, type and depth of disturbance, modification to sediment structure and recovery starting point (state of the biological community), there is a general trend of increasing recovery time with increasing spatial scale and intensity of disturbance (Figure 1). For example, studies focussing on small scale biological disturbance at the scale of $10^3$ to $10^4$ $cm^2$ report recovery times of weeks (in terms of ambient species richness and abundance) (e.g. Zajac, 2004, Table 4).

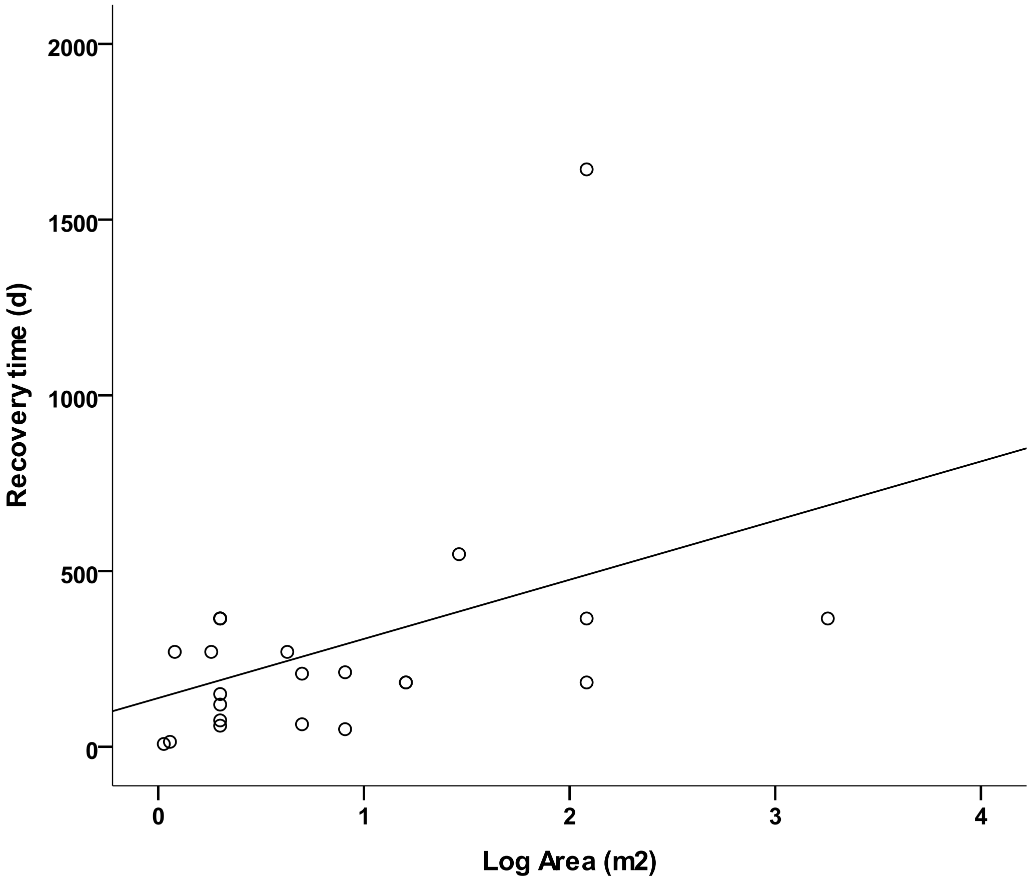

**Figure 1. Generalised recovery rate of macrofauna in relation to spatial scale of disturbance (Log(x+1) transformed).**

Where experimental disturbance has taken place, spatial scale ranges from 1 to 120 $m^2$ (Table 3) with recovery times (in terms of species richness and abundance) ranging from 4-5 months in a 1 $m^2$ area (Zajac & Whitlatch, 2003) to 6-12 months in a disturbance area of 120 $m^2$ (e.g., Beukema et al., 1999). However, these general patterns should be treated with caution (as should Figure 1) because of the variation in disturbance type and the non-uniform intensity of disturbance. For example, Bolam et al. (2004) reported recovery rates of 12 months in an area of 1 $m^2$ which is considerably smaller than the area from which

Beukema et al. (1999) reported a much quicker recovery rate. This can be explained by the fact that Bolam et al. (2004) physically modified the structure of the sediment whereas Beukema et al. (1999) monitored recovery following induced anoxia which caused a lower degree of physical disturbance. Dernie et al. (2003b) reported recovery rates of 2-7 months from a 4 $m^2$ area, the difference in recovery time, again, being attributed to the degree of sediment modification (depth of disturbance in this case). Herbert et al. (2009) suggested that where permanent or long lasting modifications to the sediment occur, recovery may not be possible and reported recovery times of >1.5 years (i.e. beyond the duration of the study) following scouring of a 28 $m^2$ area. Comparatively long recovery times (>9 months) were reported by Thrush et al. (1996), despite the small scale of disturbance (0.2-3.2 $m^2$) because of the local hydrodynamic conditions. With regard to macrofauna, the generally low dispersal ability of adults implies that colonisation rate will decrease with increasing spatial scale of disturbance. This particularly applies to seagrass species. Indeed, Beukema et al. (1999) pointed out that whilst larval recruitment could be rapid (months) over wide areas, recovery of population structure could take years.

Monitoring studies following anthropogenic disturbance suggest that spatial scale and intensity of disturbance significantly influence macrofaunal recovery with recovery periods of 6-7 years being reported following dredging (Boyd et al., 2004). In contrast, recovery rates of months were reported following intertidal cockle harvesting (Table 2). However, in many cases, recovery had not actually been achieved by the end of the study period. Furthermore, information regarding the state of the benthic community (e.g. complete or partial defaunation) or seagrass community (Table 5) was generally not given and, in some cases, the validity of the reference sites is questionable due to the fact that these areas have also been impacted in the past (e.g. Hall & Harding, 1997). These factors make it difficult to assess recovery rates according to spatial scale and intensity.

Therefore, whilst recovery rate is likely to be scale dependent, the relationship is not straight forward and other factors such as the degree and type of physical modification and the local hydrodynamic conditions need to be considered. Furthermore, the functional role of the component species needs to be considered since failure to recover by one species may prolongue the recovery process of others. This may, for example, relate to species with a role in sediment stabilisation (Thrush et al., 1996) or species with commensalistic interactions (e.g. Beukema et al., 1999). Despite this, there is a strong suggestion that recovery rates decrease with increasing area and that recovery from anthropogenic activities (which tend to cause a higher degree of modification to the sediment structure) is slow compared to that following natural or experimentally induced disturbance. Indeed, Thrush et al. (1996) showed that recolonisation rate was dependent upon the size of the disturbed area with a 64-84% reduction in colonisation by common species with a two fold increase in distance to the centre of the plot. Variability in species abundance did not increase with plot size. However, Zajac et al. (1998) suggested that the influence of spatial scale on the factors controlling colonisation and succession is likely to make extrapolation between small scale experiments and larger scale responses inaccurate. They indicated that large scale experiments may not be practical (or ethical) and that experimental design should account for multi-scale processes to improve the understanding of colonisation and successional processes.

Pressure on coastal environments is high and varied ranging from recreational activities (trampling, boating, vehicular access) to construction work (coastal defence, installation of pipelines and cables from offshore windfarm sites) and landclaim. Whilst sedimentary habitats and their communities have the potential to recover, the widespread distribution of activities causing physical disturbance must be considered, together with the fact that many habitats are already degraded. If these activities are not carefully managed and regulated and the impact and recovery process is not well understood, there is potential for habitat fragmentation to occur. This has been described as a major factor contributing to the decrease in global biodiversity although fragmentation in marine habitats has not been widely studied (Uhrin & Holmquist, 2003). The effect of patch size on macrofaunal recovery has already been discussed and it is clear that whilst larvae and juveniles can colonise disturbed areas relatively quickly (within months), adults have lower dispersal abilities and recovery of biomass and population structure can take years (Beukema et al., 1999). Similarly, recovery time for seagrass species will increase with area since recolonisation occurs exclusively as a result of rhizome growth from adjacent areas Boese et al., (2009). It is therefore reasonable to assume that the greater the area of disturbance, the greater the recovery time but also that as habitat fragmentation increases, distance to a source of colonists may also increase. Uhrin & Holmquist (2003) stated that the impact of fragmentation would be dependent upon the dispersal abilities of the species in question. Many marine invertebrate species have pelagic larval phases and therefore have high dispersal abilities and therefore have the potential to recover. In the case of seagrasses, suitable habitat between the nearest bed and the disturbed area would be necessary for rhizome growth; if such habitat did not exist, recovery may not be possible.

### 5.4.1. SUMMARY

Overall, there appears to be a general trend of increasing recovery time with increasing spatial extent and intensity of disturbance. However, direct comparison between the individual studies highlighted in Tables 1-5 is impossible due to variation in habitat type and disturbance type (e.g. natural, anthropogenic, experimental). Furthermore, the parameters measured, the timescale of the study and the authors’ definition of recovery all vary between studies. In many cases, recovery was not achieved within the timescale of the study. Therefore, there is no replication of study types, aims or circumstances (habitat type, type of disturbance etc.) and it is not possible to disentangle the actual effects of spatial scale and intensity of physical disturbance from those related to habitat type, study design and sampling technique.

## 5.5. Monitoring strategy/time

Documented recovery rates may vary according to interpretation of the data. Dernie et al. (2003b) found that fluctuations in abundance (resulting from natural disturbance) in both ambient and disturbed sediments gave rise to an apparent convergence of the treatments long before recovery was actually achieved. It was therefore recommended that longer-term monitoring may be required to accurately describe the recovery trajectory. Thrush et al. (1996) found that temporal variability in ambient community structure reduced the difference between ambient and disturbed plots and suggested that the recovering communities were responding in a similar way to seasonal variation, hydrodynamics and other environmental factors as the ambient communities. They concluded that the timescale of recovery was dependent on seasonal or longer term variations in macrofaunal density and this finding has strong implications for the timing of anthropogenic activities causing disturbance and for the monitoring of the recovery process.

Several studies were found where recovery of disturbed communities was assessed against adjacent, recently disturbed communities (e.g., Allen, 1995; Hall & Harding, 1997). Hall & Harding (1997) examined macrofaunal recovery following cockle harvesting (hydraulic and tractor towed dredges) by comparing community structure to the adjacent area which had not been dredged for 4.5 months. Given the recovery times documented in some of the studies reviewed here, it may have been reasonable to assume that the reference community had recovered from any previous disturbance. However, evidence in Beukema et al. (1999) suggests that recovery in this time period would have been unlikely.

Most of the studies reviewed here are of insufficient length to effectively document recovery, many of them stating that recovery had not been achieved by the end of the study period. Furthermore, the duration of most of the studies is insufficient to assess recovery time in any context other than the re-establishment of species abundance and community structure (in terms of the relative abundance of the component species). There are indications that full recovery of community and population structure, and therefore ecological functioning, could take years. Monitoring of colonisation in a habitat creation scheme indicated convergence of community structure after 3 years but subsequent divergence in later years as a result of continued habitat development (Mazik et al., 2010). Therefore, recovery would only be short term and monitoring programmes need to be sufficiently long to pick up this temporal variability associated with habitat evolution

Furthermore, Thrush & Dayton (2002) state that the extent and intensity of human disturbance to marine systems is a significant threat to structural and functional diversity and in many cases, this has eliminated natural systems that might serve as baselines to evaluate these impacts. That is, true reference conditions generally do not exist so recovery is likely to be evaluated against conditions which are already degraded to some extent. To ensure that maximum recovery can be achieved, it is important that all aspects of recovery are understood and this may mean longer term monitoring than has been the case in most studies.

# 6. DISCUSSION

The timescale for recovery in the studies reviewed here ranges from a few hours or days for meiofaunal communities (Cross & Curran, 2004; Johnson et al., 2007) to a few weeks or months for macrofaunal communities. Most of the studies reviewed involve either small scale experimental defaunation or monitoring after a disturbance event (e.g. dredging, cockle harvesting) and the monitoring is generally short term with recovery being declared once species richness, abundance and, in some cases, community structure have been restored (generally up to 1 year). However, simple measurements of species richness, abundance and community structure do not provide any information about the recovery of population structure (e.g. the relative proportions of adults, juveniles and older individuals of longer-lived species) or biomass and few authors have addressed this. This is an important consideration when measuring recovery since the scale of the ecological function provided by an organism is strongly related to its size (Peterson et al., 1998). Therefore, restoration of ecological function requires restoration of both population (for an individual species) and community structure, at least in terms of the functional groups present.

Those who have monitored population structure and biomass report recovery times of >1.5 – 4.5 years (e.g. Kenny & Rees, 1996, Beukema et al., 1999; Herbert et al., 2009) although in many cases, the study period is shorter than the timescale required to achieve recovery. Based on this, Beukema et al. (1999) suggested that the duration of many studies (generally up to 1 year) may be insufficient to monitor full recovery and highlighted studies where this had been the case (e.g. Bonsdorff, 1983). Furthermore, Beukema et al. (1999) stated that many studies did not assess the completeness of defaunation (baseline from which recovery starts) and monitoring was too infrequent to adequately understand the recovery process (Kenny & Rees, 1996; Lewis et al., 2002).

For seagrasses, recovery times of between 2 years (Campbell & McKenzie, 2004; Hammerstrom et al., 2007) and >18 years (Gonzålez-Correa et al., 2008) have been recorded with model predictions of between 6 and 20 years (Neckles et al., 2005) to reach comparable shoot density and biomass. In the Dutch Wadden Sea, recovery from seagrass losses in the 1930s and 1960s has not occurred, probably due to turbidity of the water column (Giesen et al., 1990) and recovery is generally slow due to the low dispersal ability of seagrass species (Holt et al., 1995).

The evidence for recovery following physical disturbance is variable due to:

- The state of the community when the processes of recruitment and colonisation leading to recovery began (total or partial defaunation). This relates to the scale and intensity of disturbance but also to the longevity, fragility and tolerance of the species present.

- Degree of physical modification to the sediment (changes in elevation, particle size, redox gradients, porosity).

- The level and nature of disturbance experienced by the reference community. That is, how representative is the reference community of undisturbed conditions. For example, Hall & Harding (1997) compared benthic community structure in an area impacted by cockle dredging with that in an area which had been dredged within the previous 6 months.

- The biological feature and the parameters being monitored. In the context of this study, biological features include meiofauna, macrofauna and seagrass.

- Differences in the scale and source of the disturbance. Recovery times appear to increase with the spatial scale of disturbance although the relationship between scale and recovery is unclear due to variation in the parameters being measured and the point at which recovery is considered to be complete.

- Temporal variability in community. Some studies showed convergence of community structure and then divergence several months later and variability in the reference community has also been found to imply recovery. However, few studies have compared temporal patterns in recovering and reference communities, or convergence/divergence between the two after the point at which recovery is considered to be complete.

- The monitoring timescale and frequency which influence the recorded recovery time in that frequent sampling will enable detection of recovery as it happens. In contrast, annual monitoring programmes may only enable detection of recovery several months after it has occurred, potentially giving a false indication of recovery time.

- The time taken to achieve recovery is habitat and disturbance type (and a combination of both) specific and differences in these parameters between studies mean that the effects of spatial scale and intensity cannot be separated from those associated with habitat, disturbance and experimental design parameters.

Importantly, the definition and interpretation of recovery is also variable between studies and largely relates to the biological parameters being recorded (Tables 1-5). Most of the studies reviewed here document recovery times of a few months (or years where the scale of disturbance has been particularly large). With respect to infaunal invertebrate communities (macro- and meiofaunal), the most widely used interpretation appears to be the achievement of comparable species richness and abundance to pre-disturbance or reference/control levels (e.g. Cross & Curran, 2004; Zajac, 2004) with many authors also accounting for community structure (e.g.Thrush et al., 1996; Hall & Harding, 1997; Dernie et al., 2003b; Lewis et al., 2002; 2003; Bolam et al., 2004). However, Zajac & Whitlatch (2003) pointed out that the interpretation of successional endpoints such as recovery should be considered and stated that factors such as population structure may not be in synch with community level measures of recovery. That population-level differences between disturbed (recovering) and ambient communities are longer lasting than differences in community structure has been documented by several authors (e.g Smith & Brumsickle, 1989; Thrush et al., 1996; Beukema et al., 1999; Zajac & Whitlatch, 2003). Paterson et al. (2011) stated that species can be categorised functionally based on their ecological roles but also in terms of the ecological scales at which they operate (i.e. their influence on the environment) which corresponds to body size. This means that population structure (individual size, biomass and age class) must recover as well as the abundance of the component species. According to Thrush & Dayton (2002), as density declines, population structure (e.g. size and age frequency of organisms) and individual distribution are altered. Whilst species may not be completely absent, they may not be able to fulfil their functional roles. Lohrer et al. (2010) found that important ecological functions (e.g. primary production and ammonium efflux) were reduced by the loss of macrofauna and stated that recovery of ecological function would be prolonged in areas where the macrofaunal communities were dominated by large, long lived species. Finally, permanent habitat changes (e.g. changes in elevation, sediment composition or hydrodynamic regime) may mean the recovery to a pre-disturbance or reference state is unrealistic. The lack of convergence between the recovering and the reference habitat should not necessarily be regarded as failure to recover (Bolam et al, 2006), provided that an alternative state is achieved and that the necessary ecological functions are provided. However, the long term sustainability of the new habitat and its associated functions should be considered, accounting for habitat change over time and the fact that it may evolve into something which represents neither the structure nor the function of the intended habitat (Mazik et al., 2010).

Almost all of the studies reviewed focus on the structural recovery of soft sediment habitats and species with very little consideration for ecological function. There are only a few studies where ecological function has been directly measured and, not necessarily in the context of recovery. For example, Bolam et al. (2002) examined the effect of diversity and biomass on nutrient flux, oxygen consumption, redox potential and sediment shear strength in estuarine mudflats and found the diversity of functional groups to be more important than species composition. Bolam et al. (2011) also used measures of secondary production as an indicator of ecological functioning in relation to the disposal of dredged material.

In an environmental management context, indices of environmental quality now frequently include a measure of function. For example, the Infaunal Quality Index (UK IQI) developed by the Environment Agency is based upon a combination of species richness, diversity and the relative abundance of organisms belonging to functional groups. These groups form the basis of the AZTI Marine Biotic Index (AMBI, a component of the UK IQI) and relate to the tolerance/sensitivity of each species to organic enrichment (Borja et al., 2000). Other authors have examined the diversity and relative abundance of functional traits (Biological Traits Analysis, e.g. Bremner et al., 2003), including bioturbation mode, feeding mode, reproductive strategy, body size, body shape, motility and mode of life in the sediment (e.g. infaunal or epifaunal). Whilst both of these approaches provide an effective means of assessment of ecological function, they can only be classed as indicative since no direct measure of function is actually made. Similarly, whilst body size forms a component of Biological Traits analysis, it is based only on general size ranges for each species and takes no account of actual size (for example, there is no indication of recruitment or population structure), despite its direct relationship with ecological function (Peterson et al., 1998).

There does not currently appear to be any clear guidance on what functions should be measured or how to interpret their importance. A better understanding of the terms ‘ecological’ and ‘ecosystem functioning’ is required since these terms are used in a wide context, often with no definition of which functions are the most critical to a particular site.

# 7. CONCLUSION AND RECOMMENDATIONS.

## 7.1. Is ‘minimising the footprint’ an effective intervention to maximise the recovery of intertidal sediments from disturbance?

The evidence presented in this review suggests that recovery time increases with increasing spatial scale and intensity of disturbance but also with increasing study detail. Simple studies of species richness and abundance indicate faster recovery times than more complex studies of population and community structure. Whilst species richness and abundance can recover within months, recovery of population structure can take several years following defaunation, even when the physical structure of the sediment has not been altered (e.g. Beukema et al., 1999). The duration of most studies has been insufficient to reliably document recovery patterns and timescales. Furthermore, compared to the intensity and scale of some human activities, the scale of most of the studies reviewed here is small suggesting that the recovery time following some forms of anthropogenic disturbance may be comparatively long. The variability in the definition of recovery between studies, together with variability in habitat, disturbance type and experimental design makes it difficult to draw any firm conclusions about the relationship between disturbance scale and intensity and recovery.

Very few studies have examined recovery following disturbance that does not cause complete defaunation or where a patchwork of disturbed and non-disturbed areas exist. Assuming that the sediment conditions are suitable and recovery is allowed to begin coincidentally with the larval recruitment period, recovery of species abundance and community structure should be rapid (6-12 months e.g. Beukema et al., 1999; Dernie et al., 2003b). Furthermore, the proximity to undisturbed patches will enhance the potential for recovery of population structure by providing a source of nearby adult colonists. It should be noted though that emigration of adults and larger organisms from undisturbed areas will result in an overall change in population structure throughout the area. With regard to seagrass species, their generally low dispersal ability means that recovery potential will decrease with increasing spatial extent of disturbance, assuming uniform removal of shoots throughout the area. In this case, low intensity disturbance over a wider area (leaving some seagrass patches intact) may enable faster recovery.

Finally, the difference between the impacts of natural and anthropogenic disturbance must be considered. Activities such as pipe line construction, bait digging and beach re-charge involve digging or smothering to depths of tens of centimetres to meters over hundreds of square meters (in the case of construction activities). They therefore have a far greater impact than natural influences such as storms and bioturbation. Due to the nature of some activities, it might be difficult to minimise the impacts by spreading the disturbance out, particularly in relation to construction work or beach nourishment. However, it may be feasible to spread the impacts associated with, for example, access.

## 7.2. Recommendations

The evidence presented here suggests that widespread impacts over large spatial areas will lead to prolonged recovery times. In most studies, the intensity of disturbance has been relatively uniform. The hypothesis that low level, widespread impacts (i.e. a patchwork of small disturbance areas over a large area) could enable greater recovery potential needs to be tested experimentally in the field and should consider timing of disturbance, temporal variation in larval supply, the mobility of adult species, the recovery potential of the habitat structure (e.g. infilling of pits, erosion of mounds, recovery of particle size distribution, organic content, sediment stability, porosity and cohesion). It is important that the potential for recovery of all structural aspects of the community is studied including species composition, relative abundance, individual and community biomass, organism size and population structure in order to assess the recovery of ecological function. Long term studies (5 years or more) may be required to assess the recovery of population structure and ecological function. Variation in habitat type and disturbance type are important considerations.

The recovery potential of soft sediment habitats which have been disturbed but not completely defaunated (i.e. an infaunal community is present) is not known and it is recommended that any study involving low intensity disturbance takes thorough account of the baseline community from which recovery is being measured. Furthermore, methods for the characterisation of sediment properties should account for differences in surface and subsurface conditions given the importance of surface sediment properties in influencing recruitment success.

In the context of environmental management within the UK, recovery needs to be clearly and uniformly defined to enable an holistic approach to understanding and monitoring the recovery process between the regulatory and conservation bodies. Finally, recovery of ecological function requires consideration.

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