DEFINING AND ASSESSING BENTHIC RECOVERY FOLLOWING DREDGING AND DREDGED MATERIAL DISPOSAL

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ABSTRACT

Assessing the recovery of benthic habitats disturbed by dredging and dredged material disposal operations is an important and growing management issue throughout the world. Although many projects have been monitored and a substantial literature on the subject exists, few generalizations can be made about typical recovery rates because biological responses are influenced by numerous factors, including site-specific bathymetry, hydrodynamics, depth of deposited sediments, the spatial scale of the disturbance, sediment type, and the timing and frequency of the disturbance. Additionally, there is no accepted definition of what constitutes "recovery." In various studies, recovery has been defined as a return of benthic resources to a baseline (pre-impact) condition, a reference (neighboring unimpacted) condition and/or both. Infaunal macroinvertebrates are most commonly monitored to assess benthic recovery, usually by sampling the substrate, preserving organisms in formalin, and identifying and enumerating the organisms in the laboratory. Methods used to analyze these data vary and may influence study conclusions just as much as the aforementioned physical factors. Study results are generally presented through some combination of three methods of data analysis, i.e., univariate statistics (very common and used almost exclusively in early studies), multivariate statistics (increasingly common), and benthic indices. We review benthic recovery rates reported for approximately 50 dredging-related (disposal and dredging sites) projects and explore the relative influence of both physical and analytical factors in the determinations of recovery status. Although early impact assessments relied heavily upon univariate diversity indices that were derived from species level identifications of macroinvertebrates, it has become increasingly apparent that multivariate analyses of the same data sets provide more sensitive measures of ecological status.

Keywords: Meta-analysis, infauna, thin-layer, salt marsh, mud flat habitat

INTRODUCTION

Impact assessment of dredging activities has been conducted for many years and over a broad geographical range, although most study results are not published in the peer-reviewed literature (Bolam and Rees 2003). There have been several recent reviews of the environmental consequences of dredging impacts (Fredette and French 2004, Bolam et al. 2006a, Brooks et al. 2006), but no general consensus has emerged of an operational definition of recovered benthic habitat. Some studies define recovery as a return of the impacted area to pre-disturbance conditions, whereas others indicate recovery is attained once the impacted area is equal to or exceeds an undisturbed reference area in terms of biological metrics, however, locating suitable reference areas is often challenging (Quigley and Hall 1999, Sanchez-Moyano et al. 2004, Fraser et al. 2006). Estimates of benthic recovery rates are summarized as ranging from several months to several years (Qian et al. 2003), but associations with disturbance types (i.e., dredging vs. disposal sites) or details of the analytical approaches (univariate vs. multivariate statistical techniques) that were used to determine recovery status are generally not provided. Although there are several environmental conditions that are commonly identified as influencing benthic recovery rates, such as sediment type, depth of overburden, frequency and timing of deposition, and receiving habitat type, the relative importance of these factors in influencing benthic recovery is not known. In addition, benthic communities in estuaries can be highly dynamic, thus making it difficult to distinguish between natural variation and changes that occur as part of the recovery process (Wildish and Thomas 1985).

A general benthic successional paradigm (Pearson and Rosenberg 1978, Rhoads and Germano 1986) states that following initial decreases to benthic diversity, abundance, and biomass that immediately follow a disturbance, pioneering (Stage I) organisms, such as small, tube-dwelling polychaetes and small bivalves colonize the surficial sediments. These opportunistic taxa occur in relatively high abundances and low diversity and over time are

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replaced by larger, longer-lived and deeper-burrowing (Stage II) species. The Stage III assemblage is comprised of a more diverse but less abundant group of larger taxa such as maldanid polychaetes. Thus, the absence of deposit feeders and mid-depth burrowers is indicative of areas that are in a state of recovery. Reviews of dredging impacts on seagrass (Erftemeijer and Lewis 2006) and unvegetated benthic habitats (Bolam et al. 2006a) indicate that site-specific factors influence impacts, thus limiting the extrapolation of results across geographic regions. We summarize the ranges of observed recovery rates across broad habitat categories and disturbance conditions and examine potential associations between biological responses and characteristics of the dredging projects while noting the influence of different analytical approaches on determinations of recovery status. In addition, we review biological responses to the intertidal placement of dredged material in "beneficial use" projects, in which dredged material is used as a resource to enhance habitats.

PHYSICAL FACTORS AFFECTING BENTHIC RECOVERY

Depth of Overburden at Disposal Sites

Some benthic organisms such as burrowing polychaetes, amphipods and molluscs can colonize newly deposited sediments through vertical migration, therefore, if dredged material depths are limited to within the vertical migration capacity of these organisms (20-30 cm), recovery rates may be quicker than if colonization is dependent upon the lateral migration of juveniles and adults from adjacent areas and larval settlement. Successful vertical migration through 15 cm of sediments occurred for benthic infauna in Auckland, NZ (Roberts et al. 1998) and mud snails in Delaware Bay (Miller et al. 2002). Successful movements through up to 32 cm have been documented for polychaetes (Maurer et al. 1982) and bivalves (Maurer et al. 1981). The amount of the deposit and the frequency of deposition are interactive factors affecting vertical migration for nematodes (Schratzberger et al. 2000).

Habitat Type (disturbance history)

Shallow benthic habitats (< 20 m depth, Hall 1994) experience relatively frequent wave, wind, and current induced disturbances and thus are typically inhabited by low-diversity, r-selected benthic assemblages that can readily re-establish themselves under conditions of high frequency disturbances (Dauer 1984, Clarke and Miller-Way 1992, Ray and Clarke 1999). These communities are naturally held in early successional stages and therefore, are able to recover more rapidly than communities in deeper, more stable environments (Newell et al. 1998, Bolam and Rees 2003).

Sediment Type

Rapid recolonization of soft-bottom benthic habitats is frequently associated with either unconsolidated fine grain sediments (Cruz-Motta and Collins 2004) or the rapid dispersion of fine-grained dredged material by currents (Van Dolah et al. 1984). Newell et al. (1998) characterized typical recovery times at 6-8 months for mud habitats and 2-3 years for sand and gravel substrata.

Spatial Scale of Disturbance

The spatial scale of the dredged or disposal area may be proportional to recovery times (Zajac et al. 1998, Guerra-Garcia et al. 2003). For small-scale disturbances, the edge/surface area ratio of the disturbed area is larger than for larger disturbances, therefore colonization through adult immigration from surrounding undisturbed areas may facilitate recovery. With larger disturbed areas, the central portion of the disturbed areas is reliant upon settlement from the water column for colonization, which is very dependent on seasonal recruitment patterns and local hydrodynamics. Guerra-Garcia et al. (2003) demonstrate a log-linear relationship between recovery times and spatial scale using 14 studies of recovery at dredging and disposal sites. For instance, recovery in small patches (1000 m^2) took 7 months, whereas recovery was projected to require years at spatial scales of 10^5 m^2 and above.

Timing and Frequency of Disturbance

Avoiding dredging activities after seasonal larval recruitment periods is a common practice when possible. Deposition of sediments in several smaller units rather than one deep deposit also may be less detrimental to the benthos. In a microcosm study, sediment deposited in a single event caused more severe changes to nematode assemblages than the same amount of sediment deposited in smaller doses (Schratzberger et al. 2000).

METHODS THAT AFFECT RECOVERY ESTIMATES

Sampling

Sampling different components of the benthos may affect determinations of an area's recovery status, for example, nematodes are more sensitive to sediment structure than macrofauna and thus may exhibit changes in community structure more readily (Boyd et al. 2000). However, changes in meiofaunal community structure do not persist as long as changes to the macrofaunal community (Coull and Chandler 1992, Somerfield et al. 1995), therefore the faunal assemblage that is targeted for sampling may affect perceptions of impact severity and recovery rates. The accuracy of assessing habitat conditions can be increased by using remote survey methods, such as sidescan sonar and photography (both of which require ground-truthing). Habitat characterizations that are based solely on biological data may be a function of sampling methods (Rees et al. 1999, Brown et al. 2001). Macrofauna are most commonly monitored to assess benthic recovery, usually by sampling the seafloor, preserving organisms in formalin, and identifying and enumerating the organisms in the laboratory. Statistical techniques, however, vary among studies and may influence recovery estimates depending on the approach taken. In addition, the sampling methods used can influence the identity and abundance of benthic organisms that are captured. Two common methods of sampling soft benthic habitats are box cores and grab samplers.

Analytical

Study results are generally presented through some combination of three methods of data analysis, i.e., univariate statistics (very common and used almost exclusively in early studies), multivariate statistics (increasingly common), and benthic indices. Less common analytical approaches include examinations of functional groups (Niemi et al. 1990, Wilber and Stern 1992) and secondary production (Wilber and Clarke 1998). The thoroughness of data analyses may affect study conclusions, for example univariate measures such as total infaunal abundances (McCauley et al. 1977) may suggest more rapid recovery than multivariate (Bolam and Whomersley 2005) or functional group (Wilber and Stern 1992) analyses of the same datasets.

Univariate measures include commonly reported parameters such as, total abundance, taxa richness, and total biomass. There are a number of diversity indices that reduce multivariate data (e.g., abundances of multiple species) into a single index, which can then be treated statistically using univariate analyses. Common diversity indices include the Shannon-Weiner diversity index (H'), Margalef's index (d), Pielou's evenness index (J'), and the Simpson index (λ). Changes between sites or over time of univariate measures (including indices) are usually plotted as means and confidence intervals for each site and time (e.g., Fig. 1, taken from Burlas et al. 2001) and recovery is indicated when values for the impacted area (in this case, black circles) are no longer significantly lower than pre-disturbance levels or those of a reference location. Multivariate analyses of community structure have historically been conducted using cluster analysis, with resultant dendograms that group samples (for example, stations or sites) such that samples within a group are more similar to each other than samples from different groups. Segregation of stations from disturbed and reference areas may indicate impacts have affected community structure. Increasingly, non-parametric multi-dimensional scaling (MDS) ordination plots using the Bray-Curtis similarity measure are created to identify groups of samples having similar faunal assemblages (Clarke and Warwick 2001). Separation of impacted from reference sites is visually apparent and can be statistically tested with multivariate tests such as ANOSIM (Clarke and Warwick 2001). In addition, impacted sites frequently have greater variability in community composition than reference areas (Warwick and Clarke 1993), therefore the relative dispersion of samples from the two habitat types can be compared (e.g., Fig. 2, taken from Boyd et al. 2004). In Fig. 2, it is apparent that benthic community structure at the reference (blue) and low frequency (green) disturbance stations is both less variable and different from that of stations where dredging is more frequent (red). By using MDS ordinations, Jewett et al. (1999) demonstrate a similar pattern of higher variability among stations in sand and cobble substrate in the

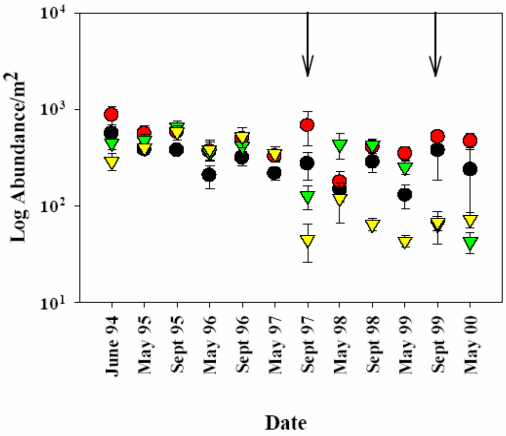

Figure 1. Example of univariate analyses that tracks a variable (total infaunal abundance) at an impacted site (black circles) before and after a disturbance (arrows) relative to reference areas (other symbols) from Burlas et al. 2001.

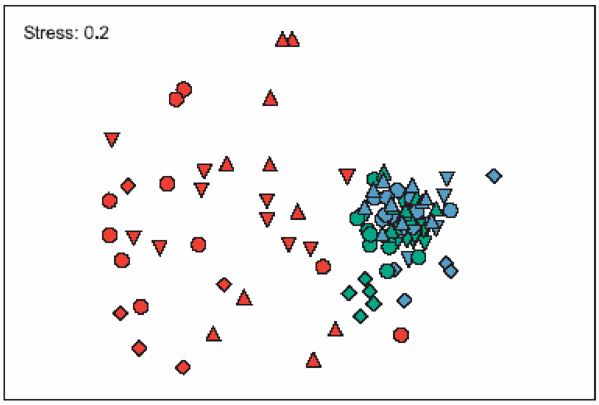

Figure 2. Example of multivariate analyses in which an MDS plot depicts variable community structure at high frequency disturbance sites (red symbols) relative to low frequency (green) and undisturbed (blue) sites over several years (denoted by different symbol shapes) from Boyd et al. 2004.

northeastern Bering Sea in a frequently mined area as compared to less variable community structure among stations in recovering habitat (2 – 7 years post-mining) and reference un-mined stations, which exhibited the smallest variation in community composition. Similar patterns of convergence in community variability are apparent as benthic gravel mining sites in the North Sea were sampled 2 weeks, one-year, and two-years after dredging and compared to reference stations (Kenny and Rees 1996).

Benthic indices have been developed to integrate macrobenthic community parameters into a single metric that can be used to distinguish between disturbed and undisturbed areas. In an effort to economize on labor, indicator species that are identified as either disturbance tolerant or intolerant are used to define areas as either disturbed or undisturbed, respectively. Relative abundances of these indicator species can be combined to provide an index of biological impact that is site- and impact-specific, thus simplifying subsequent monitoring efforts (Roberts et al. 1998). The major benefit of such an approach is the time and labor cost savings of not identifying the full macrofaunal assemblage. A potential problem with this approach is that reliance on single indicator species that may be patchily distributed can lead to erroneous interpretations. For example, the absence of a species cannot be reliably interpreted without extensive sampling. An index of biotic integrity developed for the Gulf of Mexico

(Engle et al. 1994) integrates both community structure and function. It is comprised of a benthic diversity measure (Shannon-Wiener index adjusted for salinity), proportion of tubificid oligochaetes (an indicator of organic pollution), and the proportion of bivalve molluscs (indicative of undegraded environmental conditions, Pearson and Rosenberg 1978).

RECOVERY ESTIMATES

Disposal Areas

Recovery of dredged material disposal sites has been studied throughout the world (Table 1). Longer recovery rates (up to several years) are observed at higher latitudes (Blanchard and Feder 2003, Harvey et al. 1998) where the associated stable physical environments and long-lived taxa take longer to recover from disturbances (Newell et al. 1998). Relatively rapid recovery of temperate and sub-tropical disposal areas (Table 1) is attributed to a greater community composition of opportunistic species at these latitudes (Clarke and Miller-Way 1992, Van Dolah 1984).

Theories as to the mechanisms of recovery at disposal sites, i.e., whether by adult migration or larval recruitment or some combination thereof, are usually circumstantially derived. One mode of colonization unique to disposal sites is the transfer of adult and/or juvenile organisms from the dredge site. Jones (1986) demonstrated that dredged material disposal buried the macrofaunal community at the disposal site, which was replaced by taxa common at the dredged site that had survived the dredging and dumping process. Vertical migration of juveniles and adults through the deposited sediments is also thought to contribute to relatively quick recovery rates in areas with shallow deposits or rapid dispersion of the dredged material due to currents or waves (e.g., McCauley et al. 1977, Newell et al. 1998, Ray and Clarke 1999).

Dredged Areas

Sand Mining

Recovery of dredged areas covers a broad spectrum of disturbance types, such as dredging of sand for beach replenishment projects (sometimes called borrow areas). Planning and operational aspects of the dredging practice can play an important role in influencing recovery rates. Dredging sand for beach nourishment typically results in either the creation of relatively shallow pits that are refilled by sand movement and are rapidly recolonized by opportunistic infauna or the creation of deeper pits that become depositional areas where fine sediments accumulate and sand-associated assemblages are replaced by soft-bottom fauna (Burlas et al. 2001). If borrow pits are deep enough that water circulation is restricted, hypoxic or anoxic conditions may result in a depauperate infaunal community. If possible, borrow areas can be located on bathymetric peaks so as not to create depressions on the seafloor (Burlas et al. 2001). Recovery of infaunal abundance, diversity and community composition in New Jersey occurred in one year, whereas the return of biomass to reference conditions took 1.5 to 2.5 years (Burlas et al. 2001). In Florida, relatively rapid recovery (~ 1 year) was reported for borrow areas when measured in terms of total abundance and taxonomic diversity, however, recovery times of up to three years were needed to restore functional groups, such as deposit feeders and mid-depth borrowers (Wilber and Stern 1992).

Aggregate Mining

The effects of marine aggregate mining (sand and gravel extraction) on the environment and fisheries is a concern along the eastern and southern English coastlines where the practice has been common for decades, providing aggregate for both the construction industry and a source material for beach nourishment (Boyd et al. 2004). Benthic recovery in the resultant saucer-shaped depressions that are 8-10 m deep is dependent upon the scale of the dredging operation, frequency of dredging, degree to which sediments are changed, hydrodynamics and the general nature of the habitat (reviewed in Newell et al. 1998). Dredging can be expected to reduce macrobenthic population densities 40-90% and species diversity 30-70% (Newell et al. 1998), with restoration of benthic communities on the sandy, coarse gravel substrate occurring within approximately 2-4 years following the cessation of dredging (Table 1, Newell et al. 2004, Boyd et al. 2004). Typically, recovery of biomass takes longer than other community attributes, such as total infaunal abundance (Kenny and Rees 1996, Newell et al. 2004) and changes in community structure can last for many years, especially if there is a change in sediment type (Desprez 2000). Opportunistic

Table 1. Selected marine and estuarine studies in which benthic macrofaunal recovery rates were reported.

Site	Region	Depth (m)	Sediment Type	CH[1]	Mech[2]	Recovery Time[3]	Metric[4]	Reference
Open Water Disposal Sites								
New S. Wales, Australia	Temperate	6	Fine sand	N	A	3 months	U/M	Smith and Rule 2001
Gulfport, MS, US	Temperate	3	Silt and clay	Y	A	1 year	U/M	Wilber et al. in press
Corpus Christi, TX, US	Temperate	3	Silt and clay	N	L/A	< 1 year	U/M	Ray and Clarke 1999
South Carolina, US	Temperate	13	Fine sand	Y	Un	N/A	U/M	Zimmerman et al. 2003
Coastal Louisiana, US	Temperate	3	Silt and clay	N	Un	5 months	U/M	Flemer et al. 1997
Sewee Bay, SC, US	Temperate	3	Silt and clay	Y	A	6 months	U/M	Van Dolah et al. 1979
Dawho River, SC, US	Temperate	<5	Silt and clay	Y	A	3 months	U/M	Van Dolah et al. 1984
Delaware Bay, US	Temperate	Shallow	Silt and clay	N	Un	>5 months	U	Leathem et al. 1973
Queensland, Australia	Sub-Tropical	11	Silt and clay	Y	A	3 months	U/M	Cruz-Motta and Collins 2004
New S. Wales, Australia	Temperate	Shallow	Silt,clay,sand	N	A	1 month	U	Jones 1986
Mobile Bay, AL, US	Temperate	3	Mud	N	A	3 months	U	Clarke and Miller-Way 1992
Oregon, US	Temperate	8	Silt and clay	N	A	1 month	U	McCauley et al. 1977
Mirs Bay, Hong Kong	Sub-Tropical	19	Sand and gravel	Y	Un	< 2 years	U/M	Valente et al. 1999
Quebec, Canada	Cold	55	Fine sand	Y	L/A	> 2 years	U/M	Harvey et al. 1998
Port Valdez, Alaska	Cold	15-23	Mud	N	L	> 2.5 years	U/M	Blanchard and Feder 2003
Puget Sound, WA	Cold	60	Silt,clay,sand	N	A	> 9 months	U	Bingham 1978
Western Baltic Sea	Cold	19	Fine sand	N	A	< 2 years	U/M	Powilleit et al. 2006
Liverpool Bay, UK	Cold	10	Sand and mud	N	Un	N/A	U/M	Rees et al. 1992
Weser estuary, Germany	Cold	16	Silt and sand	Y	Un	> 8 months	U/M	Witt et al. 2004
James River, VA	Temperate	3	Fluid mud	N	L/A	3 months	U	Diaz and Boesch 1977, Diaz 1994
Columbia River, OR	Cold	Shallow	Fine sand, clay	N	L/A	>10 months	U	Richardson et al. 1977
Southern Brazil	Temperate	19	Silt,clay,fine sand	Y	A	< 9 months	U/M	Angonesi et al. 2006
Dredging Site – Borrow Area								
South Carolina, US	Temperate	Shallow	Sand	Y	A	3-6 months	U/M	Jutte et al. 2002
New Jersey, US	Temperate	17	Medium fine sand	Y	L/A	1 year	U/M	Burlas et al. 2001
Florida, US	Sub-Tropical	9-12	Sand	N	Un	2-3 years	FG	Wilber and Stern 1992
North Sea, Denmark	Cold	20	Sand	Y	L/A	2-4 years	U/M	van Dalfsen et al. 2000
Bay of Brest, France	Cold	7	Sandy mud	N	L/A	2 years	U/M	Hily 1983

Table 1 (continued)

Wadden Sea, Germany	Cold	10	Sand	N	A	2 years	U/M	Schuchardt et al. 2004
North Carolina, US	Temperate	12-15	Sand	N	L/A	< 9 months	U/M	Posey and Alphin 2002
Florida, US	Temperate	10	Medium sand	Y	Un	9-12 months	U	Johnson and Nelson 1985
NW Mediterranean	Temperate	15	Coarse/med sand	Y	Un	> 2 years	U	Sarda et al. 2000
Dredging Site - Channels								
Sewee Bay, SC, US	Temperate	4	Silt and clay	Y	A	6 months	U/M	Van Dolah et al. 1979
Dawho River, SC, US	Temperate	4	Silt and clay	N	A	3 months	U	Van Dolah et al. 1984
Georgia, US	Temperate	Shallow	Silt and clay	N	A	3 months	U	Stickney and Perlmutter 1975
Oregon, US	Temperate	11	Silt and clay	N	A	1 month	U	McCauley et al. 1977
Delaware Bay, US	Temperate	Shallow	Silt and clay	N	Un	>5 months	U	Leathem et al. 1973
Sardinia, Italy	Temperate	15-20	Silt and clay	N	A	~ 6 months	U	Pagliai et al. 1985
Ceuta, North Africa	Temperate	3	Silt and clay	Y	L/A	6 months	U/M	Guerra-Garcia et al. 2003
New South Wales, Australia	Temperate	Shallow	Silt,clay,sand	N	A	1 month	U	Jones 1986
Queensland, Australia	Temperate	17	Medium/fine sand	N	Un	N/A	U	Poiner and Kennedy 1984
Southwest Finland	Cold	9	Mud	N	L/A	2-5 years	U	Bonsdorff 1980, 1983
Long Island, NY	Temperate	2	Sand,silt,clay	Y	A	> 11 months	U	Kaplan et al. 1975
Algeciras Bay, Spain	Temperate	5,15,30	Fine sand	N	L/A	4 years	U/M	Sanchez-Moyano et al. 2004
Yaquina Bay, OR	Cold	6-11	Fine sand,silt	Y	L/A	1 year	U/M	Swartz et al. 1980
North Sea, UK	Cold	9	Silt and clay	N	A	> 3 months	U/M	Quigley and Hall 1999
Southern Brazil	Temperate	3-18	Silt,clay,sand	Y	Un	> 3 months	U/M	Bemvenuti et al. 2005
Dredging Site - Aggregate Mining								
Nome, AK	Cold	9-20	Sand, cobble	Y	Un	4 years	U/M	Jewett et al. 1999
Sotheast coast, England	Cold	27-35	Sand, gravel	Y	L	2-4 years	U/M	Boyd et al. 2004
South coast of U.K.	Cold	10-20	Sand,mud,gravel	N	Un	2-3 years	U/M	Newell et al. 2004
Eastern English Channel	Cold	15	Gravel	Y	Un	> 28 months	U	Desprez 2000
Southern Baltic Sea	Cold	10-14	Sand	Y	Un	> 10 years	U/M	Szymelfenig et al. 2006.
Southern North Sea	Cold	25	Sand, gravel	Y	L	> 2 years	U/M	Kenny and Rees 1996
Botany Bay, Australia	Temperate	14-18	Mud	Y	L	> 1 year	U/M	Fraser et al. 2006

Table 1 (continued)

Defaunation - Anoxia/Hypoxia								
Hong Kong, China	Sub-Tropical	1	Sand	N	L	< 15 mo	U/M	Lu and Wu 2000
Tampa Bay, FL, US	Sub-Tropical	intertidal	Silt,clay,sand	N	A	1 month	U	Dauer 1984
Tampa Bay, FL, US	Sub-Tropical	4-5	Silt, clay	N	L/A	8 months	U	Santos and Simon 1980
Gullmar Fjord, Sweden	Cold	115	Silt and clay	N	L	> 18 months	U	Josefson and Widbom 1988
Capping								
Hong Kong, China	Sub-Tropical	5-6	Mud	N	L	3 years	U/M	Qian et al. 2003

[1]Changes (CH) to the sediment type (granulometry) by the benthic disturbances are indicated by Y (yes) or N (no).
[2]The mechanism (Mech) of recovery (usually speculated) is given as A – adults, L – larval recruitment, or Un – unknown probably due to sampling protocol.
[3]Studies in which recovery times were not reported are noted as not applicable (N/A), for instance disposal areas may have been surveyed while they were in use or years after disposal stopped.
[4]Metrics used for data analyses and determinations of recovery rates are noted as either univariate (U), multivariate (M), or functional groups FG.

colonists of gravel substrates include barnacles, whereas polychaetes are more common colonists of sandy areas (Boyd et al. 2004). Impacts have also been observed outside the boundaries of the dredged areas, for example a local enhancement of the abundance of filter feeders that may reflect organic enrichment resulting from the dredging activity (Poiner and Kennedy 1984, Robinson et al. 2005). Excavation of deep pits (10-14 m) that are approximately 4-5 times deeper than the average bay depth in the southern Baltic Sea has created isolated depauperate microhabitats that differ from unimpacted reference areas over ten years after sand extraction (Szymelfenig et al. 2006).

Channels

Studies of benthic recovery in dredged channels are restricted to relatively shallow, less-stable habitat types and thus have relatively short recovery rates (Table 1). One mechanism of recovery in dredged areas is the colonization by infauna from adjacent areas undisturbed sediment, or from "hummocks" of unexcavated sediment that remain within the footprint of dredged bottom. Re-colonization of the defaunated dredged areas may occur from adults migrating from the relatively undisturbed hummocks (McCauley et al. 1977, Jutte et al. 2002). Rapid recolonization of unconsolidated sediments in dredged channels (Stickney and Perlmutter 1975, Van Dolah et al. 1984, Jones 1986) and a muddy-maerl habitat (DeGrave and Whitaker 1999) were attributed to slumping of non-dredged sediments into the dredged furrows, thus transporting benthic infauna.

BIOLOGICAL RESPONSES TO BENEFICIAL USES OF DREDGED MATERIAL

The intentional placement of dredged material to provide habitat functions is an alternative to open-water disposal that is commonly referred to as "beneficial use." Beneficial use projects in the marine environment are commonly conducted in intertidal habitats and include sediment enhancement of mudflats and tidal marshes through either habitat creation or restoration. Assessments of these restoration efforts typically rely on comparisons to reference habitats since the pre-discharge communities occur at lower tidal levels with associated differences in environmental conditions (Bolam et al. 2006b). Because salt marsh creation success for projects constructed on dredged material has recently been reviewed (Streever et al. 2000), we restrict our assessment of biological responses to thin layer dredged material placement in marsh restoration projects (sensu Cahoon and Cowan 1988). In this placement method, dredged material is hydraulically sprayed over the marsh habitat. The thickness of the sediments and the extent to which soil characteristics are changed influences the number of roots and rhizomes that survive to generate new shoots and establish a root and rhizome mat at the newly appropriate soil depth (Wilber 1993). This response mode typically requires two growing seasons. If altered soil conditions prevent substantial root and rhizome survival and shoot penetration into the dredged material, the marsh may be recolonized by seedlings, which requires longer to establish typical marsh vegetation.

Deltaic marsh habitats are being lost at high rates due to excessive inundation that occurs in many areas from a combination of subsidence and sea level rise. Deposition of sediments in river deltas has been altered by the construction of dams, levees, and berms, thus sediments have been artificially supplied (beneficial use projects) to raise marsh elevations and reverse habitat loss. In Louisiana, dredged sediments sprayed as a slurry over marsh habitat in thin layers (typically less than 15 cm) increased salt marsh grass, *Spartina alterniflora*, percent cover and stem density (Ford et al. 1999, Slocum et al. 2005).

Intertidal mudflats dissipate tidal and wave energy and provide important feeding areas for shorebirds and migratory waterfowl. Areas with inadequate sediment supply have lost mudflat habitat, which can exacerbate the erosion problems that occur at the base of seawalls (Widdows et al. 2006). In some areas, mudflat habitats have been lost through infilling and subsequent development and excavation for facilities such as ports (Evans et al. 1998). Because of limited knowledge concerning the movement of dredged material placed in the intertidal zone and the recovery rates of buried intertidal habitat, beneficial placement of dredged material in the UK has been limited to small-scale trials (Bolam and Whomersley 2005). Biological responses of created mudflat habitats (Table 2) include infaunal communities that were comparable to reference areas within two years of construction (Ray 2000), whereas at least three years were needed to create mudflat habitat with adequate abundances of benthic prey for shorebirds (Evans et al. 1998).

Table 2. Biological responses of intertidal mudflat and salt marsh habitats following dredged material placement as part of beneficial use projects.

Mudflat Habitat		
Source	Site	Summary of results
Bolam and Whomersley 2005	Southeastern UK	Low organic content/ silt/clay,high organic content/silt/clay, multivariate – 12 months, multivariate – >12 months
Bolam et al. 2006b	Southeastern UK	Diverse macro and meiofaunal communities were re-established within three months of sediment deposition. After 42 months, these communities remained significantly different from reference sites.
Widdows et al. 2006	Southeastern UK	Sediment erosion was measured with relation to abundances of key plant species that served as bio-stabilizers.
Schratzberger et al. 2006	Southeastern UK	Nematode colonization of the dredged material mudflat resulted from settling of suspended nematodes and their subsequent reproduction and differential survival.
Evans et al. 1999	Northeastern UK	Invertebrate colonization and bird foraging behavior suggest the functional attributes of the mudflat were achieved three years following sediment recharge.
Ray 2000	Maine, US	Infaunal taxa richness, abundance, and species diversity were similar between reference and constructed sites within two years of construction. Infaunal biomass at constructed sites remained lower than reference values at two constructed mudflats.
Salt Marsh Habitat		
Source	Site	Summary of results
Wilber et al. 1992	North Carolina, US	Healthy stands of marsh vegetation were present on thin-layer disposal areas ten years after deposition, however, species composition of the recharged habitat differed from that of reference areas.
Croft et al. 2006	North Carolina, US	Sediment (0-10 cm) placed on deteriorating marsh plots increased *Spartina* stem density by second growing season to reference levels.
Leonard et al. 2002	North Carolina, US	The addition of 2-10 cm of sediment on deteriorating marsh surfaces increased vascular plant stem densities and microalgal biomass. There were no longterm impacts to the infaunal community.
Ford et al. 1999	Louisiana, US	Spraying dredged material (approximately 23 cm depth) knocked down marsh plants, but they resprouted and recolonized the site within a year.
Slocum et al. 2005	Louisiana, US	Areas of the marsh that received moderate (2-12 cm) dredged material exhibited greater *Spartina alterniflora* % cover and canopy heights
Mendelssohn and Kuhn 2003	Louisiana, US	Sediment enrichment was associated with higher percent plant cover, biomass and height. Plant species composition did not change.
Schrift 2006	Louisiana, US	Two years after sediment recharge, marshes in the low elevation areas (11-16 cm above ambient marsh) were the most similar to reference marshes in plant cover and species richness.

Recovery Following Capping

Sediment caps are constructed by covering highly contaminated sediments with uncontaminated sediments to physically isolate the contaminants from fauna, flora, and other habitats. Most monitoring efforts of sediment caps, such as the Disposal Area Monitoring System (DAMOS) program established by the US Army Corps of Engineers, New England District have focused on documenting whether contaminants were contained by sediment caps (Fredette and French 2004) rather than rates of biological recovery. Comprehensive monitoring in Long Island Sound has demonstrated that caps can effectively isolate contaminants from the marine environment, with cap material clearly distinguishable from underlying mound material in sediment cores as much as 11 years after disposal (Fredette and French 2004). Bathymetric surveys conducted with sediment profile photography have proved to be effective methods of monitoring the physical stability of sediment caps (Nakayama et al. 1998, Fredette and French 2004). One study that documented recolonization of sediment caps on the subtropical coast of Hong Kong indicated recovery took several years (Qian et al. 2003), a finding that is consistent with other studies conducted in stable areas that are sheltered from storms and currents (Table 1).

FRESHWATER SYSTEMS

Open-water dredged material disposal in freshwater habitats is not as widely studied although maintaining adequate depths for commercial navigation in rivers and lakes has been a common practice for nearly a century. As with estuarine and marine benthos, freshwater benthic macroinvertebrates are an important energy source for higher trophic levels. Tidal freshwater habitats have low species diversity and are dominated by tubificid oligochaetes and chironomid insect larvae, along with molluscs and other insect and crustacean groups (Diaz 1994). Minimizing loss of benthic productivity is an important consideration for managing dredged material disposal projects in freshwater ecosystems. Corollaries with marine responses have been observed in freshwater studies in that disposal sites are initially colonized by taxa that have short life cycles, high turnover rates and can adapt to different substrate types (Koel and Stevenson 2002). For instance, the abundances of opportunistic oligochaete species as well as variability in community composition increase following disposal (Flint 1979). Tubificid oligochaetes are subsurface deposit feeders that can undergo subsurface migrations when environmental conditions deteriorate to more suitable habitat (Diaz 1994). Likewise, recovery times as measured by macroinvertebrate densities are lower than when either biomass or taxonomic richness is the metric being used (Flint 1979, Niemi et al. 1990). Dredged material placement along the main channel of the Illinois River reduced densities of dominant taxa, which included Chironomid midges and Ephemeropteran mayflies, and recovery was not observed within one year (Koel and Stevenson 2002). In contrast, recovery from disposal of fluid mud in a tidal freshwater portion of the James River, VA was achieved within three months (Diaz 1994). Avoiding dredged material placement in reaches with high macroinvertebrate densities, or near islands, may improve riverine productivity.

CONCLUSIONS

Although impacts of dredging and dredged material disposal on benthic habitats are varied and difficult to predict, several generalities emerge when studies are carefully reviewed. Suites of factors can be categorized as either being associated with recovery measured in months (such as, shallow, naturally disturbed habitats, unconsolidated, fine grain sediments, and univariate analytical approaches) or years (e.g., deep, stable habitats, sand and gravel sediments, and multivariate or functional group analytical techniques, Table 1). Perhaps the most consistent physical parameter influencing benthic recovery rates is the prior disturbance history of the habitat in question. Benthic recovery occurs more rapidly in shallow areas where the resident species assemblages are already adapted to shifting sediments. There are no obvious differences in the biological responses to beneficial use practices in intertidal habitats compared to recovery of subtidal benthos following traditional disposal methods (Bolam et al. 2006a). However, knowledge gained to date on how recovery proceeds has not led to a consensus on how to interpret rates of benthic recovery with respect to the need to manage dredging and dredged material disposal projects. Although identifying benthic assemblages at the species level and the use of univariate diversity indices to analyze the resultant data was the norm for early impact assessments, it has become increasingly accepted that multivariate analyses of the same data sets are more sensitive to detecting clear differences in assemblage composition (Warwick and Clarke 1991, Byrnes et al. 2004). Objective means to assess whether these sometimes subtle distinctions signify important differences in ecological functions of the benthic community remain elusive. Linking various spatial and temporal scales of benthic disturbance to appropriate triggers for management decisions will remain an arbitrary process until a unified definition of "recovery" is attained that can serve as a common endpoint for monitoring efforts.

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