Ecological Functions of Shallow, Unvegetated Estuarine Habitats and Potential Dredging Impacts (with emphasis on Chesapeake Bay)

by Gary L. Ray

PURPOSE: The U.S. Army Corps of Engineers (USACE) is faced with increasing numbers of requests to dredge shallow, unvegetated estuarine waters (Figure 1). Most of these proposals involve tidal waters ranging in depth from mean low water (MLW) to 1.2 m (4 ft) below MLW, with projected post-dredging depths of 1.8 m (6 ft) below MLW. This technical note summarizes what is known about the ecological functions of these habitats and the potential impacts of dredging.

Figure 1. Aerial view of an estuary (Photo source: USACE)

BACKGROUND: In 1992, concern over potential deleterious impacts of dredging on shallow, unvegetated estuarine habitats led the U.S. Army Engineer District, Norfolk (CENAO) to request the U.S. Army Engineer Research and Development Center (ERDC) to review the ecological definition, functions, and potential sensitivity of these habitats to dredging. At that time, there was no clear definition of the term "shallow-water" and few studies specifically addressing differences in ecological functions along the depth range of interest. Since this time, there has been a concerted effort to define "shallow water" (e.g., Reilly et al. 1996) and a number of conferences have explicitly addressed ecological functions and importance of shallow-water habitats.

Definition of Shallow Water. In 1993 the U.S. Environmental Protection Agency surveyed environmental managers, regulators, and researchers in order to arrive at a consensus of what constitutes shallow water (Spagnolo et al. 1994, Reilly et al. 1996). Opinions varied widely among the participants, the agencies and geographic regions they represented, and the disciplines in which they worked. This was not surprising since "shallow" is a relative term, differing with the context in which it is used. For example, what is considered shallow by an oceanographer working on the continental shelf is quite different from someone working in estuaries or bays. After considerable debate, shallow water was operationally defined as less than 4 m (13 ft) below MLW. This definition was not meant to be an inflexible standard, but a starting point for further discussions and refinement. Despite an exhaustive search of the technical and scientific literature, no alternative definitions have been put forth since these publications.

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Estuarine Habitat Classifications. There are four major habitat classification schemes encompassing estuarine habitats of the United States: Shaw and Fredine (1956), Cowardin et al. (1979), Dethier (1992), and Allee et al. (2000). Shaw and Fredine (1956) is a descriptive scheme that differentiates among regularly and irregularly flooded intertidal habitats and open-water (subtidal) habitats, but provides no further distinction based on depth. Cowardin et al. (1979), Dethier (1992), and Allee et al. (2000) are all framed as dichotomous keys and share a similar hierarchical structure. The earliest of these, Cowardin et al. (1979), was developed as a tool to classify emergent wetlands. It has three general levels (and sublevels): system, class, and modifiers. As in Shaw and Fredine (1956), shallow-water habitats are distinguished under the subsystem level as either intertidal or subtidal, but there is no further distinction or definition based on depth. Dethier (1990, 1992) modified the Cowardin scheme for use in Washington State. She added an additional level, energy/exposure, to the hierarchy, permitting description of habitats as exposed, partially exposed, semi-protected, and protected. Marine habitats are additionally classed as high, moderate or low energy and estuarine habitats as open, partially enclosed, or channel/slough. Depth is incorporated as a habitat modifier for intertidal systems as backshore (areas above mean spring high water or MHWS) and eulittoral (areas below MHWS). Subtidal systems are described as shallow (extreme low spring water, ELWS, to less than 15 m) and deep (more than 15 m below ELWS). Allee et al. (2000) have further refined Cowardin et al. (1979) by restructuring the scheme and adding geographic, regional, geomorphic, topographic, and hydrodynamic levels. Of specific interest is the definition of offshore habitats by depth. Shallow offshore waters are defined as less than 200 m. Unfortunately, no distinction is made between habitats of inshore waters.

Jay et al. (2000) reviewed classification of estuaries and their habitats and concluded that a geomorphic approach would be more appropriate than the descriptive approach common to existing classifications. A hydrogeomorphic approach has previously been successfully employed for classification of wetlands (Brinson 1993). Whether or not habitats in the depth range of interest would be more effectively classified by such an approach is unclear at this time.

Shallow-Water Habitats. Shallow, unvegetated, estuarine habitats are presently defined primarily by their relative position within the gradient of tidal exposure (intertidal or subtidal), sediment type (e.g., mud or sand), and presence of biogenic structure (e.g., oyster shell). The depth range of interest is primarily subtidal, although it does represent a gradient of intermittent exposure at higher intertidal elevations. Beyond the terms intertidal and subtidal, there is no official designation of habitats by elevation (depth). Definition of habitats by sediment type also represents a gradient of conditions ranging from unconsolidated mud to gravel. In general, habitats are defined by dominant sediment type, i.e., sand, mud, muddy sand, etc. In Chesapeake Bay, nonvegetative biogenic structure is limited to oyster shell, whereas in other systems mussel and clam beds, worm reefs, and coral reefs may be present. Habitats can also be classified by their salinity: oligohaline, 0.5-5; mesohaline, 5-18; polyhaline 18-30; euhaline, above 30. Therefore, the major shallow, unvegetated, habitats encountered in Chesapeake Bay include oligohaline, mesohaline, polyhaline, or euhaline mud and sand flats, subtidal mud or sand bottoms of both tidal creeks and open bay waters, and oyster reefs.

Ecological Functions

Tidal flats, tidal creeks, and shallow subtidal bottoms. Appreciation of the ecological functions of shallow-water habitats has increased significantly during the past ten years. The ecological functions of tidal flats and shallow-water habitats include high primary production by benthic microalgae (e.g., diatoms), nutrient regeneration, decomposition of organic matter, secondary production by infauna (benthic invertebrates), feeding habitat and predation refuges for post-larval fishes and invertebrates, and feeding habitat for shore birds and wading birds (Peterson and Peterson 1979; Diaz et al. 1982a, 1982b; Diaz and Schaffner 1990; Short et al. 2000) (Figure 2).

Figure 2. The Blue crab (*Callinectes sapidus*), an important species of shallow water (Photo courtesy of Southeast Regional Taxonomic Center)

MacIntyre et al. (1996) reviewed the scientific literature on primary production of shallow-water benthic algae, finding that benthic microalgal biomass and production often equal that of the overlying water column, indicating that it plays an important role in system productivity. Pinckney and Zinmark (1993) have shown that microalgae of short-form *Spartina alterniflora* marsh and mudflats supply 45 percent and 22 percent, respectively, of total primary production at North Inlet, South Carolina. Shallow-subtidal (defined as less than 1 m) bottoms provide an additional 13 percent of production. Sand flats produce less (3 percent of total) than mudflats, as has also been reported by Barranguet et al. (1998). In addition to their contribution to primary production, microalgae are also important to nutrient cycling (Tyler et al. 2003) and to higher trophic levels (see review by Miller et al. (1996)). Middelberg et al. (2000) have shown that carbon derived from microalgae was incorporated into bacteria, meiofauna, and macrofauna in proportion to algal biomass, emphasizing its importance to estuarine food webs.

Benthic microalgae also are important in maintaining sediment stability. Underwood and Patterson (1993) demonstrated this attribute by purposefully eradicating diatoms from a patch of mud flat, thereby substantially increasing erosion in the treated patch. Similar conclusions have been reached by Austen et al. (1999) and Madsen et al. (1993) for intertidal and subtidal sediments, respectively. Meadows et al. (1994) reported that fungi provide a similar function in sandy intertidal sediments.

Infaunal communities of southeastern tidal creeks have been described by Lerberg et al. (2000) and Holland et al. (2004). Depths in these waterways ranged from a few centimeters to 1.5 m. Only three benthic species displayed distinct patterns of abundance within the creeks. Densities of the oligochaete *Monopylephorus rubroniverus* and the polychaete *Capitella capitata* were highest in the upper reaches of the creeks, while abundance of the polychaete *Heteromastus filiformis* was highest in the lower reaches. Infaunal community structure and habitat quality were degraded in watersheds with substantial commercial and urban development due to chemical pollution associated with runoff. Gillet et al. (2005) estimated secondary production of *M. rubroniverus* in two of the same waterways sampled by Lerberg et al. (2000) and Holland et al. (2004) and report that production was greater in upper than in lower reaches. Benthic algal production (as estimated from chlorophyll A concentrations) was also higher in upper than lower reaches.

Ewing and Dauer (1982) monitored benthic macroinvertebrate populations of shallow waters in the lower Chesapeake Bay and found that shoal and tidal creek stations tended to have fewer species, but three to five times more animals than deeper sites. West (1985) monitored tidal creek infauna of the Pamilco River estuary, North Carolina along a depth gradient ranging from 0.3 m to 1.5 m (1 to 5 ft) and also found that shallowest stations had the highest densities of infauna. Tourtellotte and Dauer (1983) reported that differences in benthic assemblages of Lynhaven Bay and surrounding areas sampled at depths between 2 m to 4.9 m (6 ft to 16 ft) were related principally to sediment type rather than depth. Chester et al. (1983) examined Newport River estuarine sites with depths of approximately 1 m and found that sediment type and salinity were the dominant factors determining abundances.

Shallow-water habitats are important nursery areas for post-larval and juvenile fish and shellfish. In Chesapeake Bay, young-of-the-year spot, silver perch, sea trout, and croaker dominate the nekton of waters 3.7 m or less (less than 12 ft) in spring and summer (Chao and Musick 1977). Weinstein and Brooks (1983) also found high densities of juvenile spot in tidal creeks in Chesapeake Bay. Flounder and blue crabs were also collected throughout shallow waters in that study. Both summer and southern flounder post-larvae have been shown to concentrate on Chesapeake tidal flats (Burke et al. 1991). In Cape Fear, North Carolina, tidal creeks and the fringes of marshes have high abundances of postlarval and juvenile spot, striped mullet, menhaden, and brown shrimp (Weinstein 1979). Allen and Barker (1990) describe subtidal creek bottoms at similar depths in South Carolina as important nursery habitats for larval spot, gobies, bay anchovy, and croaker. Burton et al. (1985) specifically recommend sampling for freshwater and oligohaline ichthyoplankton in shallow waters of Upper Chesapeake Bay. Hodson et al. (1981) and O'Neil and Weinstein (1987) have also documented the utilization of North Carolina and Chesapeake tidal creek habitats as feeding habitat for juvenile spot. Ryer (1987) and Orth and Montfrons (1987) examined utilization of tidal creeks and other shallow-water habitats in Chesapeake Bay as feeding habitat for blue crabs. Pihl et al. (1991) have also noted that fish and invertebrates migrate from the channel habitats into shallower, more oxygen-rich, habitats during hypoxic events in Chesapeake Bay. In contrast to widely held assumption that unvegetated habitats represent poorer nursery habitats than seagrass, Seitz et al. (2005) has recently shown that growth of juvenile blue crabs was greater in unvegetated mud and sand flats of the upper York River than the same habitats or seagrass beds in the lower river. In a companion study Lipcius et al. (2005) report that survival and overall abundance of juvenile blue crabs were also greatest in the unvegetated habitats of the upper river.

Clear evidence in support of the importance of shallow waters as a refuge from predation for both juveniles and small species of fish and decapods in Chesapeake Bay was provided by Ruiz et al. (1993). Sampling three depth zones, 1-35 cm, 36-70 cm, and 71-90 cm, they found that small species such as grass shrimp (*Palaemonetes pugio*), sand shrimp (*Crangon septemspinosa*), mummichog (*Fundulus heteroclitus*), striped killifish (*F. majalis*), the mud crab *Rithropanopeus harrisi*, four-spine stickleback (*Apeltes quadricus*), and naked goby (*Gobiosoma bosci*) were most abundant at depths less than 70 cm (2.2 ft). With the exception of the sand shrimp, small individuals (juveniles) of each species were also more abundant at depths less than 70 cm. Large species, especially predators such as blue crab (*Callinectes sapidus*), spot (*Leiostomus xanthurus*), and Atlantic croaker (*Micropogonias undulatus*) were generally most abundant at depths greater than 70 cm. Experiments with tethered shrimp, fish, and crabs showed that survival decreased with increased depth, suggesting higher rates of predation in deeper water. Additional support for the predation refuge hypothesis comes from the work of Paterson and Whitfield (2000). Sampling

South African salt marsh tidal creeks, they found that total fish abundance increased with depth in habitats ranging from less than 1 m to approximately 3 m. However, abundances of large predatory fish were disproportionately lower in depths less than 1m.

Shallow water also serves as an important feeding habitat for birds. Shorebirds and large wading birds obviously utilize the shallowest portions of the depths of interest while skimmers, kingfishers, terns, and gulls will feed in the deeper areas (Diaz et al. 1982b). The best feeding conditions will presumably be in the shallowest portions where prey will be more visible and concentrated. Scotts (1985) specifically identified mud flats, subtidal depths to 2 m (6 ft), and 2-m to 14-m depths as the most important for Chesapeake Bay waterfowl. In a review of the importance of shallow waters to waterbirds and shorebirds along the Mid-Atlantic coast, Erwin (1996) pointed out that intertidal flats are particularly important to migrant shorebirds and waters less than 2 m in depth are important to waterbirds. Fewer waterbirds are found in depths greater than 2 m.

Oyster reefs. Considerable advances have recently been made in understanding the ecological functions of oyster reef habitat (Coen et al. 1999, Coen and Luckenbach 2000) (Figure 3). Long valued as a commercial resource, large masses of oysters exert a significant influence on water quality, phytoplankton productivity, and nutrient cycling of estuaries (Dame 1996). For instance, Dame and Libes (1993) found decreased concentrations of ammonium, total nitrogen, total dissolved phosphorus, and total phosphorus in tidal creeks where oysters had been removed. Ecosystem modeling of Chesapeake Bay suggests that increased filtration by oysters could reduce eutrophication by lowering phytoplankton concentrations, which in turn would reduce the abundance of ctenophores and other gelatinous zooplankton (Ulanowicz and Tuttle 1992). Since gelatinous zooplankton are major predators of oyster larvae (Purcell et al. 1991), such an effect could be self-reinforcing.

Figure 3. Intertidal oyster bar (Photo source: USACE)

Oyster reefs also interact directly with local hydrodynamic conditions, affecting currents, flow conditions, and sedimentation patterns (Lenihan 1999). They provide habitat as well as a refuge from both predation and poor water quality conditions for fish and decapods. Breitburg (1999) has shown that oyster reefs provide habitat for a variety of fish species, including resident species such as gobies, blennies, and toadfish (*Opsanus tau*), facultative species including black sea bass (*Centrpristes striata*), and transients such as juvenile winter (*Pleuronecets americanus*) and summer (*Paralichthes dentatus*) flounder, striped bass (*Morone saxatilis*), spot, and silversides (*Menidia menidia*). Examining fish and decapod distributions in subtidal habitats of North Inlet, South Carolina, Lehnert and Allen (2002) found distinctive resident and transient fish communities associated with shell bottoms. They further suggest that high densities of juvenile sea bass (*Centropristes* spp.), groupers (*Mycteroperca* spp.) and snappers (*Lutjanus* spp.) indicate that shell bottoms represent essential fish habitat for these species. Glancy et al. (2003) have compared fish and decapod communities of shallow-water habitats in a North Florida estuary and concluded that decapod fauna of oyster shell habitats are distinct from that of either seagrass or marsh-edge habitats. Posey et al. (1999) have experimentally shown that grass shrimp actively seek out oyster shells when predators are present, obtaining some refuge from predation.

POTENTIAL IMPACTS OF DREDGING IN SHALLOW-WATER HABITATS: The general impacts of dredging and dredged material disposal have been reviewed by Sherk and Cronin (1970), Windom (1976), Morton (1977), Guillory (1982), Allen and Hardy (1980), Diaz et al. (1982b), and most recently by Newell et al. (1998). All of these identify removal of habitat, burial, turbidity, altered current patterns, salinity intrusion, and decreased flushing in relatively deep areas as the most common effects associated with dredging projects. Although none of these potential issues are unique to shallow waters, they do suggest which critical shallow-water habitat functions are most at risk.

First, the direct physical impact of dredging entails removal of sediment potentially resulting in alteration of ambient sediment, water quality, and hydrodynamic conditions. Depending on the spatial scale involved, changes in bottom topography can have profound effects on benthic infauna. Dernie et al. (2003) have shown that a difference of only 10 cm in the amount of material removed from a Welsh sand flat resulted in a substantial decrease in the rate of recovery. Plots where 20 cm of sediment were removed required 208 days for the infaunal community to be reestablished, whereas plots where only 10 cm was removed recovered in 64 days. Of particular concern is the potential for reduced primary production by benthic microalgae in the dredged area due to decreased ambient light at depth. As has been discussed above, benthic microalgae are a critical component of estuarine primary production and directly support much of the secondary production of benthic invertebrates and fishes that use shallow-water habitats. Likewise, increasing depth contours by dredging may not only reduce value of the habitat as a predation refuge for estuarine-dependent fish and shellfish, but may provide a conduit for predators to reach areas which were previously inaccessible. In each of the cases described (benthic, microalgal, and fisheries impacts), the actual impact of the dredging operation will be a function of the spatial extent and degree to which the post-construction environment differs from pre-construction conditions.

Second, benthic habitats differ in their characteristic rate of recovery after disturbance. Newell et al. (1998) point out that benthic assemblages in low salinity (e.g., oligohaline) habitats recover faster than those in high salinity habitats (e.g., euhaline) and assemblages in fine-grained sediments recover faster than those in coarse-grained sediments. Such generalizations are not absolute, however, as shown by Ferns et al. (2000). They report that infauna of muddy-sand flats were far more impacted and required longer to recover from passage of tractor-powered cockle harvesters than those of sandy intertidal flats.

Finally, the ultimate use of the dredged area may result in persistent, deleterious impacts to surrounding habitats. Dredging of shallow water is generally performed to provide access for small boats. Vessel traffic in shallow navigation channels has the potential to create turbulence, resuspending bottom sediments, increasing turbidity, physically disrupting bottom communities, and producing wakes (or waves) that erode surrounding habitats. Beachler and Hill (2003) report that boats moving at "near plane" speeds create the greatest impact. They provide a model relating boat size and water depth to resuspension that may be of use in evaluating potential problems. Likewise, Maynord (2005) has produced a model relating wave height to the speed and size of small boats, which may be useful for evaluating wave-induced effects.

Repeated physical disturbance has the potential to result in decreased productivity and increased susceptibility of affected communities (Odum 1985, Gray 1989). Marinelli and Woodin (2002) demonstrated experimentally that disturbing the surface of soft sediments alters sediment chemistry, making it less attractive to recruiting infauna. If such disturbances are routine, the resulting communities will most likely be less abundant and less diverse than those of undisturbed habitats. Bishop (2004) and Bishop and Chapman (2004) have detected such impacts in intertidal infauna exposed to boat wakes.

Microalgal production is also sensitive to disturbance as shown by Emerson (1989). He found that production of microalgal communities (and higher trophic levels as well) is negatively correlated with the degree of wind stress, which represents both physical disruption and increased turbidity. Primary production of benthic microalgae and vascular plants (submerged aquatic vegetation or SAV) is closely tied to light availability, and increased turbidity generated by boating activity could inhibit this process (Dennison et al. 1993, Pinckney and Zingmark 1993, Barranguet et al. 1998, Kemp et al. 2004). Gucinski (1982) has examined the effects of boat prop-induced turbidity on photosynthesis in submerged aquatic vegetation and found that small boats traveling at high speeds in water depths of 1 m can resuspend sufficient amounts of sediment to reduce seagrass productivity. Boat traffic may also directly impinge on benthic microalgae or SAVs when boaters inadvertently stray out of the channel, creating prop scars, or disrupt the bottom when anchoring their vessels. Such direct impacts are already of concern for SAV's (Asplund and Cook 1997, Blackhurst and Cole 2000). Since most SAV's in Chesapeake Bay occur in depths less than 3 m (Orth and Moore 1988, Dennison et al. 1993), increased boat access could increase the numbers and frequency of such impacts.

Chronic resuspension of sediment may also represent a threat to nearby oyster reefs. Adult oysters are adapted to living in a turbid environment and are therefore relatively insensitive to moderate increases in turbidity or suspended sediments (Shumway 1996). Settling oyster larvae, however, are reputed to be sensitive to even very thin layers (a few millimeters) of sedimentation.

Boating activities also have the potential to interfere with nesting birds and other wildlife (York 1994). Shorebirds utilize shallow water for both feeding and nesting habitat. For instance, Burger (2003) has reported how slowing recreational boat traffic in Barnegat Bay, New Jersey decreased the number of "upflights" (a startle response) by the common tern (*Sterna hirundo*), resulting in more time spent nesting and increased reproductive success.

CONCLUSIONS: Shallow-water estuarine habitats represent ecologically important and generally under-valued natural resources. Historically, problems in assessing their value resided in the fact that the definition of the term "shallow" is largely context-sensitive. For instance, what is shallow water near the outer reaches of an estuary may represent relatively deep water in the upper estuary. Recent attempts to resolve this semantic difficulty have sparked renewed interest in determining the ecological functions of shallow waters and how they change along depth gradients. To date, however, relatively little quantitative information is available in the depth range of greatest interest, 0-2 m MLW.

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