IN SITU CAPPING OF CONTAMINATED SEDIMENTS: COMPARING THE RELATIVE EFFECTIVENESS OF SAND VERSUS CLAY MINERAL-BASED SEDIMENT CAPS

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ABSTRACT

Ecological problems caused by sediment contamination occurring in deep water or wetland environments may be addressed through natural recovery, in-place containment or treatment, dredging and removal, or in some cases by *in situ* capping – which is defined as the placement of a subaqueous covering or cap of clean isolating material over an in-place deposit of contaminated sediment. While dredging and removal of contaminated sediments may be the most practical remedial method in many situations and sometimes necessary for navigational purposes, this remedial approach may not be the most environmentally protective and/or cost-effective approach. *In situ* capping approaches are often considered to be more protective of faunal and floral communities inhabiting impacted ecosystems than dredging alternatives, or when converting an impacted area to a closed cell. According to current regulatory philosophy and recommendations, the three primary functions of an *in situ* sediment cap include (1) physical isolation of the contaminated sediment from the benthic environment; (2) stabilization of contaminated sediments, preventing re-suspension and transport to other areas or sites; and (3) reduction of the flux (transport) of dissolved contaminants into the overlying water column. To date, most *in situ* capping projects appear to involve the use of primarily granular (i.e., sandy) capping materials. Although such capping materials may adequately serve to meet stated cap functions at many sites, their relatively high permeability and low organic matter and clay content may limit their ability to reduce contaminant transport into the overlying water column. Furthermore, non-cohesive, granular materials can also be prone to erosional losses and redistribution, thus minimizing their effectiveness in isolating and stabilizing contaminated sediments. Finally, the thickness required to meet performance goals, many times on the order of several feet, can have a deleterious effect on channel hydraulics and waterway uses. As an alternative to granular sediment caps, a new *in situ* capping technology, AquaBlok™, has been developed for use in either deep water or wetland ecosystems. AquaBlok™ is a clay mineral-based capping material that offers several functional advantages over granular capping materials including lower permeability, higher resistance to erosive forces, and considerably higher attenuation capacity for many types of contaminants. The likely need for thinner AquaBlok™ caps at many sites would also minimize navigational constraints. In this paper, we compare the potential relative effectiveness of a typical granular sediment cap to that of a typical AquaBlok™ sediment cap, as each would be installed into "typical" impacted deep water or wetland ecosystems.

Key words: *sediment, contamination, subaqueous, cap, AquaBlok™*

INTRODUCTION

Significant progress has been made over the last several decades in cleaning up polluted air, soil, and waters. With respect to surface-water pollution in particular, substantial reductions in non-point and point-source discharges have greatly reduced contaminant loading into many U.S. rivers, lakes, estuaries, and coastal oceanic areas, thus resulting in generally improved water quality conditions overall. Despite these relatively recent pollution-prevention efforts, sediments contaminated by a variety of toxic and enduring organic and metallic compounds – contamination which often results from decades-old releases – still remain beneath many of our nation's surface waters. Even with the primary "end-of-pipe" discharge sources eliminated, these polluted sediments can act as secondary sources of contamination, posing significant direct and indirect environmental risks through bioaccumulation in aquatic organisms and subsequent incorporation into aquatic and upland food webs. Episodic (e.g., storm-induced) physical redistribution of contaminated sediments in dynamic riverine or tidally influenced ecosystems can also literally disperse such environmental risks, impacting biological and water quality conditions far from the original sediment source.

Contaminant characterization and biotoxicity analysis of sediments collected from more than 21,000 sampling locations in river reaches of 1,372 targeted watersheds across the U.S. indicate concentrations of polychlorinated biphyenls (PCBs), polynuclear aromatic hydrocarbons (PAHs), pesticides, and/or heavy metals at levels which pose probable or potential risks to fish, wildlife, and humans (US EPA, 1997a; US EPA, 1998a). Such impacted sediments occur in a total of 96 watersheds nationwide, which comprises approximately 5 percent of all continental U.S. watersheds. Each of these impacted watersheds – in which further study of effects and sources of sediment contamination as well as possible risk reduction is deemed warranted (US EPA, 1997a) – is collectively referred to as an “area of probable concern,” or APC. In terms of volume of sediments impacted, it is estimated that approximately 1.2 billion cubic yards of significantly contaminated sediments occur throughout these particular APCs (US EPA, 1998a). Geographically, these impacted watersheds primarily occur throughout inland and coastal areas of the East and Midwest, although several also occur in Western states as well (US EPA, 1997a).

Many of the impacted watersheds (or APCs) occurring along coastal areas contain *estuarine* components. Estuaries are hydrologically dynamic and complex aquatic ecosystems occurring more or less where a river flows into either an ocean or into another large water body, such as one of the Great Lakes. In addition to abundant deep water habitat, many fresh water and salt water estuaries include significant wetland components, mainly marshes positioned adjacent to or around deep water areas. For clarification, the terms “deep water” and “wetland” are as defined by the Army Corps of Engineers (US ACE, 1987); in general, deep water areas are permanently inundated beneath at least 6.6 feet of water and/or do not support hydrophytic (water-loving) plants, whereas wetland areas are vegetated and relatively shallow. The term “marsh” can generally represent frequently or continually inundated wetlands characterized by the presence of herbaceous hydrophytic vegetation (Mitsch and Gosselink, 1993). Estuarine marshes and other coastal wetlands are critical ecosystem components in that they play important roles in providing habitat, and biological productivity, and controlling the flow of pollutants into estuaries and coastal areas in general (Herdendorf, 1992; Mitsch and Gosselink, 1993). Contaminated sediments are likely to occur in estuaries and associated wetland areas in at least some of these APCs.

Contaminated sediments occurring in deep water or wetland areas may be remediated (that is, isolated from biological receptors, thus lowering risk) through either removal and disposal or through treating, containing, or capping the contaminated sediments in place (*in situ*). A detailed technical discussion of various removal and *in situ* sediment remediation technologies, including associated economic, regulatory, and management considerations, can be found in the US EPA's Remediation Guidance Document (US EPA, 1994). Of the sediment remediation technologies typically considered, removal by dredging appears to be the most often chosen remedial approach (e.g., US EPA, 1998b). An additional, passive remedial approach also often considered (at least temporarily) for many projects is natural recovery. This generally involves isolation or removal of sediment contaminants from biological receptors through a combination of naturally occurring physical, chemical, and biological processes (US EPA, 1998c). Implicit to the natural-recovery concept is concurrent implementation of effective primary – and secondary – source-control measures, as well as the process of long-term burial and isolation of contaminated sediments by layers of clean sediment.

Sediment remediation by *in situ* capping can also be a relatively cost-effective and non-invasive approach to addressing contaminated sediments, and could be equally as effective as other more costly and environmentally invasive remedial approaches in meeting National Contingency Plan criteria for many sediment remediation projects. *In situ* capping does not involve costs for sediment removal, de-watering, treatment, and disposal – as does dredging. Furthermore, unlike dredging, sediment remediation through *in situ* capping does not involve substantial damage to, or the permanent removal of, benthic (invertebrate) habitat, or significant disturbance to the overlying water column. Depending on cap design and local benthic communities, placement of capping materials cannot only biologically isolate the underlying contaminated sediments, but can also provide clean substrate for re-colonization, assuming controls on primary and secondary contaminant sources. Finally, when compared to a relatively non-invasive remedial approach like *in situ* capping, dredging contaminated sediments from wetland ecosystems would typically involve destruction and removal of wetland plant communities, as well as removal of substrates supporting the vegetation and also perhaps related benthic communities as well; effectively restoring or replacing destroyed wetland ecosystems at a dredged site can be a technical challenge. In short, although the need for dredging and removal of extremely contaminated sediments is justified at many sites, capping can offer a viable alternative to sediment remediation in many impacted deep water and wetland ecosystems.

Fine-grained as well as granular (sandy) materials can be effective sediment caps (Brannon et al., 1985). Nevertheless, many remedial *in situ* capping projects conducted to date have apparently involved sand or sand-based caps (Palermo et al., 1998) primarily because of sand's availability, its relative ease of placement, its relative stability in sloped areas, its proven inhabitability by benthic organisms, and/or the ease with which sand can be differentiated from underlying, often finer-grained sediments during monitoring. Finer-grained material can provide better chemical barriers than sands because of its higher sorption capacity and also its relatively lower permeability (Palermo et al., 1998). However, the use of finer-grained materials in remedial cap design has been limited, apparently mainly because of logistical difficulties associated with effective placement of such material atop inundated substrates. Research has also shown that some invertebrate organisms can burrow more deeply into finer-grained sediments than into sands (Palermo et al., 1998), however, it is uncertain as to what degree this somewhat generalized observation has directly influenced the choice of granular over fine-grained materials in *in situ* capping projects; it should be noted that numerous physiological as well as various inherent and transient, substrate-related factors ultimately and collectively control burrowing depths (e.g., Palermo et al., 1998; Bosworth and Thibodeaux, 1990).

Regulatory guidance for conducting *in situ* remedial capping projects recently published by the US EPA, with assistance from the US ACE and academic researchers (Palermo et al., 1998), indicates that an *in situ* sediment cap should be effective in fulfilling three different functions: (1) physical isolation of contaminated sediments below the benthic environment; (2) stabilization of contaminated sediments, keeping them from being re-suspended and transported to other (e.g., down river) areas; and (3) reduction of transport (flux) of dissolved contaminants into the overlying water column. In this paper, we compare the relative effectiveness of two different remedial sediment caps – a sand cap and a clay mineral-based AquaBlok™ cap – in meeting the three capping functions within the context of "typical" deep water and estuarine wetland ecosystems.

DESCRIPTIONS OF AQUABLOK™, DEEP WATER, AND WETLAND ECOSYSTEMS, AND CAP DESIGNS AND PROPERTIES

Description of the AquaBlok™ Sediment-Capping Technology

AquaBlok™ is a patented technology based on a blend of clay minerals, polymers, and other additives surrounding a dense aggregate nucleus such as gravel (Figure 1). For typical product formulations, the clay component is often comprised largely of bentonite, although other clay-sized materials can be used in product preparation to address specific project needs and capping objectives. When applied in large mass to deep water or inundated wetland environments, AquaBlok™ particles settle through the water column and across the targeted sediment surface. Typically in less than two weeks, the applied layer of AquaBlok™ particles hydrates and expands, coalescing into a cohesive and low-permeability barrier cap between the contaminated sediments and the overlying deep water or wetland ecosystem (Figure 2).

The AquaBlok™ sediment-capping technology was originally developed for pilot-scale testing at a U.S. military firing range located in an estuarine salt marsh ecosystem on Eagle River Flats near Anchorage, Alaska. As a result of decades of ordinance testing, sediments in vegetated and ponded areas across the site were contaminated by white phosphorous (WP). High waterfowl mortality rates in the site area, which attracted the attention of the USDA, the US EPA, and other regulatory agencies, were directly linked to WP-contaminated sediments. Results of on-site mallard mortality studies conducted by USDA animal researchers indicated that AquaBlok™ was effective in isolating contaminated sediments to depths below which test mallards could dabble. Based on these study results, it is understood that AquaBlok™ is being considered as one of two *in situ* sediment remediation technologies for expanded use at this National Priority List site. Complete results of AquaBlok™ pilot testing at the Eagle River Flats site are summarized by the USDA in Racine and Cate (1995) and Cate and Racine (1996). A streamlined synopsis of pilot-study results can also be found in Hull et al., 1998a.

Preparations are also currently underway for pilot-scale testing of AquaBlok™ and AquaBlok™-based sediment caps in the Ottawa River, located in northwestern Ohio. This field-demonstration project generally involves installation and field-scale testing of three different capping designs along a particular section of the Ottawa River known to have elevated levels of PCBs and other contaminants in sediments. The three cap designs being tested include AquaBlok™ exclusively, AquaBlok underlain by a geogrid component, and AquaBlok plus geogrid plus a surficial stone armor layer. The primary goal of this project is to assess the relative effectiveness of AquaBlok™ and AquaBlok™-based sediment caps in physically stabilizing or isolating contaminated sediments occurring in a riverine environment representative of many Great Lakes tributaries. A secondary goal of the project, pursuant to a conditional Nationwide 38 permit issued by the USACE, is a study of the establishment of macroinvertebrate organisms above the encapsulated sediments, both as a function of cap design and over time. A detailed description of the Ottawa River site and its characteristics, including a presentation of laboratory settling-column and flume studies conducted as part of the project, can be found in Hull et al., 1998a.

General Characteristics of Typical Deep Water and Wetland Ecosystems Relevant to In Situ Capping

Site conditions, including the physical environment, hydrodynamic conditions, sediment characteristics, and waterway uses, will dictate the overall feasibility of remediating contaminated sediments by *in situ* capping (Palermo et al., 1998). In comparing the relative effectiveness of sand versus AquaBlok™ caps, differences in a targeted site's physical environment and hydrodynamic conditions are considered to be most relevant. Particular aspects of, and relative differences between, these conditions as may apply to "typical" deep water and wetland ecosystems are summarized in Table 1. For the purposes of this paper, a deep water ecosystem could be broadly represented by a major river, a large lake, or an ocean. A typical wetland ecosystem may, in turn, be represented by an emergent, fresh water or salt water estuarine marsh occurring at either the river-lake or the river-ocean interface. In that most sediment contaminants tend to be associated mainly with clay-sized particles and organic matter, it may be assumed that sediments in both environments are generally fine-grained and saturated, low-density materials prone to consolidation upon capping. And based on results of the US EPA's National Sediment Quality Survey (US EPA, 1997a), it can also be assumed that sediments contain significant concentrations of PCBs and also perhaps PAHs, pesticides, and/or metals.

Descriptions of Sand and AquaBlok™ Remedial Sediment Caps

The composition and thickness of components of a cap can be considered as the remedial cap design (Palermo et al., 1998). Cap designs should be developed after considering a number of design- and site-related factors including cap compatability with contaminated sediments, compatability amongst cap components, the potential for bioturbation (sediment mixing) by indigenous invertebrate organisms, the potential for cap erosion, and the potential for contaminant transport through the cap into the overlying water column.

The design of remedial sediment caps can be relatively simplistic – composed of monolayers of a single capping sediment or material – or they can be more complex, involving the inclusion of geotextiles and/or a variety of fine-grained or granular components or layers (Palermo et al., 1998). For the purposes of comparing the relative effectiveness of sand versus AquaBlok™ caps, we consider a simplistic cap design for each cap type, as summarized in Table 2. Also included in Table 2 are estimated values for several physical cap properties that have particular relevance in terms of cap effectiveness in minimizing contaminant transport. Methods by which sand and AquaBlok™ caps are installed into deep water and wetland ecosystems are not discussed in this paper; placement techniques used in cap construction and installation are discussed in detail in Palermo et al., 1998.

RELATIVE EFFECTIVENESS OF SAND VERSUS AQUABLOK™ SEDIMENT CAPS IN MEETING RECOMMENDED FUNCTIONS

As stated previously, an *in situ* remedial sediment cap should typically be designed to physically isolate contaminated sediments from the benthic environment, stabilize and immobilize the sediment mass, and reduce the transport of dissolved contaminants from sediments into the overlying water column (including transport into the bioturbation zone). Provided in this section are discussions of how effective the sand and AquaBlok™ cap designs (Table 2) should be at meeting each function in typical deep water and wetland ecosystems. Also included in some of these discussions are ways in which a given cap design might be modified to meet capping functions, if it appears that the cap design(s) may not do so.

Physical Isolation of Contaminated Sediments from the Benthic Environment

The benthic environment may be defined as the realm of subaqueous substrates (sediments) that can provide habitat for various epifaunal and infaunal aquatic organisms. Physical isolation of contaminated sediments from this environment and its invertebrate inhabitants will likely be a requirement of cap design if a remedial objective is to reduce risk associated with contaminant exposure to benthic invertebrates, bioaccumulation of contaminants, and/or potential movement of contaminants into the food web (Palermo et al., 1998). Risk would be reduced through designing and installing a cap that either discourages burrowing or that provides for an adequate "replacement" substrate, at a thickness exceeding typical depths of burrowing and bioturbation (sediment mixing) for indigenous invertebrate communities. Capping contaminated sediments with stone armor can discourage bioturbation into contaminated sediments and may also attract a relatively great diversity of some types of benthic organisms (Palermo et al., 1998). However, armoring would do little to offer a clean and viable replacement substrate for near-term colonization by infaunal organisms that depend on particular soft substrate for habitat.

This section focuses on the potential viability of sand and AquaBlok™ caps as habitat for colonization by macroinvertebrate organisms, which are typically greater than about 1 mm in size. Of particular concern is invertebrate colonization of (and selection for) certain substrates as well as depths to which infaunal organisms may bioturbate as a function of substrate type and conditions, the taxa involved, and the presence of sediment contamination. Other factors affecting bioturbation depths and benthic colonization (such as vegetation, current velocity, water depth, and temperature) will also be discussed briefly, to the extent that these factors will vary between typical deep water and wetland environments. Adequate published information is available regarding invertebrate colonization of various sandy materials – substrate that may be collectively considered similar to the sand cap presently discussed (Table 2). In contrast, little macroinvertebrate colonization data are currently available for AquaBlok™, although small red worms were reportedly observed to occur in capping material during the following growing season at the Eagle River Flats site. In lieu of AquaBlok™-specific benthic data, published information related to invertebrate colonization of relatively fine-grained (silt and/or clay rich) materials is considered. Such substrate may collectively be considered as broadly equivalent to AquaBlok™ in terms of general physical character and related properties (Table 2). Results of the upcoming Ottawa River macroinvertebrate study should provide important information related to invertebrate-AquaBlok™ compatability.

The type of substrate present is known to exert both direct and indirect controls on the taxa present, primarily through selecting for invertebrate organisms with particular feeding modes. Direct substrate controls on invertebrate taxa present are related largely to sediment burrowability and burrow stability, as well as potential interferences of substrates with feeding behavior. Stable burrows can be formed by subsurface feeders or insect larvae in cohesive fine-grained sediments, but gills and filtering systems of other benthic organisms may not tolerate the large amounts of suspended fines present (Hartnoll, 1983; Hynes, 1970). In contrast, sands lack abundant fines that could interfere with feeding processes, but burrowability and burrow stability in sandy material is also lower, thus limiting habitation by invertebrates that require a competent substrate for burrow establishment. Indirect substrate controls on taxa present are mainly related to the quantity and distribution of organic (food) material present as well as dissolved oxygen levels in interstitial pore waters, which typically decrease with depth even in clean sediments (Hartnoll, 1983); in general, fine-grained sediments typically contain higher organic content than sands, but also lower levels of dissolved oxygen (Hartnoll, 1983).

General substrate influences on macroinvertebrate community diversity and richness are also fairly well documented. Sand generally offers a relatively poor habitat with few specimens of few species of macrofauna while fine-grained (muddy) substrates may be very rich in biomass, but lower in diversity (Hynes, 1970). And as further illustration of direct and indirect substrate controls on benthic communities, Hynes (1970) as well as Hartnoll (1983) reference studies noting significant changes in macroinvertebrate diversity and/or populations with changes in substrate type at a given site, as may be brought about by storm-related deposition of fine-grained material overtop previously exposed stony or cobbled areas. *In situ* capping guidance stating that colonization of a sand or armored cap would be sparse until "new" sediments with sufficient organic matter are deposited on the cap (Palermo et al., 1998) may also illustrate substrate controls on macroinvertebrate colonization, at least in terms of food abundance.

Bioturbation depths of infaunal organisms, which is a critical aspect of benthic behavior from the standpoint of isolating organisms from contaminated sediments by capping, are collectively controlled by physiological factors (including burrowing abilities, body size, lifestyle) as well as environmental conditions (substrate grain size, bulk density, organic content, and pore-water geochemistry) (Bosworth and Thibodeaux, 1990; Palermo et al., 1998). Although substrate grain size is more-or-less inherent, other substrate-related parameters – particularly geochemical parameters like redox – are transient and can vary temporally as well as spatially, thus affecting bioturbation depths. Nevertheless, results of benthologic and sediment-capping studies generally indicate that most infaunal organisms – including amphipods, oligochaete (tubificid) worms, and mud-dwelling insects – typically occur within the upper 1 to 6 inches of sediments (Bosworth and Thibodeaux, 1990; Charbonneau and Hare, 1998; Hynes, 1970; Davis, 1974). In general, bioturbation by marine benthos is greater than that of freshwater benthos (Palermo et al., 1998). Some species of large bivalves, polychaete worms, and burrowing shrimp have the ability to burrow as deeply as almost 80 inches (Bosworth and Thibodeaux, 1990; Brannon et al., 1987; Palermo et al., 1998, Hynes, 1970). Few studies have apparently been conducted that relate bioturbation depths to particular substrate types and conditions while "holding" other potentially controlling variables (e.g. organic content, or the degree to which contaminated pore waters have migrated upwards) constant. However, several researchers have noted greater penetration depths into "muddy" sediments than into sandy sediments for some macroinfaunal taxa, including midge and worm species indigenous to the Great Lakes (Palermo et al., 1998; Hines and Comtois, 1985 [as referenced by Bosworth and Thibodeaux, 1990]).

Bioturbation depths are also known to be affected by the presence of sediment contaminants, not only directly through toxicity impacts, but also indirectly through further reductions in dissolved oxygen levels in interstitial pore waters (Bosworth and Thibodeaux, 1990; Hartnoll, 1983; Hynes, 1960 [as referenced in Hynes, 1970]). Bosworth and Thibodeaux (1990) found that organisms in chemically impacted (as well as physically disturbed) substrates are often limited to the upper inch or so, whereas, in less polluted or disturbed substrates, organisms tend to occur at greater depths. A situation somewhat unique to remedial capping projects which could similarly affect (limit) burrowing depths involves the upward advective seepage of contaminant-bearing pore waters into caps during cap-induced consolidation of sediments (Palermo et al., 1998; Zeman, 1994); this phenomenon typically occurs when soft and compressible sediments are encapsulated by relatively permeable capping materials like sand. In fact, the expectation of such upward advective migration of contaminated pore waters into a cap during sediment compaction – which effectively reduces the contaminant-free cap thickness available for low-risk bioturbation – requires consideration during cap design (Palermo *et al.,* 1998). Significant cap-induced consolidation of soft, fine-grained sediments has not been observed when using AquaBlok™ as the capping material (Hull et al., 1998b), perhaps because of AquaBlok™'s relatively low bulk density and very low permeability (Table 2). Consequently, consolidation-induced migration of pore waters into the AquaBlok™ cap should be insignificant, thus maximizing the contaminant-free thickness available for low-risk benthic colonization. An approximation of potential upward migration of a selected contaminant into AquaBlok™ caps through diffusive (rather than advective) processes, which could also have an impact on benthic bioturbation and cap colonization, is addressed later in this paper.

Little information appears to be available indicating that invertebrate colonization of and bioturbation depths within wetland sediments would be significantly different from that discussed above. Nevertheless, invertebrate organisms are known to be important food-web components in coastal wetlands (Mitsch and Gosselink, 1993), with many infaunal species, including oligochaetes and midge larve, also playing important roles in decompositional pathways in wetlands ecosystems as well (Krieger, 1992). Greater diversity and populations of invertebrate taxa typically occur in wetlands than in non-vegetated deep water areas, primarily due to more diverse habitat resulting from higher spatial and temporal variability in habitat-controlling factors (Hynes, 1970; Krieger, 1992; Mitsch and Gosselink, 1993). Therefore, with respect to remedial capping and benthic isolation, the same sand or AquaBlok™ caps applied to deep water ecosystems should be equally as effective in wetland ecosystems. Fundamental differences, such as cap characteristics and properties (Table 2), however, may need to be considered when evaluating potential cap impacts on wetland hydrology and vegetative establishment and growth. Furthermore, the intensity of bioturbation (over and above depth), may be greater in more biologically productive and populated wetland areas; the potential for this should be evaluated.

In summary, a one-foot thick AquaBlok™ cap should be at least as effective in isolating bioturbating invertebrate organisms from underlying contaminated sediments in deep water or wetland ecosystems as would a one-foot thick sand cap. Nevertheless, regulatory guidance (Palermo et al., 1998) recognizes the need for site-by-site characterization of indigenous benthic communities (including an estimation of their associated bioturbation depths) in light of the many environmental, substrate, and geographic controls on a site's benthos. Site-specific benthic data could ultimately indicate that such a cap thickness could, in fact, be decreased while still accommodating bioturbation depths of local organisms. However, site-specific information may instead imply that the cap thickness should be increased to maximize benthic protection, or that a geotextile layer be placed at the cap/sediment interface to minimize macroinvertebrate contact with contaminated sediments. Finally, in regards to habitat restoration, it would seem logical to use a capping material that is physically similar to the contaminated sediments being capped in order to best accommodate indigenous infaunal species, e.g. infaunal organisms requiring competent burrows. Because most sediment contaminants tend to be associated with finer-grained particles, it should follow that the capping material should be as physically similar to local (albeit contaminated) substrates as possible. In terms of general physical properties and characteristics, AquaBlok™ would probably more effectively meet this requirement at many impacted sites than would relatively non-cohesive, sandy materials.

In Situ Stabilization of Contaminated Sediments Through Capping

Minimizing the physical re-distribution of contaminated sediments within a dynamic aquatic ecosystem is important for environmental and economic reasons. First, highly contaminated sediments – such as those occurring nearest to a primary or secondary contaminant source – would remain geographically localized, thus spatially minimizing and concentrating impacted areas of highest biological risk. Secondly, a concentration of larger portions of total sediment contamination into smaller areas should provide for more cost-effective sediment remediation, regardless of the remedial technique(s) employed; large volumes of slightly to moderately impacted sediments dispersed over broad areas can pose significant technical and economic challenges to effective sediment remediation.

Stabilization of contaminated sediment through remedial capping involves preventing or limiting sediment re-suspension and subsequent transport and redistribution by current or wave action. In general, sediment re-suspension will only occur if erosive forces exerted on substrate (sediment) surfaces exceed those required to entrain non-cohesive or cohesive material into the water column (Dortch et al., 1990). The role that remedial caps can play in minimizing exposure of sediment to such erosive forces, and in enduring the impact of such forces themselves, is the principal focus of this section. In terms of cap functioning, it is assumed that, if a cap can remain in place over time, then the underlying sediments will also remain physically stabilized and in place. Adequately meeting this function would not necessarily, however, address the potential for eventual vertical sediment migration, or “winnowing” up through coarser-grained capping materials, which is a factor that may require additional consideration during cap design (Palermo et al., 1998).

The potential for erosion of capping materials upon exposure to current- or wave-related forces can be evaluated on theoretical as well as empirical bases. Predicting particle re-suspension or entrainment by turbulent forces associated with wave oscillation is more complicated, for various reasons, than modeling sediment movement under unidirectional current flow (Dortch et al., 1990; Tsai and Lick, 1986). Nevertheless, sediments generally differ in erodability depending on their non-cohesive or cohesive character. For non-cohesive materials like sand or gravel, smaller particles will typically begin to move (erode) before larger particles, mainly as a function of water velocity and turbulence as well as the drag and lift forces exerted (Dortch *et al.*, 1990). In contrast, the erosive behavior of finer-grained, cohesive materials is dictated by a variety of often interrelated factors including clay content and mineralogy, shear strength, water content, bulk density, and organic content (Jepsen et al., 1997; Kamphuis, 1990; US ACE, 1998); even the quantity and size of gas bubbles present in the sediment can affect erosion rates (Jepsen et al., 1997). Predicting the erodability of cohesive sediments is further complicated by the fact that erosion rates can vary with sediment depth, primarily through depth-related changes in shear strength, bulk density, consolidation, and moisture content (Dortsch et al., 1990; McNeil et al., 1996; Jepsen et al., 1997; US ACE, 1998); such depth-related changes are typically not characteristic of coarser-grained, non-cohesive sediments (McNeil et al., 1996). Additional research has shown that erodability of fine-grained material can also vary with pore-water chemistry as well as the exchangeable cations present (Otsubo and Muraoka, 1988; Raudkivi and Tan, 1984).

In general, empirical observations indicate that, under tidal and other unidirectional currents, loose fine sand erodes easiest and that increased resistance to erosion for silt- and clay-sized particles is attributable to the presence of cohesive and adhesive forces (Dortch et al., 1990), as discussed above. Such observations are generally confirmed by results of large-scale laboratory flume studies of AquaBlok™, sand, and gravel resistance to current flow, which are published in detail in Hull et al.(1998b). AquaBlok™ – a cohesive, clay mineral-based material – displays greater physical resistance to sustained fresh water flow than does poorly graded fine sand and poorly graded gravel. Relatively insignificant erosional loss of hydrated AquaBlok™ (losses on the order of less than one inch in thickness) have been observed after several days of continuous flow at velocities ranging from approximately 5 to 6 feet/sec (Hull et al., 1998b). In contrast, nearly complete erosion of fine-grained sand samples occurs under lower sustained flow velocities (approximately 1.5 to 2.7 feet/sec) exerted over a much smaller time frame (less than one hour). A more or less intermediate response to erosive forces is observed for gravel, in that approximately 20 percent loss of sample mass occurs under a flow of approximately 1.8 to 3.2 feet/sec over a one-hour time period (Hull et al., 1998b); the relative resistance to flow of larger-grained, non-cohesive particles like gravel or cobbles is the basis for placing armor layers atop fine-grained, non-cohesive capping materials in some cap designs (Palermo et al., 1998). Less data appear to be available related to resistance of cohesive and non-cohesive capping materials to other types of erosive forces including those associated with tides, ice and debris scouring, and engine prop wash. These topics could be addressed through further laboratory study, as could an evaluation of AquaBlok™ (versus sand) erodability under salt water or brackish conditions, or a determination of cap resistance to flowing waters containing substantial suspended material – a factor which can greatly increase erosion rates of even resistant cohesive materials (Kamphuis, 1990).

Differences in the erodability of AquaBlok™ versus sand caps in an estuarine marsh versus a deep water ecosystem would generally be dependent on the types of erosive forces involved, as well as the buffering effect exerted by wetland vegetation. Except in localized channeled areas, less current flow would be expected through an estuarine marsh, even during flood conditions, than along a major river; vegetation effects on slowing water movement and promoting sedimentation in general are well known (Mitsch and Gosselink, 1993; USFWS, 1984). In contrast, portions of estuarine or coastal marshes peripheral to open lakes or oceans would generally be more exposed to erosive tidal, wave, and ice-related forces than would riverine areas. Relative to sand, then, AquaBlok™ may offer a higher degree of sediment isolation and stabilization in deep water or wetland environments exposed to significant erosive forces related to current, wave, and/or tidal forces.

In terms of sediment re-suspension, it should also be noted here that the capping process itself – and not just the influence of flowing waters – can cause sediment re-suspension to occur (Reible et al., 1997; Palermo et al., 1998). The extent to which this occurs will generally depend on physical properties of the sediment as well as methods and rates of capping material application. Unpublished laboratory research indicates that placement of a relatively thin layer of sand atop sediments prior to AquaBlok™ application can greatly reduce sediment re-suspension during AquaBlok™ additions. Less sediment re-suspension may also occur with more gradual application of capping material. Silt curtains may also be used to minimize lateral spread of sediment re-suspension plumes from a project site, if some degree of re-suspension does occur during cap application. Another issue that deserves mention is potential interaction between *in situ* capping materials and dynamic subaqueous features like “concentrated benthic suspensions,” or CBSs. CBSs have been characterized by Toorman (1998) as highly turbid yet fluid, near-bed layers occurring atop fine-grained substrates; such near-bed layers may conceptually be considered equivalent to benthic boundary layers, which occur “on the water side” of the sediment-water interface just above the bioturbation zone (Wang et al., 1991). Research indicates that these mobile, near-bed viscous layers are a major mechanism of transport of fine-grained sediments in coastal zones and estuaries (Toorman, 1998). Thus, the need for determining the effectiveness of sand or AquaBlok™ caps in immobilizing and isolating such mobile, potentially contaminant-bearing sediment masses of inherently low-bearing capacity.

In summary, despite the fact that relatively low-energy environments are considered to be the most appropriate sites for *in situ* remedial capping projects, regulatory guidance suggests that caps be conservatively designed to accommodate periodic, high-flow events, e.g. 100-year floods (Palermo et al., 1998). Incorporation of such event-probability considerations into cap design is reasonable in that the greatest degree of sediment re-suspension and transport should be expected to occur during such infrequent but extreme events (e.g. Lick, 1992). Cohesive capping materials like AquaBlok™ have demonstrated ability in withstanding sustained, high-flow conditions. Based on laboratory flume studies (Hull et al., 1998b), a one-foot thick AquaBlok™ cap will remain intact and should be more than thick enough to stabilize underlying contaminated sediments over an extended period of time (and given numerous high-flow events). A one-foot sand cap may also be effective in stabilizing contaminated sediments at many sites (even without the presence of a cobbled armor layer), depending on the frequency, magnitude, and duration of high-flow conditions. However, sand's high erodability relative to AquaBlok™ may limit its long-term effectiveness, considering the cumulative impact of incremental erosive losses over successive high-flow events. When armoring is not included, an allowance for erosional losses is often made during the design of granular (sandy) sediment caps, usually through increased initial cap thickness to accommodate sacrificial sand losses (Palermo, 1991; Palermo et al., 1998). Because long-term erosional losses for AquaBlok™ should be relatively minimal at most sites, relatively thin caps (perhaps less than one-foot thick) could be installed to isolate and stabilize contaminated sediments – assuming that the deployed cap still provides adequate protection for benthic organisms (as discussed previously) and adequate reduction in contaminant transport into the overlying water column (as discussed in the following section). Additionally, when compared to the vertical dimensions of sand caps, which could be on the order of several feet in some cases (Palermo, 1991), relatively thin yet still effective AquaBlok™ caps should have minimal impacts on waterway uses and navigability or site hydrology/hydraulics.

Finally, the role that bioturbation can potentially play in altering substrate erodability should also be clearly recognized. In some situations, benthic colonization can increase substrate stability, e.g. through tube formation. In other cases, however, burrowing and bioturbation into substrates – including capping materials like sand or AquaBlok™ – may reduce a material's overall resistance to erosive forces through altering sediment-bed topography, or through greatly increasing near-surface water content and decreasing bulk density as well as shear strength (e.g. Bosworth and Thibodeaux, 1990). Bioturbation can also have a significant impact on the transfer of dissolved contaminants from sediments into the overlying water column, as discussed in the following section.

Reduction of Contaminant Transport from Sediments into the Water Column

In addition to physically separating invertebrate organisms from the bulk-contaminated sediment mass, *in situ* capping can also reduce biological risk by limiting the upward transport of dissolved contaminants into the bioturbation zone established within capping material. Limiting the rate and extent of contaminant transport into the bioturbation zone – as well as increasing the migration path along which the contaminants must travel – not only protects invertebrate communities, but also decreases contaminant transfer into the overlying water column, thereby minimizing impacts to surface-water quality. Once contaminants migrate into a cap's bioturbation zone, however, and perhaps regardless of cap type (with the possible exception of armored caps), burrowing and reworking of sediments greatly increases the rate of pore-water release and contaminant migration into the water column (Thoma et al., 1993; Bosworth and Thibodeaux, 1990).

Provided in this section are estimates for long-term contaminant migration through either a sand or AquaBlok™ cap after the underlying sediment, and also perhaps the cap itself, has physically settled (the issue of upward migration of contaminated pore waters resulting from cap-induced compaction and sediment de-watering was discussed previously). The advection-dispersion model published in Appendix B of Palermo et al. (1998) was used as a guide to simulate vertical contaminant (solute) migration through sand and AquaBlok™ caps. Assumed cap designs and related properties are summarized in Table 2. For the purpose of discussion, this review simulated PCB migration, as represented by the PCB congener Aroclor 1242. We focus on PCBs because they are pervasive contaminants in sediments and because little predictive data, e.g., partitioning coefficients, are available for other contaminants like heavy metals (Palermo et al., 1998). Included below are (1) general descriptions of the transport processes involved and which process(es) likely dominate under given conditions; (2) site-, cap-, and contaminant-related assumptions considered when conducting this modeling effort; and (3) results graphically summarizing simulated rates and extent of PCB transport through sand or AquaBlok™ caps.

Contaminant transport through any sediment cap can occur by advection, molecular diffusion, and mechanical dispersion (Palermo et al., 1998). Advection refers to the transport of contaminants as part of bulk pore-water flow, with flow dictated mainly by hydraulic gradient and effective porosity of the porous media. Molecular diffusion is a process involving movement of dissolved ions in response to concentration gradients. And mechanical dispersion, which can be described as "diffusion-like" mixing relative to the average pore-water velocity, occurs as a result of heterogeneity. Different mixing process(es) will dominate under different hydrologic conditions and cap properties, mainly as a function of the local hydraulic gradient, hydraulic conductivity of the capping material, and the degree of chemical interaction occurring between a dissolved contaminant and particle surfaces. Surface reactions, or sorption, of contaminants by organic carbon and/or clay surfaces in substrate can significantly control contaminant transport, regardless of which transport process(es) are dominant. In general, diffusion through a substrate (including capping material) will always occur to some degree, while advection is significant only if an upward hydraulic gradient is acting on the cap (Palermo et al., 1998) or if contaminant diffusion is relatively slow. The relative magnitude of mechanical dispersion can be quantitatively estimated through determination of the Peclet number. The Peclet number can be calculated using estimates for effective cap thickness, diffusion plus dispersion, and advective or seepage velocity, which is a function of local gradient and substrate conductivity (Palermo et al., 1998). This parameter may also be approximated using estimated mean grain-size diameter in combination with seepage velocity and diffusion/dispersion (Fetter, 1993).

A number of assumptions were made to conduct this modeling effort, in addition to the assumption that consolidation-induced advective flow of sediment pore water is not occurring into caps (discussed previously). Assumptions related to site conditions, cap and contaminant characteristics, and the dominant transport processes involved per capping scenario are summarized as follows:

- Caps are constructed in a freshwater setting, either in a deep water or an estuarine wetland ecosystem such as would occur in a Great Lakes tributary.
- Cap characteristics including thickness and related physical properties are summarized in Table 2, whereas estimates for parameters related to Aroclor 1242 are included in Table 3.
- An upward hydraulic gradient of 0.0005 (unitless) is assumed to be acting on the base of either the sand or AquaBlok™ cap.
- Calculation of Peclet numbers for each capping scenario given the above gradient and other conditions (Tables 2 and 3) indicates that dissolved contaminant movement through the sand cap is dictated by diffusion plus mechanical dispersion. In contrast, movement through the much less permeable AquaBlok™ cap is dictated more or less exclusively by diffusion; mechanical dispersion should be insignificant through AquaBlok™ in that advective flow is minimal.
- Concentrations of Aroclor 1242 in pore waters present at the sediment/cap interface (the model’s lower boundary) remain constant and equal to its estimated equilibrium solubility in fresh water (Table 3).
- Only upward migration of the contaminant through caps is considered, although lateral transport through each cap would also be occurring to some degree.
- No deposition of clean sediments is occurring over the top of either cap over time, nor is any capping material being lost through erosion over time.
- The potential for long-term contaminant biodegradation in caps is not considered.

Simulations of vertical transport of dissolved PCBs (as represented by Aroclor 1242) through either the sand or AquaBlok™ sediment cap over time – assuming a 0.5% organic carbon content uniformly throughout each cap – are graphically summarized in Figures 3 and 4. Simulation results indicate that, for both cap types, minimal to no contaminant movement would occur into zones of greatest cap bioturbation (upper 6 inches) within the first 100 years. Nevertheless, at 500 years, detectable levels of Aroclor 1242 may be present (with concentration decreasing upward) within the 3-to-6 inch depth range in the AquaBlok™ cap (Figure 3), but not within the sand cap (Figure 4). In terms of total contaminant mass transport, however, simulations also predict that, over the course of 2,000 years, a significantly larger quantity of dissolved Aroclor 1242 will migrate into the bioturbation zone of the sand cap than into the AquaBlok™ cap's bioturbation zone.

In that hydrophobic contaminants are known to have a high affinity for substrate organics, the quantity of organic carbon present can have a significant impact on contaminant movement through capping materials. Work by Thibodeaux and Bosworth (1990) indicates that "breakthrough times" for Aroclor 1242 through a similar, one-foot thick sand cap containing 0.2% organic content would be about 670 years, versus about 3,346 years at 1.0% organic content. For comparative purposes, we also simulated Aroclor 1242 movement through sand and AquaBlok™ caps assuming organic carbon levels of 0.2% (Figures 5 and 6). As expected, simulation results indicate that transport through both the sand and AquaBlok™ cap will be greater at this relatively lower organic content, although the effect of lowering organic carbon content was more significant for the sand cap. Quantitatively, results of sand-cap simulations at 0.2% organic carbon also indicated, as did earlier modeling by Thibodeaux and Bosworth (1990), that cap breakthrough (bioturbation issues aside) may occur between 500 and 1,000 years (Figure 6). This breakthrough estimate may be somewhat faster than Thibodeaux and Bosworths' estimated 670 years in that the revised simulation considered advective and dispersive mixing processes to be involved, rather than just molecular diffusion (Thibodeaux and Bosworth, 1990).

In summary, results of modeling efforts indicate that a one-foot sand cap and a one-foot AquaBlok™ cap may be equally effective in isolating bioturbating benthic organisms from migrating PCB contaminants over a 100-year time period, assuming organic carbon concentrations ranging from 0.2 to 0.5%. Over longer periods of time, however, the degree of contaminant migration into and through the two caps diverge, particularly at lower organic carbon levels. Significantly higher concentrations (and greater total masses) of dissolved PCB eventually enter into the sand cap's bioturbation zone than into that of the AquaBlok™ cap. Exposure of invertebrate organisms in the sand cap to higher concentrations of dissolved contaminants may ultimately pose a greater risk to them directly, or to food-web dynamics associated with a remedial sand cap. The conservative assumptions considered to conduct these simulations – including constant dissolved contaminant concentrations in pore waters at the cap/sediment interface as well as no contaminant biodegradation or deposition of clean sediments over time – should be kept in mind when interpreting these results, or when evaluating the apparent differences in cap performance. The potential for incorporation of additional organic matter into capping materials over time, particularly in vegetated wetland ecosystems, is also not considered; this could substantially decrease the transport of hydrophobic contaminants through either cap type. During its manufacture, AquaBlok™ would be particularly amenable to effective incorporation of specially engineered “organoclays,” which are relatively hydrophobic materials that can selectively enhance attenuation of organic contaminants like PCBs (e.g. Cadena, 1989); other materials like iron oxides can also be incorporated into AquaBlok™ to increase metal attenuation. Additional reactive organic material could also be incorporated into sand, although maintaining a homogeneous distribution of organics during the application process may present a challenge.

CONCLUSIONS

Sandy materials have been used for a number of *in situ* remedial capping projects and, when incorporated into appropriate cap designs, have proven to be effective in isolating sediment-borne contaminants from benthic organisms, physically stabilizing contaminated sediments, and/or reducing the transport of sediment-borne contaminants into the bioturbation zone and overlying water column. Sand-based remedial caps and sandy substrates in general have also been shown to be viable substrate for invertebrate colonization. Research also indicates that macroinvertebrates may tend to burrow less deeply into relatively organic-poor sands, which could minimize potential breaching of sand cap/sediment interfaces; however, many environmental and physiological factors collectively control bioturbation depths, in addition to substrate grain size and food abundance and distribution. Finally, sediment de-watering associated with compaction during sand application can effectively increase the shear strength and bearing capacity of the sediments being capped (Palermo et al., 1998); less de-watering and subsequent geotechnical stabilization of sediments may occur during capping with less permeable AquaBlok™, potentially requiring additional considerations during cap design, e.g., inclusion of a stabilizing geotextile at the cap/sediment interface; while this may require additional cost, reducing the movement of contaminated pore waters into caps may be a positive attribute in some cases.

Despite sand’s attributes as a sediment capping material, results of this comparative study also indicate that AquaBlok™ could offer several advantages over sand in capping contaminated deep water or wetland sediments in the following circumstances:

- As opposed to more permeable sand material, AquaBlok™ application does not appear to result in significant compaction-related movement of sediment pore waters into capping material, thus maximizing the effective, contaminant-free thickness of an AquaBlok™ cap.
- AquaBlok™ displays significantly higher resistance to unidirectional current flow than sand, which could give more flexibility in cap design (perhaps no armor needed) as well as the range of hydrologic environments into which AquaBlok™ caps could be applied.
- By virtue of its lower permeability and amenability to organic additions, AquaBlok™ should act

as a more effective barrier to *long-term* contaminant transport of dissolved, sediment-borne contaminants into the bioturbation zone.

- AquaBlok™ is physically similar to fine-grained contaminated sediments and could therefore be a more effective substrate than sands for colonization by local invertebrate communities.
- By virtue of its higher resistance to erosive forces and effectiveness as a chemical barrier, a relatively thin AquaBlok™ cap (one foot or less) could be deployed to collectively meet all functional objectives at a given site. Such a relatively thin, yet effective cap could minimize restrictions on waterway uses and navigation – as opposed to sand caps, which may need to be applied at thicknesses significantly greater than one foot in order to meet functional objectives.

In summary, a one-foot AquaBlok™ cap would appear to be at least as effective as a one-foot sand cap in biologically, physically, and chemically isolating sediment-borne contaminants in deep water and wetland ecosystems. Both capping materials can be viable substrate for macroinvertebrate colonization. Both capping materials can physically stabilize contaminated sediments, either with or without additional capping components (e.g. stone armor). Finally, transport simulations indicate that both sand and AquaBlok™ caps can effectively limit upward migration of hydrophobic contaminants into bioturbation zones for a period of many decades. Both cap types should also be relatively easy to deploy, monitor, and maintain over time. Cost comparisons for sand versus AquaBlok™ sediment capping can be readily determined on a site-specific basis. Costs for implementing an *in situ* capping approach can be significantly less than costs associated with sediment removal, treatment, and disposal in many applications. The appropriateness of using either sand or AquaBlok™ to cap contaminated sediments – as well as implementing the *in situ* remedial capping approach in general – should be assessed on a site-by-site basis through careful consideration of site and sediment conditions, potential disruption of existing ecosystems, navigational or other waterway use requirements, economic issues, and – perhaps most importantly – risk-based project goals.

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Table 1. Site conditions and attributes of "typical" deep water and wetland ecosystems.

SITE CONDITIONS	COMPONENT	TYPICAL DEEP WATER ECOSYSTEM		TYPICAL WETLAND ECOSYSTEM (Estuarine Marsh)
		River	Lake or Ocean	
Physical Environment	Spatial dimensions of surface water	Variable	Variable	Variable; also temporal variability
	Surface water depths	Variable, but usually > 6.6 ft.	Variable, but usually > 6.6 ft.	Typically shallow, < 6.6 feet; probable temoral variability
	Tidal and/or wave influences	Low	High	Could be high near deep water boundaries
	Ice formation and influences	Variable, depending on location	Variable, depending on location	Could be high, depending on location
	Hydrophytic vegetation	Absent	Absent	Present
	Invertebrate community	Present	Present	Present
Hydrodynamic Conditions	Current flow velocities	Can be high	Can be high	Low
	Tidal and wave energy	Low	High	Could be high, depending on location
	Potential for periodically high (storm) flow	High	Low	Low
	Groundwater influences	Variable	Variable	Variable

Table 2. General designs and estimated physical properties for sand and Aquablock™ sediment caps.

GENERAL CAP DESIGN AND ESTIMATED PHYSICAL PROPERTIES	SAND CAP	AQUABLOCK™ CAP
Cap Design	A single, 1.0-foot thick layer of poorly graded, medium-grained sand containing some fined-grained sand and silt[1].	A single, 1.0-foot thick layer of AquaBlock™ of typical formulation.
Mean grain diameter (microns)	250[2]	14[6]
Total porosity (cm/cm)	0.27[3]	0.59[7]
Effective porosity (cm/cm)	0.25[1]	0.30[8]
Wet bulk density (g/cm^3)	2.0[4]	1.30[9]
Hydraulic conductivity (cm/sec)	1.0×10^{-3} [5]	5.0×10^{-9} [9]
Organic carbon content, or f_{oc} (g/g)	0.005[4]	0.005[9]

[1] from Thibodeaux and Botsworth (1990).
[2] particle size for fine- to medium-grained sand, per USDA classification.
[3] estimated from relationship between effective and total porosity for loamy sand per Rawls et al., 1982.
[4] from Palermo et al., 1998.
[5] estimated from Heath (1984) for fine sandy material (as referenced in Ohio EPA, 1995).
[6] from Otsubo and Muraoka, 1988 (considers only clay fraction of AquaBlok™).
[7] calculated from particle density and compacted dry bulk density values obtained from commercial bentonite source (considers only clay froaction of AquaBlock™).
[8] conservatively assumed to be half of total porosity, though likely equal to a smaller percentage of total porosity, based on very low hydraulic conductivity value.
[9] average value, determined in laboratory (f_{oc} for clay fraction of AquaBlok™).

Table 3. Parameter estimates for PCB congener, Aroclor 1242.

PARAMETER	Aroclor 1242
Solubility in fresh water (µg/1)	450[1]
Organic carbon partition coeff., K_{OC} (L/kg)	198,000[2]
Diffusivity in water, D_W (cm^2/sec)	4.5×10^{-6} [2]

[1] from US EPA, 1997b
[2] from Palermo et al., 1998

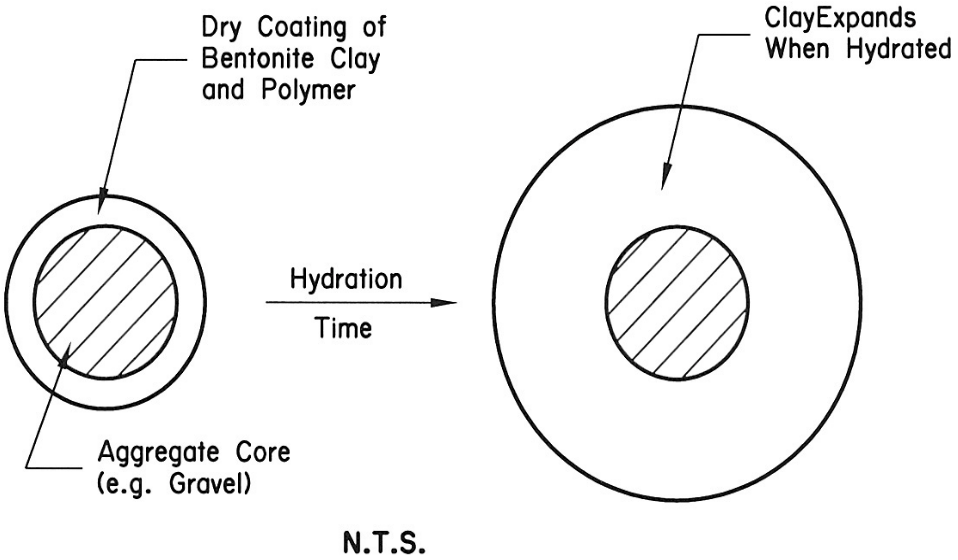

Figure 1. Configuration of a typical particle of AquaBlok™.

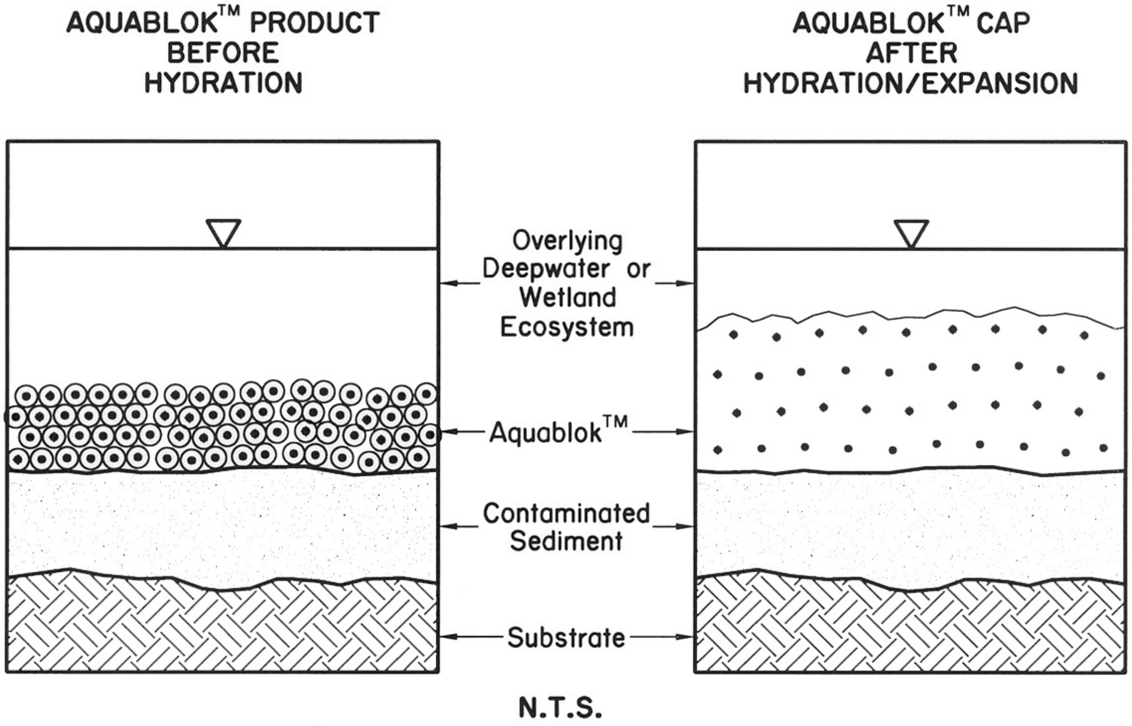

Figure 2. AquaBlok™ deployment in deepwater or wetland ecosystem.

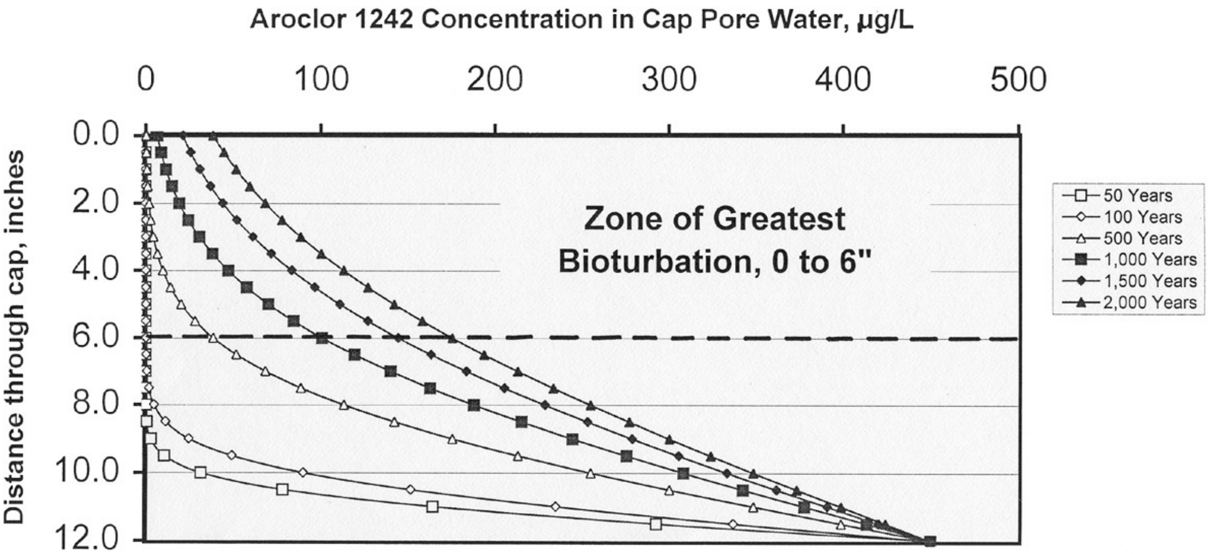

Figure 3. Simulated migration of Aroclor 1242 through one-foot AquaBlok cap (0.5% OC).

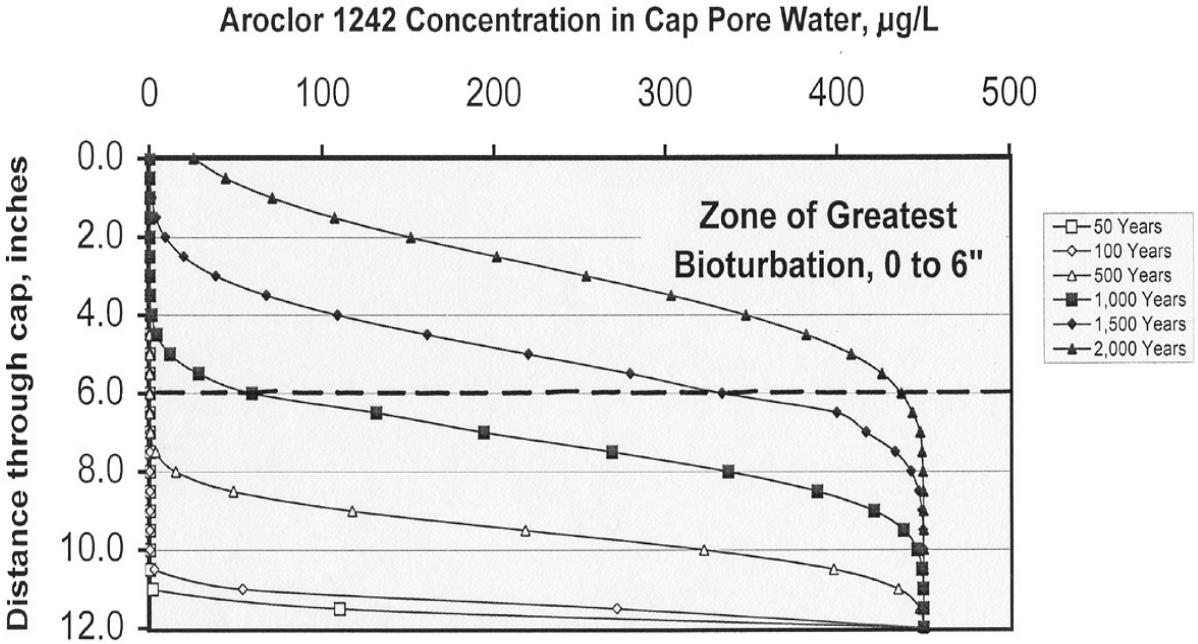

Figure 4. Simulated migration of Aroclor 1242 trhough one-foot sand cap (0.5% OC).

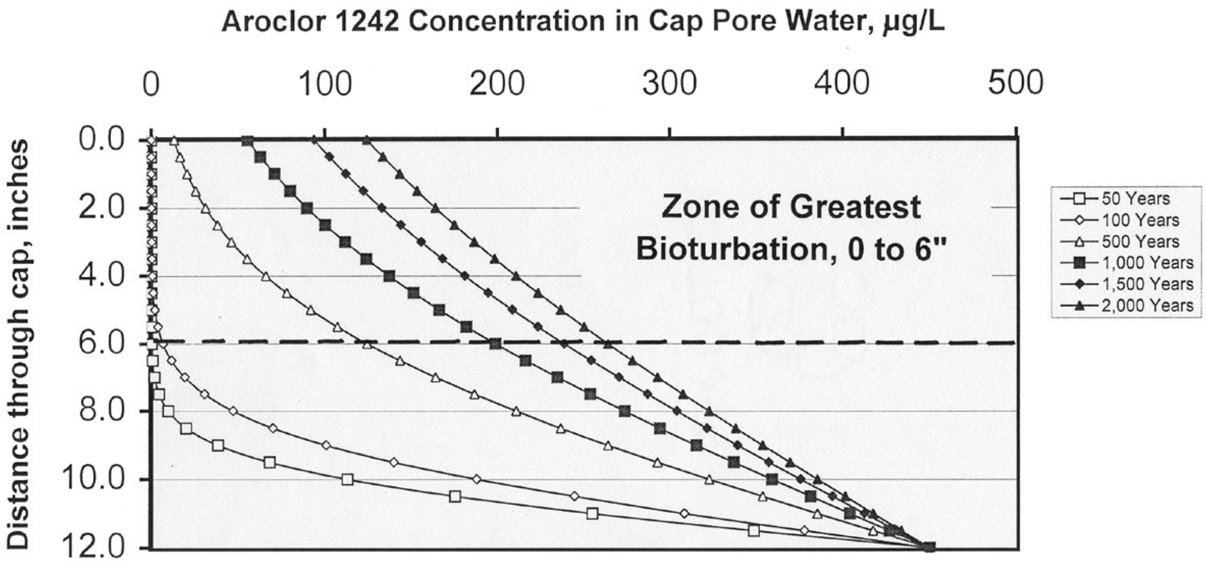

Figure 5. Simulated migration of Aroclor 1242 through on-foot AquaBlok cap (0.2% OC).

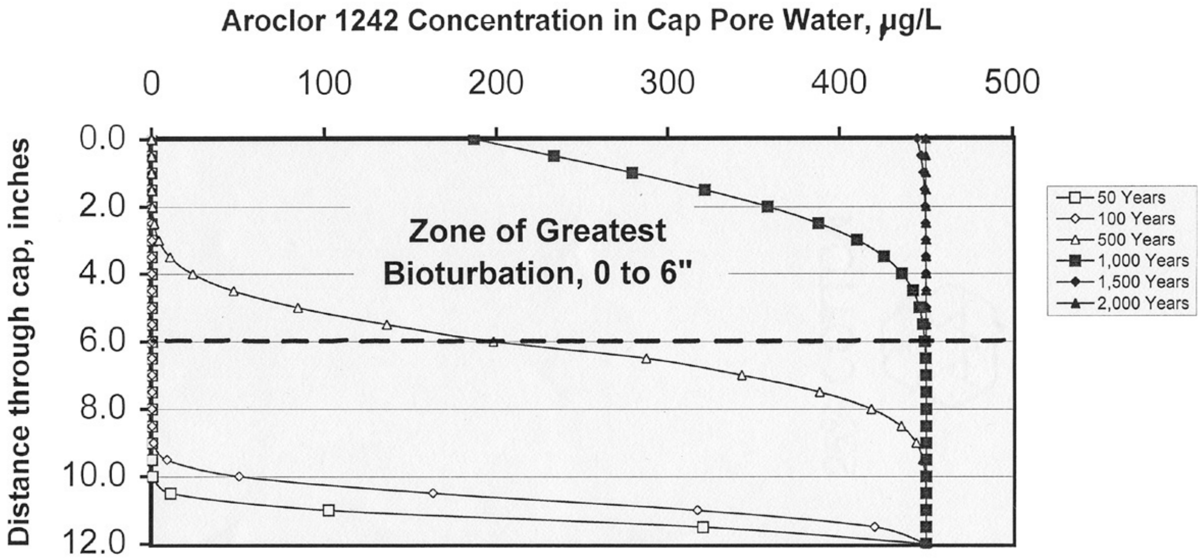

Figure 6. Simulated migration of Aroclor 1242 trhough one-foot sand cap (0.2% OC).