Part 2: Literature Review Summary
Integrated Guidance Model

Literature regarding the following assumptions of the integrated guidance model was reviewed:

1. Upland land use
   - model classifications: natural, agricultural, and developed
   - assumes impacts to water quality

2. Riparian landuse
   - model classifications: natural, agricultural, developed, industrial/paved, forested buffer
   - assumes impacts to water quality and habitat

3. Bank cover
   - model classifications: bare (<25%), partial cover (25-75%), total cover (>75%)
   - assumes impacts to water quality

4. Bank stability
   - model classifications: stable, unstable, and undercut
   - assumes impacts to water quality

5. Shoreline resources
   - model classifications: marsh, Phragmites, dune, beach
   - assumes impacts to water quality and habitat

6. Shoreline structures
   - model classifications: jetties, breakwaters, bulkheads, riprap, miscellaneous,
   - assumes impacts to water quality and habitat shoreline structures
   - model classifications: marinas, docks, boat ramps
   - assumes impacts to water quality and habitat

7. Subaqueous resources
   - model classifications: submerged aquatic vegetation, oysters, aquaculture
   - assumes impacts to water quality and habitat

8. Fetch and bathymetry

1. Upland Landuse

Land Use – Water Quality

General Impacts

Watershed development can have far-reaching impacts on water quality and hydrology that may impact aquatic communities downstream from the actual site of disturbance. Water quality factors, particularly nitrogen and phosphorus loadings, tend to be directly linked to human populations through increased nutrient production/availability and increased flow rates--two key factors in calculating nutrient loading to aquatic bodies (Smith et al. 2003). Changes in water quality (such as increased nutrient and sediment loads) due to development impact benthic invertebrate communities (Lerberg et al. 2000, Gage et al. 2004, Bilkovic et al. 2006) and eutrophication associated with upland land use increases benthic microphyte growth (Lever and Valiela 2005) and changes SAV faunal communities, with an increase in detritovores and a decrease in herbivores along a gradient of increasing eutrophication (Cardoso et al. 2004). Long-term eutrophication may lead to a loss of the SAV community. Other factors related to development (such as habitat fragmentation, increased human activity, and pollution) can impact fish populations (Scheuerell and Schindler, 2004), oyster growth (Bayen et al. 2007) and decrease marshbird (DeLuca et al. 2004) and riparian bird (Hennings and Edge 2002, Smith and Wachob, 2006) community integrity. Changes in impervious area alter stream hydrology, which is related to changes in fish community structure and fish abundance (Roy et al. 2005).

Residential

Impacts associated with residential (or suburban) development include non-point source pollution associated with diffuse runoff, which can be higher in residential than urban areas due to higher per capita impervious area (Atasoy et al. 2006, Bosch et al. 2003). Nearshore development tends to result in a loss of woody debris, emergent and floating vegetation in adjacent water bodies (Jennings et al. 2003), which can impact aquatic communities through habitat loss.

Urban

The installation of roads begins an accumulation of impacts to aquatic communities that culminates with urbanization of the watershed (Angermeier et al. 2004). Industrial and paved areas (parking lots and roads) contribute polycyclic aromatic hydrocarbons to adjacent wetlands (Kimbrough and Dickhut 2006). “Urban stream syndrome” describes a relationship between urbanization and increased nutrients and containments, increased hydrologic flashiness and altered biotic assemblages (Meyer et al. 2005, Nelson et al. 2006, Roy et al. 2005, Walsh et al. 2005). Elevated nutrient concentrations may be either due to increased non-point source inputs or reduced rates of nutrient removal in urban streams (Meyer et al. 2005).

Agriculture

Intense fertilization of upland lands, lack of groundcover and animal husbandry all contribute to aquatic impacts in agricultural areas. Nutrient and sediment concentrations increase as streams move through agricultural landscapes, particularly where animal agriculture occurs (Dukes and Evans 2006, Simon et al. 2005). Drainage of agricultural lands alters the hydroperiod of nearby wetlands, impacting amphibian growth rate and density (Gray and Smith 2005). The use of Best Management Practices may ameliorate some of the impacts of agriculture on aquatic communities (Nerbonne and Vondracek 2001).

Forested

Forested sites are generally considered to be the default landscape setting, to which other settings are compared. Vegetation slows runoff, filters groundwater and reduces hydrologic flashiness. The use of nutrients in groundwater and sediment stabilization reduces nitrogen and phosphorus loads to adjacent streams. Trees provide shade, temperature regulation and woody habitat to aquatic, terrestrial and avian species. Even moderate reductions in forest cover are associated with increases in suspended and dissolved solids, nitrate, turbidity and temperature (Price and Leigh 2006).

Threshold

There is a well-established link between increased development and increased aquatic impacts with stream quality degradation with impervious surface at as little as 10% of the catchment (Paul and Meyer, 2001). These results help to clarify the relationship between development and aquatic function, but can be problematic from a management viewpoint since they do not identify how much development is too much. Ecological thresholds mark breakpoints at which a system or community notably responds (perhaps irreversibly) to a disturbance. Threshold studies (Wang et al. 1997, Limburg and Schmidt 1990, Paul and Meyer 2001, DeLuca et al. 2004, Brooks et al. 2006, King et al. 2005, Bilkovic et al. 2006, Lussier et al. 2006) suggest that the relationship between development and ecological function is not a gradual, linear relationship and that alarmingly low levels of development (between 10-25%) can effectively render a system non-functional. The ecological thresholds identified in these studies could be critical for effective planning and management because they offer a definitive endpoint of development to manage towards.

Summary Table of Comparisons Between Forested, Agricultural and Urbanized Watersheds

	Forest	Agriculture	Urban/Developed	Sources
Invertebrate community (IBI scores)	High High*	Moderate High*	Low Low*	Bilkovic et al. 2005; Gage et al. 2004; Kratzer et al. 2006; *Synder et al. 2003
Turbidity/ Sedimentation	Low	Moderate	High	Burcher and Benfield 2006; Fisher et al. 2006; Hagen et al. 2006
Nutrient inputs/ concentration	Low	Moderate	High	Dougherty et al. 2006; Fisher et al. 2006; Hagen et al. 2006
SAV	High	Moderate	Low	Fisher et al. 2006
Hypoxia	Low	Moderate	High	Fisher et al. 2006; Hagen et al. 2006; Rodriguez et al. 2007
Species invasions		Low	High	Carpenter et al. 2007
Fish species richness		Low	High	Burcher and Benfield 2006
Aquatic habitat	High	Moderate	Low	Carpenter et al. 2007
Fine sediment	Low	High	High	Opperman et al. 2005
Temperature	Low	Moderate	High	Hagen et al. 2006
Heavy metal		Arsenic (in Cape Cod)	Silver, Cadmium, Mercury	Rodriguez et al. 2007

The literature review supports the model's assumption that upland land use can impact water quality. The relative values assigned to each of the three land use types (Natural =1, Agriculture=2, Developed=3) appear to reflect the general trend of water quality impacts summarized in the table.

2. Riparian Landuse

Riparian Land Use - Water Quality

Riparian lands in a natural state are likely to have gradual sheetflow and infiltration, deep soil carbon for denitrification and random spatial patterns of microdiversity and denitrification "hot spots" (Mayer et al 2006, Correll 1996, Klaproth & Johnson 2000). Hanson et al. 1994 found higher concentrations of NO3 in the groundwater of a developed stream buffer compared to a natural buffer across the same stream.

None of the riparian buffer studies found in this review compared the pollutant loading from different riparian land uses. Several authors concluded that absent or limited vegetative cover would increase flow rates creating the potential for channelized flows and complicating treatment options. Effective nutrient reduction & sediment removal in riparian buffers through infiltration depends more on ground and surface water hydrology and soil biogeochemistry than vegetative cover type or buffer width (Mayer et al 2006, Mankin et al, 2007).

Correll (1996) found only small increases in suspended sediments and nutrient input where a narrow forested buffer was maintained adjacent to a clear cut. Mayer et al (2006) reported that while buffers greater than 50 m are the most effective, narrower buffers (5-6 m) might still reduce subsurface nitrate by up to 80%. Several papers also referred to the value of rainfall interception, stream temperature regulation and bank stabilization provided by this type of narrow buffer.

The literature supports the assumption that a natural riparian land use should score higher than either agricultural or developed land uses. The findings in the literature were inconclusive regarding the equal score assigned by the model for developed and agriculture riparian land uses, although they are different for upland land use in general. The model clarifier for industrial/paved riparian land use assumes that intensely developed riparian buffers will not only prevent infiltration and below ground nutrient removal processes, but are likely to contribute additional nonpoint source pollution. The increase in model score for the presence of a forested buffer where the riparian land use is <30% natural is supported by the literature.

Riparian Land Use - Habitat

Several studies supported the assumption that natural riparian buffers have higher biological integrity for both terrestrial and aquatic habitats than either agricultural or developed riparian land uses, particularly mature forested buffers (Henry et al 1999, Mahan & O'Connell 2005, Teels et al 2006). Van Holt et al (2006) also found that any reduction in forested land cover in favor of an increase in agriculture, residential, or wetland land cover had a negative influence on the percent of sensitive aquatic species in New York streams.

One study indicated that the existing agricultural stream condition and the IBI responded negatively to urban land-use patterns at both restored and reference sites, particularly in a rapidly developing watershed (Teels et al 2006).

Studies on riparian lands as a primary determinant of habitat in adjacent waterways have mixed results. Diamond et al (2002) suggested that urban and agricultural land uses within a specified riparian corridor were more related to mussel species richness and fish IBI in Virginia mountain streams than land uses at a larger scale, but these latter analyses were limited. However, other studies suggested that riparian land use patterns were not as influential on the biological integrity of streams as land use at a watershed scale (Snyder et al 2003). Van Holt et al (2006) found no significant differences between the local riparian and landscape scales.

Palone & Todd (1997) report that a large number of aquatic organisms depend on the large woody debris from riparian buffers within 60 feet of the stream and that the presence or absence of riparian trees might be the single most important factor altered by humans that affects stream macroinvertebrates. They also conclude that a buffer width of 50-100 feet is adequate to improve aquatic habitat conditions and provide habitat for many terrestrial animals, with the exception of neotropical migratory birds that require wide buffers for quality breeding habitat. Other studies of bird communities in intensive agriculture areas suggest that riparian areas are very important habitats and that even narrow forest buffers support songbirds compared to herbaceous riparian vegetation (Klaproth & Johnson 2000). Yet these same authors also suggest that bird predators, brown-headed cowbirds, raccoons, domestic animals & exotic plants frequently occupy these buffers when surrounded by commercial, residential & industrial development.

There is no clear consensus on the corridor function of riparian buffers, particularly from a multi-species ecosystem level. One paper reported that the popularity of this concept was developed in the absence of supporting scientific evidence and another stated it appears to be species-dependent (Palone & Todd 1997, Klaproth & Johnson 2000).

There is some evidence to support the model assumption that agricultural land use should score higher than developed land use. There was no evidence to support the model assumption that if >50% of the assessment unit is agricultural and/or developed that the habitat value of the riparian buffer is "critically compromised". There are contrary findings related to land use scale and the influence of riparian land uses on the biological integrity of streams. The modifier score for the presence of a forested buffer adjacent to either agricultural or developed riparian land uses may be a valid assumption.

3. Bank cover (water quality only)

Shoreline erosion at the land-water interface is a natural process and source of nutrient and sediment loading. Both herbaceous (grass) and forested buffers effectively stabilize banks in the coastal plain of Virginia (Klapproth et al 2000, Palone et al 1997). On tidal shorelines in the Chesapeake Bay region, the volume of erosion tends to be highest where the soil is unconsolidated and barren of vegetation (Hardaway et al 1992, CBP 2005).

The ability of vegetation to stabilize banks and prevent erosion is dependent upon plant vigor, density and rooting depth (Ott 2000). Vegetation stabilizes banks by increasing shear strength of the soil, intercepting rainfall, and reducing water velocity (Ott 2000). Roots increase the strength of bank soils by physically binding the soil in place and by increasing soil cohesion (Dierks 2007, Wynn et al 2004). Root density at the bank toe where hydraulic shear is highest contributed to bank stability in another study (Wynn et al 2004). Easson and others (2002) found that when no root reinforcement existed on riverbanks, the slope failed marginally.

Structural erosion control methods such as revetments and bulkheads will also reduce the nitrogen, phosphorus, and suspended sediment load if designed properly (Hardaway et al 1992, Palace et al 1998). Wave reflection off hard structures also increases nearshore erosion of fine-grained sediments (Langland and Cronin 2003). Structures for reducing erosion tend to displace the bank vegetation by design, at least temporarily for installation. If the original vegetation cover continues to be suppressed, then stabilizing and nutrient removal processes provided by bank vegetation are also reduced. A constant state of biological activity above and below ground contributes to plant vigor and efficient nutrient removal (Dierks 2007).

However, the presence or absence of bank cover is not the sole predictor of erosion potential (see Bank Stability). Freeze-thaw cycles, slope hydrology and stream hydraulics also contribute to bank instability and erosion, particularly for bluffs (NRC 2007). Vegetation bank cover is also ephemeral and subject to change in response to bank failure, storms and anthropogenic modifications.

The model assumption that vegetation or structural bank cover predicts erosion potential is partially supported by the research. The presence/absence of cover is only one determining factor of erosion potential. The descending value for total, partial and bare cover is logical, but there are no research findings to confirm noticeable differences with various coverage levels. The modifier for structural instead of vegetative cover (total or partial) has not been verified with comparative studies, but may be valid because structures reduce biological contributions to water quality.

4. Bank Stability (water quality only)

Bank stability refers to the potential for bank erosion and bank failure, or the physical collapse of all or part of the bank as a result of geotechnical instabilities (Wynn et al 2004). Bank instability and failure depends on the site-specific, combined effects of gravity (bank height), wave attack, rainfall impact, surface water runoff, and wind (Ibison et al 1990), storm surge and groundwater seepage (Hardaway et al 1992, USACOE 1990), reduction in littoral drift (ACOE 1990, CBP 2005) and boat wakes (Davis et al 2000).

Excess sediment in the water column contributes to degraded water quality and habitats for living resources, yet the current understanding of sediment loads from eroding shorelines and tidal water quality effects is incomplete (CBP 2005). It is currently believed that sediment is the 3rd biggest pollutant of the Chesapeake Bay and that 57% of the total Bay sediment load is from tidal erosion (CBP 2005, Langland and Cronin 2003). Unstable banks contribute large volumes of sediment directly to the estuary (Ibison et al, 1992; Hardaway et al 1992). Excessive sediment loads can reduce clarity and introduce nutrients and toxics attached to soil particles. Ibison et al (1990, 1992) estimated the nitrogen load from tidal erosion is 5.2% and the phosphorus load is 23.6% of the "controllable" nonpoint source load, which excluded atmospheric contributions.

Bank vegetation contributes to bank stability (see Bank Cover), but the effect is variable with different vegetation types and different root-shoot architecture and belowground biomass (Ott 2000, Wynn et al 2004). Bank vegetation also provides resistance during heavy floods (Klapproth and Johnson 2000). On the other hand, large trees may locally increase failure if their weight can overcome any additional increase in shear strength due to root systems. Windthrown trees can also contribute sediment to the adjacent waterway (Ott 2000).

Undercut banks with erosion only at the toe of the bank have characteristics of both stable and unstable banks. Hydraulic shear stress from the stream channel and turbulence are highest at the bank toe (Wynn et al 2004). Water level fluctuations in tidal tributaries allow waves and currents to attack the bank at changing elevations, and the tractive force removal of material at the toe of the bank can be a failure mechanism (Davis et al 2000). An increase in bank instability and sediment loss due to undercutting caused by groundwater seepage has also been demonstrated (Wilson et al 2007). Groundwater seepage can substantially reduce stability by either forcing sediment grains apart or by facilitating slippage along discontinuities in the sediment pile (e.g. water flow along clay layers) (NRC 2007).

The concept of stability implies that a bank is stable if it does not appreciably change within a defined time frame (Ott 2000). The bank appearance at any given time may or may not reflect long-term stability. Hardaway et al (1992) found that the volume of eroded material is more significant than the erosion rate indicated by bank appearance for

nutrient and sediment loading in the Chesapeake Bay. This effect depends on bank height. An unstable low bank might have less impact on water quality than a visibly stable high bank if large volumes of sediment are eroded from the high bank only periodically (Hardaway et al 1992, Ibison et al 1992).

The model assumption that unstable banks have a high potential for continued erosion and sediment introduction is supported by research conducted in the Chesapeake Bay region. The assumption that undercut banks have a moderate potential for sediment and nutrient loading is also supported by research conducted in other riparian systems. However, the assumption that no visible erosion equates to bank stability and low potential for adverse nonpoint source pollution is not entirely supported. Additional information is needed about vegetation distribution in relation to erosion rates, historic landward recession and hydrodynamic processes such as tidal currents, wave attack and groundwater flow.

5. Shoreline Resources

Shoreline Resources - Water Quality

Numerous studies have examined the role of marshes in nutrient reduction and sediment retention. Some studies suggest higher specific rates of dentrification in marsh systems (Kaplan et al. 1979, Davis et al. 2004) while others suggest a general system-wide reduction in nutrient loading to adjacent waters (MacCrimmon 1980, Reddy & Gale 1994, Fisher & Acreman 2004, Morse et al. 2004). Multiple studies and reviews have focused on sediment retention and the assumed water quality benefits of retaining nutrient-laden sediment (Oviatt & Nixon 1975, Gleason et al. 1979, MacCrimmon 1980, Reddy & Gale 1994, Bricker 1996, Christiansen 1999, Neubauer et al. 2002, van Proosdij et al. 2006).

One study in particular concluded that *Phragmites* (when compared to Spartina communities) actually exhibited greater rates of mineral and organic sediment trapping and sediment stabilization (Chambers et al. 1999, Rooth et al. 2003). Studies have also recognized the increased production, both in aboveground (increased litter) and belowground components and the significance of this advantage in the face of sea level rise (Rooth & Stevenson 2000, Rooth et al. 2003).

Coastal primary sand dunes are assumed in the model to serve as protective barriers from flooding and erosion resulting in decreased sediment and nutrient inputs. Norstrom and Lotstein (1989) emphasized the importance of unstable, dynamic dune systems as important reservoirs of sand storage for beaches and the negative impacts that dune stabilization and active management can have on this function. They did not address positive benefits of dune stabilization on water quality. However, the relative deficiency of nutrients in sandy substrates would suggest that dune plant root systems would be very efficient at uptake and retention of available nutrients as they filter through the dune substrate (Conn & Day 1993, Heyel & Day 2006). In addition, any physical barrier to runoff is likely to result in a reduction in nutrient and nutrient-laden sediment inputs to adjacent waters.

The model assumption that vegetated dunes and marshes, including *Phragmites*, improve water quality and help reduce erosion by filtering groundwater and holding sediment in place are supported by the literature.

Shoreline Resources - Habitat

The exceptional habitat provided by salt marsh ecosystems has been extensively documented. This body of evidence includes studies and reviews of individual bird taxa or functional groups (Erwin et al.1993, Erwin 1995, Erwin 1996), invertebrate taxa (Smalley 1960), reptiles (Hurd et al. 1979) and fish (Oviatt & Nixon 1973) as well as comprehensive reviews of all vertebrate and invertebrate taxa (Pomeroy & Wiegert 1981, Wiegert & Freeman 1990).

Coastal primary sand dunes represent transitional areas that bridge marine and terrestrial habitats and provide essential habitat for plants and animals. Beaches interact with primary and secondary sand dunes and serve as habitat for benthic animals and microalgae living on or within the sand. Beaches can also serve as refuge and forage areas for finfish, blue crabs and wading shorebirds. Dunes and beaches have been studied for the habitat they provide as a continuum (Engels 1942) and in particular beaches have been studied as unique intertidal surf zone habitat for a large variety of vertebrate and invertebrate, upland, aerial, and aquatic taxa (Pearse et al. 1942, Dexter 1967, McLachlan & Brown 2006).

Phragmites marshes grow in a wide range of intertidal and nearshore areas. The typical growth pattern of monotypic stands raises questions regarding habitat value relative to more diverse communities. The non-native variety of *Phragmites* may be highly competitive, displacing native marsh vegetation. Studies of the impact of *Phragmites* on habitat have focused on decreased food availability (Able & Hagan 2000, Robertson & Weis 2005, Hunter et al. 2006, Robertson & Weiss 2007), changes on marsh surface (Jivoff & Able 2003), and the general changes in species diversity, density and richness associated with these changes (Chambers et al. 1999, Wainright et al. 2000, Angradi et al. 2001, Able & Hagan 2003, Osgood et al. 2003, Weiss & Weiss 2003). However, some studies have also asserted that some taxa, particularly those at higher trophic levels, may be unaffected by the changes (Chambers et al. 1999, Weiss & Weiss 2003).

The model identifies marshes, beaches, dunes and *Phragmites* as transitional areas between upland and subaqueous lands and assumes that they provide habitat (food and shelter) for both aquatic and terrestrial animals such as blue crabs, small fish and marsh birds. The model assumptions regarding the provision of habitat services by shoreline resources are supported by the scientific literature.

6. Shoreline Structures

Shoreline Structures - Water Quality

Bulkheads

Bulkheads have the potential to impact water quality positively by reducing upland erosion, or negatively by changing wave reflection patterns leading to suspension of bottom sediments. The change in wave patterns associated with bulkheads may negatively impact nearby SAV and marshes, both of which improve water quality. All bulkheads have the potential to impact the sediment dynamics of a system through the entrapment of sediment landward of the bulkhead (Douglass and Pickel 1999, Griggs 2005). This may be a positive impact where clay and fine sediments are prevented from entering the water column and turbidity is reduced, or may be a negative impact where the reduced erosion results in a sediment deficit downstream. The location of a bulkhead in the landscape may affect its impact; subtidal and low intertidal bulkheads promote sediment movement and an increase in sediment grain size at the base of the bulkhead (Bozek and Burdick 2005, Douglass and Pickel 1999, Spalding and Jackson 2001). Bulkheads that are located in the upper intertidal zone and landward appear to have less impact on local sediment movement (Basco et al. 1997, Griggs 2005, Spalding and Jackson 2001). Bulkheads may reduce groundwater flow from the upland, which can have an unquantifiable impact to water quality. This property may be highly dependent on the material that the bulkhead is made from, since rock seawalls do not appear to be a barrier to groundwater flow (Bozek and Burdick 2005). Bulkheads can lead to beach or marsh loss through passive erosion (Bozek and Burdick 2005, Griggs 2005) and can reduce marsh plant diversity by occupying the upper marsh elevation (Bozek and Burdick 2005). Impacts to marsh vegetation may indirectly impact water quality.

Riprap

Little work has been done on the impact of riprap revetments on water quality. Like bulkheads, riprap revetments have the potential to impact the sediment dynamics of a system through the entrapment of sediment landward of the revetment (Griggs 2005) and the effect may be positive where the sediments are fine-grained with associated nutrients, or negative where a downstream sediment deficit results. Since stone seawalls appear to have no impact on groundwater flow (Bozek and Burdick 2005), it is unlikely that riprap revetments would. Revetments can lead to beach or marsh loss through passive erosion (Griggs 2005), and may impact downdrift properties through the interaction of wave reflection with longshore wave transmission (Camfield and Briggs 1993). They can indirectly reduce water quality through the loss of natural vegetation due to riprap placement (Quigley and Harper 2004), or indirectly improve water quality by providing a substrate for filter feeders (Newell and Ott, 1999).

Jetties

Only one study was found evaluating the impact of jetty construction on water quality (Nelson et al. 2005). In the study there was a concurrent switch from septic tank to sewage collection system in the area that resulted in a beneficial effect on water quality. No impact, positive or negative, to water quality was linked to jetty construction. Jetties can indirectly reduce water quality services when natural vegetation is displaced by structure placement. However, the use of rock jetties as substrate for filter feeders may indirectly improve water quality (Goren and Benayahu 1993, Johnson and Geller 2006).

Breakwaters

Breakwaters can impact water quality through the alteration of sediment transport and wave dissipation. Breakwaters can change the mode of sediment transport (Cuadrado et al. 2005), may lead to scouring or bar formation (El Banna 2006, Ranasinghe and Turner 2006) and can cause shoreline changes both locally and on adjacent properties (Pranesh et al. 1984). These changes may be directly related to wave energy, since breakwaters have been found to have more effect on large amplitude wave energy than lower energy waves (Dickson et al. 1995). However, breakwaters tend to be built in areas with sandy sediments, so alterations in sediment movement are unlikely to be associated with increased turbidity. This limits the potential for breakwaters to impact water quality. Breakwaters can indirectly reduce water quality through the loss of SAV due to structure placement, or they may indirectly improve water quality by providing a substrate for filter feeders (Newell and Koch 2004). In the short term, wave attenuation associated with breakwaters may provide the appropriate energy conditions for SAV and saltmarsh (Allen et al. 1990, Rice et al. 1989, Rennie 1990). However, the benefit associated with wave attenuation may be negated in the long term by changes in sediment transport leading to the accumulation of fines in the breakwater lee (NRC 2007)

Debris

Many different types of debris have been used to stabilize shorelines, including concrete rubble and automobile tires. If installed properly, structures built from these materials have the potential to impact the sediment dynamics in a manner comparable to riprap revetments or bulkheads (see above). The lack of literature regarding the potential for water quality impacts from the use concrete structure in the aquatic environment suggests that the general consensus is that concrete is neutral in this regard. The impact of tires (used and new) on water quality has been extensively examined. Leachate from tires into water is toxic to a number of aquatic organisms, including: rainbow trout (Day et al. 1993, Stephensen et al. 2003), sheepshead minnows (Evans et al. 2000, Evans 1998), *Daphnia sp*. (Wik and Dave 2006) and various bacteria (Day et al. 1993). Leachate from used tires is more toxic than new tires (Day et al. 1993). The toxicity of leachate tends to be highest in freshwater (Hartwell et al. 2000), so the use of tires is of greatest concern in the upper tidal region.

Marinas and Boat ramps

There are several literature surveys and studies on the effects of marinas and boat ramps on water quality (Nixon et al. 1973, Chmura & Ross 1978, USEPA 1985, Milliken & Lee 1990, NCDEM 1991, USEPA 2001). Pollutants most documented or of greatest interest are:

- copper and TBT from antifouling paints on boats and other structures,
- other heavy metals such as lead, zinc, and mercury
- petroleum hydrocarbons (including PAHs)
- fecal coliforms as an indicator of the presence of sewage (human or animal).

Copper and TBT were generally found in greater concentrations within marinas than at non-marina locations (Grovhoug et al. 1986, Hall et al. 1987, McGee et al. 1995, Nixon et al. 1973, Young & Hessen 1974, Young et al. 1975). Chen et al. (1972) and McMahon (1989) found storm drains, maintenance area drains, and fuel docks to be important sources of heavy metals. Petroleum hydrocarbons were generally higher in marinas (Marcus et al. 1988, McGee et al. 1995, Mastran et al. 1994, Voudrias & Smith 1986). An et al. (2002) suggest that fuel spillage is greater at boat motor start-up locations than at fuel pumping facilities, suggesting that boat ramps may be important sources of this pollutant.

Many studies found that marinas are associated with have high fecal coliform concentrations in the water column and sediments (Barbaro et al. 1969, Cassin et al. 1971, Faust 1982, Fisher et al. 1987, Fufari & Verber 1969). However, Kirby-Smith & White (2006) found the highest fecal coliform levels on residential shorelines rather than at marinas, suggesting that upland runoff is a more important source of fecal coliform than boat discharge.

According to the literature reviewed, the impact of bulkheads and revetments on water quality has not yet been clearly defined. However, it does support the assumption that these structures can reduce sediment inputs from erosion by stabilizing the shoreline. Although bulkheads may alter local sediment movement and temporarily increase local turbidity, no direct negative impacts to water quality have been identified. Jetties are considered to have relatively little impact on water quality and therefore should not be considered in the water quality model. Similarly, the literature reviewed support the assumption that breakwaters have relatively little impact on water quality and therefore should not be considered in the water quality model.

Certain types of debris/miscellaneous stabilization methods can greatly impact water quality. Some debris structures may reduce sediment inputs from erosion by stabilizing the shoreline, but installation methods tend to be highly irregular, limiting their effectiveness. The model should consider possible toxic impacts associated with marine debris.

The literature generally supports the assumption that marinas and boat ramps introduce pollutants. However, several studies stressed the importance of flushing

and circulation in controlling levels of all pollutants (Kirby-Smith & White 2006, Marcus et al. 1988, McGee et al. 1995, Voudrias & Smith 1986). These shoreline/water body characteristics are not addressed in the current model. There was no literature found that directly supported the differential water quality values attributed in the model to marinas (-3), public boat ramps (-2), and private boat ramps (-1).

Shoreline Structures - Habitat

Bulkheads

Bulkheads may reduce natural habitat by direct replacement in the landscape (Bozek and Burdick 2005), through passive erosion (Griggs 2005), through active erosion and interference with sediment transport (Douglass and Pickel). Bulkheads landward of the intertidal area have little impact on sediment movement (Basco et al. 1997, Griggs 2005, Spalding and Jackson 2001), which may translate to low habitat impacts (Jarmillo et al. 2002) on beaches channelward of bulkheads. Bulkheads closer to the water correlated with sediment loss and high temperatures in the intertidal zone, resulting in impacts to organisms using those areas (Spalding and Jackson 2001, Rice et al. 2004, Rice 2006.) The reduction of natural habitat may result in habitat loss if the bulkhead cannot provide substitute habitat services. In Australia, where vertical rocky shores are prevalent, concrete bulkheads are colonized by a variety of aquatic animals, although community structure and zonation may differ from natural shorelines (Bulleri 2005a, Bulleri et al. 2005, Bulleri 2005b, Chapman 2006, Chapman 2003.) On shorelines that tend to be vegetated, bulkheads may lower invertebrate density relative to natural shorelines (Seitz et al. 2006, Toft 2005). In North Carolina, bulkheads were found to increase predation on sea urchins (Zito et al. 2004). In general, bulkheads tend to support lower density and diversity of nekton than natural sites (Bischoff 2002, Hendon et al. 2001, Peterson et al. 2000, Trial et al. 2001). Percentage of hardened shoreline is negatively correlated to the number and diversity of species (Wolter 2001). When compared with riprap, bulkheads tend to support the lowest diversity and abundance of fauna, while riprap may be intermediated or similar to natural sites (Jennings et al. 1999, Schmude et al. 1998, Seitz et al. 2006, Trial et al. 2001). Despite this, along hardened reaches, even altered marsh shorelines can serve as important habitat for some nekton (Hendon et al. 2000).

Riprap

Riprap revetments may reduce natural habitat by occupying its space in the landscape (Bozek and Burdick 2005) and through passive erosion (Griggs 2005). Riprapped shorelines are associated with the removal of riparian vegetation, which can lead to a lack of large woody debris, and important habitat, in river systems (Angradi et al 2004). However, riprap also appears to provide habitat especially along naturally rocky shorelines. Riprap may serve as habitat for filter feeders (Burke et al. 2006, Newell and Ott 1999). Compared with vegetated marshes and natural oyster reefs, riprap tends to support lower diversity and abundance of fauna (Bischoff 2002, Burke 2006, Carroll 2003, Davis 2001, Garland et al. 2002, Hendon et al. 2001, Peterson et al. 2000,

Schmetterling et al. 2001, Seitz et al. 2006). Some studies have found exceptions to this, with riprap similar to natural shoreline (Jennings et al. 1999, Trial et al. 2001) and the impact of riprap on community structure may depend on its location along the coastline (Davis et al. 2002) and the structural makeup of adjacent natural sites (i.e. rocky vs. marshy shorelines). Even altered marsh shorelines may serve as important habitat in highly developed reigns (Hendon et al. 2000). In comparison to bare sediment and created oyster reefs, riprap may support similar or higher nekton abundance and oyster settlement (Beauchamp et al. 1994, Burke 2006, Davis et al. 2001).

Jetties

Jetties provide subtidal and intertidal structure and therefore may support diverse communities. Rock jetties have been compared with reef structures in terms of the species of fish associated with them (Dolah et al. 1987), and support certain species of filter feeders (Johnson and Geller 2006, Newell and Ott 1999, Rader 1998). However, they may support lower densities of reef dependent fish than natural reefs (Hernandez et al. 2001). Rock jetties can support high algal biomass, which can represent a significant contribution to local food chains (Kaldy et al. 1995). Jetties support a different epiphytic algal community from adjacent SAV beds, suggesting that they cannot replace natural habitats, but may increase algal diversity (Sullivan 1984). In Texas waters, juvenile green turtles appear to preferentially select jetties over other habitats, which serves as summer habitat for this species (Renaud et al. 1995). In stream settings, jetties provide surfaces for some invertebrate growth and create scour hole habitat for fish (Witten and Bulkley 1975).

Breakwaters

Like jetties and riprap, breakwaters can provide structure in areas otherwise lack subtidal structural habitat. In areas that do not have natural hard structure, breakwaters can change the local community composition (Airoldi et al. 2005, Moschella et al. 2005). There is also concern that they may promote the spread of non-native or invasive species (Airoldi et al. 2005). However, they may also provide nursery habitat (Moschella et al. 2005), encourage the growth of SAV and salt marsh (Allen et al. 1990, Rice et al. 1989, Rennie 1990), and encourage recruitment of oysters (Newell and Koch 2004) and other sessile organisms (USACE et al. 1990). Conflicting information is found on the impact of breakwaters on nekton communities, with some studies reporting benefits to the community (Lincoln Smith et al. 1994, Stephens et al. 1994, Stephens and Pondella 2002), while other studies found reduced fish community diversity or impacts to particular species associated with breakwaters (Moschella et al. 2005, Seitz et al. 2005).

Debris

The role of miscellaneous structures and debris as aquatic habitat has not been clearly studied. However, broken concrete and tires cover natural habitats and therefore are assumed to negatively impact habitat function.

Marinas and boat ramps

Generally, adverse impacts of marina operations and boat ramps on habitat is attributable to filling of subtidal and wetland habitat by boat ramps, shading by piers and boats associated with marinas, and periodic dredging associated with marinas. Nixon et al. (1973) suggested that fouling communities in marinas appeared to be an important food source for fish. They found that sport fish were more abundant in marina areas than in nonmarina areas. Houseboats acted as artificial structures providing habitat for many types of organisms (Hertler et al. 2004). Houseboats, if allowed to swing 360 degrees and did not have antifouling paint, did not adversely impact seagrass growth.

Jensen et al. (2004) found that TBT-contaminated sediments were associated with decreased net photosynthetic activity and decreased relative growth rate of seagrass. Antifouling herbicides were shown to have potential adverse impacts on seagrass growth (Chesworth et al. 2004). Benthic infauna present were a reflection of environmental degradation within a marina basin (McGee et al. 1995).

Reish (1961) found that a benthic community colonized a newly dredged area within a year of dredging.

The literature reviewed supports the model's assumption that bulkheads can negatively impact habitat function in a reach by replacing and impacting natural habitat. The magnitude of the impact varies from location to location and may be somewhat dependent on the adjacent shoreline setting. Consistent with the model, bulkheads have a greater negative impact than riprap on habitat functions.

The literature reviewed suggests that the value of riprap as habitat is highly situational. In areas that are structurally simple or where shorelines are naturally rocky, riprap may provide similar or improved habitat. Riprap appears to provide better habitat than bulkheads in most circumstances. However, it almost always provides reduced habitat compared to a complex marsh shoreline. The situational nature of habitat services provided by riprap has made it a neutral element in the habitat model, neither increasing nor decreasing habitat function.

The literature reviewed suggests that rock jetties can provide habitat although it may not be equivalent to natural habitats. In areas with reduced structure, jetties may be valuable, but in areas with lots of other structure they are unlikely to be important. In the Chesapeake Bay, oyster reefs that used to provide structure no longer exists, so jetties may be serving some reef-like function.

The literature reviewed suggests that rock breakwaters can provide habitat although it may not be equivalent to natural habitats. In areas with reduced structure, breakwaters may provide valuable habitat. In the Chesapeake Bay, oyster reefs that used to provide structure no longer exists, so breakwaters may be serving some reef-like function.

No studies are available to assess the impact of miscellaneous structures and debris on habitat function.

There was no literature found that directly supported the differential habitat values attributed in the model to marinas (-3), more than one boat ramp (-2), and single boat ramps (-1).

7. Subaqueous Resources

Subaqueous Resources - Water Quality

SAV

Submerged Aquatic Vegetation (SAV) can improve water quality through uptake of excess nutrients or other pollutants, stabilization of sediments and reduction in turbidity caused by wave damping. The removal of excess nutrients is somewhat in question, studies on sediment porewater profiles (Lilleboe et al. 2006) in SAV beds suggest that the impact of vegetation is dependant on the biomass and root penetration of the plant and may vary temporally. At nighttime, SAV beds may actually contribute nutrients (particularly phosphorus) to the water column. The impact of SAV presence on turbidity has stronger support. SAV beds have been found to help stabilize unconsolidated sediments (Churchill et al. 1978) and reduce seawater velocity and wave energy (Fonseca and Calahan 1992, Gambi et al. 1990, Madsen et al. 2001, Newell and Koch 2004, Peterson et al. 2004). The reduction of flow is dependant on vegetation density (Gambi et al. 1990, Peterson et al. 2004), vegetation height (Fonseca and Calahan 1992) and existing wave climate (Paling et al. 2003). Broad, shallow grass beds are predicted to reduced wave energy substantially, and may be as effective as comparably sized salt marshes (Fonseca and Calahan 1992). The reduction in water movement within the SAV beds results in increased sedimentation and reduced turbidity in the water column (Madsen et al. 2001), but SAV may be less effective in this regard than oyster beds (Newell and Koch 2004). Along high-energy coastlines, SAV may not be effective at wave reduction or increasing sedimentation (Paling et al. 2003).

Oyster Reefs

Oyster reefs can improve water quality by acting as a wave break while the oysters can reduce sediment and nutrient content of the water column through filtration. In this regard, both created and natural oyster reefs appear to function in a similar manner (Campbell 2005, Heck et al. 2005, Piazza et al. 2003, Piazza et al. 2005). Oyster reefs dissipate wave energy (Campbell 2005) and reduce erosion along lower energy shorelines (Piazza et al. 2003, Piazza et al. 2005). However, in high energy areas, their potential for reducing erosion may be limited (Piazza et al. 2003, Piazza et al. 2005). Oyster filtration has been shown to potentially reduce turbidity by an order of magnitude, which may facilitate SAV restoration efforts (Newell and Koch 2004, Cerco and Moore 2001),

particularly in areas that already support SAV. Other bivalves also filter the water column, helping to remove excess nutrients and reduce turbidity (Ruesink et al. 2006). This suggests that the presence of any bivalve will result in water quality improvement, but oysters have been shown to be particularly effective due to their high rates of water filtration (Newell and Koch 2004)

Shellfish Aquaculture

The impact of bivalve aquaculture on water quality is not well quantified. Bivalves are capable of reducing turbidity and removing excess nutrients through water column filtration (Newell and Koch 2004, Ruesink et al. 2006). However, the impact of the bivalves on water quality may depend greatly on the species being cultured. Oysters and other fast growing bivalves have the potential to greatly impact water quality (Newell and Koch 2004, Rheault 2006, Ruesink et al. 2006), while hard clams may have only limited impacts on water quality (Newell and Koch 2004). Some bivalve aquaculture has been shown to have little to no impact on water quality properties (Vaudrey et al. 2006). Aquaculture may also impact water quality indirectly by impacting SAV survival and restoration efforts. There is a potential for aquaculture to enhance SAV growth through the reduction of turbidity (Newell and Koch 2004, Cerco and Moore 2001), and in Long Island Sound, eelgrass growth was enhanced in the presence of aquaculture (Vaudrey et al. 2006). However, oyster harvesting methods can negatively impact SAV, and oyster aquaculture has been shown to decrease eelgrass density, particularly in dredged beds (Tallis et al. 2006).

The literature reviewed supports the model's assumption that SAV can contribute significantly to water quality, and is approximately equivalent to the presence of salt marsh along a reach. The contribution of SAV to water quality may be somewhat reduced in high energy settings. According to the literature, oysters can contribute significantly to water quality. They appear to be more effective in this regard than hard clam aquaculture or SAV. The contribution of oyster reefs to water quality may be also somewhat reduced in high energy settings.

The impact of Aquaculture on water quality may be positive or neutral. The magnitude of the impact is related to the type of bivalve being cultured as well as the culturing and harvesting methods. In addition, there is the potential for indirect negative impacts to water quality through SAV reduction.

Subaqueous Resources - Habitat

SAV

The benefits of SAV as habitat are well established. SAV beds tend to support higher densities of fish and crabs than unvegetated areas (Castellanos et al. 2001, Dealteris et al. 2004, Harris et al. 2004, Hosack et al. 2004, Lipcius et al. 2005, Thayer et al. 1985). This may be a result of increased food availability in SAV (Horinouchi 2007, Leduc et al. 2006) or enhanced refuge from predation (Reid 2004). Blue crabs appear to benefit

extensively from the presence of SAV since their megalopae preferentially select SAV for settlement (Stockhausen and Lipcius 2003, Van Montgrans et al. 2003). Clams benefit from reduced predation within SAV beds (Reid 2004). SAV seems to support similar densities and abundances of crustaceans as oyster reefs, oyster aquaculture and salt marshes (Dealteris et al. 2004, Glancy 2003), but the community structure tends to be more similar to salt marshes than oyster reefs (Glancy 2003). Some bottom dwelling fish, such as winter flounder, do not appear to benefit from SAV (Goldberg et al. 2002, Phelan et al. 2000), but do not seem to be negatively impacted either.

Oyster Reefs

Oyster reefs (both natural and created) provide habitat for other oysters as well as a variety of other attached and reef-dwelling aquatic species. Overall faunal densities tend to be much higher on oyster reef than on unstructured bottom (Heck et al. 2005, Hosack et al. 2004). Comparisons between oyster reef and salt marsh or SAV suggest that all habitats may have similar value (Dealteris et al. 2004, Glancy 2003, Piazza et al. 2003). Oyster shell provides a site for oyster settlement, although the relative value of different habitat types is in question. One study showed that oyster recruitment and survival was best in salt marshes, but better on granite than on created reefs (Burke et al. 2006). While another study showed much higher settlement on created reefs (at the edge of the marsh) than natural reefs (Meyer and Townsend 2000). Epifaunal, macrofaunal and sessile macrofaunal density may also be enhanced on created reefs relative to natural ones (Rodney and Paynter 2005, Rodney et al. 2006)

Shellfish Aquaculture

Aquaculture of bivalves provides structure in the water column and therefore has some potential as habitat. Shellfish aquaculture gear and cultivated oysters provide substrate for sessile invertebrates and refuge for juvenile fish (Dealteris et al. 2004, Rheault 2006). Aquaculture gear is considered to have higher habitat value than unvegetated bottom and may provide services equivalent to SAV (Dealteris et al. 2004). Aquaculture has been associated with enhance eelgrass growth (Vaudrey et al. 2006) and increased seedling recruitment (Wisehart et al. 2006). However, impacts associated with aquaculture harvest may negatively impact SAV density, particularly in dredged beds (Tallis et al 2006).

The literature reviewed supports the model's assumption that SAV and oyster reefs can provide significant comparable aquatic habitat, equal to salt marshes. The relative scarcity of SAV in the Chesapeake Bay, combined with its importance as a habitat to many aquatic and fishery species, suggests that this habitat should be preferentially conserved. The magnitude of the habitat contribution by oyster reefs may depend on whether the reef is natural or created, the material used for the reef, and its location in the landscape.

The literature reviewed suggests that aquaculture should be included as an element for the habitat model. Habitat value may be highest when aquaculture is located away from SAV beds, to reduce the potential for negative harvesting impacts.

8. Fetch and Bathmetry

Fetch and water depth are recognized as elements significant to wave climate (Knutson et al.1982, Knutson et al. 1981). Fetch can be described as a simple measure of relative wave energy (Hardaway and Byrne 1997). van der Wal and Pye (2004) suggest erosion within estuaries can result from relatively small waves generated over short fetches, and that flats in the UK provide little shoreline protection during storm tides when they are submerged by up to 4m of water. Williams (2001) suggests limiting fetch to <300 m when trying to establish salt marsh plantings and insure natural sedimentation. Hardaway and Byrne (1997) characterized low energy shorelines as those where fetch <1 nautical mile.

Shallow nearshore depths, such as tidal flats and sand bars are able to attenuate incoming wave energy before reaching the shoreline better than deeper waters (Hardaway and Byrne, 1997). Hardaway et al. (1992) used distance to the 6 ft. contour to characterize nearshore water depths.

Wave height is a good indicator of the amount of energy reaching a shoreline (Roland et al. 2005), with wave energy related to the square of wave height.
Empirical models have been developed (Basco and Shin 1993, Hardaway et al. 1992, Keddy 1982, Knutson et al. 1981, Shafer et al. 2003) to characterize relative wave energy reaching a shoreline using numerous metrics such as fetch, wave height, wave period, wind speed, wind duration, and shoreline geometry. Some of these models require the use of wind-wave hindcasting to provided wave height or wave period input to the specific model. This approach requires wind data along with establishing a tide and wave gauge at the area(s) of interest in order to measure the effect of winds on wave height and wave period over some extended period of time, usually for a minimum of one year or more in order to capture seasonal variation in wind patterns. Walton and Adams (1976) used significant wave height and significant wave period to derive a measure of wave energy and separates shorelines into different energy environments. Another accepted wave energy model is the Relative Exposure Index (REI) (Keddy, 1982), that is based upon mean annual wind speed, percent frequency that the wind blew from 16 cardinal and subcardinal compass directions, and fetch distance in each of the 16 compass directions. Fonseca (1996) modified REI for identifying suitable sites for seagrass restoration projects in Florida Bay. However, Roland et al. (2003) noted that the potential effect of water depth is not explicitly accounted for in the REI model, and because wave height can decrease as a wave propagates from deep to shallow water, the inability of this model to account for the effect of water depth on wave climate reduces the applicability of REI for determining the sustainability of marshes or locating potential wetland planting sites.

Given the multitude of available wave climate models, each with its own rationale for characterizing fetch and water depth somewhat differently, the shoreline protection model developed by VIMS is designed to be conservative in evaluating the effect that each of these metrics has on the ability of vegetated tidal wetlands to provide shoreline erosion protection. NOAA bathymetry data were used to calculate the water depths and

distance to the 2 m contour. Shallow water habitat is defined by the U.S. Army Corps of Engineers as <2 m depth, and dominate the nearshore depths throughout most of the Chesapeake Bay. The critical distance to the 2 m contour was defined as 100 m for the shoreline protection model. As with water depths, the characterization of fetch in the various models discussed previously is somewhat relative to the scale of the system for which the model was developed. The shoreline protection model developed by VIMS uses a critical fetch distance of 1000 m (@0.5 nautical miles) to represent a considerable percentage of protected shorelines within the Chesapeake Bay and its tributaries where vegetated wetlands can be expected to be located.