Integrated Guidance for Tidal Shorelines

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Introduction to the Integrated Guidance Concept

Tidal shorelines are the site of complex interactions between terrestrial and aquatic systems. These areas have values that far outweigh their relative size in the larger ecosystem. They are exceptionally important habitat for a wide variety of organisms, some living primarily on land, others that live in water, and a few that are found only in the intertidal zone between land and water. Tidal shoreline systems provide important filtration capacity for materials carried in runoff and groundwater. They are uniquely valued by human users of coastal systems.

In Virginia, tidal shoreline systems are managed in small segments, rather than as a whole unit. Local governments implementing the Chesapeake Bay Preservation Act manage the riparian zone, intertidal areas fall under the purview of local wetland boards, and the subaqueous environment is the responsibility of the Virginia Marine Resources Commission. Each of these programs tends to seek avoidance of impacts in areas under their jurisdiction. This preference for the status quo can be in conflict with shoreline management that optimizes the tradeoffs in public and private benefits.

Recognition that particular shoreline management options may not be uniformly desirable from different regulatory perspectives means coordination among management agencies will be essential. The basis for coordination is logically the rationale for establishment of the various regulatory programs – sustaining public benefits from environmental services. The desire to maintain the capacity of the natural system to do things that are important and valuable to the general citizenry of the Commonwealth underpins the riparian, intertidal and subaqueous lands management programs operating in Virginia. These programs uniformly seek to accommodate private development interests within the broader goal of sustaining ecological services.

There are currently a variety of guidelines developed by local and state programs managing shoreline development activities. These include the Virginia Marine Resources Commission guidelines for tidal wetlands, subaqueous lands and coastal primary sand dunes. In addition, the Department of Conservation and Recreation's Chesapeake Bay Local Assistance Division and Shoreline Erosion Advisory Service have both issued guidelines for riparian land management. There are, however, no comprehensive guidelines that synthesize the objectives of all these programs.

It has become increasingly apparent that in order to reduce the cumulative and secondary impacts of activities within the multiple jurisdictions and multiple management programs affecting the littoral and riparian zones, better coordination and integration of policies and practices is necessary. It may be possible to address the gap of the jurisdictional limitations of the various programs that manage the shoreline by providing enhanced technical guidance to promote integrated management decision-making.

The Integrated Guidance Model

Part 1: Ecosystem Services Assessment Model

Introduction

The model integrates water quality and habitat features with shoreline risk through a cross-section of the coastal landscape, from the upland through the subaqueous zone. Each element of the model individually impacts water quality and habitat services, allowing integrated assessment of the ecosystem services provided along a given reach of shoreline. Water quality and habitat functions were modeled separately, because elements may impact the two services independently. Shoreline risk was also modeled separately because it represents a potential threat to the shoreline, not a service provided by the shoreline.

The assessment units were 500 meters in length and of variable width, beginning 90 meters landward of the shoreline and continuing to some distance off-shore as determined by the 2 meter depth contour or 200 meters distance, whichever comes first. They are divided into Upland (90 meters from shoreline), Riparian (30 meters from shoreline), Banks (within the riparian zone), Shoreline, and Subaqueous (channelward of shoreline, variable width).

Each element and its known impacts on water quality and habitat services and shoreline risk are described below. Also included is a brief description of how the element affects the overall score generated by the model. Rankings and calculations associated with the elements are shown in the table. The total value for an assessment unit is a sum of the element values.

The final model scores represent the average of the value of the assessment unit plus half the value of each adjacent unit. This running average method of assessing the shoreline allows for better integration of the effects of adjacent shoreline features and avoids segmentation of the shoreline. In this first iteration of the model, scores are divided into 5 categories that reflect the current level of water quality or habitat functions along a given shoreline. The categories are: Good, High moderate, Moderate, Low moderate and Poor.

Two figures are provided, which show the results of the water quality and habitat models for a stretch of shoreline in Gloucester, Virginia. The range of final scores is shown as different colors along the shoreline.

Model Elements

Upland Landuse

Upland Landuse was estimated from remotely sensed land cover data assessed from the shoreline interface landward 90 meters. For the purposes of the model land cover was compiled into 3 categories: Natural, Agricultural, and Developed. Natural land cover includes wetlands, forest, scrub-shrub, and timbered cover types. Agricultural land cover includes agricultural and grassland cover types. All other cover types were classified as Developed. Land use estimates were taken from the National Land Cover Dataset (RESAC 2000).

Riparian Landuse

Riparian Landuse was estimated from observed data (CCI Shoreline Inventory) assessed from 30 meters landward to the shoreline interface. For the purposes of the model land cover was compiled into 4 categories: Natural, Agricultural, Developed, Industrial/Paved. Natural land cover includes wetlands, forest, scrub-shrub, and timbered cover types. Agricultural land cover includes agricultural and grassland cover types. Industrial/Paved land cover includes large industry and roads/parking lots adjacent to the shoreline interface. All other cover types were classified as Developed.

Forested Buffer

Forested Buffer refers to the presence or absence of a forest fringe in the riparian zone (30 meters landward of the shoreline interface). Forested Buffer presence was a riparian area modifier calculated from a combination of field-observed (CCI Shoreline Inventory) and remotely sensed data (RESAC 2000) and assessed only where combined agricultural or developed cover comprised at least 70% of the assessment unit.

Bank Cover

Bank Cover was estimated from the CCI Shoreline Inventory and refers to either vegetative or structural cover on the bank, defined here as the area of transition between the shoreline and upland. Bank Cover is divided into 3 categories based on percent cover: Bare (< 25%), Partial Cover (25-75%), and Total Cover (> 75%).

Bank Stability

Bank Stability was estimated from the CCI Shoreline Inventory and refers to the amount of erosion on the bank face or bank toe and ranked by severity. Bank Stability is divided into 3 categories based on severity: Stable, Unstable, and Undercut. Stable banks may include banks with minimal or no erosion on the bank face or toe. Unstable banks may include slumping, scarps, or exposed roots on the bank face. Undercut banks may include otherwise stable banks with erosion observed only at the toe.

Shoreline Resources

Shoreline Resources observed in the field were identified by 4 categories: Marsh, Dune, Beach, *Phragmites.* These resources were included in the model score when present but did not reduce the total score when absent. The linear extent of Marsh, Beach, and Phragmites resources were observed in the field and interpreted in the model as a percentage of the linear assessment unit. The extent of Dune features within an assessment unit was determined from remotely sensed data (Hardaway Dune Inventory).

Shoreline Structures

Features added to the shoreline by property owners were recorded as a combination of points or lines. These features include defensive structures, constructed to protect the shoreline from erosion; offshore structures, designed to accumulate sand from longshore transport; and recreational structures, built to enhance recreational use of the water. All features were recorded by presence or absence. Point features include marinas, docks, jetties, breakwaters, and boat ramps. Linear features include bulkheads, riprap, miscellaneous, and debris. All point features were scored as discrete values. All other linear features were scored as a percentage of the linear feature within the assessment unit.

Subaqueous Resources

All subaqueous resources were identified from remotely sensed data (Submerged Aquatic Vegetation (SAV), VIMS; Oysters and Aquaculture, VMRC). For each assessment unit, percentage SAV coverage was calculated between the shoreline interface and the 2 meter depth contour or 200 meters offshore. Presence/absence of Oyster and Aquaculture sites were identified within these same areas.

Fetch

Fetch was calculated using shoreline coverage data from the Department of the Interior's National Wetland Inventory (NWI). GIS arc(s) were created for wetlands intersecting the shoreline. Coordinate geometry (COGO) was used to create short arcs in 16 wind directions (N, NNE, NE, ENE, E, ESE, SE, SSE, S, SSW, SW, WSW, W, WNW, NW, NNW). These arcs are then extended to intersect with the bathymetry and shoreline. Directions and distances are then assigned back to the wetland. If two midpoints are measured, the midpoint with the longest fetch is identified and assigned to the wetland. If there are three or more shoreline segments for a single wetland polygon, the maximum fetch and direction for each midpoint is determined. The 16 wind directions are then condensed into four quadrants (NE, SE, SW, NW). The predominant fetch direction is then determined based upon the number of points in each quadrant. The longest fetch is selected from the predominant quadrant and assigned to the wetland. If two or more quadrants have an equal number of points, then the longest fetch is selected from among those quadrants.

The assessment of wetland islands, where a single wetland is completely surrounded by open water, requires a slightly different analysis. A centroid point is established within the wetland. Arcs are created from this point and radiate out in 16 wind directions to intersect with the wetland's perimeter. From each of these intersection points, 16 additional arcs are created and extended to the nearest shoreline and 2m bathymetric contour. The arc with the longest fetch is assigned to the wetland. The direction of the arc with the longest fetch is then used to determine the distance to the 2m contour.

On the Chesapeake Bay and its tributaries, fetches greater than 1,000 m (1 km) are considered unlimited, and form the basis for the scoring system used in this model. Scores are divided into three categories: Long (≤1000m), Short (>1000m), and None.

Bathymetry

NOAA bathymetry data (-2m depth contour) was used to identify the distance from the wetland polygon to the 2m depth contour. Bathymetry was divided into 4 depth categories based on the distance that the 2m depth contour is located from the shoreline: Deep (≤100m), Moderate (>100m and <fetch), Shallow (=fetch), and None (when the wetland polygon is contiguous with the upland shoreline).

Water Quality Model

Upland Landuse

Upland Landuse was considered for the water quality model because upland areas contribute to nonpoint sources pollution through contaminated upland runoff and groundwater. Upland landuses were complied into 3 categories (natural, agricultural, and developed), which reflect general contributions of non-point source pollution. The categories were ranked (1=least important, 2=moderate importance, 3=most important) based on their relative importance for the maintenance of water quality. Natural landuse was given a value of 3 because it includes wetland, scrub-shrub, and forested cover types which are identified as contributing the least excess nutrients while also removing pollutants and retaining sediment from adjacent upland areas. Agricultural landuse was given a value of 2 because it has the potential to retain sediments, however may be associated with excess nutrient inputs. Developed landuse was given a value of 1 because it offers the lowest potential for sediment retention and nutrient removal and may increase contaminated surface runoff. A value was calculated for each assessment unit by adding the relative percentages of each landuse type multiplied by its ranking.

Riparian Landuse

Riparian Landuse was considered for the water quality model because riparian areas provide capacity for mitigating nonpoint source pollution by reducing upland runoff and intercepting groundwater. Landuses were complied into 4 categories (natural, agricultural, industrial/paved, and developed), which reflect the buffering capacity of the riparian lands. In the water quality model, the natural category was given a value of 3 because the vegetation associated with it has high buffering capacity, while the other two categories (developed and agriculture) were given a ranking of 1 because they were considered to have reduced buffering capacity due to lack of vegetation and/or excess nutrient inputs. The fourth category (Industrial) was added to the water quality model and observed by presence/absence. When this category was present, the final score was modified. If the riparian landuse was Industrial and the upland (inland) landuse was greater than 20% developed, then the score for upland landuse went to zero (regardless of the remaining 79% cover). If upland landuse was less than 20% developed, the score was reduced by 1 point. This reflects the lack of buffering value and potential for increased pollution associated with industrial sites. The 20% threshold reflects a conservative estimate based on current understanding of the adverse effects of development on water quality. A value was calculated for each assessment unit by adding the relative percentages of each landuse type multiplied by its ranking.

Forested Buffer

Forested Buffer presence was a riparian area modifier considered for the water quality model since even narrow bands of riparian trees may provide improved ecological services. Buffers were only applied in areas where the Riparian Landuse was less than 30% natural. The presence of a buffer was ranked as a 3 (high buffering capacity) and the absence of a buffer was ranked as 1 (reduced buffering capacity).

Bank Cover

Bank Cover was considered for the water quality model because vegetative cover on a bank helps to stabilize the bank, reducing sediment inputs to the waterway. Bank cover was separated into 3 categories, which reflect general contributions of non-point source pollution. The categories were ranked (1=bare, 2=partial cover, 3=total cover) based on their relative importance for the maintenance of water quality. Total cover was given a value of 3 because vegetation and structures help to prevent erosion and sediment introduction. Partial cover was given a value of 2 because a portion of the bank was unprotected or exposed, resulting in potential for erosion and sediment introduction. Bare banks were given a value of 1 because of their high potential for erosion and sediment introduction. A value was calculated for each assessment unit by adding the relative percentages of each cover category multiplied by its ranking.

Bank Stability

Bank Stability was considered for the water quality model because stable banks are less susceptible to erosion and failure, reducing sediment inputs to the waterway. Bank stability was separated into 3 categories, which reflect general contributions of non-point source pollution. The categories were ranked (1=unstable, 2=undercut, 3=stable) based on their relative importance for the maintenance of water quality. Stable banks were given a value of 3 because a lack of observed erosion suggests low potential for sediment introduction. Undercut banks were given a value of 2 because the minimal toe erosion indicated a moderate potential for sediment introduction. Unstable banks were given a value of 1 because of their high potential for continued erosion and sediment introduction. A value was calculated for each assessment unit by adding the relative percentages of each cover category multiplied by its ranking.

Shoreline Resources

Three resources were considered that provide water quality services in varying degrees: Dunes, Marsh, and *Phragmites*.

Coastal primary sand dunes serve as protective barriers from flooding and erosion resulting in decreased sediment and nutrient inputs. Marshes are transitional areas between upland and sub-aqueous lands that improve water quality and help reduce erosion by filtering groundwater and holding sediment in place. From a water quality perspective, *Phragmites* are highly productive, trapping and binding sediments, intercepting run-off and stabilizing shorelines.

These resources were ranked by their importance to water quality according to their proximity to the shoreline interface and relative opportunity for water quality improvement: Marsh and *Phragmites*=3 and Dune=2. A value was calculated for each assessment unit by adding the relative percentages of each cover category multiplied by its ranking.

Shoreline Structures

Six categories of structures were considered for their potential to impact water quality in varying degrees: Boat ramps, Marinas, Bulkheads, Riprap, Miscellaneous, and Debris (car tires, trash, appliances, etc.). The impact of structures on water quality is variable and may be positive or negative (improve or degrade water quality). Structures that stabilize shorelines (including Miscellaneous and Debris) and reduce erosion may improve water quality and were given a value of 2. Marinas and Boat ramps introduce pollutants associated with boating and therefore were assigned negative values. Presence of a Marina within an assessment unit automatically scored a –3, while boat ramps were scored based on the number of such structures within the assessment unit (public ramps = -2 each; private ramps = -1 each).

In the model, these structures may modify Bank Cover by reducing the overall water quality score if the cover is provided by a manmade structure (Bulkheads, Riprap, Miscellaneous, and Debris) rather than vegetation. This reflects the greater capacity of natural bank cover to impact water quality through the reduction of erosion and interception of runoff and groundwater.

Subaqueous Resources

Three resources were considered to provide water quality services in varying degrees: Submerged Aquatic Vegetation (SAV), Oyster Reefs, and Aquaculture. Both SAV and oysters were once prevalent throughout the Chesapeake Bay and the surrounding watersheds, however they have become increasingly rare. They both have limited capabilities to dampen waves and stabilize nearshore sediments. Oysters also remove pollutants through filtration while SAV may help reduce excess nutrients. These ecosystem services justify a high ranking for these resources. Percent SAV coverage was multiplied by a factor of 6 in the model. Oyster and aquaculture were identified by presence/absence and contributed scores of 3 and 2 respectively which was added to the total model score for each assessment unit.

Habitat Model*

Riparian Landuse

Riparian Landuse was considered for the habitat model because riparian vegetation can provide essential habitat for terrestrial and avian species. Land uses were compiled into 3 categories (natural, agricultural, and developed), which reflect habitat value. In the model, the natural landuse type was given a value of 6 because it provides native or unaltered habitat for terrestrial and avian species. The agricultural landuse type was assigned a value of 4 because it was considered to be in an altered state which may result in reduced availability of suitable habitat. The developed landuse type was assigned a score of 2 because disturbance from development has likely resulted in reduced available habitat. In areas where combined agricultural and development landuse was greater than 50% of the assessment unit, the entire unit was considered developed. This reflects the concept of a minimal threshold of disturbance beyond which habitat is critically compromised. A value was calculated for each assessment unit by adding the relative percentages of each landuse type multiplied by its ranking

Forested Buffer

Forested Buffer presence was a riparian area modifier considered for the habitat model since even narrow bands of riparian trees may provide some habitat. Buffers were only applied in areas where the Riparian Landuse was less than 30% natural. The presence of a buffer was ranked as a 3 because the buffer provides a habitat corridor on otherwise developed land. The absence of a buffer was assigned a rank of 1.

Shoreline Resources

Four resources were considered that provide habitat in varying degrees; the resources are: Dunes, Beaches, Marsh, and *Phragmites*. 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. Marshes are transitional areas between upland and subaqueous lands. They provide habitat (food and shelter) for both aquatic and terrestrial animals such as blue crabs, small fish and marsh birds. *Phragmites* marshes grow in a wide range of intertidal and nearshore areas. They generally represent a monotypic community, which limits their habitat value relative to more diverse communities. The non-native variety of *Phragmites* may be highly competitive, displacing native marsh vegetation. In the habitat model, the resources were ranked by their relative habitat value as follows: Dunes=1, Beach and Marsh=3 and *Phragmites*=2.

Shoreline Structures

Seven categories of structures were considered for their potential to impact habitat in varying degrees: Boat ramps, Marinas, Bulkheads, Breakwaters, Miscellaneous, Debris, and Jetties. Generally, structures have an adverse impact on habitat because they displace native environments or interrupt the marine-terrestrial interface. The two exceptions are Breakwaters and Jetties, which involve the placement of stone in the subaqueous zone. These structures may provide attachment surfaces for aquatic animals such as oysters, barnacles, and jingle shells. Jetties and breakwaters were assigned the only positive values (+1) because they provide some habitat value. Boat ramps were given a value of –1 when a single ramp was present and a value of –2 when more than one ramp was present. Marinas were assigned a value of –3. All other structures were assigned a value of –3 multiplied by the linear extent of the feature expressed as a percentage of the assessment unit.

Subaqueous Resources

Two resources were considered to provide habitat in varying degrees; they are Submerged Aquatic Vegetation (SAV) and Oyster Reefs. Both SAV and oysters were once prevalent throughout the Chesapeake Bay and the surrounding watersheds, however they have become increasingly rare. They are important components of the coastal ecosystem, providing critical forage and nursery habitat for a wide variety of estuarine species. These ecosystem services justify a high ranking for these resources. Rankings were positive and relatively high for both categories. Percent SAV coverage was multiplied by a factor of 6 in the model. Oysters were identified by presence/absence and contributed a score of 2, which was added to the total model score for each assessment unit.

*When riparian landuse within an assessment unit was 100% natural and no shoreline structures were present, an additional 10 points was added to the final model score to reflect the diversity of habitat supported by this unaltered landscape condition. It also reflects the relative scarcity of unaltered shorelines along the entire coastline.

Shoreline Risk Model

Fetch

Fetch was used as an element in the model because it influences the wave climate in a given reach. Fetch >1,000 m receive the highest score = 1. Though larger fetches are common, no justification could be provided for weighting greater distances higher. In the shoreline protection model, the highest score (1) represents that tidal wetlands are most valued where they are subject to the greatest potential wave energy. Tidal wetlands located where fetch distances are less than 1,000m receive a moderate value score = 0.5. Our rationale is that fetches <1,000m are less significant and are more easily mitigated.

Bathymetry

Bathymetry data was used in the shoreline protection model because, in addition to fetch, shallow water habitat, water depths < 2m as defined by the US Army Corps of Engineers, can enhance the ability of tidal wetlands to provide shoreline protection by forcing waves to break offshore, thereby dispersing a significant portion of the wave's energy before it reaches the shoreline. Where the distance to the 2m depth contour is less than or equal to 100m, the wetland receives the highest score (1) because the nearshore exerts less wave-reducing influence, and the wetlands are therefore more valuable in providing shoreline protection. Distances greater than 100m are scored progressively lower, to represent the increased ability of nearshore bathymetry to enhance the shoreline protection of tidal wetlands. Therefore, when the distance to the 2m contour is greater than 100m, the nearshore shallow water habitat contributes significantly to wave reduction and the role of the wetland to provide shoreline protection is reduced.

WATER QUALITY ELEMENTS

Shoreline Element	**Model Values**	**Element Rules**
Upland Landuse		
natural	3	% landuse * value added to score
agriculture	2	% landuse * value added to score
developed	1	% landuse * value added to score
Riparian Landuse		
natural	3	% landuse * value added to score
agriculture	1	% landuse * value added to score
developed	1	% landuse * value added to score
industrial or paved	0 or -1	Score for upland landuse when present: 0 if developed > 20%, subtract 1 if dev ≤ 20%
Forest Buffer		
yes	3	Applied when Ag + Dev RL ≥ 70% of buffer area
no	1	Applied when Ag + Dev RL ≥ 70% of buffer area
Bank Cover		
bare	3	% of unit * value added to score
partial	2	% of unit * value added to score
total	1	% of unit * value added to score
Bank Stability		
stable	3	% of unit * value added to score
undercut	2	% of unit * value added to score
unstable	1	% of unit * value added to score
Shoreline Resources		
dunes	2	% of unit * value added to score
marsh	3	% of unit * value added to score
phragmites	3	% of unit * value added to score
Shoreline Structures		
boat ramp	-1, -2	public= -2, private = -1: all ramps totaled
marina	-3	if present -3 added to score
bulkhead	2	
riprap	2	
miscellaneous	2	Structure modifies bank cover: subtract (% of unit
debris	2	* value) from cover value
Subaqueous Resources		
SAV	3	(% area * 3) added to score
oyster	3	if present +3 added to score
aquaculture	2	if present +2 added to score

HABITAT MODEL ELEMENTS

Shoreline Element	Model Values	Element Rules
Riparian Landuse		
natural	6	When % (Ag+Dev) > 50% = developed
agriculture	4	When % (Ag+Dev) > 50% = developed
developed	2	
industrial or paved	0	
Forest Buffer		
Yes	3	Applied when Ag + Dev RL ≥ 70% of buffer area
No	1	Applied when Ag + Dev RL ≥ 70% of buffer area
Shoreline Resources		
dunes	1	% of unit * value added to score
beach	3	% of unit * value added to score
marsh	3	% of unit * value added to score
phragmites	2	% of unit * value added to score
Shoreline Structures		
boat ramp	-1, -2	One ramp= -1, > one ramp = -2, added to score
marina	-3	If present -3 added to score
bulkhead	-3	% of unit * value added to score
miscellaneous	-3	% of unit * value added to score
debris	-3	% of unit * value added to score
jetty	1	If present +1 added to score
breakwater	1	If present +1 added to score
Subaqueous Resources		
SAV	6	(% area * 6) added to score
oyster	2	if present +2 added to score

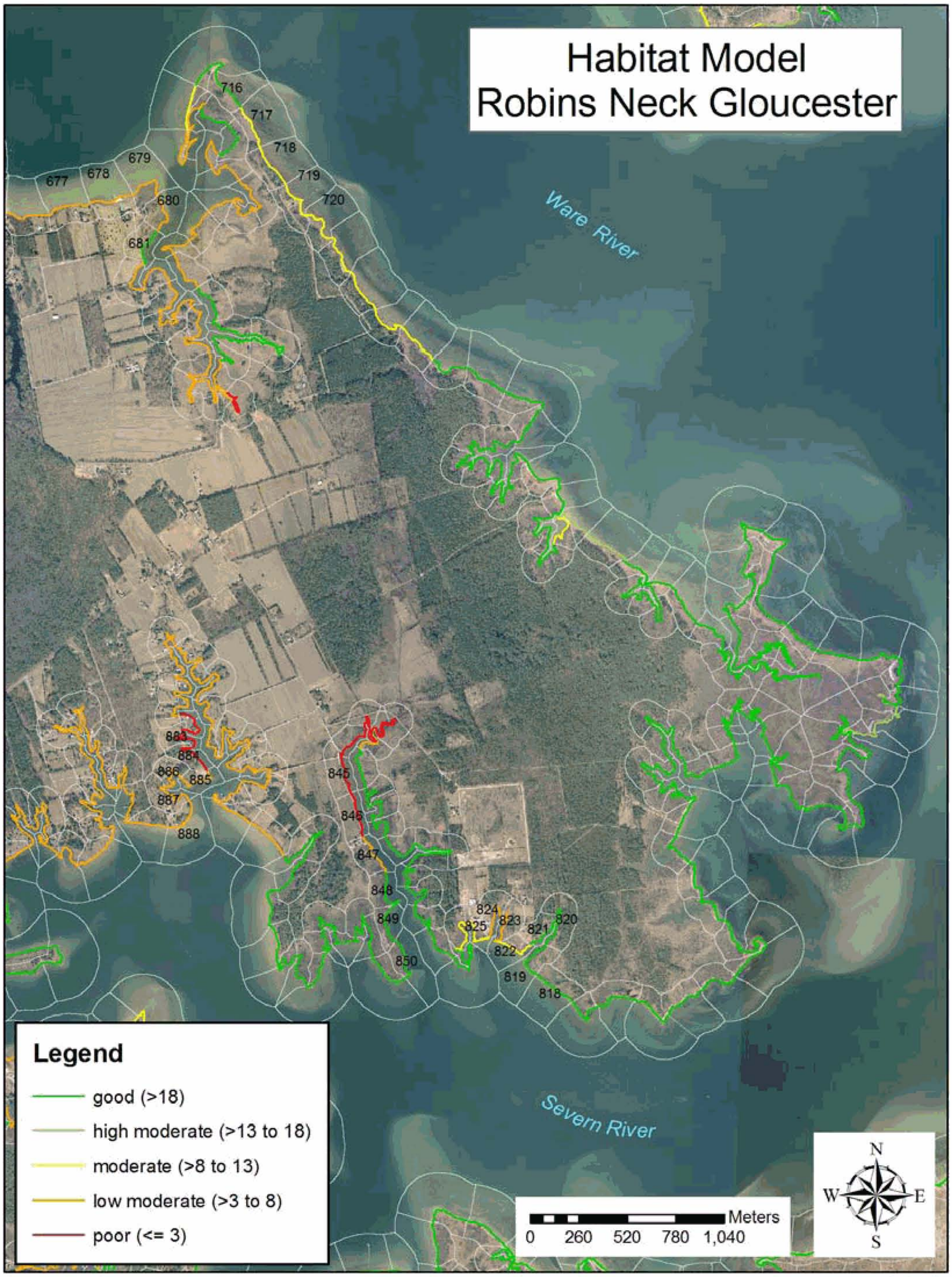
Add 10 points Assessment unit with LU = 100 % natural, no structures

SHORELINE RISK MODEL ELEMENTS

Shoreline Element	Model Values	Element Rules
Bathymetry (Distance to 2 mile contour)		
Deep	1.0	Unit value added to score
Moderate	0.5	Unit value added to score
Shallow	0.25	Unit value added to score
None	0	Unit value added to score
Fetch		
Long	1.0	Unit value added to score
Short	0.5	Unit value added to score
None	0	Unit value added to score


Water Quality Model
Robins Neck Gloucester
Ware River
Severn River
677
678
679
680
718
719
720
883
884
885
886
887
888
845
846
847
848
849
850
824
825
823
821
820
822
819
818
Legend
high moderate (>13 to 18)
moderate (>8 to 13)
low moderate (>3 to 8)
poor (< = 3)
N
W
E
S
Meters
0 270 540 810 1,080

11/10/06


Habitat Model
Robins Neck Gloucester
Ware River
Severn River
716
717
718
719
720
677
678
679
680
681
883
884
885
886
887
888
845
846
847
848
849
850
824
825
823
821
820
822
819
818
Legend
good (>18)
high moderate (>13 to 18)
moderate (>8 to 13)
low moderate (>3 to 8)
poor (<= 3)
Meters
0 260 520 780 1,040
N
W
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S

11/10/06

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 9<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 depend 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. 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 agriculture, residential, or wetland 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 is mixed. 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, 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, then 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 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). Eason 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). 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 subject to change in response to bank failure and storms and it may conceal bank slumping or other evidence of bank failure.

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 (WRP 2000).

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. 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). Byrne et al 1982 estimated that shore erosion accounted for 6% of the suspended sediment in the total inorganic sediment budget for the Virginia portion of Chesapeake Bay. 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.

These studies did not include stable banks for comparison to verify differences in nutrient and sediment loadings between unstable and stable banks. 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). 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). The tractive force removal of material at the toe of the bank can be a failure mechanism (Wetlands Engineering Handbook, 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 for nutrient and sediment loading in the Chesapeake Bay than the erosion rate indicated by bank appearance. This effect depends on bank height. An unstable low bank might not actually have greater 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 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 nonmarina 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.

Part 3: ***Annotated Bibliography***
Integrated Guidance Model

1. Upland Landuse

Angermeier, P.L., Wheeler, A.P., and Rosenberger, A.E. 2004. A conceptual framework for assessing impacts of roads on aquatic biota. Fisheries. 29(12): 19-29.
A review and development of a two-dimensional framework to organize impacts of roads on aquatic biota. One dimension recognizes three phases of road development, each with distinctive ranges of spatial and temporal scales. The second dimension recognizes five classes of environmental impacts associated with road development.

Atasoy, M., Palmquist, R.B., and Phaneuf, D.J. 2006. Estimating the effects of urban residential development on water quality using microdata. J. Environ. Manage. 79(4): 399-408.
Both the density of residential land use and the rate of land conversion have a negative impact on water quality. The impacts of these non-point sources are found to be larger in magnitude than those from urban point sources.

Bayen, S., Kee Lee, H., and Philip Obbard, J. 2007. Exposure and response of aquacultured oysters, *Crassostrea gigas*, to marine contaminants. Environ. Res. 103(3): 375-382.
Low growth rates in oysters exposed to polluted waters, and significant differences in the burden of persistent organic pollutants (POPs) were discovered. On a positive note, the effects of pollution on oysters were found to be reversible.

Bilkovic, D.M., Roggero, M., Hershner, C.H., and Havens, K.H. 2006. Influence of land use on macrobenthic communities in nearshore estuarine habitats. Estuaries Coasts. 29(6B): 1185-1195.
For biotic indices applied in the nearshore, the highest scores were associated with forested watersheds (W-value, B-IBI sub(N)). Ecological thresholds were identified with nonparametric change- point analysis, which indicated a significant reduction in B-IBI sub(N) and W- value scores when the amount of developed shoreline exceeded 10% and developed watershed exceeded 12%, respectively.

Bosch, D.J., Lohani, V.K., Dymond, R.L., Kibler, D.F., and Stephenson, K. 2003. Hydrological and fiscal impacts of residential development: Virginia case study. J. Water Resour. Plann. Manage. 129(2): 107-114.
Low density development has the greatest hydrological impact due to highest per capita impervious area. Higher density settlements reduce hydrological impacts but bear a high cost to local governments in reduced property tax revenues.

Boward, D., Kazyak, P., Stranko, S., Hurd, M., Prochaska, A., and Environmental Protection Agency, Washington, DC (USA). Office of Research and Development. 1999.

From the mountains to the sea: The state of Maryland's freshwater streams. U.S. Environmental Protection Agency, Center for Environmental Research Information Cincinnati, OH 45268 (USA).
The report describes the impact of urbanization on fish and benthic macroinvertebrate communities (using indices of biotic integrity), herpetofauna, and stream temperatures. The report also describes the extent of physical habitat degradation, including riparian buffer conditions. Other topics include acidification; nutrient enrichment; biodiversity; introduced fish; and rare, threatened, and endangered fish species.

Brooks, R.P., D.H. Wardrop, K.W. Thornton, D. Whigham, C. Hershner, M.M. Brinson and J.S. Shortle, eds. 2006. Integration of ecological and socioeconomic indicators for estuaries and watersheds of the Atlantic Slope. Final Report to U.S. Environmental Protection Agency STAR Program, Agreement R-82868401, Washington, DC. Prepared by the Atlantic Slope Consortium, University Park, PA. 96pp.+ attachments (CD).

Burcher, C.L., and Benfield, E.F. 2006. Physical and biological responses of streams to suburbanization of historically agricultural watersheds. J. N. Am. Benthol. Soc. 25(2): 356-369.
Stormflow total suspended solids were significantly lower and substrate inorganic matter content was significantly higher in streams influenced by suburban development. Fish taxa richness and the density of nonguarding fishes were significantly higher in sites in suburban watersheds than in sites in agricultural watersheds. Biotic assemblages at sites in suburban watersheds were distinct from those at sites in agricultural watersheds. Suburbanization near historically agricultural southern Appalachian streams induces subtle changes to inorganic sediment dynamics, substrate composition, and fish and macroinvertebrate assemblage structure.

Cardoso, P.G., Pardal, M.A., Lilleboe, A.I., Ferreira, S.M., Raffaelli, D., and Marques, J.C. 2004. Dynamic changes in seagrass assemblages under eutrophication and implications for recovery. J. Exp. Mar. Biol. Ecol. 302(2): 233-248.
Eutrophication related modifications include a decrease in species diversity along the *eutrophication gradient, an increase in detritivores and a decline in herbivores together with a significant increase in small deposit-feeding polychaetes. In the long term, sustained eutrophication of this estuary is expected to lead to complete replacement of seagrass habitat by unvegetated coarser sediments, occasionally covered by green macroalgal blooms and dominated by opportunistic invertebrate taxa.*

Carpenter, S.R., Benson, B.J., Biggs, R., Chipman, J.W., Foley, J.A., Golding, S.A., Hammer, R.B., Hanson, P.C., Johnson, P., Kamarainen, A.M., Kratz, T.K., Lathrop, R.C., McMahon, K.D., Provencher, B., Rusak, J.A., Solomon, C.T., Stanley, E.H., Turner, M.G., Zanden, M., Wu, C., and Yuan, H. 2007. Understanding regional change: A comparison of two lake districts. BioScience, Vol. 57(4): 323.
We compared long-term change in two lake districts, one in a forested rural setting and the other in an urbanizing agricultural region, using lakes as sentinel ecosystems. Biotic changes such as habitat loss, species invasions, and poorer fishing were prevalent in the rural region, and lake hydrology and biogeochemistry responded to climate trends and

landscape position. Similar biotic changes occurred in the urbanizing agricultural region, where human-caused changes in hydrology and biogeochemistry had conspicuous effects.

DeLuca, W. V., C. E. Studds, L. L. Rockwood, and P. P. Marra. 2004. Influence of land us on the integrity of marsh bird communities of the Chesapeake Bay, USA. Wetlands 24:837-847.

Dougherty, M., Dymond, R.L., Grizzard, T.J.,Jr, Godrej, A.N., Zipper, C.E., and Randolph, J. 2006. Quantifying long-term NPS pollutant flux in an urbanizing watershed. J. Environ. Eng. 132(4): 547-554.
Compared an urbanized watershed in D.C with 3 mixed Ag-forested watersheds. The urbanized watershed had higher annual NPS sediment and nutrient fluxes linked to increased soil disturbance from urban construction and increased storm volumes resulting from increased mean impervious percent. Higher NPS pollutant fluxes during the growing season, indicate a seasonal impact of replacing vegetated cover with impervious surface.

Dow, C.L., Arscott, D.B., and Newbold, J.D. 2006. Relating major ions and nutrients to watershed conditions across a mixed-use, water-supply watershed. J. N. Am. Benthol. Soc. 25(4): 887-911.
Anthropogenic land uses primarily defined ion and nutrient baseflow chemistry patterns at regional and watershed levels. Individual analyses indicated no dominant watershed-scale landscape attribute that could be used to predict instream inorganic chemistry concentrations, and no single ion or nutrient was identified as the best indicator of a given anthropogenic land use.

Dukes, M.D., and Evans, R.O. 2006. Impact of agriculture on water quality in the North Carolina middle coastal plain. J. Irrig. Drain. Eng. 132(3): 250-262.
Nutrient and sediment concentrations increased along the streams draining the agricultural lands. Surface water was impacted dramatically by seepage from an anaerobic swine wastewater lagoon.

Fisher, T.R., Hagy, J.D.,III, Boynton, W.R., and Williams, M.R. 2006. Cultural eutrophication in the Choptank and Patuxent estuaries of Chesapeake Bay. Limnol. Oceanogr. 51(1): 435-447.
Hydrochemical modeling and land-use yield coefficients suggest that current input rates are 4- 20 times higher for N and P than under forested conditions existing 350 yr ago. Sewage is a major cause of increased nutrients in the Patuxent; agricultural inputs dominate in the Choptank. These loading have caused (1) increased nutrients, phytoplankton, and turbidity; (2) decreased submerged grasses due to higher turbidity and epiphyton shading; and (3) bottom-water hypoxia due to respiration of excess organic matter.

Gage, M.S., Spivak, A., and Paradise, C.J. 2004. Effects of land use and disturbance on benthic insects in headwater streams draining small watersheds north of Charlotte, NC. Southeast. Nat. 3(2): 345-358.
Increasing development north of Charlotte, NC, threatens aquatic life in streams by reducing riparian zones and increasing runoff. Runoff, sedimentation from erosion, and poor construction practices are principal sources of pollution. Sensitive taxa were found in streams with extensively forested watersheds, but were nonexistent in extensively developed watersheds. Disturbances occurring in streams caused declines in diversity, often eliminating sensitive taxa.

Gray, M.J., and Smith, L. 2005. Influence of land use on postmetamorphic body size of playa lake amphibians. J. Wildl. Manage. 69(2): 515-524.
Agricultural land use may indirectly affect the body size of amphibians by altering the hydroperiods of nearby wetlands and influencing amphibian densities—both factors which can limit the larval and post-metamorphic growth rates of amphibians. Thus, if cultivation of landscapes surrounding wetlands negatively influences post-metamorphic body size of amphibians, restoration of native grasslands surrounding playa wetlands may help prevent local amphibian declines.

Groffman, P.M., Dorsey, A.M. and Mayer, P.M. 2005. N processing within geomorphic structures in urban streams. J. N. Am. Benthol. Soc. 24(3): 613-625.
Denitrification potential was highest in organic debris dams and organic-rich gravelbars—structures with high organic matter content. Organic debris dams in suburban streams had higher denitrification than debris dams in the forested reference stream, likely because of higherNO_3^- concentrations in suburban streams. However, such denitrifying structures as organic debris dams may be difficult to maintain in urban streams because of high storm flows and downstream displacement.

Hagen, E.M., Webster, J.R., and Benfield, E.F. 2006. Are leaf breakdown rates a useful measure of stream integrity along an agricultural landuse gradient? J. N. Am. Benthol. Soc. 25(2): 330-343.
Leaf breakdown rates were related to landuse category (forested, light agriculture, moderate agriculture, and heavy agriculture) but did not differ significantly. Nutrient concentration, temperature, and sedimentation increased, and dissolved O_2 decreased along the landuse gradient from forest to heavy agriculture.

Henning, L.A. and W.D. Edge. 2003. Riparian bird community structure in Portland, Oregon: Habitat, urbanization, and spatial scale patterns. The Condor 105:288-302.

Inwood, S.E., Tank, J.L., and Bernot, M.J. 2005. Patterns of denitrification associated with land use in 9midwestern headwater streams. J. N. Am. Benthol. Soc. 24(2): 227-245.
The effects of land use on the relationships among denitrification, NO sub (3) super (-)-N, dissolved organic C (DOC),and other environmental parameters were examined in 9 headwater streams (3 each in forested, agricultural, and urban-dominated sub-watersheds). Agricultural streams had high concentrations of NO sub(3)super(-)-N, DOC, soluble reactive P, and NH sub(4) super(+)-N, whereas forested streams had the

lowest concentrations of these nutrients, and urban streams generally exhibited intermediate concentrations. Sediment denitrification rates were highest in agricultural streams and lowest in forested streams throughout the study period. Despite higher denitrification rates in agricultural and urban streams compared to forested streams, sediment denitrification removed a smaller proportion of the stream NO sub(3) super(-)-N load in agricultural and urban streams relative to forested streams.

Jennings, M.J., Emmons, E.E., Hatzenbeler, G.R., Edwards, C., and Bozek, M.A. 2003. Is littoral habitat affected by residential development and land use in watersheds of Wisconsin lakes? Lake Reserv. Manage. 19(3): 272-279.
Looked at differences in nearshore littoral zone habitat among lakes with different amounts of residential development and different patterns of watershed land use. Quantity of woody debris, emergent vegetation and floating vegetation decreased at developed sites and in lakes with greater cumulative lakeshore development density. Littoral sediments contained more fine particles at developed sites and in lakes with greater development density. Sediment composition, quantity of vegetation, and woody debris were weakly associated with differences in watershed land use. Cumulative changes to watersheds and riparian zones are associated with measurable differences in littoral habitat that may not be detectable at smaller scales.

Kaushal, S.S., Lewis, W.M., Jr, and McCutchan, J.H., Jr. 2006. Land use change and nitrogen enrichment of a rocky mountain watershed. Ecol. Appl. 16(1): 299-312.
The biotic capacity of headwater ecosystems to assimilate increases in inorganic N from residential development may be insufficient to prevent nitrogen enrichment over considerable distances and multiple aquatic ecosystems downstream. Nitrogen enrichment was studied in a Rocky Mountain watershed undergoing rapid expansion of population and residential development. From a watershed perspective, total loading of N to the Blue River catchment from septic and municipal wastewater (2 kg times ha super(-1) times yr super(-1)) is currently less than the amount from background atmospheric sources (3 kg times ha super(-1) times yr super(-1)).

Kimbrough, K.L., and Dickhut, R.M. 2006. Assessment of polycyclic aromatic hydrocarbon input to urban wetlands in relation to adjacent land use. Mar. Pollut. Bull. 52(11): 1355-1363.
The relationship between polycyclic aromatic hydrocarbons (PAHs) in wetland surface sediments and adjacent land use was assessed in the Elizabeth River, VA. Wetlands adjacent to parking lots and petroleum industrial sites exhibited the highest PAH concentrations of all wetlands examined. Overall, commercial land uses had the highest PAH concentrations and automotive sources dominated (52- 69%) PAH input to wetland surface sediments irrespective of adjacent land use.

Kratzer, E.B., Jackson, J.K., Arscott, D.B., Aufdenkampe, A.K., Dow, C.L., Kaplan, L.A., Newbold, J.D., and Sweeney, B.W. 2006. Macroinvertebrate distribution in relation to land use and water chemistry in New York City drinking-water-supply watersheds. J. N. Am. Benthol. Soc. 25(4): 954-976.

Macroinvertebrate communities were examined in conjunction with landuse and water-chemistry variables. Urbanized macroinvertebrate communities varied primarily with specific conductance, population density, and agricultural and urban land use, but communities were not classified as impaired along these gradients. In other areas, conditions ranged from forested to urban, and distinctive communities were associated with point-source discharges, road density, and lake outlets, resulting in impaired macroinvertebrate communities.

Kuehl, S., Alexander, C., Carter, L., Gerald, L., Gerber, T., Harris, C., McNinch, J., Orpin, A., Pratson, L., Syvitski, J., and Walsh, J.P. 2006. Understanding sediment transfer from land to ocean. EOS Trans. Am. Geophys. Union. 87(29): 281.
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Leithold, E.L., Perkey, D.W., Blair, N.E., and Creamer, T.N. 2005. Sedimentation and carbon burial on the northern California continental shelf: The signatures of land-use change. Cont. Shelf Res. 25(3): 349-371.
The burial of organic carbon (OC) on continental margins is strongly coupled to the supply and accumulation of inorganic mineral particles. The record reveals a 6-11-fold increase in the rate of sediment accumulation on the mid-shelf beginning about 1955, and a concomitant decrease in grain size and increase in flood-layer preservation. At the same time, the age of buried wood fragments abruptly decreased and their stable carbon isotopic composition became enriched in super(13)C. It is argued that these changes can be explained largely as the result of altered land use in the Eel watershed. Increases in mass wasting and input of bedrock material following the onset of intensive industrial logging in the Eel watershed may have resulted in a lower loading of terrestrial plant OC in the clay fraction deposited after.

Lerberg, S. B., A. F. Holland, and D. M. Sanger. 2000. Responses of tidal creek macrobenthic communities to the effects of watershed development. Estuaries 23:838-853.

Lever, M.A. and Valiela, I. 2005. Response of microphytobenthic biomass to experimental nutrient enrichment and grazer exclusion at different land-derived nitrogen loads. Mar. Ecol. Prog. Ser. 294: 117-129.
Effects of eutrophication on the relative importance of nutrients and macroherbivores as controls of microphytobenthic standing crop show that nitrogen + phosphorus addition increased sediment chlorophyll alpha (chl alpha) content (herein used as a proxy for biomass) by a similar magnitude across estuaries. Grazer exclusion also increased chl a, but to a different extent across estuaries: the magnitude of the response increased with increasing nitrogen loading rates. We found no interactions between nutrients and grazing.

Limburg, K. E., and R. E. Schmidt. 1990. Patterns of fish spawning in Hudson River tributaries: response to an urban gradient? Ecology 71:1238-1245.

Locke, B.A., Cherry, D.S., Zipper, C.E., and Currie, R.J. 2006. Land use influences and ecotoxicological ratings for upper clinch river tributaries in Virginia. Arch. Environ. Contam. Toxicol. 51(2): 197-205.
The Clinch River system; all tributary watersheds are predominately forested, but agricultural, mining, and developed land uses (urban, transportation) are also present. ETRs indicated that the tributaries draining mining-influenced watersheds had greater potential impact on the mainstem than those draining agricultural or forested watersheds, and the presence of developed land uses had no significant relationship with ETRs.

Lussier, S.M., Enser, R.W., Dasilva, S.N., and Charpentier, M. 2006. Effects of habitat disturbance from residential development on breeding bird communities in riparian corridors. Environ. Manage. 38(3): 504-521.
This study assessed the relationship among land use, riparian vegetation, and avian populations at two spatial scales. Bird guilds were correlated with riparian vegetation metrics, percent impervious surface, and percent residential land use, revealing patterns of breeding bird distribution. The number of intolerant species predominated below 12% residential development and 3% impervious surface, whereas tolerant species predominated above these levels.

Mallin, M.A., Johnson, V.L., Ensign, S.H., and MacPherson, T.A. 2006. Factors contributing to hypoxia in rivers, lakes and streams; eutrophication of freshwater and marine ecosystems. Limnol. Oceanogr. 51(1): 690-701.
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Meyer, J.L., Paul, M.J., and Taulbee, W.K. 2005. Stream ecosystem function in urbanizing landscapes. J. N. Am. Benthol. Soc. 24(3): 602-612.
Ecologists have described an urban stream syndrome with attributes such as elevated nutrients and contaminants, increased hydrologic flashiness, and altered biotic assemblages. Both NH4 and soluble reactive P uptake velocities decreased as indicators of urbanization (i.e., % of catchment covered by high-intensity urban development) increased. The amount of fine benthic organic matter (FBOM) also decreased with increasing urbanization, and uptake velocities were directly related to FBOM. Uptake velocities were not related to ecosystem metabolism (gross primary production [GPP], community respiration [CR], or net ecosystem production). Measures of ecosystem function responded differently to urbanization: ecosystem metabolism was not correlated with indicators of urbanization, although breakdown rate of Acer barbatum leaves was positively correlated and nutrient uptake velocities were negatively correlated with indicators of urbanization. Elevated nutrient concentrations associated with urbanization are usually attributed to increased inputs from point and non-point sources; our results indicate that concentrations also may be elevated because of reduced rates of nutrient removal.

Paul, M. J. and J. L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32:333-365.

Moore, A.A., and Palmer, M.A. 2005. Invertebrate biodiversity in agricultural and urban headwater streams: Implications for conservation and management. Ecological Applications: A Publication of the Ecological Society of America. 15(4): 1169-1177.
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Nelson, P.A., Smith, J.A., and Miller, A.J. 2006. Evolution of channel morphology and hydrologic response in an urbanizing drainage basin. Earth Surf. Process. Landforms. 31(9): 1063-1079.
The Dead Run catchment in Baltimore County, Maryland, has undergone intense urbanization since the late 1950s. Trend analyses of discharge records in Dead Run show that urban development and stormwater control measures have had significant impacts on the hydrologic response of the catchment.

Nerbonne, B.A., and Vondracek, B. 2001. Effects of local land use on physical habitat, benthic macroinvertebrates, and fish in the whitewater river, Minnesota, USA. Environ. Manage. 28(1): 87-99.
Best management practices (BMPs) have been developed to address soil loss and the resulting sedimentation of streams, but information is lacking regarding their benefits to stream biota. We compared instream physical habitat and invertebrate and fish assemblages from farms with BMP to those from farms with conventional agricultural practices. Sites were classified by upland land use (BMP or conventional practices) and riparian management (grass, grazed, or wooded buffer). Physical habitat characteristics differed across buffer types, but not upland land use, using an analysis of covariance, with buffer width and stream as covariates. Stream sites along grass buffers generally had significantly lower percent fines, embeddedness, and exposed streambank soil, but higher percent cover and overhanging vegetation when compared with sites that had grazed or wooded buffers.

Opperman, J.J., Lohse, K.A., Brooks, C., Kelly, N.M., and Merenlender, A.M. 2005. Influence of land use on fine sediment in salmonid spawning gravels within the Russian River basin, California. Can. J. Fish. Aquat. Sci. 62(12): 2740-2751.
Relationships between land use or land cover and embeddedness, a measure of fine sediment in spawning gravels, were examined at multiple scales. Agricultural and urban land uses and road density were positively associated with embeddedness, while the opposite was true for forest cover. Land use within a more restricted riparian corridor generally did not relate to embeddedness, suggesting that reach-scale riparian protection or restoration will have little influence on levels of fine sediment.

Price, K., and Leigh, D.S. 2006. Comparative water quality of lightly- and moderately-impacted streams in the southern Blue Ridge Mountains, USA. Environ. Monit. Assess. 120(1-3): 269-300.
For less-developed regions like the Blue Ridge Mountains, data are limited that link basin-scale land use with stream quality. Two pairs of lightly-impacted (90-100% forested) and moderately-impacted (70-80% forested) sub-basins were identified for comparison. Statistically significantly higher mean values of suspended and dissolved

solids, nitrate, specific conductivity, turbidity, and temperature were observed in the moderately impacted streams versus the lightly impacted streams in both pairs, while dissolved oxygen levels were lower in the moderately-impacted streams. No significant differences were demonstrated in orthophosphate or ammonium concentration. However, the demonstration that moderate reductions in forest cover are associated with stream water quality degradation carries important implications for stream management in this rapidly developing mountainous region.

Rodriguez, W., August, P.V., Wang, Y., Paul, J.F., Gold, A., and Rubinstein, N. 2007. Empirical relationships between land use/cover and estuarine condition in the northeastern United States. Landscape Ecol. 22(3): 403-417.
Patterns of coastal urban and agriculture gradients were measured and their relationship with indicators of estuarine condition was modeled statistically. Moderate ($0.4 < | r | < 0.7$) to strong (| r | greater than or equal to 0.7) linear associations were found between total urban area and measures of estuarine condition. Within regions, total urban area was positively associated with Silver ($r = 0.59$), Cadmium ($r = 0.65$), and Mercury ($r = 0.47$) in Cape Cod, and inversely related to DO ($r = -0.65$) in the Hudson/Raritan region. Total area of agriculture showed a moderate association with Arsenic in Cape Cod, but no other associations were found in the other two regions.

Roy, A.H., Freeman, M.C., Freeman, B.J., Wenger, S.J., Ensign, W.E., and Meyer, J.L. 2005. Investigating hydrologic alteration as a mechanism of fish assemblage shifts in urbanizing streams. J. N. Am. Benthol. Soc. 24(3): 656-678.
Stream biota in urban and suburban settings are thought to be impaired by altered hydrology; however, it is unknown what aspects of the hydrograph alter fish assemblage structure and which fishes are most vulnerable to hydrologic alterations in small streams. Increased imperviousness was positively correlated with the frequency of storm events and rates of the rising and falling limb of the hydrograph (i.e., storm "flashiness") during most seasons. Increased duration of low flows associated with imperviousness only occurred during the autumn low-flow period, and this measure corresponded with increased richness of lentic tolerant species. Altered storm flows in summer and autumn were related to decreased richness of endemic, cosmopolitan, and sensitive fish species, and decreased abundance of lentic tolerant species. Species predicted to be sensitive to urbanization, based on specific life-history or habitat requirements, also were related to stormflow variables and% fine bed sediment in riffles. Overall, hydrologic variables explained 22 to 66% of the variation in fish assemblage richness and abundance.

Scheuerell, M.D and D.E Schindler. 2004. Lakeshore residential development alters the spatial distribution of fishes. Ecosystems 7(1): 98-106.

Scott, D., Harvey, J., Alexander, R., and Schwarz, G. 2007. Dominance of organic nitrogen from headwater streams to large rivers across the conterminous United States. Global Biogeochem Cycles. 21(1).
We find that organic nitrogen is the dominant nitrogen pool within rivers across most of the United States and is significant even in basins with high anthropogenic sources of nitrogen.

Simon, N.S., Bricker, O.P., Newell, W., McCoy, J., and Morawe, R. 2005. The distribution of phosphorus in popes creek, VA, and in the Pocomoke River, MD: Two watersheds with different land management practices in the Chesapeake Bay basin. Water, Air, Soil Pollut. 164(1-4): 189-204.
This paper compares phosphorus (P) concentrations in sediments from two watersheds, one with (Pocomoke), and one without (Popes Creek), intensive animal agriculture. Concentrations of total P and P extracted with 1N HCl are significantly larger in main-stem bottom sediments from the Pocomoke River than in main-stem bottom sediments from Popes Creek.

Smith, C.M., and Wachob, D.G. 2006. Trends associated with residential development in riparian breeding bird habitat along the Snake River in Jackson Hole, WY, USA: Implications for conservation planning. Biol. Conserv. 128(4): 431-446.
Throughout North America, bird population declines may be attributable to loss of habitat on the breeding grounds. They determined the effects of housing densities on avian community parameters, guilds, individual species distributions, and environmental variables. Landscape-level features were most affected by residential development and trends associated with increasing housing densities, such as anthropogenic habitat fragmentation primarily structured local bird communities. Overall species richness and diversity declined with increasing residential development. Neotropical migrant species were most negatively impacted and consistently declined in proportional representation on forested plots as residential development densities increased. Food generalists, ground gleaners, and avian nest predators all increased with increasing residential development.

Smith, S.V., D.P. Swaney, L. Talaue-McManus, J.D. Bartley, P.T. Sandhei, C.J. McLaughlin, V.C. Dupra, C.J. Crossland, R.W. Buddemeier, B.A. Maxwell and F. Wulff. 2003. Humans, hydrology, and the distribution of inorganic nutrient loading to the ocean. Bioscience 53(3):235-245.

Snyder, C.D., Young, J.A., Villella, R., and Lemarie, D.P. 2003. Influences of upland and riparian land use patterns on stream biotic integrity. Landscape Ecol. 18(7): 647-664.
We found that index of biological integrity (IBI) scores were strongly associated with extent of urban land use in individual catchments. Sites that received ratings of poor or very poor based on IBI scores had > 7% of urban land use in their respective catchments. Habitat correlations suggested that urban land use disrupted flow regime, reduced water quality, and altered stream channels. In contrast, we found no meaningful relationship between agricultural land use and IBI at either whole-catchment or riparian scales. Urban land use was more disruptive to biological integrity in catchments with steeper channel slopes.

Torbick, N.M., Qi, J., Roloff, G.J., and Jan Stevenson, R. 2006. Investigating impacts of land-use land cover change on wetlands in the Muskegon river watershed, Michigan, USA. Wetlands, Vol. 26, no. 4. 26(4): 1103-1113.
No Abstract?

Ulseth, A.J., and Hershey, A.E. 2005. Natural abundances of stable isotopes trace anthropogenic N and C in an urban stream. J. N. Am. Benthol. Soc. 24(2): 270-289.
Important ecological services of low-order streams are greatly affected by urbanization. Thus, specific influences of point sources of N could be distinguished in food web components. Nonpoint sources and stormwater influenced seston delta super (15)N during storm events, but these sources could not be distinguished in consumers by using natural abundances of stable isotopes.

Walsh, C.J.,Roy, A.H., Feminella, J.W., Ottingham, P.D., Groffman, P.M., and Morgan, R.P. 2005. The urban stream syndrome: Current knowledge and the search for a cure. J. N. Am. Benthol. Soc. 24(3): 706-723.
Symptoms of the urban stream syndrome include a flashier hydrograph, elevated concentrations of nutrients and contaminants, altered channel morphology, and reduced biotic richness, within creased dominance of tolerant species. Most impacts can be ascribed to a few major large-scale sources, primarily urban stormwater runoff delivered to streams by hydraulically efficient drainage systems. Other stressors, such as combined or sanitary sewer overflows, wastewater treatment plant effluents, and legacy pollutants (long-lived pollutants from earlier land uses) can obscure the effects of stormwater runoff. Most research on urban impacts to streams has concentrated on correlations between instream ecological metrics and total catchment imperviousness. Recent research shows that some of the variance in such relationships can be explained by the distance between the stream reach and urban land, or by the hydraulic efficiency of stormwater drainage.

Wang, L., J. Lyons and P. Kanehl. 1997. Influences of watershed land use on habitat quality and biotic integrity in Wisconsin streams. Fisheries 22:6-12.

2. Riparian Landuse

Riparian Land Use - Water Quality

Correll, D.L. 1997. Buffer Zones and water quality protection: general principles. Pages 7-20 In: Buffer Zones: Their Processes and Potential in Water Protection. The Proceedings of the International Conference on Buffer Zones September 1996. Edited by N.E. Haycock et al.
This paper is a review of world literature on riparian buffer zones with emphasis on streams and water quality. One of the conclusions was that grass or dense herbaceous vegetation is most effective at removing or trapping particulates from surface flow but only if there is sheet flow. Woody vegetation may be more effective at removing nitrate from groundwater due to the availability of organic carbon deep in the soil for microbial denitrification. For effective nitrate and acidity removal, groundwater must move at a slow speed and at a shallow depth to be intercepted by root zone.

Hefting, M.M., R. Bobbink, M.P. Janssens. 2006. Spatial Variation in Denitrification and N20 emission in relation to nitrate removal efficiency in a N-stressed riparian buffer zone. Ecosystems. 9(4): 550-563.
Riparian zone management that has as its goal an increase of the denitrification activity in buffers is worthy from the perspective of water quality improvement; however, the risk of N_2O emission remains inevitable. Simultaneous minimization of N_2O emissions is only possible if riparian zone management is combined with source-directed measures designed to drastically reduce the nitrate concentration in agricultural runoff.

Henry, M., H. Stevens, & K.W. Cummins. 1999. Effects of Long-Term Disturbance on Riparian Vegetation and In-Stream Characteristics. Journal of Freshwater Ecology. 14(1): 1-18.
This paper evaluated the influence of riparian disturbance on 26 stream variables in six Pennsylvania tributaries. Their primary conclusion was that anthropogenic disturbance destabilized stream ecosystem function. Specifically, agricultural land uses caused detrital processing rates to be less predictable than in forested streams. Mature woodland streams had more stable channel substrates and populations of intolerant species. Disturbed sites tended to have more silt and higher populations of Gammurus sp. Land use in particular had a great effect on detritus processing and storage.

Klapproth, J.C. & J.E. Johnson. 2000. Understanding the Science behind Riparian Buffers: Effects on Water Quality. Virginia Cooperative Extension Publication 420-151.
This is one paper in a series that summarizes the current scientific understanding of riparian buffers in Virginia. Streams in the coastal plain of Virginia particularly benefit from forested riparian buffers due to low-gradient topography and shallow aquifers that allows for effective sediment and nutrient removal. Both grass and forested buffers are effective for sediment removal, forested buffers also offer greater resistance during heavy floods. Buffer widths of 50-100 feet are recommended for sediment trapping. The ability of buffers to remove nutrients is highly variable and is tied to soils and hydrology.

Mayer, P.M., S.K. Reynolds, Jr., M.D. McCutchen and T.J. Canfield. Riparian Buffer Width, Vegetative Cover, and Nitrogen Removal Effectiveness: A Review of Current Science and Regulations. EPA/600/R-05/118. Cincinnati, OH, U.S. Environmental Protection Agency, 2006
Identifies causation & trends in the relationship between buffer width & nitrogen removal capacity extracted from peer-reviewed studies with empirical data on buffer effectiveness. Their main conclusions were that buffer widths of 10-50 m are effective nutrient filters, narrower buffers (5-6 m) may still reduce subsurface nitrate by up to 80%, but buffer widths > 50 m are the most effective. Also, vegetation characteristics did not determine level of pollutant removal, soil type, ground and surface water flow patterns & subsurface biogeochemistry were more determining than vegetation type or width.

Lowrance, R. & J.M. Sheridan. 2005. Surface Runoff Water Quality in a Managed Three Zone Riparian Buffer. J. Environ. Qual. 34: 1851-1859.

This study measured surface runoff volumes and nutrient concentrations and loads in a 3-zone riparian buffer system consisting of a grass strip at the edge of an agricultural field, a managed forest and an unmanaged forest by the stream. The largest percentage reducing of incoming nutrient load (at least 65% for all nutrients) took place in the grass zone because of the large decrease in flow.

Mankin, K.R., D.M. Ngandu, C.J. Barden, S.L. Hutchinson, W.A. Geyer. 2007. Grass-shrub riparian buffer removal of sediment, phosphorus, and nitrogen from simulated runoff. Journal of the American Water Resources Association. 43(4).
This study assessed the effectiveness of different types of grass-shrub buffers for reducing the pollutant load from agricultural fields in Iowa where farmers more readily accept this type of buffer than forested buffers. Their findings suggest that grass-shrub buffers were very efficient in removing sediments, N, and P with the removal efficiencies strongly linked to infiltration, which is consistent with other studies. They also found that a buffer width of only 8 m provided water quality improvement, particularly if adequate infiltration is achieved.

Palone, R. S. & A.H. Todd (editors). 1997. Chesapeake Bay riparian handbook: a guide for establishing & maintaining forest buffers. USDA Forest Service. NA-TP-02-97. Radnor, PA. Revised 1998.
This comprehensive guide provides information about the function, design, establishment, and management of riparian forest buffers in the Chesapeake Bay watershed. Different land use considerations are included for agriculture, forestry and urban settings. They recommend a flexible buffer width design that incorporates floodplain areas, undevelopable steep slopes, and adjacent wetlands.

Schoonover, J.E., K.W.J. Williard, J.J. Zaczek, J.C. Mangun & A.D. Carver. 2006. Agricultural Sediment Reduction by Giant Cane and Forest Riparian Buffer. Water, Air, and Soil Pollution Vol. 169(1-4): 303-315.
High infiltration rates of riparian soils was primary factor controlling sediment trapping, more than particle settling (i.e. variable sediment concentration, surface deposition) Greatest sediment reduction in forest during fall with leaf litter, no significant reduction of sediment mass during the spring at all, during remainder of year only significant at 6.6m into buffer, cane buffer sediment mass reductions significant by 3.3m during all seasons.

Riparian Land Use - Habitat

Diamond, J.M., D.W. Bressler, V.B. Serveiss. 2002. Assessing relationships between land uses and the decline of native mussels, fish and macroinvertebrates in the Clinch and Powell River Watershed, USA. Environmental Toxicology and Chemistry. 21(6): 1147-1155.
This paper was investigating the cause of mussel population declines in mountain streams of Virginia. Limited analyses in two subwatersheds suggested that urban and agricultural land uses within a specified riparian corridor were more related to mussel species richness and fish IBI than land uses in the entire catchments. Based on land uses

within a riparian corridor of 200 m × 2 km for each biological site in the watershed, fish IBI was inversely related to percent cropland and urban area and positively related to pasture area.

Klapproth, J.C. & J.E. Johnson. 2000. Understanding the Science behind Riparian Buffers: Effects on Plant and Animal Communities. Virginia Cooperative Extension Publication 420-152.
This is one paper in a series that summarizes the current scientific understanding of riparian buffers in Virginia. A review of scientific papers suggests that the vegetation complexity of riparian buffers supports a wide variety of wildlife and the loss of complex structure will reduce overall diversity. Forested riparian buffers adjacent to agricultural fields and in developed settings provide habitat to non-riparian dependent wildlife. Studies of bird communities in intensive agriculture areas suggest that riparian areas are very important habitats. Even narrow forest buffers support songbirds compared to herbaceous riparian vegetation, particularly short distance migrants. They also found variable results and conclusions in the literature regarding the terrestrial corridor function. They also concluded that species diversity & biomass of aquatic benthic communities decrease significantly as forest cover is removed from riparian buffers, primarily due to increased water temperature.

Lee, P., C. Smyth, and S. Boutin. 2004. Quantitative review of riparian buffer width guidelines from Canada and the United States. J. Environ. Manag. 70: 165–180.
This review of riparian buffer guidelines compared management trends with ecological recommendations for terrestrial and aquatic species. Buffer widths for most jurisdictions were found to be adequate to protect aquatic biota and habitats but were generally less than recommended for terrestrial communities. Most notably core habitat for medium and large mammals and birds were wider than most current guidelines. Fixed-width buffer requirements to accommodate all possible habitats would potentially require buffers wider than is warranted by local site conditions.

Mahan, C.G. & T.J. O'Connell. 2005. Small Mammal Use of Suburban and Urban Parks in Central Pennsylvania. Northeastern Naturalist. 12(3): 307-314.
This study evaluated the importance of riparian buffers for small mammals in Pennsylvania parks. Mature riparian forest sites contained more small mammal species. Species richness and diversity were lowest in parks containing manicured habitats and surrounded by human-modified landscapes.

Snyder, C.D., J.A. Young, R. Villella, D.P. Lemarie. 2003. Influences of upland and riparian land use patterns on stream biotic integrity. Landscape Ecology 18: 647-664.
This paper evaluated urban, agriculture & forested land use patterns at the riparian-reach scale & riparian-site scale. At the riparian-site scale, 2 of the 3 fish cover measures, LWD & undercut banks, exhibited strong positive correlations with forest land cover and negative associations with agriculture land use. Riffle-pool ratios also positively associated with forest and negatively with agriculture land use. Riparian land use patterns failed to explain a significant amount of the remaining variation in site IBI scores suggesting that forested buffer zones were of little value in mitigating the

deleterious effects of urban land use on fish communities. Efforts to moderate impacts of urban land use by protecting riparian buffers may not be sufficient to maintain biotic integrity.

Teels, B.M., C.A. Rewa, J. Myers. 2006. Aquatic Condition Response to Riparian Buffer Establishment. Wildlife Society Bulletin. 34 (4): 927-935.
This study evaluated the effects of recently established riparian buffers on the aquatic condition of agricultural streams in Northern Virginia. They collected baseline data that indicated influences of different land uses. The baseline IBI scores corresponded negatively to percent cropland and percent pasture and positively to percent nonagricultural land. The IBI also responded negatively to urban land-use patterns, particularly in a rapidly developing watershed where dramatic declines in the visible stream condition (SVAP) and the IBI were observed for both the restored and reference sites.

Van Holt, T., D.M. Murphy, L. Chapman. 2006. Local and Landscape Characteristics in the Great Swamp, New York. Northeastern Naturalist. 13(3): 353-374.
This study used local and landscape models to predict fish assemblages in the Great Swamp, New York region, which was undergoing rapid development. Four different scales were evaluated including reach, segment, network and watershed scales. No single model best predicted fish assemblages. The segment scale (100-m buffer for 1 km upstream) was recommended as the best predictor. They also suggested that forest cover is essential to protect fish assemblages. The percent forest positively influenced percent intolerant species and the IBI metrics for benthic insectivores, terete minnows, and dominant species. Any diminishing of forest land cover in favor of increasing agriculture, residential, or wetland negatively influenced the percent intolerant species.

Wigley, T.B. & M.A. Melchiors. Date? (circa 1995 based on studies conducted 1980-1993) Wildlife Habitat and Communities in Streamside Management Zones: A Literature Review for the Eastern United States. Excerpted from Riparian Ecosystems in the Humid U.S.: Functions, Values and Management.
This paper included a literature review of studies related to buffer width for streamside management zones (SMZ) in forestry settings. One of the main findings was that the corridor function of riparian buffers is variable and it is not clear if SMZs provide this function. They also suggest that the existing database is not adequate for SMZ width policy development and that most of the studies came to conclusions that were not supported by the data.

3. Bank Cover

Chesapeake Bay Program (CBP). 2005. Sediment in the Chesapeake Bay and management issues: tidal erosion processes. Prepared by the Tidal Sediment Task Force of the Sediment Workgroup under the Chesapeake Bay Program, Nutrient Subcommittee, May 2005.
This document provides background information on sediment processes and data to determine the effects of tidal sediment in a watershed and can serve as a general targeting tool for shoreline management and sediment controls that can improve water quality. It also suggests that tidal erosion should be viewed as an integral part of the natural ecosystem processes in the Bay and a necessary component of a properly functioning ecosystem, while also considering that excessive sediment loads can be detrimental to water quality.

Dierks, S. 2007. Not all green space is created equal: the hydrologic benefits of native landscapes. Stormwater, the journal for surface water quality professionals, March/April 2007 8(2).
This paper investigates the transformative effects of native plants and landscapes on soil properties that affect the soil's strength, growing medium capacity, and hydrologic properties. Two case studies were used to demonstrate the impact of native plants on soil properties in agricultural watersheds. Native plants have the capacity to improve structure and infiltration over time in all soils, including clays, but additional research beyond typical runoff studies is needed to quantify these effects.

Easson, G, Yarbrough, LD. 2002. The effects of riparian vegetation on bank stability. Environmental & Engineering Geoscience, 8(4) pp.247-260, Nov 2002.
This investigation quantifies root tensile strength of the sweet gum (Liquidambar styraciflua) in a cohesive, fine-grained, primarily loess-derived fluvial material in northern Mississippi. Estimating root reinforcement and root-soil matrix interactions allows for the determination of whether bank vegetation is beneficial or detrimental. The modeling results showed a contrast between root-reinforced and unreinforced soil. When no root reinforcement existed, the slope failed marginally. When simulated root reinforcement of 20 kPa was applied, the slope was shown to be completely stable.

Hardaway, C.S., Jr. G.R. Thomas, J.B. Glover, J.B. Smithson, M.R. Berman, A.K. Kenne. 1992. Bank erosion study. Special report in Applied Marine Science and Ocean Engineering, No. 319. Virginia Institute of Marine Science. April 1992.
This study analyzed fastland bank erosion along 383 miles of tidal shoreline in the Virginia portion of Chesapeake Bay. The authors discovered that the erosion tends to be greatest where a vegetation cover is absent. Sediment volume loading was considered to be halted where defensive shoreline structures were installed. For the James River, 18% of the shoreline studied was defended with structures between 1985-1990 which reduced the annual estimated nutrient load by 5%.

Klapproth, J.C. & J.E. Johnson. 2000. Understanding the science behind riparian buffers: Effects on water quality. Virginia Cooperative Extension publication 420-151.
This is one paper in a series that summarizes the current scientific understanding of riparian buffers in Virginia. Streams in the coastal plain of Virginia particularly benefit from forested riparian buffers due to low-gradient topography and shallow aquifers that allows for effective sediment and nutrient removal. Both grass and forested buffers are effective for sediment removal, forested buffers also offer greater resistance during heavy floods. Buffer widths of 50-100 feet are recommended for sediment trapping. The ability of buffers to remove nutrients is highly variable and is tied to soils and hydrology.

Langland, M. and T. Cronin, editors. 2003. A summary report of sediment processes in Chesapeake Bay and watershed. U.S. Department of the Interior, U.S. Geological Survey, Water-Resources Investigations Report 03-4123.
This report summarizes the most relevant studies concerning sediment sources, transport and deposition in the Chesapeake Bay watershed and estuary, sediments and relation to water clarity, and provides an extensive list of references for those wanting more information.

National Research Council. 2007. Mitigating shore erosion on sheltered coasts. The National Academies Press, Washington DC. 174 pp.
This report examines the impacts of shoreline management on sheltered coast environments and strategies to minimize potential negative impacts to adjacent or nearby coastal resources. It includes background information about sediment processes and basic physical laws that control erosion, such as conservation of sediment mass and control of sediment fluxes.

Ott, R A. 2000. Factors affecting stream bank and river bank stability, with an emphasis on vegetation influences. An annotated bibliography compiled for the Region III Forest Practices Riparian Management Committee.
This literature review focuses on factors affecting stream bank and riverbank stability in Alaska, with an emphasis on vegetation influences. Vegetation has been shown to stabilize banks of rivers and streams in some systems. It has been suggested that the harvest of riparian timber in interior Alaska can increase riverbank erosion rates. It has also been suggested that timber harvest near watercourses will decrease the supply of large woody debris (LWD) that is recruited into a river through natural erosion processes.

Palace, M.W., J.E. Hannawald, L.C. Linker, and G.W. Shenk. 1998. Chesapeake Bay watershed model applications & calculation of nutrient & sediment loadings - Appendix H: Tracking best management practice nutrient reductions in the Chesapeake Bay. Program report of the modeling subcommittee. August, 1998. Chesapeake Bay Program Office, Annapolis, MD.
This document provides a summary of the methodologies used in tracking nutrient reduction goals for the Chesapeake Bay watershed. Both tidal structural and non-structural erosion controls reduce the nitrogen, phosphorus, and suspended sediment loads by an estimated 75 percent.

Palone, R. S. & A.H. Todd (editors). 1997. Chesapeake Bay riparian handbook: a guide for establishing & maintaining forest buffers. USDA Forest Service. NA-TP-02-97. Radnor, PA. Revised 1998.
This comprehensive guide provides information about the function, design, establishment, and management of riparian forest buffers in the Chesapeake Bay watershed. Different land use considerations are included for agriculture, forestry and urban settings. They recommend a flexible buffer width design that incorporates floodplain areas, undevelopable steep slopes, and adjacent wetlands.

Wynn, T.M., S. Mostaghimi, J.A. Burger, A.A. Harpold, M.B. Henderson, L. Henry. 2004. Variation in root density along stream banks. J. of Environ. Qual. 33(6): 2030-2039.
The purpose of this study was to determine the type and density of vegetation that provides the greatest protection against erosion by determining the density of roots in stream banks of the Appalachian Mountains. Root length density with depth and aboveground vegetation density were measured for both forested and herbaceous sites. The results indicated that forested vegetation may provide better protection against stream bank erosion because it had a significantly higher concentration of fine roots, as compared to herbaceous sites.

4. Bank Stability

Byrne, R.J. C.H. Hobbs, III, M.J. Carron. 1982. Baseline sediment studies to determine distribution, physical properties, sedimentation budgets and rates in the Virginia portion of the Chesapeake Bay. US EPA R806001010, Virginia Institute of Marine Science, Gloucester Point, VA. p.155
This study estimated that shore erosion accounted for 6% of the suspended sediment in the total inorganic sediment budget for the Virginia portion of Chesapeake Bay. The authors also gave a measured value for the sand component from shore erosion as $4.0x10^5$ *metric tons per year.*

Chesapeake Bay Program (CBP). 2005. Sediment in the Chesapeake Bay and management issues: tidal erosion processes. Prepared by the Tidal Sediment Task Force of the Sediment Workgroup under the Chesapeake Bay Program, Nutrient Subcommittee, May 2005.
This document provides background information on sediment processes and data to determine the effects of tidal sediment in a watershed and can serve as a general targeting tool for shoreline management and sediment controls that can improve water quality. It also suggests that tidal erosion should be viewed as an integral part of the natural ecosystem processes in the Bay and a necessary component of a properly functioning ecosystem, while also considering that excessive sediment loads can be detrimental to water quality.

Davis, J.E., S.T. Maynord, J. McCormick and T.J. Olin. 2000. Shoreline protection and erosion control. In Section 5, Establishing Proper Hydrologic Conditions, Wetlands Engineering Handbook. Compiled by TJ Olin et al, prepared for U.S. Army Corps of Engineers Wetlands Research Program, ERDC/EL TR-WRP-RE-21, March 2000, 719 pp.
This chapter in the wetlands engineering handbook describes methods for erosion control and shoreline protection for created and restored wetland sites. Various causes and mechanisms of erosion and bank failure are described. Erosion protection alternatives and design considerations are also provided.

Hardaway, C.S., Jr. G.R. Thomas, J.B. Glover, J.B. Smithson, M.R. Berman, A.K. Kenne. 1992. Bank erosion study. Special report in Applied Marine Science and Ocean Engineering, No. 319. Virginia Institute of Marine Science. April 1992.
This study analyzed fastland bank erosion along 383 miles of tidal shoreline in the Virginia portion of Chesapeake Bay. The authors discovered that the volume of eroded material was more significant than the erosion rate, which depends on bank height (i.e. high bank with low erosion rate may contribute larger volume than low bank with higher erosion rate.) Sediment volume loading was reduced where defensive shoreline structures were installed. For the James River, 18% of the shoreline studied was defended with structures between 1985-1990, which reduced the annual estimated nutrient load by 5%.

Ibison, N.A., C.W. Frye, J.E. Frye, C.L. Hill, N.H. Burger. 1990. Sediment and nutrient contributions of selected eroding banks of the Chesapeake Bay estuarine system.

Virginia Department of Conservation and Recreation, Division of Soil and Water Conservation, Shoreline Programs Bureau. January 1990.
This study evaluated 14 eroding banks to examine sediment and nutrient inputs from tidal shoreline erosion. They estimated 1.37 million pounds per year of nitrogen and 0.94 million pounds per year of phosphorus is entering the Bay ecosystem through shoreline erosion. This quantity of nitrogen is 5.2% of the "controllable" nonpoint source nitrogen load, as defined by the 1987 Chesapeake Bay Agreement (Chesapeake Executive Council, 1988).

Ibison, N.A., J.C. Baumer, C.L. Hill, N.H. Burger, J.E. Frye. 1992. Eroding bank nutrient verification study for the lower Chesapeake Bay. Virginia Department of Conservation and Recreation, Division of Soil and Water Conservation, Shoreline Programs Bureau. February 1992.
This research verified and expanded a 1990 study by examining 44 additional eroding banks along the lower Chesapeake Bay for grain size, nitrogen & phosphorus loads. The impacts of land use on nutrient loading characteristics were also examined (active farms, fallow farms, wooded and rural residential). They discovered that the mean nutrient loading rates from the previous study were approximately twice those calculated in this study. The reaches sampled had lower erosion rates and eroded soil volumes.

Klapproth, J.C. & J.E. Johnson. 2000. Understanding the science behind riparian buffers: Effects on water quality. Virginia Cooperative Extension publication 420-151.
This is one paper in a series that summarizes the current scientific understanding of riparian buffers in Virginia. Streams in the coastal plain of Virginia particularly benefit from forested riparian buffers due to low-gradient topography and shallow aquifers that allows for effective sediment and nutrient removal. Both grass and forested buffers are effective for sediment removal, forested buffers also offer greater resistance during heavy floods. Buffer widths of 50-100 feet are recommended for sediment trapping. The ability of buffers to remove nutrients is highly variable and is tied to soils and hydrology.

Langland, M. and T. Cronin, editors. 2003. A summary report of sediment processes in Chesapeake Bay and watershed. U.S. Department of the Interior, U.S. Geological Survey, Water-Resources Investigations Report 03-4123.
This report summarizes the most relevant studies concerning sediment sources, transport and deposition in the Chesapeake Bay watershed and estuary, sediments and relation to water clarity, and provides an extensive list of references for those wanting more information.

National Research Council. 2007. Mitigating shore erosion on sheltered coasts. The National Academies Press, Washington DC. 174 pp.
This report examines the impacts of shoreline management on sheltered coast environments and strategies to minimize potential negative impacts to adjacent or nearby coastal resources. It includes background information about sediment processes and basic physical laws that control erosion, such as conservation of sediment mass and control of sediment fluxes.

Ott, R A. 2000. Factors affecting stream bank and river bank stability, with an emphasis on vegetation influences. An annotated bibliography compiled for the Region III Forest Practices Riparian Management Committee.
This literature review focuses on factors affecting stream bank and riverbank stability in Alaska, with an emphasis on vegetation influences. Vegetation has been shown to stabilize banks of rivers and streams in some systems. It has been suggested that the harvest of riparian timber in interior Alaska can increase riverbank erosion rates. It has also been suggested that timber harvest near watercourses will decrease the supply of large woody debris (LWD) that is recruited into a river through natural erosion processes.

US Army Corps of Engineers (USACOE). 1990. Chesapeake Bay shoreline erosion study, feasibility report, October 1990.
The purpose of this study was to evaluate shoreline protection measures that will protect both land and water resources of the Chesapeake Bay from the adverse effects of continued erosion. Both structural and nonstructural solutions and their suitability for different shoreline types were examined. Five demonstration projects were also constructed and monitored for about two years, including two in Maryland and three in Virginia.

Wilson, G V; Periketi, R K; Fox, G A; Dabney, S M; Shields, F D; Cullum, R F Soil properties controlling seepage erosion contributions to stream bank failure**.** Earth Surface Processes and Landforms, vol.32, no.3, pp.447-459, Mar 2007
The objective of this study was to determine the impact of soil properties on seepage erosion and the resulting stream bank failure along the banks of a deeply incised stream in northern Mississippi. It was suggested that the USDA-ARS Stream bank Stability model demonstrated the increase in instability of banks due to undercutting by seepage erosion, but failed to account for the sediment loss due to sapping for stable banks and overestimated the sediment loads for failed banks.

Wynn, T.M., S. Mostaghimi, J.A. Burger, A.A. Harpold, M.B. Henderson, L. Henry. 2004. Variation in root density along stream banks. J. of Environ. Qual. 33(6): 2030-2039.
The purpose of this study was to determine the type and density of vegetation that provides the greatest protection against erosion by determining the density of roots in stream banks of the Appalachian Mountains. Root length density with depth and aboveground vegetation density were measured for both forested and herbaceous sites. The results indicated that forested vegetation might provide better protection against stream bank erosion because it had a significantly higher concentration of fine roots, as compared to herbaceous sites.

5. Shoreline Resources

Marshes - Water Quality

Bricker S. B. 1996. Retention of sediment and metals by Narragansett Bay subtidal and marsh environments; an update; Transport and accumulation processes of contaminants in estuarine and coastal waters. The Science of the Total Environment. 179: 27-46.
Study of Narragansett Bay suggests that salt marshes retain virtually all incoming sediment.

Christiansen T. 1999. Sediment deposition on a tidal salt marsh. Thesis Publ. Date: 1998, 134 pp University of Virginia, Charlottesville, VA, USA
Student thesis suggests that Spartina alterniflora has a significant dampening effect on the turbulence of the flow, promoting deposition of suspended particles. Again concludes that more sediment is deposited on marsh edge than interior, with 27% sediment deposited on the marsh surface by storms; the rest deposited during normal high spring tides.

Davis J. L., B. Nowicki, and C. Wigand. 2004. Denitrification in fringing salt marshes of Narragansett Bay, Rhode Island, USA. Wetlands. 24:870-878.
Results of this field survey show the potential of New England fringe salt marshes to intercept and transform land-derived nitrogen loads, but calls for a better understanding of the natural and anthropogenic factors controlling denitrification and net N losses.

Fisher J., M. C. Acreman. 2004. Wetland nutrient removal: a review of the evidence. Hydrology and Earth System Sciences. 8:673-685.
Survey of data from 57 wetlands from around the world showed that the majority of wetlands reduced nutrient loading Marshes were also shown to be slightly more effective at nutrient reduction than riparian zones.

Gleason M. L., D. A. Elmer, N. C. Pien, and J. S. Fisher. 1979. Effects of stem density upon sediment retention by salt marsh cord grass, Spartina alterniflora Loisel. Estuaries. 2(4), 271-273.
Greatest accretion of sediment results from higher density low marsh.

Kaplan W., I. Valiela, and M. Teal. 1979. Denitrification in a salt marsh ecosystem. Limnology and Oceanography. 24:726-734.
Seasonal denitrification measured in Great Sippewissett Marsh, Mass. The rate of denitrification was related to temperature and was highest in the wettest habitats. 60 % of denitrification occurred in the creek bottoms. Suggest greater rates of dentrification than fixation, indicating a significant source of atmospheric nitrogen. Upshot for water quality is less nitrogen in the water column.

MacCrimmon H. R. 1980. Nutrient and sediment retention in a temperate marsh ecosystem. Internationale Revue der Gesamten Hydrobiologie. 65:719-744.

11-month study of a marsh to determine its retention capability for incoming nitrogen, phosphorus, silica, and suspended solids from the Wye River, Ont., agricultural watershed. Results indicated that the marsh is a substantial reservoir for nutrients, sediments.

Morse, J.L., P. Megonigal, and M.R. Walbridge. 2004. Sediment nutrient accumulation and nutrient availability in two tidal freshwater marshes along the Mattaponi River, Virginia, USA. Biogeochem. 69: 175-206.
Sediment accumulation is greater in downstream, channelward marsh edge as opposed to upstream or interior marsh because of close proximity to turbid water.

Neubauer S. C., I. C. Anderson, J. A. Constantine, and S. A. Kuehl. 2002. Sediment deposition and accretion in a Mid-Atlantic (U.S.A.) tidal freshwater marsh. Estuarine, Coastal and Shelf Science. 54:713-727.
A study of sediment deposition and accretion rates in a fresh marsh indicated that deposition was greatest at the creek margin rather than in marsh interior. Seems to indicate most sediment comes from water column and not from upland sources. Also concluded the deposition was not from resuspension/erosion of existing marsh but was from outside sources. Rapid mineralization rates contributed to a decrease in accretion relative to deposition, with the remainder resulting from erosion.

Oviatt C. A., S. W. Nixon. 1975. Sediment resuspension and deposition in Narragansett Bay. Estuar.Coast.Mar.Sci. 3:201-217.
Study concluded that sediment activity increase as the sampling points moved from the head to the mouth of the estuary. Although organics of deposited materials decreased from head to mouth of estuary. most deposited material was from resuspension of bottom sediments rather than fresh inputs from the water column. This seems to indicate the importance of the upper reaches of estuaries (tidal creeks) for trapping organic laden sediments.

van Proosdij D., R. G. D. Davidson-Arnott, and J. Ollerhead. 2006. Controls on spatial patterns of sediment deposition across a macro-tidal salt marsh surface over single tidal cycles. Estuarine, Coastal and Shelf Science. 69:64-86.
A field study examining the spatial patterns of sediment deposition across a salt marsh surface in the Bay of Fundy. Describes the area around the mean high water level as a transition zone for processes of sediment transport and deposition, with most deposition taking place around this region. Somewhat limited in direct application to local Virginia marshes because of the huge difference in tide range.

Reddy K. R., P. M. Gale. 1994. Wetland processes and water quality: a symposium overview. Journal of Environmental Quality. 23:875-877.
Review paper states that wetlands act as buffer between upland and aquatic areas and protect aquatic systems from upland environments through sedimentation and filtration of runoff providing environments for nutrient assimilation.

Marshes - Habitat

Erwin R. M. 1995. The ecology of cormorants: Some research needs and recommendations. Colonial Waterbirds. 1:240-246.
Review of the significance of Cormorants (Phalacrocorax auritus) in coastal systems and the need for more research.

Erwin R. M., G. M. Haramis, D. G. Krementz, and S. L. Funderburk. 1993. Resource protection for waterbirds in Chesapeake Bay. Environmental Management. 17:613-619.
Describes waterbird usage in the Bay and need for more focus on habitat conservation in Bay cleanup efforts. Describes habitat concerns for American black ducks (Anas rubripes), great blue herons (Ardea herodias), and other associated wading birds, wood ducks (Aix sponsa), canvasbacks (Aythya valisineria) and redheads (Aythya americana), and for bald eagles (Haliaeetus leucocephalus)

Erwin R. 1996. Dependence of waterbirds and shorebirds on shallow-water habitats in the mid-Atlantic coastal region: An ecological profile and management recommendations. Estuaries. 19(2A):213-219.
Describes the importance of salt and brackish marsh shallow water areas as habitat for waterbirds (waterfowl, colonially nesting wading and seabirds, ospreys [Pandion haliaetus], and bald eagles [Haliaeetus leucocephalus]) and shorebirds (sandpipers, plovers, and relatives).

Hurd, L. E., G. W. Smedes, et al. (1979). "Ecological Study of a Natural-Population of Diamondback Terrapins (Malaclemys-Terrapin-Terrapin) in a Delaware Salt-Marsh." Estuaries. 2(1): 28-33.
A 2-year study of a population of Diamondback Terrapin in a salt marsh in Delaware. Estimated size of the terrapin population from this study indicates that terrapins may be important components of the marsh food web.

Oviatt C. A., S. W. Nixon. 1973. The demersal fish of Narragansett Bay: an analysis of community structure, distribution and abundance. Estuar.Coast.Mar.Sci. 1:361-378.
Monthly samples of demersal fish resulted in 9000 individuals representing 99 spp Winter flounder, Pseudopleuronectes americanus was most abundant.

Pomeroy L. R.,[1925-], R. G. Wiegert. 1981. The Ecology of a salt marsh / edited by L.R. Pomeroy.
Review of salt marsh ecology and the flora and fauna of the salt marsh.

Smalley, A.E. 1960. Energy flow of a salt marsh grasshopper population. Ecology. 41(4): 672-677.
Classic paper on the energetics of a salt marsh invertebrate at Eugene Odum's Sapelo Island research sites. Often sited in reviews of animals that inhabit salt marshes as Orchelimum is considered a dominant primary consumer in Spartina marshes

Wiegert, R.G. and B.J. Freeman. 1990. Tidal salt marshes of the southeast Atlantic Coast: a community profile. U.S. Fish Widl. Serv., Biol. Rep. 85 (7.29). 70 pp.
This publication series by U.S. Fish and Wildlife Services issues reports on research and inventories, and the effects of land-use on various fish and wildlife resources. This volume by University of Georgia researchers looks at the ecology of tidal marshes in the southeast U.S. Contains lists of vascular plants found in salt marshes as well as invertebrates, fish, reptiles, birds, and mammals that inhabit southeastern salt marshes and in specific community types.

Phragmites - Water Quality

Rooth J. E., J. Court Stevenson, and J. C. Cornwell. 2003. Increased sediment accretion rates following invasion by Phragmites australis: the role of litter. Estuaries. 26:475-483.
"Greater rates of mineral and organic sediment trapping were associated with the P. australis community in both a subsiding creek bank marsh (34 g times m super(-2) times day super(-1) in P. australis vs. 18 g times m super(-2) times day super(-1) in Spartina spp.) and a laterally eroding marsh (24 g times m super(-2) times day super(-1) in P. australis vs. 15 g times m super(-2) times day super(-1) in Spartina spp.). Litter accumulation in P. australis stands is responsible for the higher depositional pattern observed. Additionally, below ground accumulation in P. australis communities (as much as 3 mm in 6 months) appears to substantially increase substrate elevation over relatively short time periods. Thus P. australis may provide resource managers with a strategy of combating sea-level rise and current control measures fail to take this into consideration."

Rooth J. E., J. C. Stevenson. 2000. Sediment deposition patterns in Phragmites australis communities: Implications for coastal areas threatened by rising sea-level. Wetlands Ecology and Management. 8:173-183.
Phragmites was found to enhance rates of marsh accretion and sheds some doubt on the view of Phragmites invasion as a purely undesirable change in the face of sea level rise .Implications for water quality and erosion protection.

Phragmites - Habitat

Able K. W., S. M. Hagan. 2003. Impact of common reed, Phragmites australis, on essential fish habitat: Influence on reproduction, embryological development, and larval abundance of mummichog (*Fundulus heteroclitus*). Estuaries. 26:40-50.
Phragmites invasion in brackish marshes may be having deleterious effects on fish populations and possibly on predators that prey upon F. heteroclitus, and as a result, marsh secondary production.

Able K. W., S. M. Hagan. 2000. Effects of common reed (Phragmites australis) invasion on marsh surface macrofauna: Response of fishes and decapod crustaceans. Estuaries. 23:633-646.

As a result of these observations, with different sampling techniques, it appears there is an overall negative effect of Phragmites on larval and small juvenile fish but less or no effect on larger fish and decapods crustaceans.

Angradi T. R., S. M. Hagan, and K. W. Able. 2001. Vegetation type and the intertidal macroinvertebrate fauna of a brackish marsh: Phragmites vs. Spartina. Wetlands. 21:75-92.
Total macroinvertebrate density, mean taxa richness, stem density, microtopographic relief, density of microhabitats was greater in Spartina. Detrital and above-ground vegetative biomass and water velocity were greater in Phragmites marsh. Difference in invert dominance between the communities. Greater density of intertidal standing-water microhabitats is important as a source of faunal variation between marsh types. Fewer refugia from predators at high tide in Phrag.

Chambers R. M., L. A. Meyerson, and K. Saltonstall. 1999. Expansion of Phragmites australis into tidal wetlands of North America. Aquatic Botany. 64:261-273.
"Rapid spread of Phragmites has been documented in freshwater (<0.5 ppt), oligohaline (0.5-5 ppt) and mesohaline (5-18 ppt) tidal wetlands…A fundamental concern regarding Phragmites expansion, particularly into tidal freshwater wetlands, is the observed reduction in biodiversity as many native species of plants are replaced by a more cosmopolitan species. Commensurate with a shift in habitat type is a reduction in insect, avian and other animal assemblages. Ecosystem services, including support of higher trophic levels, enhancement of water quality and sediment stabilization, however, are not diminished when a tidal wetland becomes dominated by Phragmites, provided that tidal flooding is retained."

Hunter K. L., D. A. Fox, L. M. Brown, and K. W. Able. 2006. Responses of resident marsh fishes to stages of Phragmites australis invasion in three mid Atlantic estuaries. Estuaries and Coasts. 29:487-498.
Patterns studied suggest a decline in habitat function for larval and juvenile F. heteroclitus as invasion progresses.

Jivoff P. R., K. W. Able. 2003. Blue crab, *Callinectes sapidus*, response to the invasive common reed, Phragmites australis: Abundance, size, sex ratio, and molting frequency. Estuaries. 26:B 587-595.
Crabs prefer the marsh surface in Spartina Marshes, so phrag can negatively affect crab usage through different marsh surface. Restoration can have positive influence on marsh crab through change in marsh surface.

Osgood D. T., D. J. Yozzo, R. M. Chambers, D. Jacobson, T. Hoffman, and J. Wnek. 2003. Tidal Hydrology and Habitat Utilization by Resident Nekton in Phragmites and Non-Phragmites Marshes. Estuaries. 26:522-533.
Shrimp and Mummichog greater in Spartina. Large fish effected when Phrag invasion effects flood frequency, water depth.

Robertson T. L., J. S. Weis. 2005. A comparison of epifaunal communities associated with the stems of salt marsh grasses Phragmites Australis and Spartina Alterniflora. Wetlands. 25:1-7.
S. alterniflora stems support more epifaunal animals than P. australis stems overall.

Robertson T. L., J. S. Weis. 2007. Interactions between the grass shrimp *Palaemonetes pugio* and the salt marsh grasses *Phragmites australis* and *Spartina alterniflora*. Biological Invasions. 9:25-30.
Attempted to demonstrate the epifauna differences were due to greater effectiveness of grazing Phragmites. However, found grazing to be proportional. Rules out top-down control of epifauna on Phrag.

Wainright S. C., M. P. Weinstein, K. W. Able, and C. A. Currin. 2000. Relative importance of benthic microalgae, phytoplankton and the detritus of smooth cordgrass Spartina alterniflora and the common reed Phragmites australis to brackish-marsh food webs. Marine Ecology Progress Series. 200:77-91.
Although there were negative effects of invasion on benthic microalgae from shading, the study does indicate that P. australis may contribute to aquatic food webs in tidal marshes.

Weis J. S., P. Weis. 2003. Is the invasion of the common reed, Phragmites australis, into tidal marshes of the eastern US an ecological disaster? Marine Pollution Bulletin. 46:816-820.
Review of studies on Phragmites invasion suggest that Phragmites dominated marshes are comparable to Spartina marshes in benthics and nekton. Some negative effects on Mummichog and epifauna, but overall refutes the idea that Phrag is 'useless' as a habitat and contributor to food webs.

Dunes - Water Quality

Conn C. E., F. P. Day. 1993. Belowground biomass patterns on a coastal barrier island in Virginia. Bulletin of the Torrey Botanical Club. 120:121-127.
Structural attributes of dune plant root systems result in reduced nutrient losses through leaching.

Heyel S. M., F. P. Day. 2006. Long-term residual effects of nitrogen addition on a barrier island dune ecosystem. Journal of the Torrey Botanical Society. 133:297-303.
Belowground reserves of nitrogen from a single fertilization persisted for more than ten years. Suggest that dune plants are efficient at uptake and storage of nutrients given the limitations in the sandy, well drained substrate.

Dunes - Habitat

Engels, W. L. 1942. Vertebrate fauna of North Carolina coastal islands. A study in the dynamics of animal distribution I. Ocracoke Island. American Midland Naturalist. 28(2): 273-304.

Monograph on the geology, climate, ecology of Ocracoke Island and the flora and fauna of the dunes and beaches on the island.

Dunes - Sand Storage

Nordstrom, K.F. and E.L. Lotstein. 1989. Perspectives on resource use of dynamic coastal dunes. The Geographical Review. 79(1): 1-12.
This review discusses the use and management of dunes systems and describes the system as dynamic. Questions the stabilization or control of dunes as diminishing the resource value of dunes as sand reserves.

Beaches - Habitat

Dexter, D.M. 1967. Distribution and niche diversity of Haustoriid Amphipods in North Carolina. Chesapeake Science. 8(3): 187-192.
Review of amphipod fauna of North American Atlantic Coast from Gulf of Saint Lawrence to Florida. Indicates that 91% of interidal beach fauna consists of amphipods in the family haustoriidae.

McLachlan, A. and A. Brown. 2006. 2nd Edition. The Ecology of Sandy Shores.
Gives a global perspective on sand shore environments, with some bias toward South Africa where the most work has been done. Also treats the pacific coast (California) and the N. American Atlantic Coast (Maine) among other locations. Gives the general ecology of sandy shores and the flora and fauna that inhabit them.

Pearse, A.S., H.J. Humm, and G.W. Wharton. 1942. Ecology of sand beaches at Beaufort, N.C. Ecological Monographs. 12(2): 135-190.
A comprehensive ecological survey of the flora and fauna on the beaches around Ft. Macon, N.C. Lists a large number of inverts including Haustoriidae which is again cited as abundant in the surf zone. Also lists fish species found in the near shore as well as bird species foraging in the intertidal beach zone.

6. Shoreline Structures

Seawalls/Bulkheads – Water Quality

Basco, D.R., Bellomo, D.A., Hazelton, J.M., and Jones, B.N. 1997. The influence of seawalls on subaerial beach volumes with receding shorelines. Coast. Eng. 30(3-4): 203-233.
Sandbridge, VA; Do bulkheads increase erosion an adjacent non-bulkheaded shorelines? No, but do affect seasonal variability of sand volume in front of bulkhead.

Bozek, C.M., and Burdick, D.M. 2005. Impacts of seawalls on saltmarsh plant communities in the Great Bay estuary, New Hampshire USA. Wetlands Ecol. Manage. 13(5): 553-568.
How do seawalls impact fringing marshes? Also looks at sediment movement and groundwater flow with bulkheads.

Camfield, F.E., and Briggs, M.J. 1993. Longshore transmission of reflected waves. Journal of Waterway, Port, Coastal and Ocean Engineering (ASCE) JWPED5. p 575-579: 7 ref.
Waves reflected from coastal structures may become trapped by refraction, impacting downshore properties

Dellapenna, T.M., Allison, M., Robb, B., Bronikowski, J.L., Cerf, M., Majzlik, E.J., and Noll, C. 2004. Architecture of barrier island shore face in response to hard structures, beach nourishment, and subsidence; investigations along Galveston and Follet's islands, upper Texas coast. Geological Society of America, south-central section, 38th annual meeting. Abstracts with Programs - Geological Society of America. 36(1): 23.
Compared bulkheads and beach nourishment sites with natural shorelines.

Douglass, S.L. and B.H. Pickel. 1999. "The Tide Doesn't Go Out Anymore"—The Effect of Bulkheads on Urban Bay Shorelines. University of South Alabama, Civil Engineering and Marine Sciences Departments, Mobile. [Online] Available at: http://www.southalabama.edu/cesrp/Tide.htm [January 25, 2006].
Sediment deficit leads to shoreline erosion, when bulkhead is constructed, the erosion continues in front of the bulkhead leading to reduced intertidal areas. Discusses societal impacts and habitat impacts associated with loss of intertidal areas. Seawalls also impact littoral transport by preventing erosion.

El Banna, M.M. 2006. Responses of ras el bar seafloor characteristics to the protective engineering structures, Nile delta, Egypt. Environ. Geol. 49(5): 645-652.
Scouring impacts associated with breakwaters and bulkheads.

Griggs, J.F. and G.B. Tait. 1991. Beach Response to the Presence of a Seawall; Comparison of Field Observations. Contract Report CERC 91-1. U.S. Army Corps of Engineers, Waterways Experiment Station, Vickburg, Mississippi.

Griggs, G.B. 2005. The impacts of coastal armoring. Shore Beach. 73(1): 13-22.
Impacts from coastal hardening along the California coastline.

Kraus, N.C. and W.C. McDougal. 1996. The effects of seawalls on the beach: Part I, An updated literature review. Journal of Coastal Research. 12(3):691-701.

Kraus, N.C. and Heilman, D.J. 1998. Comparison of beach profiles at a seawall and groins, Corpus Christi, Texas. Shore and Beach. 66(2): 4-13.
Very general abstract, need to look at the paper for detail.

Kraus, N.C. and O.H. Pilkey. 1988. The effects of seawalls on the beach. Journal of Coastal Research. Special Issue 4:1-28.

Spalding, V.L. and N.L. Jackson. 2001. Field investigation of the influence of bulkheads on meiofaunal abundance in the foreshore of an estuarine sand beach. Journal of Coastal Research. 17:363-370.

Tait, J.F. and G.B. Griggs. 1990. Beach response to the presence of a seawall: a comparison of field observations. Shore and Beach. 58(2):11-28.
Systematic study of seawall effect on active beach erosion. Looks at seasonal beach changes and the physical processes that impact them.

Weggel, J.R., and Lesnik, J. 2006. Did this seawall cause erosion? Shore Beach. 74(3): 24-25.
No abstract.

Wilber, R.J., and Bush, D.M. 2003. Mesoscale response of a mobile sedimentary shoreline; basis for new directions in coastal management at the community level; Falmouth, MA. Geological Society of America, 2003 annual meeting. Abstracts with Programs - Geological Society of America. 35(6): 469.
Looked at structure impact on sediment movement; a case study.

Seawalls/Bulkheads – Habitat

Bischoff, A., and Humboldt-Universitaet zu Berlin. 2002. Juvenile fish recruitment in the large lowland river Oder: Assessing the role of physical factors and habitat availability. Shaker Verlag GmbH, Aachen.
Sampled juvenile fish assemblages in several different habitats; natural habitat is best.

Bozek, C.M., and Burdick, D.M. 2005. Impacts of seawalls on saltmarsh plant communities in the Great Bay estuary, New Hampshire USA. Wetlands Ecol. Manage. 13(5): 553-568.
How do seawalls impact fringing marshes? Also looks at sediment movement and groundwater flow with bulkheads.

Bulleri, F. 2005. Experimental evaluation of early patterns of colonisation of space on rocky shores and seawalls. Mar. Environ. Res. 60(3): 355-374.
Australia; The different habitat types modify the environment and affect recruitment.

Bulleri, F., Chapman, M.G., and Underwood, A.J. 2005. Intertidal assemblages on seawalls and vertical rocky shores in Sydney harbour, Australia. Austral Ecol. 30(6): 655-667.
Australia, patterns of abundance and variation differ between bulkheads and natural sites.

Bulleri, F. 2005. Role of recruitment in causing differences between intertidal assemblages on seawalls and rocky shores. Mar. Ecol. Prog. Ser. 287: 53-65.
Australia, Intrinsic differences between seawalls and natural shorelines affect recruitment of algae and inverts.

Chapman, M.G. 2006. Intertidal Seawalls as Habitats for Molluscs. J.Molluscan Stud. 72(3): 247-257.
Research has shown that many components of intertidal assemblages live on seawalls, but patterns of abundance and diversity are very variable. Seawalls differ physically from natural shores in a number of ways that are likely to influence distribution and abundances of intertidal molluscs, assemblages varied between tidal heights and among locations, but when data were combined across locations, there were some general patterns. Sessile bivalves (oysters and mussels) and many limpets were found in similar numbers on both habitats, or patterns varied inconsistently. Many coiled snails, in contrast, including whelks and grazing gastropods, plus opisthobranchs, which were either common or relatively sparse on horizontal shores, were not found on seawalls and found in intermediate frequencies on vertical shores. Similarly, common species of molluscs were found in natural and artificial boulder-fields in similar numbers, or patterns were not consistent, although rarer species were not found in these boulder-fields.

Chapman, M.G. 2003. Paucity of mobile species on constructed seawalls: Effects of urbanization on biodiversity. Mar. Ecol. Prog. Ser. 264: 21-29.
Australia, there is a lack of mobile species using seawalls relative to natural shorelines; also a lack of rare species.

Douglass, S.L. and B.H. Pickel. 1999. “The Tide Doesn’t Go Out Anymore”—The Effect of Bulkheads on Urban Bay Shorelines. University of South Alabama, Civil Engineering and Marine Sciences Departments, Mobile. [Online] Available at: http://www.southalabama.edu/cesrp/Tide.htm [January 25, 2006].
Sediment deficit leads to shoreline erosion, when bulkhead is constructed, the erosion continues in front of the bulkhead leading to reduced intertidal areas. Discusses societal impacts and habitat impact associated with loss of intertidal areas. Seawalls also impact littoral transport by preventing erosion.

Hendon, J.R., Peterson, M.S., and Comyns, B.H. 2001. Seasonal distribution of gobiids in waters adjacent to estuarine marsh-edge habitats: Assessing the effects of habitat alteration. Proc. Gulf Caribb. Fish. Inst.(52): 428-441.
Lower goby abundance in “altered” marsh habitats compared with natural.

Hendon, J.R., Peterson, M.S., and Comyns, B.H. 2000. Spatio-temporal distribution of larval gobiosoma bosc in waters adjacent to natural and altered marsh-edge habitats of mississippi coastal waters. Bull. Mar. Sci. 66(1): 143-156.
Even altered marsh habitats are important habitats for goby reproduction and life cycle.

Jaramillo, E., Contreras, H., and Bollinger, A. 2002. Beach and faunal response to the construction of a seawall in a sandy beach of south central Chile. J. Coast. Res. 18(3): 523-529.
Found that bulkheads did not impact beach macrofaunal community over the course of the study.

Jennings, M.J., Bozek, M.A., Hatzenbeler, G.R., Emmons, E.E., and Staggs, M.D. 1999. Cumulative effects of incremental shoreline habitat modification on fish assemblages in north temperate lakes. N. Am. J. Fish. Manage. 19(1): 18-27.
Species richness increased with increasing habitat complexity; assemblage structure changed with cumulative effect.

Peterson, M.S., Comyns, B.H., Hendon, J.R., Bond, P.J., and Duff, G.A. 2000. Habitat use by early life-history stages of fishes and crustaceans along a changing estuarine landscape: Differences between natural and altered shoreline sites. Wetlands Ecology and Management. 8: 209-219.
Nekton population was less abundant along bulkhead and riprap shorelines than marsh grass shorelines.

Pilkey, O. and H. Wright. 1988. Seawalls vs. beaches. Journal of Coastal Research SI4:41-64.

Rice, C., Sobocinski, K., and Puget Sound Action Team, Olympia, WA (USA). 2004. Effects of shoreline modification on spawning habitat of surf smelt (hypomesus pretiosus) in Puget Sound, Washington. Puget Sound Action Team, PO Box 40900 Olympia WA 98504 USA.
Higher temperatures were found on unvegetated, armored shorelines compared with natural.

Rice, C.A. 2006. Effects of shoreline modification on a northern Puget Sound beach: Microclimate and embryo mortality in surf smelt (hypomesus pretiosus). Estuaries Coasts. 29(1): 63-71.
Higher temperatures on altered beaches impact smelt egg survival.

Scheuerell, M.D., and Schindler, D.E. 2004. Changes in the spatial distribution of fishes in lakes along a residential development gradient. Ecosystems. 7(1): 98-106.
Shoreline modification is modifying the spatial distribution of aquatic organisms.

Schmude, K.L., Jennings, M.J., Otis, K.J., and Piette, R.R. 1998. Effects of habitat complexity on macroinvertebrate colonization of artificial substrates in north temperate lakes. J. N. Am. Benthol. Soc. 17(1): 73-80.
Riprap shorelines have a larger and more diverse community than bulkheaded shorelines.

Seitz, R.D., Lipcius, R.N., Olmstead, N.H., Seebo, M.S., and Lambert, D.M. 2006. Influence of shallow-water habitats and shoreline development on abundance, biomass,

and diversity of benthic prey and predators in chesapeake bay. Mar. Ecol. Prog. Ser. 326: 11-27.
Benthic abundance and diversity are highest in natural sites, intermediate at riprap sites, and lowest at bulkhead sites, this is also true for blue crabs.

Toft, J. 2005. Benthic macroinvertebrate monitoring at Seahurst Park 2004, pre-construction of seawall removal. Rep. Fish. Res. Inst. Wash. Univ.(0502).
Seawall sites have lower invertebrate density compared to vegetated sites.

Toft, J., and Cordell, J. 2006. Olympic Sculpture Park: Results from pre-construction biological monitoring of shoreline habitats. Rep. Fish. Res. Inst. Wash. Univ (0601) .
Abstract is vague—is this the results of a study or a proposal for a study?

Trial, P.F., Gelwick, F.P., and Webb, M.A. 2001. Effects of shoreline urbanization on littoral fish assemblages. Lake Reserv. Manage. 17(2): 127-138.
Bulkhead communities were different from natural shorelines, while riprap was similar.

Watkinson, D.A., Franzin, W.G., Podemski, C.L., and Department of Fisheries and Oceans Canada, Winnipeg, MB (Canada) Centr. Arctic Reg. 2004. Fish and invertebrate populations of natural, dyked and riprapped banks of the Aassiniboine and Red rivers, Manitoba. 2524.
Fish abundance was higher at armoured than natural shorelines and invertebrate and fish diversity was similar at all sites.

Wolter, C. 2001. Conservation of fish species diversity in navigable waterways. Landscape Urban Plann. 53(1-4): 135-144.
The percent of hardened shoreline was inversely correlated to the number of species, species diversity and the number of intolerant species.

Zito, A.N., Welch, J.M., and Kirby-Smith, W.W. 2004. The impact of avian predation on sea urchins arbacia punctulatan inhabiting a sea wall in Beaufort NC. Ohio J. Sci. 104(1): A-20.
Gulls preferentially prey on sea urchins living on seawalls (compared to what?).

Riprap – Water Quality

Camfield, F.E., and Briggs, M.J. 1993. Longshore transmission of reflected waves. Journal of Waterway, Port, Coastal and Ocean Engineering (ASCE)
JWPED5. p 575-579: 7 ref.
Waves reflected from coastal structures may become trapped by refraction, impacting downshore properties

Griggs, G.B. 2005. The impacts of coastal armoring. Shore Beach. 73(1): 13-22.
Impacts from coastal hardening along the California coastline.

Newell, R.I.E. and J. Ott. 1999. Macrobenthic Communities and Eutrophication. Pp. 265-293 in Ecosystems at the Land-Sea Margin: Drainage Basin to Coastal Sea. Coastal and Estuarine Studies, Malone, T.C., A. Malej, L.W. Harding, Jr., N. Smodlaka and R.E. Turner (eds). American Geophysical Union, Washington, DC.

Quigley, J.T. and D.J. Harper (eds.). 2004. Stream bank protection with rip-rap: An evaluation of the effects on fish habitat. Canadian Manuscript Report of Fisheries and Aquatic Sciences Report no. 2701. Canada Department of Fisheries and Oceans, Ottawa, Ontario, Canada. 76 pp.

Wilber, R.J., and Bush, D.M. 2003. Mesoscale response of a mobile sedimentary shoreline; basis for new directions in coastal management at the community level; Falmouth, MA. Geological Society of America, 2003 annual meeting. Abstracts with Programs - Geological Society of America. 35(6): 469.
Looked at structure impact on sediment movement; a case study.

Riprap – Habitat

Angradi, T.R., Schweiger, E.W., Bolgrien, D.W., Ismert, P., and Selle, T. 2004. Bank stabilization, riparian land use and the distribution of large woody debris in a regulated reach of the upper Missouri river, North Dakota, USA. River Res. Appl. 20(7): 829-846.
Riverine; Looked at LWD distribution in areas with and without riprap shorelines; speaks to the environmental impacts of riprap with reduces the abundance of LWD along stabilized shorelines.

Beauchamp, D.A., Byron, E.R., and Wurtsbaugh, W.A. 1994. Summer habitat use by littoral-zone fishes in Lake Tahoe and the effects of shoreline structures. N. Am. J. Fish. Manage. 14(2): 385-394.
Looked at fish communities near piers and "rock-cribs?" and found higher fish abundance in rock-cribs than no crib areas.

Bischoff, A., and Humboldt-Universitaet zu Berlin. 2002. Juvenile fish recruitment in the large lowland river oder: Assessing the role of physical factors and habitat availability. Shaker Verlag GmbH, Aachen.
Sampled juvenile fish assemblages in several different habitats; natural habitat is best.

Burke, R., Lipcius, R., Luckenbach, M., Ross, P.G., Woodward, J., and Schulte, D. 2006. Eastern oyster settlement and early survival on alternative substrates along intertidal marsh, rip rap, and manmade oyster reef. J. Shellfish Res. 25(2): 715.
Oysters recruit and survive less on riprap than natural marsh, but more than on man-made oyster reefs, riprap had the highest recruitment.

Carroll R (2003). Nekton utilization of intertidal fringing salt marsh and revetment hardened shorelines. Masters thesis, The College of William and Mary, Virginia Institute of Marine Science, Gloucester Point, VA

Fringing marsh versus riprap revetment in relation to nekton abundance, biomass and diversity. Significantly greater abundance of most species occurred in fringing marsh which also had higher diversity. Total abundance and biomass were higher in fringing marsh than riprap

Davis, J.L.D., Levin, L.A., and Walther, S.M. 2002. Artificial armored shorelines: Sites for open-coast species in a southern California bay. Mar. Biol. 140(6): 1249-1262.
Riprap communities were more different from natural communities at open coast sites than in protected bays.

Davis, J., Kramer, M., Young-Williams, A., and Hines, A. 2001. Effects of habitat type and size on species composition, nursery function, and refuge quality for an estuarine fish and macroinvertebrate community. Transactions, American Geophysical Union.
Compared riprap to oyster reef, SAV, woody debris and bare sediment; looked at fish and invertebrate; riprap and bare sediment can't substitute for oyster reef, SAV or woody debris.

Garland, R.D., Tiffan, K.F., Rondorf, D.W., and Clark, L.O. 2002. Comparison of subyearling fall chinook salmon's use of riprap revetments and unaltered habitats in Lake Wallula of the Columbia River. N. Am. J. Fish. Manage. 22(4): 1283-1289.
More fish found on natural than altered shorelines.

Hendon, J.R., Peterson, M.S., and Comyns, B.H. 2001. Seasonal distribution of gobiids in waters adjacent to estuarine marsh-edge habitats: Assessing the effects of habitat alteration. Proc. Gulf Caribb. Fish. Inst.(52): 428-441.
Lower goby abundance in "altered" marsh habitats compared with natural.

Hendon, J.R., Peterson, M.S., and Comyns, B.H. 2000. Spatio-temporal distribution of larval G*obiosoma bosc* in waters adjacent to natural and altered marsh-edge habitats of Mississippi coastal waters. Bull. Mar. Sci. 66(1): 143-156.
Even altered marsh habitats area important habitats for goby reproduction and life cycle.

Jennings, M.J., Bozek, M.A., Hatzenbeler, G.R., Emmons, E.E., and Staggs, M.D. 1999. Cumulative effects of incremental shoreline habitat modification on fish assemblages in north temperate lakes. N. Am. J. Fish. Manage. 19(1): 18-27.
Species richness increased with increasing habitat complexity; assemblage structure changed with cumulative effect.

Mueller, O. 2004. Riprap of groynes as habitats for larvae of Ophiogomphus cecilia (odonata: Gomphidae). Libellula. 23(1-2): 45-51.
Riprap is used by O. cecilia as a microhabitat and this is considered unusual.

Peterson, M.S., Comyns, B.H., Hendon, J.R., Bond, P.J., and Duff, G.A. 2000. Habitat use by early life-history stages of fishes and crustaceans along a changing estuarine landscape: Differences between natural and altered shoreline sites. Wetlands Ecology and Management. 8: 209-219.

Nekton population was less abundant along bulkhead and riprap shorelines than marsh grass shorelines.

Scheuerell, M.D., and Schindler, D.E. 2004. Changes in the spatial distribution of fishes in lakes along a residential development gradient. Ecosystems. 7(1): 98-106.
Shoreline modification is modifying the spatial distribution of aquatic organisms.

Schmetterling, D.A., Clancy, C.G., and Brandt, T.M. 2001. Effects of riprap bank reinforcement on stream salmonids in the western United States. Fisheries. 26(7): 6-23.
River bank study; riprap provides some habitat, but not the intricate habitat provided by vegetated shorelines.

Schmude, K.L., Jennings, M.J., Otis, K.J., and Piette, R.R. 1998. Effects of habitat complexity on macroinvertebrate colonization of artificial substrates in north temperate lakes. J. N. Am. Benthol. Soc. 17(1): 73-80.
Riprap shorelines have a larger and more diverse community than bulkheaded shorelines.

Scholle, J., and Schuchardt, B. 1999. Vertical gradients of the macrozoobenthos on riprap embankments of the tidal lower Weser River. Hydrologie Und Wasserbewirtschaftung /Hydrology and Water Resources Management-Germany. 43(2): 60-66.
Compared vertical distribution patterns of benthos on riprap with natural bottom.

Seitz, R.D., Lipcius, R.N., Olmstead, N.H., Seebo, M.S., and Lambert, D.M. 2006. Influence of shallow-water habitats and shoreline development on abundance, biomass, and diversity of benthic prey and predators in Chesapeake Bay. Mar. Ecol. Prog. Ser. 326: 11-27.
Benthic abundance and diversity are highest in natural sites, intermediate at riprap sites, and lowest at bulkhead sites, this is also true for blue crabs.

Stewart, T.W., Shumaker, T.L., and Radzio, T.A. 2003. Linear and nonlinear effects of habitat structure on composition and abundance in the macroinvertebrate community of a large river. Am. Midl. Nat. 149(2): pp.293-305.
Looks at stones as habitat structure; evidence for riprap as habitat?

Toft, JD, Cordell, JR, Simenstad, CA, Stamatiou, LA (2007). Fish Distribution, Abundance, and Behavior along City Shoreline Types in Puget Sound. North Am J Fish Manage 27:465–480
Sampled fish in five main habitat types: cobble beach, sand beach, riprap extending into the intertidal zone, deep riprap extending into the subtidal zone, and the edge of overwater structures. In Puget Sound, Washington.

Toft, J., and Cordell, J. 2006. Olympic Sculpture Park: Results from pre-construction biological monitoring of shoreline habitats. Report # 0601.
Abstract is vague—is this the results of a study or a proposal for a study?

Trial, P.F., Gelwick, F.P., and Webb, M.A. 2001. Effects of shoreline urbanization on littoral fish assemblages. Lake Reserv. Manage. 17(2): 127-138.
Bulkhead communities were different from natural shorelines, while riprap was similar.

Watkinson, D.A., Franzin, W.G., Podemski, C.L., and Department of Fisheries and Oceans Canada, Winnipeg, MB (Canada) Centr. Arctic Reg. 2004. Fish and invertebrate populations of natural, dyked and riprapped banks of the Assiniboine and Red Rivers, Manitoba. 2524.
Fish abundance was higher at armoured than natural shorelines and invertebrate and fish diversity was similar at all sites.

Wolter, C. 2001. Conservation of fish species diversity in navigable waterways. Landscape Urban Plann. 53(1-4): 135-144.
The percent of hardened shoreline was inversely correlated to the number of species, species diversity and the number of intolerant species.

Jetties – Water Quality

Nelson, K.A., Scott, G.I., and Rust, P.F. 2005. A multivariable approach for evaluating major impacts on water quality in Murrells and North inlets, South Carolina. Journal of Shellfish Research, Vol. 24, no. 4. 24(4): 1241-1251.
Urbanization poses a particular threat to the coastal areas of the southeastern United States where uplands surrounding wetlands are still relatively undeveloped compared with other regions. The approach used for this study involved a historical comparison of land use change and fecal coliform bacterial densities on Murrells Inlet (MI) (urbanized site) (n = 2026 samples) and North Inlet (NI) (pristine site) (n =1656 samples), both bar-built estuaries located on the northern coast of South Carolina south of Myrtle Beach. Regression models used the(WQ) parameters and a change in trend term that accounted for both instantaneous and gradual changes in water quality that may arise from a particular environmental intervention. For MI, the 1980 environmental intervention consisted of the construction of a jetty and the conversion from septic tanks to a main sewer line of approximately 92% of all residences. For NI, the 1973 and 1977 interventions were the construction of Baruch Laboratory and urban development of Debidue Island, respectively. For MI, there was a significant decrease in the increasing trend of fecal coliform bacteria and the conversion to the sewage collection system had a beneficial effect on water quality and probably dominated the jetty effect. For NI, the laboratory construction had no overall impact on water quality so background natural sources of bacteria probably masked any small increases from human sources.

Jetties – Habitat

Dolah, R.F.V., Wendt, P.H., Wenner, C.A., Martore, R.M., Sedberry, G.R., and Army Engineer Waterways Exp. Stn., Vicksburg, MS (USA). 1987. Environmental impact research program. Ecological effects of rubble weir jetty construction at Murrells inlet,

south Carolina. Volume 3. Community structure and habitat utilization of fishes and decapods associated with the jetties.
Quarrystone jetties at Murrells Inlet, South Carolina, were studied. The jetties attracted fish species normally associated with reef structures, species commonly found around estuarine inlets, and species that seasonally migrate along the coast. The jetties also serve as nursery habitat for a variety of fish species. More fish captured (recreational fishing) around the jetty structures than in non-jetty areas.

Goren, R., and Benayahu, Y. 1993. Benthic communities on artificial substrates at Elat (Red Sea). Isr. J. Zool. 39(1): 59.
The present study deals with recruitment of benthic communities on the oil jetties underwater constructions at Elat. The vertical pipes of the northern oil jetty have the highest living coverage along the mid-water parts along each pipe. The living coverage is composed of scleractinians, octocorals, sponges, ascidians, and actinians. Size distribution of the most abundant recruits on the pipes clearly shows high abundance of juveniles. The vertical seaward pipes are primarily covered by octocorals.

Hernandez, F.J.,Jr, Shaw, R.F., Cope, J.S., Ditty, J.G., and Farooqi, T. 2001. Do low-salinity, rock jetty habitats serve as nursery areas for pre-settlement larval and juvenile reef fish? Proc.Gulf Caribb. Fish. Inst.(52): 442-454.
This study investigated a coastal rock jetty system as a low-salinity, landward end member of artificial reefs along a transect of three oil and gas platforms extending from the inner continental shelf to the shelf break. Clupeiformes (engraulids and clupeids) comprised approximately 95% and 70% of the total light-trap and pushnet catch, respectively. Reef (or structure-dependent) fish, though not as abundant, included blennies, gobies, eleotrids, sparids, and lutjanids. Significantly lower densities and CPUEs were observed at sampling stations located within the jetty walls versus stations located externally. This result may be related to possible differences in environmental parameters (turbidity, temperature, salinity, and dissolved oxygen) between inner (estuarine) and outer (coastal) sampling stations. Preliminary results indicate that the jetty may serve as a refuge area for pre-settlement reef fish in the absence of other structurally-complex habitat.

Johnson, S.B., and Geller, J.B. 2006. Larval settlement can explain the adult distribution of Mytilus californianus conrad but not of M. galloprovincialis lamarck or M. trossulus gould in moss landing, central California: Evidence from genetic identification of spat. J. Exp. Mar. Biol. Ecol. 328(1): 136-145.
We investigated the spatial distribution of adult and newly settled mussels (Mytilus galloprovincialis Lamarck, Mytilus trossulus Gould and Mytilus californianus Conrad) on the shore at Moss Landing, California to test the hypothesis that adult distributions are a result of settlement patterns. Adult M. californianus were most abundant on a wave-exposed rocky jetty and adults of Blue mussels (M. trossulus and M. galloprovincialis) were more abundant inside the protected Moss Landing harbor.

Kaldy, J.E., Dunton, K.H., and Czerny, A.B. 1995. Variation in macroalgal species composition and abundance on a rock jetty in the northwest Gulf of Mexico. Bot. Mar. 38(6): 519-527.
Seasonal variation in algal species composition and biomass was examined on a rock jetty at Port Mansfield, Texas.). Over 30 algal species were collected during the study period, with the overall species composition dominated by rhodophytes, especially Bryocladia spp., Gelidium crinale/Pterocladia bartlettii complex, Hypnea spp., Centroceras clavulatum and Polysiphonia spp. All sites exhibited greater than 50% similarity of algal species, indicating a relatively homogeneous algal distribution. Red algae accounted for 94% of the average annual biomass compared to 5% for green algae and 1% for brown algae. The high biomass of algae recorded on the north jetty at Port Mansfield indicates that these plants represent a substantial food and habitat resource to marine animals, both invertebrates and vertebrates, which is available year-round.

Rader, W.L. 1998. Faunal list of shelled marine mollusks inhabiting the northern jetty, marina Del Rey, Los Angeles County, California. Festivus. 30(10): 105-112.
Over a seven-year period, at irregular intervals during wintertime low tides, 75 species of shelled marine mollusks living along the western end of the northern jetty at the southern tip of the Marina Peninsula, Los Angeles County, were recorded. A pronounced ecological zonation was noted among the inhabitants, both upon the jetty and within the ecological habitats at its base, and certain species were found to be characteristic of each habitat. A faunal list documenting all taxa that were recorded during this study was assembled.

Renaud, M.L., Carpenter, J.A., Williams, J.A., and Manzella-Tirpak, S.A. 1995. Activities of juvenile green turtles, Chelonia mydas, at a jettied pass in south Texas. Fish. Bull. 93(3): 586-593.
Texas nearshore and inshore waters are important habitats for juvenile and subadult sea turtles. Recent tracking and mark-recapture studies on green turtles indicate that jetties and channel entrances along the south Texas coast serve as summer developmental habitats for this species. Turtles using jetties and channel entrances could interact with human activities, such as channel dredging, shrimping, and recreational fishing and boating. We hypothesized that juvenile green turtles select jetty habitat over other habitats within Brazos-Santiago Pass.

Sullivan, M. 1984. Community structure of epiphytic diatoms from the gulf coast of Florida, U.S.A. Otto Koeltz, Koenigstein.
Epiphytic diatoms were collected on 13 March 1981 and 10 March 1982 from macrophytic algae (Cladophora sp. and Microcoleus lyngbyaceus) on a submerged jetty rock and an immediately adjacent seagrass bed (Syringodium filiforme and Thalassia testudinum) in Fred Howard Park on the Gulf Coast of Florida. The jetty samples were dominated by Synedra barbatula, Licmophora dalmatica, and Achnanthes brevipes var. intermedia, while the most abundant seagrass epiphytes were Cocconeis scutellum var. parva, Amphora tenuissima, and Mastogloia crucicula. Besides sharing no dominant taxa in common, structural similarity between the jetty and seagrass samples as

quantified by the SIMI index was very low and ranged from 0.012 to 0.152. This pronounced dissimilarity in the epiphytic diatom communities of the jetty and seagrass beds was independent of whether diversity (H' & S) was high or low in the former habitat.

Witten, A.L., and Bulkley, R.V. 1975. A study of the effects of stream channelization and bank stabilization on warmwater sport fish in Iowa. Subproject no. 2. A study of the impact of selected bank stabilization structures on game fish and associated organisms. Fish and Wildlife Service Office of Biological Services Report 76/12, May, 1975.116 p, 7 Fig, 15 Tab, 23 Ref.
Four types of stream bank stabilization structures (revetments, retards, permeable jetties, impermeable jetties) were studied to determine their impact upon game fish habitat. Permeable jetties and retards deepened the channel near the structures 7 to 110% greater than the maximum depth in control sections. No other significant differences in either physical parameters or in mean body length or abundance of game fish were found between structured and non-structured stream sections. Rock revetments and impermeable jetties fostered the growth of some invertebrates, primarily mayflies and caddis flies. Revetments, which presented the most rock surface for invertebrate colonization, had the greatest impact on invertebrate abundance. A long rock jetty, extending far enough into the stream to produce a scour hole, would combine most of the advantages noted in the structures studied. For habitat improvement, rock was superior to steel as a construction material, and structures which cause the formation of scour holes superior to those that do not deepen the stream.

Breakwaters – Water Quality

Allen, H.H., R.L. Lazor and J.W. Webb. 1990. Stabilization and development of marsh lands. Beneficial Uses of Dredged Material. Pp. 101-112 in Proceedings of the Gulf Coast Regional Workshop, April 26-28, 1988, Galveston, Texas. Technical Report D-90-3. U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, Mississippi.

Cuadrado, D.G., Gomez, E.A., and Ginsberg, S.S. 2005. Tidal and longshore sediment transport associated to a coastal structure. Estuar. Coast. Shelf Sci. 62(1-2): 291-300.
Found different modes of sediment transportation on the 2 ends of a breakwater.

Dickson, W.S., Herbers, T.H.C., and Thornton, E.B. 1995. Wave reflection from breakwater. J. Waterway Port Coast. Ocean Eng. 121(5): 262-269.
Large amplitude waves are dissipated more than lower energy waves across a breakwater structure.

El Banna, M.M. 2006. Responses of Ras el bar seafloor characteristics to the protective engineering structures, Nile Delta, Egypt. Environ. Geol. 49(5): 645-652.
Scouring impacts associated with breakwater and bulkhead structures.

Newell, R.I.E. and E.W. Koch. 2004. Modeling seagrass density and distribution in response to changes in turbidity stemming from bivalve filtration and seagrass sediment stabilization. Estuaries 27:793-806.

Pranesh, M., Venugopalan, P., and Marine Technology Soc., Manoa, HI (USA). Hawaii Sect. 1984. Ocean structure-shoreline interaction.
Predictions of shoreline change due to construction of a detached shoreline structure.

Ranasinghe, R., and Turner, I.L. 2006. Shoreline response to submerged structures: A review. Coastal Engineering. 53: 65-79.
Review of the impact of submerged breakwaters on local sediment transport.

Rennie, T.H. 1990. Using new work and maintenance material for marsh creation in the Galveston District. Beneficial Uses of Dredged Material. Pp. 184-187 in Proceedings of the Gulf Coast Regional Workshop, April 26-28, 1988, Galveston, Texas. Technical Report D-90-3. U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, Mississippi.

Rice, D.W., T.A. Dean, F.R. Jacobsen and A.M. Barnett. 1989. Transplanting giant kelp Macrocystis pyrifera in Los Angeles Harbor: productivity of the kelp population. Bulletin of Marine Science. 44:1070

Breakwaters – Habitat

Airoldi, L., M. Abbiati, M.W. Beck, S.J. Hawkins, P.R. Jonsson, D. Martin, P.S. Moschella, A. Sundelof, R.C. Thompson and P. Aberg. 2005. An ecological perspective on the development and design of low-crested and other hard coastal defense structures. Coastal Engineering. 52:1073-1087.
Consequences of breakwaters can be seen on a local scale, as disruption of surrounding soft-bottom environments and introduction of new artificial hard-bottom habitats, with consequent changes to the native assemblages of the areas. Proliferation of coastal defense structures can also have critical impacts on regional species diversity, removing isolating barriers, favoring the spread of nonnative species and increasing habitat heterogeneity. Advice is provided to meet specific management goals, which include mitigating specific impacts on the environment, such as minimizing changes to surrounding sediments, spread of exotic species or growth of nuisance species, and/or enhancing specific natural resources, for example enhancing fish recruitment or promoting diversity. The DELOS project points out that the downstream effects of defense structures on coastal processes and regional-scale impacts on biodiversity necessitate planning and management at a regional (large coastline) scale.

Allen, H.H., R.L. Lazor and J.W. Webb. 1990. Stabilization and development of marsh lands. Beneficial Uses of Dredged Material. Pp. 101-112 in Proceedings of the Gulf Coast Regional Workshop, April 26-28, 1988, Galveston, Texas. Technical Report D-90-3. U.S. Army Corps of Engineers, Waterways Experiment Station, Vickburg, Mississippi.

Lincoln Smith, M.P., C.A. Hair and J.D. Bell. 1994. Man-made rock breakwaters as fish habitats: comparisons between breakwaters and natural reefs within an embayment in southeastern Australia. Bulletin of Marine Science. 55:1344.

Moschella, P.S., M. Abbiati, P. Åberg, L. Airoldi, J.M. Anderson, F. Bacchiocchi, F. Bulleri, G.E. Dinesen, M. Frost, E. Gacia, L. Granhag, P.R. Jonsson, M.P. Satta, A. Sundelöf, R.C. Thompson and S.J. Hawkins. 2005. Low-crested coastal defense structures as artificial habitats for marine life: using ecological criteria in design. Coastal Engineering. 52:1053-1071.
Analysis of the effects of breakwaters on the surrounding intertidal and subtidal infaunal assemblages and mobile fauna. Changes in sediment and infauna seem to be inevitable and usually tend to induce negative changes, particularly on the landward side and in the presence of additional structures or after beach nourishment. The presence of species either coming from the new hard bottoms or associated to physical disturbances is viewed as a negative impact, while the potential nursery role is a positive one.

Newell, R.I.E. and E.W. Koch. 2004. Modeling seagrass density and distribution in response to changes in turbidity stemming from bivalve filtration and seagrass sediment stabilization. Estuaries. 27:793-806.

Rennie, T.H. 1990. Using new work and maintenance material for marsh creation in the Galveston District. Beneficial Uses of Dredged Material. Pp. 184-187 in Proceedings of the Gulf Coast Regional Workshop, April 26-28, 1988, Galveston, Texas. Technical Report D-90-3. U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, Mississippi.

Rice, D.W., T.A. Dean, F.R. Jacobsen and A.M. Barnett. 1989. Transplanting giant kelp Macrocystis pyrifera in Los Angeles Harbor: productivity of the kelp population. Bulletin of Marine Science. 44:1070

Stephens, J. and D. Pondella. 2002. Larval productivity of a mature artificial reef: the ichthyoplankton of King Harbor, California, 1974-1997. ICES Journal of Marine Science. 59:S51-S58.

Stephens, J., P.A. Morris, D. Pondella, T.A. Koonce and G.A. Jordan. 1994. Overview of the dynamics of an urban artificial reef fish assemblage in King Harbor, California, USA, 1974-1991: a recruitment driven system. Bulletin of Marine Science. 55:1224-1239.

U.S. Army Corps of Engineers (USACE), State of Maryland and Commonwealth of Virginia. 1990. Chesapeake Bay Shoreline Erosion Study. U.S. Army Corps of Engineers, Baltimore District, Maryland.

Misc and Debris – Water Quality

Day, K.E., Holtze, K.E., Metcalfe-Smith, J.L., Bishop, C.T., and Dutka, B.J. 1993. Toxicity of leachate from automobile tires to aquatic biota. Chemosphere. 27(4): 665-675.
A laboratory study was conducted to determine if automobile tires immersed in fresh water leach chemicals which are toxic to aquatic biota. Three tire types were examined - tires obtained from a floating tire breakwater; road-worn tires from the same vehicle; and new tires. Overlying water from both new and used tires was lethal to rainbow trout (Oncorhynchus mykiss) but leachate from used tires was more toxic. In addition, leachate remained relatively toxic to rainbow trout over time (8 d for new and 32 d for used) after tires were removed from the aquaria indicating that the chemicals responsible for toxicity degrade slowly and are non-volatile. No toxicity to cladocerans or fathead minnows was observed with these same leachates. Tires from a floating tire breakwater which had been installed for several (10) years did not release chemicals which were toxic to any species tested. Concentrated (10X) leachate from tires immersed for 25 d in water inhibited bioluminescence in the marine bacterium, Photobacterium phosphoreum , the enzyme, beta -galactosidase, in mutant Escherichia coli and the enzyme, NADH-coenzyme Q reductase, in the inner membrane of mitochondria.

Evans, J.J. 1997. Rubber tire leachates in the aquatic environment. Reviews of Environmental Contamination and Toxicology. 151: 67-115.
This review discusses the background of scrap tires discarded in the environment, including tire composition, adverse environmental effects, threats to public health and safety, and solid waste management. A review of the available information on chemical characterization of tire leachate from tire storage facilities, manufacturing, usage in recycling applications, and toxicity exposure studies, of vegetation surveys from waste tire areas and reviews of mammalian tire product toxicity, and of toxicity, mutagenicity, and carcinogenicity of tire exposure in experimental aquatic animals, microbes, and organelles is presented.

Evans, J.J., Shoemaker, C.A., and Klesius, P.H. 2000. In vivo and in vitro effects of benzothiazole on Sheepshead minnow (Cyprinodon variegatus). Mar. Environ. Res. 50(1-5): 257-261.
Benzothiazole, a common chemical associated with tire manufacturing and industrial wastewater, is a principal component of both fresh water and estuarine tire leachate, a neurotoxicant to larval sheepshead minnows (Cyprinodon variegatus) in in vivo estuarine studies. Fish mortality occurred after 5 days of exposure. Significant decreases in larval growth were noted at all concentrations. Histologically, gills had cellular alterations but the central nervous system lacked the severe cellular damage seen in previous tire leachate exposure studies.

Evans, J. 1998. Toxicity of shredded tire leachate to Sheepshead minnow (Cyprinodon variegatus): A morphological and cytological evaluation.
Analyses of larval sheepshead minnows (C. variegatus) exposed to sequential shredded tire leachate extractions at 5, 15 and 25 ppt salinities indicated that tire leachate acted as a central nervous system neurotoxin. Lesions and decreases in DNA synthesis were most severe after 1 and 2 days of exposure to 2 and 3 day post hatch larvae.

Hartwell, S.I., Jordahl, D.M., and Dawson, C.E.O. 2000. The effect of salinity on tire leachate toxicity. Water, Air, Soil Pollut. 121(1-4): 119-131.
The toxicity of leachates decreased with increasing salinity up to 15 ppt, with no significant change at higher salinities. Tire leachates are probably a greater threat to freshwater habitats than brackish or marine habitats, but bioaccumulation of persistent organic contaminants from tires is an unknown.

Li, W., Seifert, M., Xu, Y., and Hock, B. 2006. Assessment of estrogenic activity of leachate from automobile tires with two in vitro bioassays. Fresenius Environ. Bull. 15(1): 74-79.
This study examined whether automobile tires immersed in fresh water can leach chemicals that display estrogenic activity. All tire leachates obviously showed estrogenic activity, which was increased with duration of immersion. As tire leachates contain estrogenic compounds, they could be important pollution sources, potentially harmful to wildlife and human health. Thus, use of shredded tires as road fill or in landfill sites should arouse our attention.

Nelson, S.M., Mueller, G., and Hemphill, D.C. 1994. Identification of tire leachate toxicants and a risk assessment of water quality effects using tire reefs in canals. Bull. Environ. Contam. Toxicol. 52(4): 574-581.
Due to the paucity of literature on the effects of fire reefs on water quality, a study was conducted using Toxicity Identification Evaluation procedures as designed by the EPA. Three series of analyses utilizing plugs cut from tires and whole tires were performed. Tire leachate was taken from tire plugs and utilized for toxicity evaluation. Meanwhile, a different set of plug leachate was assessed for possible bioconcentrated organic contaminants. The whole tires were applied in depuration studies.

Stephensen, E., Adolfsson-Erici, M., Celander, M., Hulander, M., Parkkonen, J., Hegelund, T., Sturve, J., Hasselberg, L., Bengtsson, M., and Foerlin, L. 2003. Biomarker responses and chemical analyses in fish indicate leakage of polycyclic aromatic hydrocarbons and other compounds from car tire rubber. Environ. Toxicol. Chem. 22(12): 2926-2931.
This study focused on sublethal effects of tire leacheate (PAH). We kept rainbow trout (Oncorhynchus mykiss) in tanks with two types of tires: a tire containing HA oils in the tread or a tire free of HA oils in the tread. After 1 d of exposure, an induction of cytochrome P4501A1 (CYP1A1) was evident in both exposed groups. After two weeks of exposure, EROD activity and CYP1A1 mRNA were still high in fish exposed to leachate from HA oil-containing tire, whereas the effect was somewhat lower in fish exposed to leachate from HA oil-free tread tire. Compounds in the tire leachates also affected antioxidant parameters. Total glutathione concentration in liver as well as hepatic glutathione reductase, glutathione S-transferase, and glucose-6-phosphate dehydrogenase activities were markedly elevated after two weeks of exposure in both groups. The responses were greater in the group exposed to leachate from HA oil-free tread tire. Vitellogenin measurements did not indicate leakage of estrogenic compounds from the tires. Chemical analyses of bile from exposed fish revealed the presence of

hydroxylated PAH as well as aromatic nitrogen compounds indicating uptake of these compounds by the fish.

Wik, A., and Dave, G. 2006. Acute toxicity of leachates of tire wear material to Daphnia magna - variability and toxic components. Chemosphere. 64(10): 1777-1784.
Large amounts of tire rubber are deposited along the roads due to tread wear. Several compounds may leach from the rubber and cause toxicity to aquatic organisms. UV exposure of the filtered tire leachates caused no significant increase in toxicity. The acute toxicity of tire wear for Daphnia magna was found to be -40 times a predicted environmental concentration based on reports on the concentration of a tire component found in environmental samples, which emphasizes the need for a more extensive risk assessment of tire wear for the environment.

7. Marinas, Docks, Boatramps

An, Y.J., D.H. Kampbell, & G.W. Sewell. 2002. Water quality at five marinas in Lake Texoma as related to methyl tert-butyl ether (MTBE). Environmental Pollution. 118(3):331-336.
Concentrations of a gasoline additive (suggesting gasoline spillage) were greatest at boat docks, then fuel pumping facilities, at 5 reservoirs. Suggested importance of gasoline spillage at boat motor start-up, as well as spillage during fueling.

Barbaro, R.D., B.J. Carroll, L.B. Tebo, & L.C. Walters. 1969. Bacteriological water quality of several recreational areas in the Ross Barnett Reservoir. J. Pollution Control Federation. 41(7):1330-1339.
Ref. by Chmura & Ross 1978. Higher fc in marina waters than in nonmarina waters.

British Waterways Board. 1983. Waterway ecology and the design of recreational craft. Inland Waterways Amenity Advisory Council, London, England.
Ref. by USEPA 2001. Boats can cause bank erosion, therefore effect WQ.

Brown, C.L. & R. Clark. 1968. Observations on dredging and dissolved oxygen in a tidal waterway. Water Resources Research. 4(6):1381-1384.
Ref. by Chmura & Ross 1978. DO temporarily decreased 16-33% during dredging of a tidal waterway, due to oxidation of resuspended sediments & decreased light for photosynthesis.

Cassin, J., K. Smith & K. Frenke. 1971. Sanitary implications of small boat pollution in an Atlantic estuary. Environmental Letters. 2(2):59-63.
Ref. by Chmura & Ross 1978. FC increased in water column & shellfish in direct relation to boat numbers, in estuarine area on NY coast.

Chen, K.Y., F.R. Bowerman, & M. Petridis. 1972. Environmental impact of storm drain on a semi-enclosed coastal water. Preprint of the 8th Annual Conference of the Marine Technology Society. (Sea-Grant Library abstract SCU-R-72-003).

Ref. by Chmura & Ross 1978. Heavy metals settled out of stormwater within short distance of discharge, but storm drain discharge had little direct effect on marina WQ. Marina breakwaters accumulated organic debris, depleting DO in bottom water.

Chesworth, J.C., M.E. Donkin, & M.T. Brown. 2004. The interactive effects of the antifouling herbicides Irgarol 1051 and Diuron on the seagrass Zostera marina. Aquatic Toxicology. 66(3):293-305.
Antifouling herbicides applied to boats to prevent algae growth are shown to have the potential to adversely affect seagrass growth.

Chmura, G.L. & N.W. Ross. 1978. The environmental impacts of marinas and their boats: A literature review with management considerations. Marine Advisory Service, Univ. of R.I., Narragansett.
Fairly extensive lit. review. As ref. by USEPA 2001: boat operation increases turbidity.

Faust, M.A. 1982. Contribution of pleasure boats to fecal coliform bacteria concentrations in the Rhode River estuary, Maryland, USA. The Science of the Total Environment. 25(2):255-262.
FC increased over a holiday weekend due to increased boat traffic.

Fisher, J.S., R.R. Perdue, M.F. Overton, M.D. Sobsey, & B.L. Sill. 1987. Comparison of water quality at two recreational marinas during a peak-use period. UNC Sea Grant College Prog. Raleigh, NC.
Ref. by USEPA, 2001. Boats were source of fc in areas w/high boat density & poor flushing.

Fufari, S.A. & J.L. Verber. 1969. Boat waste survey, Potter Cove, Rhode Island, summer 1968. Northeast Marine Health Services Lab., U.S. Public Health Svc. Davisville, RI. 27pp.
Ref. by Chmura & Ross 1978. Sampled water, shellfish, sediment in RI tidal cove, found that primary source of fc was boat waste.

Grovhoug, J.G., P.F. Seligman, G. Vafa, & R.L. Fransham. 1986. Baseline measurements of butyltin in US harbors and estuaries. In Proceedings Oceans 86, Vol.4 Organotin Symposium, Inst. of Electrical & Electronics Engineers, Inc. NY. pp.1283-1288.
Ref by USEPA, 2001. Toxic levels of butyltin found in marina waters.

Hall, L.W., Jr., M.J. Lenkevich, W.S. Hall, A.E. Pinkney, and S.T. Bushong. 1987. Evaluation of butyltin compounds in Maryland waters of Chesapeake Bay. Marine Pollution Bulletin. 18(2):78-83.
Ref. by USEPA, 2001. Toxic concentrations of dissolved copper found at CBay marinas.

Hertler, H., J. Spotila, & D.A. Kreeger. 2004. Effects of houseboats on organisms of the La Parguera Reserve, Puerto Rico. Environmental Monitoring and Assessment. 98:391-407.

Houseboats, at moorings that allowed 360 deg. swing, did not adversely impact seagrass growth. Proximity to new development on shoreline did appear to affect seagrass growth, possibly due to runoff. Houseboats, w/o antifouling paint, acted as artificial structures providing habitat for many types of organisms.

Jensen, H.F., M. Holmer, & I. Dahlof. 2004. Effects of tributyltin (TBT) on the seagrass *Ruppia maritima*. Marine Pollution Bulletin. 49(7-8):564-573.
Denmark. Net photosynthetic activity decreased up to 60%, relative growth rate –25% lower in TBT-contaminated sediments.

Kirby-Smith, W.W. & N.M. White. 2006. Bacterial contamination associated with estuarine shoreline development. J. of Applied Microbiology. 100: 648-657.
Compared fecal colif. along developed, undeveloped, and marina/cmty pier shorelines. Found highest fc at older developed shoreline, then new developed shl., then large marina. Suggest upland runoff more impt. source of fc than boats. Extent of circulation/flushing deemed important. Suggest that oyster gardeners not eat oysters they raise, due to likely high fc. Implications for shellfish closure areas.

Marcus, J.M., G.R. Swearingen, A.D. Williams, & D.D. Heizer. 1988. Polynuclear aromatic hydrocarbons and heavy metals concentrations in sediments at coastal South Carolina marinas. Archives of Environmental Contamination and Toxicology. 17:103-113.
Ref. by USEPA, 2001. Petroleum HCs in high concen. in marinas, but lower in sed. of well-flushed marinas.

McGee, Beth L., Christian E. Schlekat, Daniel M. Boward, and Terry L. Wade. 1995. Sediment contamination and biological effects in a Chesapeake Bay marina. Ecotoxicology. 4(1):39-59.
Found PAHs, copper, TBT elevated in marina sediments. Benthic infauna reflected environmental degradation within marina basin. Marina basin design limited flushing & contaminant export.

McMahon, P.J.T. 1989. The impact of marinas on water quality. Water Science & Technology. 21(2):39-43.
Ref. by USEPA, 2001. Higher levels of metals near maintenance area drains & fuel docks.

Mastran, T.A., A.M. Dietrich, & D.L. Gallagher, & T.J. Grizzard. 1994. Distribution of polyaromatic hydrocarbons in the water column and sediments of a drinking water reservoir with respect to boating activity. Water Research. 28(11):2353-2366.
Found PAHs in water column and sediment correlated with boating activity. Greater in marinas than nonmarina sites. Urban runoff and atmospheric deposition also contributed PAHs.

Milliken, A.S. & V. Lee. 1990. Pollution impacts from recreational boating: A bibliography and summary review. Rhode Island Sea Grant Publications, URI Bay Campus, Narragansett, RI.
Good lit. review.

Nixon, S.W., C.A. Oviatt, & S.L. Northby. 1973. Ecology of small boat marinas. Marine Tech. Rpt. Series No. 5, Univ. of R.I., Kingston, RI.
Rhode Island. Compared marina basin to saltmarsh cove. Found no differences in several factors. Copper levels higher in marina. Fouling communities in marinas exerted signif. oxygen demand. Fouling communities appeared to be impt. food source for fish. Atl. menhaden seldom found in marina. But sport fish more abundant in marina areas.

NCDEM. 1990. North Carolina coastal marinas: Water quality assessment. Report No. 90-01. North Carolina Division of Environmental Management. Raleigh, NC.
Cited in USEPA, 2001. Found lower DO in marinas compared to adjacent water bodies. Copper most common metal found in toxic concentrations in marina waters. PAHs. Boats source of fc in areas w/high boat density & poor flushing. Petroleum HCs in high concen. in marinas, but lower in sed. of well-flushed marinas.

NCDEM. 1991. Coastal marinas: Field survey of contaminants and literature review. Report No. 91-03. North Carolina Division of Environmental Management, Raleigh, NC.
Ref. by USEPA, 2001. Copper most common toxic metal in marina waters. Higher levels of metals near maintenance area drains & fuel docks.

Nixon, S.W., C.A. Oviatt, & S.L. Northby. 1973. Ecology of small boat marinas. Univ. of R.I. Marine Tech.Rpt. No.5. Narragansett, RI. 20pp.
Ref. by Chmura & Ross 1978. Fouling communities on piers & other structures contribute to biological productivity (supplement food), but may not replace pre-existing salt marsh condition. Copper found in high levels in water, sediment, & fouling communities in marinas.

Reish, D.J. 1961. A study of benthic fauna in a recently constructed boat harbor in southern California. Ecology. 42(1):84-91.
Ref. by Chmura & Ross 1978. In a marina newly dredged in part from upland, found that colonization with benthic community similar to those in nearby similar soft sediments occurred within 1 yr. of dredging.

USEPA. 1974. Assessing effects on water quality by boating activity. USEPA, National Environmental Research Ctr, Cincinnati, OH.
Ref. by USEPA, 2001. Boat propwash increases turbidity/susp.sed.

USEPA. 1985. Coastal Marinas Assessment Handbook. USEPA Region IV. EPA 904/6-85-132.

USEPA. 2001. National Management Measures Guidance to Control Nonpoint Source Pollution from Marinas and Recreational Boating. USEPA Office of Water. EPA 841-B-01-005.
Most recent summary, but most recent reference cited is 1992. Good summary of pollutant types and impacts.

Voudrias, E.A. 1981. Influence of marinas on hydrocarbons in sediments of two estuarine creeks. VIMS M.A. Thesis.

Voudrias, E.A., & C.L. Smith. 1986. Hydrocarbon pollution from marinas in estuarine sediments. In Estuarine, Coastal, and Shelf Science. 22:271-284.
Ref. by USEPA, 2001. Petroleum HCs in high concen. in marinas, but lower in sed. of well-flushed marinas.

Young, D.R. & T.C. Hessen. 1974. Inputs and distributions of chlorinated hydrocarbons in three southern California harbors. p.51-67 In Proceedings of the 4th Annural Technical Conference on Estuaries of the Pacific Northwest, March 14-15, 1974. Oregon Expt. Sta. Circular No. 50, Corvallis, OR.
Ref. by Chmura & Ross 1978. Copper higher in mussels taken from boat harbors.

Young, D.R. & D.J. McDermott, T.C. Hessen & T.K. Jan. 1975. Pollutant inputs and distributions off southern California. p.424-439 In Church, T.M. (ed.). Marine chemistry in the coastal environment. ACS Symposium Series 18, American Chemical Society, Wash. DC. 710pp.
Ref. by Chmura & Ross 1978. Copper higher in mussels taken from boat harbors.

8. Subaqueous Resources

SAV – Water Quality

Cerco, C. F. and K. Moore. (2001) System-Wide Submerged Aquatic Vegetation Model for Chesapeake Bay. Estuaries. 24(4):522-534.
A predictive model of submerged aquatic vegetation (SAV) biomass is coupled to a eutrophication model of Chesapeake Bay. Domain of the model includes the mainstem of the bay as well as tidal portions of major embayments and tributaries. Three SAV communities are modeled: ZOSTERA, RUPPIA, and FRESHWATER. Sensitivity analysis to reductions in nutrient and solids loads indicates nutrient controls will enhance abundance primarily in areas that presently support SAV.

Churchill, A. C., A. E. Cok and M. I. Riner. (1978) Stabilization of subtidal sediments by the transplantation of the seagrass Zostera marina L. NYSSGP; Albany, NY (USA). 46p. Report Series NY Sea Grant Institute.
The seagrass Zostera marina has potential for stabilizing unconsolidated sediments, esp dredge spoils.

Fonseca, M. S. and J. A. Calahan. (1992) A preliminary evaluation of wave attenuation by four species of seagrass. Estuar.Coast.Shelf Sci. 35(6):565-576.
Seagrasses are able to modify current flow and sediment composition. Percent wave energy reduction per meter of seagrass bed equaled 40% when the length of these seagrasses was similar to the water depth. Seagrasses are approximately equal to saltmarshes in reducing wave energy on a unit distance basis, but only when water depth is scaled to plant size. When seagrass beds occur as broad, shallow meadows, the influence of seagrasses on wave energy will be substantial.

Gambi, M. C., A. R. M. Nowell and P. A. Jumars. (1990) Flume observations on flow dynamics in Zostera marina (eelgrass) beds. Marine ecology progress series. 61(1-2):159-169.
Flow dynamics in Zostera marina (eelgrass) were studied in a large seawater flume. Mean velocity increased above the canopy, while within the bed water speed dropped distinctly below the canopy-water interface. Depending on shoot density, water speed was from 2 to 10 times lower under the canopy than upstream of the seagrass bed.

Lilleboe, A. I., M. R. Flindt, M. A. Pardal and J. C. Marques. (2006) The Effect of Zostera noltii, Spartina maritima and Scirpus maritimus on Sediment Pore-water Profiles in a Temperate Intertidal Estuary. Hydrobiologia. 555:(1)175-183.
Sediment profiles of loss on ignition (LOI) showed an increase of the organic matter contents from sand-flat, to Zostera, Spartina, mud-flat and Scirpus. These results suggest that there is an intense mobility of nutrients in the sediment, showing a day-night variation of nutrient concentrations in the pore-water. In the plants' rhizosphere, the day-night variation of nutrients seemed dependent on plant biomass and penetration of the roots. Additionally, coupling between plant and sediment seems to be a species-specific process. The top 10 cm of the sediment in the Spartina salt marsh and in the Zostera beds may contribute to the efflux of nutrients during the night period, especially phosphate.

Madsen, J. D., P. A. Chambers, W. F. James, E. W. Koch and D. F. Westlake. (2001) The interaction between water movement, sediment dynamics and submersed macrophytes. Hydrobiologia. 444(1-3):71-84.
This review defines known relationships and identifies areas that need additional research on the complex interactions among submersed macrophytes, water movement, and sediment dynamics. Water movement has a significant effect on macrophyte growth; in turn, macrophyte beds reduce current velocities both within and adjacent to the beds, resulting in increased sedimentation and reduced turbidity. Additionally, macrophytes affect the distribution, composition and particle size of sediments in both freshwater and marine environments. Therefore, establishment and persistence of macrophytes in both marine and freshwater environments provide important ecosystem services, including: (1) improving water quality; and (2) stabilizing sediments, reducing sediment resuspension, erosion and turbidity.

Newell, R. and Evamaria W. Koch. (2004) Modeling Seagrass Density and Distribution in Response to Changes in Turbidity Stemming from Bivalve Filtration and Seagrass Sediment Stabilization. Estuaries. 27(5):793-806.
Development of a simple model to calculate how changes in the balance between sediment sources (wave-induced resuspension) and sinks (bivalve filtration, sedimentation within seagrass beds) regulate turbidity. The model predicted that eastern oysters reduced suspended sediment concentrations by nearly an order of magnitude. Hard clams and seagrass were less effective. Our model predicted that restoration of eastern oysters has the potential to reduce turbidity in shallow estuaries, such as Chesapeake Bay, and facilitate ongoing efforts to restore seagrasses.

Paling, E. I., M. Van Keulen, K. D. Wheeler, J. Phillips and R. Dyhrberg. (2003) Influence of spacing on mechanically transplanted seagrass survival in a high energy wave regime. Restor.Ecol. 11(1):56-61.
This study indicates that the ability of seagrasses to influence sediment would appear to vary with the prevailing hydrodynamic regime and that a reappraisal of the notion that all seagrass communities trap sediment is necessary.

Peterson, C. H., R. A. Luettich Jr, F. Micheli and G. A. Skilleter. (2004) Attenuation of water flow inside seagrass canopies of differing structure. Mar.Ecol.Prog.Ser. 26:881-92.
A model of the effects of seagrass habitat structure on mean flow within and above the canopy. The field data demonstrated greater flow reductions inside the canopy with increasing vegetation density.

SAV – Habitat

Cardoso, P. G., D. Raffaelli and M. A. Pardal. (2007) Seagrass beds and intertidal invertebrates: an experimental test of the role of habitat structure. Hydrobiologia. 575(1):221-230.
Evidence of the importance of habitat structure for invertebrate community composition and dynamics. Seagrass beds enhanced survival of snails due to protection from avian or fish predators which can have a significant effect on population structure and hence biomass and productivity of key species in this system.

Castellanos, D. L. and L. P. Rozas. (2001) Nekton Use of Submerged Aquatic Vegetation, Marsh, and Shallow Unvegetated Bottom in the Atchafalaya River Delta, a Louisiana Tidal Freshwater Ecosystem. Estuaries. 24(2):184-197.
Vegetated areas generally supported much higher nekton densities than unvegetated sites, and may be important nursery areas for blue crabs and other estuarine species.

Dealteris, J. T., B. D. Kilpatrick and R. B. Rheault. (2004). A comparative evaluation of the habitat value of shellfish aquaculture gear, submerged aquatic vegetation and a non-vegetated seabed. J.Shellfish Res. 23(3):867-874.
Comparison of the habitat value of modified rack and bag, shellfish aquaculture gear (SAG) used for the grow-out phase of the American oyster, Crassostrea virginica,

submerged aquatic vegetation (SAV), Zostera marina, and a shallow nonvegetated seabed (NVSB). The physical structure of the SAG habitat protects juvenile fish from predators and provides substrate for sessile invertebrates that serve as forage for fish and invertebrates. Therefore, we conclude that shellfish aquaculture gear has substantially greater habitat value than a shallow nonvegetated seabed, and has habitat value at least equal to and possibly superior to submerged aquatic vegetation.

Glancy, T. P., T. K. Frazer, C. E. Cichra and W. J. Lindberg. (2003). Comparative Patterns of Occupancy by Decapod Crustaceans in Seagrass, Oyster, and Marsh-edge Habitats in a Northeast Gulf of Mexico Estuary. Estuaries. 26(5):1291-1301.
Decapod crustacean assemblages associated with oyster reef were distinct from seagrass and marsh-edge habitats (which were similar).

Goldberg, R., B. Phelan, J. Pereira, S. Hagan, P. Clark, A. Bejda, A. Calabrese, A. Studholme and K. W. Able. (2002). Variability in Habitat Use by Young-of-the-Year Winter Flounder, *Pseudopleuronectes americanus*, in Three Northeastern U.S. Estuaries. Estuaries. 25(2):215-226.
YOY winter flounder were found in higher densities in unvegetated areas adjacent to eelgrass. The exception was in the Hammonasset River in 1995 when densities were higher in eelgrass. We conclude that the type of habitat most important to YOY winter flounder varies among estuaries and as a result, care should be taken in defining EFH, based only on limited spatial and temporal sampling.

Harris, L. A., B. Buckley, S. W. Nixon and B. T. Allen. (2004). Experimental studies of predation by bluefish *Pomatomus saltatrix* in varying densities of seagrass and macroalgae. Mar.Ecol.Prog.Ser. 28:1233-239.
Eelgrass significantly increased the survivorship of silversides, tautog and cunner at very low shoot densities. Experiments using macroalgae did not result in significantly different survival rates. This information increases our understanding of the relative value of eelgrass habitat to fish stocks.

Horinouchi M. (2007). Distribution patterns of benthic juvenile gobies in and around seagrass habitats: effectiveness of seagrass shelter against predators. Estuar.Coast.Shelf Sci. 72:(4)657-664.
Differences in juvenile abundance are likely due to differences in food availability not predator exclusion by SAV.

Hosack, G., D. Armstrong, B. Dumbauld, J. Ruesink, B. Semmens and I. Fleming. (2004). The effect of *Zostera marina* and *Crassostrea gigas* culture on intertidal communities in a Northeast Pacific estuary. J.Shellfish Res. 23(2):655.
In Pacific Northwest estuaries, commercial oysters are often cultivated at the same tidal elevation as seagrass meadows, and thus management decisions concerning resource use require insight into their comparative value. Meiofauna densities were more than seven times higher in structured habitats and composition was significantly affected by habitat.

Leduc, D., P. K. Probert, R. D. Frew and C. L. Hurd. (2006). Macroinvertebrate diet in intertidal seagrass and sandflat communities: a study using C, N, and S stable isotopes. N.Z.J.Mar.Freshw.Res. 40:(4)615-629.
Z. capricorni was a potentially important contributor (24-99%) to the diet of most consumers sampled at the seagrass site, whereas microphytobenthos dominated the diet of the same consumers at the sandflat site.

Lipcius, R.N., Rochelle D. Seitz, Michael S. Seebo and Duamed Colon-Carrion. (2005). Density, abundance and survival of the blue crab in seagrass and unstructured salt marsh nurseries of Chesapeake Bay. J.Exp.Mar.Biol.Ecol. 319(1-2):69-80.
Juvenile blue crab density was nearly an order of magnitude lower in mudflats and sandflats than in SAV habitats; density was lowest in DCM. We conclude that shallow subtidal mud and sand flats near upriver salt marshes and in marsh coves are vital nursery grounds for the blue crab, and thus warrant conservation and restoration efforts at the level provided to SAV.

Phelan, B. A., R. Goldberg, A. J. Bejda, J. Pereira, S. Hagan, P. Clark, A. L. Studholme, A. Calabrese and K. W. Able. (2000). Estuarine and habitat-related differences in growth rates of young-of-the-year winter flounder (*Pseudopleuronectes americanus*) and tautog (*Tautoga onitis*) in three northeastern US estuaries. J.Exp.Mar.Biol.Ecol. 147:(1)1-28.
Comparisons across nominal habitats within and among estuaries did not show any one habitat with consistently higher growth, and growth was relatively independent of whether a habitat was vegetated or adjacent to vegetation.

Reid, Carolyn Cristine. (2004). The effects of submerged aquatic vegetation as habitat on the survivorship of clams: Field surveys in St. Mary's River, Maryland and laboratory predation experiments (*Callinectes sapidus, Mya arenaria*). Masters Abst.Int. 4241204
Greater predation pressure on clams in lower SAV biomass may be causing differences in clam abundance since SAV presence significantly reduced predation. Habitat studies tracking behavior revealed crabs spent more time in vegetation but consumed more clams outside SAV.

Stevenson, J. C. and N. M. Confer. (1978). Summary of Available Information on Chesapeake Bay Submerged Vegetation. Fish and Wildlife Service 747 ref.
A review of impacts which may contribute to the reduction of SAV in the Chesapeake Bay.

Stockhausen, W. T. and R. N. Lipcius. (2003). Simulated effects of seagrass loss and restoration on settlement and recruitment of blue crab postlarvae and juveniles in the York River, Chesapeake Bay. Bull.Mar.Sci. 72(2):409-422.
Seagrass meadows provide important settlement habitat, food and refuge for postlarvae and young juveniles of the blue crab, Callinectes sapidus. This is a model of the impacts of SAV losses or creation on blue crab settling.

Thayer, G.W., MS Fonseca and W. Judson Kenworthy. (1985). Restoration of seagrass meadows for enhancement of nearshore productivity. Int. Symp. on Utilization of Coastal Ecosystems: Planning, Pollution, and Productivity, Rio Grande, RS (Brazil), 21 Nov 1982
Eelgrass exhibited an exponential growth and coverage rate and attained densities comparable to natural meadows. Bottom coverage rate, carbon fixation, detrital production and faunal recruitment are enhanced by seagrass restoration, helping to maintain nearshore productivity.

Valesini, F. J., I. C. Potter and K. R. Clarke. (2004). To what extent are the fish compositions at nearshore sites along a heterogeneous coast related to habitat type? Estuar.Coast.Shelf Sci. 60(4):737-754.
Extents of the differences in the fish compositions among the various habitat types parallel those in the environmental data for the corresponding habitat types.

Van Montfrans, J., C. H. Ryer and R. J. Orth. (2003). Substrate selection by blue crab Callinectes sapidus megalopae and first juvenile instars. Mar.Ecol.Prog.Ser. 260:209-217.
The initial non-random distribution of blue crabs in Chesapeake Bay may be deterministic and due to habitat-selection behavior by megalopae. Selection for seagrass assures the greatest likelihood of maximal survival and accelerated growth.

Oysters/Bivalves – Water Quality

Campbell, M. (2005). The use of oyster reef restoration as a method of erosion control. J.Shellfish Res. 24(1):318.
The design of a structure that would aid in oyster reef restoration in the shape of a submerged breakwater, termed an "oysterbreak," is designed to stimulate the growth of biologic structures in a wave dissipating shape. In the same way native oyster reefs can form immense structures that protect shorelines and coastal communities from storms, this structure would provide the same function.

Heck Jr., K. L., J. Cebrian and S. P. Powers. (2005). Ecosystem services provided by oyster reefs: An experimental assessment. J.Shellfish Res. 24(1):323.
Experimental assessment of many of the key expectations associated with oyster reef restoration. Specifically, we are assessing whether there are differences in water clarity, nutrient dynamics, benthic primary and secondary production, and nursery value for fish and mobile invertebrates that are associated with the presence of experimentally planted oyster reefs.

Newell, R. and Evamaria W. Koch. (2004). Modeling Seagrass Density and Distribution in Response to Changes in Turbidity Stemming from Bivalve Filtration and Seagrass Sediment Stabilization. Estuaries. 27(5):793-806.
Development of a simple model to calculate how changes in the balance between sediment sources (wave-induced resuspension) and sinks (bivalve filtration, sedimentation within seagrass beds) regulate turbidity. The model predicted that eastern

oysters reduced suspended sediment concentrations by nearly an order of magnitude. Hard clams and seagrass were less effective. Our model predicted that restoration of eastern oysters has the potential to reduce turbidity in shallow estuaries, such as Chesapeake Bay, and facilitate ongoing efforts to restore seagrasses.

Piazza, B., J. Plunket, J. Supan and M. La Peyre. (2003). Using created oyster reefs as a sustainable coastal protection and restoration tool. J.Shellfish Res. 22(1):349.
The value of reefs for protecting shorelines was determined by tracking shoreline position and adjacent marsh health (vegetation biomass, redox, sediment accretion) at paired cultched and non-cultched sites. Reef sustainability was determined by measuring recruitment and survival of oyster spat. Fisheries value of the reef was quantified by sampling nekton. Fish community usage of cultched and non-cultched sites was similar. Shoreline retreat appears to be slightly higher in high energy, non-cultched sites. Minimal movement and reworking of shell through two tropical storm events showed that reefs were stable.

Piazza, Bryan P., Patrick D. Banks and Megan K. La Peyre. (2005). The Potential for Created Oyster Shell Reefs as a Sustainable Shoreline Protection Strategy in Louisiana. Restor.Ecol. 13(3):499-506.
Shoreline retreat was reduced in cultched low-energy shorelines as compared to the control low-energy shorelines (analysis of variance; $p < 0.001$) but was not significantly different between cultched and noncultched sites in high-energy environments. Recruitment and growth rates of oyster spat suggested potential reef sustainability over time. Small fringing reefs may be a useful tool in protecting shorelines in low-energy environments. However, their usefulness may be limited in high-energy environments.

Ruesink, J. L., B. E. Feist, C. J. Harvey, J. S. Hong, A. C. Trimble and L. M. Wisehart. (2006). Changes in productivity associated with four introduced species: ecosystem transformation of a 'pristine' estuary. Mar.Ecol.Prog.Ser. 311:203-215.
The introduction of non-native species can lead to higher local richness. The replacement of native oysters in Willapa Bay by 2 new bivalve species has increased secondary production of harvested suspension feeders by 250% over peak historic values. These changes in production are also associated with altered detritus, water filtration, and biogenic habitat. Because other stressors are largely absent from Willapa Bay, the addition of particular exotic species has dramatically enhanced system production, while fundamentally reshaping the ecological character of the estuary. These strong ecological impacts of introduced species have rarely been measured at whole- ecosystem scales, and they occur in part because new species occupy habitats where similar native species were not present.

Oysters/Bivalves – Habitat

Burke, R., R. Lipcius, M. Luckenbach, P. G. Ross, J. Woodward and D. Schulte. (2006). Eastern oyster settlement and early survival on alternative substrates along intertidal marsh, rip rap, and manmade oyster reef. J.Shellfish Res. 25(2):715.

Surveys within the Lynnhaven Bay system indicate that artificial oyster shell reefs created in the early 1990s are producing poor to marginal oyster densities relative to densities on nearby granite and concrete riprap, and on oyster clusters along marsh-fringed shores. Overall, replicates showed a distinct recruitment/early survival pattern between sites: marsh > rip rap > artificial oyster reef. We therefore propose that granite may be a favorable oyster reef construction material, since it appears to enhance oyster settlement and early post-settlement survival. Additional biological benefits may accrue from granite reefs as community structure develops on the reefs.

Heck Jr., K. L., J. Cebrian and S. P. Powers. (2005). Ecosystem services provided by oyster reefs: An experimental assessment. J.Shellfish Res. 24(1):323.
Experimental assessment of many of the key expectations associated with oyster reef restoration. Specifically, we are assessing whether there are differences in water clarity, nutrient dynamics, benthic primary and secondary production, and nursery value for fish and mobile invertebrates that are associated with the presence of experimentally planted oyster reefs.

Hosack, G., D. Armstrong, B. Dumbauld, J. Ruesink, B. Semmens and I. Fleming. (2004). The effect of Zostera marina and Crassostrea gigas culture on intertidal communities in a Northeast Pacific estuary. J.Shellfish Res. 23(2):655.
Intertidal habitat structure created by seagrass and oysters is broadly considered to play an important role in the shaping of many biologic communites. Meiofauna densities were more than seven times higher in structured habitats, and composition was significantly affected by habitat.

Meyer, D. L. and E. C. Townsend. (2000). Faunal Utilization of Created Intertidal Eastern Oyster (Crassostrea virginica) Reefs in the Southeastern United States. Estuaries. 23(1):34-45.
Oyster cultch was added to the lower intertidal marsh-sandflat fringe of three previously created Spartina alterniflora salt marshes. Eastern oyster (Crassostrea virginica) settlement at one site of created reef exceeded that of the adjacent natural reefs within 9 mo of reef creation. The created reefs also had a higher number of molluscan, fish, and decapod species than the adjacent natural reefs. Created oyster reefs can quickly acquire functional ecological attributes of their natural counterparts.

Piazza, B., J. Plunket, J. Supan and M. La Peyre. (2003). Using created oyster reefs as a sustainable coastal protection and restoration tool. J.Shellfish Res. 22(1):349.
The value of reefs for protecting shorelines was determined by tracking shoreline position and adjacent marsh health (vegetation biomass, redox, sediment accretion) at paired cultched and non-cultched sites. Reef sustainability was determined by measuring recruitment and survival of oyster spat. Fisheries value of the reef was quantified by sampling nekton. Fish community usage of cultched and non-cultched sites was similar. Shoreline retreat appears to be slightly higher in high energy, non-cultched sites. Minimal movement and reworking of shell through two tropical storm events showed that reefs were stable.

Rodney, W. and K. T. Paynter. (2005). Benthic macrofaunal assemblages on restored and unrestored eastern oyster (Crassostrea virginica) reefs in Chesapeake Bay: Implications for finfish species. J.Shellfish Res. 24(1):335.
Motile macrofaunal density was an order of magnitude higher on restored reefs, epifaunal density was more than twice as high on restored reefs, and sessile macrofaunal density was two orders of magnitude higher on restored reefs. Since reef macrofauna include many important fish prey species, oyster reef restoration has the potential to augment energy transfer to higher trophic levels by increasing fish prey densities and fish foraging efficiency.

Rodney, William S. and Kennedy T. Paynter. (2006). Comparisons of macrofaunal assemblages on restored and non-restored oyster reefs in mesohaline regions of Chesapeake Bay in Maryland. J.Exp.Mar.Biol.Ecol. 335(1):39-51.
Density of macrofauna was an order of magnitude higher on restored reefs, epifaunal density was more than twice as high on restored reefs and sessile macrofaunal density was two orders of magnitude higher on restored reefs. Mean amphipod density was 20 times higher on restored plots and densities of xanthid crabs and demersal fish were both four times greater on restored plots. Two out of four functional feeding groups: suspension feeders and carnivore/omnivores, were more abundant on restored plots. Since reef macrofauna include many important fish prey species, oyster reef restoration may have the potential to augment fish production by increasing fish prey densities and fish foraging efficiency.

Seaman, William. (2007). Artificial habitats and the restoration of degraded marine ecosystems and fisheries. Hydrobiologia. 580(1):143-155.
A review of habitat restoration. Principles of ecological restoration are summarized, from planning through to evaluation. Alternate approaches to facilitate ecological recovery include land-use and ecosystem management and determining levels of human population, consumption and pollution.

Zeug, S. C., V. R. Shervette, D. J. Hoeinghaus and S. E. Davis. (2007). Nekton assemblage structure in natural and created marsh-edge habitats of the Guadalupe Estuary, Texas, USA. Estuar.Coast.Shelf Sci. 71(3-4):457-466.
We conclude that lower substrate complexity (specifically oyster) and soil organic content in the created marsh reduced measures of nekton similarity and recommend that these features be addressed in future restoration efforts.

Aquaculture – Water Quality

Rheault, R. B.. (2006). Ecological services rendered by cultured eastern oysters. J.Shellfish Res. 25(2):766.
It has been shown that wild and cultured oysters provide many ecological services that benefit both the environment and wild oyster stocks. (Shumway et al. 2003, Newell 2004, 2002) The services rendered by commercial oyster aquaculture include: removal of nutrients (both by harvest and enhanced bacterial deni-trification); improving water clarity and water quality by enhancing sedimentation; adding larvae to the wild;

sequestering tons of carbon and creating a structurally diverse habitat for other marine organisms.

Ruesink, J. L., B. E. Feist, C. J. Harvey, J. S. Hong, A. C. Trimble and L. M. Wisehart. (2006). Changes in productivity associated with four introduced species: ecosystem transformation of a 'pristine' estuary. Mar.Ecol.Prog.Ser. 311:203-215.
The introduction of non-native species can lead to higher local richness. The replacement of native oysters in Willapa Bay by 2 new bivalve species has increased secondary production of harvested suspension feeders by 250% over peak historic values. These changes in production are also associated with altered detritus, water filtration, and biogenic habitat. Because other stressors are largely absent from Willapa Bay, the addition of particular exotic species has dramatically enhanced system production, while fundamentally reshaping the ecological character of the estuary. These strong ecological impacts of introduced species have rarely been measured at whole- ecosystem scales, and they occur in part because new species occupy habitats where similar native species were not present.

Tallis, H., J. Ruesink, B. Dumbauld, S. Hacker and L. Wisehart. (2006). Eelgrass responds to oysters and grow-out methods in an aquaculture setting. J.Shellfish Res. 25(2):782.
Aquaculture has been shown to have negative impacts on eel-grass. We argue that the magnitude of tradeoffs between aquaculture and biodiversity depends on the ecological details of the production system. We found lower eelgrass density in all aquaculture systems relative to uncultivated areas in 2002-2004. Dredged beds had the lowest eelgrass densities (similar to 50% < uncultivated) while hand picked and long line beds were intermediate. Additionally, eelgrass density declined with oyster density in cultivated beds, although oysters were sparsely planted (similar to 20% cover).

Vaudrey, J., T. Getchis and B. Britton. (2006). Assessing impacts of shellfish aquaculture on eelgrass populations in eastern Long Island Sound. J.Shellfish Res. 25(2):785.
Bivalve aquaculture, specifically the utilization of submerged cultivation and depuration gear such as cages, has been implicated as a potential source of negative impacts to eelgrass populations. However, shellfish aqua-culture gear has also been shown to provide an equivalent or greater degree of ecosystem services as submerged aquatic vegetation such as eelgrass. Preliminary results suggest an increase in eelgrass growth rate, measured as sheath length. No treatment effect was seen for water column properties, sediment % organics, or benthic microalgae concentrations.

Aquaculture – Habitat

Dealteris, J. T., B. D. Kilpatrick and R. B. Rheault. (2004). A comparative evaluation of the habitat value of shellfish aquaculture gear, submerged aquatic vegetation and a non-vegetated seabed. J.Shellfish Res. 23(3:)867-874.
Comparison of the habitat value of modified rack and bag, shellfish aquaculture gear (SAG) used for the grow-out phase of the American oyster, Crassostrea virginica, submerged aquatic vegetation (SAV), Zostera marina, and a shallow nonvegetated

seabed (NVSB). The physical structure of the SAG habitat protects juvenile fish from predators and provides substrate for sessile invertebrates that serve as forage for fish and invertebrates. Therefore, we conclude that shellfish aquaculture gear has substantially greater habitat value than a shallow nonvegetated seabed, and has habitat value at least equal to and possibly superior to submerged aquatic vegetation.

Rheault, R. B.. (2006). Ecological services rendered by cultured eastern oysters. J.Shellfish Res. 25(2):766.
It has been shown that wild and cultured oysters provide many ecological services that benefit both the environment and wild oyster stocks. (Shumway et al. 2003, Newell 2004, 2002) The services rendered by commercial oyster aquaculture include: removal of nutrients (both by harvest and enhanced bacterial deni-trification); improving water clarity and water quality by enhancing sedimentation; adding larvae to the wild; sequestering tons of carbon and creating a structurally diverse habitat for other marine organisms.

Tallis, H., J. Ruesink, B. Dumbauld, S. Hacker and L. Wisehart. (2006.) Eelgrass responds to oysters and grow-out methods in an aquaculture setting. J.Shellfish Res. 25(2):782.
Aquaculture has been shown to have negative impacts on eel-grass. We argue that the magnitude of tradeoffs between aquaculture and biodiversity depends on the ecological details of the production system. We found lower eelgrass density in all aquaculture systems relative to uncultivated areas in 2002-2004. Dredged beds had the lowest eelgrass densities (similar to 50% < uncultivated) while hand picked and long line beds were intermediate. Additionally, eelgrass density declined with oyster density in cultivated beds, although oysters were sparsely planted (similar to 20% cover).

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Bivalve aquaculture, specifically the utilization of submerged cultivation and depuration gear such as cages, has been implicated as a potential source of negative impacts to eelgrass populations. However, shellfish aqua-culture gear has also been shown to provide an equivalent or greater degree of ecosystem services as submerged aquatic vegetation such as eelgrass. Preliminary results suggest an increase in eelgrass growth rate, measured as sheath length. No treatment effect was seen for water column properties, sediment % organics, or benthic microalgae concentrations.

Wisehart, L. M., S. D. Hacker, B. R. Dumbauld and J. L. Ruesink. (2006). Oyster aquaculture may positively affect eelgrass (Zostera marina L.) through enhanced seed production and germination. J.Shellfish Res. 25(2):792.
Past studies have identified both positive and negative effects of shellfish aquaculture on eelgrass but researchers have yet to address how such activities may affect eelgrass recruitment. We found higher seedling densities in dredged beds than in longlines or eelgrass beds. Germination was highest in the eelgrass beds, where, interestingly, eelgrass removal had a positive effect. Greater recruitment in dredged beds may thus be due to both enhanced seed densities as well as removal of neighboring adults. Together

these studies suggest ground culture practices may positively affect eelgrass recruitment while longlines may have a negative effect.

9. Fetch and Bathymetry

Basco, D.R. and C.S. Shin. 1993. Design wave information for Chesapeake Bay and major tributaries in Virginia. Report No. 93.1. The Coastal Engineering Institute, Old Dominion University, Norfolk, VA. 49 p.
This report discusses the development and application of a wave climate model based upon the Automated Coastal Engineering System (ACES), developed by the Coastal Engineering Research Center of the U.S. Army Corps of Engineers. This model uses historical wind data, generically referred to as hindcasting, to calculate wave climate. This model was used to produce twelve (12) wave energy maps of the Chesapeake Bay and tributary rivers in Virginia. A significant limitation of this model is that is does not consider nearshore wave transformation processes such as shoaling, refraction and wave breaking processes, i.e. water depth. Therefore, the model is intended to indicate boundary conditions to be used in finer resolution, nearshore wave transformation models.

Cooper, N.J. 2005. Wave dissipation across intertidal surfaces in the Wash tidal inlet, eastern England. Journal of Coastal Research. 21(1): 28-40.
A one-year study was conducted by the authors to characterize the role of the intertidal saltmarsh and mudflats in dissipating wave height and energy. Wave measurements were taken along three (3) different topographic transects, perpendicular to the shoreline of the Wash tidal inlet, located in eastern England. Results indicated that on average, the Wash is 83% effective in dissipating wave height and 91% in dissipating wave energy. The authors conclude that shoreline management should move away from the current structural line of defense approach and move towards management of a "deference zone" that consists of both natural and structural elements in the intertidal zone.

Hardaway, C.S. and R. Byrne. 1997. Shoreline management in Chesapeake Bay. Report. College of William and Mary, Virginia Institute of Marine Science, Gloucester Point, VA. 26 p.
In this report, the authors discuss topics such a shoreline evolution, shoreline processes, reach assessment, shoreline management strategies, and shoreline management goals and applied strategies. Under shoreline processes, the authors discuss wave climate and shoreline erosion. Fetch is described as a simple measure of relative wave energy. For their purposes, low energy shorelines have an average fetch of <1 nautical mile (nm); medium energy shorelines have an average fetch of 1-5 nm; and high-energy shorelines have an average fetch of >5 nm. 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. The authors suggest planted marshes as a viable shoreline protection strategy along low-energy shorelines where fetch exposure is <0.5 nm.

Hardaway, C.S., J. Posenau, G.R. Thomas, and J.C. Baumer. 1992. Shoreline Erosion Assessment Software (SEASware) report. College of William and Mary, Virginia Institute of Marine Science, Gloucester Point, VA. 47 p.
The authors used thirteen (13) shoreline parameters to produce the Chesapeake Bay shoreline erosion potential scale (CBSEPS), with categories of low (<2 ft./yr.), intermediate (2-4 ft./yr.), and high (>4 ft./yr.) erosion rates. These metrics include fetch, longest fetch and direction, distance from MHW to 6 ft. contour, shoreline orientation, shoreline geometry, boat wakes, bank height above MHW, bank base composition, shore zone characterization (presence/absence and width of marsh or beach), backshore zone, fastland bank condition, nearshore morphology, an nearshore aquatic vegetation. Each metric was scored, weighted, and then aggregated with all others to derive the shoreline erosion index (SEI). The fetch distances were categorized into seven (7) ranges; 0-0.1 mi., 0.1-0.33 mi., 0.33-1.0 mi., 1-3 mi., 3-10 mi. 10-20 mi., and >20 mi. Distances to the 6 ft. contour were categorized into five (5) ranges; >3000 ft., 2000-3000 ft., 1000-2000 ft., 300-1000 ft. and <300 ft. The authors note that much of the shoreline erosion observed is a direct result of high-energy storm events, which exhibit considerable temporal variability. Consequently, the cumulative erosion potential value (CEPV) depends upon storm frequency, storm type and direction, intensity and duration, and the resulting wind tides, currents and waves. Also, man-made erosion control structures have the ability to modify this erosion potential.

Keddy, P.A. 1982. Quantifying within-lake gradients of wave energy: Interrelationships of wave energy, substrate particle size, and shoreline plants in Axe Lake, Ontario. *Aquatic Botany*. 14: 41-58. (REI methodology)

Knutson, P.L. and M.R. Inskeep. 1982. Shore erosion control with salt marsh vegetation. CETA-82-3. U.S. Army Corps of Engineers, Coastal Engineering Research Center, Ft. Belvoir, VA. 23 p.

Knutson, P.L., R.A. Brochu, W.N. Seelig and M.R. Inskeep. 1982. Wave dampening in Spartina alterniflora marshes. Wetlands. 2: 87-104.
The authors developed a model based on empirical estimates of the fluid drag forces occurring on vertical cylinders, intended to simulate Spartina alterniflora stems, to describe the decay of wave energy in tidal salt marshes. Field experiments were conducted to test and calibrate the empirical model in S. alterniflora marshes on the eastern shore of the Chesapeake Bay in Virginia. The results of the study suggest that marshes create a buffer against shoreline erosion, dissipating 64% of wave energy within the first 2.5 m of marsh, and virtually no wave energy persisting beyond 30 m. Impacts to and the filling of salt marshes, especially S. alterniflora fringe marshes, will adversely affect the stability of the shoreline in many cases.

Knutson, P.L., J.C. Ford, M.R. Inskeep and J. Oyler. 1981. National survey of planted salt marshes (vegetative stabilization and wave stress). Wetlands. 1: 129-157.
The frequency and magnitude of severe wave conditions is largely influenced by local climatological patterns, fetch, and water depth. In a survey of 104 saltmarsh planting

sites (23 in Virginia) from 12 coastal states, the relative stability of the plantings was compared with various metrics comprising wave climate such as fetch, nearshore slope, offshore depth, shoreline configuration, sediment grain size, shoreline orientation with respect to prevailing and storm winds, and boat wakes. The authors found that of these parameters, fetch, shoreline configuration and sediment grain size were the most highly correlated with planting stability. The authors use these three parameters to develop a model, the Vegetative Stabilization Site Evaluation Form, for evaluating the likely success of potential marsh planting sites. Cumulative scores for four metrics (two used for fetch) are characterized into three probabilities for success: 15%, 50% and 100%. This model is appropriate for calculating the likelihood of a successful planting effort, and can be utilized to determine a site's suitability for vegetative treatments in lieu of rock and wooden structures. The authors acknowledge that both fetch and water depth play significant roles in wave climate for a site.

Möller, I. 2006. Quantifying saltmarsh vegetation and its effect on wave height dissipation: Results from a UK East coast saltmarsh. Estuarine, Coastal and Shelf Science. 69: 337-351.
Wave attenuation across any surface is known to vary with relative wave height. This study uses digital imaging to quantify density, height and structure of three (3) macrotidal saltmarshes in Dengie peninsula, Essex, UK. At two (2) of the study sites dominated by Spartina spp., the relative incident wave height (wave height/water depth) exerted a statistically significant positive control on wave attenuation up to a threshold value of 0.55, beyond which no additional wave attenuation was observed. However, variability in wave attenuation was high, and different vegetation characteristics between the three (3) transects did not exert a statistically significant influence on wave attenuation. The authors report that wave attenuation across the three (3) 10 m transects sampled in the study varied from 0.08% in 0.4 m of water depth to 33% in 0.2 m of depth.

Perry, J.E., T.A. Barnard, Jr., J.G. Bradshaw, C.T. Friedrichs, C.T. Havens, P.A. Mason, W.I. Priest, III, G.M. Silberhorn. 2001. Creating tidal salt marshes in the Chesapeake Bay. Journal of Coastal Science. Special Issue No. 27: 170-191.
The authors provide a review of the existing knowledge of tidal salt marsh creation in the Chesapeake Bay, and provide recommendations for the appropriate siting, design, and construction methods. Three case studies are presented along with a discussion of lessons learned and suggestions for improvements for future design and construction. The authors include fetch in their discussion of recommended design and construction methods. They discuss work done by Hardaway et al. (1980, 1981) that suggested that marshes planted on naturally nonvegetated, intertidal flats did poorly if fetch exceeded 1,600 m. Hardaway and Byrne (1997) recommend planting marshes for erosion control on low energy shorelines with fetches of less than 800 m. The vegetative stabilization site evaluation methodology developed by Knutson et al. (1981) is also discussed, which uses average and greatest fetch distances across open water, along with shoreline geometry and sediment grain size to determine the percent success rate of salt marsh creation sites. Using their site evaluation method, Knutson et al. (1982) suggest that a tidal salt marsh with of fetch >3,000 m has only a 44% probability of survival.

Roland, R.M. and S.L. Douglass. 2005. Estimating wave tolerance of Spartina alterniflora in coastal Alabama. Journal of Coastal Research. 21(3): 453-463.
The authors compared two (2) existing methodologies to estimate wave climate and site suitability for marsh creation in coastal Alabama. Nine (9) sites were selected for this study, with and without Spartina alterniflora. The Keddy (1982) wave exposure index, which uses wind-wave hindcasting, and the Knutson et al. (1981) vegetative stabilization site evaluation methods were used to determine wave climate and suitability of the site to sustain a marsh. The results from the Keddy (1982) model indicated that vegetated wetland sites had the smallest hindcast wave heights and nonwetland (intertidal beach) shorelines had the highest. Sites where the marsh was eroding exhibited intermediate hindcast wave heights. The authors caution that although the Keddy (1982) method (or REI) may be an effective tool in some situations, it may be difficult for those wishing to apply the model to determine the appropriateness of their situation, which limits the model as a tool for wave climate evaluation of wetlands. Incorporating water depth, which can limit wave heights, could improve the utility of this model. Likewise, the authors believe it is unclear how the Knutson et al. (1981) method could be applied to the design of created marshes.

Shafer, D.J., R. Roland and S.L. Douglass. 2003. Preliminary evaluation of critical wave energy thresholds at natural and created coastal wetlands. WRP Technical Notes Collection (ERDC TN-WRP-HS-CP-2.2), U.S. Army Engineer Research and Development Center, Vicksburg, MS.
The purpose of this study was to evaluate a shallow-water wave hindcasting method for determining wave energy at natural and created wetland sites in order to identify critical wave climate thresholds for success of created wetlands. In addition, the critical wave heights identified in this study can be used to help design minimally sized defensive structures that are capable of protecting wetland creation sites. The authors developed their hindcast approach to address shortcomings of various wave climate models such as Relative Exposure Index (REI) which 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. The authors note that the potential effect of water depth is not explicitly accounted for in the REI methodology, and because wave height can decrease as a wave propogates from deep to shallow water, the inability of this methodology to account for the effect of water depth on wave climate reduces the applicability of REI for siting potential wetland planting sites. To overcome this deficiency, the authors generated the nearshore wave climate using wave hindcasting, where wave height, fetch, wind speed, and average water depth data are input to a computer program based on shallow-water wave generation models (WaveGen; Weggel and Douglass, 1985). With no previous studies to help guide the selection of a wave height statisic, the authors arbitrarily selected the 50th and 20th percentile exceedence wetland wave heights in this study for comparison with observed shoreline stabilization and vegetation characteristics. The 50th percentile represents "average" conditions (exceeded 50% of the time) and the 20th percentile represents more extreme conditions. Using a hindcast method to calculate a critical wave height (wave height statistic), the authors found good agreement with the REI model (Keddy, 1982).

van der Wal, D. and K. Pye. 2004. Patterns, rates and possible causes of saltmarsh erosion in the greater Thames area (UK). Geomorphology. 61: 373-391.
The authors describe a number of factors affecting lateral erosion and interior dissection of salt marshes along the greater Thames estuary, among others. Several causes are attributed to this loss such as the filling of wetlands and construction of embankments that have increased tidal range and current velocities. The large amount of erosion experienced during the 1970's is attributed to a change in the wind-wave climate by the authors. The basic methodology employed is an analysis of published and unpublished documentary evidence along with a time series comparison of historical aerial photographs. This information was used to identify changes in mean low and mean high water along with morphological changes to the shoreline, thus allowing for status and trends analysis. The authors 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, P. 2001. Restoring physical processes in tidal wetlands. Journal of Coastal Research. Special Issue No. 27: 149-161.
This paper discusses the development of a methodology for the restoration of physical processes to create and sustain tidal wetland habitat values based on a geomorphic understanding of the natural evolutionary trajectory of restored tidal marshes. The restoration design methodology presented by the authors includes limiting fetch to a maximum of 300 m to insure natural sedimentation of the marsh is not impeded and to allow for vegetation to naturally colonize and maintain the perimeter of the site.

Shoreline Project Review

Project Review Protocol

Individual permit review is based on three primary considerations:

- the need for shoreline management created by the existing or intended upland use;
- the risks created by shoreline and upland management alternatives; and
- the goal of preserving or enhancing ecosystem services that provide public benefits.

Individual permit reviews begin from the assumption that the intended use represents an informed local decision about the consequences of development options for the shoreline reach and local watershed. From this basis, project review is intended to identify preferred management alternatives that:

1. allow the use permitted by zoning
   This step involves elimination of shoreline management alternatives that would prohibit intended use of the site. It does not, however, avoid consideration of altered site planning or reduced intensity of use that may lessen risk and/or minimize impacts to ecosystem services.
2. reduce on-site risks to both use and ecosystem services
   This involves preserving and/or enhancing the riparian buffer to the maximum extent possible consistent with the intended use. It also involves consideration of the long-term impacts of the site design for water quality, habitat, and sediment stabilization in the riparian and littoral zones.
3. reduce off-site risks to existing uses and ecosystem services
   This step seeks to ensure that the on-site shoreline management alternatives do not increase risks on adjacent properties for existing uses. This includes consideration of increased erosion potential, decreased sediment supply, and increased risk to existing defensive structures. This assessment also considers the impacts of alternative management strategies on the ecosystem services (particularly water quality and habitat) currently provided by adjacent properties.
4. maximize the potential for the site to continue to provide ecosystem services that benefit the public
   Within the constraints of the foregoing considerations, the management alternatives that provide the greatest potential for sustained ecosystem services on-site will be identified as the preferred strategy.

To accomplish this we employ several models:

An **erosion vulnerability model** is used to classify shoreline reaches according to the probability that the intertidal and riparian features will persist in the face of natural events. This model assesses the potential for shoreline retreat due to erosion and/or inundation, and the potential for shoreline features, such as marshes and forested buffers, to persist. This model is used to assess the need for shoreline management to support the intended site use.

The erosion vulnerability model is based on the probability that site conditions will permit significant wave energies to strike the shore. This assessment is based on an integration of: fetch (unobstructed distance over open water), nearshore bathymetry (the slope of the bottom next to the shoreline), orientation (predominate direction the shoreline reach faces), and the existing erosion protection on site whether natural (marsh, reef, sand bar), or anthropogenic (bulkhead or other revetment). The assessment characterizes the shoreline segments as being at high, medium, or low risk for continuing shoreline erosion. As such, the assessment evaluates the relative need for managing a shoreline based on natural processes.

A **site development impact model** is used to characterize the potential for a realized site plan to impact:

- littoral zone water quality through alteration of storm/groundwater flows and quality;
- riparian and littoral habitat services through alteration of land use/land cover; and
- riparian and intertidal sediment stability through alteration of storm water flows.

This model is used to identify alternative site development plans that can minimize impacts to a site's long-term capacity to provide ecosystem services with public benefits.

The model is based on existing site conditions. The location and type of existing structures on the site is considered in light of the erosion vulnerability assessment. This determines if there is an obvious need for shoreline management. In the case of new development, the site plan is considered to determine if risk is being unnecessarily created in locating structures. Potential impacts to ecosystem services are evaluated by considering existing riparian and intertidal vegetation, and current bank condition (stable, eroding, undercut). Alternative development strategies are indicated based on: reduction in long-term risk to structures; preservation/enhancement of vegetative cover; preservation/enhancement of contact between vegetation and runoff/shallow groundwater flows; and minimization of any disruption of connections between riparian, intertidal and subaqueous environments.

A **management strategy impact model** is used to characterize the potential for any particular shoreline management plan to affect conditions in adjacent properties. This model considers the potential of management alternatives to increase erosion on adjacent properties, diminish beneficial sediment transport, diminish the effectiveness of adjacent existing shoreline management efforts, increase flooding potential on adjacent properties, or create some other detrimental off-site impacts.

The model is based on existing management strategies on adjacent properties. If adjacent shorelines are unmanaged, then the preferred management strategy will be one that does not reflect energy or significantly alter sediment transport pathways. If adjacent shorelines have defensive structures, then preferred strategies will be ones that allow structures along the entire reach to work together effectively. This may result in avoidable short-term impacts to ecosystem services on the subject property in the interest of sustained performance of existing management strategies on adjacent properties.

Ecosystem service models are used to evaluate the potential that a site has for providing beneficial water quality, habitat, and sediment stabilization services to the local system. The models are based on the combination of physical and biological features that create and sustain capacity to deliver these services. As such, the models provide guidance for the maintenance and/or creation of desirable physical and biological features in shoreline systems.

Project Review Guidance

1. Opening Statement – Project Assessment

Inconsistent with integrated guidance, impacts can be avoided, no action

We have determined that the proposed project is inconsistent with an integrated approach to shoreline management and that the impacts can be avoided. It is our opinion, given the site conditions, that no action is necessary at this time.

Inconsistent with integrated guidance, impacts can be reduced

We have determined that the proposed project is inconsistent with an integrated approach to shoreline management and that the impacts associated with the proposed project can be reduced.

Inconsistent with integrated guidance despite minimal impacts

We have determined that although the impacts to wetland resources will be relatively minor, the proposed project is inconsistent with an integrated approach to shoreline management.

Consistent with integrated guidance, impacts have been minimized

We have determined that the proposed project is consistent with an integrated approach to shoreline management and that the impacts associated with the proposed project have been minimized to the extent possible given the site conditions.

Consistent with integrated guidance, impacts can be reduced

We have determined that the proposed project is consistent with an integrated approach to shoreline management. However, it is our opinion that the impacts associated with the proposed project can be reduced.

Consistent with integrated guidance, impacts justified

We have determined that the proposed project is consistent with an integrated approach to shoreline management and that the impacts, though significant, are justified given the site conditions.

2. Risk Assessment

Erosion Risk
Based on an assessment of various parameters including fetch, orientation, nearshore bathymetry, bank condition and existing natural or man-made erosion protection, we have determined that the risk of continued shoreline erosion at this location is low.

Based on an assessment of various parameters including fetch, orientation, nearshore bathymetry, bank condition and existing natural or man-made erosion protection, we have determined that the risk of continued shoreline erosion at this location is moderate.

Based on an assessment of various parameters including fetch, orientation, nearshore bathymetry, bank condition and existing natural or man-made erosion protection, we have determined that the risk of continued shoreline erosion at this location is high.

Development Risk
Existing upland improvements are at risk for damage or loss due to shoreline erosion at this location.

The proposed upland development includes improvements that will be at risk for damage or loss due to their proposed location.

Various elements of the proposed upland use including siting, impervious surface, landscaping, and the accommodation of effective and preferred erosion control and bank stabilization, create the potential for adverse risk to adjacent properties and water quality and habitat ecosystem services.

Proposed Action Risk
Revetment
The proposed revetment will sever the connection between riparian, intertidal and subaqueous areas and convert native soils and vegetated areas to non-native rock. The result is a loss in the provision of water quality improvement processes and a change in the benthic community and associated forage animals.

Bulkhead
The proposed bulkhead will sever the connection between riparian, intertidal and subaqueous areas, alter the natural curve of the shoreline, remove undercut crevice habitat, reduce shallow water habitat, and may result in the loss of wetland and upland vegetation during or following construction. Bulkheads also change nearshore wave dynamics, may cause increased erosion on adjacent properties, and typically contribute to their own demise by reflecting wave energy to erode the substrate channelward of the structure.

Breakwater

The proposed breakwater will cause the conversion nearshore shallow waters to rock and sandy shoreline. This will cause a shift in the benthic community and associated forage by crustaceans and shorebirds. The construction of the breakwater will cause temporary water quality impacts and may interrupt sediment transport. Breakwaters are effective in certain shoreline settings and when designed for a shoreline reach.

Groin
The proposed groin(s) will, by design, interrupt sediment transport along shore. This may result in a downdrift sediment deficit associated with increased erosion risk and the loss of intertidal habitats.

3. Automated Language for Project Review

Definitions

Water Dependent

In granting or denying any permit for use of State-owned submerged lands and the waters overlying those lands, the Commission will consider, among other things, the effect of the proposed project upon: other reasonable and permissible uses of State waters and State-owned submerged lands; marine and fisheries resources, wetlands, adjacent or nearby properties; anticipated public and private benefits, submerged aquatic vegetation, and water quality. The Commission will also consider the water-dependency of the project and alternatives for reducing any anticipated adverse impacts.

As defined by the Commission, water dependent means "those structures and activities that must be located in, on, or over State-owned submerged lands." When applying this definition, both of the following questions must be answered affirmatively:

1. Is it necessary that the structure be located over water? and,
2. Is it necessary that the activity associated with the structure be over the water?

Use of the definition for water dependency does not necessarily preclude issuance of a permit for non-water dependent structures over State-owned submerged lands. At public hearing, the Commission may determine that, while a structure is not water dependent, it is a reasonable use of State-owned submerged lands. These types of projects are evaluated on a case-by-case basis.

1. Shoreline Erosion

<u>Descriptions of shoreline types / projects</u>

1. No Action
    a. No erosion
    b. Marsh spit

    c. Under cut banks

2. Non-Structural
    a. Minor erosion
        i. Plantings: eroding marsh. Narrow, low energy
        ii. Veg. Modification: Shading
    b. Bank modification and planting:
        i. Banks > 6 feet
        ii. Upland erosion
    c. Beach nourishment – Sandy shoreline
3. Hybrid
    a. Marsh Sill: Eroding marsh wide
    b. Marsh enhancement w/ sill: Eroding marsh narrow, mod energy
    c. Fiber logs: Undercut banks
    d. Marsh/ Flat Creation w/ sill: Banks > 6 feet
    e. Nourish w/ structure: Sandy shoreline
    f. Revetment: Low bank/ Riparian forest
3. Serviceable Bulkhead replacement, restricted navigation
4. Serviceable Bulkhead, no navigational restrictions
5. Qualifying Conditions
    a. Sea Level Rise
    b. Off-shore SAV beds
    c. Upland modifications

Eroding dune (this option will be handled under the Dunes section)

Shoreline types comments

Opening statement (for inclusion with all projects)

Natural shorelines tend to be dynamic and interconnected with the surrounding landscape and vegetative and animal life. Intertidal and riparian areas provide numerous water quality and habitat benefits. Sandy shorelines and banks contribute to the overall sand budget of tidal rivers, bays and the ocean. Any action on one shoreline has the potential to impact adjacent and downdrift shorelines; therefore, activities that impact sub-aqueous, intertidal and riparian zones should be avoided whenever possible. When erosion along a shoreline has the potential to result in significant loss of property and upland improvement, the consideration of shoreline erosion protection activities may be appropriate. Preserving, creating or enhancing natural systems such as marshes, beaches and dunes is always the preferred approach to shoreline erosion protection.

1. No Action

a. No erosion/minor erosion

We question the need for the proposed action along this shoreline. There are no structures at risk and minor or no shoreline erosion. The location, orientation and morphology of the shoreline are indicative of quiet, low energy conditions.

b. Marsh spit

Marsh spits are notably mobile over time in response to patterns in prevailing winds and currents as well as catastrophic storm events. In addition, changes in composition and location of the vegetated community due to sea level rise are part of the natural process. Efforts to protect the marsh are likely to be ineffective and will result in some direct adverse impacts, and un-quantifiable indirect impacts, to the marsh and the adjacent marine system.

c. Sustainable erosion

2. Non-Structural

a. Minor erosion

i Marsh and or riparian plantings: Eroding marsh, narrow, low energy

We recommend the use of vegetation to abate minor erosion. The planting of shoreline vegetation provides natural shoreline stabilization as well as habitat and water quality benefits.

When erosion is present in areas with narrow marsh fringes, shoreline protection may be increased by widening the marsh. This may be accomplished by planting existing sediment with wetland vegetation (where existing elevations are appropriate) or creating additional intertidal area and planting it.

ii. Upland erosion

Erosion caused by overland flow (generally runoff from rain events) is frequently found on shorelines with high or steep banks or a lack of riparian vegetation. Erosion can become worse when the property is developed; increased impervious surface (roofs, driveways, etc.) prevent water from soaking into the ground. Evidence of upland erosion includes rills, gullies and exposed roots at or near the top of the bank.

Shoreline structures are not an effective method for dealing with upland erosion because they are placed channelward of where the erosion is occurring. Managing the quantity and rate of runoff in the upland is the most effective method for preventing continued erosion. Two of the easiest methods of slowing runoff are: planting additional vegetation at the top of the bank and allowing grass near the edge the top of the bank to grow without mowing. Other methods for handling runoff include: gutters, drains, rain gardens, dry wells, berms, and level spreaders. More information on methods for managing runoff can be found at the Department of Conservation and Recreation's website (http://www.dcr.virginia.gov/soil_&_water/index.shtml). Additional information on rain gardens can be found at the Virginia Department of Forestry's website (http://www.dof.virginia.gov/rfb/rain-gardens.shtml).

ii. Vegetation Modification: Shading Effects

Wetland vegetation is generally found in areas with low to moderate fetch and sufficient sunlight. In many areas with appropriate wave climates, wetland vegetation is excluded from growing in the intertidal area due to insufficient light generally resulting from overhanging tree branches. On shorelines without erosion, it is preferable to leave the riparian vegetation in place.

Minor erosion along these shorelines can often be stopped by planting or encouraging the growth of marsh vegetation in the intertidal area. Overhanging branches must be removed to allow sufficient light to enter the intertidal area. Trees should be preserved where possible to prevent loss of riparian services. If marsh grass is found along adjacent shorelines, the grass may colonize the intertidal area naturally. However, colonization is generally faster if marsh grasses are planted along the shoreline.

If the scarp or undercut is too high, vegetation alone may not be enough to protect the toe of the bank. In these situations, additional protection may be gained by increased the width of the intertidal area prior to planting marsh vegetation. In moderate and some low energy settings, a rock sill may be needed to sustain the intertidal area.

b. Bank Modification and Planting

i. Eroding banks: no cover/lawn

The preferred alternative for bank stabilization is to grade the bank to a maintainable slope and used in combination with vegetative plantings to provide the desired bank stabilization. Bank modifications should be limited to those portions of the bank that need stabilization.

c. Beach Nourishment

The preferred approach to shoreline protection for sandy shoreline is to enhance the natural capacity of the sand to provide the desired erosion protection through beach nourishment.

The placement of sand on the beach can impact habitat of protected species (e.g. sea turtles, Northeastern beach tiger beetle). Time of year restrictions may be required for protected species. Temporary water quality impacts are also possible if fill contains a large amount of fine-grained material and it is exposed to wind or wave erosion.

Manipulation of beach fill may also result in artificial dune lines that are not reliable indicators of suitable building and access locations. Structures should be sited based on indicators of the natural dune line.

3. Hybrid

a. Undercut banks

The preferred shoreline management approach for an undercut bank which is otherwise stable bank is to allow the natural process to continue unchanged.

The undercut bank appears unstable and the potential for bank failure causes a threat to upland improvements, therefore treatment of the undercut is appropriate. Natural fiber logs can be placed at the undercut to protect against erosive forces. The logs eventually decay while trapping sediment and raising the elevation of the eroding area.

If the riparian cover is sparse, or lawn, the undercut can be eliminated by grading the bank back from the undercut toe. The newly graded area should be vegetated with wetland and native riparian species at appropriate elevations. The vegetation will serve to stabilize the bank, prevent further bank failure and increase the capacity to trap and treat overland runoff.

b. Eroding Banks > 6 feet: nourish and sill/ veg or not

The toe of the bank is at or a short distance above mean high water. The lack of distance between the base of the bank and tidal waters allow erosion of the bank. The preferred option is to create distance between the bank and tidal waters by building a wetland. This may be a non-vegetated sand flat, or a vegetated marsh. In moderate and some low energy settings, a rock sill may be needed to sustain the flat.

Erosion at the base of the bank has caused slumping and failure of the bank. Some bank grading may be necessary to create a more stable slope. Bank modifications should be limited to those portions of the bank that need stabilization. The graded bank should be densely re-vegetated with native small trees, shrubs and deep-rooted grasses.

In the process of trying to reach a stable slope, some banks have already slumped to a slope of two to one or flatter at the bottom. These areas may be colonizing with vegetation or already vegetated. The top of the bank remains vertical and unstable. Upland runoff and heavy trees can contribute to continuing, or catastrophic bank failure. Selective grading of the unstable portions followed by re-vegetation with native small trees, shrubs and deep-rooted grasses is appropriate.

c. Sandy shorelines

The preferred approach to shoreline protection for sandy shorelines is to enhance the natural capacity of the sand to provide the desired erosion protection. The critical element of this approach is beach nourishment in combination with a rock structure. The rock structure can be a nearshore sill, or offshore breakwater(s).

The placement of sand on the beach can impact habitat of protected species (e.g. sea turtles, Northeastern beach tiger beetle). Time of year restrictions may be required for protected species. Temporary water quality impacts are also possible if fill contains a large amount of fine-grained material and it is exposed to wind or wave erosion.

Manipulation of beach fill may also result in artificial dune lines that are not reliable indicators of suitable building and access locations. Structures should be sited based on indicators of the natural dune line.

d. Eroding marsh, wide (greater than 15 feet)

To protect the toe of a wide, eroding marsh, the preferred alternative is a marsh toe structure. This may be constructed of biologs, rock, or other appropriate material. The structure should be small (no more than one foot higher than the marsh scarp) placed at the channelward edge of the marsh.

The length of the proposed structure warrants a gap, or dip in structure height, to allow better connection to tidal waters. An off-set gap, with a slight overlap of the structure sections, may be the least subject to erosion of the marsh.

e. Eroding marsh, narrow, medium energy

The preferred approach is to enhance shoreline protection by widening the existing marsh to provide more wave attenuation and therefore more effective upland protection. The installation of a stone sill channelward of the existing marsh grass may be necessary to provide long-term protection for the created marsh.

f. Eroding low bank, non-veg: forested

The preferred choice for erosion protection on this heavily forested shoreline is a small riprap revetment placed at the toe of the bank. In this setting, grading the bank would result in the removal of a large square footage of forest, which would have more ecological impact than the conversion of nonvegetated wetland to revetment.

3. Serviceable Bulkhead replacement, restricted navigation

Bulkheads are generally not the preferred option for stabilizing a shoreline. However, in some cases, restricted navigation makes bulkheads the most logical option for shoreline stabilization. In this situation, the bulkhead should be replaced in the current alignment wherever possible and no further than 2-feet channelward of the existing bulkhead where necessary.

4. Serviceable Bulkhead, no navigational restrictions

In areas of high wave energy, areas with high/steep banks or where upland improvements are too close to the shoreline to permit grading, the preferred alternative is to either to remove the bulkhead and replace it with a revetment, aligned landward of the existing bulkhead, or to construct a bulkhead toe. Although revetments may cover more square footage than bulkheads, they are less environmentally intrusive, provide habitat opportunities and have less impact on nearshore wave dynamics.

5. Special Conditions

a. Sea-level rise (tidal flooding)

The shoreline shows no evidence of erosion, but shows signs of tidal flooding. As sea level rises, continued encroachment of tidal waters has resulted in the conversion of upland lawn to vegetated wetland. Shoreline erosion techniques are generally not appropriate to address tidal flooding. The preferred option is to relocate structures inland from the flooding waters or raise the elevation above flood level. Consideration may be given to the use of a revetment or soil berm (levee) placed landward of the wetlands to provide protection from flooding.

b. Off-shore SAV bed

Submerged Aquatic Vegetation (SAV) beds are highly productive ecosystems which provide food and habitat for several fisheries species and help improve water quality by stabilizing sediments and reducing turbidity. The range of SAV beds in the Chesapeake Bay watershed has been greatly reduced from the range in the 1930s, which makes these beds of prime concern for conservation. Impacts to SAV beds should be avoided by re-locating structures either to the side of the beds or on-shore. Although the creation of intertidal areas can have beneficial consequences, the conversion of SAV beds to intertidal land is not considered an appropriate habitat conversion.

c. Upland Limitations

The use of some management options is limited by choices already made or planned on the upland. Houses and other upland improvements should always be placed well landward of the wetlands, the riparian buffer, the dune or the top of a failing bank. A structure placed too close to the waterway eliminates the use of often-preferred erosion control options and complicates access for construction. Consideration should be given to relocating existing structures to minimize or eliminate the risk of losing the structure.

Absent upland modifications to accommodate preferred options, the impacts resulting from the proposed structure are considered acceptable.

Mitigation/Compensation

Bridges

2. Beaches & Dunes

Descriptions of project types

1. Residential, Commercial & Accessory Structures
2. Beach Access Structures
3. Dune Leveling & Relocation
4. Dune Restoration
5. Stabilization Structures
6. Beach Nourishment
7. Dredging on Sand Beaches

Beach and Dune Comments

Opening Statement (for inclusion with all projects)

Coastal Primary Sand Dunes perform a host of ecological services that benefit adjacent ecosystems as well as the human inhabitants of these areas. As impacts to these areas represent both an ecological and an economic impact, any proposed activities should be examined carefully to weigh the consequences for both public and private interests in these areas. Coastal primary sand dunes serve as protective barriers from flooding and erosion, provide reservoirs of sand to replenish the beach zone, and provide habitat for a variety of plants and animals. Plants adapted for life on coastal primary sand dunes must tolerate very limited amounts of fresh water, constant salt spray, and withstand marked variations in temperature. The natural vegetation occurring on sand dunes can act as a baffle, slowing wind speed and causing wind-borne sand to settle and be trapped in the vegetation resulting in accretion of the dune.

1. Residential, Commercial & Accessory Structures

Single & multi-family residences, commercial buildings, swimming pools, parking areas, gazebos, and other structures can adversely affect the composition, form and function of dunes because they interfere with wind and sand deposition patterns and natural dune building processes. Structures may also shade or displace dune vegetation. Dune vegetation is important because it slows wind speed and causes wind-blown sand to settle and be trapped resulting in accretion of the dune.

Structures on slab foundations or designs other than open piling may exhibit structural failure and do not allow for the natural migration of the dune. They usually require excavation, which reduces stability of the primary dune and reduces the amount of sand available for flood and erosion protection. They may also be subject to burial, which may result in frequent excavation and movement of sand (see Dune Relocation).

All structures should be located landward from the dune system. If encroachment is necessary, then the footprint should be minimized and limited to the dune backface. The dune crest, dune face and beach backshore should be avoided. Since the dune backface is the most active sand deposition area, only elevated open-pile foundations should be used; slab foundations and other structures that require excavations should be avoided. If sand must be excavated, it should remain in the local vicinity and be strategically placed and stabilized with vegetation to enhance the existing beach and dune features.

2. Beach Access Structures

Pedestrian or vehicle traffic through dune vegetation will create a breach for storm waves through the dune line. Open-pile walkways & decks over dune vegetation will have shading impacts. Access structures are subject to failure during storms, resulting in scattered solid waste debris.

Elevated open-pile structures are preferred to minimize disturbance of vegetation and natural dune building processes. Open-pile structures should be as direct and narrow as possible. The elevation depends on dune crest and vegetation height. Designated access points are preferred over multiple paths or elevated walkways. Construction mats placed on grade can be used for temporary vehicle access.

3. Dune Leveling & Relocation

Leveling a dune removes the buffering capability and reduces the amount of sand available for the adjacent beach. The protective capabilities of dunes are reduced not only at the site, but for adjacent areas as well. If dune vegetation is removed, then the associated sand accretion, stabilization and habitat functions are also reduced.

Dunes should be maintained as a relatively uniform, uninterrupted dune line in order to offer the maximum flood and erosion protection. Relocating part of a dune line creates a breach and hazard for property behind the relocated dune and adjacent properties.

Dune alterations are not advised. If they are considered necessary, then strategically place the sand to enhance natural beach and dune features and stabilize with vegetation.

4. Dune Restoration

Actions to create more extensive, better-stabilized dunes are generally beneficial. They should be designed to enhance natural dune lines, height and vegetation communities. Clean, coarse-grained sand should be imported from an upland source; existing beach and dune sand should not be “borrowed”.

5. Stabilization Structures

The natural position of a dune is the result of a balance of wind, waves and storms at any given time. Continuous sand movement between offshore sand bars, the beach and dune is necessary for dynamic dune building processes plus flood and erosion protection. Structures to stabilize a dune in a particular location inhibit this natural fluctuation and buffering capability. Reflected wave action may increase beach erosion and habitat for beach-dependent species may be impacted.

Stabilization methods and structures that enhance the natural dynamics of the beach-dune ecosystem are preferred (e.g. sand fences, offshore breakwater system, sometimes groins). Stabilization structures are not advised on the dune face, dune crest or beach backshore.

If armoring is considered absolutely necessary, the structure should be covered with sand and dune vegetation planted where it will not be subject to regular wave action. Dune vegetation disturbed by construction activity should be restored. Fill material for geo-textile bags should not be obtained from other beach and dune areas.

6. Beach Nourishment

The placement of sand on the beach can impact habitat of protected species (e.g. sea turtles, Northeastern beach tiger beetle). Existing dune vegetation may be buried. Manipulation of beach fill may also result in artificial dune lines that are not reliable indicators of suitable building and access locations. Temporary water quality impacts are also possible if fill contains a large amount of fine-grained material and it is exposed to wind or wave erosion.

Only clean sand fill that contains at least 90% coarse-grained sand should be used. Time of year restrictions may be required for protected species. The beach nourishment area should be stabilized with appropriate vegetation. Structures should be sited based on indicators of the natural dune line.

7. Dredging on Sand Beaches

Dredging to remove sand or change sand transport patterns to maintain navigation channels reduces the sand supply available for storm protection and interferes with natural dune building processes. Dredging on sand beaches should be avoided. Sand bypass systems should be considered.

3.Utility Crossings

Subaqueous

For crossings that may occur in tidal waters where the activity impacts subaqueous bottom, we anticipate a disruption of the benthic community and a localized increase in turbidity. We expect these impacts to be relatively short-lived, especially in the larger waterways. Conducting the work quickly and as cleanly as possible may minimize the quantity and duration of the adverse effects from increased turbidity. We concur with the Department of Game and Inland Fisheries on the recommended time of year restrictions

Wetlands: Brackish

For crossings that may occur in tidal waters where the activity impacts wetlands, we anticipate a disruption of the benthic community and a localized increase in turbidity. We expect these impacts to be relatively short-lived. Conducting the work quickly and as cleanly as possible may minimize the quantity and duration of the adverse effects from increased turbidity. We concur with the Department of Game and Inland Fisheries on the recommended time of year restrictions

Wetlands: Fresh water

Staging Area

All areas impacted during repair should be identified including the actual area of excavation, equipment staging and dredged material stockpiles.
The restoration plan should include a monitoring protocol and timeline. Milestones should be provided as to the chosen protocol for assessment restoration success (i.e., vegetated cover, density, stem count, etc.). In areas where adjacent *Phragmites* elevates the risk of invasion, the protocol should include options for *Phragmites* control.

4.Aquaculture Activities

Aquaculture comments

Shellfish Aquaculture

Shellfish are an important component of the Chesapeake Bay ecosystem. They help increase water clarity by filtering their surrounding water, contribute to the aquatic food chain and beds and reefs serve as habitat for other aquatic species. While generally considered beneficial, impacts expected to result from aquaculture projects include temporary re-suspension of sediments resulting from aquaculture practices and the loss of aquatic bottom for other resources.

Use of aquaculture BMPs, appropriate to the particular aquaculture operation, can minimize adverse environmental impacts.

Aquaculture over SAV beds

VIMS Submerged Aquatic Vegetation (SAV) reports for the most recent 5 years time span indicates that the proposed deployment area is located immediately adjacent to or within SAV habitat. This SAV data may be accessed at http://www.vims.edu/bio/sav.

Submerged Aquatic Vegetation (SAV) beds are highly productive ecosystems which provide food and habitat for several fisheries species and help improve water quality by stabilizing sediments and reducing turbidity. The range of SAV beds in the Chesapeake Bay watershed has been greatly reduced from the range in the 1930s, which makes these beds of prime concern for conservation. Impacts to SAV beds should be avoided by moving the operation to deeper water or a different area.

Artificial reefs

Construction of artificial reefs is one component of aquatic resources management. Generally considered beneficial, there are some anticipated impacts from this activity. The construction activity temporarily disrupts the bottom and causes localized increased turbidity. The structure will cause a change from the existing bottom type to another. However, the structure will also create habitat for attached organisms attracting the desired finfish and crustacean predators.

5.Temporary Impact areas

<u>Temporary Impact comments</u>

General Comments for Temporary Impact Areas

Temporary impacts are defined as any activities that result in a temporary loss of ecosystem services such as habitat or water quality functions but do not result in a permanent loss of these functions. These activities may include staging areas, equipment crossings, stockpiling, or excavations for the installation of utility crossings, or other such activities that do not involve the permanent loss of marine resources.

Wetlands and subaqueous lands should not be used to stockpile dredged material, construction materials, or equipment. We recommend that any temporary impact in wetlands or subtidal areas be limited to only that which is necessary for construction or installation of the proposed project and that appropriate erosion and sedimentation controls be installed outside of the impact areas to minimize additional secondary impacts to adjacent wetlands and waterways. We would recommend that all impact areas be restored to their pre-construction contours and planted with appropriate wetland plantings on 12-18 inch centers to aid in the reestablishment of wetland vegetation.

We recommend that the applicant provide a detailed restoration plan including scaled, geographically referenced drawings for any temporary impacts to wetlands resulting from permitted activities. This plan should include adequate details to allow for the assessment of the likely success of wetland restoration. All areas to be temporarily impacted should be identified including any areas of excavation, equipment staging, or dredged material stockpiles. The restoration plan should include a monitoring protocol and timeline. Milestones should be provided as to the chosen protocol for assessment restoration success (i.e., vegetated cover, density, stem count, etc.). In areas where adjacent *Phragmites* elevates the risk of invasion, the protocol should include options for *Phragmites* control.

1. Temporary Impacts to Subtidal Areas

For any temporary impacts to subaqueous bottom, we anticipate a disruption of the benthic community and a localized increase in turbidity. We expect these impacts to be relatively short-lived. Conducting the work quickly and as cleanly as possible may minimize the quantity and duration of the adverse effects from increased turbidity.

2. Temporary Impacts to Freshwater Wetlands

The following additional recommendations are pertinent to any proposed temporary impacts in tidal freshwater wetlands. These wetland communities have soils that are typically fine-grained, comprised of silts and clays. Loading these soils with heavy equipment, even on mats, may result in compaction and or displacement in the form of a "mud wave". A restoration plan for these wetlands may need to provide for the post-construction removal or replacement of displaced soils and/or the addition of compatibly sized material to achieve the appropriate grade for compacted areas. In addition, the restoration of tidal freshwater vegetation is difficult as the plants are highly sensitive to variations in elevation, and often have reduced timelines for successful vegetative growth and re-establishment

6. Dredging Activities

Dredging checklist

1. Dredging not justified/no action
2. Dredging avoidance/minimization by piering out
3. Dredging avoidance/minimization by lessening extent of dredging
4. Dredging in wetlands
5. Dredging adjacent to wetlands (4x buffer)
6. Dredging in shallow, narrow creeks and coves
7. Dredging within shellfish areas, SAV beds, and other areas of high productivity
8. Dredging in anadromous fish spawning and nursery areas (TOY)
9. Dredging in shellfish resource areas (TOY)
10. Dredging in industrial areas, other areas of potential contaminated sediments, including, but not limited to:
    a. Parts of the Elizabeth River
    b. Hopewell
    c. Paradise Creek
    d. Ft. Eustis/Skiffes Creek (PCB's, & others)
    e. Almost any military installation
    f. Newport News shipyard
11. Dredging deeper than controlling depth
12. Non-hydraulic dredging
13. Use of dredged material as bulkhead backfill
14. Beneficial use of dredged material

Dredge Comments

Opening statement for all dredging projects

Dredging has the potential to impact many of the services provided by and for the natural marine/estuarine ecosystem. The marine and aquatic organisms that live in and near the subtidal bottom are one component of the ecosystem most at risk from dredging operations. The normal assemblage of organisms varies with location and depth, but all can be considered an integral part of the marine/aquatic ecosystem. Resources of special interest include submerged aquatic vegetation (SAV) and shellfish, which play an important role in maintaining water quality and providing habitat. The water column above these bottom-dwelling organisms provides habitat for both swimming and drifting life forms, including both resident and migratory fish and invertebrates, and the larval forms of fish and shellfish. Good water quality is required for these organisms and the healthy functioning of this environment.

Wetlands, both vegetated and nonvegetated, are transitional areas between upland and subtidal areas. They provide habitat (food and shelter) for both aquatic and terrestrial animals such as blue crabs, small fish, and marsh birds. They are highly productive systems and contribute to aquatic food webs through the growth of algae and the export of detritus (dead plant material). Wetlands also improve water quality and help reduce erosion. Plant roots help to improve water quality by filtering groundwater and holding sediment in place. The aboveground portions of marsh plants remove sediment and pollutants from overland flow and help to attenuate wave action.

Dredging can cause a significant disruption of the marine environment, and it often must be repeated in order to maintain water depths. Dredging re-suspends bottom sediments in the water column, which adversely impacts water quality. When material to be dredged includes fine-grained sediments such as silt and clay that remain in suspension for a long time, the adverse impact to water quality can be widespread in both area and time. In addition, dredging eliminates the existing bottom-dwelling organisms. The timeline for recovery of this community and the ecological services it provides is not well known.

Dredged material disposal is another potentially damaging aspect of dredging projects. Dewatering and disposal of dredged material in upland sites away from the shoreline is preferable to overboard disposal. Disposing of dredged material overboard lengthens the time, and often the distance, that suspended sediments are at elevated levels. Placement of dredged material in properly sized and contained upland disposal sites gets the material out of the system, so it is less likely to fill in dredged areas and lead to frequent maintenance dredging. Rehandling of the dredged material should be minimized as it is transported to disposal sites in order to lessen the reentry of material into the aquatic system. Design specifications for dredged material disposal areas are available at (link to website reference).

Wherever possible, dredging should be avoided or minimized by examining both the need for dredging and the possibility of lessening the extent of dredging by constructing open pile piers to reach existing navigable depths rather than dredging to create required depths.

Include as needed:

1. Dredging not justified/no action (edit to include reasoning)

It is our opinion that dredging is not justified in this situation and should be avoided.

2. Dredging avoidance/minimization by piering out

It is our opinion that dredging can be avoided or minimized by constructing open pile pier(s) to reach existing navigable depth, rather than dredging to create the required depth.

3. Dredging avoidance/minimization by lessening extent of dredging

It is our opinion that dredging can be minimized by lessening the extent of the proposed dredging to only what is required for safe navigation.

4. Dredging in wetlands

Tidal wetlands, both vegetated and nonvegetated, are highly productive and valuable components of the marine ecosystem. Dredging converts wetlands to subtidal bottom. Over time, it would be expected that a new shallow water community would be established, with organisms adapted to the new depth, sediment composition, and available food supply. However, the unique and valuable functions of the wetland are lost. Therefore, dredging through wetlands should be avoided.

5. Dredging adjacent to wetlands (4x buffer)

Dredging that takes place adjacent to wetlands should maintain an adequate buffer between the dredge cut and the wetlands in order to prevent slumping and loss of the wetlands. Generally, the toe of the side slope of the design channel should be located at a horizontal distance from the channelward edge of the wetland (i.e., mean low water) that is at least 4 times the depth of dredged material to be removed.

6. Dredging in shallow, narrow creeks and coves

Not all waterfront property is conducive to navigation or appropriate for deep draft boat traffic. The shallow areas of creeks and coves provide important habitat and refuge for marine organisms. Dredging these areas destroys shallow habitat and exposes prey species to increased predation from larger fish. In addition, it is often difficult to maintain appropriate buffers from wetlands in narrow creeks and coves. Tidal flushing may not be adequate to prevent pockets of poor water quality from developing. Dredging of shallow habitat should be avoided.

7. Dredging within shellfish areas, SAV beds, and other areas of high productivity

Dredging within shellfish areas, beds of submerged aquatic vegetation (SAV), and other areas of high productivity, should be avoided. These are highly valuable resources that contribute significantly to the health of estuarine ecosystems. The greater depths created from dredging, and the frequency of maintenance dredging, often preclude recovery of these resources.

8. Dredging in anadromous fish spawning and nursery areas (TOY)

(need current info from fish folks; tailor recommendations as much as possible to specific tributaries)

During the spring spawning run (approximately mid-March through June, but varying between river systems, and dependent on water temperatures which change from year to year), the fish and eggs of anadromous fish can be adversely affected by higher than normal levels of suspended sediments. In anadromous fish spawning and nursery areas, dredging and overboard disposal operations should be avoided during mid-March through June. It is also important to avoid unnecessary water quality impacts as much as possible during the nursery period, July through October, when larvae develop into juveniles and begin their downstream journeys.

9. Dredging in shellfish resource areas (TOY)

High levels of suspended solids caused by dredging can interfere with the development and survival of shellfish larvae. Resulting sedimentation can cover existing shellfish beds and make substrates unsuitable for shellfish. In oyster and clam growing areas, dredging should be avoided during the months of July, August, and September when the majority of oyster spawning and spatfall occurs. Dredging in these areas should also be avoided during December, January, and February, when the pumping activity of shellfish is reduced and they are less able to clear away rapidly accumulating silt.

10. Dredging in industrial areas, other areas of potentially contaminated sediments

Sediments in industrial areas are often contaminated with substances that can be hazardous to marine organisms when the sediments are resuspended and the substances are remobilized. Sediments should be sampled for hazardous substances prior to dredging. If sediments are contaminated, and dredging cannot be avoided, the dredged material should be carefully handled and properly disposed of in an upland disposal site, such as Craney Island, approved for handling contaminated sediments.

11. Dredging deeper than controlling depth

Dredging a channel or basin to a depth that is deeper than that of the waterway to which it is connecting has the potential to result in secondary impacts to the marine environment and is therefore not recommended. Tidal flushing and circulation is reduced in these areas, and the dredged area can become a sink for organic material. Consequently, the potential exists for a reduction in dissolved oxygen levels, adversely affecting water quality.

12. Non-hydraulic dredging

Dredging by bucket, clamshell, dragline, or other non-hydraulic methods creates greater suspended sediment concentrations in the vicinity of the dredging. For large projects, hydraulic dredging is generally preferred because the direct water quality impacts are lessened.

13. Use of dredged material as backfill for bulkhead

Use of dredged material as backfill for a bulkhead is undesireable. The resulting hydraulic back pressure can lead to structural failure of the bulkhead and uncontrolled reentry of dredged material into the marine environment.

14. Beneficial use of dredged material

Provided the dredged material is good quality sand, it may be appropriate for use as beach nourishment, marsh creation, or marsh enhancement where ecologically suitable. Placement of material may be subject to time-of-year restrictions for threatened/endangered species such as the tiger beetle, least tern, and sea turtle.

7.Marina Activities

Marina Comments

Opening Statement for All Marina Projects

The proposed project involves various shoreline and waterway activities associated with a marina operation. Marina activities adversely impact the water quality and habitat ecosystem services of shoreline and coastal resources. These activities include wet storage of boats, commercial structures, boating, fuel handling, solid waste and garbage disposal, shoreline stabilization structures, dredging and upland improvements. Activities associated with marinas should be water dependent in nature if proposed over water.

Marina Location

Direct and indirect adverse impacts on ecosystems services will vary from location to location. Water quality effects depend upon the capacity of the waterway to assimilate the pollutants generated by a marina. Habitat loss, or adverse effect, depends upon the resources in the project area such as wetlands, SAV and riparian forest. The Marina Siting Suitability Tool employs a model of environmental parameters to assess the suitability of shoreline segments for marina development. The preference is to locate marina activity in areas that are ranked high for suitability. These sites will have less adverse environmental impacts, fewer habitat resources, no SAV and good flushing to reduce impacts to water quality. http://ccrm.vims.edu/marinasiting.html

Wet Slips: Commercial Facility

Wet slips and concentrated boat handling introduce petroleum products, toxicants, bacteria, and garbage into the waterway. Petroleum products may enter the water from regular fueling, engine exhaust, bilge water or large oil spills. Plans to address oil spills in the waterway, and on the upland as necessary, should be in place. Provision of pump-out facilities and restrooms and promotion of their use can reduce bacterial pollution. Marinas should provide and maintain sufficient garbage receptacles to reduce solid waste in the waterway. Use of pump-out facilities and the proper handling of garbage should be promoted with signage.

Wet Slips: Community Facility

Wet slips and concentrated boat handling introduce petroleum products, toxicants, bacteria, and garbage into the waterway. Petroleum products may enter the water from regular fueling, engine exhaust, bilge water or large oil spills.

Wet slips should be limited to that number that corresponds to riparian waterfront lots. Plans to address oil spills in the waterway, and on the upland as necessary, should be in place. Provision of pump-out facilities and restrooms and promotion of their use can reduce bacterial pollution. Marinas should provide and maintain sufficient garbage receptacles to reduce solid waste in the waterway. Use of pump-out facilities and the proper handling of garbage should be promoted with signage.

Piers and Commercial Structures

Piers and other structures have shading impacts on the wetlands and waters. The limitation on sunlight exposure reduces the production of all photosynthetic plants including marsh grasses and micro algae. The loss of the vegetated community structure and the primary production results in an adverse change in the habitat services of the impacted area.
Locating the piers where there is no wetlands vegetation or ensuring the piers are a minimum height above vegetated wetlands can reduce shading impacts. In general, piers should be at least the pier width, minus one foot, over the marsh (pier is 6 feet wide, 6 ft minus one = 5 feet elevation).
Sufficient garbage receptacles should be provided and maintained to reduce solid waste in the waterway. Signs to encourage proper handling of garbage and waterway stewardship should be posted. Where the amenity is part of a larger plan that includes upland riparian approach paths, the plan should be designed to minimize additional clearing and grading.

Riparian and Upland Modifications

Impervious surfaces and buildings in the upland, as well as boat repair, painting and cleaning operations, are a source of nonpoint source pollution. Appropriately designed, located and maintained BMPs can minimize the risk of polluted runoff entering the waterway. It is preferable to maintain, or create, a vegetated riparian area and locate upland improvements outside the Resource Protection Area.

Shoreline Stabilization

The shoreline along the marina and immediately adjacent shorelines, from shallow waters to upland edge, may be subjected to erosion from an increase in boating activity. No-wake zones adjacent to the marina can reduce the risk of erosion. If erosion protection structures are used, the least impacting approach is recommended. Vegetation, both wetland and upland, should be used to the maximum extent, in combination with bank modification and rock structures as necessary.

Dredging

Dredging activities adversely impact the aquatic environment by creating increased turbidity through the physical removal of benthic habitat and it's inhabitants. Dredging projects in anadromous fish spawning reaches should comply with time of year restrictions to protect the environmentally sensitive spawning and larval life stages. Dredge material should not be rehandled, but removed, transported and disposed in a proper handling location.

Appropriate location along the waterway, proper site planning and use of piers to reach intended water depth is preferred to dredging. Marinas should be located where there is already adequate water for navigation. If the marina is dependant upon dredging to achieve navigable waters, it is expected that there will be a need for repeated maintenance dredging. Dredging volume and areal extent should be limited to that necessary for safe navigational access. Pier placement and orientation should be designed to eliminate or minimize dredging. Slips should be designed such that the shallower draft vessels are docked in the landward locations with deeper draft vessels nearer the channel or open waterway.

Community Fishing Piers and Passive Recreational Facilities

Piers and other structures over water have shading impacts on the wetlands and waters. The limitation on sunlight exposure reduces the production of photosynthetic plants from marsh grasses to micro algae. The loss of this production, and vegetative community results in a change in the habitat services of the impacted area.

Locating the piers where there is no wetlands vegetation or ensuring the piers are a minimum height above vegetated wetlands can reduce shading impacts. In general, piers should be at least the pier width, minus one foot, over the marsh (pier is 6 feet wide, 6 ft minus one = 5 feet elevation).

Sufficient garbage receptacles should be provided and maintained to reduce solid waste in the waterway. Signs to encourage proper handling of garbage and waterway stewardship should be posted. Where the amenity is part of a larger plan that includes upland riparian approach paths, the plan should be designed to minimize additional clearing and grading.

Boat ramps

Boat ramps cause the conversion of riparian area, wetlands and shallow water to a hardened landform. Generally constructed of concrete or gravel, boat ramps do not provide water quality or habitat services associated with the natural shoreline.

It is preferable is to use existing ramps, rather than construct new ones. Boat ramps should not be located where dredging or sand trapping is required to provide navigation or where major bank grading is necessary for upland access. Boat ramps should be located on the shoreline where loss of vegetated wetlands, dunes and native riparian vegetation is avoided.

8.Time of Year Restrictions

Time of Year Restrictions

Location	**Resource**	**J**	**F**	**M**	**A**	**M**	**J**	**J**	**A**	**S**	**O**	**N**	**D**
Lynhaven River *Below Trans Point and Hebdon Cove*													
	Shellfish	x	x	x			x	x	x	x			x
Lynnhaven River *Lynnhaven Inlet*													
	Summer flounder			x	x	x	x	x	x	x			
James River *Upper Brandon to fall line*													
	Anadromous fish		x	x	x	x	x						
Beaches Hampton, Mathews, Middlesex, Lancaster, Northumberland, Northampton, Accomack *Lower river and Bay front*													
	Tiger Beetles						x	x	x	x			
	Piping Plover			x	x	x	x	x	x				
	Terns/black Skimmer				x	x	x	x	x				
	Loggerhead Sea Turtle					x	x	x	x	x	x	x	
Various locations													
	Bald Eagles	x	x	x	x	x	x	x					x
Wading Bird Rookeries	Great Blue Heron			x	x	x	x	x					
	Great Egret				x	x	x	x					
	Green Heron				x	x	x	x	x				
	Yellow-Crowned Night Heron					x	x	x					

Ecosystem Services

Marshes

Marshes are transitional areas between upland and sub-aqueous lands. They provide habitat (food and shelter) for both aquatic and terrestrial animals such as blue crabs, small fish and marsh birds. They are highly productive systems and contribute to aquatic food webs through the growth of algae and the export of detritus. Marshes also improve water quality and help reduce erosion. Grass roots help to improve water quality by filtering groundwater and holding sediment in place. The shoots remove sediment from overland flow and help to attenuate wave action.

Saltbush

Saltbushes provide habitat for wildlife, particularly for nesting birds. Saltbush roots help to stabilize sediments, slowing erosion.

Mown wetland vegetation

Marshes are transitional areas between upland and sub-aqueous lands. They provide habitat (food and shelter) for both aquatic and terrestrial animals such as blue crabs, small fish and marsh birds. They are highly productive systems and contribute to aquatic food webs through the growth of algae and the export of detritus. Marshes also improve water quality and help reduce erosion. Grass roots help to improve water quality by filtering groundwater and holding sediment in place. The shoots remove sediment from overland flow and help to attenuate wave action. When mown, the ability of the marsh to provide ecosystem services is greatly compromised. However, services should return when mowing stops.

Marsh Spits

Unvegetated wetlands

Unvegetated wetlands provide foraging areas for shorebirds, crabs and young fish. Although important habitat, they tend to be less productive than vegetated subaqueous and intertidal areas.

Undercut bank with forested riparian area

Riparian trees provide habitat for birds, mammals and fish and contribute to water quality by slowing runoff, taking up groundwater, and stabilizing sediments. In addition, unvegetated intertidal areas also provide unique habitat and water quality functions.

Beaches and Sandy Shorelines

Sandy shorelines are a dynamic component of tidal rivers, the Bay and Atlantic shoreline. They are typically associated with moderate to high-energy conditions and can contribute to local sediment dynamics through two processes. There is sand that moves along the shoreline that comes from eroding bluffs and sand that moves on and off shore between the flats, the shallows and off shore bars. As sand moves on and off shore beaches also interact with primary and secondary sand dunes and sandy berms. Beaches can provide natural shoreline protection by forcing waves to shoal and break before reaching the upland. Beaches are habitat for benthic animals and microalgae living on or within the sand. The beaches serve as refuge and forage area for finfish, blue crabs and wading shorebirds.

Dunes

Coastal primary sand dunes serve as protective barriers from flooding and erosion, provide reservoirs of sand to replenish the beach zone, and provide habitat for a variety of plants and animals. Plants adapted for life on coastal primary sand dunes must tolerate very limited amounts of fresh water, constant salt spray, and withstand marked variations in temperature. The natural vegetation occurring on sand dunes can act as a baffle, slowing wind speed and causing wind-borne sand to settle and be trapped in the vegetation resulting in accretion of the dune.

Structures can adversely affect the structure, form and function of dunes because they interfere with wind and sand deposition patterns and natural dune building processes. Structures may also shade or displace dune vegetation.

Tidal intrusion behind bulkhead

Landward of the deteriorating bulkhead, tidal intrusion has created mudflat wetlands. Mudflats can be a foraging area for both waterfowl and aquatic animals and play an important role in nutrient cycling. The mudflats on this property are very small and aquatic access is blocked by the bulkhead, therefore they are expected to have limited habitat value but may still contribute to other ecological services.

Landward of the deteriorating bulkhead, tidal intrusion has created vegetated wetlands. Wetlands provide habitat (food and shelter) for both aquatic and terrestrial animals such as blue crabs, small fish and marsh birds. They are highly productive systems and contribute to aquatic food webs through the growth of algae and the export of detritus. Marshes also improve water quality and help reduce erosion. Grass roots help to improve water quality by filtering groundwater and holding sediment in place. The shoots remove sediment from overland flow and help to attenuate wave action. The wetlands on this property are very small and aquatic access is blocked by the bulkhead, therefore they are expected to have limited habitat value but may still contribute to other ecological services.

Forested Riparian

Forested riparian zones provide habitat for birds, fish and terrestrial animals. They intercept rain and runoff, helping to slow erosion, and intercept groundwater flow, removing nutrients. Their roots help to stabilize banks and reduce erosion.

Eroding Banks

Eroding banks contribute to the sediment budgets of rivers and the Bay. High levels of clay in the water may contribute to turbidity; however, sandy bluffs are frequently the sand source for adjacent and neighboring beaches. Stabilizing these banks permanently removes the sediment source from the system. Sand flats can provide natural shoreline protection by forcing waves to shoal and break before reaching the upland. Sand flats are habitat for benthic animals and microalgae living on or within the sand. The flats serve as refuge and forage area for finfish, blue crabs and wading shorebirds.

SAV

Submerged Aquatic Vegetation (SAV) beds are highly productive ecosystems which provide food and habitat for several fisheries species and help improve water quality by stabilizing sediments and reducing turbidity. The range of SAV beds in the Chesapeake Bay watershed has been greatly reduced from the range in the 1930s, which makes these beds of prime concern for conservation.

Oyster Reef

Oyster reefs are highly productive ecosystems which provide food and habitat for several fisheries species and help improve water quality by reducing turbidity. The number of oyster reefs in the Chesapeake Bay watershed has been greatly reduced from the range in the 1930s, which makes these beds of prime concern for conservation.

Subaqueous shallows

Subaqueous shallow areas provide erosion protection for upland and intertidal areas by forcing waves to break, reducing wave energy. They provide habitat for forage fish, juvenile fishery species, and several invertebrates. Although important habitat, they tend to be less productive than vegetated subaqueous and intertidal areas.